

Article

Strategic Grazing in Beef-Pastures for Improved Soil Health and Reduced Runoff-Nitrate-A Step towards Sustainability

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Abstract: Generally, improvement in the soil health of pasturelands can result in amplified ecosystem services which can help improve the overall sustainability of the system. The extent to which specific best management practices have this effect has yet to be established. A farm-scale study was conducted in eight beef-pastures in the Southern Piedmont of Georgia, from 2015 to 2018, to assess the effect of strategic-grazing (STR) and continuous-grazing hay distribution (CHD) on soil health indicators and runoff nitrate losses. In 2016, four pastures were converted to the STR system and four were grazed using the CHD system. Post-treatment, in 2018, the STR system had significantly greater POXC (by 87.1, 63.4, and 55.6 mg ha⁻¹ at 0–5, 5–10, and 10–20 cm, respectively) as compared to CHD system. Soil respiration was also greater in the STR system (by 235 mg CO₂ m⁻² 24 h⁻¹) and less nitrate was lost in the runoff (by 0.21 kg ha⁻¹) as compared to the CHD system. Cattle exclusion and overseeding vulnerable areas of pastures in STR pastures facilitated nitrogen mineralization and uptake. Our results showed that the STR grazing system could improve the sustainability of grazing systems by storing more labile carbon, efficiently mineralizing soil nitrogen, and lowering runoff nitrate losses.

Keywords: soil health; grazing systems; runoff-nitrate; strategic grazing

1. Introduction

The growing cognizance of scientists and producers on soil health and its implication on agricultural sustainability has propelled research works on creating management strategies for improving soil health [1–3] in various agroecosystems. Given the vast ecosystem services grazing lands provide [4–6], more effort is needed toward the development of soil health-focused grazing management strategies. The United States has about 308 million ha of grazing lands, which is approximately 31% of the total land area [7,8]. This area provides numerous ecosystem services and is an important contributor to the national GDP (gross domestic product). The Southeast region has approximately 11.6 million ha grazing lands [9] and is home to 20% of the national beef-cattle herd [10]. Moreover, between 2005 and 2015, almost 4000 ha of row-crop land was converted to intensive grazing farms in Georgia [11]. Cow-calf production in pastures is common in this region due to climatic suitability and forage availability [12], however, most of the beef-pastures are in fragmented marginal land that is vulnerable to erosion and usually not suitable for row-crop production [10]. Thus, there is a need for more studies to better understand these livestock systems in an effort to create grazing management systems that are



more sustainable and suitable for these ecoregions. In the Southeastern USA, we define conventional grazing systems as pastures continuously grazed with little control over grazing time in specific locations within pastures. This results in cattle-preference to certain areas [13,14], leading to uneven nutrient distribution [15,16] and inefficient land utilization. High cattle activity near pasture equipages (water, shade, hay, mineral blocks, etc.) in the conventional system results in nutrient hotspots, soil compaction [17,18], and erosion resulting in poor fertility and vegetative cover in these steep, marginal areas of the pastures which exacerbates the ecosystem's vulnerability to runoff losses.

In a recent study, it was suggested that the US cattle industry was significantly affected by drought conditions in 2017 [19]. Additionally, high-intensity rains caused by hurricanes can increase the amount of runoff resulting in further loss of soil and nutrients. In changing global climatic scenario, a greater number of extreme weather events such as droughts and hurricanes are predicted to occur more frequently [20], which might have overwhelming impacts on these already marginal lands.

Soil health assessment is foundational in determining the sustainability of grazing lands, thus an improvement, or at least maintenance, of the current state of soil health is important for improving the overall sustainability of pastoral systems [21-23]. The soil health indicators we measured in this study have been reported as reliable as well as sensitive to management changes in agroecosystems. The active carbon fraction measured as POXC (permanganate oxidizable carbon) [24] and soil respiration are recommended measures of soil health [25,26]. A study in Georgia cattle pastures [27] reported a range of 201–1468 mg kg⁻¹ POXC at 0–5 cm soil depth and it has been reported [27,28] that a larger POXC pool is an indicator of a healthy and more sustainable system. mineralization of the mineralizable nitrogen pool to the plant available form and effective plant uptake also indicates a grazing system that is more sustainable [22,23,29]. We expect an increase in plant-available nitrogen, lower runoff-nitrogen, and improved forage productivity. Pilon et al. [30] reported that average nitrate losses from continuously grazed pastures and rotationally grazed pastures with fenced riparian buffer ranged from 0.3–1.8 kg NO₃⁻N ha⁻¹ and 0.03–5.53 NO₃⁻N kg ha⁻¹, respectively. The Pilon et al. [30] study was conducted in broiler litter fertilized pastures and authors were unsure of the underlying cause of higher nitrate in rotational pastures. They speculated that higher forage biomass in rotational pastures trampled during the wet period contributed to increased nitrogen losses. We expect to reduce runoff-nitrate by using better management practices.

Our study hypothesized that a collection of better grazing management practices would improve soil health indicators and reduce runoff-nitrogen as compared to the existing conventional grazing systems or rotational grazing only in Southeastern USA.

2. Materials and Methods

2.1. Study Site

The experiment was conducted from May 2015–June 2018 in eight beef-pastures (Figure 1), including (i) four pastures in Animal and Dairy Science Department Eatonton Beef Research Unit (33.420759° N, 83.476555° W) in Eatonton, GA (referred to as Eatonton pastures for the remainder of the manuscript), and (ii) four pastures in J. Phil Campbell Sr. Research and Education Center (33.887487° N, 83.420966° W) in Watkinsville GA (referred to as Watkinsville pastures for the remainder of the manuscript). The study areas, (Eatonton or Watkinsville), have a humid subtropical climate with an average minimum and maximum annual temperature of 10.4 °C or 11.1 °C and 22.5 °C or 25.6 °C, respectively. The mean annual rainfall for Eatonton and Watkinsville pastures were 1190 and 1230 mm respectively. The soils in Eatonton pastures were classified as fine, kaolinitic, thermic Rhodic Kandiudults, loamy, mixed, active, thermic, shallow Typic Hapludalfs with textural class of sandy loam, clay loam, and loam. The average sand, silt, and clay content of Eatonton soils were 45%, 35%, and 20% respectively. The soils in Watkinsville pastures were classified as fine, kaolinitic, thermic Typic Kanhapludults with predominately sandy loam and sandy clay loam textural classes. The average sand, silt, and clay content of Watkinsville soils were 50%, 22%, and 28% respectively.

Historically (>15 years), the tall fescue (*Festuca arundinacea* Schreb)/bermudagrass (*Cynodon dactylon* L.) mixed-pastures in both sites were managed using a continuous grazing system with a variable stocking density of 1.2-1.8 cows ha⁻¹ in Eatonton and 1.8-2.4 cows ha⁻¹ in Watkinsville.



Figure 1. Two study sites, (**left**) Eatonton, and (**right**) Watkinsville, showing the treatment arrangements, soil sampling locations, water/powerlines, watersheds, runoff collectors, exclusions, and rotational paddock delineation. STR = strategic-grazing pastures, CHD = continuously grazed with hay distribution.

2.2. Treatments

The experiment was conducted using before-after as well as with-without approach. In farm-scale studies, it is often difficult to control various environmental and managerial factors; thus, a baseline study was conducted before the implementation of treatments to allow comparison between pastures before the implementation of treatments. Two grazing systems; (i) strategic-grazing (STR) and (ii) continuous-grazing hay distribution (CHD) were designed and implemented in May 2016 as shown in Figure 1. The CHD system was a modification of a continuous grazing system where hay was distributed in various locations across the pasture to reduce the time cattle spent in historical "cattle camping areas" and to distribute the carbon associated with the feeding of hay. The STR system was a combination of multiple better grazing practices listed in Table 1 and was designed for efficient distribution of cattle associated nutrients. Hay was to be fed only when needed and need was determined by farm managers at both locations. Hay was fed only during the drought of 2016. In Eatonton, 34 and 90 hay bales were distributed in the STR and CHD pastures, respectively. In Watkinsville, six dry hay bales and six silage hay bales were distributed in CHD pastures and no hay was fed in STR pastures.

Better Grazing Practices	Description of the Grazing Practice				
Manure distribution by lure management of cattle	Portable hay rings, waterers, and shade structures were strategically rotated in various locations of pasture. The placement of these pasture equipages was driven by the nutrient distribution and pasture health.				
Exclusion of vulnerable areas of pasture	Compacted and/or nutrient-rich areas (areas of interest: AOIs) in pastures caused by high cattle activity/preference we excluded using an electric fence. AOIs were either uphill depositional areas and erosional landscape positions or down depositional and erosional landscape positions.				
Over-seeding the exclusions	The excluded areas were over-seeded with productive forage mix (grass-legume) during summer and spring every year. The exclusions were over-seeded as follows: (i) May 2016: pearl millet, crabgrass, and cowpea, November 2016: oat (<i>Avena sativa</i> L.), ryegrass (<i>Lolium multiflorum</i> L.), crimson clover (<i>Trifolium incarnatum</i> L.), forage rape (<i>Brassica napus</i> L.); May 2017: crabgrass (<i>Digitaria sanguinalis</i> L.), pearl millet (<i>Pennisetum glaucum</i> L.), and cowpea (<i>Vigna unguiculate</i> L).				
Flash/Mob grazing of the exclusions	After full growth, the exclusions were flash grazed (4 hours in the morning) and cattle were taken out from exclusion, every day until all forage was consumed.				
Moderate rotational grazing	Each STR pasture divided into 8 smaller sub-paddocks and moderate rotational grazing (7–10 days) was followed to allow forage regrowth.				

Table 1.	Descrit	ption	of better	grazing	practices	imple	mented i	in the	STR	grazing sv	zstem
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2.3. Soil Sampling and Soil Respiration Measurement

A 50-m grid was laid out in all pastures, and 18% of the 50-m grid points (a total of 10 points in each pasture) were randomly selected for soil sampling (which will be referred to as "Matrix" samples for the remainder of the manuscript). Additional samples (10–15 points in each pasture) were also collected from areas frequented by cattle and vulnerable to nutrient loss (Table 1) from the pasture (referred to as "AOIs": Areas of Interest samples for the remainder of the manuscript). In May 2015, two replicate soil samples were collected from each sampling location at three soil depths (0–5 cm, 5–10 cm, and 10–20 cm; baseline). Soil samples were collected using a 5-cm-diam Giddings Hydraulic Probe mounted on a truck. On the day of soil sampling, in situ soil respiration was measured at each sampling location using the alkali trap method as described by Anderson [31]. One mol L⁻¹ sodium hydroxide (NaOH) traps were put inside a PVC chamber pushed 5 cm into the soil. After 24 hours, the traps were harvested and analyzed for CO₂ by titrating with hydrochloric acid following the addition of barium chloride (BaCl₃). After treatment soil samples were collected, and in situ soil respiration was measured in summer (June–July) of 2017 and 2018 in a similar manner.

2.4. Soil Analysis

All soil samples were air-dried, ground and sieved (2-mm mesh), and stored in airtight plastic bags. Three grams of soil was extracted using 2 mol L^{-1} KCl (potassium chloride) as described by Maynard and Kalra, [32]. From the extract, ammonium (NH₄⁺-N) was measured as suggested by Kempers and Zweers [33], and the nitrate (NO₃⁻-N) was measured as described by Doane and Horwath [34]. Potentially mineralizable nitrogen (PMN) was measured using the hot-KCl extraction method described by Picone et al. [35]. PMN was calculated as the difference of NH₄⁺-N measured from the hot-KCl extraction method and NH₄⁺-N measured from the cold-KCl extraction method. Inorganic nitrogen (IN) was calculated as the sum of NH₄⁺-N and NO₃⁻-N measured from the cold-KCl extraction by Weil et al. [24].

2.5. Runoff Collection and Analysis

In each pasture, 3–4 pour-point runoff collectors were established at the edge-of-field, downhill of AOIs. Contributing areas to each runoff collector were delineated using ArcGIS 10.6 (Figure 1). Runoff was collected immediately after each runoff event, filtered, and analyzed for nitrate (NO₃⁻) throughout the study period. The runoff collectors had 3 to 5 Nalgene bottles placed 5-cm apart vertically to allow for vertical amalgamation of runoff nitrate concentrations (mg L⁻¹). Nitrate concentrations from each bottle, from each respective collector, and for each event were averaged to get a representative concentration. The runoff volume from each watershed, during each event, was calculated by using the curve number method as suggested by USDA [36]. The nitrate load was calculated by multiplying the concentration during each event and the associated runoff volume.

An Index of the time spent by cattle in pastures was measured using wildlife GPS Collars [32] set to record cow location every 5-min. Two to three GPS collars were deployed in each pasture at 28-day intervals. After 28 days, the collars were removed for data download and battery recharging. The collars were on throughout the year except when cows were breeding or calving or when batteries were charging. The location data were downloaded using GPS 3000 Host [37], corrected, georeferenced, and processed using a Continuously Operating Reference Station (Athens, GA: 33.95237° N, 83.32563° W). The location data was projected in NAD 1983 Universal Transverse Mercator Zone 17N using ArcGIS 10.x for further analysis. Using the location dataset, cattle density (m⁻²) rasters were created for each measurement period for all collars using ArcGIS 10.x (Point Density Tool). The rasters were normalized for maximum possible location fixes 8064 (28 days \times 24 hours \times 60 min/5 min) for each 28-day period, because some collars did not collect all the possible location records due to technical errors. The two/three replicate collars from each pasture and for each measurement period were averaged. Those rasters were then multiplied by the total number of cows in each pasture. The rasters within each project year (May to April), were summed to get an annual cattle density raster for four project years (2015 to 2018). Total data collection days were not the same for all project years (May–April); thus, they were standardized for 365 days to get annual cattle density raster. The cattle density raster was multiplied with 5/60 to get hour spent by cattle.

2.7. Statistical Analysis

Data processing was done in Microsoft Excel and data analysis was done using R Statistical software [38]. Cattle locus data were processed and analyzed using ArcGIS 10.x (ESRI). The student's t-test was used to compare the treatments ($\alpha \le 0.05$ were considered significant). A non-parametric version of the t-test (Wilcoxon Rank Sum test) was used to compare runoff data as it was right-skewed. A linear regression model was used for establishing the relationship between runoff-nitrate vs. soil nitrate and cattle locus index.

3. Results

3.1. Weather Information during the Study Period

In 2016, the study area experienced an eight-month drought (April–November) which lowered the annual rainfall to 754 mm compared with the 100-year average of 1190 mm in Eatonton and 883 mm compared with the 100-year average of 1230 mm, in Watkinsville (Figure 2). The low precipitation resulted in an extreme negative annual water balance in the soil at both study sites. Typically, in this region, there is a negative water balance during summer months but an annual positive water balance. It was also hotter in 2016 as compared to the historical average which resulted in higher average annual maximum temperature and soil temperature at 20 cm depth. The combined effect of high soil and air temperature and low precipitation and annual water balance had discernible impacts on the pastures, which will be described later in the manuscript (Sections 3.1–3.4).

3.2. Active Carbon

After the random assignment of STR and CHD treatments to the pastures, the treatment groups were compared to assess any initial differences in POXC values using baseline (2015) measurements. During the baseline, there was no significant difference between STR and CHD pastures, at all soil depths (Figure 3). After treatment, STR pastures had significantly higher POXC as compared to CHD pastures, at all depths, in both years 2017 and 2018. When POXC was compared within treatment, there was a significant decrease in POXC in year 2017, at all depths, in both treatments. This might be attributed to a prolonged drought (8 months) in 2016 followed by a wet year in 2017 resulting in rapid mineralization. In the year 2018, the POXC values were statistically similar to the baseline values. These results indicate grazing system's ability to quickly recover from extreme weather events. The

STR pastures had greater POXC in 2018 (889 mg kg⁻¹) as compared to baseline (845 mg kg⁻¹), however, not statistically different. In 2018, the CHD pastures experienced a decrease in POXC (-50 mg kg⁻¹) as compared to baseline, but not significantly.



Figure 2. Meteorological conditions of **(A)** Eatonton, and **(B)** Watkinsville during the study period (2015–2018), showing average annual maximum temperature, average annual soil temperature at 20 cm depth, annual rainfall and annual water balance.



Figure 3. Comparison of active carbon (POXC) between treatments and within treatment across years, at three soil depths (0–5, 5–10, and 10–20 cm). The upper-case letters compare the year within treatments, whereas the lower-case letters compare treatments within respective years. Different letters suggest a significant difference between compared groups, whereas, NS indicates no statistical difference at $\alpha = 0.05$. STR = strategic-grazing pastures, CHD = continuously grazed with hay distribution.

3.3. Soil Respiration

As soil is a dynamic living system, soil respiration is one of the most important indicators of soil health. During the baseline, and the year 2017, the STR and CHD pastures were not significantly different (Figure 4). In 2018, the STR pastures had significantly higher soil respiration as compared to CHD. When compared within treatments, CHD pastures experienced a significant increase in soil respiration in 2017 and a decrease in 2018 which was significantly below the baseline respiration. The STR pastures also experienced an increase in 2017, however, it reverted to the baseline level in 2018. The higher respiration in 2017, in both treatments, might be attributed to a prolonged drought in 2016, followed by a wet year (2017) causing rapid mineralization of soil carbon [39]. Orchard and Cook [40] reported a rapid increase in soil respiration upon rewetting of dried soil. The soil respiration after

reaching an equilibrium phase reverted to a level that was below the initial level and they attributed this to substrate depletion during the rewetting phase. Overall STR pastures had more stable respiration throughout the drought and extremely wet weather suggesting that STR management could help stabilize pastoral systems under changing climates.



Figure 4. Comparison of soil respiration between treatments, and across years within treatments. The upper-case letters compare the years within treatments, whereas the lower-case letters compare treatments within years. Different letters suggest a significant difference between compared groups, whereas, NS indicates no statistical difference at $\alpha = 0.05$. STR = strategic-grazing pastures, CHD = continuously grazed with hay distribution.

3.4. Potentially mineralizable Nitrogen and Inorganic Nitrogen

PMN was compared between treatments for 3 years and no significant differences were detected between treatments in PMN in all years, at any soil depth, including the baseline (Figure 5). At 0–5 cm depth, the STR pastures experienced a significant reduction in PMN (-7.8 mg kg^{-1}) and the CHD pastures also experienced a reduction of (-9.5 mg kg^{-1}). At the 5–10 cm depth, STR experienced -3.5 mg kg^{-1} reduction and CHD experienced -2.9 mg kg^{-1} reduction. At 10–20 cm depth, no change was observed in both treatments.

Inorganic N was compared between treatments across years and no significant differences between treatments were detected (Figure 5). During the post-treatment years, there was a significant increase from baseline in IN at 0–5 cm soil depth in both treatments. The STR and CHD pastures experienced 24.7 mg kg⁻¹ and 24.4 mg kg⁻¹ increase, respectively, from baseline to 2018 at 0–5 cm soil depth. This result corroborates with our argument that the pool of PMN was efficiently mineralized to add to the inorganic N pool. The findings from another part of this study showed that the soil organic carbon pool was mineralized, and in the process, nitrogen was released to add to the inorganic nitrogen pool [41]. There was no difference from baseline to post-treatment years, at 5–10 cm and 10–20 cm soil depths, in both treatments, in terms of inorganic nitrogen.

To study the effect of exclusions on PMN and IN, soil samples inside the overseeded exclusions were compared across years (Figure 6). At all soil depths, there was a decrease in PMN from baseline to the year 2018, however, the reduction was significant at 0–5 cm (-10.2 mg kg^{-1}). In terms of IN, there was a significant increase, at 0–5 cm, from baseline to the year 2018 (28.6 mg kg⁻¹). This result highlights the potential of exclusion and overseeding of vulnerable areas in pasture in mineralizing the nitrogen to make it available to plants. The forage mix included legumes which could be responsible for the additional increase in IN during post-treatment years.



Figure 5. Comparison of (**A**) potentially mineralizable nitrogen, and (**B**) inorganic nitrogen between treatments and within treatment across years, at three soil depths (0–5, 5–10, and 10–20 cm). The upper case letters compare the years within treatments. NS indicates no statistical difference at $\alpha = 0.05$. STR = strategic-grazing pastures, CHD = continuously grazed with hay distribution.



Figure 6. Comparison of (**A**) potentially mineralizable nitrogen, and (**B**) inorganic nitrogen across years, at three soil depths (0–5, 5–10, and 10–20 cm) in soil samples inside the overseeded exclusions (in STR system). Different lower-case letters suggest a significant difference between compared groups at $\alpha = 0.05$.

3.5. Nitrate in Runoff

Runoff nitrate was compared between treatments before and after the treatment application. The baseline-CHD, baseline-STR, after-CHD had statistically similar runoff nitrate loss per event, whereas, the STR pastures after treatment had significantly lower (0.08 kg ha^{-1}) than all other groups (Figure 7).



Figure 7. Comparison of runoff nitrogen (kg ha⁻¹) between treatments, and before and after the treatment application. Different letters suggest a significant difference between compared groups at $\alpha = 0.05$. STR = strategic-grazing pastures, CHD = continuously grazed with hay distribution.

To further explore results, the runoff nitrate losses from the pastures were regressed against time spent by cattle within 50-m of the runoff collectors (Figure 8). Using a 50-m buffer created around each runoff collector using ArcGIS. 10.6., the time spent by cattle in that buffer was calculated (cattle locus index) by extracting the cattle index map (Section 2.6) for each year (2015–2019). For all runoff nitrate values for a particular year, the respective cattle locus index was used. Similarly, the soil nitrate values for each runoff collector were calculated by averaging the soil nitrate from all sampling locations from respective watershed on a yearly basis. The same soil nitrate value was used to regress with all runoff nitrate values from a particular runoff collector for a particular year. A larger slope of the regression line suggests that cattle activity near the runoff collectors affects the amount of runoff nitrate that is lost from the field. The slopes of the regression line for baseline-STR, baseline-CHD, and after-CHD were significant (Table 2) and not statistically different from each other. However, the regression line for after-STR had slope equal to zero suggesting that even with increased cattle activity near runoff collectors, no corresponding increase in runoff nitrate was observed. This illustrates the ability of over-seeding of AOI exclusions to protect and utilize nitrogen from animal manure.

Table 2. Regression equations showing the relationship between runoff nitrate vs. (i) cattle hour spent within 50-m of the runoff collector, and (ii) soil nitrate, before and after the treatment application.

Treatments	Runoff NO ₃ ⁻ vs. Cattle locus	Runoff NO ₃ ⁻ vs. soil NO ₃ ⁻
Baseline CHD	Runoff NO3 = $-0.03 + 0.21^* \times Cattle locus$	Runoff NO3 = $-0.039 + 0.015^* \times \text{Soil NO3}$
After CHD	Runoff NO3 = $-0.03 + 0.24^* \times Cattle locus$	Runoff NO3 = $-0.077 + 0.005^* \times \text{Soil NO3}$
Baseline STR	Runoff NO3 = $-0.09 + 0.27^* \times Cattle locus$	Runoff NO3 = $0.05 + 0.007^* \times Soil NO3$
After STR	Runoff NO3 = $0.05 - 0.00 \times Cattle locus$	$Runoff NO3 = 0.021 + 0.00 \times Soil NO3$

* indicates significant slope at $\alpha = 0.05$.

Similarly, the effect of soil nitrate on runoff nitrate loss was also assessed using linear regression (Figure 8B). The before-STR, before-CHD, and after-CHD had significant slopes, whereas after STR did

not have a significant slope (Table 2). In STR pastures, there was a significant reduction in the slope (from 0.007 to 0), which shows the ability of over-seeded AOI exclusions to protect nitrogen losses in runoff. Similarly, in CHD pastures, there was also a significant reduction in slope (from 0.015 to 0.005).



Figure 8. Relationship of runoff nitrate with (**A**) cattle locus and (**B**) soil nitrate at 0-5 cm soil depth, across treatments, before and after the treatment application. STR = strategic-grazing pastures, CHD = continuously grazed with hay distribution.

4. Discussion

The results are in agreement with our hypotheses that strategic grazing improves or at least maintains soil health in pastures and reduces nitrate losses via runoff. Usually, conventional beef pastures are continuously grazed marginalized lands and pasture equipages such as water, shade, mineral, and hay are stationary/permanent [42,43]. The grazing systems we proposed were a collection of several better grazing practices. A historical study [44] did not find any improvement from rotational grazing in cattle productivity. However, more recent studies assessing rotational grazing systems or better grazing management systems have found positive benefits in terms of forage and animal productivity and soil health parameters [43,45–48]. Improved water quality and soil health from cattle exclusion of vulnerable areas in pastures have also been reported in previous studies [6,14,49,50]. Lure management of cattle using pasture equipages have been used to distribute the cattle activity in pastures [14] and has the potential to distribute animal manure in the pasture while also protecting riparian areas from nutrient build-up.

The weather during this research likely had a significant impact on the results of this study. The year extremes of 2016 in terms of temperature and precipitation, as shown in Figure 2, resulted in an eight-month drought with unusually high air and soil temperatures and an extreme negative annual soil water balance. This was followed by a relatively wet year in 2017, which included hurricane "Irma."

The active fraction of soil carbon, measured as POXC, was a highly sensitive indicator of soil health in these grazing systems. There was a significant reduction in POXC in the year 2017 in both treatments, at all depths; however, STR pastures had a significantly greater POXC as compared to CHD pastures at all depths. In the year 2018, POXC in STR pastures was higher than the baseline (44.35, 9.29 and 18.36 mg kg⁻¹ at 0–5, 5–10, and 10–20 cm, respectively) though not statistically significant. In the STR pastures, we speculate that there was a downward movement of POXC beyond 20 cm due to improved forage and root growth, darker soil colors, and the presence of mycelial networks that were not there in 2015 (Figure 9). However, this information was not quantified because the sampling depth was only 20 cm. Another part of this study including more soil samples has shown a significant increase in POXC in the STR pastures at all three depths under consideration. As noted in the results, POXC in CHD pastures also recovered from extreme events but not to the extent that STR pastures recovered. This result highlights the resilience and ability of both systems to improve carbon sequestration by increasing the active carbon pool in soils at deeper depths thereby improving the

overall volume of soil (increased depth of activity) in which the rhizosphere of the grassland is active. The positive trend of POXC, at all soil depths, in the STR grazing system is promising and stirs the need for longer-term research with a periodic sampling below 20 cm.



Figure 9. A soil core (A) showing root and mycelial growth at 30 cm soil depth and (B) movement of carbon across the soil profile showing carbon breaking in the core.

Soils are dynamic living systems, and soil respiration is a crucial biological indicator of soil health [25]. Although difficult to measure, in situ soil respiration, measured using static chambers, was a sensitive indicator of management changes in pastures. The average soil respiration across years and grazing systems in our research was 1092 mg CO_2 m⁻² 24 h⁻¹, which is lower than the $2628 \text{ mg CO}_2 \text{ m}^{-2} 24 \text{ h}^{-1}$ reported by Chiavegato et al. [51] during July–August in grass-legume mixed pastures in northwest Michigan. However, the comparison of soil respiration with other studies is complicated because of the differences in climate, forage type, soil type, temperature, and moisture regimes. During the baseline and the year 2017, the CHD and STR systems had similar soil respiration. However, in 2018, the STR system had significantly more (235 mg CO_2 m⁻² 24 h⁻¹) soil respiration as compared to the CHD system. Both systems underwent a stress period during the drought of 2016 and a wet period of 2017 resulting in increased soil respiration and a reduction in POXC. Previous studies [39,40,52] have reported increased soil respiration in wet periods that followed drought. The CHD system experienced a significant increase in soil respiration in the year 2017 and a significant reduction in the year 2018. Whereas, the STR system also experienced an increase in soil respiration in the year 2017; however, it reverted to the baseline level. We have a reason to believe that the STR system performed well because it had low variation in soil respiration across years as compared to the CHD system.

The PMN and IN fraction of soil nitrogen are crucial for any agroecosystem because nitrogen remains one of the most critical and expensive agricultural inputs. These results (Figure 6) illustrate the ability of STR systems (exclusion and overseeding of vulnerable areas) to mineralize the PMN pool in areas that initially were more compacted from heavy animal traffic [53] and N was not as plant available to areas that are more productive with more readily available N for plant uptake. The increase in plant-available N might be attributed to improved root growth and ground cover in the overseeded exclusions. It has been reported [54] that most of the nitrogen in the excluded riparian areas can be lost via denitrification, thus overseeding of excluded areas critical for utilizing the nitrogen. The available nitrogen can be readily utilized by the overseeded forage helping to prolong annual grazing duration [55]. In another part of this study, Subedi et al. [53] noted an approximately 4% reduction in loss-on-ignition carbon during a three-year period. The mineralization of soil organic carbon releases nitrogen in the process which might also be attributed to the overall increase in inorganic nitrogen in both systems.

Nutrients in the runoff, especially nitrogen and phosphorus, are the leading cause of eutrophication and groundwater contamination [56]. Surface deposition of feces and urine and associated nitrogen in low-lying portions of pastures that have high cattle activity are prone to runoff losses [15,57]. However even with the greater concentration of inorganic N in the 0–5 cm soil layer of the STR system, runoff-nitrate was significantly reduced (from 0.17 to 0.08 kg NO_3^- ha⁻¹) which can be attributed to controlled cattle activity in low-lying vulnerable areas, improved ground cover in the AOIs, deeper root growth, and plant utilization of the available nitrogen. The CHD pastures had a small reduction in runoff nitrate after treatment, but it was not statistically significant. Before treatment, per unit increase in time spent by cattle, within 50-m the runoff collectors, would result in 0.27 kg ha⁻¹ increase in runoff nitrate. After the treatment, in the STR system, there was no effect of cattle locus on runoff nitrate which shows the efficacy of controlled utilization (overseeded and limited grazing) of low-lying areas vulnerable to erosion in reducing runoff-nitrate losses. Previous studies [57–59] have reported the effectiveness of riparian buffer in agricultural watersheds to protect and utilize nitrate. It should be noted that exclusions were not placed adjacent to streams, but they were in concentrated flowpaths adjacent to riparian areas which had little vegetation within the concentrated flowpaths. The exclusions in this study not only protected the nitrogen [60] from leaving the field but also provided extra forage [55] where little to no vegetation was present during baseline. During the baseline, the positive relationship between soil-nitrate and runoff-nitrate indicated that regions with greater soil-nitrate tend to lose more nitrate in runoff. After the treatment, in STR pastures, that relationship was not evident, again illustrating the effectiveness of flash grazing of overseeded exclusions in vulnerable areas for improving surface water quality.

Strategic grazing has several evident advantages in terms of soil health and surface water quality over the existing conventional grazing system and its slightly improved version, the CHD system. Moreover, the ability of the STR grazing management system to recuperate from extreme weather events such as droughts and hurricanes, in terms of soil health, establishes the potential of this system to improve the overall sustainability of managed pastoral systems in the changing climatic scenario.

5. Conclusions

Promising positive changes in ecosystem services came from the strategic grazing system including an increase in active carbon, consistent respiration rate, and cleaner runoff water with a reduction in nitrate in runoff water. The comparatively higher active carbon in the strategically grazed pastures compared to continuously grazed with rolling out of hay showed strategically grazed system's ability to sequester carbon, which can be used as a tool in our fight with changing climate and extreme weather events. The reduced nitrate in runoff has twofold advantages: (i) dollars saved by reduced loss of expensive nitrogen and (ii) healthy environment from cleaner streams. These results indicate that strategic grazing can improve the overall sustainability of the beef-production system in Southeastern USA and make it more resistant to extreme weather events. Further research is required at farmers' fields to assess the actual applicability. A long-term study is recommended to evaluate the beneficial effect of STR and CHD grazing systems to improve carbon sequestration, soil health, and nitrogen cycling.

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