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MULTI-DECADAL IMPACTS OF GRAZING ON SOIL PHYSICAL AND BIOGEOCHEMICAL PROPERTIES IN SOUTHEAST UTAH

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Abstract. Many soils in southeastern Utah are protected from surface disturbance by biological soil crusts that stabilize soils and reduce erosion by wind and water. When these crusts are disturbed by land use, soils become susceptible to erosion. In this study, we compare a never-grazed grassland in Canyonlands National Park with two historically grazed sites with similar geologic, geomorphic, and geochemical characteristics that were grazed from the late 1800s until 1974. We show that, despite almost 30 years without livestock grazing, surface soils in the historically grazed sites have 38–43% less silt, as well as 14–51% less total elemental soil Mg, Na, P, and Mn content relative to soils never exposed to livestock disturbances. Using magnetic measurement of soil magnetite content (a proxy for the stabilization of far-traveled eolian dust) we suggest that the differences in Mg, Na, P, and Mn are related to wind erosion of soil fine particles after the historical disturbance by livestock grazing. Historical grazing may also lead to changes in soil organic matter content including declines of 60–70% in surface soil C and N relative to the never-grazed sites. Collectively, the differences in soil C and N content and the evidence for substantial rock-derived nutrient loss to wind erosion implies that livestock grazing could have long-lasting effects on the soil fertility of native grasslands in this part of southeastern Utah. This study suggests that nutrient loss due to wind erosion of soils should be a consideration for management decisions related to the long-term sustainability of grazing operations in arid environments.

Key words: arid; carbon; desert; erosion; grazing; nitrogen; phosphorus; soil; soil crust; Utah.

Introduction

Soil stability is a primary control over the fertility, productivity, and sustainability of managed ecosystems in arid and semi-arid parts of the United States. Most arid/semi-arid lands have sparse vegetation cover and low surface-soil organic matter content (Follett 2001). Disturbance to surface soils by activities such as livestock grazing can influence arid-land ecosystem fertility in many ways including through the alteration of vegetation cover, soil physical properties, microbial communities, carbon (C) cycling, nitrogen (N) fixation, and hydrologic properties (e.g., Schlesinger et al. 1990, Verstraete and Schwartz 1991). Another potential impact of grazing in arid-land soils is the disruption of biological soil crusts (BSC), which influence nutrient cycling and stabilize surface soils (Belnap and Gillette 1998, Belnap and Lange 2001).

Biological soil crusts are made up of associations of cyanobacteria and cyanolichens, sometimes also including mosses, microfungi, green algae, and bacteria (Marble and Harper 1989, Belnap and Lange 2001). Desert soil nutrient availability is often closely related to BSCs, which, in addition to increasing soil N content through N fixation, also exude organic acids and organic compounds that increase cation exchange capacity (CEC) of sandy soils, decrease soil pH, and increase soil water holding capacity, all of which may facilitate the weathering of rock-derived nutrients (Schwartzman and Volk 1991, Belnap 2003). Semiarid soils in general, and the grasslands of southeastern Utah specifically, are characterized by relatively low nutrient availability and low soil organic matter (SOM) content (Schlesinger et al. 1996, Reynolds et al. 1999). In addition, the rock-derived nutrients in desert soils are concentrated in the soil fine fraction that is most prone to erosion by wind and water (Caravaca et al. 1999). The combination of a protective cover of BSCs and the proportionally high nutrient content in the fine soil fraction most prone to wind erosion makes desert nutrient cycling particularly susceptible to surface disturbance and subsequent loss of soil resources.

Unlike mesic ecosystems, and particularly grasslands, which often show higher productivity and organic matter content in grazed vs. ungrazed ecosystems (e.g., Johnson and Matchett 2001), grazing in arid ecosystems can lead to lower productivity and soil nutrient content (Turner 1998, Sparrow et al. 2003). Arid land responses to grazing are closely linked to climate because periodic drought may cause rapid changes in vegetation composition and productivity (Illius and

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TABLE 1. Characteristics of site soil and vegetation in southeastern Utah, USA.

Site	Geologic substrate	Soil type	Vegetation cover
Never grazed	Cedar Mesa Sandstone	Coarse-loamy, mixed, superactive, mesic Ustic Haplocambids	*Stipa comata* and *Stipa hymenoides*/*Hillaria jamesii* with scattered shrubs including *Ephedra viridis* and *Atriplex canescens*
Historically grazed 1	Cedar Mesa Sandstone	Coarse-loamy, mixed, superactive, mesic Ustic Haplocambids	*Stipa comata* and *Stipa hymenoides*/*Hilaria jamesii* with scattered shrubs including *Coleogyne ramosissima, Ephedra viridis,* and *Atriplex canescens*
Historically grazed 2	Cedar Mesa Sandstone	Mixed, mesic Typic Torripsamments	*Stipa comata* and *Stipa hymenoides*/*Hilaria jamesii* with scattered shrubs including *Coleogyne ramosissima, Ephedra viridis,* and *Atriplex canescens*

Notes: The three sites have similar vegetation types, but the density of shrub cover is somewhat higher on the HG2 site relative to the NG or HG1 sites. Nomenclature follows Welsh et al. (1993).

O'Connor 1999, Briske et al. 2003). Another reason to expect greater vegetation sensitivity to grazing in arid lands is that, in contrast to many mesic grasslands that supported native grazers prior to human settlement, southeastern Utah ecosystems experienced very light grazing prior to human settlement (Mack and Thompson 1982). Finally, the soils of arid lands may be more impacted by disturbance than are soils in mesic environments because of the loss of BSCs during grazing (Anderson et al. 1982, Memmott et al. 1998, Hiernaux et al. 1999, Harris and Asner 2003). If grazing leads to disturbance of BSCs, regeneration typically requires decades for the initial colonization and hundreds of years for a crust lichen community to form (Anderson et al. 1982, Belnap and Warren 2002). During this long regeneration period, wind erosion of exposed soils could increase (e.g., Belnap and Gillette 1998), although the long term effects of soil loss are not well documented.

In this paper, we examine the hypothesis that grazing leads to a destabilization of soil surfaces and leads to subsequent losses of soil nutrients to wind and water erosion. Specifically, we show that grazing has decadal-scale implications for desert soil fertility by examining sites where grazing ended about 30 years ago. Undisturbed sites in desert environments can be difficult to find as most of the Canyonlands National Park region of the Colorado Plateau was once grazed. It is thus difficult to assess the predisturbance conditions of areas to provide a benchmark for studies of grazing impacts. In this study, we examine a site never grazed by domestic livestock and two nearby historically grazed sites, to evaluate the impacts of grazing on desert biogeochemical cycles and the resilience of these systems to livestock grazing.

METHODS

Site description and field sampling

Our research sites for this study are located ~100 km south of Moab, Utah, USA in the Needles district of Canyonlands National Park (CNP). The soils on all sites are derived from the weathering of sandstones and are classified as Begay fine sandy loams (never grazed [NG] and historically grazed 1 [HG1]) and a Sheppard fine sand (historically grazed 2 [HG2]) (U.S. Department of Agriculture Soil Conservation Service 1991). Additional soil details are available in Table 1. The CNP (~1500 m elevation) environment is a cold semiarid ecosystem (mean annual precipitation of 207 mm, mean annual maximum temperature of 20°C and minimum temperature of 3.3°C [data from Western Regional Climate Center, *available online*]).[5] Canyonlands National Park was created in 1964 and the grazed sites measured here were taken out of use in 1974. There are very few sites in this region that have never been grazed by livestock and even fewer for which it is possible to pair sites in similar geologic substrates and geomorphic settings. In this study, we present data from sites formed from the Cedar Mesa Sandstone of the Cutler Group. Soils derived from this formation are sandy, and are high in carbonate and Ca content but low in P, Mo, Mn, and other biologically important trace elements.

Large numbers of cattle were introduced to this portion of southeastern Utah in the 1880s (Hindley et al. 2000), and these sites were used by cattle until the cessation of grazing in the mid-1970s. Grazing use in these low-elevation areas typically occurred during winter and spring, although some year-round use may have occurred. It is problematic to estimate the intensity of the historical grazing regimes in this area because existing records do not describe specific stocking rates that occurred before 1970. However, prior to grazing retirement, the historically grazed sites were managed within the same grazing operation, and likely experienced similar grazing regimes. Current stocking rates in surrounding public lands are about four to five animal unit months per 100 ha (M. Miller, *personal communication*), and prior to the 1940s probably exceeded current stocking rates by 20% or more (Hindley et al. 2000).

The never-grazed (NG) site in Virginia Park is surrounded by a high rock wall, and lacks water to support livestock (Kleiner and Harper 1972, 1977, Belnap and

[5] ⟨http://www.wrcc.dri.edu/summary/climsmut.html⟩

Phillips 2001). The site is a large (>200 ha) native grassland with two dominant grass communities: one dominated by *Stipa comata* and *Stipa hymenoides* and the other by *Hilaria jamesii*. *Hilaria* communities within Virginia Park have recently been invaded by *Bromus tectorum*. Our sample areas were located in the uninvaded *Stipa* community.

The historically grazed sites HG1 and HG2, also developed from the Cedar Mesa Sandstone, are located at the same elevation ~5–10 km away from the NG site and have similar vegetation-species composition but a higher amount of *Bromus tectorum*, a lower density of grass species, and somewhat higher densities of shrub species such as blackbrush (*Coleogyne ramosissima*). The three sites are nearly identical with respect to bedrock sources of soils. The sites, moreover, are fundamentally similar in geomorphic setting. The surficial deposits at HG2 have experienced more landform modification (incipient formation of coppice dunes, in places) by wind. HG2 is located in a more open, wind-exposed setting than either the NG or HG1 site and may experience more input of material from local dust sources. However, at all sites, the chemical and textural properties of deeper (50 cm soil depth) soil samples are similar. From this observation and the similarities in geomorphology (position relative to headwalls and slope angles), we infer that the uppermost sediments at each site were also closely similar before disturbance.

On the ungrazed and historically grazed sites we stratified our sampling across shallow hillslopes that grade away from sandstone headwalls. In each site, we established seven to nine plots that included proportional representation of hilltop and midslope positions (geomorphically paired for the three sites). We sampled the top 10 cm of soils at each plot, following removal of the BSC if present, as well in increments to depths to 50 cm in at least three positions on each transect. Soil respiration measurements were made within a meter of these sampling sites in areas that did not contain BSCs with lichen or moss cover.

We sampled rock headwalls above the sites in order to determine the background chemistry of source rock and took samples of rock debris (physically weathered material) from underneath rock slabs to examine the partitioning of rock nutrients into different size classes. We screened this material for dust inputs using magnetic properties (described in *Magnetic properties, texture, and bulk density*). Briefly, a sample of physically weathered rock free of dust has magnetic properties identical to bedrock values. Only samples of physically weathered material that were magnetically identical to bedrock were used for the analysis of physically weathered headwall chemistry. In this way, we were able to compare the chemical properties of unweathered vs. physically weathered rock. To compare the potential inputs of nutrients in dust vs. weathered rock, we took samples from exposed potholes (small depressions on a sandstone surface) where dust tends to accumulate (Reynolds et al. 2001).

Magnetic properties, texture, and bulk density

Changes in soil isothermal remanent magnetization (IRM) can be used as an indicator of soil stability in settings where local bedrock lacks strongly magnetic minerals and where strongly magnetic iron oxides do not form in significant amounts by pedogenesis, such as in these near-surface arid environments. In these sites, stable soil surfaces tend to accumulate fine-grained (silt-to-clay size fraction) mineral dust over decades to centuries of deposition. Previous work in this area (Reynolds et al. 2001) revealed elevated IRM values (typically $>1 \times 10^{-3}$ $A \cdot m^2 \cdot kg^{-1}$, where A is amperes) for undisturbed sandy soil having ~20% silt. The much higher IRM of the soil relative to bedrock is caused mainly by the presence of silt-size magnetite and related titaniferous iron oxide minerals that were formed initially in igneous rocks as confirmed by reflected-light petrographic examination of magnetic grains separated from the soil (Reynolds et al. 2001).

Magnetic property measurements, on dried bulk sediment packed into 3.2-cm^3 plastic cubes and normalized for sample mass, included IRM acquired at 0.3 Tesla (T) and magnetic susceptibility (MS). Both methods provide primarily a measure of magnetite content, but IRM is preferable in this study because it is sensitive to detrital magnetite grains in mineral dust that are large enough hold remanence, whereas MS responds also to ultrafine, possibly pedogenic, iron oxide as well as to iron-bearing silicate minerals (see Thompson and Oldfield 1986). We used MS to test for the presence of infiltrated dust into weathered rock. IRM was measured using an Agico JR-5A spinner magnetometer (AGICO, Prague, Czech Republic) and MS using a Sapphire II susceptometer (Sapphire Instruments, Ruthven, Ontario, Canada).

Particle size was determined as volume percentage using a laser-light scattering method capable of measuring particles between 0.03 and 2000 μm. Organic matter was removed from sediment using a 30% solution of hydrogen peroxide and magnesium chloride. Carbonate was removed using a 15% hydrochloric acid solution to eliminate any pedogenic carbonate in the soil. Such a treatment also removes from the sediment any eolian carbonate dust and detrital calcite, which might be locally derived from bedrock. We measured bulk density using a Soil Moisture Equipment (Santa Barbara, California, USA) model 0200 soil core sampler.

Carbon and nutrient measurements

Soil C and N content was measured with a Leco high-temperature combustion instrument (Leco, St. George, Michigan, USA) at the Natural Resource Ecology Laboratory at Colorado State University and soil carbonate was measured using a Chittick apparatus at the U.S.

TABLE 2. Isothermal remanent magnetism (IRM) and soil textural attributes of the sites.

Site	IRM (1×10^{-3} $A \cdot m^{-2} \cdot kg^{-1}$)	Sand content (%)	Silt content (%)	Clay content (%)
Never grazed	1.45a (0.22)	71.6a (2.1)	22.1a (2.2)	6.3a (0.3)
Historically grazed 1	0.24b (0.03)	81.2b (1.7)	13.8b (1.3)	4.9b (0.5)
Historically grazed 2	0.86c (0.06)	83.9b (4.3)	12.8b (3.9)	3.2b (0.5)

Notes: IRM values are for surface soils on the never-grazed and historically grazed sites in CNP. IRM is a proxy for the accumulation of far-traveled eolian material in surface soils such that high IRM values indicate the accumulation of eolian dust on stable soil surfaces through time. All values are given for the top 10 cm of soil. Standard errors for site means are given in parentheses. Significant differences between sites were determined from planned comparisons between sites following MANOVA and are shown by different letters within a column.

Geological Survey in Denver (Machette 1986). Carbonate was measured because it is a common component of bedrock in southeastern Utah. Total K, Na, Mg, Ca, P, and Mn concentrations in rocks and soils were determined at the USGS laboratories in Denver. All samples were ground to -100 mesh (<150 μm) and 0.2-g aliquots were dissolved using a four-acid (HF, HCl, HNO_3, $HClO_4$) total digestion procedure. Elemental concentrations were determined using inductively coupled plasma atomic emission spectroscopy (Briggs 1996). We measured soil CO_2 fluxes in May and again in July of 2002 using a LiCor 6400 (LiCor, Lincoln, Nebraska, USA) fitted with a 6400-09 soil respiration chamber and previously installed PVC rings at field sites.

We measured chloroform-labile C on soils two weeks prior to the May CO_2 sampling using the chloroform fumigation direct extraction (CFDE) technique (Beck al. 1997, Brookes et al. 1985). This technique involved extraction of nonfumigated soils with 0.5 mol/L K_2SO_4 followed by fumigation of paired subsamples with ethanol-free chloroform for 72 h followed by K_2SO_4 extraction. The total C in nonfumigated and fumigated samples was determined by high-temperature oxidation of soluble C to CO_2 followed by detection with an infrared gas analyzer (TOC V_{cpn}, Shimadzu Instruments, Columbia, Maryland, USA). Chloroform-labile C content is reported without conversion to microbial biomass.

Statistical tests

We examined site differences for this study using multivariate multiway, between-groups analysis of variance (MANOVA). Variables used in the analysis included surface soil C, N, and carbonate contents; chloroform-labile C content; percent sand, silt, and clay; IRM; and Ca, K, Mg, Na, P, and Mn concentrations. The design included transect (NG, HG1, HG2) and hillslope (upper-slope, mid-slope and toe-slope) factors. Following the MANOVA, we carried out planned comparisons of site differences (NG vs. HG1 vs. HG2) if significant differences were found. These comparisons included the effects of site across the entire multivariate model as well as between subject effects to examine differences in individual dependent variables across sites. Significance was determined at the $P < 0.05$ level for all tests. All statistical tests were performed using the Statistica software package (Statsoft, Tulsa, Oklahoma, USA).

RESULTS

Site differences

Site differences were an important control over surface soil elemental composition (Wilks' lambda = 0.000, $F_{30,8} = 16.88$, $P < 0.001$), whereas transect position was not (Wilks' lambda = 0.011, $F_{30,8} = 2.252$, $P = 0.116$). Planned comparisons of contrast vectors for the three sites indicate differences between NG and HG1 (Wilks' lambda = 0.008, $F_{15,4} = 31.833$, $P = 0.002$), NG and HG2 (Wilks' lambda = 0.030, $F_{15,4} = 8.520$, $P = 0.026$) and between HG1 and HG2 (Wilks' lambda = 0.012, $F_{15,4} = 20.516$, $P = 0.005$). If the planned comparisons are limited to soil variables that are more strongly controlled by physical (sand content, IRM, Ca, K, Mg, Na, total P, and Mn) rather than biological processes (soil C, N, CO_2 flux, chloroform-labile C) then HG1 and HG2 are no longer significantly different from one another (Wilks' lambda = 0.844, $F_{15,4} = 3.316$, $P = 0.085$), whereas NG remains different from both HG1 (Wilks' lambda = 0.314, $F_{15,4} = 39.253$, $P < 0.001$) and HG2 (Wilks' lambda = 0.604, $F_{15,4} = 11.784$, $P = 0.002$).

Soil texture

We examined the relative difference in surface (0–10 cm and 10–30 cm) and deep soil (30–50 cm) texture for the three sites. For surface soils, the historically grazed sites tended to be higher in sand content relative to the NG site (Table 2). By normalizing to the deep-soil sand content, we can determine whether surface disturbance (which would likely have larger impacts on surface vs. deep soils) has led to a loss of the soil fine fraction (increase in the sand fraction) relative to the deep soils at each site. The surface 0–10 cm layers of historically grazed sites were 8–20% enriched in sand content relative to the 30–50 cm layer, compared to a 4% depletion in sand (enrichment in fines) relative to deeper soils on the never-grazed site. Values from the 10–30 cm layers show similar patterns among sites

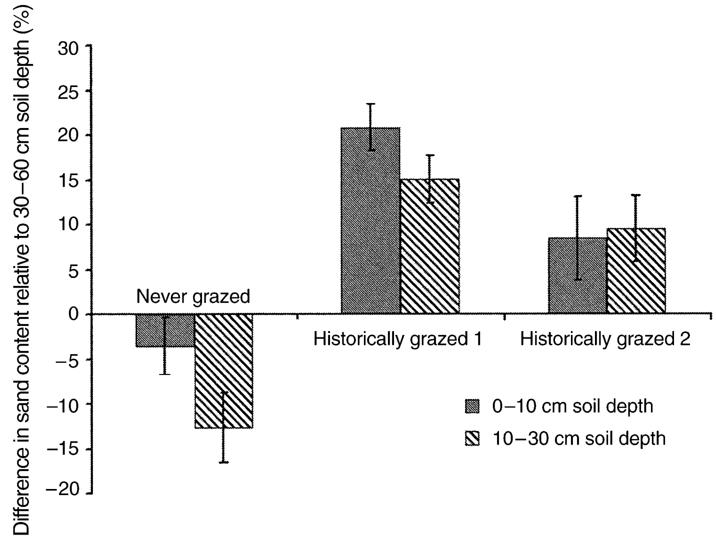

FIG. 1. Relative differences (mean ± 1 SE) between subsurface and surface sand contents. Bars indicate enrichment (positive) or depletion (negative) in soil sand content relative to the 30–50 cm soil depths for the 0–10 and 10–30 cm soil depths on the three sites. Enrichment indicates that surface soils have greater sand content than subsurface soils, implying depletion of the soil fine (silt + clay) fraction.

(Fig. 1). Surface bulk density was similar for the three sites and averaged 1.4 g/cm^3.

Rock and soil nutrient and magnetic properties

Rock-derived nutrients are concentrated in the silt and clay fractions of soils in the Canyonlands area (Fig. 2). In the freshly deposited rock fragments from the Cedar Mesa Sandstone described here, the <63-μm size fraction contains the highest concentrations of potentially available plant-essential nutrients, with means (±1 SE) of 79% ± 7% of Ca, 67% ± 4% of K, 73% ± 7% of Mg, 79% ± 6% of Mn, and 79% ± 2% of P are located in the <63-μm size fraction. The sedimentary bedrock that underlies the study area contains virtually no magnetite and has very low IRM values (typically $<1 \times 10^{-4}$ A·m^2·kg^{-1}). Preliminary evidence suggests that the IRM of incoming dust is $\sim 1.1 \times 10^{-2}$ A·m^2·kg^{-1}.

The IRM of surface soils was an average of 68% lower in the historically grazed sites than the never-grazed site (Table 2). Carbonate was lower on both HG sites relative to the NG site (Table 3). For rock-derived nutrients, the historically grazed sites were higher in Ca, variable in K, and lower in Mg, Na, P, and Mn relative to the NG site (Table 4).

Carbon, N, chloroform-labile C, and soil respiration tended to be lower on the HG sites relative to the NG site though soil respiration was not significantly so (Table 3). Chloroform-labile C was an average of 46% lower in HG1 and HG2 relative to the NG site (Table 3), paralleling at 59% reduction in soil carbon (Table 3). Nitrogen content was lower on both historically grazed sites with only 7.2–28.2 g N/m^2 in the top 10 cm on average compared to two to five times higher concentrations in the never-grazed site (Table 3). Soil K was variable across the three sites ranging from 1.7 and 2.5 kg K/m^2 for the three sites (Table 4). Soil Mg was between 0.45 and 0.57 kg/m^2 on the historically grazed sites and averaged 26% lower than on the never-grazed site relative to HG1 and HG2 (Table 4). P, Mn, and Na were lower in both HG1 and HG2 compared to NG (Table 4) but Ca was higher (Table 4).

DISCUSSION

The extensive use of both public and private lands of the southwestern United States for livestock grazing

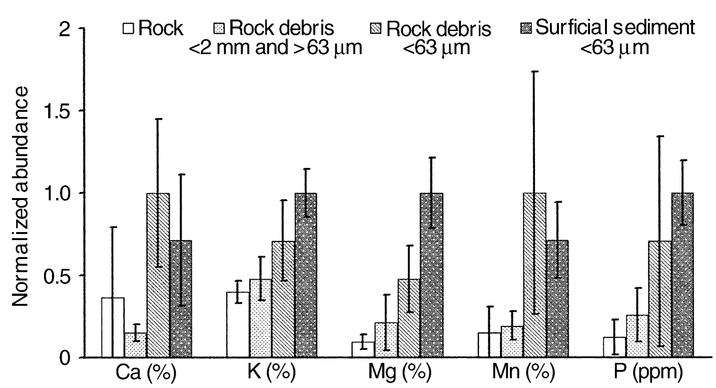

FIG. 2. Relative element content (mean ± 1 SE) of material from the Cedar Mesa Sandstone in southeast Utah. All values are normalized to the material with the highest element content. Rock indicates bulk rock chemistry, the two rock debris values provide a measure of the elemental content of rock debris in areas unexposed to dust deposition, and the surficial sediment shows the relative elemental content of a depositional environment where atmospheric dust is accumulating.

TABLE 3. Soil C and N content, soil respiration, and chloroform-labile C content (CLC).

Site	Soil C (g/m^2)	Soil N (g/m^2)	Soil C:N	Soil CO_3 (g/m^2)	CLC (mg C/g soil)	Soil respiration (g/m^2)
Never grazed	501.8a (65.1)	47.2a (8.1)	11.8 (1.4)	4.55a (1.03)	0.10a (0.008)	0.33a (0.04)
Historically grazed 1	296.4b (36.4)	28.2b (5.5)	11.4 (1.1)	6.73b (0.14)	0.04b (0.01)	0.27a (0.05)
Historically grazed 2	153.2b (26.2)	7.2c (1.7)	23.3 (4.3)	8.52b (0.87)	0.06b (0.004)	0.20a (0.05)

Notes: Soil C and N content and C:N ratios are shown for the top 10 cm. Soil respiration measurements are averages from measurements between April and July, and chloroform-labile C (CLC) measurements are from April. Standard errors for site means are given in parentheses. Significant differences between sites were determined from planned comparisons between sites following MANOVA and are shown by different letters within a column. No contrasts are provided for soil C:N ratio because it was not a variable used in statistical tests of site differences.

through the 20th century has resulted in numerous (and ongoing) changes in grassland environments, including soil loss (Anderson et al. 1982, Belnap and Gilette 1998) and alteration of carbon and nitrogen cycling (e.g., Harris and Asner 2003). In southeastern Utah, and much of the southwestern United States, native grasslands once covered large areas but were converted to grazing lands following settlement (Cottam 1948) and are becoming increasingly rare due to shrub and tree encroachment (Archer et al. 1995, Belsky 1996) and nonnative grass invasion (Evans et al. 2001, Booth et al. 2003). Vegetation from packrat middens in Capital Reef National Park shows that highly palatable plant species were abundant prior to the introduction of sheep and cattle grazing, and this finding suggests that 20th century grazing has led to increases in shrub species in southcentral Utah (Cole et al. 1997). Work in southern Utah at Grand Staircase-Escalante National Monument illustrates the potential for vegetation and soil changes associated with grazing, including declines in BSC cover and increased woody biomass (Harris and Asner 2003). The underlying mechanisms that drive these changes in ecosystem properties following grazing remain uncertain but soil stability appears to be an important factor influencing both the retention of rock derived nutrients, and the dynamics of soil organic matter in arid land soils.

Wind erosion and changes in soil nutrients

The Colorado Plateau experiences a constant input of far-traveled dust and probably has for most of the Holocene (Reheis et al. 2002). This dust accumulates in soils in the region, and the presence of the magnetite in upland soil but not in local bedrock can be used to examine eolian dust accumulation in soils (Reynolds et al. 2001). When soils are disturbed, erosion preferentially removes fine-grained material relative to coarse-grained (sand) material (Pye 1987) and work by Breshears et al. (2003) suggests that the majority of erosion in the semiarid southwestern United States occurs via wind rather than water. In this study, the depletion of magnetite associated with the soil fine fraction in historically grazed sites serves as a useful proxy for the erosion of this previously stabilized eolian material. Although both rock weathering and dust deposition can contribute to the nutrient content of soils in the CNP area, the relevant points here are that the fine fraction of these soils is the dominant reservoir for rock-derived nutrients in this region and that loss of the soil fine fraction due to erosion following disturbance appears to be an important mechanism leading to nutrient depletion in disturbed sites. Although we cannot conclusively differentiate between eolian and downslope water-driven erosion of soil fine particles in this study, landforms in this region show geomorphic evidence of construction and modification by eolian processes. In addition, modern observations of wind erosion and dust emission from nearby grazed sites demonstrate the potential for increased wind erosion in grazed vs. ungrazed environments (Reynolds et al. 2003).

Loss of soil C, N, and microbial biomass

Changes in soil C, N, and other biological properties may be influenced by soil erosion but are likely also related to vegetation and soil crust cover changes with

TABLE 4. Soil nutrient content for the top 10 cm for the NG, HG1, and HG2 sites in CNP.

Site	Soil Ca (kg/m^2)	Soil K (kg/m^2)	Soil Mg (kg/m^2)	Soil Na (g/m^2)	Soil P (g/m^2)	Soil Mn (g/m^2)
Never grazed	2.05a (0.42)	2.50a (0.1)	0.66a (0.02)	341.33a (39.73)	56.88a (9.28)	42.00a (4.09)
Historically grazed 1	2.82a,b (0.04)	1.67b (0.14)	0.45b (0.07)	211.20b (24.47)	47.04a,b (6.74)	24.25b (3.35)
Historically grazed 2	3.17b (0.42)	2.4a (0.1)	0.57c (0.01)	151.12c (9.28)	37.33b (2.66)	18.47b (0.89)

Notes: Standard errors for site means are given in parentheses. Significant differences between sites were determined from planned comparisons between sites following MANOVA and are shown by different letters within a column.

rangeland use. Crust cover on the historically grazed sites is characterized by spotty distributions of cyanobacterium, *Microcoleus vaginatus*, with little moss or lichenized crust development despite 30 years without livestock. While some regeneration of the lichen/moss component of the crusts is now occurring, this process takes several hundred years in these environments (Belnap and Eldridge 2001). Biological soil crusts dominated by *Microcoleus* are estimated to fix less than 1 kg $N\cdot ha^{-1}\cdot yr^{-1}$. In contrast, lichen-dominated crusts found in ungrazed areas such as Virginia Park are estimated to fix 6–9 kg $N\cdot ha^{-1}\cdot yr^{-1}$ (Belnap 2001). Loss of crusts (and crust N fixation) due to grazing probably reduces N input and may be partially responsible for the changes in soil N content on the historically grazed sites. Reductions in soil N content were also reported by Evans and Belnap (1999) when they compared soils in Virginia Park with a nearby area that had been intermittently grazed in winter.

Reductions in soil carbon content, CO_2 flux, and chloroform-labile C all point to diminished C input and/or increased microbial decomposition rates in historically grazed vs. never-grazed ecosystems. Multiple factors probably drive these changes in organic matter cycling, including loss of BSC cover and increased patchiness of vegetation in disturbed arid ecosystems (Schlesinger et al. 1996). Because decomposition rates are relatively high in desert environments (e.g., Parker et al. 1984), organic matter in previously undisturbed soils can be rapidly decomposed if grazing leads to increased vegetation patchiness and decreased inputs of carbon. These findings contrast with studies in more mesic environments where nutrient cycling and organic matter content often increase with grazing (e.g., Johnson and Matchett 2001). Grazing appears to affect arid systems differently than mesic environments with a history of grazing by large native ungulates and where grazing appears to play an important role in the maintenance of productivity and nutrient cycling (Seastedt and Knapp 1993). The effects of grazing on arid-land productivity and nutrient cycling are significantly more variable than in mesic environments. In arid lands, grazing may lead to reduced productivity, vegetation change, and other forms of environmental degradation particularly when accompanied by drought events (Belsky 1996, Illius and O'Connor 1999, Briske et al. 2003, Harris and Asner 2003).

The interaction of erosion and climate

Dryland ecosystem productivity is sensitive to relatively small variations in moisture availability, and this sensitivity could be exacerbated when nutrients are lost to erosion (e.g., Westoby et al. 1989). Over the past 100 years, Utah has experienced at least four major droughts including one that was particularly severe during 2001 and 2002 (Paulson et al. 1991, National Weather Service's Climate Prediction Center web site).[6] When combined with higher rates of wind erosion of actively grazed sites compared to ungrazed or historically grazed areas (Reynolds et al. 2003), drought, soil erosion, and nutrient loss may represent an important mechanism influencing rangeland sustainability in drylands. There is clear evidence that climatic variation interacts with soil properties to determine how arid-land ecosystems respond to grazing pressure (e.g., Westoby et al. 1989, Van de Koppel et al. 1997, Briske et al. 2003). In dryland systems where interannual variation in precipitation is large, the relative strength of nutrient vs. water limitation varies from year to year with greater nutrient limitation in wet years and water limitation in dry years (Le Houérau et al. 1988, Van de Vijver 1993, Guevara et al. 2000). In this study, the lower concentrations of soil fine particles and many of the rock-derived nutrients in the historically grazed sites relative to the never-grazed site may reflect a state change in site hydrologic and biogeochemical characteristics. The impacts of processes such as soil erosion can be irreversible on management timescales (Westoby et al. 1989, Van de Koppel et al. 1997) and match expectations from the state and transition paradigm (e.g., Westoby et al. 1989, Illius and O'Conner 1999, Briske et al. 2003) of grazing system dynamics. Further research is needed to examine the long-term impact of nutrient depletion on the productivity and sustainability of these ecosystems.

In the southeastern part of Utah, there are strikingly few examples of never-grazed systems and most ecological studies have been done on sites with some history of disturbance. This study is one of a handful that documents the dynamics of ''never-grazed'' ecosystems (see Harris and Asner [2003] for another example) but because such sites are few, extrapolation to a larger area remains problematic. The large declines in soil nutrient and organic matter content in this study suggest that predisturbance nutrient status may be significantly different from modern observations for erosion-prone areas of the western United States and highlights the need to find additional remnant ecosystems for comparison to grazed and historically grazed sites.

Conclusions

This study indicates that erosion of nutrient-rich soil fine materials results in depletion of rock-derived nutrient content and suggests that historical grazing in southwestern Utah was responsible for a period of soil loss that continues to impact current soil biogeochemical characteristics three decades after grazed ended. Using soil magnetic properties as a tracer of dust accumulation in soil, we show that stable soil surfaces accumulate silt-sized particles and that disturbance leads to the loss of these materials. We also show that the majority of the rock-derived nutrient capital of these sites resides in these fine soil particles that are

[6] ⟨www.cpc.noaa.gov⟩

present from either accumulated atmospheric dust or the physical weathering of surrounding sandstones. Lower contents of soil fine particles in historically grazed sites leads to lower concentrations of many of the rock-derived nutrients. Lower IRM measurements in historically grazed sites suggest that erosion is responsible for the loss of these fine soil particles and associated nutrients. Soil organic matter C and N content and microbial biomass also are lower in the historically-grazed vs. never-grazed site. The causes of organic matter decline may be due to destruction of BSCs and wind erosion of soils but could also be due to long-term changes in vegetation cover/composition. Because of extensive grazing, undisturbed sites are difficult to find and this study is based on a single ungrazed site. For that reason, it is difficult to extrapolate the results of this study to the broader region of southern Utah. However similarities in soil geochemical, geomorphological, and geological characteristics provide strong support for the hypothesis that grazing triggers wind erosion and results in significant nutrient loss in this semi-arid setting. From a land management perspective, these changes illustrate the potential sensitivity of arid land biogeochemical cycling to land use change and highlight the need to improve our understanding of long-term grazing impacts in the arid southwestern United States. This study also illustrates the potential of wind erosion to contribute to loss of soil nutrients and the need to consider wind erosion in monitoring and management decisions.

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