ASSESSING STRATEGIC GRAZING EFFECTS ON IMPROVING THE SUSTAINABILITY OF PASTURES BY MEASURING KINETIC SOIL HEALTH INDICATORS AND RUNOFF WATER QUALITY

by

SUBASH DAHAL

(Under the Direction of Dorcas H. Franklin)

ABSTRACT

Soils are central to sustainable management of any agroecosystem including grazing lands, thus development of innovative grazing management systems and scientific studies focused on improving soil health is of prime importance. A study was conducted in eight beef-pastures of Southern Piedmont, Georgia, USA, from 2015 to 2018, to assess the effects of existing grazing system (continuous/conventional), a slightly improved continuous system (continuous grazing with hay distribution: CHD), and a collection of better grazing practices (strategic grazing: STR) on soil health, spatial distribution of labile carbon and nitrogen, and runoff water quality. In 2015, a baseline study was conducted to assess soil health indicators (in-situ soil respiration, labile carbon, potentially mineralizable nitrogen, labile nitrogen), spatial distribution of labile carbon and nitrogen, and runoff water quality in historically (>10 years) continuously grazed pastures. In 2016, STR grazing system was implemented in four pasture and CHD grazing system was implemented in four pastures. We found that management factors (such as location of hay, water, and shade), cattle locus index, and landscape factors significantly affected the spatial distribution of soil inorganic nitrogen (was highly uneven) in continuously grazed pastures. Post-treatment, STR pastures experienced increased labile carbon down to 20-cm soil depth, and less yearly variation of soil respiration, as compared to CHD pastures. The exclusion and over-seeding of nutrient-rich vulnerable areas in STR pastures facilitated the mineralization of potentially mineralizable nitrogen (PMN) and soil organic carbon to make the nitrogen readily available for plant uptake. The STR system also significantly reduced runoff nitrate losses as compared CHD system, which was mainly attributed to cattle exclusion and continuous vegetative cover of vulnerable low-lying areas of pasture. The STR system significantly improved the concentration and spatial distribution of labile carbon (POXC) down to 20-cm soil depth. Moreover, the STR system also improved the spatial distribution of inorganic nitrogen. We conclude that the STR grazing system has shown potential to improve the sustainability of grazing systems by enhancing soil health and water quality, and we recommend longer-term studies to fully assess its potential.

INDEX WORDS: Strategic Grazing, Continuous Grazing, Sustainable Grazing, Lure Management, Soil Health, Spatial Distribution, Water Quality

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by

SUBASH DAHAL

BS, Tribhuvan University, 2010

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SUBASH DAHAL

Major Professor:	Dorcas H. Franklin
Committee:	Miguel L. Cabrera
	Dennis W. Hancock
	Deepak R. Mishra
	Cheolwoo Park

Electronic Version Approved:

Ron Walcott
Interim Dean of the Graduate School
The University of Georgia
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DEDICATION

I would like to dedicate this dissertation to my parents, my wife, and the happiness of my life, “Khusi”.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Grazing Systems

Beef cattle farming is an important contributor to US agricultural production and economy. In 2018, beef-cattle production represented eighteen percent of the total cash receipts from agriculture commodities (USDA-ERS, 2019). Globally, not only is the US the largest grain-fed cattle industry, but also the largest consumer of beef. In 2018, 33.7 million head of cattle were harvested, and the total value of beef production was 60.8 billion dollars (USDA-ERS, 2019). To support the massive beef cattle industry there are more than 316 million hectares of grazing land in the US of which 3.36 million hectares are in Southeastern US (USDA, 2011). Of all the private lands in the US, 47% are grazed lands (USDA, NRCS, 2003), and pasture-based beef farming is an important driver of the economy in the Southern Plains and Southeastern region of US (McBride and Kenneth, 2011).

Typically, in the southeastern US, pastures are established on lands that are not suitable for crop production which suggests that pastures might have steep slopes and likely have been eroded (Drouillard, 2018). Nutrient losses (N, P, and C) from grazing systems is a common problem, and it occurs through various avenues. For example, the fertilizers added in the pasture to increase the forage yield might get lost through leaching, runoff, and volatilization. N and P losses are caused by several factors: timing, type and rate of fertilizer used, field slope and aspect, rainfall intensity and vegetative cover (Brennan et al. 2012). Nutrient losses and/or retention are also attributed to management activities and cattle movement in the farm (Franklin et al. 2009; Byers et al. 2005).

The conventional cattle production in the southeastern US generally follows continuous grazing or lax rotational grazing (Butler et al. 2010). In continuous grazing, the animals are allowed to access all parts of the pasture for multiple years, without any control over the grazing time and intensity, which leads to reduced forage availability and inefficient land utilization. Bellows (2003) has provided a comprehensive guide on how uncontrolled grazing affects overall pasture health including riparian areas. We define lax-rotational grazing as a system where pastures are divided into two to three sub-pastures, and the cattle are rotated every few months. This method has a slight advantage over the continuous grazing because the rotation of cattle allows the forage to regenerate. A typical pasture-based beef farm in Georgia managed in a conventional manner may cause negative environmental impacts, soil deterioration and nutrient runoff leading to poor sustainability of the farm systems. Osmond et al. (2007) suggested that many farmers in this region feed hay for five or more months per year, which results in poor distribution of nutrients in the pasture, and overall soil health deteriorates.

The four natural laws of grazing management (a) shorter shoot means shorter root, (b) nature does not like bare spots, (c) bare soils have lower moisture-holding capacity, and (d) if given a chance nature recovers the damage, should be of prime consideration for developing grazing systems (USDA-NRCS, 2016). Moreover, stocking rate, rotational grazing, rate of forage utilization, and forage recovery period are some of the easily understood and commonly used indicators of sustainable grazing. Different management practices are being developed for better production and sustainability in the pasture-based beef farms. Best management practices (BMPs) such as; controlled rotational grazing, stream exclusion, and buffer strips have shown promising results in enhancing soil and pasture health (Undersander et al. 1993; Bellows, 2003). The USDA has adopted rotational grazing as one of the instruments for improving grazing systems. Lure management/rotational grazing of animals using pasture equipages seeks to utilize the pasture resources by mimicking activities of wildlife (Bellows, 2003). Rotational grazing systems are tested and found to have more stable forage production, greater yield potential, better quality forage, lower weeds, lower soil erosion, and more uniform soil fertility (Undersander et al. 1993). Better grazing practices that help conserve water and redistributes and recycles the nutrients could benefit both producers and the environment to achieve sustainability (La Maitre et al. 2007). However, only moderate successes have been reported in efficiently managing the pastures for improved productivity and sustainability of the farm. The primary concern with the BMPs is the higher time required for management. We believe that if we could reduce the amount of time required for management activities, farmers will feel more comfortable adopting these methods (Undersander et al. 1993). The proposed strategic grazing involves lure management techniques (i.e. use of movable shades, appropriate placement of waterers, minerals, and hay bales) to rotate cattle, which reduces the time required for moving the animals in a rotational grazing system. Farmers are more likely to adopt the management practices which are both beneficial and input-efficient.

In a study by Porath et al (2002) in Oregon, portable water troughs and mineral feeders were used for lure management of cattle. They reported that there was a time of the day effect on cattle preference to the riparian stream and provided water troughs. Cattle preferred riparian stream during the morning, whereas water troughs were preferred in the afternoon. Also, forty-five percent of total water use was contributed by the alternate water troughs. This study has an important implication for reducing fencing requirements in the pasture for protecting the stream water quality. In addition to better water quality, calf-weight-gain and forage productivity were improved in pastures with alternative water troughs and mineral feeders. In tall fescue pastures of Georgia, Franklin et al. (2009) reported that provision of alternative water troughs as a means of lure management significantly reduced time spent by cattle in riparian streams. Only when the temperature-humidity index was extremely high did more cattle spend more time near the streams and the impact on water quality was minimal as those periods were short.

Animal exclusion in the riparian areas of pastures is widely practiced and recommended by USDA to protect soil and water quality in pastures and other agroecosystems (Clary and Webster. 1989). In a recent study by Danvir et al. (2018) in New Mexico pastures, significant improvement in vegetation cover was observed by excluding animals from riparian areas. The above-mentioned study was conducted in large ranches, hence grazing acreage loss by excluding riparian areas might not be significant, however, the loss could be significant for smallholder farmers. To address acreage loss, strategic grazing (using better grazing practices such as flash or mob grazing) should be practiced in excluded riparian areas and/or vulnerable areas of pastures. In another study by Larson et al. 2016, it was reported that exclusion of riparian areas significantly improved the stream water quality. Exclusion of low-lying vulnerable areas has dual advantages; (a) continuous soil cover and root growth which facilitates water infiltration and reduces erosion, and (b) the plants uptake the nutrients deposited in the riparian area and provide extra forage to the animals. As the excluded low-lying vulnerable areas can be sinks for nutrients eroded from upper parts of the pasture that can be rich in nutrients (via cattle manure deposition or runoff deposition), there is an opportunity for either large pulse of these materials to the streams during extreme events or for utilization of these nutrients through the use of exclusions over-seed with grass-legume mixes to produce protein-rich forage. Improvement in live weight of calves attributed to high protein leguminous forage has been reported (Baxter et. al 2017). In a study by Bridges et al. 2019, the grass-legume mix was over-seeded for ten years in Bahia grass pastures where they reported an increase in soil organic matter (by 6%) and microbial diversity and abundance.

Short-term flash grazing/mob grazing is an intensive grazing management technique which involves grazing a small area of the paddock for a brief period, typically ranging from few hours to a day (Lemus, 2011; Haan and Bartlet, 2010). Generally, this practice is done in small exclusion areas that have been over-seeded with a diverse forage mix. Paine (1999) reported that flash grazing of exclusion areas improved nutrition distribution, weed control, soil quality and forage biomass in the exclusion areas. However, this practice is relatively new, and further research is required to quantify potential benefits and drawbacks. In our study exclusion areas are generally located at lower elevation areas where nutrients are deposited through sedimentation and deposition from cattle camping. In some cattle camping areas, soil can get compacted due to cattle activities, deposited nutrients remain on the soil surface resulting in soil low in nutrients. In areas of concentrated flow nutrients can get swept away by flow of water (Lyons et al. 2000). We hypothesized that areas vulnerable to nutrient loss (either in the soil or on the surface of the soil) overseeded with mixed forages and grazed for a brief periods, will improve water quality and some nutrients will be redistributed via animal manures (as the animals later go to other parts of pasture and defecate and urinate).

Soil Health

USDA-NRCS, 2014 defines soil health as the “continuous capacity of soil to function as the vital living ecosystem that sustains plants, animals and humans.” Soil is a complex, dynamic and living ecosystem and is the building block of all agroecosystems. Soil provides several ecosystem services such as; carbon cycling, soil nutrients storage, water filtration, germplasm storage, and support all terrestrial life (Lal, 2016). Hence, improved soil health in pastures and rangeland is crucial for addressing agricultural and environmental challenges of the 21st century (Shafer et al. 2016). In recent times, the growing awareness of scientists and producers on soil health and its implication on agricultural sustainability (Moebius-Clune et al. 2016; Shafer et al. 2016) has boosted research works on creating management strategies for improving soil health (Acharya et al. 2019; Plastina et al. 2018) in various agroecosystems.

Derner et al. (2018) suggested using the four-principle approach of soil health designed for croplands, which consists of (a) plant diversity, (b) reduced soil disturbance, (c) prolonged crop growth period, and (d) soil cover, in grazing systems. However, they also mentioned that the inherent spatiotemporal variations, complexity of ecological relationships, and limited scientific knowledge about grazing lands would challenge the applicability. Compared to the vast number of ecosystem services grazing lands provide (Bretagnolle et al. 2011), very little effort has been made towards development of soil health-oriented grazing management strategies. There is no single soil health indicator that can describe the overall state of soil health or quality (Roper et al. 2017); thus, several indicators are used to address the dynamic and complex nature of soil ecosystem. Moreover, it has been reported that soil health is significantly affected by climate and management factors (Bhowmik et al. 2016; Byrnes, et al. 2018; Bhandari et al. 2018, Ghimire et al. 2019), thus a deeper understanding of interrelationship between soil health indicators and management factors such as fertilizer source and grazing system, is highly important (USDA, 2019).

Soil texture (Damasa et al. 2015) is an important driver of other soil health indicators, whereas, bulk density (Paul Obade and Lal, 2016) and aggregate stability (Idowu et al. 2008) are most commonly used physical soil health indicators. Chemical indicators of soil health are the oldest and most studied indicators and remain crucial for crop and forage production. Nitrogen, Phosphorus, Potassium, Calcium, Magnesium, Iron, Manganese, Zinc, and PH are recommended chemical indicators (Moebius-Clune et al. 2016), however other trace elements have been used to assess soil health. Biological indicators are often most complex, dynamic, and difficult to measure yet reliable and sensitive indicators of soil health. Active carbon (Weil et al. 2003), soil microbial biomass (Sangha et al. 2005), soil respiration (Haney et al. 2008), potentially mineralizable nitrogen (Franzluebbers et al. 1994) are widely used and recommended biological indicators of soil health.

Moebius-Clune et al. (2016) has provided a comprehensive framework for assessing soil health in various agroecosystems. They have recommended using chemical and physical soil health indicators with a special focus on biological indicators their interactions for assessing soil health. They suggested that a healthy soil, in general, should have (a) good soil tilth, (b) sufficient depth, (c) good water storage and good drainage, (d) sufficient supply but no excess nutrients, (e) small population of plant pathogen, insects and pests, (f) larger population of beneficial insects, (g) low weed pressure, (h) free of chemicals and toxins, (i) resistant to degradation, (j) resilient when unfavorable conditions occur. In order to develop healthy soil, we also need to understand the constraints which are (a) soil compaction, (b) poor aggregation, (c) weed pressure, (d) high pathogen pressure, (e) low water and nutrient retention, (f) salinity and sodicity, and (g) heavy metal contamination. They have recommended four physical, four biological, and routine chemical analysis for soil health assessment. This dissertation is a part of a bigger collaborative research and is focused on the biological soil health indicators (a) soil respiration, (b) active carbon, (c) potentially mineralizable nitrogen, and (d) inorganic nitrogen.

Soil Carbon

Soil carbon is the fraction of soil which supports the living components of soil, and a healthy soil can have as high as 5 Mg ha^{-1} live biomass (Lal, 2016). Soil carbon can be classified in two parts; (a) the inorganic carbon and (b) the organic carbon (45-60% of soil organic matter) which represents the remains of plant and animals at various stages of decomposition and the microbial biomass and its byproducts (Lal, 2016). Soil organic carbon is a complex, dynamic and reactive component of the soil ecosystem and provides a basis for several soil processes. Soils store two to three times more carbon than the atmosphere which makes them excellent sinks of carbon (Minasny et al. 2017). It has been estimated that 700-3000 gigatons of soil organic carbon can be stored in the terrestrial earth, whereas atmosphere only contains 720 gigatons of carbon (Bouwman, 1990). In the United States, the soil organic carbon stock is 54.5 gigatons in the 0-30 cm soil depth, of which 30.0 gigatons is in the agricultural lands (Minasny et al. 2017). Grasslands are largest reservoirs of terrestrial carbon. It has been estimated that globally, there is more than 343 peta-grams carbon within 1m of the grassland soil (Sombroek et al. 1993).

It has been reported that a 1^0 C rise in global temperature would drive a loss of 203 peta-grams carbon from upper soil surface, within 35 years, provided there is no change in other variables (Crowther et al. 2016). There are several documented benefits of increased soil organic carbon, such as; improved soil aggregation, better water retention, increased nutrient holding capacity, and reduced compaction (Wander, 2004). The rate and amount of changes in soil carbon stock is highly variable and difficult to generalize due to inherent differences in soil type, climate, and site-specific management. These challenges drive the need for research related to carbon sequestration in grazing systems because they are one of the most important avenues for carbon sequestration (Sanderman et al. 2017).

It is important to quantify indicators of soil organic carbon for efficient quantification of change caused by various management activities. Two divisions of soil organic carbon; (a) the recalcitrant carbon and the (b) labile carbon (Weil et al. 2003) are important to understand. One of the most commonly used, also highly debated, method for estimating soil carbon is the loss-on-ignition carbon method. This method constitutes combusting the organic carbon in soil at very high temperatures approximately 500^{o}C for 8-12 hours. However, De Vos et al. (2005) suggested that the loss-on-ignition method removes the hygroscopic moisture, release carbon dioxide from carbonates, remove water from hydroxyl groups, and destroy elemental carbon, which leads to an overestimation soil organic carbon. Also, the loss-on-ignition carbon is not considered suitable for measuring short term changes in organic carbon. To create a measurable change in loss-on-ignition carbon induced by changes in management could take decades (Grandy and Robertson, 2006). Jensen et al. (2018) conducted research to determine relationships and correction factors between loss-on-ignition carbon and soil organic matter calculated from original Walkley-Black method (Walkley and Black, 1934). Hoogsteen et al. (2015) conducted research to determine relationships between LOI and SOC while (Hendricks et al. (2019) examined the relationships between LOI, POXC and SOC. Both LOI and POXC were found to be fractions of SOC. To address these concerns, scientists have suggested also measuring the labile fractions of soil carbon. The labile fraction has a short turnover time, and it is sensitive to management changes and external efforts as compared to the recalcitrant fraction (Grandy and Robertson, 2006). Past literature (Weil et al. 2003; Cozzolino and Moron, 2006) have shown that using the labile fraction can accurately capture changes in soil carbon in just a few years. Particulate organic matter (less than 2 mm and greater than 0.53 mm in size) while often a relatively small proportion of the soil is the most dynamic fraction of labile carbon

(Cambardella and Elliot, 1992). Particulate organic matter is a USDA recommended indicator of soil health and is used to detect changes in soil organic matter over short periods of time (3-5 years) (Gregorich et al. 2006; Baldock et al. 2018).

Permanganate oxidizable carbon (POXC) is another commonly used and widely accepted measurement of soil carbon. Weil et al. 2003 developed this rapid and inexpensive method to measure soil carbon which is applicable to both field and laboratory settings. It makes use of 0.2 M potassium permanganate ($KMnO_4$) which is used to oxidize soil carbon for 10 min, and the absorbance of the soil solution is measured at 550 nm. The absorbance value of soil solution is then compared to blank $KMnO_4$ solution to calculate soil carbon. To assess the usefulness of POXC method a meta-analysis was conducted (Culman et al. 2012) where they reported a wide range of R^2 (0.01-0.95) between POXC and soil organic carbon. They attributed this variation to different fields, soil type, climatic variation and management practices. Overall, they concluded that POXC is a reliable and sensitive indicator of soil carbon and soil health. In a study by Weil et al. (2003), which compared differences between two cropping systems in terms of soil carbon, no significant difference was observed in the soil organic carbon. However, in terms of POXC, a significant difference was observed between cropping systems; this signifies the use of POXC method to assess changes in carbon in various agroecosystems.

Soil Nitrogen

Nitrogen is one of the most important elements in agriculture production as it affects the yield as well as the quality of crops (Parfitt et al. 2005). Nitrogen mineralization is considered the heart of nitrogen cycle which controls the ability of soil to supply nitrogen for plant growth and development (Aber and Melillo, 2001). It is a multi-step enzymatic process by which organic nitrogen is converted to ammonium form by microbes. In simple terms, it is a process by which microorganisms decompose the organic nitrogen to inorganic form which is readily available to the plants for uptake. The plant available nitrogen fraction is also crucial for agricultural production and is routinely assessed by extracting the soil with a 1 M or 2 M KCl solution and colorimetric or spectrophotometric determination of nitrate and ammonium. Nitrogen mineralization/potentially mineralizable nitrogen could be a useful tool for assessing available nitrogen (Piatek and Allen, 2000). Nitrogen mineralization and plant available nitrogen are good indicators of soil fertility (Vitousek and Matson, 1985), and soil health (Moebius-Clune et al. 2016), and the grazing management practices heavily affect the dynamics of soil nitrogen (Franzluebbers, 2005).

Stanford and Smith (1972) incubated soils at 35^{o}C over a 30-week period to determine rate of nitrogen mineralization in several types of soils. They reported a linear relationship of cumulative net nitrogen mineralization with square root of time. They concluded that 12 weeks was the time required for one half of nitrogen to be mineralized. This method provides accurate estimate of nitrogen mineralization rate; however, it is a lengthy and resource consuming process. By the time results come out, it is too late to provide a timely nitrogen fertilizer recommendation.

Efforts have been made to develop rapid methods for determining nitrogen mineralization. Gianello and Bremner (1986) used 2 M KCl solution at 100^{o}C for 4 hours to extract the ammonium in soil and suggested using it as a proxy for potentially mineralizable nitrogen. Campbell et al. 1997 evaluated the use of hot KCl extraction method for predicting mineralizable nitrogen in soils of Saskatchewan, Canada. They found a good prediction with a R^2 of 0.78. In another study by Picone et al. 2002, in Georgia, USA, a R^2 of 0.79 was reported between cumulative nitrogen mineralization determined from hot KCl extraction method and the 24-day incubation method.

Haney et al. (2001) suggested that nitrogen mineralization is crucial for effective nitrogen fertilizer recommendation and introduced a rapid method for estimating nitrogen mineralization in soil. The rapid method was based on evolution of carbon dioxide evolved in 24 hours under optimum laboratory condition following rewetting of dried soil. They reported a high correlation of nitrogen mineralization and the carbon dioxide evolved (R^2=0.78) and suggested this rapid method as a proxy of nitrogen mineralization.

A study was conducted by Cabrera (1993) to study the effect of drying and rewetting of soils in nitrogen mineralization. He used replicate soil samples; the first group was dried and rewetted before incubation, and the second group was air dried (the standard procedure). He incubated the soils at 30^{o}C for 20 days and measured nitrogen mineralization periodically. He reported that the dried soils required zero order kinetics to model the cumulative nitrogen mineralization, whereas, the dried and rewetted soil required a combination of two models (one following zero order kinetics, and another following first order kinetics). Also, the first order nitrogen mineralization model superimposed on the zero order one. He highlighted the importance of wetting of soil (which adds to the initial substrate and increases the background mineralization rate) in modeling nitrogen mineralization.

Nitrogen mineralization is well studied in various cropping systems however, there are limited studies on grazing systems effect on nitrogen mineralization. Contrasting results have been reported about how grazing effects nitrogen mineralization (Liu et al. 2011), which might be due to variability in stocking density, management systems, forage type, soil, and climate.

Shan et al. 2011 conducted a 5-year study to assess the effect of seasonality on nitrogen mineralization in sheep grazed with two stocking densities; very low (1.5-4.5 sheep ha^{-1}) and very high (6-9 sheep ha^{-1}). During early grazing season, nitrogen mineralization was increased by 107% and 128% in very low grazed and heavily grazed plots, respectively. During peak grazing season the heavy grazing plots had higher nitrogen mineralization (108%) as compared to light grazing plots (71%). They also found net nitrogen was reduced by 181% (low) and 147% (high) when pastures where covered in snow and sheep were not grazing the pastures. They concluded that heavily grazed pastures hasten nitrogen mineralization.

In a study by Biondini et al. 1998 in a mixed-grass prairie of North Dakota, effect of grazing intensity (no-grazing, moderate grazing, and heavy grazing) on nitrogen mineralization was studied over 8 years. They reported that heavily grazed pastures had significantly low nitrogen mineralization rendering the system less sustainable as compared to moderate and no-grazing treatments. Augustine and McNaughton, (2006) reported that grazing increased inorganic pool at onset of growing season, but reduced nitrogen mineralization in early growing season.

Spatial Distribution of Soil Carbon and Nitrogen

While much of the past research is focused on developing and comparing grazing management systems in terms of concentration of soil nutrients and soil health indicators, the spatial distribution is equally important but overlooked dimension of grazing system research. Spatial distribution is one of the most important considerations in grazing systems for optimizing land use and maximizing farm profitability. Study of spatial distribution of soil carbon in grazing systems are limited due to large farm size, and expensive sampling and analysis. Schuman et al. (1999) utilized a 50-m transect sampling scheme to study the spatial distribution of soil carbon in non-grazed pastures, high stocked and low stocked cattle pastures. After 12 year of treatment implementation, there was no change in the total soil carbon and nitrogen, however the distribution of both carbon and nitrogen changed significantly. They reported 6000-9000 kg ha^{-1} higher soil carbon and 450-700 kg ha^{-1} higher soil nitrogen in the grazed treatments as compared to excluded treatments. They attributed this increase to redistribution and recycling of carbon and nitrogen due to grazing activity.

A study was conducted by Sigua and Coleman (2010) to assess the spatial distribution of soil carbon in managed beef pastures of subtropical Florida. They studied the effect of slope aspect and slope position on spatial distribution of soil organic carbon at 0-20 cm and 20-40 cm soil depths. They reported a significant interaction of soil aspect and soil position on soil organic carbon, and highest soil organic carbon was observed at top (8.4 g kg^{-1}) and middle slope (7.8 g kg^{-1}). They found that south facing soils had more concentration of SOC as compared to north facing soils. They attributed these differences to microclimatic variations, clay content, and cattle preference to south facing soils.

In another study by Sanderson et al. 2010 in beef farms in Maryland, USA, livestock concentration areas and its effect on spatial distribution of organic matter and other soil nutrients were studied. They utilized 100 m transects and sampled at 0-5 cm and 0-15 cm soil depths. They found that feeding areas were the largest cattle congregation areas and accounted for 48% of total congregation areas. The congregation areas had significantly greater concentrations of soil organic matter (by 2%) and phosphorus (by 36%) as compared to the rest of the pasture. Spatial distribution of soil carbon and nitrogen pool was assessed by Franzluebbers et al. 2000 in beef pastures of Georgia, USA. They collected soil samples in transects at varying distances from shade or water sources at 0-2.5 cm, 2.5-7.5 cm, and 7.5-15 cm soil depths. They reported that a zone within 30-m radius around shade or watering station becomes rich in carbon and nitrogen at shallow depths (0-2.5 cm). They reported that particulate organic matter, at 0-2.5 cm depth, within 10-m of shade or water was 26% greater than the rest of the pasture. At a deeper depth (30 cm) and within 10-m of water or shade structure, the soil organic carbon was 12% greater than the areas farther in the pastures. There was a gradual decrease in inorganic nitrogen with increasing distances from shade or water structures. They attributed this nutrient enrichment to the high frequency of cattle defecation and urination around the pasture equipages.

Soil Respiration

Soil respiration reflects the degree of microbial activity in soil and its tendency of soil to lose carbon in the form of CO_2 (Kennedy and Papendick, 1995). Soil respiration is the major pathway for cycling carbon from lithosphere to the atmosphere. It has been reported that approximately 80% of the photosynthate is respired back the atmosphere (Law et al. 2002).

Soil respiration depends on various soil properties, climatic factors, and management factors. Generally, lower respiration indicates stable soil however it might not always be the case (Kennedy and Papendick, 1995). Also, high respiration might not always indicate better soil health; it might be indicative of unstable system and/or recent disturbance or stress (USDA-NRCS). There is a reduction in photosynthesis during stress conditions such as droughts and freezing, which leads to utilization of stored non-structural carbohydrates causing root degeneration followed by a reduction of soil respiration (Hogberg et al. 2001). Role of management activities is important for determining soil respiration. Higher respiration suggests higher microbial diversity and/or activity and is a good indicator of soil health (Arias et al. 2005).

Best estimate of soil respiration could be achieved by measuring it in actual field condition. In-situ alkali absorption (Gupta and Singh, 1977; Cropper et al. 1985) is an old but accepted method of measuring soil respiration. It is an inexpensive method which can be employed a large spatial area because it needs less instrumentation. A sodium hydroxide solution is put inside a static poly vinyl chloride chamber driven in the soil at field condition. After certain hours (typically 24 hours), the solution is brought back to lab and titrated with hydrochloric acid to calculate amount of CO_2 absorbed.

Grazing is reported to have significant impact in soil respiration. A study was conducted by Bremer et al. (1998) in tallgrass prairie in Kansas to assess the effects of clipping and grazing on soil respiration. They found that annual soil respiration was lowered by 17.5% due to clipping or grazing, as compared to non-grazed plots. They also reported that daily soil respiration was 20-37% less in early grazed pastures as compared to non-grazed pastures. Their results highlight the potential of controlled or strategic grazing to maintain a moderate soil respiration. USDA (2019b) recommends 1000-2000 mg CO_2-C kg^{-1} soil $week^{-1}$ for ideal plant growth. It has been reported that soil microbial biomass reduced in response to defoliation of grasses, and it might be attributed to root death following defoliation (Garcia and Rice, 1994). In another study by Allard et al. 2007, soil respiration was compared between continuously grazed low and high stocking density management. They found that intensive grazing will gradually reduce the net carbon storage potential due to increased respiration. After three years in the study, the intensive system which mimics our continuous grazing had higher system respiration, whereas the extensive grazing which mimics our strategic grazing.

A study was conducted in grass meadows of Tibetan Plateau (Chen et al. 2016) to study the effect of grazing exclusion in soil respiration. They reported an increase in aboveground biomass and soil moisture but a decrease in soil respiration (by 23%, annually) and microbial biomass. They attributed reduced soil temperature, reduced microbial biomass, and increased soil moisture as probable causes of reduction in soil respiration. However, the temperature sensitivity was higher for the exclusions as compared to the grazed regions. Thus, they suggested that grazing exclusions and proper management might reduce carbon loss in short term, but as the temperature increases due to global warming, the exclusions might lead to increased carbon emissions. In another study by Nie et al. (2019), in temperate steppe of China, three grazing systems were compared in terms of in-situ soil respiration and temperature sensitivity (a) rest-rotation grazing, (b) grazing exclusion, and (c) continuous grazing. They reported that soil respiration was highest in rest-rotation grazing (1.26 mmol m^{-2} s^{-1}) followed by exclusion grazing (0.98 mmol m^{-2} s^{-1}), and continuous grazing (0.94 mmol m^{-2} s^{-1}). They attributed the difference in soil respiration to differences in soil moisture, soil temperature, potential substrate, and soil microbial activity. They recommended rest-rotation grazing as a more sustainable grazing system. Garcia et al. 2011 compared continuous grazing and rotational grazing with a control with no grazing. They reported a higher soil respiration in rotational pastures (53 mg CO_2 per 100 g soil) as compared to control (45 mg CO_2 per 100 g soil) and continuous grazing (46 mg CO_2 per 100 g soil). They attributed higher accumulation of animal excreta in the rotational paddocks as the reason for higher soil respiration. Franzluebbers et al. 2019 conducted a study in Oklahoma pastures to compare continuous and rotational grazing. They reported a significantly higher basal-soil respiration at 12-20 cm soil depth in one of the years, however this result was not consistent across years.

Cattle Locus and GPS Collars

In an extensive review of GPS collars use in studying animal behavior, Bailey et al. (2018) suggested that in the past two decades, tremendous advances have been made in use of GPS collars for studying livestock behavior. Reduced cost of GPS collars has provided scientists opportunities to study cattle behavior at greater temporal and spatial resolution with many replications. Successful use of GPS collars in cattle to monitor their movements in the managed pastures has been reported (Franklin et al. 2009; Byers et al. 2005).

Bailey et al. 1996, in a synthesis paper addressing mechanisms affecting grazing patterns in grazing animals, reported that grazing behavior and spatial distribution of animals depend on various biotic and abiotic factors. Abiotic factors include management factors such as distances to water, slope, aspect, weather etc., whereas, biotic factors include forage quality and quantity, animal behavior etc.

Franklin et al. (2009) used GPS 2200LR livestock GPS collars to measure cattle location every 5-min for several 17-day periods. The cattle location data was used to assess the effect of alternative water troughs on cattle activity in riparian areas. They reported that during times of low temperature humidity index, time spent by cattle in the riparian areas was significantly reduced (by 63%). Turner et al. (2000) used GPS2000 wildlife collars (Lotek Engineering) to (a) assess the accuracy of location measurement in beef cattle and (b) discuss livestock behavior in tall fescue pastures. They reported that increasing the frequency of location measurement will increase error on location measurement (by 7%), hence recommended frequent measurement of location (5 min). They also mentioned the importance of using replicate collars because of high animal variability (can introduce up to 70% more error).

Wesley et al. 2012 used GPS 2000 and 3300 (Lotek Engineering) for studying animal behavior in beef pastures in New Mexico. They separated cows in fast eaters and slow-eaters group and assessed cattle behavior. The fast-eater cows spent less time at watering stations and covered larger area in the pastures, and had greater weight as compared to slow-eater groups. Grazing behavior and spatial distribution of cattle was studied by Perez et al. 2017 in Mexico pastures using GPS collars. They found that GPS collars were effective in studying the spatial and temporal pattern of cattle activity, and recommended GPS collars for real time herd management.

Runoff Water Quality

Nutrients in runoff, especially nitrogen and phosphorus, are the leading cause of eutrophication and groundwater contamination (Paerl, 2009). Typically, in agricultural systems, fertilizers are applied at a rate greater than they are required which might lead to nonpoint-source pollution. In grazing systems, soil nutrient hotspots are created due to cattle preference to certain areas of pasture (Franklin et al. 2009; Sanderson et al. 2010; Hendricks et al. 2019). If such hotspots are present in the concentrated flow areas or areas vulnerable to erosion, they tend to get washed by runoff water. Previous studies indicated that soil nutrients deposited in low-lying parts pastures of high cattle activity are prone to runoff loses (Dahal et al. 2018; Wilcock et al. 2012). In the southeastern US, pastures have been established on vulnerable lands with steep slopes and likely have been eroded (Drouillard, 2018). Soil nutrient losses from grazing systems is a common problem, and it occurs through various avenues. For example, excess fertilization of pasture with broiler litter and inorganic fertilizer is common in Southeastern US, thus; loss of excess nutrients via leaching, runoff, and volatilization is also common. Pierson et al. 2001 reported that broiler litter application in pastures increased the ammonium nitrogen in runoff from 0.5 mg L^{-1} to >18 mg L^{-1}. The runoff losses are caused by several factors such as: timing, type and rate of fertilizer used (Franklin et al. 2006), field slope and aspect, rainfall intensity and vegetative cover (Brennan et al. 2012).

Nutrient losses are also attributed to grazing management activities and cattle behavior (Christensen et al. 2019) in the farm (Franklin et al. 2009; Byers et al. 2005). Vegetative cover and soil health are important parameters for quality and quantity of runoff water (Franzluebbers et al. 2012). Smith (1989) assessed the effect of using riparian buffer in surface nitrogen losses from pastures. She found that the pastures with no riparian buffers had median nitrate nitrogen ranging from 24-106 ppm, whereas, pastures with riparian buffers recorded 13-33 ppm. Overall, there was a significant reduction of nitrogen and nitrate in runoff water in pastures with riparian buffers.

In a rainfall simulation study by in North Carolina, Butler et al. (2007) assessed the impact of groundcover on nitrogen losses in runoff using four levels of ground covers (0, 45, 70, and 95% ground cover). They applied 200 kg N ha^{-1} from beef steer feces and urine to simulate usual pasture environment. They reported that total nitrogen export from the bare plots was maximum and the lowest ground cover (45%) reduced losses by 34%. Also, there was no statistical differences between low, moderate or high cover in terms of total nitrogen losses. They showed that all ground covers under consideration reduced the ammonium losses in runoff by at least 85% as compared to bare soil. Hence, they recommended that time of rainfall following fertilizer application and vegetative cover are important consideration for reducing nitrogen losses in pastures.

In a long-term study (11 years) by Pilon et al. 2019 in Arkansas, USA, five grazing treatments were designed and implemented (hayed, continuously grazed, rotationally grazed, rotationally grazed with riparian buffer, and rotationally grazed with fenced riparian buffer) in 15 historically continuously grazed, broiler litter fertilized pastures to study nitrogen runoff losses. Contrary to common belief, they reported that rotationally grazed pastures had most nitrogen losses in runoff. They also found that the rotationally grazed pastures with riparian fencing significantly reduced nitrate losses in runoff by 52%. In a study by Lambert et al. (1985) two grazing management systems; (a) rotational grazing with cattle, (b) rotational grazing with sheep, and (b) continuous grazing with sheep, high and low fertilizer application, were compared in terms of various water quality parameters. They found that rotational grazing with cattle pastures had higher nitrogen losses in runoff (12.1 kg ha^{-1} $year^{-1}$) as compared to sheep grazed pastures (8.7 kg ha^{-1} $year^{-1}$). They suggested that optimum stocking density is important for reducing runoff nitrogen losses and it also depends on animal type.

Objectives

The objectives of this study were to investigate:

1. Understand the effect of continuous grazing on spatial distribution of inorganic nitrogen, through the analysis of:

    a. Management factors (distance to hay, shade and water)

    b. Landscape parameters (slope and elevation)

    c. Cattle locus index

2. The effect of strategic grazing and continuous grazing with hay spreading on soil health and water quality in pastoral systems, through the analysis of:

    a. Soil respiration

    b. Active carbon

    c. Potentially mineralizable nitrogen

    d. Inorganic nitrogen

    e. Runoff nitrate

3. The effect of strategic grazing and continuous grazing with hay distribution on soil carbon and nitrogen, in managed pastures, through analysis of:

    a. Concentration and spatial distribution of active carbon

    b. Concentration and spatial distribution of inorganic nitrogen

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CHAPTER 2

SPATIAL DISTRIBUTION OF INORGANIC NITROGEN IN PASTURES: AS AFFECTED BY MANAGEMENT, LANDSCAPE, AND CATTLE LOCUS[1]

[1] Dahal, S., *Franklin, D.H., Cabrera, M.L., Hancock, D.W., Stewart, L., Ney, L.C., Subedi, A., & Mahmud, K.H. 2018. *Journal of Environmental Quality*. 47: 1468-1477.
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ABSTRACT

Uneven spatial distribution of soil nitrogen (N) in conventionally managed pastures is a function of various biotic and abiotic factors and results in poor land use efficiency. In this study, we measured soil inorganic N (0-5, 5-10, and 10-20 cm depth) in a 50-m grid and specific areas of interests (AOIs) from eight conventionally managed beef-pastures (~ 17 ha each), four near Eatonton and four near Watkinsville in the Southern Piedmont of Georgia, USA, to assess the effect of management, landscape, and cattle locus on spatial distribution of soil inorganic N. Significant spatial autocorrelation was observed in the soil inorganic N indicating that the regions of high inorganic N deposition were near (within 91 m) one or more pasture equipage (hay, shade, and water). In the Watkinsville pastures, inorganic N, down to a 10 cm soil depth, was 65% greater within 5 m of shade than in the rest of the pastures. In the Eatonton pastures, inorganic N (0-5 cm) was 22% greater within 30 m of hay-feeding areas than in the rest of the pasture. Cattle locus calculated as cattle density (cow ha^{-1} yr^{-1}) was a function of pasture equipage and had a significant positive relationship with soil inorganic N. Landscape parameters (slope and elevation) significantly affected inorganic N distribution however the effect was small and was masked by management factors. Our results suggest that strategic placement of pasture equipage (hay, shade, and water) can effectively distribute N where needed in beef-pastures thereby increasing land use efficiency.

INTRODUCTION

Pasture-based beef cattle production is a significant contributor to the economy of southeastern USA (McBride and Kenneth, 2011) as well as the national agricultural GDP (USDA, 2015). Many beef producers follow a conventional grazing system that includes continuous grazing or lax-rotational grazing (Butler et al. 2010). Spatial variability of soil nitrogen (N) is a major concern in continuously grazed pastures because it results in less than optimum land use. Researchers (Franklin et al. 2009; Byers et al. 2005) have indicated that nutrient losses from pasture are attributed to management activities and cattle preference for certain areas of the pasture (Matthews et al. 1994; DelCurto et al. 2005). It has been documented that cattle concentration areas have significant impact on nutrient hot-spots in the pastures (Sanderson et al. 2010; Matthews et al. 1994; Bailey et al. 2001) but very few efforts have been made to quantify the effects of management activities and landscape parameters on spatial distribution of nitrogen. Franzluebbers et al. (2000) reported that there was spatial variability in N with respect to shade and water sources in small paddocks.

The goal of this research was to assess the spatial variability of soil inorganic N and to quantify the effects of management activities, cattle locus, and landscape parameters in a multi-location, representative farm-size setting via extensive grid-soil sampling and use of sophisticated GIS technologies.

MATERIALS AND METHODS

Study Sites

The study was conducted at 1) J. Phil Campbell Sr. Research and Education Center (33.887487, -83.420966), Watkinsville, Georgia, and 2) Animal and Dairy Science Department Eatonton Beef Research Unit (33.420759, -83.476555), Eatonton, Georgia. Each location had 4 pastures (approximately 17 ha each) and managed as separate herds for a total of eight experimental units. These locations were selected mainly because they were representative of conventional beef-farming system in the Southern Piedmont area: fescue (*Festuca arundinacea* Schreb) and bermudagrass (*Cynodon dactylon* L.) mixed pastures.

The soil type, soil texture, and erosion potential were extracted from Web Soil Survey (USDA, NRCS) and verified with a first-order soil survey (conducted by NRCS, USDA). The historic management information was accessed from the farm records of respective experimental stations.

Soil Sampling

In the Fall 2014 to Spring 2015, a 50-m grid (referred to as ‘matrix’) was laid out in the pastures and soil samples were taken using a 5-cm diameter Giddings hydraulic probe (Giddings Machine Corporation, Inc., Windsor, CO) at 3 different depths (0-5 cm, 5-10 cm and 10-20 cm). Soil samples were also taken from predefined “areas of interests” (referred to as “AOI”). The AOIs were selected based on field observation and historical records with respect to cattle preference to certain areas of pasture, and landscape parameters. In general, the areas selected as AOIs had high cattle activity with (i) uphill depositional and erosional landscape positions, and (ii) downhill depositional and erosional landscape positions.

Management Parameters

Geographic coordinates of hay-feeding stations, natural and artificial shades, and waterers were recorded using an R10 GPS (Trimble Inc., Sunnyvale, California, USA). The distance of each sampling location to the nearest hay feeding station, waterer, and shade was calculated in ArcGIS 10.x (ESRI, Redlands, California, USA) using Near Tool. The Near Tool iterates each sampling location and measures the planar distance (straight line) between sampling location and the nearest pasture equipage specified. These distances were measured within each pasture for each of the pasture equipages.

Cattle Locus

Use of GPS collar to assess animal locations is already a well-established method (Bailey et al. 2001; Byers et al. 2005; Turner et al. 2000) and it provides a reasonably accurate dataset with high temporal resolution. In Eatonton, two to three cows were fitted with GPS Collars (Lotek Engineering, Inc., Newmarket, Ontario, Canada) on each pasture, which recorded the location of those cows in decimal degrees (WGS-1984 geographic coordinate system) every 5 minutes. Location data were not available for January-early April because cows were removed from the pastures for calving and breeding. The collars were removed every 28 days for data download and battery recharging. The uncorrected location-data files were downloaded using GPS 3000 Host Program (Lotek Engineering, Inc., Newmarket, Ontario, Canada). The location data were corrected, georeferenced, and processed using a Continuously Operating Reference Station (COORS) located near Athens (33.95237, -83.32563) Georgia, operated by National Geodetic Survey (NGS). The corrected location-datafiles were imported to ArcGIS 10.x (ESRI, Redlands, California, USA) and projected to NAD 1983 UTM (Universal Transverse Mercator) Zone 17N for further analysis.

Cattle density per hectare for each measurement period (bottom-right of Fig. 3) was calculated in a 5 x 5 m grid using Point Density Tool in ArcGIS10.x (ESRI, Redlands, California, USA). The Point Density Tool calculates the density of point features inside a defined grid. As mentioned in the methodology section, two or three collars were put on cows in each pasture. Thus, for each pasture, 2 or 3 cattle density maps (depending on how many collars were put in each pasture) were created for each measurement period. Those rasters were added using Raster Calculator Tool, in ArcGIS 10.x (ESRI, Redlands, California, USA), and divided by the number of collars (2 or 3) to get an average cattle density raster for each respective measurement period. Raster Calculator Tool was used to sum five cattle density rasters from five measurement periods (bottom-right of Fig. 3) to create final the cattle density (cattle locus) raster for each pasture. Visually comparing Fig.1 and Fig. 3 suggests that cattle spend a majority of time near the pasture equipages (hay, shade, and water).

Soil Analysis

Inorganic Nitrogen was measured using 2 M cold KCL (Potassium Chloride) extraction method (Maynard and Kalra, 1993). The ammonium (NH_4^+) and nitrate (NO_3^-) fractions were measured using Salicylate method (Kempers and Zweers, 1986) and Cadmium Reduction (Huffman and Barbarick, 1981*)* procedures respectively, using an Alpkem RFA300 AutoAnalyzer® (Astoria-Pacific, Oregon). Inorganic N was obtained by adding the ammonium and nitrate fractions. Bulk density was measured for three soil depths (0-5 cm, 5-10 cm, and 10-20 cm) in accordance with the USDA Soil Survey Laboratory Methods Manual (USDA, 2004).

Landscape Parameters

Elevation information was collected every 2-m using an R10 GPS (Trimble Inc., Sunnyvale, California, USA) throughout the pastures with sub-centimeters (0.004 m) accuracy.

The elevation data was then imported in ArcGIS Desktop 10.x (ESRI, Redlands, California, USA) to create Digital Elevation Models (DEMs) for each pasture (Fig. 2). Slopes (%) for each sampling location were calculated using Spatial Analyst and 3D Analyst in ArcGIS 10.x (ESRI, Redlands, California, USA). The Watkinsville and Eatonton pastures were similar in terms of slope with average slopes of 6% (Table 2).

RESULTS AND DISCUSSION

Spatial Distribution of Inorganic N

The Point Interpolation Tool in ArcGIS 10.x (ESRI, Redlands, California, USA) was used to create N distribution maps for three soil depths. The Tool uses the Empirical Bayesian Kriging interpolation method, which in turn uses repeated simulations to account for the error in the semivariogram. The NH_4^+, NO_3^-, and inorganic N (the sum of NH_4^+ and NO_3^-) were mapped separately for three soil depths.

Locations of greater N concentrations were observed in both forms of inorganic N and at all depths symptomatic of spatial autocorrelation, which is indicative of uneven distribution of inorganic N. In general, areas with greater soil N concentrations were nearby one or more pasture equipages (hay, shade, and water).

To further quantify the degree of spatial autocorrelation, spatial weight matrices were created for each study site using different neighborhood structures. A total of 12 spatial weight matrices (i) exactly 4, 8, and 12 nearest neighbors, (ii) at least 4, 8, and 12 neighbors using fixed distance of 50 m, (iii) at least 4, 8, and 12 neighbors using inverse distance at 50 m, and (iv)) at least 4, 8, and 12 neighbors using inverse distance with power 2, at 50 m, were created for each study site. When creating the distance-based neighborhood structure, the distance condition (50

m) was overridden to ensure a minimum number of neighbors for all sampling locations. Moran's I (measure of spatial autocorrelation) was computed for each neighborhood matrix using Inorganic N at three soil depths. Neighborhood structures with highest spatial autocorrelation are reported in Table 3 as suggested by (Chi and Zhu, 2008).

A Moran' I near 0.2, -0.2, or further from 0 indicates spatial autocorrelation. The results in Table 3 suggest that there is significant spatial autocorrelation in the distribution of inorganic nitrogen at all depths in Watkinsville, especially for the 4-neighborhood weight structure. However, in Eatonton, spatial autocorrelation was evident only for the 0-5 cm soil depth.

Distribution of Inorganic N around Pasture Equipages

To quantify the role of management activities on distribution of inorganic N, distances of sampling locations to nearest hay, shade, or water (equipage) were used. Recursive Partitioning was used to classify the distances from each sampling location to the nearest hay feeding station, waterer and shade structure, based on inorganic nitrogen using the 'rpart' package (Therneau et al. 2017) in R statistical software (R Core Team, 2013). This technique uses a simple nonparametric regression approach to classify the data in such a way that similar response values are recursively grouped based on the predictor variable (Strobi et al. 2009). The response variable (inorganic N) was log transformed. The first split was taken as the threshold distance and the distances were classified into two groups based on that threshold to create a new variable. In most cases the threshold distances were less than 100 m. As we were primarily interested in knowing how the N was distributed around the pasture equipages, we selected 100-m as threshold distance whenever the first split was more than 100 m. The threshold distances were used to classify the distance variables in two groups to create new variables. For example, the distance to hay was split at 44 m (threshold distance) by Recursive partitioning (Figure 5).

Thus, all sampling locations with "Distance to hay" less than or equal to 44 m were classified as 1 and the rest of the sampling locations were classified as 0 to create a new hay variable.

The threshold distances suggest that the inorganic N distribution was significantly different within 100 m of all pasture equipages (Table 4). Different equipage influenced N distributions differently. In Watkinsville, N distribution differed significantly within 45 m of hay and within 11 m of shade, at 0-10 cm depths. In Eatonton, N distribution was significantly different within 83 m of hay and within 5 m of shade, for the 0-10 cm depths. Very small threshold distances were found for shade (less than 12 m) in both locations, for the depths 0-10 cm, which suggests significant inorganic N accumulation near and under the shade structures. Moderate threshold distances from hay (within 50 m, except 5-10 cm depth at Eatonton) were observed at both locations. The greater threshold distance from water, at both locations, is indicative of lesser N accumulation near waterers as compared to shade and hay areas. This might be due to availability of multiple waterers within pastures.

Effect of Management, Cattle Locus, and Soil Bulk Density

To assess the effect of pasture management activities, cattle locus, and bulk density on inorganic nitrogen, multiple linear regression was fit for the log of inorganic N for each soil depth using seven predictor variables (Log of Cattle Density, Classified Distance to Hay, Classified Distance to Shade, Classified Distance to Water, Bulk density (0-5 cm), Bulk density (5-10 cm), and Bulk density (10-20 cm). In Watkinsville pastures, the cattle density variable was not available.

$$Y = X\beta + \epsilon$$

Where Y denotes the response variable (inorganic N), X denotes the matrix of explanatory variables, β denotes the vector of regression coefficients, and ϵ denotes the error term. Residual spatial autocorrelation was checked for each model and whenever there was significant spatial autocorrelation in residuals, a spatial lag model was fit.

In Watkinsville, there was significant residual spatial autocorrelation in the multiple regression models. Thus, spatial lag models were fit for each soil depth with the spatial weight matrix (Table 3) resulting in highest autocorrelation as suggested by Chi and Zhu (2008). The spatial lag model was specified as:

$$Y = pWY + X\beta + \epsilon$$

Where, Y denotes the response variable (inorganic N), p denotes the spatial autocorrelation coefficient, W denotes the spatial weight matrix and ϵ denotes the error term, X denotes the matrix of explanatory variables, β denotes the vector of regression coefficients. No significant residual autocorrelation was observed in any spatial lag models.

A larger regression coefficient suggests a greater influence on the distribution of inorganic N. The regression coefficients listed in Table 5 suggest that there is significant effect of pasture management activities of hay, shade, and water for all soils and cattle density for the soils sampled in Eatonton. Regression coefficients also indicate that bulk density in the upper 10 cm has a significant effect on concentration of soil inorganic N, at both study sites.

Effect of Cattle Locus

There was direct and significant impact of cattle locus on the inorganic N distribution at 0-5 cm and 10-20 cm depth (Table 5). The effect at 5-10 cm was direct but statistically non-significant. This signifies the role of animals in manure distribution in pastures. It has been reported that cattle distribution patterns closely match the defecation and urination pattern (White et al. 2001; Dragnova et al. 2012) in managed dairy pastures. Our results are in agreement with other studies (Haan et al. 2010; DelCurto et al. 2005; da Costa et al. 2017) that also observed cattle preference for a certain pasture area and further indicated that cattle urination and deposition of manure was similar to observations of cattle distribution.

Effect of Hay Placement

There was significant effect of hay in distribution of inorganic N at both locations. In Eatonton, at the 0-5 cm depth, the areas within 28 m of hay were 22% greater in inorganic N than the rest of the pasture. The effect was extended to the 10-20 cm depth as well. The area within 54 m of hay was 17% greater than the rest of the pasture. In Watkinsville, the effect of hay placement was not evident at 0-5 cm, however at 5-20 cm depth, N content was 34% greater within 30 m of hay. This highlights the importance of rotating the hay feeding area for better distribution of N in the pastures. The deposition of N around hay feeding areas might be attributed to both N in leftover hay and the feces and urine deposited by cattle while eating. Hay feeding in Eatonton was predominately done near the top of the hill whereas hay in Watkinsville was more often fed lower in the landscape and, in part, in concentrated flow areas. It is reasonable to speculate that nitrogen in left-over hay, urine, and feces was likely washed away during heavy rainfall events (> 2.5 cm) in Watkinsville.

Effect of Shade

The areas near shade were significantly greater in inorganic N at all depths and both study sites with and exception of Eatonton (10-20 cm). The threshold distances (Table 4) were very small (within 11 m) at the 0-10 cm depths, suggesting the inorganic N is very high near and under the shade. In Eatonton, the inorganic N were 25% and 17% greater within 5 m of shade structures as compared to the rest of the pasture at 0-5 cm and 5-10 cm depths, respectively. In Watkinsville, the effect was even greater. The inorganic N content within 11 m of the shade structures were 65% and 64% greater than the rest of the pastures at 0-5 cm and 5-10 cm depths, respectively. Franzluebbers et al. (2000) also reported significantly greater N deposition within 30 m radius of shade in small paddocks.

Effect of Water Source

In Eatonton, the areas near water source were significantly greater (67%) in inorganic N at the 5-10 cm depth. The effect of water might have been masked by other management variables at other depths. However, in Watkinsville, this effect of water was not evident, as there was no significantly greater deposition of inorganic N within 100 m of water sources. This was in contrast with our assumptions. Areas with greater inorganic N accumulation were evident (Figure: 4 B) near some of the water sources, however it was not the case for all waterers. This might be attributed to multiple water sources available in Watkinsville pastures, some nearer to shade and others further from shade or the presence of heavy-use protections (i.e. geotextile fabric with crushed stone) on some of the waterers and not on others.

Effect of Bulk Density

The bulk density had significant impact on the inorganic N content. Bulk density at 0-5 cm had a negative effect on inorganic N at 0-5 cm depth at both study sites. This supports the theory that compacted soils reduce N infiltration, encourage nitrogen runoff, have a lower nutrient holding capacity. In managed pastures, the compaction mainly occurs around animal camping areas which are known to be prone to loss of N in runoff (Franklin et al. 2009; Byers et al. 2005). At 5-10 and 10-20 cm depths, no definitive conclusions could be made about the impact of bulk density in inorganic N. This might be attributed to different soil types and texture in these pastures.

Effect of Landscape Parameters

The landscape parameters could not be modelled collectively because the paddocks were different in terms of elevation, where whole pastures had lower elevations than upper pastures at the same location, and Eatonton generally had lower elevations than Watkinsville being further south in the Georgia Piedmont. Multiple regressions were carried out for each pasture for log of NO_3^- and NH_4^+ fractions, at each depth, to assess the effect of landscape parameters. There was significant effect of landscape factors in the distribution of inorganic N, and in most cases the effects of landscape parameters were either exaggerated or confounded by management factors.

Effect of Elevation

In Eatonton, elevation had a significant positive impact in distribution of NO_3^- in EP2, and NH_4^+ in EP4 (Table 6), which might be attributed to the flat/uphill hay feeding area and waterer. In other paddocks in Eatonton, no definitive effect of elevation was observed. In Watkinsville, mixed results were obtained. The NO_3^- fraction was negatively affected by elevation whenever it was statistically significant. In contrast, the NH_4^+ fraction was positively affected by elevation whenever the effect was significant with one exception of the NE paddock at 5-10 cm. This might be attributed to higher mobility of NO_3^- as compared to NH_4^+.

Effect of Slope

In Eatonton, slope was inversely related to both NO_3^- and NH_4^+ fractions with exception of EP4. This result in EP4 might be attributed to the location of waterer and hay feeding area in a greater slope region. Our findings support the common fact that areas with greater slope are prone to losing nutrients and the areas with smaller slope tend to accumulate nutrients.

In Watkinsville, no definitive effect of slope was observed in the distribution of inorganic N.

CONCLUSION

The spatial distribution of inorganic N in pastures is shaped by management, landscape, and soil factors. A typical conventionally grazed pasture is prone to uneven spatial distribution of inorganic N with greater soil N concentrations near water, hay, or shade. The uneven spatial distribution was evident even at 20 cm depth suggesting a need to move pasture equipages on a regular basis. The cattle density in a conventionally grazed pasture suggests that cattle spend the majority of their time near one or more of pasture equipages. Slopes within pastures can have a significant negative effect on the amount and spatial distribution of inorganic N. The effect of elevation was not consistent, and the effect of landscape factors were either exaggerated or masked by management factors.

Although the landscape and soil factors cannot easily be changed, these findings suggest that appropriate planning and placement of hay, pasture, and shade should result in an improved distribution of inorganic nitrogen and potentially improve nutrient use efficiency in grazing systems. Also, temporary exclusion of nitrogen rich areas can be practiced facilitating nitrogen uptake by plants.

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CHAPTER 3

STRATEGIC GRAZING IN BEEF-PASTURES FOR IMPROVED SOIL HEALTH AND REDUCED RUNOFF-NITRATE: A STEP TOWARDS SUSTAINABILITY[2]

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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 with hay distribution (CHD) on soil health indicators and runoff nitrate losses. In 2016, four pastures were converted to STR system, and four were grazed using CHD system. Post-treatment, in 2018, the STR system had significantly greater POXC (by 87.1, 63.4, 55.6 mg ha^{-1}, at 0-5, 5-10, and 10-20 cm, respectively) as compared to CHD system. Also, soil respiration was greater in the STR system (by 235 mg CO_2 m^{-2} 24 hr^{-1}), and less nitrate was lost in runoff (by 0.21 kg ha^{-1}) as compared to 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.

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 [Acharya et al. 2019; Thapa et al. 2018; Plastina et al. 2018] in various agroecosystems. Given the vast ecosystem services grazing lands provide [Bretagnolle et al. 2011; Lemaire et al. 2005], 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 [Joshi et al. 2019; Havstad et al. 2007]. This area provides numerous ecosystem services and is an important contributor to the national GDP. The Southeast region has approximately 11.6 million ha grazing lands [USEPA, 2015] and is home to 20% of the national beef-cattle herd [Drouillard, 2018]. Moreover, between 2005 and 2015, almost 4000 ha of row-crop land was converted to intensive grazing farms in Georgia [Machmuller et al. 2015]. Cow-calf production in pastures is common in this region due to climatic suitability and forage availability [Adkins and Riley, 2012], however, most of the beef-pastures are in fragmented marginal land that are vulnerable to erosion and usually not suitable for row-crop production [Drouillard, 2018]. Thus, there is a need for more studies to better understand these livestock systems to create grazing management systems more sustainable and suitable for this region. In 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 [Sanderson et al. 2010; Franklin et al. 2009], leading to uneven nutrient distribution [Dahal et al. 2018; da Costa et al. 2017] and inefficient land utilization. High cattle activity near pasture equipages (water, shade, hay, mineral blocks, etc.) in the conventional system results in nutrient hotspots and soil compaction [Hendricks et al. 2019; Franzluebbers et al. 2000] in depositional landscapes and loss of nutrients deposited in concentrated flow paths or in areas vulnerable to erosion resulting in poor fertility and vegetative cover in these marginal and steep areas of the pastures which exacerbate 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 [USDA-ERS, 2018]. 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 [IPCC, 2012], 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 overall sustainability of pastoral systems [Derner et al. 2018]. 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) [Weil et al. 2003], and soil respiration are recommended measures of soil health [Moebius-Clune et al. 2016; Melero et al. 2009]. A study in Georgia cattle pastures [Culman et al. 2012] reported a range of 201-1468 mg kg^{-1} POXC at 0-5 cm soil depth, and we believe 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. We expect an increase in plant-available nitrogen, lower runoff-nitrogen, and improved forage productivity. Pilon et al. [2019] reported that average nitrate losses from continuously grazed pastures and rotationally grazed pastures with fenced riparian buffer ranged from 0.3-1.8 kg NO_3^-N ha^{-1} and 0.03-5.53 NO_3^-N kg ha^{-1} respectively. The Pilon et al. [2019] 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 and reduce runoff-nitrogen as compared to the existing conventional grazing systems or rotational grazing only in Southeastern USA. Particularly, active carbon (POXC), soil respiration, potentially mineralizable nitrogen (PMN), inorganic nitrogen (IN), and runoff nitrate were monitored before and after implementation of two “better” grazing systems in historically continuously grazed pastures.

MATERIALS AND METHODS

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 soils in Watkinsville pastures were classified as Fine, kaolinitic, thermic Typic Kanhapludults with textural class of sandy loam and sandy clay loam. 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.

<u>Treatments</u>

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 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 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 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.

Soil Respiration Measurement and Soil Sampling

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). Soil samples were collected using a 5-cm-diam Giddings Hydraulic Probe mounted in a truck. Additional samples (10-15 points in each pasture) were also collected from areas frequented by cattle and that were vulnerable to nutrient loss (Table 1) from the pasture (referred to as "AOIs": Area 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). On the day of soil sampling, in-situ soil respiration was measured at each sampling location using alkali trap method as described by Anderson [1982]. 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 CO_2 by titrating with hydrochloric acid following the addition of barium chloride ($BaCl_3$). 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.

Soil Analysis

All soil samples were air-dried for 14 days, 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, [1993]. From the extract, ammonium (NH_4^+-N) was measured as suggested by Kempers and Zweers [1986], and the nitrate (NO_3^--N) was measured as described by Doane and Horwath [2003]. Potentially mineralizable nitrogen (PMN) was measured using hot-KCl extraction method described by Picone et al. [2015]. PMN was calculated as the difference of NH_4^+-N measured from hot-KCl extraction method and NH_4^+-N measured from cold-KCl extraction method. Inorganic nitrogen (IN) was calculated as the sum of NH_4^+-N and NO_3^--N measured from cold-KCl extraction method. Permanganate oxidizable carbon/active carbon (POXC) was measured as described by Weil et al. [2003].

<u>Runoff Collection and Analysis</u>

In each pasture, 3-4 pour-point runoff collectors were established at edge-of-field, downhill of AOIs. Contributing areas to each runoff collector were delineated using ArcGIS 10.x. Runoff was collected immediately after each runoff event, filtered and analyzed for nitrate (NO_3^-) 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 [1986]. The nitrate load was calculated by multiplying the concentration during each event and the associated runoff volume.

<u>Cattle Locus Index</u>

An Index of the time spent by cattle in pastures was measured using wildlife GPS Collars [Lotek Engineering] 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 [Lotek, Engineering, 2000], 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 x 24 hours x 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 Apr), were summed to get an annual cattle density raster for four project years (2015 to 2018). Total data collection days were not 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.

Statistical Analysis

Data processing was done in Microsoft Excel and data analysis was done using R Statistical software [R Core Team, 2013]. Cattle locus data were processed and analyzed using ArcGIS 10.x (ESRI). Student's t-test was used to compare the treatments. A non-parametric version of t-test (Wilcoxon Rank Sum test) was used to compare runoff data as it was right-skewed. Linear regression model was used for establishing the relationship between runoff-nitrate vs. soil nitrate and cattle locus index.

RESULTS

Weather Information During the Study Period

In 2016, the study area experienced an eight-month drought (April-November) which lowered the annual rainfall of 754 mm as compared 100-year average of 1190 mm, and 883 mm as compared to 100-year average of 1230 mm in Eatonton and Watkinsville, respectively (Figure 2). The low precipitation resulted in an extreme negative water balance in 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 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 water balance had discernible impacts on the pastures, which will be described later in the manuscript.

Active Carbon

After 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, however not statistically significant.

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 a rapid mineralization of soil carbon [Borken and Matzner 2009]. Orchard and Cook [1983] reported a rapid increase 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 through the drought and extremely wet weather suggesting that the STR management could help stabilize pastoral systems under changing climates.

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 -2.94 mg kg^{-1} reduction. At 10-20 cm depth, no change was observed in both treatments.

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 (-10.24 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. Also, the forage mix included legumes which could be responsible for additional increase in IN during post-treatment years.

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).

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.x., the time spent by cattle in that buffer was calculated (cattle locus index) by extracting the cattle index map (section 2.4) 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 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.

Similarly, the effect of soil nitrate on runoff nitrate loss was also assessed using linear regression (Figure 7B). 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 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).

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 [de Vries et al. 2015; Paine et al. 1999]. The grazing systems we proposed is a collection of several better grazing practices. A historical study by Mcilvain and Savage, 1951 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 quality [Zhou et al. 2019; Stanley et al. 2018; Oates et al. 2011; Paine et al. 1999; Walton et al. 1981]. Improved water quality and soil quality from cattle exclusion of vulnerable areas in pastures have also been reported before [Malan et al. 2018; Olson et al. 2011; Franklin et al. 2009]. Lure management of cattle using pasture equipages have been used to distribute the cattle activity in pastures [Franklin et al. 2009] 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 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 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 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 presence of mycelial networks that were not there in 2015 (Figure 9). However, this information was not quantified because the sampling depth was only to 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 the 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 STR grazing system is promising and stirs the need for longer-term research with periodic sampling below 20 cm.

Soils are dynamic living systems, and soil respiration is a crucial biological indicator of soil health [Moebius-Clune et al. 2016]. 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 hr^{-1}, which is lower than the 2628 mg CO_2 m^{-2} 24 hr^{-1} reported by Chiavegato et al. 2015 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 the year 2018, STR system had significantly more (235 mg CO_2 m^{-2} 24 hr^{-1}) soil respiration as compared to the CHD system. Both systems underwent a stress period during the drought of 2016 and wet period of 2017 resulting in increased soil respiration and a reduction in POXC. Previous studies [Orchard and Cook 1983; Borken and Matzner 2009; Borken et al. 2006] 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. Although, longer-term research is required to better understand the dynamics of soil respiration in these two systems.

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 [Subedi et al. 2019]) 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 the improved root growth and ground cover in the overseeded exclusions. It has been reported [Green and Kauffman, 1989] 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. In another part of this study Subedi et al. [2019] noted 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 runoff, especially nitrogen and phosphorus, are the leading cause of eutrophication and groundwater contamination [Paerl, 2009]. The nitrogen deposited in low-lying portions of pastures which have high cattle activity are prone to runoff loses [Dahal et al. 2018; Wilcock et al. 2012]. 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 collector, 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 of low-lying areas vulnerable to erosion in reducing runoff-nitrate losses. Previous studies [Wilcock et al. 2012; Hill et al. 2014; Hill, 1996] have reported the effectiveness of riparian buffer in agricultural watersheds to protect and utilize nitrate. The exclusions did not only protect the nitrogen [Groffman et al. 2002] from leaving the field but also provide extra forage [Dahal et al. 2019]. 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 exclusion, overseeding, and flash grazing of vulnerable areas for improving surface water quality.

The 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 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.

CONCLUSION

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 showed this 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 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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CHAPTER 4

STRATEGIC GRAZING FOR IMPROVING SPATIAL DISTRIBUTION OF ACTIVE CARBON AND INORGANIC NITROGEN[3]

[3] Dahal, S., *Franklin, D.H., Subedi, A., Cabrera, M.L., Hancock, D.W., Mahmud, K., Ney, L.C., Park, C., & Mishra, D.R. To be submitted to *Geoderma*.

ABSTRACT

Ecosystems service that could be provided by Grazing systems, both regulating and supporting, warrant more scientific studies dedicated to greater understanding of the breadth and magnitude of soil health parameters in these agroecosystems. Of particular need are studies that identify management systems that could improve spatial distribution of these parameters to optimize full use of these pastoral systems. A study was conducted in eight beef-pastures (~17 ha each) in the Southern Piedmont of Georgia that had been historically continuously grazed (>10 years) to study the effect of strategic grazing on concentration and spatial distribution of labile carbon (POXC) and inorganic nitrogen (IN). In the year 2015 (baseline), soil samples at three soil depths, 0-5 cm, 5-10 cm, and 10-20 cm, were collected from all pastures in a 50-m grid, as well as pre-defined areas of high cattle congregation. In May 2016, a strategic grazing system (STR) was devised and randomly implemented in four of the pastures, and a continuously grazed system with hay distribution (CHD) was implemented in the four remaining pastures. In 2018 (post-treatment), soil samples were again collected as was done in the baseline. Both baseline and post-treatment soil samples were analyzed in the laboratory to measure the POXC and IN. Moran’s I was used as an indicator of spatial autocorrelation and spatial distribution of POXC and IN. In STR pastures, post-treatment, experienced a significant increase in POXC (by 60.1, 43.6, and 58.8 mg kg^{-1}, in 0-5, 5-10, and 10-20 cm soil depths, respectively). Moreover, the STR pastures improved spatial distribution, 56% of the instances (4 pastures x 3 depths) of POXC, whereas the CHD pastures improved the spatial distribution only 25% of the instances. Similarly, spatial distribution of IN, in STR pastures was also improved (78 % of instances). Both “better”

grazing systems experienced an increase in IN concentration which was attributed to greater nitrogen release via mineralization of soil organic carbon, however, the increase was significantly greater in the STR system. Our results indicate that both concentrations and spatial distribution of POXC and IN could be improved by using the strategic grazing management system in managed beef pastures.

INTRODUCTION

Grazing lands have tremendous carbon and nitrogen cycling capacity (Zhang et al. 2018), and appropriate management of these lands is central for sustainable livestock production. Spatial distribution of soil nutrients is a crucial consideration for optimizing land-use and forage productivity of grazing lands. In continuously grazed pastures, animals tend to spend more time near pasture equipages such as hay, shade (Gerrish et al. 1993; Dahal et al. 2018), and water (White et al. 2001), which leads to an uneven deposition of feces and urine (Richards and Wolton ,1976; da Silva et al. 2013). Depending on the landscape position, such deposition of animal manure could either create nutrient hotspots (Franzleubbers et al. 2000; Dahal et al. 2018; Hendricks et al. 2019) if it is in a depositional landscape, or nutrients could be lost with runoff water if they are in a concentrated flow path (da Silva et al. 2001). The study of spatial variability of soil nutrients is important for recycling nutrients in the pasture, variable rate fertilization, and reducing fertilizer requirements and risk of stream pollution.

Nitrogen remains one of the most important inputs in grazing lands (Parfitt et al. 2005), whereas active carbon is another equally important component of grazing systems and a sensitive indicator of soil health (Haynes, 2000; Moebius-Clune et al. 2016). Cattle feces and urine are rich in carbon and nitrogen (Haynes and Williams, 1993; Oenema et al. 2005); hence effective dispersion of cattle manure across the pasture, by manipulating pasture equipages, rotational grazing, and exclusion of vulnerable areas, has a vast potential to improve spatial distribution of nitrogen and carbon.

While much of the studies are focused on understanding the spatial variability of soil carbon and nitrogen, very few studies (Soussana et al. 2004; Jensen et al. 2012; Bernardi et al. 2017) have been conducted to develop grazing systems for improving spatial distribution. The objective of this study was to compare the concentration and spatial distribution of inorganic nitrogen and active carbon between strategic grazing system and continuous grazing with hay distribution, at varying soil depths.

MATERIALS AND METHODS

Study Sites

The study was conducted in eight beef-pastures located at Animal and Dairy Department Eatonton Beef Research Unit, Eatonton, Georgia (33.420759° N, 83.476555° W) and J. Phil Campbell Sr. Conservation Research and Education Center, Watkinsville, Georgia (33.887487° N, 83.420966° W), USA. The climate is humid subtropical, and the average annual precipitation in Eatonton and Watkinsville are 1190 mm and 1230 mm, respectively. The average annual minimum and maximum temperatures range from 10.4° C – 11.1° C and 22.5° C -25.6° C, respectively.

Each site consisted of four historically (<10 years before 2016) continuously grazed pastures (average size 17 ha). Both study sites were tall fescue (*Festuca arundinacea* Schreb)-Bermudagrass (*Cynodon dactylon* L.) mixed pastures which are representative of conventional beef farming in the southern Piedmonts of United States. Before 2016, the pastures were grazed using a put-and take system where the stocking densities in Eatonton and Watkinsville ranged from 1.2-1.8 cow-calf pair ha^{-1} and 1.8-2.4 cow-calf pair ha^{-1}, respectively.

The soil information for the pastures was collected from Web Soil Survey (Soil Survey Staff) and was verified by a first-order soil survey (conducted by NRCS-USDA). In Eatonton, soil was sandy loam, clay loam, and loam, classified as Fine, kaolinitic, thermic Rhodic Kandiudults; loamy, mixed, active, thermic, shallow Typic Hapludalfs. In Watkinsville, soil was sandy loam, sandy clay loam, and was classified as Fine, kaolinitic, thermic Typic Kanhapludults. In terms of erosion potential, Eatonton soils were eroded to moderately eroded, whereas Watkinsville pastures were eroded to severely eroded.

Experimental Design

Two grazing systems; (i) Strategic-Grazing (STR), and (ii) Continuous-Grazing with 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 of pasture for manipulating cattle location. The STR system was a combination of multiple better grazing practices listed in Table 1. 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.

Soil Sampling and Analysis

An extensive baseline soil sampling was conducted in March 2015 and December 2014 in Eatonton and Watkinsville pastures, respectively. A 50-m grid (referred to as “Matrix”) was laid in the pastures (Figure 1) and two soil cores were collected from each location using a 5-cm diameter Giddings hydraulic probe (Giddings Machine Corporation) at three soil depths (0-5, 5-10, and 10-20 cm). In addition to the Matrix samples, soil samples were also collected from predefined vulnerable areas of pastures with high cattle activity. Soil samples were air-dried, ground, sieved (2-mm mesh), and analyzed for permanganate oxidizable carbon and inorganic nitrogen. The active carbon/active carbon (POXC) was measured as described by Weil et al. 2003. Inorganic nitrogen (IN) was calculated by adding the ammonium (NH_4^+) and nitrate (NO_3^-) fraction extracted using 2 mol L^{-1} cold-KCl extraction method (Maynard and Kalra, 1993). All sample weights were corrected for air-dried moisture content. From the extract, ammonium (NH_4^+-N) was measured as suggested by Kempers and Zweers (1986), and the nitrate (NO_3^--N) was measured as described by Doane and Horwath (2003). In February 2018, soil samples were collected and analyzed again in similar manner as the baseline to allow comparison between and within treatments. A portion of the SW pasture in Watkinsville was not sampled in 2018 because of unplanned nitrogen fertilizer application during sampling.

Data Analysis

Data analysis was done in R statistical software (R Core Team, 2013) and ArcGIS 10.x (ESRI). The treatments were compared using Wilcoxon-Mann-Whitney test (Mann and Whitney, 1947). Spatial autocorrelation, measured as Moran's I (Moran, 1950), was calculated for each pasture at each sampling depths using a 55-m threshold for the neighborhood structure (ArcGIS 10.x). Nutrient distribution maps were created by using Point Interpolation Tool (ArcGIS 10.x) which uses empirical Bayesian kriging interpolation method, using repeated simulations to account for the error in the semivariogram (Krivoruchko, 2012).

To assess the change in nitrogen from 2015 to 2018, soil organic carbon, POXC, and bulk density data (Subedi et al. 2019) were utilized to calculate amount of nitrogen released from mineralization of soil organic carbon. The loss-on-ignition carbon (LOI) was converted to soil organic carbon (SOC) using the relationship (SOC = -0.6289 + 0.4687 x LOI) established by Hendricks et al. (2019). The POXC fraction was subtracted from the soil organic carbon to account for the change in the POXC. The C: N of the pasture soils (13.5) was utilized from existing literature (Awale et al. 2017; Franzluebbers et al. 2000) to convert the soil organic carbon to nitrogen. The nitrogen was converted to kg ha^{-1} by multiplying with the bulk density. The potential increase in soil nitrogen from SOC was calculated by subtracting 2018 values from 2015 values.

RESULTS AND DISCUSSION

Permanganate Oxidizable Carbon

In the baseline year (2015), there was no significant difference between the treatments, in terms of POXC, at 0-5 and 10-20 cm soil depths (Figure 2). At 5-10 cm depth, the CHD pastures had significantly lower POXC (335 mg kg^{-1}) as compared to the STR pastures (398 mg kg^{-1}). In the year 2018, we found that STR pastures had greater POXC than CHD pastures by 125, 109, and 107 mg kg^{-1}, at 0-5, 5-10, and 10-20 cm soil depths, respectively. When compared within treatments, the STR pastures experienced significant increases in POXC in the year 2018 as compared to the year 2015 at all threes soil depths (60, 44, and 59 mg kg^{-1}, at 0-5, 5-10, and 10-20 cm soil depths, respectively). The CHD pastures experienced a decrease in POXC at all soil depths, however statistically not significant (the p-values were 0.0612, 0.0837, and 0.4877 at 0-5, 5-10, and 10-20 cm, respectively).

Our results agree with Conant and Paustian (2003) who reported an increased particulate organic matter in rotational grazing as compared to continuous grazing, up to 10 cm soil depth.

In a long-term grazing study, Reeder et al. (2004) observed that continuous grazing reduces the labile fraction of soil organic matter. They reported that at deeper depths, the particulate organic matter was still higher in rotational grazing but not statistically significant. Within three years of change in grazing management, our study showed positive changes in POXC in the STR pastures. This increase in POXC does not only show the ability of STR system's ability to sequester more carbon but also improve overall soil health. It should be noted that the increase was observed down to 20 cm soil depth; thus, there is a possibility that the increase might be evident at deeper soil depths. Although, we do not have data for deeper layers, there was visual evidence, during 2018 soil sampling, showing roots and mycelium tens of cm below the 20 cm sampling depth as well as evidence of other soil biota that was not obvious in the 2015 soil sampling.

Inorganic Nitrogen

The inorganic nitrogen (IN) was significantly greater in 2018 as compared to 2015 in both treatments at all soil depths. However, no significant difference in IN was observed between treatments in both years; thus, data was analyzed separately for two locations. It is important to note that Watkinsville pastures received 56 kg ha^{-1} nitrogen every year (2013-2017) and 45 kg ha^{-1} in 2018, on the upper half of all pastures. Eatonton pastures did not receive any inorganic fertilizer during the study period (2015-2018). During baseline, Watkinsville pastures were greater in terms of IN at all soil depths. Post-treatment IN concentration was either comparable or greater than Watkinsville pasture at all soil depths. This highlights the ability of pastures under STR grazing management to more efficiently retain and mineralize organic N to available forms without the addition of mineral fertilizers. In Eatonton pastures, during baseline at 0-5 cm soil depth, STR pastures had significantly greater IN as compared to CHD pastures (by 0.75 mg kg^{-1}). In 2018, the difference (5.03 mg kg^{-1}) was still significant. At the 10-20 cm soil depth, during baseline, the STR pastures had significantly more IN (by 0.76 mg kg^{-1}) as compared to CHD pastures (Table 2). In the year 2018, STR pastures did not significantly differ from CHD pastures, which indicates removal of nitrogen from 10-20 cm (Table 2). This might be attributed to better forage shoot and root growth in the STR treatments and utilization of nitrogen in deeper soil depths as well as redistribution to other parts of the pasture as a result of flash grazing and lure management. Such differences were not evident at 5-10 cm soil depth. Improved root and shoot growth due to rotation of cattle and over seeding of exclusions might be attributed to higher availability of inorganic nitrogen in the STR pastures. In Watkinsville pastures, no such differences between the treatments were observed in both years at any soil depth.

There was an overall decrease of loss-on-ignition carbon in another part of this study (Subedi et al. 2019) from the baseline year (2015) to post-treatment year (2018). The nitrogen released from organic matter mineralization is reported in Table 3 which shows that there was potentially significantly greater nitrogen availability in STR pastures as compared to CHD pastures at 5-10 cm and 10-20 cm soil depth. We attribute higher inorganic nitrogen in the post-treatment sampling to the carbon mineralization and release of nitrogen in the process. The change in IN from 2015 to 2018 (Mean $_{in}$) in Table 3 shows that there was a greater increase in IN in the STR pastures as compared to CHD pastures at all depths. However, the potential increase in N/actual increase in IN was significantly greater only at the 10-20 cm depth (2.3 mg kg^{-1} in CHD as compared 3.9 mg kg^{-1} in STR pastures).

<u>Spatial Distribution of Permanganate Oxidizable Carbon and Inorganic Nitrogen</u>

Spatial autocorrelation (Moran's I), as an indicator of clustering in POXC, was calculated for each pasture at three soil depths, resulting in 24 values for the baseline and 21 values for the post-treatment (Table 3). For each pasture and corresponding three soil depths, the Moran's I was compared before and after treatment. A Moran's I of 0.2 was used as the threshold; Moran's I greater than 0.2 was considered to have spatial autocorrelation. The STR pastures experienced a decrease in spatial autocorrelation 56 % of the total instances, whereas it remained the same 44 % of the total instances (Figure 3). The CHD pastures experienced an increase, decrease, or no change for 8.3%, 25%, and 66.7% of the total instances, respectively. This result shows that the STR system was able to either improve or maintain the spatial distribution of POXC, and a more uniform spatial distribution of POXC is desirable and is an indicative of healthier soil which might ultimately lead to better forage productivity and better utilization of the whole pasture. The comparison of Moran's I for inorganic N (Table 4 and Figure 3) shows that STR pastures were able to either improve or maintain the spatial distribution of inorganic nitrogen in pastures. In STR pastures, 78 % of the total instances Moran's I decreased, whereas 22 % of the time no change was observed. In the CHD pastures the spatial autocorrelation mostly remained the same (58 %) or experienced a decrease (25%).

Cattle feces and urine are important source of soil carbon and nitrogen in grazing systems. Past studies have reported that cattle activity closely matches defecation and urination patterns (White et al. 2001; Draganova et al. 2012). Well distributed cattle activity, due to lure management using pasture equipages and moderate rotational grazing, might have promoted a more uniform distribution of cattle excreta resulting in a more spatially uniform concentrations of POXC (Figure 4, 5 and Table 4) and IN (Figure 6, 7 and Table 4). The nutrient rich exclusions, over seeded with productive legume-grass mix, utilized the available soil nutrients and reduced compaction (Subedi et al. 2019) and soil health via cattle exclusion and root growth. The unrestricted nutrient uptake by plants from the overseeded excluded areas (which were flash grazed) might have attributed to the improved spatial distribution of POXC and IN via cattle excreta redistribution. In addition, the rest time provided by cattle rotation allowed forage shoot and root regrowth which facilitated the improvement in POXC down to 20 cm soil depth.

CONCLUSION

Uneven spatial distribution of soil nutrients and soil health indicators, and the issues triggered by this inefficient use of nutrient in conventional grazing systems have been well identified. However, there are relatively few studies dedicated to develop and assess grazing management systems for improving spatial distribution of soil health indicators. The strategic grazing system devised and evaluated in this study was aimed at not only improving the concentration of soil nutrients but also their spatial distribution for optimize land use and efficient nutrient retention and cycling. The strategic grazing system significantly improved the concentration and spatial distribution of POXC down to 20 cm soil depth. This result has an important implication in the changing climatic scenario for sequestering carbon in soil for mitigating effects of extreme weather events. Moreover, increased labile carbon has been proven to have demonstrated positive impact on pasture health, soil health, and productivity of grazing systems. The strategic grazing system also improved spatial distribution of inorganic nitrogen, the readily available fraction of soil nitrogen, down to 20 cm soil depth. A more uniform spatial distribution of inorganic nitrogen is indicative of pasture’s ability to provide nitrogen, uniformly across the pasture, for forage growth. These promising results within three years of implementing strategic grazing highlights a need for a longer-term study to fully understand the potential of strategic grazing to improve the quantity and spatial distribution of labile carbon and inorganic nitrogen.

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CHAPTER 5

CONCLUSIONS

The multitude of ecosystem services provided by pastures and grazing lands mandate more scientific research for improving sustainability of these complex agroecosystems. The overall goal of this study was to improve the existing grazing system in Southern Piedmonts of Georgia to enhance the soil health, and water quality and improve the spatial distribution of labile carbon and nitrogen in beef pastures. This dissertation documents several promising results and advantages of strategic grazing (STR), in terms of soil health and water quality, over the existing continuous grazing system and a slightly modified continuous grazing with hay distribution (CHD) grazing system.

It is important to note that two soil sampling schemes were used in this study. For the ease of understanding, the two studies will be referred to as (ii) Matrix study which includes an extensive 50-m grid soil sampling plus extra soil samples collected from pre-identified vulnerable areas of interest with high cattle activity (AOI), in 2015 (Pre) and 2018 (Post), and (ii) Soil Health study which includes the 18% locations of the 50-m grid (approximately 10 locations per pasture) and all AOI locations (10-15 locations per pasture), sampled more frequently (2015, 2017, and 2018).

In 2015, to assess the current situation of soil health in pastures, we conducted the Matrix sampling, up to 20-cm soil depth, in eight historically continuously grazed beef-pastures. The spatial modeling of inorganic nitrogen (IN) showed that location of pasture equipages, grazing management, cattle locus index, and landscape parameters significantly affect the spatial distribution of IN. Whenever the sampling location was near hay, shade or water, the concentration of IN was significantly higher highlighting the ability of cattle locus and excreta to modify spatial distribution of soil IN. Although landscape parameters are difficult to modify, we hypothesized that cattle locus could be easily manipulated by using portable pasture equipages to bring positive change in grazing systems. This led to the conception of an innovative grazing system: Strategic-Grazing.

The soil health study was conducted in 2015 (baseline), 2017, and 2018 to assess the effect of STR and CHD systems in labile carbon (POXC), in-situ soil respiration, potentially mineralizable nitrogen (PMN), inorganic nitrogen (IN), and runoff nitrate. The comparatively greater labile carbon sequestration capacity of STR system, as compared to the CHD system, highlights its potential resistance to extreme weather and adaptability in our fight with changing climate and extreme weather events. Reduced nitrate losses in the STR pastures, attributed to cattle-exclusion and over seeding of the low-lying, vulnerable to erosion, and nutrient-rich areas with historically high cattle activity, doesn't only reduce stream pollution, but also provides economic benefit by retaining the expensive nitrogen in the field. The over seeded exclusions also effectively mineralized the nutrients deposited in the AOIs and made it more available for plant uptake and provided continuous vegetative cover to protect those vulnerable areas. The vegetative cover in exclusion was attributed to the low hay requirement in the STR system which highlights this system's ability to adapt to extreme weather events as well to provide economic benefit by reducing external hay inputs. The STR system had consistent soil respiration and was less affected by drought (2016) followed by a wet year (2017), as compared to the CHD system. Overall, the Soil Health Study showed that promising positive changes in ecosystem services could be achieved from STR grazing system including increased labile carbon with soil depth, consistent soil respiration, efficient nitrogen mineralization, and cleaner runoff water.

Post-treatment, in 2018, the Matrix soil sampling was conducted again to assess the effect of STR and CHD grazing systems in the concentration and spatial distribution of POXC and IN. A more uniform distribution of labile carbon and nitrogen is indicative of a healthy soil and productive pasture. The STR grazing significantly improved the concentration and the spatial distribution of POXC down to 20 cm soil depth. Our study was limited to 20 cm soil depth, but further studies including deeper soil depths and longer timeframes are recommended to fully understand the carbon sequestration potential of STR system. The STR pastures also improved the concentration and spatial distribution of labile nitrogen (plant available/inorganic nitrogen) down to 20 cm soil depth by efficient mineralization of soil organic matter and potentially mineralizable nitrogen.

Overall, the STR grazing, in a relatively short timeframe, could improve the sustainability of grazing systems by generating positive changes in soil health, water quality, and spatial distribution of labile carbon, and nitrogen in grazing systems. We recommend a longer-term study to understand the full potential of STR system to improve pasture sustainability, and we also suggest implementing the STR system in farmers' field to fully understand its applicability and the extent to which it will have positive results.

Table 2.1

Site characteristic and historic management system of Eatonton and Watkinsville pastures

	Eatonton	Watkinsville
Soil Type	Fine, kaolinitic, thermic Rhodic Kandiudults Loamy, mixed, active, thermic, shallow Typic Hapludalfs	Fine, kaolinitic, thermic Typic Kanhapludults
Soil Texture	Sandy Loam, Clay Loam, Loam	Sandy Loam, Sandy Clay Loam
Erosion Potential	Eroded to Moderately eroded	Eroded to Severely Eroded
Grazing System	Put and Take	Put and Take
Stocking Density	20-30 head per pasture	30-40 head per pasture
Forage	Tall Fescue/Bermuda	Tall Fescue/ Bermuda

Table 2.2

Summary Statistics of the variables used in the analysis for pastures in Eatonton and Watkinsville.

	Mean	SD	Median	Min	Max	n	Abbreviations
Eatonton							
Inorganic N (0-5 cm; mg N/kg)	13.67	8.89	11.16	4.13	69.45	233	N05
Inorganic N (5-10 cm; mg N/kg)	7.69	6.63	6.14	2.46	63.20	233	N510
Inorganic N (10-20 cm; mg N/kg)	5.20	3.30	4.74	2.20	33.76	233	N1020
Elevation (m)	147.87	6.55	147.15	136.01	166.73	233	
Slope (%)	6.86	3.26	6.33	0.93	16.68	233	
Distance to Hay (m)	201.42	138.04	189.28	0.00	579.69	233	
Distance to Shade (m)	37.82	41.54	25.29	0.00	215.04	233	
Distance to Water (m)	202.07	133.63	175.57	5.66	543.33	233	
Cattle Density (ha^{-1})	30797	43229.58	17640	1247	288914	233	
Bulk Density (0-5 cm; g/cm^3)	1.39	0.35	1.35	0.71	2.51	233	BD05
Bulk Density (5-10 cm; g/cm^3)	1.62	0.27	1.58	0.82	2.54	233	BD510
Bulk Density (10-20 cm; g/cm^3)	1.46	0.26	1.42	0.98	2.35	233	BD1020
Watkinsville							
Inorganic N (0-5 cm; mg N/kg)	20.65	15.40	15.97	3.33	95.80	255	N05
Inorganic N (5-10 cm; mg N/kg)	9.27	9.33	6.73	2.63	94.45	255	N510
Inorganic N (10-20 cm; mg N/kg)	6.62	6.74	5.20	1.57	57.94	255	N1020
Elevation (m)	226.34	7.51	227.58	206.81	240.35	255	
Slope (%)	6.33	2.96	5.72	1.20	16.65	255	
Distance to Hay (m)	201.82	118.96	197.58	0.00	505.17	255	
Distance to Shade (m)	128.77	101.27	108.49	0.00	414.35	255	
Distance to Water (m)	180.59	108.03	174.64	6.99	403.81	255	
Bulk Density (0-5 cm; g/cm^3)	1.24	0.38	1.17	0.40	2.98	255	BD05
Bulk Density (5-10 cm; g/cm^3)	1.64	0.22	1.66	0.58	2.44	255	BD510
Bulk Density (10-20 cm; g/cm^3)	1.40	0.18	1.41	0.56	2.30	255	BD1020

SD: Standard Deviation, n: number of samples

Table 2.3
Neighborhood matrices with highest spatial autocorrelation with respect to inorganic N (the sum of NH_4^+. NO_3^-) at 3 soil depths, for Eatonton and Watkinsville.

	Eatonton		Watkinsville	
Soil Depth	Spatial weight matrix	Moran's I	Spatial weight matrix	Moran's I
0-5 cm	Exactly 4 nearest neighbors	0.17***	Inverse distance with at least 4 nearest neighbors	0.19***
5-10 cm	Exactly 4 nearest neighbors	0.06	Inverse distance with at least 4 nearest neighbors	0.29***
10-20 cm	Inverse distance with power 2 with at least 8 neighbors	0.04	Exactly 4 nearest neighbors	0.23***

*, ** and *** suggest that the regression coefficients are significantly different than zero at α equal to 0.1, 0.05, and 0.01 respectively.

Table 2.4
Threshold distances (at the first split) from hay, shade, and water, calculated from recursive partitioning for Eatonton and Watkinsville pastures, for 0-5 cm, 5-10 cm, and 10-20 cm soil depths

	Eatonton			Watkinsville		
	0-5 cm	5-10 cm	10-20 cm	0-5 cm	5-10 cm	10-20 cm
Hay	28	83	54	44	27	29
Water	91	100	91	100	25	100
Shade	4.5	4.5	100	3.6	11	100

Table 2.5
Coefficients for multiple regression of soil inorganic N with Cattle Density (Cow), Distance to Hay, Shade, and Water, and Bulk Density for Eatonton, and coefficients of spatial lag regression of soil inorganic N with Distance to Hay, Shade, and Water, and Bulk Density from spatial lag model for Watkinsville, at threes soil depths (0-5 cm, 5-10 cm, and 10-20 cm)

	Eatonton			Watkinsville		
Model	Multiple Reg	Multiple Reg	Multiple Reg	Spatial Lag	Spatial Lag	Spatial Lag
	0-5 cm	5-10 cm	10-20 cm	0-5 cm	5-10 cm	10-20 cm
Intercept	0.78*	1.46**	0.98***	2.41***	2.01***	1.40***
Cow	0.14***	0.04	0.07**			
Hay	0.22**	0.12	0.17*	0.21	0.34***	0.34***
Shade	0.25***	0.17**	-0.22**	0.65***	0.64***	0.14*
Water	0.04	0.66***	-0.01	-0.26***	-0.30***	-0.42***
BD05	-0.24*	-0.28**	0.00	-0.16*	0.25***	0.24**
BD510	0.30**	0.12	0.09	-0.12	-0.29**	-0.06
BD1020	0.04	0.09	-0.04	0.05	-0.28*	-0.11
n	233	233	233	255	255	255

*, ** and *** suggest that the regression coefficients are significantly different than zero at α is equal to 0.1, 0.05, and 0.01 respectively.

Table 2.6
Coefficients of multiple regression of inorganic N with landscape parameters (Elevation and Slope) for Eatonton and Watkinsville, for NH_4^+ and NO_3^- , at 3 soil depths (0-5 cm, 5-10 cm and 10-20 cm).

		NO_3^-			NH_4^+		
Eatonton	Parameters	0-5 cm	5-10 cm	10-20 cm	0-5 cm	5-10 cm	10-20 cm
EP1	Elevation	-0.02	-0.01	-0.02	-0.01	0.00	0.01
	Slope	-0.08*	-0.14***	-0.10**	0.00	-0.02	0.00
EP2	Elevation	0.12***	0.03	0.07**	0.01	-0.02*	-0.02*
	Slope	-0.10*	-0.06*	-0.01	-0.03***	-0.04***	-0.02*
EP3	Elevation	-0.02	-0.05	0.03	0.00	0.00	0.02
	Slope	0.03	0.00	-0.02	0.02	0.03	0.02
EP4	Elevation	0.04	0.03	0.03*	0.04***	0.02*	0.01**
	Slope	0.19***	0.07	0.08*	0.02	0.00	0.01
		NO_3^-			NH_4^+		
Watkinsville	Parameters	0-5 cm	5-10 cm	10-20 cm	0-5 cm	5-10 cm	10-20 cm
NE	Elevation	-0.06***	-0.03	-0.05***	0.02**	-0.05**	0.01
	Slope	-0.06	0.00	-0.05	0.01	-0.09	0.01
SE	Elevation	-0.03	-0.06*	-0.04	0.04***	0.03*	0.04***
	Slope	-0.03	0.10	0.04	0.00	0.03	0.03
SW	Elevation	-0.06	-0.10*	-0.07	0.05**	0.01	0.03
	Slope	-0.03	-0.11	-0.10*	0.00	-0.03	-0.01
NW	Elevation	-0.02	-0.10*	-0.08	0.04*	0.04**	0.01
	Slope	-0.10	-0.13*	-0.11	-0.02	0.01	-0.01

*, ** and *** suggest that the regression coefficients are significantly different than zero at α is equal to 0.1, 0.05, and 0.01 respectively.

Table 3.1

Description of better grazing practices implemented in STR grazing system.

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 (Area of Interests: AOIs) in pastures caused by high cattle activity/preference were excluded using an electric fence. AOIs were either uphill depositional areas and erosional landscape positions or downhill 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 (*Avena sativa* L.), ryegrass *(Lolium multiflorum* L.), crimson clover (*Trifolium incarnatum* L.), forage rape (*Brassica napus* L.), May 2017: Crabgrass (*Digitaria sanguinalis* L.), pearl millet (*Pennisetum glaucum* L.), and cowpea (*Vigna unguiculate* 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 a moderate rotational grazing (7-10 days) was followed to allow forage regrowth.

Table 3.2

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^-**
Before STR	Runoff NO3 = -0.09 + 0.27* x Cattle locus	Runoff NO3 = 0.05 + 0.007* x Soil NO3
After STR	Runoff NO3 = 0.05 – 0.00 x Cattle locus	Runoff NO3 = 0.021 + 0.00 x Soil NO3
Before CHD	Runoff NO3 = -0.03 + 0.21* x Cattle locus	Runoff NO3 = -0.039 + 0.015* x Soil NO3
After CHD	Runoff NO3 = -0.03 + 0.24* x Cattle locus	Runoff NO3 = -0.077 + 0.005* x Soil NO3

* indicates significant slope at α=0.05.

Table 4.1

Description of better grazing practices implemented in STR grazing system.

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.
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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 a moderate rotational grazing (7-10 days) was followed to allow forage regrowth.

Table 4.2

Comparison of inorganic N (mg kg^{-1}) between treatments in 2015 and 2018, and between years across treatments, for Eatonton and Watkinsville pastures, at 0-5, 5-10, and 10-20 cm soil depths.

Depth	Year	Treatment	Median	Mean	p-value	Treatment	Median	Mean	p-value
		Eatonton				*Watkinsville*			
0-5 cm	2015	CHD	10.5 b	14.9	0.0124	CHD	16.7 b	23.6	0.5201
		STR	11.6 B	15.7		STR	15.2 B	21.7	
	2018	CHD	72.7 a	91.5	0.858	CHD	74.2 a	89.8	0.5813
		STR	79.9 A	96.5		STR	79.2 A	87.4	
5-10 cm	2015	CHD	5.9 b	8.9	0.2048	CHD	6.6 b	8.4	0.1968
		STR	6.5 B	6.9		STR	6.7 B	11.1	
	2018	CHD	16.1 a	22.2	0.221	CHD	9.5 a	14.1	0.99
		STR	14.2 A	16.8		STR	9.9 A	11.8	
10-20 cm	2015	CHD	4.3 b	5.5	0.001	CHD	5.1 a	6.4	0.1726
		STR	5.0 B	5.4		STR	5.3 A	7.3	
	2018	CHD	8.5 a	10.3	0.274	CHD	4.8 a	6.3	0.9531
		STR	9.9 A	11.3		STR	5.1 A	5.3	

The p-value is reported for the Wilcoxon-Mann-Whitney test for two samples (comparing between treatments). The p-value less than 0.05 indicates a statistically significant difference between compared groups at α= 0.05. The lowercase letters compare the years within CHD system, whereas, the uppercase letters compare years within STR system for the particular soil depth. CHD = Continuous grazing hay distribution, STR = Strategic Grazing, SE = Standard Error of mean.

Table 4.3
Comparison of change in nitrogen (kg ha^{-1}) across treatments, calculated from change in soil organic carbon ($Mean_{soc}$), and ($Mean_{in}$) from 2015 to 2018, at 0-5 cm, 5-10 cm, and 10-20 cm soil depth. The p-value less than 0.05 indicates a statistically significant difference between compared groups at α= 0.05.

Depth	Treatment	$Mean_{soc}$ kg ha^{-1}	SE	p-value	$Mean_{in}$ (mg kg^{-1})	SE	p-value
0-5 cm	CHD	197.5	39.6	0.9602	69.4	3.9	0.125
	STR	200.5	46.2		76.5	4.7	
5-10 cm	CHD	203.4	35.5	0.0478	7.5	1	0.27
	STR	377.1	44.1		8.3	0.7	
10-20 cm	CHD	225.4	62.9	0.0016	2.3	0.5	0.001
	STR	381.3	68.9		3.9	0.4	

CHD = Continuous grazing with hay distribution, STR = Strategic-Grazing, in = Inorganic nitrogen, soc = soil organic carbon, and SE = Standard error of mean. $Mean_{soc}$ = Mean of difference in nitrogen from 2015-2018, calculated from soc mineralization. $Mean_{in}$ = Mean of change in inorganic nitrogen from 2018-2015.

Table 4.4
Comparison of Moran's I (indicator of spatial autocorrelation), for POXC and inorganic N, before and after the treatments, within each pasture, at three soil depths (0-5, 5-10, and 10-20 cm).

Treatment	Pasture	Depth (cm)	Moran's I (Before)	Moran's I (After)	Pasture	Depth	Moran's I (Before)	Moran's I (After)
				POXC				
CHD	EP1	0-5	-0.05	0.32***	SE	0-5	-0.09	0.33***
	EP1	5-10	0.08	0.08	SE	5-10	0.09	0.07
	EP1	10-20	0.45***	0.22**	SE	10-20	0.22***	0.13*
CHD	EP3	0-5	0.07	0.12	NW	0-5	0.15*	-0.01
	EP3	5-10	-0.07	-0.06	NW	5-10	0.20**	-0.02
	EP3	10-20	0.02	0.03	NW	10-20	0.23***	0.11
STR	EP2	0-5	0.22**	0.35***	NE	0-5	0.20**	0.10
	EP2	5-10	0.31***	0.05	NE	5-10	0.24***	-0.07
	EP2	10-20	0.27**	0.00	NE	10-20	0.22***	-0.01
STR	EP4	0-5	-0.02	0.03	SW	0-5	0.27***	
	EP4	5-10	0.09	0.11	SW	5-10	0.13*	
	EP4	10-20	0.20**	0.14	SW	10-20	-0.01	
				Inorganic N				
CHD	EP1	0-5	-0.02	0.117*	SE	0-5	0.078	-0.039
	EP1	5-10	0.05	0.04	SE	5-10	0.059	0.002
	EP1	10-20	-0.027	-0.06	SE	10-20	0.34***	-0.064
CHD	EP3	0-5	-0.024	0.29***	NW	0-5	0.052	-0.09
	EP3	5-10	0.014	-0.016	NW	5-10	0.177**	-0.005
	EP3	10-20	-0.002	0.059	NW	10-20	0.138**	-0.004
STR	EP2	0-5	0.22***	0.024	NE	0-5	-0.042	0.023
	EP2	5-10	-0.04	-0.012	NE	5-10	0.09	0.17**
	EP2	10-20	0.35***	0.015	NE	10-20	0.036	-0.004
STR	EP4	0-5	0.49***	0.18*	SW	0-5	-0.02	NA
	EP4	5-10	0.18*	0.05	SW	5-10	-0.096	NA
	EP4	10-20	0.26***	0.13	SW	10-20	0.085	NA

*** indicates significant Moran's I at α = 0.001. ** indicates significant Moran's I at α =0.01. * indicates significant Moran's I at α=0.05. NA=Not available (the SW pastures were not sampled in 2018 due to unintended fertilizer application right before soil sampling).

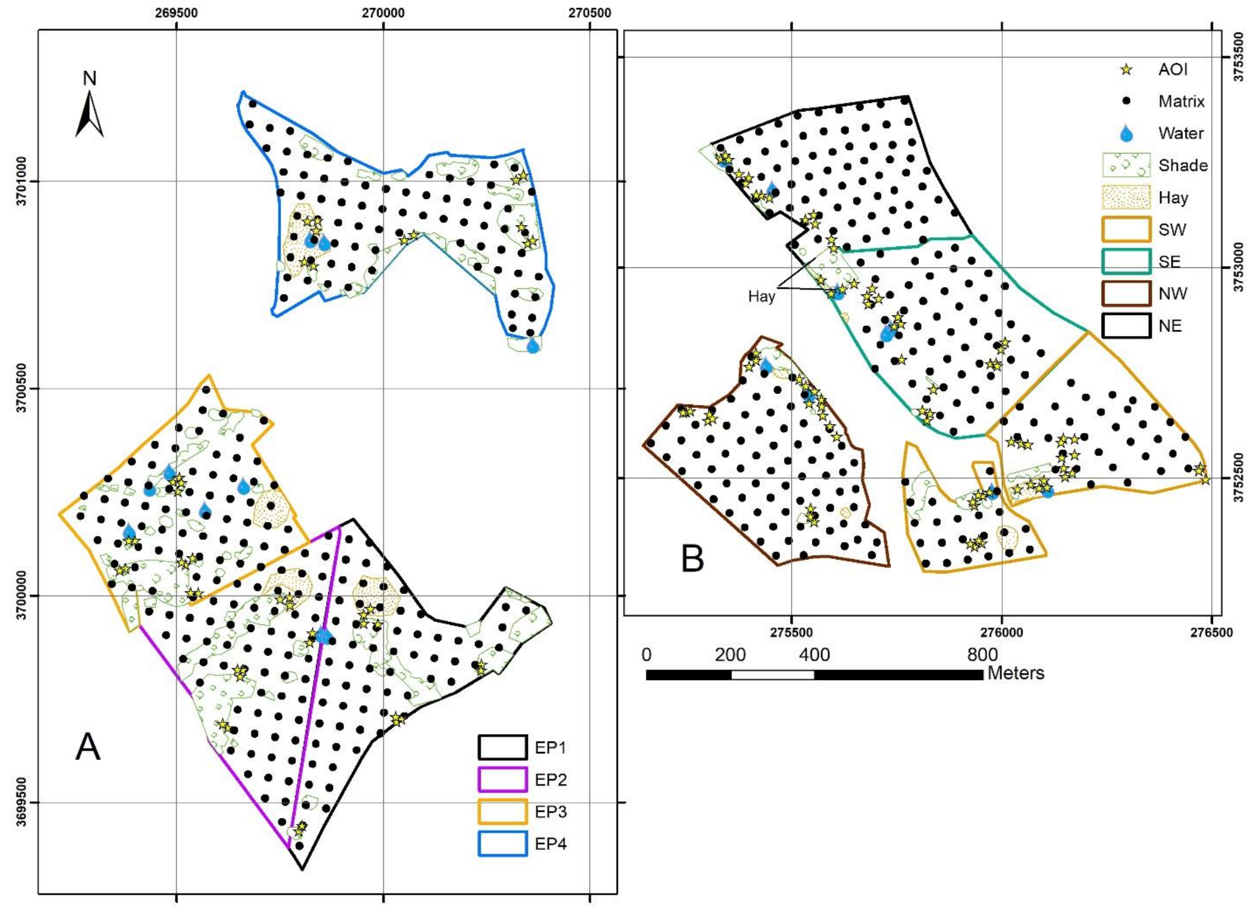


Figure 2.1 Study sites showing the paddock delineation, location of hay, shade and water, and sampling locations (A) Eatonton and (B) Watkinsville. The map unit is meters and the projection system is NAD 1983 UTM Zone 17 N.

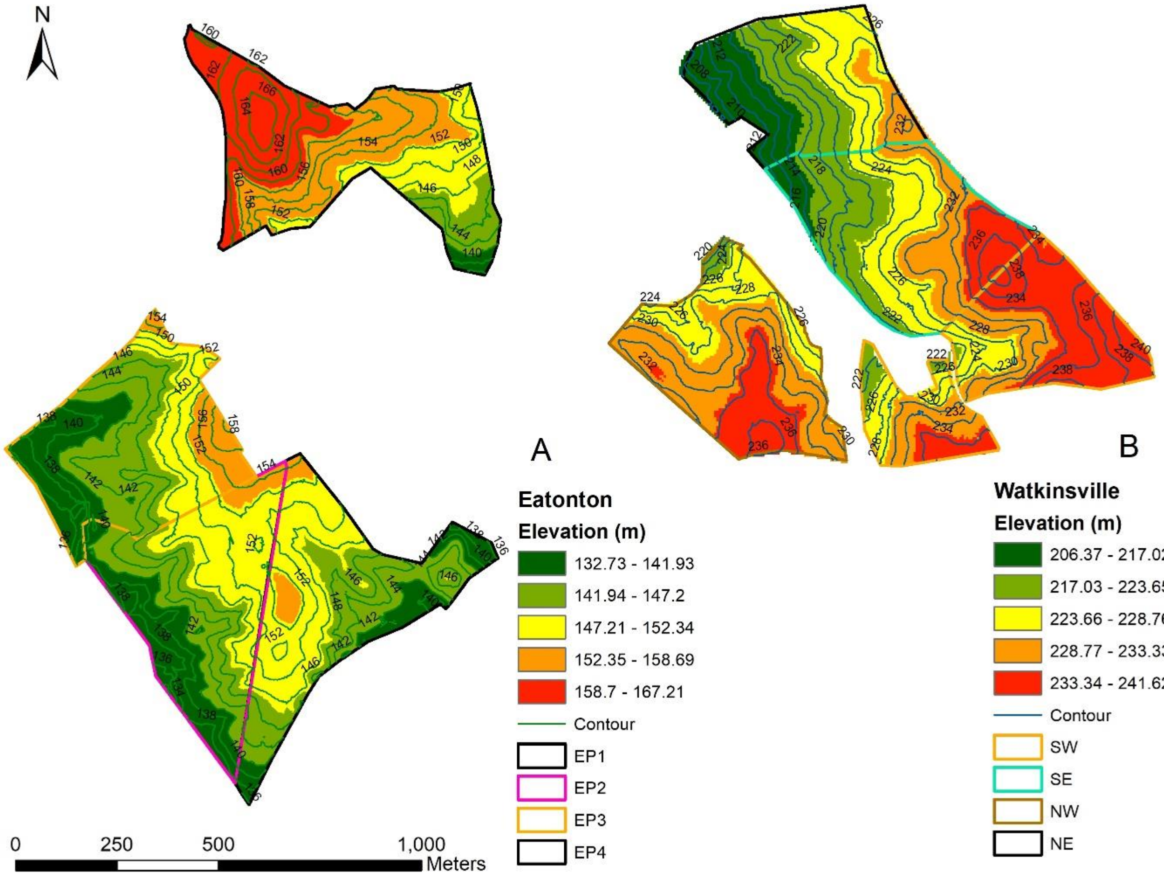
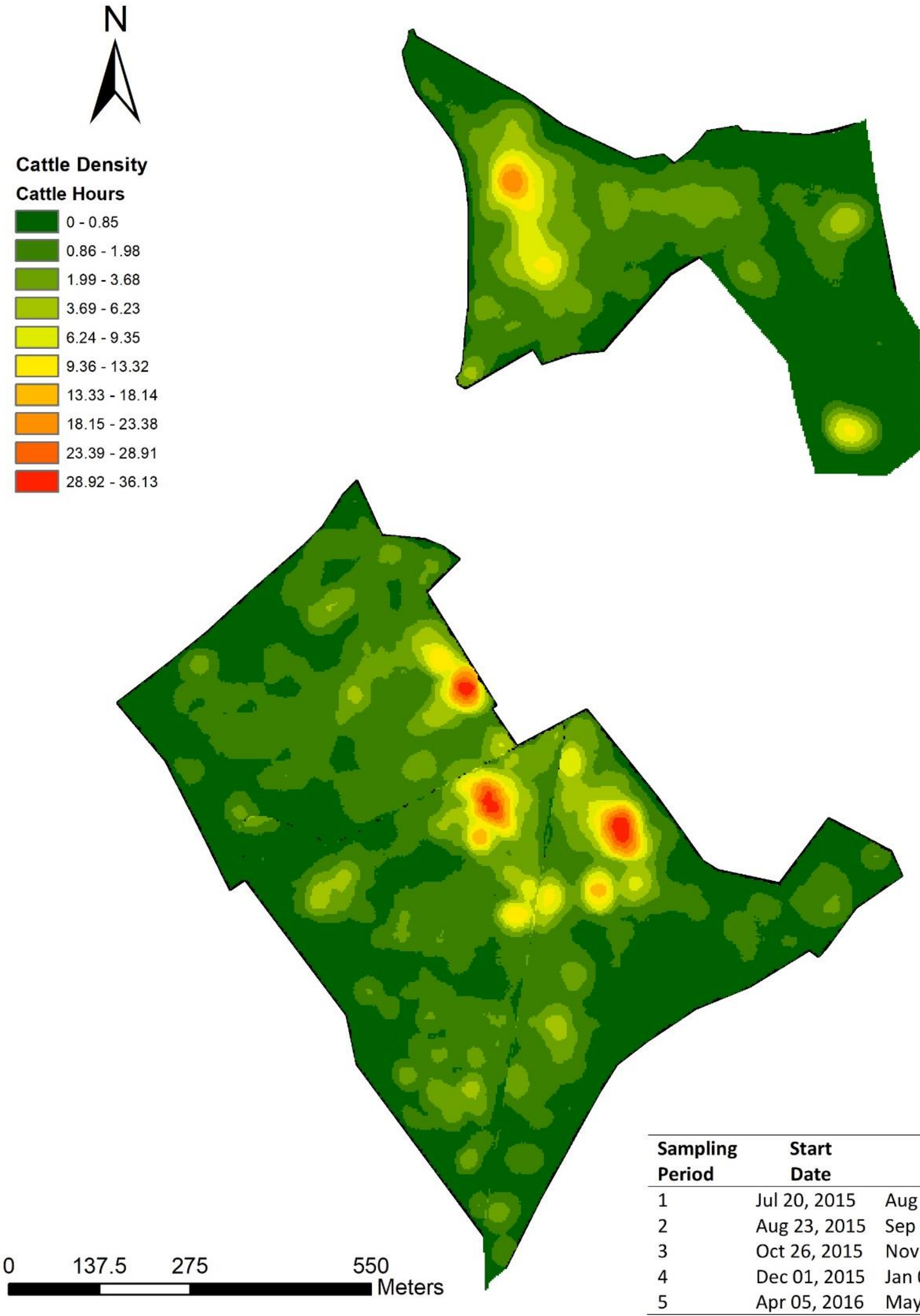


Figure 2.2 Digital Elevation and contour map of (A) Eatonton and (B) Watkinsville.



Sampling Period	Start Date	End Date
1	Jul 20, 2015	Aug 17, 2015
2	Aug 23, 2015	Sep 23, 2015
3	Oct 26, 2015	Nov 23, 2015
4	Dec 01, 2015	Jan 05, 2016
5	Apr 05, 2016	May 04, 2016

Figure 2.3 Cattle density (hours $ha^{-1}yr^{-1}$) while cattle were on the Eatonton pastures and not calving. The map unit is meters and the projection system is NAD 1983 UTM Zone 17 N. The table on the bottom right shows the start day and end day for each measurement period.

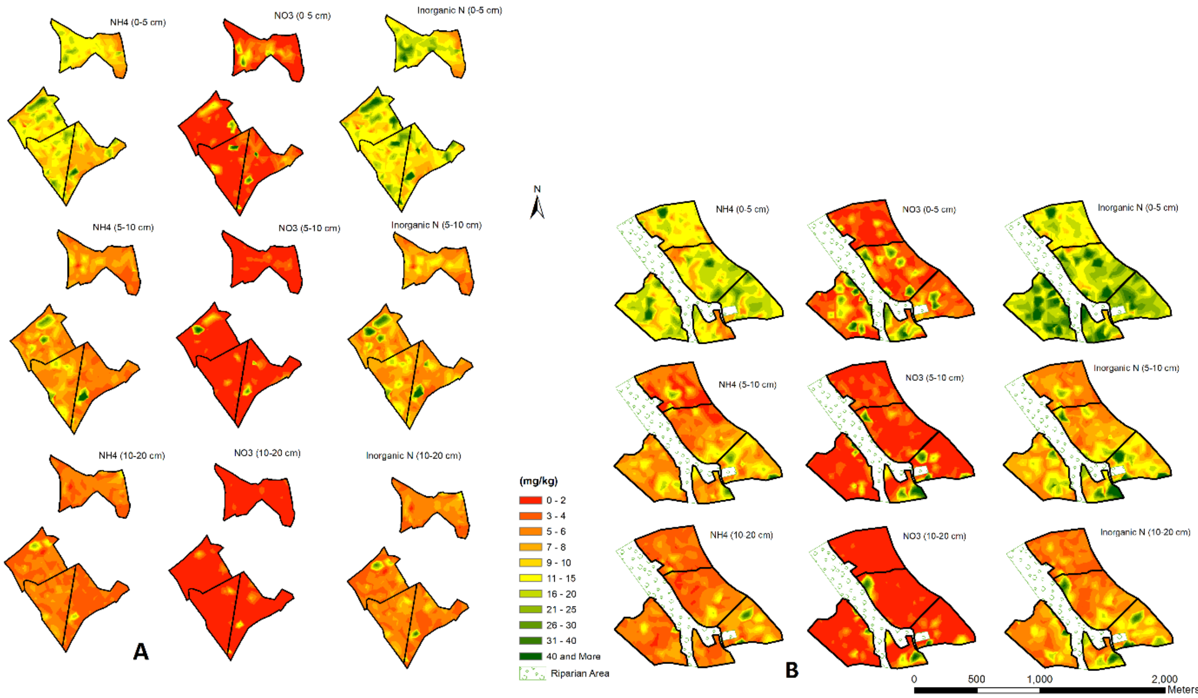


Figure 2.4 Spatial distribution of soil inorganic Nitrogen at three soil depths (0-5, 5-10, and 10-20 cm) NH_4^+. NO_3^-, and inorganic N (the sum of NH_4^+ and NO_3^-) (A) Eatonton and (B) Watkinsville.

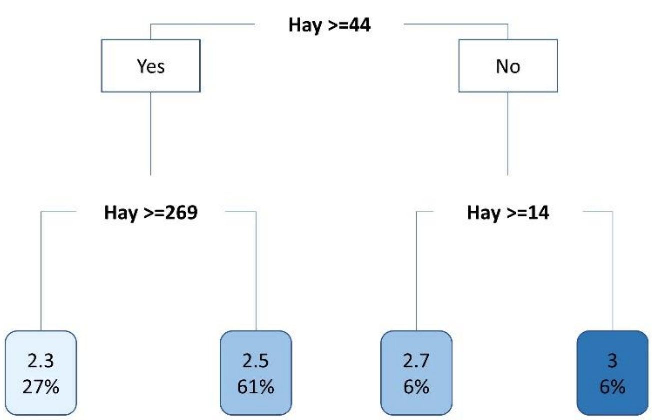


Figure 2.5 Decision tree from recursive partitioning of "Distance to Hay" variable for 0-5 cm depth, in Watkinsville.

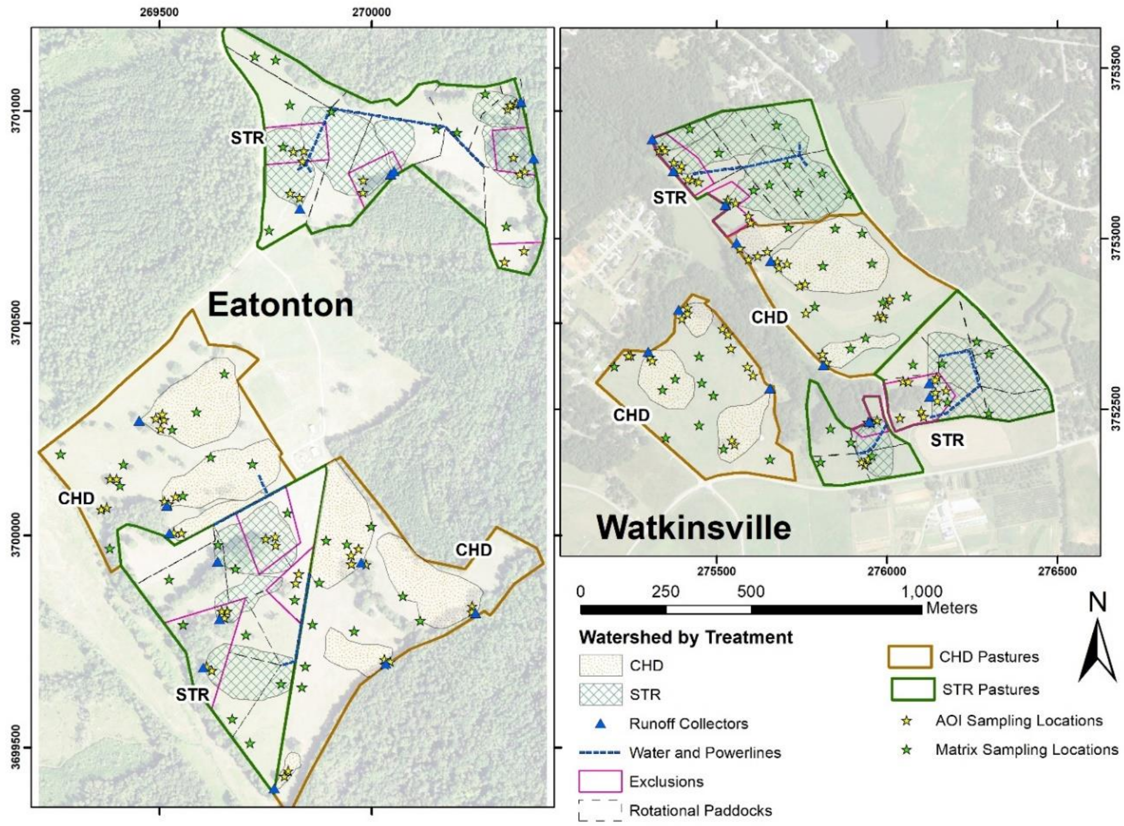


Figure 3.1 Two study sites, (left) Eatonton, and (right) Watkinsville, showing the treatment arrangements, soil sampling locations, water/powerlines, watersheds, runoff collectors, exclusions, and rotational paddock delineation. STR = Strategic-Grazing pastures, CHD = Continuously grazed with hay distribution.

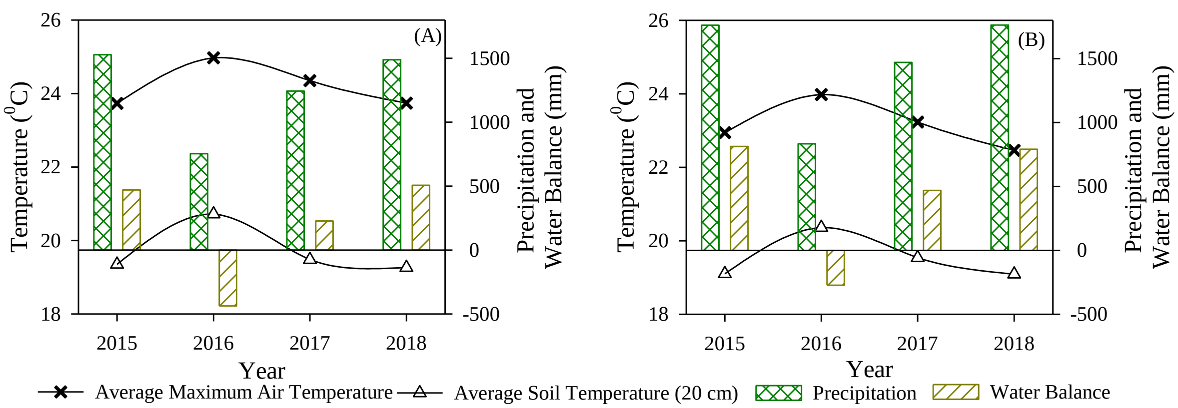


Figure 3.2 Meteorological conditions of (A) Eatonton, and (B) Watkinsville during the study period (2015-2018), showing average annual maximum temperature, average annual soil temperature at 20 cm depth, annual rainfall and annual water balance

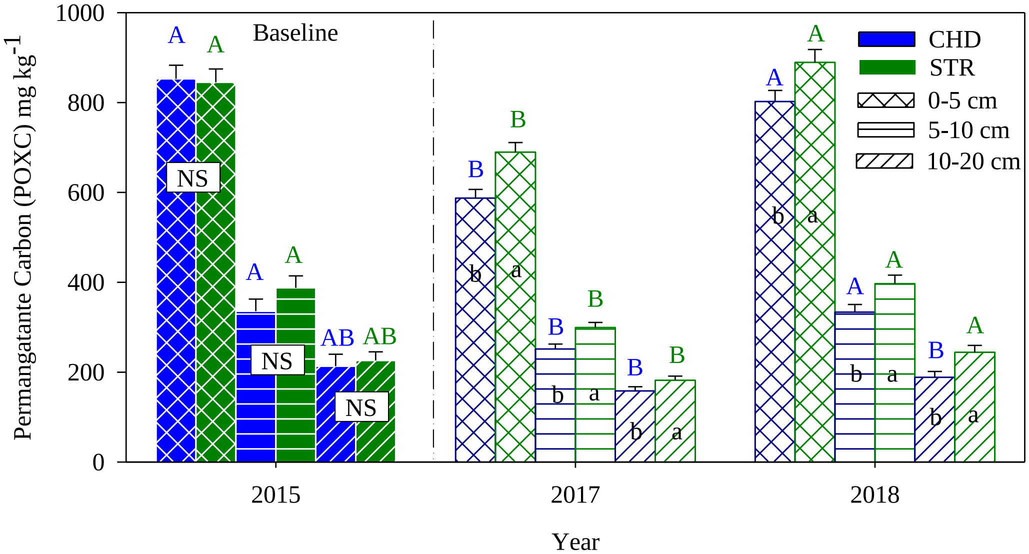


Figure 3.3 Comparison of active carbon (POXC) between treatments and within treatment across years, at three soil depths (0-5, 5-10, and 10-20 cm). The upper-case letters compare the year within treatments, whereas the lower-case letters compare treatments within respective years. Different letters suggest a significant difference between compared groups, whereas, NS indicates no statistical difference at α=0.05. STR = Strategic-Grazing pastures, CHD = Continuously grazed with hay distribution.

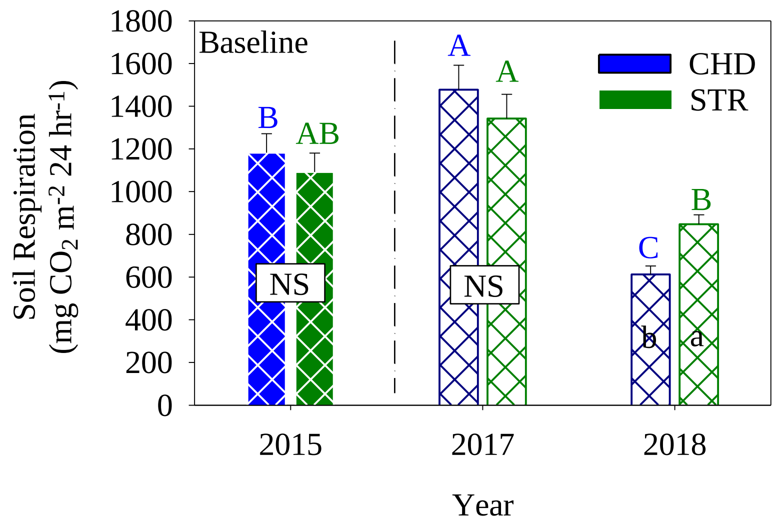


Figure 3.4 Comparison of soil respiration between treatments, and across years within treatments. The upper-case letters compare the years within treatments, whereas the lower-case letters compare treatments within years. Different letters suggest a significant difference between compared groups, whereas, NS indicates no statistical difference at α=0.05. STR = Strategic-Grazing pastures, CHD = Continuously grazed with hay distribution.

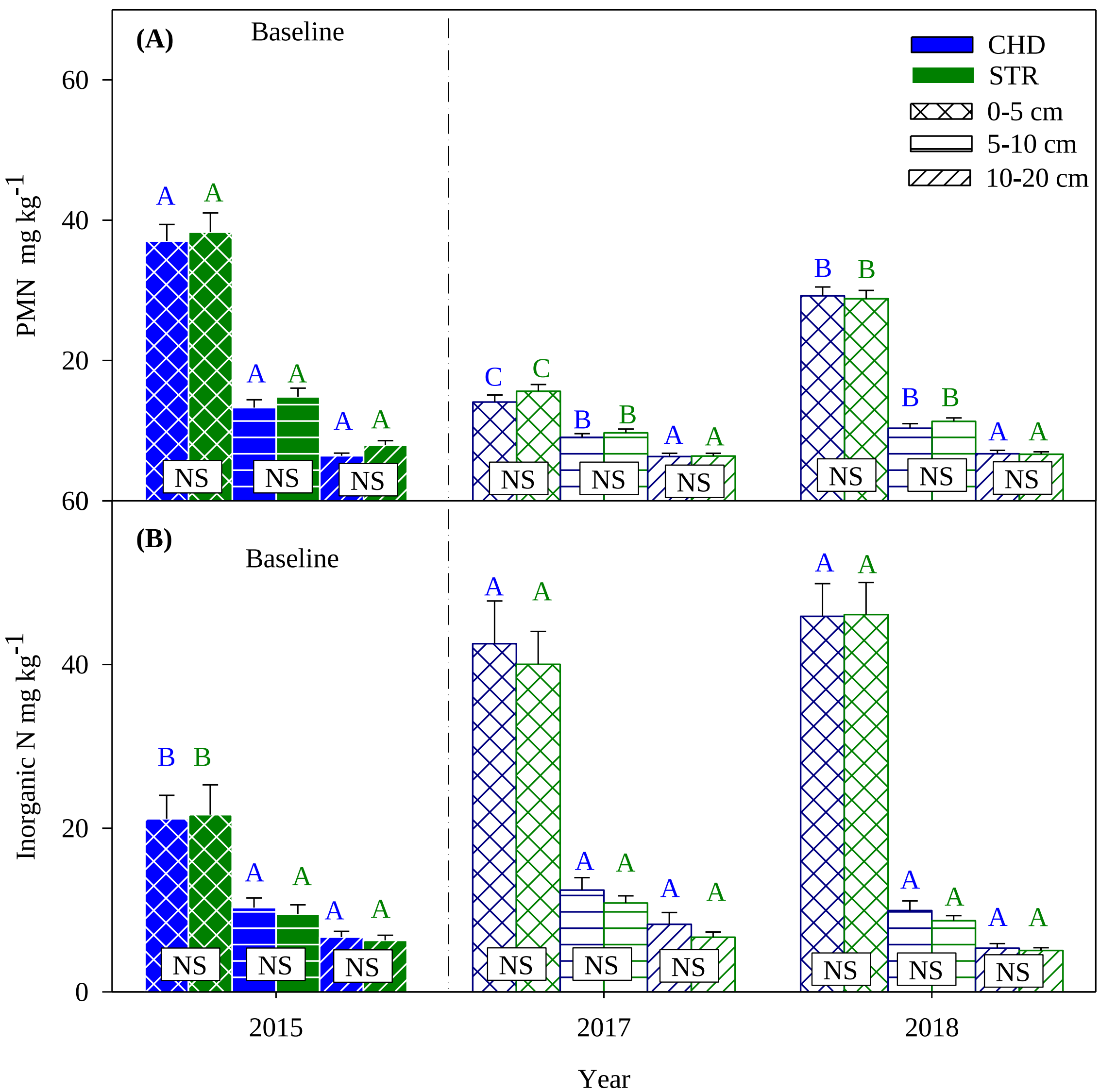


Figure 3.5 Comparison of (A) potentially mineralizable nitrogen, and (B) inorganic nitrogen between treatments and within treatment across years, at three soil depths (0-5, 5-10, and 10-20 cm). The upper case letters compare the year within treatments, whereas the lower-case letters compare treatments across years. Different letters suggest a significant difference between compared groups, whereas, NS indicates no statistical difference at α=0.05. STR = Strategic-Grazing pastures, CHD = Continuously grazed with hay distribution.

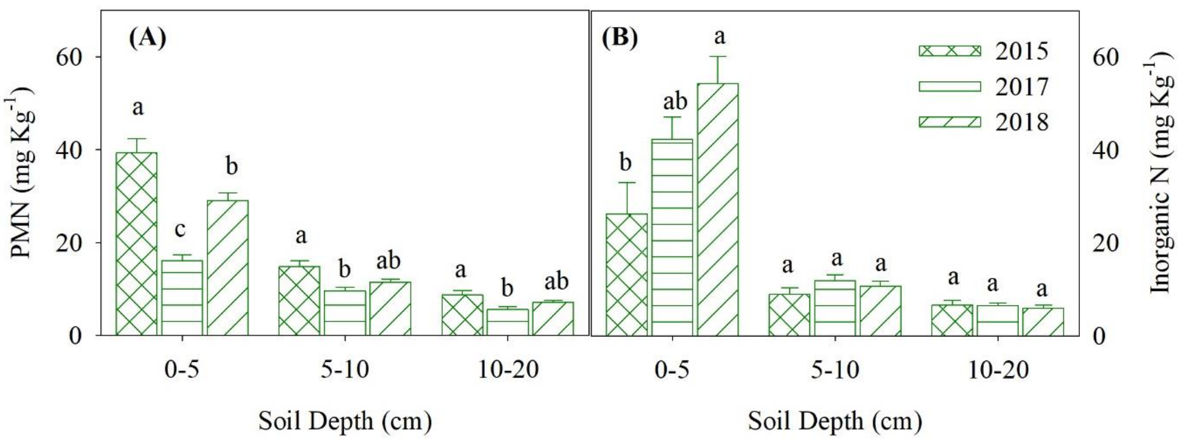


Figure 3.6 Comparison of (A) potentially mineralizable nitrogen, and (B) inorganic nitrogen across years, at three soil depths (0-5, 5-10, and 10-20 cm) in soil samples inside the overseeded exclusions (in STR system). Different lower-case letters suggest a significant difference between compared groups at α=0.05.

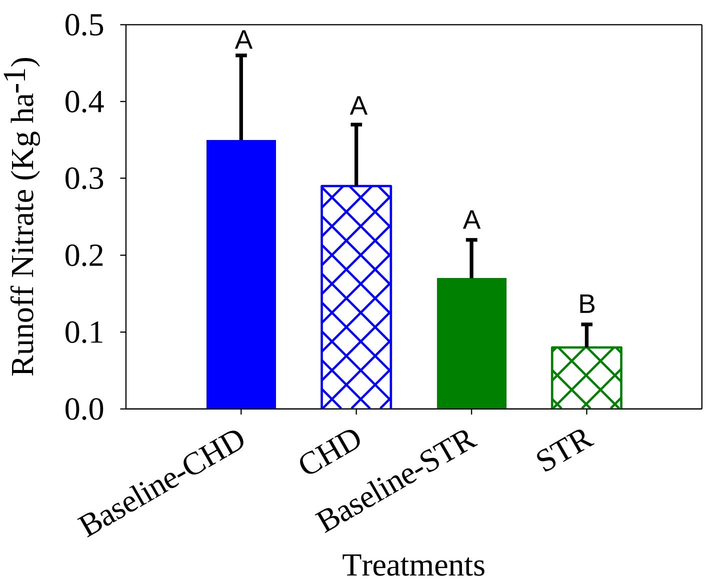


Figure 3.7 Comparison of runoff nitrogen (kg ha $^{-1}$) between treatments, before and after the treatment application. Different letters suggest a significant difference between compared groups at α=0.05. STR = Strategic-Grazing pastures, CHD = Continuously grazed with hay distribution.

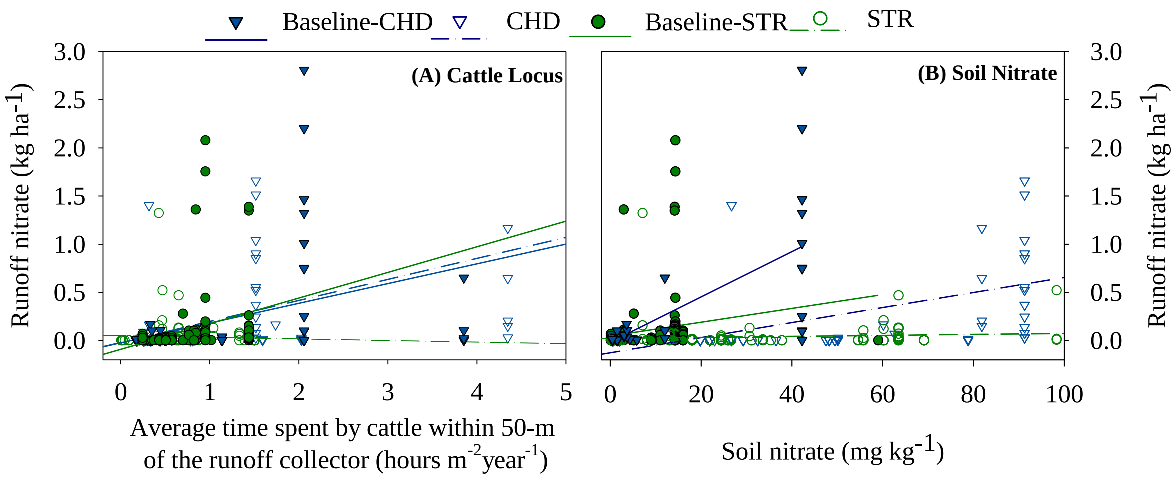


Figure 3.8 Relationship of runoff nitrate with (A) cattle locus, and (B) soil nitrate at 0-5 cm soil depth, across treatments, before and after the treatment application. STR = Strategic-Grazing pastures, CHD = Continuously grazed with hay distribution.

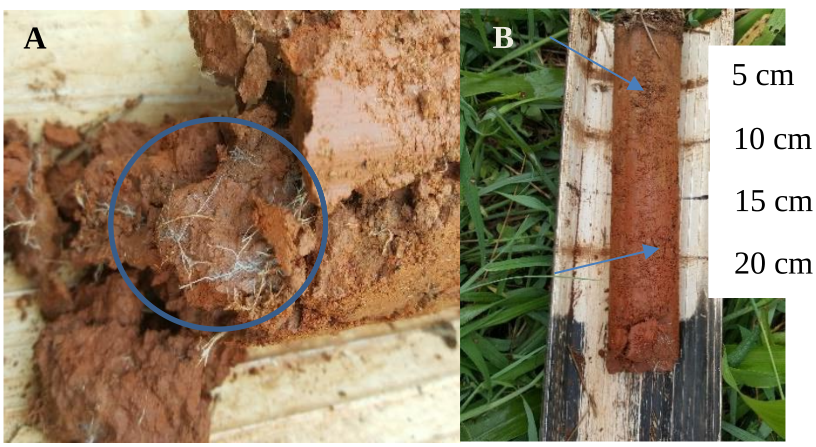


Figure 3.9 A soil core (A) showing root and mycelial growth at 30 cm soil depth and (B) movement of carbon across the soil profile showing carbon breaking in the core.

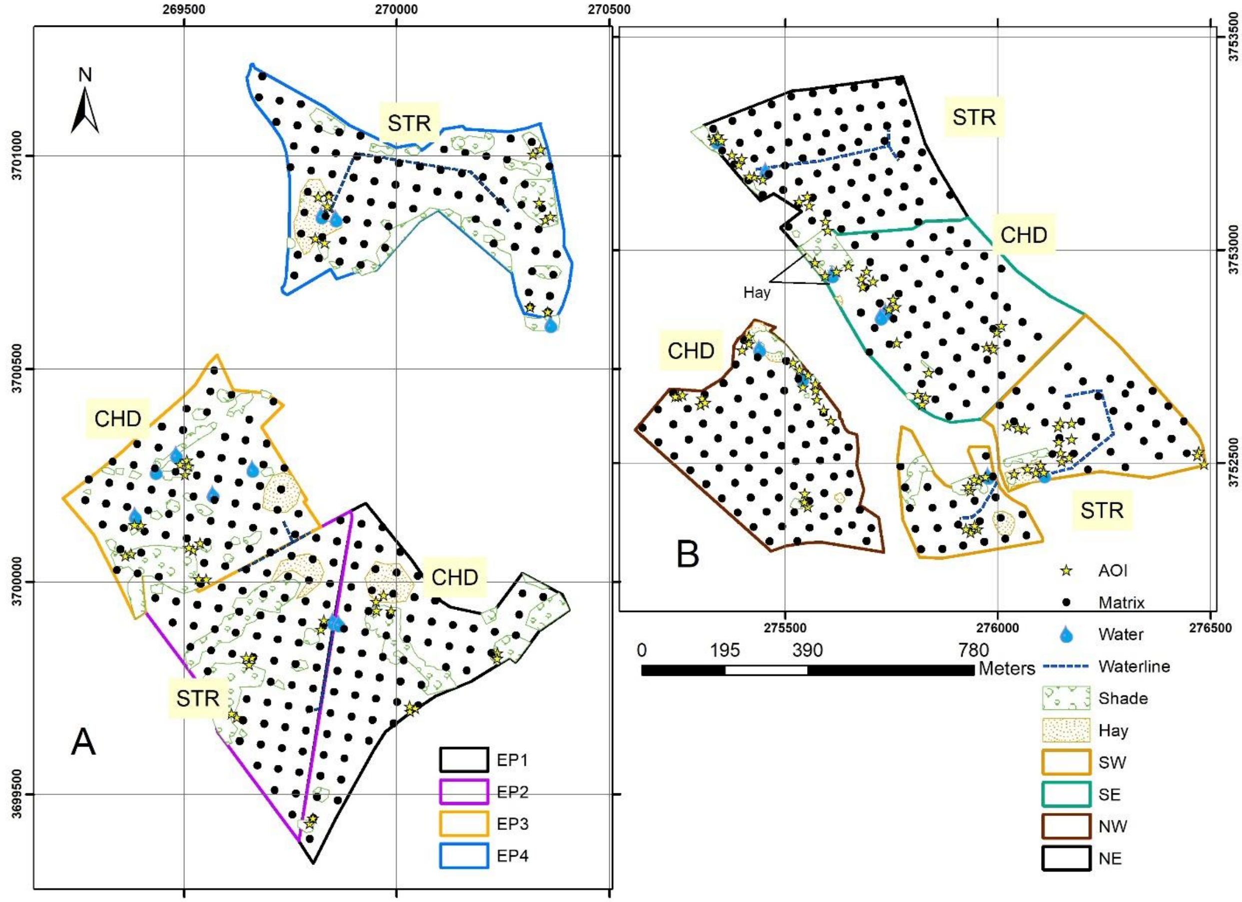


Figure 4.1 Study sites showing the pastures, sampling locations, hay feeding areas, shade, waterer, waterlines, and treatment allocations for (A) Eatonton and (B) Watkinsville pastures. STR = strategically grazed pastures and CHD = continuously grazed with hay distribution. The map unit is meters, and the projection system is NAD 1983 Universal Transverse Mercator Zone 17 N.

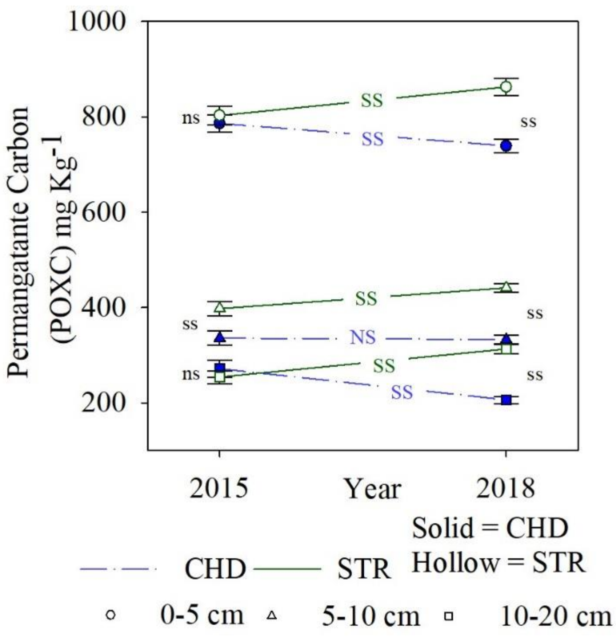


Figure 4.2 Comparison of POXC (mg kg^{-1}) within treatments across years (2015 and 2018). The lowercase ns and ss denote no statistical difference and statistical difference between treatments, respectively, within the year, at α= 0.05. The uppercase NS and SS denote no statistical difference and statistical difference across years, respectively, within treatments, at α= 0.05. STR = Strategic-Grazing, CHD = Continuous grazing with hay distribution.

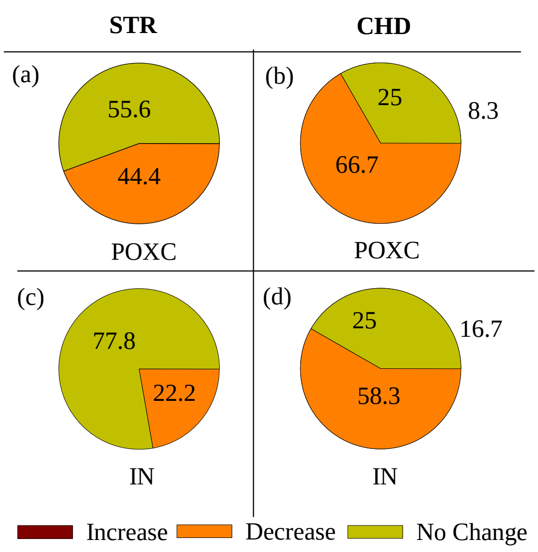


Figure 4.3 Pie charts showing change in spatial autocorrelation (a) STR pastures in POXC, (b) CHD pastures in POXC, (c) STR pastures in IN, and (d) CHD pastures in IN, after the treatment across all three soil depths (0-5, 5-10, and 10-20 cm). Increase indicates the percentage of total instances when spatial autocorrelation in the dataset increased after treatment, Decrease indicates the percentage of total instances when spatial autocorrelation in the dataset decreased, and No Change indicates the percentage of total instances where no increase or decrease in spatial autocorrelation was observed.

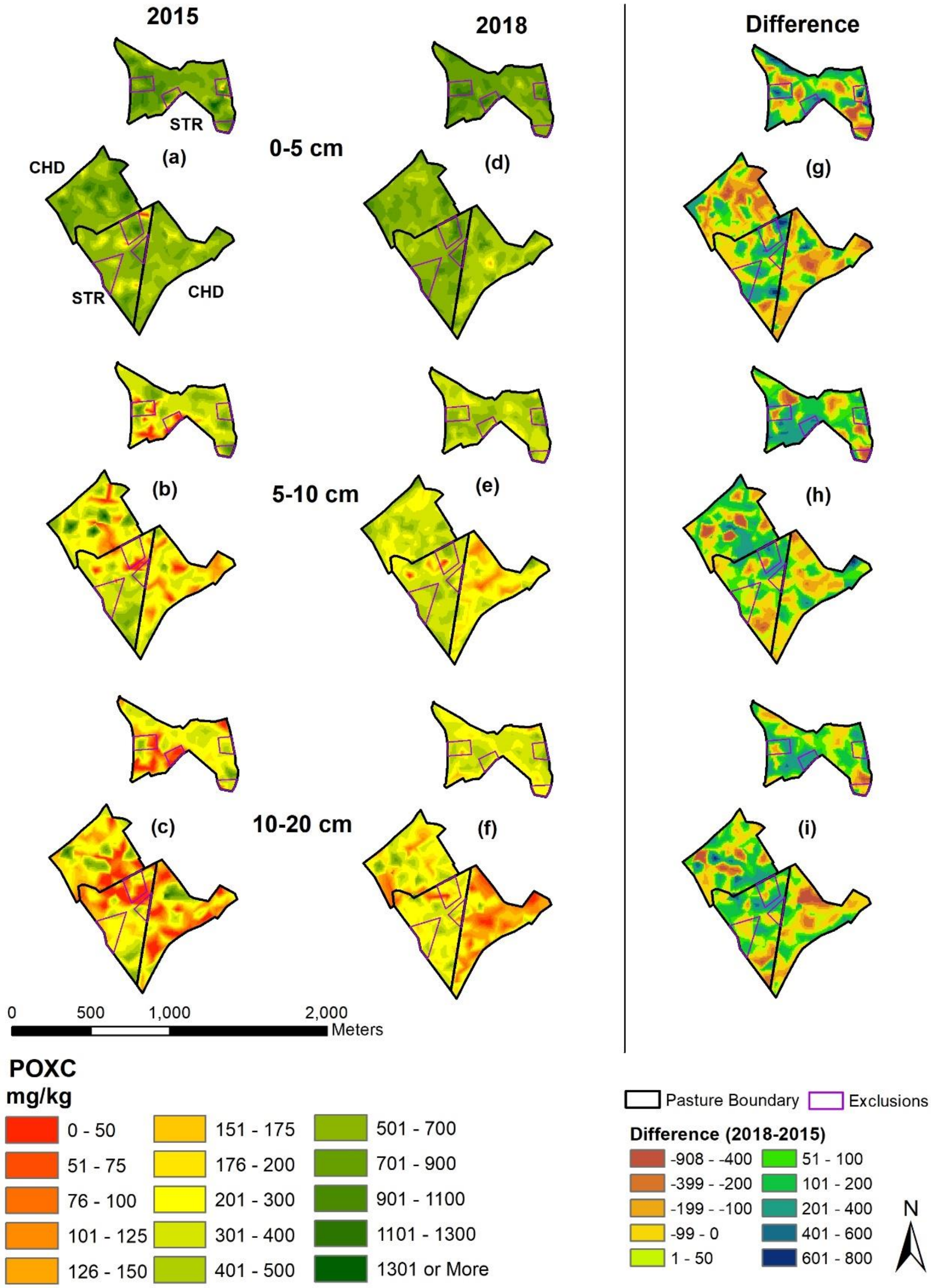


Figure 4.4 Spatial distribution of POXC in Eatonton in the year 2015 at (a) 0-5 cm, (b) 5-10 cm, and (c) 10-20 cm, and in the year 2018 at (d) 0-5 cm, (e) 5-10 cm, and (f) 10-20 cm soil depths. The difference in POXC in Eatonton between year 2018 and 2015 at (g) 0-5 cm, (h) 5-10 cm, and (i) 10-20 cm.

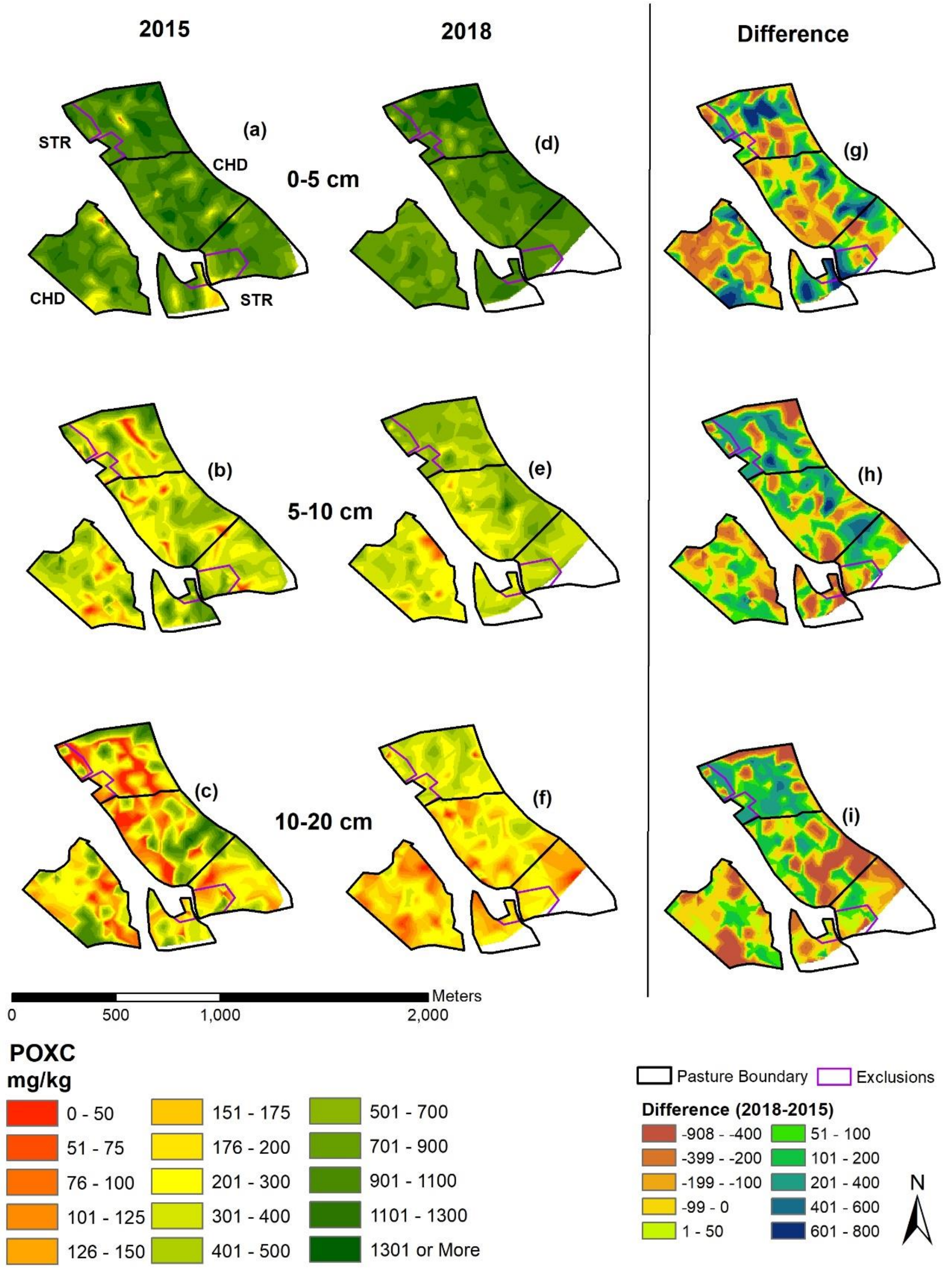


Figure 4.5 Spatial distribution of POXC in Watkinsville in the year 2015 at (a) 0-5 cm, (b) 5-10 cm, and (c) 10-20 cm, and in the year 2018 at (d) 0-5 cm, (e) 5-10 cm, and (f) 10-20 cm soil depths. The difference in POXC in Watkinsville between year 2018 and 2015 at (g) 0-5 cm, (h) 5-10 cm, and (i) 10-20 cm.

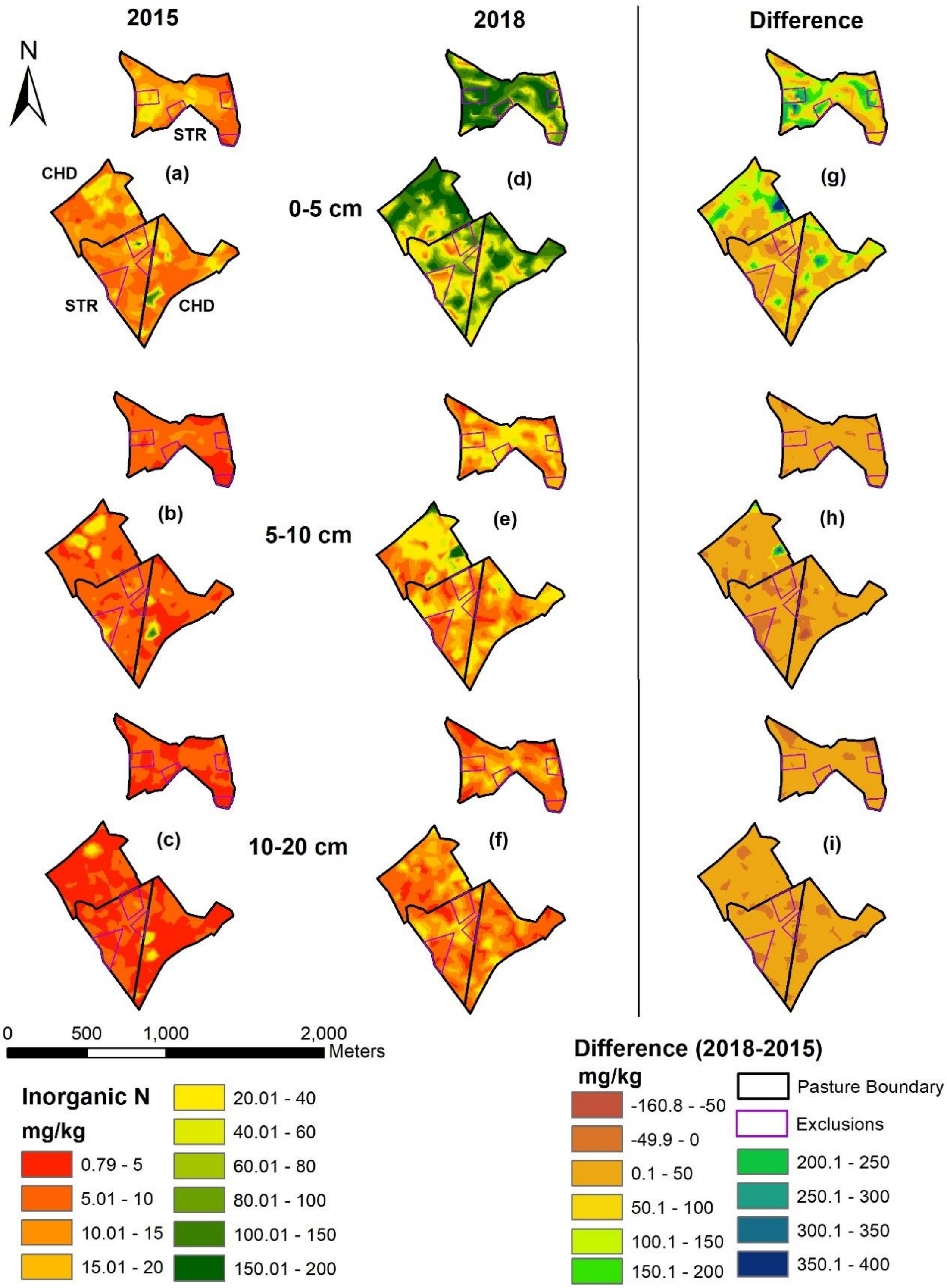


Figure 4.6 Spatial distribution of Inorganic N in Eatonton in the year 2015 at (a) 0-5 cm, (b) 5-10 cm, and (c) 10-20 cm, and in the year 2018 at (d) 0-5 cm, (e) 5-10 cm, and (f) 10-20 cm soil depths. The difference in Inorganic N in Eatonton between year 2018 and 2015 at (g) 0-5 cm, (h) 5-10 cm, and (i) 10-20 cm.

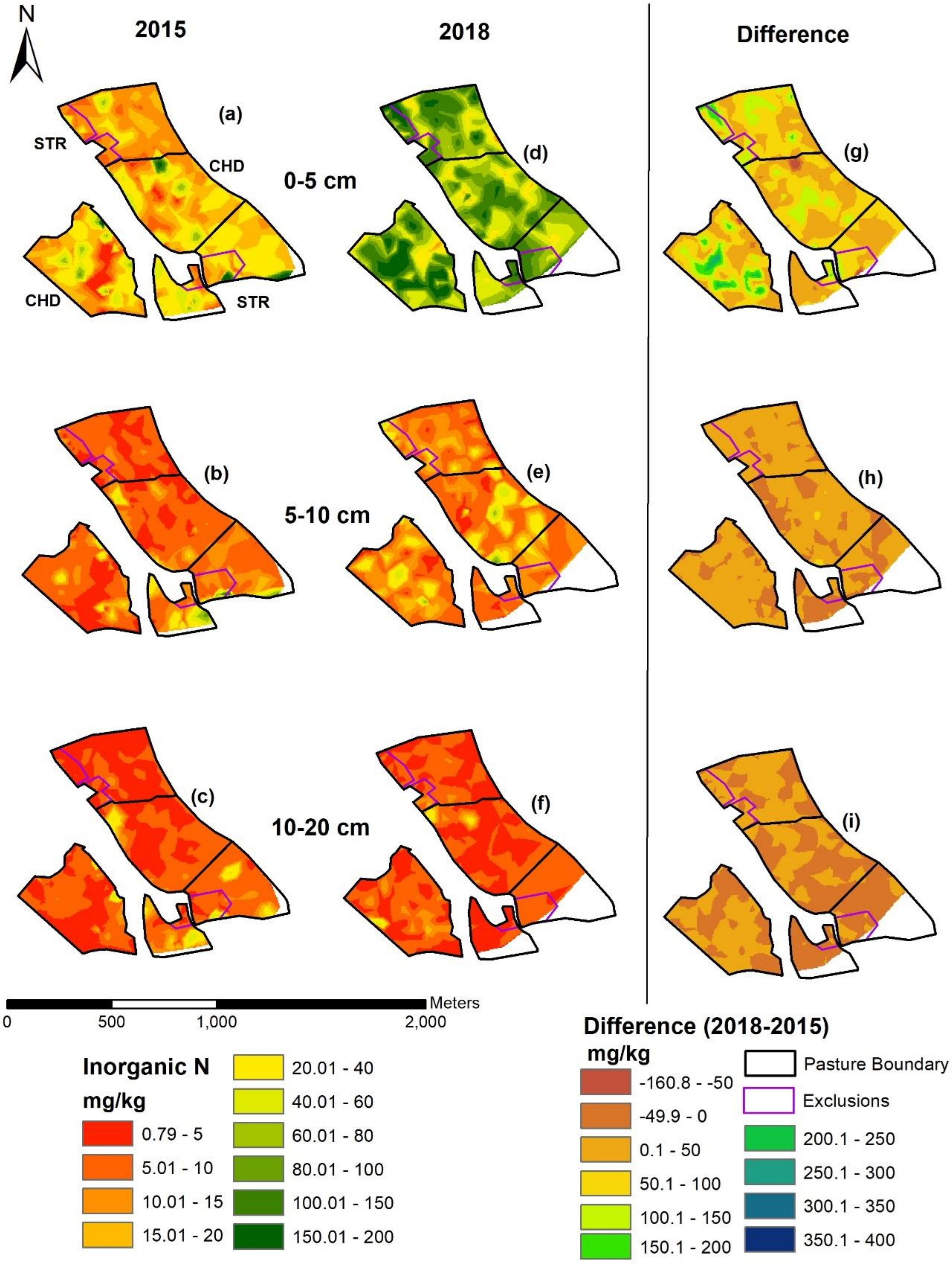


Figure 4.7 Spatial distribution of Inorganic N in Watkinsville in the year 2015 at (a) 0-5 cm, (b) 5-10 cm, and (c) 10-20 cm, and in the year 2018 at (d) 0-5 cm, (e) 5-10 cm, and (f) 10-20 cm soil depths. The difference in Inorganic N in Watkinsville between year 2018 and 2015 at (g) 0-5 cm, (h) 5-10 cm, and (i) 10-20 cm.