

To: McAlear, Christopher[cmcalear@blm.gov]
From: Miller, Mark
Sent: 2017-08-02T11:54:33-04:00
Importance: Normal
Subject: Re: Former Grand Staircase colleague - touching base from Alaska BLM - potential meeting in DC next week?
Received: 2017-08-02T11:54:51-04:00
[Miller 2008 GSE.pdf](#)
[Miller etal 2011 AlternativeStates.pdf](#)

Hi Chris - thanks for your response. After I contacted you yesterday, a conflict arose so that I'll be returning to AK early on the 8th instead of staying over another day. We'll have to hook up some other time, when I'm next in DC or ...if you should happen to travel here to learn more about AK conservation lands.

It's an interesting time for NLCS these days. I spent more than 15 yrs of my career working on or in Grand Staircase and Bears Ears and related issues (attached).

Best - Mark

Mark E. Miller, PhD | Deputy Director
North Slope Science Initiative | <http://www.northslope.org>
Email: memiller@blm.gov | Office: 907-271-3212 | Mobile: 907-231-9427
c/o Bureau of Land Management | Alaska State Office | State Director's Office
222 West 7th Avenue, #13 | Anchorage, AK 99513

"We are drowning in information, while starving for wisdom. The world henceforth will be run by synthesizers, people able to put together the right information at the right time, think critically about it, and make important choices wisely."

E. O. Wilson, *Consilience*

On Tue, Aug 1, 2017 at 9:56 AM, McAlear, Christopher <cmcalear@blm.gov> wrote:

Hi Mark,
Sounds like you have been very busy doing great work. Looks like you followed Kate over to the Moab area before coming back from the dark side ;<)

Would love to catch up, let me know when on the 8th works for you. Between 1 3 works well or coffee before 0900?

Hope all is well,
Chris m

On Tue, Aug 1, 2017 at 12:59 PM, Miller, Mark <memiller@blm.gov> wrote:

Chris -
I hope that you remember the time that we were colleagues on staff at Grand Staircase back in the early days. Since then, we've both moved around a bit.

I'm going to be in DC next week and wonder whether you might be available to meet with me sometime on Wed 9th. I've also reached out to Karen Prentice and Gordon Toeys (see below) but have not yet received a response.

I look forward to hearing from you.

Mark

Mark E. Miller, PhD | Deputy Director
North Slope Science Initiative | <http://www.northslope.org>
Email: memiller@blm.gov | Office: 907-271-3212 | Mobile: 907-231-9427
c/o Bureau of Land Management | Alaska State Office | State Director's Office
222 West 7th Avenue, #13 | Anchorage, AK 99513

"We are drowning in information, while starving for wisdom. The world henceforth will be run by synthesizers, people able to put together the right information at the right time, think critically about it, and make important choices wisely."

E. O. Wilson, *Consilience*

----- Forwarded message -----

From: **Miller, Mark** <memiller@blm.gov>
Date: Tue, Aug 1, 2017 at 8:56 AM
Subject: Touching base from Alaska BLM - potential meeting in DC next week?
To: Karen Prentice <kprentic@blm.gov>
Cc: Gordon Toevs <gtoevs@blm.gov>

Karen -

Among others here in our office, I had hoped that you would be able to travel to Alaska next week to participate in the planned field visit. But it turns out that I will be in DC for training on Mon-Tue 7-8 Aug, and I'm wondering whether you and possibly others in WO would be available to meet with me sometime on Wed 9th. I've already reached out to Gordon Toevs, but he may be out of the office and I've not yet heard from him.

I'm new to my position here, having arrived in March after ~ 20 years on the Colorado Plateau - with BLM at Grand Staircase, NPS Inventory & Monitoring program, USGS (based again at Grand Staircase), and most recently with NPS again as Chief of Resources for Arches, Canyonlands, Hovenweep, and Natural Bridges. I believe you know my close colleague Mike Duniway.

I look forward to hearing from you.

Thanks in advance.

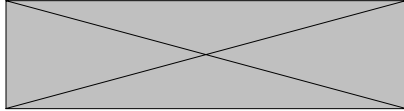
Mark

Mark E. Miller, PhD | Deputy Director
North Slope Science Initiative | <http://www.northslope.org>
Email: memiller@blm.gov | Office: 907-271-3212 | Mobile: 907-231-9427
c/o Bureau of Land Management | Alaska State Office | State Director's Office
222 West 7th Avenue, #13 | Anchorage, AK 99513

"We are drowning in information, while starving for wisdom. The world henceforth will be run by synthesizers, people able to put together the right information at the right time, think critically about it, and make important choices wisely."

E. O. Wilson, *Consilience*

--



Christopher McAlear
Assistant Director
National Conservation Lands
and Community Partnerships
(W) 202-208-4731
(C) 775-722-9539

Broad-Scale Assessment of Rangeland Health, Grand Staircase–Escalante National Monument, USA

Mark E. Miller

Author is Research Ecologist, US Geological Survey, Southwest Biological Science Center, Kanab, UT 84741, USA.

Abstract

Over a 3-yr period, the qualitative assessment protocol “Interpreting Indicators of Rangeland Health” was used to evaluate the status of three ecosystem attributes (soil/site stability, hydrologic function, and biotic integrity) at over 500 locations in and adjacent to Grand Staircase Escalante National Monument (Utah). Objectives were to provide data and interpretations to support the development of site-specific management strategies and to investigate broad-scale patterns in the status of different rangeland ecological sites. Quantitative data on ground cover, plant community composition, and soil stability were collected to aid the evaluation of qualitative attributes and improve consistency of the assessment process. Ecological sites with potential vegetation dominated by varieties of big sagebrush (*Artemisia tridentata* Nuttall) had the highest frequencies (46.7%–75.0%) of assessments with low ratings (moderate or greater departure from expected reference conditions) for all three ecosystem attributes. In contrast, sites with potential vegetation characterized by Utah juniper (*Juniperus osteosperma* [Torrey] Little) and/or Colorado pinyon (*Pinus edulis* Engelman) had low frequencies (0.0%–7.8%) of assessments with low ratings for all attributes. Several interacting factors likely contributed to the development of patterns among ecological sites, including 1) potential primary production and thus long-term exposure to production-oriented land uses such as livestock grazing; 2) the presence of unpalatable woody plants capable of increasing and becoming persistent site dominants due to selective herbivory, absence of fire, or succession; 3) soil texture through effects on hydrologic responses to livestock grazing, trampling, and other disturbances; and 4) past management that resulted in high livestock use of ecological sites with sensitive fine-loamy soils following treatments designed to increase forage availability. This case study illustrates an extensive application of an assessment technique that is receiving increasing use worldwide, and results contribute to an understanding of factors contributing to patterns and processes of rangeland degradation.

Resumen

Durante un período de tres años, se siguió el protocolo *de interpretación de Indicadores de Salud de Pastizales*, para evaluar el estado de tres atributos del ecosistema (Suelo /Estabilidad del Sitio, función hidrológica e integridad biótica) en mas de 500 áreas del Grand Staircase Escalante National Monument (en Utah USA) y en áreas adyacentes. Con los objetivos de proporcionar datos e interpretaciones que apoyen el desarrollo de estrategias de manejo a sitios específicos, y para investigar los patrones a gran escala del estado de diferentes sitios ecológicos de pastizal. Se recolectaron datos cuantitativos sobre cobertura de suelo, composición vegetal de la comunidad, y estabilidad del suelo para ayudar a la evaluación de los atributos cualitativos y para mejorar la consistencia en el proceso de evaluación. Sitios ecológicos con la vegetación potencial dominada por el arbusto (*Artemisia tridentata* Nuttall) tuvieron las mayores frecuencias con los índices de evaluación mas bajos (46.7%–75%) con una diferencia moderada a grande en relación a la esperada con las áreas de referencia, para los tres atributos del ecosistema. En contraste, sitios con vegetación potencial caracterizados por el táscale (*Juniperus osteosperma* [Torrey] Little) y/o el Piñón colorado (*Pinus edulis* Engelman) presentaron bajas frecuencias (0.0%–7.8%) de evaluación con bajos índices para todos los atributos del ecosistema. La interacción de algunos factores probablemente contribuyó al desarrollo de patrones entre los sitios ecológicos, incluyendo 1) producción potencial primaria y por lo tanto largo tiempo que estas áreas estuvieron expuestas a la producción orientada del ganado en pastoreo; 2) la presencia de plantas leñosa de baja palatabilidad capaces de incrementar su población, llegando a ser dominantes y permanentes del sitio, debido al pastoreo selectivo, ausencia de fuego, o sucesión; 3) las textura del suelo y su efecto sobre respuesta hidrológica al pastoreo, pisoteo y otros disturbios; y 4) Historial de manejo, que da como resultado un alto grado de uso por el ganado en sitios ecológicos con suelos susceptibles de textura fina, seguidos por tratamientos diseñados para incrementar la disponibilidad de forraje. Este estudio ilustra una extensiva aplicación de una técnica de evaluación que está siendo utilizada más y más en todo el mundo y cuyos resultados contribuyen a un mejor entendimiento de los factores y patrones que causan la degradación de las áreas de pastizal.

Key Words: *Artemisia tridentata* Nuttall, big sagebrush, ecological sites, ecosystem assessment, rangeland condition, soil properties

Field work for this research was funded by the Bureau of Land Management. Data analyses and manuscript preparation were funded by the US Geological Survey (Southwest Biological Science Center and Earth Surface Dynamics Program) and supported by the Bureau of Land Management.

The use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the US Government.

At the time this research was initiated, Miller was ecologist, Bureau of Land Management, Grand Staircase Escalante National Monument, Kanab, UT 84741, USA.

Correspondence: Mark E. Miller, US Geological Survey, Southwest Biological Science Center, c/o Grand Staircase Escalante National Monument, Kanab, UT 84741, USA. Email: mark_miller@usgs.gov

Manuscript received 27 September 2007; manuscript accepted 24 February 2008.

INTRODUCTION

Over the past 15 yr, there has been a focused effort to develop new methods for assessing the status of rangeland ecosystems. This effort has been driven by increased recognition that 1) the dynamics of such ecosystems often are much more complex than previously assumed and 2) sustainable management requires consideration of a broader suite of ecosystem attributes than production of key forage species and similarity of the existing plant community to a single idealized climax community (see reviews by Pyke et al. 2002; Pyke and Herrick 2003; and Briske et al. 2005 for historical perspectives). In the United States, much of this effort directly followed recommendations made by expert panels convened by the National Research Council (NRC; NRC 1994) and the Society for Range Management Task Group on Unity in Concepts and Terminology Committee (SRM Task Group; SRM Task Group 1995). The NRC panel recommended that rangeland assessments should focus on indicators of soil stability, watershed function, nutrient cycling, energy flow, and recovery mechanisms (NRC 1994). The SRM Task Group observed that because the sustainable management of rangeland ecosystems depends primarily on soil conservation, assessments should evaluate rangeland plant communities in terms of their ability to protect a site against accelerated soil erosion (SRM Task Group 1995). Both panels recommended that assessments should be conducted and interpreted on the basis of a common system for classifying land units on the basis of soil, landscape setting, and climate analogous to the ecological site concept of the US Department of Agriculture Natural Resources Conservation Service (NRCS; NRCS 2003).

Both in the United States and in Australia, there has been rapid growth in research focusing on conceptual and applied aspects of rangeland assessment and monitoring, with a strong emphasis on indicators of ecosystem or landscape capacity to capture and retain soil and water resources. The majority of this work has focused on field-based indicators (Whitford et al. 1998; de Soyza et al. 2000a; Pyke et al. 2002; Rosentreter and Eldridge 2002; Tongway and Hindley 2004; Herrick et al. 2005; Pellant et al. 2005), but the need for approaches that can be applied affordably and effectively across expansive landscapes also has led to efforts focused on the development of indicators that can be reliably detected with remotely sensed imagery (de Soyza et al. 2000b; Ludwig et al. 2002, 2007). Rather than being a stand-alone activity, assessment increasingly is recognized as a key component of an integrated framework designed to support science-based management of rangeland ecosystems (Herrick et al. 2006).

To date, the most widely adopted assessment approach in the United States has been the technique "Interpreting Indicators of Rangeland Health" (IIRH; Pellant et al. 2000, 2005; Pyke et al. 2002). In this technique, an interdisciplinary team of resource specialists evaluates three ecosystem attributes (soil/site stability, hydrologic function, and biotic integrity) on the basis of a suite of qualitative indicators. IIRH is widely applied by NRCS, the Bureau of Land Management (BLM), and the National Park Service (NPS), and protocols have been translated into Spanish, Chinese, and Mongolian (J. Herrick, personal communication, August 2007).

Despite its widespread adoption and increasing use worldwide, there are no published examples of how the IIRH technique has been applied to evaluate the status of rangeland ecosystems across broad spatial extents characteristic of public lands in the western United States. The purpose of this paper is to describe one such project as a case study in which the technique was applied at over 500 locations in and adjacent to Grand Staircase-Escalante National Monument, Utah (hereafter, the Monument), over a 3-yr period. Objectives of this assessment project were 1) to provide data and interpretations to support the development of site-specific management strategies for the improvement of resource conditions and 2) to investigate broad-scale patterns in the status of different rangeland ecological sites across the entire Monument. The second objective is the focus of this paper. This case study illustrates an extensive application of the IIRH technique, and results provide insights into factors affecting patterns and processes of rangeland degradation.

METHODS

Study Area

The Monument covers approximately 760 000 ha in southern Utah and the west-central portion of the Colorado Plateau physiographic province (Hunt 1974) between lat 37°N, lat 38°N, long 111°W, and long 112.5°W. Elevation ranges from 1 164 to 2 625 m, and mean annual precipitation (MAP; 1961–1990) ranges from 17 to 61 cm. (Precipitation estimates are based on the PRISM model, <http://www.ocs.orst.edu/prism>; Daly et al. 1994.) Approximately 90% of the Monument receives less than 36 cm MAP. As a proportion of MAP, May–September precipitation varies from 33.1% in Kanab (1 509 m elevation, 37.9 cm MAP, 16 km west of the Monument boundary) to 44.2% in Escalante (1 771 m elevation, 25.4 cm MAP, north-central edge of the Monument). Tremendous geologic and topographic heterogeneity (Doelling et al. 2000), as well as gradients in elevation and precipitation, together are responsible for generating a diversity of soils and ecological settings across the Monument. In a recent soil survey for the Monument, the NRCS described 136 distinct soil types and 50 distinct ecological sites (NRCS 2005).

Livestock grazing has been an important economic activity on lands within the Monument since the time of Euro-American settlement in the 1870s (Bradley 1999), and it remains the most extensive land use on the Monument today. Monument lands are subdivided into 91 grazing allotments, some of which extend onto adjoining public lands managed by the NPS (Glen Canyon National Recreation Area) and the US Department of Agriculture Forest Service (Dixie National Forest). Allotments are divided into two or more fenced pastures to facilitate livestock management. Pastures represent the smallest management units in the Monument, although they are typically larger than 5 000 ha and range in size up to 54 288 ha.

Sampling Design

A major objective of the assessment project was to collect data that would contribute to an evaluation of resource conditions

in grazing allotments and to the development of future strategies for meeting resource-management objectives. As a consequence, assessments were conducted in all pastures and allotments across the Monument. Within these management units, it was assumed that ecosystem conditions could vary among different soils and ecological sites due to potential differences in past livestock use and in ecosystem responses to livestock use, management activities, and climate variability. Thus digital spatial data delineating soils and ecological sites were used to stratify each pasture into soil-based sampling units.

Within sampling units in pastures, specific assessment locations were identified subjectively rather than probabilistically. This approach was chosen because time and resources were judged to be inadequate for obtaining a statistically adequate number of randomly located assessments for each sampling unit in all pastures and allotments, given the overall scope of the project. For each pasture, soil map units were ranked in descending order according to their total area in the pasture, and at least one assessment was conducted in the predominant ecological site in the soil map units that cumulatively accounted for at least 75% of the pasture area. Assessments also were conducted in areas expected to receive relatively high livestock use even where these areas were associated with minor soil components or soil map units that fell below the 75% cut-off in a particular pasture. Water sources and similar areas with concentrated livestock use were excluded from sampling. The assessment team selected one or more representative assessment locations associated with each targeted ecological site, with representativeness evaluated by examining aerial photographs with superimposed soil map unit delineations and by surveying conditions on the ground prior to conducting assessments. Assessment locations were approximately 0.5–1.0 ha in size.

Field Methods

Assessments were conducted following the technique IIRH, version 3 (Pellant et al. 2000; Pyke et al. 2002). The standard technique calls for the evaluation of three ecosystem attributes (soil/site stability, hydrologic function, and biotic integrity; Table 1) on the basis of 17 qualitative indicators (Pellant et al. 2000; Pyke et al. 2002; Table 2). Indicators and attributes for a particular assessment area are evaluated and rated according to the degree to which they depart from benchmark (reference) conditions described in ecological site descriptions prepared by NRCS and/or observed at one or more ecological reference areas (Pellant et al. 2000; Pyke et al. 2002), and on the basis of the combined experience and professional judgment of the interdisciplinary assessment team. In all cases, benchmark conditions are identified and applied on an ecological-site basis, thus requiring assessment teams to properly identify soil types and ecological sites. An ordinal, five-class rating system is used, with degree of departure rated as none to slight (NS), slight to moderate (SM), moderate (M), moderate to extreme (ME), or extreme (E). In the project described here, assessment teams identified relatively few reference areas. Thus ratings primarily were based on NRCS ecological site descriptions for those indicators related to plant community composition, ground cover, and potential primary production. For indicators not described in existing site descriptions (e.g., frequency and

Table 1. Three attributes of rangeland health and their definitions (from Pellant et al. 2000; Pyke et al. 2002).

Attribute	Definition
Soil/site stability	The capacity of a site to limit redistribution and loss of soil resources (including nutrients and organic matter) by wind and water.
Hydrologic function	The capacity of a site to capture, store, and safely release water from rainfall, run-on, and snowmelt (where relevant), to resist a reduction in this capacity, and to recover this capacity following degradation.
Biotic integrity	Capacity of a site to support characteristic functional and structural communities in the context of normal variability, to resist loss of this function and structure due to a disturbance, and to recover following such disturbance.

spatial distribution of erosional features such as rills, pedestals, and terracettes), indicator ratings primarily were based on team members' collective field observations and experience. Interdisciplinary assessment teams ranged in size from two to five members, with botanists, ecologists, geologists, wildlife biologists, and rangeland management specialists serving as the primary team members.

The IIRH protocol allows for the use of additional indicators where necessary to meet local assessment needs (Pellant et al. 2000). For this project, the integrity of biological soil crusts (BSCs) was included as an 18th indicator applicable to all three ecosystem attributes (Table 2) because of important BSC contributions to soil stabilization (Belnap 1995; Williams et al. 1995a, 1995b), hydrologic processes (Warren 2003; Belnap et al. 2005), nutrient cycling (Evans and Lange 2003), and biological diversity (Rosentreter and Belnap 2003) in rangeland ecosystems on the Colorado Plateau. Ratings for this indicator were based on the distribution and abundance of soil lichens, soil mosses, and dark cyanobacterial crusts in comparison with reference areas and team members' collective field observations and experience (Table 3). During the 2002 field season, ratings for biological soil crusts also were informed by preliminary results from a concurrent project being conducted to develop a spatial predictive model of BSC cover, composition, and function in relation to precipitation and substrate characteristics (Bowker et al. 2006).

To inform the evaluation of qualitative indicators and increase consistency of the assessment process, quantitative data on ground cover (e.g., percentage of cover of bare ground/mineral soil, BSC, litter, plant bases, and rock), plant community composition (percentage of live and dead canopy and basal cover by species and plant functional groups), and soil stability were collected prior to evaluating indicators and attributes (Pyke et al. 2002). Data on ground cover and plant community composition were collected following the step-point technique (Coulloudon et al. 1999). Cover data were recorded for 50–100 subsample points (approximately 1-mm diameter) placed at 4-pace intervals along a pace transect walked by one or two team members. The pace transect crossed the assessment area three to five times, with total transect length ranging from 150 to 300 m. Surface and subsurface soil stability

Table 2. Brief description of 18 rangeland health indicators, their applicability to rangeland health attributes, and associated quantitative data collected during assessments conducted on Grand Staircase–Escalante National Monument (adapted from Pyke et al. 2002).

Indicator and brief description	Attributes ¹			Quantitative data
	S	H	B	
1. Rills – frequency and spatial distribution of linear erosional rivulets	X	X	—	—
2. Water flow patterns – amount and distribution of overland flow paths that are identified by litter distribution and visual evidence of soil and gravel movement	X	X	—	—
3. Pedestals and/or terracettes – frequency and distribution of rocks or plants where soil has been eroded from their base (pedestals), and/or occurrence of erosional terracettes	X	X	—	—
4. Bare ground – size and connectivity among areas of soil not protected by vegetation, biological soil crusts, litter, standing dead vegetation, gravel, or rocks	X	X	—	Percentage of bare ground
5. Gullies – amount of channels cut into the soil and the amount and distribution of vegetation in the channel	X	X	—	—
6. Wind-scoured areas, blowouts, and/or deposition areas – frequency of areas where soil is removed from under physical or biological soil crust or around vegetation OR frequency of accumulation areas of soil associated with large structural objects, often woody plants	X	—	—	—
7. Litter movement – frequency and size of litter displaced by wind and overland flow of water	X	—	—	—
8. Soil surface resistance to erosion – ability of soils to resist erosion through the incorporation of organic material into soil aggregates	X	X	X	Soil aggregate stability
9. Soil surface loss or degradation – frequency and size of areas missing all or portions of the upper soil horizons that normally contain the majority of organic material of the site	X	X	X	—
10. Plant community composition and distribution relative to infiltration and runoff – the community composition or distribution of species that restrict the infiltration of water on the site	—	X	—	Percentage of composition by functional group
11. Compaction layer – thickness and distribution of the structure of the soil near the soil surface (≤ 15 cm)	X	X	X	—
12. Functional / structural groups – the number of groups, the number of species within groups, or the rank of order of dominance of groups	—	—	X	Relative composition and dominance of functional groups (based on cover)
13. Plant mortality/decadence – frequency of dead or moribund (dying) plants	—	—	X	Percentage of standing-dead cover
14. Litter amount – deviation in the amount of litter	—	X	X	Percentage of cover of litter
15. Annual aboveground production – amount relative to the potential for that year based upon recent climatic conditions	—	—	X	—
16. Invasive plants – abundance and distribution of invasive plants regardless if they are noxious weeds, exotic species, or native plants whose dominance greatly exceeds that expected for the ecological site	—	—	X	Percentage of cover and relative composition of invasive plants
17. Reproductive capability of perennial plants – evidence of the inflorescences or of vegetative tiller production relative to the potential for that year based upon recent climatic conditions	—	—	X	—
18. Biological soil crusts – amount, spatial distribution, and degree of development	X	X	X	Percentage of cover and relative composition of biological soil crusts

¹S indicates soil/site stability; H, hydrologic function; and B, biotic integrity.

beneath plant canopies and in interspaces among plants was measured using a soil aggregate stability field kit (Herrick et al. 2001). Nine pairs of surface and subsurface samples were collected from three to six interspace locations and three to six subcanopy locations that were selected as visually representative of conditions across the assessment area.

Assessments were conducted from July 2000 through December 2002, with about 80% of the field work conducted during April–October periods in 2001 and 2002. Amounts of precipitation received in Kanab and Escalante respectively were 32% and 43% below the 1971–2000 average during the 2000 water year, 13% and 27% above average during the 2001

Table 3. Evaluation matrix for biological soil crusts (from Pellant et al. 2000).

Indicator	Degree of departure from ecological site description and/or ecological reference area(s)				
	Extreme	Moderate to extreme	Moderate	Slight to moderate	None to slight
Biological soil crusts	Found only in protected areas; very limited suite of functional groups	Largely absent, occurring mostly in protected areas	In protected areas and with a minor component in interspaces	Evident throughout the site, but continuity is broken	Largely intact and nearly matches site capability

water year, and 53% and 64% below average during the 2002 water year (Western Regional Climate Center 2007).

Data Analyses

Chi-square analysis (Zar 1999) was used to examine whether the three attributes of rangeland health had different rating distributions for all assessment locations combined (507 assessments and 1 521 attribute ratings). For ecological sites with five or more assessments, χ^2 analyses also were used to determine whether some ecological sites were characterized by ecosystem conditions that were better (i.e., a greater proportion of assessments with a small degree of departure from expected reference conditions) or worse (greater proportion of assessments with a large degree of departure from expected reference conditions) than typical conditions described on the basis of the combined data set for all 507 assessment locations. For each ecological site, separate χ^2 analyses were conducted for each of the three attributes of rangeland health.

Extensive areas within the Monument were mechanically treated in the past to reduce the cover of unpalatable woody vegetation such as big sagebrush (*Artemisia tridentata* Nuttall), Utah juniper (*Juniperus osteosperma* [Torrey] Little), and Colorado pinyon (*Pinus edulis* Engelman). In conjunction with mechanical treatments, treated areas (hereafter referred to as “seedings”) generally were seeded with nonnative forage grasses such as crested wheatgrass (*Agropyron cristatum* [L.] Gaertner) and Russian wildrye (*Elymus junceus* Fischer). (Taxonomic nomenclature follows Welsh et al. 2003.) For ecological sites with five or more assessments in seedings and in comparable untreated areas, separate χ^2 analyses were conducted to examine whether there was a tendency for seedings or untreated areas to be characterized by better or worse ecosystem conditions in comparison with all 507 assessments combined. For all χ^2 analyses, rating classes E and ME were combined into a single class (E–ME) because of the infrequent occurrence of E ratings. Multivariate analysis of variance (MANOVA) also was used to test for differences between mean values of selected quantitative measures for seeded and comparable untreated ecological sites. Dependent variables were log-transformed [$x' \ln(x+1)$] prior to analysis because variances were proportional to means (Zar 1999). Stepwise multiple regression analysis was used to examine potential factors contributing to general patterns in ecosystem condition among ecological sites (Zar 1999).

Ecosystems dominated by varieties of big sagebrush are of particular interest to resource managers on the Colorado Plateau and throughout the Intermountain West because of their diversity and habitat value, and because they have been widely degraded by cumulative effects of land use, invasive exotic plants, and altered fire regimes (Knick et al. 2003; Connelly et al. 2004; Welch 2005). Five of the 50 distinct ecological sites found in the Monument are characterized by potential vegetation dominated by varieties of big sagebrush (Table 4; NRCS 2005). Of these five sites, the Semidesert Loam (Wyoming big sagebrush) site had a relatively large sample size ($n = 55$) and was characterized by a wide range of rangeland-health conditions. For these reasons, data for this ecological site were examined in greater detail to evaluate relationships between quantitative data and qualitative ratings of rangeland health. Principal components analysis (PCA; McCune and

Grace 2002) with varimax normalized factor rotation was used to describe variability among the 55 assessments in terms of 12 quantitative variables: interspace soil aggregate stability; percentage of total live cover; total plant cover; percentage of bare ground; percentage of BSC cover; percentage of litter cover; percentage of relative cover of annual exotic plants, total exotic plants, and woody plants; functional group richness; diversity (H'); and evenness (J' ; Zar 1999). Spearman's rank correlation coefficients (Zar 1999) were calculated to describe relationships between quantitative variables and ordinal qualitative ratings assigned to the three rangeland-health attributes. MANOVA was used to test whether log-transformed mean values for selected quantitative variables were significantly different among rating classes for individual rangeland health attributes. For rangeland health attributes determined to have significant effects by MANOVA, Tukey's honestly significant difference (HSD) post hoc analysis was used to test for differences between mean quantitative measures associated with different attribute rating classes (Zar 1999). With the exception of the χ^2 analyses, all statistical analyses were conducted using the software package STATISTICA™ version 6.1 on a Windows® platform (Statsoft 2004). For all analyses, results with $P \leq 0.05$ were considered statistically significant.

RESULTS

Overall Patterns Among Ecological Sites

For all 507 assessments combined, SM was the modal rating class for each of the three rangeland health attributes (Fig. 1). The rating distributions for all three attributes were similar, but the distribution for biotic integrity was significantly different than the distribution for all 1 521 attribute ratings combined. Overall, biotic integrity tended to receive NS ratings less frequently and M and SM ratings more frequently than soil/site stability and hydrologic function attributes (Fig. 1). Of the 507 assessments, 226 (44.6%) were assigned a low rating (moderate or greater departure from expected reference conditions) for at least one of the three attributes, and 100 (19.7%) were assigned low ratings for all three attributes.

Of the 26 ecological sites with five or more assessments (including seeded and untreated areas for two ecological sites), 10 had one or more attributes with rating distributions that were significantly different than the overall distributions for all 507 assessments (Tables 4 and 5). Of the five ecological sites with significantly higher frequencies of low ratings relative to the overall distributions, four were deep-soil ecological sites with high potential production and potential vegetation dominated by varieties of big sagebrush (Tables 4 and 5). In contrast, all five ecological sites with significantly lower frequencies of low ratings relative to the overall distributions were shallow-soil ecological sites with relatively low potential production and potential vegetation characterized by the presence of juniper and/or pinyon. Only the seeded Upland Loam and seeded and untreated Semidesert Loam ecological sites had rating distributions that were significantly different from overall distributions for all three rangeland health attributes. Potential dry-weight production (Table 4; $\beta = 0.447$, $P = 0.003$) and treatment (seeded vs. untreated, from Table 4;

Table 4. So -depth class, potential dry-weight production by ecological site, and χ^2 values by ecological site and range and height attribute (soil stability, hydrologic function, and botanical integrity) for sites with five or more range and height assessments (n), Grand Staircase-Escalante National Monument. Chi-square values were calculated to test the hypothesis that the distribution of ratings for a particular ecological site and range and height attribute was not different than the overall distribution for a 507 assessment locations. Values in bold type are statistically significant. Ecological sites are ranked in descending order according to the percentage of assessments that received ratings of moderate or greater departure from expected reference conditions for a three attributes of range and height. Potential production includes perennials, grasses and shrubs that provide livestock forage. Unpublished production includes unpalatable woody and suffrutescent plants.

Ecological site ¹	Site no	Soil -depth class	Potential dry-weight production (kg · ha ⁻¹ · yr ⁻¹) ²		n	Soil stability χ^2	Hydrologic function χ^2	Botanical integrity χ^2	Assessments with moderate or greater departure from expected attributes (%)
			Total	Potential					
Up and Loam (mountain big sagebrush) – seeded	035XY308UT	Deep	1009	504	20	22.8***	29.0***	26.2***	75.0
Sem desert Loam (Wyoming big sagebrush) – seeded	035XY209UT	Deep	757	378	24	25.5***	31.3***	23.7***	62.5
Sem desert Loam (Wyoming big sagebrush) – untreated	035XY209UT	Deep	757	378	31	49.3***	33.8***	25.9***	58.1
Loamy Bottom (basin big sagebrush)	035XY011UT	Deep	1793	986	15	3.2	5.2	18.2***	46.7
Sem desert Sandy Loam (blackbrush)	035XY218UT	Deep	532	213	7	13.2**	2.9	11.1*	42.9
Sem desert Sandy Loam (Wyoming big sagebrush)	035XY214UT	Deep	532	399	15	6.5	4.4	2.0	33.3
Up and Loam (mountain big sagebrush) – untreated	035XY308UT	Deep	1009	504	15	4.6	5.7	3.1	26.7
Desert Sandy Loam (fourwing shrub)	035XY215UT	Deep	476	357	5	0.7	0.3	2.8	20.0
Sem desert Gravelly Shallow Breaks	—	Shallow	No data	No data	5	6.9	1.3	3.5	20.0
Sem desert Shallow Sandy Loam (blackbrush)	035XY233UT	Shallow	364	73	5	1.7	1.9	1.2	20.0
Desert Shallow Clay (mat shrub)	035XY223UT	Shallow	213	32	6	5.8	2.0	2.1	16.7
Sem desert Sandy Loam ³ (black grama)	035XY219UT	Deep	532	426	14	7.1	5.8	4.8	14.3
Sem desert Sand (fourwing shrub)	035XY212UT	Deep	644	451	32	0.6	3.0	2.3	12.5
Sem desert Sandy Loam (fourwing shrub)	035XY215UT	Deep	532	453	12	1.7	2.1	4.7	8.3
Sem desert Shallow Loam (Utah juniper-nyon)	—	Shallow	504	151	64	7.6*	5.7	12.8**	7.8
Up and Sand (mountain big sagebrush)	—	Deep	672	202	15	1.8	3.2	4.7	6.7
Up and Shallow Dissected Slope ³ (nyon-Utah juniper)	035XY311UT	Shallow	196	59	30	4.9	2.9	11.3*	6.7
Sem desert Steep Shallow Loam (Utah juniper-nyon)	—	Shallow	252	88	16	4.9	5.2	20.4***	6.3
Up and Shallow Loam (nyon-Utah juniper)	035XY315UT	Shallow	616	216	34	13.5**	6.2	11.2*	2.9
Desert Shallow Sandy Loam ³ (shadscale)	035XY130UT	Shallow	252	126	5	2.8	2.2	0.9	0.0
Sem desert Shallow Clay (shadscale-Utah juniper)	—	Shallow	168	101	5	2.5	5.4	4.7	0.0
Sem desert Shallow Sand (Cutter Mormon tea)	035XY225UT	Shallow	420	273	15	2.6	6.0	5.5	0.0
Sem desert Shallow Sand (Utah juniper-nyon)	035XY227UT	Shallow	252	101	8	1.5	1.9	1.9	0.0
Sem desert Shallow Shale (Utah juniper-nyon)	—	Shallow	252	88	9	7.5	12.7**	23.5***	0.0
Up and Loam ⁴ (nyon-Utah juniper)	—	Deep	644	226	5	2.7	2.1	2.6	0.0
Up and Stony Loam (nyon-Utah juniper)	035XY321UT	Deep	560	140	11	5.1	3.6	2.4	0.0

¹Scientific names of associated plant species: mountain big sagebrush (*Artemisia tridentata* var. *vaseyana* [Rydb.] B. Boivin), Wyoming big sagebrush (*A. tridentata* var. *wyomingensis* [Beetle & A. Young] S. Welsh), basin big sagebrush (*A. tridentata* Nuttall), blackbrush (*Coleogyne ramosissima* Torrrey), fourwing shrub (*Atriplex canescens* [Pursh] Nuttall), mat saltbush (*Atriplex corrugata* S. Watson), black grama (*Bouteloua eriopoda* [Torr.] Torrey), Utah juniper (*Juniperus osteosperma* [Torr.] Little), piñon (*Pinus edulis* Engelm.) shadscale (*Atriplex confertifolia* [Torr.] & Frémont) S. Watson), Cutter Mormon tea (*Ephedra viridis* var. *viscida* [Culter] L. Benson)

²Potential production in normal-precipitation years with plant community composition similar to that described by the Natural Resources Conservation Service (NRCS) as the historical climax plant community (data from NRCS 2005 unless otherwise noted)

³Production estimates from NRCS ecological site description

⁴Production estimates from draft ecological site description prepared by the Bureau of Land Management

* $P \leq 0.05$ ** $P \leq 0.01$ *** $P \leq 0.001$

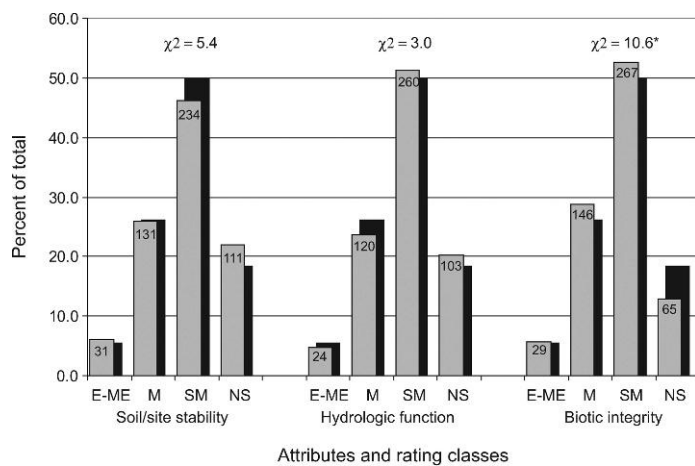


Figure 1. Overall distributions (gray bars) of ratings assigned to three rangeland-health attributes at 507 assessment locations on Grand Staircase–Escalante National Monument. Numerals in gray bars indicate numbers of assessments that received associated ratings. Black bars behind each rating distribution indicate the overall distribution of all 1 521 ratings and the null distributions that were used in χ^2 analyses for each of the three attributes (reflected by χ^2 statistics above each rating distribution; * $P < 0.05$). For attribute ratings, E indicates extreme departure; ME, moderate to extreme departure; M, moderate departure; SM, slight to moderate departure; and NS, no departure to slight departure from expected reference conditions.

β 0.556, P 0.0004) both were significant in a stepwise multiple regression model predicting for each ecological site the percentage of assessment locations that was assigned low ratings for all three attributes of rangeland health (adjusted R^2 0.62, df 2,22, F 20.34, P 0.00001). Log-transformed means for percentage of bare ground, BSC cover, and interspace soil aggregate stability were not significantly different between seeded and untreated Semidesert Loam (Wilks' λ 0.94, F 0.94, df 3,46, P 0.43) and Upland Loam (Wilks' λ 0.84, F 1.70, df 3,27, P 0.19) ecological sites.

Patterns Within the Semidesert Loam Ecological Site

Two PCA axes explain 49.3% of the variability in 12 quantitative variables sampled in conjunction with 55 assessments of the Semidesert Loam ecological site (seeded and untreated areas combined; Fig. 2). Axis 1 represents a gradient of decreasing bare ground and increasing total plant cover, total live cover, and functional group richness and diversity (Fig. 2a). Axis 2 represents a gradient of decreasing relative cover of exotic plants (including nonnative forage grasses, which accounted for 72.0% of total exotic cover, on average) and increasing interspace soil aggregate stability and cover of BSCs (Fig. 2a). Qualitative ratings assigned to the three attributes of rangeland health tended to be higher (lesser degree of departure from expected reference conditions) at assessment locations characterized by higher scores for PCA axes 1 and 2, but there was considerable variability in PCA scores among assessment locations that were assigned the same qualitative rating for a particular attribute (Figs. 2b–2d). Ratings for the three attributes of rangeland health were more strongly correlated with site scores for PCA axis 2 than with site scores for PCA axis 1 (Table 6).

Seven of twelve quantitative variables were significantly correlated with ratings assigned for one or more rangeland health attributes (Table 6). Measures of functional group richness and diversity (H') were important in the PCA but not correlated with assigned ratings for any of the three attributes (Table 6). However, both variables were significantly correlated with assigned ratings for the individual indicator pertaining to functional and structural groups (richness: ρ 0.42, $P < 0.01$; diversity: ρ 0.38, $P < 0.01$). Percentage of bare ground, total live cover, BSC cover, and interspace soil aggregate stability had the highest rank correlations with assigned attribute ratings (Table 6). MANOVA results for these four variables were statistically significant for each of the three rangeland health attributes (soil/site stability: Wilks' λ 0.26, F 6.28, effect df 12, error df 114.1, $P < 0.001$; hydrologic function: Wilks' λ 0.29, F 5.63, effect df 12,

Table 5. Percentages of assessments by rating class¹ for three rangeland health attributes (soil/site stability, hydrologic function, and biotic integrity) at 10 rangeland ecological sites and for all sites combined, Grand Staircase–Escalante National Monument. Values are only reported for those ecological sites and attributes with rating distributions that are significantly different than the associated distribution for all sites combined (see Table 4 for significant χ^2 values). Bold, underlined print indicates percentages that exceed corresponding percentages for all sites combined.

Ecological site	n	Soil/site stability				Hydrologic function				Biotic integrity			
		E-ME	M	SM	NS	E-ME	M	SM	NS	E-ME	M	SM	NS
All sites combined	507	6.1	25.8	46.2	21.9	4.7	23.7	51.3	20.3	5.7	28.8	52.7	12.8
Upland Loam (mountain big sagebrush) – seeded	20	20.0	60.0	20.0	0.0	20.0	60.0	20.0	0.0	5.0	80.0	15.0	0.0
Semidesert Loam (Wyoming big sagebrush) – seeded	24	16.7	62.5	20.8	0.0	16.7	62.5	16.7	4.2	25.0	45.8	29.2	0.0
Semidesert Loam (Wyoming big sagebrush) – untreated	31	35.5	29.0	19.4	16.1	22.6	45.2	22.6	9.7	22.6	48.4	22.6	6.5
Loamy Bottom (basin big sagebrush)	15	—	—	—	—	—	—	—	—	13.3	73.3	13.3	0.0
Semidesert Sandy Loam (Blackbrush)	7	0.0	85.7	14.3	0.0	—	—	—	—	0.0	85.7	14.3	0.0
Semidesert Shallow Loam (Utah juniper–pinyon)	64	1.6	17.2	48.4	32.8	—	—	—	—	0.0	14.1	67.2	18.8
Upland Shallow Dissected Slope (pinyon–Utah juniper)	30	—	—	—	—	—	—	—	—	3.3	3.3	80.0	13.3
Semidesert Steep Shallow Loam (Utah juniper–pinyon)	16	—	—	—	—	—	—	—	—	0.0	12.5	37.5	50.0
Upland Shallow Loam (pinyon–Utah juniper)	34	0.0	8.8	47.1	44.1	—	—	—	—	2.9	5.9	67.6	23.5
Semidesert Shallow Shale (Utah juniper–pinyon)	9	—	—	—	—	0.0	0.0	33.3	66.7	0.0	11.1	22.2	66.7

¹E ME indicates extreme or moderate to extreme departure; M, moderate departure; SM, slight to moderate departure; and NS, no departure to slight departure from expected reference conditions.

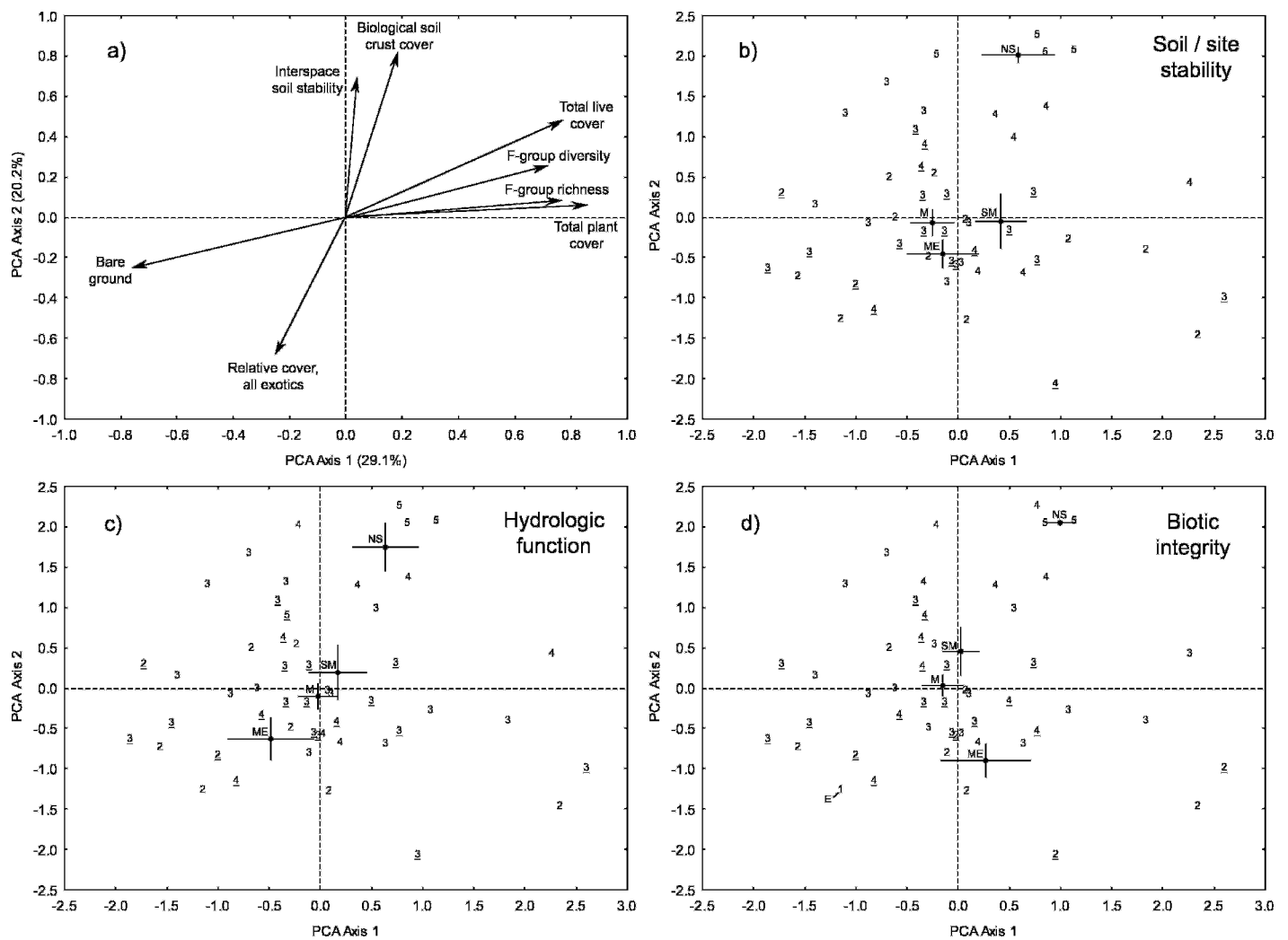


Figure 2. Principal components analysis (PCA) results for data associated with 12 quantitative variables measured at 55 Semidesert Loam assessment locations, Grand Staircase–Escalante National Monument. **a**, vectors indicate loadings (Pearson correlation coefficients, r) of eight variables on axes 1 and 2 (only those variables with $r \geq 0.60$ are shown; F-group indicates functional group). In the remaining panels, numbers 1–5 indicate attribute ratings (1 and E indicate extreme departure; 2 and ME, moderate to extreme departure; 3 and M, moderate departure; 4 and SM, slight to moderate departure; and 5 and NS, no departure to slight departure from expected reference conditions) assigned for **b**, soil/site stability; **c**, hydrologic function; and **d**, biotic integrity at each of the assessment locations. Underlined ratings are for assessments associated with seedlings. Coordinates of the attribute ratings in ordination space indicate PCA scores associated with the corresponding assessment location. Points indicate centroids (mean PCA scores ± 1 SE) for each set of assessment locations receiving the same attribute rating.

error df 114.1, $P < 0.001$; biotic integrity: Wilks' λ 0.32, F 3.62, effect df 16, error df 128.9, $P < 0.001$), but Tukey's HSD analyses found relatively few significant differences among log-transformed mean values for different attribute rating classes because of the high degree of variability in quantitative measures among assessments that were assigned the same rating for a particular attribute (Fig. 3). Mean quantitative measures for assessment locations that were assigned NS ratings for rangeland health attributes were statistically different than means associated with locations that were assigned lower rangeland health ratings in most cases, whereas means for locations that were assigned ME, M, or SM ratings were statistically different from one another less frequently (Fig. 3). This finding is consistent with PCA results showing that centroids for locations that were assigned ME, M, or SM ratings tended to be clustered together in the center of

the ordination space defined by the quantitative variables, whereas the centroids for locations assigned NS ratings were relatively distinct in ordination space (Fig. 2).

DISCUSSION

Results of this broad-scale assessment project indicate patterns in qualitative attributes and quantitative measures of rangeland health across a 760 000-ha landscape that represents a significant proportion of the Colorado Plateau physiographic province. Because of the large numbers of assessment locations and ecological sites included in the project, data resulting from this effort represent a valuable resource for examining general patterns in ecosystem condition among and within different ecological sites, and for developing hypotheses about factors

Table 6. Spearman rank correlations between 12 quantitative variables included in the principal components analysis (PCA; Fig. 2), site scores for PCA axes 1 and 2, and ordinal qualitative ratings (extreme, moderate-to-extreme, moderate, slight-to-moderate, and none-to-slight departure from expected reference conditions ranked 1–5, respectively) for rangeland health attributes soil/site stability, hydrologic function, and biotic integrity at 55 Semidesert Loam assessment locations on Grand Staircase–Escalante National Monument ($n=50$ for interspace soil aggregate stability). Bold type indicates statistically significant relationships.

Variable	S	H	B
Bare ground %	–0.65***	–0.65***	–0.40**
Total live cover %	0.54***	0.55***	0.48***
Total plant cover %	0.35**	0.38**	0.31*
Biological soil crust cover %	0.52***	0.47***	0.57***
Litter cover %	0.16	0.19	0.15
Interspace soil aggregate stability	0.50***	0.40**	0.50***
Functional group richness	0.09	0.19	0.19
Functional group diversity (H')	0.15	0.24	0.17
Functional group evenness (J')	0.05	0.02	–0.15
Relative annual exotic cover %	0.01	–0.10	–0.42**
Relative total exotic cover %	–0.18	–0.20	–0.31*
Relative woody plant cover %	–0.05	–0.06	–0.11
PCA axis 1 site scores	0.33*	0.33*	0.13
PCA axis 2 site scores	0.38**	0.42**	0.58***

¹S indicates soil/site stability; H, hydrologic function; and B, biotic integrity.

* $P \leq 0.05$; ** $P \leq 0.01$; *** $P \leq 0.001$.

that may have contributed to the development of these patterns.

Factors Contributing to Patterns Among Ecological Sites

Production Potential and Relative Use. At the scale of the entire Monument, upland ecological sites with the greatest production potential tended to be the most degraded, as measured by percentages of assessment locations that were assigned low ratings for all three attributes of rangeland health. Productivity has been widely cited as a factor affecting ecosystem responses to grazing by large herbivores (Milchunas et al. 1988; Cingolani et al. 2005; Lunt et al. 2007) and to disturbance in general (Huston 1979). In the Monument, production potential likely was an indirect factor contributing to general patterns of ecosystem status among different ecological sites because of correlations with land use and plant community composition.

In this rocky dryland environment characteristic of much of the Colorado Plateau, ecological sites with the greatest production potential account for a relatively small proportion of the landscape and thus have tended to receive a disproportionate level of use for livestock grazing—the predominant production-oriented land-use activity on the Monument. For example, estimates based on soil-survey data (NRCS 2005) indicate that productive Upland Loam, Semidesert Loam, and Loamy Bottom ecological sites cumulatively account for approximately 7.4% (56 461 ha) of the total Monument area. In contrast, relatively unproductive ecological sites with low frequencies of low rangeland health ratings (those with significant χ^2 values in Table 4) account for approximately 33.8% (257 378 ha) of the total Monument area. Relative to

the productive big sagebrush ecological sites, the unproductive ecological sites typically have received low levels of use for livestock grazing or other land-use activities except on a very localized basis. On the basis of existing data, it is difficult to quantify differences in livestock use among ecological sites because use is recorded by allotment and allotment boundaries do not correspond with ecological site boundaries.

Plant Community Composition. The relative abundance of different plant functional types is an important factor that affects ecosystem responses to drivers such as livestock grazing (Díaz et al. 2002; Lunt et al. 2007). In the Monument, rangeland ecological sites with the greatest production potential are characterized by the presence of big sagebrush, with that species accounting for a significant proportion of standing biomass and annual production (20%–30%) in historic climax plant communities described by NRCS (2005). Except for some formerly grazed reference areas and seedings where sagebrush was removed or thinned in the past, most assessments conducted in big sagebrush ecological sites found much higher ratios of sagebrush to perennial grasses than expected on the basis of NRCS ecological site descriptions—a factor that contributed to the assignment of low ratings for biotic integrity at such locations.

Big sagebrush is relatively unpalatable to livestock, and livestock grazing (selective herbivory) has long been cited as a process that has facilitated increases in shrub:grass ratios in sagebrush ecological sites throughout the Intermountain West due to effects of grass removal on competitive relations and fire frequency (USDA Forest Service 1937; Miller et al. 1994). But successional trends resulting in increasing shrub:grass ratios have been reported for ungrazed sagebrush ecosystems in some settings, a pattern that may be attributable to landscape characteristics that naturally protect such sites from fire (West and Yorks 2006). Baker (2006) reviewed the evidence for natural fire regimes in sagebrush ecosystems and concluded that fire exclusion (whether due to grazing or fire suppression) probably has had little effect on vegetation trends in most sagebrush systems because of natural fire-return intervals that are likely to be much longer than commonly assumed. In a study conducted on the Monument, Harris et al. (2003) found significantly higher sagebrush:grass ratios in a grazed area relative to a comparable area on an ungrazed mesa top (both associated with the Upland Loam [mountain big sagebrush] ecological site), suggesting that livestock grazing has played a role in increasing shrub:grass ratios in some settings.

No matter the cause, increases in shrub density can be accompanied by a greater concentration of soil impacts in interspaces among shrubs if such areas are used by livestock and/or large numbers of mule deer (*Odocoileus hemionus*). In many sagebrush-dominated areas associated with the Semidesert Loam ecological site in the Monument, trampling of interspaces has resulted in erosion and the loss of relatively sandy surface horizons, the exposure of relatively fine-textured subsurface horizons, and the subsequent development of “playettes” (Eckert et al. 1986) with vesicular structure (M. Miller, personal observation, August 2001). Interspace playettes have been reported for sagebrush settings elsewhere (Eckert et al. 1986; Pierson et al. 1994), and their presence can indicate altered hydrologic functioning (i.e., transition from

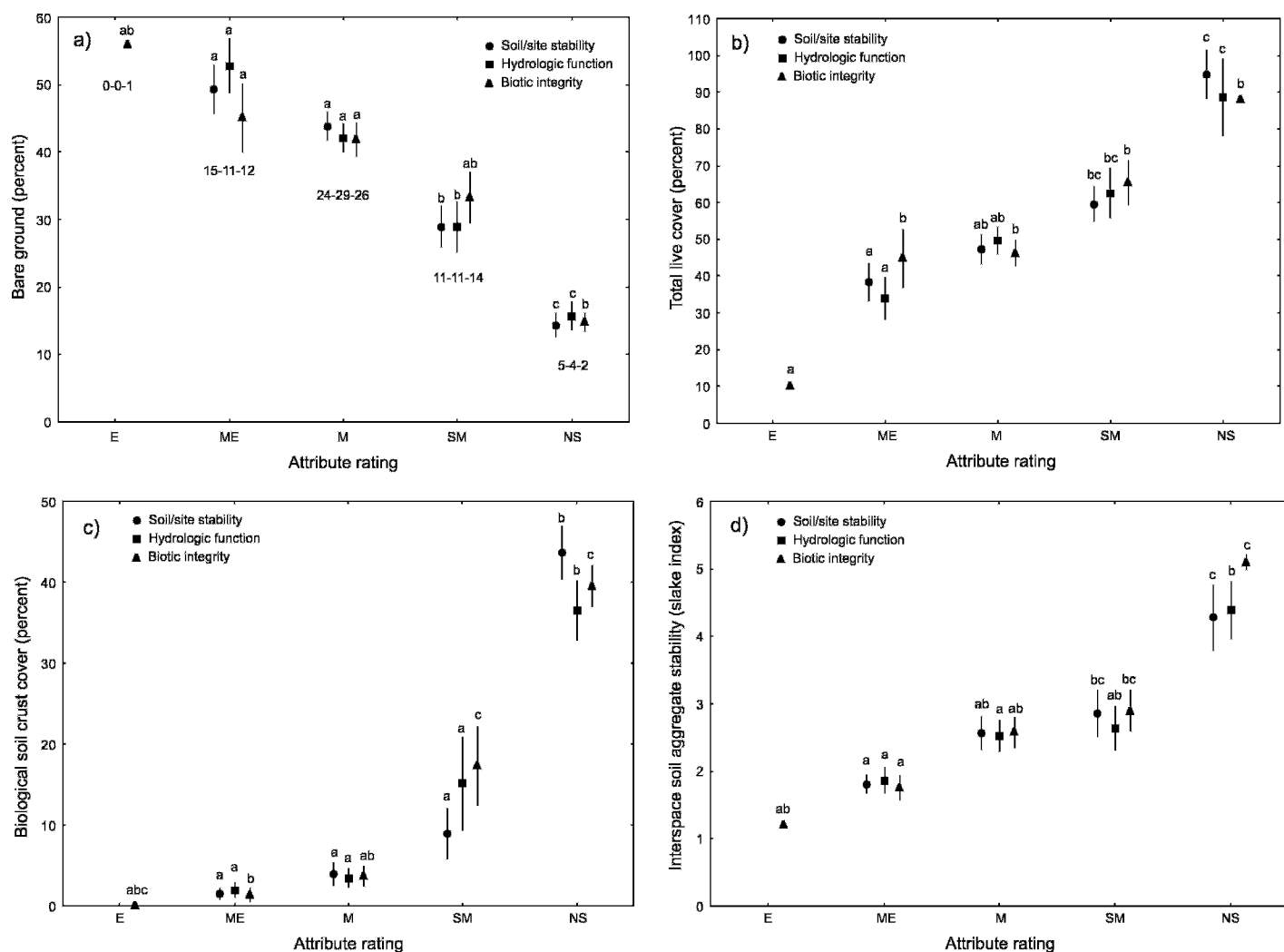


Figure 3. Relations between qualitative ratings assigned for rangeland health attributes (soil/site stability, hydrologic function, and biotic integrity) and quantitative measures (means \pm 1 SE) of **a**, percent bare ground; **b**, percent total live cover; **c**, percent biological soil crust cover; and **d**, interspace soil aggregate stability for 55 Semidesert Loam rangeland health assessments, Grand Staircase–Escalante National Monument. In **a**, numbers indicate sample sizes for multivariate analysis of variance and numbers of assessments that received particular ratings for particular attributes. For each quantitative measure and rangeland health attribute, means annotated with the same letter (a–d) are not significantly different. (Attribute ratings: E indicates extreme departure; ME, moderate to extreme departure; M, moderate departure; SM, slight to moderate departure; and NS, no departure to slight departure from expected reference conditions.)

infiltration to runoff generation; Pierson et al. 1994), accelerated erosion, and diminished potential for seedling establishment (Eckert et al. 1986). All of these were factors that contributed to low ratings for the three attributes of rangeland health.

Assessment results for big sagebrush ecological sites contrast with those for several ecological sites characterized by grassland physiognomic structure (Desert Sandy Loam [fourwing saltbush], Semidesert Sand [fourwing saltbush], Semidesert Sandy Loam [black grama], and Semidesert Sandy Loam [fourwing saltbush]). These grassland sites also tend to receive preferential use by livestock in the Monument because of high levels of forage production relative to production of unpalatable woody plants, but they all had lower frequencies of low rangeland health ratings than all of the big sagebrush ecological sites except the Upland Sand site (Table 4). This result may be due to the fact that these grassland ecological sites differ from many other semiarid grasslands (e.g., Van Auken 2000) in that

they generally lack unpalatable, long-lived woody plants that have the capacity to increase and become persistent site dominants due to succession, absence of fire, or selective herbivory by livestock. In some settings where palatable shrubs such as winterfat (*Ceratoides lanata* [Pursh] J.T. Howell) and fourwing saltbush (*Atriplex canescens* [Pursh] Nuttall) are major components in these ecological sites, moderate livestock grazing actually tends to maintain grassland physiognomic structure whereas release from grazing can result in conversion to shrubland structure (Rasmussen and Brotherson 1986; Floyd et al. 2003).

Soil Texture. Among the five big sagebrush ecological sites, assessment results varied systematically in relation to soil texture. Sagebrush sites primarily associated with fine-loamy soils (seeded Upland Loam and seeded and untreated Semidesert Loam) had higher frequencies of assessments with low

ratings for all rangeland health attributes than sites primarily associated with coarse-loamy (Loamy Bottom and Semidesert Sandy Loam [Wyoming big sagebrush]) or sandy (Upland Sand) soils (Table 4; soil textural family classes from NRCS 2005). Livestock grazing and trampling can have adverse impacts on rangeland hydrologic processes and erosion where they cause reductions in ground cover, soil aggregate stability, soil structure, and soil-surface roughness (Thurrow 1991; Spaeth et al. 1996; Ward and Trimble 2004). Assessment results reported here for sagebrush ecological sites are consistent with Walker's (2002) proposition that relatively sandy soils are inherently more resistant to livestock impacts on hydrologic processes than soils with lots of silt and clay because infiltration rates are inherently greater in relatively sandy soils. Grassland ecological sites in the Monument also are characterized by coarse-loamy or sandy soils, thus this same soil-hydrologic principle may have contributed to the finding that these sites had relatively low frequencies of low ratings for all three attributes of rangeland health.

Management. Seeded areas associated with the two sagebrush ecological sites on fine-loamy soils had the highest frequencies of low ratings for all three attributes of rangeland health (Table 4). This suggests that past vegetation treatments associated with these two ecological sites generally have not provided long-term ecological benefits compared with untreated areas, although without further research it is difficult to know the relative degree to which degraded conditions in seedings are attributable to pretreatment land uses, long-term effects of mechanical treatments themselves, or posttreatment management. However, it is likely that interactions between soil properties and posttreatment management played a role in the development of poor rangeland-health conditions documented in Semidesert Loam and Upland Loam seedings on the Monument.

Allotment management plans in the past typically have allowed higher levels of forage utilization by livestock in seedings than in comparable untreated areas (P. Chapman, personal communication, June 2007), largely because nonnative forage grasses such as *A. cristatum* are more tolerant of heavy grazing than some native grasses (e.g., Richards and Caldwell 1985). This high-use management strategy inadvertently may have contributed to the relatively degraded conditions found in seedings because of the inherent sensitivity of fine-loamy soils to adverse hydrologic changes, as well as their susceptibility to compaction caused by trampling or other compressive forces (Hillel 1998). Of the ecological sites in Table 4, the seeded Upland Loam, untreated Semidesert Loam, and seeded Semidesert Loam sites had the highest frequencies of assessments with low ratings (moderate or greater departure from reference conditions) for soil compaction (35.0%, 22.5%, and 20.8%, respectively), which is one of the four qualitative indicators that applies to all three attributes of rangeland health (Pellant et al. 2000; Pyke et al. 2002). On the Monument, the typical seasons of livestock use are winter and spring (when soils are most likely to be moist and thus most susceptible to compaction) for the Semidesert Loam site and summer and fall for the Upland Loam site. Because of elevational differences, winter mule deer use of the Semidesert Loam ecological site also tends to be greater than that of the Upland Loam ecological site. Drier soils during summer and fall use may

explain why low ratings for soil compaction were less frequent (13.3%) for untreated Upland Loam assessments than for untreated Semidesert Loam assessments.

Patterns Within the Semidesert Loam Ecological Site

Multivariate Gradients in Ecosystem Condition. Analyses of quantitative data collected during assessments of the Semidesert Loam (Wyoming big sagebrush) ecological site describe two multivariate gradients in ecosystem condition (Fig. 2a). Inter-space soil aggregate stability and BSC cover tended to vary independently of total plant cover, functional-group richness and diversity, and percentage of bare ground (Fig. 2a). These results support approaches to rangeland assessment and monitoring that focus on multiple indicators of soil stability, hydrologic function, and biotic integrity rather than on plant community composition alone (Pellant et al. 2000, 2005; Herrick et al. 2005). Soil aggregate stability is related to several ecosystem processes associated with concepts of soil quality and rangeland health including erosion resistance, infiltration capacity, and soil biotic activity (Herrick et al. 1999, 2001). Likewise, BSCs are important contributors to soil stability (Belnap 1995; Williams et al. 1995a, 1995b), nutrient cycling (Evans and Lange 2003), and biological diversity (Rosentreter and Belnap 2003). Because soil-surface roughness increases residence time of runoff on hillslopes (Ward and Trimble 2004), roughness attributable to well-developed BSCs also has been cited as a factor that can enhance runoff retention and infiltration relative to comparable soils without well-developed BSCs (Belnap 2003; Warren 2003). This provides strong rationale for including BSCs (abundance, spatial continuity, and degree of roughness) as indicators of hydrologic functioning for ecological sites with high BSC potential.

Consistent with results of Bowker et al. (2006), data reported here (Fig. 3c) indicate the high BSC potential of soils associated with the Semidesert Loam ecological site. Three distinct soils (Barx series; Progresso series, cool phase; and Ruinpoint series) were found to have BSC cover greater than 40%, with maximum BSC cover of 56% on the Barx series, which is the dominant soil associated with this ecological site in the Monument. Because of the hydrologic sensitivity and high BSC potential of fine-loamy soils associated with this ecological site, the functional significance of BSCs for runoff retention and erosion resistance is particularly high. The steep decline in mean BSC cover between assessment locations assigned NS ratings and those assigned SM ratings for the three attributes of rangeland health (Fig. 3c) also indicates the low resistance and resilience of well-developed BSCs to disturbance (Belnap and Eldridge 2003). In combination, these factors suggest that BSC loss and the degradation of hydrologic and soil-stabilization functions performed by BSCs on fine-loamy soils likely played a role in the development of poor rangeland-health conditions documented for this ecological site.

Relations Between Quantitative and Qualitative Data. Quantitative data exhibited a large degree of variability among Semidesert Loam locations that were assigned the same qualitative ratings by assessment teams (Figs. 2b–2d). Some of this variability probably reflects the fact that ratings for the three qualitative attributes were based on suites of multiple

indicators, several of which are difficult to measure and thus were not addressed by the quantitative sampling (Pellant et al. 2000; Pyke et al. 2002). Accordingly, variations in the status of indicators that were evaluated solely on a qualitative basis could have caused variations in rangeland-health ratings among assessment locations that might have been similar with respect to the quantitative variables.

It is also probable that the assessment process was not as consistent as it might have been had qualitative ratings been linked more explicitly with the quantitative data. Although quantitative data certainly were useful during the assessment process, they would have been more effective in improving assessment consistency on a real-time basis if thresholds between rating classes (NS, SM, M, ME, and E) were defined by ranges in values for one or more quantitative variables. The reference worksheet included in version 4 of the IIRH technique (Pellant et al. 2005) is a significant improvement that seeks to establish such a quantitative framework for rating indicators. This approach will work well for indicators that are easily quantified (e.g., percentage of bare ground) but will be less effective for indicators that are difficult to quantify (e.g., amount and distribution of overland flow paths; Table 2). Ideally, quantitative rating frameworks would be developed through process-based studies conducted on an ecological-site basis, but resources are insufficient to support this work for more than a small number of rangeland ecological sites. An alternative is to develop quantitative rating frameworks for specific ecological sites on the basis of existing, published research and through the use of standardized sampling techniques (e.g., Herrick et al. 2005) to acquire regional data sets describing ranges of variability across gradients of land use and condition, including sites heavily impacted by human activities as well as relatively unimpacted reference sites (Whitford 1998; Tongway and Hindley 2004). Quantitative data describing ecosystem-specific condition gradients (e.g., Figs. 2a and 3; Bosch and Kellner 1991) would be of utility to a wide range of institutions and stakeholders involved in assessment, monitoring, and sustainable management of rangeland ecosystems (e.g., Parrish et al. 2003), as well as to scientists engaged in related research activities (Herrick et al. 2006; Vavra and Brown 2006). The absence of such contextual data sets constrains the interpretation of data from moment-in-time ecological assessments, whether based on qualitative or quantitative techniques.

Additional Lessons Learned From Application of the Technique

As applied in this project, the IIRH technique had two important and related strengths. First, it was effective in broadening many practitioners' perspectives concerning the number and types of ecological attributes encompassed by the notion of "rangeland health." Staff who had previously focused primarily on key forage species or measures of plant community composition became attuned to soil and hydrologic processes and their importance for evaluating the status of rangeland ecosystems. Second, the technique proved valuable as a tool for facilitating discussion among diverse practitioners and stakeholders about ecological processes in rangelands.

Four factors would improve application of the IIRH technique relative to its application in this project. As discussed

above, consistency would be improved by greater integration of quantitative data in the assessment technique. Second, a probabilistic sampling design (e.g., Theobald et al. 2007) would enable spatial analyses and inferences not possible with the judgment-based design used in this project. Third, the prominence of soil and hydrologic indicators in the IIRH technique calls for practitioners to have greater professional knowledge of these topics. Soil expertise is lacking in most BLM field offices (B. Ypsilantis, personal communication, July 2007), and a trained soil scientist participated in only 7 of 507 assessments in this project. As a consequence, it is probable that there was a tendency for assessment teams to understate the degree to which particular soil indicators (e.g., soil instability, soil surface degradation, and compaction) were expressed across the project area. Finally, conceptual models of ecosystem dynamics (e.g., Bestelmeyer et al. 2004) need to play a stronger, more explicit role in the assessment process to enhance the information content of assessment results and thus their value for informing the development of effective strategies for management and restoration (Briske et al. 2005; Herrick et al. 2006; Hobbs 2007).

MANAGEMENT IMPLICATIONS

The qualitative IIRH technique used in this project yielded meaningful data regarding the status of three ecosystem attributes (soil/site stability, hydrologic function, and biotic integrity) and how the status of these attributes varied among and within a large number of ecological sites across a 760 000-ha landscape. Patterns among ecological sites in terms of the frequency of assessments with low ratings for all three attributes appear attributable to several interacting factors including 1) potential primary production and long-term exposure to production-dependent land-use activities such as livestock grazing; 2) the presence of unpalatable woody plants that have the capacity to increase and become persistent site dominants due to selective herbivory, absence of fire, or succession; 3) soil texture through effects on hydrologic responses to grazing, trampling, and other disturbances; and 4) past management that resulted in high livestock use of ecological sites with sensitive fine-loamy soils following treatments designed to increase forage availability. In particular, results indicate that big sagebrush ecological sites with relatively high production potential had high frequencies of assessments with low ratings for all three ecosystem attributes, whereas shallow-soil ecological sites with relatively low production potential and the presence of Utah juniper and/or Colorado pinyon had low frequencies of assessments with low ratings for all three attributes. Areas where fine-loamy big sagebrush ecological sites were seeded in the past to increase livestock forage were characterized by frequencies of low rangeland health ratings that were higher than or similar to comparable untreated areas, suggesting that these treatments have not provided long-term ecological benefits relative to untreated areas. For seeded areas, it is likely that interactions between soil properties and posttreatment management played a role in the development of poor rangeland-health conditions documented by assessments. These results—that sites with the greatest production potential tended to be the most degraded,

and that net effects of past management treatments have not been ecologically beneficial—suggest that ongoing management, restoration treatments, and posttreatment management of these ecological sites should be tailored to account for their sensitivity to degradation.

ACKNOWLEDGMENTS

More than 50 BLM, NPS, and NRCS staff assisted with planning and implementation of this project. Major contributions were made by Laura Fertig, Walter Fertig, Dennis Pope, Doug Powell, Mary Lou Zimmerman, Bill Falvey, Allan Bate, Sean Stewart, Rick Oyler, Harry Barber, Jeff Chynoweth, Sarah Yarborough, Alan Titus, Kezia Nielsen, Chris Killingsworth, Gregg Christensen, Andrew Dubrasky, Cory Black, Dan Yarborough, and Kent Sutcliffe. Former Monument managers Kate Cannon and Dave Hunsaker were instrumental in the initiation and completion of this effort. Jayne Belnap, Brandon Bestelmeyer, Matthew Bowker, Marietta Eaton, Walter Fertig, Sam Fuhlendorf, Jeffrey Herrick, David Pyke, Roger Rosentreter, and two anonymous reviewers provided comments that improved the quality of the manuscript.

LITERATURE CITED

- BAKER, W. L. 2006. Fire and restoration of sagebrush ecosystems. *Wildlife Society Bulletin* 34:177–185.
- BELNAP, J. 1995. Surface disturbances: their role in accelerating desertification. *Environmental Monitoring and Assessment* 37:39–57.
- BELNAP, J. 2003. Comparative structure of physical and biological soil crusts. *In*: J. Belnap and O. L. Lange [EDS.]. *Biological soil crusts: structure, function, and management*. 2nd ed. Berlin, Germany: Springer-Verlag. p. 177–191.
- BELNAP, J., AND D. J. ELDRIDGE. 2003. Disturbance and recovery of biological soil crusts. *In*: J. Belnap and O. L. Lange [EDS.]. *Biological soil crusts: structure, function, and management*. 2nd ed. Berlin, Germany: Springer-Verlag. p. 363–383.
- BELNAP, J., J. R. WELTER, N. B. GRIMM, N. BARGER, AND J. A. LUDWIG. 2005. Linkages between microbial and hydrologic processes in arid and semiarid watersheds. *Ecology* 86:298–307.
- BESTELMEYER, B. T., J. E. HERRICK, J. R. BROWN, D. A. TRUJILLO, AND K. M. HAVSTAD. 2004. Land management in the American Southwest: a state-and-transition approach to ecosystem complexity. *Environmental Management* 34:38–51.
- BOSCH, O. J. H., AND K. KELLNER. 1991. The use of a degradation gradient for the ecological interpretation of condition assessment in the western grassland biome of southern Africa. *Journal of Arid Environments* 21:21–29.
- BOWKER, M. A., J. BELNAP, AND M. E. MILLER. 2006. Spatial modeling of biological soil crusts to support rangeland assessment and monitoring. *Rangeland Ecology and Management* 59:519–529.
- BRADLEY, M. S. 1999. A history of Kane County. Salt Lake City, UT, USA: Utah State Historical Society. 380 p.
- BRISKE, D. D., S. D. FUHLENDORF, AND F. E. SMEINS. 2005. State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. *Rangeland Ecology and Management* 58:1–10.
- CINGOLANI, A. M., I. NOY-MEIR, AND S. DIAZ. 2005. Grazing effects on rangeland diversity: a synthesis of contemporary models. *Ecological Applications* 15:757–773.
- CONNELLY, J. W., S. T. KNICK, M. A. SCHROEDER, AND S. J. STIVER. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. Available at: [http://sagemap.wr.usgs.gov/Docs/Greater Sage-grouse Conservation Assessment 060404.pdf](http://sagemap.wr.usgs.gov/Docs/Greater_Sage-grouse_Conservation_Assessment_060404.pdf). Accessed 25 March 2008.
- COULLOUDON, B., K. ESHELMAN, J. GIANOLA, N. HABICH, L. HUGHES, C. JOHNSON, M. PELLANT, P. PODBORNY, A. RASMUSSEN, B. ROBLES, P. SHAVER, J. SPEHAR, AND J. W. WILLOUGHBY. 1999. Sampling vegetation attributes. Denver, CO, USA: US Department of the Interior, Bureau of Land Management, National Science and Technology Center, Interagency Technical Reference 1734-4. 163 p.
- DALY, C., R. P. NEILSON, AND D. L. PHILLIPS. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. *Journal of Applied Meteorology* 33:140–158.
- DE SOYZA, A. G., J. W. VAN ZEE, W. G. WHITFORD, A. NEALE, N. TALLENT-HALLSEL, J. E. HERRICK, AND K. M. HAVSTAD. 2000a. Indicators of Great Basin rangeland health. *Journal of Arid Environments* 45:289–304.
- DE SOYZA, A. G., W. G. WHITFORD, AND A. R. JOHNSON. 2000b. Assessing and monitoring the health of western rangeland watersheds. *Environmental Monitoring and Assessment* 64:153–166.
- DIAZ, S., D. D. BRISKE, AND S. MCINTYRE. 2002. Range management and plant functional types. *In*: A. C. Grice and K. C. Hodgkinson [EDS.]. *Global rangelands: progress and prospects*. Wallingford, UK: CABI Publishing. p. 81–100.
- DOELLING, H. H., R. E. BLACKETT, A. H. HAMBLIN, J. D. POWELL, AND G. L. POLLOCK. 2000. Geology of Grand Staircase–Escalante National Monument, Utah. *In*: D. A. Sprinkel, T. C. Chidsey, Jr., and P. B. Anderson [EDS.]. *Geology of Utah's parks and monuments*. Salt Lake City, UT, USA: Utah Geological Association. p. 189–231.
- ECKERT, R. E., JR., F. F. PETERSON, M. S. MEURISSE, AND J. L. STEPHENS. 1986. Effects of soil-surface morphology on emergence and survival of seedlings in big sagebrush communities. *Journal of Range Management* 39:414–420.
- EVANS, R. D., AND O. L. LANGE. 2003. Biological soil crusts and ecosystem nitrogen and carbon dynamics. *In*: J. Belnap and O. L. Lange [EDS.]. *Biological soil crusts: structure, function, and management*. 2nd ed. Berlin, Germany: Springer-Verlag. p. 263–279.
- FLOYD, M. L., T. L. FLEISCHNER, D. D. HANNA, AND P. WHITEFIELD. 2003. Effects of historic livestock grazing on vegetation at Chaco Culture National Historic Park, New Mexico. *Conservation Biology* 17:1703–1711.
- HARRIS, T. A., G. P. ASNER, AND M. E. MILLER. 2003. Changes in vegetation structure after long-term grazing in pinyon–juniper ecosystems: integrating imaging spectroscopy and field studies. *Ecosystems* 6:368–383.
- HERRICK, J. E., B. T. BESTELMEYER, S. ARCHER, A. J. TUGEL, AND J. R. BROWN. 2006. An integrated framework for science-based arid land management. *Journal of Arid Environments* 65:319–335.
- HERRICK, J. E., J. W. VAN ZEE, K. M. HAVSTAD, L. M. BURKETT, AND W. G. WHITFORD. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. Volume I: quick start. Las Cruces, NM, USA: USDA-ARS Jornada Experimental Range. 36 p.
- HERRICK, J. E., M. A. WELTZ, J. D. REEDER, G. E. SCHUMAN, AND J. R. SIMANTON. 1999. Rangeland soil erosion and soil quality: role of soil resistance, resilience, and disturbance regime. *In*: R. Lal [ED.]. *Soil erosion and soil quality*. Boca Raton, FL, USA: CRC Press. p. 209–233.
- HERRICK, J. E., W. G. WHITFORD, AND M. WALTON. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. *Catena* 44:27–35.
- HILLEL, D. 1998. Environmental soil physics. San Diego, CA, USA: Academic Press. 771 p.
- HOBBS, R. J. 2007. Setting effective and realistic restoration goals: key directions for research. *Restoration Ecology* 15:354–357.
- HUNT, C. B. 1974. Natural regions of the United States and Canada. San Francisco, CA, USA: W. H. Freeman. 725 p.
- HUSTON, M. 1979. A general hypothesis of species diversity. *The American Naturalist* 113:81–101.
- KNICK, S. T., D. S. DOBKIN, J. T. ROTENBERRY, M. A. SCHROEDER, W. M. VANDER HAEGEN, AND C. VAN RIPER III. 2003. Teetering on the edge or too late: conservation and research issues for avifauna of sagebrush habitats. *The Condor* 105: 611–634.
- LUDWIG, J. A., G. N. BASTIN, V. H. CHEWINGS, R. W. EAGER, AND A. C. LIEDLOFF. 2007. Leakiness: a new index for monitoring the health of arid and semiarid landscapes using remotely sensed vegetation cover and elevation data. *Ecological Indicators* 7:442–454.
- LUDWIG, J. A., R. W. EAGER, G. N. BASTIN, V. H. CHEWINGS, AND A. C. LIEDLOFF. 2002. A leakiness index for assessing landscape function using remote sensing. *Landscape Ecology* 17:157–171.

- LUNT, I. D., D. J. ELDRIDGE, J. W. MORGAN, AND G. B. WITT. 2007. Turner Review No. 13. A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia. *Australian Journal of Botany* 55:401–415.
- MCCUNE, B. P., AND J. B. GRACE. 2002. Analysis of ecological communities. Gleneden Beach, OR, USA: MJM Software Design. 300 p.
- MILCHUNAS, D. G., O. E. SALA, AND W. K. LAUENROTH. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *The American Naturalist* 132:87–106.
- MILLER, R. F., T. J. SVEJCAR, AND N. E. WEST. 1994. Implications of livestock grazing in the Intermountain sagebrush region: plant composition. In: M. Vavra, W. A. Laycock, and R. D. Pieper [EDS.]. Ecological implications of livestock herbivory in the West. Denver, CO, USA: Society for Range Management. p. 101–146.
- [NRC] NATIONAL RESEARCH COUNCIL. 1994. Rangeland health: new methods to classify, inventory, and monitor rangelands. Washington, DC, USA: National Academy Press. 180 p.
- [NRCS] USDA NATURAL RESOURCES CONSERVATION SERVICE. 2003. National range and pasture handbook. Revision 1. Washington, DC, USA: US Department of Agriculture, Natural Resources Conservation Service. 575 p.
- [NRCS] USDA NATURAL RESOURCES CONSERVATION SERVICE. 2005. Soil survey of Grand Staircase–Escalante National Monument area, parts of Kane and Garfield counties, Utah. Salt Lake City, UT, USA: US Department of Agriculture, Natural Resources Conservation Service. 577 p.
- PARRISH, J. D., D. P. BRAUN, AND R. S. UNNASCH. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. *BioScience* 53:851–860.
- PELLANT, M., P. L. SHAVER, D. A. PYKE, AND J. E. HERRICK. 2000. Interpreting indicators of rangeland health. Version 3. Denver, CO, USA: US Department of the Interior, Bureau of Land Management, Interagency Technical Reference TR-1734-6. 118 p.
- PELLANT, M., P. L. SHAVER, D. A. PYKE, AND J. E. HERRICK. 2005. Interpreting indicators of rangeland health. Version 4. Denver, CO, USA: US Department of the Interior, Bureau of Land Management, Interagency technical reference TR-1734-6. 122 p.
- PIERSON, F. B., W. H. BLACKBURN, S. S. VAN VACTOR, AND J. C. WOOD. 1994. Partitioning small scale spatial variability of runoff and erosion on sagebrush rangeland. *Water Resources Bulletin* 30:1081–1089.
- PYKE, D. A., AND J. E. HERRICK. 2003. Transitions in rangeland evaluations: a review of the major transitions in rangeland evaluations during the last 25 years and speculation about future evaluations. *Rangelands* 25:22–30.
- PYKE, D. A., J. E. HERRICK, P. L. SHAVER, AND M. PELLANT. 2002. Rangeland health attributes and indicators for qualitative assessment. *Journal of Range Management* 55:584–597.
- RASMUSSEN, L. L., AND J. D. BROTHERTON. 1986. Response of winterfat (*Ceratoides lanata*) communities to release from grazing pressure. *Great Basin Naturalist* 46:148–156.
- RICHARDS, J. H., AND M. M. CALDWELL. 1985. Soluble carbohydrates, concurrent photosynthesis and efficiency in regrowth following defoliation: a field study with *Agropyron* species. *Journal of Applied Ecology* 22:970–920.
- ROSENTRER, R., AND J. BELNAP. 2003. Biological soil crusts of North America. In: J. Belnap and O. L. Lange [EDS.]. Biological soil crusts: structure, function, and management. 2nd ed. Berlin, Germany: Springer-Verlag. p. 31–50.
- ROSENTRER, R., AND D. J. ELDRIDGE. 2002. Monitoring biodiversity and ecosystem function: grasslands, deserts, and steppe. In: P. L. Nimis, C. Scheidegger, and P. A. Wolseley [EDS.]. Monitoring with lichens—monitoring lichens. Dordrecht, Netherlands: Kluwer. p. 223–237.
- SPAETH, K. E., T. L. THURLOW, T. H. BLACKBURN, AND F. B. PIERSON. 1996. Ecological dynamics and management effects on rangeland hydrologic processes. In: K. E. Spaeth, F. B. Pierson, M. A. Weltz, and R. G. Hendricks [EDS.]. Grazingland hydrology issues: perspectives for the 21st century. Denver, CO, USA: Society for Range Management. p. 25–51.
- [SRM TASK GROUP] SOCIETY FOR RANGE MANAGEMENT TASK GROUP ON UNITY IN CONCEPTS AND TERMINOLOGY COMMITTEE. 1995. New concepts for assessment of rangeland condition. *Journal of Range Management* 48:271–282.
- STATSOFT [computer program]. 2004. STATISTICA. Version 6. Available at: <http://www.statsoft.com>. Accessed 7 March 2008.
- THEOBALD, D. M., D. STEVENS, D. WHITE, N. S. URQUHART, A. OLSEN, AND J. NORMAN. 2007. Using GIS to generate spatially balanced random survey designs for natural resource applications. *Environmental Management* 40:134–146.
- THURLOW, T. L. 1991. Hydrology and erosion. In: R. K. Heitschmidt and J. W. Stuth [EDS.]. Grazing management: an ecological perspective. Portland, OR, USA: Timber Press. p. 141–159.
- TONGWAY, D. J., AND N. L. HINDLEY. 2004. Landscape function analysis: procedures for monitoring and assessing landscapes. Canberra, Australia: CSIRO Sustainable Ecosystems. 82 p.
- USDA FOREST SERVICE. 1937. Range plant handbook. Washington, DC, USA: US Department of Agriculture, Forest Service. 844 p.
- VAN AUKEN, O. W. 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and Systematics* 31:197–215.
- VAVRA, M., AND J. BROWN. 2006. Rangeland research: strategies for providing sustainability and stewardship to the rangelands of the world. *Rangelands* 28:7–14.
- WALKER, B. H. 2002. Ecological resilience in grazed rangelands. In: L. H. Gunderson and L. Pritchard, Jr. [EDS.]. Resilience and the behavior of large-scale systems. Washington, DC, USA: Island Press. p. 183–193.
- WARD, A. D., AND S. W. TRIMBLE. 2004. Environmental hydrology. 2nd ed. Boca Raton, FL, USA: Lewis Publishers. 475 p.
- WARREN, S. D. 2003. Synopsis: influence of biological soil crusts on arid land hydrology and soil stability. In: J. Belnap and O. L. Lange [EDS.]. Biological soil crusts: structure, function, and management. 2nd ed. Berlin, Germany: Springer-Verlag. p. 349–360.
- WELCH, B. L. 2005. Big sagebrush: a sea fragmented into lakes, ponds, and puddles. Fort Collins, CO, USA: US Department of Agriculture, Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-144. 210 p.
- WELSH, S. D., N. D. ATWOOD, S. GOODRICH, AND L. C. HIGGINS [EDS.]. 2003. A Utah flora. 3rd ed., revised. Provo, UT, USA: Brigham Young University. 912 p.
- WEST, N. E., AND T. P. YORKS. 2006. Long-term interactions of climate, productivity, species richness, and growth form in relictual sagebrush steppe plant communities. *Western North American Naturalist* 66:502–526.
- WESTERN REGIONAL CLIMATE CENTER. 2007. Western Regional Climate Center. Available at: <http://www.wrcc.dri.edu>. Accessed 27 March 2007.
- WHITFORD, W. G. 1998. Validation of indicators. In: D. J. Rapport, R. Costanza, P. R. Epstein, C. Gaudet, and R. Levins [EDS.]. Ecosystem health. Malden, MA, USA: Blackwell Science. p. 205–209.
- WHITFORD, W. G., A. G. de SOYZA, J. W. VAN ZEE, J. E. HERRICK, AND K. M. HAVSTAD. 1998. Vegetation, soil and animal indicators of rangeland health. *Environmental Monitoring and Assessment* 51:179–200.
- WILLIAMS, J. D., J. P. DOBROWOLSKI, D. A. GILLETTE, AND N. E. WEST. 1995a. Microphytic crust influence on wind erosion. *Transactions of the American Society of Agricultural Engineers* 38:131–137.
- WILLIAMS, J. D., J. P. DOBROWOLSKI, AND N. E. WEST. 1995b. Microphytic crust influence on interrill erosion and infiltration capacity. *Transactions of the American Society of Agricultural Engineers* 38:139–146.
- ZAR, J. H. 1999. Biostatistical analysis. 4th ed. Upper Saddle River, NJ, USA: Prentice Hall. 663 p.

Alternative states of a semiarid grassland ecosystem: implications for ecosystem services

MARK E. MILLER,^{1,4,†} R. TRAVIS BELOTE,^{2,5} MATTHEW A. BOWKER,² AND STEVEN L. GARMAN³

¹U.S. Geological Survey, Southwest Biological Science Center, 190 E. Center Street, Kanab, Utah 84741 USA

²U.S. Geological Survey, Southwest Biological Science Center, Box 5614, Northern Arizona University, Flagstaff, Arizona 86011 USA

³U.S. Geological Survey, Rocky Mountain Geographic Science Center, P.O. Box 25046, MS 516 Denver, Colorado 80225 0046 USA

Abstract. Ecosystems can shift between alternative states characterized by persistent differences in structure, function, and capacity to provide ecosystem services valued by society. We examined empirical evidence for alternative states in a semiarid grassland ecosystem where topographic complexity and contrasting management regimes have led to spatial variations in levels of livestock grazing. Using an inventory data set, we found that plots ($n = 72$) cluster into three groups corresponding to generalized alternative states identified in an a priori conceptual model. One cluster (*biocrust*) is notable for high coverage of a biological soil crust functional group in addition to vascular plants. Another (*grass bare*) lacks biological crust but retains perennial grasses at levels similar to the *biocrust* cluster. A third (*annualized bare*) is dominated by invasive annual plants. Occurrence of *grass bare* and *annualized bare* conditions in areas where livestock have been excluded for over 30 years demonstrates the persistence of these states. Significant differences among all three clusters were found for percent bare ground, percent total live cover, and functional group richness. Using data for vegetation structure and soil erodibility, we also found large among cluster differences in average levels of dust emissions predicted by a wind erosion model. Predicted emissions were highest for the *annualized bare* cluster and lowest for the *biocrust* cluster, which was characterized by zero or minimal emissions even under conditions of extreme wind. Results illustrate potential trade offs among ecosystem services including livestock production, soil retention, carbon storage, and biodiversity conservation. Improved understanding of these trade offs may assist ecosystem managers when evaluating alternative management strategies.

Key words: biological soil crusts; *Bromus tectorum*; Colorado Plateau; drylands; dust; ecosystem services; functional groups; livestock grazing; resilience; state and transition model; wind erosion.

Received 28 January 2011; revised 29 March 2011; accepted 15 April 2011; **published** 19 May 2011. Corresponding Editor: J. Morgan.

Citation: Miller, M. E., R. T. Belote, M. A. Bowker, and S. L. Garman. 2011. Alternative states of a semiarid grassland ecosystem: implications for ecosystem services. *Ecosphere* 2(5):art55. doi:10.1890/ES11 00027.1

Copyright: This is an open access article distributed under the terms of the Creative Commons Attribution License, which permits restricted use, distribution, and reproduction in any medium, provided the original author and sources are credited.

⁴ Present address: National Park Service, Southeast Utah Group, 2282 Southwest Resource Boulevard, Moab, Utah 84532 USA.

⁵ Present address: Research Department, The Wilderness Society, 503 West Mendenhall Street, Bozeman, Montana 59715 3450 USA.

† E-mail: mark e miller@nps.gov

INTRODUCTION

Ecosystems can shift between alternative states or dynamic regimes that are characterized by persistent differences in structure and function

(Beisner et al. 2003, Scheffer and Carpenter 2003, Mayer and Rietkerk 2004). Such shifts are caused by factors that independently or interactively trigger relatively major changes in functional group structure, disturbance regimes, and/or

resource regimes (Chapin et al. 1996). In the context of ecosystem management, alternative states are of concern for two primary reasons. First, shifts between alternative states may occur as relatively abrupt, nonlinear responses to factors such as climate and human land use (Scheffer and Carpenter 2003, Briske 2006). The potential for abrupt changes in ecosystem properties generates a high degree of uncertainty and unpredictability in management (Holling 1996). Second, alternative states invariably differ from one another in their capacity to provide ecosystem services and support different management objectives. Once a state shift has occurred, restoration of previous conditions and management options may be difficult, costly, or effectively impossible (Whisenant 1999, Suding and Hobbs 2009).

Two major research themes have developed around the phenomenon of alternative ecosystem states. The first has focused on biotic and abiotic attributes that confer resilience to perturbations and thus, reduce ecosystem susceptibility to state shifts (Walker 1992, Carpenter et al. 2001). Resilience, defined as the magnitude of perturbation that a system can withstand while maintaining its fundamental structure and function (Holling 1996), is a dynamic property that can change in response to human impacts or climatic conditions (Scheffer and Carpenter 2003). Resilience has become a central concept for work on ecosystem sustainability in the context of global climate change and increasing human pressures on the environment (Chapin et al. 2009). The second theme has focused on thresholds between alternative states, often with an emphasis on predicting thresholds to inform ecosystem management (Westoby et al. 1989, Bestelmeyer 2006, Briske 2006). Challenges in the identification and prediction of threshold behaviors have led to questions regarding the practical applicability of the threshold concept to ecosystem management (Groffman et al. 2006). These same challenges have led to recommendations for greater management emphasis on maintaining resilience of ecosystem states that provide the broadest and most valued range of ecosystem services rather than focusing efforts on the identification of thresholds (Briske et al. 2008).

Alternative states have been described for many types of ecosystems (Folke et al. 2004,

Mayer and Rietkerk 2004), but drylands are among the most susceptible to this phenomenon due to low and variable amounts of precipitation in combination with effects of human land-use activities (Schlesinger et al. 1990, van de Koppel et al. 1997, Reynolds et al. 2007). Published examples of alternative states in drylands represent variations on three common syndromes (Okin et al. 2009). The first is characterized by a persistent increase in the ratio of woody plants to perennial grasses, with woody plant dominance reinforced by feedbacks involving decreased fire frequency and/or the loss or redistribution of soil resources (Schlesinger et al. 1990, Archer et al. 1995). The second is characterized by a persistent shift in dominance from perennial plants to invasive annual plants (especially grasses), often accompanied by a feedback with increased fire frequency (D'Antonio and Vitousek 1992). The third is reflected by feedbacks between soil degradation and a persistent decline in total vegetative cover (van de Koppel et al. 1997).

We examine evidence for the existence of alternative states in a semiarid grassland ecosystem on the Colorado Plateau, USA, where livestock grazing, climate, and invasive annual plants have contributed to persistent changes in ecosystem properties. This ecosystem is characterized by the presence of biological soil crusts (biological crust, hereafter), which are soil-surface assemblages of cyanobacteria, mosses, and lichens that are functionally significant for soil stabilization (Belnap 1995, Warren 2003), nutrient cycling (Evans and Lange 2003), hydrologic processes (Eldridge et al. 2002, Warren 2003), and mediation of vascular plant establishment (Belnap et al. 2003, Escudero et al. 2007). The functional significance of biological crust is countered by its high vulnerability to surface disturbances that can result in long-term reductions of crust structure and functionality (Belnap and Eldridge 2003). In sparsely vegetated drylands, disturbance-induced declines in biological crust often are accompanied by accelerated soil erosion and persistent changes in soil physical and biogeochemical properties (Neff et al. 2005). Dust emitted from unstable drylands also can have downwind impacts on air quality and human health, ecosystem biogeochemistry (Neff et al. 2008), and regional-scale hydrologic processes (Painter et al. 2010). Our objectives were to

(1) validate our a priori conception of possible alternative states using empirical field data collected across a range of conditions and land uses; (2) evaluate the functional outcome of state changes, focusing on modeled potential wind erosion; (3) relate our results to principles of resilience theory; and (4) examine implications for ecosystem services and management.

METHODS

Study area and ecological site

Field studies were conducted in plots distributed throughout a 1500-km² area located on the central Colorado Plateau in southeastern Utah, USA (Fig. 1). Approximately 25% of the study area is located within Canyonlands National Park (CNP), portions of which were grazed by livestock (cattle) from the late 1880s until 1974. The remainder of the study area encompasses the adjacent Indian Creek grazing allotment of the Dugout Ranch, where livestock grazing continues to be the dominant land use. Elevation ranges from 1470 to 2044 m. Ranges of climate variables (from Western Regional Climate Center, <http://www.wrcc.dri.edu>), unless otherwise noted) are as follows: (1) mean annual precipitation (MAP), 210 to 255 mm; (2) mean annual temperature, 10.7 to 12.1°C; and (3) the ratio of MAP to potential evapotranspiration, 0.18 to 0.34 (Flint and Flint 2007; 0.20 is defined as the division between arid and semiarid zones, Reynolds and Stafford Smith 2002).

We used the U.S. Department of Agriculture Natural Resources Conservation Service (USDA NRCS) ecological site system as a framework for landscape stratification and ecosystem classification (Herrick et al. 2006, Bestelmeyer et al. 2009). In this system, ecological sites are differentiated by physical attributes including inherent soil properties (texture, depth, and horizonation), geomorphic setting, and climate, and the potential (rather than current) vegetation associated with these physical attributes within a specific ecoregion (Herrick et al. 2006, Bestelmeyer et al. 2009). Despite the term “ecological site,” they do not correspond to a particular study site or plot on the landscape but rather to a class of land. In this study, we focused on the Semidesert Sandy Loam (SDSL hereafter) ecological site because of its broad spatial extent and high degree of past

and present use for livestock grazing throughout the region. We further restricted our analyses to the Begay soil series (a coarse-loamy, mixed, superactive, mesic Ustic Haplocambid), which is the most common soil attributed to the SDSL site in the region (USDA NRCS 1991). The Begay soil is formed in eolian and alluvial deposits derived from calcareous sandstone and is found in broad valleys and on structural benches with gentle slopes. Surface textures range from fine sandy loams to loamy fine sands, depths range from 100 to over 150 cm, and surface pH is moderately alkaline.

In relatively undisturbed settings, the vascular plant community of the SDSL site is characterized by a mixture of perennial grasses, shrubs, and annual herbaceous species. Common perennial grasses include *Stipa hymenoides* Roemer & Schultes and *S. comata* Trinius & Ruprecht (C₃ bunchgrasses; all nomenclature follows Welsh et al. 2003), *Sporobolus* R. Br. spp. (short-lived C₄ bunchgrasses), and *Hilaria jamesii* (Torrey) Benth and *Bouteloua gracilis* (Humboldt, Bonpland, & Kunth) Lagasca ex Steudel (rhizomatous C₄ grasses). Common shrubs include *Atriplex canescens* (Pursh) Nuttall and *Krascheninnikovia lanata* (Pursh) Meeuse & Smit (both palatable to livestock and may exceed perennial grasses), as well as the subshrub *Gutierrezia sarothrae* (Pursh) Britton & Rusby (unpalatable to livestock). Common exotic annuals include the invasive C₃ grass *Bromus tectorum* L., the invasive C₄ forbs *Salsola tragus* L. and *S. paulsenii* Litvinov, and the C₃ forb *Erodium cicutarium* (L.) L'Hertier. Biological crust (cyanobacterially dominated but containing lichens such as *Collema* and *Placidium*, and mosses such as *Syntrichia*) is an important functional group associated with the SDSL and many other ecological sites on the Colorado Plateau (Bowker and Belnap 2008, Bowker et al. 2008, Miller 2008).

Plots sampled for this study were classified as never grazed, formerly grazed, or currently grazed based on past or current accessibility and evidence of livestock use. However, the relative intensities of past and current grazing use are highly variable spatially due to deep canyons and high sandstone walls that limit livestock movements and access to forage and water.

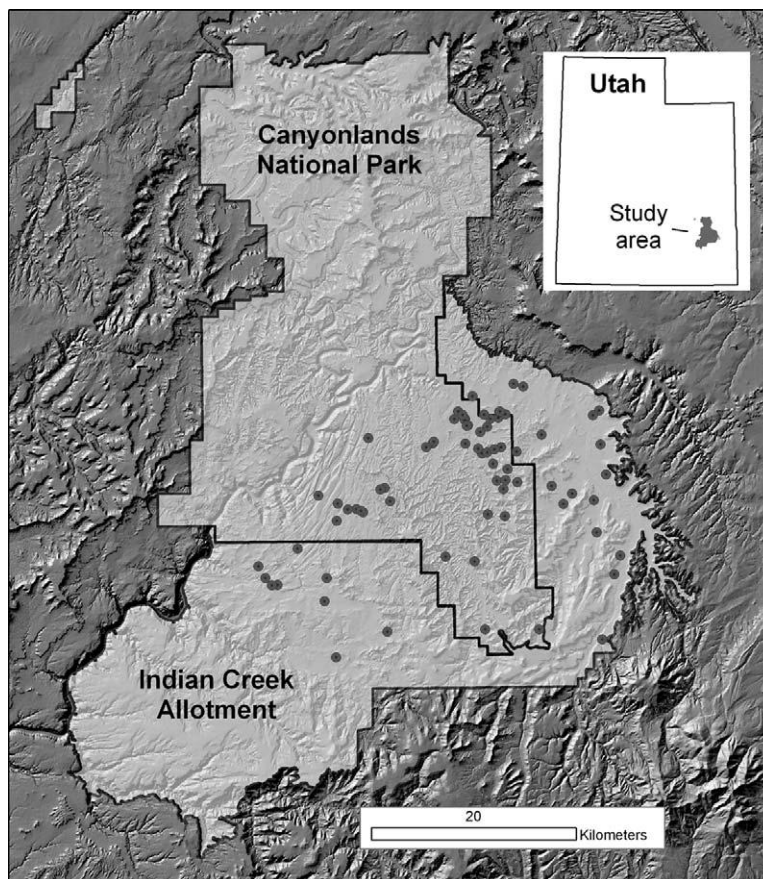


Fig. 1. Map showing the location of the study area in southeastern Utah, USA, and the distribution of plots (points) in Canyonlands National Park and the Indian Creek grazing allotment.

Conceptual model of ecosystem dynamics

We developed an a priori state-and-transition model (STM) describing putative alternative states associated with the SDSL ecological site (Fig. 2). STMs serve to describe alternative states and general processes most likely to have caused state transitions in the past (Westoby et al. 1989). Our model articulates hypotheses about temporal patterns and processes based upon observed patterns of spatial variability. Dynamics and associated processes depicted in the model motivated our selection of particular field measurements, and they provided a framework for our analytical approach. The model asserts the existence of four alternative states (one historical and three extant) and is based on field observations and previous investigations of this ecological site in the study area (Kleiner and Harper 1972, Belnap and Phillips 2001, Neff et al. 2005,

Miller et al. 2006, Belnap et al. 2009). States in our conceptual model are differentiated by the relative abundance of generalized functional groups of biota that differ in their effects on ecosystem processes and in their responses to livestock grazing, surface disturbances, and climate. These three generalized groups consist of biological crust, perennial grasses and shrubs, and invasive annual plants. The specific composition of each of these three groups can vary spatially in relation to elevation and subtle soil-geomorphic properties, and temporally in response to climate and disturbance history. We accommodate this degree of natural variability in the model through our generalized characterization of functional groups, thereby ensuring the plausibility that divergent extant states derive from the same initial conditions.

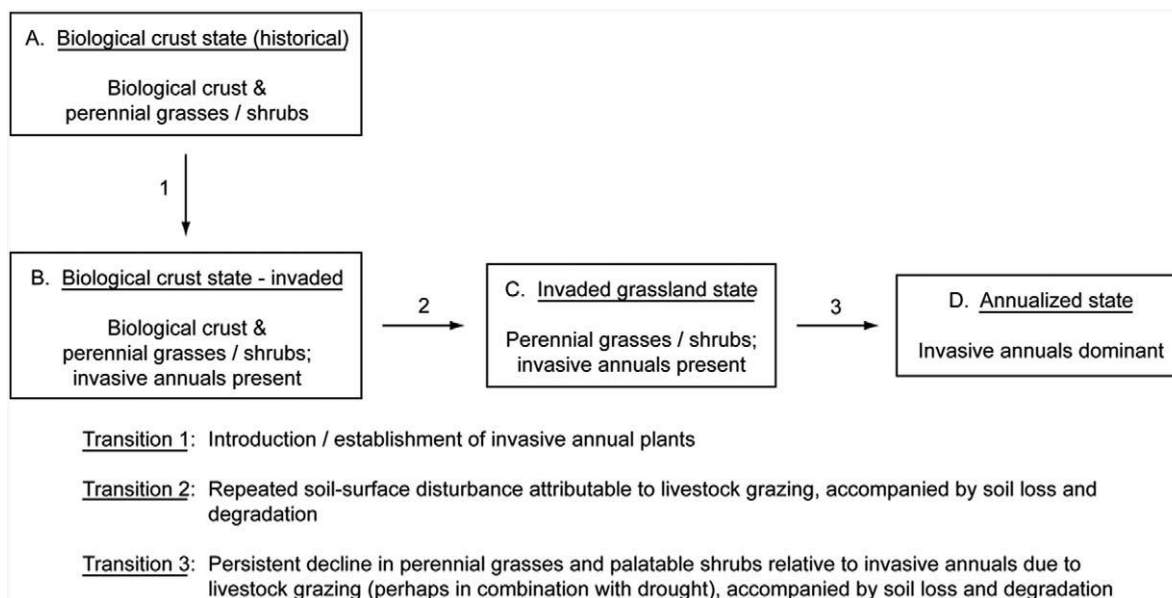


Fig. 2. Conceptual state and transition model for the SDSL ecological site. Boxes A–D represent putative alternative states and numbered arrows reflect hypothesized causal processes responsible for persistent transitions among states.

Sampling design

We sampled 72 SDSL plots as part of a larger study of ecological site variability. We used the Generalized Randomized Tessellation Stratification (GRTS) method (Stevens and Olsen 2004) to select spatially balanced sampling locations within strata that consisted of soil map units (excluding units dominated by rock outcrops) delineated by the SSURGO (soil survey geographic database) order-three soil survey (USDA NRCS 1991) for areas inside and outside of CNP, and fenced pastures on the Indian Creek allotment ($n = 48$). The GRTS method minimizes clustering of sampling locations that can occur with simple random sampling. To capture the full range of variability in the SDSL site, we also used targeted sampling to select an additional 24 plot locations. In the field, the ecological site membership of each plot was determined based on soil properties and landscape setting. At each location, a sampling plot consisted of three parallel 50-m transects separated by 25 m and oriented parallel to the hillslope contour. Where initial transect alignments were found to cross soil-geomorphic boundaries separating different ecological sites, plot locations were adjusted objectively to ensure that sampling was restricted

to the SDSL ecological site. Sampling was conducted from late May through October in 2006–2008. For all analyses described below, data for GRTS plots and targeted plots were combined in a single data set.

Field measures

Field measures were selected specifically to quantify structural and functional attributes related to the states and processes depicted in our conceptual model.

Biotic composition, ground cover, spatial structure of vegetation, livestock use.—At each plot, live foliar cover of vascular plants and cover of biological crust (differentiated as dark cyanobacteria, moss, or lichen), litter, rocks, and bare ground were estimated by line-point intercept sampling with 1-m sampling intervals (150 points per plot; Herrick et al. 2005). As an indicator of wind-erosion resistance (Okin 2008), gaps between perennial plant canopies were measured using line-intercept sampling following procedures described by Herrick et al. (2005). The frequency of livestock dung was acquired from 1×1 m quadrats placed at 5-m intervals along each transect (30 quadrats per plot). Dung frequency provides an index of

recent but not past livestock use. We also recorded a list of all plant species observed in the plot.

Soil-surface attributes.—As an additional indicator of erosion resistance and soil biotic activity, surface soil aggregate stability was measured using a field-based soil stability kit (Herrick et al. 2001), with plot-level averages based on measurements at six random points per transect (18 subsamples per plot). Several ecological functions of biological crust are attributable to interspace soil-surface roughness associated with well-developed crust communities. Fine-scale soil-surface roughness facilitates the retention of overland water flow (Ward and Trimble 2004), the retention of litter and plant propagules, and the creation of safe sites for seed germination and establishment (Harper et al. 1965). We measured soil roughness in plant interspaces by draping a 20-cm jewelry chain with 2-mm chain links across surface microtopographic features and measuring the horizontal distance between the ends of the chain. Measures were acquired at 10-m intervals along each transect (15 subsamples per plot) and averaged to derive a plot-level mean. A soil roughness index (in percent) was calculated for each plot as

$$\text{Soil Roughness Index} = (1 - L_2/L_1) \times 100$$

where L_2 is the mean horizontal distance in cm and L_1 is the length (20 cm) of the chain (Saleh 1993).

In sandstone-derived soils in our study area, magnetic minerals in soil are attributable to deposits of far-travelled eolian dust that contribute significant amounts of silt, clay, and rock-derived nutrients (Reynolds et al. 2006). As an indicator of dust, soil fines, and rock-derived nutrients, we measured the magnetic susceptibility of the soil surface with a MS-20 magnetic susceptibility (MS) meter (GF Instruments, s.f.o.; Czech Republic) with a sensitivity of 10^{-6} SI units. Measures were acquired at 10-m intervals along each transect (15 subsamples per plot), and averaged to derive a plot-level mean. When compared among sites with similar landscape settings and soils, lower MS readings are interpreted to indicate depletion of eolian silts, clays, and associated soil resources following soil destabilization and wind erosion (Neff et al. 2005, Reynolds et al. 2010). In combination, soil

aggregate stability, soil-surface roughness, and MS all are interpreted as relative indicators of soil health (high stability, roughness, and MS) and soil degradation (low stability, roughness, and MS).

Statistical analysis

Cluster analysis.—As an empirical examination of our state-and-transition model we applied a fuzzy cluster analysis (Equihua 1990) to group and classify plots according to their degree of similarity in biophysical attributes. In fuzzy clustering (as opposed to “hard” clustering), observations are assigned membership values for all clusters, where membership values sum to one and cluster identity is determined by the maximum value (Equihua 1990). In ecological applications, this classification approach explicitly acknowledges variability and the fact that samples naturally will differ in their degree of affinity for a given cluster, and may display some affinity for multiple clusters simultaneously (Roberts 1989, Equihua 1990). Fuzzy clusters were derived using four variables as classification criteria and Euclidean distances among plots in NCSS 2001 software (Hintze 2004). This method is compatible only with Euclidean distance; use of this distance measure is justified by approximate normal distributions of data, approximate linear intercorrelation among variables, and few zero values. To facilitate a linkage between our conceptual model and the cluster analysis, we used a parsimonious set of classification variables that was based on the compositional attributes of states depicted in the model. Classification variables included (1) percent cover of biological crust, (2) percent live cover of perennial grasses and palatable shrubs, (3) percent relative live cover of invasive annual plants, and (4) percent cover of bare ground. We used principal components analysis (PCA) to visualize and describe underlying differences among clusters in terms of the four classification variables. PCA was conducted using PC-ORD 5.0 (McCune and Mefford 2006).

Patterns among and within clusters.—To characterize clusters quantitatively and interpret them with respect to states outlined in the STM, we conducted multivariate and univariate analyses on a suite of 26 variables. We used univariate analysis of variance (ANOVA), or Kruskal-Wallis

tests when the assumptions of normality were not met, to determine which specific measurements differed among clusters. Contrasts were performed using the Tukey HSD test.

To determine if samples within clusters were distinguishable based upon grazing status, we conducted a multi-response permutation procedure (MRPP) of currently and historically grazed plots within the *grass-bare* and *annualized-bare* clusters; because only two *biocrust* plots were currently grazed, an MRPP analysis of grazing status was not conducted for the *biocrust* cluster. (Cluster names are described in Results.) MRPP uses distance measures (Euclidean in our case, for consistency with other techniques) and randomization tests to determine if groups are different. It calculates chance corrected within-group agreement (A , ranging from 0–1, with values >0.1 often considered to indicate a strong degree of agreement within groups). MRPP was conducted in PC-ORD 5.0 (McCune and Mefford 2006).

To determine if individual variables differed by grazing status within cluster assignments, we used a two-way mixed-effects ANOVA with cluster assignment, grazing status, and cluster \times grazing as fixed effects, and year as a random effect. Year was included to control for differences in annual precipitation among the years of field sampling. Because the ANOVA was unbalanced (due to unequal sample sizes among fixed and random effects), the restricted maximum likelihood approach was used to estimate parameters (Spilke et al. 2005). In the *biocrust* cluster all but two plots were classified as never or as formerly grazed. Because this prohibits investigating interactive effects of cluster assignment and grazing status, data from the *biocrust* cluster were omitted from the analysis. The mixed-effects models were conducted in SAS 9.2 (Littell et al. 2006).

Simulation modeling

We used a wind erosion model (WEMO hereafter) to investigate effects of measured biophysical attributes on predicted rates of wind-driven soil movement at our plots (Okin 2008). WEMO predicts horizontal dust flux ($\text{g}\cdot\text{cm}^{-1}\cdot\text{d}^{-1}$) on the basis of wind velocity, plant height, the size-class distribution of gaps between plant canopies, total plant cover, threshold

shear velocity (TSV; Gillette et al. 1982), and a suite of other variables. TSV is the surface wind velocity required to initiate soil movement and thus is a measure of soil erodibility. Direct measurement of TSV requires a wind tunnel or similar apparatus. For our plots, we estimated TSV from soil aggregate stability measures using data from wind tunnel observations on soils similar to those of the SDSL ecological site (J. Belnap, *unpublished data*). In the wind tunnel data, TSV variability increased with increasing soil aggregate stability, and residuals were not normally distributed around a least-squares model. Therefore, we used quantile regression to fit separate linear models through the 10th, 50th, and 90th percentiles of the empirical TSV data (Cade and Noon 2003). Here we report WEMO predictions based on the 10th percentile model because fluxes predicted with this model were most consistent with flux observations from a continuous monitoring effort (since 1999) at two of our plots (VP and NR in Belnap et al. 2009). The rate of wind erosion is proportional to the cube of wind velocity above TSV (Bagnold 1941), so we used a range of wind velocities (17.5, 26.25, and 35.0 m/s, measured at 10 m above the surface) in WEMO to examine relative increases in predicted dust fluxes with increasing wind velocity. The highest velocity value we used corresponds to the maximum wind velocity reported in the study-area region (Williams et al. 1995). We used perennial plants only as the basis for WEMO inputs for canopy gaps, plant cover, and plant height because production and cover of annual plants are highly responsive to precipitation variability and contribute little to erosion resistance during periods of drought (Belnap et al. 2009). Thus predicted dust fluxes represent relative measures of susceptibility to wind erosion during drought.

RESULTS

Cluster analysis

Fuzzy cluster analysis resulted in three clusters with minimized within-cluster variance and maximized among-cluster variance. Short-hand notation for clusters used hereafter are *biocrust* (biological crust, perennial grasses, and palatable shrubs), *grass-bare* (perennial grasses and bare ground), and *annualized-bare* (invasive annual

grasses or forbs, and bare ground); these clusters correspond well with states B, C, and D, respectively, in the STM (Fig. 2). PCA results illustrate how the three clusters differed on the basis of the four classification variables (Fig. 3). The *biocrust* cluster contained 21 plots, the *grass-bare* cluster contained 24 plots, and the *annualized-bare* cluster contained 27 plots. Table A1 (Appendix) summarizes the sample numbers by year, water-year precipitation, cluster, and grazing status.

ANOVA results highlighted numerous distinguishing characteristics of each cluster. Of the 26 variables considered, only perennial forb cover and unpalatable shrub cover did not differ between at least two clusters (Table 1). The *biocrust* cluster was characterized by 5.3 times greater biological crust cover than the *grass-bare* cluster and 7.9 times greater crust cover than the *annualized-bare* cluster. As a result, averages for soil aggregate stability, soil surface roughness, and magnetic susceptibility also were highest in the *biocrust* cluster. The *grass-bare* cluster did not differ strongly from the *biocrust* cluster in terms of the perennial plant community (with the exception of less palatable shrub cover), but bare ground in the *grass-bare* cluster was 2.6 times greater than the *biocrust* cluster. Cover of native annual forbs in the *grass-bare* cluster was only 28% of that found in the *biocrust* cluster. Bare ground in the *annualized-bare* cluster was two times greater than in the *biocrust* cluster. Average relative cover of invasive exotic annuals in the *annualized-bare* cluster was five times greater than in the *biocrust* cluster and 7.7 times greater than in the *grass-bare* cluster; the *annualized-bare* cluster also was characterized by higher litter cover than the *grass-bare* cluster. Average total live cover of all vascular plants in the *biocrust* cluster was twice the average found in the *grass-bare* cluster. Both richness and total live cover of all functional groups (including biological crust) in the *biocrust* cluster were significantly higher than in the other two clusters. Average vascular plant richness in the *biocrust* cluster tended to be greater than in the *grass-bare* cluster and was significantly higher than in the *annualized-bare* cluster. Additional differences among clusters are presented in Table 1.

Empirical patterns within clusters

MRPP indicated that formerly grazed and currently grazed groups within the *annualized-bare* cluster ($A = 0.24$, $P < 0.0001$) and the *grass-bare* cluster ($A = 0.04$, $P = 0.015$) differed overall. Within the *grass-bare* and *annualized-bare* clusters, ANOVA revealed that grazing status (currently versus formerly grazed) was statistically significant for seven of 26 variables (Table 2 and Appendix: Table A2). Compared to currently grazed *grass-bare* plots, formerly grazed *grass-bare* plots were characterized by greater abundance of biological crusts as well as greater magnetic susceptibility and surface roughness—both of which are functionally related to biological crust cover. Currently and formerly grazed *annualized-bare* plots primarily differed in higher relative abundance of exotic grasses and forbs, respectively. This difference may be partially accounted for by a difference in elevation among these groups, as formerly grazed *annualized-bare* plots were lower and drier.

Predicted levels of wind erosion

Predicted levels of wind erosion differed among and within clusters as a function of soil stability and vegetation structure (Fig. 4; Appendix: Table A3). For the *biocrust* cluster, no erosion (zero flux) was predicted by WEMO except at the maximum wind velocity for two plots with relatively low soil aggregate stability values (4.8 and 4.9, compared with cluster mean 5.5; Appendix: Table A3) and thus relatively high wind erodibility. In the *grass-bare* cluster, higher average levels and greater frequency (percentage of plots with flux) of wind erosion were predicted for currently grazed plots than for formerly grazed plots with significantly higher levels of biological crust. Predicted erosion frequency in currently grazed plots in the *grass-bare* cluster also tended to be greater than in currently or formerly grazed plots in the *annualized-bare* cluster at all three wind velocities (Appendix: Table A3). At intermediate and maximum wind velocities, formerly grazed plots in the *annualized-bare* cluster were predicted to have the highest average levels of wind erosion, with maximum fluxes predicted for a plot with a median gap size of 2153 cm and soil aggregate stability value of 3.3 (Fig. 4; Appendix: Table A3). In contrast, no wind erosion was predicted for a

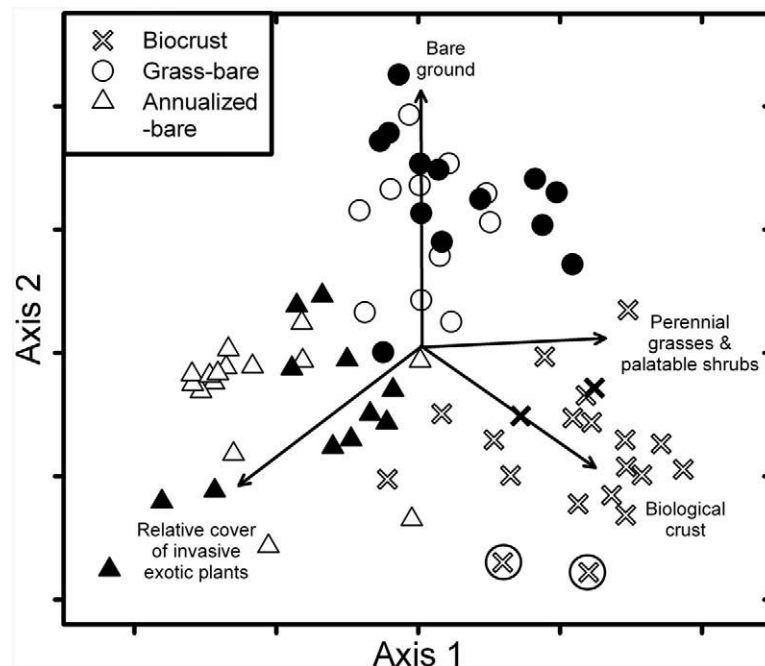


Fig. 3. PCA ordination of plots based on the four classification variables used in the fuzzy cluster analysis. Clusters are noted as *biocrust*, *grass bare*, and *annualized bare*. These clusters closely match states B, C, and D, respectively in Fig. 1. For each cluster symbol, closed symbols indicate plots that are currently accessible to grazing, open symbols indicate plots that formerly were grazed, and circled open symbols indicate two plots that were never grazed. Vectors indicate loadings of four classification variables on the two axes. Axis 1 accounts for 42.6% of the variability and is most highly correlated with relative cover of invasive exotics ($r = 0.85$), cover of perennial grasses and palatable shrubs ($r = 0.70$), and cover of biological crust ($r = 0.69$). Axis 2 accounts for 36.2% of the variability and is most highly correlated with bare ground ($r = 0.97$).

plot in the *biocrust* cluster that also was characterized by large canopy gaps (median gap size 1885 cm) but that had a higher soil aggregate stability value of 5.2 (Fig. 4).

DISCUSSION

Empirical evidence for alternative states

Our empirical results document the existence of alternative states defined by significant differences in functional group structure for the SDSL grassland ecosystem (Fig. 5). Our analysis substitutes space for time, and we infer that states represented by the *grass-bare* and *annualized-bare* clusters reflect persistent changes in ecosystem structure and function triggered by interactions of livestock grazing (reduction of perennial grasses and palatable shrubs through selective herbivory), associated soil disturbances (depletion of soil resources through trampling,

loss of biological crust, soil destabilization, and accelerated erosion), and climate (drought-induced reduction in grazing tolerance of preferred forage species) (Fig. 2). The fact that *grass-bare* and *annualized-bare* plots in CNP have been protected from livestock impacts for more than 30 years strongly suggests that the striking among-cluster differences in structure (Fig. 3) and function (Fig. 4) can be persistent for at least multiple decades and are effectively irreversible at a time scale relevant to current management without costly investments in ecological restoration. Among-cluster differences in WEMO results are consistent with patterns in magnetic susceptibility, which was significantly higher in the *biocrust* cluster than in *grass-bare* and *annualized-bare* clusters—suggesting greater wind erosion and soil depletion in these two clusters. These results also support and expand upon other recent research that has examined legacy effects

Table 1. Means (standard errors) of 26 variables for three fuzzy clusters, and ANOVA results.

Variables	Biocrust	Grass bare	Annualized bare
Biological crust (cover %)	34.1 ^A (3.0)	6.4 ^B (1.4)	4.3 ^B (1.0)
Perennial grasses and palatable shrubs (cover %)	20.6 ^A (2.8)	14.9 ^A (2.0)	11.4 ^B (2.0)
Relative cover of invasive exotic annuals (%)	11.7 ^A (2.7)	7.7 ^A (1.5)	59.3 ^B (3.9)
Bare ground (cover %)	18.7 ^A (1.7)	48.6 ^B (2.0)	36.6 ^C (3.3)
Annual grasses, native (cover %)	2.6 ^A (0.6)	1.3 ^A (0.4)	0.1 ^B (0.04)
Annual grasses, exotic (cover %)	4.9 ^{AB} (1.7)	0.6 ^B (0.2)	13.1 ^A (3.2)
Annual forbs, native (cover %)	16.7 ^A (3.6)	4.7 ^B (1.2)	5.1 ^B (1.6)
Annual forbs, exotic (cover %)	1.1 ^A (0.5)	1.4 ^A (0.5)	9.2 ^B (1.6)
Perennial forbs (cover %)	1.4 ^A (0.6)	1.5 ^A (0.7)	0.5 ^A (0.1)
Bunchgrasses C ₃ (cover %)	3.7 ^A (1.3)	2.5 ^A (0.7)	0.6 ^B (0.3)
Bunchgrasses C ₄ (cover %)	3.2 ^A (1.0)	3.1 ^A (1.3)	6.0 ^B (1.3)
Rhizomatous grasses C ₄ (cover %)	4.6 ^A (0.8)	5.2 ^A (1.0)	2.4 ^B (0.9)
Palatable shrubs (cover %)	5.5 ^A (0.9)	1.2 ^B (0.5)	1.1 ^B (0.4)
Unpalatable shrubs (cover %)	2.3 ^A (0.7)	1.5 ^A (0.5)	0.9 ^A (0.3)
Total live cover, perennial plants (%)	20.7 ^A (2.8)	14.9 ^A (1.9)	11.4 ^B (1.9)
Total live cover, vascular plants (%)	46.0 ^A (4.6)	22.9 ^B (2.9)	38.9 ^A (3.6)
Total live cover, all functional groups (%)	80.1 ^A (4.8)	29.3 ^B (2.6)	43.3 ^C (4.1)
Functional group richness (no. of functional groups)	8.3 ^A (0.3)	7.0 ^B (0.3)	6.2 ^C (0.3)
Species richness (no. of vascular plant species)	23 ^A (1.4)	19.4 ^A (1.3)	16.7 ^B (1.2)
Soil aggregate stability (index)	5.5 ^A (0.1)	3.8 ^B (0.2)	4.3 ^B (0.2)
Soil surface roughness (index)	11.9 ^A (0.7)	4.8 ^B (0.6)	4.5 ^B (0.6)
Litter (cover %)	49.7 ^A (2.5)	28.4 ^B (1.9)	42.7 ^A (3.4)
Magnetic susceptibility (10 ⁻⁶ SI units)	0.18 ^A (0.0)	0.10 ^B (0.0)	0.12 ^B (0.0)
Median size of perennial canopy gaps (cm)	165.9 ^{AB} (86.2)	78.3 ^A (6.5)	253.7 ^B (83.8)
Elevation (m)	1670.1 ^A (72.1)	1557.1 ^B (114.7)	1623.9 ^B (152.3)
Livestock dung (% frequency)	1.0 ^A (3.6)	11.3 ^B (3.4)	13.3 ^B (3.2)

Notes: Means superscripted with different letters are statistically different at the $\alpha = 0.05$ level using the Tukey HSD test. The first four variables are the classification criteria used to generate clusters.

Table 2. Means (standard errors) of variables by grazing status for the *grass bare* and *annualized bare* clusters.

Variables	Grass bare		Annualized bare	
	Currently grazed (n 13)	Formerly grazed (n 11)	Currently grazed (n 12)	Formerly grazed (n 15)
Biological crust (cover %)	2.8 (1.4)	10.7 (1.9)	4.5 (1.4)	4.2 (1.5)
Perennial grasses and palatable shrubs (cover %)	19.6 (3.0)	9.4 (1.3)	15.8 (2.8)	8.0 (2.5)
Relative cover of invasive exotic annuals (%)	7.0 (1.9)	8.5 (2.4)	54.6 (6)	63.1 (5.1)
Bare ground (cover %)	48.8 (3.0)	48.4 (2.8)	29.7 (3.8)	42.1 (4.7)
Annual grasses, native (cover %)	2.0 (0.5)	0.5 (0.3)	0.2 (0.1)	0.1 (0.1)
Annual grasses, exotic (cover %)	0.8 (0.2)	0.4 (0.2)	23.7 (4.3)	4.6 (3.3)
Annual forbs, native (cover %)	3.6 (0.8)	5.9 (2.5)	2.8 (0.9)	6.9 (2.7)
Annual forbs, exotic (cover %)	1.0 (0.5)	1.8 (0.9)	1.6 (0.6)	15.3 (1.6)
Bunchgrasses C ₃ (cover %)	4.0 (1.1)	0.7 (0.3)	1.1 (0.6)	0.2 (0.1)
Bunchgrasses C ₄ (cover %)	5.2 (2.2)	0.5 (0.2)	8.7 (2.3)	3.9 (1.2)
Rhizomatous grasses C ₄ (cover %)	5.1 (1.5)	5.3 (1.4)	2.1 (0.9)	2.6 (1.5)
Palatable shrubs (cover %)	0.8 (0.3)	1.6 (1)	1.8 (0.8)	0.4 (0.2)
Unpalatable shrubs (cover %)	2.4 (0.9)	0.3 (0.1)	1.6 (0.6)	0.2 (0.1)
Total live cover, perennial plants (%)	19.6 (6)	9.4 (1.3)	15.8 (2.8)	7.9 (2.5)
Total live cover, vascular plants (%)	27.0 (3.8)	17.9 (4.3)	44.1 (4.3)	34.9 (5.4)
Total live cover, all functional groups (%)	29.8 (3.6)	28.7 (4)	48.5 (4.3)	39.1 (6.5)
Functional group richness (no. of groups)	7.6 (0.2)	6.4 (0.4)	7.0 (0.4)	5.5 (0.5)
Species richness (no. of vascular plant species)	22.1 (2.0)	16.3 (1.8)	19.1 (1.6)	14.8 (1.1)
Soil aggregate stability (index)	3.2 (0.3)	4.4 (0.2)	4.5 (0.3)	4.1 (0.2)
Soil surface roughness (index)	4.1 (0.6)	5.6 (0.5)	6.2 (0.7)	3.2 (0.7)
Litter (cover %)	28.4 (3.0)	28.4 (2.5)	48.1 (5.2)	38.4 (4.4)
Magnetic susceptibility (10 ⁻⁶ SI units)	0.085 (0.010)	0.118 (0.012)	0.150 (0.015)	0.093 (0.001)
Median size of perennial canopy gaps (cm)	65.0 (6.7)	94.0 (10)	87.1 (9.6)	387.1 (143.4)
Elevation (m)	1566.4 (41.1)	1546.2 (18.1)	1752.4 (41.9)	1521.1 (8.2)
Livestock dung (% frequency)	18.5 (4.6)	2.7 (2.1)	29.7 (7.5)	0.2 (0.2)

Notes: Values in boldface indicate $P < 0.05$, based on two way mixed effects ANOVA models with year as a random effect block. Exact P values are in Appendix: Table A2.

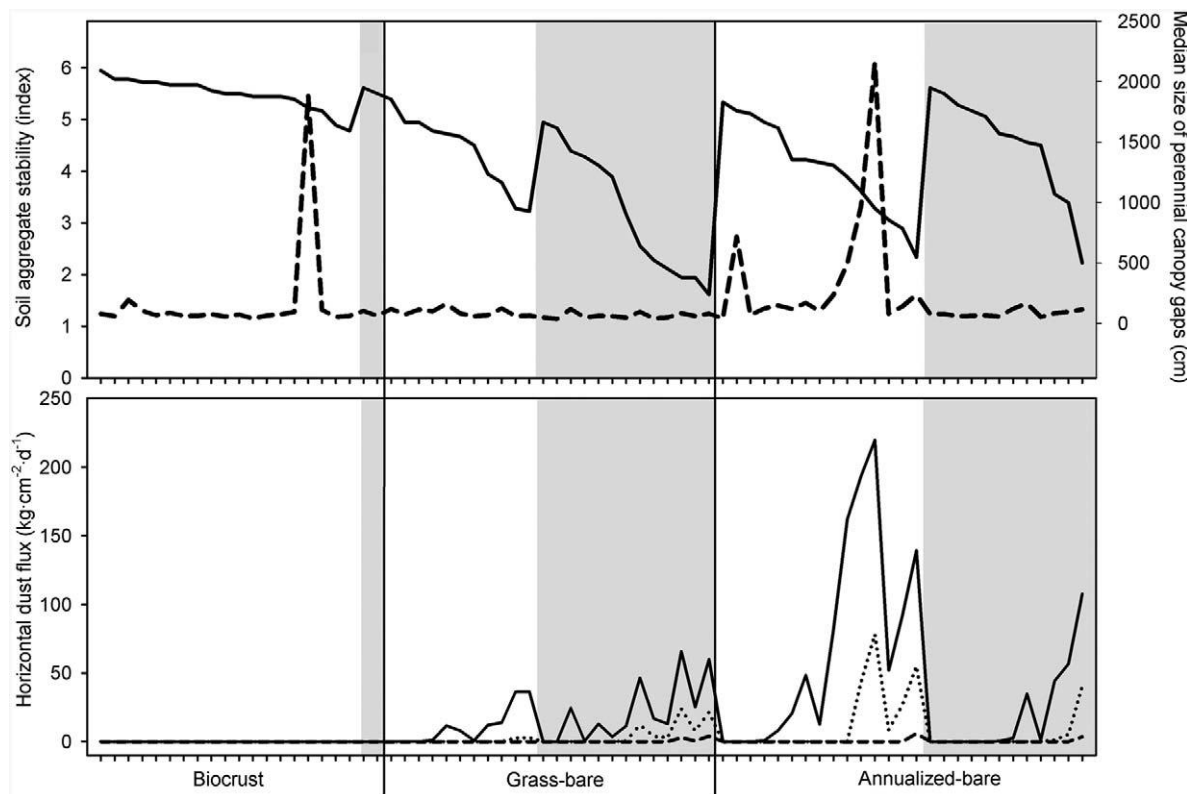


Fig. 4. Soil aggregate stability (top panel, solid line), median gap size between perennial plant canopies (top panel, dashed line), and predicted rates of wind erosion (horizontal dust flux) at three wind velocities (bottom panel; dashed line = 17.5 m/s, dotted line = 26.25 m/s, solid line = 35.0 m/s) for plots categorized by cluster and grazing status (shading indicates plots currently accessible to grazing). Within each cluster and grazing category, plots are ordered from left to right by decreasing soil aggregate stability. Data are discrete points, indicated by tick marks on the X axis, but are represented as lines for easier viewing.

of livestock grazing on soil biogeochemical properties and erosional processes in our study area (Neff et al. 2005, Belnap et al. 2009).

The *biocrust* and *grass-bare* states differed significantly with respect to biological crust cover and related soil attributes. But these differences were not accompanied by differences in perennial grass cover and composition despite our interpretation that lower measures of soil aggregate stability, surface roughness, and MS were indicative of degraded soil conditions in the *grass-bare* state relative to the *biocrust* state. In terms of the vascular plant community, evidence for consequences of soil differences may be reflected in the much higher cover of native annual forbs in the *biocrust* state (16.7%) versus the *grass-bare* state (4.7%, Table 1). This pattern could be attributable to greater retention of

propagules and availability of safe sites in roughened interspaces dominated by biological crust, as well as to greater resource availability to seedlings in undisturbed interspaces that retain higher levels of eolian fines. This hypothesis is consistent with data linking fine-scale patterns in MS and eolian fines to distributional patterns of annual plants including the exotic *Bromus tectorum* (Reynolds et al. 2010). In the *grass-bare* state, formerly grazed plots had higher measures of biological crust cover, surface roughness, and MS (Table 2) relative to currently grazed plots, suggesting some recovery of soil attributes following 30 years of rest from livestock disturbance.

In the *grass-bare* and *annualized-bare* states, there was an unexpected tendency for currently grazed plots in the Indian Creek allotment to

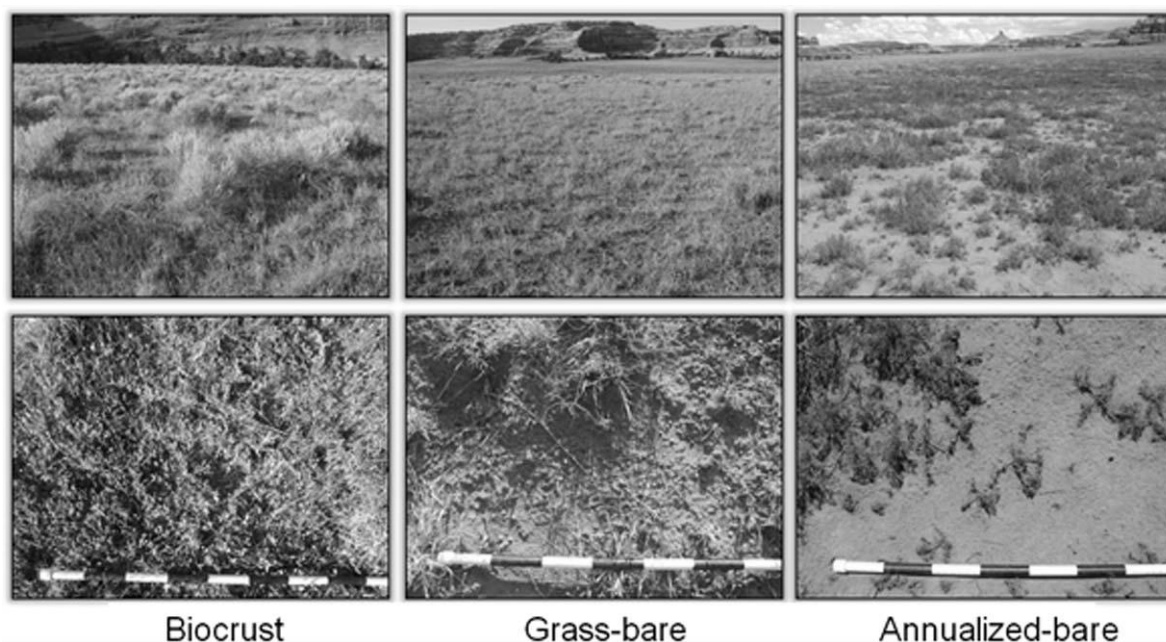


Fig. 5. Example photos from plots classified into three clusters representing ecosystem states *biocrust* (left), *grass bare* (middle), and *annualized bare* (right) within the SDSL ecological site. Top photos show general vegetation characteristics. Bottom photos show soil surface characteristics including differences in roughness and in the relative abundance of biological crust and bare ground.

have higher cover of perennial grasses than formerly grazed plots in CNP (Table 2), suggesting potential facilitation of grass establishment by recent livestock disturbance. Instead, we interpret this pattern primarily as an artifact of sampling year precipitation conditions. Combining both states, 19 of 26 (73%) formerly grazed plots in CNP were sampled in the dry year of 2006 (Appendix: Table A1), whereas 21 of 25 (84%) of currently grazed plots in the Indian Creek allotment were sampled in comparatively wet years of 2007 and 2008 that were more favorable for grass establishment and growth.

Relation to resilience theory

Functional group structure is recognized as the key biotic control of ecosystem resilience and sustainability (Chapin et al. 1996). This case study is unique for its incorporation of the biological crust functional group in an alternative state framework and for its documentation of a state characterized by high areal coverage of biological crust relative to vascular plants. Though underreported, the current or former

existence of such a state is widespread among many forms of dryland ecosystems (Bowker and Belnap 2008, Bowker et al. 2008, Miller 2008), including at least five of the eight most common ecological sites sampled in our study area in conjunction with the current study (M. E. Miller, *unpublished data*). Biological crust effects on soil stability, nutrient cycling, hydrologic processes, and vascular plant establishment indicate a need for explicit consideration of this functional group in ecosystem analyses and management, particularly in systems characterized by a high degree of biological crust coverage and functionality relative to the vascular plant community.

Resilience is promoted both by redundancy in the performance of key ecosystem functions and by diversity in biotic responses to perturbations (Walker 1992, Walker et al. 1999, Elmqvist et al. 2003). These principles are well-illustrated by the biological crust functional group and *biocrust* cluster in our case study. Sparsely vegetated drylands with high coverage of biological crust lack redundancy with respect to ecosystem functions performed by the crust functional

group. In such systems, key functions including the capture and retention of mobile soil resources and the mediation of vascular plant establishment are effectively lost from interspaces when biological crust is lost from the system. The likelihood that high-functioning biological crust will be eliminated from a system by surface disturbance (as evidenced by the *grass-bare* and *annualized-bare* clusters) is heightened by the fact that most biological crust components and functions tend to recover slowly or not at all following extensive surface disturbance and destabilization (Belnap and Eldridge 2003). Thus as a functional group, biological crust generally lacks diversity in its responses to surface disturbance—perhaps the most ubiquitous human impact on drylands.

Implications for ecosystem services and management

Dryland degradation is widely recognized as having a biophysical component (reflecting reductions in soil resources, biological diversity, or other ecosystem attributes) as well as a socioeconomic component (reflecting reductions in the valued services that society derives from ecosystems) (Reynolds and Stafford Smith 2002, Reynolds et al. 2007). Biophysical state changes described here may or may not be recognized as socioeconomic degradation, depending on societal perceptions of how such changes affect services that are most highly valued (Reynolds and Stafford Smith 2002, Walker et al. 2002). Of the 6000 ha of SDSL in the currently grazed Indian Creek allotment, we estimate that 15% (900 ha) is in *biocrust* condition, whereas 50 and 35% are in *grass-bare* and *annualized-bare* condition, respectively. Of the 4500 ha of SDSL in CNP, we estimate that 45% (2025 ha) is in *biocrust* condition, with 21 and 34% in *grass-bare* and *annualized-bare* condition, respectively. Together, these proportions can be viewed as comprising an investment portfolio of ecosystem goods and services because each state has distinct biophysical attributes that have the potential to support distinct sets of socioeconomic values. Investment in a particular state, and therefore in a particular set of goods and services, may result in trade-offs and synergies relative to other goods and services.

The SDSL ecological site, like many drylands

worldwide, by tradition has been valued primarily for its provision of livestock forage in support of the livelihoods and cultural traditions of local residents. With respect to total perennial livestock forage, the *biocrust* and *grass-bare* clusters both appear to exhibit greater value than the *annualized-bare* cluster (Table 1). Yet maintenance of the *biocrust* state requires that surface disturbances be limited, thereby greatly constraining access to available forage and potentially reducing the perceived value of this state relative to states with greater forage accessibility. Thus, an investment portfolio emphasizing livestock forage favors a landscape dominated by the *grass-bare* state.

Increasingly, drylands are recognized as providing a suite of ecosystem services (Havstad et al. 2007), and investment in one may incur trade-offs with others either in space or time (Rodríguez et al. 2006). For example, use of forage for livestock production has the potential to incur costs in terms of diminished erosion resistance. In this case study, SDSL clusters clearly differed with respect to modeled levels of wind erosion, with the *biocrust* cluster emitting essentially no dust, the *grass-bare* cluster consistently emitting dust, and the *annualized-bare* cluster potentially becoming a major dust source when drought conditions limit cover of annual plants. Dust emissions from unstable drylands can have downwind consequences through effects on ecosystem biogeochemistry (Neff et al. 2008), mountain snowpack and downstream water delivery (Painter et al. 2010), air quality and human health, and atmospheric dust concentrations that can affect the global energy balance (Field et al. 2010). Economic costs of these downwind consequences are potentially great but rarely considered in local decision making where management for dust abatement is at odds with maximizing livestock production.

Carbon (C) storage is another ecosystem service with increasing importance in the context of climate change mitigation efforts, and represents another service compromised by investment in pastoralism. Data for total live cover plus litter (Table 1) approximate relative C stocks for the three SDSL clusters. The *biocrust* cluster emerges as clearly superior in this way, supporting the greatest average cover of vascular plants, biological crust, litter, and their sum (130% total).

Although the *annualized-bare* cluster also may support high total live cover plus litter (86% total), live cover and litter both fluctuate greatly in response to precipitation due to dominance by annual plants. The *grass-bare* state appears to be the most depauperate in terms of C-storage potential (58% total), although the *annualized-bare* cluster likely would be lower during drought years. Neff et al. (2005) demonstrated a 60–70% difference in soil C between plots in *biocrust* condition and plots in *grass-bare* condition in our study area, and Barger et al. (2006) documented significant C loss with surface disturbance of plots in *biocrust* condition.

In addition to trade-offs, synergies may emerge when managing for multiple outcomes. For example, there is no conflict between management emphasizing dust abatement and C storage; if management actions are taken to enhance one, the other is likely to be enhanced as well. Other non-traditional valuations of ecosystem states include biological diversity, a supporting service which ensures long-term sustainability of forage and other values (Chapin et al. 2009, Stafford Smith et al. 2009). In terms of biodiversity, we found *biocrust* > *grass-bare* > *annualized-bare* in functional group and species richness, indicating another synergy with C storage and dust abatement.

Future strategies for ecosystem management would benefit from explicit evaluation of existing ecosystem states, the breadth of ecosystem services that each can support, and potential risks, trade-offs, and synergies associated with alternative management strategies. Applying such an approach to the SDSL ecological site examined in our study could result in a range of different management prescriptions that also could change over time in response to ever-shifting valuations of costs and benefits due to climate change, for example. Canyonlands National Park currently is invested primarily in a portfolio emphasizing biodiversity, C storage, and dust abatement provided by the *biocrust* state, but the legacy of past disturbance remains apparent in the high coverage of the *annualized-bare* state. Because grazing is no longer permitted in CNP, the *annualized-bare* state has little economic value. Active restoration of *annualized-bare* areas is warranted to attain states with higher functional diversity to enhance resiliency

to climate change and drought, as well as to enhance long-term capacity for C storage, dust control, and biodiversity conservation. In the case of the Indian Creek allotment, where the current portfolio favors the provision of livestock forage, lands in *annualized-bare* condition might continue to be grazed only if analysis indicates that benefits for livestock production exceed costs attributable to dust emissions and diminished capacity for C storage and biodiversity conservation. Likewise, where benefits exceed costs, lands in *grass-bare* condition might be managed for livestock production but in a careful manner that minimizes risks of dust emissions and enhances resilience to mitigate risks of further degradation to the *annualized-bare* state. If the balance of the cost-benefit calculus changes, then these lands might be retired or rested from grazing, or actively restored. Acknowledging the interplay between alternative ecosystem states and economic forces will illuminate management strategies which maximize the provision of ecosystem goods and services.

ACKNOWLEDGMENTS

Funding support was provided by the U.S. Geological Survey (Southwest Biological Science Center, Status and Trends of Biological Resources Program, and Global Change Program), The Nature Conservancy of Utah, and the National Park Service. Additional logistical support was provided by the Bureau of Land Management. We thank Rebecca Mann, Mary Moran, Hillary Hudson, Ralph Ferrara, and Ole Bye for assistance with field work and data management. Jayne Belnap provided wind tunnel data. Brandon Bestelmeyer, David Eldridge, Barry Baker, Seth Munson, Nichole Barger, and Jack Morgan provided comments that improved the quality of the manuscript. We thank Vicki Webster for editorial assistance. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

LITERATURE CITED

- Archer, S., D. S. Schimel, and E. A. Holland. 1995. Mechanisms of shrubland expansion: land use, climate or CO₂? *Climatic Change* 29:91–99.
- Bagnold, R. A. 1941. *The physics of blown sand and desert dunes*. Methuen, London, UK.
- Barger, N. N., J. E. Herrick, J. Van Zee, and J. Belnap. 2006. Impacts of biological soil crust disturbance on C and N loss from water erosion. *Biogeochemistry*

- 77:247–263.
- Beisner, B., D. Haydon, and K. Cuddington. 2003. Alternative stable states in ecology. *Frontiers in Ecology and the Environment* 1:376–382.
- Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. *Environmental Monitoring and Assessment* 37:39–57.
- Belnap, J., and D. J. Eldridge. 2003. Disturbance and recovery of biological soil crusts. Pages 363–383 in J. Belnap, and O. L. Lange, editors. *Biological soil crusts: Structure, function, and management*. Second edition. Springer Verlag, Berlin, Germany.
- Belnap, J., and S. L. Phillips. 2001. Soil biota in an ungrazed grassland: response to annual grass (*Bromus tectorum*) invasion. *Ecological Applications* 11:1261–1275.
- Belnap, J., R. Prasse, and K. T. Harper. 2003. Influence of biological soil crusts on soil environments and vascular plants. Pages 281–300 in J. Belnap and O. L. Lange, editors. *Biological soil crusts: Structure, function, and management*. Second edition. Springer Verlag, Berlin, Germany.
- Belnap, J., R. L. Reynolds, M. C. Reheis, S. L. Phillips, F. E. Urban, and H. L. Goldstein. 2009. Sediment losses and gains across a gradient of livestock grazing and plant invasion in a cool, semi arid grassland, Colorado Plateau, USA. *Aeolian Research* 1:27–43.
- Bestelmeyer, B. T. 2006. Threshold concepts and their use in rangeland management and restoration: The good, the bad, and the insidious. *Restoration Ecology* 14:325–329.
- Bestelmeyer, B. T., A. J. Tugel, G. L. Peacock, D. G. Robinett, P. L. Shaver, J. R. Brown, J. E. Herrick, H. Sanchez, and K. M. Havstad. 2009. State and transition models for heterogeneous landscapes: A strategy for development and application. *Rangeland Ecology and Management* 62:1–12.
- Bowker, M. A., and J. Belnap. 2008. A simple classification of soil types as habitats of biological soil crusts on the Colorado Plateau, USA. *Journal of Vegetation Science* 19:831–840.
- Bowker, M. A., M. E. Miller, J. Belnap, T. D. Sisk, and N. C. Johnson. 2008. Prioritizing conservation effort through the use of biological soil crusts as ecosystem function indicators in an arid region. *Conservation Biology* 22:1533–1543.
- Briske, D. D. 2006. A unified framework for assessment and application of ecological thresholds. *Range Ecology and Management* 59:225–236.
- Briske, D. D., B. T. Bestelmeyer, T. K. Stringham, and P. L. Shaver. 2008. Recommendations for development of resilience based state and transition models. *Rangeland Ecology and Management* 61:359–367.
- Cade, B. S., and B. R. Noon. 2003. A gentle introduction to quantile regression for ecologists. *Frontiers in Ecology and the Environment* 1:412–420.
- Carpenter, S., B. Walker, J. M. Anderies, and N. Abel. 2001. From metaphor to measurement: Resilience of what to what? *Ecosystems* 4:765–781.
- Chapin, F. S., III, et al. 2009. Ecosystem stewardship: sustainability strategies for a rapidly changing planet. *Trends in Ecology and Evolution* 25:241–249.
- Chapin, F. S., III, M. S. Torn, and M. Tateno. 1996. Principles of ecosystem sustainability. *American Naturalist* 148:1016–1037.
- D'Antonio, C. M., and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review Ecology and Systematics* 23:63–87.
- Eldridge, D. J., E. Zaady, and M. Shachak. 2002. Microphytic crusts, shrub patches and water harvesting in the Negev Desert: the Shikim system. *Landscape Ecology* 17:587–597.
- Elmqvist, T., C. Folke, M. Nystrom, G. Peterson, J. Bengtsson, B. Walker, and J. Norberg. 2003. Response diversity, ecosystem change, and resilience. *Frontiers in Ecology and the Environment* 1:488–494.
- Escudero, A., I. Martinez, A. de la Cruz, M. A. G. Otalora, and F. T. Maestre. 2007. Soil lichens have species specific effects on the seedling emergence of three gypsophile plant species. *Journal of Arid Environments* 70:18–28.
- Equihua, M. 1990. Fuzzy clustering of ecological data. *Journal of Ecology* 78:519.
- Evans, R. D., and O. L. Lange. 2003. Biological soil crusts and ecosystem nitrogen and carbon dynamics. Pages 263–279 in J. Belnap and O. L. Lange, editors. *Biological soil crusts: Structure, function, and management*. Second edition. Springer Verlag, Berlin, Germany.
- Field, J. P., J. Belnap, D. D. Breshears, J. C. Neff, G. S. Okin, J. J. Whicker, T. H. Painter, S. Ravi, M. C. Reheis, and R. L. Reynolds. 2010. The ecology of dust. *Frontiers in Ecology and the Environment* 8:423–430.
- Flint, L. E., and A. L. Flint. 2007. Regional analysis of ground water recharge. Pages 29–60 in D. A. Stonestrom, J. Constantz, P. A. Ty, and S. A. Leake, editors. *Ground water recharge in the arid and semiarid southwestern United States*. U.S. Geological Survey Professional Paper 1703 B (<http://pubs.usgs.gov/pp/pp1703/b/>).
- Folke, C., S. R. Carpenter, B. Walker, M. Scheffer, T. Elmqvist, L. Gunderson, and C. S. Holling. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution and Systematics* 35:557–581.
- Gillette, D. A., J. Adams, D. Muhs, and R. Kihl. 1982. Threshold friction velocities and rupture moduli

- for crusted desert soils for the input of soil particles into the air. *Journal of Geophysical Research* 87:9003 9015.
- Groffman, P., et al. 2006. Ecological thresholds: The key to successful environmental management or an important concept with no practical application? *Ecosystems* 9:1 13.
- Harper, J. L., J. T. Williams, and G. R. Sagar. 1965. The behaviour of seeds in soil. I. The heterogeneity of soil surfaces and its role in determining the establishment of plants from seed. *Journal of Ecology* 53:273 286.
- Havstad, K. M., D. P. C. Peters, R. Skaggs, J. Brown, B. Bestelmeyer, E. Fredrickson, J. Herrick, and J. Wright. 2007. Ecological services to and from rangelands of the United States. *Ecological Economics* 64:261 268.
- Herrick, J. E., B. T. Bestelmeyer, S. Archer, A. J. Tugel, and J. R. Brown. 2006. An integrated framework for science based arid land management. *Journal of Arid Environments* 65:319 335.
- Herrick, J. E., J. W. Van Zee, K. M. Havstad, L. M. Burkett, and W. G. Whitford. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems. Volume I. Quick start, USDA ARS Jornada Experimental Range, Las Cruces, New Mexico, USA.
- Herrick, J. E., W. G. Whitford, and M. Walton. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. *Catena* 44:27 35.
- Hintze, J. 2004. NCSS, Number Cruncher Statistical Systems, Kaysville, Utah, USA. (<http://www.ncss.com>)
- Holling, C. S. 1996. Surprise for science, resilience for ecosystems, and incentives for people. *Ecological Applications* 6:733 735.
- Kleiner, E. F., and K. T. Harper. 1972. Environment and community organization in grasslands of Canyon lands National Park. *Ecology* 53:299 309.
- Littell, R. C., G. A. Milliken, W. W. Stroup, R. D. Wolfinger, and O. Schaberberger. 2006. SAS for mixed models. Second edition. SAS Institute Inc., Cary, North Carolina, USA.
- Mayer, A. L., and M. Rietkerk. 2004. The dynamic regime concept for ecosystem management and restoration. *BioScience* 54:1013 1020.
- McCune, B. P., and M. J. Mefford. 2006. PC ORD: Multivariate analysis of ecological data, version 5.27. MJM Software Design, Gleneden Beach, Oregon, USA.
- Miller, M. E. 2008. Broad scale assessment of range land health, Grand Staircase Escalante National Monument, USA. *Rangeland Ecology and Management* 63:249 262.
- Miller, M. E., J. Belnap, S. W. Beatty, and R. L. Reynolds. 2006. Performance of *Bromus tectorum* L. in relation to soil properties, water additions, and chemical amendments in calcareous soils of southeastern Utah, USA. *Plant and Soil* 288:1 18.
- Neff, J. C., A. P. Ballantyne, G. L. Farmer, N. M. Mahowald, J. L. Conroy, C. C. Landry, J. T. Overpeck, T. H. Painter, C. R. Lawrence, and R. L. Reynolds. 2008. Increasing eolian dust deposition in the western United States linked to human activity. *Nature Geoscience* 1:189 195.
- Neff, J. C., R. L. Reynolds, J. Belnap, and P. Lamothe. 2005. Multi decadal impacts of grazing on soil physical and biogeochemical properties in south east Utah. *Ecological Applications* 15:87 95.
- Okin, G. S. 2008. A new model of wind erosion in the presence of vegetation. *Journal of Geophysical Research* 113 F02S10.
- Okin, G. S., A. J. Parsons, J. Wainwright, J. E. Herrick, B. T. Bestelmeyer, D. C. Peters, and E. L. Fredrickson. 2009. Do changes in connectivity explain desertification? *BioScience* 59:237 244.
- Painter, T. H., J. S. Deems, J. Belnap, A. F. Hamlet, C. C. Landry, and B. Udall. 2010. Response of Colorado River runoff to dust radiative forcing in snow. *Proceedings of the National Academy of Sciences* 107:17125 17130.
- Reynolds, J. F., and D. M. Stafford Smith. 2002. Do humans cause deserts? Pages 1 25 in J. F. Reynolds and D. M. Stafford Smith, editors. *Global desertification: Do humans cause deserts?* Dahlem University Press, Berlin, Germany.
- Reynolds, J. F., et al. 2007. Global desertification: Building a science for dryland development. *Science* 316:847 851.
- Reynolds, R. L., M. C. Reheis, J. C. Neff, H. Goldstein, and J. D. Yount. 2006. Late Quaternary eolian dust in surficial deposits of a Colorado Plateau grass land: Controls on distribution and ecologic effects. *Catena* 66:251 266.
- Reynolds, R. L., H. L. Goldstein, and M. E. Miller. 2010. Atmospheric mineral dust in dryland ecosystems: Applications of environmental magnetism. *Geochemistry Geophysics Geosystems* 11:Q07009.
- Roberts, D. W. 1989. Fuzzy systems vegetation theory. *Vegetatio* 83:71 80.
- Rodríguez, J. P., T. D. Beard, Jr., E. M. Bennett, G. S. Cumming, S. Cork, J. Agard, A. P. Dobson, and G. D. Peterson. 2006. Trade offs across space, time, and ecosystem services. *Ecology and Society* 11:28.
- Saleh, A. 1993. Soil roughness measurement: chain method. *Journal of Soil and Water Conservation* 48:527 530.
- Scheffer, M., and S. R. Carpenter. 2003. Catastrophic regime shifts in ecosystems: Linking theory to observation. *Trends in Ecology and Evolution* 18:648 656.
- Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological feedbacks in

- global desertification. *Science* 247:1043–1048.
- Spilke, J., H. P. Piepho, and X. Hu. 2005. Analysis of unbalanced data by mixed linear models using the mixed procedure of the SAS System. *Journal of Agronomy and Crop Science* 191:47–54.
- Stafford Smith, D. M., N. Abel, B. Walker, and F. S. Chapin. 2009. Drylands: Coping with uncertainty, thresholds, and changes in state. Pages 171–195 in F.S. Chapin, G.P. Kofinas, and C. Folke, editors. *Principles of ecosystem stewardship: Resilience based natural resource management in a changing world*. Springer, New York, New York, USA.
- Stevens, D. L., Jr., and A. R. Olsen. 2004. Spatially balanced sampling of natural resources. *Journal of the American Statistical Association* 99:262–278.
- Suding, K. N., and R. J. Hobbs. 2009. Threshold models in restoration and conservation: A developing framework. *Trends in Ecology and Evolution* 24:271–279.
- USDA NRCS. 1991. Soil Survey of Canyonlands Area, Utah: Parts of Grand and San Juan Counties. U.S. Department of Agriculture, Natural Resources Conservation Service, Salt Lake City, Utah, USA.
- van de Koppel, J., M. Rietkerk, and F. J. Weissing. 1997. Catastrophic vegetation shifts and soil degradation in terrestrial grazing systems. *Trends in Ecology and Evolution* 12:352–356.
- Walker, B. H. 1992. Biodiversity and ecological redundancy. *Conservation Biology* 6:18–23.
- Walker, B. H., A. Kinzig, and J. Langridge. 1999. Plant attribute diversity, resilience, and ecosystem function: The nature and significance of dominant and minor species. *Ecosystems* 2:95–113.
- Walker, B. H., D. M. Stafford Smith, N. Abel, and J. L. Langridge. 2002. A framework for the determinants of degradation in arid ecosystems. Pages 75–94 in J. F. Reynolds and D. M. Stafford Smith, editors. *Global desertification: Do humans cause deserts?* Dahlem University Press, Berlin, Germany.
- Ward, A. D., and S. W. Trimble. 2004. *Environmental hydrology*. Second edition. Lewis Publishers, Boca Raton, Florida, USA.
- Warren, S. D. 2003. Synopsis: Influence of biological soil crusts on arid land hydrology and soil stability. Pages 349–360 in J. Belnap and O. L. Lange, editors. *Biological soil crusts: Structure, function, and management*. Second edition. Springer Verlag, Berlin, Germany.
- Welsh, S. L., N. D. Atwood, S. Goodrich, and L. C. Higgins. 2003. *A Utah flora*. Third edition, revised. Brigham Young University, Provo, Utah, USA.
- Westoby, M., B. Walker, and I. Noy Meir. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* 42:266–274.
- Whisenant, S. G. 1999. *Repairing damaged wildlands: A process oriented, landscape scale approach*. Cambridge University Press, Cambridge, UK.
- Williams, J. D., J. P. Dobrowolski, D. A. Gillette, and N. E. West. 1995. Microphytic crust influence on wind erosion. *Transactions of the American Society of Agricultural Engineers* 38:131–137.

APPENDIX

Additional results

Table A1. Water year precipitation at a long term weather station in the study area (elevation 1537 m), and numbers of plots by year and grazing status within the *biocrust*, *grass bare*, and *annualized bare* clusters.

Year	Water year precipitation		Biocrust Never or formerly grazed	Grass bare		Annualized bare	
	mm	% of MAP		Currently grazed	Formerly grazed	Currently grazed	Formerly grazed
2006	167	77.6	3	4	9	0	10
2007	283	132.1	10	7	1	2	3
2008	239	111.1	8	2	1	10	2

Note: MAP is mean annual precipitation.

Table A2. *P* values for main and interactive effects of cluster assignment and grazing status for the *grass bare* and *annualized bare* clusters, based on two way mixed effects ANOVA models with year as a random effect block.

Variables	Effect		
	Cluster	Grazing	Cluster \times grazing
Biological crust (cover %)	0.167	0.004	0.001
Perennial grasses and palatable shrubs (cover %)	0.001	0.393	0.405
Relative cover of invasive exotic annuals (%)	<0.001	0.236	0.573
Bare ground (cover %)	0.040	0.219	0.137
Annual grasses, native (cover %)	<0.001	0.654	0.117
Annual grasses, exotic (cover %)	0.016	0.010	<0.001
Annual forbs, native (cover %)	0.647	0.037	0.885
Annual forbs, exotic (cover %)	<0.001	<0.001	0.002
Bunchgrasses C ₃ (cover %)	0.001	0.286	0.372
Bunchgrasses C ₄ (cover %)	0.001	0.002	0.393
Rhizomatous grasses C ₄ (cover %)	0.002	0.328	0.563
Palatable shrubs (cover %)	0.634	0.316	0.977
Unpalatable shrubs (cover %)	0.133	<0.001	0.485
Total live cover, perennial plants (%)	0.001	0.393	0.405
Total live cover, vascular plants (%)	<0.001	0.085	0.480
Total live cover, all functional groups (%)	0.061	0.069	0.237
Functional group richness (no. of groups)	0.011	0.156	0.627
Species richness (no. of vascular plant species)	0.011	0.706	0.289
Soil aggregate stability (index)	0.565	0.038	0.070
Soil surface roughness (index)	0.035	0.060	0.004
Litter (cover %)	0.005	0.814	0.386
Magnetic susceptibility (10 ⁻⁶ SI units)	0.328	0.282	0.025
Median size of perennial canopy gaps (cm)	<0.001	0.109	0.855
Elevation (m)	0.164	0.025	0.001
Livestock dung (% frequency)	0.800	<0.001	0.046

Notes: *P* values less than 0.05 are in boldface. Means are in Table 2 in main text.

Table A3. Descriptive statistics for predicted rates of wind erosion (horizontal dust flux) at three wind velocities in plots categorized by cluster and grazing status.

Cluster	Grazing status (n plots)	Flux statistic	Wind velocity (m/s at 10 m height)		
			17.5	26.25	35.0
Biocrust	Never or formerly grazed (21)	Frequency (%)	0.0	0.0	9.5
		Mean	0.0	0.0	0.4
		CV	0.0	0.0	450.2
Grass bare	Formerly grazed (11)	Frequency (%)	0.0	18.2	81.8
		Mean	0.0	494.2	10,939.7
		CV	0.0	222.8	124.7
	Currently grazed (13)	Frequency (%)	38.5	53.8	100.0
		Mean	558.8	5571.6	21,527.6
		CV	231.2	151.7	104.3
Annualized bare	Formerly grazed (15)	Frequency (%)	6.7	33.3	86.7
		Mean	388.4	14,120.6	68,711.5
		CV	387.3	177.5	111.0
	Currently grazed (12)	Frequency (%)	8.3	25.0	58.3
		Mean	292.9	3897.6	20,604.4
		CV	346.4	297.5	166.0

Notes: Frequency indicates the percentage of plots with predicted fluxes greater than zero. Mean flux values are in g cm⁻¹ d⁻¹. CV = coefficient of variation.