Dear Ms. Little,

Please accept these additional comments and an attachment in pdf format from me on behalf of the Alliance for the Wild Rockies, Native Ecosystems Council and Montana Ecosystems Defense Council.

Please find attached a paper titled, "Characteristics of browsed aspen forests following wildfire and implications for management: a case study" by James R. Biggs, Dawn M. VanLeeuwen, Jerry L. Holechek, Sherri L. Sherwood and Raul Valdez.

They found that high severity fires are the best way to regenerate aspen.

They also found that elk browsing usually occurs in winter when the aspen are dormant but cattle browse them in summer. That's an important difference and shows that cattle grazing have a much bigger impact on the decline of aspen.

Thank you for your attention to these concerns.
Sincerely yours,

Mike Garrity
Alliance for the Wild Rockies
P.O. Box 505
Helena, Montana 59624
406-459-5936

And on behalf of:
Sara Johnson Ph.D.
Native Ecosystems Council
P.O. Box 125 Willow Creek, MT 59760

and for

Steve Kelly
Montana Ecosystems Defense Council
P.O. Box 4641 Bozeman
MT 59772
Tel: (406) 586-4421
Characteristics of browsed aspen forests following wildfire and implications for management: a case study

James R. Biggs,¹ Dawn M. VanLeeuwen,² Jerry L. Holechek,³ Sherri L. Sherwood,⁴ and Raul Valdez⁵

¹Visiting Assistant Professor, Forestry Dept., New Mexico Highlands University, Las Vegas, New Mexico 87701, USA
²Professor, Agricultural Biometric Service, New Mexico State University, Las Cruces, New Mexico 88003, USA
³Professor, Dept. Animal and Range Sciences, New Mexico State University, Las Cruces, New Mexico 88003, USA
⁴Environmental Technician, Los Alamos National Laboratory, Los Alamos, New Mexico, 87545
⁵Professor, Dept. Fishery and Wildlife Sciences, New Mexico State University, Las Cruces, New Mexico 88003, USA

Correspondence address. Dept. of NRM, New Mexico Highlands University, PO Box 9000, Las Vegas, New Mexico 87701, USA. Tel: (505) 454-3320; Fax: (505) 454-3103; E-mail: jrbiggs@nmhu.edu

Running title: Characteristics of browsed aspen forests
Abstract

Aims

Recognizing ungulate browsing thresholds between viable and declining aspen (*Populus tremuloides* Michx.) stands are critical to ensuring long-term persistence of this biologically important plant community. Studies have shown declines in vigor and regeneration when as few as 30% of current annual twigs are browsed while other studies have shown higher limits. Although the effects of ungulate herbivory are of concern in aspen forests, few studies have assessed browsing effects following wildfire and few criteria exist for determining potential effects of ungulate browsing on aspen forests following wildfire. We evaluated the effects of ungulate abundance and foraging intensity on regenerating aspen 1 to 6 years post-fire and assessed the use of abundance and foraging intensity indicators in predicting impacts to regenerating aspen. 

**Rocky Mountain elk (Cervus elaphus nelson)** was the primary ungulate in the study area.

Methods

The study area was located within the 17,500 ha Cerro Grande Fire burn area in the Jemez Mountains, New Mexico, USA. We used percent aspen twigs browsed and pellet-group counts to evaluate relationships between these indicators and aspen patch structure (height, size). We collected data in **randomly generated** 3 x 33 m plots 5-6 years post-fire. We also established 4 fenced exclosures (25 x 55 m and 3.3 m in height) with paired unfenced plots 1 year post-fire to monitor aspen regeneration. Each spring, we recorded percent browsed twigs from the previous fall through the early spring period which coincided with the highest ungulate use period within the study
area. We assessed associations between percent twigs browsed and pellet-group density and patch size and height of aspen using Spearman’s correlation coefficients.

**Important Findings**

Mean percent twigs browsed and ungulate pellet-group density across the burn area was ≤ 31% and 1 pellet-group 100 sq m⁻¹, respectively. Patch size and height decreased with increasing browsing and pellet-group density 5-6 years post-fire. However, mean aspen heights were approaching or exceeding a minimum browsing level of 2 m and, therefore, ungulate browsing did not appear sufficient to cause significant impacts to aspen across the burn area. We observed a positive correlation between pellet-group density and twig browsing suggesting that one or both measures could be used to assess potential effects of browsing on regenerating aspen following fire.

**KeyWords:** ungulate, regeneration, transitory range, pellet-group, foraging

**INTRODUCTION**

The effects of herbivory in determining forest community composition can be as significant as episodic agents and more important than site and other disturbance factors (Riggs *et al.* 2000). Aspen and shrub recruitment has been negatively associated with ungulate herbivory (Augustine and McNaughton 2004; Bartos *et al.* 1994; Kay 2001a, b; Romme *et al.* 2005; Wambolt *et al.* 1998; White *et al.* 1998). The effects of browsing on western U.S. forests, including aspen, have been of particular concern due to the typically high diversity of vertebrates and invertebrates inhabiting these plant communities (Baker *et al.* 1997; Bailey and Whitham 2003; Kay and Bartos 2000;
Riggs et al. 2000; Schoenecker et al. 2004; Kleintjes et al. 2007; others). Much of the impact to aspen has been attributed to elk but although impacts of elk browsing and grazing on vegetation in relatively undisturbed plant communities are well documented (Baker et al. 1997; Kay 1993, 1995; Ripple and Beschta 2012; Ripple and Larsen 2000; Romme et al. 1995), fewer studies on aspen browsing have occurred immediately following large-scale fires. Wan et al. (2014) reported that aspen regeneration was positively related to fire size and high burn severity and recruitment was strongly influenced by fire severity, pre-fire aspen cover area, and stand density. Wan et al. (2014) also found that interaction of large ungulate browsing and fire size played a role in aspen recruitment. Although mature aspen stands can experience high production and rapid growth of aspen shoots immediately following fire (Larsen and Ripple 2005; Romme et al. 2005; Schier et al. 1985), ungulate browsing can be a major contributing factor suppressing aspen growth (Romme et al. 2005).

Incorporating an adaptive management strategy for aspen should account for varying levels of ungulate browsing coupled with the expected warming and drying trends due to a changing climate (O’Brien et al. 2010). General guidelines have been developed for managing elk (Cervus elaphus L.) abundance with respect to effects specifically on quaking aspen (Populus tremuloides Michx.) in elk summer ranges of the central and northern Rocky Mountains (Rogers et al. 2015; Weisberg and Coughenour 2003; White 2001; White et al. 2003). In some cases, general categories of browsing have been used to assess effects on aspen. For example, Kaye et al. (2005) assessed browsing effects on aspen in the central Rocky Mountains based on either high browsing (visible signs of use on sprouts and stems) or low browsing (no visible signs of use) by ungulates and Rogers et al. (2015) used both subjective (stand type,
number of vertical aspen layers, percent aspen cover, recent disturbance) and objective (including tree composition, regeneration and recruitment, browse level, herbivore use) measures to assess recruitment of aspen in stands in Utah experiencing ungulate browsing. Identifying relationships between ungulate density and impacts on plant communities is critical in setting management objectives related to population levels (Augustine and McNaughton 1998). The combined use of indirect and direct measurements of plant use may provide a measure of browsing intensity as it relates to effects on plant community structure (White et al. 2003).

The May 2000 Cerro Grande Fire (CGF) in north-central New Mexico was a mixed severity fire that resulted in high mortality of conifer species. Area natural resource managers expressed concern that elk (the primary ungulate species in this area) would negatively impact aspen and shrub recruitment in areas previously dominated by conifer species. Significant impacts to woody species such as aspen, oak (Quercus spp.), chokecherry (Prunus virginiana L.), gooseberry (Ribes spp.), willow (Salix spp.) and others have been observed in areas nearby the Cerro Grande Fire burn that were exposed to moderate to heavy elk use. In some cases, the loss of aspen in this region has had negative impacts on the diversity of organisms associated with these communities (Kleintjes et al. 2007).

The objectives of this study were to: 1) evaluate the effects of large ungulate population abundance and foraging intensity on aspen patch structure in a recently burned area and, 2) determine if individual indicators of ungulate abundance (pellet group counts) and foraging intensity (browsed twigs,) could be used to assess impacts to regenerating aspen. We defined a patch as overlapping cover of aspen without
breaks in the canopy regardless of how many individual sucker or sapling stems were present.

MATERIALS and METHODS

Study area

The study area was located within the CGF burn area in the eastern Jemez Mountains, north-central New Mexico, USA (Fig. 1). The CGF was ignited as a prescribed fire in May 2000 but grew out of control due to high winds, eventually burning about 17,500 ha. Elevations range from about 1,620 m to 3,500 m within and immediately adjacent to the burn area. Annual precipitation ranges from about 46 cm to 76 cm and is strongly influenced by the topography of this area (Bowen 1990). Most precipitation occurs in the form of rain in the summer months but annual snowfall can exceed 127 cm per year in the higher elevations. Elk were reintroduced into the Jemez Mountains from the late 1940s through the 1960s but did not significantly increase their numbers until the 1980s and 1990s. This increase was, in large part, attributed to the 1977 La Mesa Fire that burned 6,178 ha of forested areas in the eastern Jemez Mountains resulting in large areas of grass and other early successional plant species favored by elk (Allen 1996). The fire occurred in an area that generally separates the higher elk summer range of the Valles Caldera National Preserve (VCNP) and the lower lying winter range of the Pajarito Plateau (referred to hereafter as transitory range). The majority of use of the CGF by elk primarily occurs during seasonal movement periods and winter with the degree of use almost completely dependent on snow depth on the transitory range and the summer range. Although extensive aerial surveys adjacent to the burn area have been conducted by The New Mexico Dept. of Game and Fish, no reliable estimates of
population size is available for the burn area. An extensive effort to assess elk movement and distribution patterns within the study area occurred from 2001 through 2004 using animals equipped with GPS collars (Biggs et al. 2010). Results of this monitoring effort suggested that the southern portion of the study area received a disproportionately higher level of use relative to the overall burned area and that elk use of the burned area did not change significantly during that period. The study concluded that elk occurring in the burn area is expected to be highly concentrated during seasonal movements between winter and summer range with considerably less use over extended seasonal periods.

Of the total area burned by the CGF, about 5,800 ha were classified as high severity burn, 1,390 ha moderate severity burn, and 10,100 ha low severity burn. Regardless of the severity level, in most areas, midstory (shrubs) and understory vegetation burned. Following the fire, an intensive effort was made to stabilize soil loss in the moderate and high severity burned areas through seeding, raking, contour felling of trees, aerial hydromulching, and other techniques. The seed mixture consisted of 30% each of annual ryegrass (Lolium perenne L.), mountain brome (Bromus marginatus Nees ex Steud.), and slender wheatgrass (Elymus trachycaulus Gould ex Shinners) and 10% barley (Hordeum vulgare L.) (Burned Area Emergency Rehabilitation Team, unpublished report, U.S. Forest Service). The mid and higher elevations (> 1,980 m) of the study area were characterized by plant species associated with ponderosa pine (Pinus ponderosa Douglas ex C.Lawson), mixed conifer (Pseudotsuga menziesii (Mirb.) Franco, Abies concolor (Gordon) Lindl. Ex Hildebr.), and spruce-fir (Picea engelmannii Parry ex Engelm.-Abies lasiocarpa (Hook.) Nutt.) forests. Quaking aspen also were associated with these cover types and made up about 8% of the cover in the burned area.
immediately following the fire, primarily above 2,220 m. Common shrubs and graminoïds associated with the burned area included chokecherry, locust (*Robinia neomexicana* A.Gray), maple (*Acer glabrum* Torr.), oak, sagebrush (*Artemisia* spp.), bluegrass (*Poa pratensis* L.), brome grass (*Bromus* spp.), danthonia (*Danthonia parryi* Scribn.), fescue (*Festuca* spp.), Junegrass (*Koeleria nitida* Nutt.), mutton grass (*Poa fendleriana* (Steud.) Vasey), and sedge (*Carex* spp.).

**Data collection**

We randomly generated 85–3 × 33 m plots within the southern and northern portions of the burned area using ArcGIS of the Geographical Information System (ESRI, v8.3). We did not specifically identify aspen stands for plot establishment so not all plots contained aspen. *Of these 85 plots, 17-20 contained aspen in 2005 and 2006.* These areas were generally similar vegetatively prior to the fire (ponderosa pine and mixed conifer overstory). The central portion of the burned area was not sampled because of steep, relatively inaccessible terrain and due to low pre-fire use by radio collared elk (Biggs *et al.* 2010). All plots were placed in areas where understory, midstory, and overstory vegetation had experienced 100% mortality from the fire. We avoided areas of high burn severity where re-generating vegetation was absent. In no case was there more than one plot in any individual aspen patch or stand.

In addition to the *randomly generated* plots, we *identified 4 aspen stands in 2001* in the southern portion of the burned area where the highest concentration of elk use occurred within the burned area and established fenced exclosures (25 × 55 m and 3.3 m in height) with paired unfenced plots during fall 2001 within *those* stands. We used the fenced exclosures as controls for no browsing or grazing. We selected the *aspen stands*
for placement of exclosures based on accessibility, dominant stand type, similar topographic characteristics, burn severity, known elk use patterns, and similarities in plant species composition. Only aspen stands that experienced 100% mortality were selected for placement of the exclosures and their paired unfenced plots. Fence mesh size allowed small and medium size wildlife species access into exclosures but prevented larger species such as elk and deer (*Odocoileus hemionus*) from entering. We randomly placed the paired unfenced plots within adjacent aspen stands at least 10 m from each fenced exclosure in an attempt to minimize vegetation effects from potential behavioral responses by animals to the fences (i.e., animals moving along fence line when traversing the area) while ensuring the unfenced plots were close enough to the fenced exclosure to retain similar site and vegetation characteristics. We established a 3 m x 33 m plot within each exclosure and paired plot for the collection of data.

We measured aspen patch size and height, and counted the total number of stems within each plot. Multiple aspen patches could occur within the 3 m x 33 m plot so we identified and measured the size and maximum height of each individual aspen patch. All patch measurements were taken during summer 2005 and 2006. We defined a “patch” as any overlapping aspen canopy cover regardless of the total number of sucker or sapling stems, assuming the stems were living tissue and at least 10 cm in height. Aspen suckers were defined as < 2.0 m tall, the height to which an elk is expected to browse, and saplings as 2.0 – 3.5 m tall. Any stem greater than 3.5 m was considered tree form. Aspen patch size (sq m) was based on width and length measurements of each individual patch occurring within the plot. We determined patch cover by first measuring the longest length of the patch (X axis) followed by measuring the width perpendicular to the first measurement (Y axis). In most cases, this
resulted in an overestimation of the patch cover unless the patch was virtually square in shape with no gaps between cover. In cases where the patch shape would have resulted in gross errors in calculation of cover (i.e. L- or U-shaped), we took two separate measures and combined them to more accurately reflect the true cover value. In cases where openings occurred within a patch, the size of the opening was estimated and subtracted from the total patch cover estimate. If the patch size was greater than the plot size, then we recorded the patch size as a value equal to the plot size (99 sq m).

Maximum patch height was based on the height of the tallest stem (m) in each individual patch within the plot. Height was measured to the nearest centimeter using a PVC pole delineated in centimeters. We also recorded the density of all stems within each patch by counting the number of live sucker and sapling stems > 10 cm tall within each individual patch. Regenerating aspen frequently occur in clumps of at least two stems and although only one stem is expected to survive to tree form (Larsen and Ripple 2005), we included all live stems in our counts. Immediately following establishment of the exclosures and paired unfenced plot, we counted all live stems within each patch located in the 3 m x 33 m plot. All stems were less than 2.0 m tall at the time of initial sampling.

We collected data on the percent of browsed aspen twigs at all plots in early spring 2005 and 2006. Browsing level was used as an indicator of forage use intensity. Each spring, we recorded percent browsed twigs since the previous fall on the tallest stem of each patch. We examined all lateral twigs and terminal twigs up to 2 m in height on the stem (the height up to which elk browsing was expected to occur) for browsing. We measured lateral twigs because repeated use of these twigs may cause a decrease in the size and nutritious quality of the aspen shoot (van Beest et al. 2010).
We used pellet-group density as an indicator of animal abundance (Neff 1968; White and Eberhardt 1980) and related it to aspen patch structure. A pellet-group was defined as ≥ 5 pellets compressed together (Neff 1968; Jenkins and Manly 2008). We estimated pellet-group density within each plot each spring and again at the end of the summer. We cleared transects of all pellets immediately following data collection in spring and late summer. We did not include an analysis of summer pellet groups due to very low numbers and focused on the primary seasons of use (fall through early spring).

Data analyses

We analyzed data from all randomly placed plots containing aspen, including the 4 plots adjacent to the fenced exclosures. We assessed correlations between previous percent twigs browsed and the following summer patch structure by plot measured in August using Spearman’s rank correlation coefficients, a non-parametric test (SAS v9.13). We repeated this analysis for fall-winter-early spring pellet-group density and summer patch structure. We could not match the exact size of individual patches measured during summer to the same individual patches measured in April due to the absence of leaves during the April sampling effort. The absence of leaves did not allow for clear delineations of patch boundaries potentially leading to incorrect patch size estimates. Therefore, we used mean percent twigs browsed by plot to relate previous fall-spring browsing to summer patch size and height. We estimated pellet-group density for each plot based on the mean number of pellet-groups 100 sq m⁻¹. We assessed the correlation between the percent of twigs browsed and pellet-group density to determine if one measure could be used as a surrogate for the other using Spearman’s rank correlation coefficients. We compared differences in aspen characteristics between
the 4 exclosures and their paired unfenced plots using paired t-tests. **We also**
compared stem counts within exclosures and unfenced areas by year and between
years using t-tests. Probability values $p \leq 0.05$ are considered statistically significant.

RESULTS

**Ungulate occurrence and initial aspen stem density**

Mule deer (*Odocoileus hemionus*) occurred in the burn area but made up < 7% of the
total number of pellet groups found in plots monitored from 2002 to 2006 (elk made up
the remainder). Additionally, elk pellets were recorded in 4x more plots than deer
pellets. Therefore, we did not expect deer to have a significant contribution to
herbivory levels recorded in the study area. Cattle were not present in the southern half
of the study area and little to no use occurred in the northern portion. In no case were
cattle feces found within or near any plot.

During an initial estimate of aspen within the exclosures and paired plots in 2001,
most stems were less than 2.0 m and, therefore, considered suckers. **The number of
stems in the fenced exclosures did not differ from the unfenced plots in 2001 ($p = 0.742$).**
The average number of suckers for the exclosures and the unfenced paired
plots in 2001 were 44,066 ha$^{-1}$ and 51,364 ha$^{-1}$, respectively, and the average number of
aspen suckers for all plots combined was 47,715 ha$^{-1}$ (Table 1).

**Ungulate browsing and abundance and aspen patch size and height**

In 2005, patch height and twig browsing were negatively correlated ($r = -0.563$, Table 2,
**Fig. 2**). Patch size of aspen was similarly negatively correlated to twig browsing in
2005 and 2006 but only statistically significant in 2006 ($r = -0.451$, Table 2, **Fig. 3**).
The mean patch size and height in 2005 for all plots was 7.9 sq m (se = 4.17) and 1.78 m (se = 0.24), respectively, and the mean percent twigs browsed was 14.11 (se = 4.99). In 2006, the mean patch size and height was 5.11 sq m (se = 1.49) and 1.90 m (0.25), respectively, and the mean percent twigs browsed was 30.64 (se = 5.98).

Aspen patch size was not correlated to pellet-group density in 2005 but there was a negative correlation between patch size and pellet-group density in 2006 ($r = -0.682$, Table 2, Fig. 4). There also was no correlation between patch height and pellet-group density in 2005 but there was a negative correlation in 2006 ($r = -0.582$, Table 2, Fig. 5). Pellet-group density and percent aspen twigs browsed were positively correlated in both 2005 and 2006 ($r = 0.619$ and $0.474$, respectively, Table 2, Figs. 6 and 7). The mean pellet-group density per 100 m$^2$ for all plots in 2005 was 4.35 (se = 1.48) and in 2006, the mean density was 1.09 (se = 0.30).

**Exclosure and paired unfenced plots**

There was no evidence to suggest the number of stems, height, or size of aspen patches differed between exclosures and their paired unfenced plots in 2005 ($n = 4$, $p = 0.524$, 0.106, and 0.443, respectively). There also were no differences between exclosures and unfenced plots in the number of aspen stems, patch height, or patch size in 2006 ($n = 4$, $p = 0.417$, 0.352, and 0.436, respectively). The reduction in stems within both the exclosures and unfenced plots decreased substantially from 2001 to 2006 (Table 1) and the decrease was nearly significant when comparing the unfenced areas in 2001 to 2005 and 2006 ($p = 0.061$ and 0.063, respectively). The lack of detectable statistical difference is likely due to the small sample size ($n = 4$) and high variability in stem counts among controls and exclosures.
DISCUSSION

The objectives of this study focused on assessing relationships between large ungulate abundance, foraging intensity, and aspen patch characteristics within a recently burned area. We predicted that patch size and height would decrease with increasing levels of browsing intensity within 5-6 years post-fire. Although aspen regeneration has been reported to be strong following large-scale fires such as what we observed, significantly lower regeneration can occur in browsed stands compared to unbrowsed stands (Smith et al. 2011). Our results suggested that as browsing on aspen increased across the burned area, patch height and size decreased 5-6 years post-fire. However, average patch heights generally exceeded the maximum expected browsing level of ungulates (2 m) in our study area. Also, we observed no differences between fenced exclosures and their paired unfenced plots with respect to patch height and size of aspen despite two of the exclosures being located in an area where elk were known to concentrate. In most unfenced plots, percent twigs browsed were less than 14% during both years of study reflecting low levels of ungulate browsing despite this area typically experiencing a higher concentration of ungulates relative to other parts of the study area. Our results suggest that the number of elk, the primary ungulate species occurring within the burned area, was not high enough to cause overall negative impacts to regenerating aspen stands across the burn area during our study period. The lack of significant effects on regenerating aspen could also be due to the large densities of suckers produced after the fire relative to the number of ungulates present thereby swamping the effects of browsing. Wan et al. (2014) reported a net loss in aspen cover...
for fires less than 700 ha in areas prone to aspen regeneration whereas aspen cover increased for many fires greater than 700 ha. **The increase of aspen cover in larger burned areas was presumably due to a greater number of aspen stems surviving browsing and reaching mature tree form.** The CGF burned over 5,000 ha of mixed conifer forest in the southern part of the study area where aspen regeneration was occurring. We also predicted that aspen patch size and height would be inversely related to ungulate abundance and found that patch size and height decreased with increasing pellet-group density in 2006. Additionally, we observed a positive correlation between pellet-group density and twig browsing in both 2005 and 2006.

The **maximum** mean percent of twigs browsed across our study area 5-6 years post-fire was **31% in 2006**. Our browsing estimates included both terminal and lateral twigs so the percent of terminal twigs browsed would likely be less than 31%. However, we observed significant negative correlations between browsing and aspen height in 2005 and patch size in 2006 suggesting that browsing levels could be approaching a threshold where significant effects on regenerating aspen can occur. Thresholds between viable and declining aspen stands have been observed to occur as low as when approximately 30% of the plant’s current annual twigs were browsed (Olmstead 1979) but Rogers *et al.* (2015) reported that the majority of stands in their study area exceeding 20% summer browsing were not experiencing sustainable regeneration. Most browsing in our study area took place during primarily the inactive plant growth period outside of the summer and could likely sustain higher browsing levels. In addition to the inverse relationships we observed between browsing and aspen patch structure, we found that pellet-group density was inversely related to patch size and height in 2006. Prior studies of aspen have reported median aspen stem heights of 1.5 – 1.8 m six years post-
fire and greater than 3.1 m nine years post-fire (Kilpatrick et al. 2003). Regenerating aspen sucker and sapling heights > 2 m were common in our study area six years post-fire. Aspen subjected to moderate and high ungulate browsing would not be expected to produce stems > 2 m in height (Kay and Bartos 2000). Ungulate populations in our study area appeared to be low enough relative to the amount of aspen regeneration in the burned area that overall impacts to aspen were below the threshold where stand regeneration would be impacted negatively. Although our findings suggest that increasing ungulate browsing and abundance (based on pellet-group density) was correlated to decreasing patch size and height of regenerating aspen following fire, these effects did not appear to be limiting overall aspen development as evidenced by mean sapling heights across our study area approaching that of the expected maximum browsing height of 2 m. Additionally, we observed an increase in the mean aspen patch height for all plots from 1.78 m in 2005 to 1.90 m in 2006 when while also recording a browsing level of 31\% in 2006.

Expectedly, high levels of browsing (> 50\%) have been found to result in very low levels of aspen recruitment (Rogers and Mittanck 2014; Zeigenfuss et al. 2008) and stem densities of at least 500 ha\(^{-1}\) (O’Brien et al. 2010) and as much as 1250 ha\(^{-1}\) (Bartos and Campbell 1998) have been suggested as a minimum to maintain aspen stands. We recorded considerably higher densities of regenerating aspen suckers and saplings within our study area six years post-fire suggesting these aspen stands are sufficiently regenerating. However, the total number of regenerating sucker and sapling stems did decrease substantially from 2001 to 2005 within both the exclosures and their paired plots and this decline could be attributed to at least two factors. First, we measured all live stems greater than 10 cm in height regardless of their health status. In many cases,
this included short (< 30 cm) suckers in poor condition that likely did not survive from 2001 to 2005. A severe drought occurred in 2002 which may have also influenced the survival of smaller stems. Second, although one study reported aspen growth six years post-fire similar to our findings (Kilpatrick et al. 2003), many of the regenerating aspen suckers in our study area experienced greater growth from 2001 to 2005 (0.4 to 0.5 m on average per year) compared to other studies (Ripple and Beschta 2007; Romme et al. 2005). This rapid growth could have been due to the high availability of soil nutrients immediately following the fire (Romme et al. 2005). High production of suckers immediately following disturbance is not uncommon but their numbers decline rapidly within 1-2 years and by the fifth year, most clumps are reduced to only one or two stems due to the intense competition among suckers that cause the least vigorous ones to die (Schier et al. 1985). Typically, only one of the stems will eventually grow to tree form (Larsen and Ripple 2005). Other studies have reported rapid declines in stem densities following initial growth and, in addition to competition among stems, reasons for the declines have varied from heavy ungulate browsing to climatic and site conditions (Kaye et al. 2005; Romme et al. 2005; Romme et al. 1995). It is possible that ungulates could have caused some reduction in the number of stems from 2001 to 2006. However, we believe browsing was not a primary factor due to the initial rapid reduction in aspen suckers from natural thinning observed in other studies, no significant change in elk movement patterns observed in the study area from 2001 through 2004 (Biggs et al. 2010), and mean twig browsing levels recorded at or below a minimum threshold for causing impacts to aspen.

White (2001) and White et al. (2003) quantified elk densities while at the same time measuring an index of browsing over a range of these densities. They found that aspen
saplings were only abundant at low elk densities (< 1 pellet group 100 sq m\(^{-1}\)). At moderate and high elk densities (1-3 pellet groups and > 3 groups 100 sq m\(^{-1}\), respectively), abundance of aspen saplings decreased exponentially. In both studies, the authors speculated that the transition toward significant impacts to aspen saplings probably occurs at a browsing rate equivalent to > 3 pellet groups 100 sq m\(^{-1}\). We estimated 1.1 pellet-groups 100 sq m\(^{-1}\) for all plots in 2006 when a significant correlation was observed between increasing pellet-groups and decreasing patch size and height. In other studies, young European aspen (\(P. tremula\)) stands excluded from ungulate browsing were significantly taller and experienced significantly greater growth rates than areas exposed to browsing (Edenius et al. 2011). We did not observe any differences between areas excluded from browsing and areas accessible to browsing 5-6 years post-fire which may have been due to relatively low densities of ungulates, especially elk, occurring in the study area either during peak aspen growth (summer) or on an extended seasonal basis. However, it should be noted that there were only four pairs of fenced-unfenced plots limiting the statistical power to detect browsing effects. Based on our results, regenerating aspen stands throughout the burn area may be able to sustain elk use under the distribution and abundance patterns observed for ungulates during our study period but the population may be approaching a threshold where negative effects could occur.

Pellet-group density and aspen twig browsing were strongly positively correlated and both were correlated with patch size and height in one or both years of study suggesting that it may be possible to use these two variables either independently or in combination with one another to assess potential impacts to regenerating aspen. Either of these two indicators could be used to monitor long-term effects of elk on aspen
community structure and development by providing both a measure of elk abundance and a measure of forage use intensity. Recent studies have found a correlation between total pellet counts of elk and aspen stand condition (Rogers and Mittanck 2014, Rogers et al. 2015) and we believe this method can help provide a measure of potential browsing effects on aspen regeneration if applied sufficiently across landscapes. Use of these indicators are intended to be applied to similar conditions in which they were developed (recently burned semi-arid montane ranges used by large ungulates primarily during the non-active plant growth period).

FUNDING

Research was funded in part by the Environmental Stewardship Division, Los Alamos National Laboratory and the Espanola Ranger District, Santa Fe National Forest, U.S. Forest Service.

ACKNOWLEDGEMENTS

We thank Marwin Shendo, Leonard Sandoval, Leslie Hansen, Susan Rupp, Heather Alexander, Marjorie Wright, Sam Loftin, Carey Bare and other members of the Ecology Group at LANL for field and project support, and Suzanne Gifford of the Valles Caldera National Preserve for field support. We also thank Mary Orr of the U.S. Forest Service, Espanola District, for partial funding support, access permits, and assistance in identifying locations for placement of fenced exclosures.
REFERENCES

Experiment Station.

Augustine DJ, McNaughton SJ (1998) Ungulate effects on the functional species
composition of plant communities: herbivory selectivity and plant tolerance. J. Wildl.
Manage. 62:1165-1183.

Augustine DJ, McNaughton SJ (2004) Regulation of shrub dynamics by native


Bailey JK, Whitham TG (2003) Interactions among elk, aspen, galling sawflies and

communities following fire. J. Range Manage. 47:79-83.

Bartos DL, Campbell RB, Jr. (1998) Decline of quaking aspen in the Interior West-

Biggs JR, VanLeeuwen DW, Holechek JL, Valdez R (2010) Multi-scale analyses of
habitat use by elk following wildfire. Northwest Sci. 84:20-32.

LA-11735-MS, Los Alamos, NM, USA.


Table 1. Estimated aspen mean stem density (ha$^{-1}$) for 2001, 2005, and 2006 for fenced exclosures (n = 4), unfenced paired adjacent plots (n = 4), and both combined, Cerro Grande Fire burn area, New Mexico.

<table>
<thead>
<tr>
<th>Year</th>
<th>Exclosures (se)</th>
<th>Unfenced Plots (se)</th>
<th>Combined (all plots) (se)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2001</td>
<td>44,066 (20,444)</td>
<td>51,364 (13,213)</td>
<td>47,715 (11,352)</td>
</tr>
<tr>
<td>2005</td>
<td>17,374 (7,159)</td>
<td>20,076 (3,827)</td>
<td>18,725 (3,793)</td>
</tr>
<tr>
<td>2006</td>
<td>15,202 (6,166)</td>
<td>21,465 (4,781)</td>
<td>18,333 (3,802)</td>
</tr>
</tbody>
</table>
Table 2. Correlations between aspen summer patch size and height and previous winter twig browsing and elk pellet-group density, Cerro Grande Fire burn area, New Mexico 2005-2006.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Year</th>
<th>n</th>
<th>P-value</th>
<th>r^a</th>
</tr>
</thead>
<tbody>
<tr>
<td>Patch size-twig browsing</td>
<td>2005</td>
<td>21</td>
<td>0.068</td>
<td>-0.406</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>20</td>
<td>0.046</td>
<td>-0.451</td>
</tr>
<tr>
<td>Patch height-twig browsing</td>
<td>2005</td>
<td>21</td>
<td>0.008</td>
<td>-0.563</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>20</td>
<td>0.085</td>
<td>-0.395</td>
</tr>
<tr>
<td>Patch size-pellet group density</td>
<td>2005</td>
<td>23</td>
<td>0.143</td>
<td>-0.315</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>24</td>
<td>≤ 0.001</td>
<td>-0.682</td>
</tr>
<tr>
<td>Patch height-pellet group density</td>
<td>2005</td>
<td>23</td>
<td>0.072</td>
<td>-0.383</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>24</td>
<td>0.003</td>
<td>-0.582</td>
</tr>
<tr>
<td>Pellet group density-twig browsing</td>
<td>2005</td>
<td>21</td>
<td>0.003</td>
<td>0.619</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>20</td>
<td>0.035</td>
<td>0.474</td>
</tr>
</tbody>
</table>

^a Correlations between variables were assessed using Spearman’s rank correlation test. Correlations are considered significant at p ≤ 0.05.
Figure 1
Figure 3
Figure 5

![Graph showing the relationship between Aspen patch height (cm) and Pellet-group density (per 100 sq m). The graph displays a scatter plot with points indicating varying densities at different heights.]
Figure 6
Figure 7

![Graph showing percent aspen twig browsing vs. pellet-group density (per 100 sq m).](image-url)