

Testing rangeland health theory in the Northern Great Plains

Kurt O. Reinhart  | Matthew J. Rinella | Richard C. Waterman  | Mark K. Petersen  | Lance T. Vermeire 

United States Department of Agriculture-
Agricultural Research Service, Fort Keogh
Livestock & Range Research Laboratory,
Miles City, Montana

Correspondence

Kurt O. Reinhart
Email: kurt.reinhart@ars.usda.gov

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Abstract

1. Correctly assessing whether rangeland ecosystem services are stable, improving, or degrading is of global importance. Soil aggregate stability (SAS) is widely used to infer rangeland health, partly because high SAS is thought to reduce run-off by increasing infiltration. We studied the sensitivity of SAS to grazing and other disturbances, the effects of SAS on infiltration, and the utility of alternative indicators of infiltration in the Northern Great Plains.
2. To test grazing effects on SAS, we compared SAS between paired areas that were lightly to moderately grazed or excluded from grazing for 6 years. Additionally, we compared SAS between grazed and not-grazed plots of a 2-year controlled grazing experiment with moderate and severe grazing. We also applied herbicide, mowing, and fungicide treatments to test SAS responses to disturbances more generally, as well as effects of SAS and other factors on infiltration. To more generally test for a SAS–infiltration relationship, we performed a meta-analysis of our data combined with other data from the region.
3. Grazing often reduced stability of small macroaggregates (0.25–1 mm) in the controlled grazing experiment but not the paired grazing area experiment. Grazing had no detectable effect on SAS of larger macroaggregates (1–2 mm). Herbicide tended to reduce SAS, and mowing sometimes increased SAS. Infiltration exhibited high plot-to-plot variation and was not significantly affected by treatments. Variation in infiltration was best explained by plant community composition variables and was not explained by either SAS or other soil properties. Our meta-analysis revealed no general SAS–infiltration relationship.
4. *Synthesis and applications.* Our findings counter prevailing expectations that soil aggregate stability (SAS) is consistently sensitive to rangeland disturbance(s) and a leading indicator of soil water transport. Plant community composition properties were better predictors of infiltration. Our findings support the theory that excessive grazing increases the prevalence of a grazing tolerant species, which was associated with low levels of infiltration irrespective of SAS.

KEYWORDS

biocompaction, bioturbation, ecosystem management, hydrologic function, locally applicable indicator, rangeland health, soil aggregate stability, soil health

1 | INTRODUCTION

Rangelands are the most widespread terrestrial biome in the world (Ellis & Ramankutty, 2008), and determining how to manage these systems requires accurate methods for assessing ecosystem function and health (Rezaei, Gilkes, & Andrews, 2006; Tongway & Hindley, 2004). For more than a decade, USA federal agencies have relied on the “Interpreting Indicators of Rangeland Health” (IIRH) system to detect early signs of rangeland degradation on >800,000 sites (Herrick et al., 2010). IIRH uses 17 indicator variables to score three “attributes” (i.e., biotic integrity, hydrologic function, and site stability) of rangeland health (*sensu stricto* Pyke, Herrick, Shaver, & Pellant, 2002; Pellant, Shaver, Pyke, & Herrick, 2005). The IIRH assessment implicitly predicts that the indicator, soil aggregate stability (SAS) (also known as “soil stability”), is positively related to all three rangeland health attributes (Pellant et al., 2005; Pyke et al., 2002). SAS is also relevant to other rangeland and soil health assessments (Idowu et al., 2008; Tongway & Hindley, 2004). Seminal work by Tisdall and Oades (1982) presented conceptual theory on the functional importance of SAS, including SAS’s role in soil water transport.

Because SAS has been an inconsistent predictor of core rangeland health variables (e.g., water infiltration, plant production), it is increasingly unclear that SAS is a reliable indicator of degradation caused by overgrazing and other disturbances (e.g., Pierson, Williams, Kormos, & Al-Hamdan, 2014; Reinhart, Nichols, Petersen, & Vermeire, 2015; Reinhart & Vermeire, 2017). For example, prior studies have reported conflicting results on whether livestock grazing consistently reduces SAS. Specifically, some studies have shown negative (Beever, Huso, & Pyke, 2006; Wen, He, & Zhang, 2016; Wiesmeier et al., 2012), positive (Franzluebbers, Wright, & Stuedemann, 2000), or no appreciable effects of grazing on SAS (Duchicela et al., 2012; Schmalz et al., 2013; Wang, 2010; Warren, Blackburn, & Taylor, 1986). Also, SAS–infiltration relationships in rangelands are rarely quantified, are often from observational studies, and results are mixed (e.g., Collins, Stone, & Cratic, 2015; Reinhart et al., 2015). Reinhart et al.’s (2015) empirical data from Montana, USA and systematic literature review indicated conflicting evidence on the link between SAS and infiltration. However, Collins et al. (2015) reported that SAS was negatively associated with run-off and sediment yield in Arizona, USA.

Scientists have rarely used field experiments to test for relationships between SAS and soil water transport in rangelands. Among the few examples are an experiment in Nevada, USA which found no relationship between SAS and run-off (Pierson et al., 2014). Another study in Arizona reported their best multiple regression model for predicting sediment yield included SAS and two other variables (O’Dea, 2007). A mesocosm experiment in the UK reported that fine rooted grasses were associated with higher levels of SAS but thick rooted legumes had higher levels of infiltration (Gould, Quinton, Weigelt, De Deyn, & Bardgett, 2016), and they confirmed that root properties, not SAS, influenced infiltration (Gould I.J., personal communication [Aug. 17, 2016]).

In this study, we set out to test if SAS is a reliable indicator of rangeland function in North America’s Northern Great Plains. Robust predictive indicators of ecosystem function are important for regions like the Northern Great Plains which is expansive (>22 million ha) and ecologically and economically important, e.g., support ~11 million animal unit months of livestock grazing (Reeves & Baggett, 2014; Reinhart & Vermeire, 2017; Rezaei et al., 2006). We looked for changes to SAS caused by a number of disturbances, particularly grazing, mowing, and biocides. If SAS is sensitive to these disturbances and correlated to key rangeland function variables (e.g., infiltration, forage production), this will suggest managers can rely on SAS as a proximate indicator of rangeland health. If not, other indicators are needed. To quantify effects of grazing on SAS, we relied on two experiments. One experiment carefully controlled the stocking rate and the timing of grazing, and we hereafter refer to this as the “controlled grazing experiment.” The other grazing experiment consisted of plots excluded from grazing for 6 years paired with nearby plots that have been grazed by cattle every year for over 50 years, and we hereafter refer to this as the “paired plot grazing experiment.” To quantify effects of disturbances on SAS and relationships between SAS and infiltration, we relied on an experiment with mowing, fungicide and herbicide treatments, and we hereafter refer to this as the “disturbance experiment.” Biocide applications that reduce either mycorrhizal fungi or roots have also been reported to reduce SAS (Druille, Omacini, Golluscio, & Cabello, 2013; Wilson, Rice, Rillig, Springer, & Hartnett, 2009; Zaller, Heigl, Ruess, & Grabmaier, 2014) and may aid detection of positive SAS–infiltration relationships. This experiment was also used to test the utility of alternative indicators (e.g., soil carbon, plant diversity) of infiltration. To look at the effects of SAS on infiltration more generally, we combined data from the disturbance experiment with data from past research in the Northern Great Plains and conducted a meta-analysis.

Past research found that SAS depends on plant roots and mycorrhizal fungi, and intense grazing is hypothesized to reduce roots and mycorrhizal fungi (Gehring & Whitham, 2002; but see Barto & Rillig, 2010) and presumably reduce SAS (Beever et al., 2006; Chaudhary et al., 2009; Jastrow, Miller, & Lussenhop, 1998; Wilson et al., 2009). We hypothesized SAS would be negatively affected by livestock grazing and other disturbances and positively associated with infiltration. One of the disturbance experiment’s treatments was intended to act as a conceptual bridge with studies that have described the functional importance of SAS in more arid rangelands. Specifically, studies in arid rangelands often compare properties of vegetated versus unvegetated patches (e.g., Bird, Herrick, Wander, & Murray, 2007; Boix-Fayos, Calvo-Cases, Imeson, Soriano-Soto, & Tiemessen, 1998; Chaudhary et al., 2009; Van De Koppel, Rietkerk, & Weissing, 1997). Herbicide-treated plots acted as proxies for areas that are persistently without vegetation (see Supporting Information Figure S1). We predicted that herbicide treated plots would have the lowest SAS values, followed by fungicide (e.g., Druille et al., 2013; Wilson et al., 2009; Zaller et al., 2014). Biocides were predicted to cause soil aggregates to become unstable (particularly in water)

and for soil fragments to obstruct macropores and preferential flow paths that would ultimately result in less water infiltration.

2 | MATERIALS AND METHODS

2.1 | Study system

Research was conducted at the USDA-Agricultural Research Service's Fort Keogh Livestock and Range Research Laboratory (Fort Keogh), Montana, USA. Fort Keogh (22,402 ha) is centrally located in the Northern Great Plains Steppe ecoregion which spans five states in the USA and two Canadian provinces (Martin et al., 1998). Most sites on Fort Keogh are dominated by perennial C_3 graminoids, and most plant growth occurs between early April and early July. Mean annual precipitation is 34 cm (1937–2011) with most occurring during May and June.

2.2 | Plant and soil properties

Methods describing the measurement of several plant and soil properties are provided in Appendix S1. Researchers familiar with the IIRH system should be aware that we measured SAS primarily by a quantitative laboratory method, which is like the laboratory method used to originally justify the field method for qualitative scoring of SAS (Herrick et al., 2001) used by IIRH.

2.3 | Livestock grazing experiments

The controlled grazing experiment occurred on a calcareous grassland site with a Pinehill loam soil (United States Department of Agriculture, 2003) (105°57'59.1"W, 46°24'24.8"N). The experiment consisted of two grazing intensity treatments (moderate, severe) factorially combined with two grazing seasons (June, October/November) arranged in a grid in a randomized complete block design with five replications ([2 grazing intensities \times 2 grazing seasons \times 5 replications] + 20 ungrazed controls = 40 sampled areas). Each grazed paddock (60 \times 30 m, 20 total paddocks) contained a randomly placed, 1 \times 2-m grazing exclusion cage. Cattle (*Bos taurus*) grazing intensities approximated recommended (i.e., moderate; 1 AUD/ha) and severe (1.5 AUD/ha) stocking rates, with an estimated 656 and 309 kg/ha of post-grazing residual standing biomass respectively. Species composition of the site was dominated by *Hesperostipa comata* (31%), *Pascopyrum smithii* (17%), *Bouteloua gracilis* (12%), and *Carex filifolia* (12%). In mid-November 2014, after implementing the grazing treatments in 2013 and 2014, three 31.75 (diam.) \times 100-mm soil cores were collected inside and 1.5 m around (in three cardinal directions) each cage in each paddock to quantify SAS.

The paired plot grazing experiment occurred at four sites separated by 1.9–5.6 km (105°47'58.5"W, 46°17'36.0"N; 105°49'1.1"W, 46°16'51.7"N; 105°51'53.4"W, 46°18'18.6"N; 105°50'31.5"W, 46°19'39.8"N). Each site occurred on mixed-grass prairie and loamy soils (i.e., Yamacall-Delpoint-Cabbart loams, Busby-Blacksheep-Twilight fine sandy loams, Yamacall-Delpoint loam,

and Yamacall-Delpoint-Cabbart loams respectively). The sites were dominated by *P. smithii* (36%), *C. filifolia* (23%), *H. comata* (13%), and *B. gracilis* (12%). Each site was grazed at light to moderate levels and included two pairs of plots (4.9 \times 4.9 m). Each "site" was intersected by a fenceline, established over 50 years ago. On opposite sides of the fence, we placed a plot pair in patches of similar vegetation and separated by \geq 3 m. We erected a grazing enclosure around one randomly selected plot per pair in 2009. To quantify SAS, we collected three soil cores (31.75 \times 100 mm) per plot (16 total plots) on August 28, 2014.

2.4 | Disturbance experiment

This experiment (46°18'13.9"N, 105°58'39.2"W) was on an upland plain (elevation = 850 m) with calcareous grassland dominated by four graminoid species (*B. gracilis* [2015% cover mean = 28%], *P. smithii* [18%], *C. filifolia* [11%], and *H. comata* [9%]). The experimental site had a gentle slope (variation in plot-to-plot elevation across site = 0.9 m) and loam soil (Eapa loam; 48% sand; 38% silt; 13% clay). Most of the experiment was contained in a fenced area (0.34 ha; established in 2009) that prevented grazing by livestock. The experiment was positioned over two adjacent pastures with one soil series. This was done to maximize local similarity in static soil (e.g., soil texture) and climatic properties while maximizing among-plot variation in dynamic plant (e.g., community composition) and soil properties (e.g., organic matter, aggregate stability). At the pasture level, both pastures had similar stocking rates and were grazed moderately based on (NRCS) regional standards from April to November.

The disturbance experiment was a randomized complete block design with three replications conducted in two adjacent pastures (i.e., sites). Experimental plots (3 \times 3 m) within the fenced area (i.e., 42 total plots) were separated by a 3-m buffer. The experiment included a herbicide treatment and 2 \times 3 factorial of fungicide (control and fungicide) and mowing (no mowing, mowed once, and mowed twice per year) treatments ($n = 3$ per pasture). For the mowing \times fungicide treatment portion, mowing was to simulate non-selective grazing. Mowed plots were mowed in June (blade height 7.3 cm) and plots mowed twice per year were also mowed once in May. Since mowing was to simulate non-selective grazing, we removed clippings from plots every other year to approximate irregular inputs of organic matter. In addition, grazing treatment plots were established (randomly) outside the fenced area in the two adjacent pastures. These plots were separated by $>$ 3 m, and avoided cattle trails and harvester ant mounds. Overall, the experiment included 48 total plots.

Biocides were applied annually from 2010 to 2016. The fungicide used, Topsin M: thiophanate-methyl (dimethyl [(1,2-phenylene)-bis(iminocarbonothioyl)] bis[carbamate]), its rate, method of application, and frequencies of application followed Wilson and Williamson (2008). Topsin M (70% active ingredient; Cerexagri Inc., Philadelphia, Pennsylvania) was applied as a soil drench (9 L per plot) with watering cans at the rate of 1.25 g a.i. \times m⁻² every 3 weeks from April 15–July 8 [5 applications per year] (Supporting Information Figure S1). Non-fungicide plots received equal water from separate

watering cans. The non-specific herbicide glyphosate (Roundup, Monsanto) was applied to herbicide plots (approximately 2.7 kg a.i. \times ha⁻¹) in May and included additional follow-up applications. We collected three soil cores (31.75 \times 100 mm) per plot August 25–26, 2014 (after 5 years of treatments) to quantify several soil properties. Since we also aimed to determine the best predictors of infiltration, we also measured infiltration, soil penetration resistance, plant properties (e.g., species % cover, peak biomass), and additional soil properties (i.e., inorganic carbon, surface soil stability, total carbon, and total nitrogen).

2.5 | Analyses

Data from the controlled grazing experiment, paired plot grazing experiment, and disturbance experiment were analysed with multivariate linear models fit with a FORTRAN program (Intel Corporation, 2018). The controlled grazing experiment model had fixed effects for block and the five grazing treatments and a random effect for paddock. The paired plot grazing experiment model had a fixed effect for grazing treatment (not grazed, grazed) and a random effect for site. One of the disturbance experiment responses (surface soil stability) was not normally distributed, so this response was analysed with a bootstrap procedure (Efron & Tibshirani, 1993) implemented in Wolfram Mathematica (Wolfram Research, 2017). The model for the other responses had fixed effects for site, treatment, and site by treatment. Our main aim for analyses was to test whether SAS was affected by disturbance treatments. In addition, we tested the effect of treatments on additional response variables which included peak plant biomass (2015 and 2016 herbaceous vegetation data averaged), infiltration, soil organic carbon, and soil resistance to penetration.

Data for the disturbance experiment were also used to determine the best plant and/or soil property predictors of variation in infiltration (i.e., simple pairwise correlations, multiple regression) and to test for a general positive SAS–infiltration relationship (i.e., meta-analysis). Analyses were performed in R (R Development Core Team, 2011). We determined the Pearson correlation coefficients of all would-be predictors of infiltration and peak plant biomass (Supporting Information Tables S1 and S2). To identify the best minimum dataset for predicting infiltration, we then performed multiple linear regression analyses using plant and soil properties for the full dataset, except the plots treated with herbicide ($n = 42$). To include data on plant community composition, we first performed a principal component (PC) analysis of vascular plant community composition (i.e., % cover) data using the VEGAN package (Oksanen et al., 2015). Plant abundance data were transformed with the command “deco-stand(hellinger)” in vegan, and total inertia was 0.23. Then, PC scores with eigenvalues greater than the mean (i.e., PC1–13) were incorporated into multiple linear regression (MLR) models. MLR predictor variables included: plant community dissimilarity (13 PC scores), diversity (species richness and Shannon H), soil penetration resistance, % soil organic carbon, % soil total nitrogen, % soil inorganic carbon, and % water-stable aggregates (0.25–1 and 1–2 mm). We

used the “regsubsets” function in the LEAPS package (Lumley & Miller, 2009) with the exhaustive search method to determine the five best models per level of parameters. Leaps uses an efficient branch-and-bound algorithm to rapidly determine the best models. We then used Schwarz’s Bayesian information criterion to identify the five best models. We report traditional parametric statistics for models. Residual analyses included visual confirmation that the assumptions of normality and homoscedasticity of residuals were not violated. We assessed multicollinearity among parameters in selected models with variance inflation factors. We also conducted an outlier analysis using Cook’s distance (Cook & Weisberg, 1982).

Lastly, we were interested in performing a synthesis of many SAS–infiltration relationships. The Pearson correlation coefficients (r) of individual experiments were used as effect sizes in meta-analysis (Schulze, 2004) to obtain a weighted mean correlation coefficient (\bar{r}) and 95% confidence intervals. The meta-analysis pooled 14 correlation coefficients (r) describing SAS–infiltration relationships for multiple datasets including: disturbance experiment, published (Reinhart et al., 2015; Wang, 2010), and unpublished data from the focal research station. [We requested additional data from Dr. Guojie Wang’s dissertation but received no reply.] We used a fixed-effect model based on Fisher’s r -to- z transformation (Laliberté et al., 2010). A fixed-effect model was used because the inference space was limited by the sample space, e.g., most data were from a single field station in Montana. This transformation is normalizing and variance stabilizing so that the variance depends only on sample size. Individual effect sizes were weighed by the inverse of their variance (Borenstein, Hedges, Higgins, & Rothstein, 2009). The meta-analysis was implemented with the “metacor” function in the META package with the DerSimonian-Laird method to estimate the between-study variance (DerSimonian & Laird, 1986). If more data were available for the meta-analysis, then moderator variables could have been used to distinguish which SAS measurements (e.g., small water-stable aggregates) are most often correlated with infiltration. (Presented results were back-transformed.)

3 | RESULTS

3.1 | Effects of grazing on SAS

Livestock grazing effects on SAS depended on experiment, treatment, and macroaggregate size class. In the controlled grazing experiment, grazing treatments had no effect on medium-sized macroaggregates (1–2 mm) (Figure 1a). However, all fall grazing treatments and severe summer grazing reduced the stability of small (0.25–1 mm) macroaggregates (Figure 1b). Since soil samples were all collected in the fall, interpreting differences among grazing season treatments is difficult, because treatments also differed in the time soil had to recover. Another study also reported effects of fall grazing on SAS (Evans, Krzic, Broersma, & Thompson, 2012). In the paired plot grazing experiment, grazing had no effect on small- and medium-sized macroaggregates (Supporting Information Figure S2).

3.2 | Disturbance experiment

Several soil structure properties (i.e., surface soil stability, % water-stable aggregates) were reduced by herbicide (Figure 2). However, soil structure properties and infiltration were similar for control and grazed plots (Figure 2 and Supporting Information Figure S3), thereby suggesting no appreciable effects of livestock grazing. Infiltration exhibited high levels of plot-to-plot variation as indicated by the size of the confidence intervals and was not affected by disturbance treatments (Supporting Information Figure S3). Peak herbaceous plant biomass and soil organic carbon were reduced by herbicide (Supporting Information Figure S3).

3.3 | Predictors of infiltration

Of several would-be predictors of infiltration, only three plant properties, and no soil properties, were weakly correlated with infiltration (Supporting Information Table S1, $\alpha = 0.05$). Specifically, infiltration was positively correlated with plant community diversity ($r = 0.31$) and peak herbaceous plant biomass ($r = 0.34$) and negatively correlated with community dissimilarity principle component 1 ($r = -0.32$; Supporting Information Table S1). Multiple linear regression analyses also indicated that slight amounts of variation in infiltration ($0.10 \leq R^2 \leq 0.27$) were explained by relatively simple and equally parsimonious models that included metrics of plant community diversity and dissimilarity (Table 1). Among the five best models, measures of plant community dissimilarity (PC1, PC5) and diversity (species richness and Shannon H) were the only predictor variables. For the two principle components, we then identified the two plant species that contributed most to each principle component. Simple linear regressions were then used to determine whether the per cent cover of each of the four plant species explained appreciable variation in infiltration. Infiltration was negatively correlated with a perennial C_4 grass (*B. gracilis*; $p = 0.005$, $r^2 = 0.18$) and positively correlated with an annual/biennial exotic forb (*Tragopogon dubius*; $p = 0.02$, $r^2 = 0.13$) (Figure 3).

3.4 | SAS–infiltration relationship meta-analysis

A meta-analysis was used to pool 14 correlation coefficients (r) describing SAS–infiltration relationships. The pooled weighted correlation coefficient (\bar{r}), for measures of SAS and infiltration, equalled -0.0189 which did not differ from zero ($p = 0.71$) and 95% confidence intervals overlapped zero (Figure 4). This indicates no general correlation between SAS and infiltration in the Northern Great Plains. However, if we analysed only a subset of the data (square symbols in Figure 4) that are seemingly most relevant to the standardized methods for assessing rangeland health (see table 2 of Reinhart et al., 2015) then a small but significant pooled correlation coefficient is detected ($\bar{r} = 0.1621$ [0.0228; 0.2952] and $p = 0.02$). [Note: This last synthesis result should be interpreted with caution because the quantity of data used in the analysis is below recommendations

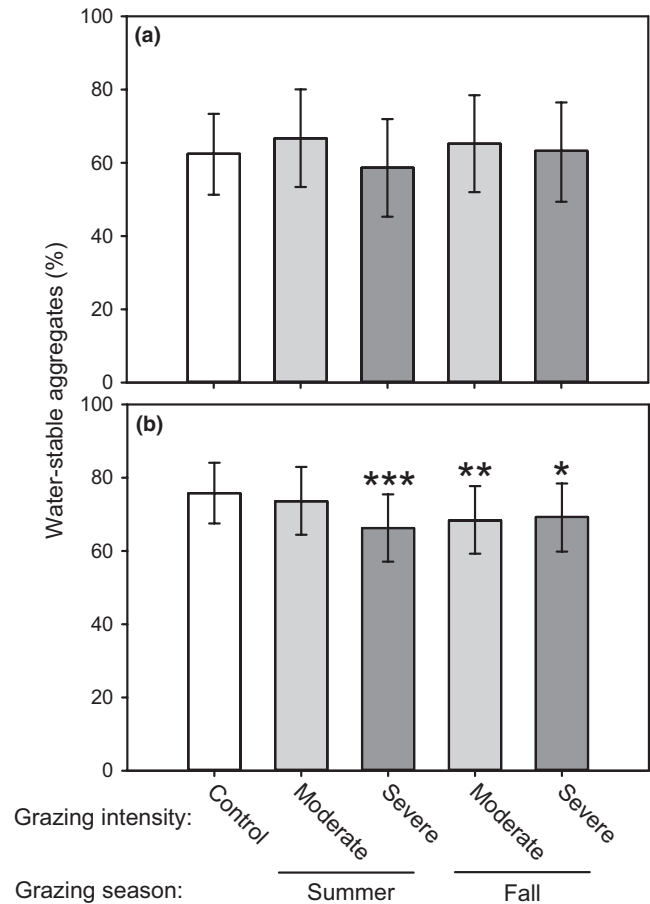


FIGURE 1 Effect of manipulating livestock grazing intensity and season on SAS. Values represent mean and 95% normal-based confidence intervals. Panels show results for medium- (a; 1–2 mm) and small-sized macroaggregates (b; 0.25–1 mm) for the controlled grazing experiment. Asterisks denote significant differences from control (* $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$)

(Field, 2001), and we know additional contradictory data exist (Wang, 2010) that would likely change the result.]

4 | DISCUSSION

4.1 | Rangeland ecosystem function

In the USA, SAS is widely thought to be a critical indicator of rangeland and soil health. Unfortunately, simplified measurements of soil structure, like measures of SAS, may not consistently predict important soil functions (e.g., Bennett, Mele, Annett, & Kasel, 2010; Letey, 1991; Reinhart et al., 2015). While livestock are known to reduce infiltration in portions of the Northern Great Plains (Abdel-Magid, Schuman, & Hart, 1987; Emmerich & Heitschmidt, 2002; Wang, 2010), we observed no effect of field treatments on infiltration (Supporting Information Figure S3). SAS was not consistently affected by livestock grazing, and the pooled weighted correlation coefficient (\bar{r}) for SAS–infiltration relationships of regional datasets did not differ from zero.

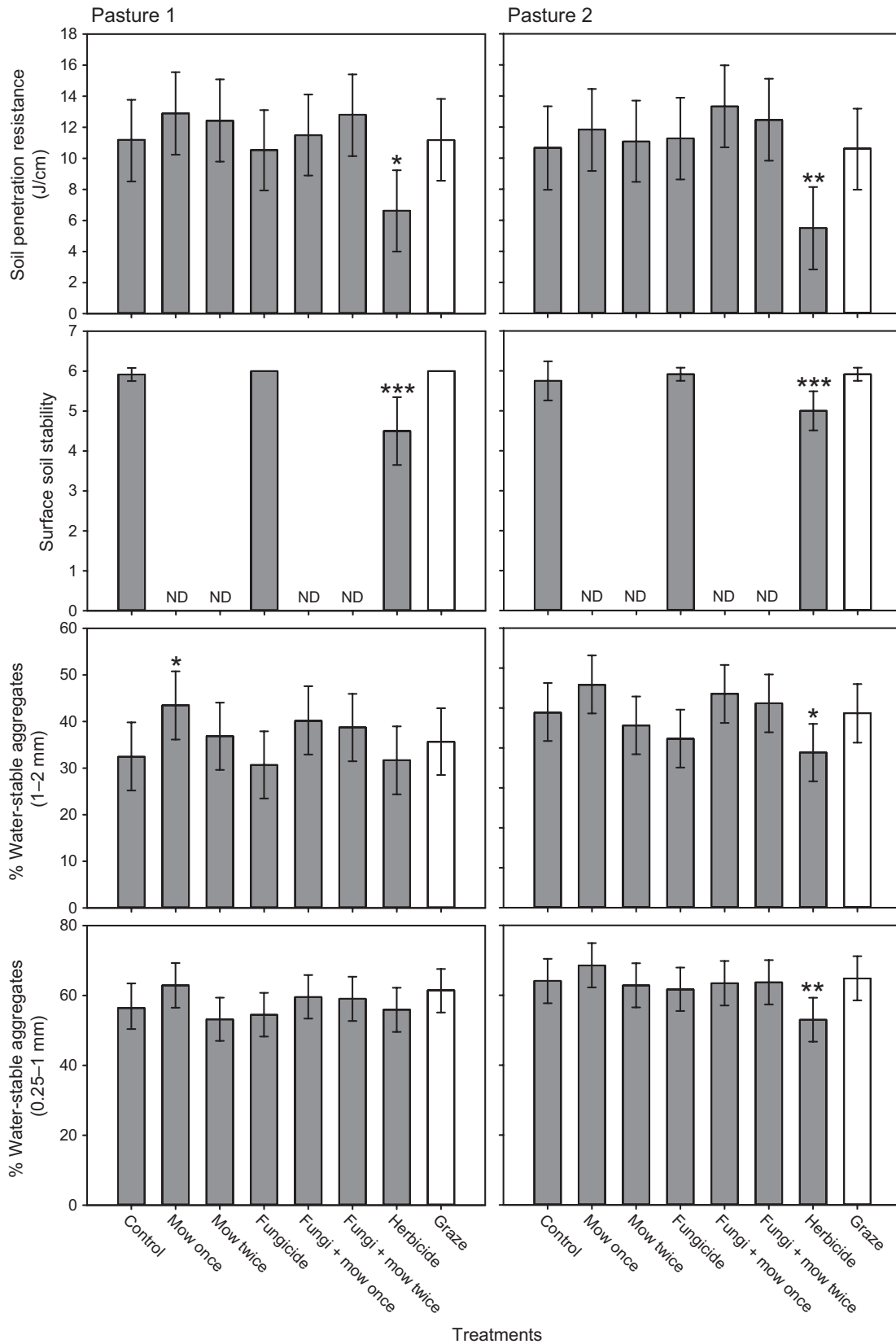


FIGURE 2 Effect of disturbance treatments on soil structure properties. Treatments within the grazing exclusion (grey bars) included: control, herbicide, and fungicide (none or fungicide) \times mowing (none, once, or twice per year). Bars represent means with 95% confidence intervals ($n = 3$ per pasture). Soil structure properties include: soil penetration resistance, surface soil stability (Pellant et al., 2005), and water-stable aggregates (1-2 and 0.25-1 mm macroaggregates). ND = no data. Asterisks denote significant differences from control (* $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$)

TABLE 1 Five best multiple regression models, based on Schwarz Bayesian Criterion scores (BIC) to explain variation in field-saturated hydraulic conductivity (i.e., infiltration) for the disturbance experiment ($n = 42$, excluded data for herbicide-treated plots)

Models	Independent variables	BIC	t-value	F	p	R ²
1 ^a	PC5	-	2.42	-	0.020	-
	Richness	-	-2.31	-	0.026	-
	Diversity	-	2.75	-	0.009	-
	Total	1.85	-	4.64	0.007	0.27
2	PC1	-	2.18	-	0.036	-
	PC5	-	-1.87	-	0.070	-
	Total	2.89	-	4.11	0.024	0.17
3	PC1	-	-2.11	-	0.042	-
	Total	3.06	-	4.43	0.042	0.10
4	Diversity	-	2.07	-	0.045	-
	Total	3.19	-	4.30	0.045	0.10
5	PC5	-	1.79	-	0.082	-
	Diversity	-	1.99	-	0.053	-
	Total	3.63	-	3.86	0.030	0.17

^aCollinearity between principle component 5 and diversity measures (richness and diversity) was detected. Total = properties of the full model; PC = principle component [1, 5] for plant community composition. Significance of linear model was tested with ANOVA. Numerator and denominator degrees of freedom, respectively, were 3,38 for model 1; 2,39 for models 2 and 5; 1,40 for models 3 and 4.

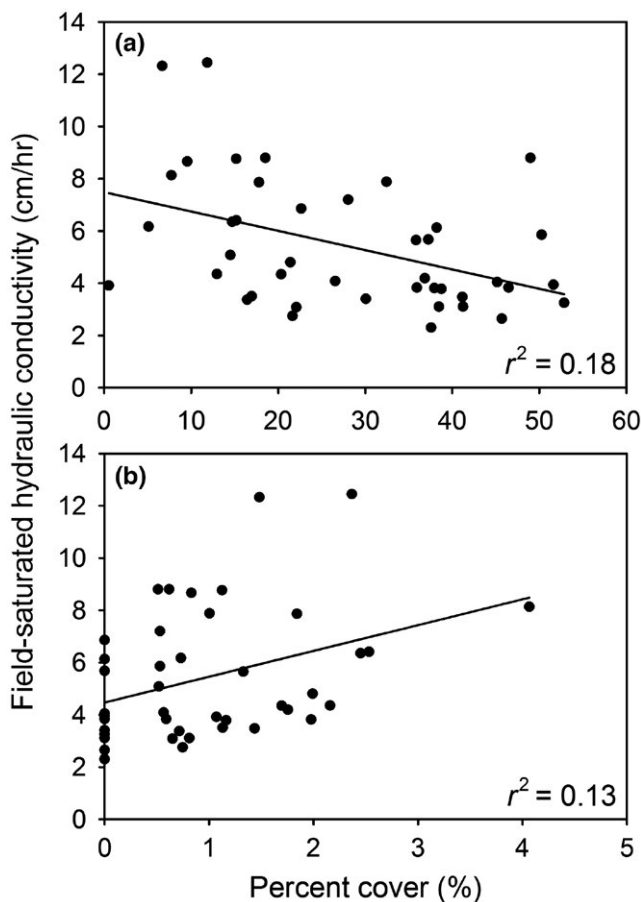


FIGURE 3 Relationship between field-saturated hydraulic conductivity (i.e., infiltration) and plant cover (*Bouteloua gracilis*, a; *Tragopogon dubius*, b) for the disturbance experiment. Each panel contains a best fit line, based on ordinary least squares, and regression coefficient

Instead of validating a key (putative) indicator of popular systems for assessing rangeland health (e.g., IIRH, Landscape function analysis (Tongway & Hindley, 2004)), we uncovered evidence linking changes in soil water transport to gradual shifts in vegetation that are likely driven by variation in disturbance by livestock grazing (e.g., Porensky, Mueller, Augustine, & Derner, 2016). We found that local variation in infiltration (i.e., within a site and soil series) was explained more by plant than soil properties. Specifically, we found that infiltration was negatively correlated ($r^2 = 0.18$) with cover of a dominant C_4 grass *Bouteloua gracilis*. Since *B. gracilis* is tolerant of grazing and increases with grazing pressure (e.g., Porensky et al., 2016), this result is seemingly consistent with biocompaction theory which predicts high levels of grazing pressure select for short stature plant species tolerant of compacted soils and high levels of grazing pressure (Howison, Olf, Van De Koppel, & Smit, 2017). However, biocompaction is not clearly the driver of regional grazer effects (Abdel-Magid et al., 1987). Alternatively, if disturbance by grazers select for plant species with unique rooting characteristics that affect infiltration then bioturbation may be the driver of rangeland soil water transport (Gould et al., 2016; Howison et al., 2017; Thompson, Harman, Heine, & Katul, 2010).

Our findings are generally at odds with those from more arid rangelands which have interpreted the functional importance of SAS by comparing vegetated versus unvegetated patches (e.g., Bird et al., 2007; Boix-Fayos et al., 1998; Chaudhary et al., 2009; Van De Koppel et al., 1997). We predicted that herbicide treated plots would have the lowest SAS (and presumably lowest infiltration) values, followed by fungicide (e.g., Druille et al., 2013; Wilson et al., 2009; Zaller et al., 2014). We also expected that soil penetration resistance would increase due to degraded SAS. Though herbicide tended to reduce SAS (Figure 2), infiltration was unaffected by treatments (Supporting Information

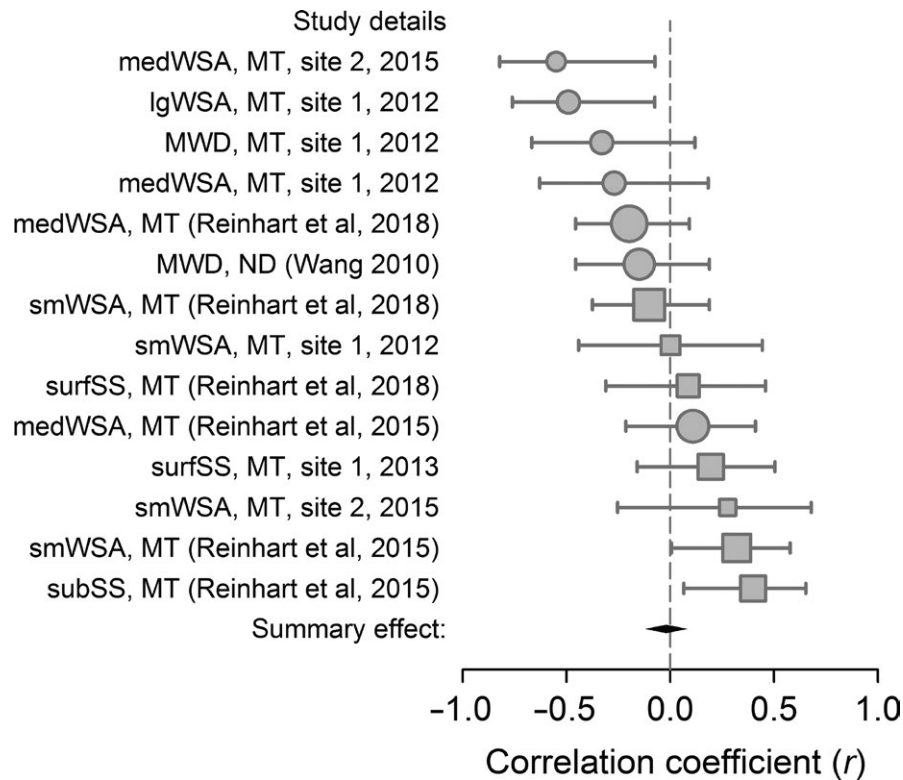


FIGURE 4 Effect sizes for the SAS–infiltration relationship across several studies and sites in the Northern Great Plains. Study information is provided to the left of the figure, the center of each symbol corresponds with the effect size (correlation coefficient r , x-axis), and the whiskers indicate lower and upper 95% confidence intervals. The (grey) symbol size indicates the weighting for each experiment in the analysis. The summary effect (pooled weighted correlation coefficient, \bar{r}) is indicated by a diamond at the bottom of the plot, where $\bar{r} = -0.0189$ (-0.1196 ; 0.0821) and $p = 0.71$. lgWSA = water-stable aggregates (2–4 mm size class); medWSA = water-stable aggregates (1–2 mm); MT = Montana; MWD = mean weight diameter; ND = North Dakota; Reinhart et al. (2018) = this study (disturbance experiment); smWSA = water-stable aggregates (0.25–1 mm); subSS = subsurface soil stability (Pellant et al., 2005), and surfSS = surface soil stability (Pellant et al., 2005). Field-saturated infiltrability (Ogden, Van Es, & Schindelbeck, 1997) was the infiltration data for site 1 (2012) and Reinhart et al. (2015) and field-saturated hydraulic conductivity (K_{sat}) was the infiltration measurement for other comparisons. Square symbols indicate SAS measurements from the Interpreting Indicators of Rangeland Health (Pellant et al., 2005) assessment guide or % water-stable aggregates (0.25–1 mm) which are positively correlated with subsurface soil stability measures (see Reinhart et al., 2015)

Figure S3). Herbicide effects on soil penetration resistance were the opposite of predictions (Figure 2), which may be attributed to the decomposition of root biomass. [Note: Decomposition of roots may have had offsetting effects on infiltration, e.g., temporally increased paths (i.e., continuous macropores left by decomposed roots) for preferential water flow.] If the herbicide treated plots had been placed in the grazed pastures (and not the grazing enclosure), then livestock biocompaction of the soil may have been adequate to increase penetration resistance (Schmalz et al., 2013) and decrease infiltration relative to vegetated plots. Research in the adjacent state of Wyoming, however, interpreted that bulk density (which is often positively correlated with penetration resistance) was not a useful predictor of infiltration (Abdel-Magid et al., 1987). Overall, we found it difficult to affect infiltration and confirm the importance of popular would-be predictors of rangeland ecosystem function in the field.

It may be easy to collect SAS data and to conceptualize their functional importance in rangelands, but their predictive power is surely context-specific (e.g., Boix-Fayos, Calvo-Cases, Imeson, &

Soriano-Soto, 2001; Letey, 1991) and likely eclipsed by more important putative indicators of ecosystem function such as plant species and total biomass, bare-ground cover, and litter cover (Goff, Bent, & Hart, 1993; Hart & Frasier, 2003; Thompson et al., 2010). For example, bare-ground and litter cover are much stronger predictors of related measures of soil water transport, i.e., run-off ($r^2 = 0.83$ – 0.99 of Hart & Frasier, 2003). In the Northern Plains, soil water transport is affected by many bioturbation factors related to preferential flow paths and soil macroporosity (Abdel-Magid et al., 1987; Fischer et al., 2014; Gould et al., 2016; Reinhart et al., 2015). We hypothesize that bioturbation (i.e., root and invertebrate tunnel properties) is a key driver of soil pores, channels, and ultimately soil water transport.

4.2 | Implications for range management

Trust in any rangeland health assessment system should be proportional to the system's predictive accuracy (i.e., % truth) and quantity of evidence to support defined relationships (e.g., Reeves & Baggett, 2014; Reinhart et al., 2015; Rezaei et al., 2006). The gold standard

for evidence-based assessments of rangeland function and health are to include routine empirical validation necessary to estimate predictive uncertainty and to facilitate continual innovation (e.g., Houlahan, McKinney, Anderson, & McGill, 2017), e.g., identify unique indicator sets (or predictive models) for specific rangeland types.

While IIRH is technically an “evidence-based” system (e.g., collects qualitative data on many plant and soil properties), one concern is that rangeland health attributes are not directly measurable (Pyke et al., 2002) which indicates IIRH is not amenable to empirical validation. According to Pyke et al., attributes constitute, “ecosystem component[s] that cannot be directly measured, but can be approximated by a set of observable indicators...” and go on to define indicators as, “observable components of an ecosystem that are related to 1 or more [rangeland health] attributes... and used in combination with other indicators as an index of the status of that attribute.” The next step for monitoring rangeland health should be to quantify predictive accuracy (Reeves & Baggett, 2014; Rezaei et al., 2006) which may require either new or alternative approaches. For example, the Landscape Function Analysis system includes routine validation, e.g., its “infiltration index” was a moderately strong predictor of soil water transport ($r^2 = 0.64\text{--}0.75$, Tongway & Hindley, 2004). More work is needed validating would-be indicators of rangeland function to determine which should be modified, replaced, or discarded (Rezaei et al., 2006).

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AUTHORS' CONTRIBUTIONS

All authors helped conceive the study ideas and methods; K.R. collected the data; K.R. and M.R. analysed the data; K.R. led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

DATA ACCESSIBILITY

Data available via the Dryad Digital Repository <https://doi.org/10.5061/dryad.25v031b> (Reinhart, Rinella, Waterman, Petersen, & Vermeire, 2018)

ORCID

Kurt O. Reinhart  <http://orcid.org/0000-0002-7985-6738>

Richard C. Waterman  <http://orcid.org/0000-0002-2044-3517>

Mark K. Petersen  <http://orcid.org/0000-0002-1249-7152>

Lance T. Vermeire  <http://orcid.org/0000-0001-9147-0099>

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