Synthesis Paper:

An Assessment of Research on Rangeland Fire as a Management Practice

Ryan F. Limb¹, Samuel D. Fuhlendorf², David M. Engle², and Richard F. Miller³

Authors are ¹Assistant Professor, School of Natural Resource Sciences, North Dakota State University, Fargo, ND 58102, USA; ²Regents Professor, Department of Natural Resource Ecology & Management, Oklahoma State University, Stillwater, OK 74078, USA; ³Professor Emeritus, Department of Animal and Rangeland Sciences, Oregon State University, Corvallis, OR 97337, USA.

Key words: ecosystem management, fire effects, prescribed burning, wildland fire

Correspondence: Ryan F. Limb NDSU Dept 7680 PO Box 6050, Fargo, ND 58108

Email: ryan.limb@ndsu.edu

Introduction

Fire has played a key role in the formation of most rangeland ecosystems in North America (Axelrod 1985) and the world (Bond et al. 2003; Keeley and Runcel 2005). Alteration of fire regimes on US rangelands since European settlement has created cases of severely altered ecosystems that can eventually result in no-analog, novel, or emerging ecosystems (House et al. 2003; Hobbs et al. 2009). According to LANDFIRE (an interagency vegetation, fire, and fuel characteristics mapping program sponsored by both the US Departments of Interior and Agriculture), three-fourths of US lands dominated by native plants show moderate or high departure from reference conditions as a result of altered fire regimes (The Nature Conservancy 2009). Because most rangelands are considered fire-dependent ecosystems, restoring historical fire regimes is fundamentally important when the management goal is to restore or maintain the potential (or historical) natural community. The historical plant community for most ecological sites was maintained by fire, and altered fire regimes are a dominant cause of ecological sites moving across a threshold of increased woody plant dominance (Twidwell et al. 2013), leading to reduced livestock production and loss of other ecosystem services such as pollination (Chi and Molano-Flores 2015) and soil stabilization (Puttock et al. 2014). Rapid and extensive woodland expansion on rangelands clearly reflects the essential role of fire in the maintenance of historical rangeland ecosystems (Archer 1994, Limb et al. 2010).

However, implementation of prescribed burning as a conservation practice in the rangeland management profession has been overshadowed by implementation of other practices, especially prescribed grazing. Disproportionate implementation of these two

categories of practices is influenced by complexity of social interactions among agencies, public perception of risk vs reward, and public policy (Twidwell et al. 2015). Social and policy concerns differ greatly across rangeland regions, ranging from complete acceptance in fire cultures (e.g., Flint Hills of Kansas and Oklahoma) to attempts to completely remove fire from the landscape (e.g., Great Basin).

With a few exceptions, fire regimes in grassland ecosystems are altered through intentional fire suppression and by grazing that uniformly reduces fuel loads. In the Great Basin, reduction of fine fuels through grazing have also reduced fire occurrence in the wetter and cooler sagebrush communities where fuels would be otherwise sufficient to carry fire (Miller et al. 2011; Balch et al. 2013). In contrast, the warmer and drier sagebrush communities that are invaded by exotic annual grasses display fine-scale fuel homogeneity, which enables more frequent fire. State-and-transition models suggest conversions to woody plant dominance and to exotic annuals can eventually become irreversible and result in alternative stable states (Twidwell et al. 2013).

Rangeland ecosystems are characterized generally as working landscapes and are subject to diverse management, often to meet economic objectives. However, management of rangeland ecosystems is increasingly difficult with the growing demand for goods and services coupled with declining rangeland area (Anderson and Inouye 2001; Fischer and Lindenmayer 2007). Conserving historic species and processes, while maintaining economic stability, requires management practices that are consistent with historic disturbances (Limb et al. 2011b), although exceptions exist where ecosystems

have crossed ecological or social threshold boundaries (Davies et al. 2009, Limb et al. 2014).

The Natural Resource Conservation Service (NRCS) commissioned the Conservation Effects Assessment Project (CEAP) to determine if conservation practices align with current science. The rangeland CEAP addressed the need to evaluate current rangeland management practices and broadly assess the impact of management activities on environmental parameters and rangeland conservation. Prescribed burning (Prescribed Burning, Code 338) was one of the NRCS National Conservation Practice Standards (USDA NRCS 2015a) evaluated through peer-reviewed scientific literature. A complete evaluation of the standard is available (Briske 2011). Herein, we updated the recent review of prescribed fire as a land management practice (Fuhlendorf et al. 2011), include a subset of the National Handbook of Conservation Practices (NHCP) standards, and discuss fire in the context of a global management practice.

Defining Our Literature Database

Rangeland ecosystems across the globe vary substantially by climate, soil type, floral and faunal composition as well as goods and services produced. Given the broad disparity of rangeland ecosystems across the globe, we analyzed the research literature to establish the ecological effects of prescribed fire as a management practice on rangelands. We intentionally took a broad perspective to include research articles on plants, soil, water, wildlife, arthropods, livestock, fire management, fire behavior, smoke management, socio-economics, air quality, fire history and human health. We also addressed spatial scale and temporal scale of field studies and other descriptions of field studies (private or public land,

grazing status, and ecosystem type) that compose the body of rangeland fire research, and we then relate our findings to the practice of prescribed burning on rangeland.

Evaluation of the peer-reviewed literature on prescribed fire first required determining methods to query the entire body of scientific literature on the topic. We wanted to include all relevant papers, but we limited the scope of the search to exclude fire research from forested ecosystems, which dominates the fire research literature. Many papers that report fire research on rangelands do not include the term *prescribed*, and many relevant papers do not use the term *rangeland*. The data set built from the search with the term *prescribed fire* indicated that numerous important papers were omitted from the pool, and many of the papers included some discussion of fire but with minimal or no data related to fire. Therefore, our final search used the term *fire*, which also located articles with *prescribed fire* in the title, to broaden the search. Although this approach likely excluded some papers that reported research from regionally important ecosystem types (e.g. shinnery oak or chaparral vs. shrubland) and papers in which the title contained other key fire-related words (e.g., burned, burning, and prescribed burning) but not fire, the search located more than 1000 papers. We do not believe that this is an exhaustive dataset, but we argue that it provides an adequate, unbiased sample from which we could determine the nature of information available through the peer-reviewed literature. Because fire can both positively and negatively influence certain components of ecosystems, fire should be evaluated to address a broad array of ecosystem components. We evaluated and summarized the entire dataset from a global perspective, and for this article, we narrowed the focus to North American rangelands regions to illustrate

differences and similarities on key topics. As with the comprehensive search, we used Web of Science to search for papers on a particular topic. We justified limiting our search to these topics on the basis that indexed articles are widely accepted as scientifically valid peer-reviewed literature.

Of the 1039 papers (available in October 2015) from our query through Web of Science, 846 papers were accessible and confirmed to be peer-reviewed research papers (Appendix 1). Of these 846 papers, less than 10 papers were published annually from 1967 through 1989, which was followed by an exponential increase with 50 or more papers published each year from 2009 to 2014 (Figure 1). In contrast, when we replaced the search term “fire” with “grazing,” more papers were available (14,912), but the increase was linear rather than exponential for the same time period. This suggests that the research community may view fire with increasing importance.

Evaluation of the Dataset

An important outcome of the search was that rangeland fire research literature is dispersed among numerous ecosystems and across most continents. More than 270 journals published rangeland fire research, with studies primarily located in North America (48%), but substantial research was conducted in Africa (14%), Australia (14%), and Europe (11%). Fewer studies were conducted in Asia and South America (5 and 8% respectively). The majority of papers reported research on plants, fire management, soils, fire behavior, socio-economics, and wildlife (Table 1). Authors described their papers as addressing a variety of vegetation types, with almost half of the papers classified as savannas and grasslands.

The scientific literature generally supports the value and need to include prescribed fire as a component of conservation practice recommendations. However, one of the most revealing outcome of our search was that it uncovered a general misalignment of research to management application, a fundamental barrier to constructing research-based information to support prescribed burning. In particular, most of research was conducted at temporal and spatial scales that do not apply to management—48% of the studies were less than 4 years in duration and almost 25% of the studies were less than 1 ha in area. Many studies were conducted on plots much smaller than 1 ha or on individual plants. However, even more problematic is that over half of studies did not report the time period treatments were in place or area of the experimental unit (62% and 56%, respectively). Transferring information from small research plots to landscape-level management is challenging and is not limited to fire research (Twidwell et al. 2013). Approximately 20% of the studies were based on modeling and 8% on Geographic Information Systems with minimal field evaluations or immediate application to management. Reliance on information resulting from single fires applied on small plots tracked for a relatively short time interval greatly constrains inferences and application to ecosystem management, and this information should be applied with caution.

The fire regime, features that characterize fire as a disturbance within an ecosystem—fire frequency, severity, behavior (i.e., fire intensity), predictability, size, seasonality, and spatial pattern (Morgan et al. 2001), was rarely evaluated. Only 11.5% of the papers focused on fire frequency, and only 12% focused on understanding changes that occur over variable times since fire. Fire season was evaluated in 7.5% of the studies

and fire intensity was evaluated in 15% of the studies. Many studies (33%) simply compared a burn treatment without any description of fire behavior and/or intensity to an unburned control, which simplifies fire to the point of irrelevance to management. Further, the literature generally ignores fire as a dynamic disturbance process that occurs on complex landscapes in conjunction with other disturbances such as grazing and drought (Limb et al. 2011a).

Fire effects on ecosystem components

Plants

The dataset included 338 papers focused on plants and plant communities. Specific topics varied greatly, but papers generally evaluated change in species composition and diversity, productivity, individual plant mortality, germination, or establishment. A few papers (<1%) considered forage quality. From the dataset, we conclude that plant productivity, plant nutrient content, plant diversity and plant mortality responses to fire are highly variable (Blair 1997; Reich et al. 2001). Metrics used and how productivity is defined (i.e. standing crop, aboveground net primary production (ANPP) or biomass) also varied.

Production - Production of herbaceous communities following fire were highly variable across studies with increases, decreases and no changes being reported depending on ecosystem, fire intensity/season, and weather conditions. Forage or herbaceous production often decreases for 1-2 years after fire followed by a positive to neutral effect (Table 2). An exception occurs in the tallgrass prairie where productivity can be enhanced after one spring fire following several years without fire (Blair 1997). Total

above ground biomass appears to be negatively associated with fire frequency although this depends on the ecosystem. Several studies indicate an increase in forage quality in mesic grasslands following fire (Hobbs et al. 1991; Allred et al. 2014), but these patterns are uncertain in more xeric regions. Because individual species and communities differ widely in response to fire, we could not draw generalized conclusions about species diversity and plant mortality following fire.

Community composition - Almost half of plant papers focused on plant community composition, and we grouped papers to identify regional differences and similarities among deserts, grasslands, savanna and shrublands. One of the strongest arguments for the use of prescribed fire is to maintain or restore a desired successional community or pattern of communities in different stages of recovery following fire. Of the papers reviewed, 65% reported results of prescribed burns, 21% reported results of wildfires and most (>75%) fire treatments were applied in spring and summer. Several studies in grassland ecosystems recognized that season of fire mostly had a minimal or temporary effect (Engle and Bidwell 2001, Limb et al. 2011a). However, mixed results have been reported in the Intermountain region (Miller et al. 2013; Bates and Davies 2016). A major concern identified in the database is the limited number of long-term studies. Only twenty-four percent of the studies extended past 10 yrs, of which a majority substituted space for time (comparing vegetation across fires of different ages), with 50% of studies not extending beyond 3 years.

Response of individual plants to fire was highly variable both across and within regions and ecosystems (Table 2). A large portion of this variability can be attributed to the

interaction of site characteristics, fuel characteristics, climate, community composition, time since fire, fire season, fire intensity, and post-fire management. However, several patterns are evident in fire related plant responses across regions and ecosystems.

First, perennial grasses declined in abundance in the first post-fire growing season in 76% of the studies, but usually recovered within the second or third year. Abundance of perennial grasses increased in only 11% of the studies in the first post-fire growing season and 5% in the second or third year following fire, but no studies reported long-term declines in perennial grasses. Differences between rhizomatous and caespitose growth form can account for much of the variance in post-fire response of grasses. Given that rhizomatous grasses are more common in the Great Plains and caespitose grasses more common in the Intermountain West, we discuss these groups in greater detail within the respective regional summaries. Second, annual grasses were usually more abundant in the first, second and third years following fires compared to unburned stands. Third, forbs were inconsistent in their response the first year following fire, but they were more abundant in 4 out of 6 studies by the second or third year. Fourth, abundance (biomass, cover, or volume) of both resprouting and non-sprouting shrubs was lower during the first 10 years following fire. However, density of sprouting shrubs usually equaled or exceeded that of unburned communities within 3 years following fire suggesting little or no mortality. Full recovery of sprouting shrubs occurred within 3-20 years, 25-35 years for non-sprouting shrubs on relatively wet sites (e.g. mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*) in the 300-400 mm precipitation zone), and greater than 45 years on dry

sites (e.g. Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*) in the 200-300 mm precipitation zone).

Species groups - Fire can be used to change plant composition (e.g. the proportion of C₃:C₄ plants, herbaceous:woody, and forbs:grasses) and reduce excessive litter buildup. Reduced litter results in an increase of light to basal tillers, often an issue restricted to highly productive sites (Howe 1994, 1995; Ansley 2008). The literature is mixed on the issue of fire effects on invasive species. Fire can act as a trigger to force a desirable stable-state across a threshold to an undesirable invasive plant state. Cheatgrass (*Bromus tectorum* L.) provides an excellent example of this dynamic in the Intermountain West (Keane et al. 2008). Fire, on the other hand, can be used to control invasive species through direct control or by focusing herbivory on a relatively small burned area within a landscape that has higher palatability and forage quality after burning (Cummings et al. 2007).

Woody species - Prescribed fire is particularly relevant to the issue of woody plant encroachment in grassland and savannah ecosystems. Woody plant abundance (biomass, cover, volume) consistently declined the first year following fire and was generally less than the control 3 years after fire (Table 3). Density of resprouting shrub species recovered or exceeded pre-burn levels within 3 years following fire, and grassland may transition to woodland with as little as 30 years without fire. Most shrubs in the Great Plains are resprouting and fire return intervals of 2-5 years may be required to maintain herbaceous dominance. Non-sprouting encroaching trees, primarily Ashe juniper (*Juniperus ashei*) and eastern redcedar (*Juniperus virginiana*), increase without fire and

gain dominance after about 30 years (Limb et al. 2010). The effects of these fires depend on grazing intensity, which constrains fuel load and fire intensity.

The majority of studies from the Intermountain West evaluated summer burns (57%, many of which were wildfires), but more than half also considered fall or spring prescribed burns (35% fall, 21% spring). Thirteen studies reported total perennial grass response in the first post-fire growing season, of which ten reported a decline in cover, biomass, or density, one an increase in cover, and two no change in cover. The majority of these studies (7 of 9) showed that perennial grass recovered to that of unburned plots within 2-3 years, while one study showed a decline in cover and one study an increase in cover. Perennial forbs generally increased as did annual grasses in the first post-fire growing season. However, a review on fire effects in the Great Basin indicated increases in perennial forb abundance typically occurred in the cool-moist sagebrush communities, with little to no change in the warm-dry sagebrush communities (Miller et al. 2013). Broadleaved grasses and smaller bunches typically were more resistant to fire than fine leaved grasses or large bunches. In the Intermountain West, pre-fire composition is a major driver in determining post-fire composition and ecological sites on the cooler-wetter soils (e.g. mountain big sagebrush) more resilience to fire disturbance (Miller et al. 2013; Chambers et al. 2014). Sites invaded with exotic grass species may require post-fire seeding to assist the reestablishment of native species and reduce further invasion (Davies et al. 2013). Coupled herbicide and seeding applications are common, but timing of treatments and weather conditions strongly influence outcomes (Munson et al. 2015).

Woody plants have captured much attention in the Intermountain West. Non-sprouting shrubs, primarily mountain big sagebrush and bitterbrush (*Purshia tridentata*) (a weak resprouter), consistently decreased with fire typically recovering within 25 to 35 years. However, recovery of Wyoming big sagebrush is typically longer with one study reporting only 5% sagebrush cover after 23 years following fire. This is especially the case on landscapes receiving <200-mm of precipitation. Few studies evaluated resprouting shrubs *Chrysothamnus viscidiflorus*, *Ericameria nauseosa*, horsebrush (*Tetradymia* spp.). However, limited work indicates biomass declines of these species in the first several years following fire but density typically recovers to preburn levels within 3 years.

Density of juniper (*Juniperus* spp.) and piñon pine (*Pinus* spp.) is reduced following fire, but tree size often influences the effectiveness of burning. Juniper cover of individual trees increased slowly for the first 45 yrs followed by rapid increase during the next 45-70 years. Closed canopies can develop within 80 to 120 years (Johnson and Miller 2006). Fire in woodlands is typically followed by an increase in perennial grasses and a reduction of woody plants. Sagebrush (*Artemesia* spp.) reached pre-burn levels within 40 yrs and then often declined if piñon pine and/or juniper became established on the site. Understory cover declined to 5% of the adjacent vegetation by 100 yrs following fire as piñon and juniper woodlands developed (Barney and Frischknecht 1974; Wangler and Minnich 1996). However, understory composition following fire is highly dependent on the composition and abundance of the understory prior to the burn, time since fire, and fire severity, especially on the lower resilient warm-dry sagebrush ecological sites (Chambers et al. 2014). Targeted management strategies will often incorporate mechanical treatments

followed by seasonal burning to increase woody species control and accelerate herbaceous recovery (Bates et al. 2014).

Soil and hydrology

The literature on the effects of fire on rangeland soils is abundant. However, only 16 of the 51 studies reported research from North American rangelands limiting the inferential base for applying the results to management. Fortunately, these were distributed more-or-less evenly across North America and across vegetation types (grasslands, shrublands, etc.). Small plots (0.0003 - 1 ha) were the general rule, and studies often reported effects from a single fire (22 papers). Only ten of the 51 studies encompassed time periods of ≥ 10 years.

DeBano (2000) reviewed over 700 published papers reporting on either various aspects of water repellency that contribute information directly related to understanding or the basic processes that underlie soil water repellency. The influence of fire on soil depends largely on the pre-fire and post-fire environment, interaction with other factors including grazing and invasive species, and the evolutionary history of the ecosystem with regard to fire frequency and intensity and grazing. However, it is notable that most studies on rangeland that can be characterized as lacking a long evolutionary history of both frequent and expansive fire are conducted after wildfire without treatment controls. For example, portions of the Great Basin shrub-steppe have had substantial increases in fire frequency and burn area over the past century (Miller et al. 2011). In contrast, prescribed burning and the ecological role of fire are the context of studies on rangelands characterized by a long evolutionary history of frequent fire, specifically the Great Plains.

Soil organic matter has long been a subject of interest to rangeland fire researchers (e.g., Reynolds and Bohning 1956; Owensby and Wyrill 1973). Research has increasingly reported the influence of fire on soil organic carbon, CO₂ ecosystem flux, and carbon sequestration, which is tied to atmospheric properties related to global climate change. In the single study located in the U.S. that appeared in our sample (Ansley et al. 2006), carbon storage in soil increased with fire, likely the result of a shift in species composition. Burning in the Intermountain region typically has limited impact on total carbon, since the majority of the carbon pool is below ground (Rau et al. 2009, 2010). Studies in the Intermountain shrub-steppe are mixed in their findings regarding burning and soil C and N. Several suggest plant invasions (i.e., *Bromus tectorum*) can reduce soil carbon (Bradley et al. 2006; Prater et al. 2006) or influence the vertical distribution of carbon (Rau et al. 2011), but this did not occur in a similar shrub community when perennial native grasses dominated post-fire (Davies et al. 2007). However, while repeated burning of invaded sites changed litter C and N content, burning did not reduce soil nutrients (Jones et al. 2015).

Burning in a semiarid Great Plains rangeland can alter soil carbon, but the magnitude of change was inconsequential partly because of a relatively low CO₂ flux (MacNeil et al. 2008). In sub-humid Great Plains rangeland where CO₂ flux is markedly greater, soil carbon flux increases with periodic burning over non-grazed rangeland. This is because burning removes accumulated litter that creates temperature and light-limiting conditions for plant growth, but annual burning will reduce both soil organic matter and nitrogen mineralization (Ojima et al. 1994; Blair 1997). Annual burning over perhaps 20 to 100 years may increase the fraction of passive soil organic matter at the expense of more

active fractions, which might ultimately reduce total soil organic carbon (Ojima et al. 1994). Most ecosystem carbon loss in rangeland results from combustion of above-ground organic material (i.e., fuel), with the time to reach pre-fire levels dependent on primary productivity (MacNeil et al. 2008). Soil carbon response to the ecological interaction of fire and grazing, rather than the statistical interaction, has not been investigated. However, because nitrogen and carbon are coupled in the organic matter pool, soil carbon might show similar increases to soil nitrogen following fire-grazing disturbances (Anderson et al. 2006).

Nitrogen in aboveground biomass is volatilized in fire, and the degree of volatilization varies with both environmental conditions and fire characteristics. Specifically, drier fuels and soils coupled with hotter fires result in more intense combustion and more nitrogen volatilization (DeBano et al. 1979). Because most prescribed burning objectives call for conditions that consume most above ground herbaceous fuel, it is often assumed that fire depletes ecosystem nitrogen (Anderson et al. 2006). Indeed, post-burn soil inorganic nitrogen (NO_3 and NH_4) is often less, but greater herbaceous aboveground annual production and vegetation cover at some point after burning suggests plant-available nitrogen increases following burning. Research in sub-humid rangeland (Blair 1997) and semi-arid rangeland (Davies et al. 2007) indicates that burning increases nitrogen mineralization and enhances other mechanisms that result in increased nitrogen availability. Therefore, burning indirectly enhances plant ability to utilize nitrogen. However, in the Central Great Basin, total aboveground nitrogen declined by 78%, but no change occurred in total belowground nitrogen (Rau et al. 2010; Jones et al.

2015). The amount of total nitrogen volatilized was directly related to the amount of organic matter consumed. Annual burning of sub-humid rangeland over a period of 20 to 100 years has been predicted to reduce mineralizable nitrogen similar to the effect on soil organic carbon (Ojima et al. 1994). Because grazing reduces the amount of nitrogen available for volatilization by fire and because nitrogen loss is proportional to biomass available for combustion, grazing lessens the effects of fire on soil nitrogen (Hobbs et al. 1991). This explains why nitrogen fertility was not diminished with annual burning coupled with long-term moderate grazing (Owensby and Anderson 1967). Moreover, this mediating effect of grazing is subject to the effect of scale and preferential grazing of patches (McNaughton 1984; Hobbs et al. 1991). When fire and grazing interact spatially (i.e., the ecological fire-grazing interaction) in a sub-humid rangeland, plant-available nitrogen increases in recently burned, grazed patches (Anderson et al. 2006), but unburned patches with minimal grazing pressure have low levels of available nitrogen. No published research on the effects of the fire-grazing interaction on soil nitrogen is available for other rangelands.

Some fire prescriptions, wildfire conditions, and high fuel situations in rangelands can result in extreme soil heating, which can markedly change soil chemical and physical properties. Brush piles and thinning slash, in particular, create intense heat that can change biological, chemical and physical properties of soil and can facilitate undesirable vegetation change including plant invasions (Neary et al. 1999; Haskins and Gehring 2004). In the Intermountain region, soil surface temperatures during fire are spatially

heterogeneous due to large differences in fuel abundance and structure in interspaces and beneath grasses, shrubs, and tree canopies (Korfmacher et al. 2003; Bates et al. 2011).

Hydrologic variables related to fire effects on soil were water repellency, water quality, hydraulic conductivity or infiltration, and erosion/runoff. Generally, limited information is available on rangelands, and the majority of studies were monitored for ≤ 3 years: 1 year (52%), 2 years (20%), and 3 years (12%). Variables that influenced the effects of fire on soil hydrology were aspect, fire severity, and microsite location. A large decrease in infiltration and increased erosion can occur after fire in coppice dunes beneath shrubs and trees. However, recovery for most processes occurred within 2 years following fire. The largest decrease in infiltration rate and increase in erosion following fire can occur in coppice dunes beneath shrubs and trees. Fire often has little effect on these two variables in shrub or tree interspaces. Water repellency usually occurred on both burned and unburned sites but usually increased, particularly beneath shrub and tree canopies, following fire. Hydrophobicity was reported to decline within several months to near preburn levels following wetting. One study reported rill erosion as the primary source of sediment and several studies reported rills readily formed in the coppice dunes. However, post-fire wind erosion in the drier-warmer sagebrush steppe, has been found to move significant amounts of soil (Miller et al. 2012; Sankey et al. 2012). In general, these studies suggest that immediate effects of fire are largely negative on soil hydrologic properties, but effects are short-lived.

Livestock

Among the 846 papers evaluated, most studies did not specifically discuss the interaction of fire with grazing, and only 47% of the studies specifically stated that they included non-grazed or grazed sites. Only thirteen papers evaluated the influence of fire on livestock behavior or performance, of which seven studies were conducted in North America. The small number and geographic scope of the studies limit the inferences that can be made. Grazing was a part of the experimental design in only 16% of the studies. Because the vast majority of rangeland is grazed, the failure of research to address the interaction of fire and grazing severely limits applying the research to rangeland management.

Early research from the Great Plains region focused on livestock performance and indicates that mid spring and late spring burning increased steer weight gains, particularly early in the growing season (Anderson et al. 1970; Smith and Owensby 1976). Burning can improve livestock performance as much as 75% over unburned pastures, but break-even returns are not always immediate (Kirk et al. 1976; Limb et al. 2011b; Allred et al. 2014). The increased benefit from burning alone was comparable to supplemental feeding. Increased livestock performance is mostly attributed to increased forage quality from increased plant crude protein and decreased fiber content following fire (Grelen and Epps 1967; Allen et al. 1976; Kirk et al. 1976). Within the fire-grazing interaction, cattle are attracted to recently burned locations (within the same growing season), which provide higher quality forage than the surrounding matrix of unburned vegetation, and cattle devote 75% of grazing time within recently burned areas (Fuhlendorf and Engle 2004). The disturbance created by the interaction of cattle grazing and fire mimics historical grazing behavior of large ungulates and creates a structurally diverse landscape.

Wildlife

Fire effects on faunal assemblages (100 studies) are largely dominated by avifauna (27%) and small mammals (15%). However, since 2000, a growing number of studies are related to invertebrate communities (37%). With limited exceptions within the invertebrate literature, the vast majority of research targeted the influence exerted by fire on habitat structure to which these faunal assemblages are especially sensitive. With a diverse assemblage of habitat requirements among rangeland wildlife species, broadly characterizing the effects of fire on wildlife habitat is not realistic. Generally, conditions post-fire will temporarily create favorable opportunities for a specific subset of species and unfavorable conditions for other species as a function of time since fire. For example, many, but not all, rangeland small mammal and bird species are more suited to areas not recently burned and grazed because these areas provide vegetative cover required for concealment or nesting. However, recently burned locations do favor upland sandpiper (*Bartramia longicauda*), horned lark (*Eremophila alpestris*) and the gopher tortoise (*Gopherus polyphemus*), whereas greater prairie chicken requires a matrix of burned and non-burned prairie (Fuhlendorf et al. 2006; Reinkensmeyer et al. 2007; Ashton et al. 2008; McNew et al. 2015).

The landscape context in which prescribed burning is applied on rangeland will affect the wildlife species in question. For example, greater prairie chicken responds positively when portions of a landscape are burned, but negatively when entire landscapes are burned (McNew et al. 2015). Additionally, wildlife species in a given area have variable habitat requirements, so positive response by one species will likely cause other species

to decline. However, because rangeland and rangeland wildlife evolved with periodic fire and because periodic fire is required to maintain habitat suitable for wildlife species native to a particular rangeland region, fire is essential for maintaining rangeland wildlife populations. The sage-grouse (*Centrocercus urophasianus* and *C. minimus*) is in a precarious situation being both dependent on non-fire tolerant sagebrush habitat (Crawford et al. 2004) but also limited by woody species encroachment following fire suppression (Farzan et al. 2015). Unnaturally long fire-return intervals often lead to tree encroachment and other changes that reduce habitat suitability for native wildlife species that are habitat specialists (Coppedge et al. 2001; Reinkensmeyer et al. 2007), some of which are species of conservation concern. Fire-return intervals greater than those with which an ecosystem evolved can have correspondingly deleterious effects on habitat and populations of habitat specialists (Robbins et al. 2002; Pedersen et al. 2003; Fuhlendorf et al. 2012; Rowland et al. 2006).

Air quality/smoke management

Air quality influenced by rangeland fire and associated smoke management are relatively new fields of study, and therefore, limited information is available. A total of 34 studies were reviewed, of which 94% were dated after 2000. Rangeland fire generates a wide variety of by-products that fall into two broad categories, gasses and particulates. Smoke, the visible product of partially combusted fuel material, contains an array of organic and inorganic airborne particulates. Airborne particles can be a nuisance, reducing visibility hundreds of kilometers downwind from emission sources (Ferguson et al. 2003; McKenzie et al. 2006) and degrading air quality (Martin et al. 1977). Prescribed fire produces 15 to

25% of airborne particulates and 7 to 8% of hydrocarbons emitted to the atmosphere annually (Martin et al. 1977; Barnard and Sabo 2003). However, environmental conditions, fuel characteristics, and fire behavior influence the amount of non-combusted material produced. Relatively slow-moving fires, in which more complete combustion is expected, were modeled to produce about 50% less smoke than fires with higher rate of spread and less complete combustion (Glitzenstein et al. 2006). Fuel consumption and fire intensity clearly influence emissions from fire.

Smoke particle size influences the period of suspension in the atmosphere. Relatively large particles, between 0.07 μm and 1.0 μm diameter, may take days to settle out, whereas small particles $<0.07 \mu\text{m}$ do not settle under natural conditions (Martin et al. 1977). Particles $>2.5 \mu\text{m}$ do not remain suspended for long time periods, yet can be problematic for individuals with asthma or other chronic respiratory conditions (Dockery et al. 1993). While smoke management is important as it relates to air quality, our review of the literature revealed that only six papers addressed smoke management on rangeland, so conclusive evidence is limited. Further investigation is needed to provide a complete understanding of how prescribed fire influences air quality.

Conclusion and Management Implications

Fire is as important as climate and soils to the maintenance of structure and function in many ecosystems, and altered fire regimes are a primary cause of ecological sites moving across a threshold (Twidwell et al. 2013). Recognition of fire as a dynamic regime that has variable effects depending on the interaction with grazing and climate patterns is critical to applying management that will optimize conservation effectiveness in rangeland

ecosystems. Fire effects on ecosystems are often considered to be static over time, even though the research literature indicates that fire effects vary over time (both in terms of time since fire and time between successive burns). The literature supporting prescribed burning needs greater managerial relevance that can be obtained by directly addressing spatial scale, temporal scale, and interaction with other disturbances, including drought and grazing.

Although rangeland management professionals generally support using fire in rangeland ecosystems, a long history of exclusion, uncertainty about the effects of fire, increased wildland-urban interface, socio-economics, and natural resource policy are formidable barriers to reintroducing fire. Moreover, the effects of prescribed fire are commonly described as unpredictable and uncertainty about fire effects commonly forces managers to look for more predictable options for land management. Rangeland ecosystems evolved under specific fire regimes rather than in response to individual fires, which requires that management plans include more comprehensive fire planning that addresses both short-term and long-term outcomes.

Ecosystem responses to fire is strongly influence by temporal scale and must be carefully considered in conservation planning. Much of the prescribed fire literature does not provide sufficient ecosystem information to support predictive inferences beyond a site specific description of gross treatment effects (Kral et al. 2015). The majority of studies (83%) reported non-descript treatments as simply burned or non-burned and 85% failed to provide sufficient information to adequately characterize fire treatments. In contrast, livestock grazing treatments descriptions are often highly detailed, including the stocking

rate, kind and class of animal, season and duration of use and ecosystem type.

Consequently, grazing management practices can be conducted with more precision. To have predictive power with prescribed fire, it is imperative that enough data is reported to adequately characterize each fire including: fuel model (type and load), percent fuel moisture (live and dead), relative humidity, air temperature, slope, wind speed and direction, and fire date. Many of these variables can be used to model fire behavior and estimate intensity. However, measuring fire intensity is preferred. The lack of sufficient descriptive information on treatments is a major limitation to a generic approach to prescribed fire recommendations.

The complex interaction of scientific knowledge, social concerns, and variable policies across regions are major limitations to successful restoration of fire regimes on rangeland. Smoke characteristics and smoke management, effects of fire on wildlife and insects, and fire behavior and fire management are ecosystem specific and highly variable across the landscape. A lack of ecosystem-specific data on these topics and high spatial variability in fire effects make it difficult to inform prescribed burning practices at a regional or national level.

Acknowledgments

This paper was written with funding support by the North Dakota Agricultural Experiment Station and the Oklahoma Agricultural Experiment Station and by grants from the USDA-NRI Managed Ecosystems Program (#02-00777), USDA-AFRI Managed Ecosystems (#2010-85101-20457), Joint Fire Science Program (#201814G905; #11-1-2-19), and National Science Foundation/Experimental Program to Stimulate Competitive Research (#

NSF-1301789, and NSF Dynamics of Coupled Natural and Human Systems (CNH) program (DEB-1413900).

Literature cited

- 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 grass. *Journal of Range Management* 29:306-308.
- Allred, B. W., J. D. Scasta, T. J. Hovick, S. D. Fuhlendorf, and R. G. Hamilton. 2014. Spatial heterogeneity stabilizes livestock productivity in a changing climate. *Agriculture Ecosystems & Environment* 193:37-41.
- Anderson, J. E., and R. S. Inouye. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. *Ecological Monographs* 71:531-556.
- Anderson, K. L., E. F. Smith, and C. E. Owensby. 1970. Burning bluestem range. *Journal of Range Management* 23:81-&.
- Anderson, R. H., S. D. Fuhlendorf, and D. M. Engle. 2006. Soil nitrogen availability in tallgrass prairie under the fire-grazing interaction. *Rangeland Ecology & Management* 59:625-631.
- Ansley, R. J., T. W. Boutton, and J. O. Skjemstad. 2006. Soil organic carbon and black carbon storage and dynamics under different fire regimes in temperate mixed-grass savanna. *Global Biogeochemical Cycles* 20:11.

- Ansley, R. J., W. E. Pinchak, and D. L. Jones. 2008. Mesquite, Tobosagrass, and Common Broomweed Responses to Fire Season and Intensity. *Rangeland Ecology & Management* 61:588-597.
- Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. In: M. Vavra, W.A. Laycock, and R.D. Piper [EDS.]. Ecological implications of livestock herbivory in the west. Denver, CO, USA: Society for Range Management. p.13-68.
- Ashton, K.G., B.M. Engelhardt, and B.S. Branciforte. 2008. Gopher tortoise (*Gopherus polyphemus*) abundance and distribution after prescribed fire reintroduction to Florida scrub and sandhill at Archbold Biological Station. *Journal of Herpetology* 42:523-529.
- Axelrod, D. I. 1985. Rise of the grassland biome, central north-america. *Botanical Review* 51:163-201.
- Balch, J. K., B. A. Bradley, C. M. D'Antonio, and J. Gomez-Dans. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980-2009). *Global Change Biology* 19:173-183.
- Barnard, W., and W. Sabo. 2003. Review of 1999 NEI (Version 2, final) and recommendations for developing the 2002 VISTAS inventory for regional haze modeling (area and point sources). Prepared for VISTAS, Asheville, NC, USA.
- Barney, M. A., and Frischkn.Nc. 1974. Vegetation changes following fire in pinyon-juniper type of west-central utah. *Journal of Range Management* 27:91-96.
- Bates, J. D., and K. W. Davies. 2016. Seasonal burning of juniper woodlands and spatial recovery of herbaceous vegetation. *Forest Ecology and Management* 361:117-130

- Bates, J. D., K. W. Davies, and R. N. Sharp. 2011. Shrub-steppe early succession following juniper cutting and prescribed fire. *Environmental Management* 47:468-481.
- Bates, J. D., R. O'Connor, and K.W. Davies. 2014. Vegetation recovery and fuel reduction after seasonal burning of western juniper. *Fire Ecology* 10:27-48.
- Blair, J. M. 1997. Fire, N availability, and plant response in grasslands: A test of the transient maxima hypothesis. *Ecology* 78:2359-2368.
- Bond, W. J., G. F. Midgley, and F. I. Woodward. 2003. The importance of low atmospheric CO₂ and fire in promoting the spread of grasslands and savannas. *Global Change Biology* 9:973-982.
- Bradley, B. A., R. A. Houghtonw, J. F. Mustard, and S. P. Hamburg. 2006. Invasive grass reduces aboveground carbon stocks in shrublands of the Western US. *Global Change Biology* 12:1815-1822.
- Briske, D. D. [ed.]. 2011. Conservation benefits of rangeland practices: assessment, recommendations, and knowledge gaps. Washington, DC, USA: USDA-NRCS. 429 p.
- Brooks, M. L., C. M. C'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. DiTomaso, R. J. Hobbs, M. Pellant, and D. Pyke. Effects of invasive alien plants on fire regimes. *BioScience* 54:677-688.
- Chambers, J. C., R. F. Miller, D. I. Board, D. A. Pyke, B. A. Roundy, J. B. Grace, E. W. Schupp, and R. J. Tausch. 2014. Resilience and resistance of sagebrush ecosystems: implications for state and transition models and management treatments. *Range Ecology and Management* 67:440-454.

- Chi, K., and B. Molano-Flores. 2015. Degradation of habitat disrupts plant-pollinator interactions for rare self-compatible plant. *Plant Ecology* 216:1275-1283.
- Coppedge, B. R., D. M. Engle, R. E. Masters, and M. S. Gregory. 2001. Avian response to landscape change in fragmented southern Great Plains grasslands. *Ecological Applications* 11:47-59.
- Davies, K. W., and A. M. Nafus. 2012. Exotic annual grass invasion alters fuel amounts, continuity and moisture content. *International Journal of Wildland Fire* 22:353-358.
- Davies, K. W., A. M. Nafus, and D. D. Johnson. 2013. Are early summer wildfires an opportunity to revegetate exotic annual grass-invaded plant communities? *Rangeland Ecology and Management* 66:234-240.
- Davies, K. W., T. J. Svejcar, and J. D. Bates. 2009. Interaction of historical and nonhistorical disturbances maintains native plant communities. *Ecological Applications* 19:1536-1545.
- Davies, K. W., J. D. Bates, and R. F. Miller. 2007. Short-term effects of burning Wyoming big sagebrush steppe in southeast Oregon. *Rangeland Ecology & Management* 60:515-522.
- DeBano, L. F. 2000. The role of fire and soil heating on water repellency in wildland environments: a review. *Journal of Hydrology* 231:195-206.
- DeBano, L.F., R.M. Rice, and C.E. Conrad. 1979. Soil heating in chaparral fires: effects on soil properties, plant nutrients, erosion, and runoff. USDA Forest Service Research Paper PSW-145, 21p.

- Dockery, D. W., C. A. Pope, X. P. Xu, J. D. Spengler, J. H. Ware, M. E. Fay, B. G. Ferris, and F. E. Speizer. 1993. An association between air-pollution and mortality in 6 united-states cities. *New England Journal of Medicine* 329:1753-1759.
- Engle, D. M., and T. G. Bidwell. 2001. The response of central North American prairies to seasonal fire. *Journal of Range Management* 54:2-10.
- Farzan, S., D. J. N. Young, A.G. Dedrick, M. Hamilton, E.C. Porse, P. S. Coates, and G. Sampson. 2015. Western juniper management: assessing strategies for improving greater sage-grouse habitat and rangeland productivity. *Environmental Management* 56:675-683.
- Ferguson, S. A., R. L. Collins, J. Ruthford, and M. Fukuda. 2003. Vertical distribution of nighttime smoke following a wildland biomass fire in boreal Alaska. *Journal of Geophysical Research-Atmospheres* 108:12.
- Fischer, J., and D. B. Lindenmayer. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* 16:265-280.
- Fuhlendorf, S. D., and D. M. Engle. 2001. Restoring heterogeneity on rangelands: Ecosystem management based on evolutionary grazing patterns. *Bioscience* 51:625-632.
- Fuhlendorf, S. D., and D. M. Engle. 2004. Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. *Journal of Applied Ecology* 41:604-614.
- Fuhlendorf, S. D., D. M. Engle, R. D. Elmore, R. F. Limb and T.G. Bidwell. 2012. Conservation of pattern and process: developing an alternative paradigm of rangeland management. *Rangeland Ecology and Management* 65:579-589.

- Fuhlendorf, S. D., W. C. Harrell, D. M. Engle, R. G. Hamilton, C. A. Davis, and D. M. Leslie. 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. *Ecological Applications* 16:1706-1716.
- Glitzenstein, J. S., D. R. Streng, G. L. Achtemeier, L. P. Naeher, and D. D. Wade. 2006. Fuels and fire behavior in chipped and unchipped plots: Implications for land management near the wildland/urban interface. *Forest Ecology and Management* 236:18-29.
- Grelen, H. E., and E. A. Epps. 1967. Season of burning affects herbage quality and yield on pine-bluestem range. *Journal of Range Management* 20:31-&.
- Haskins, K. E., and C. A. Gehring. 2004. Long-term effects of burning slash on plant communities and arbuscular mycorrhizae in a semi-arid woodland. *Journal of Applied Ecology* 41:379-388.
- Hobbs, N. T., D. S. Schimel, C. E. Owensby, and D. S. Ojima. 1991. Fire and grazing in the tallgrass prairie - contingent effects on nitrogen budgets. *Ecology* 72:1374-1382.
- Hobbs, R. J., E. Higgs, and J. A. Harris. 2009. Novel ecosystems: implications for conservation and restoration. *Trends in Ecology & Evolution* 24:599-605.
- House, J. I., S. Archer, D. D. Breshears, and R. J. Scholes. 2003. Conundrums in mixed woody-herbaceous plant systems. *Journal of Biogeography* 30:1763-1777.
- Howe, H. F. 1994. Response of early-flowering and late-flowering plants to fire season in experimental prairies. *Ecological Applications* 4:121-133.
- Howe, H. F. 1995. Succession and fire season in experimental prairie plantings. *Ecology* 76:1917-1925.

- Jones, R. J. C. Chambers, D. W. Johnson, R. R. Blank. and D. I. Board. 2015. Effect of repeated burning on plant and soil carbon and nitrogen in cheatgrass (*Bromus tectorum*) dominate ecosystems. *Plant and Soil* 386:47-64.
- Johnson, D. D., and R. F. Miller. 2006. Structure and development of expanding western juniper woodlands as influenced by two topographic variables. *Forest Ecology and Management* 229:7-15.
- Keane, R. E., J. K. Agee, P. Fule, J. E. Keeley, C. Key, S. G. Kitchen, R. Miller, and L. A. Schulte. 2008. Ecological effects of large fires on US landscapes: benefit or catastrophe? *International Journal of Wildland Fire* 17:696-712.
- Keeley, J. E., and P. W. Rundel. 2005. Fire and the Miocene expansion of C-4 grasslands. *Ecology Letters* 8:683-690.
- Kirk, W.G., E.M. Hodges, F.M. Peacock, L.L. Yarlett, and F.G. Martin. 1976. Production of cow-calf herds: effect of burning native and supplemental feeding. *Journal of Range Management* 27:136-139.
- Korfmacher, J. L., J. C. Chambers, R. J. Tausch, B. A. Roundy, S. E. Meyer, and S. Kitchen. 2003. Technical note: A technique for conducting small-plot burn treatments. *Journal of Range Management* 56:251 - 254.
- Kral, K. A., R. F. Limb, T. J. Hovick, D. A. McGranahan, A. L. Field and. P. L. O'Brien. 2015. Simulating grassland prescribed fires using experimental approaches. *Fire Ecology* 11:34-44.

- Limb, R. F., D. M. Engle, A. L. Alford, and E. C. Hellgren. 2010. Tallgrass prairie plant community dynamics along a canopy cover gradient of eastern redcedar (*Juniperus virginiana* L.). *Rangeland Ecology & Management* 63:638-644.
- Limb, R. F., D. M. Engle, A. L. Alford, and E. C. Hellgren. 2014. Plant community response following removal of *Juniperus virginiana* from tallgrass prairie: testing for restoration limitations. *Rangeland Ecology & Management* 67:397-405.
- Limb, R. F., S. D. Fuhlendorf, D. M. Engle, and J. D. Kerby. 2011a. Growing-Season Disturbance in Tallgrass Prairie: Evaluating Fire and Grazing on *Schizachyrium scoparium*. *Rangeland Ecology & Management* 64:28-36.
- Limb, R. F., S. D. Fuhlendorf, D. M. Engle, J. R. Weir, R. D. Elmore, and T. G. Bidwell. 2011b. Pyric-Herbivory and Cattle Performance in Grassland Ecosystems. *Rangeland Ecology & Management* 64:659-663.
- Macneil, M. D., M. R. Haferkamp, L. T. Vermeire, and J. M. Muscha. 2008. Prescribed fire and grazing effects on carbon dynamics in a northern mixed-grass prairie. *Agriculture Ecosystems & Environment* 127:66-72.
- Martin, R.E., R.W. Cooper, A.B. Crow, J.A. Cuming, and C.B. Phillips. 1977. Report of task force on prescribed burning. *Journal of Forestry* 75:297-301.
- Mckenzie, D., S. M. O'Neill, N. K. Larkin, and R. A. Norheim. 2006. Integrating models to predict regional haze from wildland fire. *Ecological Modelling* 199:278-288.
- Mcnaughton, S. J. 1984. Grazing lawns - animals in herds, plant form, and coevolution. *American Naturalist* 124:863-886.

- Mcnew, L. B., V. L. Winder, J. C. Pitman, and B. K. Sandercock. 2015. Alternative Rangeland Management Strategies and the Nesting Ecology of Greater Prairie-Chickens. *Rangeland Ecology & Management* 68:298-304.
- Miller, M. E., M. A. Bowker, R. L. Reynolds, and H. L. Goldstein. 2012. Post-fire land treatments and wind erosion -- lessons from the Milford Flat Fire, UT, USA. *Aeolian Research* 7:29-44.
- Miller, R. F., J. C. Chambers, D. Pyke, F. B. Pierson, and C. J. Williams. 2013. A review of fire effects on vegetation and soils in the Great Basin Region: Response and ecological site characteristics. USDA Forest Service, Rocky Mountain Research Station RMRS-GTR-308.
- Miller, R. F., S. T. Knick, D. A. Pyke, C. W. Meinke, S. E. Hanser, M. J. Wisdom, and A. L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. Pages 146-184. *In*: S. T. Knick and J. W. Connelly, editors. Greater Sage-Grouse: ecology and conservation of landscape species and its habitats. University California Press, Berkley CA.
- Morgan, P., C. C. Hardy, T. W. Swetnam, M. G. Rollins, and D. G. Long. 2001. Mapping fire regimes across time and space: Understanding coarse and fine-scale fire patterns. *International Journal of Wildland Fire* 10:329-342.
- Munson, S. M., A. L. Long, C. Decker, K. A. Johnson, K. Walsh, and M. E. Miller. 2015. Repeated landscape-scale treatments following fire suppress a non-native annual grass and promote recovery of native perennial vegetation. *Biological Invasions* 17:1915-1926.

- Neary, D. G., C. C. Klopatek, L. F. Debano, and P. F. Folliott. 1999. Fire effects on belowground sustainability: a review and synthesis. *Forest Ecology and Management* 122:51-71.
- Ojima, D. S., D. S. Schimel, W. J. Parton, and C. E. Owensby. 1994. Long-term and short-term effects of fire on nitrogen cycling in tallgrass prairie. *Biogeochemistry* 24:67-84.
- Owensby, C. E., and K. L. Anderson. 1967. Yield responses to time of burning in Kansas flint hills. *Journal of Range Management* 20:12-&.
- Owensby, C. E., and J. B. Wyrill. 1973. Effects of range burning on Kansas flint hills soil. *Journal of Range Management* 26:185-188.
- Pedersen, E. K., J. W. Connelly, J. R. Hendrickson, and W. E. Grant. 2003. Effect of sheep grazing and fire on sage grouse populations in southeastern Idaho. *Ecological Modelling* 165:23-47.
- Prater, M. R., and E. H. Delucia. 2006. Non-native grasses alter evapotranspiration and energy balance in Great Basin sagebrush communities. *Agricultural and Forest Meteorology* 139:154-163.
- Puttock, A., J. A. J. Dungait, C. J. A. Macleod, R. Bol, and R. E. Brazier. 2014. Woody plant encroachment into grasslands leads to accelerated erosion of previously stable organic carbon from dryland soils. *Journal of Geophysical Research-Biogeosciences* 119:2345-2357.
- Rau, B. M., D. W. Johnson, R. R. Blank, and J. C. Chambers. 2009. Soil carbon and nitrogen in a Great Basin pinyon-juniper woodland: influence of vegetation, burning, and time. *Journal of Arid Environments* 73:472-479.

- Rau, B. M., D. W. Johnson, B. R. Blank, A. Luccheis, T. G. Caldwell, and E. W. Schupp. 2011. Transition from sagebrush steppe to annual grass (*Bromus tectorum*): influence on belowground carbon and nitrogen. *Rangeland Ecology & Management* 64:139-147.
- Rau, B. M., R. J. Tausch, A. Reinen, D. W. Johnson, J. C. Chambers, R. R. Blank, and A. Lucchest. 2010. Influence of prescribed fire on ecosystem biomass, carbon, and nitrogen in pinyon juniper woodland. *Rangeland Ecology & Management* 63:197-202.
- Reich, P. B., P. Bakken, D. Carlson, L. E. Frelich, S. K. Friedman, and D. F. Grigal. 2001. Influence of logging, fire, and forest type on biodiversity and productivity in southern boreal forests. *Ecology* 82:2731-2748.
- Reinkensmeyer, D. P., R. F. Miller, R. G. Anthony, and V. E. Marr. 2007. Avian community structure along a mountain big sagebrush successional gradient. *Journal of Wildlife Management* 71:1057-1066.
- Reynolds, H. G., and J. W. Bohning. 1956. Effects of burning on a desert grass-shrub range in southern Arizona. *Ecology* 37:769-777.
- Robbins, M. B., A. T. Peterson, and M. A. Ortega-Huerta. 2001. Major negative impacts of early intensive cattle stocking on tallgrass prairies: the case of the greater prairie-chicken (*Tympanuchus cupido*). *North American Birds* 56:239-244.
- Rowland, M. M., M. J. Wisdom, L. H. Suring, and C. W. Meinke. 2006. Greater sage-grouse as an umbrella species for sagebrush-associated vertebrates. *Biological Conservation* 129:323-335.

Sankey, J. B., M. J. Germino, S. G. Benner, N. F. Glenn, and A. N. Hoover. 2012.

Transport of biologically important nutrients by wind in an eroding cold desert. *Aeolian Research* 7:17-27.

Smith, E. F., and C. E. Owensby. 1978. Intensive-early stocking and season-long stocking of Kansas flint-hills range. *Journal of Range Management* 31:14-17.

The Nature Conservancy. 2009. The LANDFIRE Project: Supporting fire and land management across the nation. Available at www.landfire.gov (viewed October 1, 2015).

Twidwell, D., B. W. Allred, and S. D. Fuhlendorf. 2013. National-scale assessment of ecological content in the world's largest land management framework. *Ecosphere* 4:27.

Twidwell, D., C. L. Wonkka, M. T. Sindelar, and J. R. Weir. 2015. First approximations of prescribed fire risks relative to other management techniques used on private lands. *Plos One* 10:e0140410.

USDA NRCS 2015a National Conservation Practice Standards

(<http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/cp/ncps/>, 1 October 2015).

Wangler, M.J., Minnich, R.A., 1996. Fire and succession in pinyon-juniper woodlands of the San Bernardino Mountains. California. *Magronño* 43:493-514.

Table 1. Number of papers published reporting research on topics related to prescribed fires conducted within rangeland types globally based on a Web of Science search. The wildland-urban interface is listed as WUI.

Table 2. Summary of number of studies for all years (1967-2014) and plant group (Total Herbs, Perennial Grasses, etc.) across global rangeland ecosystem type (Grassland/Desert, Savanna and Shrubland). Numbers of studies indicating negative (-), positive (+), and no change (=) in response to fire. Plant group responses (-, + or =) can be production, cover, abundance, etc.

Figure 1. Number of papers published per year from a total of 846 papers published on rangeland fire between 1967 and 2014. See text for an explanation of papers selected.

Figure

Table

Topic area	Not specific	Desert	Forest	Grassland	Grassland/ Forest	Grassland/ Shrub	Savanna	Shrub	WUI	Total
Soil	4	1	16	29	1	5	18	17	1	92
Water			3	3			2	1		9
Plants	9	2	61	125	5	7	82	45	1	338
Wildlife	4	1	13	25	1	5	12	4		65
Arthropods	4		2	14			13	2		35
Soil invertebrates		1		3	1			1		6
Livestock	1			8	1		1	2		13
Fire Management	37		32	9			10	9	16	113
Fire behavior	29		37	20	4	1	14	20	9	133
Smoke management	4		6							10
Econ/Policy/Social	36		15	6			3	4	20	84
Air quality	14		8	4	1		6	1		34
Fire history	4		3	5	4	1	7	6		31
Human health	7		6					1		18
Other	13		7	8	1		5	1		35
Total	166	5	209	259	19	19	173	114	52	

Table

Ecosystem	Plant group	1 year post fire			2-3 years post fire			≥ 4 years post fire		
		(-)	(+)	(=)	(-)	(+)	(=)	(-)	(+)	(=)
Grassland/Desert	Total herbaceous	7	4	3	2	2	3	0	1	1
	Annual grass	1	2	1	0	2	0	1	0	0
	Perennial grass	11	5	7	6	3	22	0	9	6
	Forbs	5	2	5	2	3	8	1	6	2
	Resprouting shrubs	3	2	0	1	0	0	2	1	2
	Non-resprouting shrubs	2	1	0	0	0	1	0	0	0
	Trees	6	0	0	4	1	0	8	1	2
Savanna	Total herbaceous	0	0	0	0	0	0	0	0	0
	Annual grass	0	0	0	0	0	0	0	0	0
	Perennial grass	1	0	4	0	0	1	0	0	10
	Forbs	0	0	0	4	0	0	1	0	0
	Resprouting shrubs	0	0	0	0	0	0	1	0	0
	Non-resprouting shrubs	0	0	0	0	0	0	2	0	0
	Trees	0	0	0	0	0	0	0	2	0
Shrubland	Total herbaceous	0	0	1	0	1	0	0	0	0
	Annual grass	0	2	3	0	3	1	0	1	3
	Perennial grass	10	1	5	1	2	8	0	0	2
	Forbs	1	5	1	0	2	0	1	0	0
	Resprouting shrubs	1	0	0	0	0	0	4	1	1
	Non-resprouting shrubs	9	0	0	5	0	0	0	0	0
	Trees	3	0	0	2	0	0	1	1	0