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Soil aggregate stability was an uncertain predictor of ecosystem functioning in a temperate and semiarid grassland

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Abstract. We estimate rangeland managers assessing ecosystem health have measured soil stability >800,000 times. Our aim was to use quantitative data from a site in the Northern Great Plains, USA and a semi-quantitative literature search to demonstrate the robustness of soil stability as an indicator of ecosystem functioning. Empirical data included measurements of plant and soil properties along a local livestock grazing gradient to determine whether soil stability (e.g., % water-stable aggregates) explained primary productivity and soil water transport for a mixed-grass prairie site in the Northern Great Plains. We measured: annual net primary productivity (ANPP), elevation, % soil moisture, measures of soil stability, and soil water transport (field-saturated infiltrability and sorptivity) across points spanning a local gradient in livestock grazing intensity (none vs. light to moderate stocking rates; mean distance separating points = 39.9 m [range = 5.2–71.3 m]). Across the sampled gradient, variation in ANPP was best explained by a model with field-saturated infiltrability and % soil moisture. Infiltrability explained slightly more of the variation. We then determined that moderate amounts of variation in infiltrability were explained by ANPP, % soil moisture, and % water-stable aggregates. We determined that most of this variation was explained by ANPP and then soil moisture. Our empirical findings indicate that plant production was correlated with infiltration though we could not determine whether variation in plant production was caused by variation in infiltration or vice versa. We generally failed to show that soil stability (e.g., % water-stable aggregates) was a useful predictor of primary productivity and soil water transport. Our semi-quantitative literature review also indicated that soil stability was not a consistent predictor of either plant production or infiltration. The varying evidence reported here on whether soil stability is a predictor of ecosystem function illustrates the difficulty in identifying an indicator of ecosystem health that (1) is a predictor of ecosystem function across grassland types, (2) is sensitive to rangeland management, and (3) can be easily implemented by non-experts.

Key words: ecosystem function; indicators of rangeland health; rangeland; semi-arid grassland; soil stability; soil water transport; water infiltration; water-stable aggregates.

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Introduction

Efficacious natural resource management of forests and rangelands requires science-based information to help gauge the impact of multiple factors (e.g., management, climate, pest outbreaks) and inform decision making (Park and Cousins 1995, Andrews and Carroll 2001, Bone et al. 2014). Ideally, such measurements would be direct measures of ecosystem functioning (e.g., annual primary productivity) and easy to collect. Direct measurements of ecosystem functioning may not always be feasible to collect. Alternative measurements may be required which we refer to as "indicator" (proxy) variables. Indicator variables are assumed to be highly correlated with the ecosystem function(s) of interest. In some cases, metrics of ecosystem health are derived from composites of several indicators (Tongway and Hindley 2004*b*, Pellant et al. 2005), ideally a minimum set of indicators. In such cases, they are assumed to collectively represent an optimal and minimum set of variables for explaining the variance in the focal ecosystem function(s), each is individually important, and pairs have minimal collinearity. Scientists have helped determine the minimum set of variables for predicting ecosystem functions for row crop agriculture (Andrews and Carroll 2001, Idowu et al. 2008) and rangelands (Rezaei et al. 2006).

Several authorities have proposed that soil stability (e.g., water-stable aggregates) is a critical indicator of ecosystem processes (e.g., Rillig et al. 2002, Six and Paustian 2014). As part of standardized rangeland health assessments, land managers routinely use field kits to measure soil stability (i.e., rangeland health soil stability tests also called "slack test") in Australia (Tongway and Hindley 2004*b*) and the USA (Pellant et al. 2005). We estimate from a recent study (Herrick et al. 2010) that land managers in the western United States have measured soil stability using rangeland health soil stability tests >800,000 times as part of a multi-agency approach to quantify metrics of ecosystem health.

Recognizing that confidence in soil stability as an indicator depends on the amount of evidence validating its use, we were interested in uncovering the scientific evidence validating it as an indicator of hydrologic function (e.g., water infiltration) and biotic integrity (e.g., primary productivity). Our own attempts to uncover literature linking soil aggregate stability to measures of ecosystem function revealed inconsistencies, gaps, and controversy (e.g., Letey 1991, Young et al. 2001). For example, we used the Web of Science to perform a systematic review of a subset of relevant journals (*Ecological Applications*, *Geoderma*, *Plant and Soil*, and *Soil Biology & Biochemistry*) from 2001 to 2010. We identified 205 papers with the (topic) terms: soil aggregate*, water stable aggregat*, macroaggregat*, or soil stability (the asterisk [*] represents any group of characters, including no character). We determined that 53 (26%) papers included the term (water) "infiltration" used to characterize an important hydrologic function. The three journals with the most relevant articles were *Geoderma* (34), *Plant and Soil* (9), and *Soil Biology & Biochemistry* (9). Of the 53 total papers, only five provided quantitative data on soil (or aggregate) stability and water infiltration (Hoyos and Comerford 2005, Pilatti et al. 2006, Kapur et al. 2007, Shrestha and Lal 2008, Hallett et al. 2009). Two reported a positive relationship between water-stable aggregates and water infiltration (Kapur et al. 2007, Shrestha and Lal 2008), another found no relationship (Hoyos and Comerford 2005), and the others did not provide correlations of the two soil properties. Related searches with other journals uncovered empirical studies with landscape-level comparisons that found positive associations between soil stability and infiltration (Six et al. 2002, Shukla et al. 2003) but others found no relationship (Franzluebbers 2002, Wang 2010).

Soil stability is also hypothesized to be positively correlated with rangeland "biotic integrity" (Pellant et al. 2005) and by extension should be positively correlated with primary productivity. There is varying empirical evidence to support whether plants and/or mycorrhizae contribute to soil stability (i.e., soil aggregate stability). Some posit that roots and mycorrhizae stabilize macroaggregates (>0.25 mm; Tisdall and Oades 1982, Elliott 1986). Roots and mycorrhizae may have a synergistic positive effect on aggregate stability (Jastrow et al. 1998). Others propose that aggregates are stabilized by roots first and mycorrhizae second (Daynes et al. 2013). Arbuscular mycorrhizal fungi (AMF) abundance has been linked to aggregate stability values (Rillig et al. 2002, Wilson et al. 2009, Wu et al. 2014), and 80% of plant species associate with AMF (Smith and Read 2008). Others have speculated that AMF colonization, and presumably AMF abundance, declines with increasing intensity of herbivory (Wallace 1987, Gehring and Whitham 2002). This suggests measures of soil (aggregate) stability should be positively correlated with ANPP and a useful indicator of effects of management (e.g., grazing intensity) on rangeland health. However, soil stability values are not a consistent indicator of grazing (e.g., Bird et al. 2007, Li et al. 2007, Wang 2010, Duchicela et al. 2012). There are also cases of no AMF effects on grassland productivity. Specifically, grassland productivity was equal among plots treated with fungicide, to suppress AMF, relative to control plots (e.g., Hartnett and Wilson 1999, Yang et al. 2014) while the fungicide treated plots had lower soil aggregate stability values (Wilson et al. 2009). Thus, it is conceivable that adjacent plant communities (on the same soil series) but with varying compositions, AMF biomass, and soil stability values may actually have identical aboveground productivities, leaf litter, bare ground, and even hydrologic functions (Fig. 1A).

An informal sampling of the literature revealed varying empirical support for the prediction that measurements of soil stability are positively correlated with plant productivity. For example, one study identified positive relationships between measures of soil stability and plant cover (Chaudhary et al. 2009). This study, like related studies for the southwestern USA, compared measures of soil stability in vegetated vs. unvegetated areas. Another study of seeded plots also found a positive correlation between plant cover and water-stable aggregates (Rillig et al. 2002). A study of Iranian rangelands found a positive correlation between soil stability and plant yield, but determined other factors were better overall predictors of yield (Rezaei et al. 2006). Others have failed to correlate measures of soil stability with crop yield (Letey 1985) and percent cover of pasture plants (O'Dea 2007). There is also mixed evidence for whether belowground measures of plant productivity are (Jastrow et al. 1998, Rillig et al. 2002) or are not (Piotrowski et al. 2004, Barto et al. 2010) positively correlated with measures of soil stability. Comparisons across vegetated areas are likely to document idiosyncratic associations between productivity and soil stability because aggregate stabilization varies by specific pairings of plant and AMF species (Piotrowski et al. 2004) and both are likely to be spatially variable.

Our overall impression from the literature was that many accept that soil stability metrics are predictors of many ecosystem functions despite the limited and often varying empirical evidence that might be used to inform such a belief. As we contemplated what this meant to monitoring rangeland health, we became increasingly concerned that linkages between soil stability and measures of ecosystem function (when present) might primarily exist when comparing extremely different areas (e.g., vegetated vs. unvegetated patches [Fig. 1A vs. B], different habitats with different soils [or till vs. no till in row crop agriculture]) and might not be detectable across more moderately different areas relevant for monitoring and inferences on rangeland management practices (Fig. 1A). We were also concerned by the apparent absence of relevant empirical data to help interpret the likely importance of soil stability as an indicator of rangeland health in the expansive grasslands and shrublands (>22 million ha) of the Northern Great Plains (but see Wang 2010).

Here our aim was to use soil stability to predict two ecosystem processes, annual net primary productivity (ANPP) and measures of soil water transport. These two critical measures of ecosystem functioning are well integrated into monitoring systems for assessing rangeland health (e.g., Adams et al. 2003, Tongway and Hindley 2004*b*, Pellant et al. 2005). Our study emphasized validating the importance of soil stability because it is one of the only indicators (beside "compaction layer") used to quantify the three attributes of rangeland health (Pellant et al. 2005) and to provide data relevant to an expansive rangeland system. Predicated by interpretations in rangeland monitoring guides, we predicted measures of water infiltration and water-stable aggregates (and measures of rangeland health soil stability) would be positively correlated with ANPP. However, we anticipated multicollinearity between soil properties. Like related studies (Rezaei et al. 2006), we used correlation and multiple regression to detect relationships. Since our aim

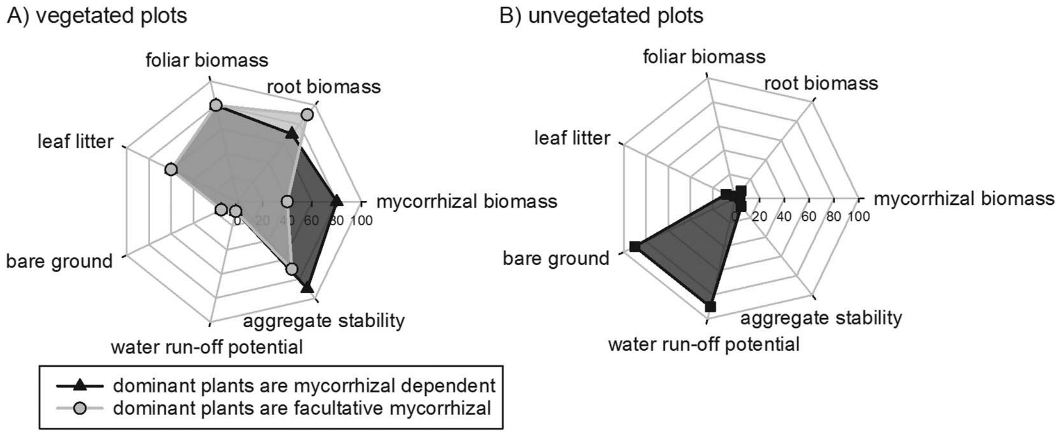

Fig. 1. Conceptual framework for the context dependence of soil aggregate stability as a predictor of ecosystem functioning (i.e., aboveground [foliar] production and potential water run-off) in grasslands/shrublands. The variation in several plant, soil, and hydrologic properties is illustrated with radar diagram plots. Each axis is the % of a theoretical maximum for each variable, and maxima were assumed equal across comparison groups (e.g., A vs. B). The illustration was designed to highlight two hypothetical scenarios. The comparison of vegetated (A) vs. unvegetated areas (B) indicates vegetated areas support greater values for foliar biomass, root biomass, leaf litter, mycorrhizal biomass, and aggregate stability while having minimal bare ground and water run-off potential. The second comparison is of two plant communities (A) with varying allocations to mycorrhizal fungi (which secondarily affect aggregate stability values) and roots. A critical assumption is that changes in mycorrhizal abundance may not affect foliar biomass at the community-level (e.g., Hartnett and Wilson 1999, Yang et al. 2014). The illustration also suggests a trade-off in carbon allocation to roots vs. mycorrhizal fungi.

was relatively focused, our study measured a relatively narrow set of factors unlike studies attempting to elucidate the minimum set of variables for predicting ecosystem functioning (e.g., Rezaei et al. 2006). Our comparison is different from others in more arid regions (Bird et al. 2007, Chaudhary et al. 2009) which compared the soil properties of vegetated vs. unvegetated areas. Here all of the sampled areas were vegetated, and the magnitude of the gradient was less extreme, but representative of typical grasslands in the region. Our study also differs from those that compared plant and soil properties across habitats. We sampled across local-gradients within a relatively small (0.3 ha, <72 m between sampled points) and homogeneous area which provided better control of abiotic factors (e.g., soil, climate, geomorphology) than studies across broader scales. This study should provide valuable scientific information on the utility of soil stability as an indicator for predicting primary productivity and soil water transport. These findings should also be relevant to policy on natural resource management monitoring in an expansive and intact grassland ecosystem predominantly used to support rangeland agriculture.

Methods

Study site and system

The study was conducted at the USDA-Agricultural Research Service's Fort Keogh Livestock and Range Research Laboratory (Fort Keogh, 22,000 ha), Montana, USA. Average annual precipitation was 34 cm (1937–2011). Peak annual productivity for this system occurs between June and July reflecting its dominance by perennial C_3 graminoids (Vermeire et al. 2009). Fort Keogh is centrally located in the Northern Great Plains Steppe ecoregion which is dominated by temperate and semiarid mixed-grass prairie that covers more than 22 million ha in five states in the USA and two Canadian provinces (Martin et al. 1998). These intact grasslands are used to support an estimated 11 million animal unit months of livestock grazing. Average annual precipitation for this region ranges from 25–50 cm with most occurring during the growing season (May and June).

The study site (46°18′20.8″ N, 105°58′42.8″ W) is an upland plain, has a gentle slope (1.05° slope), and has loamy soil (Eapa fine loam, frigid Aridic Argiustolls). The vegetation consisted of one of the dominant regional grassland types (*Hesperostipa comata*, *Bouteloua gracilis*, and *Carex filifolia*) (Martin et al. 1998). The study site was selected because it represented one of the most common grassland types in the Northern Great Plains (e.g., Coupland 1961, Martin et al. 1998), and it balanced our desire to control abiotic factors while still sampling across local gradients in plant productivity. Approximately one third of the area was within a livestock exclosure established in 1999 (Fig. 2). The other two thirds were equally divided among two adjacent pastures that on average were grazed at light to moderate levels (based on USDA-NRCS recommendations) primarily from May through October (Fig. 2). In terms of pasture area per cow, pasture "A" averaged 16 ha per cow (522 kg = 1,150 lbs) during May and October while pasture "B" averaged 14 ha per cow (between 1991–2011 the lowest unit area per cow per month was 2.2 and 2.8 ha per cow, respectively). The average distance separating sampling points within an area was 19.6 m, the average distance between points in the two adjacent pastures was 29.7 m, and the average distance separating points in the livestock exclosure relative to either pasture was 38.0 m. We fenced off the remaining sampling area (i.e., portions of pastures "A" and "B" shown in Fig. 2) from livestock in 2011 to prevent removal of pin flags and confounding of annual plant productivity measures.

We successively sampled measures of soil water transport, soil stability, and vegetation of 39 randomly selected points in 2013. Sampling intensity was approximately equal across the three areas (Fig. 2). The three areas had similar vegetation and soil properties, but had distinct elevations (Fig. 2) and soil moistures (Table 1). A main difference was that the livestock exclosure had drier soils and was up slope (1.05°) of the two adjacent pastures (Table 1, Fig. 2).

The aim of our study was to validate the importance of soil stability as an indicator of ecosystem functioning while incorporating a limited number of additional variables and did not include bulk density. Livestock are routinely thought to affect soil bulk density and possibly other soil properties (e.g., soil nitrogen from urine/feces). While we did not directly measure bulk density, we can infer from the infiltrability data, which should be negatively correlated with bulk density, that there was no difference between the ungrazed exclosure relative to the grazed pastures (Table 1, overlap of 95% CI). A study in North Dakota found that bulk density was negatively related to ANPP ($r = -0.44$; Wang 2010). However, Wang found no significant relationship between bulk density and infiltration ($r = 0.16$) and determined bulk density did not vary across grazing treatments. Others noted that grazing effects are not consistent across large spatial scales and vary by system (e.g., Milchunas and Lauenroth 1993). We suspect the importance of bulk density as a predictor variable of ecosystem functioning is context dependent. For example, grazer effects on bulk density are more likely in more mesic systems with greater forage production because the number of animals per unit area is likely to be much greater and soils are more often moist and vulnerable to compaction.

Soil water transport

Soil water transport was quantified using a Cornell Sprinkle Infiltrometer (CSI) which is capable of measuring time-to-runoff, sorptivity, and field-saturated infiltrability (Ogden et al. 1997). We focused on sorptivity, a universal soil hydraulic property describing early infiltration, and field-saturated infiltrability. These are useful measures because they quantify different phases (early vs. late) of water infiltration and are independent of simulated rainfall rate (Ogden et al. 1997). Time-to-runoff was disregarded because it is influenced by rainfall rate which inevitably varies during field measurement.

In brief, the CSI consisted of a portable rainfall simulator placed onto a single 235 mm (inner diameter) infiltration ring permitting the simulation of rainfall at predetermined rates. The large volume of deaired water required was obtained by cooling water initially heated to 51°C in a conventional domestic hot water system. CSI measures were taken for all 39 locations during 5–13 June 2013. We avoided sampling locations with obvious forms of soil disturbance (e.g., cattle trails or harvester ant mounds) that were

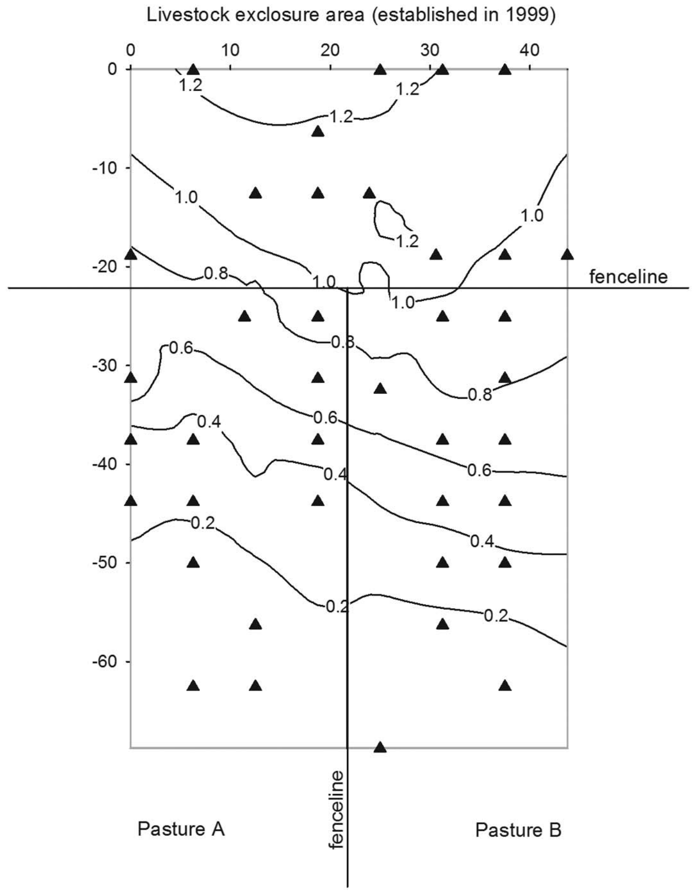

Fig. 2. Map of the sampled area. Sampling was divided equally among three adjacent areas: a livestock exclosure and two adjacent pastures (A, B) grazed annually by cows. Solid triangles identify individual sampling points, axes' units are meters, and contour lines represent elevation gain (m) relative to the lowest point in the sampled area.

Table 1. Average (±95% CI) plant and soil properties for three adjacent subsections (a grazing exclosure and two adjacent pastures; Fig. 2) of a relatively flat upland plain site of mixed-grass prairie vegetation sampled in 2013. Data were pooled and used in subsequent analyses (Tables 2–4) to determine whether soil stability was a strong predictor of ANPP and infiltrability. Variation among the three subsections can be interpreted by whether 95% CI overlap.

Site properties	Grazing exclosure	Pasture A	Pasture B
ANPP	20.6 ± 2.68	22.7 ± 2.40	19.94 ± 1.91
Sorptivity	1.18 ± 0.12	1.09 ± 0.13	1.09 ± 0.14
Infiltrability	45.2 ± 15.6	62.3 ± 15.4	42.8 ± 10.2
SmWSA	63.9 ± 2.5	68.5 ± 3.0	68.0 ± 4.6
MedWSA	50.9 ± 4.3	55.5 ± 4.7	58.0 ± 3.0
Soil moisture	23.4 ± 2.0	28.6 ± 1.5	28.1 ± 1.0
Elevation	1.1 ± 0.07	0.4 ± 0.14	0.5 ± 0.17
Subsurface soil stability	5.4 ± 0.4	5.8 ± 0.2	5.5 ± 0.5

Notes: ANPP is annual net primary productivity (g × 0.2 m^{-2}), sorptivity is a unitless measure of early infiltration independent of rainfall rate, infiltrability is field-saturated infiltrability (mm × hr^{-1}), smWSA is small size class (0.25–1 mm) of % water-stable aggregates, medWSA is medium size class (1–2 mm) of % water-stable aggregates, volumetric soil moisture (%) [5–13 June 2013], elevation (meters) relative to the lowest point in the study site (see Fig. 2), and subsurface soil stability is a field assessment of soil stability commonly used by land managers to assess rangeland health (Pellant et al. 2005).

Table 2. Pearson product-moment correlations (r) and p-values (in parentheses) among factors measured in a mixed-grass prairie in 2013. In bold are correlations coefficients with significant ($\alpha \leq 0.05$) p-values.

Site properties	ANPP	Infiltration	Sorptivity	SmWSA (0.25–1 mm)	MedWSA (1–2 mm)	Soil moisture	Elevation
Infiltration	**0.39 (0.02)**	...	...	...	...	...	...
Sorptivity	0.10 (0.54)	**0.42 (0.01)**	...	...	...	...	...
SmWSA	0.26 (0.12)	**0.42 (0.01)**	0.02 (0.89)	...	...	...	...
MedWSA	0.03 (0.85)	0.04 (0.82)	−0.20 (0.22)	0.13 (0.44)	...	...	...
Soil moisture	0.23 (0.16)	0.08 (0.66)	**−0.40 (0.013)**	**0.34 (0.034)**	−0.02 (0.92)	...	...
Elevation	−0.26 (0.13)	**−0.44 (0.01)**	0.05 (0.76)	**−0.59 (<0.001)**	−0.10 (0.55)	**−0.56 (<0.001)**	...
Subsurface soil stability	0.28 (0.14)	0.28 (0.14)	0.13 (0.49)	**0.47 (0.01)**	0.26 (0.17)	0.29 (0.12)	**−0.46 (0.01)**

Note: Factors are defined in Table 1.

likely to affect soil water transport. The instruction manual notes that measurements may be influenced by soil moisture (Ogden et al. 1997) and prior trials indicated likely spatial variability in soil moisture, especially as the area dries (K. O. Reinhart, *unpublished data*). To avoid this, we did two things. First, we conducted all measurements within a relatively short period of time. Second, we sampled in the early summer following recent rainfall events (total monthly precipitation is typically greatest in June) when conditions were still relatively moist and presumably uniform. To further monitor and possibly statistically control for this potential problem, we measured the percent volumetric water content (FieldScout TDR 100 Soil Moisture Meter; Spectrum Technologies, Aurora, Illinois, USA) from three locations immediately surrounding the CSI ring the morning of measurement. We detected a negative correlation between mean % soil moisture and sorptivity, but no relationship between % soil moisture and field-saturated infiltrability (Table 2). Further details on the equipment, standard methods, and formulas to determine sorptivity and field-saturated infiltrability can be found elsewhere (Ogden et al. 1997).

Soil stability tests

We measured soil stability using (1) measures of water-stable (soil) aggregates for two macroaggregate size classes (0.25–1 and 1–2 mm; e.g., Kemper and Rosenau 1986) and (2) rangeland health tests of surface and subsurface soil stability (Pellant et al. 2005). We used these two rather distinct methods for measuring soil stability to ensure the findings were broadly relevant, to determine if the two measures were correlated, and to determine the best predictors of two ecosystem functions. To help ensure measurements of soil stability were directly relevant to measures of soil water transport, we sampled soil from within the infiltration ring area ($433.7\ cm^2$) to quantify water-stable aggregates. Plant material was clipped and detritus was removed from the area prior to collection. Studies that quantify soil stability routinely collect soil samples representing the surface soil (e.g., 0–5, 0–20 cm; e.g., Wilson et al. 2009, Duchicela et al. 2012). We carefully extracted a larger than needed soil sample using a narrow bladed (13 cm) shovel on 24 June 2013. The soil sample was then cut down to $10 \times 7 \times 6$ cm (10 cm length = 0–10 cm soil depth). This resulted in a cuboid soil sample free of soil compaction from the extraction process. The soil samples were then brought back to the lab and lightly massaged to break up the soil along natural breakpoints to prevent the sample from cementing together during air drying. The samples were air dried to constant weight and stored in polyethylene bags until dry-sieved.

The samples were dry sieved in 20.3-cm sieves on a mechanical sieve shaker (RO-TAP, RX-29; W.S. Tyler, Mentor, Ohio, USA). Each aggregate size class (1–2 and 0.25–1 mm) was collected individually from largest to smallest. The shaker accommodated four nested pairs of a sieve atop a collection pan. Samples were shaken for 1 minute (278 oscillations $\times$ min^{-1}, 150 taps $\times$ min^{-1}). Individual samples representing separate aggregate size classes per sample were then stored in polyethylene bags until wet-sieved.

The percentage of water-stable aggregates was determined for two macroaggregate size classes (0.25–1 and 1–2 mm) following the operators manual for a wet-sieving apparatus (Eijkelkamp, Giesbeek, Netherlands) which is similar to existing methods (e.g., Kemper and Rosenau

1986). Aggregates from the dry sieving portions (1 g) were placed onto mesh screens a quarter the size of the smallest aggregates (0.0625 and 0.25 mm, respectively). Four subsamples per size class were used to estimate percent water-stable aggregates per sample. Subsamples were rewetted by capillary action for 10 min. Subsamples were wet-sieved for 8 min on a mechanical wet-sieving apparatus (stroke = 1.3 cm, at 34 strokes $\times$ min^{-1}). After wet-sieving, the material collected in the cans was washed gently into weigh boats. Cans were then re-filled with a dispersing agent (0.2 g NaOH), and the coarse material persisting on the sieve was wet-sieved again. At the end of this wet-sieving, a watchman was used to try and manually breakup any persisting coarse material. The sample was then wet-sieved for an additional two minutes in the dispersing agent solution. The material in the can containing the dispersing solution was then washed gently into another set of weigh boats. The weigh boats were then dried in a convection oven at 110°C, until the water had evaporated, and weighed. The formula for determining the percent water-stable aggregates (corrected for coarse material) was

$$\mathrm{WSA}_i = \frac{W_2 - 0.2}{W_1 + (W_2 - 0.2)} \times 100$$

where WSA_i = water-stable aggregation for each size class i; W_1 = mass of material collected in the cans after the first wet sieving in water in size class i; W_2 = mass of the material collected in the cans after the second wet sieving in the dispersing agent (weighing 0.2 g) in size class i.

We also measured surface and subsurface soil stability with a soil stability field kit (Synergy Resource Solutions, Belgrade, Montana, USA) following the methods for monitoring rangeland health indicators (Pellant et al. 2005). We made three surface and subsurface measurements (subsamples) from 33 of the 39 points on 20 June 2013. Measurements were made in undisturbed areas immediately outside of where water infiltration rings had been inserted. The field kit provides reliable measures of ped stability relative to laboratory measurements of similar peds (Herrick et al. 2001). However, selection of peds may introduce varying forms of bias. An issue that we encountered was that the test requires an extraction of a soil ped of sufficient size and stability for measurement. These were often challenging to collect, and we found ourselves deliberately selecting areas with visible biological soil crusts. In addition, we often had to mist subsurface soil prior to cutting and extracting peds. The instructions allow for misting, but call for air drying the ped prior to testing since moisture content can affect ped stability. We briefly air dried peds. The original methods description (Pellant et al. 2005) did not specify a procedure for air drying peds. The lack of standards for air drying peds in the field is likely to add to data variability and bias. The surface soil stability values had little variability and were not incorporated into further analyses. The subsurface values contained sufficient variability to be incorporated into analyses (data range: 2–6 [categorical data's potential range: 0–6]).

Annual net primary production

Plant productivity and community composition were assessed from clipped 0.1-m^2 quadrats. Since prior soil measurements generated both soil disturbance and matted down vegetation, subsamples were clipped from two quadrats adjacent to (~1–2 m) each soil measurement. We selected areas with unmatted vegetation and similar species compositions based on visual comparisons. The vegetation in the quadrat was clipped and separated by dominant species and functional groups on 15–17 July 2013. Dominant species included four graminoids (*Carex filifolia*, *Koeleria macrantha*, *Hesperostipa comata*, and *Pascopyrum smithii*) and one cactus (*Opuntia polyacantha*). Additional species were grouped as annual grasses (*Bromus* spp.), forbs, other grasses, or shrubs. Plant material was dried to constant weight, separated into current-year and older material, and weighed. A correction calculation (0.2 $\times$ fresh weight), derived by researchers at Fort Keogh, was used to calculate the dry weight of *Opuntia*. Data for subsample quadrats were summed per location (g $\times$ 0.2 m^{-2}).

Analysis

We provide descriptive comparisons (mean and 95% CI; Table 1) of the three sampled areas. The primary purpose of this study was to uncover plant-soil correlative associations and not to test livestock management treatments

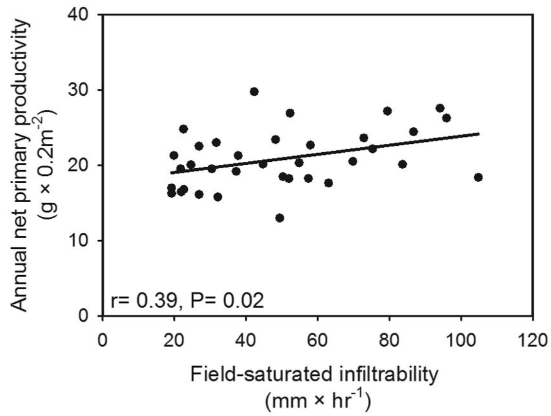

Fig. 3. Relationship between field-saturated infiltrability and plant productivity for a temperate and semiarid grassland. Correlation coefficient and P-value are shown (from Table 2).

(which were not replicated). Prior to analyses, outliers were identified and removed based on the maximum normal residual method (Snedecor and Cochran 1989) and $P < 0.05$. We then used multiple linear regressions (MLR) to identify likely factors driving variability in ANPP and soil water transport. We were specifically interested in seeing whether measures of soil stability were correlated with ANPP and infiltration. Many variables were linearly correlated (Table 2), which raised concerns of multicollinearity among variables. MLR was used to identify independent variables that best predicted ANPP and field-saturated infiltrability. MLR analyses were performed using Proc Reg with SAS version 9.3 (SAS Institute, Cary, North Carolina, USA). Best MLR models were determined by Akaike information criterion (AIC) scores. Residual analyses included visual confirmation that the assumptions of normality, homoscedasticity, and independence of residuals were not violated. Multicollinearity analyses included quantifying variance inflation and condition index scores. To help differentiate the relative importance of variables with mild degrees of collinearity that may be due to either a true synergy among variables or the confounding of one by another, we determined squared semi-partial correlation coefficients of each independent variable. MLR for the dependent variable ANPP included the independent variables: field-saturated infiltrability, sorptivity, two size classes of % water-stable aggregates, soil moisture measured in early June, and elevation. MLR for the dependent variable field-saturated infiltrability included the independent variables: ANPP, two size classes of water-stable aggregates, soil moisture measured in early June, and elevation. Separate MLRs were conducted as above but with subsurface soil stability as an additional independent variable. These analyses were performed separately because this independent variable was missing data ($n = 28$ instead of 35 or 37, respectively). This was an issue for explaining variation in ANPP and infiltration. For the best fit model for ANPP, the restricted data set identified an additional independent variable (water-stable aggregates [1–2 mm]) not identified as an important parameter in the analysis of the full data set. For the best fit model for infiltration, the best model selected subsurface soil stability in place of water-stable aggregates (0.25–1 mm).

Results.—The grassland soils exhibited moderate levels of soil stability based on % water-stable aggregates (e.g., Table 1) which ranged from 54% to 86% for macroaggregates between 0.25-1 mm and 39–70% for 1–2 mm macroaggregates. A related study in neighboring North Dakota measured % water-stable aggregates across varying landscape positions and grazing management treatments and detected less data variability and a grand mean value of 92% water-stable aggregates (1–2 mm size class; Wang 2010). We also found no apparent differences in % water-stable aggregates between the grazed and ungrazed portions of the site (Table 1; overlap of 95% CI). Annual net primary productivity (ANPP) and infiltrability also did not differ among the three adjacent areas (Table 1), but exhibited ecologically important variation with ANPP values ranging from 13 to 39 $g \times 0.2\ m^{-2}$ and infiltrability values ranging from 19 to 155 $mm \times hr^{-1}$ (Fig. 3).

We identified multiple correlations between ANPP, measures of infiltration, elevation, soil moisture, and water-stable aggregates (Table 2, Figs. 3 and 4) indicating possible problems with multicollinearity among variables. Despite the relatively small difference (1.26 m) in elevation over the 0.3 ha site (Fig. 2), we determined soil moisture was negatively correlated with elevation (Table 2). One size class of % water-stable aggregates (0.25–1 mm) was also negatively

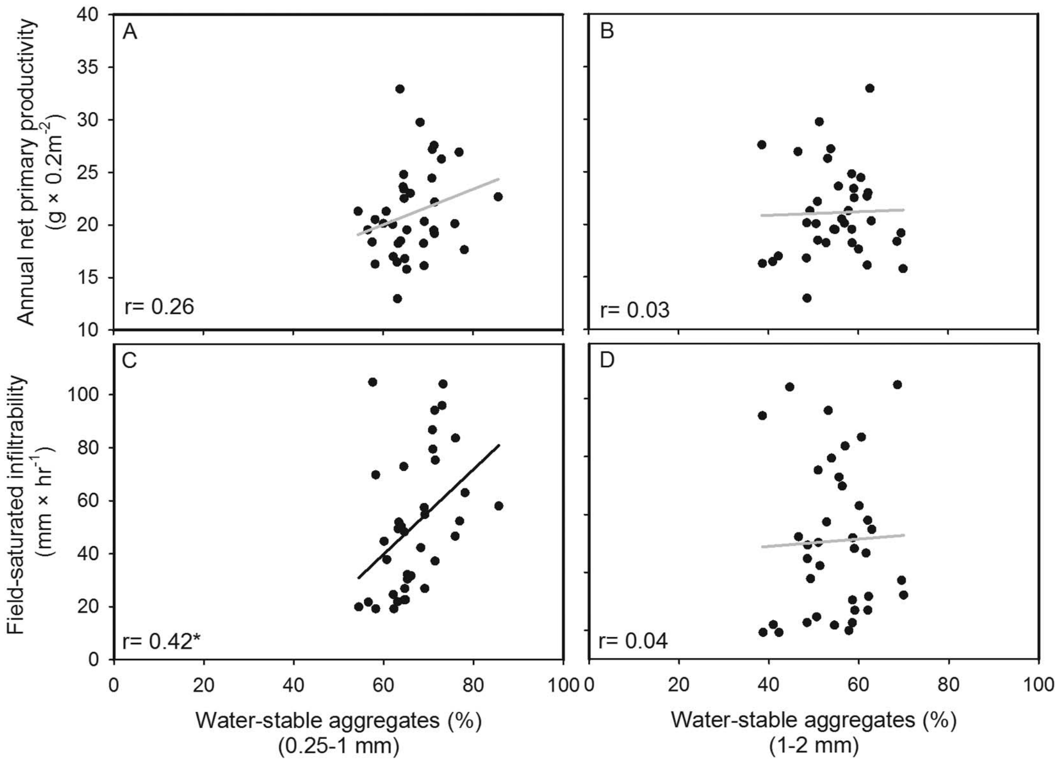

Fig. 4. Relationships between % water-stable aggregates and measures of plant productivity (A, B) and field-saturated infiltrability (C, D). Correlation coefficients are provided (see Table 2) which can be contrasted with multiple linear regression results (see Tables 3 and 4). Best-fit lines are black when linear regressions were significant ($P \leq 0.05$) or gray when nonsignificant ($P > 0.05$).

correlated with elevation and to a lesser extent positively correlated with soil moisture (Table 2).

A moderate amount of variation in ANPP was best explained by a simple multiple linear regression (MLR; $F_{2,35} = 10.81$, $P = 0.0002$, $R^2 = 0.40$; Table 3) and included the parameters field-saturated infiltrability and soil moisture. We detected no direct collinearity between measures of infiltrability and soil moisture (condition index scores < 15.4), however, both were negatively correlated with elevation (e.g., Table 2). The MLR model indicated that both independent variables were positively correlated with ANPP. We then evaluated the importance of the individual parameters by comparing (1) their standardized beta coefficients which standardize the parameter estimates correcting for variation in scales per parameter and (2) their squared semi-partial correlation coefficients which characterize the relationship between the dependent and independent variable after the contributions of the other independent variables have been removed (e.g., Table 3). Our comparison of beta coefficients indicated that infiltrability was only a slightly better predictor (0.60 > 0.38) than soil moisture (Table 3). Evaluation of the squared semi-partial correlation coefficients ($_{semi}r^2$) determined a greater difference between the two independent variables (Table 3). We can interpret that removal of the infiltrability parameter would reduce the overall model R^2 from 0.40 to 0.06 (0.40−0.34 = 0.06) while removal of the soil moisture parameter would reduce the R^2 from 0.40 to 0.26 (Table 3). Despite this, we urge caution in interpreting causation of infiltrability on ANPP since the comparisons are correlative. When subsurface soil stability was added as an independent variable, the data set was reduced and a slightly different best model was selected which included infiltrability, soil moisture, and water-stable aggregates (1–2 mm). However, water-stable aggregates (1–2 mm) was only

Table 3. Best multiple regression model, based on Akaike's information criterion (AIC) values, to explain variation in annual net primary productivity (ANPP).

Independent variables	Parameter estimates (beta coefficients)	t-value	$F_{2,35}$	P	r^2	$_{semi}r^2$
Infiltrability	0.07 (0.60)	4.32	...	0.0001	0.36	0.34
Soil moisture	0.46 (0.38)	2.75	...	0.01	0.19	0.14
Total	...	...	10.81	0.0002	0.40	...

Notes: Infiltrability is field-saturated infiltrability and $_{semi}r^2$ is squared semi-partial correlation coefficients. Significance of linear models was tested with ANOVA.

identified as a useful predictor of ANPP with the reduced data set, was negatively related to ANPP (after accounting for the other parameters), and was the weakest predictor of the three (analysis not shown).

Moderate amounts of variation in field-saturated infiltrability were best explained by a MLR model ($F_{3,35} = 10.01$, $P < 0.0001$, $R^2 = 0.48$) including parameters for ANPP, % soil moisture, and one size class of water-stable aggregates (0.25–1 mm). Two independent variables (soil moisture and water-stable aggregates) were negatively correlated (Table 2), but were determined to have only mild collinearity (condition index scores < 11.1). A comparison of standardized beta coefficients indicated that ANPP was a slightly more influential variable for predicting variation in infiltrability than either soil moisture or water-stable aggregates (Table 4). Evaluation of the squared semi-partial correlation coefficients ($_{semi}r^2$) which better (but not fully) accounts for their collinearity revealed that ANPP contributed the most ($_{semi}r^2 = 0.26$) and water-stable aggregates ($_{semi}r^2 = 0.08$) the least to explaining variation in infiltrability. The MLR results revealed possible drivers of infiltrability that could not have been inferred by examining the partial correlation coefficients alone (Table 2). When subsurface soil stability was added as an independent variable, the data set was reduced, and a qualitatively similar model was selected. The main distinction between best models was that water-stable aggregates (0.25–1 mm) was replaced by subsurface soil stability as a variable in the best overall model (analysis not shown).

Discussion

Though we assumed confidence in soil stability as an indicator would be tied to the amount of evidence validating its use, our semi-quantitative literature search revealed a widespread belief that measures of soil stability are important indicators of multiple ecosystem functions despite the often limited and varying levels of empirical support. Links between soil stability and annual net primary productivity (ANPP) or field-saturated infiltrability appear context dependent (see below and Fig. 1). Despite this, soil stability tests have been frequently used to quantify the three attributes of rangeland health (>800,000 times in the USA; Herrick et al. 2010) over vast areas of grassland and shrubland. We were then interested in assessing the importance of soil stability as an indicator in mixed-grass prairie of the Northern Great Plains—an area of expansive grasslands but limited relevant empirical data (but see Wang 2010). Contrary to

Table 4. Best multiple regression model, based on Akaike information criterion (AIC) values, to explain variation in field-saturated infiltrability. Independent variables included ANPP, two size classes of water-stable aggregates, soil moisture, and elevation.

Independent variables	Parameter estimates (beta coefficients)	t-value	$F_{3,35}$	P	r^2	$_{semi}r^2$
ANPP	4.11 (0.53)	4.00	...	0.0004	0.33	0.26
Soil moisture	−4.32 (−0.46)	−3.47	...	0.0015	0.27	0.19
Water-stable aggregates (0.25–1 mm)	1.56 (0.31)	2.29	...	0.029	0.14	0.08
Total	...	...	10.01	<0.001	0.48	...

Notes: ANPP is annual net primary productivity and $_{semi}r^2$ is squared semi-partial correlation coefficients. Significance of linear models was tested with ANOVA.

predictions, we found limited evidence that soil stability was a useful predictor of ANPP or field-saturated infiltrability. Specifically, we failed to detect correlations between ANPP and measures of soil stability (i.e., subsurface soil stability and % water-stable aggregates; Table 2, Fig. 4). Others have found no relationship between % water-stable aggregates and (1) crop yield (Letey 1985), (2) % cover of pasture plants (O'Dea 2007), and (3) root production (Piotrowski et al. 2004, Barto et al. 2010). A related study in North Dakota also found no significant relationship between ANPP and mean (aggregate) mass diameter (another measure of soil stability, $r = -0.24$; Wang 2010). The lack of relevant regional empirical evidence and previously mentioned uncertainties may explain why rangeland health assessments for the Canadian portion of the Northern Great Plains do not include soil stability tests (e.g., Adams et al. 2003).

We used multiple-linear regression to identify the best predictors of ANPP. We determined field-saturated infiltrability was a main predictor of ANPP (Fig. 3, Table 3). Others have noted that infiltration is consistently greater in vegetated than unvegetated patches (Boix-Fayos et al. 1998) and included infiltration in their best model for predicting ANPP (Wang 2010). For our measure of hydrologic function, we found mixed evidence that measures of soil stability were useful predictors of field-saturated infiltrability. We detected a positive correlation between water-stable aggregates (0.25–1 mm size class) and field-saturated infiltrability (Table 2, Fig. 4C), and the best MLR model explaining variation in infiltrability (Table 4) included this same measure of soil stability. However in a previous year (2012) with different sampling methods (e.g., systematic sampling), we identified a negative correlation between water-stable aggregates (2–4 mm) and infiltrability ($r = -0.49$, $n = 21$, $P = 0.02$) and no correlation between small (0.25–1 mm) and medium (1–2 mm) sized water-stable macroaggregates and infiltrability ($P \geq 0.24$; K. O. Reinhart, *unpublished data*). Here the MLR analysis also revealed that the water-stable aggregates variable was a relatively weak predictor (squared semi-partial correlation coefficients, ${}_{\text{semi}}r^2 = 0.08$) of infiltrability after accounting for other variables such as ANPP (${}_{\text{semi}}r^2 = 0.26$) and soil moisture (${}_{\text{semi}}r^2 = 0.19$, Table 4). Contrary to simple Pearson correlation coefficients (Table 2), the MLR results revealed that ANPP was actually the best predictor of infiltration after accounting for other variables. This suggests that monitoring plant cover or related measures will likely explain as much or more of the variation in infiltration (and run-off; Hart and Frasier 2003). Other factors (e.g., invertebrate tunnels, root structure/biomass) related to preferential flow are likely to explain more variation in infiltrability than measures of soil stability (Letey 1991). Thus, our study generally failed to show that soil stability was a strong predictor of either ANPP or infiltrability.

An evaluation of the literature revealed the links between soil stability measures and ANPP (narrative review) or infiltration (systematic review) were context dependent—depended on the range of variation, magnitude of variation among the data, and inclusion of samples from highly disturbed areas or areas with different environments. Stands of perennial plants were likely to produce moderate to high levels of soil stability (Jastrow et al. 1998, Hallett et al. 2009, Wang 2010) whereas areas that were persistently without vegetation were likely to have low levels of soil stability (Bird et al. 2007, Chaudhary et al. 2009). Studies comparing vegetated with unvegetated plots were likely to detect significant differences in soil stability measurements while comparisons across less extreme gradients might not (Fig. 1). When the variability among soil stability values is profound (Fig. 1A vs. B; Six and Paustian 2014) then soil stability measures are likely a useful indicator of dramatic shifts in ecosystem functioning and health. Dramatic shifts in ecosystem properties are by definition the easiest to document and may be accounted for equally well or better with a variety of other soil or plant variables than soil stability. Rates of aggregate stabilization are also known to vary by specific pairings of plant and AMF species (Piotrowski et al. 2004) which adds an additional layer of uncertainty as to whether soil stability can be a robust indicator of rangeland health. We interpret that measures of soil stability are unlikely to detect shifts in local ecosystem processes for relatively stable ecosystems responding to changes in management practices that are unlikely to cause rapid changes in perennial plant cover and community composition (Wang 2010). Ecosystems like the Northern Great Plains, with relatively stable grassland plant communities and cover by perennial plants may require new indicators of ecosystem processes to improve ecological and economic sustainability of regional rangelands.

We failed to determine that (sub)surface soil stability, a simplified soil stability test (Pellant et al. 2005), were useful predictors of ANPP or infiltration. While soil stability was positively correlated with water-stable aggregates (0.25–1 mm, $r = 0.47$), we found that other factors explained disproportionately more of the variation in ANPP and infiltration than measures of soil stability (e.g., see ${}_{\mathrm{semi}}r^2$ values for Table 4). This finding is similar to work in Iranian rangelands, which determined that while soil stability was positively correlated with yield that other variables were among the best minimum data set for explaining variation in rangeland yield (Rezaei et al. 2006). We did, however, identify a positive correlation between subsurface soil stability and one measure of water-stable aggregates (0.25–1 mm size class; $r = 0.42$, Table 2). This is interesting since we are unaware of other studies that have correlated measures of soil stability based on two entirely different methods for isolating (e.g., dry sieving vs. ped selection in the field) and re-wetting aggregates/peds. Our results suggest that the smaller (0.25–1 mm) macroaggregate size class is more ecologically important than the larger (1–2 mm) size class (Table 2, Fig. 4; and K. O. Reinhart, *unpublished data*) in temperate grasslands.

These findings should help improve methods for inventorying the ecosystem health of expansive grassland and shrubland systems used primarily as rangelands. For example, measures of soil stability are currently one of many indicators used to enumerate two of three metrics of ecosystem health for the Landscape Function Analysis (Tongway and Hindley 2004*b*) system and all three interrelated attributes of rangeland health for the Indicators of Rangeland Health method (Pellant et al. 2005). Implied is that both systems assume measures of soil stability are positively correlated with metrics of ecosystem function and are among the best minimum set of indicators for predicting variation in measures of ecosystem health/functioning. Soil stability tests are considered a direct measure of soil or site stability (Tongway and Hindley 2004*b*, Pellant et al. 2005). However, these metrics may possibly be less important than other relevant indicators (e.g., bare ground cover, litter cover, gullies). For example, research in shortgrass and mixed-grass prairie determined that variation in surface water run-off was largely explained by either leaf litter ($r^2 = 0.99$ and 0.94, respectively) or bare ground ($r^2 = 0.99$ and 0.83, respectively; Hart and Frasier 2003). Here we determined that soil stability measures were poor predictors of ANPP and infiltrability, which may add error to existing metrics of ecosystem health. Others have expressed similar opinions that measures of soil stability may not be a useful predictor of soil functional processes (Letey 1991, Young et al. 2001). Based either on the data presented here, summaries of available literature, or the metrics of ecosystem health themselves, we cannot conclude soil stability measures are a robust indicator. The systems for measuring ecosystem health either lack appropriate validation or rely on composites of six to eight indicator variables which complicates deciphering the best minimum set of predictor variables (but see Rezaei et al. 2006). We are aware of one study that tested the power of these and other soil properties to predict rangeland yield (Rezaei et al. 2006). Their first model determined ($R^2 = 0.67$) that rangeland production was primarily driven by (1) soil profile effective thickness and (2) total nitrogen percentage—soil stability was not among the best predictors despite it being positively correlated with yield. We suspect it is possible to determine the best minimum set of predictor variables for the Landscape Function Analysis monitoring system which has validated its composite metrics of ecosystem functioning (e.g., Tongway and Hindley 2004*a*). An analysis of existing indicator data would be extremely useful and help quantify the relative importance of individual indicator variables. Such an analysis could help reveal the best minimum set of indicators to monitor rangelands and assess change in ecosystem functioning.

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