Version of Record: https://www.sciencedirect.com/science/article/pii/S1550742419300648
Manuscript_39a5935899c6ebed4d00ad8faf22cfe7

Grazing-Induced Changes to Biological Soil Crust Cover Mediate Hillslope Erosion in a Long-Term Exclosure Experiment

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ABSTRACT

Dryland ecosystems are particularly vulnerable to erosion generated by livestock grazing. Quantifying this risk across a variety of landscape settings is essential for successful adaptive management, particularly in light of a changing climate. In the Upper Colorado River Basin (UCRB), there are nearly 25,000 km^2 of rangelands with underlying soils derived from Mancos Shale, an erodible and saline geologic parent material. Salinity is a major concern within the Colorado River watershed, much of which is attributed to runoff and leaching from Mancos Shale deposits. In a 60-year paired-watershed experiment in western Colorado, we used silt fences to measure differences in saline hillslope erosion, including both total sediment yield and concentrations of primary saline constituents (Na and Se), in watersheds which were either exposed to grazing or where livestock was excluded. After accounting for the strong effects of soil type, slope, and antecedent precipitation, we found that grazing increased sediment loss by approximately 50% across our 8-year timeseries (0.1 – 1.5 tn ha^{-1}), consistent with levels reported at the watershed scale in early published work from studies at the same location. Eroded sediment Se levels were low and unaffected by grazing history, but Na concentrations were significantly

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reduced on grazed hillslopes, likely due to depletion of surface Na in soils exposed to chronic soil disturbance by livestock. Variable selection and path analysis identified that biological soil crust (BSC) cover, more than any other variable, explained the differences in sediment yields between grazed and ungrazed watersheds, partially through the enhancement of soil aggregate stability. Our results suggest that BSC cover should be granted heightened consideration in rangeland decision support tools (e.g. State and Transition Models) and that measures to reduce surface disturbance from livestock such as altering the timing or intensity of grazing may be effective for reducing downstream impacts.

KEYWORDS

Badger Wash, Ecological States, Paired Watershed Experiment, Salt Desert, Salinity, Upper Colorado River Basin

INTRODUCTION

Livestock grazing is a culturally and economically important activity worldwide, yet the effects of improperly managed grazing systems may have severe repercussions for nature and society at both local and global scales (Asner et al. 2004). The impacts of improper grazing may be particularly severe in dryland ecosystems, where arid conditions exacerbate the effects of grazers on vegetation and soil, leading to heightened levels of erosion by wind and water (Turner 1971, Field et al. 2011, Aubault et al. 2015, Nauman et al. 2018). In the arid and semi-arid western United States, grazing is the historically dominant land use by area, comprising over 2 million km^2, or 50% of the land area (Bigelow and Borchers 2017). In these systems, overgrazing has been associated with persistent shifts in local species composition and vegetative cover (van de Koppel et al. 2002), declines in productivity (Schlesinger et al. 1990, Neff et al. 2005), as well as increased regional atmospheric dust emissions (Neff et al. 2008, Nauman et al. 2018) and reduced water quality (Fleischner 1994, Belsky et al. 1999). Understanding and quantifying the effects of grazing on soil loss, and the conditions under which soil loss is reduced or heightened remains a high-priority management concern, especially with projected increases in aridity under climate change (Ault et al. 2016).

Grazing primarily influences erosion by altering vegetative cover and quality, but also the physical state of the soil surface through trampling (Branson et al. 1981, Duniway et al. 2018). Vegetation and litter serve to anchor and shield soils against erosive energy from wind and water, as well as restrict the overland flow of water and sediment. Reductions in the size and/or type of stabilizing vegetation by grazing can change the patterns of soil loss on a landscape, potentially leading to exponential changes in erosion due to greater connectivity between bare patches (Kefi et al. 2007, Okin et al. 2009). Mechanical disruption of soil surfaces by grazers may further reduce the stability of soil surfaces and also reduce water infiltration via subsurface compaction and breakdown of soil aggregates (Branson et al. 1981, Warren et al. 1986, Duniway et al. 2018), leading to overland water flow and reduced water availability to plants. In dryland ecosystems, the breakdown of biological soil crusts (BSC) by trampling may be particularly damaging, as well-developed crusts protect soil surfaces and in some contexts, slow the overland movement of wind and water via the generation of microtopographic roughness (Belnap and Lange 2003). Despite the importance of dynamic soil surface properties in mediating the erosion-response of drylands to land disturbances (e.g. aggregate stability, soil roughness, biological soil crust cover), there remains significant knowledge gaps which limit the inclusion of soil surface quality measures in commonly applied rangeland erosion models (Nearing et al. 2011). Namely, the varying importance of dynamic soil surface attributes across ecosystems limits our ability to quantitatively account for many soil quality attributes in process-based soil erosion models (Pierson et al. 2002).

The impacts of grazing on erosion are the result of an interplay between dynamic ecosystem properties vulnerable to disturbance by grazers (surface cover, soil surface stability and roughness) and more static system properties unlikely to be directly impacted by grazing on management timescales (climate, static soil properties, topography). The distinction between these static and dynamic properties is often conceptualized as the difference between ecological 'sites', which dictate the range of potential ecological configurations based on relatively permanent physical constraints (Duniway et al. 2010), and ecological 'states', reflecting configurations of transitory ecosystem properties (Bestelmeyer and Brown 2010). Together, ecological sites and states are fundamental to models of ecosystem responses to climate and management such as state and transition models (Bestelmeyer et al. 2004) which are widely applied in rangeland management. Much work has been done identifying the drivers and indicators of transitions between states (Beisner et al. 2003, Briske et al. 2006), however there is increasing interest in quantifying the ecosystem services (or risks) rendered among states (Webb et al. 2014, Williams et al. 2016). Identifying the relative contribution of static vs dynamic properties to management concerns such as erosion rates is important for planning land use activities and mitigating impacts across the landscape.

In the watershed of the Colorado River in the southwestern US, grazing and other land uses have come under scrutiny due to the high levels of sediment loading and salinization in the river (Miller et al. 2017). Currently, the Colorado River exceeds its legally mandated levels of sediment, salinity, and selenium, risking violation of international treaties and posing a hazard for downstream water users (U.S.

Bureau of Reclamation 2017). Areas in the upper reach of the Colorado, termed the Upper Colorado River Basin (UCRB), are known to contribute large quantities of sediment and salinity due the presence of erodible soils derived from evaporate-rich marine deposits, subject to continued land use since the 1880s (Tuttle and Grauch 2009, Leib et al. 2012). While irrigated fields are known to be large point sources of salinity (Tuttle et al. 2014a), the largest, and most common land use in the region is grazing of domestic cattle and sheep (Copeland et al. 2017). Identifying the potential conditions where grazing impacts on river sediment and salinity loading are greatest are critical for developing sustainable management plans that meet national water quality targets (Nauman et al. 2019).

The Badger Wash paired watershed study, located in western Colorado and part of the UCRB, was established in 1953 to better quantify grazing impacts on soil loss (Lusby et al. 1964). Following in the tradition of paired watershed studies (Ziemer and Ryan 2000), the study measured changes in runoff, sediment yield, and vegetation at the watershed scale between recently fenced watersheds and watersheds open to use by cattle and sheep (Lusby 1970). Initial results in the first five years indicated rapid decreases in sediment and runoff from fenced watersheds, with modest differences in vegetation but larger changes to ground cover (Lusby et al. 1964). Recent surveys of the watersheds indicate that these trends in cover and vegetation have persisted, some 60 years later (Duniway et al. 2018). In this study we seek to determine if the trends in sediment yield reported in early work persist to this day, using a timeseries of hillslope-scale sediment loss measurements. We also explore how levels of salinity and selenium in collected sediment vary among the experimental watersheds. Finally, we assess mechanisms by which grazing history influences sediment yields evaluating the influence of hillslope surface variables responsive to grazing, such as vegetation, soil stability, and biological soil crust to variables unaffected by grazing history such as soil texture, slope and weather.

METHODS

Study Location

The Badger Wash paired watershed experiment is located in Western Colorado (39.3397 N latitude, 108.9339W longitude; approximately 1530 m elevation) and within the Colorado Plateau physiographic province (Fig. 1). The climate is that of a semi-arid cool-desert with an average winter minimum temperature of -9.5 °C, summer maximum temperature of 33.6 °C, and mean annual evaporative demand (1406 mm) exceeding mean annual precipitation (239 mm) by nearly six-fold (aridity index = 0.17, Trabucco and Zomer 2009; climate data from approximately the same elevation but 27 km southeast in Fruita, Colorado; 1970-2000 Averages, Western Regional Climate Center, http://www.wrcc.dri.edu). Precipitation is divided approximately evenly between frontal storms in the winter and spring season (November – April) and more intense monsoonal thunderstorms in the summer and fall (May – October), the latter being more likely to induce runoff and erosion.

Badger Wash is situated along a band of rolling hills and structural benches composed of evaporite-rich, fine-textured soils (regionally referred to as the 'dobie' badlands, Tuttle et al. 2014b) predominantly derived from Mancos shale parent material, a Late Cretaceous marine deposit. Mancos Shale is found throughout the Upper Colorado River Basin and is a dominant source of salinity, selenium, and other trace salt contaminants to the Colorado River (Tuttle et al. 2014a). The steep and erosive hillslopes formed by Mancos Shale at Badger Wash are inter-tongued with coarse-grained, non-saline sandstone deposits, which tend to form gently sloping benches. Soils in the area have been classified into four distinct groups (Lusby et al. 1964, Duniway et al. 2018): 1) high-pH, fine-textured shale; 2) less-alkaline, coarse-textured Sandstone; 3) Mixed soils composed of both Shale and Sandstone-derived sediment; and 4) Alluvium, occurring in topographic lows and encompassing a variety of textures and salinity from diverse alluvial source material.

Vegetation in the Badger Wash study area is typical of the salt-desert shrub type (Lusby et al., 1964; Alzerreca-Angelo et al., 1998). In the mixed and sandstone soils, shrubs such as *Atriplex* *confertifolia*, *Atriplex gardneri*, *Gutierrezia sarothrae*, and smaller *Eriogonum* (*E. bicolor* and *E. contortum*) are common. Grasses (including *Leymus salinus* and *Pleuraphis jamesii*) may be the dominant cover or codominant with shrubs. On the shale soils, *Atriplex gardneri* and *Atriplex corrugata* dominate with sparse cover of perennial grasses. 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. All soils have considerable microtopography due to the presence of mosses and lichens and soil heaving resulting from freeze-thaw cycles.

Historically, Badger Wash has been used for seasonal livestock grazing for herds en route between winter and summer ranges, beginning in the 1880's (Lusby 1979). Early grazing rates by sheep and cattle were heavy, although later diminished by the Taylor Grazing Act (1934) and closing of the stock driveway in 1957 due to improved infrastructure available for trucking (Lusby 1979). After 1988, only cattle were allowed to graze on the allotment, which is currently managed by the Bureau of Land Management (BLM). Stocking rates have varied during the experiment (detailed in Duniway et al. 2018), starting near 6500 AUM permitted during the winter months in the 136 km^2 allotment and dwindling to 800 AUM or no animals in times of drought or rest.

In 1953, four pairs of adjacent watersheds with similar edaphic and topographic characteristics were identified for experimental livestock exclosure and control comparison (Fig 1.). Watersheds range in size from 4.9 ha (watershed 4B, Fig. 1) to 40.9 ha (2B), and total relief (highest to lowest points) ranges from 23 m (4B) to 60 meters (2A). In the original study (Lusby et al. 1964), rates of sediment yield and runoff were estimated from water levels and infill in collection ponds constructed at the base of each watershed. In 2004-2005, watersheds were re-surveyed to assess biophysical changes between watershed pairs (Duniway et al. 2018).

Field and laboratory methods

Hillslope soil loss was assessed using silt fence sediment traps. The silt fences consisted of staked nylon fabric sheets (0.6 mm apparent opening size, Geotex 2130, Propex, Chattanooga, Tennessee, USA) suspended across constriction points subtending small hillslopes (Robichaud and Brown 2002). In November 2007, 20 silt fences were installed in watershed three (Fig 1). In November 2012, an additional 63 silt fences were installed across all watersheds bringing the total to 83 fences (Fig. 1). Silt fence material was inspected and repaired as needed every six months and replaced every three to four years. Hillslopes sampled by the silt fences range from 26 to 387 m^2 with average slopes ranging from 4.3 to 46.5% (Fig. S1; available online at [insert URL here]). Silt fence position and associated hillslope boundary were recorded using a high precision GPS in October 2015 (Trimble GeoExplorer 3000, Westminster, CO, USA).

Sediment trapped by the silt fences was excavated and weighed in the field at least once per year from 2008 to 2016 (Fig. S2; available online at [insert URL here]). Subsamples of collected sediments were retained for gravimetric water content determination and field weights were later adjusted to reflect oven-dry weight. Additional subsamples were collected for chemical analysis. Electrical conductivity, pH, and soluble salts were measured on a saturated soil paste solution (Rhoades 1982). Phosphorus was extracted with $NaHCO_3$ (Olsen 1954, Schoenau and Karamanos 1993). Total nitrogen concentration was measured using Kjeldahl analysis (Bremner et al. 1996) and elemental analysis using inductively coupled plasma mass spectrometry (Johnson and Ulrich, 1959; ICP-OES, Thermo Electron Corporation, Franklin, Maryland, USA).

To better understand the interaction of hillslope soil and vegetation conditions on sediment yield, a subset of hillslopes was assessed for vegetation and soil cover, soil aggregate stability, and soil roughness at the end of the study. To assure hillslopes sampled for soil and vegetation attributes spanned a range of erosional settings, hillslopes were selected for sampling based on residuals from a multiple regression model predicting sediment yield data from 2013 and 2014 based on sample year, soil type, grazing history, hillslope area, and average slope. A stratified random sample design was then used to select 34 of the 83 hillslopes based on watershed pair (four levels), grazing history (two levels), soil type (two levels), and residual quantile (three levels: under estimate [lower quartile], medium fit [middle two quartiles], overestimate [upper quartile]). In selected hillslopes, four equally spaced transects were established perpendicular to the hillslope and transect length adjusted such that transects did not extend outside of the small watershed of the associated silt fence. Plant and ground cover data was collected along transects using line point intercept (0.5 m point spacing) and canopy and basal gap (following Herrick et al. 2005). The total transect length sampled in each hillslope ranged from 12 to 74 m. Surface soil aggregate stability was measured along the transects at 18 pre-determined and evenly distributed locations using the field aggregate stability test kit (Herrick et al. 2001). Additionally, given the high surface roughness of some Mancos Shale soils as well as additional roughness provided by strong biological soil crust development, a surface roughness index was measured at 15 to 20 locations per hillslope (using the jewelry chain method; Saleh 1993). Hillslope measurements were conducted in November, with 16 measured in 2015 and 18 measured in 2016. Due to soil disturbance during sampling, silt fences below hillslopes measured in 2015 were not sampled for sediment yield in 2016.

Rainfall data was compiled from an on-site meteorological station. Rainfall intensity was quantified as the average hourly rainfall between sediment sampling events, excluding all hours where no precipitation was recorded.

Data Analysis

Analysis workflow is presented in figure S3 (available online at [insert URL here]). To examine the broad patterns in sediment yields across the timeseries, sediment parameters, including mass per hillslope area, sediment electrical conductivity (EC), concentrations of nutrients and mass yields of elements by hillslope area (concentration x sediment mass) were regressed against soil type, treatment, slope, hillslope area, and accumulated precipitation and precipitation intensity between sampling dates. Linear mixed effects regression models were fit to sediment data using the ‘lmer’ package in R and

Maximum Likelihood ("ML") estimation. A nested random effect of silt fence within watershed pair was included to account for repeated sampling of silt fences through time and spatial adjacency of silt fences within watersheds. An initial model containing all covariates and first-order interactions was compared to reduced models with each factor removed in turn via a likelihood-ratio test using the function `drop`. Factors were sequentially removed from the model, at each stage dropping the least significant term. The model with the lowest Aikake information criterion (AIC) was selected from the list of reduced models for interpretation. Due to the high number of sediment samples with no Se detected, a two-stage approach was taken to model Se concentrations: First, a logistic model was fit for the presence/absence of any detected Se. Second, a Gaussian model was fit for the subset of sediment samples with Se present (ppm Se | ppm Se > 0). Models were parameterized and selected as above. Continuous predictors were centered and scaled to facilitate comparison of effect sizes. Cumulative rainfall, as well as average hourly rain gauge bucket 'tips' during storms were tabulated for intervening periods between silt-fence samplings, using the on-site meteorological station. Response variables were tested for normality with a Shapiro-Wilks test and log transformed as necessary prior to model fitting. A small constant (0.001) was added to each value prior to log transformation to include zero-valued measurements in the analysis. Model coefficients of determination (both marginal for fixed effects and conditional on random effects) were calculated using the R package `MuMIn`, estimating observation-level variance with the delta method (Barton 2018).

Results from the silt fence timeseries were compared to watershed-scale sediment yields tallied by Lusby (1979), which analyzed sediment and runoff yields between 1953 and 1973. Predicted sediment yields were upscaled from silt-fence data using similar linear models as above, except fixed effects for year and watershed were substituted for rainfall predictors and the random effect of watershed pair, respectively. Predictions for hillslopes with 'average' area and slope were made for each watershed, treatment, soil type, and year, then weighted by the proportion of each soil type per watershed reported in Lusby (1979). Sediment volumes reported in Lusby (1979) were converted to mass using a conversion rate of 1.5 g cm^{-2} for wet loam soils (https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/office/ssr10/tr/?cid=nrcs144p2_074844). Recorded sediment yield estimates for watersheds 2 and 4 between 1966 and 1973 were excluded from consideration, since livestock were experimentally excluded from previously grazed watersheds 2A and 4A during this period.

The effects of grazing on sediment yields were further investigated with regression and path analysis, focusing on the dynamic surface properties measured on the subset of plots in fall 2015 and 2016. Path analysis is a special case of Structural Equation Modeling (SEM). SEM is a large-sample procedure and efforts were made to reduce model complexity relative to sample size (n = 34). To account for factors and processes not typically influenced by livestock activity, a separate 'static' model was fit relating sediment yield to precipitation volume, precipitation intensity, slope, aspect, watershed identity, and soil class. This was done using all records except those which received transect measurements in fall 2015 or 2016 (N = 49). Based on initial evaluation of cross-validation scores, we chose to include predictions from a random forest model (Breiman 2001) as a variable in path analysis to approximate expected sediment yields based on static hillslope properties and precipitation alone. To reduce the complexity of path models, potential variables were first screened with backwards model selection as described above, only without including random effects (as watershed effects should be included in random forest predictions). Significant parameters from the lowest-AIC model were considered for path analysis. As with other models yields were log transformed to approximate normality. Selected variables and their posited relationships to yields were modeled with the R package `lavaan` (Rosseel 2012). Variables for roughness and sediment yield were scaled to reduce the magnitude of variances, although this did not affect fit or model coefficients. First, a path model which included grazing as a potential driver of the significant terms from model selection was fit. Second, properties related to biological soil crust (BSC cover, aggregate stability, surface roughness) were modeled with the residuals from a regression of variables in the first path model against sediment, excluding BSC (weight ~ litter + rock cover + 'static' predictions) to isolate effects associated with BSC. Path models were evaluated using a variety of tests including a 'badness of fit' model chi-square test (where p-values greater than 0.05 are preferred), the root-mean squared error of approximation (RMSEA, with low values indicating a better fit), the Bentler Comparative Fit Index (CFI; with higher values indicating improvement of fit over baseline), and PCLOSE, a one sided test of whether RMSEA = 0.05 (with significant values indicating a non-close fit; Kline 2015).

RESULTS

Patterns in sediment yield and composition

Average sediment yields varied widely between years, treatments, and soil types (Fig 2). Mixed model results show that soil-type, rainfall characteristics, slope, and grazing history all are important determinates of sediment yields and/or composition (Table 1). Hillslopes in grazed watersheds were found to have sediment yields approximately 1.5 times that of ungrazed watersheds across years (after accounting for slope, soil-type, and previous rainfall volumes and intensity; Table 1). Soil type had a relatively larger effect on yields, with shale soils losing roughly 2.5 times that of mixed soils. The effect of grazing history did not significantly differ by soil type (lack of significant interaction). Prior rainfall was the strongest predictor of sediment accumulation (one standard deviation increase in rainfall was associated with a 4.5-fold increase in sediment yield; Fig. 3). Mean rainfall intensity and slope were also positively related to sediment, although predicted sediment yields declined proportionally to the square of slope, resulting in an attenuation of yields at very steep slopes (Fig 3). For sediment yield and nearly all other sediment variables, values declined with increasing hillslope area.

The EC of collected sediment, representing concentrations of dissolvable solutes, was significantly greater in shale soils than mixed soils. The EC of sediment from shale soils was reduced when exposed to grazing (grazing x soil type interaction), although the main effect of grazing was not significant. Grazed watershed sediments had reduced concentrations of elemental Na while Se concentration was unaffected by grazing history. Selenium concentration, when detected, was positively associated with shale soils. Patterns in other element concentrations generally followed that of EC, and results for other elements tested are listed in Table S1 (available online at [insert URL here]). Coefficients for modeled masses of individual nutrients found in trapped sediments followed a similar pattern to overall sediment yields, although there was no detected effect of grazing history on selenium mass per hillslope area, and overall R^2 was relatively weak (0.19).

Relationship to historical measurements

Silt fence-based estimates of sediment yield from grazed versus ungrazed watersheds measured in this study followed a similar pattern to sedimentation records reported in Lusby (1979; Fig. 4). The log-response ratio of grazed to ungrazed sediment yield in the early years (1954 to 1965) was associated with time since livestock exclusion and patterns in precipitation (Fig. 4a). During the period of this study with hillslope erosion data in all eight watersheds (2013 – 2016), we only see significant differences between grazed and ungrazed in 2014 (Fig. 4a). Comparing relative yield between paired grazed/ungrazed watersheds, we see the output from grazed watersheds exceeded that of ungrazed in both the current study and in sediment yields from the earlier studies, but the ratio dampened in high-yield conditions, as indicated in the linear trends relative to the 1:1 line (Fig. 4b).

Mechanisms underlying the patterns in erosion

Rock cover, BSCs, litter cover, aggregate stability, as well as predictions from the random forest model built on ‘static’ properties (Fig. S3, available online at [insert URL here]) were identified as important variables related to sediment yield, based on model selection (Table 2). BSC was perhaps the most important variable, being consistently retained across all models (Table 2). Litter and aggregate stability were retained in the most parsimonious model based on AIC, but their modeled effects were not significantly greater than zero and thus they were excluded from the initial path model.

Path analysis of the hypothesized relationship between grazing history and select dynamic system properties are presented in Figure 5a. The model has an acceptable fit (X^2 = 5.349, d.f. = 3, p = 0.148, CFI = 0.938, RMSEA = 0.14, PCLOSE = 0.183) and highlights the indirect impact of grazing on sediment yield via disruption of BSC, as compared to other direct and indirect effects. While rock cover was strongly related to sediment yields (after accounting for other geophysical factors), there was not strong evidence that this variable itself was strongly influenced by grazing history. A closer look at hypothesized effects of BSC on sediment yields is displayed in the path diagram in Figure 5b. The fit for this model was also acceptable (X^2 = 0.687, d.f. = 1, p = 0.407, CFI = 1, RMSEA = < 0.001, PCLOSE = 0.425), and suggests that BSC cover affects sediment (after accounting for other types of cover and geophysical-based predictions) both directly and indirectly through improving aggregate stability, but not via generation of surface roughness, as measured in this study.

DISCUSSION

Persistent reductions in soil loss from protected watersheds

Sixty years after the initiation of the exclosure experiment at Badger Wash, watersheds exposed to grazing continue to have roughly 50% greater sediment loss than those protected from grazing, after accounting for the effects of slope, soil type, and rainfall. Although conducted at a different scale (hillslope vs. whole watershed), the relative differences observed in this study are comparable to that of Lusby et al.(1979), who found relative increases of sediment yield around 63% at the watershed scale from 1953-1973. Heightened sediment yields associated with grazing are frequently reported in literature, although relative yields are often several times or orders of magnitude greater than reported in our study (McCalla et al. 1984, Warren et al. 1986, Weltz et al. 1989, Naeth and Chanasyk 1996, Emmerich and Heitschmidt 2002). In other cases, high variability in sediment yields within treatments statistically mask similar patterns (e.g. Buckhouse and Gifford 1976, Wood and Blackburn 1981).

Lusby (1979) attributed differences in yields to changes in ‘ground cover’ (comprised of plant canopy, plant basal cover, lichens, mosses, and plant litter), rather than changes in vegetation composition. Recent surveys found a similar pattern, with differences in dynamic soil surface properties (aggregate stability, BSC cover, roughness, compaction), much more apparent that vegetative differences between grazed and ungrazed watersheds in the soil types we examined (Duniway et al. 2018). These differences have persisted despite significant reductions in herd numbers in recent years (although exact data is lacking), suggesting that even low-intensity grazing may be sufficient to destabilize some soils. This result is consistent with the effects of repeated trampling of biological soil crusts seen in other studies, as biocrusts are known to improve soil aggregate stability and reduce erosion but take many years to recover after disturbance (see discussion below, Belnap and Eldridge 2001). Differences between grazed and ungrazed watersheds observed by Duniway et al. (2018) are largely due to recovery of biological soil crusts in ungrazed areas, in addition to continued disturbances of soil surfaces in grazed watersheds.

Apart from the effects of grazing, soil type had a strong influence on sediment yields, with soil losses in the steep, finer-textured shale hillslopes exceeding those of more gently sloping 'mixed' soil slopes by a factor of ~2.5. Interestingly, the effects of grazing were not found to vary among soil types. This may be related to lower levels of cattle use on steep shale soils due to rugged topography and lower amounts of palatable vegetation. Lower activity on these soils may be compensated for by relatively higher vulnerability to grazing-related disturbance on these soils. Despite our inability to detect a grazing-by-soils interaction, we did observe greater overall sediment yields associated with shale-derived soils which suggests soils play an important role in determining overall sediment yields across the landscape.

Predictably, precipitation quantity and intensity, slope, and hillslope area were found to be important correlates of sediment yields across watersheds. The reduction of recorded sediment yields with increasing hillslope area is consistent with yield-area scaling relationships found in other studies (Parsons et al. 2006, Moreno-de las Heras et al. 2010). In both our study and that of Lusby (1979), there was considerable inter-annual variability in sediment yields among watersheds, which was reliably linked to precipitation patterns in our study (Fig. 2). In both studies, there was a trend for the relative difference in sediment yield between grazed and ungrazed watersheds to decline at greater absolute sedimentation yields (approaching 1:1; Fig. 4b). We attribute this decreasing signal of grazing history in high-yield scenarios to high erosive rainfall events or seasons overwhelming the differences in protective cover between grazed and ungrazed pastures (Duniway et al. 2018). Although the condition of surface soils and amount of protective cover is an important determinant of site erodibility, extremely intense rain events likely overwhelm the differences in surface characteristics observed in Badger Wash.

Differences in sediment chemistry related to grazing history and texture

Throughout the time series, eroded sediment collected from ungrazed watersheds had significantly greater concentrations of Na (Table 1). It is likely that the lower levels of salinity observed in sediment recovered from grazed watersheds were the result of grazed soils having depleted levels of labile Na ions to begin with, rather than any strong difference in the rate of loss between treatments. Duniway et al. (2018) found that grazed soils had significantly lower levels of Na than soils in ungrazed watersheds, likely reflecting the consequences of repeated soil disturbance by livestock. In Mancos-shale derived soils, capillary rise concentrates solutes such as Na in the soil surface (Whittig et al. 1982, Hillel 1998) often forming crusts. Mechanical disturbance of surface crusts by trampling enhances rates of erosion and liberates accumulated solutes to be exported in runoff (Elliott et al. 2008). Rainfall simulation studies on Mancos soils have found total dissolved solids in runoff to be linearly related to sediment yield (Cadaret et al. 2016b), suggesting that sediment may be a consistent proxy for cation loss in these soils. When accounting for the mass of sediment lost among hillsides in our study, net volumes of Na lost were greater in hillslopes exposed to grazing (Table 1). Recurrent depletion of salts due to grazing influence may also explain the reduced EC associated with grazed shale soils, although EC integrates a wide variety of solutes. Taken together, these results suggest that exposure of long-rested or undisturbed watersheds to livestock or other soil disturbing land uses common in the region (e.g., off-highway vehicles, oil and gas exploration and development; Copeland et al. 2017) may result in larger flushes of Na than expected based on monitoring or modelling conducted in continually disturbed watersheds.

It is also possible that consumptive export of Na through grazing could be partially responsible for lower soil Na values observed in grazed watersheds, and hence lower concentrations of Na in eroded sediment. Many of the dominant halophytes in the region, such as *Atriplex* spp., concentrate salts from the soil in shoots, which may be palatable to livestock and other herbivores (Nemati 1977). In grasslands, consumption by ungulates has been proposed as a dominant pathway of nutrient loss, at least for N (Woodmansee 1978). Alternatively, trampling of salt-laden surface soils in grazed watersheds may effectively dilute salt concentrations at the surface, leading to reduced salt concentrations in grazed soils (but greater net salt export via greater sediment losses).

The levels of Se recovered in sediment were unexpectedly low, with over 84% of samples (459 / 545) having no Se detected. The samples with observed Se concentrations tended to be on the lower range of values found in other rangelands situated on Mancos shale in the Colorado Plateau (e.g. 4 ppm in Tuttle et al. 2014a). Both detection probability for Se and the concentration where present were linked to antecedent precipitation volume and intensity in a similar manner to total sediment, suggesting that observed values may be linked at least in part to the quantity of sediment recovered. As expected, Se concentrations were positively associated with shale-derived soils, with around 60% greater concentration in shale vs. mixed soils. Selenium concentrations in the Mancos shale are highly variable across the landscape, linked to differing patterns of deposition and erosion during pedogenesis (Tuttle et al. 2014b). The lack of any observed influence of grazing history on Se yields, including the effects grazing on of overall sediment yields, suggests that soil context may be significantly more important for potential loadings of water sources, at least in Badger Wash.

Effects of grazing history on erosion via reductions in biological soil crust

The path analysis model examining the indirect effects of grazing history on sediment yields via changes to dynamic system ‘state’ variables (biophysical variables represented in Table 2 and Fig. 5a), suggests that reduction in BSC cover is the dominant mechanism by which grazing increased sediment yields, as opposed to reductions in other types of cover (e.g., burial of rocks by trampling as reported in Turner, 1971). After accounting for other variables, neither shrub, perennial grass, nor litter cover were found to be significantly associated with erosion rates in model selection. This result is consistent with previous watershed-scale associations between grazing history and biophysical measures at Badger Wash, in which soil surface properties not related to vegetation were found to differ the most between grazed and ungrazed watersheds (Turner 1971, Lusby 1979, Duniway et al. 2018). In particular, Duniway et al. (2018) found that variables associated with BSC, including cover of moss, lichen and cyanobacteria, as well as aggregate stability and surface microtopography differed markedly between watersheds with contrasting grazing history.

Further examination of the influence of BSC on sediment yields (after accounting for the effects of rock and other geophysical effects) revealed that BSC cover was linked to reduced soil erosion both indirectly through changes in soil aggregate stability and directly (Fig. 5b), but not through surface roughness as measured in this study. The physical structure of BSC organisms such as lichen thalli and moss biomass are known to directly shield soil surfaces (Belnap 2006) and may explain at least part of the direct influence of BSC on residual sediment reduction. BSC greatly improve the tensile strength and aggregate stability of soil surfaces partially through the growth of filamentous cyanobacteria (Belnap 2003b) . These early-colonizer organisms grow throughout the upper 5 mm of the soil surface and bind soil particles together with gelatinous sheaths derived from exopolysaccarides (Belnap 2003a). In many cases the greater aggregate stability conferred by BSC is associated with improved infiltration (Chamizo et al. 2016, Whitney et al. 2017, Fick et al. 2019), but this largely depends on the successional state of the BSC (Barger et al. 2006, Belnap et al. 2013, Faist et al. 2017). Further, early colonizing cyanobacteria are often not visible in field assessments, making their potential growth and influence much greater than estimates based on cover of more visible BSC components (lichens, mosses, later-colonizing ‘dark’ cyanobacteria; Belnap et al. 2008). The high (but not perfect) correspondence between BSC cover and aggregate stability (Bowker et al. 2008, Carpenter and Chong 2010), as well as the strong negative effect of aggregate stability on sediment yield (Le Bissonnais 2016) is thus expected.

BSC is associated with greater levels of surface roughness related to freeze-thaw cycles in this cold-desert system (Belnap et al. 2003), and it was expected that roughness, in turn, would have a significant association with residual sediment loss. Undulations, pits, and pinnacles in the soil surface may reduce sediment loss by decreasing the energy of overland flow and increasing the residence time of water to pool and infiltrate (Govers et al. 2000). In some studies, surface roughness characteristics of well-developed BSC are associated with improved hydrologic function and reduced runoff and sediment yield (Barger et al. 2006, Belnap et al. 2013 p. 2013, Whitney et al. 2017, Young et al. 2019). However, in a detailed examination of erosion from BSC surfaces as related to surface roughness, Rodríguez-Caballero et al. (2012) found that none of the roughness indices examined related well to recorded sediment yield. Average roughness values observed on both grazed and ungrazed hillslopes in this study were comparable to roughness values of well-developed crusts in the region (Miller et al. 2011), and substantial roughness in Mancos Shale is imparted by freeze-thaw cycles even without BSCs (Turner 1971). Although the ungrazed hillslopes tended to have greater roughness measures than grazed slopes, this added roughness did not have had significant bearing on sediment yields.

Despite the importance of BSC for controlling key ecosystem processes such as erosion and biogeochemical cycling in arid systems, BSC are rarely included as key 'state' variables in rangeland state and transition models. Notable exceptions include several ecological sites on the Colorado Plateau in southeastern Utah, where persistent reductions in well-developed BSC have been cited as indicators of transitions to degraded states (Miller et al. 2011, Duniway et al. 2016). The prominence ascribed to BSC in Colorado Plateau ecosystems is likely due to the large gaps between perennial vegetation in this dryland region, the highly pinnacled stature and diversity of BSCs communities, and the low resilience of BSCs to disturbances regionally. The present research has identified that BSC are important indicators of ecosystem services (soil conservation) in Mancos shale-derived soils and associated plant communities, which are found throughout the intermountain west. These results suggest BSC cover should be included in existing rangeland assessments as well as models of ecosystem dynamics (state-and-transition models).

In the path analysis, BSC cover had a stronger association with sediment yields than the modeled effects of variables unaffected by grazing (soil texture, slope, aspect, antecedent precipitation, Fig. 5a). In part, this reflects the relatively high importance of dynamic 'state' variables in determining erosive outputs, despite the strong influence of more static 'site' factors related to climate and topography. This somewhat contradicts the assessment of Branson and Owen (1970) who found that at the watershed scale, geomorphic variables, rather than cover, had the highest correlations with observed sediment at Badger

Wash. Other studies have identified dynamic state properties, particularly vegetative cover (Branson et al. 1981, Pierson et al. 2007), as important controls on erosion generation. However, these patterns tend to be inconsistent across ecological contexts, with soils and geomorphology having strong controls on these relationships (Pierson et al. 2002) . Part of the relatively modest association between site factors and erosion in the path model may also be due to poor extrapolations using a model based on data excluding hillslopes used in the path model. Variables associated with precipitation were given large weight in the random forest model (Fig. S4; available online at [insert URL here]) which may reflect overfitting to differences in interannual precipitation, rather than more subtle effects related to soils or topography. Predictions using other modeling techniques, including the Rangeland Hydrology and Erosion Model (RHEM; Nearing et al. 2011) produced similarly weak correlations to observed sediment levels (Fig. S5; available online at [insert URL here]), and the global linear model for the sediment timeseries had an estimated R^2 of 0.50 (Table 1), leaving a large portion of unaccounted variance. Prediction of sediment yields may be particularly difficult in this study given the relatively high noise of sediment yield data in general, the overall low vegetative cover of plots in this system (Cadaret et al. 2016a), distortions in hydrologic behavior caused by salinity (Nouwakpo et al. 2018), and small scale of the hillslopes measured.

MANAGEMENT IMPLICATIONS

The results of this study suggest that watersheds open to grazing have increased sediment loss in Mancos shale-derived soils when compared to long-term livestock excluded watersheds. Analysis of hillslope conditions suggests that differences in BSC cover and soil aggregate stability are primarily responsible for the differences in erosion processes between grazed and ungrazed watersheds. Although other aspects of ground cover were found to explain variation in sediment yield, this study suggests BSC cover and aggregate stability are primary rangeland indicators that differ with grazing history and explain important differences in sediment yield. The lower BSC cover and soil stability with grazing is likely due to both continued disturbance of surface soils with livestock use and long-term recovery of BSCs with long-term livestock exclusion (Duniway et al. 2018). These results support the collection of BSC and soil aggregate stability data in rangeland assessments. Further, these variables should also be considered in state and transition model development for Mancos soil ecological sites, and any location where BSCs are naturally abundant.

Changes in the timing and intensity of livestock are effective strategies for limiting negative impacts of grazing on erosion processes in dryland ecosystems. Limiting grazing activity to winter months was found to give plant and soil communities a chance to recover before erosion-inducing summer rains took effect (Lusby 1979). Reduced stocking rates, especially in times of drought are recommended to maintain forage production and protect developing BSCs. Lusby (1970) found that after exclosures were installed, reductions in runoff occurred in roughly 3 years, suggesting that a few years of rest from livestock grazing may have large benefits. Given greater predicted variability in precipitation for the region with climate change, developing adaptive grazing regimes will be important for rangeland resource conservation (Seager et al. 2007, Joyce et al. 2013, Polley et al. 2013).

In this study, soil type was more important than grazing history in predicting sediment yields across the hillslope silt-fence timeseries. This suggests that understanding differences in sediment yields related to landscape-scale patterns in geologic parent materials is an important aspect of livestock and range management in this system, potentially necessitating within-pasture management tools (e.g. modular fencing) to move animals away from the more erosive soils. Spatially-explicit risk mapping for sediment loading may be a useful exercise for prioritizing efforts to reduce sources of sediment and salinity loading (Nauman et al. 2019).

ACKNOWLEDGEMENTS

We would like to thank Tamara Minnick, Richard Grauch, Sue Phillips and the BLM and USGS Water Grand Junction Field Offices, who helped design and initiate the larger Badger Wash study, and the many assistants and technicians who assisted with field work and data collection over the years, including Erika Geiger, Adam Kind, Philip Adams, Matt Van Scoyoc, Ed Grote, Hilda Smith, and many others.

Funding for this work was provided by the Bureau of Land Management and the US Geological Survey Ecosystems Mission Area. Use of trade, product, or firm names is for information purposes only and does not constitute an endorsement by the US Government.

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TABLES

	Sediment yield (kg/m2)	EC	Na yield (mg/m2)	ppm Na	Se yield (mg/m2)	ppm Se > 0	ppm Se $\mid$ ppm Se > 0
Intercept	-2.807 (0.182)*	0.066 (0.152)	3.012 (0.195)*	5.961 (0.043)	-5.835 (0.302)*	-0.155 (0.414)	-15.587 (2.396)*
Soil = Shale	1.063 (0.197)*	0.667 (0.112)*	1.138 (0.213)*		-0.129 (0.272)		0.457 (0.178)*
Grazing	0.41 (0.174)*	0.104 (0.094)	0.42 (0.188)*	-0.15 (0.053)*			
Shale x Grazing		-0.365 (0.14)*					
Rainfall	1.542 (0.079)*	-0.273 (0.027)*	1.773 (0.087)*	0.232 (0.028)*	0.885 (0.109)*	3.341 (0.737)*	0.079 (0.013)*
Rainfall^2						-2.592 (0.637)*	-9.3e-5 (1.8e-5)*
Intensity	0.379 (0.062)*		0.642 (0.069)*	0.21 (0.025)*	0.418 (0.115)*	2.298 (0.497)*	-1.039 (0.426)*
Intensity^2					-0.157 (0.108)	-2.117 (0.471)*	1.155 (0.445)*
Slope	0.343 (0.153)*	0.109 (0.046)*	0.365 (0.167)*	0.073 (0.051)	0.611 (0.398)		
Slope^2	-0.182 (0.097)		-0.152 (0.104)	0.048 (0.028)			
Area	-0.227 (0.098)*	-0.18 (0.069)*	-0.221 (0.107)*		-0.478 (0.193)*	-0.311 (0.147)*	
Area^2		0.044 (0.021)*			0.096 (0.063)		
Shale x Slope					-0.82 (0.401)*		
Grazing x Slope				-0.088 (0.057)			
AIC	2171	517	2209	1029	2323	364	228
R2m	0.47	0.54	0.51	0.21	0.19	0.87	0.39
R2c	0.5	0.73	0.54	0.21	0.2	0.88	0.42

Table 1. Standardized stepwise regression coefficients and standard errors (parentheses) plus goodness of fit estimates for sediment collected in silt fences. Continuous response variables (columns) were log-transformed prior to model fitting. Yields indicate estimates of mass per hillslope area and ppm indicates concentration by mass. EC indicates electrical conductivity, rainfall represents accumulated rainfall prior to sampling, Intensity represents mean rainfall intensity (mm/hr) during storms, Area indicates hillslope area feeding each silt fence. R^2m indicates the estimated marginal coefficient of determination for the fixed effects of the model while R^2c includes the contribution of both fixed and random effects. Asterisks indicate significant effects ($p \leq 0.05$).

Stage	BSC	Static Model	Rock	Litter	Stab-ility	Graz-ing	Gaps	Shrub	Rough-ness	Peren-nial Grass	AIC	BIC
1	0.006*	0.045*	0.02*	0.38	0.112	0.392	0.562	0.581	0.778	0.919	100.62	117.83
2	0.005*	0.035*	0.015*	0.37	0.087	0.333	0.547	0.577	0.769		98.64	114.41
3	0.003*	0.02*	0.012*	0.096	0.083	0.215	0.517	0.614			98.64	113.3
4	0.002*	0.021*	0.012*	0.074	0.09	0.22	0.469				97	110.19
5	0.002*	0.009*	0.012*	0.07	0.117	0.23					95.72	107.44
6	0.004*	0.003*	0.021*	0.087	0.098						95.6*	105.86*
7	< 0.001*	0.002*	0.009*	0.128							101.7	110.86
8	< 0.001*	0.005*	0.005*								102.46	110.09
9	< 0.001*	< 0.001*									109.56	115.66
10	0.002*										120.3	124.88
Select-ed Model Coefs	-13.63	1.12	-3.83	-3.75	-0.38							

Table 2. Model selection results for (log) sediment yield as a function of select hillslope surface variables. Cells represent p-values from a Chi-Square test with the selected term dropped from the model, with p-values < 0.05 highlighted with an asterisk. Asterisks in AIC and BIC columns designate global minima. The static model indicates random forest model predictions using only precipitation, watershed pair, soil type and topographic data (e.g. hillslope, aspect). Gaps indicate percentage of basal gap lengths greater than 100 cm. Coefficients from the most parsimonious model presented in the last row.

FIGURES

Figure 1. Map of experimental watershed area. ‘A’ watersheds are open to livestock. ‘B’ watersheds have had livestock excluded since 1953.

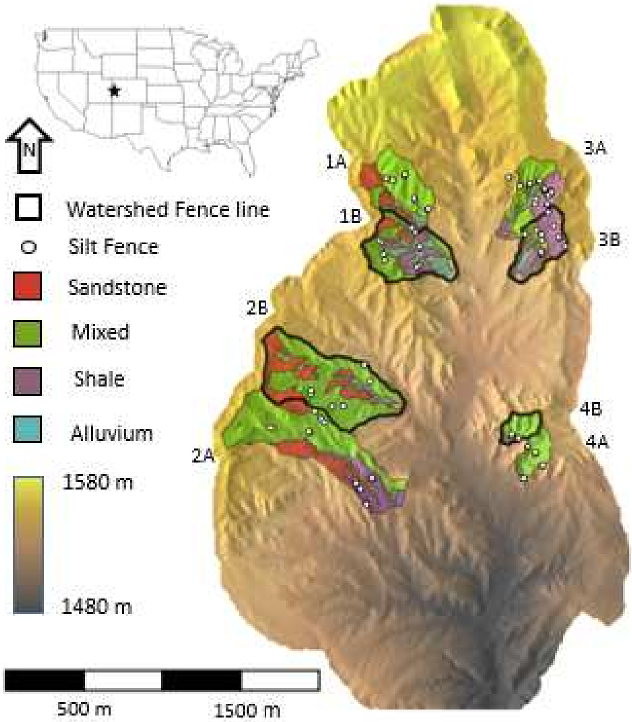
Figure 2. Sediment accumulation by treatment and soil type aggregated to fall sampling date (top panel, note the log scale). Middle panel: accumulated precipitation in periods between sampling. Bottom panel: mean hourly rainfall intensity (mm/hr) by month for duration of the experiment. Full details of sampling schedule are available Fig. S2 (available online at [insert URL here]).

Figure 3. Partial effect plots and 95% confidence intervals for AIC-selected predictors in the hillslope sediment yield time series. Effects calculated holding other predictors at their average values in the R package `effects` (Fox and Weisberg 2018). For discrete predictors different letters indicate a significant difference (alpha = 0.05, R package `emmeans`, Lenth 2018). Distribution of observed predictor values indicated by ticks on x axis.

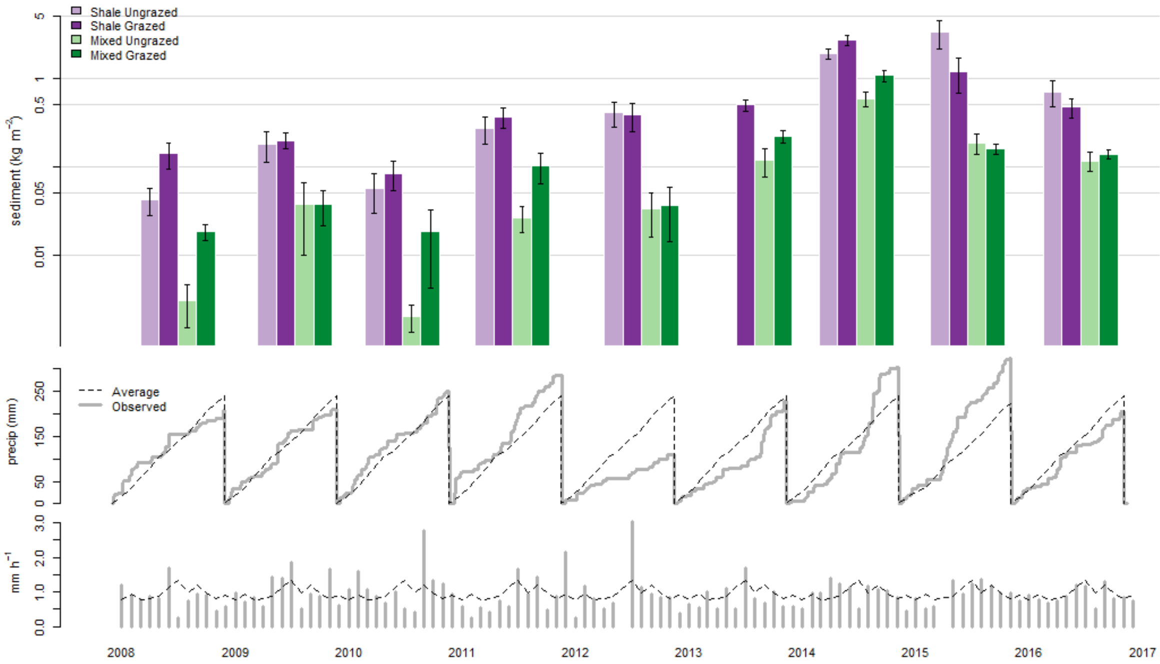
Figure 4. Comparison of Grazed vs. Ungrazed sediment yields for silt fence data and historical watershed-level yields published in Lusby (1979). Watershed-level silt-fence predictions were extrapolated from linear models of yields by year, watershed and soil type, weighted by the proportion of soil types within each watershed. Cases where either ungrazed or grazed watershed yields were zero were omitted from analysis. Data from Lusby (1979) converted from acre-ft mi^{-2} to kg m^{-2}, using a bulk density estimate of 1.336 g m^{-3}, based on a weighted average by soil type across all watersheds.

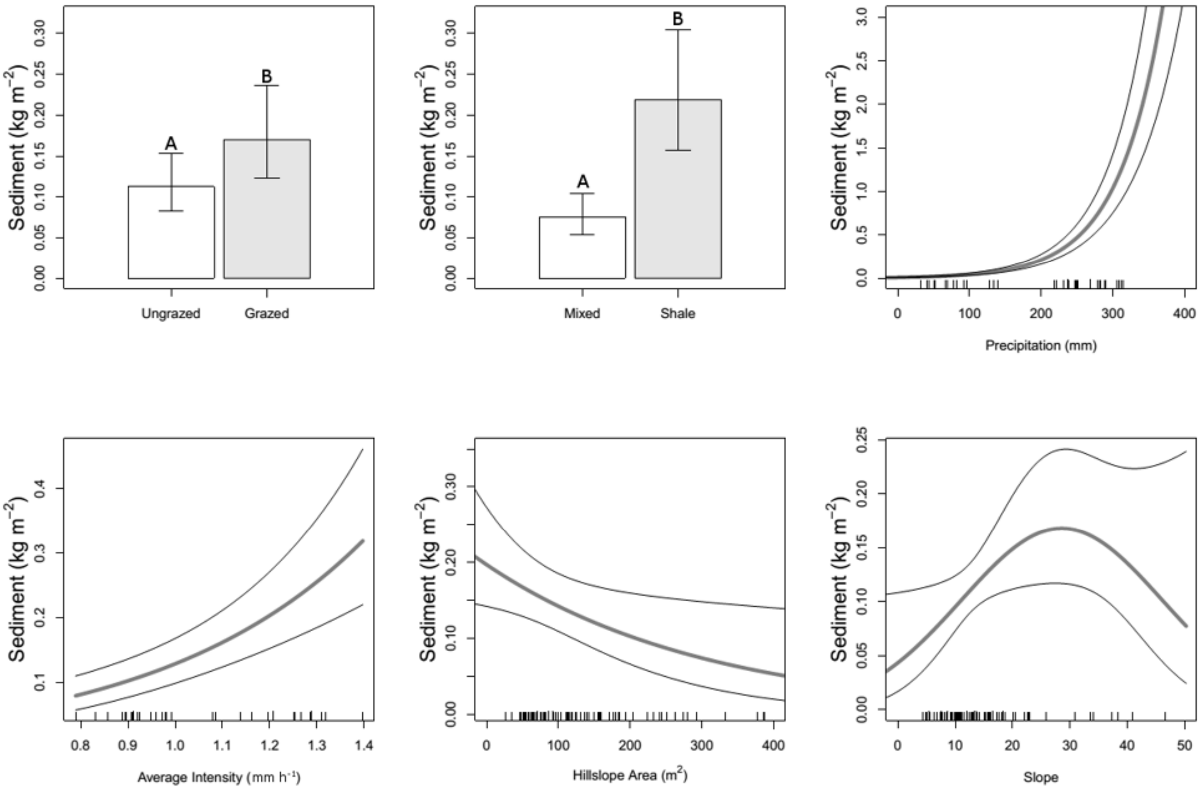
Figure 5. Influence of grazing on sediment yields via mediation of biological soil crust, rock cover, aggregate stability and soil roughness. In panel b, the residual sediment yields (subtracting the effects of geophysical variables and rock cover identified in panel a) are modeled as the dependent variable to isolate effects associated with biological soil crust. Path analysis variables are represented by boxes, hypothesized causal relationships between variables represented by single-headed arrows, and

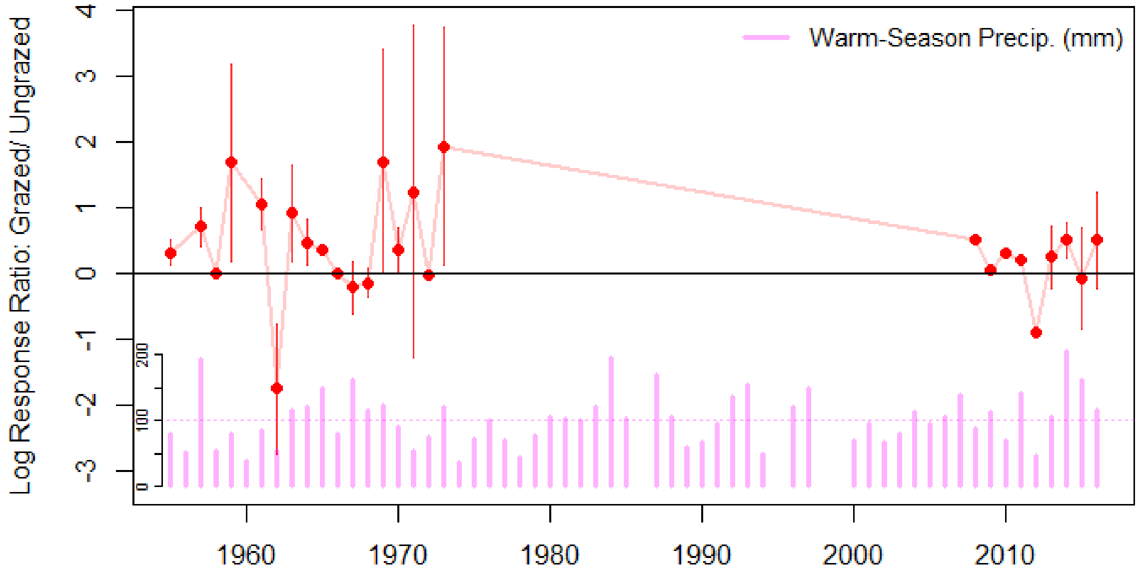
correlations specified by two-headed arrows. Arrow thicknesses are proportional to standardized coefficients which are also printed adjacent. Nonsignificant coefficients represented by dashed arrows. R-squared values for endogenous variables are printed in parentheses within boxes. Model fit statistics printed at lower left.



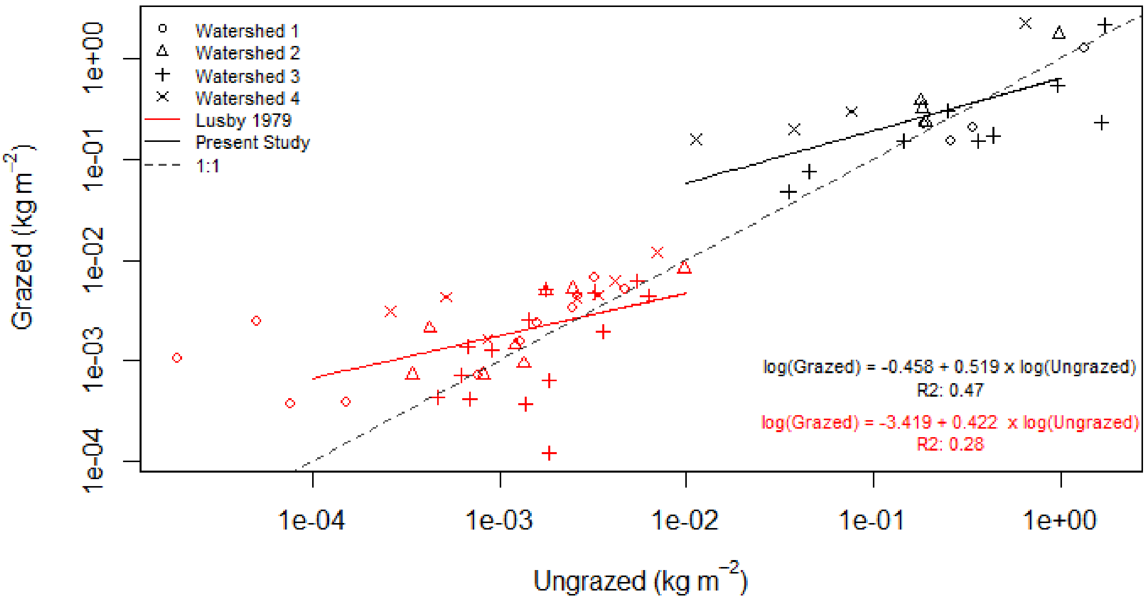
Watershed Fence line
Silt Fence
Sandstone
Mixed
Shale
Alluvium
1580 m
1480 m
500 m
1500 m
N
1A
1B
2A
2B
3A
3B
4A
4B


Shale Ungrazed
Shale Grazed
Mixed Ungrazed
Mixed Grazed
sediment (kg m^{-2})
5
1
0.5
0.05
0.01
Average
Observed
precip (mm)
250
150
50
0
mm h^{-1}
3.0
2.0
1.0
0.0
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017

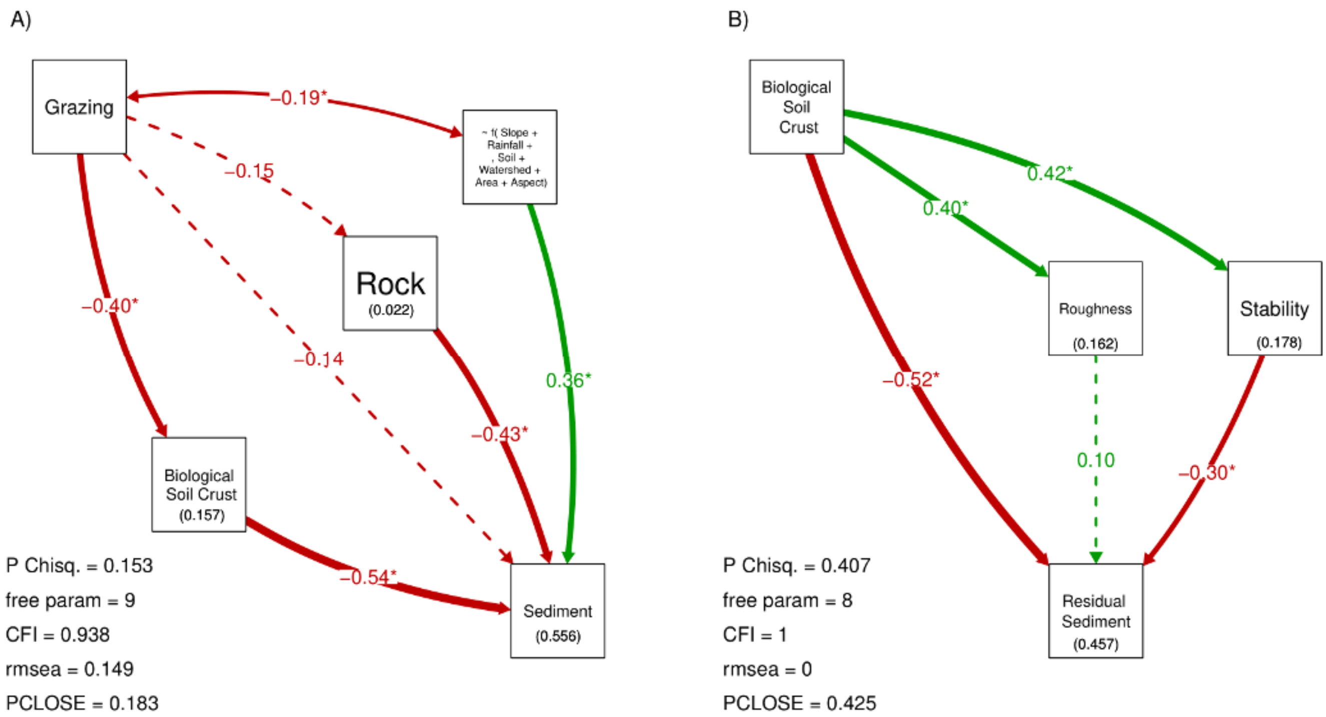

Sediment (kg m^{-2})
A
B
Ungrazed
Grazed
Mixed
Shale
Precipitation (mm)
Average Intensity (mm h^{-1})
Hillslope Area (m^2)
Slope



Log Response Ratio: Grazed/ Ungrazed
Warm-Season Precip. (mm)
4
3
2
1
0
-1
-2
-3
200
100
0
1960
1970
1980
1990
2000
2010



Watershed 1
Watershed 2
Watershed 3
Watershed 4
Lusby 1979
Present Study
1:1
Grazed (kg m^-2)
Ungrazed (kg m^-2)
1e+00
1e-01
1e-02
1e-03
1e-04
1e-03
1e-02
1e-01
1e+00
log(Grazed) = -0.458 + 0.519 x log(Ungrazed)
R2: 0.47
log(Grazed) = -3.419 + 0.422 x log(Ungrazed)
R2: 0.28



A)
Grazing
−0.19*
~ f(Slope + Rainfall + , Soil + Watershed + Area + Aspect)
−0.15
Rock
(0.022)
−0.40*
−0.14
0.36*
−0.43*
Biological Soil Crust
(0.157)
−0.54*
Sediment
(0.556)
P Chisq. = 0.153
free param = 9
CFI = 0.938
rmsea = 0.149
PCLOSE = 0.183
B)
Biological Soil Crust
0.42*
0.40*
Roughness
(0.162)
Stability
(0.178)
−0.52*
0.10
−0.30*
Residual Sediment
(0.457)
P Chisq. = 0.407
free param = 8
CFI = 1
rmsea = 0
PCLOSE = 0.425