

Landscape-Level Impacts of Livestock on the Diversity of a Desert Grassland: Preliminary Results From Long-Term Experimental Studies

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Abstract—This work is undertaken as a portion of long-term large-scale studies developed to determine how climate and disturbance (primarily fire and grazing) interact to structure desert grasslands. The results presented here are the initial grazing portions of the study. The analysis presented here indicates that following the reintroduction of cattle to the research area in 2000 (following a decade of rest) that the abundance and diversity of vegetation and small mammals increased significantly on the treatment plots ($P < 0.05$), while remaining unchanged on the control plots.

Introduction

Conservationists, land managers, and scientists have debated the role of livestock grazing in the degradation of rangelands for more than a century (Powell 1878; Bentley 1898; Leopold 1924; Sears 1935; USDA 1936; National Research Council 1994; Laycock 1994; Donahue 1999; Curtin et al. 2002; Knight et al. 2002). This debate has peaked in recent years as conservationists and researchers increasingly view ranching and the associated livestock grazing as either a crucial conservation strategy (Starrs 1998; Knight et al. 2002; Maestas et al. 2002) or a major threat (Fleischner 1994; Donahue 1999; Wuerthner and Matheson 2002). It is impossible to alter the fact tease-out the effects of a century of grazing or how the introduction of cattle may have altered the land at the time of European settlement. Yet, landscape level studies can provide important insights into the current effects of livestock. This analysis asks: Do cattle reduce the abundance and diversity of key taxa in a desert grassland? Acceptance of the hypothesis would be the result of demonstrably lower biomass and diversity; a negative answer would be the result of no effect or demonstrably higher biomass and diversity following reintroduction of cattle.

Studies such as those at Konza Prairie have investigated the role of grazing and fire in more temperate grasslands in the Eastern Great Plains (Knapp et al. 1998). Yet relatively little work has experimentally examined the landscape-level role of grazing in arid lands west of the 100th meridian where most public lands ranching, and the debate of the appropriateness of grazing, occurs (Fleischner 1994; Laycock 1994; Donahue 1999; Wuerthner and Matheson 2002; Curtin et al. 2002; Knight et al. 2002). Generally grazing has been documented to be more damaging in more arid ecosystems (Milchunas and Laurenroth 1993), with grazing in landscapes with rainfall below the 300–360 mm (12–15 inches) threshold often considered intrinsically damaging and unsustainable (Fleischner 1994; Donahue 1999). The conflict over the ecological effects of grazing is compounded by troubles with experimental design and statistical analysis that has plagued much of the grazing

literature (National Research Council 1994; Hurlbert 1984; Brown and McDonald 1995; Stohlgren et al. 1999; Jones 2000; Curtin 2002a). In this study we seek to mitigate many of the short-comings of previous studies by conducting replicated experimental research at a landscape level.

Methods

In 1998 we initiated a study on the McKinney Flats grassland on the Gray Ranch (Diamond A) in Hidalgo County in southwestern New Mexico, U.S.A. (E721033, N3472587). The study is designed to continue for at least 15 to 20 years and forms the anchor for cross-site studies developed in collaboration with The Nature Conservancy to examine the biotic and abiotic interactions associated with grazing and fire across the Intermountain West. Ungrazed from 1990 until cattle were reintroduced in 2000, the McKinney Flats pasture is located at an elevation of 1,767 m (5,800 ft). It contains a gradient from Plains-Great Basin grasslands (Bouteloua association), to semidesert grasslands (Bouteloua-Hilaria-Sporobolus association), to Chihuahuan Desert grassland/shrubland (Prosopis association). The average rainfall on McKinney Flats measured at four recording stations once a month (one at each study block) between 1999 and 2002 was 292 mm (11.3 in). The period from 2000–2002 is considered a drought according to the Palmer Drought Index (Center Assessment for the Southwest 2002). Soils on McKinney Flats range from gravelly loams (aridisols) in the uplands to silty clay loams (mollisols) in drainage basins.

The fundamental underpinning of our research design was the need for independent replication of study plots (Hurlbert 1984; Hairston 1989). This means that there must be a minimum of four replicates of each treatment. Each treatment must, while being comparable to others in biotic and abiotic components, be independent of the others. In this paper analysis of results was conducted using paired t-test through the statistical program Statview™. The sampling unit for both vegetation and vertebrate samples were the study plots containing the

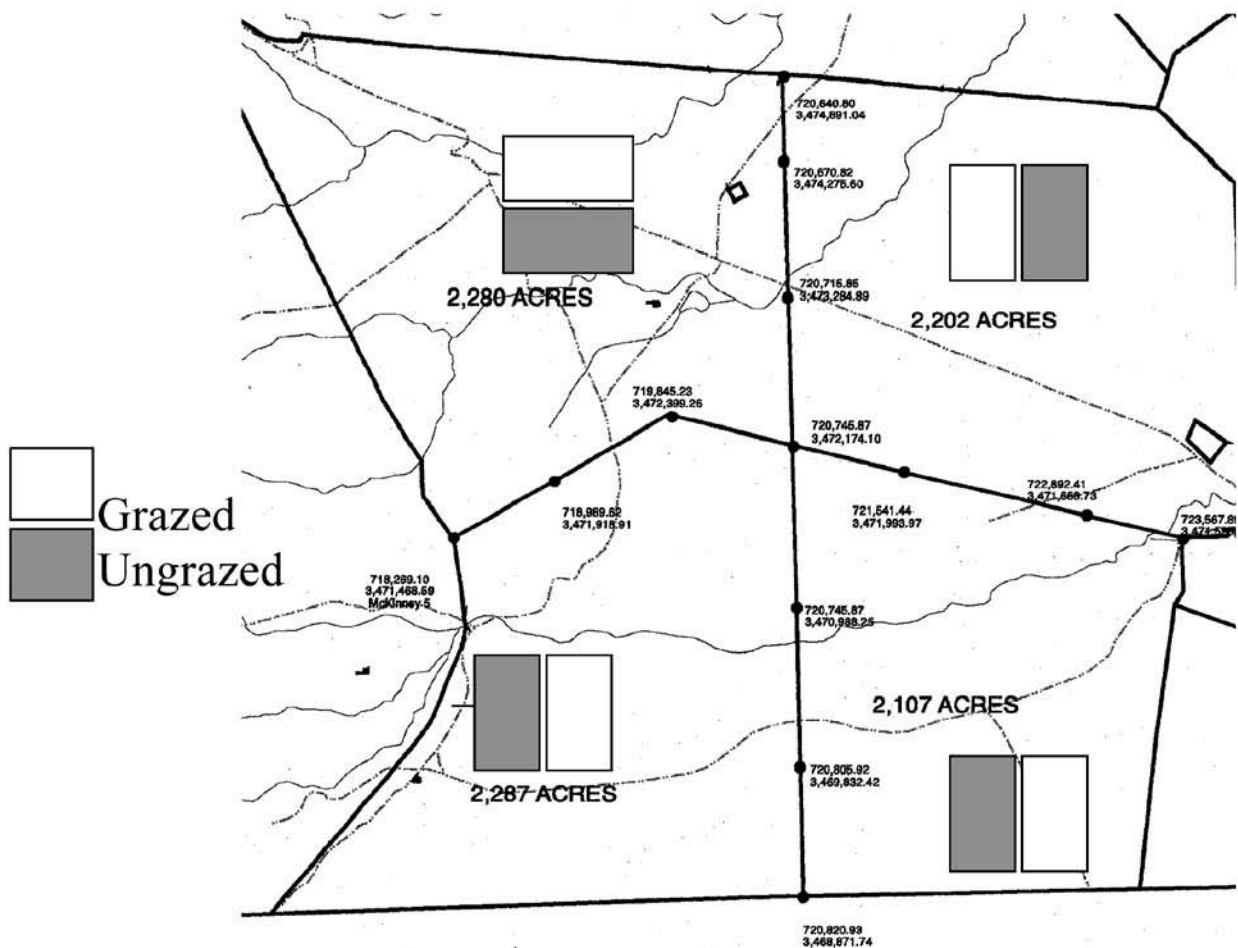


Figure 1—Map depicting the 3,668 ha (8,800 ac) McKinney Flats research site. The pasture was divided to create four, roughly 917 ha (2,200 ac), research blocks. The blocks, in addition to creating a four pasture rest-rotation grazing system typical of conventional land management, also creates four replicate landscapes for research purposes. Within each block a 1 x 1 km study area was divided into research plots containing grazed and ungrazed sections.

sampling area, with results pooled for each season resulting in 16 treatment plots each year (8 grazed and 8 ungrazed). After applying grazing to treatment blocks in 2000 and 2001, in 2002 the blocks were rested one season to measure the post-treatment response to removal of livestock.

The original 3,668 ha (8,803 ac) pasture was divided into four research blocks of about 917 ha (2,200 ac) through the use of a three strand “wildlife fence” in which the top strand is set low, and the bottom strand is smooth and set high to facilitate deer and antelope movement (figure 1). To mimic conventional livestock management of the region, and to create the four replicate treatment blocks, a four-pasture rest-rotation grazing system was used in which three pastures are grazed and one is rested each season. In this portion of the study we timed the treatment to include grazing of all four pastures within each calendar year to establish complete replication in each of the treatment years. Rather than removing cows from a grazed landscape, we instead introduced cattle into an ungrazed matrix following 10 years of rest (figure 2). Baseline sampling was conducted for two years prior to reintroduction of cattle, and the treatments were stocked at 200 to 250 head (cow/calf pairs). Cattle used in the study were primarily Hereford F1

crosses that are typical of herds in the borderlands. Targeted vegetation utilization was 50% as measured by conventional ocular estimates used by the U.S. Forest Service and local land managers. This approach to livestock management was selected because it is consistent with Federal land management guidelines and is typical of grazing practices on public and private lands in the region.

Data are collected at each of the treatment plots within 200 x 200 m focus areas creating a 100 m buffer around each sampling plot (figure 3). Driving variables measured are rainfall, livestock activity, and soils (not used in this analysis). The major response variables measured are vegetation (the primary production in the system), small mammals (primary consumers and keystone guild in many grassland and shrubland ecosystems), and lizards (secondary consumers and an assay of invertebrate abundance).

Vegetation (Primary Production)

Vegetation composition was sampled once a year (October–November) following the growing season by measuring frequency and cover within 0.40 m² quadrates set at two



Figure 2—1 km x 0.5 km grazing enclosure in the southeast study block is one of four distributed across the research area located on a northeast facing bajada adjacent the Sierra San Louis in Chihuahua, Mexico. In contrast to many grazing studies, the reintroduction of cattle into an ungrazed matrix allows a landscape level analysis of the effects of grazing, and its interaction with other processes in desert grasslands.

m intervals along 5, 150 m transect lines within each sampling block (Curtin 2002b). Vegetation biomass was measured at 30 m intervals along the 150 m transect lines within 0.40 m² square quadrates using techniques developed by the USDA ARS research center in Tucson, AZ (R.C. Marsett, AR –Tucson, personal communication 2002). Following this technique representative samples were clipped and weighted until the sampler's ability to estimate cover was consistently accurate with less than 5% variation. After the sampler's eye was trained, the technique was applied to measurements taken within the study plots. Samples are periodically retested to assure that the ability to estimate biomass is consistent throughout the sampling period. Though less accurate than weighing and clipping every sample, the variance in ocular estimates is considerably lower than the variation in biomass due to placement of sampling frames. Because this is a long-term study where repeated measures are sought through time from the same plots, nondestructive approaches to sampling were our only option to assure the continuity of the project.

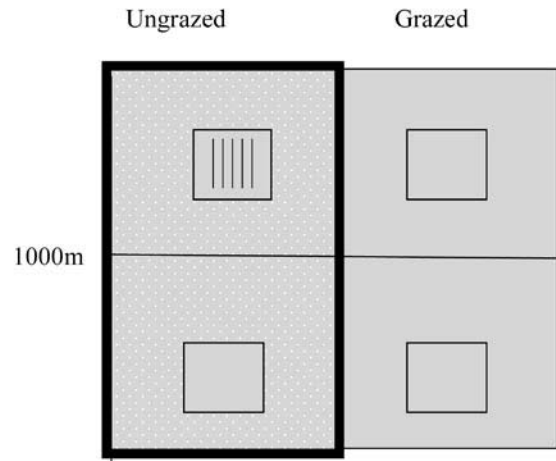


Figure 3—Experimental design of the McKinney Flats project is intended to examine at a landscape level the interaction of grazing and climate, with additional fire treatments scheduled for later in the life of the project (Curtin 2003). This core experimental protocol with five 150 m transect lines we repeated in blocks in all four sub-pastures as depicted in figure 1.

Small Mammals (Primary Consumers)

Three times a year Sherman traps were placed one meter to the east of the base of an orange 7/16 inch fiberglass stake located at 30 m intervals along the five 150 meter transects in each study plot. This sampling coincides with the lizard sampling to more efficiently use resources and to make lizards and rodents as comparable as possible. To ensure the traps are all picked up by the heat of the day only one-half of the site was trapped at a time (240 traps per night). The duration of trapping is three days in each location, three times a season for a total of 9 sampling nights per year. Due to relatively high mammal densities and diversities on the site (roughly 12 species on the site at a given time and two to ten captures per 200 x 200 m sampling area), this approach proved effective at documenting small mammal species composition. After capture species, sex, weight, body and tail length, and hind foot length are measured. All mammals are individually marked using ear tags.

Reptiles/Amphibians (Secondary Consumers)

In order to facilitate direct comparison between lizard and mammal populations, we have elected to place pit-fall traps along the same mammal trap lines one meter west of the stakes used for small mammal sampling. Pit-fall traps were censused three times yearly for three days each (total field time is 9 days to allow for lizard processing and data collection). These periods include the late spring, after adults emerge and become active (early June), in early summer before the hot dry periods prior to the monsoon (early July), and in mid August after the monsoon (when heat and drought sensitive species are likely to be active). After capture weight, sex, snout-vent length, tail length and condition, and morphometric measurements

to analyze changes in body size are recorded. All animals are individually marked through a system of toe clips.

Results

A significant difference in the vegetation biomass in grazed and ungrazed portions of the pasture ($P = 0.0001$) existed with mean biomass per 0.40 m² quadrat 41.6 (SD = 28.6) and 61.9 (SD = 37.6) gms in grazed and ungrazed plots, respectively. These recorded differences are conservative because fall rains cause some vegetative regrowth prior to sampling. Following a season of rest from livestock the mean biomass of grazed (30.5 gms, SD = 19.9) and ungrazed (29.9 gms, SD = 1.4) plots were not significantly different ($P = 0.77$). Vegetation richness was not significantly different between grazed and ungrazed plots in 1999 and 2000 prior to livestock reintroduction ($P = 0.69$ and 0.18 , respectively), was significantly higher on grazed plots in 2001 following reintroduction ($P = 0.03$), and returned to non-significant levels in 2002 after a season of rest ($P = 0.84$) (table 1). Climatic factors correlate with greater change than grazing effects with species number in 1999 prior to the drought in the low 30s, whereas by 2002 species number had dropped to the low 20s. Increases in species number on grazed plots were not the result of colonization of exotic species for

Table 1—Average vegetation species richness on treatment plots for two years prior, immediately following, and one year after removal of livestock grazing.

	Grazed	Ungrazed	P-Value
Pre-treatment (1999)	32.8 (3.8) (N = 8)	33.2 (4.1) (N = 8)	$P = 0.69$
Pre-treatment (2000)	23 (4.3) (N = 8)	20.5 (3.1) (N = 8)	$P = 0.18$
Treatment (2001)	13.7 (1.4) (N = 8)	13.1 (1.3) (N = 8)	$P = 0.03^*$
Post-treatment (2002)	20.3 (3.7) (N = 8)	20.5 (3.7) (N = 8)	$P = 0.84$

Table 2—Values of Mean and standard deviation (inside parenthesis) Biomass and Species Richness of small mammals two years before and following livestock grazing treatments. Due to the relative timing of grazing treatments and vertebrate sampling, the 2001 season did not contain sufficient replicates for all the study plots and was therefore excluded from the analysis.

Small Mammal Biomass			
	Grazed	Ungrazed	P-Value
Pre-treatment (1999-2000)	405.1 (232) (N = 8)	354.2 (213) (N = 8)	$P = 0.34$
Post-treatment (2002)	971.7 (429) (N = 8)	662.5 (214) (N = 8)	$P = 0.01^*$
Small Mammal Species Richness			
	Grazed	Ungrazed	P-Value
Pre-treatment (1999 -2000)	7.0 (1.2) (N = 8)	7.1 (1.6) (N = 8)	$P = 0.61$
Post-treatment (2002)	6.1(1.3) (N = 8)	4.8 (0.75) (N = 8)	$P = 0.02^*$

no detectable shift in species composition occurred during, or following, implementation of the grazing treatments. A detailed analysis of species composition is beyond the scope of this paper and is currently being prepared as part of a separate analysis of the interaction of climate, grazing, and fire (Curtin and Traphagen, in preparation).

Small mammal biomass was not significantly different between plots in 1999 and 2000 prior to livestock reintroduction ($P = 0.34$), yet was significantly higher on grazed plots in 2002 following reintroduction ($P = 0.01$). Small mammal richness (species number) was also not significantly different prior to livestock reintroduction ($P = 0.61$), but was significantly higher on grazed treatment plots following reintroduction ($P = 0.02$; table 2). Overall mammal biomass on both treatment and control plots increased during the sampling period (from 1998 to 2002), while diversity declined ($P < 0.05$).

Response to grazing by lizards was non-significant with biomass 341.8 gms (SD = 204) in grazed, and 408.8 (SD = 262) in ungrazed treatments ($P = 0.78$). Species richness per plot averaged 5.2 (SD = 1.3) in grazed and 4.6 (SD = 0.9) in the ungrazed treatment ($P = 0.32$).

Discussion and Conclusions

Globally, and particularly in North America, rangelands composed of grasslands and savanna have been disproportionately damaged or lost to human activities (Manning 1997; Frank et al. 1998; Dinerstein et al. 2000; Curtin et al. 2002). At the core of the debate over how best to develop long-term, large-scale conservation strategies to sustain rangelands is the role of livestock grazing. On the one hand grazing has been a leading cause of declines in biodiversity and ecosystem function (USDA 1936; Bahre and Shelton 1993; McPherson and Weltzin 2000; Curtin et al. 2002). On the other hand ecological theory states that moderate levels of disturbance maintain biodiversity (Darwin 1872; Lewontin 1969; Connell 1978; Hobbs and Huenneke 1992). Understanding the effects of livestock as a disturbance agent is crucial to understanding if grazing is an appropriate, or inappropriate, conservation strategy in the arid West.

The initial results of the McKinney Flats study reviewed here are consistent with those from other large-scale studies at Konza Prairie in Kansas (Collins 1987; Knapp et al. 1998) and in more arid landscapes Southern and Eastern Africa (McNaughton 1984; 1985; Walker 1988; Frank et al. 1998). The results of McKinney Flats and these other landscape-level studies support the “Intermediate Disturbance Hypothesis” (Darwin 1872; Lewontin 1969; Connell 1978; Hobbs and Huenneke 1992) by documenting a positive effect of grazing disturbance on biomass and diversity. While the exact mechanisms by which grazers increase biomass and diversity are not experimentally addressed here, the mechanism has long been generally understood. As stated by Charles Darwin in *The Origin of Species* (1872), “If turf which has long been mown be let grow, the more vigorous plants gradually kill the less vigorous, though fully grown plants; thus out of twenty species grown on a little plot of mown turf (three feet by four feet), nine perished, from the other species allowed to grow freely.”

This pattern is well documented in both marine and terrestrial ecosystems and illustrates the important role engineering species play in sustaining ecological systems (Jones et al. 1994). In research at McKinney Flats the results are particularly significant because lower rainfall levels have often been associated with negative response to livestock (Donahue 1999; Milchunas et al. 1993). The results presented here indicate that moderate stocking levels and rotational grazing programs even in periods of drought can maintain or contribute to system diversity in rangelands at or below the 300 mm (12 inch) threshold.

The results of this study should in no way be interpreted as blanket support for grazing in arid lands. In contrast to many desert grasslands, the region of our study has a recent history of large native herbivores with records of bison (*Bison bison*) on the Chihuahuan frontier extending back to the early 1800s (List 2002). Because an evolutionary history of interaction with large grazers is considered an important factor in determining a system's ability to sustain grazing (Archer and Smeins 1991; Milchunas and Lauenroth 1993), additional studies are needed across sites without a recent history of large native herbivores, and with different climatic patterns or levels of exotic species colonization (such as the Great Basin), to test the broader applicability of the findings presented here. Longer term and cross-site studies of not just grazing, but grazing in interaction with other disturbance factors such as climate and fire, are essential for more accurately documenting the viability of livestock grazing as a conservation strategy in the American West.

Acknowledgments

This research was supported in part by funds provided by the Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture, the Joint Fire Science Program, and the National Interagency Fire Center. Private organizations supporting this research include the Animas Foundation, Arid Lands Project, Claiborne-Ortenberg Foundation, Hewlett Foundation, Malpai Borderlands Group, Gordon and Betty Moore Foundation, Thaw Charitable Trust, The Nature Conservancy, Turner Foundation, Wallace Research Foundation, and the World Wildlife Fund.

References

Archer, S.; F. Smeins. 1991. Ecosystem-level processes. In: *Grazing management: an ecological perspective*. R. K. Heitschmidt; J. W. Smith, eds. Timber Press, Portland, OR: 109–140.

Bahre, C. J.; M. L. Shelton. 1993. Historic vegetation change, mesquite increases, and climate in southeastern Arizona. *Journal of Biogeography*. 20: 489–504.

Bentley, H. L. 1898. Cattle ranges of the Southwest: A history of exhaustion of the pasturage and suggestions for its restoration. *USDA Farmer's Bulletin* No. 72.

Brown, J. H.; W. McDonald. 1995. Livestock grazing and conservation on Southwestern rangelands. *Conservation Biology*. 9: 1644.

Center Assessment for the Southwest, Institute for the Study of Planet Earth, University of Arizona, Tucson, AZ. (www.cpc.ncep.noaa.gov/trndtext.html).

Collins, S. L. 1987. Interaction of disturbances in tall grass prairie: a field experiment. *Ecology*. 68: 124–1250.

Connell, J. H. 1978. Diversity of tropical rain forest and coral reefs. *Science*. 199: 1302–1310.

Curtin, C. G. 2002a. Livestock grazing, rest, and restoration in arid landscapes. *Conservation Biology*. 16: 880–842.

Curtin, C. G. 2002b. Integration of science and community-based conservation in the Mexico-US borderlands. *Conservation Biology*. 16: 840–886.

Curtin, C. G.; N. F. Sayre; B. D. Lane. 2002. Transformations of the Chihuahua borderlands: Grazing, fragmentation, and biodiversity conservation in desert grasslands. *Environmental Science and Policy*. 5: 55–68.

Darwin, C. 1872. *The origin of species*. 6th London ed. Thompson & Thomas, Chicago.

Dinerstein, E.; D. Olson; J. Atchley; C. Loucks; S. Contreras-Balderas; R. Abell; E. Inigo; E. Enkerlin; C. E. Williams; G. Castilleja. 2000. *Ecoregion-based conservation of the Chihuahuan Desert: a biological assessment*. The Nature Conservancy, World Wildlife Fund, Commission National para el Conocimiento y Uso de la Biodiversidad (CONBIO), PRONATURA Noreste, Instituto Tecnológico y de Estudios Superiores de Monterrey (ITESM).

Donahue, D. L. 1999. *The Western Range revisited*. University of Oklahoma Press, Norman, OK.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in Western North America. *Conservation Biology*. 8: 629–644.

Frank, D. A.; S. J. McNaughton; B. F. Tracy. 1998. The ecology of the Earth's grazing ecosystems. *BioScience*. 48: 513–521.

Hairston, H. G., Sr. 1989. *Ecological experiments: purpose, design, and execution*. Cambridge University Press, Cambridge, UK. 370 p.

Hobbs, R. J.; L. F. Huenneke. 1992. Disturbance, diversity, and invasion: Implications for conservation. *Conservation Biology*. 6: 324–337.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Monographs*. 54: 187–211.

Jones, A. 2000. Effects of cattle grazing on North American arid ecosystems: a quantitative review. *Western North American Naturalist*. 60: 155–164.

Jones, C. G.; J. H. Lawton; M. Shacker. 1994. Organisms as engineers. *Oikos*. 69: 373–286.

Knapp, A. K.; J. M. Briggs; D. C. Hartnett; S. L. Collins. 1998. *Grassland dynamics: Long-term ecological research in tallgrass prairie*. Oxford University Press, New York.

Knapp, A. K.; M. Smith. 2001. Variation among biomes in temporal dynamics of above ground productivity. *Science*. 291: 481–484.

Knight, R. L.; W. C. Gilgert; E. Marston. 2002. *Ranching west of the 100th Meridian*. Island Press, Washington, DC.

Laycock, W. A. 1994. Implications of grazing vs. no grazing in today's rangelands. In: *Ecological implications of livestock herbivory in the West*. Vavra, M.; W. A. Laycock; R. D. Pieper, eds. Society for Range Management, Denver, CO.

Leopold, A. 1924. Grass, brush, timber, and fire in southern Arizona. *Journal of Forestry*. 22: 1–11.

Lewontin, R. C. 1969. The meaning of stability. In: *Diversity and sustainability in ecological systems*. Brookhaven Symposia in Biology 22, Springfield, VA: 13–14.

List, R. 2002. *The bison from Janos, Chihuahua: The only free ranging herd in Mexico*. Unpublished Report, Instituto de Ecología, Universidad Nacional Autónoma de México.

Maestas, J. D.; R. L. Knight; W. C. Gilgert. 2002. Cows, condos, or neither: What's best for rangeland ecosystems? *Rangelands*. 24: 36–42.

Manning, R. 1997. *Grassland: the history, biology, and the promise of the American prairie*. Penguin Books, Middlesex, UK.

McNaughton, S. J. 1984. Grazing lawns: Animals in herds, plant form, and convolution. 124: 863–885.

McNaughton, S. J. 1985. Ecology of a grazing ecosystem: The Serengeti. *Ecological Monographs*. 55: 259–294.

McPherson, G. R.; J. F. Weltzin. 2000. Disturbance and climate change in the United States-Mexico borderland plant communities: A state-of-knowledge review. *USDA Forest Service RMRS-GTR-50*.

- Milchunas, D. G.; W. K. Laurenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environment. *Ecological Monographs*. 63: 327–366.
- National Research Council, 1994. Rangeland health: New Methods to classify, inventory, and monitor rangelands. National Academy Press, Washington, DC.
- Powell, J. W. 1878. Report on the lands of the arid region of the United States. House Doc. 73. 45th Congress, GPO, Washington, DC.
- Stohlgren, T. J.; L. D. Schell; B. Vanden Heuvel. 1999. How grazing and soil quality affect native and exotic plant diversity in rocky mountain grasslands. *Ecological Applications*. 9: 45–64.
- USDA, Forest Service. 1936. The Western Range: A report on the Western Range—A great but neglected resource. 74th Cong., 2d session. Senate Document No. 199.
- Vavra, M.; W. A. Laycock; R. D. Pieper. 1994. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, CO.
- Walker, B. H. 1988. Autecology, synecology, climate and livestock as agents of rangelands dynamics. *Australian Rangelands Journal*. 10: 69–75.
- Western, D. 2000. Conservation in a human-dominated world. *Issues in Science and Technology*. [On-line]. Spring.
- Wuerthner, G. 1994. Subdivision versus agriculture. *Conservation Biology*. 8: 905–908.
- Wuerthner, G.; M. Y. Matheson, 2002. Welfare ranching: The subsidized destruction of the American West. Center for Deep Ecology, Distributed by Island Press, Washington, DC.