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Effects of feral free-roaming horses on semi-arid rangeland ecosystems: an example from the sagebrush steppe

K. W. Davies,[1,]† G. Collins,[2] and C. S. Boyd[1]

[1]*Eastern Oregon Agricultural Research Center, United States Department of Agriculture, Agricultural Research Service, Burns, Oregon 97720 USA*
[2]*Sheldon-Hart Mountain National Wildlife Refuge Complex, United States Department of Interior, Fish and Wildlife Service, Lakeview, Oregon 97630 USA*

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Abstract. Feral horses (*Equus caballus*) are viewed as a symbol of freedom and power; however, they are also a relatively unmanaged, non-native grazer in North America, South America, and Australia. Information about their influence on vegetation and soil characteristics in semi-arid rangelands has been limited by confounding effects of cattle (*Bos taurus*) grazing and a lack of empirical manipulative studies. We compared vegetation and soil surface characteristics in feral horse grazed areas and ungrazed exclosures at five sagebrush (*Artemisia*) steppe sites in northern Nevada. Horse grazed areas had lower sagebrush density and plant diversity, greater soil penetration resistance, and lower soil aggregate stability than ungrazed areas. Herbaceous cover and density generally did not differ between grazed and ungrazed treatments, with the exception of heavily grazed sites in which perennial grass cover was reduced. The cumulative effect of feral horses on soil characteristics suggests that they may affect the ecological function of semi-arid rangelands by increasing the risk of soil erosion and potentially decreasing availability of water for plant growth. The two-fold increase in sagebrush density with horse exclusion suggests that feral horses may limit sagebrush recruitment and thereby negatively impact Greater Sage-grouse (*Centrocercus urophasianus*) and other sagebrush associated wildlife. The effects of feral horses on sagebrush and other semi-arid ecosystems should be considered when developing conservation plans for these ecosystems and associated wildlife.

Key words: *Artemisia*; *Equus caballus*; exclosures; grazing; herbivory; soil aggregate stability; trampling; wild horses.

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† **E-mail:** kirk.davies@oregonstate.edu

Introduction

Free-ranging horses (*Equus caballus*) are a management and conservation concern in North America (Turner 1987, Beever 2003, Girard et al. 2013) and other locations around the globe (e.g., Rogers 1991, Zalba and Cozzani 2004, Nimmo and Miller 2007, de Villalobos and Zalba 2010). Management of free-ranging horses in the United States is contentious because they are viewed as a symbol of freedom and strength and an icon of the American West (Beever 2003), but they are also a non-native, feral grazer. The concern with feral horses is that unmanaged and poorly managed non-native grazers can have substantial impacts on ecosystem integrity as seen with poorly managed livestock in the western United States (Fleischner 1994, Jones 2000). The potential for feral horse effects may be large in some areas even with low populations because feral horse grazing is largely unmanaged and on the landscape year round; where, in contrast, domestic livestock grazing is much more intensively managed through fencing, rotation grazing, herding, salt and mineral supplementation, and water sources (Beever 2003). A large portion of the western United States has the potential to be affected by feral horses as there are currently over 40,000 free-ranging horses and burros occupying 12.8 million ha (BLM 2014).

It is well established that ungulates affect ecosystems and can modify plant community composition and structure (Hobbs 1996, Augustine and McNaughton 1998). The effects of herbivory depend on the complex interaction of the characteristics of the ungulate, plant community, soil, climate, and landscape (Hobbs 1996). Thus, some of the controversy regarding feral horses is the result of a limited number of studies investigating their effects, especially in arid and semi-arid ecosystems such as the sagebrush (*Artemisia* L.) ecosystem. Beever et al. (2003) found that disturbance sensitive variables could collectively be used to separate feral horse occupied areas from horse removed areas in multivariate ordination space, despite indistinguishability of the two types of sites using data on 'key' forage species monitored by the Bureau of Land Management. Horse removed areas had greater shrub, native grass, and total plant cover and species richness than horse occupied sites (Beever et al. 2008). Penetration resistance of the soil surface (an indicator of soil compaction) was less where feral horses had been removed compared to horse occupied sites (Beever and Herrick 2006). Though these studies (Beever et al. 2003, 2008, Beever and Herrick 2006) provide valuable information regarding horse occupied and horse removed areas, they do have some limitations. Horse occupied and horse removed areas occurred in separate mountain ranges, though they had similar slope, aspect, dominant shrub, fire history, and grazing pressure by cattle (*Bos taurus*; Beever et al. 2003, 2008, Beever and Herrick 2006). Without rigorous experimental control there are undoubtedly some unaccounted for confounding variables and cause and effect relationships cannot be determined. When treatments are not randomly assigned and before and after treatment data in both control and experimental plots are not acquired, it is not possible to attribute differences to the treatments (Beever and Brussard 2000). Though grazing pressure by cattle was estimated to be minimal to none across their study sites (Beever et al. 2003, 2008, Beever and Herrick 2006), cattle presence may obscure some of the horse effects or make it difficult to separate. Therefore, there is a critical need to evaluate feral horse effects in areas without domestic livestock. Though covering large landscapes, all these studies (Beever et al. 2003, 2008, Beever and Herrick 2006) also used the same study sites. Thus, there is a need for further analysis of feral horse effects in sagebrush rangelands to broaden the scope of research to other areas occupied by feral horses.

In grasslands in Argentina, feral horses reduced grass seed density and richness in the seedbank (Loydi et al. 2012) and decreased plant species richness and grass abundance and biomass production (de Villalobos and Zalba 2010). Feral horse use decreased biomass production in marshes (Turner 1987, 1988) and islands (Rogers 1991, Seliskar 2003). In contrast, Fahnestock and Detling (1999) reported feral horses did not have consistent effects on plant cover and that abiotic factors were more important than feral horse use in determining plant community dynamics in the Pryor Mountain Wild Horse Range in southern Montana and northern Wyoming. In addition, plant species diversity was slightly greater in horse grazed compared to ungrazed sites in the Pryor Mountain lowlands (Fahnestock and Detling 1999). The inconsistency in the reported effects of feral horse grazing suggests that vegetation response likely varies by site characteristics. Furthermore, feral horse effects in more mesic habitats may contribute little to our understanding of feral horse effects in sagebrush (Beever and Aldridge 2011) and other arid and semi-arid ecosystems.

The purpose of this study was to develop a better understanding of feral horse effects in semi-arid rangeland ecosystems. We compared mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana* (Rydb.) Beetle) steppe communities where feral horses were excluded for 5 years with adjacent continuously grazed areas on the Sheldon National Wildlife Refuge in northern Nevada. Cattle were excluded from the refuge since the early 1990s. We hypothesized that: (1) perennial herbaceous and sagebrush cover and density would be lower in feral horse grazed compared to ungrazed areas, (2) plant species richness and diversity would be less in grazed compared to ungrazed areas, and (3) soil surface penetration resistance would be higher and soil aggregate stability less in grazed compared to ungrazed areas.

Materials and Methods

Study area

We conducted this study from 2008 to 2013 on Sheldon National Wildlife Refuge in northern Nevada. The study area is mountainous with elevations at the study sites ranging around 1900 m above sea level. Climate is typical of the northern Great Basin with cool, wet winters and hot, dry summers. Average annual precipitation was between 300 and 400 mm at the study sites (NRCS 1998). Annual precipitation in 2008, 2009, 2010, 2011, 2012, and 2013 was 68%, 88%, 148%, 110%, 70%, and 69% of average at the Winnemucca, NV weather station, respectively (Weather Underground 2014), approximately 180 km southeast of Sheldon National Wildlife Refuge. Feral horse populations at Sheldon National Wildlife Refuge from 2004–2013 ranged from 700 to 1200 individuals. The study area within Sheldon National Wildlife Refuge was 80,300 ha with feral horse population ranging from 387 to 673 horses with an average of 528 horses between 2007 and 2013. The study area is an unfenced area in southern portion of the refuge (Fig. 1) and horses can travel into and out of the study area. All study sites were mountain big sagebrush steppe. However, two of the study sites were lacking a sagebrush overstory because they were in the interior of the 1999 Badger Fire. Common perennial grasses included bluebunch wheatgrass (*Pseduoroegneria spicata* (Pursh) A. Löve), Idaho fescue (*Festuca idahoensis* Elmer), Thurber's needlegrass (Achnatherum thurberianum (Piper) Barkworth), bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey), and Sandberg bluegrass (*Poa secunda* J. Presl). Common perennial forbs included milkvetches (*Astragalus* L.), fleabanes (*Erigeron* L.), biscuitroots (*Lomatium* Raf.), hawksbeards (*Crepis* L.), phloxes (*Phlox longifolia* Nutt. and *P. hoodii* Richardson), and lupines (*Lupinus* L.).

Experimental design and measurements

A randomized complete block design was used to determine the response of mountain big sagebrush steppe communities to feral horse use and exclusion. Five blocks, dispersed within the 80,300 ha study area, were selected for this experiment. Blocks were separated by up to 20 km and varied by elevation, slope, aspect, and soil. The different blocks included north, south, east, and west aspects. Elevation of blocks ranged from 1856 to 1914 m above sea level. Prior to treatment herbaceous and shrub cover and density varied by block, but treatments within a block were similar ($P > 0.10$). Each block consisted of two adjacent 50×60 m plots separated by a 10-m buffer. Each block was within 0.1 km of a riparian area with a perennial spring or small creek. Both treatment plots were the same distance from the riparian area at each block. Treatments were randomly assigned to one of the plots at each block and were: (1) feral horse grazing exclosure (ungrazed) and (2) unrestricted (free-roaming) feral horse grazing. Exclosures were metal pipe buck-and-pole fences constructed in 2008 to prevent feral horse use. Exclosures were constructed with three poles with the bottom and top pole placed 43 cm and 107 cm above the ground, respectively. Wildlife were not excluded from the exclosures and motion sensor cameras recorded no difference in visits by native ungulates inside compared to outside exclosures.

Vegetation cover and density was measured in mid-June 2012 and 2013, the fourth and fifth year after exclosures were constructed. Herbaceous cover and density was measured by species in 60 0.2-m^2 quadrats in each treatment plot. The 0.2-m^2 quadrats were placed at 3-m intervals on four 50-m transects (starting at 3 m and ending at 45 m) in each plot (15 quadrats per transect). The four 50-m transects were placed parallel to each other at 5-m intervals. Cover of biological soil crust, bare ground and litter were also measured in the 0.2-m^2 quadrats. Shrub cover by species was measured on the four 50-m transects using the line-intercept method (Canfield 1941). Shrub density by species was measured by counting plants rooted inside 2×50-m belt transects laid over the 50-m transects. Sagebrush density was separated into juvenile and mature sagebrush. Sagebrush was considered to be mature if

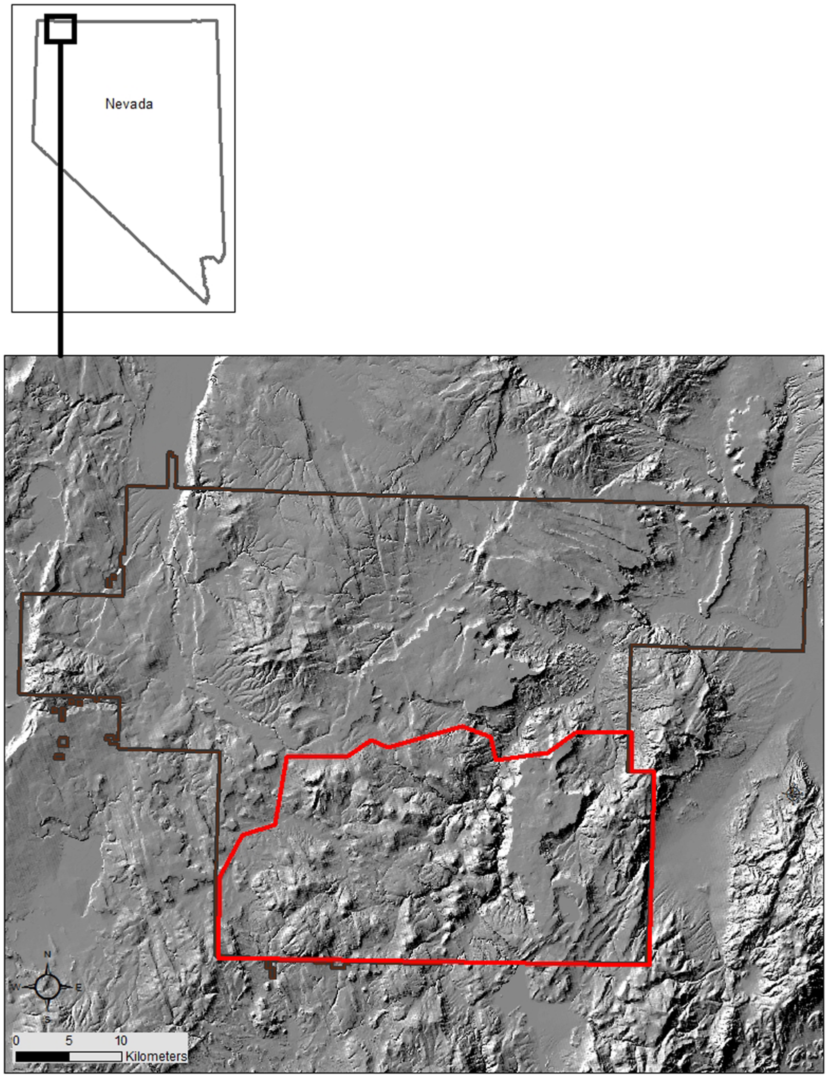

Fig. 1. Map of the study area (red perimeter polygon) inside of Sheldon National Wildlife Refuge (brown perimeter polygon) with the location of the refuge identified on a map of Nevada, USA.

reproductive stems were present and juvenile if it did not have reproductive stems. Density data was used to calculated plant species richness and diversity. We calculated plant diversity as the Shannon Diversity Index (H) (Krebs 1998). Forage utilization was measured in late September of 2009–2013 following the method described in Anderson and Curreir (1973) to quantify the level of horse use at each block. Soil aggregate stability and penetration resistance of the soil surface were measured in June 2013. Soil aggregate stability was measured at five locations in each treatment in each block with a Soil Stability Test Kit (Synergy Resource Solutions, Belgrade, MT). Penetration resistance of the soil surface was measured at five locations in each treatment in each block with a Geotester Pocket Penetrometer (Novatest, Italy).

Statistical analyses

We used repeated measures analysis of variance (ANOVA) models in PROC MIXED in SAS v.9.2 (SAS Institute, Cary, NC) with year as the repeated variable to evaluate the response of variables that were repeatedly measured with feral horse grazing and exclusion. Blocks and

Table 1. Average herbaceous utilization by feral horses at each site measured in September each year using the method described in Anderson and Curreir (1973). Sites grouped into grazing levels based on average utilization levels: low-moderate, <40%; high, >40%.

Site (block)	Utilization		Grazing level
	Mean (%)	SE (%)	
Tenmile	14	5	low-moderate
Buckaroo	61	8	high
Corral	53	4	high
Smith	48	10	high
Cottonwood	29	5	low-moderate

block by treatment interactions were treated as random variables. We also analyzed level of grazing influence on treatment response. Level of utilization was separated into two categories based on average utilization measured in September of 2009–2013: low-moderate (<40% utilization; n = 2) and high (>40% utilization; n = 3) (Table 1). Our categorization of utilization level was based on Holechek et al. (1999) synthesis of grazing studies. Covariance structures used in the repeated measure ANOVAs were selected using the Akaike's Information Criterion (Littell et al. 1996). Response variables that were not repeatedly sampled (soil aggregate stability and penetration resistance) were analyzed with ANOVAs in PROC MIXED in SAS v. 9.2 (SAS Institute, Cary, NC). Data that violated assumptions of ANOVAs were log-transformed prior to analyses. All figures and text report original data (i.e., non-transformed). For analyses, herbaceous cover and density were grouped into five functional groups: perennial grasses, Sandberg bluegrass, annual grasses, perennial forbs, and annual forbs. Sandberg bluegrass was treated as a separate functional group from the other perennial grasses because it is much smaller in stature and matures earlier (Davies 2008). Cheatgrass (*Bromus tectorum* L.) was the only annual grass detected at the study sites. Total herbaceous cover was the sum of all the herbaceous species cover. Shrub cover and density was separated into sagebrush and other shrubs. The other shrub category included all shrubs except for sagebrush. The two blocks that did not contain sagebrush were excluded from analyses of treatment effects on sagebrush. Differences between treatment means were considered significant at $P \leq 0.05$. Means are reported with one standard error.

Results

Perennial grass cover response to grazing treatments varied by grazing level (Fig. 2; $P = 0.047$). In areas with high levels of grazing, perennial grass cover was half of ungrazed exclosures ($P = 0.02$). We did not find evidence that perennial grass cover varied between areas with low-moderate levels of grazing and ungrazed exclosures ($P = 0.682$). Perennial grass density did not varied by treatment (Fig. 3) and did not vary by grazing level ($P = 0.227$ and 0.291, respectively). We also did not find any evidence that density (Fig. 3) or cover (Fig. 4) of any other herbaceous plant functional groups varied between grazed and ungrazed treatments or was influenced by grazing level or year ($P > 0.05$). Total herbaceous cover, bare ground, and litter did not vary between grazed and ungrazed treatment ($P = 0.195$, 0.304, and 0.979). Cover of biological soil crust did not differ between treatments ($P = 0.332$). Grazing level and year had no effect on total herbaceous cover, bare ground, litter, and biological soil crust cover ($P > 0.05$).

Sagebrush and other shrub cover did not differ between grazed and ungrazed treatments (Fig. 5) or among years ($P > 0.05$). In sagebrush occupied blocks, density of mature sagebrush was 2-fold greater in ungrazed exclosures compared to the grazed treatment (Fig. 6; $P = 0.005$). Sagebrush juvenile density followed a similar pattern, but statistically there was no difference between treatments (Fig. 6; $P = 0.124$). Other shrub density did not differ between treatments ($P = 0.768$). Grazing level and year had no effect on any of the shrub cover and density responses ($P > 0.05$).

Plant species richness did not differ between ungrazed exclosures (22.6 ± 1.2) and grazed treatments (22.5 ± 1.3) ($P = 0.990$). Plant species diversity (H) was 1.2-fold greater in exclosures (1.8 ± 0.2) compared to the grazed treatment (1.5 ± 0.2) ($P = 0.042$). Grazing level and year did not influence plant species richness or diversity ($P > 0.05$).

Soil aggregate stability averaged 1.5 greater in

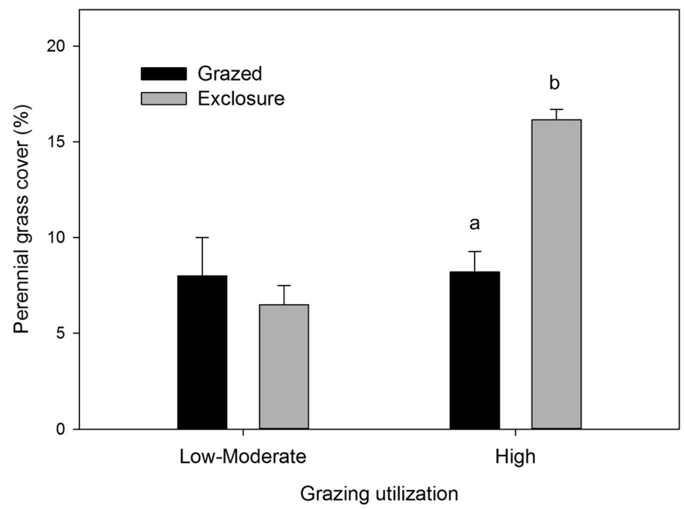

Fig. 2. Perennial grass cover (mean + SE), excluding Sandberg bluegrass, in feral horse grazed (grazed) and ungrazed (exclosure) treatments split by average grazing utilization levels: low-moderate, <40% utilization; high, >40% utilization. Utilization was determined in September in each year. Data presented from the fourth and fifth years after exclosures were constructed. Lowercase letters indicate difference between treatments ($P \leq 0.05$) at that grazing level.

ungrazed exclosures compared to the grazed treatment (Fig. 7; $P = 0.002$), but was not affected by grazing level ($P = 0.724$). Averaged across blocks, the grazed treatment required 2.5-fold more force to penetrate the soil surface compared to ungrazed exclosures ($P < 0.001$). Grazing level did not affect treatment response ($P = 0.200$).

Discussion

Our results suggest that feral horse grazing can affect some aspects of semi-arid ecosystems.

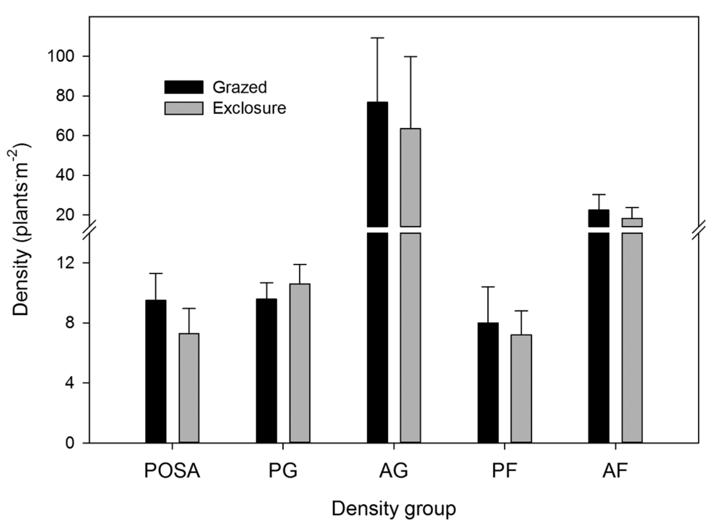

Fig. 3. Density (mean + SE) of different plant functional groups in feral horse grazed (grazed) and ungrazed (exclosure) treatments. POSA, Sandberg bluegrass; PG, perennial grasses excluding POSA; AG, annual grasses; PF, perennial forbs; and AF, annual forbs. Data presented from the fourth and fifth years after exclosures were constructed. Lowercase letters indicate difference between treatments ($P \leq 0.05$).

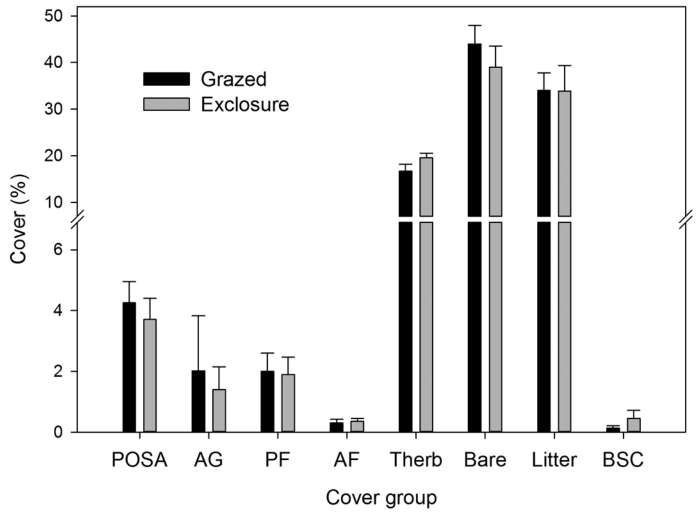

Fig. 4. Cover (mean + SE) of different cover groups in feral horse grazed (grazed) and ungrazed (exclosure) treatments. POSA, Sandberg bluegrass; AG, annual grasses; PF, perennial forbs; AF, annual forbs; Therb, total herbaceous; Bare, bare ground; Litter, ground litter; and BSC, biological soil crusts. Data presented from the fourth and fifth years after exclosures were constructed. Lowercase letters indicate difference between treatments ($P \leq 0.05$).

Similar effects have been reported for other large herbivores (Hobbs 1996, Augustine and McNaughton 1998, Olff and Ritchie 1998). Although previous research (Beever et al. 2003, 2008, Beever and Herrick 2006) suggested that feral horse use affects the sagebrush ecosystem, our results are the first study that empirically measured effects of free-roaming feral horses on the sagebrush steppe. One limitation of our study was that our study sites were relatively close to riparian areas with permanent springs or small creeks that can concentrate horse use (Crane et al.

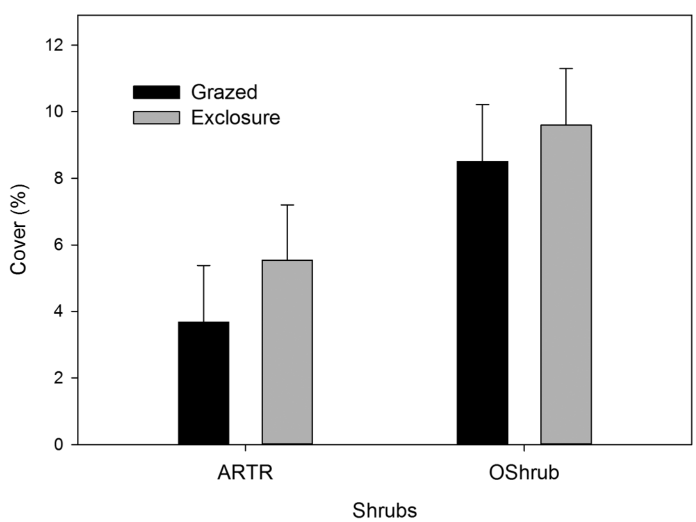

Fig. 5. Cover (mean + SE) of different shrub groups in feral horse grazed (grazed) and ungrazed (exclosure) treatments. ARTR, big sagebrush; Oshrub, shrubs other than ARTR. Data presented from the fourth and fifth years after exclosures were constructed. Lowercase letters indicate difference between treatments ($P \leq 0.05$).

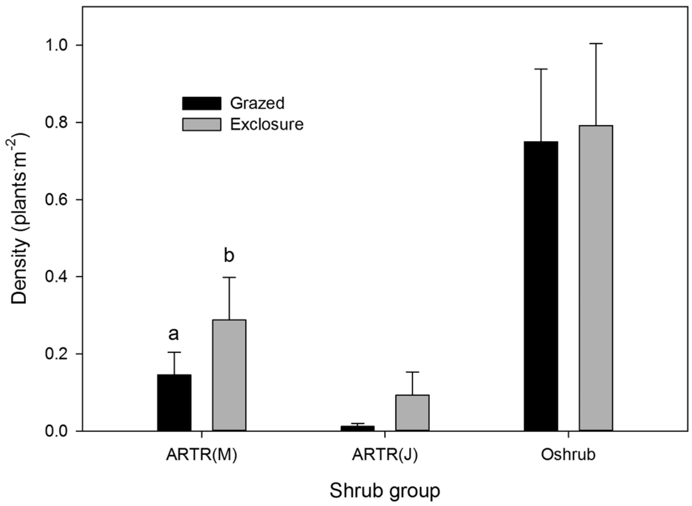

Fig. 6. Density (mean + SE) of different shrub groups in feral horse grazed (grazed) and ungrazed (exclosure) treatments. ARTR(M), mature big sagebrush; ARTR(J), juvenile big sagebrush; Oshrub, shrubs other than ARTR. Big sagebrush data are only from the three blocks that contained big sagebrush. Data presented from the fourth and fifth years after exclosures were constructed. Lowercase letters indicate difference between treatments ($P \leq 0.05$).

1997) and thus, horse effects may dissipate further from water sources. However, Ganskopp and Vavra (1986) reported that feral horses rapidly vacate watering areas after drinking. Feral horses will also travel long distances (up to 55 km) from water if forage is scarce and even disperse up to 8 km from water when forage is plentiful (Hampson et al. 2010). Herbaceous use at our study sites spanned a wide range of utilization levels (Table 1). Thus, our results can likely be extrapolated to a broad array of different feral horse use scenarios in arid and semi-arid ecosystems.

At higher levels of horse use there was about half the perennial grass cover in grazed compared to ungrazed exclosures; however, herba-

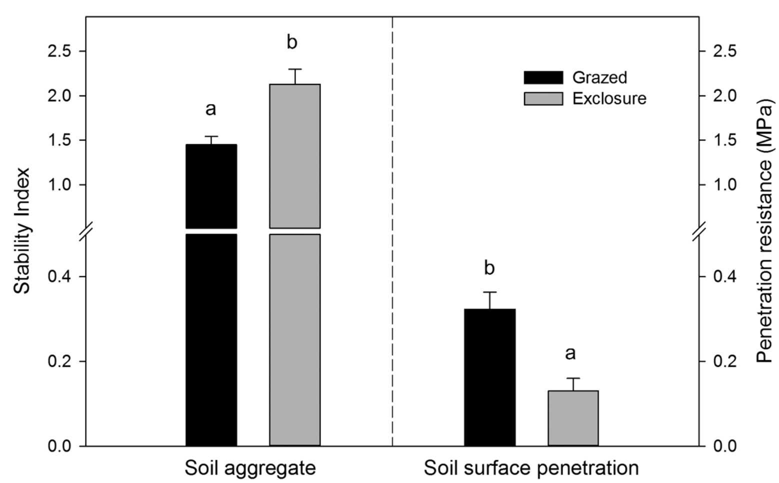

Fig. 7. Soil aggregate stability (left) and soil surface penetration resistance (right) in the feral horse grazed (grazed) and ungrazed (exclosure) treatments five years after exclosures were constructed. Lowercase letters indicate difference between treatments ($P \leq 0.05$).

ceous cover and density were generally not different between horse grazed and ungrazed areas in our study. In contrast, Beever et al. (2008) reported herbaceous cover and abundance generally varied between horse removed and horse occupied areas. Contrasting our results of no difference in perennial grass density, abundance of perennial grasses was greater in horse excluded areas in grasslands in Argentina (de Villalobos and Zalba 2010). In our study feral horses may have been affecting herbaceous cover and density, but recovery of herbaceous plant communities may not be rapid enough to detect differences with short-term exclusion of horses. Arid and semi-arid plant communities can be relatively slow to recover from disturbances. For example, sagebrush steppe plant communities can take several decades once a disturbing agent is removed for even partial recovery (Sneva et al. 1980, West et al. 1984, Anderson and Inouye 2001). Our study sites had less than half and about a third the density of perennial grasses and perennial forbs reported for relatively intact mountain big sagebrush communities in Davies and Bates (2010); further suggesting these plant communities may need more time to recover. Horses had been removed for two to three times longer in areas evaluated by Beever et al. (2008), thus over a longer time period we may see a similar herbaceous response to feral horse exclusion in our study area. De Villalobos and Zalba (2010) were also comparing longer term (7 years) horse exclusion to continuous feral horse grazing. Differences between de Villalobos and Zalba's (2010) and our results could also have been the result of different plant community and site characteristics.

In contrast to the lack of measured effects of feral horse grazing exclusion on herbaceous vegetation abundance, sagebrush density was two-fold higher in horse exclosures suggesting sagebrush recruitment may be limited by feral horses, probably though physical damage of sagebrush. Consumption of sagebrush may also be influencing its recruitment as feral horses will consume small quantities of sagebrush, though they primarily consume grass (Krysl et al. 1984, McInnis and Vavra 1987). In contrast to our results, Beever et al. (2008) did not find a difference in sagebrush frequency between horse grazed and horse removed areas. However, sagebrush density at our plots was low, about half of the density Davies and Bates (2010) reported for relatively intact mountain big sagebrush communities, suggesting that sagebrush was recovering from past disturbances. Feral horse effects on sagebrush density may be more evident when sagebrush is recovering from a disturbance event; thus, potentially explaining why we found a difference is sagebrush density between feral horse grazed and ungrazed areas and Beever et al. (2008) did not. Beever et al. (2008) also measured the frequency of sagebrush using a line-intercept, which may produce a different result than counting individual plants in a belt transect as we did.

Non-sprouting shrubs, such as sagebrush, usually do not recover before herbaceous vegetation after a disturbance (e.g., Davies et al. 2007, Bates et al. 2011). However, this type of research has primarily investigated the recovery of plant communities after fire, not the removal of an herbivore. Fire often completely removes non-sprouting shrubs, but not herbaceous vegetation (Davies et al. 2007, Bates et al. 2011). In contrast, areas where an herbivore was removed may have a residual population of the non-sprouting shrub to expedite its recovery. Anderson and Holte (1981) reported that with the cessation of heavy grazing by sheep and cattle in areas with sagebrush that sagebrush cover increased 42% in the next 15 years while no detectable change occurred in perennial grass cover, however, 10 years later perennial grass cover had increased more than 20-fold. Similar to our results, Manier and Hobbs (2006) found that the long term exclusion of large herbivores in shrub-steppe communities in the Rocky Mountains resulted in increased shrub cover, but did not affect grass cover. Thus our research and the results from Anderson and Holte (1981) suggest that sagebrush, if present in the community, may recover faster than perennial herbaceous vegetation or alternatively, some herbaceous plant groups may not change much with herbivore exclusion (Manier and Hobbs 2006). Additional research would be valuable to determine the mechanisms underlying these varying responses to herbivore removal.

We did not measure a statistically significant difference in juvenile sagebrush density, which seems to contradict our results of higher densities of mature sagebrush. However, average density of juvenile sagebrush in ungrazed areas was 7.8-fold greater than in horse grazed areas (Fig. 6). Thus, juvenile sagebrush density results are similar to mature sagebrush density, but not statically significantly different due to large variability. This may be due to the episodic nature of sagebrush recruitment (Perryman et al. 2001). The lack of a statistically significant difference may have also been due, in part, to the way we classified juvenile and mature sagebrush. Sagebrush was considered mature if it had reproductive stems, which can occur by the second growing season (Davies et al. 2014). Thus, relatively small, young sagebrush were classified as mature because they had reproductive stems. We saw a similar pattern in sagebrush cover, where we did not measure a statistically significant difference in sagebrush cover between grazed and ungrazed areas, but average sagebrush cover was 1.5-fold greater in grazed compared to ungrazed areas. Many of the sagebrush plants were relatively small and thus, we suspect as sagebrush grows larger that its cover may increase in ungrazed exclosures because of the higher density of sagebrush and become statistically different between ungrazed and grazed areas. The first year mountain big sagebrush produces reproductive stems plants are only about half the height and their canopy cover is less than a fourth of what they will grow to in the next couple of years (K. W. Davies, *unpublished data*). In support of our speculation, Beever et al. (2008) found that sagebrush cover was 1.1 to 2.2 times greater in long term (10–14 yr) horse removed areas compared to horse occupied areas.

Our results suggest effects of feral horses on species richness and diversity may vary by plant community and site characteristics (grazing-plant coevolution, climate, soils, etc.). Plant species diversity was greater in exclosures than feral horse grazed areas in our study. Beever et al. (2008) reported that species richness was greater in feral horse removed compared to horse occupied areas, but they found no difference in diversity between areas. Plant species richness and diversity response to excluding feral horses has been mixed. Species richness and diversity were greater in mountain grasslands in Argentina after seven years of feral horse exclusion (de Villalobos and Zalba 2010). Other studies have found that feral horse grazing exclusion promoted a decline in species richness in montane grasslands in Argentina (Loydi et al. 2012) and plant diversity in the Pryor Mountain Wild Horse Range (Fahnestock and Detling 1999). Variation in intensity and persistence of feral horse grazing likely also correlates to differences in diversity responses (de Villalobos and Zalba 2010). Similarly, Olff and Ritchie (1998) in synthesis of the literature reported that herbivore effects on plant diversity vary by abundance and type of herbivore, plant community composition, and environment characteristics.

Greater penetration resistance of the soil surface in feral horse grazed areas suggests that horse trampling compacted the soil. Beever and Herrick (2006) reported that penetration resistance of the soil surface was 3–15 times higher in horse occupied compared to horse removed areas. This difference was greater than we observed, but exclosures in our study had only excluded horses for half to a third of the time that horses had been absent from horse removed areas used by Beever and Herrick (2006). The greater penetration resistance of the soil surface in feral horse grazed areas than ungrazed exclosures may be a concern, because the soil environment influences many ecosystem processes (Beever and Herrick 2006). We found that soil aggregate stability was 1.5 times greater in feral horse exclosures compared to grazed plots, further demonstrating that feral horses were affecting important soil characteristics. In the Mojave Desert, removal of feral burros (*Equus asinus* L.) and domestic cattle did not produce a consistent soil aggregate stability response (Beever et al. 2006). However, Beever et al. (2006) only investigated soil aggregate stability with one to two years of grazing removal. Soil aggregate stability may take longer to respond and this likely explains why we found a difference with five years of feral horse exclusion.

Horse grazed areas are likely at an elevated risk of soil erosion compared to horse excluded areas. Infiltration rates are slower in areas with higher soil penetration resistance leading to increased runoff risk (Maestre et al. 2002, Aksakal et al. 2011). Similarly, a decline in aggregate stability increases the risk of erosion (Herrick et al. 2001). Soil erosion has the potential to result in irreversible declines in site productivity and stability and high environmental and economic costs (Pimentel et al. 1995). This suggests that the effects of feral horse trampling may have as much or even more influence on ecosystems than their selective consumption of plants. In support of this argument, Turner (1987) reported in marshes in Georgia that trampling, as it altered the soils, may be the more destructive component of grazing by feral horses.

Our results suggest that soil characteristics may need to recover to facilitate an herbaceous response. Alternatively, herbaceous vegetation may not have been substantially affected by feral horse grazing and a significant change in herbaceous vegetation with feral horse exclusion should not be expected. Other authors have also found that the exclusion of herbivores did not generally increased herbaceous cover and density (e.g., West et al. 1984, Manier and Hobbs 2006, Davies et al. 2009). However, Manier and Hobbs (2006) cautioned that their results should not be interpreted as evidence that herbivory had not influenced the herbaceous plant community because herbivory effects could be long lasting. Our study sites had less than half the density of perennial grasses found in relatively intact mountain big sagebrush communities (Davies and Bates 2010), thus, feral horses may have significantly influenced the herbaceous communities at our sites. Soil recovery may have to occur first because increased soil penetration resistance and decreased aggregate stability can reduce the amount of water available for plant growth (Boxell and Drohan 2009). In addition, high soil penetration resistance can restrict root growth (Ehlers et al. 1983, Bengough and Mullin 1991, Villamil et al. 2001). Soil compaction and trampling by large herbivores can also inhibit plant diversity (Olff and Ritchie 1998).

Our research agrees with previous research (Beever et al. 2003, 2008, Beever and Herrick 2006) that suggested feral horse grazing can cause significant ecological effects in sagebrush steppe and that these effects on sagebrush may influence sagebrush associated wildlife species (Beever and Brussard 2004, Beever and Aldridge 2011). Similarly, feral horses have been reported to negatively impact native wildlife in Argentina (Zalba and Cozzani 2004). Our research suggests that feral horse grazing is limiting the recruitment of sagebrush which likely prolongs the time that the plant community is unsuitable habitat for sagebrush associated wildlife species. This is especially concerning because sagebrush habitat is already limited for many sagebrush associated wildlife, such as Greater Sage-grouse (*Centrocercus urophasianus*) (Connelly et al. 2000, Crawford et al. 2004, Davies et al. 2011) and that fires, which remove sagebrush, are expected to become larger and more frequent with global climate change (Fulé 2008). The role feral horse grazing plays in limiting sage-grouse and other sagebrush associated wildlife conservation needs to be considered (Beever and Aldridge 2011). Additional research investigating sagebrush recruitment and longer term evaluation of the response of sagebrush and herbaceous cover and density to feral horse exclusion is needed to fully understand potential effects of feral horses on habitat for sagebrush associated wildlife.

Conclusions

Feral horse exclusion increased sagebrush density and plant species diversity and promoted recovery of important soil surface characteristics. Collectively, these results suggests that feral horse grazing at the utilization levels occurring in this study can affect the ecological function of semi-arid rangelands and may degrade the habitat value of these communities for associated wildlife. Feral horse grazing elevates the risk of soil erosion by increasing soil penetration resistance and decreasing soil aggregate stability. However, our study demonstrated that relatively short-term feral horse exclusion can initiate recovery of some variables. Other variables (e.g., perennial grass density and forb cover and density) may increase with extended horse exclusion, but long term studies are needed for verification. Though feral horse effects likely vary by intensity and frequency of use as well as a host of other factors, our results suggest that feral horses have some ecological impacts on semi-arid rangelands across a range of levels of utilization. Our results agree with Beever and Aldridge (2011) assertion that feral horses' value to society must be weighed against their ecological costs. The collective results from our study suggest that the effects of feral horses should be considered when developing conservation plans for sagebrush steppe rangelands and other semi-arid and arid ecosystems.

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