Grazing and Biodiversity

By Christopher D. Allison and Louis C. Bender[1]

aces.nmsu.edu/pubs • Cooperative Extension Service • CR-686

The College of Agricultural, Consumer and Environmental Sciences is an engine for economic and community development in New Mexico, improving the lives of New Mexicans through academic, research, and extension programs.

All About Discovery!™
New Mexico State University
aces.nmsu.edu

Figure 1. **Common use grazing by cattle and sheep on Chihuahuan desert grassland. Biodiversity can be encouraged with dietary and area selectivity differences between cattle and sheep.**

Maintaining biological diversity of rangelands is an important and appropriate land management objective. The Linebery Policy Center for Natural Resource Management supports research, education, and development of management approaches that address and facilitate biological diversity (i.e., biodiversity) in rangeland ecosystems. This requires an understanding of what biodiversity is and the processes that contribute to biodiversity and its maintenance on rangelands, and development of strategies that can maintain all relevant ecological processes that support healthy rangelands and all the products and attributes of healthy rangelands, including biodiversity.

WHAT IS BIODIVERSITY?

The Society for Range Management (2002) defines biodiversity as "the variety and variability of the world's organisms, the ecological complexes in which they occur and the processes and life support services they mediate. [Biodiversity] is a complex phenomenon influenced by the kinds of organisms (i.e. plants, animals, microbes), their genetic variation, spatial distribution (e.g. ecosystem, landscape, regional, global), structural organization

[1]Respectively, Rangeland Scientist, Linebery Policy Center for Natural Resource Management; and Senior Research Scientist (Natural Resources), Department of Extension Animal Sciences and Natural Resources, New Mexico State University.

(e.g. vertical stratification) and functional role (e.g. nutrient and water cycling, energy flow)." Biodiversity is described "in many different ways including species richness, evenness, community processes and organization structure. No one expression is intrinsically superior to another. No single expression of [biodiversity] is sufficient nor is one scale of consideration paramount" (Society for Range Management, 2002). Perhaps most importantly, biodiversity varies in time and space and is influenced by many natural processes and management activities (Fuhlendorf and Engle, 2001, 2004; Gillson, 2004; Derner et al., 2009).

"There is no simple relationship between [biodiversity] and properties of ecological systems such as stability for all rangeland sites. Loss of biological diversity, however, may reduce future land use options and the ability to maintain sustainable systems" (Society for Range Management, 2002) and resilience of rangelands to resist change or recover from extreme disturbance (Walker, 1989; Fuhlendorf and Engle, 2004). Consequently, biodiversity is of fundamental importance to the maintenance of ecological function and thereby "directly provides for human wants and needs" (Society for Range Management, 2002).

"The Society for Range Management recognizes the value of [biodiversity] to ecosystem structure and function and promotes the inclusion of biological diversity in the array of facts to be considered in management of rangelands" (Society for Range Management, 2002). However, "maximizing [biodiversity] is not always possible or desirable [or necessary] at all levels of biological or spatial organization" (Society for Range Management, 2002). Biodiversity is also hierarchical, a result of differing ecological processes dominating at the site, patch, local, landscape, and geographic scales, many of which are poorly understood. Management for biodiversity usually focuses on the landscape level since this can retain the large-scale ecological processes (Ryan, 1990; Curtin et al., 2002; Gillson, 2004; Fuhlendorf and Engle, 2004) that drove the development of rangelands at all finer spatial scales. This approach recognizes that landscapes are not static, but rather dynamic systems that exhibit a mosaic of ecological states shaped by various biotic and abiotic factors, most fundamentally topography, soils, fire, and the grazing, trampling, and excreta of herbivores (Wiens, 1985; Ryan, 1990; Pieper, 1994; Fuhlendorf and Engle, 2001; Metera et al., 2010; Fynn et al., 2016). While climate sets the limits to grassland formation at geographic scales (i.e., continental), it combines with these other factors to shape grasslands at more localized scales from the landscape to the patch (Stoddart et al., 1975; Ryan, 1990). The result is a dynamic mosaic that results from and reflects disturbance associated with the above and many other factors mediated by a variety of ecological processes (Hart, 1990; Ryan, 1990; Gillson, 2004; Fuhlendorf and Engle, 2004; Derner et al., 2009).

GRAZING AND VEGETATION COMPOSITION, STRUCTURE, AND FUNCTION

Herbivores impact rangeland vegetation through defoliation, physical trampling and other soil disturbances, and excretion (Wiens, 1985; Pieper, 1994; Derner et al., 2009; Metera et al., 2010; Fynn et al., 2016). All herbivores are selective grazers, and show three types of selectivity: area selectivity, species selectivity, and, as illustrated in Figure 1, intra-species selectivity (Allison, 2004). Through the above, "managed livestock grazing can have 4 general impacts on vegetation: (1) alter the composition of the plant community, (2) increase the productivity of selected species, (3) increase the nutritive quality of the forage, and (4) increase the diversity of the habitat by altering its [physical] structure" (Vavra, 2005, p. 128). Intensity of defoliation varies among individual plants as a function of the unique combination of plants and herbivores present at any given time (Van Soest, 1996). As a result of non-uniform use of plants (site scale) and patches (patch, local, and landscape scales), grazing can increase the diversity of habitats for both plants and animals (Milchunas et al., 1998; Fuhlendorf and Engle, 2001; Derner et al., 2009; Rosenthal et al., 2012; Borer et al., 2014; Fynn et al., 2016). Such patchiness is characteristic of rangelands, and spatially variable disturbance is critical to the function of rangelands (Fuhlendorf and Engle, 2004).

Maintaining or increasing the heterogeneity of vegetation is essential for biodiversity, ecosystem goods and services, and long-term sustainability of ecosystems and wildlife populations (Laycock, 1994; Fuhlendorf and Engle, 2001; Derner et al., 2009; Metera et al., 2010; Rosenthal et al., 2012). Vegetation heterogeneity refers to variability in the structure and composition of plant communities over space and time (Fuhlendorf and Engle, 2001; Derner et al., 2009), which results from differences in physical characteristics (climate, soils, topography) and disturbance processes (grazing, fire, burrowing mammals, ant hills, etc.) (Ryan, 1990; Pieper, 1994; Fuhlendorf and Engle, 2001, 2004; Borer et al., 2014; Lwiwski et al., 2015). Grasslands with more vegetation heterogeneity support a greater number of plant and animal species because they contain additional structural complexity and/or diverse plant communities, which provide added spatial and temporal niches (Toombs et al., 2010; Fynn et al., 2016). Thus, management strategies that maintain or increase vegetation heterogeneity at appropriate scales can increase habitat diversity for wildlife (Severson and Urness, 1994; Laycock, 1994; Fuhlendorf and Engle, 2001; Vavra, 2005; Derner et

Figure 2. **Lesser prairie chicken (left; *Tympanuchus pallidicinctus*) and dunes sagebrush lizard (right; *Sceloporus arenicolus*) are two species that often occupy common rangeland areas, but require vastly different habitats. Grazing is a tool that can create and maintain vegetative cover and structure needs of these two animal species.**
(Image credit: left, Larry Lamsa, flickr.com; right, Mark Watson, flickr.com.)

al., 2009; Fynn et al., 2016). In contrast, management practices that emphasize more homogeneous use of plant communities can result in decreased vegetation heterogeneity and wildlife diversity (Fuhlendorf and Engle, 2001, 2004). Clements (1905) may have been the first to point out that diversity, in terms of number of species, is not linearly correlated with stage of succession. Heterogeneity and biodiversity tend to be greatest at intermediate levels of succession in grasslands and lower in early successional and climax grasslands (Clements, 1905; Huston, 1979; Ryan, 1990), although this is not always the case (Milchunas et al., 1998).

The above relationship between diversity and stage of succession is often seen in grasslands (Ryan, 1990; Milchunas et al., 1998). Grazing, through influences on vegetation heterogeneity, can maintain a variety of successional or ecological states in grasslands (Pieper, 1994; Fuhlendorf and Engle, 2001; Derner et al., 2009; Metera et al., 2010; Rosenthal et al., 2012; Lwiwski et al., 2015). This increases both heterogeneity and biodiversity locally by limiting the proportion of climax communities in favor of increased and varied intermediate ecological states (Fuhlendorf and Engle, 2001; Rosenthal et al., 2012). Grazing can also increase heterogeneity and biodiversity at the landscape scale because non-uniform use creates a variety of patches in differing ecological states (Fuhlendorf and Engle, 2001; Lwiwski et al., 2015).

The effects of grazing on plant communities and biodiversity thus reflect some basic ecological principles (Laycock, 1994). These include: (1) plants are distributed in patches, and the status and distribution of patches depend upon the processes, such as grazing, that create them (Watt, 1947); (2) grazing can increase heterogeneity of plant communities by reducing dominance by a few species, which are replaced by numerous secondary species (Laycock, 1994; Pieper, 1994; Fuhlendorf and Engle, 2001; Rosenthal et al., 2012; Lwiwski et al., 2015); and (3) habitat diversity (patchiness) and resultant ecotones or edges are important as wildlife habitat for many species (Leopold, 1933), but not all (Reese and Ratti, 1988). Non-uniform use of rangelands by livestock contributes to these effects (Fuhlendorf and Engle, 2001). For example, many areas, such as uplands, receive much less use than other areas of rangelands, such as lowlands (Stoddart et al., 1975; Lwiwski et al., 2015); consequently, less vegetatively diverse but still important patches of later successional or climax communities are retained on the landscape in addition to intermediate ecological states. While edge is important to a large number of wildlife species, a number of species require larger patches of more homogenous vegetation (Reese and Ratti, 1988; Ribic et al., 2009). This is particularly true for species that require late successional states, and these states need to be maintained in adequate sizes on the landscape mosaic (Ribic et al., 2009); although not well documented, this is likely around 200 ha (NRCS, 1999). However, even species that require later successional patches often require diversity within home ranges. For example, the lesser prairie chicken (*Tympanuchus pallidicinctus*; Figure 2) requires both late successional tall grassland for nesting and early successional short grassland for brood rearing within their 200–600 ha home ranges (Derner et al., 2009). Using livestock to alter vegetation structure for wildlife habitat is thus a valuable tool for land managers to achieve desired objectives for rangelands (Derner et al., 2009).

Because factors that determine rangeland vegetation and its attributes (including species composition and heterogeneity) are hierarchical, and different factors predominate at different scales, biodiversity is hierarchical (e.g., Gillson, 2004). The lowest level is the microsite or individual plant level, where plant species composition and vigor are affected by several factors, including microclimate (e.g., shading, etc.), selective herbivory, and among plant interactions such as competition or facilitation (Noy-Meir, 1981; Pieper, 1994; Van Soest, 1996; Gillson, 2004; Borer et al., 2014; Fynn et al., 2016). The next level is the patch or local scale (such as a pasture), where plant species composition and heterogeneity are determined by characteristics of the site (e.g., soils, slope, moisture, etc.) and disturbances such as fire and differing grazing intensity within patches (Watt, 1947; Wiens, 1985; Pieper, 1994; Fuhlendorf and Engle, 2001; Lwiwski et al., 2015). At the landscape scale, differences in environmental variables like hydrology, topography, local rainfall, and soils combine with historical disturbances such as fire, varying use by herbivores among patches (including non-grazed patches), and differences in herbivore density across the landscape to create a mosaic comprised of many patches in differing successional or ecological states (Watt, 1947; Bormann and Likens, 1981; Fuhlendorf and Engle, 2001, 2004; Gillson, 2004; Lwiwski et al., 2015). This landscape mosaic, comprised of differing patches of differing sizes in differing ecological states, is the primary determinant of biodiversity on rangelands at the scales influenced by management.

GRAZING AND BIODIVERSITY

Grazing and system biodiversity

There is no question that historical, largely unregulated grazing often caused serious damage to rangelands (e.g., Milchunas, 2006). Generalizing these largely historic effects to contemporary practices is unwarranted, however. To the contrary, much work has clearly shown that proper livestock grazing can maintain biodiversity and other rangeland functions, and in some cases is necessary to maintain and enhance these processes and attributes (e.g., Severson and Urness, 1994; Laycock, 1994; Milchunas et al., 1998; Fuhlendorf and Engle, 2001; Metera et al., 2010; Borer et al., 2014). Among likely future land-use options on rangelands, ranching is the most viable means of conserving the functionality of the large-scale ecological processes that formed rangelands (Curtin et al., 2002). These are also the very processes (e.g., fire, herbivory, etc.) that create biodiversity on rangelands.

Grazing does not necessarily result in negative impacts on all or even a majority of wildlife species. For example, Kie and Loft (1990) predicted likely wildlife responses to grazing-induced changes from tall to short herb structure in annual grasslands in California. They noted that decreased vegetation height would likely benefit 52 species, harm 29, and not affect 171. They found similar likely responses to grazing wet meadows. Milchunas et al. (1998) found that light to moderate grazing enhanced conditions for some species and degraded conditions for others, limited invasion of grasslands by noxious weeds, and provided early successional habitat for some species of concern (e.g., mountain plover [*Charadrius montanus*]). Many studies have focused on diversity of plant, bird, and animal species in adjacent grazed and ungrazed riparian areas (e.g., Medin and Clary, 1990; Krueger, 1985; Schulz and Leininger, 1990; Knopf et al., 1988; Kauffman and Krueger, 1984). In general, moderate grazing usually had little to no effect on bird populations and often had positive effects on vegetation and bird biomass, density, and diversity. Livestock can also be used to enhance conservation of species requiring certain successional states (see below; Severson and Urness, 1994; Milchunas et al., 1998; Derner et al., 2009; Fynn et al., 2016). Metera et al. (2010) found that grazing created favorable conditions for the formation of habitat structure preferred by many endangered birds, small mammals, and invertebrates, positively impacting biodiversity of grasslands.

Work with specific guilds has found similar results. Elwell et al. (2016) found that livestock grazing did not negatively affect pollinator or flowering plant abundance, richness, diversity, or community composition. Derner et al. (2009) noted that livestock could be used as “ecological engineers” to enhance habitat availability and quality for grassland birds. Livestock grazing can also enhance plant community biodiversity, even in arid environments; for example, Naveh and Whittaker (1979) found greater plant diversity for several rangeland sites on moderately grazed areas as compared to ungrazed or heavily grazed areas in Israel. Rosenthal et al. (2012) found that low-intensity grazing was a valuable tool to maintain and restore plant diversity, and livestock were also an effective means of transporting plant propagules to maintain connectivity between isolated plant populations. Similarly, Török et al. (2014) found that Hungarian Grey cattle were able to create and maintain grassland heterogeneity, suppress noxious species, and create a mosaic of vegetation structures of short and tall species, maintaining high species richness in the landscape via extensive grazing. Consequently, extensively managed pastures are of critical importance for sustaining grassland biodiversity across Europe (Török et al., 2014). This also highlights the importance of matching livestock species to grazing goals (Rook et al., 2004; Metera et al., 2010).

Grazing can also mitigate human impacts on rangelands. Ellenberg (1988) proposed that livestock grazing may have to be increased to enhance diversity of the herbaceous plant communities of central Europe, which are experiencing dramatic nitrogen and sulfur enrichment due to air pollution. This results in dominance by a few species that thrive under high nutrient inputs.

Grazing and featured species

Numerous examples exist of grazing benefiting a broad array of wildlife species. While we highlight several examples because this issue is frequently ignored in the antigrazing literature, we urge readers to reject the static management paradigm that featured species management invariably leads to. Rather than focusing on needs of individual species, the focus should be on maintaining the ecological processes that developed the rangelands of North America, including herbivory and fire (Ryan, 1990; Fuhlendorf and Engle, 2001, 2004). Focusing on featured species rather than ecological processes can lead to unnecessary conflicts when multiple rare species occupy the same ecological sites but require radically different structures or ecological states (see below). Moreover, trying to benefit a featured species by maintaining a site in the same ecological state in perpetuity is a futile battle against nature (Hart, 2001; Fuhlendorf and Engle, 2004; Derner et al., 2009). Rather, management should strive to maintain the ecological processes that create varying ecological states across pastures and landscapes, and embrace the variability in disturbance and other processes that change existing states to other ecological states (Ryan, 1990; Fuhlendorf and Engle, 2001, 2004; Gillson, 2004). These same processes, if conserved, would concurrently be recreating these altered states at other sites on the landscape, conserving all habitat features (e.g., Bormann and Likens, 1981; Fuhlendorf and Engle, 2004; Gillson, 2004).

As noted above, changes in vegetation structure can affect habitat quality for wildlife (Kie and Loft, 1990). Determining the true cause-and-effect relationships driving changes in structure can be complicated, however, particularly when multiple factors are potentially at work. For example, expansion of tall shrubs such as velvet mesquite (*Prosopis velutina*) into desert grasslands along the Arizona-Sonora border has been associated with declines in habitat quality for the endangered masked bobwhite quail (*Colinus virginianus* ssp. *ridgwayi*) (King et al., 1996; Zornes and Bishop, 2009; USFWS, 2014). Expansion of velvet mesquite in the Southwest is frequently attributed to livestock grazing (Mensah, 2010).

However, the increase in shrub cover has been a long-term process (Van Devender, 1990; Pieper, 1994), and the relative impact of livestock versus climate and other causes of change is unknown (Bahre, 1991; Pieper, 1994). Moreover, population and habitat viability assessments for the masked bobwhite noted that livestock grazing could be proactively used to decrease dominance of the invasive exotic Lehmann lovegrass (*Eragrostis lehmanniana*), which would enhance habitat quality by favoring forbs and other desert legumes, increasing vegetation diversity and forage resources (King et al., 1996). However, the U.S. Fish and Wildlife Service opted to use prescribed fire to accomplish these goals, even though fire has been shown to actually favor Lehmann lovegrass (Sayre, 2005). Subsequently, the masked bobwhite has declined to near absence in the wild on the Buenos Aires National Wildlife Refuge despite the elimination of livestock grazing (USFWS, 2014), and other rangeland degradations attributed to grazing, such as large arroyos, have similarly not recovered (Sayre, 2005).

The desert tortoise (*Gopherus agassizii* and *Gopherus morafkai*) is another herbivore declining in Southwestern deserts (West, 1993). While this decline is often attributed to livestock grazing (Berry, 1978), livestock use declined well before tortoise densities, and studies have shown that the tortoise prefers grazing on introduced annuals (Bostick, 1990). Likely, other influences, such as increased predator densities, increased off-road vehicular traffic, diseases, and direct human removals, were as or more responsible for declines than were vegetation changes attributed to grazing and cattle trampling of tortoises and dens (Huxtable, 1992; West, 1993).

Some rare species are clearly favored by grazing. The mountain plover nests only in relatively heavily grazed shortgrass steppe (Graul, 1973, 1975; Ryder, 1980; Leachman and Osmundson, 1990; Milchunas et al., 1998; Derner et al., 2009), likely an adaptation to the historical influences of bison (*Bison bison*) grazing that only cattle can now replace. However, species of concern seldom occur in isolation, and conflicts can arise when multiple species with differing requirements occur on the same site. An example is the lesser prairie chicken and the dunes sagebrush lizard (*Sceloporus arenicolus*; Figure 2). The lesser prairie chicken requires tall grasses for nesting cover while the dunes sagebrush lizard prefers bare ground and active dunes (USFWS, 2008). Grazing creates these "heavy" and "light" utilization areas that are beneficial to these respective species (Derner et al., 2009), but not necessarily in the same patch. Selective management of a site for one of these species would be detrimental to the other. Alternatively, a landscape mosaic would provide suitable states for each species.

Livestock grazing can also facilitate foraging by other herbivores by removing coarse and senescent growth of dense or cured grasses through a process called facilitative herbivory (Vesey-Fitzgerald, 1960; Fynn et al., 2016). Forage quantity and quality are often inversely related (Hobbs and Swift, 1985; Noy-Meir, 1981), and wildlife are more selective feeders that require higher-quality foods than do livestock. By removing grasses that are high in crude fiber, grazing enhances foraging efficiency for wildlife like mule deer (*Odocoileus hemionus*) that require foods such as forbs and nutritious regrowth of grasses, which are higher in cell solubles like sugars, proteins, and fats, and lower in crude fiber (Hanley, 1997; Hoenes and Bender, 2012; Fynn et al., 2016). Dietary needs of wild herbivores result in far more time spent searching for food than seen in cattle, so the facilitative effect of cattle grazing can increase foraging efficiency, diet quality, and overall energy balance of wild herbivores by decreasing search times (Fynn et al., 2016). The facilitative effect of livestock herbivory can be a key component in conserving biodiversity by enhancing foraging opportunities for wild herbivores and increasing grassland heterogeneity (Western and Gichohi, 1993; Severson and Urness, 1994; Fynn et al., 2016).

Some species of concern can benefit from very heavy grazing. While inappropriate use of pasture—both overgrazing and undergrazing—can pose a threat to biodiversity (Fuhlendorf and Engle, 2001; Metera et al., 2010; Milchunas, 2006), there are examples where heavy grazing benefits both biodiversity and ecosystem function. Prairie dogs (*Cynomys* spp.) prefer heavily grazed patches for their colonies ("prairie dog towns"), and the presence of these colonies is critical for the endangered black-footed ferret (*Mustela nigripes*; Hillman et al., 1979; Uresk et al., 1981; Sierra-Corona et al., 2015). Historical overgrazing by livestock contributed significantly to the development of large areas of big sagebrush (*Artemisia tridentata*) shrublands throughout western North America (Laycock, 1967; Hull and Hull, 1974). This habitat type is currently of intense management interest because of the dependence of sage grouse (*Centrocercus urophasianus*) on these shrublands. Ironically, livestock grazing (and fire) are currently considered by some to be detrimental to big sagebrush communities and sage grouse (Shroeder et al., 2006; USFWS, 2015).

Rare plant species can similarly benefit from proper grazing. A species of buttercup (*Ranunculus ophioglossifolius*) in Great Britain can only be sustained where heavy sheep grazing is maintained (Frost, 1981). Sheep grazing was also found to benefit populations of the early spider orchid (*Ophrys sphegodes*) in England, apparently because maintenance of a short turf reduces interspecies competition (Hutchings, 1987). Cattle grazing had been resulting in declines of this species previously, apparently because of less close-cropping of competitors and mechanical hoof damage.

Clearly, grazing as an ecological process is not inherently incompatible with wildlife or biodiversity. While evaluation of species or guild responses is valuable, rangelands and their attributes such as biodiversity fundamentally depend upon conserving the ecological processes that controlled the development of grasslands, including herbivory (Ryan, 1990; Pieper et al., 1994; Fuhlendorf and Engle, 2001, 2004; Curtin et al., 2002; Derner et al., 2009; Metera et al., 2010; Rosenthal et al., 2012; Fynn et al., 2016; Teague et al., 2016). The latter is more conducive than a "charismatic" or "featured" species approach to conserving landscapes since conservation of ecological processes at appropriate scales provides for maintenance of larger tracts of land with habitat or ecosystems capable of supporting suites of sensitive species (West, 1993; Fuhlendorf and Engle, 2001; Gillson, 2004; Derner et al., 2009).

Managing rangelands for biodiversity and multiple uses

Maintaining or creating vegetation heterogeneity at both pasture and landscape scales is the key to healthy rangelands and biodiversity. Any attempt to impose a static management paradigm on rangeland will ultimately result in decreased biodiversity and other products of healthy rangelands because such an approach fights the ecological processes that controlled the development of North American grasslands (Ryan, 1990; Fuhlendorf and Engle, 2001, 2004; Derner et al., 2009). This is true whether the static management is intended to benefit a species of concern by conserving some preferred ecological state, or whether it results from desires for uniform use of rangelands by livestock.

At local (pasture) scales, grazing management can enhance conditions for achieving biodiversity and other goals. If pastures are large, like those associated with Southwestern ranches (Figure 3), management of supplements, minerals, salt, and water can be used to enhance local heterogeneity. If water and minerals are available only at a few, widely scattered points, then gradients of disturbance and consequent successional states will be available across the pasture because of non-uniform use by livestock (Andrew, 1988; Laycock, 1994; Fuhlendorf and Engle, 2001, 2004; Derner et al., 2009). Other local treatments, particularly prescribed burning, can further enhance community and landscape diversity on rangelands (Mueggler, 1984; Fuhlendorf and Engle, 2004; Derner et al., 2009; Metera et al., 2010). To illustrate, Toombs et al. (2010) listed several practices that may be used to enhance vegetation heterogeneity and biodiversity at the pasture level. These include:

- **Water.** "Manipulating the availability of water sources throughout the year depends on a pasture's having more potential water sources than are needed by livestock at any given time, and being sufficiently large that livestock cannot access all portions of the pasture while using only one of the water sources. Alternating the availability of water sources during several years can produce a nonstatic mosaic of heavily grazed and undergrazed areas. This strategy provides additional

Figure 3. **Cattle grazing high-elevation range in Rio Arriba County, NM.**

range benefits by allowing plant communities near water sources to recover when those water sources are not in use" (Toombs et al., 2010, p. 13). "Uneven livestock distribution and forage use is already occurring [in every pasture]: areas close to water receive consistent high use [and fertilization from excreta], and areas furthest from water receive less use" (Toombs et al., 2010, p. 14). While the large size of pastures also helps to maintain within-pasture vegetation heterogeneity, tall- and short-structure areas will remain static over time if water locations are held constant.

- **Supplemental feed.** "Managers can also use supplemental feed sites strategically to create intensively disturbed patches on the order of several hectares within pastures, while reducing livestock use of distant portions of the pasture. This is particularly effective when forage quality is low (e.g., dormant season or plant maturity [or periods of drought]) and can be used to influence the amount of standing dead vegetation within pastures" (Toombs et al., 2010, p. 13).

- **Fire.** "Prescribed fire is another way to create a shifting mosaic of vegetation within a pasture. Burning influences vegetation structure in ways that differ from livestock grazing," provides a significant enhancement in forage quality through fertilization and earlier green-up, and increases digestibility of foods by decreasing crude fiber content (Toombs et al., 2010, p. 13; Bender, 2011). Combined with grazing, fire can produce a multitude of vegetative conditions that otherwise would not occur. In a pasture context, "patch" burning has been suggested as a preferred management practice for implementing prescribed fire treatments. The managerial idea behind patch burning with grazing is to burn different areas each year and allow livestock access to both burned and unburned areas during the next growing season (Toombs et al., 2010; Fuhlendorf and Engle, 2004). Livestock will then selectively graze the burned areas, promoting deferment of the unburned range and resulting in greater spatial heterogeneity of vegetation (Toombs et al., 2010; Smart, 2008).

This illustrates how two of the fundamental ecological processes that shaped the development of rangelands—herbivory and fire—can be used to enhance vegetation heterogeneity at multiple scales to achieve resource goals such as biodiversity. At both local and landscape levels, fire burns with differing intensity across rangelands because of differences in fuel loading, topography, etc. (Whelan, 1995; Fuhlendorf and Engle, 2001, 2004). The result is a mosaic of unburned and burned patches, with burned patches similarly showing differing degrees of impact on vegetation because of varying fire intensity (Whelan, 1995; Fuhlendorf and Engle, 2001, 2004). This increases vegetation heterogeneity at both local and landscape levels, similar to non-uniform use by livestock.

Heterogeneity can be further increased at the landscape (multi-pasture) level through varied use of pastures across space and time, including rests, deferments, prescribed burning, varying the intensity of grazing from light to heavy, etc. (Fuhlendorf and Engle, 2001; Derner et al., 2009). Herbivory is a powerful tool for managing rangelands for biodiversity because varied grazing treatments among pastures or management units can be combined with other treatments, such as mowing, cutting, or burning, to enhance heterogeneity at the landscape level. Which treatment or combination of treatments is most suitable and feasible in a particular area depends on local biological and socioeconomic factors (Fuhlendorf and Engle, 2004; Derner et al., 2009; Metera et al., 2010). However, we encourage the use of the natural disturbance agents that shaped rangelands in their future management whenever possible. Fire is perhaps the most fundamental of these natural disturbances (Ryan, 1990; Fuhlendorf and Engle, 2004).

At the landscape level, burning selected management units, while varying those units in space and time, combined with varied grazing strategies (rest; deferment; light, moderate, and heavy grazing; etc.), can create highly heterogeneous landscapes (not even including the within-pasture heterogeneity noted above). Such treatments result in substantial differences in vegetation structure among pastures within a larger management unit, i.e., a landscape (ranging from a ranch to a region) (Derner et al., 2009). The result is a mosaic of vegetation patches and structure resulting in greatly increased vegetation diversity (Vavra, 2005). By alternating treatments like burning and grazing practices among pastures or management units, a dynamic and shifting landscape mosaic is created, driven by patch-selective grazing by herbivores (Fuhlendorf and Engle, 2004). Such a heterogeneous mosaic reflects the historic role of disturbances in North American rangelands, which created vegetation heterogeneity across space and time (Fuhlendorf and Engle, 2004). Selective herbivory then drove differing grazing pressure among sites, resulting in a dynamic landscape of habitats that varied in time since disturbance, and ranged from heavily disturbed to essentially climax communities. The dynamic landscape mosaic thus is constantly changing in response to out-of-phase successional changes among sites, but always includes heavily disturbed areas, essentially undisturbed areas, and a matrix of differing ecological (successional) states that vary due to differing time since disturbance (Fuhlendorf and Engle, 2004).

Many other considerations can facilitate development of heterogeneity within and among pastures on the landscape mosaic. The conservation of rangelands and their varied products requires careful selection of grazing species, grazing treatments, and appropriate stocking levels of grazing animals (Rook et al., 2004; Derner et al., 2009; Metera et al., 2010). Grazing species differ in their preference of habitat and plant species, which can enable the effective use of mixed grazing systems with different animal species to achieve management goals (Rook et al., 2004; Metera et al., 2010). For example, cattle grazing can be used to enhance the shrub component of rangelands to increase vertical structure and browse for wildlife (Severson and Urness, 1994). Conversely, browsing by goats can favor herbaceous species over woody (Stoddart et al., 1975; Hart, 2001; Rook et al., 2004). The selection of livestock thus depends upon the management goals regarding specific habitat needs for conservation of biodiversity. The same is true with fire; when employed at appropriate intervals, fire can limit the expansion of woody species (Ryan, 1990; Scifres and Hamilton, 1993; Bender, 2011), and both grazing and fire can be employed to favor the development of woody species where appropriate (Severson and Urness, 1994; Bender, 2011). At larger scales, further enhancement of landscape-level biodiversity can be achieved by carefully designed prescription grazing on protected areas to compliment the mosaic on adjacent ownerships (Fynn et al., 2016).

Implications

Livestock grazing is not inherently incompatible with maintenance of biodiversity. However, maintaining or creating vegetation heterogeneity at both pasture and landscape scales is key to healthy rangelands and biodiversity. This requires maintaining the ecological processes that shaped the rangelands of North America, including grazing, in order to maintain the health and productivity (for all products) of rangelands. To summarize:

- **Herbivory is an important process in rangeland ecosystems.** Livestock grazing at varied intensities can increase the variety of patch states or habitats at both pasture and landscape scales, increasing both habitat diversity and edge. Even if plant and animal species diversity is reduced in smaller, heavily grazed areas, grazing can increase habitat diversity on the landscape, providing a variety of habitat attributes for wildlife. A mosaic of ungrazed, lightly grazed, moderately grazed, and heavily grazed areas probably would maximize diversity in most landscapes, as long as adequate patch size is maintained.

- **Grazing creates living conditions that are varied and will necessarily be "good" for some species and "bad" for others at a given place and time.** Striving for a uniform level of area utilization or a uniform stubble height is not only impossible to achieve under ranching situations but would create a static landscape that is "good" for only a few species. By increasing heterogeneity, grazing can increase overall wildlife diversity on the landscape, unless grazing has been so severe and prolonged that the landscape has been simplified. In some cases, very heavy grazing is necessary to provide needed habitat structure for some species of concern.

- **A shifting mosaic of intensively grazed, climax, and intermediate states is necessary for the maintenance of structural heterogeneity and biodiversity of rangeland ecosystems.** To emphasize, distribution of the various seral states on the landscape should not be static; this would result only from the removal of the ecological processes that drove development of rangelands. In contrast, a dynamic mosaic is the result of these ecological processes, including herbivory and fire. If these processes are conserved, all relevant ecological states can be maintained on the landscape, and while individual patches will constantly shift from one ecological state to another (the dynamic mosaic), overall proportions of the various states will be conserved on the landscape, providing habitats for the full continuum of rangeland wildlife.

- **Land management involves choices, decisions, and resulting consequences.** For example, when given the choice between two endangered or threatened species that may be present on a particular landscape, are there right choices as to which species should prosper? Management that focuses on maintenance of the processes that developed rangelands, and not on a static "featured species" approach, provides a solution to such conflicts by providing for a variety of ecological states on all landscapes.

- **While persistence of all species cannot be guaranteed by any management approach, a healthy and functional rangeland system should be.** Ultimately, it is the health and functionality of the entire rangeland system that is important, not the presence of a particular species in a particular area. By promoting a healthy and dynamic rangeland system, all relevant ecological states (habitats) should be present on the landscape, and ecological functions preserved. Maintaining healthy and functional processes in a dynamic rangeland landscape is the only guarantee of a healthy rangeland and the varied products of healthy rangelands, including biodiversity.

LITERATURE CITED

Allison, C.D. 2004. Continuous vs. rotational grazing on arid lands: What 70+ years of research tells us. In *Proceedings of the Southwest Beef Symposium, Odessa, Texas, December 14–15, 2004* (pp. 2–5).

Andrew, M.H. 1988. Grazing impact in relation to livestock watering points. *Trends in Ecology & Evolution*, 3, 336–339.

Bahre, C.J. 1991. *A legacy of change: Historic human impact on vegetation of the Arizona borderlands*. Tucson: University of Arizona Press.

Bender, L.C. 2011. Burning for big game [Circular 657]. Las Cruces: New Mexico State University Cooperative Extension Service.

Berry, K.H. 1978. Livestock grazing and the desert tortoise. *Transactions of the North American Wildlife and Natural Resources Conference*, 43, 505–519.

Boivin, N.L., M.A. Zeder, D.Q. Fuller, A. Crowther, G. Larson, J.M. Erlandson, T. Denham, and M.D. Petraglia. 2016. Ecological consequences of human niche construction: Examining long-term anthropogenic shaping of global species distributions. *Proceedings of the National Academy of Sciences*, 113, 6388–6396.

Borer, E.T., et al. 2014. Herbivores and nutrients control grassland plant diversity via light limitation. *Nature*, 508, 517–520.

Bormann, F.H., and G.E. Likens. 1981. *Pattern and process in a forested ecosystem: Disturbance, development, and the steady state based on the Hubbard Brook Ecosystem Study*. New York: Springer.

Bostick, V. 1990. The desert tortoise in relation to cattle grazing. *Rangelands*, 12, 149–151.

Callenbach, E. 1996. *Bring back the buffalo! A sustainable future for America's Great Plains*. Berkeley: University of California Press.

Clements, F.E. 1905. *Research methods in ecology*. Lincoln, NE: The University Publishing Company.

Curtin, C.G., N.F. Sayre, and B.D. Lane. 2002. Transformations of the Chihuahuan Borderlands: Grazing, fragmentation, and biodiversity conservation in desert grasslands. *Environmental Science and Policy*, 5, 55–68.

Derner, J.D., W.K. Lauenroth, P. Stapp, and D.J. Augustine. 2009. Livestock as ecosystem engineers for grassland bird habitat in the western Great Plains of North America. *Rangeland Ecology and Management*, 62, 111–118.

Ellenberg, H., Jr. 1988. Floristic changes due to nitrogen deposition in central Europe. In J. Nilsson and P. Grennfelt (Eds.), *Critical loads for sulfur and nitrogen* (pp. 375–383). Copenhagen: Nordic Council of Ministers.

Elwell, S.L., T. Griswold, and E. Elle. 2016. Habitat type plays a greater role than livestock in structuring shrub-steppe plant-pollinator communities. *Journal of Insect Conservation*, 20, 515–525.

Freilich, J.E., J.M. Emlen, J.L. Duda, D.C. Freeman, and P.J. Cafaro. 2003. Ecological effects of ranching: A six-point critique. *BioScience*, 53, 759–765.

Frost, L.C. 1981. The study of *Ranunculus ophioglossifolius* and its successful conservation in the Badgeworth Reserve, Gloustershire. In H. Synge (Ed.), *Biological aspects of rare plant conservation* (pp 481–489). New York: John Wiley and Sons.

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.

Fynn, R.W.S., D.J. Augustine, M.J.S. Peel, and M. de Garine-Wichatitsky. 2016. Strategic management of livestock to improve biodiversity conservation in African savannahs: A conceptual basis for wildlife-livestock coexistence. *Journal of Applied Ecology*, 53, 388–397.

Gillson, L. 2004. Evidence of hierarchical patch dynamics in an east African savanna? *Landscape Ecology*, 19, 883–894.

Graul, W.D. 1973. Adaptive aspects of the mountain plover social system. *The Living Bird*, 12, 69–94.

Graul, W.D. 1975. Breeding biology of the mountain plover. *The Wilson Bulletin*, 87, 6–31.

Hanley, T.A. 1997. A nutritional view of understanding and complexity in the problem of diet selection by deer (Cervidae). *Oikos*, 79, 209–218.

Hart, S.P. 2001. Recent perspectives in using goats for vegetation management in the USA. *Journal of Dairy Science*, 84 (Supplement), E170–E176.

Hillman, C.N., R.L. Linder, and R.B. Dahlgren. 1979. Prairie dog distribution in areas inhabited by black-footed ferrets. *American Midland Naturalist*, 102, 185–187.

Hobbs, N.T., and D.M. Swift. 1985. Estimates of habitat carrying capacity incorporating explicit nutritional constraints. *Journal of Wildlife Management*, 49, 814–822.

Hoenes, B.D., and L.C. Bender. 2012. Factors influencing foraging habitats of mule deer (*Odocoileus hemionus*) in the San Andres Mountains, New Mexico. *The Southwestern Naturalist*, 57, 370–379.

Hull, A.C., Jr., and M.K. Hull. 1974. Presettlement vegetation of Cache Valley, Utah and Idaho. *Journal of Range Management*, 27, 27–29.

Huston, M. 1979. A general hypothesis of species diversity. *The American Naturalist*, 113, 81–101.

Hutchings, M.J. 1987. The population biology of the early spider orchid *Ophrys sphegodes* Mill. I. A demographic study from 1975 to 1984. *Journal of Ecology*, 75, 711–727.

Huxtable, N. 1992. The desert tortoise. *Rangelands*, 14, 33–34.

Kauffman, J.B., and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications...A review. *Journal of Range Management*, 37, 430–438.

Kie, J.G., and E.R. Loft. 1990. Using livestock to manage wildlife habitat: Some examples from California annual grasslands and wet meadow communities. In K.E. Severson (Tech. Coord.), *Can livestock be used as a tool to enhance wildlife habitat?* [General Technical Report RM-194] (pp. 7–24). Fort Collins, CO: U.S. Forest Service, Rocky Mountain Research Station.

King, N., et al. 1996. Habitat management. In B. Kuvlesky, Jr., J. Lewis, and P. Miller (Eds.), *Masked bobwhite (*Colinus virginianus ridgwayi*) population and habitat viability assessment workshop* (pp. 46–48). Albuquerque, NM: U.S. Fish and Wildlife Service, Southwest Region.

Knopf, F.L., J.A. Sedgwick, and R.W. Cannon. 1988. Guild structure of a riparian avifauna relative to seasonal cattle grazing. *Journal of Wildlife Management*, 52, 280–290.

Krueger, H.O. 1985. Avian response to mountainous shrub-willow riparian systems in southeastern Wyoming [Ph.D. dissertation]. Laramie: University of Wyoming.

Laycock, W.A. 1967. How heavy grazing and protection affect sagebrush-grass ranges. *Journal of Range Management*, 20, 206–213.

Laycock, W.A. 1994. Implications of grazing vs. no grazing on today's rangelands. In M. Vavra, W.A. Laycock, and R.D. Pieper (Eds.), *Ecological implications of livestock herbivory in the West* (pp. 250–280). Denver, CO: Society for Range Management.

Leachman, B., and B. Osmundson. 1990. *Status of the mountain plover: A literature review*. Golden, CO: U.S. Fish and Wildlife Service

Leopold, A. 1933. *Game management*. New York: Charles Scribner & Sons.

Lwiwski, T.C., N. Koper, and D.C. Henderson. 2015. Stocking rates and vegetation structure, heterogeneity, and community in a northern mixed-grass prairie. *Rangeland Ecology & Management*, 68, 322–331.

Medin, D.E. and W.P. Clary. 1990. Bird and small mammal populations in a grazed and ungrazed riparian habitat in Idaho [Research Paper INT-425]. Ogden, UT: U.S. Forest Service, Intermountain Research Station.

Mensah, K.K. 2010, May. Masked bobwhite quail reintroduction ahead of schedule. *Arizona Highways*. Available at https://arizonahighways.files.wordpress.com/2010/05/the-ambitious-masked-bobwhite-quail-re.pdf

Metera, E., T. Sakowski, K. Słoniewski, and B. Romanowicz. 2010. Grazing as a tool to maintain biodiversity of grassland—A review. *Animal Science Papers and Reports,* 28, 315–334.

Milchunas, D.G. 2006. Responses of plant communities to grazing in the Southwestern United States [General Technical Report RMRS-GTR-169]. Fort Collins, CO: U.S. Forest Service, Rocky Mountain Research Station.

Milchunas, D.G., W.K. Lauenroth, and I.C. Burke. 1998. Livestock grazing: Animal and plant biodiversity of shortgrass steppe and the relationship to ecosystem function. *OIKOS*, 83, 65–74.

Milchunas, D.G., O.E. Sala, and W.K. Lauenroth. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *American Naturalist*, 132, 87–106.

Mueggler, W.F. 1984. Diversity of western rangelands. In J.L. Cooley and J.H. Cooley (Eds.), *Natural diversity in forest ecosystems* (pp. 211–217). Athens: University of Georgia.

Naveh, Z., and R.H. Whittaker. 1979. Structural and floristic diversity of shrublands and woodlands in northern Israel and other Mediterranean areas. *Vegetatio*, 41, 171–190.

Noy-Meir, I. 1981. Responses of vegetation to the abundance of mammalian herbivores. In P.A. Jewell and S. Holt (Eds.), *Problems in management of locally abundant wild mammals* (pp. 233–246). New York: Academic Press.

NRCS. 1999. Grassland birds [Fish and Wildlife Habitat Management Leaflet No. 8]. Madison, MS: USDA Natural Resources Conservation Service, Wildlife Habitat Management Institute.

Pieper, R.D. 1994. Ecological implications of livestock grazing. In M. Vavra, W.A. Laycock, and R.D. Pieper (Eds.), *Ecological implications of livestock herbivory in the West* (pp. 177–211) Denver, CO: Society for Range Management.

Popper, D.E, and F.J. Popper. 1994. The buffalo commons: A bioregional vision of the Great Plains. *Landscape Architecture*, 84, 144.

Reese, K.P., and J.T. Ratti. 1988. Edge effect: A concept under scrutiny. *Transactions of the North American Wildlife and Natural Resources Conference*, 53, 127–130.

Ribic, C.A., et al. 2009. Area sensitivity in North American grassland birds: Patterns and processes. *The Auk*, 126, 233–244.

Rook, A.J., B. Dumont, J. Isselstein, K. Osoro, M.F. WallisDeVries, G. Parente, and J. Mills. 2004. Matching type of livestock to desired biodiversity outcomes in pastures—A review. *Biological Conservation*, 119, 137–150.

Rosenthal, G., J. Schrautzer, and C. Eichberg. 2012. Low-intensity grazing with domestic herbivores: A tool for maintaining and restoring plant diversity in temperate Europe. *Tuexenia*, 32, 167–205.

Ryan, M.R. 1990. A dynamic approach to the conservation of the prairie ecosystem in the Midwest. In J.M. Sweeney (Ed.), *Management of dynamic ecosystems* (pp. 93–106). West Lafayette, IN: The Wildlife Society.

Ryder, R.A. 1980. Effects of grazing on bird habitats. In R.M. DeGraff and N.G. Tiighmann (Eds.), *Management of western forests and grasslands for nongame birds* [General Technical Report INT-86] (pp. 51–66). Ogden, UT: U.S. Forest Service, Intermountain Region.

Sayre, N.F. 2005. Interacting effects of landownership, land use, and endangered species on conservation of Southwestern U.S. rangelands. *Conservation Biology*, 19, 783–792.

Schroeder, M.A., J.W. Connelly, C.L. Wambolt, C.E. Braun, C.A. Hagen, and M.R. Frisina. 2006. Society for Range Management Issue Paper: Ecology and management of sage grouse and sage grouse habitat: A reply. *Rangelands*, 28(3), 3–7.

Schulz, T.T. and W.C. Leininger. 1990. Differences in riparian vegetation structure between grazed areas and exclosures. *Journal of Range Management*, 43, 295–299.

Scifres, C.J., and W.T. Hamilton. 1993. *Prescribed burning for brushland management*. College Station: Texas A&M University Press.

Severson, K.E., and P.J. Urness. 1994. Livestock grazing: A tool to improve wildlife habitat. In M. Vavra, W.A. Laycock, and R.D. Pieper (Eds.), *Ecological implications of livestock herbivory in the West* (pp. 232–249). Denver, CO: Society for Range Management.

Sierra-Corona, R., A. Davidson, E.L. Fredrickson, H. Luna-Soria, H. Suzan-Azpiri, E. Ponce-Guevara, and G. Ceballos. 2015. Black-tailed prairie dogs, cattle, and the conservation of North America's arid grasslands. *PLoS One*, https://doi.org/10.1371/journal.pone.0118602.

Smart, A. 2008. Managing grasslands for structural heterogeneity. *Center for Grassland Studies*, 14(4), 1, 5.

Society for Range Management. 2002. Policy statements, position statements and resolutions. Available at http://www.rangelands.org/pdf/position%20and%20policy%20statement.pdf

Stoddart, L.A., A.D. Smith, and T.W. Box. 1975. *Range management*, 3rd ed. New York: McGraw-Hill.

Teague, W.R., et al. 2016. The role of ruminants in reducing agriculture's carbon footprint in North America. *Journal of Soil and Water Conservation*, 71, 156–164.

Toombs, T.P., J.D. Derner, D.J. Augustin, B. Krueger, and S. Gallagher. 2010. Managing for biodiversity and livestock: A scale-dependent approach for promoting vegetation heterogeneity in western Great Plains grasslands. *Rangelands*, 32(3), 10–15.

Török, P., O. Valkó, B. Deák, A. Kelemen, and B. Tóthmérész. 2014. Traditional cattle grazing in a mosaic alkali landscape: Effects on grassland biodiversity along a moisture gradient. *PLoS One*, https://doi.org/10.1371/journal.pone.0097095.

Uresk, D.W., J.G. MacCracken, and A.J. Bjugstad. 1981. Prairie dog density and cattle grazing relationships. In R.M. Timm and R.J. Johnson (Eds.), *Proceedings of the Fifth Great Plains Wildlife Damage Control Workshop, October 13–15, 1981, University of Nebraska, Lincoln* (pp. 199–201).

USFWS. 2008. Candidate conservation agreement for the lesser prairie-chicken (*Tympanuchus pallidicinctus*) and sand dune lizard (*Sceloporus arenicolus*). Available at www.fws.gov/southwest/es/NewMexico/documents/CCA_CCAA_LPC_SDL_2008_final_signed.pdf

USFWS. 2014. Masked bobwhite (*Colinus virginianus ridgwayi*) 5-year review: Summary and evaluation. Available at https://www.fws.gov/southwest/es/Documents/R2ES/MaskedBobwhiteQuail_5-yr_Review_Mar2014.pdf

USFWS. 2015, February 5. Memorandum: Service position on livestock grazing and working with rangeland owners to conserve sage-grouse. Available at www.fws.gov/greatersagegrouse/documents/Landowners/20150205_FWS_%20Sage%20grouse%20Ranching%20ARD%20Review_%20Sign%20Final.pdf

Van Devender, T.R. 1990. Late quaternary vegetation and climate of the Sonoran Desert, United States and Mexico. In J.L. Betancourt, T.R. Van Devender, and P.S. Martin (Eds.), *Packrat middens: The last 40,000 years of biotic change* (pp. 134–165). Tucson: University of Arizona Press.

Van Soest, P.J. 1996. Allometry and ecology of feeding behavior and digestive capacity in herbivores : A review. *Zoo Biology*, 15, 455–479.

Vavra, M. 2005. Livestock grazing and wildlife: Developing compatibilities. *Rangeland Ecology and Management*, 58, 128–134.

Vesey-FitzGerald, D.F. 1960. Grazing succession among East African game animals. *Journal of Mammalogy*, 41, 161–172.

Walker, B. 1989. Diversity and stability in ecosystem conservation. In D. Western and M.C. Pearl (Eds.), *Conservation for the twenty-first century* (pp. 121–130). New York: Oxford University Press.

Watt, A.S. 1947. Pattern and process in the plant community. *Journal of Ecology*, 35, 1–22.

West, N.E. 1993. Biodiversity of rangelands. *Journal of Range Management*, 46, 2–13.

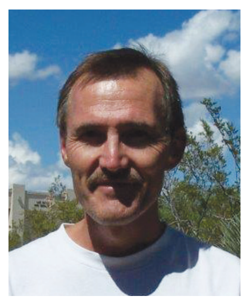
Western, D., and H. Gichohi. 1993. Segregation effects and the impoverishment of savanna parks: The case for ecosystem viability analysis. *African Journal of Ecology*, 31, 269–281.

Whelan, R.J. 1995. *The ecology of fire*. Cambridge: Cambridge University Press.

Wiens, J.A. 1985. Vertebrate responses to environmental patchiness in arid and semiarid ecosystems. In S.T.A. Pickett and T.S. White (Eds.), *The ecology of natural disturbance and patch dynamics* (pp. 169–192). San Diego, CA: Academic Press.

Zornes, M, and R.A. Bishop. 2009. *Western quail conservation plan*. Washington, D.C.: Association of Fish and Wildlife Agencies.

Christopher D. Allison *is a range scientist for the Linebery Policy Center for Natural Resource Management. He earned his Ph.D. in range science from Texas A&M University. His professional interests and expertise include range animal nutrition, grazing management, and range plant toxicology.*



Lou Bender *is a Research Scientist (Wildlife) with the Department of Extension Animal Sciences and Natural Resources at NMSU. He earned his Ph.D. from Michigan State University. His research and management programs emphasize ungulate and carnivore management, integrated wildlife and livestock habitat management, and wildlife enterprises in the Southwest and internationally.*

October 2017 **Las Cruces, NM**