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Author(s): C.S. Boyd, K.W. Davies, G.H. Collins
Source: Rangeland Ecology & Management, 70(4):411-417.
Published By: Society for Range Management
URL: http://www.bioone.org/doi/full/10.1016/j.rama.2017.02.001

#912

Rangeland Ecology & Management 70 (2017) 411–417

Contents lists available at ScienceDirect

Rangeland Ecology & Management

journal homepage: http://www.elsevier.com/locate/rama

Original Research

Impacts of Feral Horse Use on Herbaceous Riparian Vegetation Within a Sagebrush Steppe Ecosystem

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ARTICLE INFO

Article history:
Received 19 April 2016
Received in revised form 26 January 2017
Accepted 9 February 2017

Key Words:
bare ground
herbivory
springs
wild horses
wildlife habitat

ABSTRACT

Feral horses inhabit rangeland ecosystems around the world, and their impacts on riparian ecosystems are poorly understood. We characterized impacts of a free-ranging horse population on the structure and composition of riparian plant communities in the sagebrush steppe ecosystem in the western United States. We used a randomized block design with single 25 × 50 m exclosures and grazed plots on four study sites within Sheldon National Wildlife Refuge in northwestern Nevada. Exclosures were constructed in 2008. Herbaceous plant utilization was measured from 2009 to 2013 by clipping within excluded and grazed plots. Herbaceous production and vertical structure were measured in 2013, and plant functional group and ground cover components were estimated in 2012–2013. Herbaceous utilization ranged from 27% to 84%, and herbaceous production did not differ by grazing treatment ($P = 0.472$). Grazed plots had seven-fold higher bare ground cover ($P < 0.001$), 60% less litter cover ($P < 0.001$), and the basal cover index was 65% higher. Grazing increased rush density by 50% ($P = 0.041$) but did not affect sedge density ($P = 0.514$). Grazing decreased herbaceous stubble height up to 80% and visual obstruction by about 70% ($P < 0.05$). Deep-rooted hydrophytic plant species did not increase with grazing exclusion, but greater vertical structure in excluded plots could improve hiding and nesting habitat for some riparian-associated wildlife species. Additionally, decreased bare ground with grazing exclusion could reduce erosion potential and susceptibility to invasive plant species.

Introduction

During the Pleistocene, the range of a variety of species of wild horses (*Equus* sp.) covered much of Europe, Asia, North America, and perhaps North Africa (Kavar and Dove, 2008). Wild horse populations underwent a significant range contraction at the end of the Pleistocene and became extinct in North America by approximately 10,500 years before present (Guthrie, 2003, 2006). Free-ranging domestic horses (*E. caballus)* were introduced to North America by Spanish explorers during the 16th and 17th centuries (Haines, 1938) and by inadvertent and purposeful releases associated with the ranching industry and perhaps the military during the late 19th and early 20th centuries (Young and Sparks, 2002). Similar introductions of domestic horse stock have occurred globally, and current feral horse (horses whose ancestors are of domesticated lineage; Ostermann-Kelm et al., 2009) populations are stable or increasing in both the United States and Australia (Nimmo and Miller, 2007; Garrott and Oli, 2013).

Feral horses are currently a management issue on rangelands throughout much of the world. In the western United States, horses lack effective predators to control populations and periodic gathering of horses on federal lands has been necessary to control horse numbers and their associated impact on plant and soil resources (Garrott and Oli, 2013). Gauging the ecological impact of feral horses can be difficult because grazing by these animals often coincides in space and time with utilization by other large herbivores, particularly domestic livestock. In the western United States, most public lands are grazed by cattle and cattle diets overlap substantially with those of horses (Krysl et al., 1984; Scasta et al., 2016). Unlike domestic livestock, feral horse grazing is difficult to manage on a rotational or deferred basis due to difficulties associated with moving horses, resulting in continuous or near continuous use of rangeland plant communities. Previous work suggests that feral horse grazing can alter upland vegetation and soil resources within rangeland ecosystems at local (Fahnestock and Detling, 1999a; Ostermann-Kelm et al., 2009) and landscape scales (Beever et al., 2008; Zeigenfuss et al., 2014). Horse grazing has also been linked to community scale changes in composition and demographics of insect (Beever and Herrick, 2006), small mammal (Beever and Brussard, 2004), avian (Zalba and Conzzani, 2004), and estuarine fauna populations (Levin et al., 2002), as well as competition for water resources with a variety of native wildlife species (Hall et al., 2016).

Impacts of grazing animals on plant and soil resources can be particularly acute in and around riparian areas that serve as an attractant to herbivores due to forage and water availability (Kauffman and Krueger, 1984; Turner, 2015). Riparian areas are critical for maintaining a broad

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http://dx.doi.org/10.1016/j.rama.2017.02.001
1550-7424/Published by Elsevier Inc. on behalf of The Society for Range Management.

suite of plant and animal species, contain biotic and abiotic habitats that differ from the surrounding landscape, and may serve as important travel corridors for a variety of wildlife species (Gregory et al., 1991; Naiman et al., 1993; Chambers and Miller, 2011). In the sagebrush steppe of the US Great Basin these habitats harbor the majority of regional biodiversity (Sada, 2008). Research suggests preference of feral horses for riparian areas during summer months with the potential for reduction in plant species richness, height, and cover and for alteration of plant community composition (Ganskopp and Vavra, 1986; Crane et al., 1997; Beever and Brussard, 2000). In desert environments typical of the US Great Basin, upland vegetation surrounding riparian areas may also be impacted by feral horse grazing; for example, Davies et al. (2014) reported reduced sagebrush density in riparian-adjoining upland areas exposed to feral horse grazing relative to grazing exclosures.

Sheldon National Wildlife Refuge in northern Nevada offers a unique opportunity to examine the impacts of feral horses on riparian plant communities that are not currently grazed by domestic livestock. Our objective was to determine habitat structure and composition of plant communities in riparian areas exposed to and excluded from feral horse grazing. We hypothesized that plant basal cover would increase in response to horse grazing exclusion, and that grazing exclusion would shift plant community composition away from grazing-tolerant rush (*Juncus* sp.) species. We further hypothesized that vertical structure of habitat would increase with grazing exclusion in association with increased shrub density, herbaceous stubble height, and visual obstruction.

Methods

Study Area

Our study took place within the Sheldon National Wildlife Refuge (SNWR) in northwestern Nevada. Since the 1880s cattle, sheep, and horses (>20,000 animals) grazed on the SNWR, but cattle and sheep were removed between 1990 and 1994 (U.S. Fish and Wildlife Service, 2013). SNWR covers an area of 230,000 ha of largely sagebrush steppe rangeland, and our study sites were located within an unfenced and contiguous 80,000-ha management unit. Feral horse density within the management unit varied from 0.5 to 0.8 horses.km^{-2} between 2007 and 2013 (USFWS, unpublished data). Free-ranging burros occurred within SNWR, but not within our study area. Elevations within SNWR average approximately 1900 m, and climate is characterized by warm, dry summers and cold, wet winters with most precipitation occurring during the winter and spring periods. Average water year precipitation (1 October to 30 September) for the study area is approximately 300 mm, and during the study period it was 67%, 88%, 148%, 110%, 70%, and 69% of normal for 2008, 2009, 2010, 2011, 2012, and 2013, respectively. Climate information for the study period was collected from remote weather stations located within 50 km of study sites (RAWS USA Climate Archive).

Dominant plant species across study sites included the sedges *Carex nebrascensis* (Dewey) and *Carex microptera* (Mack.), the grasses *Agrostis* sp. and *Poa sp*., and the rush *Juncus balticus* (Willd. Var.). Common forb species included *Veronica americana* (Schwein.), *Polygonum bistortoides* (Pursh.), and *Achillea millefolium* (L). The only shrub species encountered within the riparian zone of the study plots was *Rosa woodsii* (Lindl.). Non-native plant species were almost nonexistent within study plots.

Experimental Design

We used a randomized block design consisting of four blocks (sites) and two treatments. We established 25 × 100 m sites near the origin of perennial springs in 2008. Study sites were chosen by random selection from the population of approximately 40 springs within the 80,000-ha management unit. The long axis of each site was bisected by the dominant riparian channel. Sites were then divided into two 25 × 50 m plots, and plots were randomly assigned to be either excluded from ("excluded") or exposed to ("grazed") feral horse use. Excluded plots were fenced with 1-m high fencing consisting of evenly spaced metal pipe rails. This fence did not completely restrict access of native large mammals (mule deer and pronghorn), but field observations (i.e., lack of fecal matter and lack of utilization in excluded areas) suggest minimal utilization of excluded and grazed plots by these species. Sites varied in elevation from 1 850 m to 1 900 m and consisted of obligate or facultative wetland vegetation (Lichvar et al., 2014) communities associated with outflow from perennial springs. The end of season (October) wetted width (flowing or standing water) of riparian zones varied across both study sites and years and ranged from 0 m to 3.9 m in a dry year (2013) and from 1.4 m to 6.1 m in a wet year (2011; Table 1). Three sites had a flat aspect, and one was located on a northerly aspect. Extreme distance between sites was < 15 km. Most sites were characterized by multiple channels with a single dominant channel. Although springs were perennial, above-ground channel flow through sites was generally ephemeral and associated with the seasonal period of snow melt. However, water availability was sufficient to allow persistence of hydrophytic vegetation species within a defined riparian zone (Winward, 2000) at all sites.

Data Collection

Within each plot, vegetation measurements occurred along two 30-m long sampling transects placed within the riparian zone on both sides of and running parallel to the dominant channel (i.e., two transects total per plot). In grazed plots, utilization of herbaceous biomass by herbivores was estimated by clipping herbaceous materials to ground level in five randomly located 40 × 50 cm quadrats located within the riparian zone on each sampling transect in October of 2009–2013 and expressing remaining biomass as a percentage of ungrazed biomass. Ungrazed biomass was estimated by clipping three 40 × 50 cm quadrats within a 1-m diameter circular wire cage that was randomly located within each grazed plot and moved annually to a new location. Utilization was subjectively characterized as indicative of grazing that is suitable for providing a relatively high level of herbaceous vegetation structure ("light" grazing, < 30%), will allow for limited vertical structure development ("moderate" grazing, 30–60%), or will not allow for significant vertical structure ("heavy" grazing, > 60%, Table 2). Annual herbaceous production was estimated within exclosures in 2013 by separating dead from live plant material in five randomly located 40 × 50 cm quadrats along each sampling transect and weighing clipped materials. Herbaceous production in grazed areas was estimated by randomly locating a total of five (for each plot) 50 × 50 cm wire exclosures along sampling transects in April of 2013 and clipping and weighing live herbaceous plant material in October of 2013. All clipped material was oven-dried to constant weight before weighing.

Ground cover of bare ground and litter was estimated (ocular) in June or July 2012–2013 within thirty 20 × 50 cm quadrats spaced at 1-m intervals along each sampling transect; obstruction of the ground surface when viewed from above due to canopy vegetation was not considered when making these estimates (i.e., we looked under the

Table 1
Wetted width during an above (2011) and below (2013) average precipitation year for riparian study sites in northern Nevada. Values represent site averages of plots that were excluded from or accessible to horse grazing

	2011		2013	
Site	Mean (cm)	SE (cm)	Mean (cm)	SE (cm)
Tenmile	138.3	26.1	0.0	0.0
Buckaroo	607.2	177.8	389.8	92.0
Corral	167.7	45.6	0.0	0.0
Smith	227.8	31.3	302.0	39.4

Table 2
Percent utilization of aboveground biomass for riparian study plots grazed by feral horses in northern Nevada. Data are site averages from 2009 to 2013

Site	Mean (%)	SE (%)	Grazing level
Tenmile	46.9	17.0	moderate
Buckaroo	72.7	6.0	heavy
Corral	27.0	15.4	light
Smith	83.9	5.9	heavy

vegetation canopy). Percent plant basal cover (i.e., "basal cover index") was estimated for each quadrat by summing percent cover of bare ground and litter and subtracting the total from 100. Density of herbaceous plant species was estimated by direct count of either plants for nonrhizomatous species or stems for rhizomatus species within 10 × 20 cm quadrats nested within those used to estimate ground cover attributes. Shrub density was by direct count within 1 × 30 m belt transects centered on each sampling transect. Maximum height was measured for all shrubs within the belt transects.

Herbaceous stubble height was estimated in October 2012–2013 at 1-m intervals along sampling transects by measuring the maximum height of the nearest sedge, grass, or rush. Visual obstruction was measured in July 2013 by noting the highest (from ground) strata of a 2.5-cm diameter pole visible from a distance of 5 m and an observation height of 1 m. The pole was demarcated into 2.5-cm strata of alternating black and white bands, and readings were taken at 1-m intervals along each vegetation transect (Robel et al., 1970).

Data Analysis

For analysis purposes, plant density data were collated by functional group. Functional groups included sedges, rushes, *Eleocharis* sp., grasses, forbs, and shrubs. All statistics were performed using Statistical Analysis Software (v. 9.2; SAS Institute, Cary, NC). The effects of herbivory on variables that were repeated in time were analyzed using repeated measures analysis of variance (ANOVA) with time as a repeated variable, and nonrepeated factors were analyzed using one-way ANOVA (PROC Mixed). Site and the site × treatment interaction were considered random factors and treatment a fixed factor. Because shrubs were only recorded at one site, we did not analyze treatment effects on the abundance of this functional group. Nonproportional data not meeting ANOVA assumptions were weighted by the inverse of the treatment variance during analysis (Neter et al., 1990; James and Drenovsky, 2007). Proportion data not meeting ANOVA assumptions were transformed using the arcsine-squareroot (Sokal and Rohlf, 1995). Nontransformed means are used for presentation and discussion. Covariance structure was selected as detailed by Littell et al. (1996). Main effects and interactions were considered significant at $\alpha = 0.05$. Data are reported as treatment means with associated standard errors.

Results

Herbaceous production varied by site ($P < 0.001$) but not grazing treatment ($P = 0.472$) and ranged from approximately 85.3 $g \cdot m^{-2}$ (± 13.8) at the Tenmile site to 356.4 $g \cdot m^{-2}$ (± 44.1) at the Smith site (Fig. 1). Utilization across study sites varied from 27.0% (± 15.4) at the Corral site to 83.9% (± 5.9) of standing crop at the Buckaroo site (Table 2; Fig. 2).

Cover of bare ground was approximately seven-fold higher in grazed versus excluded plots ($P < 0.001$, Fig. 3) but did not vary by year ($P = 0.524$). Litter cover was > 60% less in the grazing treatment ($P = 0.010$) and was unaffected by year ($P = 0.093$). The basal cover index varied by both treatment ($P = 0.053$) and year ($P = 0.035$), was about 65% higher in grazed plots (see Fig. 3), and was highest in 2013 (2013 = 38.4% [± 4.0], 2012 = 31.1% [± 5.5], data not shown).

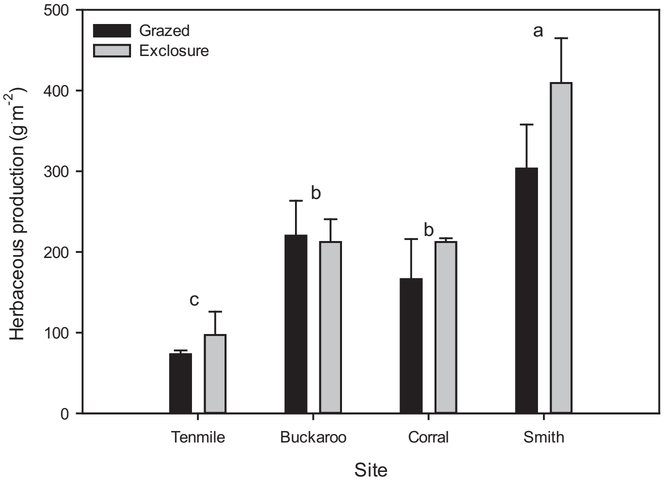

Figure 1. Herbaceous aboveground plant production for riparian study plots in northern Nevada. Data are means for 2013 with associated standard errors. Plots were excluded from horse grazing from 2009 to 2013 or were accessible to grazing during that time period. Sites without a common letter are different at $\alpha = 0.05$.

Sedge stem density was unaffected by treatment ($P = 0.514$) or year ($P = 0.233$) and averaged approximately 30 plants·0.1m^{-2} (Fig. 4). On the basis of density, rushes were the most abundant herbaceous functional group. Stem density of rushes was 50% higher in grazed plots ($P = 0.041$) but was unaffected by year ($P = 0.921$). *Eleocharis* sp. did not vary between years ($P = 0.826$) but was almost 150-fold higher in grazed plots ($P = 0.001$). Grass density in grazed plots was almost double that of excluded ($P = 0.032$) but did not change between years ($P = 0.457$). Density of forbs did not vary by treatment ($P = 0.305$) or year ($P = 0.103$).

Herbaceous stubble height was affected by year ($P = 0.005$) and the year by treatment interaction ($P = 0.016$, Fig. 5). Stubble height values ranged from 8.5 ± 2.5 cm for grazed plots in 2012 to 40.6 ± 3.1 cm for exclosure plots in 2012. Stubble height for grazed plots increased by 72% from 2012 to 2013 ($P < 0.05$). Visual obstruction was about threefold less ($P = 0.038$) in grazed plots (see Fig. 5). This change was associated primarily with reduced presence of sedge vegetation in the upper herbaceous canopy with grazing (see Fig. 2).

Discussion

Our results suggest that the impact of feral horses on riparian plant communities varies depending on the parameter under consideration. We found that exclusion from horse grazing had no impact on sedge density and, consistent with our hypothesis, rush stem density was higher in grazed plots, suggesting that maintenance of key herbaceous plant species is possible under the horse density and grazing patterns encountered in the current study. It should be noted that our study sites were primarily associated with springs, as opposed to perennial riparian drainages with more defined dominant channel systems and higher flows; under higher flow conditions, similar levels of herbivory may impair riparian function if they resulted in compromised bank structure and associated site desiccation followed by loss of deep-rooted plant species (George et al., 2012). This scenario is supported by others who have noted the importance of abiotic factors, specifically water availability, in modulating the impact of herbivory on plant species composition (Fahnestock and Detling, 1999b).

In our study, greater density of *Eleocharis* and *Rush* sp. in grazed areas was associated with greater bare ground. Both of these functional groups have been reported to increase in abundance in response to grazing, perhaps in association with increasing bare ground (Leege et al., 1981; Dobkin et al., 1998; Loucougaray et al., 2004). Changes in functional group abundance can also be associated with preferential

Figure 2. Horse-excluded (left image) and adjacent horse-grazed (right image) plots at Buckaroo Spring, Sheldon National Wildlife Refuge, Nevada, in summer 2012. The exclosure was constructed in summer 2008. A strong reduction in herbaceous plant vertical structure with horse grazing is apparent.

selection by grazing ungulates (Kauffman and Krueger, 1984), however, horses are not highly selective grazers (Hanley and Hanley, 1982). Other work has indicated sedges (*Carex* sp.) and perennial bunchgrasses are key dietary components for free-ranging horses (Crane et al., 1997).

Published studies indicating the impacts of horse grazing on spring-associated riparian plant species are limited. Our work is in agreement with Dobkin et al. (1998), who found that rush abundance increased with horse use of riparian vegetation around springs. However, dissimilar to Dobkin et al. (1998), we did not find decreased grass and sedge density with horse grazing. Our results may differ from Dobkin et al. (1998) because we measured herbaceous plant response based primarily on density as opposed to foliar cover, the latter being more sensitive to plant canopy structural alterations associated with grazing; such alterations in cover may or may not relate to changes in biotic potential of a plant functional group to produce biomass. Beever and Brussard (2000) also recorded reduced herbaceous plant abundance, as well as reduced plant diversity with horse grazing, but levels of utilization and trampling appear to have been much higher in that study compared with average utilization in the present study.

Shrubs were only present at one of our study sites, and only one species of shrub (*R. woodsia*) was encountered. Previous work suggests that this species may be somewhat tolerant to herbivory, and thus its singular presence could represent initial shrub recovery from long-term herbivory (Bailey et al., 1990). The fact that shrubs remained absent throughout our 5-year study on the remaining sites suggests that shrub propagules were not present at these sites. Because shrub distribution was limited across study sites, we are not able to make definitive statements regarding the impacts of horse grazing on shrub abundance. However, limited riparian-associated shrub populations (both *R. woodsia* and *Salix* sp.) were present in the vicinity of our study sites on similar slopes, gradients, and landscape positions (in some cases within the same drainage way). Thus, it is possible that historic livestock and horse grazing or perhaps other management practices may have led to reduced shrub abundance for long enough to limit propagule availability. Strong decreases in shrub populations with unrestricted horse grazing have been previously reported for spring-associated plant communities in Nevada (Beever and Brussard, 2000), and Davies et al. (2014) found decreased sagebrush density in horse-grazed uplands adjacent to riparian areas.

From a plant production standpoint, exclusion from feral horse grazing did not change herbaceous productivity within the time-frame of our study. This does not preclude the possibility that longer-term exposure to horse utilization could impact plant production. Lack of a difference in aboveground production between grazed and excluded areas suggests that grazing is not impairing basic ecological processes of these sites. Alternatively, exclusion for 5 years may not have been

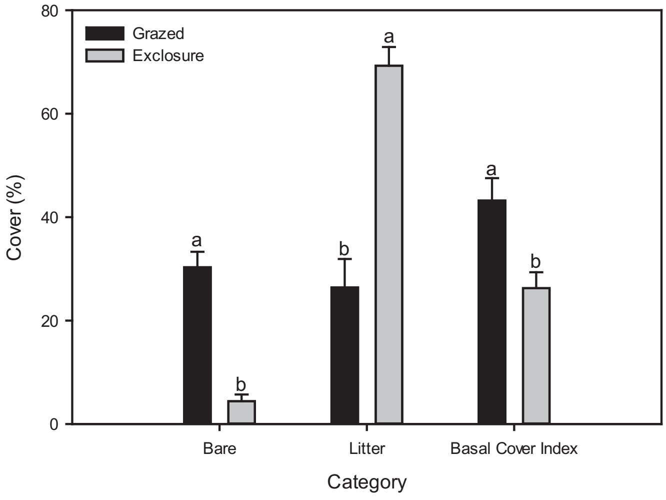

Figure 3. Ground cover for riparian study plots in northern Nevada. Data are means for 2012−2013 with associated standard errors. Plots were excluded from horse grazing from 2009−2013 or were accessible to grazing during that time period. Within a category, bars without a common letter are different at α = 0.05.

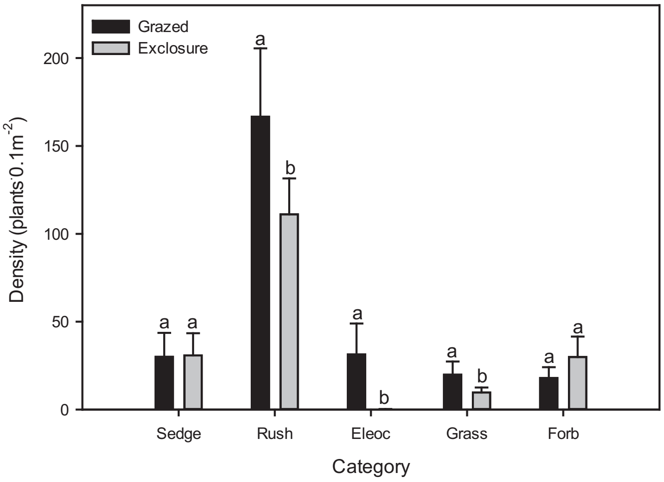

Figure 4. Herbaceous plant density for riparian study plots in northern Nevada. Data are means for 2012−2013 with associated standard errors. Plots were excluded from horse grazing from 2009 to 2013 or were accessible to grazing during that time period. Bars represent treatment means with associated standard errors. Within a category, bars without a common letter are different at α = 0.05.

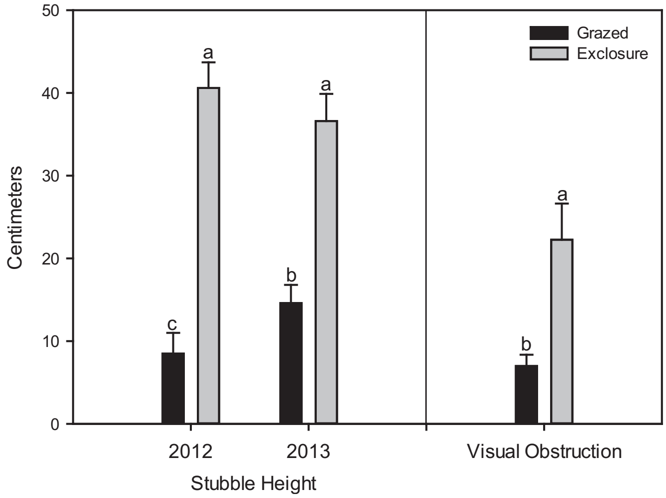

Figure 5. Herbaceous stubble height and visual obstruction for riparian study plots in northern Nevada. Stubble height is the average maximum height of herbaceous plants, and visual obstruction represents the highest (from ground) portion of a 2.5-cm diameter pole visible from a distance of 5 m and an observation height of 1 m. Data are means and associated standard errors for 2012—2013 (stubble height) and 2013 (visual obstruction). Plots were excluded from horse grazing from 2009 to 2013 or were accessible to grazing during that time period. Within a panel, bars without a common letter are different at α = 0.05.9

long enough to have elicited vegetation recovery from decades of horse use (and cattle and sheep use prior to the 1980s). We did not measure below ground production and thus cannot speculate on the effects of horse herbivory on primary production of study sites. Our results are consistent with those of others reporting either no effect of grazing or clipping on aboveground (Leege et al., 1981) or total (above and below-ground) plant production (Boyd and Svejcar, 2012) or increased aboveground production with grazing (Boyd and Svejcar, 2004). In contrast, Fahnestock and Detling (1999a) reported that aboveground plant production in arid and montane grasslands could be decreased by horse grazing during wet years, but increased with grazing on montane sites during dry years via compensatory production. In this same study, belowground biomass was invariant to horse grazing treatment. Working within the same area of the Great Basin sagebrush steppe as our study and over a longer time period, Zeigenfuss et al. (2014) reported that aboveground plant biomass decreases as horse density increases and that this effect is amplified during dry years.

Contrary to our hypothesis, the plant basal cover index was higher in grazed areas (see Fig. 3), which is somewhat counterintuitive given that percent cover of bare ground was also higher in grazed areas. However, the majority of the ground surface within exclosures was covered by litter (see Fig. 3), which may have acted to limit sites available for plant establishment and growth. This litter was predominantly from previous years' production of sedges, and to a lesser extent rushes and grasses, and we commonly observed litter buildup to 10-cm depth. Litter accumulation of this magnitude could reasonably be expected to limit establishment of new plants (Loucougaray et al., 2004). Additionally, because bare ground was higher in grazed areas, this may have acted to create establishment sites for more grazing-tolerant (*Eleocharis* sp. and *Juncus* sp.) plant species (Krueger-Mangold et al., 2006). Put another way, effective area for establishment of plant propagules may have been higher in grazed versus excluded areas. Whether or not this facilitates or detracts from the long-term sustainability of these plant communities depends on 1) whether available sites are colonized by plant species that promote soil stability (e.g., sedges and rushes; Shafroth et al., 2002) and 2) if increased bare ground results in soil loss due to elevated erosion potential (Pearce et al., 1998). In contrast, in a companion study looking at horse grazing effects on upland vegetation adjacent to riparian areas, Davies et al. (2014) reported decreased plant cover with heavy utilization (> 40%) by horses. However, these sites were dominated by caespitose perennial bunchgrasses growing in a more arid environment with low potential for litter buildup in interspace locations.

As predicted by our hypothesis, vertical structure was strongly influenced by the presence of horse grazing, with both stubble height and visual obstruction increasing with horse exclusion. Proportionally, visual obstruction values were within about 60% of stubble heights (see Fig. 5), reflecting the dense nature of vegetation at study sites (i.e., limited horizontal visibility below the upper herbaceous canopy). The degree of vertical structure in our study plots was influenced almost entirely by herbaceous vegetation. Increases in shrub abundance at some point in the future could significantly increase vertical structure of the riparian plant communities in our study. The maximum herbaceous vertical structure recorded in our study was approximately 60 cm, and *R. woodsii* (the only shrub species encountered in our study) can reach heights of 3 m (Pavek and Skinner, 2013).

Changes in vertical structure and plant functional group composition of wetland plant communities associated with horse grazing could have strong guild-dependent impacts on availability and suitability of habitat for a variety of wildlife species (Knopf et al., 1988; Dobkin et al., 1998; Levin et al., 2002). Avian species in particular are tightly linked to vertical structure in riparian habitats (Ammon and Stacey, 1997), and thus differences in vertical structure noted between horse grazing treatments in the current study, as well as other work (e.g., Beever and Brussard, 2000), could logically be expected to impact avifaunal habitat use. For example, Brewer's blackbird *(Euphagus cyanocephalus)* often nests in dense vegetation bordering wetland habitat (Martin, 2002), and this species was frequently found nesting in sedge-dominated plant communities within our riparian exclosures, even though the exclosures occupied a small area. This species was not observed nesting in horse-grazed areas. In contrast, greater sage-grouse *(Centrocercus urophasianus),* a resident species of critical conservation status within our study area, has been reported to preferentially forage in moderately grazed riparian areas during brood rearing (Klebenow, 1982). Dobkin et al. (1998) observed avian guild separation between cattle-grazed and ungrazed riparian habitats, noting that upland species increased in grazed areas while wetland and riparian species dominated nongrazed plots.

Our study area was grazed by horses throughout the year with no fencing to restrict access. Therefore, from a conservation standpoint, the relatively limited vertical structure we recorded in association with horse use suggests that horse grazing may be limiting vertical structure of riparian habitats at larger scales and, by extension, negatively impacting diversity of avian habitats. Alternatively, with no horse grazing, structural diversity of habitats could be skewed toward increased vertical structure at larger scales. This is particularly true given the absence of other resident-grazing ungulates with a dietary preference for, or capable of subsisting on, riparian graminoid vegetation. Others have suggested that avian diversity could be maximized with heterogeneous levels of grazing intensity (Loucougaray et al., 2004; Martin and McIntyre, 2007).

Management Implications

Although herbaceous plant production between grazed and excluded treatments did not vary in our study, results indicated that year-round exposure to feral horse grazing can result in changes in plant community structure and composition. The most apparent of those changes are related to plant community structural and ground cover attributes. Bare ground decreased dramatically with horse exclusion in association with a strong increase in ground cover of litter. These changes were likely related in the sense that horse grazing removed herbaceous plant material (see Fig. 2) that would otherwise have accumulated as ground litter; and as ground litter is reduced, bare ground can increase. However, herbaceous plant production did not vary between grazed and excluded treatments. Reduced vertical structure with horse grazing, associated with altered herbaceous canopy structure, may decrease habitat availability for riparian and wetland-associated avian species. Alternatively, foraging habitat for some avian species could increase with reduced vertical structure (Tewksbury et al., 2002). While 5 years of feral horse exclusion did precipitate changes in herbaceous plant species composition, density of deep-rooted species necessary for maintenance of riparian function (i.e., sedges and rushes) was either unaffected or increased with horse grazing. We were unable to experimentally determine horse impacts on shrub communities because only one shrub species was present at only one of the four study sites. Effects of horse grazing on riparian habitat will be density dependent and associated with landscape features and herd management practices that influence horse distribution across the landscape.

Acknowledgments

The authors greatly appreciate personnel from the Sheldon-Hart Mountain National Wildlife Refuge Complex for their assistance in locating study sites and constructing exclosures. We also appreciate the efforts of Clare Poulsen and Jarod Lemos in field data collection and implementation of the experiment. The findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the US Fish and Wildlife Service. The US Department of Agriculture (USDA) is an equal opportunity provider and employer. Mention of a proprietary product does not constitute a guarantee or warranty of the product by the USDA or the authors and does not imply its approval to the exclusion of other products that may also be suitable.

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