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Does Grazing Matter for Soil Organic Carbon Sequestration in the Western North American Great Plains?

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Abstract

Considerable uncertainty remains regarding grazing-induced influences on soil organic carbon (SOC) sequestration in semiarid grassland ecosystems due to three important complications associated with studying such effects: (1) Ecologically meaningful shifts in SOC pools attributable to grazing are difficult to detect relative to inherently large grassland SOC pools, (2) a lack of baseline (pre-treatment) data, and (3) frequent lack of or limited replication of long-term grazing manipulations. SOC sequestration rates were determined in 74-year-old grazing exclosures and paired moderately grazed sites, established across a soil texture gradient, in the western North American shortgrass steppe in northeastern Colorado. We sampled soils (0–20 cm) from 12 exclosures and paired grazed sites to measure SOC concentration and soil radiocarbon $\Delta^{14}C$ (‰); the latter allowed us to determine turnover of the SOC pool over a 7-decade period in the presence versus the absence of grazing. Removal of grazing for more than 7 decades substantially altered plant community composition but did not affect total soil C, SOC, soil $\Delta^{14}C$, SOC turnover rate, or total soil N. Grazing effect also did not interact with soil texture to influence any of those soil properties. Soil texture (silt + clay content) did influence total soil C and SOC, and total soil N, but not $\Delta^{14}C$ or SOC turnover. Results provide evidence that long-term removal of grazing from semiarid grassland ecosystems in the western North American Great Plains does not enhance long-term SOC sequestration, despite changes in the relative dominance of C3 versus C4 grasses.

Key words: cattle grazing; semiarid rangeland; shortgrass steppe; soil carbon; soil radiocarbon; soil carbon turnover.

Highlights

- Carbon sequestration rates in 74-year exclosures and moderately grazed sites.
- Soil radiocarbon (^{14}C) and turnover rates unaffected despite major vegetation differences.
- Long-term removal of grazing does not enhance soil carbon sequestration.

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Author contributions JDD and DJA conceived and designed study. JDD and DJA performed research. JDD analyzed data. DAF contributed methods on calculating carbon turnover rate. JDD, DJA, and DAF all contributed to writing the paper.

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Introduction

Despite advances in the understanding of pathways of grazing effects on soil organic carbon (SOC) and soil nitrogen (N) concentrations (Pineiro and others 2010), substantial knowledge gaps remain (McSherry and Ritchie 2013). For example, it remains unclear how interactions among soil texture, relative abundance of C3 versus C4 grasses, and grazing influence the potential for storing SOC (McSherry and Ritchie 2013; Abdalla and others 2018). Because grazing land ecosystems provide 70% of the forage for ruminant livestock worldwide (Holechek 2013) and domestic livestock graze about 50% of the world's land surface (Holechek and others 2011), understanding the potential of these lands for SOC sequestration is important. Research efforts to date have largely addressed the role of grazing on soil carbon (C) sequestration by comparing either (1) net ecosystem exchange (NEE) rates using Bowen ratio or eddy covariance micrometeorological technology or (2) SOC pools in grassland ecosystems with different grazing treatments.

Short-term (< 1 decade) NEE studies have determined that grazing in semiarid grassland ecosystems potentially influences C fluxes, but the effects of herbivores are strongly influenced by prevailing weather/climatic conditions, with grazers increasing SOC during wet periods, and reducing SOC during dry/drought periods. For example, grassland ecosystems of the western Great Plains are either a sink or source for C depending on yearly weather patterns (Svejcar and others 2008). In addition, these grassland ecosystems exhibit a short period (2–3 months) of high C uptake followed by long periods of C balance or small respiratory losses of C (Svejcar and others 2008). Drought reduces the periods of high C uptake, and this is exacerbated when grazing intensity (or stocking rate) is high (Morgan and others 2016). Interannual variability in NEE across the northern Great Plains can be considerable due to frequent droughts, as demonstrated during a recent 7-year period (2000–2006) (Zhang and others 2010). Variation in fluxes of NEE attributed to grazing in the semiarid, shortgrass steppe ecosystem are about half those associated between dry and wet years (Morgan and others 2016).

Studies that directly measured changes in soil SOC pools over time at varying grazing intensities revealed important grazing–climate interactions (Derner and Schuman 2007; McSherry and Ritchie 2013). Several studies in the western, semiarid portion of the Great Plains found that grazing increased SOC pools when the years preceding the soil sampling were relatively wet compared to the long-term mean (for example, Schuman and others 1999; Reeder and Schuman 2002; Reeder and others 2004). In one unusually long-term study in the northern mixed-grass prairie, heavy grazing increased SOC pools during a relatively wet period (1982–1993, Schuman and others 1999), but this grazing effect disappeared during the following decade (1993–2003) when several droughts occurred (Ingram and others 2008). Pools of SOC were more stable for grassland ecosystems experiencing light and no grazing between the climatically different decades, further exemplifying the interactions of grazing and long-term weather/climate patterns (Ingram and others 2008). Criticisms of soil sampling studies to determine effects of grazing on soil C sequestration are (1) limitations with quantifying the relatively small changes associated with grazing relative to the inherently large soil C pool, (2) lack of baseline (pre-treatment) data, and (3) frequent lack of or limited replication of long-term grazing manipulations (McSherry and Ritchie 2013).

We evaluated long-term (multi-decade) effects of livestock grazing on SOC sequestration. We used soil ^{14}C radiocarbon techniques (see Frank and others 2012) to determine the stable SOC pool turnover rate. We sampled soils (0–20 cm) in 2011 at 12 grazing exclosures established in 1937 and arrayed along a soil texture (that is, silt + clay content) gradient in moderately grazed, semiarid, shortgrass steppe of northeastern Colorado. The ^{14}C radiocarbon technique integrates SOC formation since 1950, when the atmosphere reached its peak of enrichment in ^{14}C during nuclear weapons testing. Our analysis therefore examines the effect of grazing on SOC formation over a 6-decade period that encompasses multiple wet and dry climatic cycles (Chen and others 2017), and across a soil texture gradient with its concomitant gradient in plant species composition (Augustine and others 2017) to determine the role of grazing in driving SOC sequestration in a semiarid grassland. Specifically, we were interested in how soil C, SOC, N, and C turnover rates changed along a soil texture gradient and how those variables may be affected by grazing alone or via a texture-dependent effect of grazing.

Methods

Study Site

Research was conducted on the USDA-Agricultural Research Service Central Plains Experimental

Range, a Long-Term Agroecosystem Research (LTAR) Network site, which encompasses 6270 ha of the shortgrass steppe ecosystem in northeast Colorado, USA (40°50′N, 104°43′W). Mean annual precipitation is 340 mm, and mean growing season (Apr–Aug) precipitation is 242 mm. Most of the site is subdivided into 65–130-ha pastures that have been grazed by cattle since the site was established in 1939. At that time, grazing exclosures (four-strand barbed wire; each ~ 90 m × 90 m) were constructed in more than 20 locations. Exclosures are on level terrain, with one built in most of the pastures that contained native shortgrass steppe rangeland on the site in 1939. Pronghorn antelope (*Antilocapra americana*) are present and can pass under exclosure fencing, but are rarely observed inside exclosures. Mammalian herbivores smaller than pronghorn (lagomorphs and rodents) are unimpeded by exclosures. Twelve exclosures occurring in pastures consistently managed with growing season-long (May–October) grazing at moderate stocking rates (~ 0.65 animal unit months, AUM/ha) for at least the past 30 years were sampled. Detailed stocking rate records are not available for every pasture prior to 1980. However, the sites included in this study have experienced moderate, growing season grazing for either most or all their 72-year history.

Species composition on moderately grazed lands at the site is predominately perennial grasses. *Bouteloua gracilis* (blue grama) and *B. dactyloides* (buffalograss) are the dominant perennial warm-season (C4) grasses, and these two species contribute greater than 70% of the aboveground net primary productivity (Lauenroth 2008). *Pascopyrum smithii* (western wheatgrass) is the dominant perennial cool season (C3) mid-height grass on loamy texture soils, and *Hesperostipa comata* (needle-and-thread) is the dominant on sandier soils. *Carex duriuscula* (needle leaf sedge) is the primary cool-season graminoid, *Sphaeralcea coccinea* (scarlet globemallow) is the primary forb, and *Artemisia frigida* (fringed sagewort) and *Eriogonum effusum* (buckwheat) are the main sub-shrubs across the soil texture gradient.

Soil Data

Plant communities in paired 30 m × 30 m plots located inside and outside sixteen exclosures (total of 32 plots) were sampled during July 2011 (Augustine and others 2017). For twelve of these sixteen exclosures (total of 24 plots), 0–20-cm-deep × 3.2-cm-diameter soil cores were taken at the same time. This depth included 75% of the root biomass in shortgrass steppe (Liang and others 1989). We divided each plot in half and collected cores at six systematic locations per half. The six cores per half were combined into two pooled samples per plot, passed through a 2-mm sieve, and homogenized. A subsample from each pooled sample was analyzed for sand, silt, and clay fractions using the hydrometer method (Bouyoucos 1962).

We homogenized a 1 g subsample from each pooled sample and removed all visible root material under a dissecting scope. For these cleaned subsamples, we measured soil nitrogen, total soil C, SOC and $\Delta^{14}C$ (‰) of SOC. We derived soil organic radiocarbon values as the deviation from a 1950 standard representing isotopic composition in 1950, prior to bomb-generated increases in atmospheric $^{14}CO_2$, where

$$\Delta^{14}C = [F - 1] \times 1000,$$
$$\text{and where } F = (^{14}C/^{12}C)_{sample}/(^{14}C/^{12}C)_{standard}$$

The more positive the soil $\Delta^{14}C$ value, the greater the proportion of the SOC represented by bomb-produced ^{14}C. Negative $\Delta^{14}C$ values represent SOC that was predominantly comprised of C assimilated before 1950. A sample with a $\Delta^{14}C$ equal to zero has the same isotopic composition as that of atmospheric CO_2 in 1950. Radiocarbon values were corrected for (1) isotopic fractionation by adjusting $\Delta^{14}C$ measurements to a common $\Delta^{13}C$ value of −25‰ and (2) radioactive decay of the standard after 1950. All samples were acid-pre-treated to remove mineral C, and radiocarbon measurements were taken at the Arizona AMS Laboratory (Tucson, AZ).

We calculated the turnover time (years) of the SOC with a soil C stock modeling approach (Trumbore 1993; Torn and others 2005; Frank and others 2011, 2012). In brief, this method derived the historic record of $\Delta^{14}C$ content of two soil SOC pools, active and stable, at an annual time step. We assumed steady-state dynamics so that the size of the two pools did not change over time and that the $\Delta^{14}C$ value of the C assimilated by plants was determined by the atmospheric $\Delta^{14}C$ value of CO_2 for that year (http://www.radiocarbon.org/IntCal04.htm; Levin and Kromer 2004; Graven 2008). The $\Delta^{14}C$ value for C metabolized and lost from the stable pool was equal to the radiocarbon value of the SOC pool the previous year. For each pool, the C input and output equaled the SOC pool size divided by turnover time (years). The size of the active pool was set at 3% of the total SOC pool, similar to other studies (Parton and others 1987; Torn and others 2005; Frank and others 2011), and the $\Delta^{14}C$ value of that pool was the atmospheric value for the previous year. The stable C pool (soil C–active C) and the turnover time of the active pool were known. To determine the turnover time of the stable pool, we found the value for stable C turnover time that resulted in the correct soil $\Delta^{14}C$ value for the soil collected in 2011.

Data Analysis

We used regression analyses with block (pair of grazed/exclosure sites) to evaluate the relationships between total soil nitrogen, total soil C, SOC, and $\Delta^{14}C$ and turnover rate of SOC, and the independent factors of grazing (moderately grazed vs. non-grazed, categorical), silt + clay content (%, proxy for soil texture) and their interaction. We performed these analyses on standardized data to better assess the relative importance of each independent parameter included in the models (Sokal and Rohlf 1995).

Results

Soil Carbon and Nitrogen

Grazing and silt + clay content did not interact to affect percent soil nitrogen (p =0.750, data not shown), percent total soil C (p = 0.820) or percent SOC (p = 0.470). Additional interaction analyses using sand, silt, or clay content with grazing resulted in the same findings. Silt + clay content (%) was positively associated with percent soil N (p <0.0001, data not shown), total C (p < 0.0001), and SOC (p < 0.0001) (Figure 1). Long-term moderate grazing did not affect soil nitrogen (0.1035 ± 0.0457% vs. 0.1079 ± 0.0433%, mean ± 1SD, p = 0.908), percent total C (1.0376 ± 0.6236% vs. 1.0532 ± 0.5976%, grazed vs. exclosures, p = 0.647) or SOC (0.9215 ± 0.4370% vs. 0.9349 ± 0.4122%, grazed vs. exclosures, p = 0.830).

$\Delta^{14}C$ and Turnover Rate of SOC

Grazing and silt + clay content did not interact to affect $\Delta^{14}C$ (p = 0.707) or turnover rate of SOC (p = 0.747) (Figure 1). Furthermore, grazing did not affect either $\Delta^{14}C$ (21.48 ± 24.94‰ vs. 17.99 ± 25.33‰, grazed vs. exclosures p = 0.532) or turnover rate of SOC (251 ± 92 years vs. 263 ± 88 years, grazed vs. exclosures, p = 0.598). The range of $\Delta^{14}C$ values across sites, from − 30 to + 86‰, corresponds to SOC turnover rates from 492 to 95 years, respectively. Silt + clay content was not strongly correlated with either $\Delta^{14}C$ (p = 0.816) or turnover rate of SOC (p = 0.917), although sites with high silt + clay content (> 70%) exhibited shorter SOC turnover rates (< 200 years) compared to low silt + clay (< 30%) sites, which were more variable (range of ~ 160 to 500 years; Figure 1). We found that 83.3% (n = 40 of 48) of $\Delta^{14}C$ values were positive, demonstrating that there has been substantial soil C sequestration post-1950 in the majority of this grassland ecosystem. In the remaining 16.7% of sites (n = 8 of 48), negative $\Delta^{14}C$ values indicate a relatively stable soil C pool that was predominantly assimilated before 1950.

Discussion

Soil sampling inside and outside twelve long-term (since 1939) livestock exclosures across a soil texture gradient in native shortgrass steppe rangeland provided a novel opportunity to determine the effect of moderate livestock grazing on turnover of the stable C pool in semiarid, shortgrass steppe grassland. Our primary question was ''Does grazing matter for soil C sequestration in the western North American Great Plains?'' Integrated across the seven plus decades of climatic variability, the concomitant variation in total aboveground net primary productivity (Chen and others 2017), and the effects of grazers on plant community composition (Augustine and others 2017), the answer to our primary question is a simple ''no.'' Grazing alone, or through interactions with silt + clay content, did not affect total soil C, SOC, N, or $\Delta^{14}C$ or turnover rate of SOC. We note that lands in the western Great Plains that were converted to row-crop agriculture sometime in the past 150 years often have undergone substantial loss of soil C, and recovery back to levels found in native rangeland soil can require many decades (Ihori and others 1995). Our work does not address the potential influence of grazing on soil C in formerly plowed soils, but does demonstrate that in native shortgrass steppe rangelands, grazing does not influence soil C over time scales of up to 7 decades.

The previous work documented dramatic variation in turnover rates of stable SOC among North American ecosystems, varying from a low of 59 years in subtropical forest to 2151 years in arctic tundra (Frank and others 2012). Their estimate for shortgrass steppe, based on sampling from a single location with soil clay content of 18%, was 162 years (Frank and others 2012). Across ecosystems, variation in evapotranspiration rate (ET) was most closely correlated with variation in soil C turnover rate, with rates increasing exponentially for ET below 250 mm, and rates that remain consistently low across ecosystems with ET above

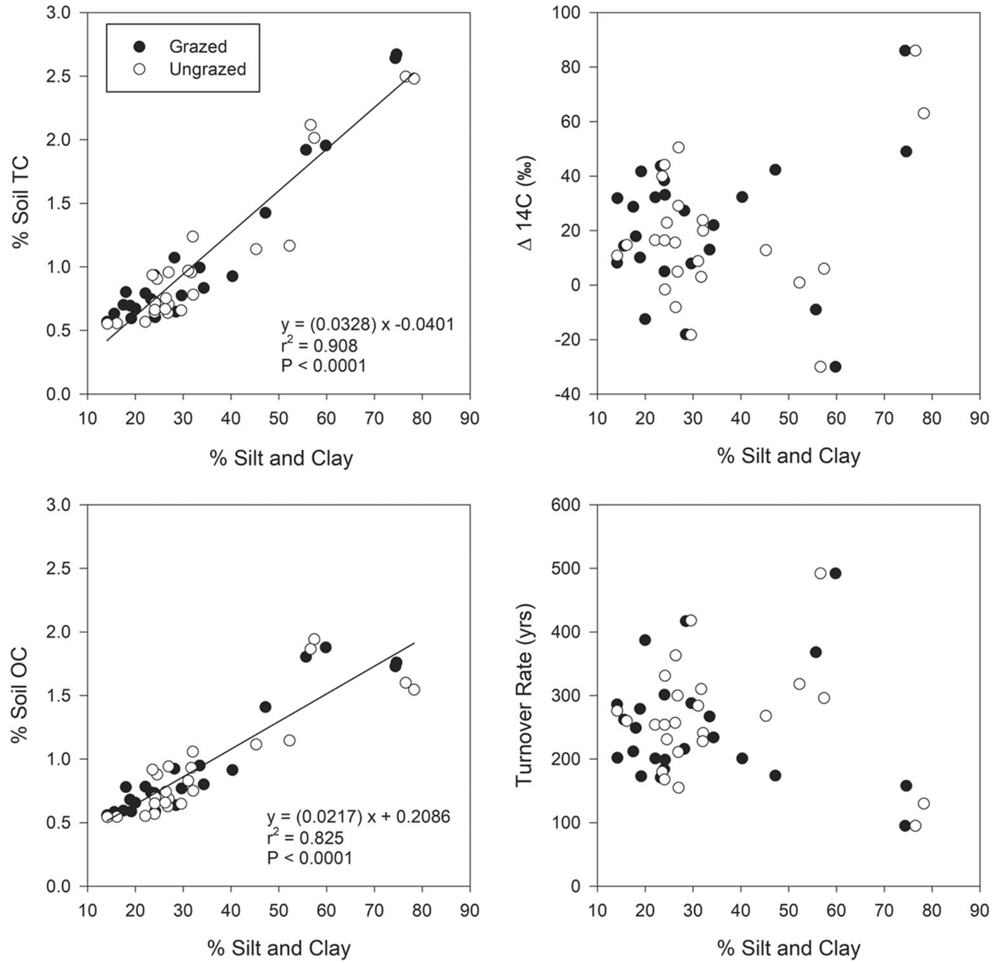

Figure 1. Soil carbon (percent total carbon and percent soil organic carbon), soil Δ^{14}C values, and turnover rate of the SOC responses to a soil texture (silt + clay) gradient and contrasting grazing treatments (moderately grazed vs. ungrazed) from 1939 to 2011 in the semiarid, shortgrass ecosystem of the western North American Great Plains.

400 mm (Frank and others 2012). For many of the semiarid grasslands and shrublands of North America, ET falls within the range of 250–400 mm (for example, ET at the Central Plains Experimental Range is 331 mm) where soil C turnover rates are most variable. Consistent with the predictions of Frank and others (2012) for such semiarid regions, we found that across a gradient of soil texture occurring within the 6270-ha Central Plains Experimental Range (from 12 to 80% silt + clay content), soil C turnover rates varied fivefold, from 95 to 492 years. Such variability in C turnover across a landscape that is similar in size to many livestock operations in the western Great Plains which have local topographic variation and spatial heterogeneity associated with plant distribution (Kelly and others 1998) illustrates the complexity of determining the potential for grazing management to influence soil C sequestration in semiarid grasslands. This variability is linked not only to soil texture effects on soil C pools (Figure 1), but also to the way that large interannual variability in precipitation inputs affects soil C turnover (Svejcar and others 2008; Ingram and others 2008). Such variability severely constrains our ability to measure and predict any additional influence of grazing management on soil C sequestration in these ecosystems.

Our study also provides novel information about the degree to which shifts in plant community composition brought about by grazing (Augustine and others 2017) may be linked to changes in SOC sequestration. Excluding cattle grazing for seven decades induced a compositional shift from dominance by a C4 shortgrass (*Bouteloua gracilis*) to co-dominance by a C3 midgrass (*Pascopyrum smithii*) and *B. gracilis*, as well as an increased abundance of annual, ruderal forbs and some sub-shrub species. Conversely, plant communities in the grazed sites contained perennial and annual growth forms that employed diverse strategies to co-exist with grazers and *B. gracilis* (Augustine and others 2017). Despite these notable differences in composition, moderately grazed versus ungrazed communities support similar levels of aboveground net primary production and respond similarly to interannual variation in spring precipitation (Irisarri and others 2016). We found that differences in plant community composition did not translate into differences in soil total C, SOC, N, or $\Delta^{14}C$ values or SOC turnover rate, supporting the primary role of soil moisture and total plant productivity in constraining plant–soil interactions in these semiarid grassland ecosystems (Burke and others 1998). Although we found that soil texture influenced total soil C, N, and SOC (consistent with Burke and others 1999), interactions between soil texture and grazing did not affect soil C variables, indicating no texture-dependent grazing effect in this semiarid grassland.

Soil C dynamics in water-limited, semiarid grassland ecosystems of the western North American Great Plains are influenced by short-term (season to multi-year) climatic variability that alters C source/sink status of the ecosystem (Svejcar and others 2008; Zhang and others 2010) with grazing exacerbating those fluxes when stocking is heavy and drought conditions are present (Morgan and others 2016). When these fluxes are evaluated over longer (decadal) time scales, similar directional trends emerge with a series of above-average precipitation and aboveground net primary production years manifesting in observed increases in SOC pools (for example, Schuman and others 1999; Reeder and Schuman 2002; Reeder and others 2004), whereas a sequence of dry/drought years result in SOC pool reductions (Ingram and others 2008). Our sampling in 2011 occurred near the end of the most recent cool phase of the Pacific decadal oscillation (PDO), which was characterized by much higher variability in and lower mean ANPP, as droughts were frequent and severe in the western Great Plains, compared to the prior warm phase of the PDO (Chen and others 2017). The use of ^{14}C radiocarbon in this study enabled the integration of two warm and cool phases of the Pacific decadal oscillation over the 6-decade period over which ^{14}C integrated SOC formation (1950–2011). Collectively, the absence of long-term differences for SOC sequestration between the grazing treatments showcases that the ecological pathways of grazing effects on SOC and nitrogen (Pineiro and others 2010) do not overcome the constraints imposed by moisture limitations in this semiarid grazing-resistant grassland.

These long-term (since 1939) findings improve an understanding of SOC formation that has been based on shorter-term (years to decade) flux experiments (for example, Morgan and others 2016) which are often primarily influenced by prevailing environmental conditions and secondarily by management. Extending the duration of flux measurement studies may provide better knowledge of how the high intra- and inter-annual variability in precipitation interacts with management to affect the long-term sink or source dynamics in shortgrass prairie. Here, we have explicit findings that unequivocally demonstrate that moderate grazing has no effect on long-term SOC sequestration in this semiarid grassland ecosystem. Moderate grazing in this ecosystem, about 12.5 animal unit days (AUD) per ha (Irisarri and others 2016), is intended to remove 40% of the ANPP each year, with mean ANPP of 680 kg per ha and a standard deviation of 210 kg per ha (Milchunas and others 1994). Long-term moderate grazing is sustainable for both livestock production (Hart and Ashby 1998) and plant community composition (Porensky and others 2016). This grazing intensity also provides economic feasibility for ranchers through optimizing livestock gains per animal and per unit land area (Bement 1969).

Collectively, a suite of US governmental policies related to private land management are emphasizing conservation-oriented management to provide a range of multiple ecosystem goods and services, including SOC sequestration, from grazing lands. Programs for private land managers through the farm bill and other non-governmental organizations emphasize monetary incentives to implement, adopt and/or modify land management practices to attain these ecosystem goods and services. The lack of responsiveness of SOC to moderate grazing in native rangeland identified here is an important and relevant message to program managers and policy makers concerning efforts to enhance SOC sequestration.

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