Grazing management effects on environmental quality of riparian and upland grassland ecosystems

by

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“You can’t prepare for everything life’s going to throw at you. And you can’t avoid danger. It’s there. The world is a dangerous place, and if you sit around wringing your hands about it, you’ll miss out on all the adventure.”

— Jeannette Walls
ABSTRACT

Grazing cattle in grasslands can impact many ecosystems services including the movement of sediment and nutrients to water bodies, biodiversity, and wildlife habitat. In riparian grassland ecosystems congregation of cattle in or near streams may increase the sediment, nutrient, and pathogen loading of surface water resources, however the impact of cattle on water bodies may be limited through pasture characteristics or management practices that reduce congregation of cattle in or near streams. The first study in this thesis was designed to determine the effects of pasture size, stream access, and off-stream water on the presence of cattle near pasture streams. In the first study the effects of an off-stream water site or limiting the stream access of cattle to stabilized sites on the presence of cattle in or near a pasture stream was measured in small (4.0 ha) and large (12.1 ha) pastures. Limiting stream access of cattle to stabilized sites reduced presence of cattle in or near streams. However, providing off-stream water sites affected congregation of cattle in or near streams relatively little. Regardless of management treatment, presence of cattle in and near the pasture stream was reduced in pastures with a larger proportion of grazing land outside of the riparian zone. As temperatures increased, the probability of cattle spending time in and near the pasture stream or tree shade increased, with a greater probability of presence in riparian shade occurring in small pastures. In upland grassland ecosystems, cattle grazing at elevated stocking densities has the potential to improve plant diversity, carbon sequestration, and wildlife habitat through soil disturbance, incorporation of plant litter into the soil profile, and removal of aboveground forage. A second study was designed to determine the effects of a single spring grazing event at two stocking densities with or without subsequent rotational grazing on plant community properties, soil characteristics, and wildlife habitat in upland grasslands. Soil structural characteristics, proportion of plant species,
and wildlife habitat were measured following no grazing or a single grazing event at elevated stocking densities with or without subsequent rotational grazing. Grazing at elevated stocking densities during periods of heavy rainfall reduced the proportion of cool season grass species for 14 months allowing succession of annual grass followed by legume species. However, after 14 months the proportion of cool season grass species returned to pre-grazing levels. The maximum height with 50% visual obstruction from vegetation was reduced for 12 months following grazing, but there were few subsequent differences. Although a single spring grazing event at either a high or moderate stocking density during periods of heavy rainfall increased soil bulk density, penetration resistance to a depth of 10 cm, and bare ground, grazing at a higher stocking density had less impact on soil structural characteristics likely because of a shorter stocking duration. Further research is necessary to determine if shade can be used to influence cattle distribution in pastures and rangelands in addition to more comprehensive research on the effects of periodic grazing at elevated stocking densities on soil aggregate stability, soil organic carbon, soil erosion, and wildlife habitat in Midwest grassland ecosystems.
CHAPTER 1. GENERAL INTRODUCTION

Grassland ecosystems are dynamic, resource rich landscapes, which comprise approximately 30% of earth’s land area and yield 35% of global plant growth (net primary productivity, NPP; Huston and Wolverton, 2009). Along with contributing a significant proportion of the world’s NPP, the dense perennial plant growth reduces water runoff velocity, while the extensive root systems and associated soil fauna increase soil stability and soil organic matter (Miller and Jastrow, 1990; Singh et al., 2004; Gyssels et al., 2005). Combined, perennial grassland plant communities and the associated soil fauna reduce soil erosion, making grasslands important ecosystems in maintaining water quality (Rillig et al., 2001; Reubens, 2007). Along with maintaining water quality, grasslands provide wildlife habitat and feed sources for bird and insect species unique to open grassy landscapes (Bond and Parr, 2010; Da Silva 1997). Furthermore, grassland ecosystems are integral landscapes for native and domestic grazing mammals as they produce more herbivore available biomass than any other terrestrial habitat (McNaughton, et. al., 1989).

The impact of grazing mammals in grassland ecosystems is the subject of many reviews as a result of their dynamic short- and long-term effects on soil characteristics and hydrology, botanical composition, nutrient flows, and wildlife habitat (McNaughton and Georgiadis, 1986; Milchunas et al., 1988; Milchunas and Lauenroth, 1993; Jones, 2000; Greenwood and McKenzie, 2001; Asner et al., 2004; Lunt et al., 2007). The weight and disturbance of grazing mammals on the soil surface impact soil characteristics such as porosity, organic matter, and hydrology (Manley et al., 1995; Mcdowell et al., 2003; Weber and Gokhale, 2011). Furthermore, runoff from congregation areas, stream bank erosion, direct fecal deposition, and stream bed
disturbance from grazing animals may impact sediment, nutrient, and pathogen loading of pasture streams (Haan et al., 2006; Schwarte et al., 2010; Belsky et al., 1999).

Grazing mammals also influence the population of dominant plant species (Todd and Hoffman, 1999) and plant community diversity (Hickman et al., 2004; Collins et al., 1998), which influence habitat and structure available to wildlife (Tews et al., 2004; Fuhlendorf et al., 2001). Because of the impact of grazing mammals on plant species diversity and their recycling of nutrients in feces and urine (Ruess and McNaughton, 1988) in grassland ecosystems, grazing may stabilize or increase nutrient and energy flows in grassland ecosystems (Loreau, 1995; Mazancourt et al., 1999) depending on landscape characteristics, precipitation patterns, and nutrient availability (Pastor and Cohen, 1997; Semmartin et al., 2004). However, grazing density and duration, as affected by biotic or abiotic pressures such as fencing or presence of water or shade, have a significant influence on the distribution and subsequent impact of grazing mammals on grassland ecosystems (Krausman et al., 2009; Sigua and Coleman, 2009).

**Thesis organization**

This thesis contains an overview of previous literature relating to the ecological significance of grasslands and the impact of large herbivores on this ecosystem. The literature review will be followed by manuscripts for submission for publication. The first manuscript evaluated the effects of pasture size on the efficacy of off-stream water or restricted stream access to alter the spatial/temporal distribution of grazing cows and has been submitted to the Journal of Animal Science. The second manuscript evaluated the impact of grazing beef cattle at high densities on botanical composition, soil quality, and wildlife habitat in grasslands and will be submitted to Rangeland Ecology and Management. The manuscripts will be followed by
general conclusions section summarizing and interpreting the impacts of grazing management on grassland ecosystems and identifying areas of future research.
CHAPTER 2. REVIEW OF THE LITERATURE

Importance of Grassland Ecosystems

Soils within grassland ecosystems are some of the most productive in the world (White et al., 2000); however, replacement of perennial grassland plants with annual cropping systems has the potential to sharply increase soil erosion and runoff of soil nutrients, reduce soil quality, and increase the eutrophication of water resources (Cambardella and Elliot, 1992; Heathcote et al., 2013). The dense growth and litter of perennial grassland plants reduces soil erosion by reducing water velocity over the soil surface which reduces the amount of sediment runoff can remove from the soil surface and transport to a new location (Blackburn et al., 1992; Huang et al., 1999; Heathcote et al., 2013). In addition, dense plant root systems and soil biota improve soil aggregate stability (Gyssels et al., 2005; Barrios, 2007). By reducing water velocity and increasing aggregate stability and formation, grassland plants and soil biota increase water infiltration rates (Franzluebbers, 2002) and storage in the soil profile (Dominati et al., 2010; Arthur et al., 2013) which increases net primary productivity (NPP; Quinton et al., 2010).

Grasslands, which comprise approximately 30% of the earth’s land area, store approximately 40% of the total soil carbon (White et al., 2000; Wang and Fang, 2009). Carbon can be stored in the soil profile through aggregation which is mediated by soil organic carbon, fungal hyphae, and other factors (Bronick and Lal, 2004) and protects soil organic matter from oxidation by microbes (Six et al., 2002). In addition to storage in soil aggregates, complex carbon molecules such as lignin from plant litter and carbon-rich polyphenols exuded by plant roots represent a significant fraction of stable soil organic carbon as a result of their own chemical stability (Six et al., 2002) Although grasslands already store large amounts of carbon,
Schuman et al. (2002) reported rangelands in the U.S. could potentially store an additional 61 million Mg C · yr\(^{-1}\) by reducing overgrazing and improving rangeland management to increase carbon storage, making rangelands a potential method to mitigate climate change (Soussana et al., 2010). However, storage of soil carbon is highly variable depending on soil type, precipitation, vegetation, management, and many other factors making it difficult to quantify (Bird et al., 2002; Derner et al., 2006; Nosetto et al., 2006).

Although grasslands are typically managed for livestock production, of 136 terrestrial ecoregions that the World Wildlife Fund-US has identified as “outstanding examples of the world’s diverse ecosystems”, 35 are considered grassland ecoregions (Olson and Dinnerstein, 1998; White et al., 2000). The plant species and functional group diversity within grasslands improves ecosystem NPP (Costanza et al., 2007) and carbon sequestration (Fornara and Tilman, 2008). Greater productivity with increased plant species diversity is likely a result of nutrient uptake from different regions of the soil profile (Tilman et al., 1997) and niche complementarities or synergistic relationships between plant species, such as legume species fixing atmospheric nitrogen in the soil which is accessible to grass species (Pirhofer-Walzl et al., 2012). In addition, plant community diversity within grasslands reduces the impact of drought on productivity and enhances the recovery of productivity following drought (Deak et al., 2009; Van Ruijven and Berendse, 2010).

The diversity of grassland plant communities and diverse mosaics of plant successional species increases the ability of grasslands to support soil biota and insects (Knops et al., 1999; Zak et al., 2003), and the habitats available for native wildlife species (Fuhelndorf and Engle, 2001; Fuhlendorf, 2006). Zak et al. (2003) observed greater microbial respiration and nitrogen mobilization in more diverse plant communities, likely a result of more available plant detritus in
more diverse plant communities. Similarly, although larger insect populations in more diverse plant communities are likely a result of greater productivity, inclusion of plants with lower carbon:nitrogen ratios in plant communities, such as forbs, are more likely to increase insect populations (Haddad et al., 2001). Insect populations of grasslands are important feed sources for many wildlife species making plant community diversity an important aspect when managing for native wildlife (Conant and Collins, 1998; Benton et al., 2002). However, spatial distribution and types of plant species within grassland habitats is important for grassland wildlife such as bobwhite quail which typically use areas less than 6 ha and prefer nesting and brood rearing areas with distinct plant communities and ground cover (Stromberg, 1990; Taylor et al., 1999).

In order to maintain the diverse range of plant species and habitats in grassland ecosystems for the wildlife, insects, and soil biota they support, periodic disturbances are necessary to reduce the dominance of more competitive plant species (Hobbs and Huenneke, 1992; Collins, 1987).

Historically native grassland ungulates, such as bison on the American Great Plains, were one source of disturbance in grassland ecosystems (Hobbs and Huenneke, 1992; Knapp et al., 1999; Truett et al., 2001). In the Konza prairie, bison preferentially graze dominant grass species and avoid forbs species, which reduces the competition from grasses for soil nutrients and allows a more diverse plant community to establish (Knapp et al., 1999). In addition to grazing, large herds of bison likely increased disturbance of grassland soil and carried plant seeds which improved the germination and distribution of native grassland plants (Milchunas, 1993; Couvreur et al., 2004; Rosas et al., 2008).
Impacts of Grazing on Water Quality

Although grasslands limit sediment, nutrient, and pathogen loading of surface water resources (Blackburn et al., 1992; Gyssels et al., 2005), cattle congregating near pasture streams to meet their need for thirst, hunger, and thermoregulation reduce stream bank stability and increase soil erosion in pastures and rangelands throughout the U.S. (Bailey, 2005; Bilotta et al., 2007; Magner et al., 2008). Localized stream bank instability from cattle entering and exiting the stream likely increases the risk of stream bank erosion which will likely increase sediment and phosphorus loads in pasture streams (Agouridis et al., 2005). In addition, soil compaction and frequent defoliation by grazing animals reduce the root growth of plants, further decreasing stream bank stability (Evans, 1973; Unger and Kaspar, 1994). Likely as a result of the impact of grazing on stream bank stability, Owens et al. (1996) found that fencing cattle out of a heavily grazed riparian zone reduced annual soil loss by 40%. However, although Schwarte et al. (2011) found stream bank erosion contributed 94.4% of phosphorus to a Midwest pasture stream, stream bank erosion was not affected by grazing or grazing management. This result suggests that fluvial processes and stream morphology also influence stream bank erosion (Allan and Castillo, 1995).

Although stream bank erosion increases sediment within pasture streams (Schwarte et al., 2011), treading on the soil surface and plant defoliation, particularly in congregation areas, can reduce the stability of soil surface particles and increase the risk of sediment movement by surface runoff (Pionke et al., 2000; Russell et al., 2001; Pande and Yamamoto, 2006). In addition, at high stocking rates, grazing can reduce the soil water infiltration rate which increases the amount of soil surface runoff (Mwendera and Saleem, 1997). As the amount of water runoff on the soil surface increases, there is a greater risk for soil detachment and sediment transport.
(Fiener and Auerswald, 2003). Furthermore, surface runoff from pastures near streams can also carry nitrates and fecal coliforms from urine and feces deposited near streams (Larsen et al., 1994; Agouridis et al., 2005).

In comparison to cattle presence surrounding pasture streams, cattle congregating in pasture streams have a direct impact on water quality (Agouridis et al., 2005). Haan et al. (2010) found as cattle spend more time in pasture streams, there is a similar increase in the proportion of fecal depositions into the stream. In addition, cattle traveling in and through pasture streams is associated with increased levels of total nitrogen and fecal indicator bacteria (Davies-Colley et al., 2004). Cattle traffic through pasture streams also resuspends sediment from the streambed which increases turbidity and likely allows greater movement of sediment-bound phosphorus (McDowell and Sharpley, 2001; Davies-Colley et al., 2004). However, the impact of grazing livestock on water bodies can be influenced by management practices which reduce the time livestock can spend in or near pasture and rangeland streams (Bailey, 2004; Agouridis et al., 2005).

**Factors Controlling the Temporal/ Spatial Distribution of Grazing Animals**

In western rangelands, cattle prefer areas with more available forage containing greater concentrations of crude protein and digestible dry matter (Smith et al., 1992; Ganskopp and Bohnert, 2008). However, forage composition and quality do not always impact the temporal and spatial distribution of cattle in Midwest pastures (Bear et al., 2012). Likely due to differences in precipitation and soil type (Sala et al., 1988; Rinehart et al., 2006), forage quality within pastures in arid ecosystems, like western rangelands, can vary more than in pastures in Midwest grasslands (Ganskopp and Bohnert, 2009; Schwarte et al., 2011). As a result, cattle distribution
in Midwest grasslands is likely less influenced by forage quality and productivity. In addition to forage quality, tree shade can also potentially influence cattle distribution. Shade reduces the amount of solar radiation which reaches cattle, thereby reducing their internal body temperature during high ambient temperatures (Blackshaw and Blackshaw, 1994; Tucker et al., 2008). Likely as a result of the effect of shade on body temperature, Franklin et al. (2009) and Bear et al. (2012) found cattle were more likely to spend time under or near trees as temperature increases.

Providing a source of off-stream water to influence the distribution of cattle in and near pasture streams has been the subject of many studies with varying results (Ganskopp, 2001; Haan et al., 2010; Schwarte et al., 2011; Kaucner et al., 2013; Rigge et al., 2013). However, streams and stream riparian areas are also sources of thermoregulation and forage, potentially influencing the effects of off-stream water sites on cow distribution (Bailey, 2005). In addition, the distribution of cattle in pastures is related to the distance and elevation from water sources (Valentine, 1947; Roath and Krueger, 1982) making water placement an important factor influencing the distribution of cows within a pasture (Rigge et al., 2013). The influence of distance and elevation from water sources on cattle distribution is likely a result of the energy required to travel to and from grazing areas to water sites (Di Marco and Aello, 1998).

Contrary to the use of off-stream water sites, restricting access of cattle to pasture streams has consistently reduced the presence of cattle near pasture streams (Bryant, 1982; Bailey, 2004; Schwarte et al., 2011). However, although restricting access of cattle to pasture streams reduces the stream load of bovine fecal coliforms (Hagedorn et al., 1999), the proportion of bare ground did not differ between non-grazed riparian areas and riparian paddocks when cattle were not allowed to graze riparian paddocks below a residual sward height of 10 cm or longer than 4 days per rotation, commonly referred to as flash grazing (Schwarte et al., 2011). Furthermore, Haan et
al. (2006) found no differences in the phosphorus loss during rainfall simulations between upland paddocks that were ungrazed or rotationally grazed to a residual height of 10 cm. Therefore, management practices such as rotational or flash grazing, which control cattle distribution and the frequency of defoliation likely improve the functioning of riparian ecosystems as compared to continuous grazing (Schwarte et al., 2011).

Although limiting the access of cattle to riparian areas with fencing reduces the potential impact of cattle on stream water quality (Haan et al., 2010), pasture shape and size can also influence the presence of cattle near pasture streams (Bear et al., 2012). In south central Iowa, Bear et al. (2012) found cattle presence near water bodies was reduced as the percentage of grazing land near water bodies was reduced. As a result, management practices that physically reduce the presence of cattle near pasture streams may be most effective in long, narrow pastures which follow a stream course.

**Impacts of Grazing on Grassland Plants**

The impacts of grazing on grassland plants depend on many factors including the plants tolerance to grazing (Guitian and Bardgett, 2000), soil moisture conditions (Drewry et al., 2008; Meneer et al., 2005), and plant stress prior to grazing (Oesterheld and McNaughton, 1991). Plants with more grazing tolerance or the ability to recover productivity following grazing have lower root mass following grazing likely because of a shift in energy allocation from below-ground processes to shoot growth. In comparison, less grazing tolerant plants increase their energy allocation to their roots (Oesterheld, 1992; Guitian and Bardgett, 2000). Furthermore, grazing of Kentucky Bluegrass (*Poa pratensis* L.), a grazing tolerant plant, promoted carbon exudates from roots which stimulated growth of rhizosphere microbial populations. Stimulating
the growth of rhizosphere microbial populations increased the plant available nitrogen and plant nitrogen uptake, potentially increasing growth rate (Hamilton and Frank, 2001). In addition to grazing tolerance, the effects of grazing also depend on the plant stress prior to grazing (Oesterheld and McNaughton, 1991). Plants in soils with lower levels of available nutrients had a lower growth rate post-grazing in comparison to pre-grazing likely because of the removal of stored nutrients and less ability to restore lost nutrients when soils have limited available nutrients (Ferraro and Oesterheld, 2002). Furthermore, grazing during periods of higher levels of soil moisture can increase the damage to plant growing points from treading while greater bulk density from grazing during wet soil conditions can reduce the ability of roots to elongate in the soil profile (Drewry et al., 2009; Meneer et al., 2005).

Although defoliation of individual plants reduced plant production by an average 52%, grazing at the ecosystem level reduced aboveground net primary productivity by less than 20% (Trlica and Rittenhouse; 1993). At the ecosystem level, grazing increases return of nutrients for plant growth and promotes plant community diversity (Ferraro and Oesterheld, 2002; Hickman et al., 2004). Grazing cattle in grassland ecosystems promote diversity in grasslands by creating disturbance at the soil surface which allows germination of forbs and annual plants (Bakker and Olff, 2003), while removing portions of the above ground plant biomass reduces the competition of dominant grasses for nutrients and sunlight (Olff and Ritchie, 1998; Wu et al., 2009). However, the response in diversity and productivity of plant communities varies between stocking densities, defined as the weight of animals per unit area (Hickman et al., 2004). As stocking density increases, grazing animals are forced to graze the available forage more evenly and be less selective, potentially decreasing the competition for nutrients from less palatable species (Barnes et al., 2008; Olff and Ritchie, 1998). In addition, at increased stocking densities,
treading damage to plants is likely more significant, thereby, reducing the regrowth of established plants (Pande and Yamamoto, 2005).

**Impacts of Grazing on Grassland Soils**

The impact of livestock grazing in grassland ecosystems can have a significant impact on soil structural characteristics and has been the subject of many reviews (Greenwood and McKenzie, 2001; Bilotta et al., 2007; Drewry et al., 2008). Improving soil structure is related to increasing macroporosity from formation of large soil aggregates (Mueller et al., 2013). However, when soil loading exceeds the soil’s strength, macroporosity is reduced, resulting in increased soil penetration resistance and bulk density, along with reduced water infiltration rates (Franzleubbers, 2002; Horn et al., 1995). Daniel et al. (2002) found rotational grazing increased penetration resistance and bulk density in the upper 10 cm of soil compared to no grazing in western rangelands. Nevertheless, rotational grazing has less impact on soil structural characteristics than continuous grazing likely as a result of rest periods which allow soil structure and plants to recover following grazing (Chanasyk and Naeth, 1995; Teague et al., 2011).

Although Teague et al. (2011) found water infiltration rate was reduced in pastures that were rotationally grazed, rotational grazing does not always reduce water infiltration rates in comparison to no grazing (Haan et al., 2006). Differences in the impact of rotational grazing on soil structural characteristics may be a result of differences in the timing and duration of grazing episodes, however antecedent soil structure is also likely a factor on the impact of grazing on soil structural characteristics (Van Haveren, 1983; Murphy et al., 2004).

The effects of grazing on soil structural characteristics are, in part, dependent on soil moisture (Bilotta et al., 2007; Drewry et al., 2008). Increasing moisture in the soil profile
increases the risk of compaction as a result of water reducing the stability of soil structure (Mulholland and Fullen, 1991; Patto et al., 1978). In addition, increasing the frequency of grazing during high soil moisture conditions reduces soil strength potentially increasing the risk of compaction (Scholefield and Hall, 1985). However, soil moisture is not the only factor affecting soil structural stability. Soil texture and mineralogy can also influence the impact of grazing on soil structure likely as a result of their impact on innate soil structural characteristics (Van Haveren, 1983; Scholefield and Hall, 1985; Bilotta et al., 2007).

In addition to the effect of grazing cattle on soil structural characteristics, cattle can have a significant impact on nutrient cycling in grassland ecosystems by increasing the rate at which nutrients are returned to the soil and changing the composition of the plant community (Chaneton et al., 1996; Pastor and Cohen, 1997; Semmartin et al., 2008). Feces and urine from grazing animals degrade at a faster rate than plant litter increasing the rate of nitrogen cycling through grazed grassland ecosystems (Reuss and McNaughton, 1987; Day and Detling, 1990). Furthermore, Semmartin et al. (2008) found litter from ryegrass (Lolium Multiflorum) decomposed at a faster rate when it was derived from plants in grasslands with a history of grazing likely a result of lower lignin: nitrogen ratios in plants from grasslands grazed in previous years than plants without a grazing history. However, the effects of a grazing history on plant chemical composition were species-specific. Litter from bahiagrass (Paspalum dilatatum) with a grazing history decomposed at a slower rate and the lignin:nitrogen ratio was greater. The decomposition of plant litter is also affected by available nutrients in the soil profile. Although grazing has no effect on the amount of nitrogen in the soil profile, grazing increased the proportion of nitrogen in the rooting zone (0-30 cm) of plants which increases the decomposition rate of plant litter (McNaughton et al., 1997; Tracy and Frank, 1998; Schuman et al., 1999).
As mentioned previously, grazing can change the species composition of plant communities which influences nutrient cycling and soil fauna through differences in plant litter chemical composition and plant root exudates (Pastor and Cohen, 1997; Marschner et al., 2001; Zak et al., 2003). Moretto et al. (2001) found the decomposition of a less palatable grass in semi-arid grasslands took longer than more palatable grasses likely as a result of a higher lignin:nitrogen ratio in less than more palatable grasses which, if grazing selectivity increases the proportion of less palatable grasses, could potentially reduce the rate of nutrient cycling. However, the rate of nutrient cycling is also dependent on the decomposition of plant litter by soil fauna (Tracy and Frank, 1998). Litter from more diverse plant communities support a larger and more diverse microbial community which potentially increases the rate of plant litter decomposition (Tracy and Frank; Bardgett and Shine, 1999). Furthermore, plant root rhizospheres support a microbial population specific to each plant species; as a result, changes in plant community composition from grazing will influence soil microbial populations (Westover et al., 1997; Grayston et al., 1998). Effects of grazing animals on soil structural characteristics can also impact soil microbial populations (Wardle, 1992). Reducing soil pore space from compaction as a result of greater stocking rates has reduced the population of micro-arthropods (Bardgett et al., 1993; Byers et al., 2000). However, Tom et al. (2006) found grazing for short periods at high stocking densities increased the population of soil organisms likely as a result of ecosystem disturbance which increased plant species diversity yet had little impact on soil structural characteristics.

Although, globally grasslands store an average 123- 154 Mg C · ha\(^{-1}\) (White et al., 2000), Schuman et al. (2002) found that US rangelands have the potential to store an additional 0.1 to 0.3 Mg C·ha\(^{-1}\)·year\(^{-1}\). To increase the storage of carbon in grassland soils, management strategies
which reduce overgrazing, improve soil aggregation, and increase species diversity of the plant community can be implemented (Balesdent et al., 2000; Conant and Paustian, 2002; Fornara and Tilman, 2008). However, Stewart et al. (2007) found storage of carbon in the soil profile likely reaches a point of saturation. Similarly, Six et al. (2002) suggested a model which decreased the capacity of soils with a greater carbon content to store additional carbon. Carbon storage in soils with a low capacity to store additional carbon may be limited by soil chemical and physical properties which are difficult to influence without intensive management and extensive soil amendments (Post and Kwon, 2000; Jastrow et al., 2007). Additionally, although grazing lands likely have potential to store added carbon in the soil profile, changes in soil carbon from management practices are small in comparison to changes from precipitation patterns and may take several years to be distinguished (Yang et al., 2008, Parsons et al., 2009).

Impacts of Grazing on Wildlife habitat

Native wildlife in grassland ecosystems rely on landscape heterogeneity and species diversity within plant communities (Fuhlendorf and Engle, 2001; Krausman et al., 2009). Grassland songbird species prefer a wide range of vertical structure for cover from predators (Patterson and Best, 1996). Furthermore, diverse plant communities, rich in forb species, attract insects and provide a large selection of seeds which are feed sources for native birds (Siemann et al., 1998; Knops et al., 1999; Martin et al., 2000). Ground nesting grassland birds prefer habitats with a diverse range of litter and plant species. Areas rich in forbs serve as feeding areas for ground nesting birds, however, litter on the soil surface limits travel of young chicks. As a result, ground nesting birds such as bobwhite quail prefer feeding areas with 25-50% bare ground (Taylor et al., 1999; White et al., 2005). In contrast, nesting areas for bobwhite quail have higher
levels of litter and perennial plant species which provide cover from aerial and land predators (Collins et al., 2009). However, the spatial distribution of habitats is also important as the home range of bobwhite quail coveys is typically less than 6 ha (Stromberg, 1990). In addition to attracting insects for grassland bird, diverse plant communities with greater proportions of forbs and legumes have the potential to increase the forage quality for large native grassland herbivores (McGraw et al., 2004).

Disturbances such as fire and grazing have the potential to create and maintain the diverse mosaics of habitats necessary for native wildlife in grassland ecosystems (Fuhlendorf and Engle, 2001; Harper, 2007). Grazing animals have the potential to improve wildlife habitat by increasing the proportion of bare ground and plant community diversity (Hickman et al., 2004; Dekeyser, 2013). Increasing the proportion of bare ground has the potential to create brood rearing habitat for ground nesting birds while establishment of annual forb species in more diverse plant communities increases available feed for grassland birds (Knops et al., 1999; Collins et al., 2009). Furthermore, grazing improves forage digestibility for wild herbivores. Frisina (1992) found grazing in a rest-rotation management system could be used to increase the forage quality in wintering areas for moose in Montana. However, grazing must be managed to promote habitat for wildlife species. Current grazing management in production livestock systems which focus on maintaining homogenous productive pastures does little to support native wildlife species (Krausman et al., 2009). Overgrazing also severely limits the ability of grassland ecosystems to support native wildlife in addition to reducing the carrying capacity of pastures for livestock (Harper, 2007; Drewry et al., 2008)
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Pasture size effects on the ability of off-stream water or restricted stream access to alter the spatial/temporal distribution of grazing beef cows\textsuperscript{1,2,3}

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Abstract

For two grazing seasons, effects of pasture size, stream access, and off-stream water source on cow distribution relative to a stream were evaluated in six 12.1-ha cool-season grass pastures. Two pasture sizes (small [4.0-ha] and large [12.1-ha]) with three management treatments (unrestricted stream access without off-stream water [U], unrestricted stream access with off-stream water [UW], and stream access restricted to stabilized sites [R]) were alternated between pasture sizes every 2 wk for five consecutive 4-wk intervals in each grazing season. Small and large pastures were stocked with 5 and 15 August-calving cows from mid-May through mid-October. At 10-min intervals, cow location was determined with GPS collars fitted on 2 to 3 cows in each pasture and identified when observed in the stream (0-10 m from the stream) or riparian (0-33 m from the stream) zones and ambient temperature was recorded with on-site weather stations. Over all intervals, cows were observed more ($P \leq 0.01$) frequently in the stream and riparian zones of small than large pastures regardless of management treatment. Cows in R pastures had 24 and 8% less ($P < 0.01$) observations in the stream and riparian zones than U or UW pastures regardless of pasture size. Off-stream water had little effect on the presence of cows in or near pasture streams regardless of pasture size. In 2011, the probability of cow presence in the stream and riparian zones increased at greater ($P < 0.04$) rates as ambient temperature increased in U and UW pastures than in 2010. As ambient temperature increased, the probability of cow presence in the stream and riparian zones increased at greater ($P < 0.01$) rates in small than large pastures. Across pasture sizes, the probability of cow presence in the stream and riparian zone increased less ($P < 0.01$) with increasing ambient temperatures in R than U and UW pastures. Rates of increase in the probability of cow presence in shade (within 10 m of tree driplines) in the total pasture with increasing temperatures did not differ between
treatments. However, probability of cow presence in riparian shade increased at greater \((P < 0.01)\) rates in small than large pastures. Pasture size was a major factor affecting congregation of cows in or near pasture streams with unrestricted access.

**Key words:** beef cow, grazing, distribution, pasture size, off-stream water, stream access

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**Introduction**

Sediment and phosphorus loading of lakes and rivers is greater from pastures and rangelands than other land uses in the Midwest (Downing et al., 2000; Alexander et al., 2008). Grazing cows in riparian areas may increase risks of sediment, nutrient, and pathogen loading of pasture streams by accelerating bank erosion (Crider, 1955; Svejcar and Christiansen, 1987; Belsky et al., 1999; Tufekcioglu, 2010) and increasing fecal cover near pasture streams (Belsky et al., 1999; Bear et al., 2012a).

Impacts of grazing cows on riparian areas are likely related to the amount of time cows spend within the area (Agouridis et al., 2005; Haan et al., 2010). Restricting stream access to stabilized sites in continuously stocked pastures or limiting stream access to a riparian paddock in rotationally stocked pastures reduced the proportion of time that cows were in or near pasture streams compared to continuously stocked pastures with unrestricted stream access (Haan et al., 2010; Schwarte et al., 2011). Although Rigge et al. (2013) found that off-stream water at distances of 200 to 1250 m from a stream was optimal to maintain live standing vegetation in ephemeral stream channels in western rangelands, the efficacy of off-stream water on the temporal/spatial distribution of cows is inconsistent (Porath et al., 2002; Byers, et al., 2005, Schwarte et al., 2011). In a study on five farms, Bear et al. (2012b) observed that pasture size,
shape, and shade distribution affected the proportion of time that grazing cows were in or near pasture streams with unrestricted stream access. These physical characteristics may impact the effectiveness of practices used to alter the temporal/spatial distribution of grazing cows.

Therefore, a study was conducted to evaluate the effects of pasture size on the influence of providing off-stream water or restricting stream access to stabilized crossings on the amount of time beef cows spend in and within 33 m of streams in Midwestern cool-season grass pastures.

**Materials and Methods**

All procedures for animal use in this experiment were reviewed and approved by the Institutional Animal Care and Use Committee at Iowa State University (Protocol Number 3-07-6325-B).

*Treatments*

Six adjoining 12.1-ha cool-season grass pastures at the Iowa State University Rhodes Research Farm (lat 42° 00’N, long 93° 25’W) were used during the 2010 and 2011 grazing seasons. Pastures were not fertilized or mowed during or at least 6 years prior to the experiment. A 141-m segment of a perennial flowing stream bisected each pasture which contained a mixture of smooth bromegrass (*Bromus inermis* L.) and reed canarygrass (*Phalaris arundinacea* L.) with lesser amounts of tall fescue (*Lolium arundinaceum* Schreb. Darbysh), Kentucky bluegrass (*Poa pratensis* L.), and legumes.

In five consecutive 4-wk intervals, cows were assigned to treatments of unrestricted stream access (U), unrestricted stream and off-stream water access (UW), or stream access restricted to a stabilized stream crossing (R) at pasture sizes of 4.0 (small) or 12.1 (large) ha.
(Figure 1). To duplicate treatments within each pasture size in each 4-wk interval, pasture sizes within each treatment were assigned to a pasture for 2 wk and switched the following the 2 wk. Cows in small pastures were limited to pasture lowlands (the stream and 2.0 ha on either side of the stream) by temporary electric fence. Cows in large pastures were allowed access to both the lowlands and uplands (4.0 ha on either side of the stream beyond the lowlands) of the pasture. By limiting cows in small pastures to lowlands, the average distance to the stream was reduced compared to large pastures, thereby, confounding pasture size with distance from the stream.

For U treatments, cows were not allowed access to off-stream water in pastures 1 and 4 in 2010 and 2 and 5 in 2011 by installing plywood covers over water tanks. In UW pastures, an off-stream water source was provided via tanks with floats at average distances (mean ± SD) of 129 ± 43.7 and 270 ± 64.3 m from the stream in small and large pastures, respectively. In R pastures, stream access to the cows was restricted to an access site consisting of a 4.9-m wide ramp that was stabilized in and to 11.3-m on either side of the stream by a geofabric base covered with 15.2-cm deep polyethylene webbing (Presto Geosystems, Appleton, WI) that was filled with crushed rock (Haan et al., 2010). Cows were not allowed access to the streamside buffer (approximately 0.91-ha) with a width of approximately 33-m on either side of the stream.

In 2010, the effects of the greater proportion of total pasture shade in the riparian zone of UW (pastures 2 and 5) than U (pastures 1 and 4) treatments seemed to supersede the effects of off-stream water (Table 1). Therefore, to evaluate the effects of off-stream water on cow distribution without the effect of shade, U and UW treatments were switched between pastures with unrestricted stream access from 2010 to 2011. However, because the stabilized stream crossings were permanent, it was not possible to rerandomize R pastures in 2011.
Sixty August-calving Angus cows (Bos Taurus L.; initial BW [mean ± SD] 592 ± 45 and 585 ± 73 kg in 2010 and 2011) were blocked by age and weight and randomly assigned to large or small R, U, or UW treatments and placed in the corresponding pastures on May 18 in both years. Small and large pastures were stocked with 5 and 15 cows, respectively. As precipitation during the study was 767 and 278-mm in 2010 and 2011, respectively, stream flow in interval 4 of 2011 was too low to support the water needs of the cows. As a result, data from interval 4 in 2011 were not included in the statistical analysis. To monitor the impact of management strategies on forage availability, sward height was measured with a falling plate meter. Cows were provided a P-free mineral (Ca maximum 30%, minimum 25%; NaCl maximum 19.4%, minimum 16.2%; Mg 1.0%; K 0.5%; Cu 1,000 mg/kg; Mn 3,750 mg/kg; Se 24 mg/kg; Zn 3,750 mg/kg; vitamin A 550,000 IU/kg; vitamin D₃ 220,000 IU/kg; and vitamin E 880 IU/kg; Kent Feeds Inc., Muscatine, IA) ad libitum in feeders located at approximately the same distances as the off-stream water sources in all pastures within a size treatment.

**Pasture characteristics**

To monitor the relationship between pasture forage characteristics on cow distribution, forage sward heights were measured with a falling plate meter (4.8 kg·m⁻²; Haan et al., 2007) at 16 sites within the lowlands and uplands in each pasture at the beginning and end of each 2-wk period in both years. In 2010, forage was hand-clipped at a height of 2.5-cm from a 0.25-m² square at 16 sites within the lowlands and uplands of each pasture at the beginning of each 4-wk interval to determine forage nutritional value.

**Laboratory analysis**

Forage samples were dried at 65°C for 48-h, weighed, ground through a 1-mm screen using a Wiley mill (Arthur H. Thomas Co., Philadelphia, PA), and subsampled for laboratory
analysis. Forage CP was calculated as 6.25 times the total Kjeldahl N (AOAC, 1990). To determine IVDMD, forage subsamples were incubated for 48-h with ruminal fluid from a fistulated steer fed a grass hay diet, and the NC-64 buffer followed by a 24-h incubation with a HCl-pepsin solution (Tilley and Terry, 1963 as modified by Barnes and Marten, 1979).

**Cow Distribution**

Collars with GPS receivers, designed and built by the Engineering Services Group of Ames Laboratory (U.S. Department of Energy, Ames, IA), were fitted on 2 or 3 cows per pasture to record cow position at 10-min intervals over 24 h·d⁻¹ for 14-d during each 2-wk period. One cow was removed from the study because of aggressive behavior in 2011, however all other cows fitted with collars were not changed within each year of the study. At the end of each 2-wk period, collars were removed and data were downloaded and evaluated for the integrity of GPS receivers. Batteries in each collar were replaced, collars were placed back onto each cow, and cows were returned to the assigned pastures.

Differential correction of the GPS data was not possible as collars only recorded date, time, and position. However, in 2010, collar accuracy was tested by placing collars on wooden stands at locations marked by a RTK-GPS unit (Agouridis et al., 2004) for 139 consecutive hours in an open field. After testing, 91, 75, and 54 percent of positions recorded by the GPS collars were within 10, 5, and 3-m, respectively, of the RTK-GPS marked position (Schwarte et al., 2011).

Data points from GPS receivers were processed using ArcGIS 10.1 (ESRI, Redlands, CA) and aerial maps (Iowa State University Geographic Information Systems Support and Research, Ames, IA). Erroneous positions (< 4% of total) including positions recorded greater than 15-m outside pasture boundaries and while cows were traveling between pastures and
working facilities, were not used in the distribution analysis. To determine cow location in relation to the pasture stream and shade, stream (0-10 m from the center of the stream), riparian (0-33 m from the center of the stream), and shade (within or 10-m from tree driplines) zones were created in each pasture as Geographic Information System buffers using ArcGIS 10.1.

**Microclimate measurements**

Weather data were recorded at 10-min intervals during the grazing season with two HOBO weather stations (Onset Comp. Co., Bourne, MA) located near the center and west of the study pastures. Weather data included ambient (Temp) and black globe (BGTemp) temperatures, relative humidity (RH), dew point, wind speed (WS), and precipitation. A temperature humidity index (THI; Mader et al. 2006), black globe temperature humidity index (BGTHI; Mader et al. 2006), and heat load index (HLI; Gaughan et al. 2008) were calculated for each 10-min interval as:

\[
\text{THI} = [0.8x\text{Temp}] + [(\text{RH}/100)x(\text{Temp}-4.4)] + 46.4 \\
\text{BGTHI} = [0.8x\text{BGTemp}] + [(\text{RH}/100)x(\text{BGTemp}-14.4)] + 46.4 \\
\text{HLIBG}_{\text{Temp}>25} = 8.62 + [0.38x(\text{RH}/100)] + (1.55x\text{BGTemp}) - (0.5x\text{WS}) + [e^{2.4-\text{WS}}] \\
\text{HLIBG}_{\text{Temp}<25} = 10.66 + [0.28x(\text{RH}/100)] + (1.3x\text{BGTemp}) - \text{WS}
\]

where RH = relative humidity, %; and WS = wind speed, m·s\(^{-1}\) (Temp and BGTemp, °C).

**Statistical analysis**

Sward height, change in sward height within period, and forage quality data were analyzed with the PROC MIXED procedure of SAS (SAS Inst. Inc., Cary, NC). Pasture was considered the experimental unit in all analyses. As previously discussed, data analysis did not include interval 4 from 2011. Forage sward height and change in sward height within each 2-wk
period was analyzed with a model that included a repeated subject effect of pasture, random effects of year, interval by year, and period nested within interval by year. Fixed effects included interval, pasture size, management treatment (U, UW, R), location (lowland and highland), interactions of interval by pasture size, interval by management treatment, pasture size by management treatment, interval by management treatment by pasture size, pasture size by location, management treatment by location, management treatment by pasture size by location, and interval by pasture size by grazing treatment by location. As forages were sampled only once per 4 wk interval in 2010, concentrations of CP and IVDMD in forage were pooled by pasture size and analyzed with a model that included a repeated subject effect of pasture, and main effects of interval, management treatment, location, and interactions of interval by management treatment, interval by location, management treatment by location, and interval by management treatment by location. Because of small differences in forage composition between management treatments and locations, forage samples were not collected in 2011. Forage sward height and composition data are reported as least square means.

Cow distribution, calculated as the proportion of total observations that cows were in the stream or riparian zones, was analyzed with the PROC GLIMMIX procedure of SAS and a model that included random effects of pasture, year, interval by year, and period nested within interval by year. Fixed effects included management treatment (U, UW, R), pasture size, interval, and the interactions of pasture size by management treatment, interval by pasture size, interval by management treatment, and interval by management treatment by pasture size. Furthermore, covariates included shade within 10 and 33 m of the pasture stream as a percent of total pasture shade for analysis of observations within the stream and riparian zones, respectively. Differences between least square means with significant treatment effects were
determined by the PDIF procedure of SAS. Distribution data are reported as least square means.

Ambient temperature, BG temperature, THI, BGTHI, and HLIBG data were paired with cow positions to determine the impact of microclimate on cow distribution in 2010. For each one unit increment of each microclimatic variable (ambient temperature, BG temperature, THI, BGTHI, and HLI), the numbers of observations recorded in the stream, riparian, total pasture shade, or shade within the riparian zone were divided by the total number of observations at that temperature (1°C) or index unit (1 index unit) increment to determine the probability of a cow being in that zone at that microclimatic variable increment. Based on the Quasilikelihood under the Independence Model Criterion (QIC) of the PROC GENMOD procedure of SAS from 2010 data, ambient temperature provided the model of best fit for the increase in the probability of cow presence in the stream, riparian, and shade zones. As a result, ambient temperatures in 2010 and 2011 were used to determine the probability of cow presence in stream, riparian, and shade zones. The probability of cows being in a zone at each microclimate increment was used to calculate an odds ratio with the binomial function of the PROC GENMOD procedure of SAS. A linear predictor was determined from the odds ratio curve based on a logit scale. Differences between linear predictors were determined with the PDIF function of LSMEANS with a repeated subject effect of pasture nested within year and a model which included fixed effects of year, pasture size, and management treatment and interactions of pasture size by management treatment, pasture size by year, management treatment by year, and pasture size by management treatment by year.
Results

Ambient temperature and pasture characteristics

In June 2010, the mean monthly ambient temperature, measured with on-site weather stations, was 3.5°C higher than the 30-yr average at a weather station in Des Moines, Iowa (Figure 2). However, mean monthly ambient temperatures in July, August, September, and October 2010 were 0.5 to 2.0°C less than the 30-yr average. Similarly, in 2011, the mean monthly ambient temperatures in May, June, August, and September were 1.0 to 4.1°C lower than the 30-yr average.

Across all pasture treatments and sizes, the mean forage sward heights were 26.3, 23.4, 19.2, 14.4, and 9.1 cm in intervals 1, 2, 3, 4, and 5 \((P < 0.05)\). Mean sward height was lower \((P < 0.01)\) in R than U and UW pastures (Table 2), possibly because 0.91 ha of the area of the R pastures was in the ungrazed riparian buffers. Similarly, mean sward heights were lower \((P < 0.01)\) in large than small pastures and in the lowlands than the uplands of pastures. However, there were no interactions between pasture treatments, sizes, locations, or intervals. Forage removal, estimated by sward height at the start and end of each period, did not differ between pasture treatments or sizes across intervals. However, while estimated forage removal did not differ between treatments in small pastures, estimated forage removal was greater from large U pastures than large R pastures \((treatment \times size, P < 0.05)\). In 2010, mean IVDMD and CP concentrations of the forage were 53.9, 11.0; 45.8, 8.2; 43.6, 8.3; 38.7, 9.6; and 39.8, 10.8% in intervals 1, 2, 3, 4, and 5 \((P < 0.05)\) across pasture size, treatments, and locations. However, over all intervals, there were no main effects or interactions of pasture size, treatment, or location on forage IVDMD or CP concentration. The lack of pasture treatment by size interactions on
forage sward height and composition implies that any effects that the switchback design used in this experiment had on forage mass and nutritive value had minimal effects on cow distribution.

**Pasture size and management effects on cow distribution**

Across pasture treatments, there were greater ($P < 0.01$) proportions of cow observations in the stream (Figure 3A) and riparian (Figure 3B) zones in small than large pastures. The effects of pasture size on the proportion of cow observations in the stream and riparian zones were greater in interval 3 than other intervals (interval x size, $P < 0.05$). Although there were no differences in the proportion of cow observations in the stream (Figure 3A) and riparian (Figure 3B) zones between the U and UW treatments in large pastures, cows in small UW pastures had a greater proportion of observations in the stream and riparian zones than cows in small U pastures (treatment x size, $P < 0.05$). Across pasture sizes, the proportion of cow observations in the stream (Figure 3A) and riparian (Figure 3B) zones in R pastures was less ($P < 0.01$) than U and UW pastures.

Likely because of the lower average temperature range in 2011 than 2010, the probability of cow presence in the stream and riparian zones increased ($P < 0.04$) at greater rates in 2011 than 2010 over the temperature range in pastures with unrestricted stream access at both sizes (Data not shown). As temperature increased across years, the probability of cow presence in the stream (Figure 4A) and riparian (Figure 4B) zones increased at greater ($P < 0.01$) rates in small than large pastures with unrestricted stream access. Compared to cows in small U pastures, the probability of cow presence in the stream (Figure 4A) and riparian (Figure 4B) zones increased at greater ($P < 0.01$) rates in small UW pastures as temperature increased. However, there were no differences in the probability of cow presence in the stream (Figure 4A) and riparian (Figure 4B) zones as temperature increased between large U and UW pastures. Within large and small
treatments, the probability of cow presence in the stream (Figure 5A) and riparian zones (Figure 4B) of U and UW pastures increased at greater \((P < 0.01)\) rates over the temperature range than R pastures. Furthermore, the probability of cow presence in the stream (Figure 4A; \(P < 0.07\)) and riparian (Figure 4B; \(P < 0.02\)) zones of small R pastures increased at greater rates than large R pastures as temperature increased.

Previous studies have implicated that riparian shade may affect the presence of cows in riparian zones (Franklin et al., 2009; Bear et al., 2012). However, the effects of pasture size on this relationship have not been well studied. Although there was no difference between treatments in the probability of cow presence in shade throughout the pasture as temperature increased (Figure 5A), the probability of cow presence in riparian shade in treatments with unrestricted stream access increased at a greater \((P < 0.01)\) rate in small than large treatments (Figure 5B).

**Discussion**

Grazing cows congregating in or near pasture streams to meet needs for thirst, hunger, and thermoregulation may increase the risk of water quality degradation in Midwest pastures and western rangelands by reducing vegetative cover and increasing fecal deposition in the streams and surrounding riparian areas (Bailey, 2005; Vidon et al., 2008; Schwarte et al., 2011). Previous studies have shown that cows were observed more often in and near the stream as ambient temperatures increased (Franklin et al., 2009; Haan et al., 2010). However, management strategies including restricted stream access or rotational stocking have reduced the concentration of cattle near pasture streams resulting in less risk to water quality (Agouridis, 2005; Schwarte et al., 2011). Although management strategies have been the subject of many
studies, relatively few considered the impact of pasture size on these strategies. Similar to the results of an on-farm, non-replicated study with pastures of varying sizes and shapes by Bear et al. (2012), as the percentage of total pasture area within 33 m of the stream in pastures with unrestricted access increased, cows were observed more often in and near the stream in the present controlled study. Combining results of these two studies, the mean proportions of observations that cattle were within the riparian zones of pastures over the grazing season were related to the proportion of pasture in the riparian zone by the regression:

\[ Y = 0.692x + 5.95; \quad (R^2 = 0.52) \]

where \( x \) = the proportion of the pasture within 30 or 33 m of the stream (Figure 6).

However, pastures on Farm C of Bear et al. (2012) were grazed by well-managed rotational stocking and appeared to contain a greater proportion of the legume, red clover (Trifolium Pratense L.) than other farms in that or the current study. These factors may have reduced the proportion of cow observations in and near the stream in comparison to the proportion of total pasture area within the riparian zone. If data from Farm C of Bear et al. (2012) were removed from the data set, the mean proportions of cow observations in the riparian zones of pastures over the grazing season were:

\[ Y = 0.827x + 4.94; \quad (R^2 = 0.74) \]

where \( x \) = the proportion of the pasture within 30 or 33 m of the stream.

This regression may be used to predict the proportion of time that cows would be in the riparian zones in pastures of different shapes and sizes under continuous grazing.

The probability of cow presence in and within 33 m of the stream increased at a slower rate as ambient temperature increased in large than small pastures in this study. The effect of larger pastures on cow distribution likely resulted from a greater proportion of grazing land being
farther from the stream. The available grazing land at a greater distance from the stream likely provided shaded areas with greater wind exposure than areas close to the stream (Bailey, 1996; Launchbaugh and Howery, 2005). Furthermore, the average distance to the stream from recorded cow observations was (mean ± SD) 189.0 ± 33.3 and 61.3 ± 8.2 m in large and small pastures respectively, potentially increasing the energy expenditure required for cows to walk to the stream in large compared to small pastures.

In addition to increased pasture size, the presence of cows near the pasture stream was further reduced throughout the grazing season by restricting stream access to stabilized crossings. Although effective at both pasture sizes, the proportion of cows spending time in the riparian zone in pastures with restricted compared to unrestricted stream access was 3 times less in small compared to large pastures. Therefore, because of greater risk for a reduction in water quality as cattle spend more time near a pasture stream (Belsky et al., 1999, Haan et al., 2010), restricting stream access to stabilized crossings is likely to be more effective at improving water quality in pastures with a larger proportion of grazing land close to a pasture stream.

In previous studies, both forage quantity and quality in western grazing lands (Wells, 2004; Ganskopp and Bohnert, 2008) and off-stream water (Godwin and Miner, 1996; Sheffield et al., 1997) influenced cow distribution and impacts on water quality in pasture streams. In the current study, although differences in forage sward height within pasture size, management treatment, and location may have impacted cow distribution, differences in forage quality were inadequate to influence cow distribution. Furthermore, compared to the present study, differences in sward heights and forage quality between the riparian and upland zones in western grazing lands are likely much greater due to differences in precipitation and soil type (Sala et al., 1988; Rinehart et al., 2006).
Similar to the effect of forage quality, there was no effect of off-stream water on cow distribution in the present study. However, the effects of off-stream water on cow distribution in previous studies are inconsistent (Franklin et al., 2009; Schwarte, 2011; Kaucner et al., 2013) and may be dependent on shade distribution and availability (Agouridis et al., 2005; Byers et al., 2005; Franklin et al., 2009).

Shade is utilized by cows to reduce exposure to solar radiation and reduce body temperature during high ambient temperatures (Blackshaw and Blackshaw, 1994; Tucker et al., 2008). Similar to previous studies (Franklin et al., 2009; Haan et al., 2010), as temperature increased, cows had more observations under or near pasture shade in the present study. However, riparian shade is in close proximity to running water which can also assist with thermoregulation at high ambient temperatures (Belsky, 1999). In the present study, the probability of cows spending time in the riparian shade increased at a greater rate in small compared to large pastures (Figure 5). Similar to the reduced probability of cow presence near pasture streams, the reduced probability of cows utilizing riparian shade in large than small pastures may be associated with lower energy expenditure for walking and greater exposure to air movement for cows present under upland shade (Bailey, 1996; Launchbaugh and Howery, 2005).

Along with management practices and pasture characteristics, the annual variability in environmental conditions may also have a significant impact on cattle distribution. Compared to 2010, in 2011 temperatures in interval 3 were an average of 2.6°C warmer and the daily temperature range was 10.5°C lower which corresponded with 17.4 and 10.0% increases in cow observations in the riparian zone of small and large pastures, respectively. The increased temperature and reduced temperature range likely reduced the ability of the cows to cool at night.
and increased the observations of cows in the riparian zone seeking relief from the heat (Mader et al., 2006).

In conclusion, fencing required to physically limit cow access to pasture streams and reduce the risks of cows impacting stream water quality would be most effective in pastures with a greater proportion of grazing land near the pasture stream.

**Literature Cited**


### Tables and Figures

Table 1. Tree shade\(^1\) within the total pasture and riparian zones\(^2\) in small and large pastures\(^3\)

<table>
<thead>
<tr>
<th>Pasture</th>
<th>Grazing treatment(^4)</th>
<th>Total pasture shade, ha</th>
<th>% of total pasture shade</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2010</td>
<td>2011</td>
<td>Small</td>
</tr>
<tr>
<td>1</td>
<td>UW</td>
<td>U</td>
<td>0.14</td>
</tr>
<tr>
<td>2</td>
<td>U</td>
<td>UW</td>
<td>0.04</td>
</tr>
<tr>
<td>3</td>
<td>R</td>
<td>R</td>
<td>0.06</td>
</tr>
<tr>
<td>4</td>
<td>UW</td>
<td>U</td>
<td>0.12</td>
</tr>
<tr>
<td>5</td>
<td>U</td>
<td>UW</td>
<td>0.45</td>
</tr>
<tr>
<td>6</td>
<td>R</td>
<td>R</td>
<td>0.39</td>
</tr>
</tbody>
</table>

\(^1\)Tree shade includes within and 0 to 10 m from tree driplines.

\(^2\)Riparian zone includes 0 to 33m from stream center.

\(^3\)Total pasture area: large (12.1 ha) and small (4.0 ha).

\(^4\)Grazing treatments within pastures were continuous stocking with unrestricted stream access (U), continuous stocking with unrestricted stream and off-stream water access (UW), and continuous stocking with stream access restricted to 4.9-m wide stabilized crossings ©.
Table 2. Effects of pasture treatment on mean compressed sward heights measured by falling plate meter at the initiation of each 2-wk period within the lowlands and uplands of small (4.0-ha) and large (12.1-ha) pastures in 2010 and 2011 (SEM = 1.02)

<table>
<thead>
<tr>
<th>Locations²</th>
<th>Pasture treatment¹ and size</th>
<th>U</th>
<th>UW</th>
<th>R</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Small</td>
<td>Large</td>
<td>Small</td>
<td>Large</td>
</tr>
<tr>
<td>Lowland</td>
<td>19.3</td>
<td>18.4</td>
<td>19.5</td>
<td>18.6</td>
</tr>
<tr>
<td>Upland</td>
<td>22.8</td>
<td>20.6</td>
<td>22.9</td>
<td>20.7</td>
</tr>
<tr>
<td>Significance</td>
<td>Treatment (t)</td>
<td>&lt;0.01</td>
<td>0.02</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td></td>
<td>Size (s)</td>
<td></td>
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<tr>
<td></td>
<td>Location (l)</td>
<td></td>
<td></td>
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</tr>
<tr>
<td></td>
<td>t x s</td>
<td>0.74</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>t x l</td>
<td>0.75</td>
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<td></td>
<td>s x l</td>
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</tr>
<tr>
<td></td>
<td>t x s x l</td>
<td>0.84</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

¹ Treatments were unrestricted stream access (U), unrestricted stream and off-stream water access (UW), or restricted stream access (©).

² Locations were the lowlands (the stream and 2.0-ha on either side of the stream) and uplands (4.0-ha on either side of the stream beyond the lowlands). The uplands were not grazed when the small size was assigned to a pasture.
Figure 1. Design of small (A) and large (B) pastures at the Rhodes Research Farm. Treatments within pasture sizes were continuous stocking with stream access restricted to 4.9-m-wide stabilized crossings (R) treatments (Pastures 3 and 6), continuous stocking with unrestricted stream access (U; Pastures 1 and 4 in 2010 and Pastures 2 and 5 in 2011), and continuous stocking with unrestricted stream and off-stream water access (UW; Pastures 2 and 5 in 2010 and Pastures 1 and 4 in 2011). Off-stream water sites were only available within UW treatments.
Figure 2. Mean monthly ambient temperature in 2010 and 2011 from 2 on-site weather stations, and 30-yr average (NOAA, Des Moines, Iowa, approximately 54 km from the study site).
Figure 3. Mean proportions of observations of cows in the stream (0 to 10 m from stream center), and riparian (0 to 33 m from stream center) zone of large (12.1 ha) or small (4.0 ha) pastures with unrestricted stream access (U), unrestricted access to stream and off-stream water (UW), or restricted stream access © during all intervals in 2010 and 2011.

A (Stream Zone)

B (Riparian Zone)

Means within the stream and riparian zones without a common superscript differ ($P < 0.05$). A 95% confidence interval of the means is shown by extending bars.
Figure 4. Odds ratio of cows presence in the stream (A), and riparian (B) zone over the temperature range in small (4.0 ha) and large (12.1 ha) treatments of continuous stocking with unrestricted stream and off-stream water access (UW), continuous stocking with unrestricted stream access (U), or continuous stocking with restricted stream access during all intervals in 2010 and intervals 1,2,3, and 5 in 2011. \(^{a-c}\) Probabilities over the temperature range without a common superscript differ \((P<0.05)\)
Figure 5. Odds ratio of cows presence in riparian zone shade (A) and total pasture shade (B) over the temperature range in small (4.0 ha) and large (12.1 ha) treatments of continuous stocking with continuous stocking with unrestricted stream access (U), unrestricted stream and off-stream water access (UW), or continuous stocking with restricted stream access © during all intervals in 2010 and intervals 1, 2, 3, and 5 in 2011. \(^{a-d}\) Probabilities over the temperature range without a common superscript differ \((P<0.05)\)
Figure 6. Mean proportion of observations of cows spent in the riparian zone (current study; 0-33 m from stream center, Bear et al., 2012; 0-30 m from stream center) compared to the proportion of total pasture area within riparian zone of pastures with unrestricted stream access with $Y = 0.692x + 5.95$; ($R^2 = 0.52$) or without $Y = 0.827x + 4.94$; ($R^2 = 0.74$) inclusion of Farm C data from Bear et al. (2012).
CHAPTER 4 EFFECTS OF STOCKING DENSITY DURING GRAZING ON
GRASSLAND ECOSYSTEMS

Beef Cattle Stocking Density and Duration Effects on Pasture Forage, Soil Quality, and Wildlife Habitat

Will be submitted to the Journal of Rangeland Ecology and Management

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Abstract
A single spring grazing event at elevated stocking densities may improve plant community diversity, soil characteristics, and wildlife habitat. Paddocks within three pastures containing cool season grass and legume species in 2 blocks without (BL1) or with (BL2) warm season grass species were assigned treatments including no grazing (NG) or a single grazing event by beef cows (*Bos Taurus* L.) in May 2011 (BL1) and 2012 (BL2) by high-density short duration-stocking (HDSD; moved 4 times daily, 498 Mg·ha\(^{-1}\) ± 84 SD) or moderate-density moderate-duration stocking (MDMD; moved once daily, 128 Mg·ha\(^{-1}\) ± 34 SD) without or with subsequent rotational stocking (HDSDR and MDMDR) until October 2013. Mean precipitation was 6.7 and 1.5 mm·d\(^{-1}\) during initial spring grazing of BL1 and BL2. Proportions of dominant plant species; soil cover, bulk density, and water infiltration rate; and forage visual obstruction were measured.
In BL1, compared to NG paddocks, grazed paddocks had lower (*P* < 0.05) proportions of cool season perennial grass species in July 2011 and May and July 2012, but had greater (*P* < 0.05) proportions of summer annual grass species in July 2011 and legume species in 2012. Soil bulk density to 7.5 cm was greater (*P* < 0.05) in MDMD, HDSDR, and MDMDR than NG paddocks and infiltration rates were less (*P* < 0.05) in HDSDR and MDMDR than NG, HDSD, and MDMD paddocks. In BL2, there were few differences in dominant plant species, soil bulk density, or water infiltration rates between treatments. In both blocks, the maximum height with 50% visual obstruction did not differ or was greater (*P* < 0.05) in HDSD and MDMD paddocks than NG paddocks in years subsequent to initial grazing. Grazing at elevated stocking densities during periods of greater rainfall temporarily reduces the proportion of cool season grass species and increases risks to soil structural characteristics.

Key Words: mob-stocking, plant diversity, stocking density, soil quality, wildlife habitat
Introduction

The dense growth and root systems of perennial grassland plants reduce soil erosion (Gyssels et al. 2005) and sequester carbon (Guo and Gifford 2002) while providing forage for grazing livestock and other herbivores (Lynch et al. 2005). However, the efficacy of grasslands to provide some ecosystem services as well as maintain productivity is dependent on plant species diversity (Tilman et al. 2006; Hodgson et al. 2005; Isbell et al. 2011). A diverse population of perennial plants in grasslands increases soil organic matter (Fornara and Tilman 2008) by sequestering carbon in the soil profile (McLauchlan et al. 2006) and, thereby, increasing soil aggregation and water infiltration and holding capacity (Franzluebbers 2002). Plant community diversity of grasslands also maintains productivity during climate stress (Sanderson et al. 2005; Deak et al. 2009) and, with establishment of legumes, increases the quantity and quality of forage for grazing livestock (Eisenhauer et al. 2011). Furthermore, diverse plant communities in grasslands provide habitat for a large variety of native wildlife including grassland birds which require a diverse range of microhabitats with significant structural complexity (Robertson et al. 2011; Ribic et al. 2009).

Conversion of grasslands from grazing to other land uses may reduce their ability to provide some ecosystem services (Fuhlendorf and Engle 2001; Brennan and Kuvlesky 2005; Fargione et al. 2009). Tillage of perennial grasslands for production of annual crops reduces aggregate stability and root structure near the soil surface, increasing oxidation of soil organic matter and risk of nutrient and soil loss (Balesdent et al. 2000; Gyssels et al. 2005; Barto et al. 2010). Chronic overgrazing increases risks of invasion by noxious weeds while reducing productivity of plant species that provide feed for livestock and habitat for native wildlife (DiTomasso 2009, Krausman et al. 2009). In contrast, without disturbance by grazing animals or fire, diversity of long-term grassland plant communities is reduced (Collins et al. 2002; Bakker et
al 2006; Burns et al. 2009) through competition for resources from cool season grasses such as tall fescue (*Lolium arundinaceum* Schreb. *Darbysh*; Rudgers and Clay 2007). Thus, properly managed grazing may enhance plant diversity, reduce plant litter, and increase bare areas to create microhabitats for brood rearing and nesting for bird species such as northern bobwhite quail (*Colinus virginianus*; White et al. 2005; Anderson and McCuiston 2008; Derner et al. 2009).

The impact of grazing livestock on plant community diversity is likely related to stocking density which influences the intensity of plant defoliation and treading on the soil surface (Hickman et al. 2004; Allen et al. 2011). Mob-stocking, defined as stocking at a high grazing pressure for a short time to rapidly remove available forage, increases stocking density and reduces grazing selectivity by livestock (Barnes et al. 2008; Allen 2011). The increased stocking density associated with mob-stocking may decrease forage regrowth and alter soil structure as a result of greater defoliation and treading damage (Brown 1968; Pande and Yamamoto 2006). Because plants are more susceptible to defoliation and treading damage during peak growth periods, early spring mob-stocking has the potential to reduce competition from cool season grasses for nutrients and light and allow less competitive plant species to establish from the soil seed bank (Hickman and Hartnett 2002, Dekeyser 2013). Although mob-stocking has the potential to improve plant community diversity, grazing at high stocking densities may increase soil compaction (Radke and Berry 1993, Drewery et al. 2008) and reduce water infiltration rate (Abdel-Magid et al. 1987). In addition to stocking density, effects of grazing on grassland plant communities and soil characteristics are likely affected by differences in the soil clay content and mineralogy (Van Haveren 1983; Harrison et al. 2003), previous land use (Pykala 2002;
The objectives of this study were to evaluate the effects of a single early spring grazing event at different stocking densities and durations on the botanical composition of the plant community, nutritional composition of the forage, physical characteristics and organic carbon content of the soil, and sward structure in south central Iowa grasslands that were subsequently not grazed to simulate grasslands maintained for wildlife habitat or grazed by rotational stocking.

**Methods**

All procedures for animal use in this experiment were reviewed and approved by the Institutional Animal Care and Use Committee at Iowa State University. (Protocol Number 3-11-7100-B)

**Study Site**

To determine the effects of a single early spring grazing event at different stocking densities and durations on subsequent plant species diversity, soil characteristics, and wildlife habitat in grasslands, two blocks were divided into three pastures with five paddocks at the Iowa State University McNay Memorial Research and Demonstration Farm (lat 40° 58’ N, long 93° 25’ W).

In block 1 (BL1), treatments were initiated in May 2011. However, because of delays in processing a request to graze government-contracted grasslands after May 15th in 2011, treatments in block 2 (BL2) were initiated in May 2012 (Fig. 1).

Block 1 consisted of three 2.02-ha pastures with slopes of 5 to 14% divided into five 0.4-ha paddocks which contained mixtures of smooth bromegrass (*Bromus inermis* L.), tall fescue, Kentucky bluegrass (*Poa pratensis* L.), reed canarygrass (*Phalaris arundinacea* L.), white clover (*Trifolium repens* L.), red clover (*Trifolium pretense* L.), and birdsfoot trefoil (*Lotus*...
corniculatus L.). Soils within BL1 were characterized as Arispe silty clay loam (Fine, smectitic, mesic Aquertic Argiudolls), Lamoni silty clay loam (Fine, smectitic, mesic Aquertic Argiudolls), and Shelby clay loam (Fine-loamy, mixed, superactive, mesic Typic Argiudolls) with a pH of 6.2 to 6.7.

Block 2 consisted of three 1.01-ha pastures with slopes of 0 to 9% divided into five 0.2-ha paddocks which contained mixtures of smooth bromegrass, tall fescue, Kentucky bluegrass, big bluestem (Andropogon gerardi Vitman), Indian grass (Sorghastrum nutans L.), switchgrass (Panicum virgatum L.), white clover, red clover, and birdsfoot trefoil. Soils within BL2 were characterized as Arispe silty clay loam, Haig silt loam (Fine, smectitic, mesic Vertic Argiaquolls), and Grundy silty clay loam (Fine, smectitic, mesic Aquertic Argiudolls) with a pH of 6.5.

**Treatments**

In each pasture, one paddock was not grazed (NG) and 4 were grazed by high-density short-duration stocking (moved 4 times per day with a back fence at 0600, 1100, 1600, and 2100 h) or moderate-density moderate-duration stocking (moved once per day with a back fence) with 10 August-calving Angus cows (Bos Taurus L.; initial BW 586 kg ± 57 SD and 648 kg ± 66 SD in 2011 and 2012 for BL1 and BL2, respectively) in the spring during the first year of the treatments for each block (Fig. 1). The duration of initial spring stocking was 49 d in BL1 and 19 d in BL2 as a result of paddocks which were twice as large in BL1 as BL2. Prior to initial stocking, live forage dry matter was estimated with a falling plate meter (4.8 kg·m⁻²; Haan et al. 2007). Cattle were allowed forage dry matter at 2.0% BW·d⁻¹, resulting in mean stocking densities of 529 Mg·ha⁻¹ ± 115 SD and 148 Mg·ha⁻¹ ± 40 SD during high-density short-duration and moderate-density moderate-duration stocking in BL1 and 470 Mg·ha⁻¹ ± 37 SD and 108
Mg·ha\(^{-1}\) ± 6 SD during high-density short-duration and moderate-density moderate-duration stocking in BL2. In each pasture following initial stocking one high-density short-duration (HDSD) and moderate-density moderate-duration (MDMD) stocked paddock was not grazed to simulate grasslands maintained for wildlife habitat. To simulate pastures for livestock production, one high-density short-duration (HDSDR) and moderate-density moderate-duration (MDMDR) stocked paddock was rotationally stocked with 35-d rest periods beginning 60 d after initial spring stocking in the first year of each block and in early May of each subsequent year (2012 and 2013 for BL1 and 2013 for BL2; Fig. 1). During rotational grazing in HDSDR and MDMDR treatments, cattle were stocked to remove 50% of the live forage dry matter as measured with a falling plate meter.

**Climatic Conditions**

Greater than average spring rainfall in 2011 resulted in an average rainfall of 6.7 mm·d\(^{-1}\) ± 15.8 SD during the 49 d of grazing by HDSD or MDMD stocking in BL1 compared to an average rainfall of 1.5 mm·d\(^{-1}\) ± 2.3 SD during the 19 d of grazing by HDSD or MDMD stocking in BL2 in the spring of 2012 (Fig. 2). However, in both 2011 and 2012, rainfall amounts from July through September were below 30-yr average. Although temperatures were close to average in most months, spring temperatures in 2012 were higher than the 30-yr average.

**Plant Community and Forage Quality Measurements**

To determine treatment effects on plant communities, the dominant plant functional group and species, or vegetative cover was identified within a 7.0 cm radius of 100 equally spaced locations on a 15.2 m string at the same 10 sites in each paddock in May, July, and October of each year. Plant functional groups include perennial cool (smooth bromeagrass, tall fescue, Kentucky bluegrass, reed canary grass) and warm (big bluestem, Indian grass, switchgrass) season grasses,
summer annual grasses (giant foxtail (*Setaria faberi*), barnyardgrass (*Echinochloa crusgalli*)), and legumes (white clover, red clover, and birdsfoot trefoil). In addition, locations dominated by forbs (not including leguminous forbs), dead forage residue, or bare soil were identified. The number of locations with different plant functional groups including perennial cool or warm season grasses, summer annual grasses, legumes, and annual forbs were divided by the number of locations with live forage to express the proportion of locations within each plant functional group as a percentage of locations with live forage. To express the proportions of species within cool and warm season grass and legume functional groups the number of locations with a majority of each plant species were divided by the number of locations with the respective functional group. The proportion of locations identified as bare or dead plant residue was expressed as a percentage of the total locations.

At six sites in each paddock in May, July, and October of each year, forage was hand-clipped from a 0.25-m² square to a height of 2.5 cm to determine treatment effects on forage nutritive value. Forage samples were dried at 65°C for 48 h, weighed, ground through a 1-mm screen using a Wiley mill (Arthur H. Thomas Co, Philadelphia, PA), and subsampled for laboratory analysis. Forage crude protein (CP) was calculated as 6.25 times the total Kjeldahl N (AOAC, 1990). To determine in vitro dry matter disappearance (IVDMD), forage subsamples were incubated for 48 h in the NC-64 buffer with ruminal fluid from a fistulated steer fed a grass hay diet, followed by a 24-h incubation period in a HCl-pepsin solution (Tilley and Terry 1963 as modified by Barnes and Marten 1979).

**Soil Physical Properties and Organic Carbon Content**

Penetration resistance was measured at 10 sites in each paddock in May and October of each year with a Field Scout SC 900 penetrometer (Spectrum Technologies, Inc, Plainfield, IL) with a
1.3-cm diameter cone tip at 2.5-cm intervals to a depth of 15-cm. Soil bulk density and organic carbon (SOC) samples were collected with a 4.8-cm diameter sampler (AMS, Inc., American Falls, ID) to a depth of 7.5-cm from 6 sites in each paddock in May and October of each year. Sample length and weights were measured and samples were divided into two subsamples. One-half of each sample was weighed, oven-dried for at least 24 h at 100°C, re-weighed, and soil gravimetric moisture calculated as the proportion of weight lost. Soil bulk density was calculated as:

\[
\text{Bulk Density, g·cm}^{-3} = \frac{(\text{soil dry matter, g})}{(L \cdot \pi \cdot 5.76)}
\]  
\[
\text{[1]}
\]

where \(L\) was the sample length, cm.

The remaining half of each soil sample was air-dried for 96 h, broken, and sieved on the 6.0, 1.2, and 0.6-mm screens of a Ro-Tap Sieve Shaker (W.S. Tyler Industrial Group, Mentor, OH) to allow manual removal of visible roots. Samples were composited by paddock and a subsample was ground with a mortar and pestle to pass through a 2-mm sieve. Soil carbon was determined by combustion with a carbon analyzer (LECO TruSpec; Leco Corporation, St. Joseph, MI) at the Iowa State University Soil and Plant Analysis Laboratory (Ames, IA). Because soils in both blocks were noncalcareous, total carbon was considered equal to SOC (Qian and Follett 2002; Schumacher et al 2002). Soil organic carbon content to a depth of 7.5-cm was calculated using the formula:

\[
\text{Soil organic carbon, Mg·ha}^{-1} = \text{SOC} \cdot BD \cdot 750
\]  
\[
\text{[2]}
\]

where SOC and BD are soil organic carbon percentage and bulk density, respectively.

Water infiltration was determined with double ring infiltrometers (Turf-Tec International, Tallahassee, FL) at the same three sites in each paddock in May and October of each year. Water was added to maintain a ponding depth between 5.1 and 2.5 cm over 90 min in the inner 15.2-cm
diameter ring. Water infiltration rate was calculated from the time and amount of water added at the last three additions of water to the infiltrometers. If water was added less than three times during the last 60 min, water infiltration rate was calculated from the total amount of water that infiltrated during the last 60 min.

Visual Obstruction

Because of the preference of bobwhite quail for herbaceous structure at greater heights during nesting and brood rearing, vertical structure was determined as visual obstruction at 6 sites in each paddock in July and October of each year (Bristow and Ockenfels 2004; Taylor et al. 1999). Digital photos of a 1 x 1 m board (Bristow and Ockenfels 2004) were taken from a distance of 4-m and height of 1-m with an Olympus Stylus 500, 5.0 megapixel camera (Olympus Corporation, Center Valley, PA) and cropped at 10-cm strata beginning at 10-cm above the soil surface. Within each 10-cm strata, pixels with vegetation were overlaid with red, and red and total pixels were summed using SigmaScan Pro 5 software (Systat Software Inc, San Jose, CA). At each 10 cm stratum, the percentage of total pixels that were red was considered the percentage of visual obstruction. Data were analyzed as the maximum height with 50% visual obstruction, as Bobwhite quail prefer sites with a minimum 50% visual obstruction at greater heights (Bristow and Ockenfels 2004; Taylor et al. 1999). Although nest incubation may occur as early as May, determination of herbaceous structure in July and October corresponded to a significant proportion of seasonal nest incubation and brood rearing periods (Burger et al. 1995). Because of forage removal from rotational stocking in HDSDR and MDMDR paddocks, structural habitat for bobwhite quail was severely limited in the treatments representing pastures and would not be satisfactory bobwhite quail habitat. Therefore, forage structure was measured in only NG,
HDSD, and MDMD paddocks that represented grasslands which would not be grazed for long periods such as those in government contracts or used for recreation.

**Statistical Analysis**

Forage plant community and nutritional composition, soil physical properties, and SOC percentage and content data from all paddocks and forage visual obstruction data from NG, HDSD, and MDMD paddocks were analyzed with the MIXED procedure of SAS within blocks (SAS Inst. Inc., Cary, NC). During many measurements the quantity of bare ground and forage residue was at or near 0 percent; as a result differences in vegetative cover were determined with the MIXED procedure of SAS within blocks following an arcsine adjustment. Differences in the proportion of species within functional group was determined Paddock was considered the experimental unit in all analyses and data were analyzed with a model that included the random effect of paddock within treatment and the fixed effects of month, year, treatment, and interactions of year by treatment, month by treatment, and year by month by treatment. Means are reported as least square means and significant treatment effects were determined by the PDIFF procedure of SAS.

**Results**

**Block 1**

**Plant Community and Forage Quality.** Across years and months, the proportions of cool season grass species were lower ($P < 0.01$) in paddocks initially grazed by HDSD or MDMD stocking with or without subsequent rotational stocking (74.7%) than NG (89.1%) paddocks (Fig. 3A). Across treatments, the proportion of cool season grass species increased from 2011 (64.7%) through 2012 (78.8%) and 2013 (89.2%; $P < 0.01$). In addition, the
proportions of cool season grass species were less \((P < 0.01)\) in July (65.2\%) than May (80.3\%) and October (87.1\%) across years. The significant main effects were largely the result of greater \((P < 0.05)\) proportions of cool season grass species in NG paddocks than paddocks initially grazed by HDSD or MDMD stocking with or without subsequent rotational stocking in July 2011 and May and July 2012 and in NG paddocks than HDSR and MDMDR paddocks in October 2013. However, there were no differences in the proportions of cool season grass species between NG, HDSD, and MDMD paddocks following July 2012. Within the cool season grass functional group the proportion of blue grass was greater \((P < 0.06)\) in HDSR (5.1\%) and MDMDR (4.9\%) paddocks than HDSD (2.4\%), MDMD (1.7\%), and NG (0.0\%) paddocks. Similarly the proportion of tall fescue was 42.9, 66.4, 73.8, 53.3, and 80.4 \% in NG, HDSD, HDSR, MDMD, and MDMDR paddocks \((P < 0.01)\).

In contrast to cool season grass species, the proportions of annual grass species in July 2011 in NG, HDSD, HDSR, MDMD, and MDMDR paddocks were 0.5, 41.4, 58.6, 47.3, and 43.7\%, respectively. However, after July 2011, there were no differences in the proportion of annual grass species between treatments, years, or months (data not shown; year x month x treatment, \(P < 0.05)\). Following the increase in annual grass species, the proportion of sites with dead forage residue in NG, HDSD, HDSR, MDMD, and MDMDR paddocks were 1.5, 23.8, 37.1, 38.0, and 23.1\% in October 2011 and 23.3, 13.2, 13.0, 17.3, and 11.5\% in October 2012, but did not differ between treatments in other months (data not shown; year x month x treatment, \(P < 0.05)\). As a result of these differences, the proportions of dead forage residue across treatments decreased \((P < 0.05)\) from 2011 (8.2\%) through 2012 (6.8\%) and 2013 (1.8\%). In addition, the proportions of dead forage residue were less \((P < 0.05)\) in May (1.3\%) and July (0.5\%) than October (14.8\%) across years.
Across years and months, there was no main effect of treatment on the proportions of legume species in the paddocks. However, while there were no differences in the proportion of legume species across treatments in 2011 and 2013, the proportion of legume species were greater in paddocks initially grazed by HDSD or MDMD stocking with or without subsequent rotational stocking than NG paddocks in 2012 (Fig. 3B; year x treatment, \( P < 0.01 \)).

Furthermore, while the proportions of legumes in HDSDR and MDMD paddocks were less than NG paddocks in July 2011, the proportions of legume species in NG paddocks were less than HDSD and MDMDR paddocks in October 2011, HDSD, HDSDR, and MDMDR paddocks in May 2012, and all initially grazed paddocks in July 2012 (year x month x treatment, \( P < 0.05 \)). However, there were no differences in the proportions of legumes species between treatments after July 2012. Within the legume functional group the proportion of birdsfoot trefoil was 51.9, 40.4, 20.4, 43.9, and 23.8 % in NG, HDSD, HDSDR, MDMD, and MDMDR paddocks (\( P < 0.05 \)). In contrast, the proportion of red clover was greater (\( P < 0.05 \)) in HDSDR (60.7 %) and MDMDR (61.6 %) paddocks than NG (12.6 %), HDSD (35.4 %), and MDMD (35.4 %) paddocks, and less (\( P < 0.05 \)) in NG paddocks than grazed paddocks. Although the proportions of forb species did not differ between treatments in 2011, the proportions of forb species in NG, HDSD, HDSDR, MDMD, and MDMDR paddocks were 0.1, 0.8, 3.2, 1.1, and 0.4% in 2012 and 0.2, 1.2, 2.6, 2.9, and 1.3% in 2013 (year x treatment, \( P < 0.05 \)).

Forage CP concentrations were greater (\( P < 0.01 \)) in HDSDR (10.2%) and MDMDR (10.4%) paddocks than NG (9.1%), HDSD (9.4%), and MDMD (9.1%) paddocks across all years and months (Fig. 4A). Across treatments, forage CP concentrations were less (\( P < 0.01 \)) in 2012 (9.2%) than 2011 (9.9%) and 2013 (9.9%) and decreased from May (11.3%) through July (9.7%) and October (7.8%; \( P < 0.01 \)). Significant main effects were largely a result of greater (\( P < 0.10 \))
forage CP concentration in HDSDR and MDMDR paddocks than NG, HDSD, and MDMD paddocks following July 2012. Similar to CP, forage IVDMD concentrations across years and months were greater \((P < 0.01)\) in HDSDR (49.2%) and MDMDR (48.6%) paddocks than HDSD (44.9%), MDMD (44.7%) and NG (42.8) paddocks (Fig. 4B). In addition, forage IVDMD concentrations were greater \((P < 0.01)\) in HDSD and MDMD paddocks than NG paddocks. Across treatments, forage IVDMD concentrations decreased from 2011 (48.4%) through 2012 (46.4%) and 2013 (43.3%; \(P < 0.01\)) and decreased from May (52.1%) through July (45.6%) and October (40.3%; \(P < 0.01\)). Significant main effects were largely the result of greater \((P < 0.10)\) forage IVDMD concentrations in paddocks initially grazed by HDSD or MDMD stocking with or without subsequent rotational stocking than NG paddocks in July 2011 and in HDSDR and MDMDR paddocks than HDSD, MDMD, and NG paddocks after July 2012.

**Soil Physical Properties and Organic Carbon Content.** Proportions of bare ground were 0, 0.5, 3.7, 0.6, and 1.7% in NG, HDSD, HDSDR, MDMD, and MDMDR paddocks across all years and months \((P < 0.01;\) Fig. 5). Across all treatments and months, bare ground was greater \((P < 0.01)\) in 2011 (1.5%) and 2012 (1.7%) than 2013 (0.7%). Furthermore, bare ground was less \((P < 0.01)\) in May (0.5%) and July (0.7%) than October (2.7%) across all years and treatments. Proportions of bare ground were greater in paddocks initially grazed by HDSD or MDMD stocking with or without subsequent rotational stocking than NG paddocks in October 2011 and in HDSDR paddocks than other paddocks thereafter \((year \times month \times treatment, (P < 0.01)\).

Over all years and months, there were no differences in penetration resistance measurements at depths of 0 to 15 cm between NG and HDSD paddocks (Table 1). However, penetration resistance measurements from 0 to 10 cm were greater \((P < 0.05)\) in HDSDR,
MDMD, and MDMDR paddocks than NG paddocks. Penetration resistance measurements were 250 and 150% greater in October than May across all depths in 2011 and 2013 and 30% greater in May than October at depths of 0 to 7.5 cm in 2012 (data not shown; year x month, $P < 0.05$).

While there were no treatment differences in soil gravimetric water content, soil gravimetric water contents were greater ($P < 0.01$) in 2013 (27.3%) than 2011 (22.4%) and 2012 (18.9%) across all treatments (data not shown). In addition, soil gravimetric water contents in May were 6.2% greater ($P < 0.01$) than October across years and treatments.

Across all years and months, soil bulk density measurements were greater ($P < 0.05$) in HDSDR, MDMD, and MDMDR paddocks than NG paddocks (Table 2), but there were no interactions of year or month with treatments. Soil bulk density measurements were greater ($P < 0.01$) in 2012 (1.0 g·cm$^{-3}$) than 2011 (0.90 g·cm$^{-3}$) and 2013 (0.85 g·cm$^{-3}$; Table 3). In addition, across years and treatments, soil bulk density measurements were greater ($P < 0.01$) in October (1.0 g·cm$^{-3}$) than May (0.84 g·cm$^{-3}$). Significant year and month effects were largely a result of soil bulk density measurements in May and October 2012 that were greater ($P < 0.01$) than May 2011 and May and October 2013 and less ($P < 0.01$) than October 2011.

There were no effects of treatment on SOC percentage or content across years and months (Table 2). However, SOC concentration and content were greater ($P < 0.05$) in 2012 (4.4%, 31.6 Mg·ha$^{-1}$) than 2011 (3.7%, 26.2 Mg·ha$^{-1}$) and 2013 (3.9%, 25.4 Mg·ha$^{-1}$; Table 3). In addition, SOC concentration and content were less ($P < 0.01$) in May (3.6%, 22.6 Mg·ha$^{-1}$) than October (4.4%, 32.8 Mg·ha$^{-1}$) across years and treatments.

Across years and months, water infiltration rates were greater ($P < 0.05$) in NG, HDSD, and MDMD paddocks than HDSDR and MDMDR paddocks (Table 2). Furthermore, water infiltration rates were greater ($P < 0.05$) in 2012 (0.47 cm·h$^{-1}$) than 2011 (0.32 cm·h$^{-1}$) and 2013
(0.20 cm·h\(^{-1}\)) across months and treatments (Table 3). Across years and treatments, water infiltration rates were greater \((P < 0.01)\) in October (0.57 cm·h\(^{-1}\)) than May (0.09 cm·h\(^{-1}\)). Significant main effects were largely a result of greater water infiltration rates in October 2011, 2012, and 2013 in NG (0.63 cm·h\(^{-1}\)) paddocks than rotationally stocked (0.23 cm·h\(^{-1}\)) paddocks \((\text{treatment x year x month, } P < 0.05)\).

**Visual Obstruction.** The maximum height with 50% visual obstruction increased from 2011 (30.9 cm) through 2012 (39.6 cm) and 2013 (53.7 cm, \(P < 0.01\)) across treatments and months and decreased from July (51.0 cm) to October (31.8 cm, \(P < 0.01\)) across treatments and years (Fig. 6). Across years and months, the maximum height with 50% visual obstruction in NG (48.9 cm) paddocks was greater \((P < 0.05)\) than HDSD (35.0 cm) and MDMD (40.3 cm) paddocks. However, while the maximum height with 50% visual obstruction in NG paddocks was greater than HDSD and MDMD paddocks in July 2011, the maximum height with 50% visual obstruction was greater in MDMD paddocks than NG paddocks in July 2013 \((\text{year x month x treatment, } P < 0.05)\).

**Block 2**

**Plant Community and Forage Quality.** Across years and months, the mean proportion of cool season grass species in BL2 paddocks was 63.1, 61.9, 67.0, 66.2, and 67.0% in NG, HDSD, HDSDR, MDMD, and MDMDR, but did not differ between treatments (Fig. 7A). However, across treatments, the proportions of cool season grass species increased from 2012 (59.4%) to 2013 (70.6%, \(P < 0.01\)) and the proportions of cool season grass species in October (82.2%) were greater \((P < 0.01)\) than May (57.4%) and July (55.5%). Across years and months, the proportions of annual grass species in NG, HDSD, HDSDR, MDMD, and MDMDR paddocks were 0, 0, 1.2, 0, and 2.4% \((P < 0.01)\), respectively, largely as a result of greater
proportions of annual grass species in HDSDR (2.5%) and MDMDR (4.9%) paddocks than NG (0%), HDSD (0%), and MDMD (0%) paddocks in 2013 (year x treatment, $P < 0.01$; data not shown). Similar to the proportion of cool season grass species, there were no treatment effects on the proportions of legume species (Fig. 7B). However, across treatments, the proportions of legume species decreased ($P < 0.01$) from 2012 (24.9%) to 2013 (6.9%), and from May (22.5%) through July (18.7%) and October (6.5%).

There were no treatment effects on the proportions of warm season grass species across both years and months and in 2012 (Fig. 7C). However, the proportions of warm season grass species were greater in NG, HDSD, and MDMD paddocks than HDSDR and MDMDR paddocks in 2013 (year x treatment, $P < 0.01$). Across treatments, the proportions of warm season grass species increased ($P < 0.01$) from 2012 (8.7%) to 2013 (20.0%) and were 11.8, 21.0, and 8.7% ($P < 0.01$) in May, July, and October, respectively.

There were no effects of treatment across years and months on forage CP concentrations (data not shown). However, while treatments did not differ in 2012, forage CP concentrations were 7.9, 8.6, 10.0, 8.2, and 10.7% in NG, HDSD, HDSDR, MDMDR, and MDMDR in 2013 across months (year x treatment, $P < 0.05$). Furthermore, forage CP concentrations across treatments and years decreased ($P < 0.05$) from May (11.2%) through July (8.7%) to October (7.8%). Forage IVDMD concentrations across years and months were 41.1, 43.0, 45.4, 43.4, and 45.2% in NG, HDSD, HDSDR, MDMDR and MDMDR paddocks, respectively ($P < 0.05$). Furthermore, although forage IVDMD concentrations did not differ in 2012, forage IVDMD concentrations were greater in HDSDR (42.0%) and MDMDR (43.6%) paddocks than NG (36.1%), HDSD (38.5%), and MDMD (37.9%) paddocks in 2013 (year x treatment, $P < 0.05$).
Across treatments, forage IVDMD decreased ($P < 0.01$) from 2012 (45.5%) to 2013 (41.7%) and from May (49.3%) through July (41.9%) to October (39.6%).

**Soil Physical Properties and Organic Carbon Content.** Across years and months, the proportions of bare ground were greater ($P < 0.05$) in HDSDR (2.6%) and MDMDR (2.8%) paddocks than NG (0.1%), HDSD (0.8%), and MDMD (1.0%) paddocks (Fig. 8). Across treatments, while the proportions of bare ground decreased ($P < 0.05$) from 2012 (2.3%) to 2013 (0.7%), the proportions of bare ground increased ($P < 0.05$) from May (0.6%) through July (1.1%) and October (2.6%). The significant main effects were largely a result of greater proportions of bare ground in HDSDR and MDMDR paddocks than NG paddocks in July 2012 and in HDSDR and MDMDR paddocks than NG, HDSD, and MDMD paddocks in October 2012. Furthermore, proportions of bare ground were greater in MDMDR paddocks than NG, HDSD, and MDMD paddocks in May 2013 and in HDSDR paddocks than NG, HDSD, and MDMD paddocks in July 2013 (year x month x treatment, $P < 0.01$).

Across years, penetration resistance measurements at 7.5 cm were greater ($P < 0.05$) in HDSDR and MDMDR paddocks than NG paddocks (Table 4). Although there were few differences in penetration resistance measurements between treatments across months in 2012, penetration resistance measurements from 2.5 to 7.5 cm were greater in HDSDR and MDMDR paddocks than NG, HDSD, and MDMD paddocks in 2013 (year x treatment, $P < 0.05$). There were no differences in penetration resistance measurements in May of both years, however, penetration resistance measurements from 2.5 through 10 cm in HDSDR and MDMDR paddocks were greater than NG, HDSD, and MDMD paddocks in October (month x treatment, $P < 0.05$). Penetration resistance measurements from 0 to 15 cm in October 2012 and 2013 were less ($P < 0.05$) than May 2013, but greater ($P < 0.05$) than May 2012 across treatments. Similarly, soil
gravimetric water contents in October of 2012 and 2013 were less than May 2013 and greater than May 2012 (year x month, \( P < 0.01 \); data not shown). There were no effects of treatment on soil gravimetric water contents.

Across years, months, and treatments, mean soil bulk density, SOC percentage, SOC content, and water infiltration rate were 1.01 gm·cm\(^{-3}\), 4.1\%, 30.8 Mg·ha\(^{-1}\), and 0.05 cm·h\(^{-1}\), but did not differ between treatments. However, across treatments, soil bulk density measurements in October 2012 were greater (\( P < 0.05 \)) than other months (Table 5). The concentration and content of SOC across treatments were less (\( P < 0.01 \)) in 2012 (3.6\%, 28.1 Mg·ha\(^{-1}\)) than 2013 (4.5\%, 33.5 Mg·ha\(^{-1}\)) and in May (3.8\%, 28.4 Mg·ha\(^{-1}\)) than October (4.3\%, 33.3 Mg·ha\(^{-1}\)). As a result, the concentration and content SOC were greater (\( P < 0.01 \)) in October 2013 (5.0\%; 37.3 Mg·ha\(^{-1}\)) than other months (3.7\%; 28.6 Mg·ha\(^{-1}\)) across treatments. Across treatments and years, water infiltration rates were greater (\( P < 0.01 \)) in October (0.07 cm·h\(^{-1}\)) than May (0.02 cm·h\(^{-1}\)).

**Visual Obstruction.** Across years and months, there was no effect of treatment on the maximum height with 50\% visual obstruction (Fig. 9). However, across treatments, the maximum height with 50\% visual obstruction increased (\( P < 0.05 \)) from 2012 (23.3 cm) to 2013 (64.3 cm) and decreased (\( P < 0.05 \)) from July (51.1 cm) to October (36.5 cm).

**Discussion**

Well managed grasslands with diverse plant communities provide consistent forage production for grazing livestock (Costanza et al. 2005) and habitat for native wildlife (Brawn et al. 2001) while reducing the loss of nutrients in soil runoff and sequestering carbon within the soil profile (Guo and Gifford 2002; Gyssels et al. 2005; Krausman et al. 2009). Grazing mammals can
increase plant community diversity in grassland ecosystems through disturbance on the soil surface (Collins et al. 1998; Hickman et al. 2004). However, the extent of soil disturbance from treading on the soil surface by grazing mammals is, in part, dependent on soil gravimetric water which increases with rainfall (Bilotta et al. 2007; Drewery et al. 2008). Mean daily precipitation during initial spring grazing in the current study was 5.2 mm lower while grazing paddocks in BL2 than BL1 which likely affected results. In this study, a single grazing event at elevated stocking densities reduced the proportion of cool season grasses in BL1, allowing annual grass species to establish, followed by greater proportions of legume species. However, cool season grass species, primarily tall fescue, returned to levels similar to those of paddocks without grazing within 2 yr after initial stocking, likely as a result of their tolerance to drought and efficient use of easily accessible soil nutrients (Milchunas and Lauenroth 1993; Olff and Ritchie 1998). Contrary to BL1, there were no effects of a single grazing event at elevated stocking densities on the proportions of cool season grass and legume species in BL2 likely because of the lesser precipitation while grazing BL2 paddocks reduced the degree of soil disturbance during grazing.

In previous studies, grazing has been shown to reduce the maturity of forage, thereby increasing forage IVDMD and CP concentrations (Balde et al. 1993; Cherney 1993). Similarly, in the current study, forage IVDMD and CP concentrations in months following initial stocking were greater in grazed than NG paddocks. However, because effects of maturity on forage quality are lessened by drought conditions (Peterson et al. 1992; Sheaffer et al. 1992), there were fewer differences in forage IVDMD or CP concentrations observed between treatments in either BL1 or BL2 in 2012 than 2011 or 2013. But, under higher precipitation in 2013, forage CP and
IVDMD concentrations were greater in paddocks grazed by rotational stocking than paddocks that were not grazed either without or with initial stocking.

Long-term grazing at higher stocking densities has been shown to increase soil penetration resistance and bulk density, and reduce water infiltration rates potentially increasing runoff on the soil surface (Dunne et al. 1991; Chanasyk and Naeth 1995; Daniel et al. 2002). Although soil penetration resistance and bulk density measurements were greater in MDMD paddocks than NG paddocks, soil penetration resistance measurements from 0 to 10 cm were on average 9.1% less in HDSD paddocks without subsequent rotational grazing than MDMD across both blocks. Lower average penetration resistance measurements may be the result of moving cows to new areas four times daily in HDSD paddocks compared to once daily in MDMD paddocks. With less frequent movements in MDMD paddocks resulting in longer stocking durations in each area, treading by the cows likely increased soil penetration resistance (Thurow 1991; Teague et al. 2011).

The impact of rotational stocking systems on soil structural characteristics in previous studies is not consistent (Gilley et al. 1996; Daniel et al. 2002; Teague et al. 2010); however, antecedent soil structural characteristics can influence the impact of grazing on soil structural characteristics (Van Haveren 1983; Murphy et al. 2004). Rotational stocking following a single grazing event at elevated stocking densities significantly impacted soil structural characteristics more in BL1 than BL2, likely, in part, a result of greater rainfall during initial stocking which increased the level of soil disturbance. Greater soil disturbance during initial stocking likely increased the risk of changes in soil structural characteristics with subsequent rotational grazing in BL1 (Greenwood and McKenzie 2001). In addition, despite similar soil classifications in BL1 and BL2, soil bulk density measurements and water infiltration rates in NG paddocks were on
average 15.1% greater and 94.6% less in BL2 than BL1 in 2012 and 2013. As a result of higher soil bulk density and lower water infiltration rates in BL2, rotational grazing was less likely to negatively impact soil structural characteristics (Murphy et al. 2004). Differences in soil structural characteristics between blocks were likely because BL2 paddocks were established in 1995 following long-term use for row crop production which was a minimum of 35 yr after the establishment of BL1 paddocks.

Schuman et al. (1999) found that, while stocking density had no impact on soil carbon storage, grazing in rangelands can increase soil carbon storage in the upper 30cm of the soil profile. In the current study, there were no differences in soil carbon as a result of grazing treatments. However, changes in soil carbon from management practices are small in comparison to changes from precipitation patterns and may take several years to be distinguished. Therefore, differences in soil carbon from grazing treatments are likely difficult to measure over a short period (Yang et al. 2008, Parsons et al. 2011).

Contrary to soil carbon, Augustine et al. (2012) found greater stocking densities of grazing livestock increased bare ground. In the current study, the proportion of bare ground was greater in grazed than NG paddocks in the fall following initial stocking at elevated stocking densities; but stocking density had no effect on the proportion of bare ground in paddocks without subsequent rotational grazing. In paddocks with subsequent rotational grazing following initial stocking, the proportion of bare ground was greater than NG paddocks likely as a result of both defoliation and treading at the soil surface (Russell et al 2001; Bilotta et al. 2007). Furthermore, bare ground was greater and lasted longer following initial stocking in BL1 than BL2 likely because of greater rainfall during initial grazing allowed more disturbance of the soil surface at higher stocking densities (Menneer et al. 2005; Bilotta et al. 2007). With more
poaching of the soil surface from initial stocking, plants in HDSD likely took longer to reestablish in bare areas (Drewery et al. 2008).

Although bare ground in pastures for grazing livestock reduces forage production and increases the risk for erosion (Russell et al. 2001; Teague et al. 2010), areas with bare ground are preferred by bobwhite quail and other grassland birds in grasslands managed for wildlife (Bristow et al. 2004; Collins et al. 2009; Derner et al. 2009). While the levels of bare ground in paddocks initially stocked at elevated stocking densities without subsequent rotational grazing were greater in the fall, the levels were not between 25 to 50%, preferred by bobwhite quail (White et al. 2005). In addition, early successional species such as annual grass species, which established following the initial stocking in BL1, provide feed and residue for habitat for bobwhite quail (Taylor et al. 1999; Collins et al. 2009). However, cool season grass species returned to levels prior to grazing within 2 yr of initial grazing providing few advantages to wildlife thereafter. These results support the observation that disturbance to maintain early successional species is recommended every 2 to 4 yr (Harper 2007). One year following initial stocking, plant structure available for protection of Bobwhite Quail from predators was similar or greater than paddocks not initially grazed. This result is likely a result of a lack of impact of a single grazing event at elevated stocking densities on warm season grass species in BL2 and greater proportions of forbs species in MDMD paddocks in BL1 in 2013. Bobwhite Quail select for areas with greater proportions of warm season grass and forbs species likely because of the cover they provide from predators (Collins et al. 2009, Osborne et al. 2011).

As a result of the change in botanical composition following grazing at elevated stocking densities during periods of heavy grazing, strategic spring grazing on government contracted land may improve botanical composition without reducing water runoff. Grasslands in programs
such as the conservation reserve program (CRP) provide habitat for many grassland birds, however, as grasslands age habitat quality may decline (Millenbah et al. 1996). Although, disturbance management regimes such as disking and burning have improved habitat quality for grassland birds (Greenfield et al. 2002, Copperedge et al. 2008), Collins et al. (1998) found burning may reduce plant species diversity. Results of the current study suggest strategic spring grazing at elevated stocking densities is another method to promote plant community diversity and habitat available for birds in government contracted grasslands without negatively impacting soil structural characteristics. In addition to improving wildlife habitat, grazing in government contracted grasslands increases the available forage for cattle producers to reduce overgrazing of their current pastures.

In conclusion, a single spring grazing event at moderate or high stocking densities has the potential to temporarily reduce the proportion of cool season grass species and increase the proportion of annual grass and legume species, if stocking occurs during periods of heavy rainfall. Larger proportions of annual grass and legume species enhance habitat for native wildlife while legume species may improve forage quality, however the longevity of legume species is dependent on weather conditions following grazing. Furthermore, the impact of grazing at moderate or high stocking densities on soil quality characteristics is likely related to the stocking duration and weather conditions during grazing.

**Implications**

While there has been interest in utilizing short duration grazing at elevated stocking densities as an alternative to use of prescribed burning to enhance the ecological services provided by grasslands, the advantages provided by a single grazing event at elevated stocking densities are
temporary and dependent on rainfall during stocking. Although a single grazing event at high
density-short duration and moderate density-moderate duration stocking were not successful in
establishing levels of bare ground preferred by bobwhite quail, stocking during periods of heavy
rainfall was successful in initiating plant community succession to improve feed resources for
birds and herbivores. The risk of changes in soil structural characteristics from short periods of
elevated stocking density is related more to stocking duration than density, in addition, grazing
by high density-short duration stocking seems less detrimental to soils than moderate density-
moderate duration stocking, particularly during periods of heavy rainfall.

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Literature Cited


Tables and Figures

Figure 1. Timeline of initial mob stocking and subsequent rotational grazing in treatments including high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing in block 1 (BL1) and block 2 (BL2).
Figure 2. Mean monthly ambient temperature and precipitation in 2011, 2012, and 2013, and 30-yr average (NOAA, Centerville, Iowa, approximately 54 km from the study site)
Figure 3. The effects of no grazing (NG), high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing, and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing on the proportion of cool season grass species (A) and legume (B) species as a proportion of total live forage in Block 1 pastures.

<table>
<thead>
<tr>
<th>A (Cool Season Grass Species)</th>
<th>NG</th>
<th>HDSD</th>
<th>HDSDR</th>
<th>MDMD</th>
<th>MDMDR</th>
</tr>
</thead>
<tbody>
<tr>
<td>% live forage</td>
<td>125</td>
<td>100</td>
<td>75</td>
<td>50</td>
<td>25</td>
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</table>

Differences between treatment (NG, HDSD, HDSDR, MDMD, MDMDR) means within month without common superscripts are significant (P<0.05)

95% confidence interval of the means is shown by extending bars.
Figure 4. The effects of no grazing (NG), high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing, and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing on the concentrations of crude protein (CP; A) and in vitro dry matter disappearance (IVDMD; B) in the forage in Block 1 pastures.

Differences between treatment (NG, HDSD, HDSDR, MDMD, MDMDR) means within month without common superscripts are significant (P<0.10)

95% confidence interval of the means is shown by extending bars
Figure 5. The effects of no grazing (NG), high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing, and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing on the proportion of bare ground in Block 1 pastures.

\[ \text{Bare ground, %} \]

\[ \text{NG} \quad \text{HDSD} \quad \text{HDSDR} \quad \text{MDMD} \quad \text{MDMDR} \]

\[ \text{May} \quad \text{July} \quad \text{October} \]

\[ \text{May} \quad \text{July} \quad \text{October} \]

\[ \text{May} \quad \text{July} \quad \text{October} \]

\[ \text{a-d Differences between treatment (NG, HDSD, HDSDR, MDMD, MDMDR) means within month without common superscripts are significant (P<0.05)} \]

\[ 95\% \text{ confidence interval of the means is shown by extending bars} \]
Figure 6. The effects of no grazing (NG), high-density short-duration stocking (HDSD), and moderate-density moderate-duration stocking (MDMD) without subsequent rotational grazing on the maximum height with 50% visual obstruction in Block 1 pastures.

\[ \text{Maximum height at 50\% obstruction, cm} \]

a, b Differences between treatment (NG, HDSD, MDMD) means within month without common superscripts are significant (P<0.05)

95% confidence interval of the means is shown by extending bars
Figure 7. The effects of no grazing (NG), high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing, and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing on the proportion of cool season grass species (A), legume species (B), and warm season grass species (C) as a proportion of total live forage in Block 2 pastures.
Differences between treatment (NG, HDSD, HDSDR, MDMD, MDMDR) means within month with different superscripts are significant (P<0.05)

95% confidence interval of the means is shown by extending bars
Figure 8. The effects of no grazing (NG), high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing, and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing on the proportion of bare ground in Block 2 pastures.

Differences between treatment (NG, HDSD, HDSDR, MDMD, MDMDR) means within month with different superscripts are significant (P<0.05)

95% confidence interval of the means is shown by extending bars
Figure 9. The effects of no grazing (NG), high-density short-duration stocking (HDSD), and moderate-density moderate-duration stocking (MDMD) without subsequent rotational grazing on the maximum height with 50% visual obstruction in Block 2 pastures.

Differences between treatment (NG, HDSD, MDMD) means within month with different superscripts are significant (P<0.05)

95% confidence interval of the means is shown by extending bars
Table 1. Effects of stocking management on penetration resistance measurements at depths of 0 to 15 cm in Block 1 pastures.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>NG</th>
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<th>HDSDR</th>
<th>MDMD</th>
<th>MDMDR</th>
<th>SEM²</th>
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<tr>
<td>Depth, cm³</td>
<td>Penetration resistance, kPa</td>
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<td>1246 ab</td>
<td>1527 b</td>
<td>1461 b</td>
<td>1494 b</td>
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<td>1447 a</td>
<td>1622 ab</td>
<td>1926 b</td>
<td>1812 b</td>
<td>1918 b</td>
<td>101.0</td>
</tr>
<tr>
<td>7.5</td>
<td>1531 a</td>
<td>1728 ab</td>
<td>2101 c</td>
<td>1924 bc</td>
<td>2039 c</td>
<td>91.7</td>
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<td>1751 ab</td>
<td>2093 c</td>
<td>1917 bc</td>
<td>1935 bc</td>
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<td>1960</td>
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<td>1782</td>
<td>1854</td>
<td>192</td>
<td>1912</td>
<td>1988</td>
<td>105.7</td>
</tr>
</tbody>
</table>

Gravimetric water, %

| 0-10 | 24.7 | 22.6 | 22.0 | 22.4 | 22.7 | 0.84 |

¹Treatments include high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing

²Standard error of the mean.

³Means followed by different letters within a row are different (P < 0.05).
Table 2. Effects of stocking management on soil bulk density, soil organic carbon, and water infiltration rate in Block 1 pastures.

<table>
<thead>
<tr>
<th>Item</th>
<th>NG</th>
<th>HDSD</th>
<th>HDSDR</th>
<th>MDMD</th>
<th>MDMDR</th>
<th>SEM</th>
</tr>
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<tbody>
<tr>
<td>Bulk density, g·cm$^{-3}$</td>
<td>0.85 a</td>
<td>0.91 ab</td>
<td>0.96 b</td>
<td>0.93 b</td>
<td>0.94 b</td>
<td>0.02</td>
</tr>
<tr>
<td>Soil organic carbon, %</td>
<td>3.79</td>
<td>4.05</td>
<td>4.08</td>
<td>3.99</td>
<td>4.06</td>
<td>0.12</td>
</tr>
<tr>
<td>Mg·ha$^{-1}$</td>
<td>24.6</td>
<td>28.6</td>
<td>28.0</td>
<td>28.3</td>
<td>29.3</td>
<td>1.66</td>
</tr>
<tr>
<td>Infiltration rate, cm·h$^{-1}$</td>
<td>0.49 a</td>
<td>0.36 a</td>
<td>0.19 b</td>
<td>0.43 a</td>
<td>0.21 b</td>
<td>0.05</td>
</tr>
</tbody>
</table>

$^1$Treatments include high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing.

$^2$Means followed by different letters within a row are different ($P < 0.05$).

$^3$Standard error of the mean.
Table 3. Effects of year and month on soil bulk density, soil organic carbon, and water infiltration rate in Block 1 pastures.

<table>
<thead>
<tr>
<th>Item</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
<th>SEM(^1)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>May</td>
<td>October</td>
<td>May</td>
<td>October</td>
</tr>
<tr>
<td>Bulk density, g·cm(^{-3})</td>
<td>0.69 a</td>
<td>1.11 b</td>
<td>1.01 c</td>
<td>0.99 c</td>
</tr>
<tr>
<td>Soil organic carbon, %</td>
<td>2.83 a</td>
<td>4.52 b</td>
<td>4.25 b</td>
<td>4.48 b</td>
</tr>
<tr>
<td>Mg·ha(^{-1})</td>
<td>14.6 a</td>
<td>37.7 b</td>
<td>29.9 bc</td>
<td>33.3 c</td>
</tr>
<tr>
<td>Infiltration rate, cm·h(^{-1})</td>
<td>0.14 a</td>
<td>0.50 b</td>
<td>0.06 a</td>
<td>0.89 c</td>
</tr>
</tbody>
</table>

\(^1\)Means followed by different letters within a row are different (\(P < 0.05\)).

\(^2\)Standard error of the mean.
Table 4. Effects of stocking management on penetration resistance measurements at depths of 0 to 15 cm in Block 2 pastures

<table>
<thead>
<tr>
<th>Treatment(^1)</th>
<th>SEM(^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NG</td>
<td>HDSD</td>
</tr>
<tr>
<td>Depth, cm</td>
<td>Penetration resistance, kPa</td>
</tr>
<tr>
<td>0.0</td>
<td>701</td>
</tr>
<tr>
<td>2.5</td>
<td>1206</td>
</tr>
<tr>
<td>5.0</td>
<td>1485</td>
</tr>
<tr>
<td>7.5</td>
<td>1638 b</td>
</tr>
<tr>
<td>10.0</td>
<td>1707</td>
</tr>
<tr>
<td>12.5</td>
<td>1788</td>
</tr>
<tr>
<td>15.0</td>
<td>1821</td>
</tr>
<tr>
<td>Gravimetric water, %</td>
<td></td>
</tr>
<tr>
<td>0-10</td>
<td>26.22 a</td>
</tr>
</tbody>
</table>

\(^1\)Treatments include high-density short-duration stocking without (HDSD) or with (HDSDR) subsequent rotational grazing and moderate-density moderate-duration stocking without (MDMD) or with (MDMDR) subsequent rotational grazing.

\(^2\)Standard error of the mean.

\(^3\)Means followed by different letters within a row are different ($P < 0.05$).
Table 5. Effects of year and month on soil bulk density, soil organic carbon, and water infiltration rate in Block 2 pastures.

<table>
<thead>
<tr>
<th>Item</th>
<th>Year and month</th>
<th>SEM$^1$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2012</td>
<td>2013</td>
</tr>
<tr>
<td></td>
<td>May</td>
<td>October</td>
</tr>
<tr>
<td>Bulk density, g·cm$^{-3}$</td>
<td>1.00 a</td>
<td>1.07 b</td>
</tr>
<tr>
<td>Soil organic carbon, %</td>
<td>3.61 a</td>
<td>3.64 a</td>
</tr>
<tr>
<td>Mg·ha$^{-1}$</td>
<td>27.0 a</td>
<td>29.2 a</td>
</tr>
<tr>
<td>Infiltration rate, cm·h$^{-1}$</td>
<td>0.03 a</td>
<td>0.08 b</td>
</tr>
</tbody>
</table>

$^1$Standard error of the mean.

$^2$Means followed by different letters within a row are different ($P < 0.05$).
CHAPTER 5. GENERAL CONCLUSIONS

Grassland ecosystems provide forage for grazing cattle throughout the U.S., in addition, grasslands also provide many ecosystem services including water filtration and storage in the soil profile, biodiversity, and habitat for native wildlife. However, congregation of cattle in pasture riparian areas and streams has the potential to increase the sedimentation, and nutrient and pathogen loading of water bodies. In upland grassland ecosystems grazing cattle has the potential to improve the diversity of plant communities, habitat for native wildlife. In both riparian and upland ecosystems the impact of grazing cattle on ecosystem services is dependent on the management practices utilized by cattle producers. The preceding research was initiated to determine cattle management practices which would enhance the functioning of grasslands or minimize the impact of grazing on those functions. The results suggest management practices which physically limit presence of cattle near streams are most effective in pastures with less available grazing land outside of the riparian area. While in uplands, grazing at elevated stocking densities has the potential to temporarily improve plant community diversity in grasslands maintained for wildlife habitat and increase the proportion of legumes in rotationally grazed pastures compared to grasslands without grazing. However, while grazing at elevated stocking densities shorter stocking durations reduce the risk of damage to soil structural characteristics.

In the first study cattle distribution was determined to measure the effects of an off-stream water site or limiting the stream access of cattle to stabilized sites on cattle presence near pasture streams in small (4.0 ha) and large (12.1 ha) pastures. Although the presence of cattle in or near pasture streams was not affected by the presence of an off-stream water site in large
pastures, limiting stream access of cattle to stabilized sites reduced the presence of cattle in and near the pasture stream in large and small pastures. However, in support of previous research, cattle in small pastures with unrestricted access to the stream were more likely to spend time in or near the stream than large pastures with more available grazing land outside the riparian area. By restricting the stream access of cattle to stabilized sites there was a greater reduction in the presence of cattle in or near the pasture stream in small than large pastures. These results suggest the efficacy of physically limiting the access of cattle to streams to reduce the potential negative impact of cattle on stream water quality would be greater in pastures with less available grazing land outside the riparian area. Similar to the results of previous studies, the probability of cattle presence in or near pastures streams increased as temperature increased. Furthermore, the probability of cattle presence in or near the pasture stream increased at a greater rate as temperature increased in small pastures with a larger proportion of pasture shade near the pasture stream than large pastures. In all pastures, as temperature increased, the presence of cattle under or near pasture shade increased suggesting the potential to use pasture shade as a mechanism to influence cattle distribution.

In the following study the effect of a single spring grazing event at elevated stocking densities with or without subsequent rotational grazing on plant species composition, soil structural characteristics and carbon storage, and wildlife habitat was determined in upland grasslands. Results of the study show grazing at elevated stocking densities during periods of heavy rainfall temporarily reduced proportion of cool season grass species which allowed a greater proportion of annual grass followed by legume species to establish in comparison to paddocks not grazed. Although there was no effect of stocking density during initial spring grazing on the proportion of legumes in rotationally grazed paddocks, direct comparisons
between conventional grazing and a grazing system which incorporated an episode of grazing at elevated stocking densities could not be made.

In paddocks without subsequent grazing to mimic grasslands maintained for wildlife habitat there was a slight increase in forbs species following spring grazing at elevated stocking densities during heavy rainfall, however, beyond greater proportions of annual grass and legume species there were few changes which suggested significant improvements in wildlife habitat. Nevertheless, reductions in the proportion of cool season grass species suggest that grazing at elevated stocking densities during periods of heavy rainfall in combination with inter-seeding of desired forbs may promote temporary establishment of plant communities preferred by ground nesting birds or other native grassland wildlife. In addition, based on previous research, strategic grazing at elevated stocking densities in grasslands on a larger scale will likely increase wildlife populations by improving habitat for native wildlife.

Interestingly, although greater disturbance at the soil surface was observed when grazing occurred during periods of greater rainfall and at higher stocking densities, when cattle were grazed at higher stocking densities soil penetration resistance and bulk density measurements were less, likely as a result of a shorter stocking duration. Greater soil penetration resistance and bulk density measurements in paddocks grazed at lower stocking densities likely resulted from a reduction in the size of soil aggregates which influence many properties of the soil profile. Differences in the effects of grazing at elevated stocking densities on soil structural characteristics were also observed between grasslands with different use histories. Although both research blocks had similar soils and were perennial pasture for at least 15 years prior to this research, significantly lower initial water infiltration rates in block 2 suggest more recent cropping, which reduces the size and stability of soil aggregates, may have led to a smaller soil
aggregates and a lower water infiltration rate. As a result grazing at elevated stocking densities had less potential to influence soil structural characteristics, however, differences in soil moisture during grazing also likely influenced the effects of grazing on soil structural characteristics.

Results from the research in this thesis suggest many areas for future research. The increase in the probability of cows spending time under or near shade as temperature increased demonstrate the potential for distribution of pasture shade to reduce the presence of cows in or near pasture streams. In upland grassland ecosystems, although grazing at elevated stocking densities increased the proportion of legume species in rotationally grazed paddocks in comparison to paddocks without grazing, a comparison of grazing systems without grazing at elevated stocking densities was not possible. In both production pastures and grasslands maintained for wildlife habitat interseeding during grazing at elevated stocking densities may increase the proportion of desired species in the plant community following grazing. In addition, grazing at elevated stocking densities in grasslands maintained for wildlife habitat may provide benefits in comparison to other methods of disturbance. However, to determine the effects of elevated stocking densities on wildlife populations more in depth measurements of wildlife habitat and counts of wildlife species would be necessary which require larger research plots for data collection. A final area of future research includes a long term study on the effects of grazing at elevated stocking densities on soil aggregation and carbon sequestration as these factors are likely influenced over longer time scales than the second study allowed.

Results from the research of this thesis suggest grazing cattle in pastures with less available grazing land outside of riparian areas during periods of elevated temperature likely pose a greater risk to stream water quality. Therefore, management practices to reduce the presence of cattle in or near pasture streams would be most effective in pastures which follow the
stream course. In upland grassland ecosystems spring grazing at elevated stocking densities has potential to improve wildlife habitat and plant community diversity, however, stocking duration should be minimized to reduce the risk to soil structural characteristics. Although grazing at elevated stocking densities may improve grassland ecosystem functioning future research at larger scales would provide more comprehensive results on the potential impacts on grassland wildlife and plant communities.
Figure 1. Demonstration of 1 x 1 m board used to determine forage structure at 10 cm intervals. Image was processed with SigmaScan Pro 5 software to determine vegetation between the camera and board.