



Soil greenhouse gas emissions as impacted by soil moisture and temperature under continuous and holistic planned grazing in native tallgrass prairie



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ABSTRACT

Adaptive multi-paddock (AMP) grazing has demonstrated the potential to substantially improve ecosystems service outcomes relative to the most commonly used grazing management of moderate (MC) and heavy continuous (HC) grazing. We hypothesize that AMP grazing would decrease net soil emissions of CO₂, CH₄ and N₂O exchange between the soil surface and the atmosphere relative to continuous grazing and the management practice options of prescribed fire (AMP-burn), and production of hay (AMP-hay) both managed using AMP grazing. Soil temperature was lower ($P < 0.009$) and soil moisture higher ($P < 0.01$) with AMP grazing than with HC and MC grazing. As CO₂, CH₄ and N₂O emissions are less with lower temperatures and increasing soil moisture, they should have declined with AMP grazing. However, AMP grazing had the highest and HC the lowest CO₂ emissions, indicating higher levels of soil respiration, an index of soil microbial activity, with AMP. Emissions of N₂O were consistent with previous research, being higher under anaerobic conditions and very low under aerobic conditions. AMP, AMP-burn and AMP-hay treatments on average had lower N₂O emissions than HC and MC ($P \leq 0.002$). Methane (CH₄) emissions were negative for most sample dates but were dwarfed by the occasional periods when soils were saturated. These elevations in CH₄ emissions occurred on 8 of the 35 dates sampled (rate > 0; $P \leq 0.05$), 7 times for HC, 4 times for MC, and 3 times for AMP. On the remaining dates sampled (27 of 35), AMP was the strongest CH₄ sink ahead of AMP-burn ($P = 0.0335$), AMP-hay ($P = 0.0232$) and HC, but was similar to MC ($P = 0.17$). MC was a stronger sink than HC ($P = 0.057$). The emissions of CO₂ and N₂O were decreased with removal of green canopy material at sampling, indicating positive responses could be achieved by adjusting grazing management. Adaptive multi-paddock grazing, but not continuous grazing, can be adjusted to maintain higher proportions of green material, and as this would also benefit energy capture by photosynthesis and livestock diet quality, multiple benefits could accrue from implementing such management. Removal of green material had no influence on CH₄ oxidation, which was greatest with AMP grazing. These results are consistent with AMP grazing having a lower intensity ecological impact than continuous grazing.

1. Introduction

The atmospheric concentrations of greenhouse gases (GHGs), carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O), have increased sharply from pre-industrial levels and the intensification of agricultural land use is a major cause of these increases (Rigby et al., 2017). Agriculture is both a source and a sink of GHGs. Different agricultural management practices have different impacts on soil microbial community composition that strongly influence CO₂ and N₂O production and CH₄ production and oxidation in soils. The net exchange rates of CO₂, CH₄ and N₂O between soil and atmosphere are determined by the balance between sources and sinks and are mainly driven by temperature, moisture, and the supply of substrates from the

plants to soil biota that mediate these ecological functions (Smith et al., 2003; Levine et al., 2011; Morriën et al., 2017).

Soils of undisturbed, late-seral grassland ecosystems under extensive management are dominated by fungi and have much greater soil biota biomass, activity and diversity than disturbed and degraded soils (Bardgett and McAlister, 1999; de Vries et al., 2012; Morriën et al., 2017). These late-seral soil and plant communities comprise complex belowground networks that enhance carbon uptake, positively influencing ecosystem services reliant on the key parameters of soil cover to reduce insolation at the soil surface, increase soil water infiltration, soil permeability, the acquisition and retention of soil nutrients and nutrient cycling in these grazing ecosystems (de Vries et al., 2012; Morriën et al., 2017). These functions directly impact soil temperature and

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moisture levels and the supply of substrates from the plants to soil biota that govern microbial activity controlling CO₂ and N₂O production and CH₄ production and oxidation in soils. Different grazing management practices and decisions directly influence all these ecological functions. Intensive agricultural management practices such as tillage, use of inorganic fertilizers and biocides, and poor grazing practices that increase bare ground, remove key high-seral grasses and reduce biodiversity, result in diminished ecological services provided by healthy soils to negatively impact the soil biota that determine the dynamics between sources and sinks influencing net GHG emissions from soils (Levine et al., 2011; de Vries et al., 2013).

Carbon dioxide emissions resulting from respiration in soil and vegetation are the principal sources from which this gas enters the atmosphere, being substantially greater than emissions from fossil fuels (Smith et al., 2003). The release of CO₂ from soil organic matter where roots are present generally increases exponentially with temperature over a wide range of soil water contents but becomes a function of water content as a soil dries out to the point where microbial activity is reduced. In general, the greater the diurnal swing in temperature, the greater will be the mean respiration rate, so exposed soil will respire more than covered soil (Smith et al., 2003).

Aerobic soils are an important sink for CH₄, contributing up to 15% of annual global CH₄ oxidation and under aerobic conditions upland native grassland sites are weak sinks of CH₄ emissions (Saggar et al., 2007). In native prairie ecosystems the major portion of the landscape is upper or mid slopes that lose rain as runoff and are aerobic most of the time and, consequently, sinks of CH₄ emissions (Saggar et al., 2007). Methane is formed in soils by the microbial breakdown of organic matter under anaerobic conditions when oxidation potential is low due to reduced gas diffusivity and O₂ availability (Smith et al., 2003). Soil temperature impact on CH₄ oxidation is relatively small because, except when soil becomes dry enough to reduce microbial activity, reduced diffusivity is the major cause of decreased oxidation (Smith et al., 2003). Grazing management to restore degraded rangelands has shown the potential to increase CH₄ oxidation in temperate rangelands across a wide range of precipitation associated with different altitudes (Wang et al., 2014).

Most N₂O is generated from mineral N in animal dung and urine, biologically fixed N₂, mineralization of soil organic N in soils of native grasslands or grazed pastures, and indirect emissions from volatilized excreta and leached N (de Klein et al., 2003). Nitrous oxide is produced in the soil by the process of nitrification of ammonium (NH₃⁺) to nitrite (NO₂⁻) and then nitrate (NO₃⁻), and denitrification of nitrate to molecular nitrogen, N₂. As nitrification is an aerobic process, the rate of N₂O production increases as soil moisture increases, water filled pore space decreases, reducing diffusivity, and as temperature increases so does respiration, resulting in decreasing soil O₂ levels causing increasingly anaerobic conditions (Davidson et al., 2002a; Smith et al., 2003). Consequently, the amount of N₂O emitted depends substantially on the wetness of the soil that is also highly dependent on soil structure.

Abandonment of all agricultural and pastoral practices is often advocated to reverse land degradation by allowing natural succession processes to restore late-succession soil microbial and plant composition and function (Morriën et al., 2017). However, soil microbial and associated plant communities in grazing ecosystems did not evolve under abandonment, they coevolved as complex, dynamic ecosystems comprising grasses and soil biota, the grazers, and their predators (Retallack, 2013). Without these coevolved elements, natural succession processes proceed incompletely and at a slow pace (Blair, 1997). Mid and tall grasses thrive and remain competitive under infrequent light to moderate defoliation but deteriorate in the absence of disturbance in the form of fire, mowing, or infrequent grazing. These periodic disturbances, followed by adequate recovery, are necessary to maintain ecosystem function in grassland ecosystems (Frank et al., 2002; Teague et al., 2013). In undefoliated or lightly defoliated prairie, light is the primary limiting factor, and competition for light quickly

favors the tallest herbaceous plants and self-shading results in domination by just a few tall grass species, a reduction in plant growth, and ecosystem diversity decline (Seastedt and Knapp, 1993). Under these conditions, water and nitrogen (N) accumulate and there is a reduction in photosynthesis and nutrient cycling that reduces ecosystem productivity. Grazing removes light as a limiting factor and enhances nutrient cycling and biodiversity as other plants can compete better and N becomes the limiting factor (Seastedt and Knapp, 1993; Blair, 1997). In tall grass and mixed grass grazing ecosystems, this can result in compensatory growth under a light to moderate defoliation regime (Blair, 1997) or pulsed grazing with multi-paddock grazing management (Teague et al., 2013).

In rangeland ecosystems, maintaining normal soil function and ecosystem health is optimized when adequate plant and litter cover are present to provide protection from soil erosion and to optimize soil microorganism performance. Consequently, grazing management has a substantial impact on soil temperatures, water infiltration, runoff and erosion (Park et al., 2017a). Historically, many rangelands have been subjected to heavy continuous livestock grazing (CG). This sustains access to plants by grazers with limited opportunity for recovery between grazing events. This has been documented to contribute to serious negative effects including increased bare ground; soil temperature; decreased soil aggregation; diminished rooting depth, biomass, and carbohydrate reserves in selectively grazed plants; reduced above-ground biomass productivity; impoverished herbaceous plant communities; decreased soil carbon; and increased soil erosion and compaction (Teague et al., 2013; Park et al., 2017a). At landscape scales these changes have contributed to lower soil water infiltration, increased runoff, erosion and downstream flooding, and thus poorer water quality (Park et al., 2017a,b).

While rangeland academics widely advocate the use of continuous grazing with low stocking rates to reduce the damage caused by heavy continuous grazing (Briske et al., 2008), there is much evidence that appropriate stocking combined with adequate recovery from grazing is necessary to minimize damage and allow recovery of degraded ecosystem function (Müller et al., 2015). Holistic planned grazing protocols were specifically designed to emulate evolved grazing ecosystem processes (Savory and Butterfield, 2016) and have been particularly effective in reversing the damage caused by continuous grazing in a timely and cost-effective manner (Teague et al., 2013; Jakoby et al., 2015). It is based on stocking according to available forage, use of short grazing periods and adequate recovery from each grazing using high stock density, adaptive multi-paddock (AMP) management (Teague and Barnes, 2017).

The use of AMP grazing principles supports increased stocking rates while improving ecological function, as paddock number increases. This promotes resource restoration and regeneration of ecological function over a range of management scenarios that result in higher net returns and lower income variability (Jakoby et al., 2015; Teague et al., 2015). Although AMP grazing supports heavier stocking, this is not intensive grazing but *intensive management* of grazing. It primarily decreases the intensity of impacts on the vegetation and soil by specifically avoiding overstocking and overgrazing in an adaptive manner in response to variable weather (Jakoby et al., 2015; Teague et al., 2015; Savory and Butterfield, 2016). Overstocking is avoided with holistic planned grazing management by ensuring that livestock numbers do not exceed the amount of forage available ensuring maintenance of essential ecosystem functions. Overgrazing is avoided by having short grazing periods followed by adequate recovery after grazing. In combination, these actions result in light to moderate grazing impact on the soil, herbaceous vegetation and ecological functions (Teague et al., 2013; Jakoby et al., 2015; Savory and Butterfield, 2016; Teague and Barnes, 2017).

Fieldwork indicates that AMP grazing provides several positive outcomes relative to both moderate continuous (MC) and heavy continuous (HC) grazing and non-grazed enclosures (EX) (Teague et al.,

2011). These include: i) Increased soil organic matter and fungal to bacterial ratio associated with superior water-holding capacity and nutrient availability and retention as discussed by Bardgett and McAlister (1999) and de Vries et al. (2006); ii) Increased late-seral tall grass species composition that are the most productive in this ecosystem and are known to be obligate mycotrophs (Hartnett and Wilson, 1999); iii) Increased productivity relative to continuous grazing; iv) Increased herbaceous biodiversity compared to non-grazed enclosures; v) Decreased bare ground with AMP that decreases soil temperature since bare ground for AMP was less (1%) than with HC (30%) and with MC (4%); vi) Improved soil physical values with AMP relative to continuous grazing (HC) at the same stocking rate with: bulk density (0.91 vs. 1.06 g cm⁻³); aggregate stability (93% vs. 81%); penetration resistance (174 vs. 246 Joules); sediment loss (4.0 vs. 18.0 g m⁻²); and soil moisture (25% vs. 15% by volume); and vii) Improved water catchment hydrologic responses with AMP grazing management to be comparable to non-grazed areas, and superior to MC and HC grazing, indicating that AMP grazing practices maintain pastures at near natural conditions (Park et al., 2017a).

Adaptive multi-paddock grazing has demonstrated the potential to substantially modify soil temperature and moisture, and the supply of substrates from the plants to soil biota that mediate ecological functions affecting CO₂ and N₂O production, and CH₄ production and oxidation in soils. Consequently, we hypothesize that AMP grazing management will likely result in improved GHG outcomes relative to continuous grazing in this ecosystem. To address this hypothesis our objective is to determine the rates of CO₂ (soil respiration), CH₄ and N₂O exchange between the soil surface and the atmosphere under AMP grazing, relative to the two most common grazing practices of moderate and heavy continuous grazing, and the management practice options of prescribed fire, and production of hay.

2. Methods and materials

2.1. Study area

The study was conducted in the Fort Worth Prairie region, located in Cooke county south of Muenster, North Central Texas, USA (33.6518°N, 97.3764°W). As per Diggs et al. (1999), the climate is continental with an average frost-free growing days and mean annual temperature of 18 °C. Mean annual precipitation is 820 mm with a bimodal distribution peaking in May-June and September. Elevation ranges from 300 m to 330 m.

The vegetation in the area is rolling tallgrass prairie on the upland and midslope catenal positions with woody vegetation along the larger watercourses. The uplands and midslopes make up the major portion of the landscape and are dominated by the original native vegetation as they are too shallow for agriculture and generally have not been previously tilled. Consequently, they are still used primarily for livestock grazing and recreational hunting (Diggs et al., 1999). On the experimental sites the vegetation was native tallgrass prairie that had not previously been tilled or fertilized.

Native tall prairie grasses dominated the upland and midslope catenal positions and are comprised of tallgrasses *Schizachyrium scoparium*, *Andropogon gerardii*, *Sorghastrum nutans*, *Panicum virgatum* and midgrasses *Bouteloua curtipendula* and *Sporobolus compositus* in association with the perennial forbs *Ambrosia psilostachya*, *Aster ericoides* and *Gutierrezia texana*. Heavy, continuous grazing pressure has resulted in a switch to the grasses *Buchloe dactyloides*, *Bothriochloa laguroides*, *Bouteloua hirsuta*, *Nassella leucotricha* and annual forbs, particularly *Gutierrezia dracunculoides*, are more common (Dyksterhuis, 1946, 1948; Diggs et al., 1999).

Soils of the uplands and midslopes are predominantly clay-loams derived from limestone and have high permeability and relatively high soil organic matter (Teague et al., 2011). The upland catenal positions are of the shallow Aledo clay-loam series, and the midslope catenal

positions that made up the larger portion of the landscape are dominated by deeper clay-loam soils of the Sanger clay series (USDA, 2009). All sites sampled on all three ranches were of the Sanger clay series. The grazing management history on each of the ranches involved in the study had been the same for 15 years prior to the beginning of this study. Consequently, the soil and plant biology had likely adjusted to the very different grazing protocols being studied as indicated by Teague et al. (2011) working in this tall grass prairie region.

2.2. Management treatments

The treatments we compared to determine their impact on soil surface GHG dynamics in native, tallgrass prairie were adaptive multi-paddock (AMP), moderately stocked continuous (MC) and heavily stocked continuous (HC) grazing as depicted by Park et al. (2017b). In addition to the different grazing management, two other management treatments were sampled, both on the property using AMP grazing management. These were, cutting of forage with removal for hay (AMP-hay), and prescribed summer fire management (AMP-burn). We implemented the following treatments at the noted locations:

- 1 AMP treatments were sampled on the Pittman ranch. Grazing was conducted with replacement heifers using holistic planned grazing protocols (Savory and Butterfield, 2016). Grazing was for periods of 1 day followed by recovery periods of 30 to 45 days in fast growth periods and 80 to 100 days in slow growth periods and were adjusted according to existing conditions. At any one time the entire number of cattle grazed in one of 41 paddocks before moving to the next to allow sufficient recovery. Stocking rates¹ in animal units per 100 ha⁻¹ averaged 27 AU 100 ha⁻¹ and were adjusted when necessary to make sure forage on offer always exceeded the requirements of the livestock leaving enough residual to cover the soil from insolation and desiccation. Replicates were at the following locations 33.5018°N, -97.38333°W; 33.54667°N, -97.38361°W; 33.54404°N, -97.39604°W; 33.53770°N, -97.39311°W.
- 2 MC treatments were sampled on the neighboring Danglemayr ranch, which was stocked with beef cows and calves at NRCS recommended levels to average 14 AU 100 ha⁻¹. Numbers were adjusted when necessary in drought years to make sure forage on offer always exceeded the requirements of the livestock and left enough residual to support basic ecological function. Replicates were at the following locations 33.54413°N, -97.39642°W; 33.54393°N, -97.39636°W; 33.53747°N, -97.39308°W.
- 3 HC grazing was sampled on the neighboring Mitchell ranch that stocked each year with stocker steers at a high stocking rate of approximately 27 AU 100 ha⁻¹. Numbers were rarely adjusted from year to year. Replicates were at the following location 33.53118°N, -97.36877°W; 33.53099°N, -97.36795°W; 33.53073°N, -97.36813°W.
- 4 Summer fire (AMP-burn) was applied in August 2015 then grazed when recovered from the burn within the AMP grazing management outlined in 1 above. Replicates were at the following locations 33.55053°N, -97.38341°W; 33.54778°N, -97.383556°W; 33.54722°N, -97.38360°W.
- 5 Cutting for hay (AMP-hay) once each year at the end of summer in August with cut forage removal, then grazed following recovery the following spring within the AMP grazing management outlined in 1 above. Replicates were at the following locations 33.55081°N, -97.38305°W; 33.54753°N, -97.38398°W; 33.54720°N, -97.38392°W.

¹ An animal unit (AU) is defined as a bovine weighing 450 kg

2.3. Soil GHG emission monitoring

The 5 grazing treatments were conducted with 16 replicates sited at 6 locations within a 3 km radius. There were 4 replicates for AMP grazing, 3 replicates for MC, 3 for AMP-burn, and 3 for AMP-hay each from separate pastures. For HC, 3 pseudo-replicates were sampled from 2 separate, but adjacent, landscapes within a single pasture with the same slope and soil series as for the AMP and MC replicates.

The effect of green foliage on gas exchange was addressed by randomly choosing a chamber location at each site then pairing it as closely as possible with another adjacent chamber position based on vegetation to compare between: (1) the intact herbaceous sward (Green), and (2) the sward with green leaves and shoots removed leaving senesced plant material (Pluck). The gas exchanges of the two adjacent plots were determined concurrently. With each chamber pair the left chamber retained foliage while the right chamber green foliage was removed. This approach was conducted to reduce sampling mistakes. Foliage removed was returned as litter into the chamber's anchor after sampling. The chambers were re-established at the start of each year.

Sampling was conducted on 17 date pairs in 2015 and 20 date pairs in 2016 at close to 2-week intervals except during the winter with sampling at monthly intervals. Sampling of the 6 sites occurred during the morning hours over 2 days. Sampling order, whether the first or second day, morning, mid-morning, or noon, was exchanged at each sampling period to reduce confounding of sampling time and temperature with treatments.

Soil CO₂, CH₄ and N₂O samples were collected using circular steel anchors made from a 125 mm length × 200 mm diameter HREW round tube which was sharpened on one edge to aid insertion. They were driven 5 cm into the soil leaving 7-cm of anchor above the soil surface. The height to the lip of the anchor was determined for each sampling period and was used to adjust the combined chamber volume. The chambers were white, vented 200 mm, Schedule 40 PVC pipe caps (447-080, Spears Manufacturing Company, Sylmar, CA) with 3500 cm³ volume and sealed with rubber tubing between the chamber and the anchor. Samples were collected at 0, 10, 20, and 30 min using a 20 ml syringe drawn twice then emptied into a clean, evacuated 12 ml Exetainer vial (Labco, Lampeter, U.K.) to over-pressurize it.

Environmental data were collected concurrently with gas sampling. Barometric pressure and relative humidity data were collected from Gainesville, Texas airport 25 km northeast from the experimental sites. At each sampling date gravimetric soil moisture was calculated from 0 to 5 cm depth soil cores and temperature probes recorded temperature within the gas chambers at 5-minute intervals. Continuous monitoring of temperature and soil moisture was made at each site with automatic data loggers. Soil moisture probes were inserted diagonally at a depth of 2 to 10 cm for a mean depth of 6 cm, providing daily soil moisture by volume proportions. Temperature loggers recorded air temperature and soil temperature at 3 cm depths for each rep and within each green foliage denuded chamber. Visual estimates of live weight of foliage within the chambers as well as site estimates of bare ground, litter cover, and vegetation cover, live proportion of vegetation, and total litter, herbaceous, and green biomass were made for each sample chamber site and date.

Vials for use in the field were prepared in the laboratory prior to field sampling. Vials were evacuated, flushed with helium (He) (99.99% pure at 2 atm.), evacuated then rinsed with He, evacuated and rinsed again with He, then evacuated. Laboratory blanks consisting of 6 vials were immediately filled with He to test the efficacy of the evacuation procedure. Field blanks consisting of 6 vials were carried to the field, and following field sampling, were shipped over-night to the Texas A&M AgriLife Amarillo laboratory then filled with He prior to gas analysis.

2.4. Laboratory analyses

Gas samples were analyzed using a Varian 450 Gas Chromatograph

(Bruker Daltronics Inc. Billerica, MA) fitted with a ⁶³Ni electron capture detector (ECD), and a thermal conductivity detector (TCD) and a flame ionization detector (FID) in series, with a Combi-PAL autosampler (CTC Analytics AG, Zwingen, CH). A 2-mL sub-sample was auto-injected and split into two sample loops, delivering 500 μl to both the ECD, and the TCD and FID in series. For the N₂O measurements, the system was configured with a 0.5 m HayeSep N backflush column (Hayes Separations, Inc., Bandera, TX) followed by a 2 m HayeSep D analytical column. The carrier gas was P10 (90% Ar, 10% CH₄). Methane and CO₂ analysis were carried out using a 0.5 m HayeSep N backflush column and a 2 m Poropak QS (Waters Corp., Milford, MA) analytical column with Ultra High Purity helium carrier gas. Hydrocarbon-free air and Ultra High Purity hydrogen gas were supplied to the FID for combustion.

Calibration curves were developed for each detector using a commercial blend of N₂O (0.26, 1.01, 5.1, 25.0, 75.1, 150, 300 ppm), CH₄ (1.50, 5.01, 10.0, 100, 499, 2000, 10,000 ppm) and CO₂ (301, 501, 998, 5000, 10,000, 20000, 50,100 ppm) in air (Scott Specialty Gases, Plumsteadville, PA). Precision analysis expressed as the coefficient of variation for 30 replicate injections of 0.26 ppm N₂O 1.5 ppm CH₄ and 301 ppm CO₂ was 1.78, 0.81, and 2.23%, respectively. Minimum detectable fluxes calculated from the results of this precision analysis and the chamber dimensions were 0.024 mg m⁻² hr⁻¹ N₂O, 0.013 mg m⁻² hr⁻¹ CH₄, and 9.57 mg m⁻² hr⁻¹ CO₂ (Parkin et al., 2012).

2.5. Data processing and statistical analyses

During field sampling any exceptions were noted regarding time intervals measured and whether a normal amount of effort was required to empty the syringe into the vials or any other physical factors that might influence the accuracy of the gas sample collected. Upon return from the field, gas data notes were compared, and errant data were identified. Gases were analyzed in the laboratory within 4 days of collection.

Emission rates were measured for 30 min at time zero and at 10-minute intervals (n = 4). Generally, the emission rate followed a curvilinear response declining with time or was linear. When the curvilinear fit was best (based on R²) the emission rate tangent at 15 min was used. If not, the linear fit for the full 30 min was used. Accelerating curves were rejected. Covariable analyses were conducted by sampling date and gas vs. time and time squared to identify the gas emission rates and what curve fit the data best. For each date, gas and replication, linear and curvilinear regressions were compared to determine best fits.

Total yearly emission estimates were made because of an imbalance in the number of days sampled during winter and drought. Estimates of CO₂ - C emitted was made by adjusting each day unsampled with the proportion of (temperature °C) × (soil moisture cm³ cm⁻³ + 0.05)^{0.5} for the unsampled day compared to the 2 immediate sample days proportioned to days between. Beginning and ending days were based only on temperature, moisture ratios for the single nearest sample date. Daily estimates were based on observations that spikes in N₂O output occurred when temperature was warm and soil moisture high. Thus, between sample dates were based on the ratio of day temperature to sample date temperature times the ratio of day soil moisture to sample date soil moisture squared. Positive values of CH₄ were adjusted daily similarly to those for N₂O, however for negative values soil moisture was fit to a shallow bell-shaped curve (Soil Moisture⁴) because nitrification was slowed when very dry or wet. Though N₂O and CH₄ correlations with temperature and moisture were poor the primary driver of between-day estimates was the nearest sample date and provided a dynamic estimate of emissions.

The statistical model was 2 foliage × 5 grazing treatment with 3 replicates except 4 replicates for AMP grazing using a completely randomized design. Data were analyzed using the MIXED MODEL (SAS Institute, 2016). Prior to analysis, data were transformed to optimize normality and homogeneity of variance was judged using the

Table 1

Mean, green and total (above and below ground) herbaceous biomass with different grazing treatments.

		Grazing Treatments					Foliage	
		HC	MC	AMP	AMP burn	AMP hay	Green	Pluck
Mean biomass (g m ⁻² y ⁻¹)	2015	159 ^c	289 ^b	421 ^a	402 ^a	368 ^a	328	–
	2016	283 ^c	422 ^b	545 ^a	510 ^{ab}	419 ^b	436	–
Green biomass (g m ⁻² y ⁻¹)	2015	97 ^c	149 ^b	218 ^a	210 ^a	199 ^a	175	–
	2016	162 ^c	229 ^b	280 ^a	267 ^{ab}	246 ^{ab}	237	–
Total plant (OM-C g m ⁻² y ⁻¹)	2015	286 ^c	465 ^b	672 ^a	620 ^a	592 ^a	523	–
	2016	580 ^c	700 ^b	856 ^a	800 ^{ab}	772 ^{ab}	741	–

Shapiro–Wilk test (Schabenberger and Pierce, 2001). Environmental values were mostly normally distributed ($W > 0.94$) and not transformed. Flux rates benefited from log transformation. Values presented are non-transformed, but probabilities associated with differences and range of standard errors are based on transformed analyses. Splitting of CH₄ emission rates into higher positive and low to negative emissions considerably improved homogeneity of variance for the latter group.

3. Results

3.1. Grazing treatment impact on soil moisture and temperature

Precipitation during the project was close to the long term mean through 2014 until the end of March 2015. Much greater than normal precipitation persisted from April 2015 to October 2016 with expected dry and hot conditions in August of both years. Monthly ambient temperatures were close to expectation throughout.

AMP averaged 2.2x the biomass of HC ($P < 0.01$; Table 1) and 1.35x MC ($P < 0.03$) while MC average 1.6x more biomass than HC grazing for sampling dates in 2015 and 2016. Green biomass was about 0.5x biomass and production was about 1.5x biomass. The conservative proxy for vegetation production was estimated from peak years aboveground standing crop doubled (to account for belowground biomass) times 0.6 for OM (organic matter) to C (carbon) conversion. AMP combinations (AMP, AMP-burn, and AMP-hay) for biomass, green biomass and production were similar both years for AMP-burn, AMP-hay, and for MC were similar in 2016.

Soil moisture was depleted during periods of active plant growth and low precipitation and was recharged during fall and winter with adequate precipitation (Fig. 1). Depletion and recharge of soil moisture varied over the dates among grazing treatments ($P < 0.001$) with HC averaging less soil moisture at 0.26 cm³ cm⁻³ compared to MC at 0.29 cm³ cm⁻³ ($P = 0.069$) and AMP at 0.30 cm³ cm⁻³ ($P = 0.005$). AMP-burn and AMP-hay, and MC treatment moisture levels were similar. There was only a single estimate of soil moisture for the 16 pairs of plots with green vs. green foliage removed was made because of the destructive nature of obtaining gravimetric soil moisture samples

precluded sampling within the anchors.

Mean daily soil temperature follows the expected seasonal trends (Figs. 2 & 3). There were grazing treatment x date and foliage removal x date interactions with differences in the summer (both $P > 0.001$). Soil temperatures of the HC averaged 1.1 °C higher than MC ($P < 0.001$) and MC averaged 0.4 °C higher than AMP ($P = 0.009$) while Plucked averaged 0.5 °C higher than Green plots ($P < 0.001$). AMP-burn, AMP-hay and MC had similar temperatures. The temperature loggers provided precise measures. However, warmer temperatures associated with plucked treatment may be affected by proximity to the metal anchors of the plots.

3.2. Greenhouse gas emissions

3.2.1. Carbon dioxide

Carbon dioxide emissions over time were associated with soil moisture and temperature (Figs. 4a & 5a). These values were corrected for the temperature at sampling to the mean daily temperature. The multiple peaks in emissions in the warm seasons of the years are associated with rainfall and subsequent rise in soil moisture. Respiration rates of grazing treatments and foliage treatments are similarly small in the winter during which time the amount of green foliage was low. Interactions of grazing treatments and green foliage with date are highly significant ($P < 0.001$).

A couple of dates on the graphs (Figs. 4a & 5a) illustrate some variability encountered. The spike in CO₂ emissions on 6 August 2015 was because 1 of 4 AMP x Pluck replications had extreme venting of 23 g CO₂ - C m⁻² day⁻¹ during a period of soil drying and cracking. The graphs were adjusted to the AMP x Green Foliage paired rep value which was about twice as high as the other AMP values. The relative increase of HC on 27 October 2016 was for all replicates. At this time, the HC treatment had high amounts of actively green, flowering annual broomweed (*Gutierrezia dracunculoides*) compared to other grazing treatments which might have added high quality litter-fall. However, these exceptions had little effect on the statistics of main effects for the entirety of the experiment. Respiration rates were > 0 for all grazing treatments and foliage treatments by date ($P < 0.001$).

Interactions with dates of grazing treatments and foliage treatments were generally associated with increased size of differences during more optimum growing conditions. Therefore, the main effects are generally applicable. Treatments with higher amounts of green foliage had higher respiration rates (Tables 1 & 2). Green treatments had 81% more respiration than Pluck ($P < 0.001$). AMP grazing had 61% greater respiration than HC ($P < 0.001$) and 29% greater than MC ($P < 0.030$) but was similar to AMP-hay and AMP-burn treatments ($P > 0.45$). MC had 28% greater respiration than HC ($P = 0.050$).

Averages of respiration without correction for temperature and moisture (CO₂ - C, Fig. 5a, Table 2) are overestimated because sampling was suspended when the soil was very dry causing wide soil cracks. Also, sampling was less frequent during winter when temperatures were lower. Both are associated with lower respiration rates. Consequently, a

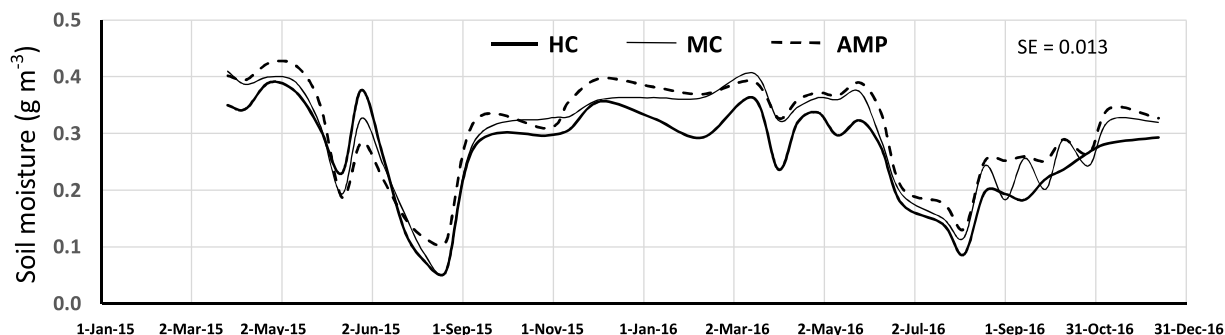


Fig. 1. Soil moisture measured at 1–10 cm depth (g cm⁻³) measured for each of the HC, MC and AMP grazing treatments.

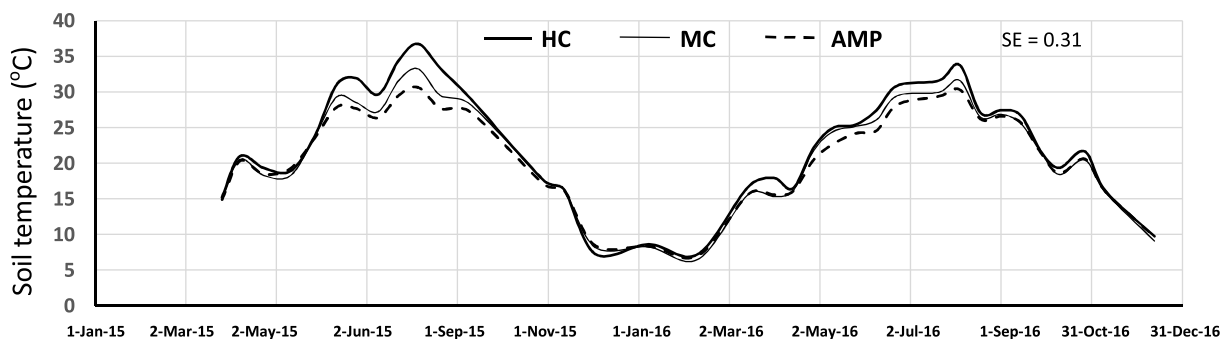


Fig. 2. Mean daily soil temperature at 5 cm (°C) measured for each of the HC, MC and AMP grazing treatments.

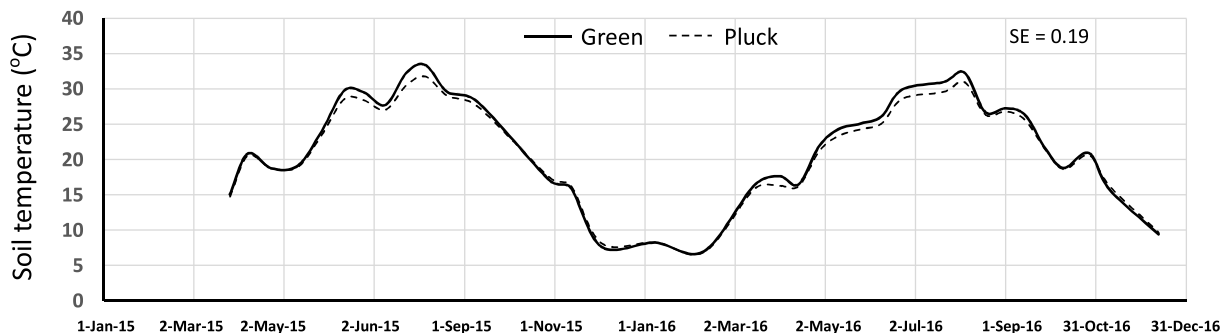


Fig. 3. Mean daily soil temperature (°C) of the intact herbaceous sward (Green) and with green leaf removed to leave senescent plant material (Pluck).

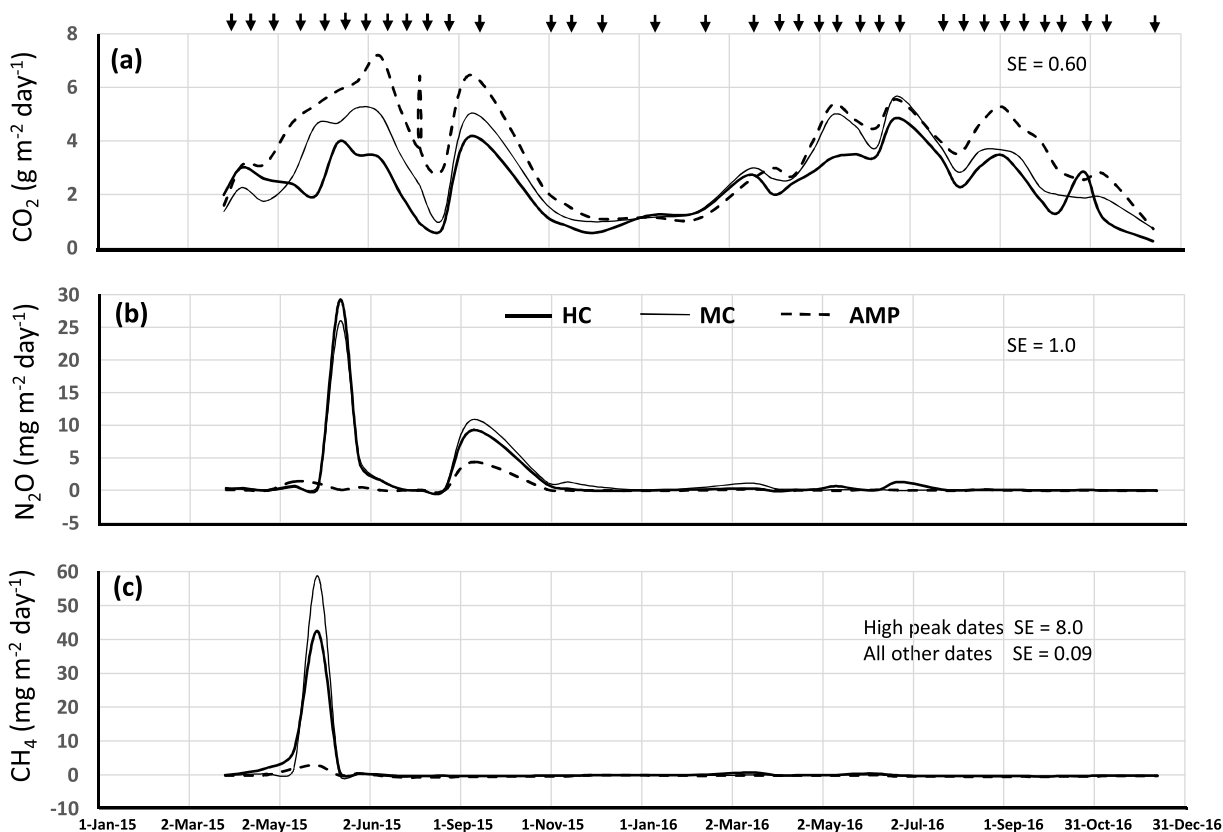


Fig. 4. Emissions for each of the HC, MC and AMP grazing treatments of: (a) carbon dioxide ($\text{CO}_2 - \text{C g m}^{-2} \text{ day}^{-1}$), (b) nitrous oxide ($\text{mg m}^{-2} \text{ day}^{-1}$) over all 35 sampling days, and (c) methane-C ($\text{mg m}^{-2} \text{ day}^{-1}$) over all 35 days sampled with negative, neutral or small positive emissions. Arrows (\downarrow) indicate sampling dates.

correction according to temperature and moisture for unsampled days between sample dates (daily $\text{CO}_2 - \text{C}$, Table 2) dates was applied (Fig. 5b). Without this correction respiration rates were overestimated

by 15–23%. Green treatments had 69% higher respiration rate than Pluck. AMP grazing had 57% greater respiration than HC and 29% greater respiration rate than MC while MC had 21% greater respiration

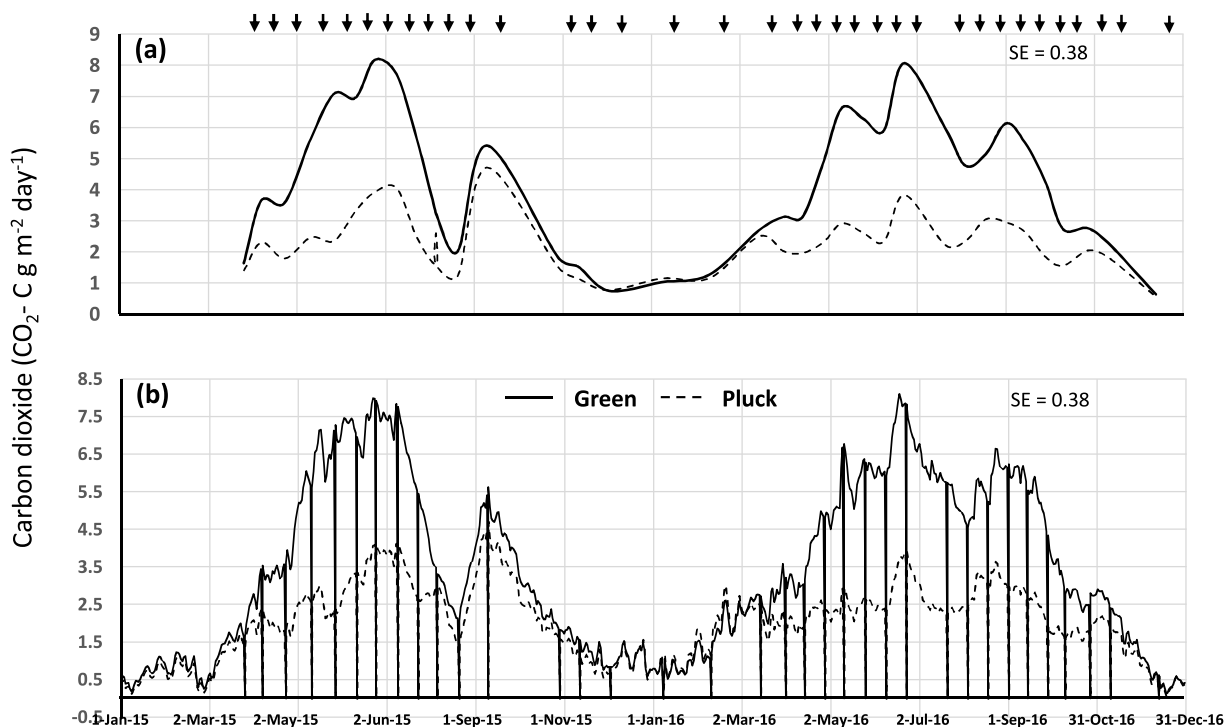


Fig. 5. Emissions of carbon dioxide ($\text{CO}_2 - \text{C g m}^{-2} \text{ day}^{-1}$) with: (a) the intact herbaceous sward (Green), and (b) with green leaf removed (Pluck) for HC, MC and AMP grazing treatments, and corrected for temperature and soil moisture on non-sampled days. Arrows (\downarrow) indicate sampling dates.

rate than HC.

Carbon release per year averaged $906 \text{ g CO}_2 - \text{C m}^{-2} \text{ year}^{-1}$ for HC, $1107 \text{ g CO}_2 - \text{C m}^{-2} \text{ year}^{-1}$ for MC, and $1436 \text{ g CO}_2 - \text{C m}^{-2} \text{ year}^{-1}$ for AMP grazing (Table 2). Mean standing crop of aboveground biomass was approximately $626 \text{ g m}^{-2} \text{ y}^{-1}$ (or production of $576 \text{ g OM-C g m}^{-2} \text{ y}^{-1}$). The respiration rate (daily $\text{CO}_2 - \text{C y}^{-1}$ of $1242 \text{ g m}^{-2} \text{ y}^{-1}$) accounted for about twice as much OM-C as estimated vegetation production OM-C. The excessive C can be attributed to plant respiration, plant exudates, and microbial breakdown of soil organic matter. The difference between respiration associated with Green and Pluck, 549 g C would likely be primarily associated with plant respiration and exudates. Respired CO_2 was related to productivity and green biomass and

does not reflect net CO_2 efflux. Sequestration potential should be greater with more productive ecosystems.

3.2.2. Nitrous oxide

Nitrous oxide emissions were episodic events with peaks in June and Sept 2015 that occurred during abnormally wet periods causing soil saturation that year (Fig. 4b). AMP treatment had substantially lower N_2O emissions during these peaks and in total in 2015, and although emissions were lower in 2016, they were statistically similar (Table 2). This was evident throughout both years of the project with average differences among grazing treatments being much smaller but still significant. AMP, AMP-burn and AMP-hay treatments had lower

Table 2
Greenhouse gasses; CO_2 -Carbon, N_2O and CH_4 -Carbon with different grazing treatments.

		Grazing Treatments					Foliage	
		HC	MC	AMP	AMP burn	AMP hay	Green	Pluck
$\text{CO}_2\text{-C}$ ($\text{g m}^{-2} \text{ y}^{-1}$)	2015	1014 ^c	1314 ^{bc}	1894 ^a	1805 ^{ab}	1779 ^{ab}	1561 ^a	886 ^b
	2016	1119 ^b	1366 ^{ab}	1695 ^a	1754 ^a	1620 ^a	1511 ^a	815 ^b
Daily $\text{CO}_2\text{-C}$ ($\text{g m}^{-2} \text{ y}^{-1}$)	2015	809	1000	1400	1265	1287	1152	643
	2016	1004	1215	1472	1537	1431	1332	742
N_2O ($\text{mg m}^{-2} \text{ y}^{-1}$)	2015	991 ^a	726 ^a	128 ^b	33 ^b	198 ^b	415 ^b	799 ^a
	2016	35 ^a	43 ^a	15 ^a	20 ^a	18 ^a	26 ^a	55 ^a
Daily N_2O ($\text{mg m}^{-2} \text{ y}^{-1}$)	2015	699	398	69	22	369	331	503
	2016	52	81	25	30	13	40	70
Daily est. N_2O $\text{CO}_2\text{-C m}^{-2} \text{ y}^{-1}$	2015	548	312	54	17	289	260	394
	2016	41	64	20	24	10	31	55
Percent of Production-C	2015	205	70	8	3	49	50	77
	2016	8%	9%	2%	3%	1%	4%	8%
$\text{CH}_4\text{-C}$ ($\text{mg m}^{-2} \text{ y}^{-1}$)	2015	1690 ^a	736 ^b	40 ^c	105 ^{abc}	582 ^{abc}	630 ^a	930 ^a
	2015	-22 ^a	-62 ^a	-80 ^a	-54 ^a	-40 ^a	-51 ^a	-47 ^a
Daily $\text{CH}_4\text{-C}$ ($\text{mg m}^{-2} \text{ y}^{-1}$)	2015	1072	516	14	85	341	400	600
	2016	-15	-67	-76	-18	-41	-51	-46
Daily est. $\text{CH}_4\text{-C}$ $\text{CO}_2\text{-C m}^{-2} \text{ y}^{-1}$	2015	71	34	1	6	22	26	39
	2016	-1	-4	-5	-1	-3	-3	-3
Percent of Production-C	2015	26%	8%	0%	1%	4%	5%	8%
	2016	0%	-1%	1%	0%	0%	0%	0%

emissions than HC and MC ($P \leq 0.002$). Presence of green foliage had a major impact on emissions with Pluck treatment averaging about twice the N_2O emissions of the Green treatment ($P = 0.001$). Of the 35 dates sampled, positive N_2O emissions (rate > 0 , $P = 0.05$) occurred 11 times for HC, 12 times for MC, 5 times for AMP, 4 times for AMP-burn, 11 times for AMP-hay, 10 times for Green, and 14 times for Pluck.

Most N_2O emissions were associated with prolonged periods of saturated soils (Fig. 4b). Saturated soils (days $\geq 0.40 \text{ cm}^3 \text{ cm}^{-3}$) occurred for 94 days in 2015 compared to 40 days in 2016. The number of dates sampled during saturated conditions happened to be proportional both years to those that occurred, however only 2015 had significant emissions of N_2O (Table 2). Contrary to saturated conditions, the 2nd largest emissions occurred September 10, 2015 following a period of prolonged soil drying ending with 12 mm precipitation boosting soil moisture from 0.05 to $0.16 \text{ cm}^3 \text{ cm}^{-3}$.

Although variability of nitrous oxide emissions was high, there were poor correlations with temperature or soil moisture levels. Day adjustments between sample dates were applied based on observations that spikes in N_2O output occurred when temperatures were warm and soil moisture was high. Daily estimates summed for 2015 indicate that N_2O output was 80% for Green and 63% for Pluck compared to dates sampled (Table 2).

The daily estimate for 2015 N_2O emission for AMP-hay was likely high based on the 1-day sampled on 27 March 2015 being $19 \text{ mg } N_2O \text{ m}^{-2} \text{ d}^{-1}$ vs 0 for the 5 other AMP-hay samples with next highest of all samples on that date being $0.85 \text{ mg } N_2O \text{ m}^{-2} \text{ d}^{-1}$. The high CO_2 equivalents associated with N_2O emissions were substantially elevated in 2015 for HC at 205% and MC at 70% of yearly production of those treatments while AMP was 8%. In 2016 N_2O emissions were significantly lower ($P < 0.06$).

3.2.3. Methane

Uptake or negative values associated with CH_4 oxidation occurred for most dates sampled, but were dwarfed by the occasional high emissions. This was particularly evident during spring 2015 that had periods with saturated soil (Fig. 4c). During the exceptionally wet period in May and June 2015 anaerobic conditions were experienced resulting in exceptionally high CH_4 emissions (up to $60 \text{ mg } CH_4 \text{ m}^{-2} \text{ d}^{-1}$) compared to maximum emissions under aerobic conditions for the remainder of 2015 and the whole of 2016 (up to $1 \text{ mg } CH_4 \text{ m}^{-2} \text{ d}^{-1}$). Lower emissions after June 2015 and through 2016 were associated with a return to aerobic conditions after the short abnormally wet period in May-June 2015. Consequently, we split analyses between these two periods of anaerobic and aerobic conditions. This had the advantage of isolating the biological process and improving variance associated with means to improve understanding of emissions during these different periods of reduction and oxidation of CH_4 .

Positive methane emissions that predominated in the anaerobic conditions in 8 of the 35 dates had high variances that made grazing treatment differences indistinguishable ($P > 0.11$) as were foliage treatment means ($P = 0.91$). A date x grazing treatment interaction ($P = 0.031$, Fig. 4c) revealed 2 dates affected: May 12, 2015 with Hay $>$ MC, AMP, and AMP-burn ($P < 0.09$) and May 28, 2015 with MC, HC, and AMP-hay $>$ AMP and AMP-burn ($P < 0.05$). Of the 8 dates sampled, positive CH_4 emissions (rate > 0 ; $P = 0.05$) occurred 7 times for HC, 4 times for MC, 3 times for AMP, 3 times for AMP-burn, 3 times for AMP-hay, 4 times for Green, and 5 times for Pluck.

The effectiveness of AMP grazing management as a methane sink was determined from the remaining 27 of 35 dates sampled under aerobic conditions that had negative, neutral or small positive CH_4 emissions. AMP was the strongest sink ahead of AMP-burn ($P = 0.0335$), AMP-hay ($P = 0.0232$) and HC ($P = 0.001$). MC was a stronger sink than HC ($P = 0.057$) but no different from AMP ($P = 0.157$). The foliage treatments were similar ($P = 0.16$), although Green was a 22% greater source of methane uptake than Pluck. Of the 27 dates sampled, negative CH_4 emissions (rate > 0 ; $P \leq 0.05$) occurred

15 times for HC, 17 times for MC, 21 times for AMP, 18 times for AMP-burn, 17 times for AMP-hay, 19 times for Green, and 17 times for Pluck. Within this data set, one additional positive methane output occurred for AMP-burn, HC, and Pluck.

The year summary indicates 3 grazing treatments had significant ($P < 0.03$) emissions of CH_4 in 2015 and day corrections were 64% that of sample date means (Table 2). In 2016 all grazing treatments were significant CH_4 sinks ($P < 0.04$). CO_2 equivalents associated with CH_4 emissions in 2015 were 26% for HC, 8% for MC and 4% for AMP-hay compared to their production C. Though significant ($P < 0.04$) sequestration of CH_4 occurred in 2016, those values were no more than 1% of production C.

4. Discussion

Differences in land use have been shown to affect soil function. Intensive land use reduces the diversity and abundance of many soil biota with negative consequences for the ecosystem services they underpin (de Vries et al., 2013). More productive grazing treatments or those treatments with greater green biomass have the highest soil respiration rates (CO_2 emissions). Although CO_2 is a GHG, its emissions at ground level does not indicate the net positive or negative sequestration in the soil / plant interface. More productive systems have greater biotic activity and greater potential to sequester C. Our yearly respiration output range was similar to that of the Davidson et al. (2002b) Amazon study that received over twice as much precipitation involving converted active pasture and degraded pasture. In the Amazon study it is not evident how material within the anchors was treated compared to ours where leaving litter and merely removing green vegetation before sampling reduced respiration by 44%.

Heavy continuous grazing (HC), the most intensive land use studied, produced the greatest emissions of N_2O and CH_4 but least CO_2 , while moderate continuous grazing (MC) had less intense impact on the land (Teague et al., 2011, 2013) and produced more moderate GHG emissions than HC. AMP grazing is used specifically to enhance key ecosystem processes and has been shown to have impact-lowering outcomes on land resources and ecological function, relative to MC and HC (Teague et al., 2011, 2013; Savory and Butterworth, 2016) and was the greatest sink for CH_4 and produced the least N_2O . These outcomes are achieved by ensuring animal numbers do not exceed the amount of forage available and using very short grazing periods (1–2 days) with adequate recovery after grazing, so the impact on the soil and plants is much less intense than caused by HC and MC.

AMP management specifically manages for: maximum amount of green leaf through the year to capture maximum energy via photosynthesis; ensuring sufficient litter and plant cover of the soil for optimal microbial function; maximizing infiltration; increasing rate of nutrient cycling and nutrient retention to enhance productivity and soil organic matter; and managing for the plant species composition that will facilitate optimal levels of these key ecological functions (Savory and Butterfield, 2016). AMP grazing and retaining green plant portions increased respiration and reduced CH_4 and N_2O emissions.

In this study, AMP grazing resulted in the lowest soil temperature and higher soil moisture than HC grazing, and yet had the highest biological activity (CO_2 emissions), with MC grazing values for both parameters intermediate between AMP and HC. According to Smith et al. (2003), increasing CO_2 emissions are usually related to increasing temperature and moisture. Previous field work at these sites (Teague et al., 2011) identified that, relative to HC and MC, AMP had: i) increased soil organic matter and cation exchange capacity; ii) increased fungal to bacterial ratio, indicating superior water-holding capacity and nutrient availability and retention; iii) increased the proportional composition of the most productive, late-seral, tall grass species; iv) increased productivity relative to continuous grazing; v) increased herbaceous biodiversity compared to non-grazed enclosures; vi) decreased bare ground for AMP ($< 1\%$) relative to HC (30%) and MC

(4%); vii) improved soil physical property values with AMP relative to continuous grazing (HC) at the same stocking rate with: better bulk density, aggregate stability, penetration resistance, sediment loss, and soil moisture.

Research has shown that having higher soil microbial activity, CO₂ emissions, and higher C sequestration are associated with less intensive impact on the land (Bardgett and McAlister, 1999; De Vries et al., 2012, 2013; Morriën et al., 2017). Therefore, in this study, responses to AMP, relative to MC and HC management, are consistent with AMP having the least intensive impact on the land, followed by MC, with HC having the most intensive impact. de Vries et al. (2013) conclude that the ecological functions associated with less intensive land use result in greater uptake and storage of C and increased efficiency of nutrient cycling with such advances in succession, and our results, in conjunction with those of Teague et al. (2011) at the same treatments and sites, are consistent with this.

Consequently, we speculate that the primary cause of AMP having the highest CO₂ emissions, followed by MC and least for HC is the elevated plant and microbial activity associated with AMP soil manifesting the most favorable, and HC the least favorable conditions for soil biota function (Teague et al., 2011). This was despite AMP treatment having the lowest temperatures and marginally higher soil moisture than HC and MC; lower temperatures are normally associated with lower CO₂ emissions (Smith et al., 2003; Cook and Orchard, 2008; Levine et al., 2011).

Nitrous oxide can be produced during nitrification under aerobic conditions or during denitrification under anaerobic conditions and increases as soil moisture increases due to reduced diffusivity within the soil, decreasing O₂ availability. As temperature increases, so does respiration, also resulting in decreasing soil O₂ levels causing increasingly anaerobic conditions (Smith et al., 2003). Consequently, the amount of N₂O emitted depends substantially on the wetness of the soil and is also highly dependent on soil structure. There were 2 spikes of N₂O emissions in our study, both of which occurred in a very dry period that followed very wet conditions (cf. Figs. 1 and 4b). Specifically, the wet periods in May 2015 followed by the very dry period in mid-June and the wet period in July followed by the very dry period in July-August 2015.

Our results are consistent with what is known regarding N₂O production as the wet periods in this study were unusually wet and visibly saturated at the time of the peaks in N₂O. This appeared to create anaerobic conditions to increase the rate of N₂O production. These were followed by dry periods that cause the soils of this area, moderately permeable clay-loams, to crack on drying that facilitates emission of the N₂O to the atmosphere. The development of cracks with drier conditions happens more completely and rapidly on soils that have less plant cover to protect the soil from insolation. The differences among HC, MC and AMP reflect this, as HC had 30% bare ground compared to MC at 4% and AMP at 1% (Teague et al., 2011). AMP exhibited far lower N₂O emissions during these peak emission periods and both years of the study, than MC and HC (Fig. 4b). The AMP-burn and AMP-hay treatments that were also grazed using AMP management, were no different from AMP and showed lower N₂O emissions than both MC and HC.

Aerobic soils are an important sink for CH₄, and upland native grassland sites are generally aerobic and weak sinks of CH₄ emissions (Saggar et al., 2007). They are net sources of CH₄ when anaerobic, such as when wet enough to reduce gas diffusivity. Soil temperature has a small impact unless soil moisture is low enough to reduce microbial activity (Smith et al., 2003). In this study there was a large spike in CH₄ production in late May-early June 2015 (Fig. 4c). This was likely caused by abnormally high rainfall causing very wet soil conditions (Fig. 1). Soil temperatures at this time ranged from around 27 °C with AMP to 33 °C with HC treatment. Unlike with N₂O, there was not a second spike in CH₄ emissions when August-September 2015 temperatures were higher at 32 °C with AMP, to 37 °C with HC treatment. It is possible that these temperatures were high enough in combination with the very low soil

moisture conditions, then to diminish soil microbial function, as explained by Smith et al. (2003).

On the 27 other dates sampled, the AMP and MC treatments were weak CH₄ sinks or small sources while HC was more of a source at least in summer of 2015. Overall AMP produced less emissions and was a larger and more frequent sink for CH₄ emissions than both MC and HC. Although AMP-burn and AMP-hay were both managed with AMP grazing, only AMP-burn was not different from AMP while Hay was the same as MC and HC (Fig. 4c).

Holistic planned grazing protocols, used in AMP management, were specifically designed to emulate evolved grazing ecosystem processes (Savory and Butterfield, 2016) and regenerate ecosystem functions degraded by the negative impacts of continuous grazing. AMP grazing management of the herbaceous plants aims at providing sufficient litter and plant cover of the soil to provide optimal microbial function, maximize rainfall infiltration, and ensure rapid rates of nutrient cycling and nutrient retention to enhance productivity and soil organic matter. These protocols manage for high biodiversity and optimum plant species composition to capture maximum sun energy via photosynthesis to facilitate optimal levels of key ecological functions.

Soil food web properties strongly influence processes of C and N cycling, but land use intensification commonly reduces the abundance of most functional groups of soil organisms (de Vries et al., 2013). This negatively impacts ecosystem services provided by healthy, functional and biodiverse soil biota and plant species as a foundation governing the processes underpinning sustainable agriculture. In previous field work, the common form of grazing management at this location, the intensive heavy continuous grazing (HC), produced the most negative impacts, while moderate continuous grazing (MC) had less intense impact on the land. In contrast with continuous grazing, AMP grazing demonstrated positive improvements in ecosystem function that resulted in higher levels of soil organic carbon (SOC) and a food web dominated by fungi (Teague et al., 2011), normally associated with extensively managed land (de Vries et al., 2013). This is consistent with other research reporting that the most extensive management of agricultural land (Kim et al., 2010; Larsen et al., 2011) showed soils dominated by fungi and high levels of C. It has been hypothesized that this was because fungal soils are presumed to be more efficient in their C use (Six et al., 2006; de Vries et al., 2012). In our study, AMP had the highest CO₂ emissions and had the highest SOC (Teague et al., 2011), which is consistent with results obtained by Ruser et al. (2006), Gelfand et al. (2015) and de Vries et al. (2013) on land managed less intensively.

Such enhanced levels of CO₂ production are a measure of soil heterotrophic activity and root respiration and represent a loss of C from soil. However, it is not a net loss of C from the soil because soil C levels are determined by the balance between the amount of C gained via plants through photosynthesis and C loss due to CO₂ respiration (de Vries et al., 2013). In this context, if domestic ruminants are managed in a way that restores and enhances grassland ecosystem function, carbon stocks in the soil are increased (Conant et al., 2001; Liebig et al., 2010; Teague et al., 2011; Machmuller et al., 2015). AMP grazing in particular, has been shown to result in more C entering the soil than is emitted indirectly or via ruminant emissions (Liebig et al., 2010; Wang et al., 2015; Rowntree et al., 2016; Paige et al., 2018). Ruser et al. (2006) and Gelfand et al. (2015) report that CO₂ emissions are higher from soils with higher carbon content and N₂O emissions are also controlled by available carbon.

The increase of CO₂ emissions and reduction of N₂O emissions with removal of green canopy material indicates positive responses could be achieved by adjusting management. Adaptive multi-paddock grazing, but not continuous grazing, could be adjusted to maintain higher levels of green material. This would also benefit energy capture by photosynthesis and livestock diet quality, so multiple benefits could accrue from implementing such management. Removal of green material had no influence on CH₄ oxidation, which was greatest with AMP grazing.

5. Conclusions

AMP grazing has previously demonstrated the potential to substantially improve ecosystems service outcomes relative to the most commonly used grazing management of moderate (MC) and heavy continuous (HC) grazing. Our goal was to determine if AMP grazing would decrease net soil emissions of CO₂, CH₄ and N₂O exchange between the soil surface and the atmosphere relative to continuous grazing and the management practice options of prescribed fire, and production of hay. Appropriate management leads to larger and more diverse populations of soil microbes, which in turn leads to greater carbon sequestration, and CH₄ oxidation. With livestock management focused on building soil health and biodiversity, grazing ruminants can create carbon negative budgets. The weather in 2015 was abnormally wet in April and May, producing spikes in GHG emissions. During these times, AMP resulted in much lower N₂O emissions than did continuous grazing. Otherwise during less extreme weather, grazing of this native prairie resulted in the soils generally being weak sinks of CH₄. At such times, AMP was almost invariably a larger CH₄ sink. Previous field work at this location indicated that AMP produced higher levels of soil organic carbon than continuous grazing and this was a major contributing factor to AMP providing a greater GHG sink because CO₂ emissions are higher from soils with higher carbon content and N₂O emissions are also decreased at higher levels of soil carbon.

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References

- Bardgett, R.D., McAlister, E., 1999. The measurement of soil fungal: bacterial biomass ratios as an indicator of ecosystem self-regulation in temperate meadow grasslands. *Biol. Fertil. Soils* 29, 282–290. <https://doi.org/10.1007/s003740050554>.
- Blair, J.M., 1997. Fire, N availability, and plant response in grasslands: a test of the transient maxima hypothesis. *Ecology* 78, 2359–2368. [https://doi.org/10.1890/0012-9658\(1997\)078\[2359:Fnaapr\]2.0.Co;2](https://doi.org/10.1890/0012-9658(1997)078[2359:Fnaapr]2.0.Co;2).
- Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash, A.J., Willms, W.D., 2008. Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangel. Ecol. Manage.* 6, 3–17. <https://doi.org/10.2111/06-159r.1>.
- Cook, F.J., Orchard, V.A., 2008. Relationships between soil respiration and soil moisture. *Soil Biol. Biochem.* 40, 1013–1018. <https://doi.org/10.1016/j.soilbio.2007.12.012>.
- Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: effects on soil carbon. *Ecol. Appl.* 11, 343–355. [https://doi.org/10.1890/1051-0761\(2001\)011\[0343:gmacij\]2.0.co;2](https://doi.org/10.1890/1051-0761(2001)011[0343:gmacij]2.0.co;2).
- Davidson, E.A., Savage, K., Verchot, L.V., Navarro, R., 2002a. Minimizing artifacts and biases in chamber-based measurements of soil respiration. *Agric. For. Meteorol.* 113, 21–37.
- Davidson, E.A., Verchot, L.V., Cattanio, J.H., Ackerman, I.L., Carvalho, J.E.M., 2002b. Effects of soil water content on soil respiration in forests and cattle pastures of eastern Amazonia. *Biogeochem.* 48, 53–69.
- de Klein, C.A.M., Barton, L., Sherlock, R.R., Li, Z., Littlejohn, R.P., 2003. Estimating a nitrous oxide emission factor for animal urine from some New Zealand pastoral soils. *Aust. J. Soil Res.* 41, 381–399. <https://doi.org/10.1071/sr02128>.
- de Vries, F.T., Thebault, E., Liiri, M., Birkhofer, K., Tsiafouli, M.A., Bjornlund, L., Jorgensen, H.B., Brady, M.V., Christensen, S., de Ruiter, P.C., d'Hertefeldt, T., Frouz, J., Hedlund, K., Hemerik, L., Hol, W.H.G., Hotes, S., Mortimer, S.R., Setälä, H., Sgardelis, S.P., Uteseny, K., van der Putten, W.H., Wolters, V., Bardgett, R.D., 2013. Soil food web properties explain ecosystem services across European land use systems. *Proc. Natl. Acad. Sci. U. S. A.* 110, 14296–14301. <https://doi.org/10.1073/pnas.1305198110>.
- de Vries, F.T., Bloem, J., Quirk, H., Stevens, C.J., Bol, R., Bardgett, R.D., 2012. Extensive management promotes plant and microbial nitrogen retention in temperate grassland. *PLoS One* 7, e51201. <https://doi.org/10.1371/journal.pone.0051201>.
- de Vries, F.T., Hoffland, E., van Eekeren, N., Brussaard, L., Bloem, J., 2006. Fungal/bacterial ratios in grasslands with contrasting nitrogen management. *Soil Biol. Biochem.* 38, 2092–2103. <https://doi.org/10.1016/j.soilbio.2006.01.008>.
- Diggs, G.M., Lipscomb, B.L., O'Kennon, R.J., 1999. *Illustrated Flora of North Central Texas*. Botanical Research Institute of Texas, Fort Worth, pp. 1623.
- Dyksterhuis, E.J., 1946. The vegetation of the Fort Worth Prairie. *Ecol. Monogr.* 16, 1–29. <https://doi.org/10.2307/1943572>.
- Dyksterhuis, E.J., 1948. The vegetation of the western cross timbers. *Ecol. Monogr.* 18, 325–376. <https://doi.org/10.2307/1948576>.
- Frank, D.A., McNaughton, S.J., Tracy, B.F., 2002. The ecology of the Earth's grazing ecosystems. *BioScience* 48, 513–521. <https://doi.org/10.2307/1313313>.
- Gelfand, I., Cui, M.D., Tang, J.W., Robertson, G.P., 2015. Short-term drought response of N₂O and CO₂ emissions from mesic agricultural soils in the US Midwest. *Agric. Ecosyst. Environ.* 212, 127–133. <https://doi.org/10.1016/j.agee.2015.07.005>.
- Hartnett, D.C., Wilson, G.W.T., 1999. Mycorrhizae influence plant community structure and diversity in tallgrass prairie. *Ecology* 80, 1187–1195.
- Jakoby, O., Quaas, M.F., Baumgärtner, S., Frank, K., 2015. Adapting livestock management to spatio-temporal heterogeneity in semi-arid rangelands. *J. Environ. Manage.* 162, 179–189. <https://doi.org/10.1016/j.jenvman.2015.07.047>.
- Kim, D.-G., Mishurov, M., Kiely, G., 2010. Effect of increased N use and dry periods on N₂O emission from a fertilized grassland. *Nutr. Cycl. Agroecosyst.* 88, 397–410. <https://doi.org/10.1007/s10705-010-9365-5>.
- Larsen, K.S., Andresen, L.C., Beier, C., Jonasson, S., Albert, K.R., Ambus, P.E.R., Arndal, M.F., Carter, M.S., Christensen, S., Holmstrup, M., Ibrom, A., Kongstad, J., Van Der Linden, L., Maraldo, K., Michelsen, A., Mikkelsen, T.N., Pilegaard, K.I.M., Priemé, A., Ro-Poulsen, H., Schmidt, I.K., Selsted, M.B., Stevnbak, K., 2011. Reduced N cycling in response to elevated CO₂, warming, and drought in a Danish heathland: synthesizing results of the CLIMATE project after two years of treatments. *Glob. Change Biol.* 17, 1884–1899. <https://doi.org/10.1111/j.1365-2486.2010.02351.x>.
- Levine, U.Y., Teal, T.K., Robertson, G.P., Schmidt, T.M., 2011. Agriculture's impact on microbial diversity and associated fluxes of carbon dioxide and methane. *ISME J.* 5, 1683–1691. <https://doi.org/10.1038/ismej.2011.40>.
- Liebig, M.A., Gross, J.R., Kronberg, S.L., Phillips, R.L., Hanson, J.D., 2010. Grazing management contributions to net global warming potential: a long-term evaluation in the northern Great Plains. *J. Environ. Qual.* 39, 799–809. <https://doi.org/10.2134/jeq2009.0272>.
- Machmuller, M.B., Kramer, M.G., Cyle, T.K., Hill, N., Hancock, D., Thompson, A., 2015. Emerging land use practices rapidly increase soil organic matter. *Nat. Commun.* 6. <https://doi.org/10.1038/ncomms7995>.
- Morriën, E., Hannula, S.E., Snoek, L.B., Helmsing, N.R., Zweers, H., de Hollander, M., Soto, R.L., Bouffaud, M.-L., Buée, M., Dimmers, W., Duyts, H., Geisen, S., Giralanda, M., Griffiths, R.I., Jørgensen, H.-B., Jensen, J., Plassart, P., Redecker, D., Schmelz, R.M., Schmidt, O., Thomson, B.C., Tisserant, E., Uroz, S., Winding, A., Bailey, M.J., Bonkowski, M., Faber, J.H., Martin, F., Lemancaeu, P., de Boer, W., van Veen, J.A., van der Putten, W.H., 2017. Soil networks become more connected and take up more carbon as nature restoration progresses. *Nat. Commun.* 8, 14349. <https://doi.org/10.1038/ncomms14349>.
- Müller, B., Schulze, J., Kreuer, D., Linstädter, A., Frank, K., 2015. How to avoid unsustainable side effects of managing climate risk in drylands - the supplementary feeding controversy. *Agric. Syst.* 139, 153–165. <https://doi.org/10.1016/j.agsy.2015.07.001>.
- Paige, L.S., Rowntree, J.E., Beede, D.K., DeLonge, M.S., Hamm, M.W., 2018. Impacts of soil carbon sequestration on life cycle greenhouse gas emissions in Midwestern USA beef finishing systems. *AG Syst.* 162, 249–258. <https://doi.org/10.1016/j.agsy.2018.02.003>.
- Park, J.Y., Ale, S., Teague, W.R., Dowhower, S.L., 2017a. Simulating hydrologic responses to alternate grazing management practices at the ranch and watershed scales. *J. Soil Water Conserv.* 72, 102–121. <https://doi.org/10.2489/jswc.72.2.102>.
- Park, J.Y., Ale, S., Teague, W.R., Jeong, J., 2017b. Evaluating the ranch and watershed scale impacts of using traditional and adaptive multi-paddock grazing on runoff, sediment and nutrient losses in North Texas, USA. *Agric. Ecosyst. Environ.* 240, 32–44. <https://doi.org/10.1016/j.agee.2017.02.004>.
- Parkin, T.B., Venterea, R.T., Hargreaves, S.K., 2012. Calculating the detection limits of chamber-based soil greenhouse gas flux measurements. *J. Environ. Qual.* 41, 705–715. <https://doi.org/10.2134/jeq2011.0394>.
- Retallack, G.J., 2013. Global cooling by grassland soils of the geological past and near future. *Annu. Rev. Earth Planet. Sci.* 41, 69–86. <https://doi.org/10.1146/annurev-earth-050212-124001>.
- Rigby, M., Montzka, S.A., Prinn, R.G., White, J.W.C., Young, D., O'Doherty, S., Lunt, M.F., Ganesan, A.L., Manning, A.J., Simmonds, P.G., Salameh, P.K., Harth, C.M., Muhle, J., Weiss, R.F., Fraser, P.J., Steele, L.P., Krummel, P.B., McCulloch, A., Park, S., 2017. Role of atmospheric oxidation in recent methane growth. *Proc. Natl. Acad. Sci. U. S. A.* 114, 5373–5377. <https://doi.org/10.1073/pnas.1616426114>.
- Rowntree, J.E., Ryals, R., DeLonge, M.S., Teague, W.R., Chiavegato, M.B., Byck, P., Wang, T., 2016. Potential mitigation of Midwest grass-finished beef production emissions with soil carbon sequestration in the United States of America. *Future Food J. Food Agric. Soc.* 4, 31–38.
- Ruser, R., Flessa, H., Russow, R., Schmidt, G., Buegger, F., Munch, J.C., 2006. Emission of N₂O, N₂ and CO₂ from soil fertilized with nitrate: effect of compaction, soil moisture and rewetting. *Soil Biol. Biochem.* 38, 263–274. <https://doi.org/10.1016/j.soilbio.2005.05.005>.
- Saggar, S., Hedley, C.B., Giltrap, D.L., Lambie, S.M., 2007. Measured and modelled estimates of nitrous oxide emission and methane consumption from a sheep-grazed pasture. *Agric. Ecosyst. Environ.* 122, 357–365. <https://doi.org/10.1016/j.agee.2007.02.006>.
- SAS, 2016. *SAS/STAT Guide for Personal Computers*. Version 9, 4 edition. Cary, NC, 2016.
- Savory, A., Butterfield, J., 2016. *Holistic Management*, 3rd ed. Island Press, Washington, DC.
- Schabenberger, O., Pierce, F.J., 2001. *Contemporary Statistical Models for the Plant and Soil Sciences*. CRC Press, Washington, DC.
- Seastedt, T.R., Knapp, A.K., 1993. Consequences of Nonequilibrium Resource Availability

- Across Multiple Time Scales: the Transient Maxima Hypothesis 141. pp. 621–633. <https://doi.org/10.1086/285494>.
- Smith, K.A., Ball, T., Conen, F., Dobbie, K.E., Massheder, J., Rey, A., 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. *Eur. J. Soil Sci.* 54, 779–791. <https://doi.org/10.1046/j.1351-0754.2003.0567.x>.
- Six, J., Frey, S.D., Thiet, R.K., Batten, K.M., 2006. Bacterial and fungal contributions to carbon sequestration in agroecosystems. *Soil Sci. Soc. Am. J.* 70, 555–569. <https://doi.org/10.2136/sssaj2004.0347>.
- Teague, W.R., Barnes, M., 2017. Grazing management that regenerates ecosystem function and Grazingland livelihoods. *Afr. J. Range For. Sci.* 2017, 1–10. <https://doi.org/10.2989/10220119.2017.1334706>.
- Teague, R., Grant, B., Wang, H.-H., 2015. Assessing optimal configurations of multi-paddock grazing strategies in tallgrass prairie using a simulation model. *J. Environ. Manage.* 150, 262–273. <https://doi.org/10.1016/j.jenvman.2014.09.027>.
- Teague, R., Provenza, F., Kreuter, U., Steffens, T., Barnes, M., 2013. Multi-paddock grazing on rangelands: why the perceptual dichotomy between research results and rancher experience? *J. Environ. Manage.* 128, 699–717. <https://doi.org/10.1016/j.jenvman.2013.05.064>.
- Teague, W.R., Dowhower, S.L., Baker, S.A., Haile, N., DeLaune, P.B., Conover, D.M., 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. *Agric. Ecosyst. Environ.* 141, 310–322. <https://doi.org/10.1016/j.agee.2011.03.009>.
- USDA Natural Resources Conservation Service, 2009. Official Soil Series Descriptions (OSD). <http://soils.usda.gov/technical/classification/osd/index.html>.
- Wang, C.J., Han, G.D., Wang, S.P., Zhai, X.J., Brown, J., Havstad, K.M., Ma, X.Z., Wilkes, A., Zhao, M.L., Tang, S.M., Zhou, P., Jiang, Y.Y., Lu, T.T., Wang, Z.W., Li, Z.G., 2014. Sound management may sequester methane in grazed rangeland ecosystems. *Sci. Rep.* 4, 4444. <https://doi.org/10.1038/srep04444>.
- Wang, T., Teague, W., Park, S., Bevers, S., 2015. GHG mitigation potential of different grazing strategies in the United States Southern Great Plains. *Sustainability* 7, 13500–13521. <https://doi.org/10.3390/su71013500>.