

Elk, Deer, and Piñon-Juniper in New Mexico: Needs, What Works, and What Doesn't

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ABSTRACT—Elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*) differ in nutritional and structural habitat needs; consequently, management of pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodlands to provide optimal habitat differs for these herbivores. Diet quality requirements are higher for deer, and are highest for both species during late gestation, lactation, and antler growth. Properly timed treatments, especially prescribed burning, can enhance forage quality and alter forage species composition, providing dietary benefits during these times. Late-winter burns provide a nutrient flush during these critical periods, and burn intervals can be manipulated to selectively benefit deer or elk by favoring browse or herbaceous species in the understory. Burning at other times can increase forage biomass, but has fewer beneficial effects on forage quality. Overstory thinning can also increase forage biomass given proper slash treatment, but excessive thinning can degrade cover attributes. For example, deer prefer higher levels of security cover, and most movements are near denser woodland patches, which ideally compose approximately 25 percent of each home range on the landscape mosaic. Elk seek thermal cover when average daily temperatures exceed their thermal neutral zone, but this can be provided by single trees or clumps. Other treatments, such as chipping or shredding, may decrease forage and reduce habitat quality if residual biomass is not treated properly.

KEYWORDS—elk, fire, forage, mule deer, New Mexico, nutrition, pinyon-juniper, thinning

INTRODUCTION

Habitat quality is the main determinant of individual and population performance of mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*), primarily through the influence of nutrition on individual body condition and consequent effects on demographic vigor (Cook 2002; Hanks 1981; Wakeling and Bender 2003). Because availability and nutrient content of forage largely determine the condition of individuals (Cook 2002; Verme and Ullrey 1984; Wakeling and Bender 2003), a direct link exists between habitat and population performance (Bender 2012; Cook et al. 2004; Gaillard et al. 2000; Hanks 1981; Hoenes and Bender 2012). Thus, attributes of habitat, including plant community composition, phenological development, and availability of cover, influence the condition of individuals and hence population performance (Hoenes

and Bender 2012; Wakeling and Bender 2003), although the mechanisms that drive these habitat-performance relationships are often poorly appreciated by managers.

Expansion and increasing density of pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodlands, particularly juniper, is a contentious issue for management of mule deer and elk habitat in the southwestern United States including New Mexico (Lutz et al. 2003; Short and McCullough 1977; Short et al. 1977). Pinyon-juniper (PJ) woodlands are one of the most extensive vegetation types in North America (Romme et al. 2009), and include nearly 3 million ha in New Mexico (Van Hooser et al. 1993), almost 10 percent of the total area of the state. Most of this woodland is actual or potential habitat for mule deer and elk, making PJ woodlands the most abundant habitat type for mule deer and

In: Malcolm, Karl; Dykstra, Brian; Johnson, Kristine; Lightfoot, David; Muldavin, Esteban; Ramsey, Marikay. 2020. Piñon-juniper habitats: Status and management for wildlife. Proceedings RMRS-P-77. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 128 p.

Papers published in these proceedings were submitted by authors in electronic media. Editing was done for readability and to ensure consistent format and style. Authors are responsible for content and accuracy of their individual papers and the quality of illustrative materials. Opinions expressed may not necessarily reflect the position of the U.S. Department of Agriculture.

elk in New Mexico. However, a number of attributes associated with these woodlands can limit their value to deer and elk, including:

1. High concentrations of secondary plant compounds, especially terpenes, in foliage. The antimicrobial properties of these limit fermentation and thus rumen function (Schwartz et al. 1980a,b). Consequently, forage value of juniper is limited despite high digestible energy.
2. Presence of other secondary plant compounds such as phenolics, which may have community-level effects on the herbaceous and woody shrub understory, such as inhibiting growth due to leachate (Ramsey 1989).
3. Limited understory development due to competition for light, water, and other nutrients (Hoenes and Bender 2012; Lutz et al. 2003; Pieper 1990; Short and McCullough 1977; Short et al. 1977). This limits the availability of ungulate forage in denser PJ stands.

The lack of understory development in particular has driven most management efforts intended to improve PJ woodlands as deer and elk habitat. Most treatments attempt to increase understory production through partial or complete removal of the PJ canopy, because an inverse relationship between overstory PJ cover and understory production is frequently seen (e.g., Brockway et al. [2002]; Halbritter and Bender [2011b]; Hoenes and Bender [2012]; Howard et al. [1987]; Pieper [1990]; Short et al. [1977]). Understory species include potentially important forages such as forbs and woody browse (Hoenes and Bender 2012).

An implicit assumption behind overstory reduction treatments is that understory biomass equals food, which ignores differing foraging strategies and forage quality requirements of large herbivores (Hofmann 1985) as well as differing nutritional quality and palatability of resultant understory communities. Moreover, the magnitude and type of understory response can be influenced by a variety of factors, including the type, timing, and size of treatments, and the composition of extant understory. (Bender 2011, 2012; Bergman et al. 2014). Consequently, the habitat-performance relationship is far more complex than the simple generalization that less PJ equals more food. Because

of this, the anticipated benefits of treatments are often not realized. In a recent review of published material that assessed population responses of wildlife to PJ treatments, Bombaci and Pejchar (2016) found that less than 10 percent of mechanical removal treatments (i.e., chaining, bulldozing) and less than 20 percent of thinning treatments resulted in any positive response by ungulates.

The lack of positive responses raises the issue of what role PJ actually plays in deer and elk habitat-performance relationships; in other words, does the presence of untreated PJ negatively affect performance of elk and deer or not? Further, inconclusive findings may not solely result from a simplistic “less PJ equals more food” assumption. Habitat-herbivore interactions and resultant individual and population dynamics are highly complex, and require understanding of how habitat attributes fundamentally affect individuals in a population, and how individual responses may (or may not) be reflected in population responses. Understanding these fundamental relationships is necessary for developing informed management treatments to benefit deer and elk.

Moreover, mule deer and elk are very different herbivores, with very different forage quality and cover requirements (Cook 2002; Hofmann 1985; Verme and Ullrey 1984; Wakeling and Bender 2003). Hence, there is no single prescription for optimal management of PJ woodlands or other habitats for elk and deer or large herbivores, as often presented in management guidelines that tend to emphasize thermal cover (e.g., Miller et al. [2005]; Thomas et al. [1979]; Wittmer et al. [1985]). Other issues may also cloud the results of herbivore-PJ studies, including the use of evaluation metrics (e.g., population size, survival, pellet group, or telemetry location density) that are either problematic indices of habitat quality or far removed from fundamental responses to habitat; a poor understanding of treatment effects on elk and deer habitat needs; and use of management treatments ostensibly intended for deer or elk but actually intended to achieve other, often unstated, management goals. A common example is the continued emphasis on winter thermal cover (Thomas et al. 1979) in Federal lands management for elk and deer, despite the demonstration that winter thermal cover provides no energetic benefit (Cook et

al. 1998; Freddy et al. 1986) and that security cover can be provided by multiple seral states other than large sawtimber or old growth.

These issues contribute to the challenge of identifying successful treatments of PJ for elk and deer. To illustrate, Bombaci and Pechjar (2016) recently reviewed more than 40 primary papers regarding PJ treatments and elk or deer. A January 2017 search of Web of Knowledge (Biological Abstracts) revealed at least 67 papers on mule deer and PJ, and at least 26 on elk and PJ, and this search did not include literature from sources such as USDA Forest Service publications. Given all of these publications and knowledge, it would be a reasonable expectation that any number of management treatments that consistently improve PJ woodlands as elk and deer habitat would have been identified. Instead, the actual efficacy of most treatments is uncertain (Bombaci and Pechjar 2016). Reasons for this uncertainty include the limitations of many of these studies, particularly the lack of relating habitat management practices to increases in individual performance, which must be present to realize population-level responses.

Consequently, understanding fundamental relationships to habitat (i.e., habitat-individual performance), how these differ between mule deer and elk, and how these are (or are not) met by PJ habitats, and why, is necessary to develop informed actions to benefit these large herbivores. Herein, I attempt to clarify these issues in order to guide effective habitat management. My goals included:

1. Clarifying the fundamental role of habitat for elk and deer
2. Illustrating how this fundamental relationship affects individual and population performance
3. Evaluating the influence of PJ on performance of deer and elk, and
4. Evaluating preliminary results from selected ongoing long-term studies of management treatments in terms of providing for elk and deer needs for forage and cover.

To do this I review the role of habitat in individual and population productivity (i.e., the

habitat-condition-productivity link), and existing literature on fundamental relationships between mule deer, elk, and PJ in New Mexico. I further use existing and new datasets from multiple populations and study areas in New Mexico to define needs of mule deer and elk relative to PJ woodlands, and to identify treatments that provide for these needs in the hotter, more arid woodlands of New Mexico.

What is Elk and Deer Habitat?

Understanding how habitat fundamentally benefits individuals, and how or why populations then may (or may not) respond to treatments, is key to designing effective habitat management strategies for deer and elk. The fundamental role of habitat is to maximize the energy balance of individuals, because every life process of homeothermic organisms is fundamentally influenced by nutrition and individual condition (e.g., Cook 2002; Cook et al. 2004; Hanks 1981; Mautz 1978; Verme and Ullrey 1984; Wakeling and Bender 2003). Habitat affects individual energetics of wildlife both directly and indirectly. The primary direct influence is providing nutrients, that is, foraging habitat, especially forage quality. The indirect influences include:

1. Minimizing disturbance, that is, hiding or security cover
2. Acquiring food, that is, ambush or stalking cover
3. Life security, that is, hiding or security cover, and
4. Moderating environmental effects, that is, thermal cover.

Each of these ultimately focuses on optimizing individual body condition through acquiring energy, minimizing loss of energy, or preventing the individual from being converted to a different individual's energy.

To maximize individual energy balance, individuals must balance food vs. cover needs. With respect to PJ woodlands, the relevant questions thus are: How does PJ affect individual energetics, why, and how can management treatments affect these fundamental relationships?

Evaluating the influences of PJ on deer and elk, and the efficacy of management treatments to enhance

individual and population performance, requires understanding of how individuals and populations respond to resource availability. Individuals (and populations) respond to resource stress in a hierarchical and predictable manner (Gaillard et al. 2000; Hanks 1981), including:

1. Declines in individual body condition
2. Declines in juvenile conception and fecundity
3. Declines in juvenile survival
4. Declines in adult fecundity, and
5. Declines in adult survival.

Thus, resource stress is first evident in declining nutritional condition of adults, which results in delayed sexual maturation of yearlings and decreased litter size of both primiparous and multiparous females (Bender and Hoenes 2017a; Bishop et al. 2009; Cook et al. 2004; Gaillard et al. 2000; Johnstone-Yellin et al. 2009). Decreased condition further results in later birth dates and lighter birth weights of neonates, which decreases preweaning survival (Bender and Hoenes 2017a; Cook et al. 2004; Lomas and Bender 2007). Only under extreme stress may adult survival be affected (Gaillard et al. 2000). Thus, individual and population productivity are most sensitive to stress, adult survival least.

Performance metrics that assess individual condition and juvenile growth and fecundity are much closer to indexing the fundamental role of habitat on deer and elk, and thus capturing real relationships between habitat attributes and deer and elk performance (Bergman et al. 2014; Hanks 1981). Conversely, metrics such as adult survival or population rate-of-increase may not be affected until extreme levels of resource stress are reached because of compensatory responses in survival dynamics of juveniles and adults, both within and among age classes (Bender 2008). Additionally, survival and population size, being further removed from the fundamental influence of habitat, can be affected by many extraneous variables, including harvest, accidents, severe density-independent events (e.g., weather, some diseases), and population age-structure, particularly in relatively small geographic areas such as most treatment test sites or study areas. Hence, the

distance of these metrics from the fundamental effect can lead to incorrect or inconclusive evaluations of habitat treatments. They are poor evaluation metrics, and their use should be avoided (Van Horne 1983).

Nutritional requirements of individuals vary throughout the annual cycle of deer and elk (fig. 1), and these differences must also be understood if management treatments are to benefit individual energetics. For both deer and elk, the greatest nutritional requirement occurs during the last trimester of gestation and during primary lactation (Cook 2002; Wakeling and Bender 2003), equivalent to early spring through mid-summer. The magnitude of this effect is apparent by the overwhelming negative effect of lactation on individual condition under all circumstances below optimal nutrition (Bender and Hoenes 2017b; Cook et al. 2004; Piasecke and Bender 2011). Because free-ranging populations rarely if ever attain optimal nutrition, lactation status has a greater effect on individual condition than most habitat or environmental influences in free-ranging mule deer and elk populations in New Mexico (Bender and Hoenes 2017b; Piasecke and Bender 2011). This impact, however, can be reduced as habitat quality increases (Piasecke and Bender 2011). The importance of the individual energetic cycle with regard to habitat management of PJ is that management treatments must provide increased nutrition or decreased disturbance when most needed. This means, for deer and elk, during late gestation and early lactation. Consequently, treatments should be designed to provide high quality forage or a nutrient flush during spring and early summer (Bender 2011, 2012).

Cover requirements also vary. Security cover is important throughout the annual cycle of deer and elk, both to minimize disturbance and to keep the individual alive. Mule deer are more vulnerable than elk to predation, and thus their preferences for security cover are much stronger (e.g., Bender et al. [2007b]; Heffelfinger et al. [2006]; Hoenes [2008]; Lomas and Bender [2007]; Lutz et al. [2003]; Short et al. [1977]). Minimizing disturbance is more critical during periods of high energetic needs or low resource availability; for example, Shively et al. (2005) documented decreased productivity of elk in response to increased disturbance during the calving season.

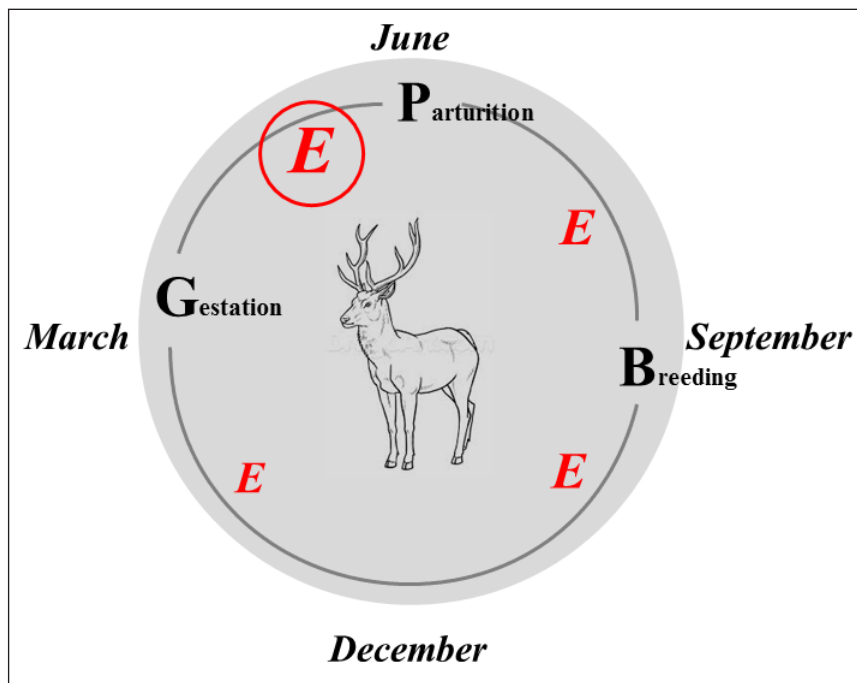


Figure 1—Energetic requirements (E) of elk relative to key periods in their life history; the larger the E, the greater the requirements. Energetic requirements are greatest during late gestation and early lactation, corresponding to late April through June in most of New Mexico. Mule deer show the same general pattern, but later parturition periods “twist” the figure clockwise approximately 0.5 to 1 month depending on latitude.

The need of elk and mule deer for winter thermal cover (Cook et al. 1998; Freddy et al. 1986) has been discounted, although the thermal tolerance of both species may be challenged in hotter climates (Parker and Gillingham 1990; Parker and Robbins 1984). In particular, relatively recent colonization (or recolonization) of hotter, more arid PJ woodlands by elk in New Mexico (e.g., Chaco Canyon area, Oscura and San Andres Mountains) makes summer thermal tolerance of elk a relevant issue in PJ management. The upper thermal neutral zone of elk is approximately 26.5 °C (Parker and Robbins 1984); above this, they need to actively cool themselves. Consequently, elk shift activity patterns when mean monthly high temperature exceeds their thermal tolerance, including avoiding activity during midday hours and seeking bedding sites with high overhead cover, particularly overstory trees (Bender et al. 2012a). Conversely, summer thermal cover needs of mule deer can be satisfied with under-story shrubs (Tull et al. 2001).

Last, mule deer and elk are not the same animal, despite being frequently treated as such, and both differ from cattle in their dietary needs. Forage quality requirements of elk and mule deer are much higher than

for cattle; for example, good diet quality for elk is 60 to 64 percent digestibility (or about 2.75–2.90 kcal/g of forage) (Cook et al. 2004). Needs of mule deer are even higher (Hanley 1997; Wakeling and Bender 2003), and thus mule deer are more sensitive to the specific characteristics and species composition of foraging environments. Consequently, cattle—not deer or elk—are often the primary beneficiary of management treatments that are intended to benefit wild ungulates but are poorly planned with respect to timing and type of forage responses.

These diet quality differences are reflected in the food habits of mule deer and elk. Woody browse and forbs are the main components of mule deer diets (Boeker et al. 1972; Krausman et al. 1997; Short et al. 1977; Wakeling and Bender 2003), whereas elk include much higher proportions of grass in their diets (Cook 2002; Short et al. 1977). Mule deer require higher quality forages (i.e., higher cell soluble:cell wall ratio) because their smaller rumeno-reticulum limits their ability to digest forages higher in cellulose (such as grasses) as compared to elk (Hanley 1997; Hofmann 1985; Putnam 1988). Thus, mule deer are far more selective of higher quality forage, spend more time

foraging, and are more sensitive to species composition and structure of plant communities (Hoenes and Bender 2012; Wakeling and Bender 2003). Because of this environmental sensitivity, woody browse is generally the predominant year-long component of deer diets (Boeker et al. 1972; Krausman et al. 1997), as its availability compared to forbs is less influenced by environmental factors such as precipitation (Hoenes and Bender 2012). But many forbs are better sources of nitrogen (N), digestible energy (DE), and other nutrients (Boeker et al. 1972; Urness et al. 1971; Wallmo et al. 1977). Optimal forage for mule deer is thus perfectly suitable for elk. However, elk can be highly productive on ranges dominated by lower quality foods such as grasses in which mule deer could not be productive.

Understanding the fundamental role of habitat helps clarify the current dynamics of cervids in New Mexico, and the importance of habitat quality is clearly seen in population demographics. Survival and performance of multiple elk and especially mule deer populations throughout New Mexico were limited by poor habitat quality through effects on individual body condition (Bender et al. 2007a, 2011, 2012b; Bender and Hoenes 2017b; Bender and Piasecke 2010; Halbritter and Bender 2011a; Hoenes 2008; Lomas and Bender 2007; Piasecke and Bender 2011), which strongly affected adult survival (Bender et al. 2007a, 2011; Halbritter and Bender 2011a), production and survival of juveniles (Bender and Piasecke 2010; Hoenes 2008; Lomas and Bender 2007), and predisposition to mortality (Bender et al. 2007a, 2011, 2012b; Bender and Rosas-Rosas 2016; Hoenes 2008; Lomas and Bender 2007). Consequently, the leading cause of mortality among mule deer populations was malnutrition (Bender et al. 2007a, 2011, 2012b; Hoenes 2008; Lomas and Bender 2007).

In response, mule deer populations have significantly declined throughout much of New Mexico (Bender et al. 2007a, 2011, 2012b), mostly because of declines in the quantity and quality of food and also because of seasonal drought (particularly from conception through parturition, approximately January–June) (Bender et al. 2007a, 2011, 2012b; Hoenes 2008). Conversely, elk are increasingly colonizing (or recolonizing) new habitats, including hotter, arid habitats such as PJ, which is typically associated with environmental conditions that challenge both thermoregulatory and water needs

of elk (Bender et al. 2010, 2012a). However, population productivity of elk is similarly strongly related to individual condition and hence forage quality in New Mexico (Bender and Piasecke 2010; Halbritter and Bender 2011a, 2015). Thus, while both cervids share the same fundamental relationships with habitat, trends of each species in response to recent past and current changes in habitats, including PJ expansion and related natural and anthropogenic changes, are dissimilar, highlighting the fundamental differences between these large herbivores.

How Does PJ Function as Elk and Deer Habitat?

In this section, I review studies from New Mexico that assessed the fundamental quality of PJ as mule deer and elk habitat, in other words, the influence of PJ on accrual of condition (e.g., body fat). I also review the influence of PJ on home range size, an index of habitat quality (Bender 2012) that is likely to reflect cover requirements more so than nutritional needs. Last, I review the role of PJ as cover for mule deer and elk. I include only data for females, since their dynamics drive population rate-of-increase (White and Bartmann 1998).

Mule deer

On the Corona Range and Livestock Research Center in east-central New Mexico, Bender et al. (2013) found that body fat of mule deer was not significantly related to either the proportion of PJ in spring-summer-autumn (SSA) home ranges or to the proportion of deer locations in PJ during the SSA period. However, body fat was positively related to the proportion of locations in mechanically thinned PJ during the SSA. Size of SSA home ranges was also not related to the proportion of PJ in SSA home ranges, but was negatively related to the number of locations of deer in PJ during the SSA period, as well as the number of locations in mechanically thinned PJ; the latter two indicate that habitat quality for deer apparently increased as more PJ was included in their home ranges. Additionally, more than 88 percent of locations of mule deer were less than 200 m from unmanaged PJ woodlands or savannas (Bender et al. 2013), indicating that deer required PJ woodlands for security cover in this site (Bender et al. 2013).

Similarly, body fat was unaffected by proportion of PJ in SSA home ranges or use of PJ during SSA in the San Andres Mountains in south-central New Mexico (Hoenes 2008; L. Bender, unpublished data). Size of SSA home ranges was negatively related to the proportion and use of PJ during the SSA. Deer were also located in PJ in SSA from 4 to 16 times more often than the availability of PJ in SSA ranges would predict, and 1.2 to 24 times more often than PJ availability in annual home ranges would predict.

In contrast, accrual of body fat was negatively impacted by increasing proportion of PJ in SSA home ranges and increasing use of PJ during the SSA period in north-central New Mexico (Bender et al. 2007b). Size of SSA ranges was not related to any PJ measure. Piñon-juniper was overrepresented by 12.4 percent in SSA ranges and 13.0 percent in annual home ranges as compared to its availability on the landscape, and deer were located in PJ at least 1.1 times more frequently than predicted by landscape availability (Bender et al. 2007b).

Results from these individual sites suggest a mixed response to PJ by mule deer, similar to the inconclusive results reviewed by Bombaci and Pejchar (2016). Condition appears to be unaffected or negatively affected by untreated PJ, yet many metrics of distribution suggest positive effects, likely driven by cover needs. Consequently, gains in mule deer habitat can be attained by management of PJ woodlands to increase forage quantity and quality (e.g., Bergman et al. [2014]), but care must be taken to preserve adequate structure for cover needs of mule deer (Bender 2012).

Elk

Results from elk studies were also variable. In the Chaco Canyon area of northwestern New Mexico, Bender et al. (2012a) found accrual of body fat to be unrelated to the proportion of PJ in SSA home ranges, while size of both annual and SSA home ranges was negatively related to the proportion of PJ in home ranges. Elk home ranges included PJ at 1.8 to 2.4 times its availability on the landscape, and elk were found in PJ 1.4 to 1.5 times more frequently than availability on the landscape would predict. Additionally, PJ woodlands were an important micro-site element of elk habitat. Twenty-seven percent of

elk bedding sites were located under pinyon or juniper trees (second only to rock ledges and caves, at 35 percent), and bedding sites averaged 46 to 56 percent overhead cover and 55 to 63 percent horizontal cover (Bender et al. 2012a).

In the Lincoln National Forest of south-central New Mexico, Halbritter and Bender (2011b) found body fat to be negatively related to the proportion of PJ in SSA home ranges. In contrast to the results from the Chaco Canyon area, elk included PJ in SSA home ranges at only 0.05 to 0.16 times its availability on the landscape. The Lincoln study area was characterized by predominant availability of more mesic forest types such as aspen (*Populus tremuloides*) and mixed conifer (Halbritter and Bender 2011b), which were not present on the Chaco site. Availability of these more mesic elk habitats very likely decreased the use of PJ woodlands for cover.

Similar to mule deer, results from these individual sites suggest a mixed response to PJ by elk. Unmanaged PJ woodlands appear to have no positive impact on body condition, but metrics of distribution suggest positive effects, at least in hotter, more arid sites such as Chaco Canyon. Also similar to mule deer, gains in elk habitat potentially could be attained by management of the PJ woodlands to increase forage quantity and quality. In hotter, more arid sites, adequate amounts of thermal cover need to be maintained for elk (Bender et al. 2012a). This management practice may be particularly relevant given that elk are increasingly using these hotter, lower elevation PJ woodlands (e.g., Chaco Canyon, the Oscura and San Andres Mountains), and the thermal benefits of PJ may be a key element to maintain elk in such areas given their thermal tolerance limitations (Bender et al. 2012a).

METHODS

Study Areas

Deer and Elk Condition and Distribution

My study areas for mule deer condition and distribution included past projects on the Corona Range and Livestock Research Center (CRLRC), the San Andres Mountains (SAM), and Colfax County (Northcentral). Elk study areas included past projects on Chaco Culture National Historic Park (Chaco) and Lincoln

National Forest (LNF), and ongoing work in the Oscura Mountains and Chupadera Mesa (Oscura). Herein, I restricted individuals from past and current studies to those with home ranges that included only untreated PJ woodlands, and eliminated individuals with home ranges that included areas of treated PJ regardless of treatment type. Brief descriptions of these sites follow; see respective citations for additional details.

The CRLRC is a 113-km² working research ranch located in east-central New Mexico. Average high temperature in July was 29 °C and average low temperature in January was −6 °C. Mean annual precipitation was 40 cm, most of which occurs in July and August as high intensity, short duration convectional thunderstorms. During this study, annual precipitation averaged 0.93 times the normal (range = 0.72–1.22 times). Pinyon-juniper (*J. monosperma*) woodlands made up approximately 20 percent of the study site (Bender et al. 2011, 2013).

The SAM study area encompassed approximately 11,000 km² primarily within White Sands Missile Range (WSMR), south-central New Mexico. Average high temperature in July was 35 °C and average low temperature in January was −3 °C at WSMR. Precipitation averaged 20 to 35 cm annually, with the bulk of moisture occurring as short, intense rainstorms from July through September. During this study, annual precipitation averaged 1.05 times the normal (range = 0.50–1.46 times). Pinyon-juniper (primarily *J. monosperma* with occasional *J. deppeana*) woodlands made up about 7 percent of the study site (Bender et al. 2012b; Hoenes and Bender 2012).

The Northcentral study area encompassed approximately 4,860 km² in Colfax County, north-central New Mexico. Average high temperature in July was 28 °C and average low temperature in January was −8 °C at Raton, New Mexico. Mean annual precipitation averaged 44 cm, with 62 percent falling from May through August. During this study, annual precipitation averaged 1.03 times the normal (range = 0.72–1.56 times). Pinyon-juniper (primarily *J. monosperma*) woodlands made up approximately 16 percent of the study site (Bender et al. 2007a,b).

The Chaco study area covered approximately 308 km² on and adjacent to Chaco Culture National Historical Park in northwestern New Mexico. Average high temperature in July was 32 °C and average low temperature in January was −11 °C. Average annual precipitation was 23 cm, with 52 percent falling from July through October. During this study, annual precipitation averaged 0.92 times normal (range = 0.84–1.02 times). Pinyon-juniper (primarily *J. monosperma*) woodlands made up approximately 15 percent of the study site (Bender and Piasecke 2010; Bender et al. 2012a).

The LNF study area encompassed much of the 1,823-km² Sacramento Ranger District in the southern Sacramento Mountains of south-central New Mexico. Average high temperature in July was 22 °C and average low January temperature was −7 °C in Cloudcroft, New Mexico. Annual precipitation averaged 67 cm with more than 50 percent falling from June through September. During this study, annual precipitation averaged 0.82 times the normal (range = 0.66–0.96 times). Pinyon-juniper (primarily *J. monosperma* with occasional *J. deppeana*) woodlands made up approximately 10 percent of the study site (Halbritter and Bender 2011a,b).

The Oscura study area encompassed approximately 870 km² of the Oscura Mountains and adjacent Chupadera Mesa in south-central New Mexico. Average high temperature in July was 33 °C and average low January temperature was −6 °C in Bingham, New Mexico. Annual precipitation averaged 27 cm with 66 percent falling from July through October, although this average underestimates actual precipitation received in the higher elevation PJ zone. During this study, annual precipitation averaged 0.97 times the normal (range = 0.74–1.07 times). Pinyon-juniper (*J. monosperma* and scattered *J. deppeana*) woodlands made up approximately 48 percent of the study site.

Pinyon-juniper treatment sites

I also include preliminary results from a subset of long-term studies assessing responses of preferred mule deer and elk forages to differing treatments of PJ woodlands on private ranches across the Southwest. Because these

studies are ongoing and several landowners preferred not to have their property identified, I refer to these ranches by general geographic location only.

Pinyon-juniper treatment areas were located in south-central (SCNM1 and SCNM2), north-central (NCNM), and central (CNM) New Mexico. Climatic and PJ relationships for the SCNM sites were similar to those listed for the Oscura site. Research on SCNM1 evaluated the influence of light long-term (i.e., decades) seasonal suitability and best pasture grazing on understory composition; this was the only site on which livestock grazing occurred. Research on the SCNM2 site has evaluated the effects of broadcast burning (i.e., understory burning of the entire ranch) on understory species composition since 2011. During this study, annual precipitation measured at the site averaged 0.86 times (range = 0.50–1.27 times) the approximate norm of 32 cm recorded at the nearest Western Regional Climate Center (wrcc.dri.edu) station. This site was previously thinned approximately 10 years before implementation of the burning trials.

Conditions for the NCNM site were similar to those listed for the Northcentral site, except that PJ woodland was more varied, ranging from savanna with less than 10 stems/ha to denser woodland with more than 200 stems/ha. During this study, annual precipitation at the site averaged 0.94 times (range = 0.67–1.17 times) the approximate norm of 41 cm recorded at the nearest Western Regional Climate Center station. Described treatments were limited to the PJ savanna on this site, where the effect of broadcast burning on understory composition has been evaluated since 2008.

Conditions for the CNM site were similar to those for the CRLRC. During this study, annual precipitation at the site averaged 1.00 times (range = 0.89–1.18 times) the approximate norm of 40 cm recorded at the nearest Western Regional Climate Center station. This site has been used to evaluate the effects of overstory thinning and chipping or mastication since 2013.

Deer and Elk Condition and Distribution

I captured adult female elk and mule deer by darting or net-gunning from a Bell 206B Jet Ranger/OH58 helicopter using carfentanil citrate and xylazine hydrochloride in late fall at the seasonal peak of condition

on each site (Bender et al. 2007a, 2011, 2012b; Bender and Piasecke 2010; Halbritter and Bender 2011a; L. Bender, unpublished data). I blindfolded individuals to minimize stress during handling, and administered penicillin, vitamin B, vitamin E and selenium (MU-SE), and an eight-way *Clostridium* bacterin to alleviate capture stress. I radio-collared each individual with a very high frequency collar (Advanced Telemetry Systems, Asanti, Minnesota) and attached unique numbered ear tags (relative to site).

Individuals were located from the ground at least once per week, and locations were recorded with a handheld GPS unit and plotted using the geographic information system ArcGIS 10.0 (ESRI, Redlands, California). I calculated 95-percent minimum convex polygons (MCPs) (Mohr 1947) for each individual annually and for SSA (April–November) locations only, and a composite 100-percent MCP with a 2.5-km buffer from all locations pooled to define the overall herbivore use area. I used annual and SSA home ranges of each individual as a clip coverage in ArcGIS 10.0 to clip 30-m raster vegetation covertype coverages to determine the vegetation composition of annual and SSA home ranges. I delineated vegetation covertypes from the U.S. Geological Survey's Southwest ReGAP land cover classification (https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap/science/land-cover-data-overview?qt-science_center_objects=0#qt-science_center_objects).

I used a SonoVet 2000 ultrasound (Medison, Seoul, South Korea) with a 5 mHz probe to measure subcutaneous fat thickness along a straight line midway between the spine, at its closest point to the coxal tuber (hip bone), and the ischial tuber (pin bone) (MAXFAT) (Cook et al. 2001; Stephenson et al. 2002). I also determined a rump body condition score (rBCS) by palpation of the soft tissue of the rump along the sacrosciatic (ischial) ligament and scored results of rBCS from standards which ranged from 1 (emaciated) to 5 (obese) in intervals of 0.25 (Bender et al. 2007a; Cook et al. 2001). I estimated body fat of elk using rBCS and MAXFAT following Cook et al. (2001), and of mule deer following Bender and Hoenes (2016).

As noted earlier, I limited this analysis to individuals that had only untreated PJ woodlands in their annual and SSA home ranges in order to assess the influence

of PJ woodlands on condition in the absence of woodland treatments. I used a linear mixed model (PROC GLIMMIX; SAS Institute Inc., Cary, North Carolina) to model the effect of proportion of PJ in annual and SSA home ranges of mule deer and elk on accrual of body fat and size of annual and SSA home ranges. I normalized data within populations to account for differences in magnitude of home ranges and fat levels among populations. Population was treated as a random effect in all models, and I used a hierarchical model for body fat to account for the effect of lactation status on condition of mule deer and elk (Bender and Hoenes 2017b; Piasecke and Bender 2011). If PJ had a significant effect on body fat, I further modeled the lactation $\times$ pinyon-juniper interaction to see if PJ cover had differing effects on condition of lactating and dry females.

Forage Responses

I used line-point intercept transects to determine plant cover and species composition during mid- to late August or September annually depending on location, near the end of the growing season. I randomly placed a randomly oriented 100-m transect in each of 15 to 20 patch or stand replicates for landscape treatments (or 10–15 transects within each stand for stand-level treatments), and recorded ground cover as bare, rock, litter, or vegetation at every 1-m interval along transects, with vegetation being further identified to species. I segregated plants into highly palatable, high quality food for deer or elk, or both (hereafter, preferred), or food for other species. I used diet studies from New Mexico (e.g., Boeker et al. 1972] Hoenes 2008; Short et al. 1977; L. Bender, unpublished data) and diets and associated diet quality data collected from many of the previously mentioned study areas used for elk and deer condition as well as ongoing studies to identify preferred forages at each site (L. Bender, unpublished data).

I estimated woody species densities using the point-center quarter method (Wyoming Game and Fish 1982). Center points were established every 10 m along transects, with the four quadrants defined by the transect line and a line perpendicular to it. The nearest shrub (<5.1 cm diameter at breast height [d.b.h.]) and tree (>5.1 cm d.b.h.) located within each quadrant was identified to species and the distance from the center point to each tree or shrub measured. Canopy coverage was also estimated at each 10-m interval along

transects by using a spherical densiometer (Lemmon 1956), taking four separate estimates (one in each cardinal direction) and using the average of all readings as the interval estimate. I also determined the volume (m^3) of preferred browse species by measuring canopy height (at the center) and diameter (mean of two perpendicular measurements) and using the formula for the shape of a solid that best approximated growth form of the shrub following Ludwig et al. (1975). Not all vegetation measures were collected at all sites.

Last, I used historical USDA Soil Conservation Service and Natural Resources Conservation Service Range Site Descriptions (RSDs) and Ecological Site Descriptions (ESDs) (<https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/>) to infer historical species cover for ecological sites that occurred on the study areas.

Pinyon-Juniper Treatments

Burn treatments were conducted in late winter or early spring (March–early April) depending on site following Bender (2011, 2012). Burn areas ranged from approximately 2 to 10 sections in the SCNM2 and NCNM sites. I assessed fire intensity using ground char and flame length classes following Ryan and Noste (1985).

Thinning and chipping or masticating treatments were performed throughout the summer on six 0.1-acre (0.04 ha) plots in the CNM site. An adjacent untreated plot was blocked with one thinned and one chipped plot into three groups. Thinned plots had slash lopped and scattered during treatment. Chip treatment (i.e., chip, and leave or remove) plots were untreated for the following two springs, then chips were manually raked and removed in the fall after the second spring because of a lack of understory growth.

I used a linear mixed model (PROC GLIMMIX; PROC GLM; SAS Institute Inc.) to analyze landscape-level data: time since burn and precipitation effects on plant cover on the NCNM site. I used patch or stand as the random effect. I used MANOVA (PROC GLM; SAS Institute Inc.) to analyze the following within-stand data: cover of forage classes preburn, 1-year postburn, and 4-years postburn in the SCNM2 site and among untreated, thinned, and chip treatments in the CNM site. I also compared preburn and 1-year

postburn cover at the NCNM site using randomization tests (Efron and Tibshirani 1993). Where appropriate, I contrasted candidate models using Akaike's information criteria (Burnham and Anderson 1998) and assessed contributions of covariates using 86-percent confidence intervals (CIs) (Arnold 2010).

RESULTS

Deer and Elk condition and Distribution

Overall, proportion of PJ in both annual ($F_{1,86} = 4.8$; $P = 0.025$) and SSA home ranges ($F_{1,86} = 5.2$; $P = 0.031$) was negatively related to the amount of body fat that mule deer were able to accrue (table 1), although lactating and dry females were differentially affected. Pinyon-juniper cover had a negative effect on body fat accrual of lactating females (annual: $P = 0.008$; SSA: $P = 0.011$), but no effect on dry females (annual: $P = 0.209$; SSA: $P = 0.194$). Proportion of PJ in annual ($F_{1,69} = 2.5$; $P = 0.122$) or SSA ($F_{1,69} = 2.0$; $P = 0.166$) home ranges did not affect accrual of body fat in elk (table 1).

Size of annual home ranges of mule deer was negatively related to proportion of PJ in annual ($F_{1,120} = 13.8$; $P < 0.001$) and SSA ($F_{1,201} = 12.7$; $P < 0.001$) home ranges. Similarly, size of SSA home ranges of mule deer was negatively related to proportion of PJ in annual ($F_{1,120} = 17.9$; $P < 0.001$) and SSA ($F_{1,201} = 20.9$; $P < 0.001$) home ranges. Proportion of PJ in either annual or SSA home ranges of elk was not related to size of either annual (annual: $F_{1,90} = 0.3$; $P = 0.610$; SSA: $F_{1,172} = 0.5$; $P = 0.475$) or SSA (annual: $F_{1,90} = 0.01$; $P = 0.936$; SSA: $F_{1,172} = 0.9$; $P = 0.343$) home ranges.

Vegetation Composition

Understory vegetation cover of mule deer and elk study sites reflected the lack of understory development in denser untreated PJ woodlands (table 2). The same pattern was present on treatment areas (table 3). Preferred forage was only a small proportion of the extant understory on all sites (tables 2 and 3). Among untreated sites, preferred grasses accounted for 65 percent (90-percent $CI_{BOOTSTRAP} = 51$ –79 percent) of total grass cover, preferred forbs 30 percent (90-percent $CI_{BOOTSTRAP} = 21$ –39 percent) of total forb cover, and

preferred shrubs 37 percent (90-percent $CI_{BOOTSTRAP} = 21$ –53 percent) of total shrub cover (tables 2 and 3). Availability of nutritious herbaceous forage during late gestation and early lactation was even rarer; cool-season grasses composed only 12 percent (90-percent $CI_{BOOTSTRAP} = 8$ –20 percent) of total grass cover.

Among the three treatment sites, treatments generally increased cover of preferred grasses (+6 percent [90-percent $CI_{BOOTSTRAP} = 3$ –9 percent]), forbs (+11 percent [90-percent $CI_{BOOTSTRAP} = 7$ –15 percent]), and shrubs (+5 percent [90-percent $CI_{BOOTSTRAP} = 0$ –10 percent]), but not cool-season grasses (+2 percent [90-percent $CI_{BOOTSTRAP} = -1$ –4 percent]). The grazing treatment (SCNM) was not included as it had only 1 year of data without a pretreatment or other control.

Preferred forage was generally comparable between present and historical plant communities (table 4), with the exception of cool-season grasses, which were less abundant in current communities in six of nine comparisons. I am uncertain how relevant these data are, however. Development of RSDs began in the 1930s and 1940s and ESDs were developed mostly starting in the late 1980s, after decades of generally static management of dynamic rangelands. Because herbivory impacts associated with overuse by large herbivores can occur rapidly (Irwin et al. 1994; Noy-Meir 1981) and eliminate preferred forages (Irwin et al. 1994; Klein 1981; Severson and Urness 1994), past herbivory may have eliminated preferred forages or altered understory compositions prior to development of ESDs.

Prescribed Burning

Fire intensity on the SCNM2 site was light (mean char class = 1.3 [standard error (SE) = 0.08]; mean flame length class = 1.6 [SE = 0.09]), corresponding to light ground char and flame lengths of less than 1 m (Ryan and Noste 1983). Burning did not significantly alter PJ density ($F_{2,42} = 0.2$; $P = 0.861$). Preburn, 1-year postburn, and 4-year postburn vegetation cover differed ($F_{4,40} = 42.8$; $P < 0.001$) (table 5). Grass ($F_{2,42} = 5.5$; $P = 0.008$) and forb ($F_{2,42} = 3.3$; $P = 0.048$) cover increased significantly after burning, then declined (table 5). Shrub cover ($F_{2,42} = 3.3$; $P = 0.047$) declined initially, then recovered by 4 years postburn. Burning decreased successional status of preferred browse species (mountain mahogany [*Cercocarpus montanus*]:

Table 1—Relationships between the proportion of pinyon-juniper woodlands in annual (annual) and spring–autumn (SSA) home ranges of mule deer and elk, and the size of annual home range (annual HR), spring–autumn home range (SSA HR), and percent body fat among study areas in New Mexico.

		Proportion of pinyon-juniper woodlands in home range									
		Annual home range					SSA home range				
Species	Variable	$P(\text{rand})^a$	β^b	SE^c	P	N^d	$P(\text{rand})^a$	β^b	SE^c	P	N^d
Mule deer	Size (annual HR)	0.922	−0.32	0.09	<0.001	122	0.939	−0.30	0.08	<0.001	205
	Size (SSA HR)	0.900	−0.39	0.09	<0.001	122	0.977	−0.31	0.07	<0.001	205
	Body fat	0.414	−0.25	0.11	0.025	91	0.311	−0.22	0.10	0.031	91
Elk	Size (annual HR)	0.554	0.46	0.89	0.610	93	0.675	0.47	0.65	0.475	176
	Size (SSA HR)	0.556	−0.05	0.58	0.936	93	0.471	0.29	0.31	0.343	176
	Body fat	0.319	−0.19	0.12	0.122	74	0.116	−0.15	0.10	0.166	74

^aProbability of random effect in model.

^bCoefficient.

^cStandard error.

^dNumber of individuals sampled.

Table 2—Proportion of mule deer and elk study areas in pinyon-juniper (PJ) cover, percent understory grass, cool-season grass (CSG), forb, and woody shrub cover; percent preferred (P) grass, forb, and shrub cover; shrub and tree density; and canopy cover (CC) of untreated pinyon-juniper woodlands in mule deer and elk study areas^a, New Mexico. Values in parentheses are standard errors.

	Mule deer			Elk		
	CRLRC	SAM	Northcentral	Chaco	LNF	Oscura
Proportion PJ	0.20	0.07	0.16	0.15	0.10	0.48
Grass	15 (7)	12 (2)	26 (13)	23 (8)	---	27 (5)
Grass (P)	9 (6)	7 (2)	12 (6)	12 (5)	---	17 (6)
Grass - CSG	2 (2)	2 (1)	3 (2)	11 (5)	---	1 (<1)
Forb	2 (2)	3 (1)	0.3 (0.9)	5 (3)	---	---
Forb (P)	<1 (<1)	1 (1)	0 (0)	2 (2)	---	---
Shrub	4 (4)	9 (1)	6 (7)	27 (9)	---	---
Shrub (P)	2 (1)	3 (3)	6 (7)	10 (6)	---	---
Shrub/ha	---	541 (40)	---	---	---	642 (179)
Shrub/ha (P)	---	274 (66)	---	---	---	257 (77)
Tree/ha	---	223 (18)	188 (41)	---	---	106 (30)
Tree (% CC)	62 (3)	25 (3)	---	33 (12)	---	---

^aCRLRC: Corona Range and Livestock Research Center; SAM: San Andres Mountains; Northcentral: Colfax County; Chaco: Chaco Culture National Historic Park; LNF: Lincoln National Forest; Oscura: Oscura Mountains and Chupadera Mesa.

Table 3—Percent understory grass, cool-season grass (CSG), forb, and woody shrub cover; percent preferred (P) grass, forb, and shrub cover; shrub and tree density; and canopy cover (CC) of pinyon-juniper treatment areas^a, New Mexico. Values in parentheses are standard errors.

	SCNM1	SCNM2	NCNM	CNM
Grass	16 (3)	15 (2)	19 (2)	2 (2)
Grass (P)	12 (3)	12 (2)	16 (2)	<1 (<1)
CSG	<1 (1)	1 (<1)	1 (<1)	0 (---)
Forb	<1 (<1)	4 (1)	1 (<1)	6 (2)
Forb (P)	<1 (<1)	1 (1)	0 (---)	1 (1)
Shrub	<1 (<1)	22 (3)	<1 (<1)	4 (1)
Shrub (P)	0 (---)	6 (4)	0 (---)	2 (1)
Tree/ha	29 (9)	131 (37)	5 (3)	---
Tree (% CC)	---	---	---	76 (6)

^aSCNM1 and SCNM2: south-central New Mexico; NCNM: north-central New Mexico; CNM: central New Mexico.

Table 4—Percent cover of all grasses, cool-season grasses (CSG), forbs, and woody browse species (shrub), and preferred forage (P) within these classes, in current (Ext) understory and in historical (Hist) plant communities in mule deer and elk study areas, New Mexico^a. Extant cover that is lower than historical is in **bold**.

	No specific treatment												Treated					
	CRLRC		SAM		Northcentral		Chaco		Oscura		SCNM1		SCNM2		NCNM		CNM	
	Ext	Hist	Ext	Hist	Ext	Hist	Ext	Hist	Ext	Hist	Ext	Hist	Ext	Hist	Ext	Hist	Ext	Hist
Grass	15	20	12	13	26	21	23	24	27	20	16	12	15	13	19	16	2	20
G (P)	9	12	7	9	12	14	12	12	17	15	12	9	12	9	16	14	<1	12
CSG	2	3	2	1	3	3	11	11	1	2	<1	2	<1	1	1	4	0	3
Forb	2	2	3	1	<1	3	5	2	---	2	<1	1	4	1	1	1	1	2
Forb (P)	<1	1	1	<1	0	1	2	1	---	1	<1	<1	1	<1	---	1	1	1
Shrub	4	3	9	8	6	13	27	10	---	12	<1	3	22	8	<1	2	4	3
Shrub (P)	2	1	3	3	6	11	10	6	---	4	0	2	6	3	0	1	2	1

^aCRLRC: Corona Range and Livestock Research Center; SAM: San Andres Mountains; Northcentral: Colfax County; Chaco: Chaco Culture National Historic Park; Oscura: Oscura Mountains and Chupadera Mesa; SCNM1 and SCNM2: south—central New Mexico; NCNM: north-central New Mexico; CNM: central New Mexico.

Table 5—Percent cover of forage classes and preferred forage (P), and tree density, prior to burning, 1 year after burning, and 4 years after burning, south-central New Mexico. Values in parentheses are standard errors. Values followed by different letters differ at the $P < 0.001$ significance level.

	Preburn	Postburn (1 year)	Postburn (4 years)
Grass	15 (2)A	31 (4)B	23 (5)AB
Grass (P)	12	25	24
Cool-season grass	<1	1	1
Forb	4 (1)A	11 (3)B	5 (2)A
Forb (P)	1	4	2
Shrub	22 (3)A	14 (2)B	23 (4)A
Shrub (P)	6	6	5
Tree (trees/ha)	131 (37)	105 (30)	115 (33)

preburn volume = 0.63 [SE = 0.11]); postburn volume = 0.30 [SE = 0.06]; $P_{\text{RAND}} < 0.001$). This decrease in plant volume was reflected in the drop in shrub cover 1 year postburn.

Fire intensity was similarly light on the NCNM site (mean char class = 1.6 [SE = 0.07]; mean flame length class = 1.9 [SE = 0.09]), corresponding to light-moderate ground char and flame lengths of less than 1 m (Ryan and Noste 1983). Burning increased cover of all grasses (+21 percent; $P_{\text{RAND}} = 0.007$) and warm-season grasses (+19 percent; $P_{\text{RAND}} = 0.014$) by 1 year postburn, but not cool-season grasses, forbs, or shrubs

($P_{\text{RAND}} < 0.166$). Among candidate models including time since burn, annual precipitation, growing season precipitation, precipitation during the previous winter, and their additive and interaction effects, only time since burn ($F_{1,34} = 23.0$; $P < 0.001$) and time since burn ($F_{1,33} = 25.4$; $P < 0.001$) + winter precipitation ($F_{1,33} = 4.7$; $P = 0.037$) were potentially informative (i.e., $\Delta\text{Akaike's information criterion [AICc]} < 3$). Time since burn (summed AIC $\omega = 1.00$) was the predominant variable influencing trends in grass cover following burning for the first 7 years postburn (table 6); grass cover declined approximately 3 percent annually each year.

Table 6—Candidate models of grass cover as influenced by time since burning (Time), precipitation (Precip), and time since burning (Time) in north-central New Mexico. Values of variables with 86-percent confidence intervals that do not include 0 are in **bold**. P(w) = winter precipitation.

Model	ΔAICc	AIC ω	Time	SE	Precip	SE
Time	0	0.81	-0.027	0.006	---	---
Time, P(w)	2.9	0.19	-0.027	0.005	0.023	0.011

Precipitation received during the previous winter (summed AIC $\omega = 0.19$) was the only other variable that had 86-percent CIs that excluded 0 in some models (table 6). Warm-season grass and all grass cover showed nearly identical relationships, whereas cover of cool-season grasses, forbs, and shrubs was not related to time since burning ($P > 0.185$) or winter or growing season precipitation ($P > 0.693$), very likely because of scarcity at this site (fig. 2).

Similar results were seen following burning in 2015. Burning increased cover of all grasses (+15 percent; $P_{\text{RAND}} = 0.013$) and warm-season grasses (+13 percent; $P_{\text{RAND}} = 0.025$) by 1 year postburn, but not cool-season grasses, forbs, or shrubs ($P_{\text{RAND}} < 0.209$).

Thinning and Chipping

Overstory canopy cover was reduced from more than 70 percent to less than 20 percent with each treatment (table 7). Vegetation responses differed among treatment combinations ($F_{3,5} = 36.2$; $P < 0.001$). Three years following treatment, the chip treatment did not differ ($P \geq 0.266$) from the control in any forage class (table 7). The thinning treatment had greater cover of all forage classes ($P \leq 0.032$) as compared to the chip treatment and control, with the exception of forb cover, which was similar between the thinning treatment and control ($P = 0.166$).

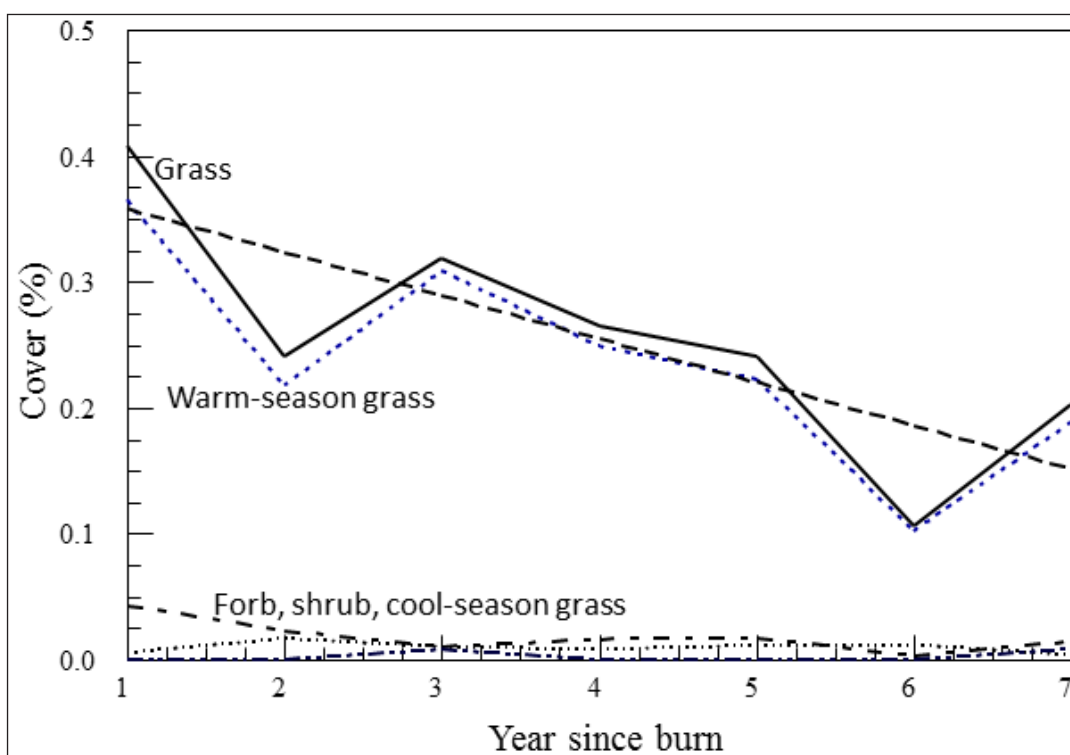


Figure 2—Trends in percent cover of all grasses, warm-season grasses, cool-season grasses, forbs, and shrubs following prescribed burning in late winter in the north-central New Mexico (NCNM) site. Also shown is the regression line for total grass cover, indicating an approximate 3-percent decline each year postburn.

Table 7—Percent cover (SE) of forage classes and preferred forage (P), and tree canopy cover, after 3 growing seasons on pinyon-juniper control sites, chip treatment sites, and thinned sites, central New Mexico. Values in parentheses are standard errors. Values followed by different letters differ at the $P < 0.001$ significance level.

	Control	Chip and leave or remove	Thin
Grass	2 (2)A	6 (3)A	27 (7)B
Grass (P)	<1	<1	8
Cool-season grass	0	0	1
Forb	6 (2)AB	4 (1)A	9 (2)B
Forb (P)	1	1	2
Shrub	4 (1)A	4 (1)A	12 (3)B
Shrub (P)	2	2	6
Tree (% CC)	76 (6)	19 (3)	17 (2)

Grazing

The SCNM1 site was included as an example of a site representative of light multipasture seasonal suitability and best pasture grazing. Differences between current and historical plant composition on this site showed patterns similar to those on the other, ungrazed sites (table 4). Cool-season grasses and preferred shrubs were particularly lacking relative to historical plant community composition.

DISCUSSION

The lack of understory associated with untreated PJ woodlands was reflected in the nutritional condition of deer and elk using these sites, as inclusion of untreated PJ woodlands in home ranges negatively affected nutritional condition of mule deer, and had no positive effect on elk (table 1). These results were similar to individual site results, which showed variable responses within species, although condition was never positively associated with untreated PJ in New Mexico (Bender et al. 2007b, 2012a, 2013; Halbritter and Bender 2011b; Hoenes 2008; see Bergman et al. 2014 for similar results in Colorado). Particularly rare in untreated PJ stands were cool-season grasses (12 percent of grass cover) and preferred browse (37 percent of shrub cover), forages that would be palatable and nutritious during the critical period of late gestation through early lactation (Short et al. 1977). Cool-season grasses were also often underrepresented in current understory communities compared to their

historical presence (table 4). Because cool-season grass was the only forage class that did not increase among treatments, preferred cool-season herbaceous native forages (or substitutes) may have to be reestablished in many PJ woodlands if forage attributes are to be enhanced significantly. This approach would particularly aid mule deer given their higher (relative to elk) diet quality requirements during the pre-monsoon late-gestation period. The generally positive responses of preferred warm-season grasses to a variety of treatments indicate that forage preferred by elk (Cook 2002; Short et al. 1977) can be significantly enhanced by many treatment options (tables 5 and 7).

Because of the current composition of most understory communities, enhancement of woody browse has the highest potential to increase the quality of PJ woodlands, especially for mule deer. Woody browse is the predominant item in mule deer diets (Boeker et al. 1972; Krausman et al. 1997; Short et al. 1977; Wakeling and Bender 2003) because it is available year-round with at least moderate dietary quality. Woody browse is also less susceptible to drought than herbaceous species, especially forbs, which is an important consideration for forage management in arid Southwestern woodlands (Bender 2012; Hoenes and Bender 2011). Although seasonal drought will always decrease productivity of cervids (Bender et al. 2007a, 2011; Hoenes 2008), the presence of sufficient relatively drought-tolerant browse can maintain survival of adults (Bender et al. 2012b; Hoenes and Bender 2011).

Similar to herbaceous forages (excluding preferred warm-season grasses, which are primarily an important forage component for elk) (Cook 2002; Short et al. 1977), preferred browse was a limited component of the overall shrub understory (table 4) and may not necessarily increase in cover in the short term with treatment of PJ woodlands (among treatments, effect = +5 percent [90-percent CI_{BOOTSTRAP} = 0–10 percent]).

Many treatments, particularly mechanical and herbicidal, have been used to increase browse in PJ woodlands (e.g., Bender et al. [2013]; Howard et al. [1987]; Miller et al. [2005]; Short et al. [1977]). However, Bender (2012) showed that mechanical removal of overstory PJ (to 11 percent CC) did not increase the cover of preferred browse as compared to unthinned PJ (62 percent CC) or PJ savanna (17 percent CC) on the CRLRC site. Despite this, body fat of mule deer was positively affected by increasing use of mechanically treated PJ sites during SSA (Bender et al. 2013). Although the mechanism behind this response was not identified, it is likely to have included both increased nutritional quality of individual plants (because of increased solar and other resource capture) and decreased successional status of individual plants (because of damage, i.e., crushing) associated with mechanical treatments (Cook and Harris 1950; Heffelfinger et al. 2006). On other sites, thinning increased protein levels of preferred shrubs by 1.2 to 1.5 times that of unthinned sites (L. Bender, unpublished data). Similarly, decreasing individual plant height and volume can increase browse availability and nutritional quality because of increased younger, less lignified, growth (Cook and Harris 1950; Heffelfinger et al. 2006; Hoenes and Bender 2011; Severson and Urness 1994).

Despite the positive relationship between mechanically treated PJ and body fat of mule deer, body fat levels of deer on the CRLRC were low (≤ 5.7 percent annually for lactating females, $< 1/3$ of what mule deer can accrue under optimal nutrition) (Bender et al. 2013). Thus, while mechanical clearing or thinning treatments may increase forage quality, increases in quantity of preferred forages are also needed to significantly enhance the quality of foraging environments, at least for mule deer. An increase in preferred browse was seen in other thinned sites such as the CNM site (table 7). But the magnitude of response (to 6 percent cover from 2 percent) was relatively low, and the total cover lower

than that seen on the CRLRC site (≤ 6 percent vs. 11 percent), although the species composition differed. It is likely that cover of preferred browse will increase with additional time since treatment, however, and further monitoring is needed to see the level ultimately attained.

Overstory reduction of PJ by herbiciding is not recommended as a management strategy for elk or deer. Broadcast application of tebuthiuron decreased live PJ canopy cover from more than 60 percent to less than 30 percent on CRLRC, but also resulted in a decrease in cover of preferred mule deer browse from 11 to 18 percent down to less than 3 percent (Bender 2012; Bender et al. 2013). Individual tree treatments also decreased preferred browse to less than 10 percent cover, because browse often is established adjacent to or under the canopy of PJ trees (Bender 2012; Bender et al. 2013).

Overstory thinning in combination with broadcast burning, or broadcast burning under previously thinned stands, has shown the most potential to enhance forage in PJ, more than doubling grass and forb cover in the first year following burning in the SCNM2 (table 5) and NCNM sites as well as several other sites in the Southwest (L. Bender, unpublished data). Although shrub cover was initially decreased, it typically recovered by the third year postburn. Further, some of the decrease in shrub cover was desirable, because it reflected loss of older, decadent canopies, as shrub volume decreased with burning. The younger growth from resprouting increased the forage quality of these shrubs significantly despite the lessened cover (see later discussion). Preferred browse followed these trends, but remained a small proportion of overall woody browse cover (table 5).

Vegetative protein levels were not assessed on the SCNM2 and NCNM sites, but burning increased protein levels of preferred grasses more than 1.6 times and shrubs more than 2 times that of unburned sites in early spring and summer following late-winter burns, although this effect was lessened by year 2 and absent by year 3 (L. Bender, unpublished data). Similar responses have been seen elsewhere (e.g., DeWitt and Derby 1955; Dills 1970; Einarsen 1946; Grelen and Epps 1967; Hobbs and Spowart 1984; Miller et al. 2005), and result from the freeing of N (Covington et al. 1991; Rau et al. 2009). Burned

sites also greened-up on average 14 days earlier than unburned across several southwestern sites (L. Bender, unpublished data). Fire speeds spring green-up, at least in the first year, because ash warms in the sun, creating a warmer microclimate for plants to germinate or initiate growth (Hobbs and Spowart 1984). Both of these responses increase diet quality of cervids during the critical late-gestation and early-lactation periods.

Burning also has nutritional benefits for elk and deer beyond those seen with mechanical treatments (which can also increase protein levels, as discussed earlier). Properly timed burns increase the quality of diets by removing dead plant material, allowing access to newly emergent forages in early phenological states (Hobbs and Spowart 1984; Severson and Urness 1994). Frequent fire can cause species shifts away from perennial warm-season grasses to forbs (Ford and McPherson 1996), which can also be preferentially increased in abundance by spring fires (Brewer and Platt 1994). Fire also tends to decrease crude fiber of plants, increasing digestibility. Fire therefore results in increased energy available, both within plants (because of increased digestibility) and overall (because of increased plant biomass) (Edwards et al. 2004). Many of these additional effects are more long-lived than the 1- to 2-year protein flush (Bender 2011), and benefit both deer and elk.

Similar to protein responses, live understory cover may be lost with increasing time since burning (fig. 2). Herbaceous responses were lost each year following burning, and lasted approximately 4 to 5 years in arid New Mexico woodlands (fig. 2, table 5) regardless of precipitation received during the winter or growing season. Thus, not only should burns be conducted at appropriate times to provide needed nutritional benefits (i.e., nutrient flush, earlier green-up, increased preferred forage cover) (Bender 2011) to cervids, but fire intervals should be timed to maintain the desired understory response(s) as well. Bender (2011, 2012) recommended that sites with moderate soil productivity be burned every 5 to 7 years, whereas sites should be burned on 10- to 15-year intervals if low productivity soils dominate or if increased woody browse is a management goal. Burning must be done with care, however, as data from several areas indicate that even light- to moderate-intensity (i.e., flame lengths ≤ 1 m) burns in thinned PJ can result in substantial mortality

of remaining trees, decreasing residual cover below desired levels. Even when the goal is increased woody browse, periodic burning is necessary to alter successional status of shrubs, increasing palatability and availability.

Additional treatments beyond properly timed prescribed burning may be needed to achieve high coverage of preferred herbaceous forage in understories. In such cases, forage seedings are frequently used to establish palatable forages (Bender 2012). For example, dryland alfalfa (*Medicago sativa* var. Ladak) was seeded in a thinned (to about 30 percent canopy cover) PJ stand at 5 to 10 lb/acre (6 to 12 kg/ha) in mid-summer following a late-winter burn near the SCNM2 site. By fall, cover of alfalfa was 12 percent (SE = 4) on the site, and increased to 20 percent (SE = 6) the next spring. Dryland alfalfa is a highly palatable legume of great nutritional quality, and hence is highly preferred by mule deer and elk. Because it is a cool-season species, it provides needed nutrition during late gestation and throughout lactation, as it maintains quality throughout the warm season. Dryland alfalfa plantings frequently fail because seeds are planted too deep (Bender 2012), and relatively shallow-rooted seedlings are susceptible to freeze-out during their first winter (L. Bender, unpublished data). However, once it or similar seedings are established, they can maintain high cover of extremely high quality forage that responds well to appropriate prescribed burning timing and intervals (Bender 2012).

In areas including livestock grazing, warm-season grazing is a tool that may favor increased cover of cool-season grasses and forbs (Chris Allison, New Mexico State University, Las Cruces, New Mexico, personal communication). Warm-season grazing may also increase the foraging efficiency of mule deer and elk due to the removal of coarse herbaceous cover (Severson and Urness 1994). Understory composition data from the SCNM1 site (table 4), however, suggested that even lightly grazed systems that include cool-season use may not maintain cover of cool-season grasses. More frequent burning of understories may decrease cover of warm-season grasses in favor of greater coverage by forbs, including cool-season species (Bender 2011; Ford and McPherson 1996; Launchbaugh 1964; Wright and Bailey 1980). Very short fire intervals (<3 years), however, can eliminate

most preferred woody shrubs (Bender 2011; Ford and McPherson 1996). Thus, this strategy should be limited to areas such as perennial grasslands that are proximate to PJ woodlands, and not the woodlands themselves, if mule deer habitat is a management goal. Burns should also be in late winter, as late-spring burns may decrease cover of cool-season grasses (Hart 1990).

Other treatments, such as chipping and other mastication treatments, may show little understory response if biomass is left on the ground, particularly in hotter and more arid PJ woodlands (table 7). Juniper wood (240:1) and leaves (55:1) have high carbon (C):N ratios (Bates 1996; Miller et al. 2005), and Bates (1996) found that C:N ratios in soil increased by 32 to 43 percent where western juniper (*J. occidentalis*) slash was deposited. High C:N ratio litter can cause soil microbes to immobilize available N (Schimel and Firestone 1989), slowing decomposition (Miller et al. 1979). Consequently, chip treatments may have lower total nitrogen, phosphorus, organic phosphorus, calcium, and potassium concentrations than untreated, lop-and-scatter, or broadcast burn treatments (Waldrop et al. 2003).

Chipping can also result in lower forest floor enzyme activity, which may slow decomposition rates up to 60 percent (Waldrop et al. 2003). Other work suggests that though chipping and mastication treatments may not be detrimental to N mineralization in the short term, slow decomposition of masticated material and widening C:N ratios may be a concern in the long term (Gottfried et al. 2009). Because decomposition can be influenced by temperature and moisture (Klopatek et al. 1995), these dynamics may be a particular challenge in the hotter, more arid PJ woodlands of New Mexico, west Texas, and northern Mexico. Studies showing positive understory responses to chipping or mastication were in cooler, moister sites, or regions with different precipitation patterns than the CNM site, which has a high (>50 percent) monsoonal index (e.g., Klopatek et al. 1995; Ross et al. 2012).

Leaching of secondary plant compounds from residual chip biomass may also inhibit understory responses. Ramsey (1989) found that phenolic leachates did not affect emergence, but generally inhibited growth, such as root and shoot biomass and tiller numbers, of several grass species (Ramsey 1989). In contrast, leaving

biomass may promote plant growth by increasing soil moisture because of the mulch effect (Benson 2006; Bergman et al. 2014; Brockway et al. 2002). However, this also makes follow-up burning more difficult, particularly in late winter or early spring, because lower chip layers and adjacent soil remain moist. For example, I was unable to burn chipped areas in the CNM site 1 year and 2 years post-chipping during late winter because of saturation of chips in contact with the ground.

A further concern with this treatment is that chipping or mastication treatments may decrease soil temperature in spring (Brockway et al. 2002; Waldrop et al. 2003). This effect is particularly relevant to the annual energetic cycle of mule deer and elk, because it would result in later green-up, delaying availability of early phenology forage during late gestation when nutritional needs increase dramatically (Cook et al. 2004; Verme and Ullrey 1984; Wakeling and Bender 2003).

Most documented responses of preferred forages (other than warm-season grasses) to treatments have been relatively small (tables 5, 7). However, even a small change in plant community composition may result in a significant change in elk or deer habitat quality. For example, it takes only an approximately 10-percent change in diet quality to significantly increase elk productivity (Cook et al. 2004). Diet and diet quality data from pronghorn (*Antilocapra americana*) suggest that even a relatively small increase—5 to 7 percent—in preferred forages may be able to produce such a change in diet quality (L. Bender, unpublished data). Similar changes were seen with cattle diet quality (Holechek and Vavra 1983; McGinty et al. 1983). However, further work is needed to identify what level of change in preferred forage can significantly increase diet quality.

For treatments that remove woodland overstory and horizontal cover, care must be taken to maintain sufficient canopy cover or stem density to meet security and thermal requirements. Inclusion of untreated PJ woodlands was negatively related to home range size of mule deer, indicating that presence of denser PJ increased habitat quality. This beneficial effect was probably because of cover attributes. Previously, Bender et al. (2013) found that the majority of mule deer activity was near unthinned PJ woodlands on the CRLRC. Elk decreased diurnal activities in the Chaco area when

mean high temperatures exceeded their thermal tolerance, with juniper trees being an important diurnal bedding site (Bender et al. 2012a). For example, radio-collared elk in the Oscura site significantly decreased use of portions of home ranges on Chupadera Mesa following prescribed burning that resulted in canopy loss of 70 to more than 90 percent and near elimination of shrubs (at least in the near term). Observations of mule deer in these areas also decreased by more than 90 percent after these burns (L. Bender, personal observation). Additionally, some cool-season grasses are positively correlated with overstory PJ canopy cover (Schott and Pieper 1985), so too much canopy reduction may negatively impact some key foods as well.

In PJ-dominated sites with few other cover options, Bender (2012) recommended a “Rule of 4s” to maintain adequate cover for mule deer when managers are treating PJ stands. This strategy recommends that for each section of PJ (which approximates the average home range size of female mule deer in New Mexico) (Bender et al. 2007a, 2011; Hoenes 2008), one-fourth should remain in untreated (i.e., PJ canopy cover > 60 percent; if existing canopy cover is <60 percent, these areas should be allowed to develop to >60 percent canopy cover) patches of 40 acres (16 ha) or more, one-fourth should be thinned to no less than 30 percent PJ canopy cover, and the remaining two-fourths should be thinned to no less than 10 to 15 percent canopy cover. This would create a mosaic in which one-fourth of each home range is in untreated PJ ideal for security cover, one-fourth is in a structural state that provides both minimal cover requirements and increased forage (30 percent canopy cover), and one-half is in a structural state that provides scattered thermal cover and optimal foraging attributes (10–15 percent canopy cover). At a relative fine scale (i.e., ≤40-acre treatments), these treatment allocations would result in 100 percent of the landscape being suitable for mule deer (Bender 2012).

While designed for the security preferences and needs of mule deer, this strategy also provides abundant summer thermal cover for elk, and probably does not decrease the nutritional carrying capacity of elk range

significantly. In areas of high disturbance, the extra security cover beyond the likely requirements of elk may be useful to avoid unnecessary movements and thus help maintain energy balance and productivity (Shively et al. 2005). Further, distribution of patches or stands should be allowed to shift across the landscape over time as succession, disturbance, or management treatments change the overstory PJ cover, thus maintaining a dynamic landscape.

The potential for enhancing extant PJ woodlands for mule deer and elk is a viable and relevant opportunity for managers of public or private land. These woodlands include almost 10 percent of the total area of New Mexico, making PJ woodlands the most abundant habitat type capable of meeting all forage and cover requirements for mule deer and elk in the State. Though current habitat quality of many woodlands may be relatively low, this may not have been true historically. In several of my study areas, agency and landowner survey data indicate much higher mule deer densities in PJ woodlands historically. For example, mule deer densities in the San Andres and Oscura Mountains were approximately 5 to 20 times greater during the 1970s and 1980s as compared to the early 2000s (Bender et al. 2012b, 2017; White Sands Missile Range: Bender, unpublished data). Similar trends have occurred in the Corona and NCNM sites (Bender et al. 2007).

Density can be a misleading index of habitat quality, as mentioned earlier (Van Horne 1983), but differences of these magnitudes probably at least partially reflect significant changes in habitat quality. While many site-specific factors can affect local deer and elk populations (e.g., disease, drought, overharvest) (Bender et al. 2007), one constant among these sites has been a lack of disturbance appropriate for rejuvenating woody forage, especially fire. Currently, deer and elk are productive in PJ habitats in New Mexico, as reviewed earlier, but not as productive as they can be. Public and private land managers who desire to increase the quality of mule deer and elk habitat in PJ woodlands in New Mexico thus have historical context indicating that habitat quality can very likely be improved substantially.

MANAGEMENT IMPLICATIONS

1. Needs and opportunity exist to improve the quality of PJ woodlands as foraging habitat for elk and mule deer.
 - Increases in understory, however, do not necessarily equal increases in food.
 - It is likely that relatively small increases in preferred forages can have a significant nutritional benefit.
2. Increased nutrients (protein, digestible energy) need to be provided during late gestation, or beginning in approximately mid-April.
 - Broadcast burning during late winter (March) appears to be the single most effective tool to achieve this goal. Burns can be done following thinning with lop-and-scatter of slash, in previously thinned stands, or in savannas.
 - Agency emphasis on “natural” lightning-driven fire regimes cannot provide a nutrient flush at appropriate times.
 - Piling and burning of slash should be avoided unless piling sites are to be seeded to a highly preferred cool-season forage (i.e., a food plot).
3. Cool-season forages should be encouraged because they provide critical early-phenology forage during late gestation through early lactation.
 - Late-winter burning and warm-season livestock grazing may help increase this component, but more information is needed.
 - Plantings are a further alternative.
4. “Xeriscaping” foraging habitats by providing an abundant and diverse browse component can mitigate the effects of drought in many cases.
 - This is particularly important for habitat for mule deer because their greater diet selectivity emphasizes forbs from herbaceous communities, which are more susceptible to drought.
5. Methods to assess responses of mule deer and elk to management treatments must be closely related to the fundamental influence of habitat.
 - Individual condition, juvenile growth, and juvenile fecundity are appropriate direct metrics.
 - Measures of forage quality of preferred forages and diet quality of mule deer and elk are appropriate indirect methods.
 - Removed metrics (i.e., adult survival, population rate of increase) that can be confounded by numerous environmental and human-caused influences should not be used.
 - Remotely sensed metrics often used to infer forage quality such as the normalized difference vegetation index are not strongly related to nutritional condition of deer or elk and should be avoided unless better indices are developed (Caltrider 2012).
6. Mule deer and elk are different herbivores with differing nutritional and cover requirements.
 - High-quality mule deer habitat will benefit elk.
 - High-quality elk habitat may not benefit mule deer.
7. Management treatments should retain adequate cover for mule deer and elk.
 - Mule deer are likely to require significant areas of high density, high canopy cover for security.
 - Elk require much less cover, primarily for summer thermal constraints.
 - Landscape prescriptions that balance forage and cover such as the “Rule of 4’s” can result in up to 100 percent of the landscape being suitable for mule deer and elk.
 - Cover and forage patches should be allowed to shift across the landscape over time in response to planned or unplanned perturbations, thereby maintaining a dynamic landscape. This subjects all forage to periodic quality-enhancing disturbance over time.

ACKNOWLEDGMENTS

Numerous individuals reviewed and contributed to this work through discussion, on-the-ground treatments, or other field work, including Dave Anderson, Chris Allison, Allen Darrow, Cervidnut, Jon Boren, Heather Halbritter, Brock Hoenes, James Lucero, Jessica Piasecke, Pat Mathis, Patrick Morrow, Mike Rearden, Cristina Rodden, Kevin Rodden, Mara Weisenberger, and the many other owners and managers of ranches in New Mexico, Texas, and Mexico.

Funding was provided by the Santa Fe Trail Mule Deer Adaptive Management Partners; ranching partners in New Mexico, Texas, and Mexico; U.S. Fish and Wildlife Service; U.S. Army; U.S. Bureau of Land Management; USDA Forest Service; National Park Service; New Mexico Department of Game and Fish; Rocky Mountain Elk Foundation; NMSU Cooperative Extension Service; and NMSU Department of Animal and Range Sciences. Activities for this project were in accordance with New Mexico State University IACUC permits 2001-022, 2002-027, 2003-013, 2003-023, 2005-023, and 2009-025.

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