Rangeland Ecology & Management 71 (2018) 492–505

Contents lists available at ScienceDirect

Rangeland Ecology & Management

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

Original Research

Insights from Long-Term Ungrazed and Grazed Watersheds in a Salt Desert Colorado Plateau Ecosystem☆

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ARTICLE INFO

Article history:
Received 4 December 2017
Received in revised form 14 February 2018
Accepted 16 February 2018

Key words:
erosion
lichens
livestock
Mancos Shale
salt desert
state change

ABSTRACT

Dryland ecosystems cover over 41% of the earth's land surface, and living within these important ecosystems are approximately 2 billion people, a large proportion of whom are subsistence agropastoralists. Improper grazing in drylands can negatively impact ecosystem productivity, soil conservation, hydrologic processes, downstream water quantity and quality, and ultimately human health and economic well-being. Concerns regarding the degraded state of western US rangelands in the 1950s resulted in an interagency committee to study the effects of land use on runoff and erosion processes. In 1953, a federal research group established four paired watersheds in western Colorado to study the interaction of grazing by domestic livestock, runoff, and sediment yield. Exclusion of livestock from half of the watersheds dramatically reduced runoff and sediment yield after the first 10 yr—primarily due to changes in ground cover but not vegetation. Here, we report results of repeated soils and vegetation assessments of the experimental watersheds after more than 50 yr of grazing exclusion. Results show that many of the differences in soil conditions between grazed and ungrazed watersheds observed in the 1950s and 1960s were still present in 2004, despite reduced numbers of livestock: few differences in vegetation cover but large differences in biological soil crusts, soil stability, soil compaction, and soil biogeochemistry. There were differences among soil types in response to grazing history, especially soil lichen cover and soil organic matter, nitrogen, and sodium. Comparisons of ground cover measured in 2004 with those measured in 1953, 1966, and 1972 suggest much of the differences between grazed and ungrazed watersheds likely were driven by high sheep numbers during droughts in the 1950s. Persistence of these differences, despite large reductions in stocking rates, suggest the combination of overgrazing and drought may have pushed these salt desert ecosystems into a persistent, degraded ecological state.

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Introduction

Arid and semiarid ecosystems (drylands hereafter) cover >41% of Earth's land surface, and approximately 2 billion people live within these important ecosystems (Millennium Ecosystem Assessment, 2005). Drylands are defined by water scarcity, and concerns regarding how land-use activities may modify ecosystem productivity and hydrologic processes are paramount (Noy-Meir, 1973). Grazing by domestic livestock is the most widespread land use in drylands globally, and a large proportion of dryland residents can be characterized as subsistence agropastoralists (Millennium Ecosystem Assessment, 2005; Steinfeld et al., 2006). Improper grazing in drylands can negatively impact ecosystem productivity, soil conservation, hydrologic processes, and downstream water quantity and quality. Of particular concern are potentially irreversible ecosystem state changes brought about by improper grazing on sensitive soils and plant communities (often referred to as "desertification"; Schlesinger et al., 1990; Bestelmeyer et al., 2015).

Grazing by large domestic herbivores affects dryland ecosystems directly through selective removal of plant biomass and physical disturbance (hoof impact) and indirectly via feedbacks with other ecosystem processes (e.g., plant competition, plant-soil feedbacks). Herbivory generally reduces overall vegetative cover and can alter vegetation composition through changes in plant competitive relationships (e.g., favoring unpalatable species) and introduction of invasive species (Fleischner, 1994). The physical effects of hoof action on soils can break up physical and biological soil crusts, increase soil erodibility, compact subsoils, and create preferential hill slope flow paths resulting in increased gully formation (due to animal trailing; Branson et al., 1981; Warren et al., 1986). Furthermore, the direct effects of grazing—especially overgrazing—on plants and soils can disrupt or alter dryland ecosystem processes and result in

☆ Funding for this work was provided by the US Geological Survey Climate and Land-Use Program and Ecosystem Mission Area and Bureau of Land Management.

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https://doi.org/10.1016/j.rama.2018.02.007
1550-7424/Published by Elsevier Inc. on behalf of The Society for Range Management.

profound, often irreversible, changes. For example, heavy grazing coupled with introduction of invasive shrubs resulted in long-term state change in Chihuahuan Desert grasslands even after many decades of rest from livestock (Schlesinger et al., 1990; Yao et al., 2006; Bestelmeyer et al., 2011). These concerns are especially salient given the increasing risk of multiyear droughts due to climate change (Cook et al., 2015) and the demonstrated low resilience of many sensitive dryland ecosystems to the combined impacts of grazing and drought.

Concerns regarding the persistent degraded state of many western US rangelands and the impact of those degraded systems on hydrologically connected streams and reservoirs led to the establishment of an interagency committee in the 1950s to study the effects of land use (primarily grazing by domestic livestock) on runoff and erosion processes (Lusby et al., 1964). In 1953 the Sedimentation Subcommittee of the Pacific Southwest Interagency Committee selected the Badger Wash basin in western Colorado as a focus area to study the interaction of land use, runoff, and sediment yield. Badger Wash was selected because it was broadly representative of rangeland conditions on much of the Colorado Plateau, and the topography facilitated measurement of catchment scale hydrology. Lusby et al. (1964) established a network of four paired watersheds with domestic livestock excluded from one watershed in each pair. The effects of excluding domestic livestock on runoff and erosion processes were dramatic—runoff was ~30% and sediment yield ~45% less in the ungrazed than grazed watersheds after only 3 yr (Lusby, 1970). A surprising result was that these relatively rapid changes in runoff and sediment yield were not accompanied by systematic changes in vascular vegetative composition or cover. A more detailed evaluation of vegetation and ground cover changes after 10 yr by Turner (1971) still detected only relatively small differences in vascular vegetative cover between grazed and ungrazed watersheds; these results were also observed in other salt desert ecosystems in the region (Alzerreca-Angelo et al., 1998). The primary changes attributable to grazing history were related to soil surface attributes (moss, litter, bare ground, and rock cover; Branson and Owen, 1970; Lusby, 1970; Turner, 1971).

A component of dryland ecosystems that was not well understood at the time of the original Badger Wash work is the diverse community of cyanobacteria, mosses, and lichens that often occupy exposed soil surfaces (Marble and Harper, 1989; Lange and Belnap, 2016; Weber et al., 2016). Biological soil crusts (BSCs) are found in most of the world's drylands and, when intact, play an important role in several critical dryland ecosystem processes (Belnap et al., 2016; Weber et al., 2016). They influence vulnerability to wind and water erosion (Belnap and Büdel, 2016), local and regional hydrologic cycles (Belnap, 2006; Painter et al., 2010; Chamizo et al., 2016), nutrient and carbon cycles (Barger et al., 2016; Sancho et al., 2016), and the establishment of vascular plants (e.g., Zhang and Belnap, 2015). Because the degree to which they perform these functions depends on their successional stage, disturbance can have severely disruptive effects (Belnap and Eldridge, 2003). Given the importance of BSCs in drylands globally (Weber et al., 2016) and in the Colorado Plateau in particular (Belnap and Gardner, 1993), it is likely that the differences in runoff and erosion between grazed and ungrazed pastures in the early work can be attributed to differences in BSC intactness (in addition to the mosses that were recorded).

Although no follow-up research on the Badger Wash study area has been published since the 1970s, the livestock exclosure fences have been maintained and the study area remains a valuable opportunity to evaluate ecological patterns and processes in watersheds with contrasting grazing histories. To provide further information on management of grazing in drylands, we report here results of soils and vegetation assessment of the experimental watersheds in the Badger Wash study area after more than 50 yr of grazing exclusion. We repeated vegetation and ground cover measurements and attempted to repeat the runoff and erosion measurements done by Lusby (1979). However, due to a combination of drought conditions and sedimentation of the retention ponds used in the original studies, we were unsuccessful in measuring runoff and erosion. Therefore, the specific objectives of this study were to 1) determine impact of grazing history on the vegetation community, BSC, soil physical properties, and soil biogeochemistry; 2) evaluate if grazing history effects vary with soil type; and 3) revisit results of previous vegetation studies in Badger Wash, last evaluated in 1972, to assess long-term ecosystem trajectories.

Methods

Study Area Description

The Badger Wash study area is located in western Colorado, within the Colorado Plateau physiographic province (39.3397 N latitude, 108.9339 W longitude; approximately 1 530 m elevation; Fig. 1**a**). The climate is dry and warm with a mean annual precipitation of 239 mm, mean annual maximum temperature of 19.2°C, and mean annual minimum temperature of 1.3°C (1971–2000 averages; climate data from approximately the same elevation but 27 km southeast in Fruita, Colorado; Western Regional Climate Center, http://www.wrcc.dri.edu). The four paired watersheds, established in 1953, range in size from 4.9 ha (watershed 4B) to 40.9 ha (watershed 2B; see Fig. 1; Lusby, 1979). Topography is generally rolling (slopes < 8°) with the exception of watershed 4, which consists of steeper erosional breaks (slopes > 10°).

Study area soils are derived from Mancos Shale, a late Cretaceous age marine sedimentary formation that occurs throughout the western United States (see Fig. 1a). The Mancos Shale is dominated by fine-textured material deposited in a deep sea basin and is characterized by high amounts of evaporate minerals (Whittig et al., 1982). The Mancos Shale is likely a major contributor of dissolved mineral salts to the Colorado River (Miller et al., 2017). At the western extent of these deposits, including in the Badger Wash area, the fine-textured shale intertongues with coarser-grained, nonsaline sandstones. Lusby et al. (1964) mapped three soil types that have formed in residuum from these contrasting parent materials, along with a fourth alluvial geomorphic surface that occurs between the structural hills and benches (Fig. 1b; Table 1). The sandstone parent materials are more resistant to erosion, and thus soils occur primarily as benches with relatively even and gentle slopes, are distinctly sandier in texture than the other soil types, and are the least alkaline. The shale soils are highly erodible, can be steeply sloping, and are typically the most alkaline and finest in texture. The mixed soil type is the most extensively mapped (Fig. 1b; see Table 1) and is formed from a mixture of sandstones and shales. The alluvial soils occur in the topographic lows and due to the range of alluvial source material encompass a variety of textures and salinity (Lusby et al., 1964).

Vegetation in the Badger Wash study area is typical salt-desert shrub type (Lusby et al., 1964; Alzerreca-Angelo et al., 1998; see Table 1). Stable alluvial soils tend to be dominated by larger shrubs such as *Artemisia tridentata* and *Atriplex confertifolia,* and the active washes are inhabited by *Sarcobatus vermiculatus* and *Chrysothamnus* spp. *Poa secunda* is the most common grass on alluvial soils, and moss cover is generally high. In the mixed soils, *Atriplex confertifolia, Atriplex gardneri, Gutierrezia sarothrae*, and smaller *Eriogonum* (*E. bicolor* and *E. contortum*) shrubs are common. Grasses may be the dominant cover or codominant with shrubs. Sandstone soils typically support *Atriplex confertifolia* and relatively high amounts of perennial grasses (including *Leymus salinus* and *Pleuraphis jamesii*). On the shale soils, *Atriplex gardneri* and *Atriplex corrugata* dominate with sparse cover of perennial grasses. All soils have considerable microtopography due to the presence of mosses and lichens and soil heaving resulting from freeze-thaw cycles.

The Badger Wash study area is administered by the Bureau of Land Management (BLM) and has been actively grazed by domestic livestock since the 1880s (see review by Lusby, 1979). Grazing up through the Taylor Grazing Act (1934) was widespread and heavy. It consisted primarily of migratory domestic sheep moving between winter and summer ranges. The location of a railway shipping point nearby and the

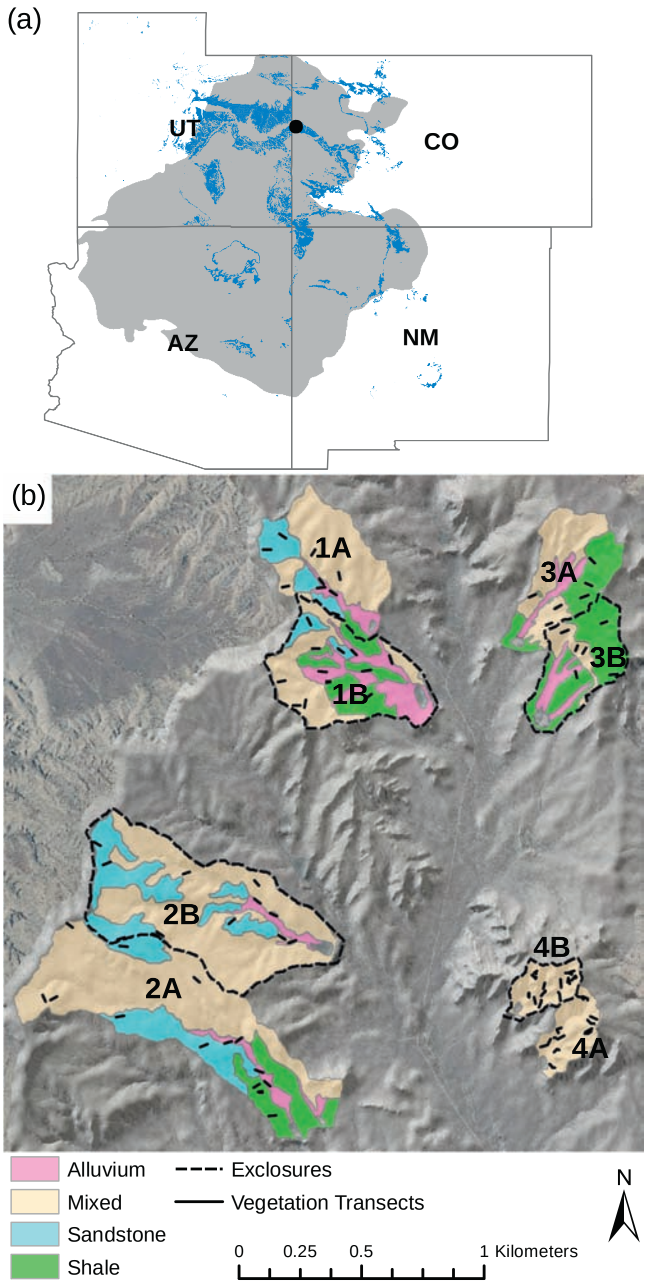

Figure 1. Map depicting location of the study area within the four-corner's region of the southwestern United States (*black dot*), Colorado Plateau (*shaded region*), and distribution of Mancos Shale deposits (*blue;***a**); experimental watershed boundaries and numbers, livestock exclosure fences (*black lines*), soil type distribution, and monitoring transects (**b**). Map of Mancos shale compiled from state geologic maps: Utah (Mancos shale and Frontier sandstones; Hintze et al., 2000), Colorado (Green, 1992), New Mexico (New Mexico Bureau of Geology and Mineral Resources, 2003), and Arizona (Mancos shale and Dakota sandstone; Richard et al., 2000).

associated Cimarron trail resulted in continued heavy use by both cattle and sheep through 1957, after which the stock driveway was closed (Lusby, 1979). The allotment in which the Badger Wash study area is located is 33 680 acres. The fences were installed to exclude livestock from the ungrazed watersheds in the fall of 1953. Exact stocking rates are unknown, but from available information when the study was implemented (1953) through the early 1970s, permitted use included approximately 3 750 sheep and 500 cattle from mid-November to the end of April (Lusby and Knipe, 1971). The watersheds open to grazing received the following use as documented by Lusby (1979): from 1954 to 1965 winter grazing by cattle and sheep (15 November to 15 May) and from 1966 to 1973 sheep only from 15 November to 15 February (1A and 3A) and no grazing in previously grazed pastures 2A and 4A. Although two of the four grazed watersheds were rested from grazing for a short period (1966 – 1973), we do not split out watersheds 1A and 3A from 2A and 4A here, given the small sample size and the fact that ≈7 yr of rest occurred 30 yr before our evaluation. From 1988 on, only cattle were permitted on the allotment. The grazing plan at the time data were collected for this study (2004 – 2005) was established circa 1999 and allows for a moderate level of use, with approximately 182 cattle from 1 January through 20 May (838 AUMs; S. Clark BLM, *personal communication*). Between 2001 and 2004 the management response to drought was to reduce the time and numbers by 32% for 2001 – 2002, no grazing for 2002 – 2003, and 50% reduction for 2003 – 2004.

Field and Laboratory Methods

Within each watershed, 6 – 13, 30-m long transects were established in 2004 for vegetation and soil sampling (see Fig. 1b). Within watersheds, transect locations were determined using a stratified random approach by soil type (as mapped by Lusby, 1979), with transect azimuths also randomly assigned. In most watershed pairs, the same soil units were available for sampling in both grazed and ungrazed areas. There were two exceptions: in watershed pair 1, a shale unit was available for sampling in the ungrazed but not the grazed, and in watershed pair 2, a shale unit was available for sampling in the grazed but not the ungrazed. A total of 85 transects were established. Our approach is similar to the earlier studies, with the primary difference being transect length (previously 15.24-m) and transect number (24 per watershed, 192 in total; Turner, 1971). In addition, earlier studies did not collect data in the Alluvial soil types.

To evaluate if grazing history has continued to affect soils and vegetation in the Badger Wash Study Area, we assessed plant cover, plant frequency, BSC cover and biomass, soil stability, soil compaction, and soil fertility along each transect in November 2004. Perennial vegetation cover, recorded by species, was measured using a continuous line intercept along each transect to the nearest 0.01 m (Elzinga et al., 1998). Rooted plant frequency was measured in 0.25-m^2 frames at 15 intervals along each transect (eight transects had more than 15 frames sampled); we used these data to calculate the frequency of occurrence of the annual invasive grass *Bromus tectorum*. Soil surface cover of BSCs was measured using a 25 × 25 cm point-frame gridded into 5-cm squares. Due to the high heterogeneity in BSCs associated with shrub canopies, soil cover sampling was stratified by under shrub canopy versus interspaces between shrubs ($n = 10$ in each strata). Every odd meter along each transect was sampled for either interspace or under canopy BSCs. If additional canopy areas were needed, shrubs within 1 m of the transect were randomly selected and sampled. In each frame, a pin flag was lowered in the corner of each square and BSC type (dark cyanobacteria, mosses, or lichens), bare ground, litter, rock (0 – 3 mm, 3 – 10 mm, 10 – 50 mm, > 50 mm), or live plant base was recorded for a total of 20 hits per frame.

Soil measurements were completed along the same transects that were used for vegetation and soil cover. The degree of surface soil crusting was determined at 20 points along each transect (every ≈1.5 m) using a fruit pressure tester (Handheld Penetrometers, QA Supplies, Norfolk, VA). This approach measures the resistance of the soil surface to penetration with an 8 mm – diameter disk and is measured in kilograms of force (using three resolutions of testers; 0 – 5 kg, 0 – 13 kg, and 0 – 20 kg). As an indicator of subsurface compaction, impact penetrometer measurements were taken in perennial plant interspaces at seven stops along each vegetation transect (every 5 m starting at 0 m; Herrick and Jones, 2002). The number of cumulative strikes required to penetrate multiple soil depths (0 – 5, 5 – 10, 10 – 15, and 15 – 20 cm) was recorded. If a designated sampling meter coincided with a perennial plant, then the nearest interspace was chosen. Surface and subsurface soil aggregate stability were evaluated using a soil stability test kit

Table 1
Badger Wash soil types and percent of grazed (G) and ungrazed (UG) watersheds, dominant plant functional groups and species, and correlated soil classifications (series, taxonomy, and ecological sites based current soil surveys; Soil Survey Staff, 2017; Spears and Kleven, 1978). Example images from soil units in grazed and ungrazed watersheds in supplementary materials (available online at https://doi.org/10.1016/j.rama.2018.02.007).

Soil unit	Dominant plant functional groups & species		Soil classifications
Sandstone G: 15% UG: 15%	Shrubs: *Atriplex confertifolia* (27%) *Artemisia tridentata* (19%)	C3 Grasses: *Bromus tectorum* (21%) *Poa secunda* (15%) C4 Grasses: *Hilaria jamesii* (16%)	Mack series, Typic Calciargids; & and Avalon series, Typic Haplocalcids; Sandy Saltdesert (R034XY402CO)
Alluvial G: 6% UG: 9%	Shrubs: *Artemisia tridentata* (49%) *Atriplex confertifolia* (12%)	C3 Grasses: *Poa secunda* (17%) *Bromus tectorum* (13%)	Fruitland series, Typic Torriorthents, Saltdesert Overflow (R034XY407CO)
Mixed G: 67% UG: 55%	Shrubs: *Atriplex gardneri* (22%) *Atriplex confertifolia* (11%) *Gutierrezia sarothrae* (6%) *Ephedra* sp. (4%)	C3 Grasses: *Leymus salina* (22%) *Elymus* sp. (13%) *Poa* sp. (9%) *Achnatherum hymenoides* (4%) C4 Grasses: *Pleuraphis jamesii* (8%)	Persayo series, Typic Torriorthents Silty Saltdesert (R034XY410CO)
Shale G: 12% UG: 21%	Shrubs: *Atriplex gardneri* (56%) *Eriogonum* sp. (4%)	C3 Grasses: *Leymus salina (12%)* *Elymus sp.* (8%) *Poa* sp. (5%)	Persayo series, Typic Torriorthents Silty Saltdesert (R034XY410CO); & Badlands

(Seybold and Herrick, 2001). Nine surface and nine subsurface samples were collected along each transect at approximately 3-m intervals. Due to time constraints, not all soil stability samples were completed during the initial sampling in 2004; 67 transects were revisited and sampled for soil stability in June — July of 2005.

Soil samples were collected along transects to both characterize static soil attributes (e.g., texture) and determine the impacts of grazing history on soil biogeochemistry. The top 0.5 cm of the soil surface in a 2.5×2.5 cm area was collected in 10 randomly selected interspaces along vegetation transects and analyzed for concentration of chlorophyll *a* as an indicator of surface soil cyanobacterial biomass (Belnap and Gardner, 1993). Any plant material, lichens, or mosses were removed, and samples were stored in dark, cool conditions before lab analysis. Chlorophyll *a* concentration was determined using a high-performance liquid chromatography (HPLC) analysis according to a slightly modified version of the method of Karsten and Garcia-Pichel (1996) and detailed in Bowker et al. (2002). Surface soil horizon (0–10 cm) samples were collected at each meter along transects using a 2.5-cm diameter soil core, composited by transect, and one subsample was analyzed for a suite of physical and biogeochemical attributes. Soil texture (percent sand, silt, and clay) was determined by the hydrometer method and the relative composition of sand size classes by dry sieving (Gee and Or, 2002). Electrical conductivity, pH, and soluble salts were measured on a saturated soil paste solution (Rhoades, 1982). Organic matter was determined using the Walkley-Black procedure of dichromate oxidation (Nelson and Sommers, 1996). Phosphorus was extracted with $NaHCO_3$ (Olsen et al., 1954; Schoenau and Karamanos, 1993). Total nitrogen concentration was measured using Kjeldahl analysis (Bremner et al., 1996).

Data Analysis

To characterize the landscape setting of each transect, basic terrain attributes were calculated using ArcGIS (ESRI, 2012) derived from a 5-m resolution digital terrain model acquired with an airborne interferometric synthetic aperture radar (IFSAR) sensor (Intermap Technologies Inc., Englewood, CO). Study area slope (in degrees) and flow accumulation (measure of upslope contributing area) were calculated for the study area using the Spatial Analyst extension. Slope and natural log of flow accumulation values were then averaged along the length of each transect (represented as a line feature). Topographic Wetness Index (TWI), a relative measure of landscape position with respect to hillslope water redistribution, was calculated for each transect using back-transformed mean flow accumulation and mean slope (in radians; Sörensen et al., 2006).

To evaluate how vegetation and soils have responded to differing grazing history, we conducted univariate analysis on select ecosystem indicators. We used an analysis of variance approach to test the null model that vegetation and soil attributes did not differ between grazed and ungrazed areas. Watershed pairs were treated as a random effect and grazing history and soil unit (plus grazing $\times$ soil interactions) as fixed effects (PROC MIXED; SAS Institute, 2012). Soil units were determined by overlaying transect locations with soil classification determined by Lusby et al. (1964). Three transects lay along the boundary of two soil types, so we compared static soil properties across soil types to determine the best soil type assignments for those transects.

Vegetation response variables tested included cover of perennial shrubs, nonwoody forbs, perennial cool and warm season grasses (C3 and C4 photosynthetic systems, respectively), total plant cover, and rooted frequency of *B. tectorum*. Perennial plant species cover was calculated as the total length of transect occupied divided by the transect length. Fractional cover of individual species were then summed by functional groups for analysis (shrubs, cool season [C3] and warm season [C4] perennial grasses, and nonwoody forbs; see Table 1). Lichen, moss, and total soil (lichen, moss, rock, litter, or plant basal) cover was calculated as the fraction of total hits within each BSC frame. For BSC, the cover of mosses and lichens and surface chlorophyll *a* concentration was used in analysis. Other dynamic soil response variables tested were total ground cover, soil aggregate stability, soil compaction (at multiple depths), soil nutrients (total nitrogen and phosphorus), soluble soil sodium, and soil organic matter. Data with multiple observations per transect (soil stability, soil compaction, *B. tectorum* frequency, and BSC frames) were averaged before statistical analysis. The BSC cover was averaged by strata (interspace and under shrub). Analysis of BSC data included microsite (canopy vs interspace) as fixed effects (including interactions). Data that were not normally distributed (assessed by looking at residuals of models run on raw data) were transformed to improve normality using either an arcsine square root (vegetation and BSC cover) or $\log_{10}$ (nitrogen, impact penetrometer, *B. tectorum* frequency).

To observe how variability in plant cover, BSC cover, and dynamic soil properties among plots related to grazing treatment, soil type, and landscape setting, we used nonmetric multidimensional scaling (NMDS) with the Sorenson (Bray-Curtis) distance measure (Kruskal, 1964; McCune and Mefford, 2011). The main matrix included plant functional group cover (shrubs, forbs, C3 and C4 grasses, and total plant cover); *B.*

tectorum frequency; as well as under canopy and interspace cover of lichens, mosses, and total soil cover. We also included in the main matrix soil variables that could vary as a response to grazing history. These included indicators of soil structure (surface and subsurface aggregate stability, surface and subsurface compaction), biogeochemical properties (nitrogen, phosphorus, organic matter, and sodium), and chlorophyll *a*, which provides an indicator of BSC cyanobacteria biomass (Karsten and Garcia-Pichel, 1996). For two missing values of surface compaction, we approximated values based on proximate transects within the same soil type and grazing treatment. In a second matrix of environmental data, we included static soil properties (soil texture, EC, slope, and topographic wetness index) and grazing history. We transformed data in the main matrix with a power transformation ($P = 0.5$) in PC-ORD and ran NMDS with the "slow and thorough" option in Autopilot. This default method performs 250 runs with real data and 250 runs with random data using a random seed number, a maximum of 500 iterations, and an instability criterion of 0.0000001. We overlaid the ordination with vectors (joint plot) that indicate the strength and direction of the relationship between variables and ordination axes (any vector with a hypotenuse length of < 0.5 was excluded). In addition, we tested differences among transects grouped by grazing treatment (grazed $n = 42$; ungrazed $n = 43$) or soil types (alluvial $n = 12$; mixed $n = 43$; sandstone $n = 18$; shale $n = 12$) using Multi Response Permutation Procedure (MRPP) on the square root transformed data matrix using the Sørensen distance measure and n/sum(n) in PC-ORD.

Finally, to provide a long-term perspective, we performed statistical comparison of our results to those of Lusby and Knipe (1971) and data recovered from the BLM. Unfortunately, the data from the original studies were not preserved and we are limited to those data discernable in the published literature (1953 and 1963) and from a 1972 progress report found in the Grand Junction BLM Field Office. The most relevant indicator from past work that we can reproduce is an index of ground cover, which is an expression of watershed cover, computed as 100 minus the number of hits on bare soil and rock not under a shrub overstory (Lusby and Knipe, 1971; Turner, 1971). To compare to this index, we used data from interspace BSC frames. Our ground cover was calculated as 100 minus the percent cover of rocks and bare ground in the interspaces, weighted by the proportion of interspace from the continuous line intercept transects. Thus, the 2004 ground cover index effectively includes plant canopy, plant basal, lichens, mosses, and plant litter. Bare ground included disturbed ground, light and dark cyanobacteria, and the vagrant cyanobacterium *Nostoc*. Because we did not have the data from previous work available for each soil type, we calculated watershed means by averaging transects within soil types for each watershed and then weighted those soil averages by the area each soil type represents within each of the eight watersheds. Past work excluded the alluvial soil type from analyses, and we have excluded alluvial transect data from 2004 in these comparisons. We used a mixed model (PROC MIXED; SAS Institute, 2012) with fixed effects of grazing history and watershed pair as random effects to compare grazed and ungrazed ground cover by year. The various field and laboratory data sets described, as well as new data generated through GIS and statistical analysis are available through the USGS ScienceBase-Catalog (Duniway and Geiger, 2018).

Results

Soils, Terrain Attributes, and Vegetation

Data collected along the transects in the Badger Wash study area demonstrate that the soil types mapped by Lusby et al. (1964) clearly differ in many soil and terrain attributes (Figs. 2 and 3; see Table 1). The sandstone transects were characterized by low slopes, low salinity, and coarser textures (sandy loams to loams in texture). The alluvial soil transects also had low slopes (Fig. 2b) and were generally in a run-in type topographic position (higher topographic wetness index; Fig. 2a) but encompass a range of salinity values and soil texture classes (Fig. 2c and 3). Transects established on the mixed soil encompassed a range of topographic positions, slopes, salinity, and textures (see Figs. 2 and 3). Finally, the shale soil type transects had the finest soil textures (clay loams to clays; see Fig. 3) and highest salinity (Fig. 2c). Vegetation composition and abundance, soil physical properties, and soil biogeochemistry were tightly coupled to soil types (see Table 1), with a significant effect of soil in all mixed models (Tables 2–5).

Effects of Grazing History on Vegetation, Soil Cover, and BSCs

Grazing history resulted in relatively few differences in vegetation and more differences in soil cover. In the vegetation responses analyzed, none had the significant main effect of grazing (see Table 2; Fig. 4). Grazing by soil interactions were detected for cool (C3) season perennial grasses, shrubs, total foliar cover, and frequency of *B. tectorum* (see Table 2). Perennial C3 grasses were much higher on ungrazed than grazed alluvial soils, while shrub cover was greater in grazed than ungrazed areas in both sandstone and alluvial soils (Fig. 5a and 5b). In contrast, shrub cover was greater in ungrazed than grazed mixed and shale transects (Fig. 5b). The only soil type where total foliar cover differed between grazed and ungrazed was in the mixed soils, in which ungrazed had greater total foliar cover (Fig. 5c). Frequency of *B. tectorum* was higher in grazed than ungrazed alluvial soils (Fig. 5d).

For soil cover and BSC biomass, the main effects of grazing history were seen in lichen and total soil cover (lichen, moss, rock, litter, and plant basal) and chlorophyll *a* concentrations, with ungrazed having higher values for all compared with grazed areas (see Table 3, Fig. 4b). Interactions of soil or microsite and grazing were only detected for lichen cover, which had a significant soil × grazing and soil × grazing × microsite interaction (see Table 3). There was greater lichen cover in ungrazed shrub interspaces of alluvial soils than other alluvial grazed or ungrazed microsites (Fig. 6). Similarly, interspaces in ungrazed mixed soils had greater lichen cover than ungrazed undercanopy locations, both of which had greater lichen cover than mixed grazed microsites (see Fig. 6). Association of lichen cover and microsite in ungrazed shale soils was opposite that of alluvial and mixed with greater cover under shrub canopies than in interspaces. Lichen cover in sandstone soils was low, with no differences detected due to microsite or grazing history (see Fig. 6).

Effects of Grazing History on Soil Physical and Biochemical Properties

Analysis of soil physical and biogeochemical properties revealed fairly consistent differences in key soil quality indicators between grazed and ungrazed watersheds. Surface soils in ungrazed watersheds had greater aggregate stability, weaker surface crusting (lower surface penetration resistance), and less subsurface compaction (lower penetration resistance at all depths; Fig. 7). Surprisingly, there was no interaction of soil type and grazing in soil aggregate stability or surface and subsurface penetration resistance (see Table 4). Among the biogeochemical responses reported here, organic matter, total nitrogen, and sodium concentrations were greater for ungrazed than grazed watersheds (top 10 cm; Fig. 8). Interactions of soil type and grazing history were detected for organic matter, total nitrogen, and sodium concentration, but not other biogeochemical responses (see Table 5). Examination of within — soil type responses to grazing history reveals that only for sandstone soils was organic matter greater in ungrazed than grazed watersheds (Fig. 9a). In contrast, sandstone soils were the only soil substrate in which no differences in total nitrogen were detected between grazed and ungrazed watersheds (Fig. 9b). The interaction of soil sodium concentration, soil type, and grazing was driven by extremely large differences in the alluvial soil type (ungrazed having more than 10-fold greater sodium) when compared with other soils (ungrazed showing 2 — 3 times greater sodium than grazed; Fig. 9c).

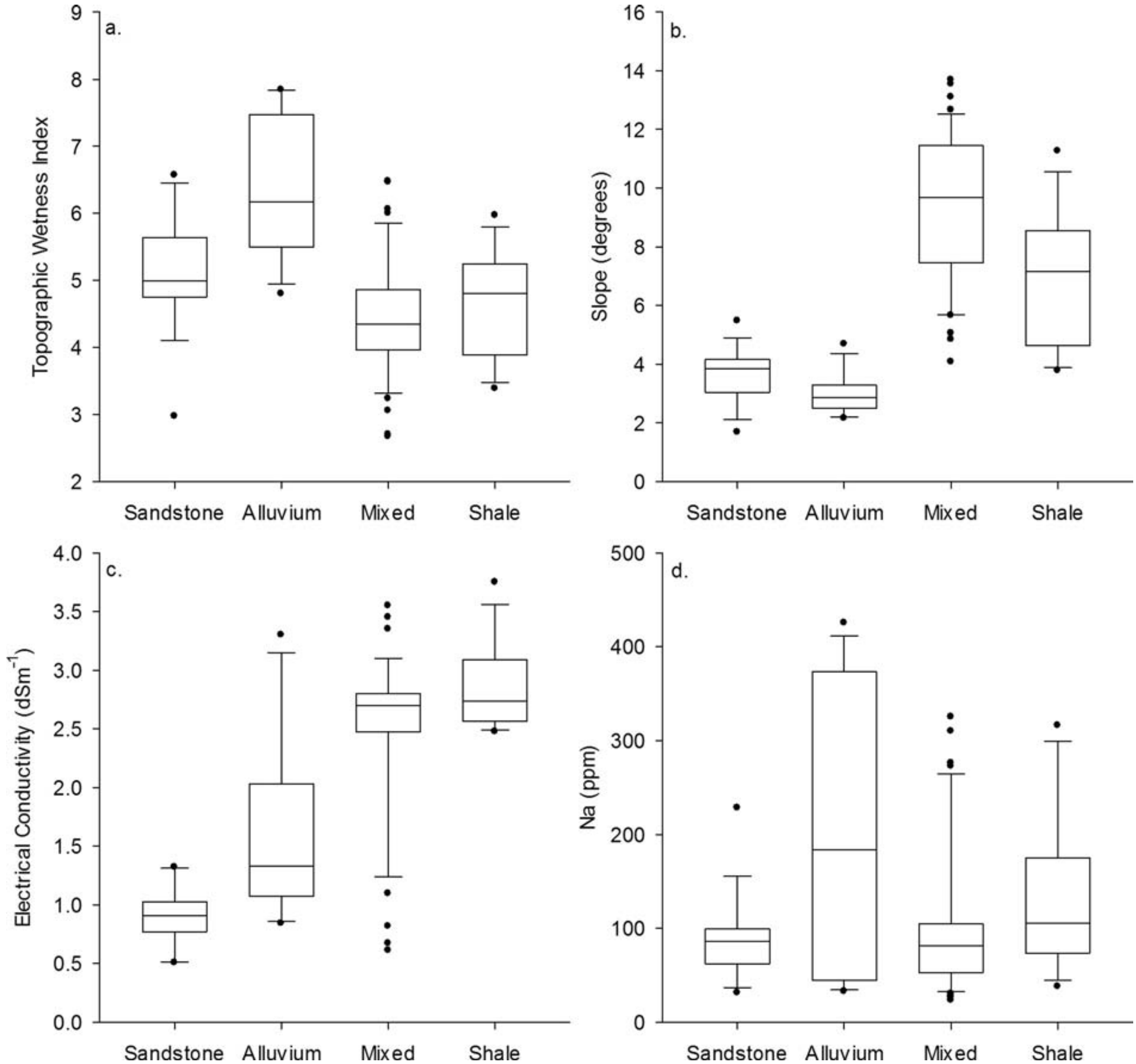

Figure 2. a, Average topographic wetness index, **b,** slope, **c,** soil electrical conductivity, and **d,** sodium along monitoring transects established in each soil type (as mapped by Lusby et al., 1964; see Table 1). Boxes represent 75th, 50th, and 25th percentiles; *whiskers* represent 90th and 10th percentiles; and outliers are indicated by *dots*.

Relationship Between Plant Community Response, Soil Quality Attributes, and Grazing History

The NMDS ordination revealed a 2-dimensional solution that explained 93.5% of the total variation, with 64.6% of that explained by the first axis (Fig. 10). The final stress was 11.35, instability 0.000, and iterations 86. The first NMDS axis was strongly positively associated with impact resistance, organic matter, and total nitrogen. The second NMDS axis was strongly positively associated with sodium and total nitrogen and negatively associated with impact resistance and abundance of *B. tectorum*. As shown by the joint plot overlaying the ordination graph, variability among plots was primarily associated with slope and EC from the secondary matrix. The multivariate analysis supports the results from our univariate analyses, with the MRPP revealing significant ($P < 0.001$) group association for both grazing history and soil type, with a slightly stronger association score for soil type (A = 0.165) than grazing (A = 0.067).

Trends in Ground Cover Over 50 Yr

Evaluation of total ground cover in the Badger Wash Study area over 50 yr shows a steady increase in ground cover in the ungrazed watersheds and an overall decrease, with high variability, in ground cover in the grazed watersheds (Fig. 11). Ground cover in the grazed watersheds was reduced to levels significantly less than in ungrazed watersheds in 1963 ($P = 0.015$) and 2004 ($P = 0.038$), but not before fence installation (1953) or in 1972.

Discussion and Conclusions

Grazing by sheep and cattle in the unfenced watersheds in the Badger Wash study area resulted in 31 − 40% increase in runoff (relative to ungrazed) after 13 yr (Lusby, 1970), and even following reduction in stocking rates from 1966 to 1973, there was still 40 − 45% higher runoff (Lusby, 1979). Sediment yield was 50% greater in the watersheds open to grazing than in the fenced (ungrazed) watersheds for much of the 20-yr study (Lusby, 1979). After 5 yr of protection, infiltration rates in the ungrazed mixed soil type were nearly double those in mixed soils open to grazing (Lusby et al., 1964). However, there were no pronounced differences in vegetative cover, and the dramatically differing hydrology and erosion processes observed between grazed and ungrazed watersheds were attributed to differences in ground cover (see Fig. 11; Lusby, 1970) and soil compaction by spring livestock use

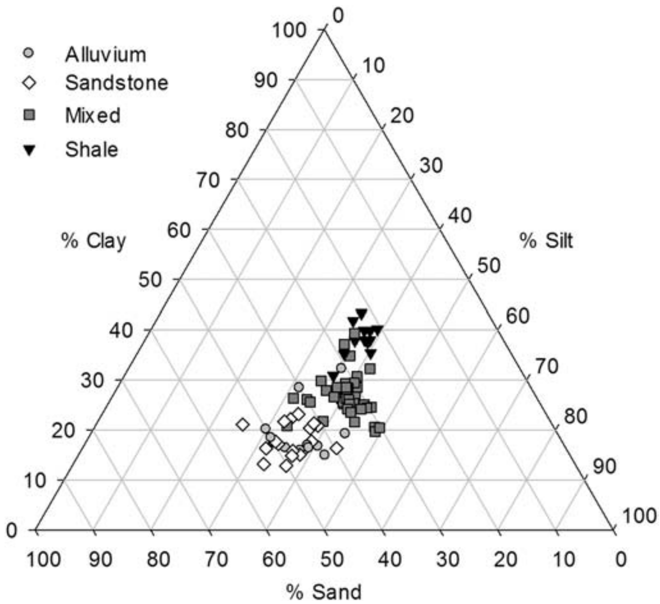

Figure 3. Average surface horizon (0–10 cm) soil texture (percent sand, silt, and clay) for each monitoring transect in each soil type (as mapped by Lusby et al., 1964; see Table 1).

(see Fig. 7; Lusby and Knipe, 1971). In addition, data collected during the first 20 yr of the Badger Wash study suggests that differing hydrology and erosion dynamics between grazed and ungrazed watersheds were driven by marked degradation of soils in watersheds open to livestock and less so by changes in the newly protected watersheds (see Fig. 11; Lusby, 1970).

Grazing History Impacts on Soils, Vegetation, and Implications for Erosion

Differences in watershed condition continue to persist 30 yr later, with the primary differences between grazed and ungrazed watersheds in variables related to soil quality (soil cover, structure, and biogeochemistry; see Tables 2–5). The soil variables that showed consistent differences

Table 2
Fit statistics from mixed linear-model univariate analysis of variance for vascular plant functional groups (see Table 1) and *Bromus tectorum* responses to grazing history and soils. Model included grazing history (Grazing; grazed vs. ungrazed watersheds), soil type (Soil; as mapped by Lusby et al., 1964), and watershed pair as a random effect.

Cover group	Effect	*df*[1]	F	*P*
Perennial forbs				
	Grazing (G)	1	1.1	0.300
	Soil (S)	3	4.3	0.008
	G x S	3	0.9	0.470
C3 Grasses				
	Grazing	1	1.3	0.262
	Soil	3	16.0	<0.001
	G x S	3	3.4	0.022
C4 Grasses				
	Grazing	1	0.5	0.504
	Soil	3	3.6	0.018
	G x S	3	0.7	0.573
Shrubs				
	Grazing	1	0.9	0.353
	Soil	3	35.6	<0.001
	G x S	3	6.5	0.001
Total plant cover				
	Grazing	1	1.7	0.193
	Soil	3	19.8	<0.001
	G x S	3	2.4	0.079
B. tectorum				
	Grazing	1	2.5	0.117
	Soil	3	33.8	<0.001
	G x S	3	2.4	0.075

[1] Numerator degrees of freedom; denominator degrees of freedom, 74.

Table 3
Fit statistics from mixed linear-model univariate analysis of variance for biological soil crust cover, total ground cover, and surface soil chlorophyll *a* responses to grazing history and soils. Soil cover models included grazing history (Grazing; grazed versus ungrazed watersheds), soil type (Soil; as mapped by Lusby et al. [1964]), microsite (under canopy vs. interspace), and watershed pair as a random effect. Soil chlorophyll *a* model was the same except for the microsite factor (sampling for chlorophyll not stratified by microsite).

Cover group	Effect	*df*[1]	F	*P*
Lichen				
	Grazing (G)	1	10.1	0.002
	Soil (S)	3	3.2	0.026
	G x S	3	2.5	0.060
	Microsite (M)	1	4.2	0.043
	G x M	1	0.0	0.956
	S x M	3	3.3	0.022
	G x S x M	3	4.0	0.009
Moss				
	Grazing	1	2.1	0.146
	Soil	3	10.8	<0.001
	G x S	3	0.5	0.655
	Microsite	1	13.1	<0.001
	G x M	1	0.1	0.720
	S x M	3	6.9	<0.001
	G x S x M	3	1.1	0.331
Total ground cover				
	Grazing	1	6.1	0.015
	Soil	3	10.9	<0.001
	G x S	3	0.1	0.931
	Microsite	1	569.3	<0.001
	G x M	1	0.3	0.608
	S x M	3	14.8	<0.001
	G x S x M	3	1.6	0.183
Chlorophyll *a*				
	Grazing	1	3.2	0.076
	Soil	3	6.0	0.001
	G x S	3	1.0	0.379

[1] Numerator degrees of freedom; denominator degrees of freedom for soil cover, 151 and for chlorophyll *a* 74.

Table 4
Fit statistics from mixed linear-model univariate analysis of variance for dynamic soil physical property responses to grazing history and soils. Model included grazing history (Grazing; grazed vs. ungrazed watersheds), soil type (Soil; as mapped by Lusby et al. [1964]), and watershed pair as a random effect.

Soil test	Effect	*df*[1]	F	*P*
Aggregate stability shallow				
	Grazing (G)	1	36.0	<.001
	Soil (S)	3	5.1	0.003
	G x S	3	0.2	0.914
Aggregate stability deep				
	Grazing	1	0.2	0.635
	Soil	3	2.7	0.051
	G x S	3	1.3	0.269
Surface crusting				
	Grazing	1	6.0	0.017
	Soil	3	4.7	0.005
	G x S	3	1.5	0.221
0-5 cm Penetrometer				
	Grazing	1	15.8	<0.001
	Soil	3	3.7	0.017
	G x S	3	1.2	0.330
5-10 cm Penetrometer				
	Grazing	1	12.4	0.001
	Soil	3	5.7	0.002
	G x S	3	1.0	0.411
10-15 cm Penetrometer				
	Grazing	1	8.0	0.006
	Soil	3	11.8	< 0.001
	G x S	3	1.8	0.151
15-20 cm Penetrometer				
	Grazing	1	5.6	0.021
	Soil	3	5.8	0.001
	G x S	3	1.5	0.228

[1] Numerator degrees of freedom, denominator degrees of freedom aggregate stability, 74; surface penetrometer, 72; and penetrometer (all depths), 68.

Table 5
Fit statistics from mixed linear-model univariate analysis of variance for biogeochemical responses to grazing history and soils. Model included grazing history (Grazing; grazed vs. ungrazed watersheds), soil type (Soil; as mapped by Lusby et al. [1964]), and watershed pair as a random effect.

Cover group	Effect	*Df*[1]	F	*P*
Organic matter				
	Grazing (G)	1	5.2	0.026
	Soil (S)	3	9.3	<0.001
	G x S	3	2.2	0.096
Electrical conductivity				
	Grazing	1	0.4	0.507
	Soil	3	26.9	<0.001
	G x S	3	0.6	0.647
Phosphorus				
	Grazing	1	1.0	0.314
	Soil	3	25.0	<0.001
	G x S	3	0.9	0.437
Total nitrogen				
	Grazing	1	44.8	<0.001
	Soil	3	7.1	<0.001
	G x S	3	5.6	0.002
Sodium				
	Grazing	1	96.2	<0.001
	Soil	3	14.1	<0.001
	G x S	3	16.2	<0.001

[1] Numerator degrees of freedom; denominator degrees of freedom, 74.

between grazed and ungrazed watersheds in 2004 (not limited to an individual soil type) are important for hydrologic function and resistance to erosion: soil stability, soil crusting, soil compactions, soil cover, biological soil crust biomass, and soil organic matter (Branson et al., 1981). This low sensitivity to vegetation and high soil sensitivity is likely due to a variety of factors, including sparse and largely unpalatable vegetation (Blaisdell and Holmgren, 1984; Alzerreca-Angelo et al., 1998), large gaps between perennial species, soils with fragile physical or biological soil crusts, and subsoils susceptible to compaction (Schumm and Lusby, 1963; Marble and Harper, 1989; Carpenter and Chong, 2010). This relatively low response of vegetation but significant impacts on soils suggests that resource managers for these systems will need to consider indicators other than forage utilization or other vegetation-based assessments when evaluating livestock management (Herrick et al., 2002; Bestelmeyer et al., 2013). Qualitative/semiquantitative approaches such as the Interpreting Indicators of Rangeland Health, when applied appropriately, do include a multitude of indicators that are sensitive to changes in soil quality such as those observed here (Pellant et al., 2005; Duniway et al., 2010).

Livestock-induced differences in runoff and erosion are particularly important for rangelands derived from Mancos Shale and similar saline geologic strata (e.g., Miller et al., 2017; see Fig. 1a). Although grazing history had no detectable effect on soil EC, soil sodium in the top 10 cm was higher in ungrazed than grazed pastures (see Fig. 8) across all four soil types (Fig. 9c). In saline soils with fine textures, capillary

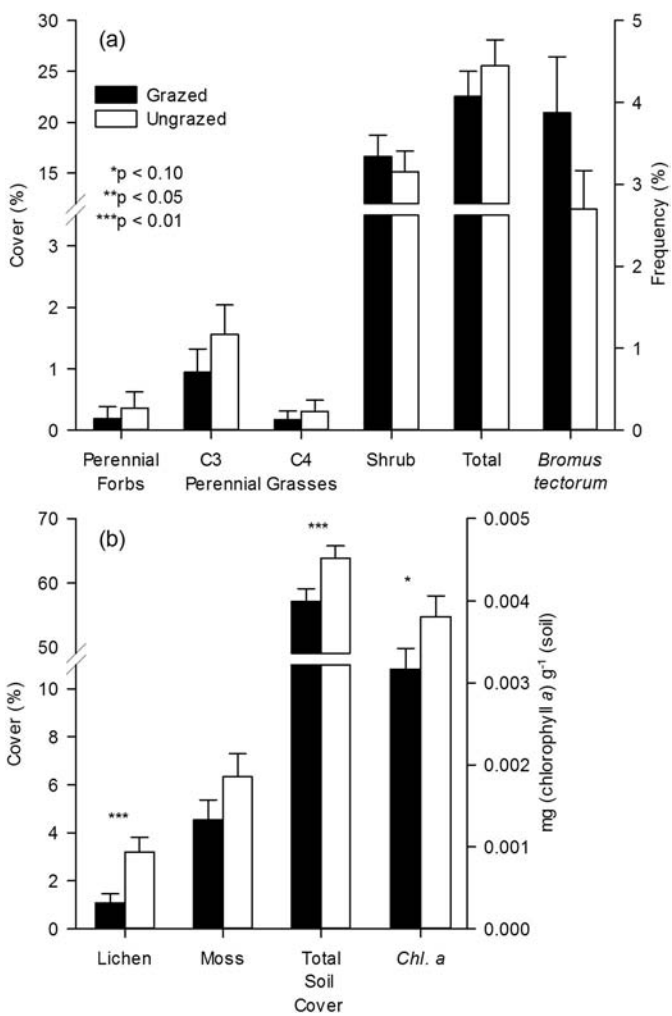

Figure 4. Average response of perennial vegetation and the invasive annual grass *Bromus tectorum* (**a**) and biological soil crusts to grazing history (**b**). Cover of perennial vegetation is grouped by functional groups (see Table 1) and is based on continuous line intercept. *B. tectorum* is reported as rooted frequency within 0.25 m^2 frames. Soil cover of lichens, mosses, and total soil cover (rocks, litter, plant basal, lichen, and moss combined) is based on point-intercept frames. Chlorophyll *a* is based on soil samples from the top 0.5 cm and provides an estimate of soil cyanobacteria biomass. Averages and standard error (error bars) are based on least-squares mean from mixed linear-model (see Tables 2 and 3).

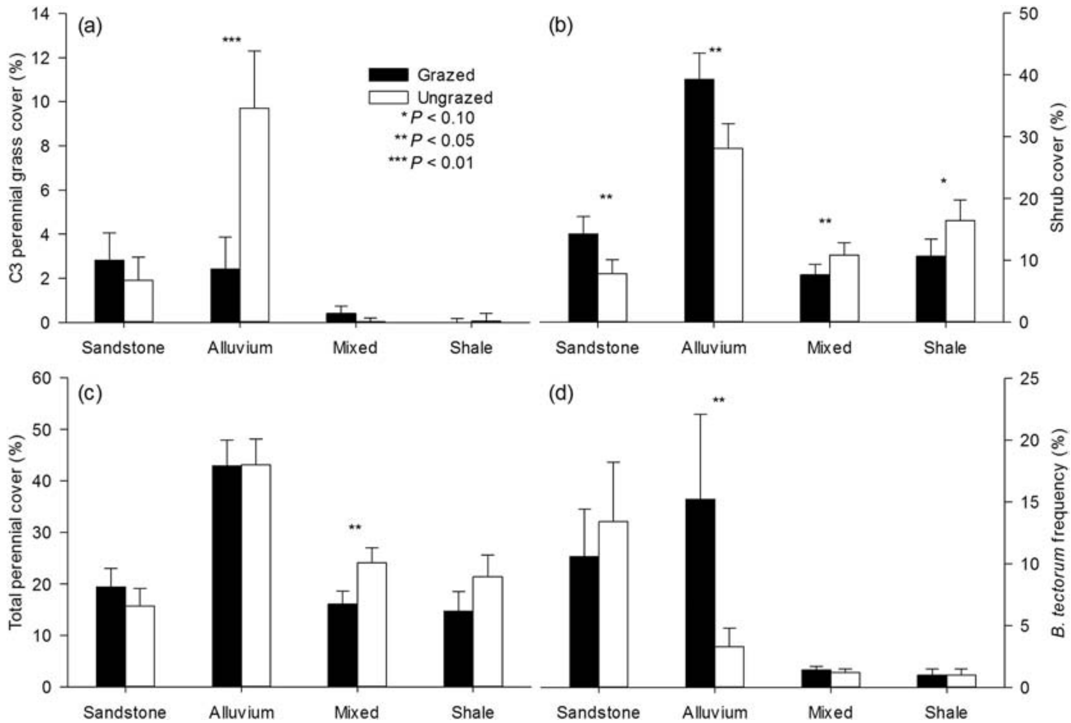

Figure 5. Vegetation indicators (see Table 1, Fig. 4) that differed in response to grazing history among soil types (significant soil by grazing interaction; see Table 2). Average C3 perennial grass cover (**a**), average shrub cover (**b**), average total perennial cover (**c**), and frequency of *Bromus tectorum* (**d**) by soil type (as mapped by Lusby et al., 1964; see Table 1). *Asterisks* indicate significant grazing effect. Averages and standard error (error bars) are based on least-squares mean from mixed linear-model (see Table 2).

movement of soil water can create high concentrations of salts at the soil surface due to evaporation (the mechanisms behind salt crusts; Whittig et al., 1982). Under these circumstances, conditions that accelerate erosion could result in loss of these high-salt surface horizons and therefore explain the differences in sodium levels observed. This effect of disturbance on surface soil salts has been observed in other studies (Elliott et al., 2008; Carpenter and Chong, 2010), and combined with past research documenting accelerated erosion in grazed watersheds (Lusby, 1970), suggests that surface soils in the grazed watersheds are continuing to erode in response to use by livestock. This is of concern in many western watersheds that contain soils with naturally high salinity levels (e.g., Mancos Shale), and especially the Colorado River, given US treaty obligations with Mexico that regulate water salinity levels, and the high economic costs of elevated salinity levels (estimated at $382 million per year in 2014; Miller et al., 2017; US Bureau of Reclamation, 2017).

Differences due to grazing history are most pronounced within the alluvial soil setting (see Table 1; see Figs. 5, 6, and 9), which were not evaluated in previous studies. Field observations suggest current livestock use (cattle) is heavier in the alluvial soils than within other areas in the grazed watersheds, likely due to low slopes, higher amounts of palatable vegetation (especially grass), and easy access to water (from retention ponds within the experimental watersheds or in points down valley). Changes in vegetation with continued livestock grazing in this sagebrush vegetation community *(Artemisia tridentata)* are similar to those observed in sagebrush communities across the west: loss of cool-season perennial grasses, increases in shrub cover, and invasion by the exotic annual grass *B. tectorum* (see Fig. 5; Germino et al., 2016). The loss of lichens in the shrub interspaces and reductions in surface soil total nitrogen and sodium in these salt desert sagebrush ecosystems has not been a previously documented impact of grazing (but see examples in other Colorado Plateau and dryland community

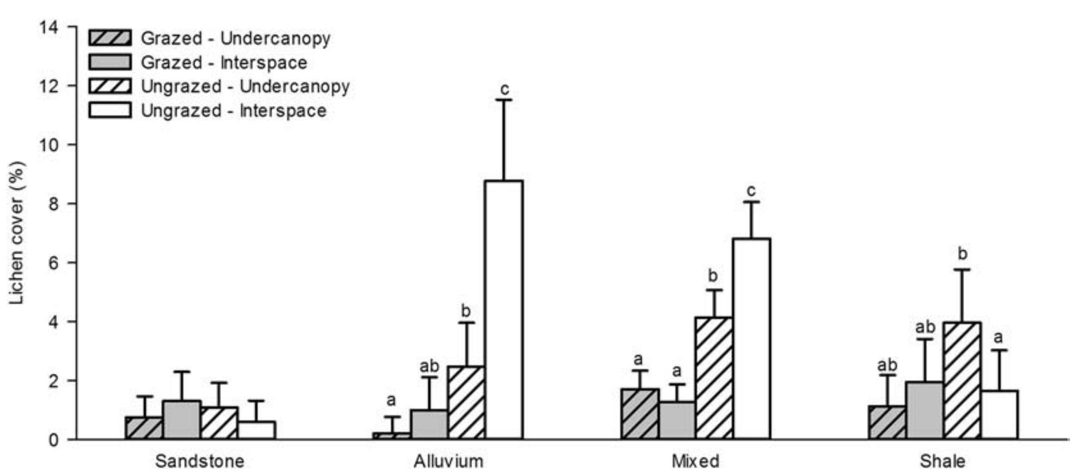

Figure 6. Soil cover of lichens by soil type, grazing history, and microsite (undercanopy and interspace; soils as mapped by Lusby et al., 1964; see Table 1). Bars with same letter within soil type are not different ($P < 0.1$). Averages and standard error (error bars) are based on least-squares mean from mixed linear-model (see Table 3).

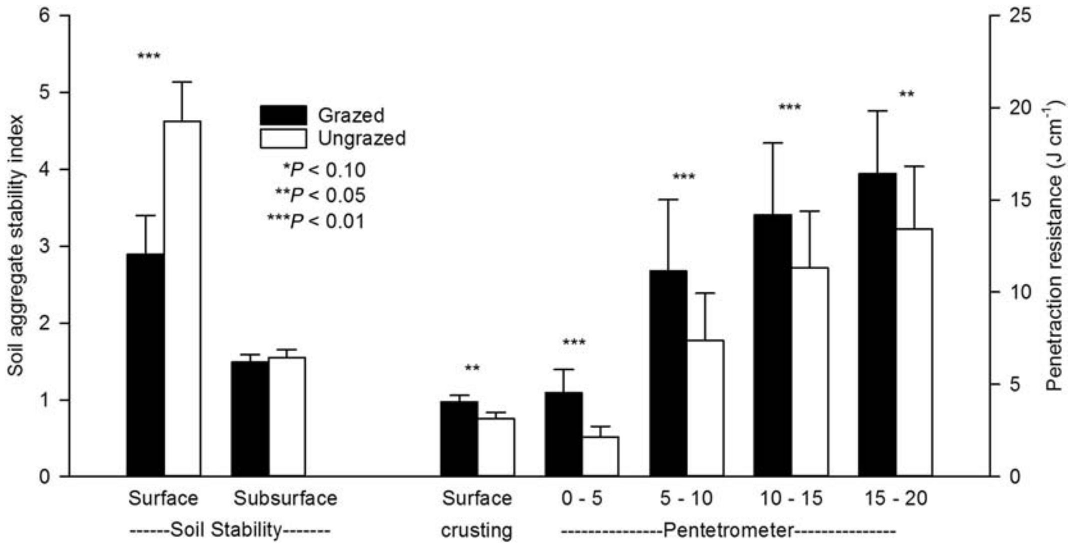

Figure 7. Average soil aggregate stability, surface crusting, and subsurface compaction in response to grazing history. Greater values of the soil aggregate stability index indicates greater stability of aggregates in water (Seybold and Herrick, 2001). Soil surface crusting was determined using a fruit pressure tester with an 8-mm diameter disk (greater resistance is associated with greater crusting). Subsurface compaction was measured with an impact penetrometer in 5-cm depth increments (Herrick and Jones, 2002). Averages and standard error (error bars) are based on least-squares mean from mixed linear-model (see Table 4).

types; Belnap and Eldridge, 2003; Collins et al., 2014; Ferrenberg et al., 2015). We attribute these changes to trampling of interspace soils by cattle, causing loss of lichens and increased erosion (Belnap and Eldridge, 2003). Livestock use had been very low in the years before the current measurements (2000 − 2004), which suggests that these differences in ecosystem condition between alluvial soil settings in grazed and ungrazed areas are due to long-term management differences and not short-term effects of grazing during the drought of the early 2000s (see Fig. 11).

Patterns in Lichen Cover and Recovery

An important revelation of this new work at Badger Wash is the interaction between soil type, microsite (under shrub vs. interspace), and grazing history on the distribution of soil lichens in salt desert ecosystems (see Fig. 6). Interestingly, there was no grazing effect on mosses (either main effects or any interactions). However, our observations and those of Turner (1971) are that mosses tend to prefer undershrub microsites, thus avoiding trampling by livestock. The very low cover of lichens on the sandstone-derived soils in all watersheds and microsites is likely due to the inherent instability of this coarse soil type. In contrast, there is high cover and strong grazing and microsite differences in the finer-textured alluvial and mixed soils. Lichens are well documented to have higher cover and faster recovery rates on soils with greater amounts of silt and nonswelling clay (e.g., Anderson et al., 1982; Belnap and Büdel, 2016). The strong differences in lichen cover between ungrazed and grazed interspaces are likely due to the high vulnerability of these soils to compressional disturbances (e.g., trampling by livestock and humans; Belnap and Eldridge, 2003; Ferrenberg et al., 2015) and likely greater use by livestock (especially compared with shale soils).

It may seem surprising that under ungrazed conditions on the alluvial and mixed soils, lichen cover was higher in the interspace than under the shrub canopy and that with grazing, cover was equalized between the two locations. There are several possible explanations: 1) because soil lichens in this region grow only a few mm high and there is substantial litter buildup under the shrubs, the lichens could easily be buried, reducing their photosynthetic capabilities and thus carbon gain when they occur in this microsite; and 2) these lichen species are tolerant of very high light (Lange, 2003) and thus shade may not be of

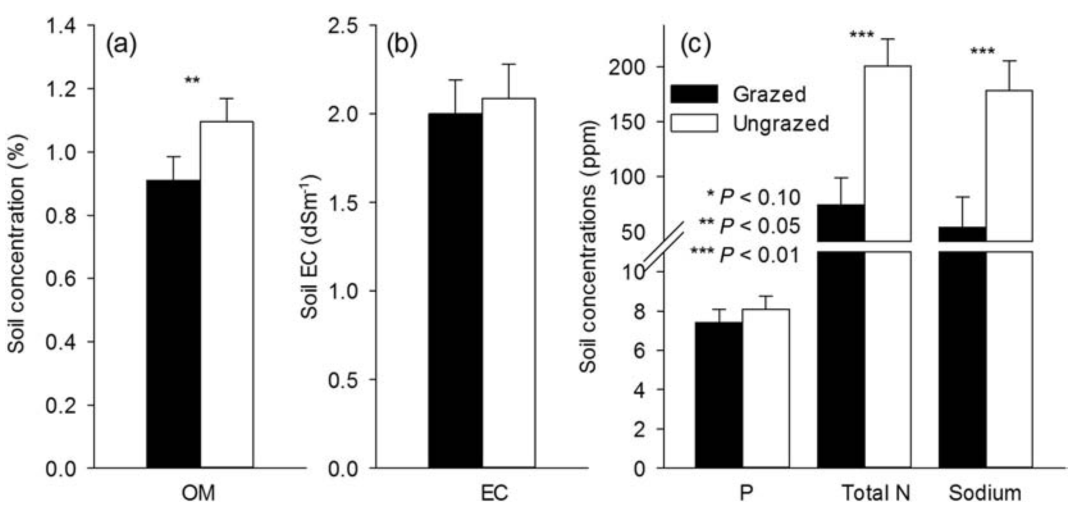

Figure 8. Average top 10 cm soil biogeochemical responses to grazing history. **a,** Concentration of soil organic matter; **b,** soil electrical conductivity (EC); and **c,** phosphorus (P), total nitrogen (N), and sodium (Na). Averages and standard error (error bars) are based on least-squares mean from mixed linear-model (see Table 5).

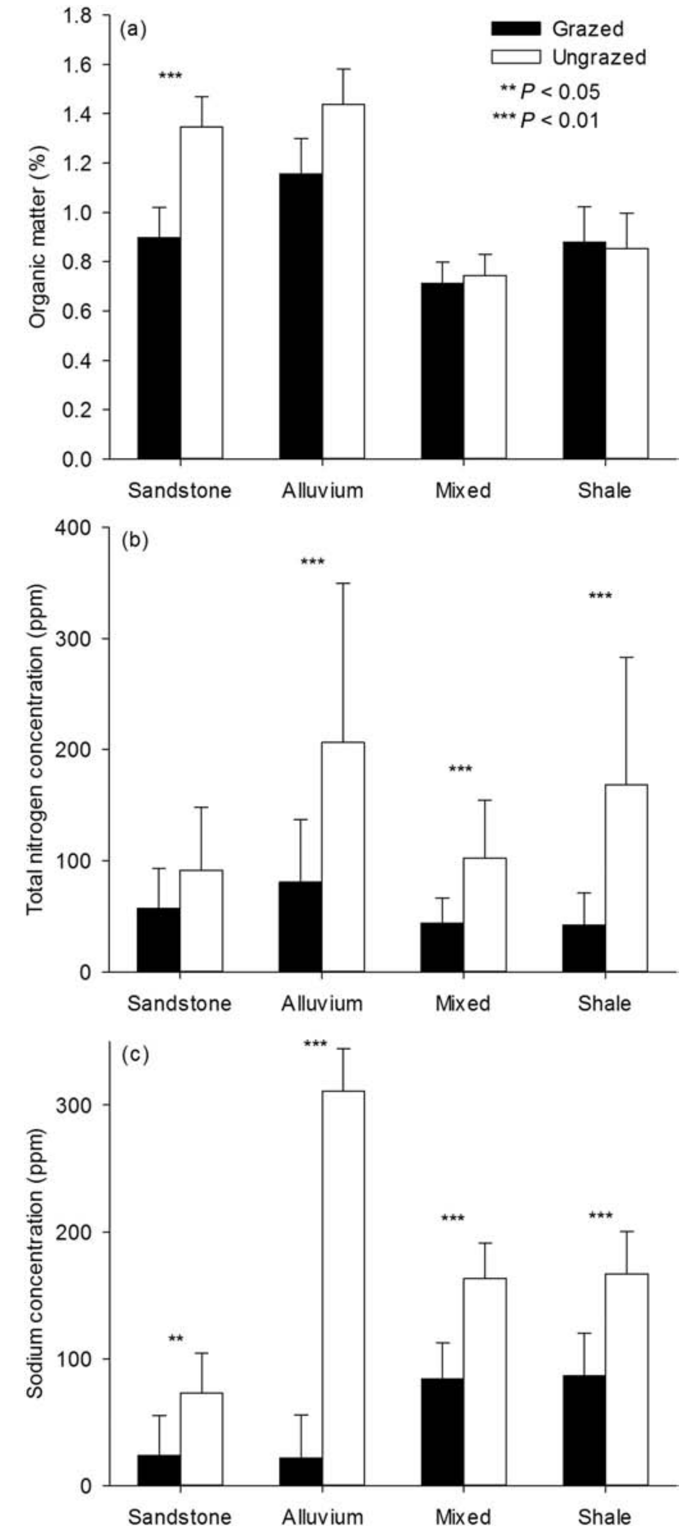

Figure 9. Soil biogeochemical indicators (see Fig. 8) that differed in response to grazing history among soil types (significant soil by grazing interaction; see Table 5). **a,** Concentration in top 10 cm of organic matter, **b,** total nitrogen, and **c,** sodium by soil type (as mapped by Lusby et al., 1964; see Table 1). Averages and standard error (error bars) are based on least-squares mean from mixed linear-model (see Table 2).

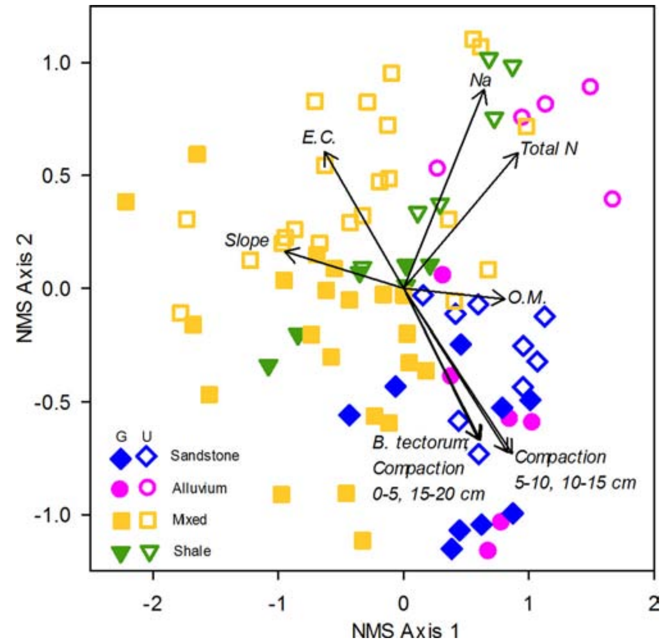

Figure 10. Nonmetric Multidimensional Scaling (NMS) ordination of transect of plant functional group cover, biological soil crust cover, and other dynamic soil properties (closed symbol, grazed; open symbol, ungrazed). Correlation of the top seven indicator species/attributes displayed as vectors. Also shown are the top two soil or topographic variables not in the primary matrix (slope and soil electrical conductivity; EC). Vector length scaled to correlation strength.

great benefit to them, especially relative to being buried under plant litter. Higher cover of lichens in interspaces has also been observed in studies in the Great Basin (Hilty et al., 2003), as well as within long-term ungrazed areas in nearby national parks (Belnap, *personal observation*).

Patterns in lichen cover on the shale soils were somewhat different (see Fig. 6). As was observed in the sandstone soils, there was no difference in lichen cover between grazed and ungrazed watersheds. Although average cover of lichens under shrub canopies was similar in mixed and shale soils, a difference in lichen cover was observed in ungrazed shale soils where cover was higher under the canopy than the interspace, unlike in the alluvial and mixed soils. This low cover of lichens in the shale soil interspaces is likely due to the erodible nature of these soils and prevalence of shrink-swell clays (Spears and Kleven, 1978; Potter et al., 1985). Shrub canopies likely provide protection from erosion (both from sheet flow and raindrop impact), and the shade provided by the shrub canopies in the shale soils appears to allow surface soils to stay wet for longer periods (particularly in the winter; Duniway, *personal observation*), slowing drying and thus limiting damage from shrink-swell.

Evidence for Degradation and Recovery

Overgrazing by domestic livestock during drought has been implicated in ecosystem state changes that exhibit hysteresis across the southwestern United States. For example, grazing during the droughts of the 1930s and 1950s in Chihuahuan Desert grasslands have been connected to persistent grass loss and increased shrub dominance, despite subsequent reduction or elimination of grazing (Herbel et al., 1972; Browning et al., 2012). However, cover of perennial grasses was shown to increase quasiexponentially when grazing was excluded from 1950 to 1975 in the sagebrush steppe (Anderson and Inouye, 2001). There is evidence of similar grazing-induced ecosystem changes on the Colorado Plateau during the same periods (Godfrey, 2008; Denis, 2012). County-level livestock data in the study area and historical accounts show extremely high stocking rates in the late 1800s and early 1900s and relatively high stocking rates through the 1950s and 1960s in some counties (see Fig. 11b; Godfrey, 2008; Copeland et al., 2017). Indeed, there is evidence from a variety of sources, including recent survey work, examination of legacy datasets, and historical accounts, of persistent vegetation and soil changes in the region following

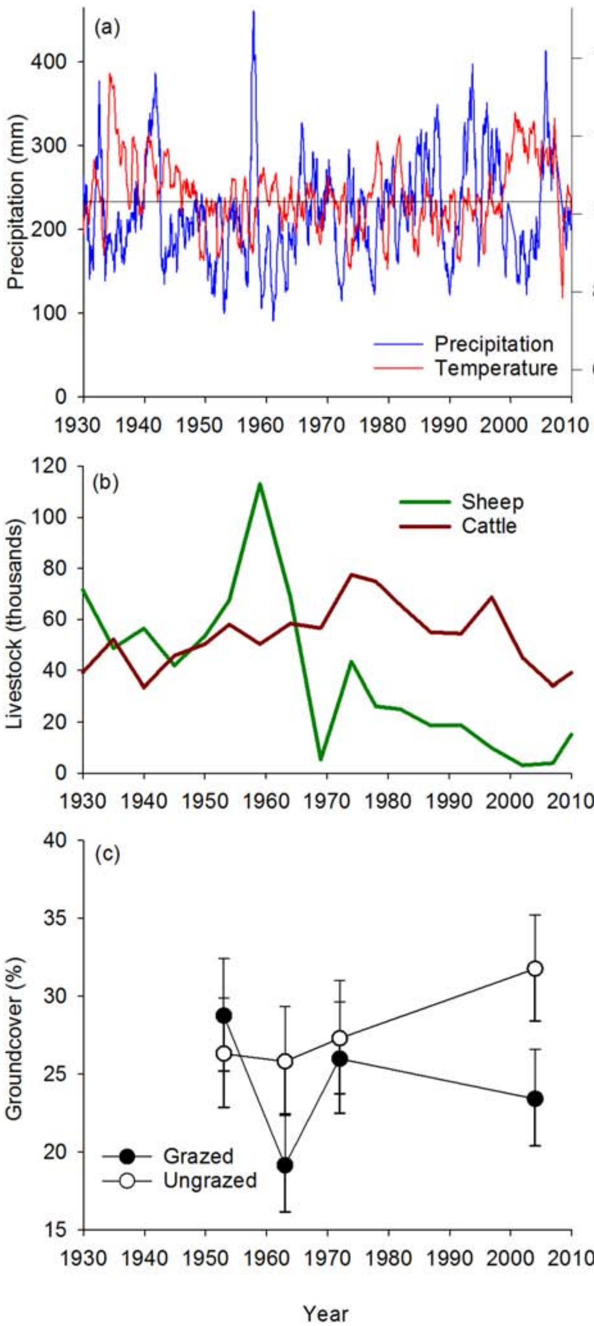

Figure 11. a, Study period climate (12-mo average precipitation and temperature, black horizontal line is long-term average); **b,** county-level cattle and sheep totals from the study area (from the USDA Census of Agriculture for Mesa County, Colorado; see Copeland et al., 2017 for details); and **c,** comparisons of ground cover in 1953 (before grazing exclosures erected), 1963, 1972, and 2004. Error bars represent standard error of the means (watersheds as the experimental units; $n = 4$ per treatment). Data from 1953 and 1963 from Lusby et al. (1964), data from 1972 based on report available from Grand Junction BLM, and data from 2004 based on new measurements reported here.

overgrazing and droughts from the late 19th through mid-20th centuries (Fernandez et al., 2008; Neff et al., 2008; Miller et al., 2011; Denis, 2012).

Studies at Badger Wash were established toward the very end of the high sheep and cattle livestock use in the study area (high use until 1957, described in Lusby et al. 1979) and provide an opportunity to understand the resilience of Colorado Plateau salt desert communities to grazing. Examination of ground cover and monitoring of runoff and erosion in the Badger Wash study area suggest a relatively rapid response of the grazed watersheds to shorter-term changes in climate (see Fig. 11a) and management (increase in county sheep numbers; Fig. 11b), with a precipitous drop in ground cover and increase in runoff and sediment yield during high use of the first 10 yr (Turner, 1971; Fig. 11c). Examination of 12-mo average precipitation and temperature deviation suggests that the regional drought of the 1950s also impacted the Badger Wash Study area between study initiation in 1953 (and fencing of the ungrazed watersheds) and the 1963 measurement (Fig. 11a); there was a concurrent doubling of county-level sheep numbers during the same period (Fig. 11b). Although there is a brief recovery of ground cover in 1972 following large reductions in livestock (Fig. 11b) and return of cooler and wetter conditions (Fig. 11a), important differences in soil quality of grazed and ungrazed watersheds persisted into 2004. These results, coupled with evidence from other ecosystems of long-term ecosystem changes triggered by combinations of drought and overgrazing during the same time periods (e.g., Herbel et al., 1972; Peters et al., 2007), suggest that much of the degradation observed in salt desert ecosystems of this regional is potentially attributable to improper management that occurred during droughts many decades ago and may represent a change in ecological state (e.g., Bestelmeyer et al., 2015).

Early work hypothesized that the large differences in runoff evident after just 2 yr of protection were due to spring trampling by livestock, compacting soils, breaking up surface soil physical and biological crusts and, in general, negating the effects of winter freeze-thaw cycles (Lusby and Knipe, 1971). This hypothesis was confirmed with the removal of livestock from two of the grazed watersheds (2A and 4A) in 1965 and finding of no difference in runoff from 1966 to 1973 between these recently ungrazed and long-term ungrazed watersheds (and continued differences between long-term grazed and ungrazed; Lusby, 1979), despite similar amounts of ground cover across all watersheds in 1972 (Fig. 11c). The rapid fluctuation in ground cover index in grazed watersheds in the 1953 − 1972 period is quite remarkable (Fig. 11c), driven by rock and bare, moss and litter, and, to a lesser extent, shrub overstory (data not shown). An aspect of the vegetation that differed between grazed and ungrazed watersheds in 1963 that is not emphasized in the earlier work is the frequency of annual grasses—*B. tectorum* was twice as frequent in grazed than ungrazed mixed soils and 20% more frequent in grazed than ungrazed sandstone soils (1.86 m^2 frames; Turner, 1971). This large difference in frequency in 1963 suggests that *B. tectorum* increased in grazed but not ungrazed watersheds from 1953 to 1963 (which is corroborated by spring measures reported in Lusby et al., 1964), but *B. tectorum* litter was not sufficient to provide ground cover during drought conditions in 1963. However, it is likely that *B. tectorum* litter could explain the rapid recovery of "moss and litter" cover class due to the sequence of wet years before the 1972 measurement. The 2004 measures were again done in a drought, with low cover and relatively low frequency of *B. tectorum* in the soil units monitored by previous studies (Sandstone, Mixed, and Shale; see Fig. 5).

Another process leading to differences between grazed and ungrazed watersheds that is now evident is the steadily increasing ground cover in the watersheds protected from livestock: ground cover in watersheds fenced in 1953 has increased an average of 1.4% per decade since 1963 ($R^2 = 0.99$; Fig. 11c), despite the drought conditions of 2004. This increase in ground cover in ungrazed watersheds is likely partially attributable to recovery of BSCs, especially in mixed soil interspaces (see Table 1, Fig. 6). Lusby and others did not differentiate cover of lichens from mosses, as there was no or little awareness of the importance of BSCs for soil stability at that point in time (Lange and Belnap, 2016). In fact, we are not aware of any study reporting natural BSC recovery anywhere in the western United States until 1984 (Ashley and Rushforth, 1984). However, given the very low cover of lichens in grazed pastures in 2004 (even with reduced stocking rates relative to those in the 1940s and 1950s; Fig. 11b), we expect lichen cover to have been very low in 1953 in all watersheds, especially in the interspaces (which would contribute to the ground cover index). Therefore, the differences in lichen cover between grazed and ungrazed microsites observed in 2004 (see Fig. 6) and increasing ground cover index from 1966 to 2004 (Fig. 11c) is likely partially attributable to recovery of lichen communities with protection from livestock and lack of recovery in the watersheds open to grazing. These rates of lichen recovery (1 to 4% per yr) are similar to those observed for recovery of BSC biomass in another study in this region (Belnap, 1993).

Implications

The long-term perspective provided by the Badger Wash studies continues to shed new insights into the resilience of Colorado Plateau salt desert ecosystems. The most pronounced and important insights from the early work are still true today: Impacts of livestock on salt desert vegetation may not be readily apparent, but livestock trampling can reduce ground cover and negatively affect soil stability and hydrologic function (Turner, 1971). Grazing by domestic livestock is one of the most widespread land-use types on the Colorado Plateau (Copeland et al., 2017). New and innovative research is needed to evaluate livestock management approaches, perhaps coupled with novel restoration techniques, that will facilitate recovery of rangeland soil stability and hydrologic function without necessitating complete removal of cattle and sheep, particularly with forecasts for more frequent and severe droughts (Cook et al., 2015). Results from the new work presented here suggest that the ground cover in these systems can increase and persist during drought when livestock is excluded. There is some evidence from the early experiments that changes in timing of use can mitigate some damage to soils (Lusby, 1979), but new research that examines alternative grazing strategies in Colorado Plateau salt desert ecosystems (beyond simply excluding livestock) is needed. Finally, the Badger Wash study results, including the new work reported here and data from the original researchers, support many current assessment and monitoring efforts that emphasize indicators of soil quality for evaluating rangeland condition (Pellant et al., 2005; Toevs et al., 2011).

Acknowledgments

We would like to thank the many assistants and technicians who contributed to the Badger Wash study, including Philip Adams, Richard Alward, Adam Atchley, Adam Collins, Chris Ely, Ed Grote, Adam Kind, John Mayo, Maegan McKee, Matt Van Scoyoc, Hilda Smith, and Natalie Melaschenko. Use of trade, product, or firm names is for information purposes only and does not constitute an endorsement by the US government. Data generated during this study are available from the USGS ScienceBase-Catalog (Duniway and Geiger, 2018).

Appendix A. Supplementary Data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.rama.2018.02.007.

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