ASSESSING SOIL HEALTH AND RUNOFF WATER QUALITY IN SOUTEHRN PIEDMONT GRAZING SYSTEMS

by

TAYLOR HENDRICKS

(Under the Direction of Dorcas H. Franklin)

ABSTRACT

Grazing systems with the ability to retain nutrients are necessary to sustainable agriculture. The objectives of this study were to determine: distribution of C and P in Southern Piedmont pastures; potential for *in situ*, visible and near-infrared (VNIR) to predict soil C as loss-on-ignition (LOI) or permanganate oxidizable C (POXC); and relationships of soil C and P to concentrations of dissolved organic C (DOC) and dissolved reactive phosphorus (DRP) in runoff. Soil samples were collected from 10 pastures, analyzed for VNIR *in situ*, and BD, LOI, POXC, and soil P in the laboratory. Runoff samples were analyzed for DOC and DRP and relationships between soil C and P were determined. When soil variables were measured at different distances from pasture equipage, various trends were found. The use of VNIR reflectance spectra will need further investigation. Runoff concentrations of DOC and DRP showed linear trends to LOI and soil P, respectively.

INDEX WORDS: Grazing systems, Sustainable agriculture, Soil carbon, Soil phosphorus, Loss-on-ignition carbon, Runoff Quality

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Grazing Importance in Agriculture and Management Tactics

Food production, especially cattle production, is an integral part of the southeastern United States' culture and economy. Approximately 31 million head of cattle (USDA, 2013) are harvested annually which account for $76 billion in cash receipts each year (USDA, 2013). Because they make up such a large portion of the agricultural infrastructure, cattle grazing systems also provide a significant source of nutrients to the environment. With the onset of major global climate changes such as increased extreme weather events and incidence of drought due to increased carbon dioxide emissions, U.S. agricultural systems need to become more sustainable, and researchers are focusing on increasing clean water and minimizing nutrient leaks.

In order to support the beef cattle industry, over 215 million hectares (~27%) of U.S. land is designated as grazing land; 5.5 million of these hectares are in the Southeast (USDA, 2013). There are two main categories to denote different types of grazing land: grassland pasture or rangeland, and forest-use grazed land (USDA, 2013). Grassland pasture and rangeland are grouped together because their vegetative diversities make distinction between the two difficult. Together they make up approximately 74% of U.S. grazing lands and are defined by the USDA (2002) as follows:

> "Grassland pasture and range comprise all open land used primarily for pasture and grazing, including shrub and brushland types of pasture, grazing land with sagebrush and scattered mesquite, and all tame and native grasses, legumes, and other forage used for pasture or grazing…."

Additionally, the grassland category may be applied to those pasture areas that can be transition areas to grazed forest land. Estimates of the acreage of these areas are created by the USDA Agriculture Census, Bureau of Land Management, Forest Service, and estimates of private land ownership.

Because grazing uses such large amounts of land, any nutrient losses can quickly compound to create widespread environmental issues including poor ground and surface water quality. Grazing cattle are usually managed using either a continuous or a traditional rotational grazed system (Butler, 2010). If poorly managed, continuously grazed systems have the potential to cause poor pasture conditions, low nutrient efficiency, and poor environmental quality of the surrounding area (Butler, 2010). In turn, this can negatively influence the production system overall by reducing forage quality and productivity, increasing the need for feed supplementation, and unnecessary loss of nutrients thereby reducing the productivity, profit, and sustainability of the entire system.

As with any other business, in order to make a grazing system profitable it is necessary to maximize the amount of product (pounds or animal units) produced on the lowest amount of financial input possible. In a grazing system, profitability is dependent on the producer's ability to grow cattle using planted forages rather than purchased hay or grain crops. In order to achieve a profitable forage system, there are several smaller goals a producer must strive to meet. These goals include, but are not limited to: focusing on efficient forage use; minimizing the use of stored feed (i.e., reducing the number of hay feeding days); reducing feeding losses; maximizing nutritional quality of available forages; feeding to match animal nutrient requirements; minimizing overall forage costs; and maximizing potential animal productivity (Georgia Forages, 2014).

Implementing a rotational stocking system or other alternative Best Management Practices (BMPs) has been shown to enhance vegetation cover; reduce soil bulk density; increase soil organic matter and aggregate stability; and enhance watershed quality. These systems, however, are often very time-consuming and expensive to implement (Osmond, 2007; Pluhar, 1987). Reducing cost and time barriers could make farmers more likely to alter management practices to improve the overall system quality – including profitability and environmental health.

Through massive efforts by extension and government agencies as well as widespread cost-share subsidies, the implementation of BMPs such as buffer strips and stream exclusions have had some success. However, the adoption of new management practices has not been widespread due to concerns about intensive time and cost inputs. It is possible that if new management systems reduce the amount of time and labor farmers must put into implementing the management strategy then these new management methods will be widely adopted.

Several strategies for improving the efficiency of grazing systems can be employed based on the needs of the production system and stage of the system. For example, a cow-calf system that relies on increasing the average daily gain (ADG) of its calves prior to weaning would have different needs, and therefore employ different management strategies, than a grazing dairy whose main focus is maximizing milk production. While general management recommendations can be made easily, determining the specific needs of a producer is not so simple; management decisions must be made with careful consideration of the needs of the specific system in which they would be implemented. Additionally, implementation barriers such as cost and time inputs must also be taken into account.

If a producer is unable to change from a continuously stocked to rotationally-stocked system, there are several management strategies that can be used to improve a continuous system. These improvements can be used to achieve main production goals (increase high-quality forage availability, reduce hay feeding days per year, and improve production and profitability) while enhancing pasture condition to reduce nutrient losses. Pasture improvements can include more evenly distributed nutrients across the pasture, increased rainfall infiltration rates, and reduced nutrient concentrations and incidence of storm runoff. Some techniques that can be added to a continuous stocking system to facilitate these pasture improvements include: stream exclusions, lure management, and the use of exclusionary areas. These techniques will increase time cattle spend grazing the pasture as a whole while also reducing the amount of time cattle spend near the stream, in concentrated flow areas, and at field edges on the downhill side of pastures.

Grazing cattle in systems with unrestricted access to areas of concentrated flow of runoff water are likely to cause surface water contamination of bacteria and nutrients, as well as damage to the stability of the stream banks and riparian areas. This damage is caused, in part, by increased cattle activity, or "camping," in riparian areas bordering the stream or in shady, low-lying areas often on pasture edges. Cattle often gather in these areas because of the easy access to drinking water and cooler, shady locations during warm weather months (Byers, 2005). This increased activity often leads to decreased vegetative cover and increased soil compaction. That combination, in turn, reduces the ability of the pasture to capture and recycle nutrients and increases the amount of runoff, sediment, and nutrients that enter the surface water (Line, 2000) and are loss to production.

While the unnecessary loss of nutrients hurts the producers' profitability, the entry of these nutrients and other elements can cause a variety of environmental and human health issues. An influx of phosphorus (and, to some extent, nitrogen) from fecal deposits close to or in streams increase the concentration of nutrients that may contribute to algae growth. Sedimentation in stream water is another environmental concern because it can inhibit aquatic vegetation growth and reduce habitat for aquatic creatures (Byers, 2005). Finally, fecal deposits near or in streams can pose a serious human and animal health risk. High levels of fecal coliforms (e.g., *Escherichia coli*) can cause serious gastrointestinal illnesses in humans, (Byers, 2005) and high levels of bacteria can pose a health risk to the cattle raised on the pasture system as well. Therefore, incorporating stream exclusions as a BMP has been a main focus of management and conservation efforts for many years.

Several studies support the use of fencing exclusions to reduce sediment and nutrient concentrations in stream waters. In a 13-year study of a continuously stocked system, the use of stream exclusions alone reduced sediment inputs from a beef cattle pasture by 50% and overall soil losses from the pasture by 40% (Owens et al. 1996). Line et al. (2000) evaluated the impact of stream exclusions using a 335-m long, 16-m wide section of stream and riparian zone that was fenced to exclude dairy cattle and heifers. The riparian zone was also planted with hard and softwood trees and the creek bank was renovated and reseeded to a healthy state prior to the study. Water quality was monitored for a three-year period and soil samples were collected from the riparian zone before and after this period. Analysis showed that this riparian exclusion reduced total phosphorus and total suspended sediments (TP and TSS, respectively) in streams following a rainfall event by 75.6 and 82.3%, respectively. Although the numbers were not significant (in part due to small sampling sizes), the soil bulk densities decreased and hydraulic conductivities increased, which indicates an increase in infiltration rate (Line et al., 2000). Additional dairy pasture studies showed that the use of stream fencing practices could reduce suspended sediments by 60.2% and some strains of bacteria (fecal coliform and enterococci) by 65.9 and 57%, respectively Line (2003).

Although stream bank fencing and riparian zone exclusion have been shown as effective means to reduce nutrient inputs to streams and improve water quality in some instances, many landscapes such as those found in the Southern Piedmont have incised channelized flow paths that would require extensive costs to rework the riparian areas and many producers are unwilling or unable to implement such practices. Major barriers include the initial startup cost as well as a long-term increase in production input cost because of the need for an alternative water source. The combination of these expenses make it somewhat unappealing unless cost-share programs are available or benefits are guaranteed.

An effective alternative to stream bank exclusions may be the use of lure management techniques. Lure management techniques can be as simple or complex as desired based on the number of elements used as lures. Common cattle lures can include: water troughs, natural or portable shades, feed/hay, and mineral sources. The use of a water trough-only lure may be used in order to reduce the amount of time cattle spend camping in riparian areas or in streams. It also provides cattle with a clean water source, rather than one with high nutrient and bacteria concentrations. Studies have shown that when provided with a water trough, cattle preferred to drink out of the trough instead of a surface water source (pond or stream) up to 92% of the time (Sheffield, 1997; Byers, 2005). Additionally, the use of water troughs alone (not in conjunction with stream fencing) reduced total suspended sediments (TSS) and total P by 90 and 81%, respectively (Sheffield, 1997).

Further study was conducted by Oregon State University (Porath et al., 2002) to determine the effects of using a combination of lure management strategies. This study investigated the use of a combination of water troughs and salt-mineral feeders to prevent cattle loafing in streams when no exclusion fencing was used. Cattle activities were monitored through visual observation, fecal distribution, and cow/calf performance. Monitoring showed that both groups of cattle – those with and without an alternate water and mineral source – began the day grazing in riparian zones nearby the stream. However, during the afternoon/non-grazing period, the cattle that had access to an alternate water source spent more time at the water troughs than in the riparian zone. It was calculated that cattle with alternate water access used the troughs for 45% of their daily water intake (Porath et al., 2002).

Finally, cattle with access to non-stream water drinking sources showed improved productivity; cows and calves with off-stream water and mineral sources gained an average of 0.27 and 0.14 kg/d, respectively, more than animals whose only water source was the stream. While the off-stream mineral source may have contributed to increased weight gain, the researchers attributed the weight gain to a more uniform grazing pattern which would allow for more forage regrowth and less patch grazing for improved grazing efficiency (Porath et al., 2002).

In some cases, however, the use of alternative water troughs is not a complete solution to mitigate cattle loafing in riparian areas or streams. In pastures where a high percentage of endophyte-infected tall fescue is present and the only available shade occurs in riparian zones near the stream, there are mixed behavioral observations. Franklin et al. (2009) showed that in Georgia pastures where endophyte-infected tall fescue was present, the time cattle spent in streams or riparian borders was significantly reduced on an annual basis, however, when the temperature and humidity index (THI) was very high (THI = 72-84) cattle loafed in riparian areas and streams regardless of the presence of available water sources in other areas of the pasture. Sheffield et al. (1997) conducted work on infected tall fescue paddocks in Virginia and still observed a 51% reduction of cattle activity in streams or riparian activity. Sheffield et al. (1997) found a reduction but they also acknowledged that previous research demonstrated that providing off-stream water sources may not always show the same reduction of in-stream loafing in pastures with endophyte-infected tall fescue. It is possible that the use of portable shade sources to provide cattle with shade away from riparian zones and areas vulnerable to nutrient loss may play a contributing role in luring cattle away from riparian zones and to areas within the pastures that could benefit from nutrients deposited by cattle.

Exclusion areas may also be used to improve environmental quality and provide a high-quality forage supplement. Exclusion areas using temporary fencing can be placed in areas of the field where long-term grazing is not ideal. These areas could be at the edge of the field, bottom of a hill, near a stream, or any other place where cattle might usually camp and cause large volumes of nutrient runoff. Planting high quality forage or a forage-legume mix in these exclusion areas will increase the available nutrients for grazing. These areas may then be grazed as higher quality forages are needed for supplementation or as they become available to optimize forage availability.

Where exclusions are used, mob stocking tactics can also be used. Mob stocking, which is sometimes referred to as Ultra-High Stock Density (UHSD) grazing or short-term flash grazing, is a grazing method that allows a large number of animals to graze a small area paddock for only a few hours at a time. Stocking densities in a mob stocking system are typically 56,000 to 84,000 kg livestock/ ha (50,000-75,000 lb/A) (NRCS, 2011). The goal of an ultra-high stock density system is to use animal behavior to mix fecal deposits, plant material, and soil in order to increase organic matter composition and improve soil aeration and mixing. Mob stocking is often conducted in an exclusionary area that has been over-seeded with nutrient dense forage. This grazing tactic has been shown to improve nutrient distribution, weed control, soil quality, uniform forage utilization, and overall forage biomass and quality (Paine, 1999).

In these exclusions, the animals are grazed at high density for a short period of time (four to eight hours) then moved to a regular pasture, often with a lower quality forage. The use of exclusions in this system is similar to a limited, or timed, grazing technique. Limit grazing is used to stretch the quantity of high quality forage and reduce the number of hay feeding days that may be necessary. In limit grazing, a high quality pasture is reserved and cattle are only allowed to graze in this pasture for short durations before being moved to their regular, lower forage quality pasture. Therefore, animals receive the nutrients required for maintenance or production without depleting all available forages or being fed a hay supplement (Ball, 2007).

Using exclusion areas can be an effective means of supplementing cattle without hay feeding and can reduce long-term hoof traffic to environmentally sensitive areas, but the use of exclusions can be quite management intensive. In order to preserve forage availability, cattle must be moved frequently from the exclusion areas to other pasture to prevent over-grazing and support regrowth potentials. Additionally, if mob stocking is used, the potential for pugging is increased. Pugging, or hoof damage, can reduce forage growth beyond the current grazing season and actually reverse any potential soil health benefits that may have occurred from the mob stocking (Ball, 2007).

Finally, if exclusions are used as a means to improve environmental health alone and are not managed to also improve cattle gains, it could become an area of economic loss. The success of an exclusion system lies in the ability of a producer to manage these areas to effectively promote sustainable animal production; this involves providing water, shade and hay in areas that will benefit from nutrient enrichment, choosing high quality forages and closely monitoring cattle activities to prevent overgrazing or soil damage, and improving forage quality for optimum cattle productivity. In a small operation, exclusionary areas alone may not be practical because of lost pasture acreage and increased labor inputs to the farmer.

Instead of using a single management method to supplement a continuous stocking system, the use of multiple grazing strategies can be used to overhaul a grazing production plan. Using multiple strategies can aid farmers in choosing a production method that incorporates environmental benefits and production needs with minimal additional inputs. Strategically, farmers can employ two or more of these grazing methods at a time in, what is herein termed, tactical grazing. Tactical grazing is a grazing system designed to effectively manage the movements of grazing animals with minimal time inputs by the farmer by combining several best management practices that have been scientifically shown to reduce environmental impacts and/or improve animal or system production. Because the use of a tactical grazing system involves a multi-management approach where each piece is dependent on other facets of management, a transformational shift away from a continuously grazed production plan would be required rather than an additive or incremental approach.

Tactical grazing incorporates the practices of excluding vulnerable source areas (areas vulnerable to erosion and nutrient losses) from continuous stocking and limited use of excluded areas by short-term mob stocking, as well as the use of rotational stocking and the strategic placement of water troughs, portable shade, and other attractants to lure livestock away from vulnerable source areas and to more uniformly distribute nutrients across the pasture. Vulnerable source areas are areas that have high potentials for erosion and nutrient losses; they can be cattle camping areas and/or areas where highly concentrated volumes of water flow through the field following precipitation. The goals of a tactical grazing system are to: (1) reduce environmental impacts, including but not limited to: improving bulk density thereby reducing incidence of runoff from pastures; reducing nutrient and sediment concentrations; and improving runoff quality; (2) improving soil health by increasing overall soil organic matter and allowing improved nutrient availability; (3) increasing available forage and overall forage quality; (4) improving animal production through the use of high quality forage supplements; (5) reducing the need for additional fertilizer application to pastures by creating a uniform nutrient distribution throughout the pasture and eliminating nutrient hot spots or deficient zones; (6) reducing time and economic inputs to the producer by reducing the amount of time used to manually move animals and provide feed supplements; and (7) improve overall producer quality of life.

A main emphasis in the tactical grazing system is the use of short-term, mob stocking in exclusion areas. Like other exclusion systems, these areas will be over-seeded with an appropriate forage-legume mix in order to improve the nutritional value and palatability of the area. To prevent an economic loss due to acreage losses, the total area per pasture dedicated to exclusions will be approximately 15% of the pasture. Exclusions can then be subdivided to create small paddocks. These areas can be mob stocked as forage targets for each are achieved.

Because the short-term mob stocking technique is often considered a “management intensive grazing” practice, the tactical grazing system incorporates lure management techniques to reduce time inputs to farmers. Lure practices include the use of movable shades, water sources, and hay sources placed in desired camping locations. As previous research into alternate water sources has shown, cattle prefer to drink out of fresh water troughs rather than streams and will most likely “camp” in areas where water sources and shade are available (Sheffield, 1997 and Franklin et al., 2009). It is possible that cattle may react similarly to shade sources during warm weather months and to hay or mineral sources or feed troughs during cold weather when additional energy may be needed for body maintenance.

By placing desirables away from the mob stocked exclusion areas, cattle will likely consume a sufficient amount of the forage-legume mix, then move away from the vulnerable exclusion areas across the pasture towards water or shade on their own. While they move, the manure deposits will be more uniform across the pasture and will act as a fertilizer to areas that may not have otherwise received nutrients (without a farmer-spread source). Additionally, the amount of fecal deposits in areas susceptible to loss will be reduced and the number and severity of nutrient “hot-spots” will likely be reduced. Compaction in these areas will also be lessened (BD will be reduced), while cattle movement may also more effectively redistribute soil, nutrients, and organic matter to other parts of the pasture.

The incorporation of lure management can reduce producer time inputs because the producers will no longer need to physically move cattle as intensively or as often as they would in a moderate or intensive rotational grazing system, however production may be increased by using a combination of both lure management and a less intense rotational stocking. Producers can simply close off the exclusion area after the mob stocking period using hotwire fencing. Lures can be repositioned on a more convenient basis to shift cattle camping throughout the field rather than physically moving cattle each day.

If the tactical grazing system is used and managed appropriately, there are many benefits to its implementation. Tactical grazing can be considered a holistic management strategy because each component of the plan – exclusions, mob stocking, rotational stocking, and lure management – works in conjunction to build off the others. Using multiple management strategies makes each individual technique more effective. Tactical grazing connects each production and conservation goal to improve forage quality, animal production, and environmental and soil health.

Unfortunately, tactical grazing systems are not without their obstacles. Because each factor plays an important role and has a cumulative effect on the success of the system, it is more challenging to implement individual portions of the system or introduce just one tactic at a time. Because the tactical grazing system is a large-scale overhaul, it takes a large-scale commitment from the producer. Each component of the tactical grazing scenario requires significant up-front financial inputs. Fencing, seeding, lure sources (portable water troughs and shades), and the set-up (water lines and electrical sources) are likely needed in higher quantities than a typical producer can access. Therefore, it is key to find a cost-share program to ease the up-front burden and reduce economic risk. Fortunately, most of the components of the system can be employed in a temporary fashion (i.e. hotwire fencing, portable shades), allowing for adjustments as the system grows and develops.

Soil Health

Soil health is defined as “the capacity of a specific kind of soil to function within natural or managed ecosystem boundaries to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation (Sharma et al., 2009).” Soil health is determined by a delicate balance of the overall physical, chemical, and biological properties of the soil (Lemenih, 2004). Soil physical properties are measurable parameters that describe the composition of the soil. These characteristics include: soil texture, soil depth, bulk density, porosity, infiltration rate, and aggregate stability (Gardiner et al., 2008).

Because cattle management systems have the potential to impact pasture conditions, nutrient efficiency, and environmental quality of the surrounding area, the use of BMPs, can significantly change the soil health of an area in a variety of ways (Butler, 2010). An agricultural area, whether cropland or pasture, has the potential to have poor soil health because of overexposure to fertilizer, pesticide, and tillage methods used in the fields, or because of soil disturbances and compaction due to the presence of machinery or livestock. Proper management techniques and the effective use of BMPs in agricultural areas with these risk factors can change soil health from poor to very good.

Several soil properties, including soil classification, bulk density, soil moisture, soil carbon, and topography can help determine the overall soil fertility, which is a major contributing factor to the definition of soil health. Each of the physical properties are interdependent, and one soil quality factor often acts as an indicator of a different property. For example, soil texture can indicate a general soil bulk density because the bulk density for sand, silt and clay varies and the percentage of mixed sand silt and clay in a soil also varies. Soil aggregation is dependent on soil textural class but also on management and traffic (cattle or vehicles). More aggregation usually results in lower BD. While sands particles have higher BD (often made of quartz which has a BD of 2.65 g cm-2) they often have lower BDs because they do not compact as easily as silt or clay but silt and clay form more stable aggregates. Additionally, bulk density can act as an indicator for other soil properties that are not as easily measured. A high bulk density could indicate that the soil has been subject to compaction; bulk densities may become restrictive to root growth range from greater than 1.47 g cm^{-3} to 1.8 g cm^{-3} (for clayey to sandy soils, respectively) (USDA, 2008). In agricultural soils, this could be due to farm equipment, constant use by domestic animals, or little bioactivity.

The soil type is determined by several forming factors including parent material, climate, organisms, topography, and time. Soils are divided into soil families, and further into soil series. The soil family is defined mainly by shared physical properties and parent material, while series are determined by creating soil groups with the same general arrangement of soil horizons in the profile (Gardiner et al., 2008). The basic properties of a soil series and its location in the landscape determine its primary use as well as the challenges of its use. For example, certain soils retain water effectively, making them highly suitable for agriculture, while other soils are not suited to crop growth because they lack this characteristic. It is important to note, however, not all properties are solely dependent on soil series. Historical management schemes may play a part in these soil properties as well.

In South Cameroon, six different land uses were studied and soil samples were taken from each land use (Monkiedje, 2006). The control group was a natural forested land that had no crop history or agrochemical use while the other tested groups included different cropping and management systems. Soils in each management system were analyzed for soil texture and were either clay, sandy clay, or clay loam soils. Systems were subjected to different fertilizer, pesticide, and insecticide practices, as well as different tillage methods. The results of the study showed that soils subjected to traditional agriculture had lower overall microbial activity in the soil and lower carbon (C) turnover, which therefore meant lower soil carbon concentrations after management strategies were enacted. On the other hand, when best management agricultural practices were put in places, soil samples showed greater amounts of both microbial activity and carbon turnover than the non-managed agricultural uses.

Additional studies conducted in South Cameroon (Agoume et al., 2009; Monkiedje et al., 2006) concluded that soil pH, total nitrogen, soil organic carbon (SOC), available P, and cation exchange were all significantly affected by the land use system of the surrounding environment. Organic matter of a soil also affected the soil's fertility. For example, when SOM increased, soil fertility increased. The study showed that by controlling erosion, fertilizer applications, and the use of agrochemicals, the soil fertility and soil health properties were improved.

A similar study was conducted in Bangladesh, where the need for increased food sources has led to an increase in crop production (Islam and Weil, 2000). Researchers were concerned about the rapid conversion of forest to agricultural land use and the effects of potential soil degradation on long-term soil productivity. Similar to the Cameroon study, different land uses were chosen for samples against a forested control group. A reforested area was also studied to determine if soil could be remediated and regain healthy soil properties. Soil bulk density, soil pH, and microbial biomass were all sampled. As seen previously, the agricultural soils had a higher bulk density and lower total nitrogen than forested soils. The increased bulk density in this study was the result of soil compaction from exposure after tilling and pesticide or herbicide applications.

Soil Carbon

The total C in soil is the sum of organic and inorganic C. Total C can be used to estimate the amount of soil organic C in the soil (Weil, 2003). Soil organic carbon (SOC) is the carbon component of soil organic matter (SOM). SOC is widely measured, but no *in situ* test method currently exists for total SOC (Weil, 2003). SOC is an important factor in determining the potential of carbon sequestration in soils. In order to accurately determine the amount of soil C, a large number of samples need to be analyzed which can be done through a variety of methods.

Soil organic matter (SOM) is a major indicator of soil health because of its interdependence with other soil physical, chemical, and biological properties which affect the soil's overall productivity potential. Soil organic matter (SOM) is defined as the amount of plant and animal residue in the soil in various stages of decomposition (Ande, 2009). When increased, SOM has been shown to benefit several soil properties, including improved water retention, increased buffering capacity, increased nutrient availability, and decreased soil bulk density (Wander, 2004). Despite the benefits of increased SOM, it is difficult to make recommendations for its management because standard SOM measures do not exist (Wander, 1994). Gradual soil carbon changes are often hard to determine because of the existing carbon content in soils in conjunction with high variability among soil types. Additionally, SOM concentrations are influenced by several factors, such as climate, soil texture, and management strategy.

To fully understand SOM dynamics in managed systems, it is important to have a way to quantify the different fractions of SOM. In the total soil C, distinctions have been made between two main fractions: the labile fraction and recalcitrant C pool. The labile fraction is also known as the active C pool. This pool, unlike the recalcitrant pool, is susceptible to change due to microbial activities – such as respiration and microbial biomass (Islam and Weil, 2000). Active SOM is a mix of living and dead organic matter that circulates through the soil biological pools (Wander, 1994). The active C pool has a short turnover time (days or weeks) while the slow C pool has a turnover rate of years or decades and a resistant pool with a centuries-long turnover rate (Grandy, 2006). To this end, several studies have been conducted to determine the effects of soil management on soil organic matter and its fractions.

SOM has no definitive chemical structure, but it can be categorized into soil organic carbon (SOC) and particulate organic matter-carbon (POM-C). POM-C is composed of partially decomposed plant and animal residue and is considered a source of energy for soil microorganisms (Marriott, 2006). In the 1980s, a method to determine particulate organic matter (POM) was developed that greatly improved the understanding of SOM changes (Cozzolino, 2006). The climate and soil texture may affect overall SOM concentrations as well as the labile fraction (Marriott, 2006).

One such method for SOC estimation is loss-on-ignition (LOI) carbon. LOI carbon analyses have both supporters and detractors. Supporters of the method cite its affordability and the large number of samples that can be run concurrently (De Vos et al., 2005; Konen et al., 2002). Scientists acknowledge that the LOI method consistently overestimates the amount of organic matter in the soil because of the indiscriminate nature of the test. Loss-on-ignition analyses routinely: 1) remove hygroscopic H_2O; 2) release CO_2 from carbonates; 3) lose H_2O from hydroxyl groups; and 4) destroy elemental C such as charcoal leading to miscalculations of the organic matter content (De Vos et al., 2005). In order to accurately predict SOC, calibrations relating SOC to LOI methods should be developed for a multitude of soil datasets.

A study of non-calcareous soils was conducted to determine the ability of LOI methodology to estimate total organic carbon (TOC), develop predictive equations for soils, and determine if initial cofactors could improve predictive equations. During this study, samples were taken from a mixed broadleaf and coniferous forest in North Belgium and samples were analyzed for TOC using an auto-analyzer equipped with a solid sample module operated at 900ºC and analysis was done according to ISO 10694 (De Vos et al., 2005).

Additionally, samples were analyzed following a LOI method (at 550ºC) and percent loss was determined; and soil textures were also determined and recorded (De Vos et al., 2005). In both the broadleaf and coniferous forests, strong linear relationships between TOC and LOI were shown (r^2 ranging from 0.9 to 0.99). However, the relationship of LOI to TOC improved when clay content was added to the equation as a covariant. Without clay content, the TOC/LOI ranged from 0.11 to 0.61 (mean value 0.42). When clay content was included in the TOC equation, a TOC/LOI ratio was determined to be 0.573 which relates strongly to the 0.58 conversion factor set by (Bhatti and Bauer, 2002).

Additionally, small changes in SOC may be difficult to measure and could take up to 10 years to demonstrate measurable differences following a change in management practice (Weil, 2003; Marriott and Wander, 2006). Measuring the particulate or microbial soil fractions rather than the SOC or total C of the soil has been shown to respond more quickly and accurately to soil changes than total carbon (Sparling, 1992). Microbial carbon has a rapid turnover rate (1-2 years), making it susceptible to changes in a management scenario.

The Rodale Institute Research Center conducted a farming systems trial (FST) experiment in which agricultural land underwent transition from a conventional management scenario to an organically managed scenario in order to help identify which SOM indices are important to the active SOM fraction (Wander, 1994). Previous research indicated that changes in management practices can affect the active SOM characteristics before changes in net organic matter content are shown (Wander, 1994). During the Rodale experiment, total SOM content was not an effective means of measuring soil quality while soils were under transition. The study demonstrated that soil POM could serve as a preliminary indicator of either soil degradation or aggradation. Treatment-based differences to the SOM are associated with differences in the extent to which active SOM was stabilized or protected through management (Wander, 1994).

In addition to soil POM, there are other methods of measuring soil C that respond more quickly to management changes than SOC. In Weil et al. (2003), two comparative studies on the effect of management on labile C pools were conducted – one between a conventional and no-till wheat rotation cropping system in Mandan, North Dakota, and the other among several Mid-Atlantic farmers. In the North Dakota study, significant differences between the two cropping systems were detected in more labile permanganate oxidizable C (POXC) pools (ANOVA F-ratio = 34), but not in the SOC pools (ANOVA F-ratio = 4). Additionally, in the Mid-Atlantic study, the proposed active C method (using 0.02 M $KMnO_4$) showed more correlation between the two soil categories than the previous method (using 0.333 M $KMnO_4$) (Weil, et al., 2003).

The POXC method developed by Weil (2003) was designed as a method that would be rapid, inexpensive, and sensitive to change. Unlike the previously developed particulate organic C (POC) and microbial biomass C (MBC) methods to determine soil C changes from management, POXC is a rapid and relatively inexpensive method; it also has the potential to be adapted for in-the-field use. In a meta-analysis of twelve studies throughout the United States, relationships between POXC and the other testing methods (POC, MBC, and SOC) were determined and the effect of additional covariates including soil type, land use, and soil depth were determined. POXC was significantly related to each of the methods of analysis, but most closely related to the SOC with 0.58 of the variation explained (Culman et al, 2012); the relationships of POXC to other analysis methods improved when compared to the individual study sites. Additionally, the size fraction at which the analyses were performed changed the strength of the relationship between POXC and POC analyses. POXC was more closely related to smaller, heavier POC fractions (53-250 µm versus 250-2000 µm; >1.7 g cm^{-3} fractions) which signifies that the method is closely tied to a pool of labile C that has already been processed to some extent.

In addition to the relationships with the microbial measurements, the POXC showed its strongest relationship with soil organic C; this could be explained, in part, because of the similarity in method of analysis (Culman et al, 2012). Because of the strong relationship between POXC and SOC, using them as covariates can further improve the accuracy of other methods of C analysis. In the meta-analysis, POXC had the largest F-statistic of analysis out of POXC, POC, MBC, and SOC analysis methods in 42% of the comparisons (13 of 31 comparisons), suggesting that POXC can be a sensitive indicator to experimental factors or management changes.

The relationships of POXC to other analysis methods demonstrates the ability of the Weil extraction method to track management changes through active soil C. Therefore, the use of dual methods (LOI and POXC) of soil C analysis will provide a rapid, cost-effective measure that will show changes to the organic carbon content of the soil, both quickly and over long periods of time.

Soil Phosphorus

Nutrient losses from agriculture to surface water can be classified by three pathways: 1) point source losses from farmyards and excessive rates of application; 2) diffuse losses from soil in relation to the soil content of nitrogen and/or phosphorus in excess of system requirements; and 3) incidental losses occurring directly from loss of fertilizer or manure during application, or when a precipitation event occurs directly following application (Brennan, 2011). Grazing cattle may contribute to incidental losses when nutrients build up in vulnerable source areas at the edge of the field and subsequently run off during rainfall events. Incidental nutrient losses are dependent on several different factors, including: rate and type of fertilizer applied; intensity of precipitation events following application; field flow path length; vegetative cover; and field slope (Brennan, 2012) and landscape position. Phosphorus has three main loss pathways: 1) P in surface runoff, 2) particulate P from erosion, and 3) soluble P in leachate.

When these nutrients enter the surface water, they often cause widespread eutrophication which, in high levels, can cause hypoxic (2-3 ppm dissolved oxygen) (USGS, 2013) or anoxic conditions following massive algae blooms. Hypoxic and anoxic zones are responsible for many large scale fish kills and the reduction of submerged aquatic vegetation (SAVs) crucial to filtering processes in these systems. Nitrogen contamination is often apparent at a distance from where nutrient loadings first occur, but because of phosphorus' reduced solubility, its effects are usually seen closer to the source of contamination.

Phosphorus losses in agricultural systems are particularly concerning because of its inherent properties and role as a potentially limiting nutrient in ecosystems. Phosphorus is considered an essential nutrient for all living organisms; it is required for both genetic makeup and the formation of adenosine triphosphate (ATP). Without phosphorus, plant production would be limited by deficiency symptoms. In order to keep crop yields at acceptable levels, phosphorus fertilizers are widely applied in agricultural systems. Phosphorus only occurs in low quantities cosmically and, unlike nitrogen, there is currently no way to manufacture inorganic phosphorus fertilizers.

To produce inorganic phosphorus fertilizers, rock phosphate must be mined and manufactured. Rock phosphate is not a readily renewable resource; stores exist only in certain areas globally, and the mining process is not only energy intensive but also environmentally destructive. Finally, phosphorus is not readily available to plants in an agricultural system because of its affinity to bind to soil surfaces (especially clays) and either become plant unavailable or run off the field as part of the suspended solids lost to soil erosion.

There are currently several possible methods to mitigate phosphorus loss issues. Those directly involving the landscape include: incorporation of P fertilizers; increasing buffer zones between fertilized land and surface waters; and improved application of timing (Brennan, 2012). Other socio-economic methods include: incentives such P-trading and placing limits on the use of these fertilizers which can help also improve nutrient efficiency and responsible fertilizer use. Finally, chemical amendments added to feed to improve P use efficiency (thus requiring less P needed in the feed) or added to organic manures to help sequester P in the soil and reduce P losses in runoff to surface waters.

To effectively manage the P involved with agricultural production, the amount of P captured and lost from these fields must be quantified but it is also crucial to quantify the plant-available P in the soil. While there are many ways to extract soil P, the Mehlich-1 method for assessing soil test P (STP; Mehlich, 1953) is widely accepted in the Southeast as an accurate measure of the P in the soil that is readily plant available.

A common method for measuring the extracted STP was developed by Murphy and Riley (1962) using molybdenum blue coloring in conjunction with the ascorbic acid. This method, however, is not without its flaws. Issues with the absorption coefficient, adaptations to the buffer conditions, and the possibility of partial hydrolysis of labile organic phosphate (P_o) have all been cited previously (He and Honeycutt, 2005). In addition, if samples are not analyzed in a timely manner, labile P hydrolysis will increase, leading to an overestimation of soil P (He and Honeycutt, 2005; Dick and Tabatabai, 1977). In extreme cases, a blue precipitate could form, rendering the supernatant colorless and interfering with sample analyses. Additionally, it may be important to add sodium dodecyl sulfate (SDS) in order to eliminate protein interferences and release all hydrolysable labile organic phosphate (He and Honeycutt, 2005).

The Use of Visual and Near-Infrared Spectroscopy to measure Soil C and P

In order to reduce the time and cost inputs to producers, research into the use of spectroscopy has become widespread. Spectroscopy is a rapid, inexpensive means to monitor various soil properties, including the overall SOM content (Rossel, 2006). Studies exploring the use of visible, near infrared, mid infrared or combined diffuse reflectance technologies for *in situ* SOM monitoring have been conducted. Many studies have been conducted throughout the entire visible and near infrared regions (400-2500 nm).

Visible near infrared reflectance spectroscopy (VNIRS) spectra are largely the result of the C – H, N – H, and O – H bonds of organic matter occurring in the 400-700 nm (visible range) range. Strong correlations of VNIRS spectra to total nitrogen (TN) and SOC have been widely reported. Some studies have even used VNIRS spectra for the prediction of microbial biomass nitrogen and water-extractable organic N with good results (St. Luce, 2014). St. Luce (2014) used variable soil samples from a managed crop rotation sample in order to determine the feasibility of a calibration model for a large region.

St. Luce (2014) found that though correlations between soil properties and wavelengths existed throughout the spectrum, the near infrared region (NIR) showed the strongest overall correlations. Strong peaks for TN and SOC were seen at 2148 and 2156 nm, respectively, and there was a strong correlation between the two properties with an r^2 value of 0.81.

Near infrared reflectance spectroscopy (NIRS) has perhaps been the most widely researched monitoring option for SOM. NIRS is also based on the C – H, N – H, and O – H organic matter groups. Cozzolino (2006) used laboratory-prepared samples in order to determine if NIR scans would be a viable alternative to traditional monitoring methods. Samples were scanned and the soil particle size distributions of each sample were also determined. In all samples, the "Clay + silt C" fraction was most highly predictive for both raw scans and its second derivative (r^2 = 0.90 and 0.90, respectively). Additionally, the C prediction potential displayed high r^2 values for each of the soil fractions (Coarse-sand C; Fine-sand C; Clay+ silt C), 0.88, 0.70, and 0.96, respectively.

NIR spectra of soil have several regions that need to be taken into account. The calibration of the Clay+ Silt C fraction is largely dependent on the 690-700 and 788 nm region, which are often associated with plant residue absorption (Cozzolino et al., 2006). Additionally, there are several ranges in the spectra which can be used to predict soil C concentrations. The 700-800 nm region are associated with humic substances and phenolic groups due to the SOM residues; OH groups are seen at 1400 and 1900 nm; CH_2 at 1700 nm; and

C – H and O – H phenolic groups are also present at 2200 and 2300 nm (Xie et al., 2011, Cozzolino et al., 2006).

Because soil samples generally have broad bands which overlap, it is virtually impossible to predict a soil parameter quantitatively; therefore, multivariate statistical techniques are required (St. Luce, 2014). Chang (2002) used a multivariate calibration and partial-least squares regression (PLSR) technique in order to correlate soil C concentrations with measured soil C values. In all of the samples (soil only, all soil mixtures, $CaCO_3$ study control) a single large absorption peak was present in the visible and near-infrared region (1100-2500 nm) followed by three major peaks at 1400, 1900, and 2200 nm. Additionally, small peaks between 2300 and 2500 nm were also present.

Using the PLSR technique, strong correlations of predicted values of organic C, inorganic C, and total C were shown (r^2= 0.96, 0.97, and 0.96, respectively, for cross-validation and r^2 =0.89, 0.96, and 0.91, for validation).

In Xie et al. (2010), a PLSR technique was also conducted on laboratory-prepared soil samples. The PLSR was used for both mid infrared (MIR) and NIR spectra. It was shown that the absorbance bands of the NIR spectra were largely produced from the overtone and combination bands of the typical C – H, O – H, and N – H organic matter groups. The combinations of these bonds have been shown to more strongly absorb in the mid infrared region. MIR scans (ranging 2500 to 25000 nm) in the Xie (2010) study showed the highest SOC calibrations at the 1479.3 to 3281.7 cm^{-1} and 3641.4 to 4002.1 cm^{-1} wavelengths of any of the identified ranges (r^2 = 0.99).

Although the MIR scans in several studies have been shown to produce consistently more accurate results, the technology is much more complex and expensive than that of visible or NIR scans (Rossel, 2006). Because infrared scanning technologies would be needed for *in situ* measurements throughout the agriculture industry, it is impractical to use a higher-cost option if a significant difference in r^2 values does not exist. Additionally, VIS and NIR spectrophotometers are readily available for field usage, while the MIR analytical spectral devices are currently too cumbersome to be a practical field sampling method.

Another issue preventing widespread spectral monitoring is the limited data on field-moist (or unprepared) sample correlations. Researchers acknowledge that soil samples were processed (oven-dried, ground, and sieved) in order to reduce scatter from variable aggregate sizes and moisture contents (Chang et al., 2002). Absorbance spectra in field-moist samples are generally characterized by lower, broader bands.

Finally, there is little calibration data for many agricultural soils. It has been shown that both *in situ* and prepared samples have higher correlations when the spectra are calibrated with a local and single soil type with similar mineralogy (Xie et al, 2010). In order to use VIS, NIR, and VNIR scans to improve SOM monitoring, large-scale calibrations for varying soil types as well as varying management scenarios would need to be created.

Therefore, the use of reflectance spectra for monitoring soil carbon has the potential to be a powerful tool for soil health monitoring. Reflectance equipment and spectra would greatly reduce the amount of time, money, and soil disturbance that is currently necessary to receive current soil parameter information. However, issues with calibration accuracy in field-moist or *in situ* measurement scenarios must be rectified before NIR, VIS, and VNIR are widely adopted monitoring methods. Additionally, improving technologies will make these methods, as well as MIR, more accessible to a large number of producers. In the future, increased research into calibration methods and technological advances to further reduce equipment costs will promote the use of near infrared spectroscopy as a widely used method of soil health monitoring throughout the globe.

Runoff and Infiltration

One of the greatest environmental threats is the eutrophication and impairment of surface freshwater systems. These systems are valuable not only as freshwater sources for human consumption, industry, and recreation, but also as ecological habitats. Therefore, the protection of surface water resources is essential for sustainable systems. The eutrophication of freshwater systems is generally limited by P. In most systems, pollution control efforts center on reducing P losses through runoff (Shigaki et al., 2007).

In agricultural systems, P losses in runoff are influenced by transport and P source factors (Shigaki et al., 2007; Sharpley et al., 2001). Previous studies have attributed increased P losses to areas with high runoff volume and soil erosion as well as high intensity rainfall events (Shigaki et al., 2007). DRP can be supplied to runoff water through the following processes: desorption of P from soil particles; release of P from chemical fertilizer; microbiological release of P from organic materials; and phosphate leaching from plant residues (Yli-Halla et al., 1995). Buffer zones at the edge of agricultural fields are reported to decrease total P and particulate P, but may increase the DRP concentrations in runoff (Uusitalo et al., 2000).

Surface cover and soil condition are largely responsible for the quantity of pasture infiltration and quantity and quality of runoff following a rainfall event. In order to improve overall environmental health and reduce risk of nutrient loss to surface runoff, maintaining adequate surface cover is imperative (Franzluebbers et al., 2011). This can be achieved through insightful management of grazing systems. In a study conducted in Oconee County, Georgia, single-ring infiltration and penetration intensity were determined in each of four pasture types (unharvested, hayed, low-intensity grazed, and high-intensity grazed). Additionally, runoff was collected following weather events in the grazed pastures and frequency of event was recorded. This study determined that while dry condition infiltrations were largely dependent on the presence of macropores, in wet soil conditions water infiltration was greater in unharvested or hayed pastures, followed by low-intensity grazed pastures; water infiltration was slowest and penetration resistance was greatest in intensely grazed pastures (Franzelubbers et al., 2011). In another study, Hahn et al. (2012) reported infiltration rates were inhibited under dry conditions and postulated that this may have been the result of soil hydrophobicity. In pastures with good ground cover, TSS concentrations in runoff were reduced as was particulate P (Uusiatlo et al., 1999). DRP concentrations did not appear correlated to TSS and therefore were variable regardless of vegetative cover (Uusitalo et al., 1999). Puustinen et al. (2005) reported DRP was actually higher in fields with year-round vegetative cover.

<u>Phosphorus and Carbon in Runoff</u>

In addition to the ground cover of the soil, the P content of the soil can also be a significant factor in predicting the DRP in runoff. In a study conducted by Vadas et al. (2005), samples were analyzed for the relationship between DRP in runoff and water-extractable soil P. The slope of these regression lines (0.006 to 0.0018 kg $^{L-1}$) covered a wide range, but did not vary significantly. This shows that extraction coefficients may be an effective means of predicting soil and runoff P relationships over a wide range of soil, topography, and management practices (Hahn et al., 2012; Vadas et al., 2005). Initial soil moisture and rainfall intensity are two main drivers of DRP concentrations in runoff. Previous research has shown DRP losses from agricultural fields were often higher when soils were previously saturated than soils that were freely draining (Hahn et al., 2012; Zheng et al., 2004). The intensity of runoff, however, was secondary to the concentration of P in the soil.

In addition to P concentration in runoff water, DOC levels in runoff are also crucial. DOC is made up of highly mobile and reactive soil organic matter (Veum et al., 2009). Links have been made between DOC and an increased sorption and mobility of pesticides from agricultural production and heavy metals. Consequently, high levels of DOC can cause problems with clean drinking water sources (Veum et al., 2009 and Pomes et al., 1999).

DOC concentrations in water can vary from 1 to 70 mg L^{-1} (Veum et al., 2009 and Zsolnay, 1996) with estimated agricultural DOC losses ranging from 14.1 to 19.5 kg ha^{-1} yr^{-1} (Dalzell et al., 2007 and Veum et al., 2009). Previous rainfall simulation studies have shown that runoff was significantly higher in mineralizable C from no-till versus plow-till soils (Jacinthe et al., 2004). This mineralizable C fraction indicates labile carbon which can be easily detected through the use of POXC soil analysis. Additionally, disturbed areas were often higher in DOC exports signifying the importance of cattle camping areas to runoff quality. By rotating the cattle camping areas or moving them away from edge-of-field flow areas, it may be possible to reduce DOC concentrations in runoff from grazing pastures. Finally, during low-intensity runoff events, the C released was 2.5 times more labile than C releases from high-energy storms (Jacinthe et al., 2004). Therefore, highly reliable methods to test labile soil C may be used in order to predict the risk of DOC losses from runoff similarly to the use of STP to predict DRP concentrations following a rainfall event.

Runoff concentrations of major nutrients in aquatic systems must be analyzed often to accurately determine the concentrations of nutrients and the risks of contamination in these systems. In order to be widely accepted, analysis methods must be accurate as well as sensitive to a range of concentrations. Auto-analyzers and flow injection techniques are commonly used, however, they are expensive and laborious (Ringuet et al., 2011). Therefore, if a 96-well microplate technique can be used to determine nutrient concentrations, it could replace the more expensive alternatives.

Ringuet et al. (2011) compared 96-well microplate and autoanalyzer techniques for the determination of orthophosphates and determined that the two methods are statistically equivalent. Additionally, sensitivity and calibration curves for orthophosphate determination are linear with a strong r-square (r^2 = 0.999). Because the methods are easily reproduced, inexpensive, rapid, and highly sensitive, they provide a viable alternative to auto-analyzer techniques for the same major nutrients in aquatic systems.

<u>Importance of the Study</u>

Because of the scale of grazing and beef cattle production in the agriculture industry in the United States and the Southeast, it is important to make these systems as efficient as possible for both animal productivity and environmental sustainability. Several methods, including stream exclusions and lure management, have been proposed and proven as effective in minimizing impacts on surface water, nutrient management, and soil health.

Although environmentally beneficial, many individual best management practices are not widely adopted because of increased time and money inputs to the producer. It is believed that if multiple-management strategies are proven to be both beneficial and input efficient, such a system is likely to be widely used. Therefore, a tactical grazing system that includes multiple management strategies – short-term mob stocking, a less intense rotational stocking, excluding vulnerable nutrient source areas, and lure management techniques – is proposed as a means to bolster forage and animal productivity, reduce detrimental environmental effects, and improve farmer quality of life.

In addition to the industry and economic effects, grazing systems have the potential to effect the environment through soil health and surface water effects. Although not a traditional cropping system, many grazed pastures are fertilized in order to improve forage yields, thus reducing the need for outside dietary amendments (i.e. hay feeding). When fertilizers are applied in excess of rates required for plant growth, the application of organic and inorganic fertilizers to pastureland can contribute to incidental nutrient losses (Brennan et al., 2012). These losses are dependent on several factors including: the type and rate of fertilizer applied; timing and intensity of rainfall events post-application; hydrologic field conditions; field slope; and type and extent of vegetative cover (Brennan et al., 2012).

Nutrient losses may also occur because of the activities of the cattle themselves. Cattle in a continuously grazed system tend to utilize the same camping or loafing areas for the entirety of a grazing season. These camping areas could be chosen for a variety of reasons, including shade, water, food and/or hay sources, mineral sources, and protection from extreme elements. When the cattle camp in a specific area, they will compact soil in the area, which reduces nutrient infiltration to the soil. In many cases, these loafing areas are located at field edges, prompting defecated nutrients to run off the field and into nearby surface waters in the event of an intense precipitation event (Byers, 2005).

Combining these management practices with rapid and inexpensive techniques to monitor soil carbon and nutrients can greatly improve the overall system. Developing a way to track the effect of management changes on soil health factors is essential. POXC techniques have been shown to be rapid and inexpensive, and also highly sensitive to changes in the active C fraction of the soil. This fraction is most rapidly changed following a switch in management practice. Additionally, monitoring soil P through Mehlich-1 extractions not only provides information on the prevalence of P in the pasture, but also a means to predict potential DRP concentrations that may leave the field during runoff events.

By developing rapid and inexpensive soil C (through POXC) and soil P determination methods and improved calibrations for VNIR technologies, the quality of life for producers could significantly increase. Additionally, by improving the overall soil health, a strong foundation for sustainable agricultural productions systems can be achieved.

Objectives

The objectives of this study are to determine:

1. the spatial distribution of bulk density and soil C (measured by LOI and POXC) in 10 Southern Piedmont pastures as affected by landscape position and management by analyzing:

    a. How distance to or from cattle camping areas or pasture equipage (hay, water, or shade) affect the distribution of soil C
    b. How different measures of soil C (LOI and POXC) are related to real-time VNIR spectral scans.
2. the distribution of soil P (Mehlich-1) based on location and soil type throughout 10 Southern Piedmont pastures by analyzing the effect of hay, water, and shade
3. the relationship of soil P (Mehlich-1) and dissolved reactive phosphorus in runoff
4. the relationship of soil C (LOI and POXC) and dissolved organic carbon in runoff

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CHAPTER 2

DETERMINATION OF CARBON DISTRIBUTION WITHIN CONVENTIONAL GRAZING MANAGEMENT SYSTEMS IN THE SOUTHERN PIEDMONT[1]

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Abstract

In order to improve production and sustainability in pasture grazed systems, the amount of carbon in the system must be able to be rapidly and reliably determined. The objectives of this study were to 1) determine spatial distribution of bulk density and soil C, measured as loss-on-ignition (LOI) carbon or permanganate oxidizable carbon (POXC), in continuously stocked Southern Piedmont pastures as affected by soil classification and management and 2) evaluate the potential of real-time, in situ VNIR scans to estimate soil C (LOI and POXC).Samples were collected from 10 pastures ranging from 9.2 to 21.8 ha. Grazing areas were subdivided into either areas of interest (AOIs), where cattle tended to frequent, or matrix (pasture), which was the remainder of each pasture. Laboratory analysis included soil bulk density (BD), loss-on-ignition carbon (LOI), and permanganate carbon (POXC). Additionally, samples were analyzed *in situ* using an ASD Vis/NIR AgriSpec (ASD Inc, Boulder, CO) portable visible and near-infrared (450-2400 nm) analytical spectral device (VNIR) to develop calibration models for soil C (in the form of LOI and POXC). Soil bulk density was greatest in the 5-10 cm soil layer. Soil C (LOI and POXC) were greatest in the 0-5 cm soil layer and both measures were strongly correlated to soil organic C. Multivariate analysis of variance (MANOVA) determined that bulk densities were lowest 40-99 m from hay, water, or shade. Soil C (LOI and POXC) values were lowest in samples 100 m or more from a shade source. Finally, It was determined that VNIR scans could be used with limited success when high response wavelengths (1695-1725 nm) could be identified.

Introduction

Beef cattle production is an important economic and cultural component of the Southern Piedmont region. In the Southeast, a large proportion of beef cattle production occurs in pasture-based grazing systems, accounting for approximately 5.5 million hectares of land (USDA, 2013) and making these systems large-scale contributors to environmental nutrient issues. With the onset of global climate change and an increase in extreme weather patterns, U.S. agricultural systems are focusing on more sustainable production management by improving access to clean water and minimizing nutrient leaks from existing systems.

Beef cattle are usually managed using either a continuous or a rotationally stocked grazing system (Butler, 2010). In continuously stocked pastures, there are areas where cattle tend to frequent and these sites have the potential to accumulate soil organic carbon and associated N and P deposited by the cattle as feces or urine. Often, however, these sites are in shaded low-lying areas or near riparian areas that are sensitive to erosion and can be compacted due to the heavy traffic. The deposition or accumulation of nutrients in these areas may become point sources of pollution. Further, the poor uniformity of nutrients in these pastures can negatively influence a producer’s ability to manage nutrients efficiently. Over application in areas of high nutrient build-up may result in environmental concerns through runoff to surface waters while under-application of nutrients can limit a system’s productivity and place the producer under financial constraints. If the variability of a grazing system is not well managed, the production system overall may suffer by reducing forage quality and productivity, increasing the need for feed supplementation, which may lead to unnecessary loss of nutrients thereby reducing the productivity, profit, and sustainability of the entire system.

Areas that animals frequent are often near grazing equipage such as: shade, waterers, minerals, and hay-feeding locations. The location of these pasture equipage may influence soil carbon concentrations and distribution. More knowledge is needed to understand if and at what distance from grazing equipage soil carbon concentrations are affected.

To make pasture-based beef cattle production systems more sustainable, producers are focusing on efficient forage use, maximizing nutritional quality of forages, and improving overall pasture quality while also maximizing animal gains. To do this, one must first have some understanding of the concentrations and distribution of nutrients within their pastures. Best management practices (BMPs) and a strong grazing program can be used to make improvements to both the immediate pasture and the surrounding environment. Improved nutrient distribution across the pasture may result in improved forage productivity, improved infiltration rates of rainfall, and reduced nutrient concentrations in runoff. The amount, type, and distribution (with depth and location within the pasture) of soil carbon can influence critically important biogeochemical measures of soil health.

Some BMPs that can facilitate pasture improvements include: stream exclusions; rotational stocking; strategic placement of shade, mineral, and water sources to lure livestock away from sensitive areas (lure management), and the use of exclusionary areas in order to reduce the amount of time cattle spend in areas vulnerable to nutrient losses. Strategic use of each of these BMPs has the potential to redistribute carbon and nutrients for positive changes throughout the pasture. Exclusion areas (whether near streams or edge-of-field) limit cattle activity in these areas and may reduce compaction (reduce bulk density), and erosion. Absence of animal traffic will allow more forage growth where none was possible when cattle frequented these sites, adding to overall forage growth for the same pasture area. Additionally, cattle without access to exclusion areas will graze in other areas of the pasture likely lower in carbon. By redistributing the cattle, carbon and nutrient rich fecal deposits are also redistributed to other areas of the pasture. To determine the potential of these BMP strategies to enhance soil C concentrations, the effect of each on soil C distribution and bulk density must be determined.

A key component for each of these sustainability goals is the pasture soil health, or the ability of the pasture soils to maintain plant and animal productivity and enhance water and air quality (Sharma et al., 2009). Several soil properties, including soil classification, bulk density, soil carbon, and topography, can affect the soil fertility, which is a major contributing factor to the definition of soil health. Each of the physical properties are interdependent, and one soil quality factor often acts as an indicator of a different property.

Soil organic carbon (SOC) is the carbon component of the soil organic matter (SOM). Soils with increased levels of soil organic matter have been shown to have improved water retention and infiltration rates, increased nutrient availability, and decreased soil bulk density (Wander, 2004). When building or banking SOC, it is often difficult to accurately quantify the rapid (annual) changes in SOC that may occur as a result of management decisions. Because carbon fractions are extremely dynamic and highly variable, precise and accurate estimates of SOC require a large number of samples (Weil, 2003). These samples are expensive and time-consuming to obtain and analyze. New, cost-effective, and rapid methods of measuring SOC and C fractions in real-time and *in situ* are needed.

Currently, SOC estimation is often conducted using the loss-on-ignition (LOI) carbon analysis. This method is frequently used because of its relative affordability and rapid turnaround time (De Vos et al., 2005; Konen et al., 2002). Because LOI is an indiscriminate analysis method and routinely overestimates the presence of SOM (De Vos et al., 2005), it must be calibrated to other methods of SOC determination.

Additionally, small changes in SOC may be difficult to measure using LOI. Some authors estimate that it may take up to 10 years to demonstrate measurable LOI differences following a change in management practice (Weil, 2003; Marriott and Wander, 2006). Therefore, measuring soil C fractions that are dependent on microbial activity have been shown to accurately predict soil C changes following management changes over a shorter term (Weil, 2003; Marriott and Wander, 2006). Unlike the particulate organic C and microbial biomass C analysis methods, the permanganate oxidizable C (POXC) method of carbon determination is a rapid and relatively inexpensive method of C estimation. This, coupled with its ability to detect management-based changes in as little as 9-12 months, makes it a valuable tool for making management decisions to improve soil C.

A dual-method approach to determine soil C can be a valuable tool in order to create a calibration curve for subsequent soil C estimates. By analyzing soil samples using both the LOI and POXC methods, soil C concentrations in pasture soils can be determined rapidly, allowing for better grazing management decisions over time.

In order to reduce the time and cost inputs to producers, research into the use of spectroscopy has become widespread. Spectroscopy is a rapid, inexpensive means to monitor various soil properties, including the overall SOM content (Rossel, 2006). Studies exploring the use of visible (350-700 nm), near infrared (700-2500 nm), mid-infrared (2500-25,000 nm) or combined diffuse reflectance technologies for *in situ* SOM monitoring have been conducted (Rossel, 2006).

Visible and near infrared reflectance spectroscopy (VNIRS) spectra are largely the result of the C – H, N – H, and O – H bonds of organic matter. Near infrared reflectance spectra of soil have several wavelength regions that need to be taken into account. The calibration of the Clay+Silt C fraction is largely dependent on the 690-700 and 788 nm regions, which are often associated with plant residue absorption. Additionally, there are several ranges in the spectra which can be used to predict soil C concentrations. The 700-800 nm region is associated with humic substances and phenolic groups due to the SOM residues (Cozzolino, 2006); OH groups are seen at 1400 and 1900 nm (Xie, et al., 2011 and Ringuet et al., 2010); CH_2 at 1700 nm; and C – H and O – H phenolic groups are also present at 2200 and 2300 nm (Cozzolino, 2006 and Chang, 2002).

A central issue preventing widespread spectral monitoring is the limited data on field-moist (or unprepared) sample correlations. Researchers commonly acknowledge that soil samples were processed (oven-dried, ground, and sieved) in order to reduce scatter from variable aggregate sizes and moisture contents (Chang et al., 2002). Reflectance spectra in field-moist samples are generally characterized by lower, broader bands.

Finally, there is little calibration data for many agricultural soils. It has been shown that both *in situ* and prepared samples have higher correlations when the spectra are calibrated with a local and single soil type with similar mineralogy (Xie et al, 2010). In order to use VIS, NIR, and VNIR scans to improve SOM monitoring, large-scale calibrations for varying soil types as well as varying management scenarios would need to be created.

Therefore, the use of reflectance spectra for monitoring soil carbon has the potential to be a powerful tool for soil health monitoring. However, issues with calibration accuracy in field-moist or *in situ* measurement scenarios must be rectified before NIR, VIS, and VNIR are widely adopted monitoring methods.

The objectives of this research were to 1) determine spatial distribution of bulk density and soil C, measured as loss-on-ignition (LOI) carbon or permanganate oxidizable carbon (POXC), in continuously stocked Southern

Piedmont pastures as affected by landscape position and management and 2) evaluate the potential of real-time, in situ VNIR scans to estimate soil C (LOI and POXC).

Methods and Materials

Study Site

This study was conducted at the J. Phil Campbell Research and Education Center's North and West Units (33.887487,-83.420966 and 33.863240,-83.457599, respectively) in Watkinsville, Georgia, and at the University of Georgia's Animal and Dairy Science Department's Beef Research Unit (33.420759,-83.476555) in Eatonton, Georgia. There were four pastures on the North Unit (14.3 to 17.8 ha); two on the West Unit (9.2 to 9.3 ha); and four in Eatonton (18.0 to 21.6 ha). Each of these pastures was an experimental unit. The North and West unit pastures in Watkinsville are characterized by Cecil (2 to 6 percent slope) sandy loam and Pacolet (6 to 10 percent slope) sandy clay loam soils that are eroded or severely eroded. Eatonton pastures are primarily characterized by Davidson clay loam (6 to 10 percent slope), Davidson loam (2 to 6 percent slope), and Wilkes sandy loam (2 to 10 percent slope) soils, although Iredell loam, Enon, Congaree, and Toccoa soils have also been reported.

The pastures in Eatonton were managed for at least 10 years using put and take grazing in continuously stocked pastures with 20-30 head per stocking herd. The pastures in Watkinsville have been managed using put and take grazing for at least 10 years, with approximately 40 head per stocking herd.

Aerial imagery (from Google Earth Pro) shows that the approximate hay feeding locations have remained the same for both Eatonton and Watkinsville since 1993. Eatonton used primarily hay rings and Watkinsville used hay rings from 2000-2006; heavy use areas from 2006-2013; and a hay un-roller from 2013-present.

Soil Sampling

Areas within these pastures where cattle tended to frequent (referred to as areas of interest AOIs) were delineated in each pasture and sampled separately from the remainder of each pasture (referred to as matrix or pasture). These AOIs were determined based on historical observations and evidence (e.g., fecal deposits, trampled or overgrazed forage, etc.) of cattle camping activity near shade or grazing equipage (water, and feed sources). Pastures contained 3-8 AOIs depending on pasture area and distribution of pasture features that attracted cattle (i.e., shade, water, food, hay sources, etc.).

Soil samples were taken randomly from the AOIs and on grid pattern within the matrix area. Samples were taken throughout the AOI during the fall and spring using a 5-cm core diameter Giddings hydraulic soil probe (Giddings Machine Corporation, Inc., Windsor, CO) at multiple points (2-5) with three replications per point within a 1-m sampling radius. Samples were taken from the remaining pasture area (matrix) on a 50-m grid with two soil cores taken within a 1-m sampling radius at each grid-point intersection. All core samples were taken from three depths: 0-5 cm, 5-10 cm, and 10-20 cm. Soil cores were split longitudinally to reveal an unaltered face and scanned for reflectance in the field using a portable ASD Vis/NIR AgriSpec (ASD Inc, Boulder, CO) VNIR analytical spectral device.

Determination of Distances from Equipage

The geometric distance (straight line) from each sampling point to closest edge of shade or grazing equipage (hay, water, and manmade or natural shades) were determined using GIS technology and ArcGIS 10.2.1 software. The analysis was completed by creating shapefiles with GPS coordinates of each sampling point and hay, water, and shade feature in the pasture. A “near tool” analysis was run through an ArcGIS model builder capability to measure the distance from each unique sampling point to the nearest hay, water, and shade source. Distance zones (m): 0 - 39, 40 - 99, 100 – 199, and 200+ were selected based on recursive partitioning and to split pastures evenly.

Soil Analysis

Soil samples were air-dried and ground (<2 mm) and oven-dried moisture and soil bulk density were calculated using the *USDA Soil Survey Laboratory Methods Manual* (USDA, 2004) for a sample from a known volume soil core. Additionally, samples were analyzed for loss-on-ignition (LOI) carbon using the method in the *USDA Soil Survey Laboratory Methods Manual* (USDA, 2004) where organic matter was combusted at 550°C for 8-10 hours. Permanganate oxidizable carbon (POXC) analyses were based on the procedure set by Weil et al. (2003).Samples were analyzed using a Biotek ELx800 absorbance reader at 550 nm (Biotek Instruments, Inc., Winooski, VT. Additionally, samples were analyzed using a standard curve ranging from 0.005 to 0.025 mg L^{-1} of 0.2 M $KMnO_4$.

Relationships among LOI, POXC, and BD were also determined through the use of recursive partitioning and PLSR statistical analyses in JMP 11 statistical software (SAS, Cary, NC) to determine which, if any, bulk densities or carbon levels were related to one another. Differences in BD with distance to shade and pasture equipage were determined using the Tukey's test for significance, while LOI and POXC were analyzed using Wilcoxon rank sum test for non-parametric statistical hypotheses.

Soil sampling points were grouped into 0-39 m, 40-99 m, 100-199 m, and >200 m from hay, water, and shade sources and the relationship of BD, LOI, and POXC to equipage were determined using multivariate analysis of variance (MANOVA) in JMP 11 (SAS, Cary, NC). Additionally, the relationships of measured parameters topographic features, soil type and soil family were determined through the use of recursive partitioning and MANOVA.

Additionally, LOI data were validated by identifying a subset of soils from each pasture that were sent to the University of Georgia Soil, Plant, and Water Test Lab and analyzed for SOC in order to create a calibration curve with LOI. SOC was analyzed by combusting the C in the samples in a chamber at 1350°C into CO_2 and determining the amount of CO_2 gas with an infrared cell. Relationships among the three carbon analyses (SOC, LOI, and POXC) were determined using linear or logarithmic regression curves where appropriate.

Vetted bulk density, LOI, POXC, and soil P point data were used to create distribution maps for each pasture.

VNIR scans were recorded as the reflectance spectra first derivative to show more prominent peaks after being processed, to remove noise from the spectra, and to eliminate misread scans where the total brightness (reflectance values of 0-0.05) or total darkness (reflectance values of 0.95-1.0) were found. Bulk density, LOI, and POXC data were compared with VNIR scans to determine correlations to full spectra scans through partial least square regression (PLSR). Bulk density, LOI, and POXC parameters were compared against each wavelength of the full VNIR spectra (450-2400 nm) individually. Recursive partitioning techniques, literature review, and PLSR were then used to determine wavelengths likely to have the strongest relationships for each BD, LOI, and POXC, and to identify signatures.

Results and Discussion

General Site Description

The six Watkinsville and four Eatonton pastures had similar trends for BD, LOI, and POXC distributions both vertically (with soil depth) and horizontally (location) (Figures 2.1-2.3). The bulk densities for all sites and depths ranged from 0.69 to 2.61 g cm^{-3}. The mean bulk density in Watkinsville was 1.41 g cm^{-3} while the mean for Eatonton was 1.46 g cm^{-3}. LOI carbon ranged from 0.1 to 35.6 g kg^{-1} for both Watkinsville and Eatonton with means of 6.3 and 7.1 g kg^{-1}, respectively. Finally, POXC values for Watkinsville and Eatonton pastures ranged from 41 to 2819 mg kg^{-1} with means of 893 and 742 mg kg^{-1}, respectively.

The mean geometric distances to the nearest hay, water, and shade sources were overall greater in Watkinsville compared with Eatonton. Bulk density and carbon (both LOI and POXC) will each be explored by location (Watkinsville or Eatonton), soil depth (0-5 cm, 5-10 cm, and 10-20 cm), soil classification, and distance to pasture equipage individually. Additionally, relationships among the measured parameters (BD, LOI, and POXC) and their relationships to VNIR spectra to the different measured parameters will also be addressed.

Bulk Density

When soil BD data were pooled across sampling depths, the data were normally distributed, and the mean BD for Watkinsville and Eatonton pastures (1.41 and 1.46 g cm^{-3}, respectively) were significantly different ($P < 0.0001$), (Figure 2.4). In both locations, bulk densities were significantly greater ($P < 0.0001$) in the 5-10 cm (1.62 g cm^{-3}) depth than in either the 0-5 cm (1.26 g cm^{-3}) or 10-20 cm (1.4 g cm^{-3}) depth (Figure 2.5). This could be explained, in part, because of a legacy plow pan existing in the 5-10 cm range. Cattle activity and forage roots in the surface (0-5 cm) and the occurrence of deep-rooted forages in the 10-20 cm sampling depth may also account for some differences in bulk density values. The surface (0-5 cm) and deeper sampling depth (10-20 cm) were often more spatially variable, possibly as a consequence of the variability in historical management and vegetation. Because of the higher BD in the 5-10 cm soil layer of both Watkinsville and Eatonton pastures, this soil layer may restrict infiltration and the capacity of these pastures to capture rainfall. Management practices that help to reduce compaction in this layer could have a significant impact on the amount of water received and banked during rainfall events in the Southern Piedmont. Because BDs were significantly different between Watkinsville and Eatonton (Figure 2.4) results will be presented for the Watkinsville and Eatonton locations separately.

When all locations and depths were combined, there were no significant differences ($P = 0.57$) on BD among soil types. However, when analyzed by depth, soil classification did have a significant effect at the 0-5 cm and 10-20 cm depths ($P < 0.0001$ and $P = 0.0064$, respectively). When the 0-5 cm sampling depth was analyzed, most of the differences due to soil classification occurred in Eatonton. The Iredell loam (2 to 6 percent slope, Eatonton only) had a significantly greater BD (1.68 g cm^{-3}) than either the eroded Cecil sandy loam (2 to 6 percent slope) (1.14 g cm^{-3}) or the severely eroded Pacolet sandy clay loam (6 to 10 percent slope) (1.12 g cm^{-3}) ($P = 0.008$ and $P = 0.02$, respectively) (Table 2.1). Additionally, the eroded Wilkes sandy loam (2 to 10 percent slope) (1.44 g cm^{-3}) had a significantly greater BD than either the severely eroded Pacolet sandy clay loam (1.12 g cm^{-3}) or the eroded Cecil sandy loam (2 to 6 percent slope) (1.14 g cm^{-3}) ($P = 0.003$ and $P = 0.0115$, respectively) (Table 2.1). In Watkinsville, the only difference ($P = 0.005$) in the 0-5 cm soil layer occurred between the severely eroded Cecil sandy loam (2 to 6 percent slope) (1.23 g cm^{-3}) and the severely eroded Pacolet sandy clay loam (6 to 10 percent slope) (1.14 g cm^{-3}). In the 10-20 cm depth, Eatonton showed differences ($P = 0.034$)

between the eroded Davidson clay loam (10 to 25 percent slope) (1.14 g cm^{-3}) and the over-wash Cecil sandy loam (0 to 2 percent slope) (1.22 g cm^{-3}).

In Watkinsville, bulk density averaged across the three depths, did not differ among distance zones from hay, water, or shade (Figures 2.6-2.8). When considered by each soil depth, (Figures 2.9, 2.11), the distance zone to nearest hay or shade did not affect the observed bulk densities. However, distance zone nearest to waterer, affected bulk density in the 0-5 cm depth (Fig. 2.10). Soils samples between 40 and 99 m from a waterer had greater BD than soils at distances greater than 200 m.

In Eatonton, bulk density averaged across the three depths were most often lower in the zone 40-99 m away from shade and pasture equipage and in the 100-199 distance zone from hay (Figures 2.6-2.8). In the 0-5 and 5-10 cm soil layer, bulk densities were significantly smaller within 0-100 m distance zone from a hay feeding area (P<0.0001 and P = 0.0424, respectively) (Figure 2.12). Although cattle are likely to congregate in the areas closest to hay feeding sources, the bulk density may actually be lowered in these areas as cattle drop hay they are fed and that hay is incorporated into the soil profile, increasing the organic matter content of the area (Figure 2.13). When Eatonton bulk densities were compared with distance to a water source, the only difference occurred between samples taken 0-39 m and 40-99 m from a water source (Figure 2.14). Bulk density was significantly greater within 40 m from the closest shade regardless of depth (P = 0.0008, 0.0006, and 0.0012 for 0-5, 5-10 and 10-20 cm, respectively) (Figure 2.15). This may be explained, in part because cattle spending increased amounts of time in or near shaded portions of the pasture will increase the overall compaction of the soil in those areas and therefore increase the bulk density.

Carbon

Soil carbon measured as loss-on-ignition (LOI) was not normally distributed; median LOI was significantly lower ($P < 0.0001$) in Watkinsville than in Eatonton (Mdns: 5.35 g kg^{-1} and 6.54 g kg^{-1}, respectively). In contrast, median POXC values were higher in Watkinsville than in Eatonton (Mdns: 731 mg kg^{-1} and 684 mg kg^{-1}, respectively $P < 0.0001$). LOI median values decreased significantly below the 0-5 cm sampling depth ($P < 0.001$) regardless of location (Figure 2.16). Median POXC values decreased significantly at each sampling depth ($P < 0.001$) in each location (Mdns: 1137, 655, and 557 mg kg^{-1} for 0-5, 5-10, and 10-20 cm samples, respectively) (Figure 2.17).

Regardless of depth or location, POXC showed the greatest variability among measured soil variables. This is to be expected because of the sensitive nature of the analysis method and also its susceptibility to show changes in management much more rapidly than the more recalcitrant LOI analysis method. Unfortunately, in soils and sampling depths where manganese concretions exist, the POXC analysis could provide falsely low carbon values because of manganese saturation in the method (Figure 2.13). This may explain, in part, why POXC values in Eatonton soils, which had several instances of manganese concretions (especially in the 10-20 cm samples), showed significantly lower values than surface soils (Figure 2.17).

When location and sample depths were combined, soil classifications showed no significant differences for either measure of carbon (LOI or POXC) ($P = 0.06$). However, when LOI was analyzed by depth, significant differences among soil classifications were found in the 5-10 cm and 10-20 cm depths ($P = 0.0141$ and 0.0096, respectively). In the 5-10 cm depth, moderately eroded Davidson loam (2 to 6 percent) had significantly greater LOI carbon (Mdn: 6.06 g kg^{-1}) than the eroded Cecil sandy loam (2 to 6 percent) (Mdn: 4.19 g kg^{-1}) ($P =0.04$) (Table 2.2). In the 10-20 cm depth, the eroded Davidson clay loam (6 to 10 percent) (Mdn: 5.36 g kg^{-1}) had significantly greater LOI than three other soil classifications: eroded Cecil clay loam (2 to 6 percent slope) (Mdn: 4.47 g kg^{-1}); severely eroded Cecil sandy clay loam (2 to 6 percent) (Mdn: 4.47 g kg^{-1}); and severely eroded Pacolet sandy clay loam (6 to 10 percent) (Mdn: 4.55 g kg^{-1}) ($P = 0.01$, 0.04, and 0.01, respectively).

Similar to LOI carbon, POXC carbon showed no significant differences ($P = 0.06$) among soil classifications when location and soil depths were combined (Tables 2.2). However, when soil classifications were analyzed by depth, significant differences were found in the 0-5 cm depth ($P < 0.0001$) but not in the 5-10 or 10-20 cm depths ($P = 0.66$ and 0.11, respectively). All differences in the soil classifications occurred at the Eatonton site. Surprisingly, the severely eroded Pacolet sandy clay loam (6 to 10 percent) (Mdn: 1397 mg kg^{-1}) had significantly greater POXC than both the eroded Davidson clay loam (6 to 10 percent) (Mdn: 915 mg kg^{-1}) and the eroded Wilkes sandy loam (2 to 10 percent) (Mdn: 993 mg kg^{-1}) ($P < 0.0001$ and $P = 0.03$). Finally, the severely eroded Cecil sandy clay loam (2 to 6 percent) (Mdn: 1354 mg kg^{-1}) and the eroded Cecil sandy loam (2 to 6 percent) (Mdn: 1315 mg kg^{-1}) showed significantly greater POXC in the 0-5 cm depth than the eroded Davidson clay loam (6 to 10 percent) (Mdn: 915 mg kg^{-1}) (P = 0.0004 and 0.002, respectively).

LOI had few differences in distance from pasture equipage when all soil depths were combined (data not shown). However, when data were sorted by location and/or depth, distance to grazing equipage had some impact on soil concentrations of LOI and/or POXC. In Watkinsville, when depths were combined, there were no differences in LOI/POXC among points occurring within 199 m from a hay, water, or shade source (Figures 2.18-2.20). Examining differences in soil depth we found that the 10-20 cm sampling depth was not significantly affected by the distance to nearest hay or water source (Figures 2.21 and 2.22). However, significantly greater LOIs were observed in Watkinsville when samples were within 200 m of the nearest shade (Figure 2.23).

In Eatonton, the LOI for the 0-5 cm depth was not affected by either distance to hay or water (Figures 2.24 and 2.25) although the LOI carbon in the 10-20 cm sample depth within 40 m of a hay was significantly less than LOI carbon in 200+ m from the closest hay source (P = 0.0178) (Figure 2.24). In the 5-10 cm depth, the 100-199 m zone had the highest LOI values; in the 10-20 cm depth the 100-199 m had the lowest LOI of any distance zone.

Additionally, in Eatonton, no sampling points fell further than 200 m from a shade source. In the 0-5 cm depth, LOI values were only different in samples 40-99 m from a shade. In the 5-10 and 10-20 cm depths, samples 100 m or further from a shade were significantly lower in LOI carbon (Figure 2.26). This is likely attributed to increased cattle activity, and therefore greater fecal depositions (and carbon content) in the areas closest to the shade.

When samples were not separated by location or depth, POXC carbon was only slightly affected by distance to the nearest hay, water, or shade feature as well (Figures 2.27-2.29). When sample depths were combined, differences occurred between samples 100-199 and 200 m or greater from a hay feeder or waterer; there were no differences in POXC between locations at different distances from shade. Samples within 0-39 m from a hay source (in the 0-5 cm zone) had significantly greater POXC than samples 40-99 m from hay (Figure 2.30).

There were no differences in POXC at the 0-5 cm depth in Watkinsville for samples at different distances from water or shade (Figures 2.31 and 2.32). Additionally, in Watkinsville, 5-10 cm and 10-20 cm depths showed significantly greater POXC within 100 m from the nearest shade source (Figures 2.31 and 2.32).

In Eatonton, POXC was only different in the samples from the 5-10 or 10-20 cm depths that occurred 40-99 m from a hay source (Figure 2.33); no differences were seen in the 0-5 cm depth. Sample POXC values were greater from 0-100 m from water; POXCs were significantly lower in samples 200 m or greater from the nearest waterer at every depth (in Eatonton) (Figure 2.34). This could be explained in part, because of increased cattle activity immediately near water sources similar to that of the lowered bulk density near hay (Figures 2.33

and 2.34). Compaction may be an immediate concern, but repeated hoof movement may actually act to break up a continuous soil surface. In Eatonton, no samples occurred further than 200 m from the nearest shade and, similarly to LOI carbon measurements, distance to shade did not appear to affect POXC (Figure 2.35).

Because of the sensitive nature of the POXC analysis, it may be more advantageous to further divide the distance from a feature to better demonstrate differences based on distance. In Eatonton, however, observed POXC values may appear more similar among the three soil sampling depths because of the presence of manganese concretions near the soil surface.

Relationship to Bulk Density

Using recursive partitioning, a bulk density of 1.26 and 1.29 g cm^{-3} (in Watkinsville and Eatonton, respectively) showed some relationship to both LOI and POXC values. Recursive partitioning was used to classify the carbon values (whether LOI or POXC) into groups based on their relationship to BD values. LOI and POXC averaged 9.36 g kg^{-1} and 1282 mg kg^{-1} for soils with bulk density less than the critical value while soils with bulk density greater than or equal to this level (1.26 or 1.29 g cm^{-3}) averaged 5.10 g kg^{-1} and 738 mg kg^{-1}, respectively. In the 0 to 5 cm depth, soil BD averaged close to 1.3 g cm^{-3} while the deeper soil layers were significantly more compacted (Figure 2.15). Both LOI and POXC were greatest in the 0 to 5 soil sampling layer. LOI values were generally inversely related to bulk density, indicating lower bulk densities often occurred where greater carbon content was present. Often the areas of highest LOI carbon occurred in known cattle camping areas positioned in the upland field positions. This could be the result of a general buildup of organic matter through fecal depositions in these camping areas and a minimal loss of nutrients via runoff during high-precipitation weather events.

Loss-on-ignition – Soil Carbon Relationships

A strong linear relationship (y= 0.0469x -0.629; r^2 = 0.8971) was found between LOI and SOC concentrations (Figure 2.36). This ratio was comparable to the findings of De Vos, et al. (2005) that showed a LOI-SOC ratio of 0.42 before clay content was included in the equation. The results from that study mirrored the LOI-SOC ratios found in Bhatti and Bauer (2002) after clay content was considered in the equation (0.57 and 0.58, respectively).

Permanganate Oxidizable Carbon – Soil Carbon Relationships

Relationships of POXC to both LOI and SOC were also found. Regression equations for these relationships were both expressed through second order polynomial equations with r^2 values of 0.51 and 0.51 for LOI and SOC, respectively (Figures 2.37 and 2.38). Outliers in these relationships may be due to saturation of manganese from soil samples that inhibited the reduction of the permanganate from Mn^{7+} to Mn^{4+}. Each of the samples that provided a POXC outlier was taken in a low-lying area that remained moist throughout most of the sampling period and therefore susceptible to the presence of manganese concretions in the soil.

Relationship of Measured Parameters to VNIR Spectral Scans

PLSR was not an effective means for predicting soil parameters of the pastures using the full 450-2400 nm spectrum without partitioning data by depth or location. Specific wavelengths in the spectrum were targeted based on a review of previous research (Xie et al., 2011; Cozzolino et al., 2006; and Chang et al., 2002) of carbon species signatures and the use of recursive partitioning techniques.

In both Eatonton and Watkinsville, the strongest relationships between scans in 1695-1720 and 1260-1280 nm ranges for soil C were in the 10-20 cm sampling depth. Additionally, strongest relationships in the 700-800 nm range were found for 0-5 and 10-20 cm sampling depths. This supports the idea that these sampling layers have high organic matter contents due to cattle activity and deep-rooted forages. The presence of carbon in the CH_2 form may have contributed to a response in the 1695-1720 nm spectral range.

VNIR relationships improved when the soil samples were divided by different factors including depth of sample, relative carbon content, soil classification, and bulk density. Spectral relationships improved from an $r^2 < 0.05$ to up to $r^2 = 0.8$ by separating by depth. Samples in the 0-5 and 5-10 cm depths may be better VNIR predictors of carbon because of they are generally more consistent across the field. The 10-20 cm depth was highly variable, even within the same bulk density range, because of deep-rooted forages occurring in some areas of the pasture while not in others. Although bulk density was not related to a range of VNIR wavelengths, recursive partitioning showed it may be a partial predictor of both LOI and POXC. LOI and POXC showed relationships in similar ranges for the Watkinsville sites, from 1695-1710 nm (r^2=0.24) and 1715-1725 nm (r^2= 0.26), respectively when all depths were combined.

Conclusions

Regardless of location, soil BD was highest in the 5-10 cm depth. High BD values in this layer may be due, in part, to a legacy plow pan layer at this depth. Additionally, BDs in the 0-5 cm soil depth were likely lower due to disturbances from cattle activity and forage root mass; 10-20 cm samples were also likely affected by deep-rooted forages that lowered BD values. High BD values at the 5-10 cm sampling depth may create a restrictive layer for both water infiltration from rainfall events and forage roots. Because this restrictive layer is so close to the surface it could be mitigated through grazing management strategies.

A strong linear relationship (y= 0.0469x -0.629; r^2 = 0.8971) was found between LOI and SOC concentrations. Therefore, using LOI values may be an effective and inexpensive way to continue to monitor pasture soil carbon. Additionally, POXC showed the most spatial variation, likely because of its susceptibility to management changes. Because of the rapid and inexpensive nature of the POXC method, it will be a valuable tool to determine soil carbon changes due to a change in management practice, although the presence of manganese concretions in the monitored soil must be taken into account.

Overall the greatest differences were seen in LOI or POXC carbon between 0-199 and >200 m from a hay, water, or shade source. Bulk densities were lower closer to hay and water sources because of increased hoof action to break up the soil surface, while bulk densities showed an increase in areas close to or within a shade source. LOI values demonstrated relatively few differences in distance from a hay, water, or shade source, and these differences often did not occur until 200 m or greater from a pasture desirable. Finally, POXC values showed many differences with depth. This could be attributed to the soil heterogeneity and variability of POXC values.

Visual and near infrared reflectance spectra from 450-2400 nm may be a useful tool in predicting LOI and POXC concentrations in the soil when target wavelengths can be determined. These wavelengths are often the CH_2 characterized responses. By grouping soils based on depth or bulk density, VNIR relationships may be strengthened, however, they will need further calibration in order to be able to fully predict soil properties. With these highly developed curves, real-time management decisions may be more easily made.

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Table 2.1. Location, surface area, and mean and median bulk density values in three soil depths for each soil type sampled. E, ME, and SE represent erosion classification according to NRCS soil survey classes "eroded," "moderately eroded," and "severely eroded," respectively.

			Mean/Median Bulk Density values (g cm^{-3})		
Soil Type	**Location**	**Area (ha)**	0-5 cm	5-10 cm	10-20 cm
Cecil soils, (Slope 0-2); overwash Fine kaolinitic thermic Typic Kanhapludults	Watkinsville	0.5	1.22/1.21	1.65/1.66	1.1/1.29
Cecil Sandy Loam (Slope 2-6); E Fine kaolinitic thermic Typic Kanhapludults	Watkinsville	21.9	1.14/1.13	1.66/1.67	1.4/1.42
Cecil Sandy Clay Loam; (Slope 2-6); SE Fine kaolinitic thermic Typic Kanhapludults	Watkinsville	28.1	1.23/1.2	1.66/1.67	1.41/1.42
Conagree and Toccoa Soils Loamy, mixed, active, nonacid, thermic Oxyaquic Udifluvents	Eatonton	1.2	1.33/1.4	1.5/1.5	1.45/1.46
Davidson Loam (Slope 2-6); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	23.0	1.17/1.21	1.59/1.39	1.39/1.51
Davidson Clay Loam (Slope 6-10); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	20.0	1.3/1.4	1.67/1.53	1.36/1.36
Davidson Clay Loam (Slope 10-25); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	6.6	1.14/1.14	1.55/1.67	1.57/1.54
Enon soils; E Fine, mixed, active, thermic Ultic Hapludalfs	Eatonton	9.0	1.0/0.99	1.45/1.52	1.52/1.45
Iredell Loam (Slope 2-6) Fine, mixed, active, thermic, Oxyaquic Vertic Hapludalfs	Eatonton	10.1	1.68/1.73	1.53/1.31	1.21/1.53
Pacolet Sandy Clay Loam (Slope 6-10); SE Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	30.6	1.12/1.14	1.65/1.41	1.39/1.67
Pacolet Sandy Clay Loam (Slope 10-15); SE Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	3.7	1.33/1.21	1.57/1.39	1.28/1.59
Wilkes Sandy Loam (Slope 2-10); E Loamy, mixed, active, thermic shallow Typic Hapludalfs	Eatonton	14.0	1.44/1.41	1.58/1.4	1.4/1.6

Table 2.2. Location, surface area, and mean and median loss-on-ignition (LOI) and permanganate oxidizable (POXC) carbon values in three soil depths for each soil type sampled. E, ME, and SE represent erosion classification according to NRCS soil survey classes “eroded,” “moderately eroded,” and “severely eroded,” respectively.

			0-5 cm Sample Depth		5-10 cm Sample Depth		10-20 cm Sample Depth	
Soil Type	**Location**	**Area (ha)**	LOI g C kg^{-1}	POXC mg C kg^{-1}	LOI g C kg^{-1}	POXC mg C kg^{-1}	LOI g C kg^{-1}	POXC mg C kg^{-1}
Cecil soils, (Slope 0-2); overwash Fine kaolinitic thermic Typic Kanhapludults	Watkinsville	0.5	9.63/9.06	1306/1281	6.07/5.74	743/739	5.83/5.74	583/586
Cecil Sandy Loam (Slope 2-6); E Fine kaolinitic thermic Typic Kanhapludults	Watkinsville	21.9	8.97/9.19	1263/1315	4.43/4.19	707/660	4.75/4.0	719/584
Cecil Sandy Clay Loam; (Slope 2-6); SE; Fine kaolinitic thermic Typic Kanhapludults	Watkinsville	28.1	9.429.53	1293/1354	5.0/4.47	701/657	5.04/4.23	701/582
Conagree and Toccoa Soils Loamy, mixed, active, nonacid, thermic Oxyaquic Udifluvents	Eatonton	1.2	5.37/4.98	1306/842	4.07/4.07	521/555	4.28/4.23	361/378
Davidson Loam (Slope 2-6); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	23.0	9.94/8.97	1093/1149	6.17/5.92	695/693	5.85/5.74	591/571
Davidson Clay Loam (Slope 6-10); ME; Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	20.0	8.72/8.58	944/915	5.83/5.36	807/698	7.49/7.41	621/571
Davidson Clay Loam (Slope 10-25); ME; Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	6.6	8.38/8.09	952/974	4.94/4.91	780/793	4.03/4.1	544/526
Enon soils; E Fine, mixed, active, thermic Ultic Hapludalfs	Eatonton	9.0	5.74/5.74	961/961	4.98/4.98	516/516	2.01/2.01	433/433
Iredell Loam (Slope 2-6) Fine, mixed, active, thermic, Oxyaquic Vertic Hapludalfs	Eatonton	10.1	9.33/7.62	1061/1002	6.45/6.46	662/691	6.68/6.64	549/528
Pacolet Sandy Clay Loam (Slope 6-10); SE; Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	30.6	9.45/9.33	1319/1397	4.97/4.55	733/690	4.77/4.31	641/557
Pacolet Sandy Clay Loam (Slope 10-15); SE; Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	3.7	8.26/8.28	1194/1285	5.78/4.89	752/704	4.48/4.17	512/567
Wilkes Sandy Loam (Slope 2-10); E Loamy, mixed, active, thermic shallow Typic Hapludalfs	Eatonton	14.0	7.13/7.44	974/993	5.24/4.56	627/609	5.9/6.41	539/507

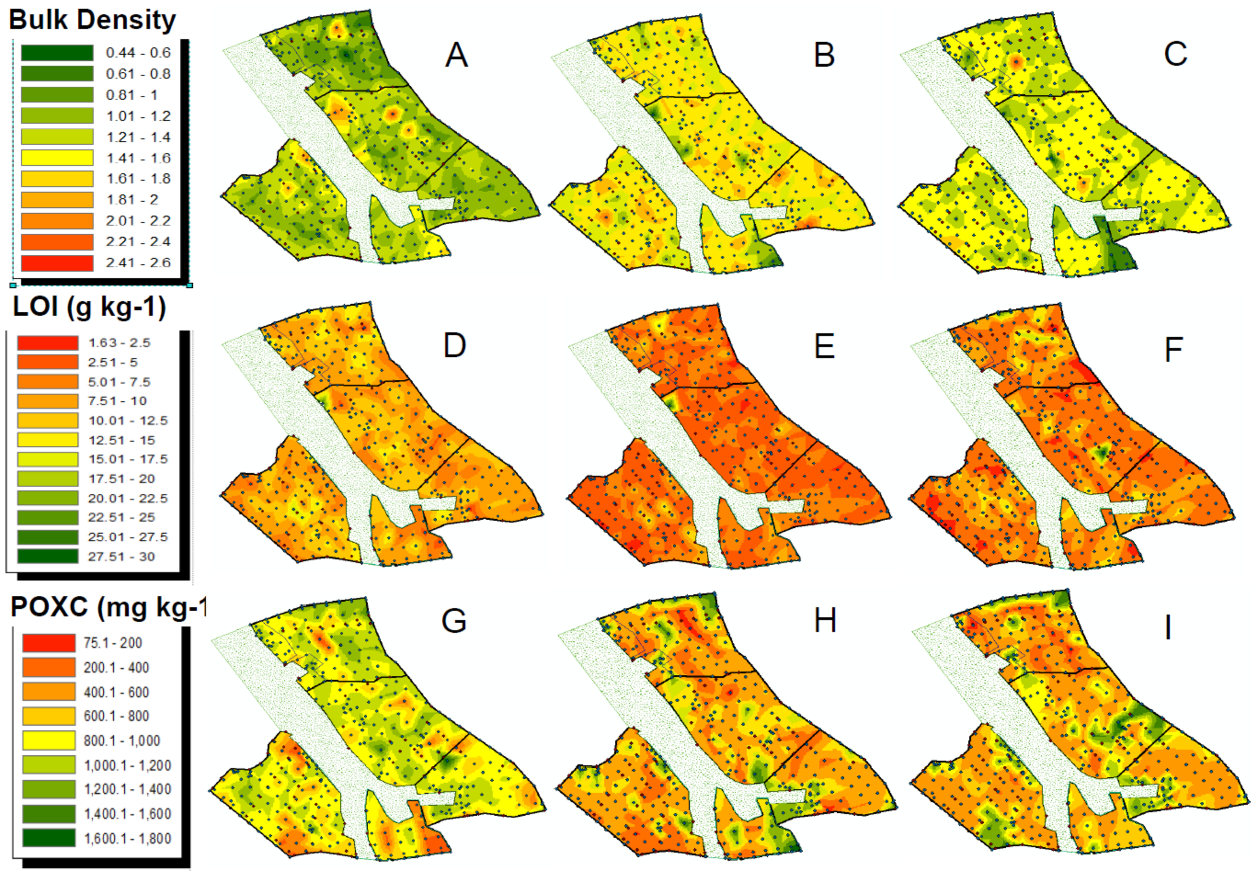

Figure 2.1. Bulk density (A,B,C), loss-on-ignition carbon (LOI; D,E,F), and permanganate oxidizable carbon (POXC; G,H,I) for each sampling depth (0-5,5-10, and 10-20 cm, respectively) for the Watkinsville North Unit. Soil bulk density is displayed in g cm^{-3}, with low bulk densities displayed in green (0.6 – 1.2 g cm^{-3}) and high bulk densities (2.21 – 2.6 g cm^{-3}) shown in red. LOI values are displayed in g kg^{-1}, with low LOI values displayed in red (1-5) and high LOI values (20-30) displayed in green. POXC values are displayed in mg kg^{-1}, with low POXC values (75-400) displayed in red and high POXC values (1400-1800) in green.

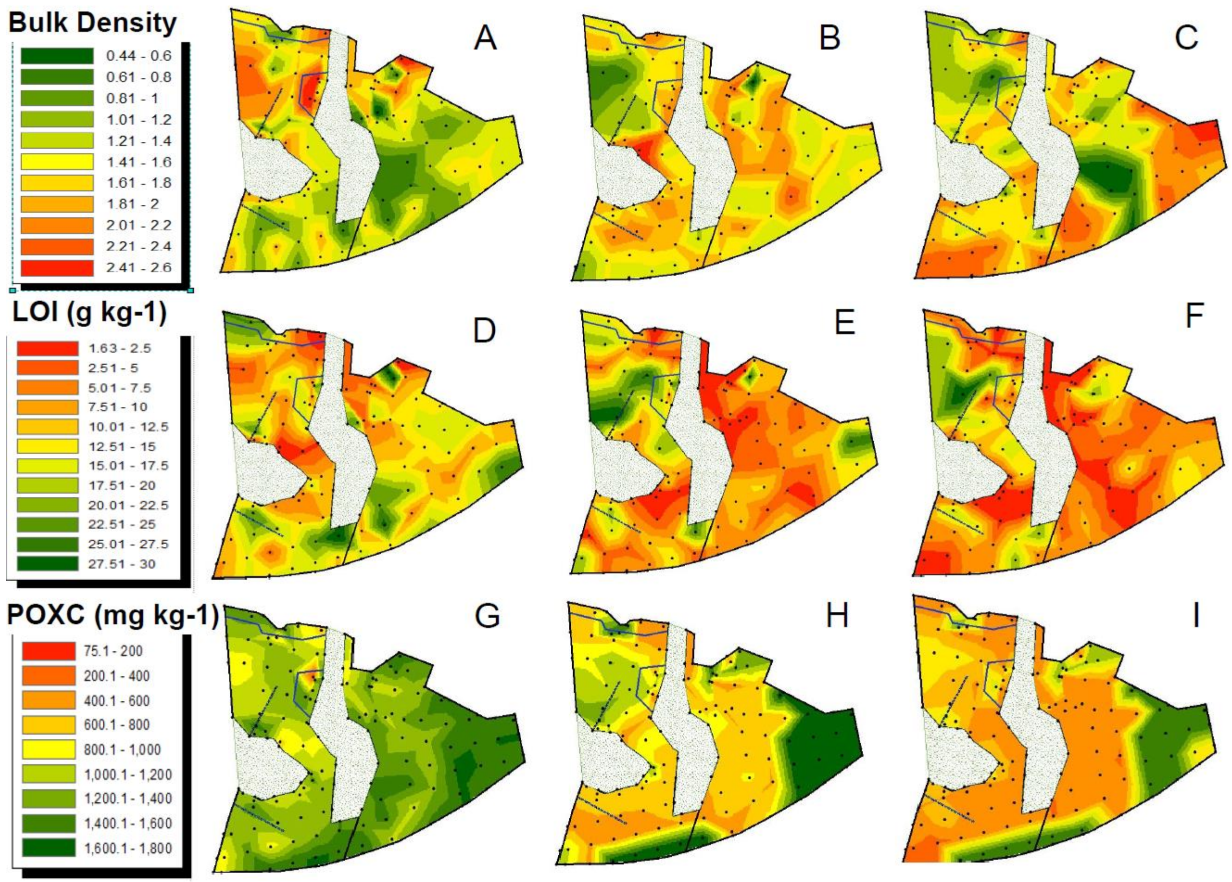

Figure 2.2. Bulk density (A,B,C), loss-on-ignition carbon (LOI; D,E,F), and permanganate oxidizable carbon (POXC; G,H,I) for each sampling depth (0-5, 5-10, and 10-20 cm, respectively) for the Watkinsville West Unit. Soil bulk density is displayed in g cm^{-3}, with low bulk densities displayed in green (0.6 – 1.2 g cm^{-3}) and high bulk densities (2.21 – 2.6 g cm^{-3}) shown in red. LOI values are displayed in g kg^{-1}, with low LOI values displayed in red (1-5) and high LOI values (20-30) displayed in green. POXC values are displayed in mg kg^{-1}, with low POXC values (75-400) displayed in red and high POXC values (1400-1800) in green.

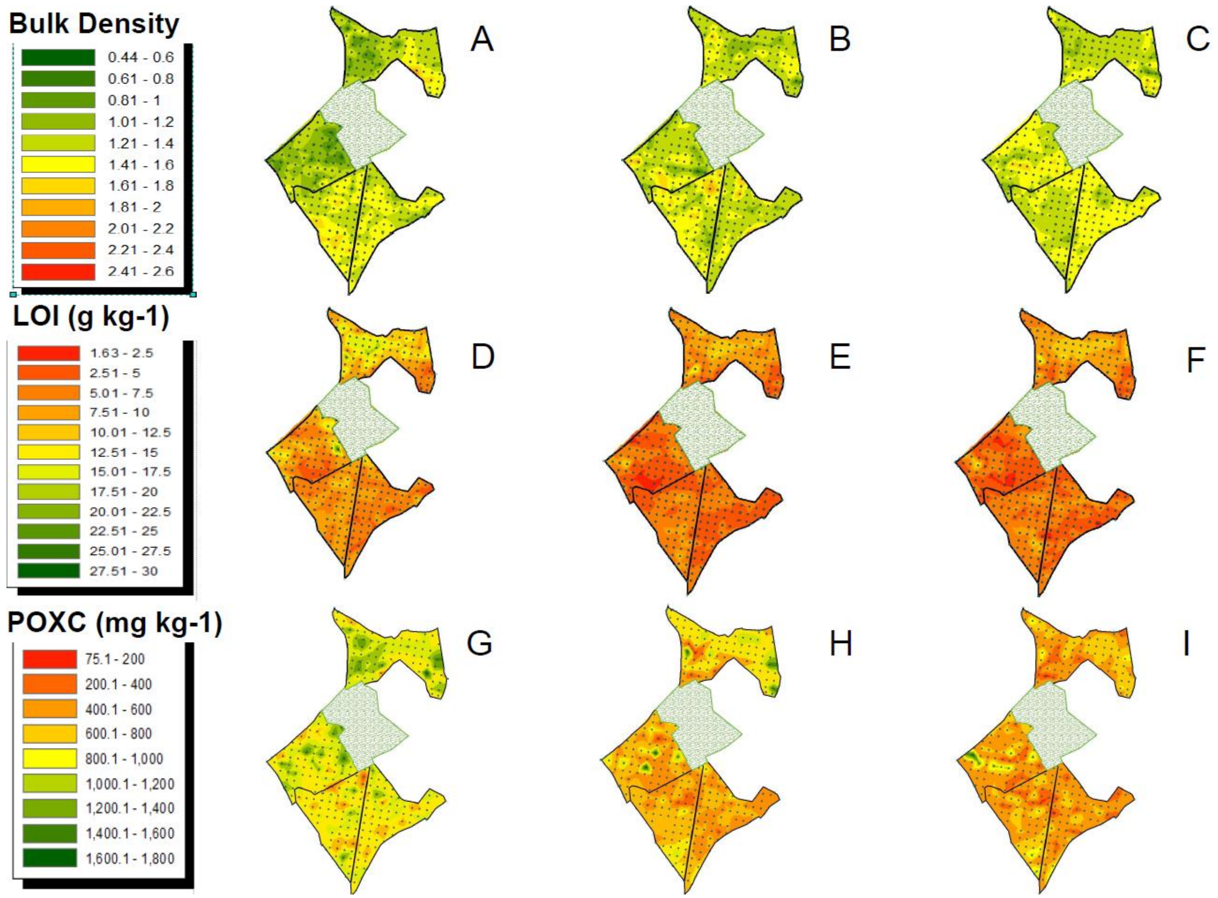

Figure 2.3. Bulk density (A,B,C), loss-on-ignition carbon (LOI; D,E,F), and permanganate oxidizable carbon (POXC; G,H,I) for each sampling depth (0-5,5-10, and 10-20 cm, respectively) for the Eatonton Unit. Soil bulk density is displayed in g cm^{-3}, with low bulk densities displayed in green (0.6 – 1.2 g cm^{-3}) and high bulk densities (2.21 – 2.6 g cm^{-3}) shown in red. LOI values are displayed in g kg^{-1}, with low LOI values displayed in red (1-5) and high LOI values (20-30) displayed in green. POXC values are displayed in mg kg^{-1}, with low POXC values (75-400) displayed in red and high POXC values (1400-1800) in green.

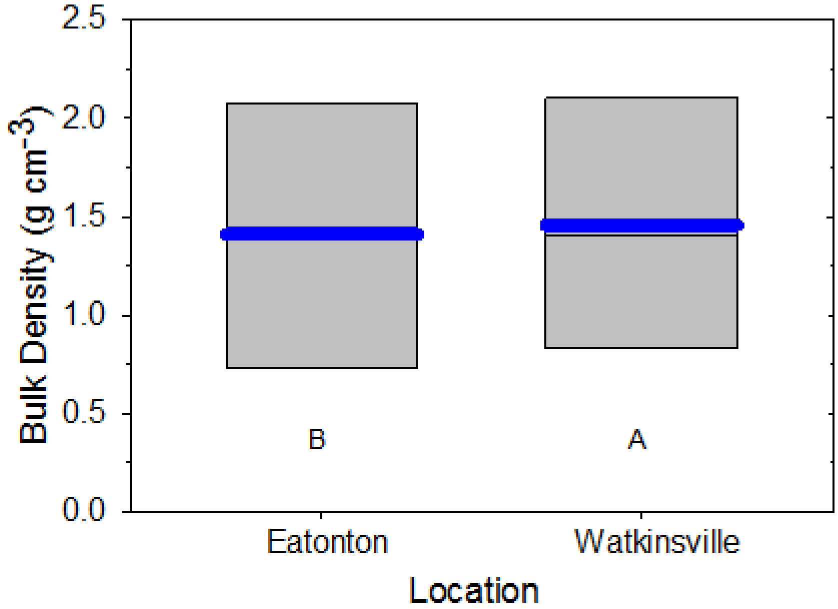

Figure 2.4. Box plots for bulk density at all depths (0-5, 5-10, and 10-20 cm) for the Eatonton and Watkinsville sites. Blue lines are means and thin black lines are medians. Different letter indicate a significant difference between means according to pairwise comparisons using Tukey’s test at p=0.05.

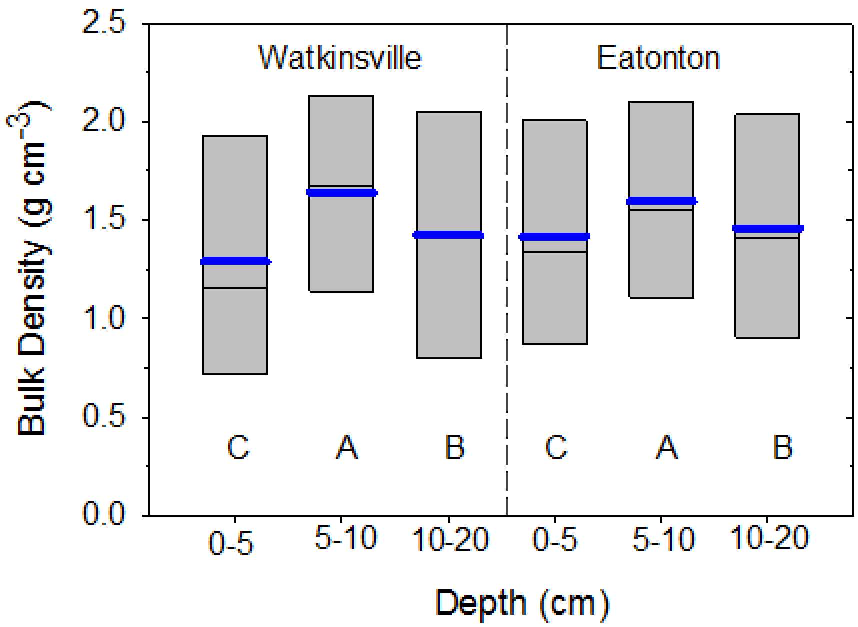

Figure 2.5. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for bulk density at each sampling depth in Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

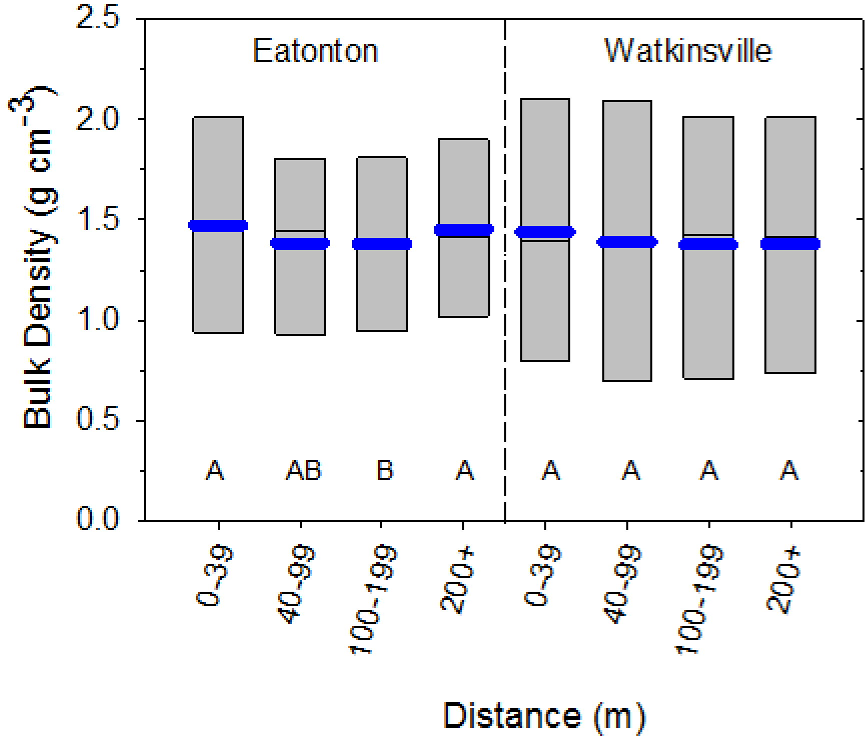

Figure 2.6. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, > 200 m from the nearest hay source in Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

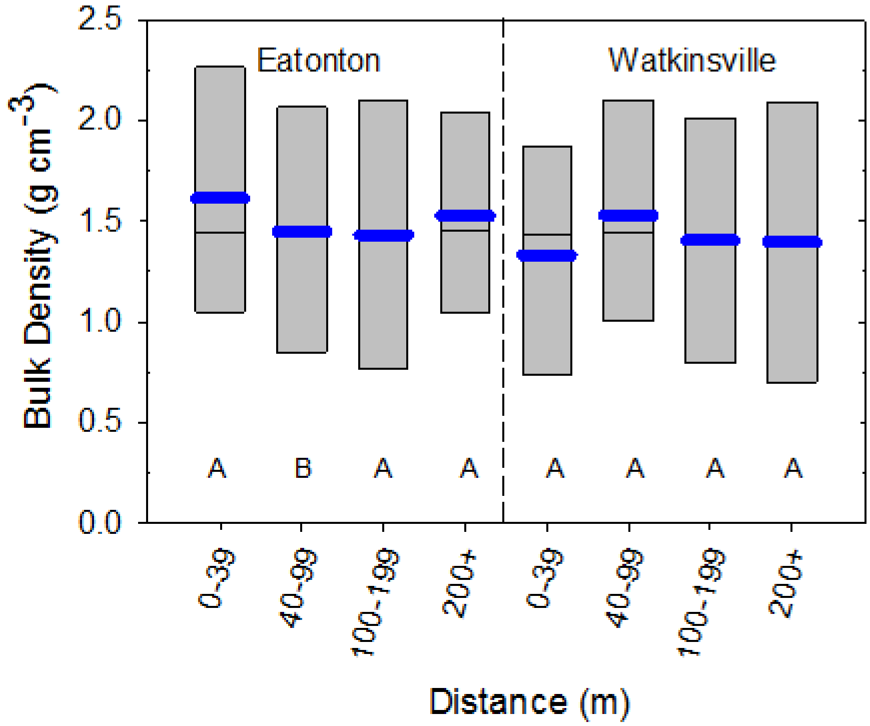

Figure 2.7. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

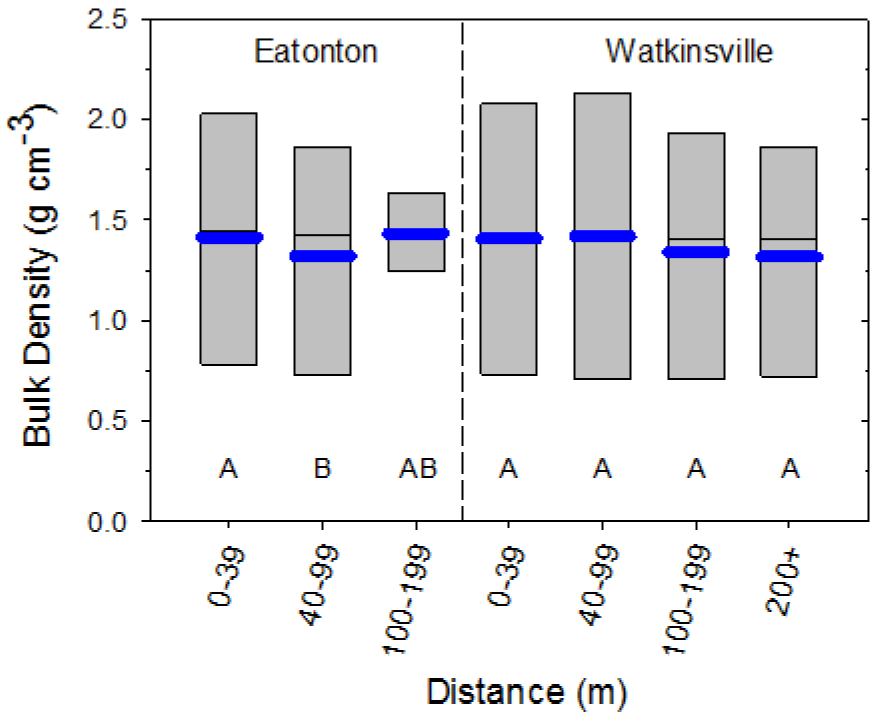

Figure 2.8. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest shade source in Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

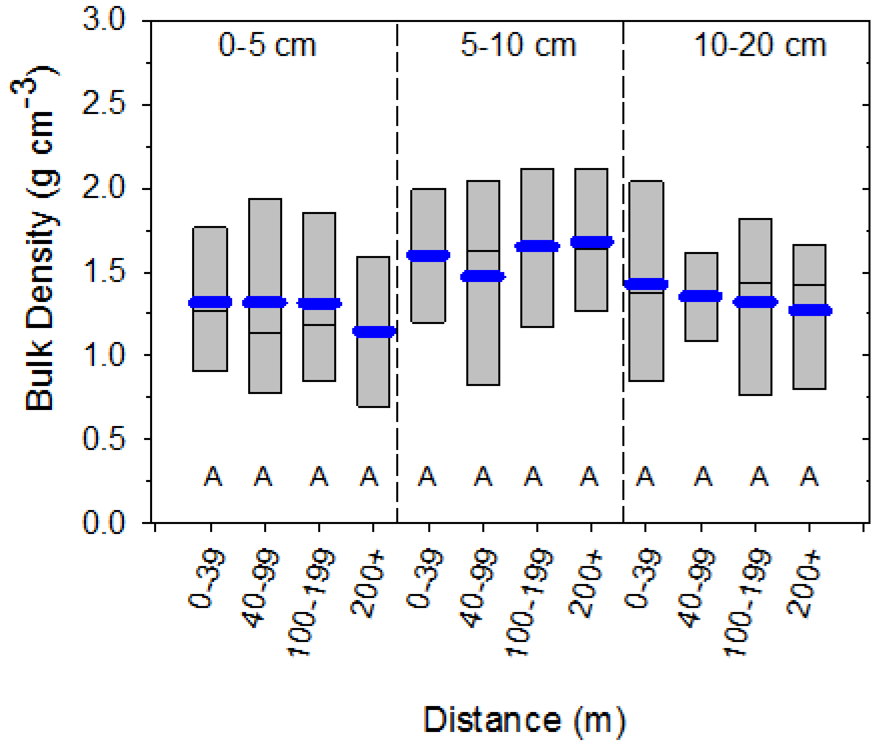

Figure 2.9. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m for each sampling depth from the nearest hay source in the Watkinsville site. Within each depth, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

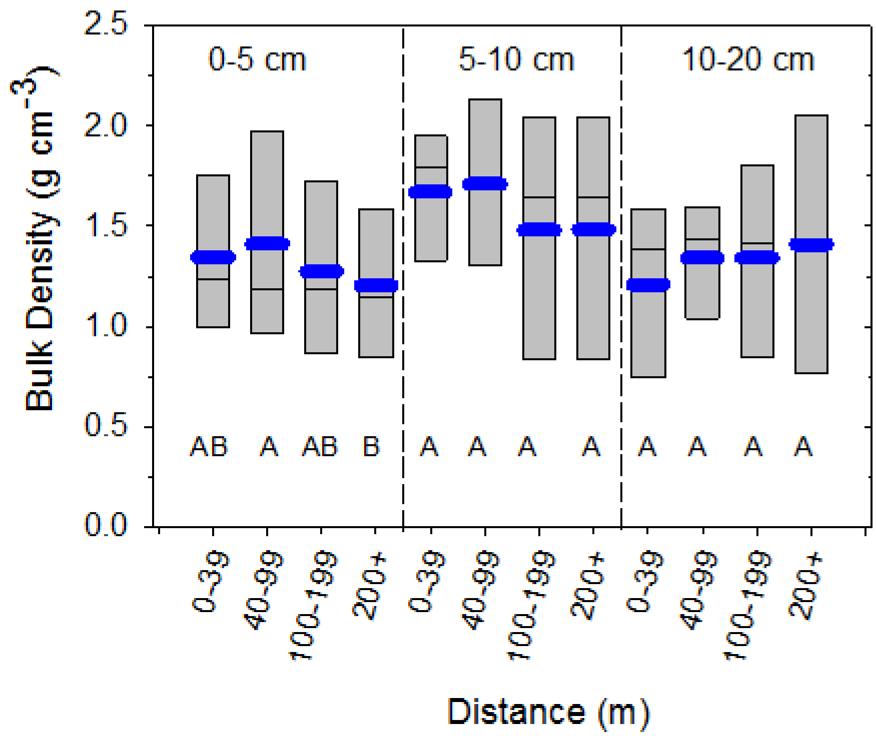

Figure 2.10. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m for each sampling depth from the nearest water source in the Watkinsville site. Within each depth, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

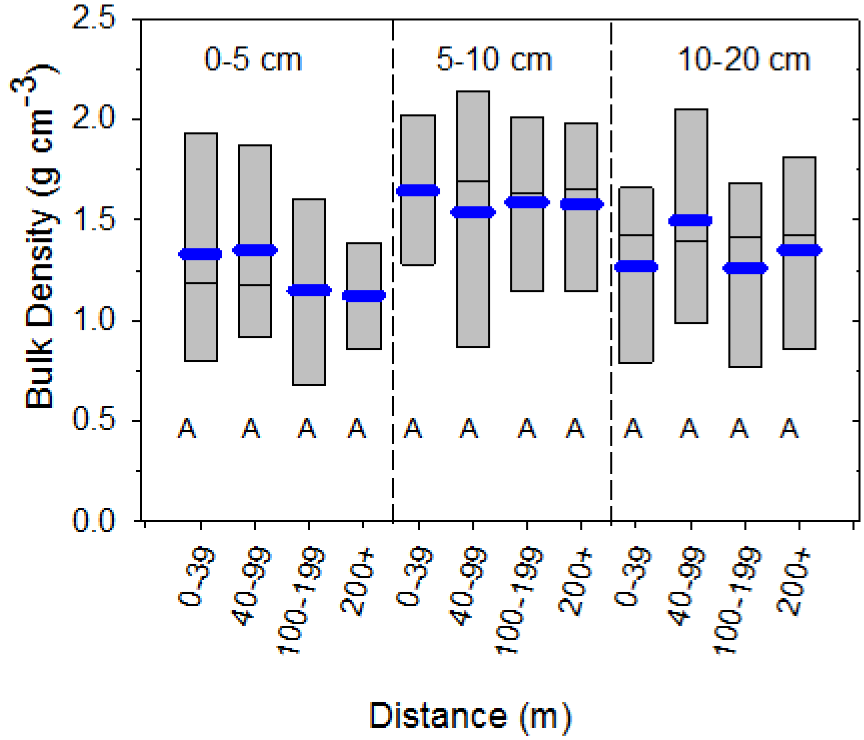

Figure 2.11. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m for each sampling depth from the nearest shade source in the Watkinsville site. Within each depth, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

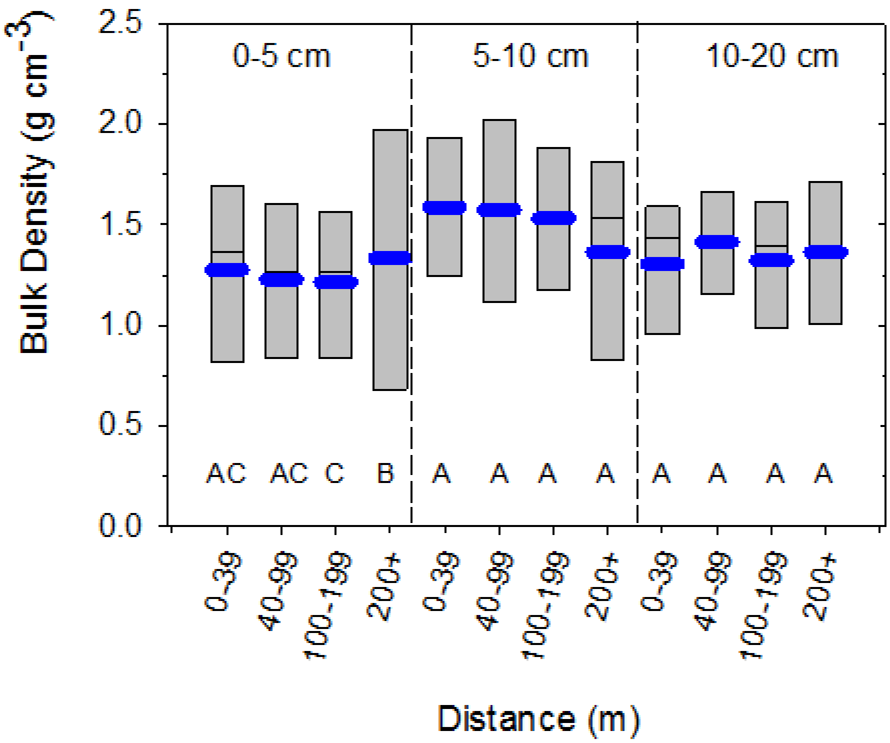

Figure 2.12. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m for each sampling depth from the nearest hay source in the Eatonton site. Within each depth, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

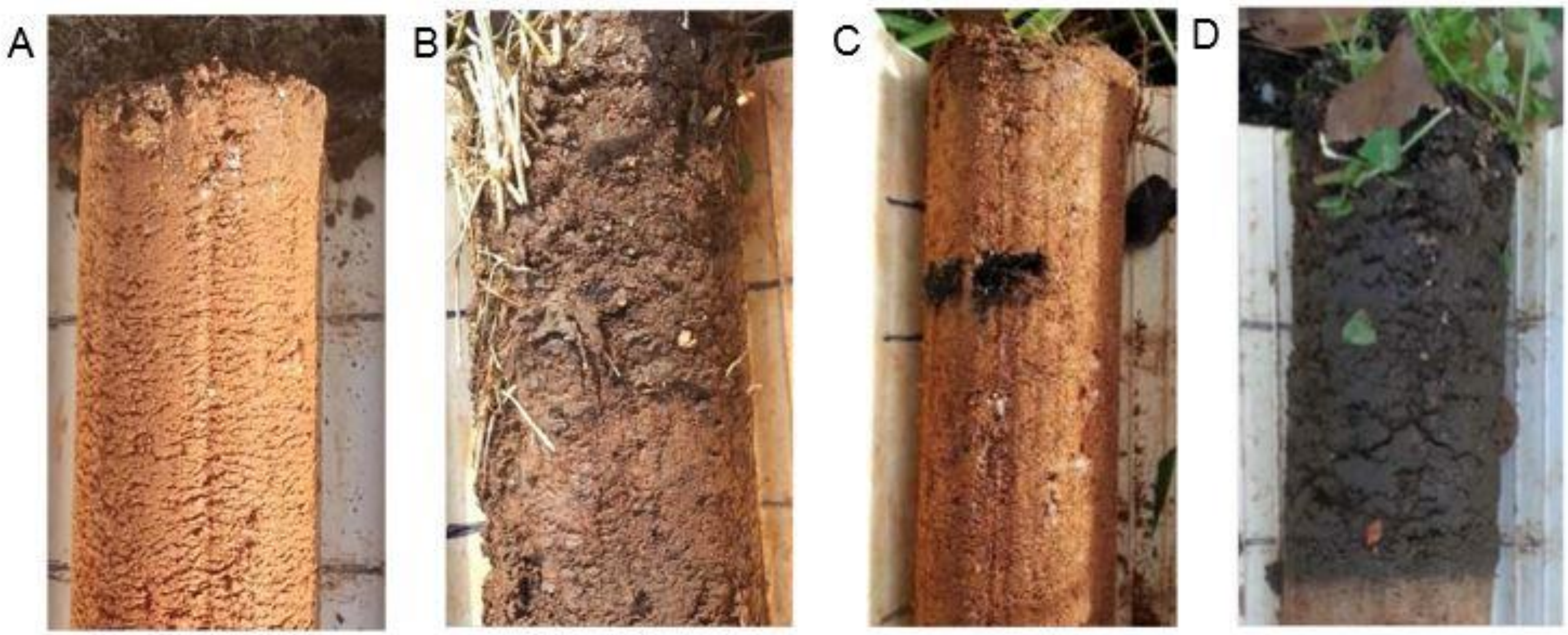

Figure 2.13. Soil Cores representing the variability of pasture soils in the 0-10 cm depth: A) a cattle camping area with hay, water, and shade in a concentrated flow path; B) an uphill hay feeding area; C) a low-lying soil with manganese concretions; and D) an uphill cattle camping area with hay, water, and shade sources.

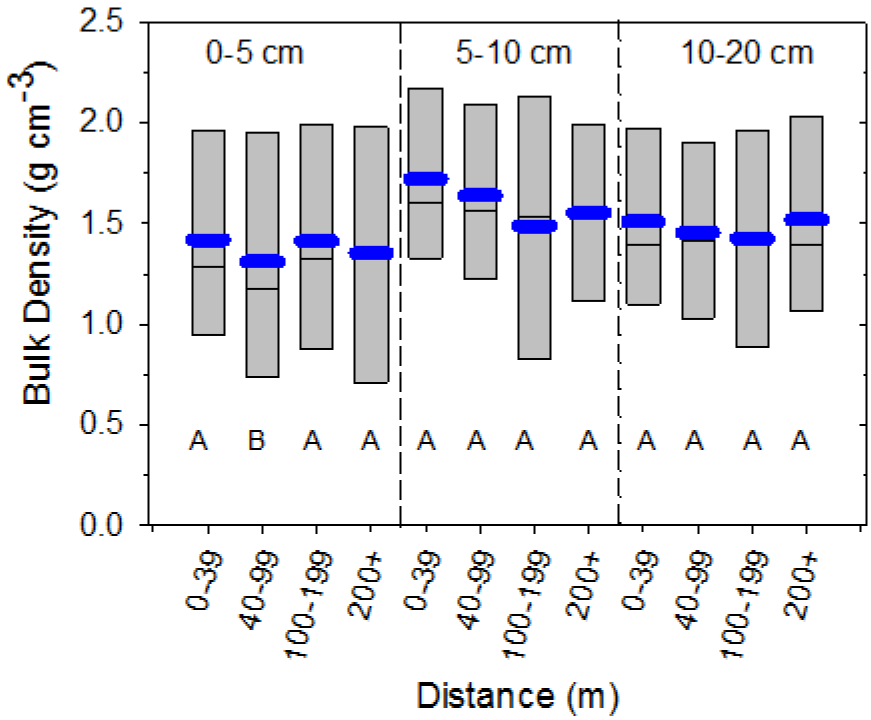

Figure 2.14. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, 100-199, and > 200 m for each sampling depth from the nearest water source in the Eatonton site. Within each depth, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

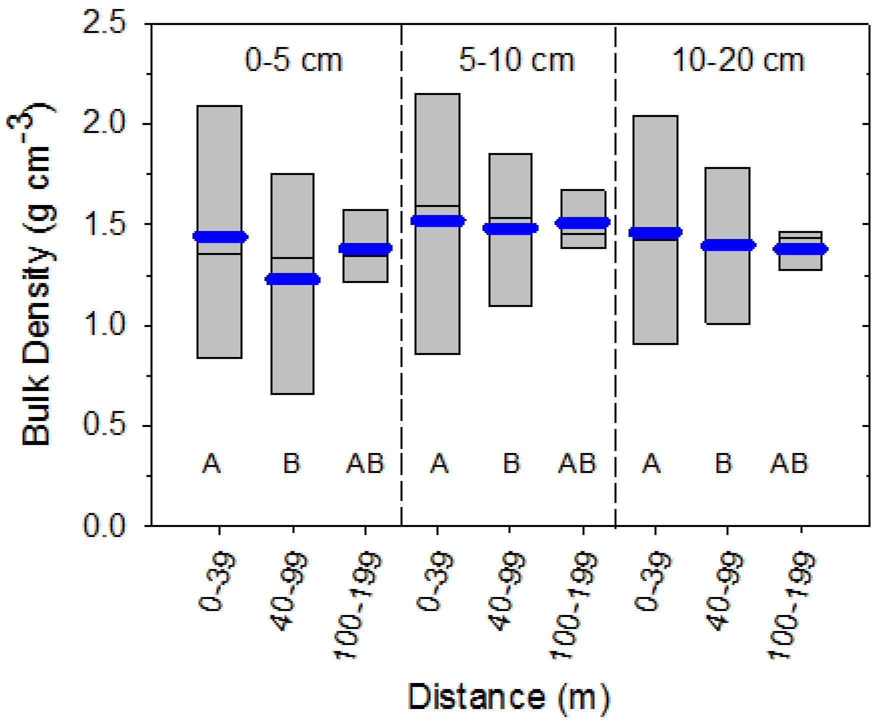

Figure 2.15. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for bulk density in geometric distances 0-39, 40-99, and 100-199 m for each sampling depth from the nearest shade source in the Eatonton site. Within each depth, different letters indicate significant differences between means according to pairwise comparisons using Tukey's test at p=0.05.

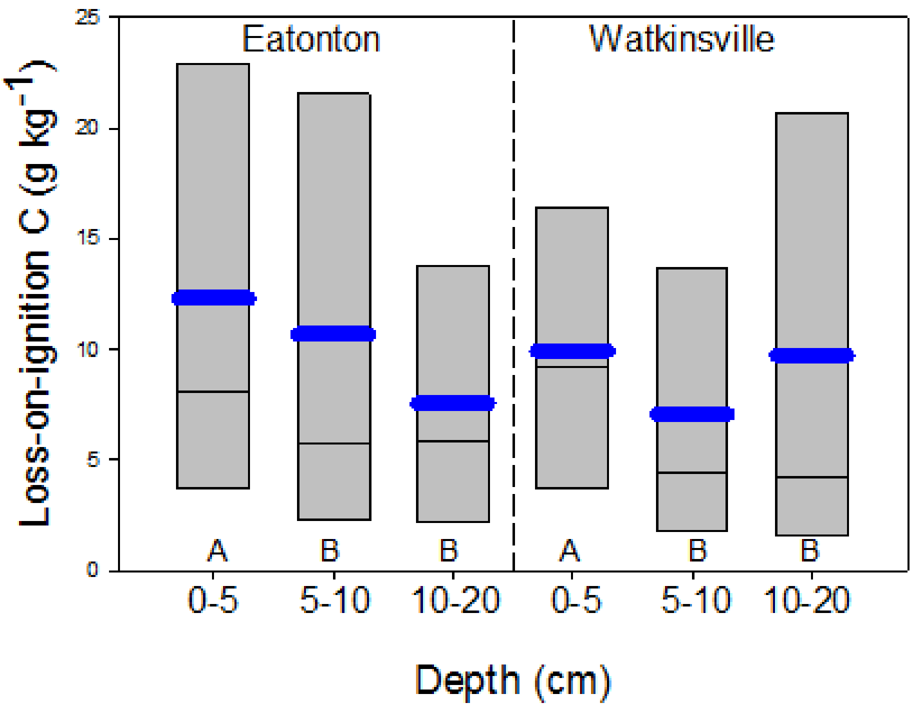

Figure 2.16. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition carbon for each sampling depth in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

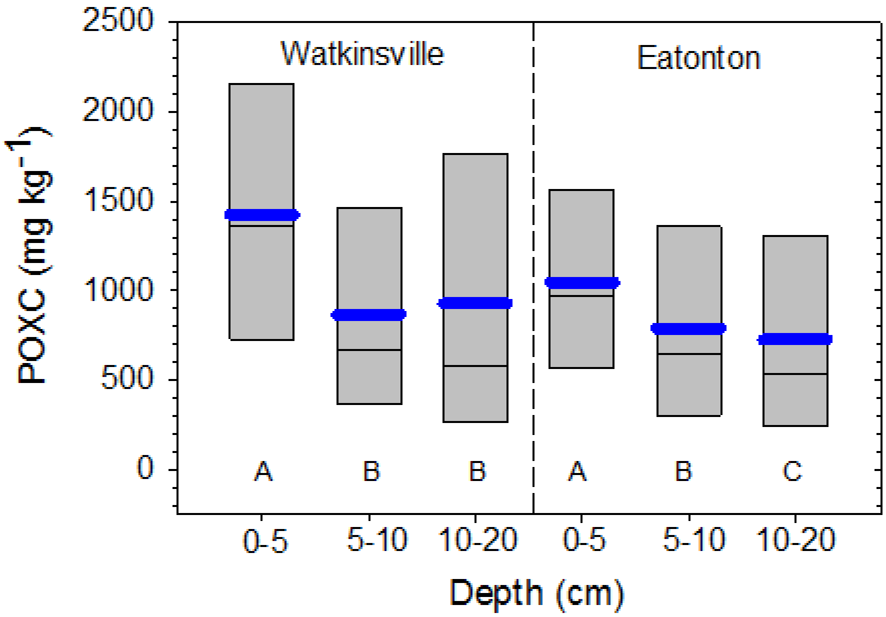

Figure 2.17. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) for each sampling depth in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

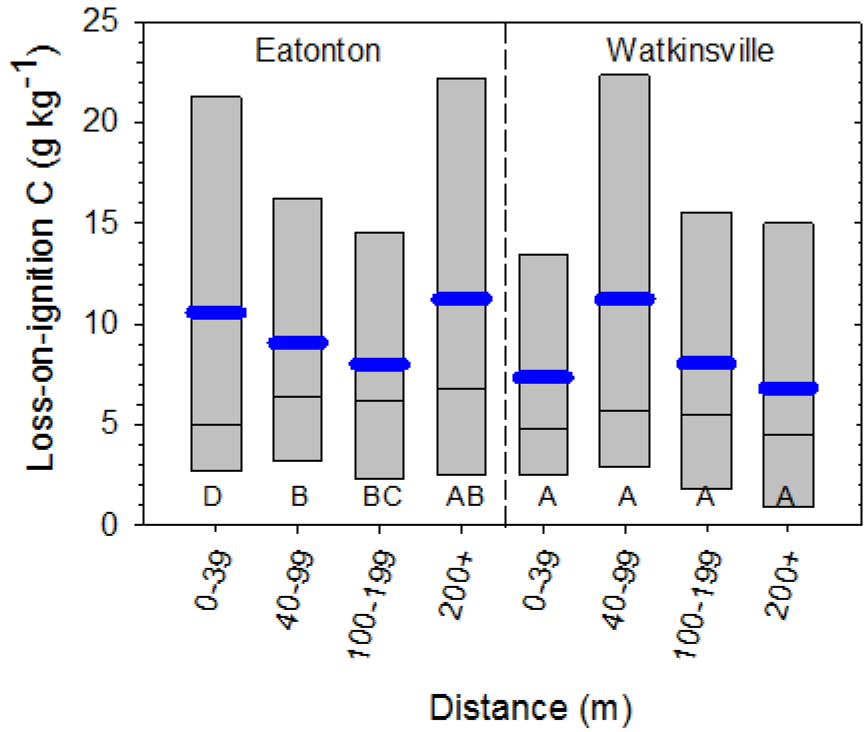

Figure 2.18. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

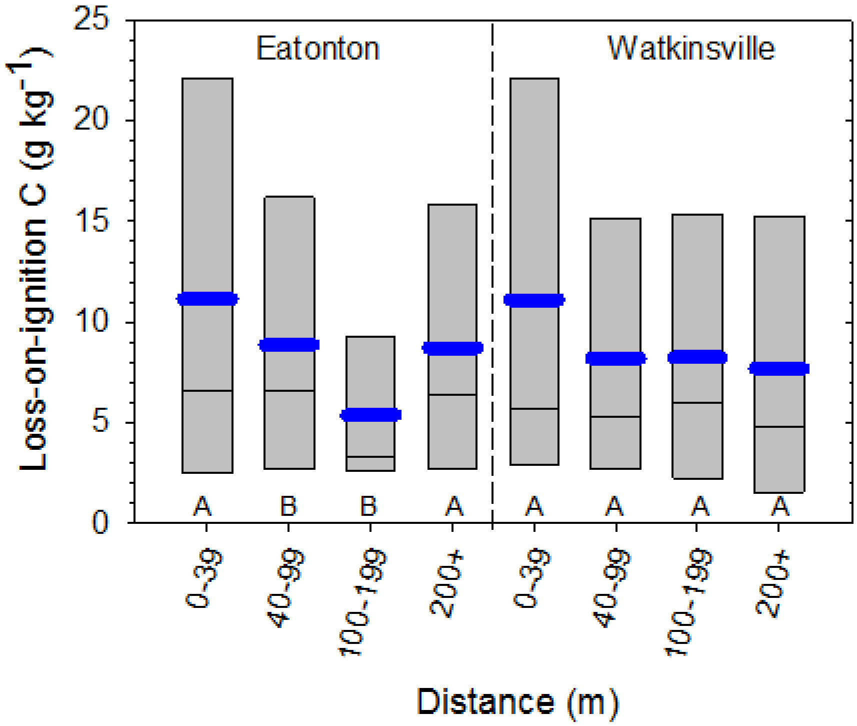

Figure 2.19. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

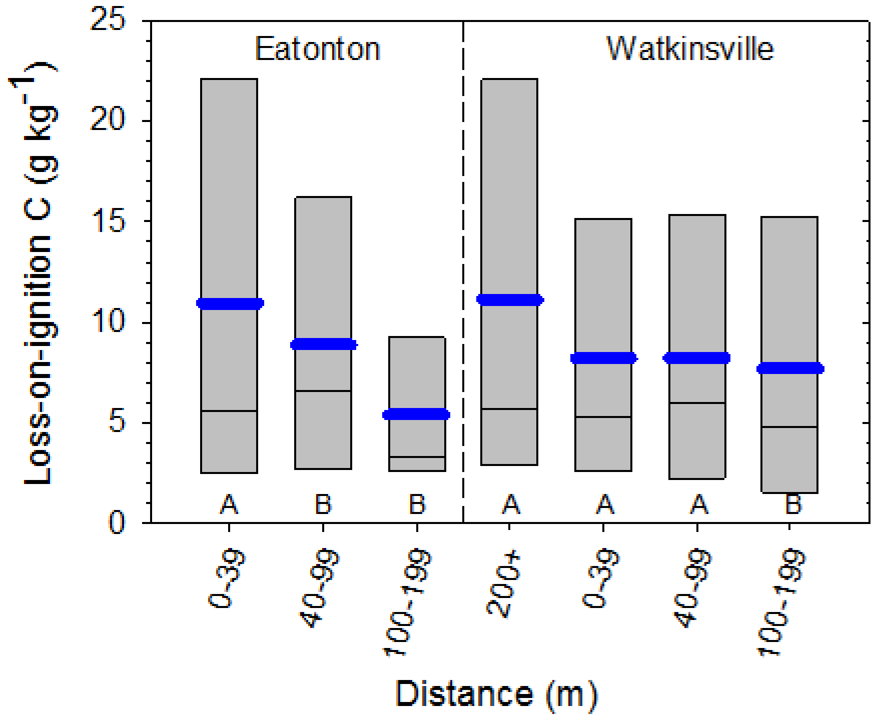

Figure 2.20. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest shade source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

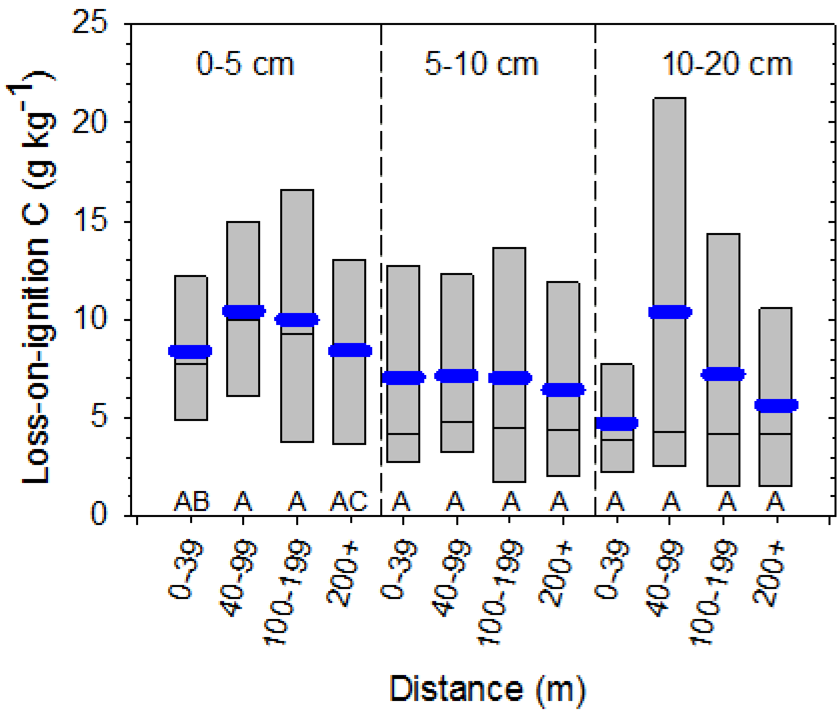

Figure 2.21. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

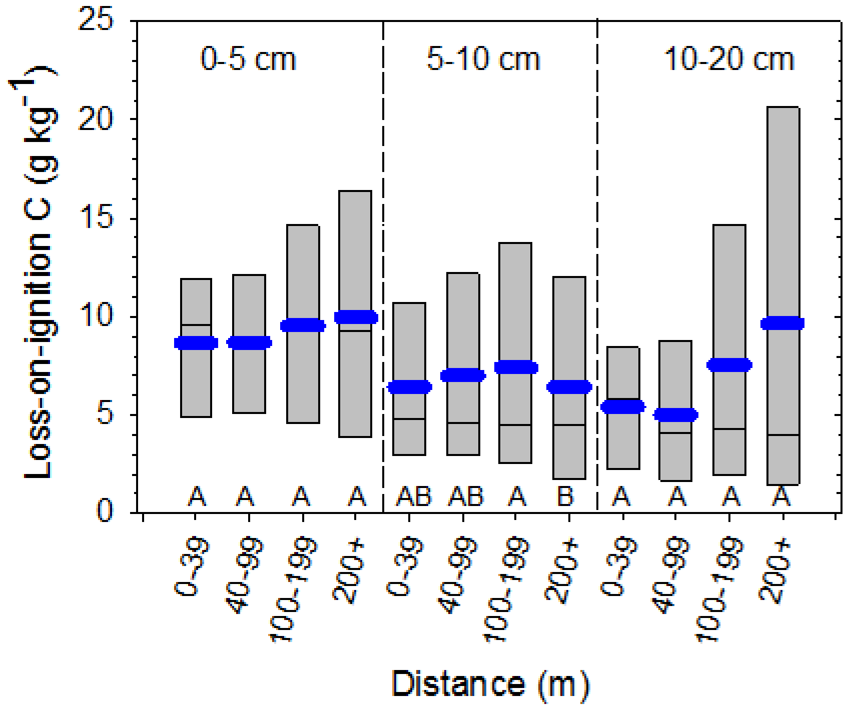

Figure 2.22. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

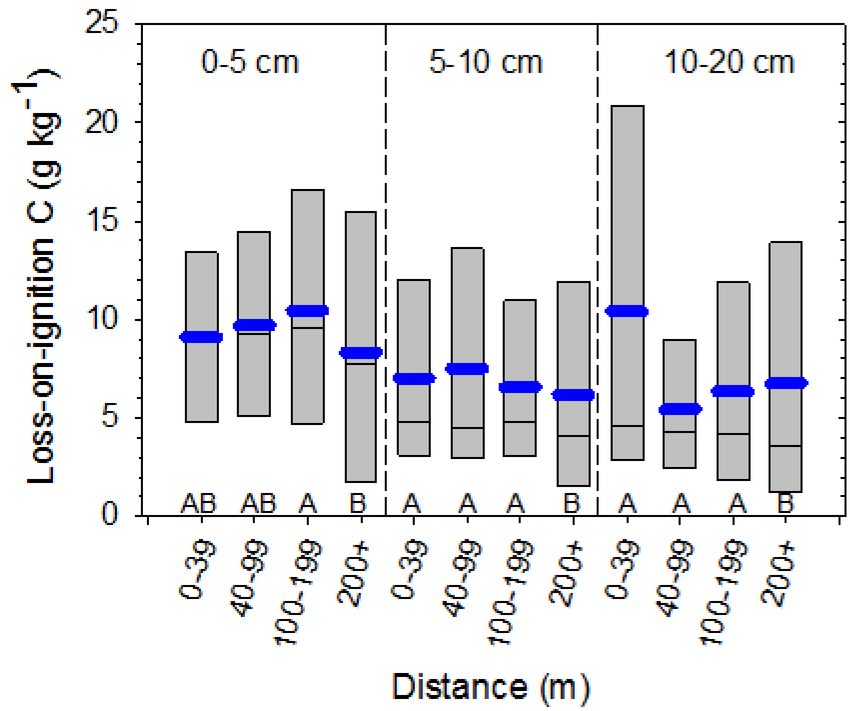

Figure 2.23. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest shade source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

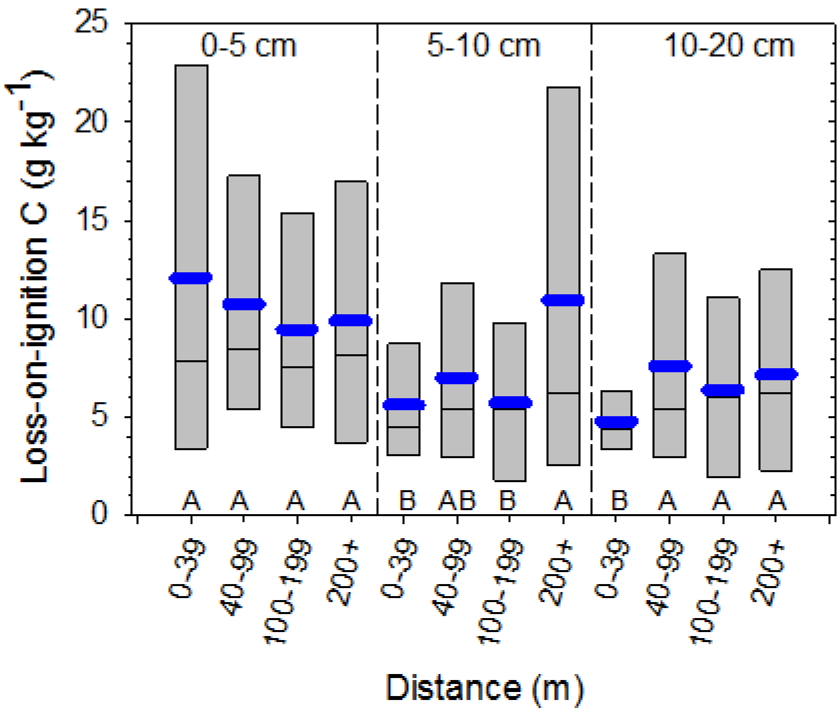

Figure 2.24. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Eatonton sites. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

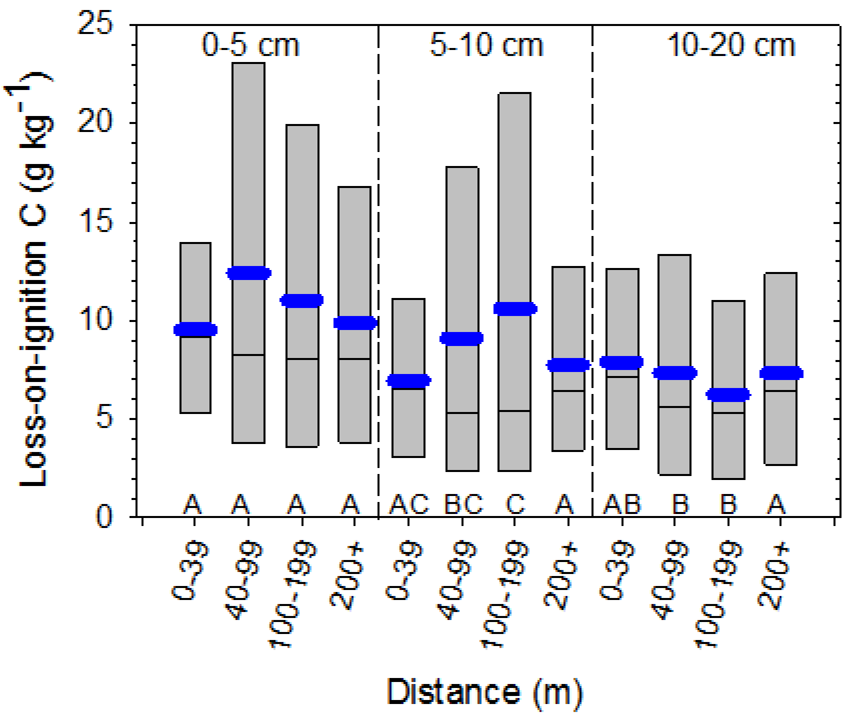

Figure 2.25. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

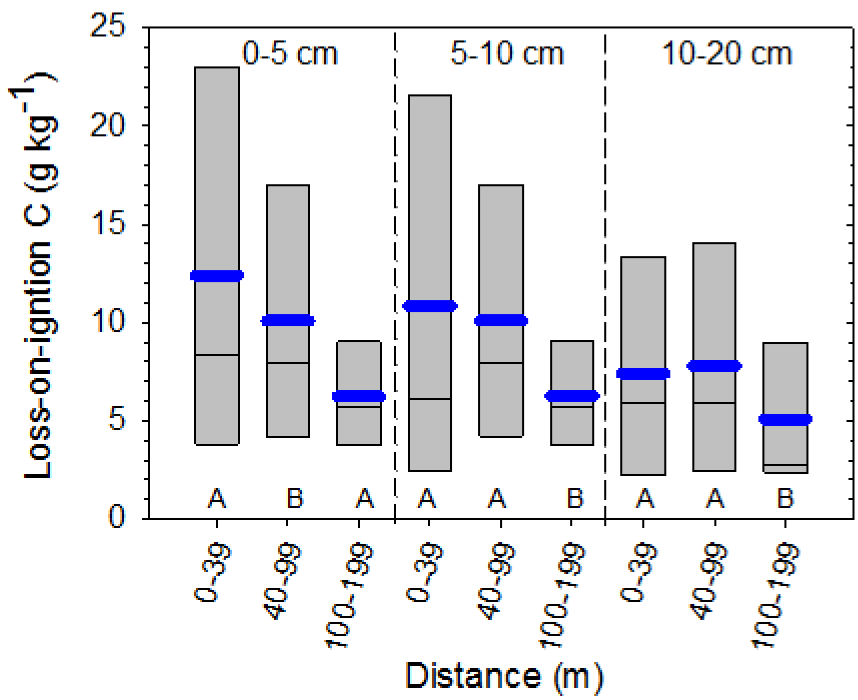

Figure 2.26. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of loss on ignition (LOI) carbon at each sampling depth in geometric distances 0-39, 40-99, and 100-199 m from the nearest shade source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

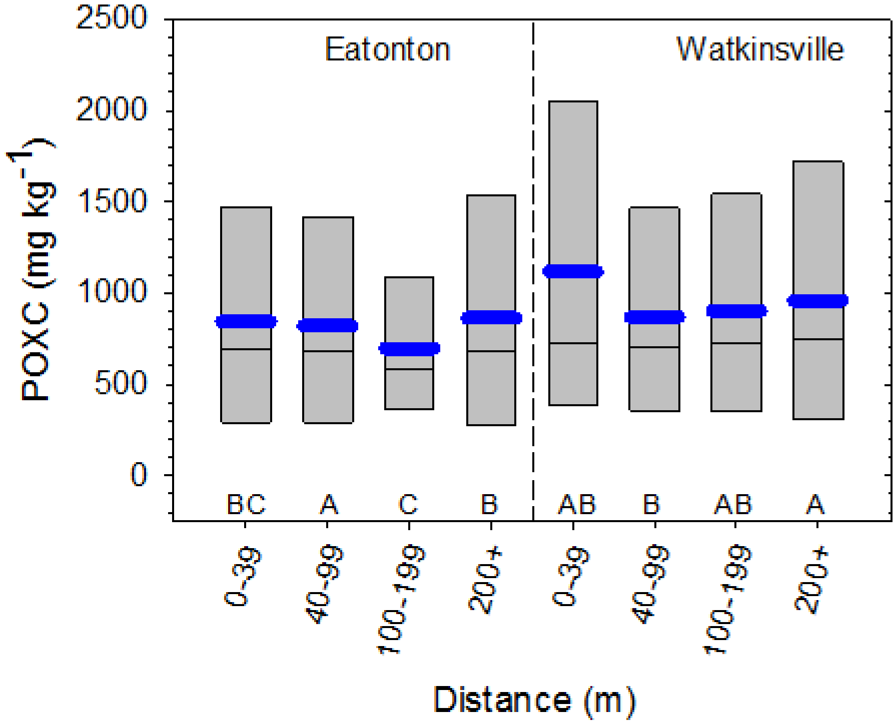

Figure 2.27. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Eatonton and Watkinsville sites. Within each site, different letters significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

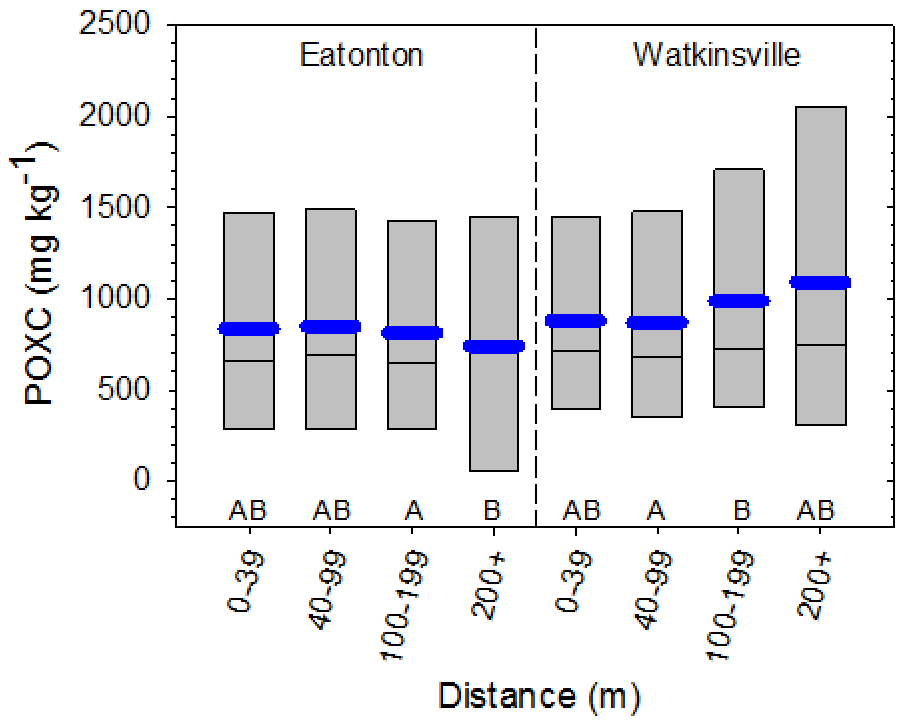

Figure 2.28. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

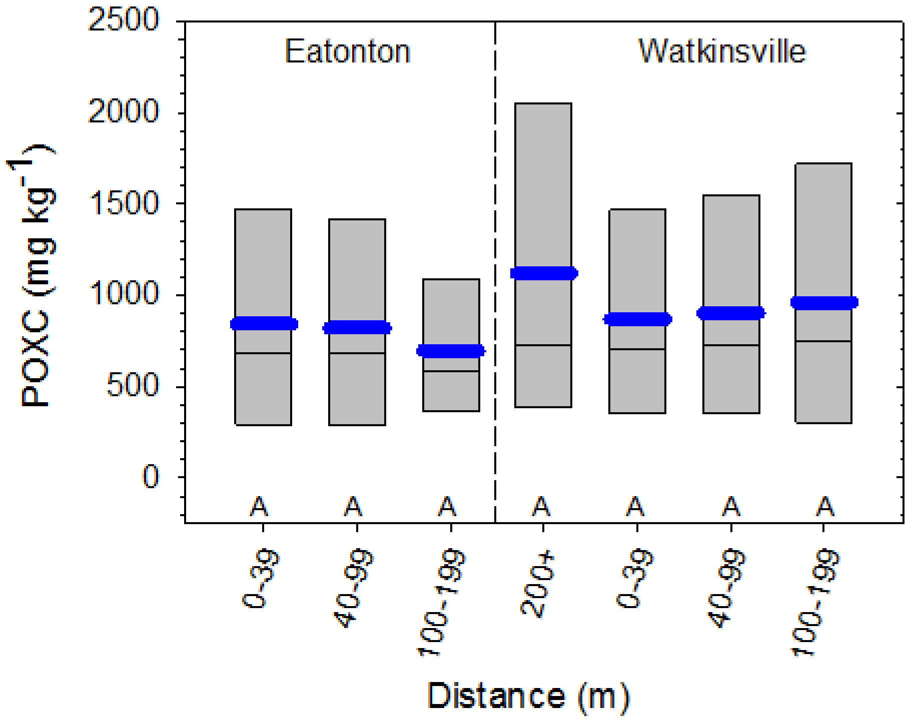

Figure 2.29. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) in geometric distances 0-39, 40-99, 100-199, and greater than 200 m from the nearest shade source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

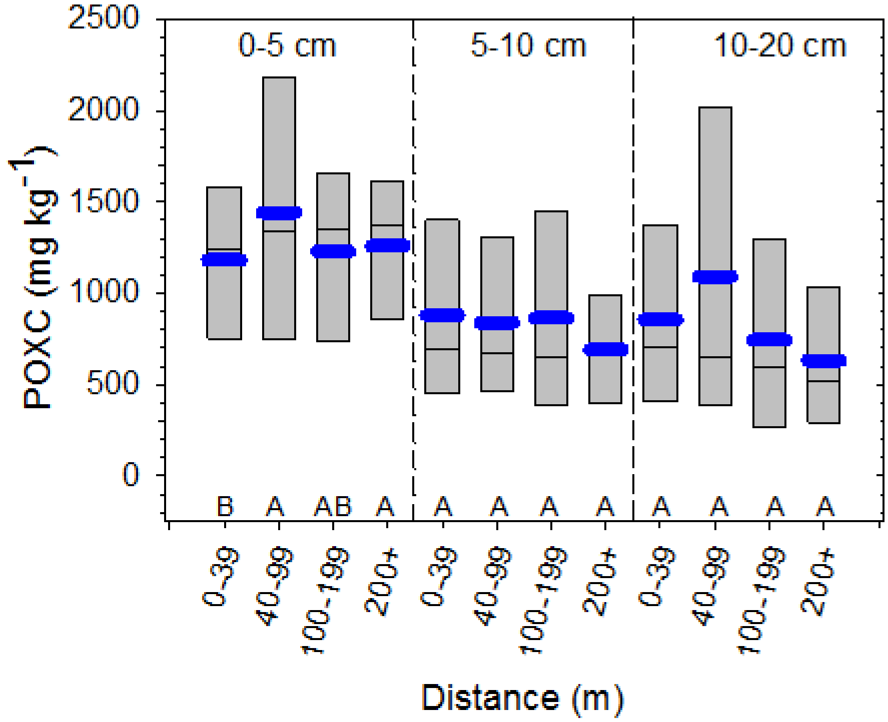

Figure 2.30. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

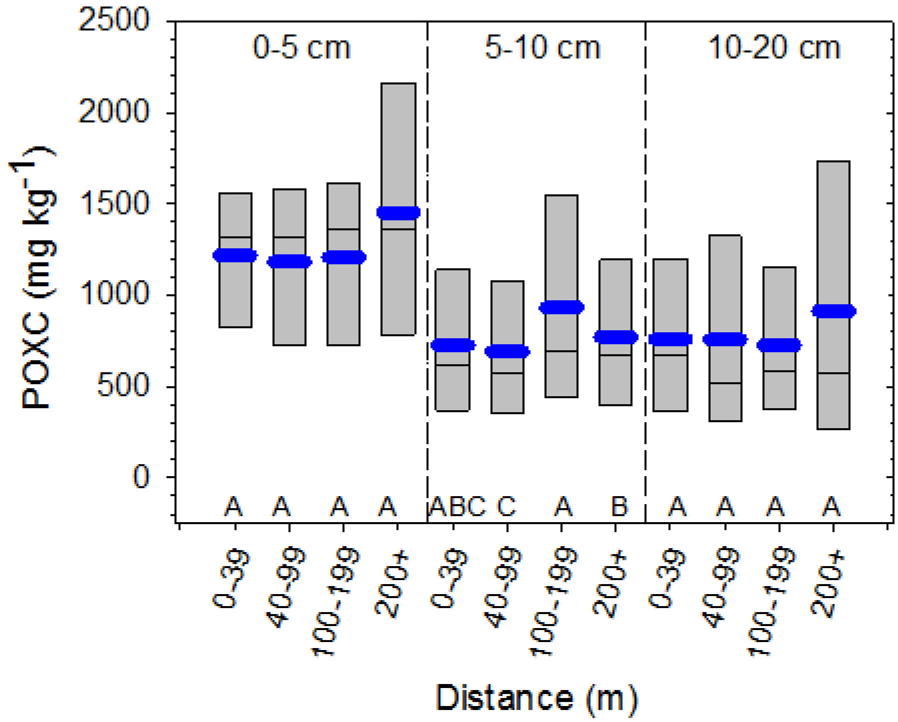

Figure 2.31. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

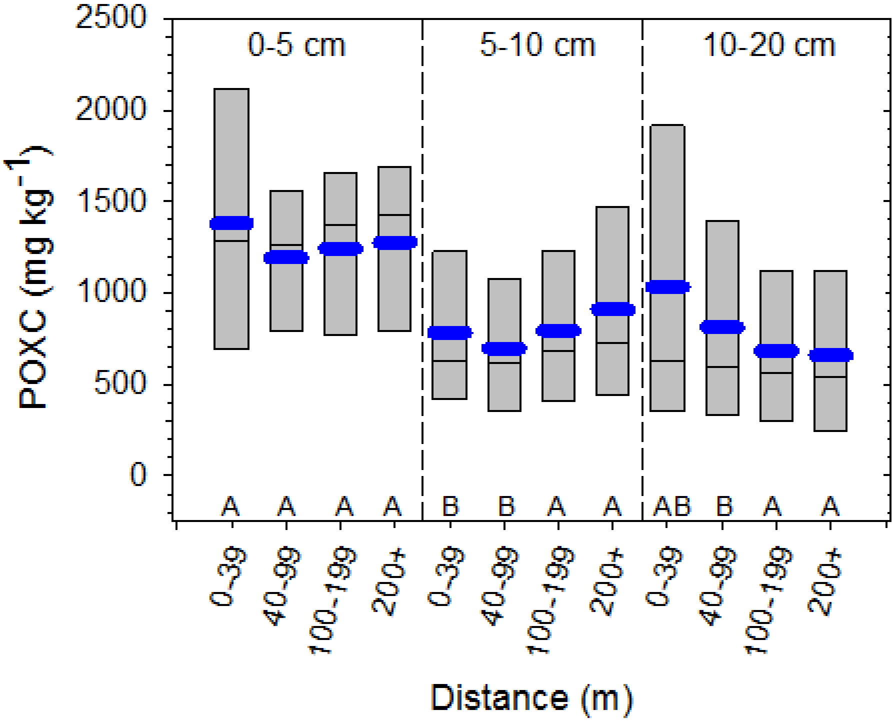

Figure 2.32. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest shade source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

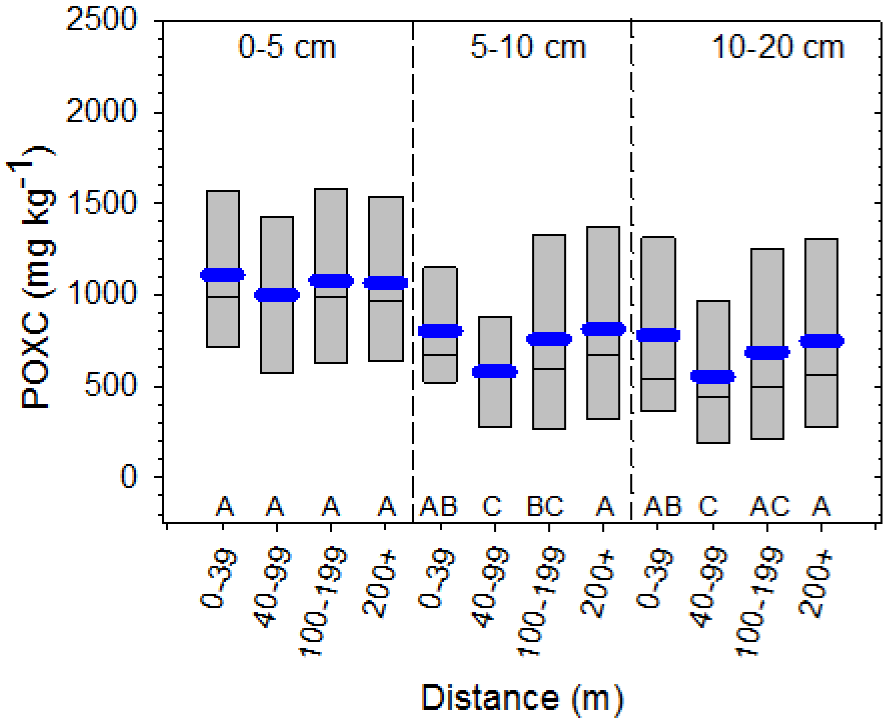

Figure 2.33. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

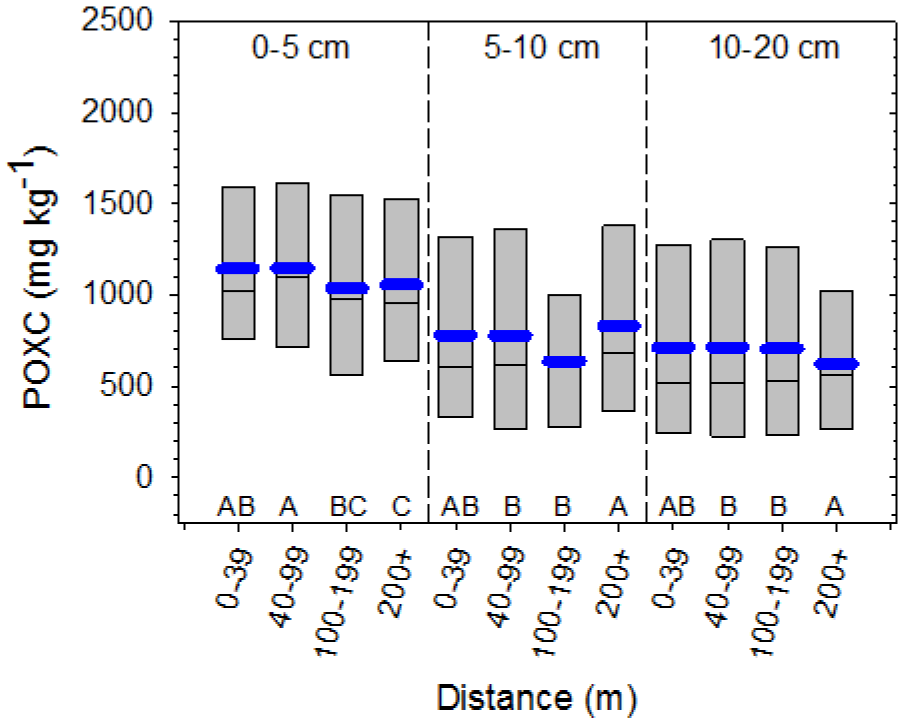

Figure 2.34. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

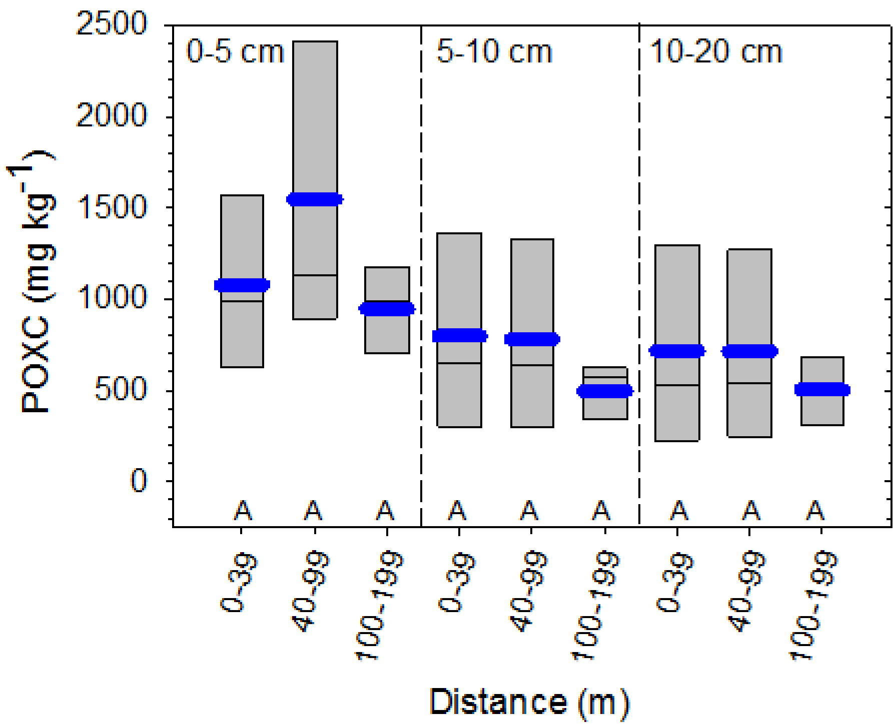

Figure 2.35. The first to third quartile (gray range), mean (bold blue line), and median (thin black line) for concentrations of permanganate oxidizable carbon (POXC) at each sampling depth in geometric distances 0-39, 40-99, and 100-199 m from the nearest shade source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

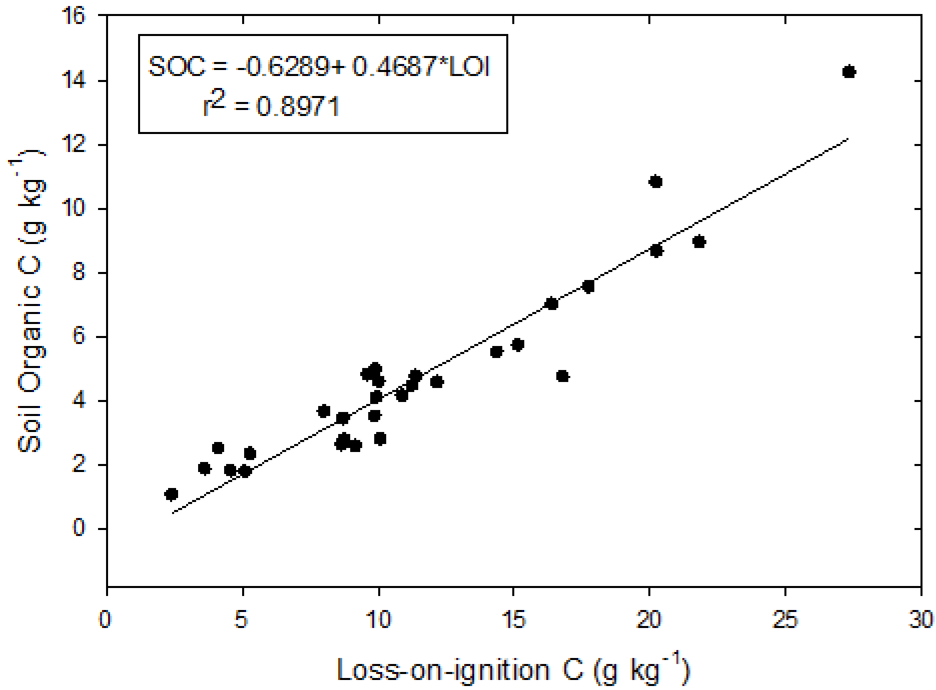

Figure 2.36. Relationship between soil organic carbon (measured by combustion, SOC) and loss on ignition carbon (LOI).

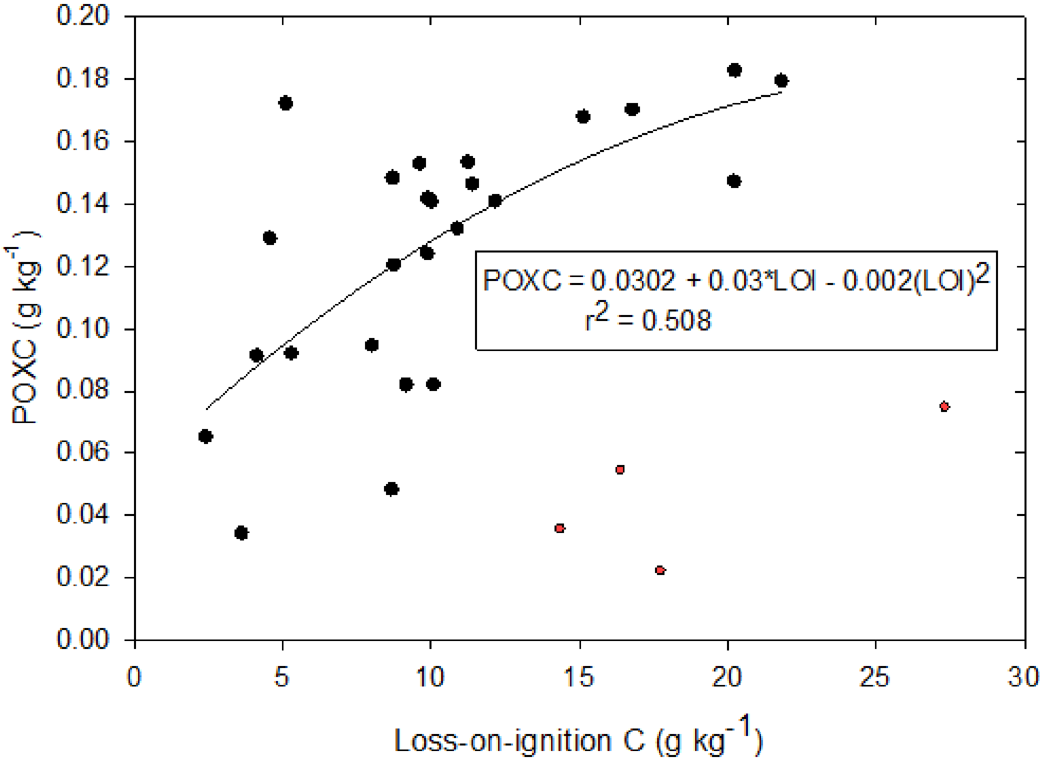

Figure 2.37. Relationship between permanganate oxidizable carbon (POXC) and loss on ignition C (LOI). Red points represent outliers probably caused by the presence of manganese concretions.

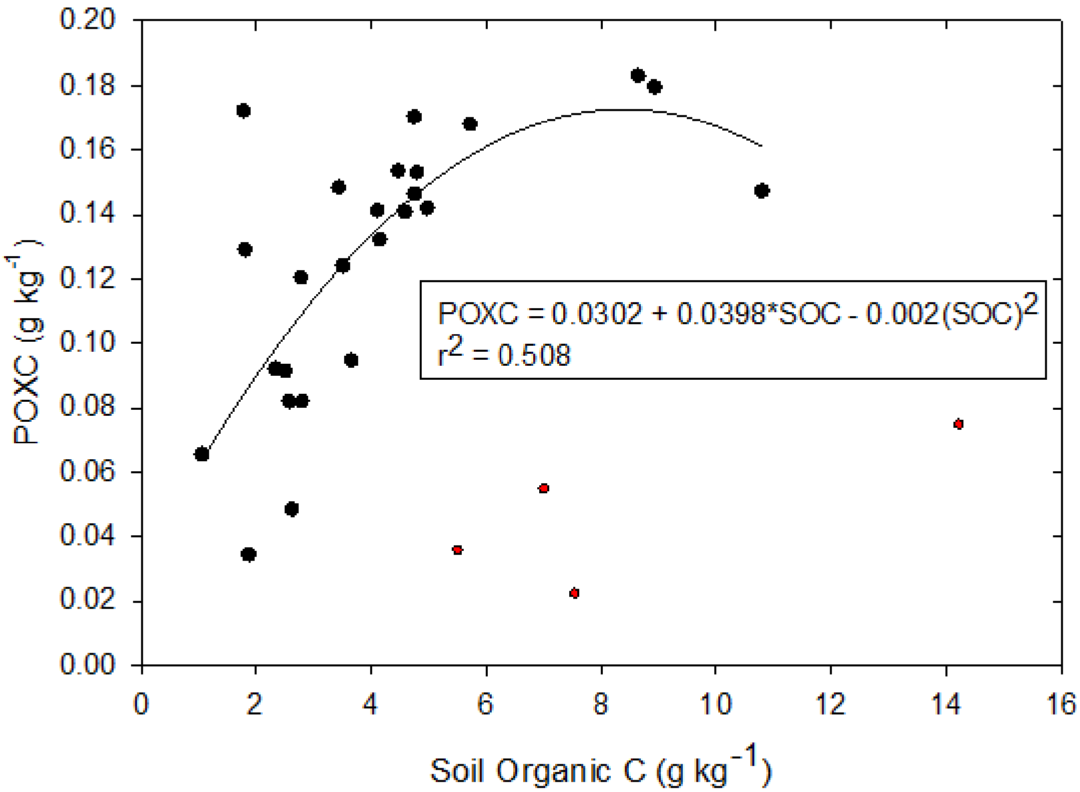

Figure 2.38. Relationship between permanganate oxidizable carbon (POXC) and soil organic carbon. Red points represent outliers probably caused by the presence of manganese concretions.

CHAPTER 3

PHOSPHORUS DISTRIBUTION, MANAGEMENT AND THE RELATIONSHIP OF RUNOFF TO SOIL PHOSPHORUS AND CARBON[2]

by

TAYLOR HENDRICKS

[2] Hendricks, Taylor, Dorcas H. Franklin, Subash Dahal, Miguel Cabrera, Dennis Hancock, Lawton Stewart, and Gary Hawkins. To be submitted to: Journal of Environmental Quality

Abstract

One of the greatest environmental threats is the eutrophication and impairment of surface freshwater systems used for consumption, industry, recreation, and ecological habitat. In the Southeast, eutrophication is generally dependent on the levels of phosphorus (P) released into surface water because P is the limiting factor to most algal growth. In addition to the ground cover and type of soil, the C and P content of an agricultural soil can also be a significant factor in predicting the TOC and DRP in runoff. The objectives of this study were to determine: (1) if soil P (in the Mehlich-1 form) varied with soil type or sampling depth; (2) the effect of distance to or from pasture equipage (i.e. hay, water, or shade) on the distribution of soil P; (3) if VNIR could predict soil P; and (4) if soil BD, soil P, or soil C (analyzed by LOI and POXC) could predict runoff quality. Soil and water samples were collected from 10 grazed pastures ranging from 9.2 to 21.8 ha. Grazing areas were subdivided into either areas of interest (AOIs), where cattle tended to frequent more often or the remainder of each pasture (pasture matrix).Soil samples were air-dried, ground, and analyzed for soil P using the Mehlich-1 soil extraction and molybdenum-blue colorimetric method developed by Murphy and Riley (1953). In addition to laboratory testing methods, visible and near-infrared (450-2400 nm) reflectance spectroscopy (VNIR) were taken on field-moist core samples *in situ* to determine if real-time estimates of P could be made based on reflectance spectra to enhance rapid nutrient measurement of soils. Additionally, using multivariate analysis of variance (MANOVA) it was determined that sample distance from a hay or water affected

the measured soil P differently based on geographic location although in both locations, samples further than 100 m from the nearest shade had the highest soil P concentrations.

Introduction

One of the greatest environmental threats is the eutrophication and impairment of surface freshwater systems used for consumption, industry, recreation, and ecological habitat. In the southeastern United States, eutrophication is generally dependent on the levels of phosphorus (P) released into surface waters because P is the limiting factor to most algal growth. When nutrients enter surface waters, they often cause widespread eutrophication which, in excess levels, can cause massive algal blooms often followed by hypoxic (dissolved oxygen of 2-3 ppm) (USGS, 2013) or anoxic conditions. These critically low levels of oxygen may be responsible for large scale fish kills and the reduction of submerged aquatic vegetation (SAVs) that is crucial to filtering these systems. Because of the low solubility of P (when compared with nitrogen), its effects are usually observed close to the source of contamination. Therefore, overall reduction of P in runoff is a main focus. However, it is not possible to create effective management plans without knowing the P loss risk of a field or location in the field based on landscape position and grazing animal activities.

Reducing environmental inputs of P from agriculture, however, is not as simple as reducing P inputs to a production system. In order to keep crop and forage yields at viable levels, P fertilization is required.

Nutrient losses from agriculture are generally classified into three pathways: 1) point source losses from a specific farmyard or from excessive rates of nutrient application; 2) diffuse losses from soil in relation to the soil content of those nutrients (usually nitrogen or phosphorus) already in excess of system requirements or from bare soil; and 3) incidental losses directly resulting from fertilizer/manure applications or when runoff occurs directly following application (Brennan, 2011). Incidental nutrient losses are highly variable because they depend on several factors that work interdependently. These factors can include, but are not limited to: rate and type of fertilizer applied; intensity of precipitation events following application; flow path length; vegetative cover; field slope; and landscape position (Brennan, 2012). Loading of P can result from P in surface runoff, particulate P from erosion, and soluble P in leachate. Therefore, it is imperative to understand where the vulnerable nutrient loss areas of a pasture exist in order to develop the most effective best management practices (BMP) to mitigate these losses from an agricultural system.

There are several methods that can be used to mitigate phosphorus losses. Landscape management methods can include: incorporation (rather than broadcasting) of P fertilizers; increasing buffer zones between fertilized land and surface waters; improved timing of fertilizer applications; and limiting the volume of runoff from the agricultural field (Brennan, 2012). In order to effectively manage agricultural P, developing a method to quantify the P captured and lost from fields is essential.

While there are many ways to extract soil P, the Mehlich-1 method for assessing soil test P (STP; Mehlich, 1953) is widely accepted in the Southeast as an accurate measure of the P in the soil that is readily plant available. Murphy and Riley (1962) developed a widely used method for measuring the extracted soil P using molybdenum blue coloring in conjunction with ascorbic acid. The Murphy and Riley molybdenum blue method, while widely accepted and reliable, is not without its flaws. Previous research has cited issues with adsorption coefficients, adaptations to the buffer conditions, and the possibility that partial hydrolysis of labile organic phosphate (P_O) may skew results (He and Honeycutt, 2005).

To reduce the time and cost inputs to producers, research into the use of spectroscopy has become widespread. Spectroscopy is a rapid, inexpensive means to monitor various soil properties and nutrients, including soil P (Rossel, 2006). Studies exploring the use of visible, near infrared, and mid infrared reflectance technologies for *in situ* nutrient monitoring have been conducted. Many studies have been conducted using the entire visible and near-infrared regions (400-2500 nm).Because the visible and NIR spectrophotometers are readily available for field usage and do not produce significantly different correlations, visible and NIR (VNIR) spectrophotometers are likely a viable sampling option.

A main issue preventing the widespread use of spectral monitoring is the limited amount of research into use on field-moist or unprepared sample correlations. Researchers agree that soil samples are generally processed (oven-dried, ground, and sieved) to reduce scatter that may be attributed to variable aggregate size and moisture content (Chang et al., 2002). The time and labor to process soil samples is often a large part of the cost of analysis and of delay in returning results to the farmer. A more rapid and less laborious method to determine soil P is needed. Reflectance spectra in field-moist soils are generally characterized by lower reflectance values and less intense differences. Additionally, there is little reflectance data for cropland soils, making strong calibrations difficult to achieve. Xie et al. (2010) showed that both *in situ* and prepared samples have stronger correlations when spectra are calibrated with a single, local soil type with similar mineralogy.

Because of these challenges, widespread calibration and validation data sets would need to be created – especially for use on *in situ* or unprocessed soil samples – to create a viable means for estimating the presence of soil P. By developing rapid and inexpensive methods to determine the presence of soil nutrients, specifically P, producers will be able to make more informed management decisions to further reduce nutrient losses from their fields. This will not only aide in making their production schemes more profitable and sustainable, but could also reduce negative environmental effects to surrounding surface waters.

In agricultural systems, losses of P via runoff are often the main concern for P conservation. P runoff losses are influenced by both transport and P source factors (Shigaki et al., 2007; Sharpley et al., 2001). Increased runoff volume, soil erosion, and high intensity rainfall events can be credited with increased P losses

(Shigaki et al., 2007). Dissolved Reactive P (DRP) can be supplied to runoff water through desorption of P from soil particles; release of P from chemical fertilizer; microbiological release of P from organic materials; and phosphate leaching from plant residue (Yli-Halla et al., 1995). While buffer zones at the edge of the field have been reported to decrease total P and particulate P, DRP concentrations in runoff may actually be increased with buffer use (Uusitalo et al., 2000).

Franzlubbers et al. (2011) determined that maintaining adequate ground cover is imperative to reduce the risk of nutrient loss through surface runoff. Ground cover management may be achieved through insightful grazing management strategies. Butler et al. (2007) showed that grassland with as little as 45% canopy cover could significantly reduce DRP concentrations in runoff. Further, pastures with good ground cover had reduced concentrations of TSS and particulate P. DRP concentrations, however, were uncorrelated to TSS and were therefore variable regardless of vegetative cover (Uusitalo et al., 1999). Some studies showed that DRP in runoff actually increased in fields with year-round vegetation (Puustinen et al., 2005).

Soil carbon is often lost through oxidation and respiration though we also lose soil carbon in runoff. Dissolved Organic Carbon (DOC) concentrations in runoff are another crucial parameter in runoff water. DOC is highly mobile and reactive component of soil organic matter, and it has been linked to increased sorption and mobility of pesticides from agricultural production and heavy metals (Veum et al., 2009). Consequently, elevated levels of DOC can cause concerns with clean drinking water sources (Veum et al., 2009; Pomes et al., 1999). DOC concentrations can range from 1 to 70 mg L^{-1} (Veum et al., 2009 and Zsolnay, 1996), with estimated agricultural DOC losses generally ranging from 14.1 to 19.5 kg ha^{-1} yr^{-1} (Dalzell et al., 2007 and Veum et al., 2009).

Previous studies have shown that DOC in runoff can be attributed to mineralizable C from a labile carbon soil fraction (Jacinthe et al., 2004). Labile soil carbon is easily detected through the use of POXC soil analysis. During low-intensity runoff events, C releases were 2.5 times more labile than C releases during high-energy storms (Jacinthe et al., 2004). Therefore, highly reliable methods to test labile soil C, such as POXC analyses, may be useful in predicting DOC concentrations in runoff similarly to the use of STP to predict DRP following a runoff event.

The objectives of this study were to determine: (1) if soil P (in the form of Mehlich-1 extractable P) varied with soil type or sampling depth; (2) the effect of distance to pasture equipage (i.e. hay, water, or shade sources) on the distribution of soil P; (3) if VNIR could predict soil P; and (4) if soil BD, soil P, and soil C (in LOI or POXC) could be used to estimate runoff concentration of P and C.

Methods and Materials

Study Site

This study was conducted at the J. Phil Campbell Research and Education Center’s North and West Units (33.887487,-83.420966 and

33.863240,-83.457599, respectively) in Watkinsville, Georgia and at the University of Georgia's Animal and Dairy Science Department's Beef Research Unit (33.420759,-83.476555) in Eatonton, Georgia. There were four pastures on the North Unit (14.3 to 17.8 ha); two on the West Unit (9.2 to 9.3 ha); and four in Eatonton (18.0 to 21.6 ha). Each of these pastures was an experimental unit. The North and West unit pastures are close to Watkinsville and are characterized by Cecil (2 to 6 percent slope) sandy loam and Pacolet (6 to 10 percent slope) sandy clay loam soils that are eroded or severely eroded. Eatonton pastures are primarily characterized by Davidson clay loam (6 to 10 percent slope), Davidson loam (2 to 6 percent slope), and Wilkes sandy loam (2 to 10 percent slope) soils, although Iredell loam, Enon, Congaree, and Toccoa soils have also been reported.

The pastures in Eatonton were managed for at least 10 years using put and take grazing in continuously stocked pastures with 20-30 head per stocking herd. The pastures in Watkinsville have been managed using put and take grazing for at least 10 years, with approximately 40 head per stocking herd. Aerial imagery (from Google Earth Pro) shows that the approximate hay feeding locations have remained the same for both Eatonton and Watkinsville since 1993. Eatonton uses primarily hay rings and Watkinsville used hay rings from 2000-2006; heavy use areas from 2006-2013; and a hay un-roller from 2013-present.

Soil Sampling

Areas within these pastures where cattle tended to frequent, or areas of interest (AOIs), were delineated from the remainder of each pasture (matrix). These AOIs were determined based on historical observations and evidence (e.g., fecal deposits, trampled or overgrazed forage, etc.) of cattle camping activity near shade or grazing equipage (water, and feed sources). Pastures contained 3-8 AOIs depending on pasture area and distribution of pasture features that attracted cattle (i.e., shade, water, food, hay sources, etc.).

Soil samples were taken from the AOIs and on a grid within the pasture area (matrix). Samples were taken throughout the AOI during the fall and spring using a 5-cm core diameter Giddings hydraulic soil probe (Giddings Machine Corporation, Inc., Windsor, CO) at multiple points (2-5) with three replications per point within a 1-m sampling radius. Samples were taken from the remaining pasture area on a 50-m grid with two soil cores taken within a 1-m sampling radius at each intersection point. All core samples were taken at three depths: 0-5 cm, 5-10 cm, and 10-20 cm. Soil cores were split longitudinally to reveal an unaltered face and scanned for reflectance in the field using a portable ASD Vis/NIR AgriSpec (ASD Inc, Boulder, CO) VNIR analytical spectral device.

Determination of Distances from Equipage

Geometric distance from each point to nearest hay-feeding area, waterer, and shade were determined using the near tool in ArcGIS 10.3.1. The analysis was completed by creating shapefiles with GPS coordinates of each sampling point and hay, water, and shade as features in the pasture. A “near tool” analysis was run through an ArcGIS model builder capability to measure the distance from each unique sampling point to the nearest hay, water, and shade source.

Runoff collection and P and C Analysis

Digital elevation models (DEMs, 4-cm horizontal and 8-cm vertical resolution) were created of each pasture using a Trimble® R8 and R10 GNSS system and ArcGIS 10.2.1 software. Areas of concentrated flow during rainfall events were determined using watershed delineation maps, DEMs, and spatial analyst tools. Runoff samples were collected during runoff-producing events (n=21) (rainfall greater than 1.25 cm) over a one-year period (May 2015 – May 2016). Runoff collectors were placed in concentrated flow paths at the edge of field downhill of AOIs. There were 3 to 5 runoff collectors in each of the 10 pastures based on pasture acreage and the number of existing sub-watersheds (Figures 3.1 – 3.5). Each runoff collector had 3-5 500-mL Nalgene bottles equipped with 0.95-cm (inside diameter) silicon tubing at 2.5 cm height increments that ran from the opening of the collector (first point of contact with runoff water) to the Nalgene bottles to create a venturi suction once runoff began to flow into and down the tubes (Figure 3.6).

Samples (n=359) were collected from the field within 24 h of the end of a runoff-producing rainfall event. Samples were refrigerated upon collection, depth of water in the bottle was measured to determine runoff quantity, and 50 mL of sample was vacuum-filtered through a 0.45-µm nitrate membrane filter and frozen within 48 h of the runoff event. Sediments in runoff were determined by filter combustion at 550ºC for 8-10 hours and measuring difference in mass. Runoff samples were then analyzed for P using the molybdate-blue method (Murphy and Riley, 1962). DOC was determined using a Shimadzu TOC-5050 analyzer where samples were injected into a combustion chamber and an NIR peak proportional to the amount of C in the sample was recorded to determine C concentration based on standard calibrations. Runoff DOC values were regressed against both AOI soil LOI and POXC concentrations as well as watershed LOI and POXC values. Regressions were determined by comparing the annual mean DOC value from a specific runoff collector to annual mean soil C concentrations of the AOI uphill of the runoff collector that was within the runoff collector contributing area. DRP in runoff was also regressed with soil P using this same method. A multivariate analysis technique, principal component analysis (PCA), was used to determine the importance of several factors (rainfall, slope, total suspended sediment, mineral-only sediment, and soil C or P values) on the amount of DOC or DRP in runoff.

Soil Analysis

Soil samples were air-dried and ground (<2 mm) and a subsample was taken to determine oven-dried soil moisture calculated using the *USDA Soil Survey Laboratory Methods Manual* (USDA, 2004). Soil phosphorus was determined by analyzing each sample for Mehlich-I soil P (Mehlich, 1953) determined by the molybdate-blue method (Murphy and Riley, 1962) modified to be analyzed in a microplate reader. Samples were run in micro-centrifuge tubes at a 1:20 of the Murphy and Riley (1962) procedure and absorbance was measured at 880 nm using an InfiniteMX200Pro NanoQuant absorbance spectrophotometer (Tecan US, Inc., Morrisville, NC).

VNIR scans were processed to remove noise from the spectra and eliminate total brightness or total darkness (0-0.05 or 0.95-1.0 reflectance values) (misread) samples and displayed as the reflectance first derivative. Mehlich-1 extractable P was compared with VNIR scans in order to determine best correlations for specific wavelengths; full spectra were considered. Recursive partitioning techniques were used to determine wavelengths of interest for soil phosphorus signatures and partial least square regression analyses were conducted using JMP 11 statistical software. Wavelengths of interest that were determined in previous research were also analyzed (1610-1635 nm; 1952-2041 nm; and 2345-2390 nm) for a response (Chang et al., 2002 and Rossel et al., 2006).

The relationships of soil P to landscape positioning, topographic features, soil type, and soil family were also determined through the use of recursive partitioning, partial least squares analyses, and multivariate analysis of variance (MANOVA). Finally, soil sampling points were grouped into 0-39 m, 40-99 m, 100-199 m, and 200 or greater meters from hay, water, and shade sources and the relationship of soil P to equipage was determined using multivariate analysis of variance (MANOVA). Because soil P data were not normally distributed, the effect of pasture equipage on soil P distribution was determined using Wilcoxon Rank Sum tests.

Results and Discussion

Soil Phosphorus

Soil extractable P was not normally distributed. Median soil P showed significant differences (P < 0.0001) between the two study locations (Mdn: 1.62 mg kg^{-1} and 3.98 mg kg^{-1} for Eatonton and Watkinsville, respectively) (Figures 3.7 and 3.8). When analyzed by depth and location, the 10-20 cm depth had the lowest median soil P (P <0.0001) and Eatonton showed significantly different soil P at each depth. The 5-10 cm samples had the greatest range in overall median extractable P (1.6 and 4.58 mg kg^{-1}, for Eatonton and Watkinsville respectively) (Figure 3.9). Recursive partitioning showed that extractable soil P may be related to soil bulk density values as follows: 1.08, 2.29, 1.2, and 1.01 g cm^{-3}. For example, using recursive partitioning, samples with a BD equal to or greater than 1.08 g cm^{-3} had a mean extractable P of 3.12 mg kg^{-1} while BD less than 1.08 g cm^{-3} had a mean extractable P of 4.63 mg kg^{-1}. The bulk density of 1.2 g cm^{-3} was close to the bulk density that was also most strongly related to previously studied loss-on-ignition and permanganate oxidizable carbon values (1.26 g cm^{-3}). Samples with bulk densities greater than 1.2 g cm^{-3} had a mean extractable P of 1.53 mg kg^{-1} while bulk densities of 1.2 g cm^{-3} or less had a mean extractable P of 3.2 mg kg^{-1}. These were the soil samples with greater soil LOI and POXC concentrations.

The differences in soil P based on location may be explained, in part, by differences in soil type between the two geographic locations (Table 3.1). The Eatonton pastures are comprised of primarily Davidson loam and Wilkes sandy loam soils while the Watkinsville pastures are primarily Cecil sandy loam and Pacolet sandy clay loam soils with various degrees of slope and erosion. When soil P was characterized based on soil type, the only significant differences (both when divided by depth, divided by location, and when depths and locations were analyzed together) occurred between the Davidson loam (2 to 6 percent slope; moderately eroded or the 6 to 10 percent slope; eroded) and the Cecil sandy loam or Pacolet sandy clay loam soils. While all the soil P concentrations were low, Cecil sandy loam soils had more than twice the extractable soil P as the Davidson loam.

Relationship of Soil Phosphorus to Equipage

The relationships of soil P to hay, water, and shade follow similar trends regardless of sample depth or soil type. Soil P showed differences when related to the distance to a pasture hay source. When soil samples were split into zones based on distance (zones = 0-39, 40-99, 100-199, and 200+ m) to hay, there were significant differences among distance zones both when divided and not divided by sampling depth and location (Figure 3.10).

Median soil P concentrations in Eatonton had a significant relationship to distance from the nearest hay source, zone 0-39 m samples had significantly ($P < 0.0001$) higher soil P (Mdn: 2.75 mg kg^{-1}) than samples from 40-199 m (Mdns:1.82 mg kg^{-1} and 1.76 mg kg^{-1} for 40-99 and 100-199 m, respectively) (Figure 3.10) and samples 0-199 m also had greater ($P < 0.0001$) soil P values than sample points that were 200 m or more from the nearest hay source (Mdn: 1.59 mg kg^{-1}). In Eatonton, the 5-10 and 10-20 cm depths followed the same trend (closer to hay higher soil P) as when samples were not divided by depth (0-39 m > 40-99 = 100-199 m > 200+ m; $P < 0.0001$) while 0-5 cm sampling depth was significantly higher ($P = 0.003$, 0.002, and $P < 0.0001$ for 40-99, 100-199 m, and 200+ m, respectively) in the 0-39 m distance (Mdn: 3.48 mg kg^{-1}) than each successive distance rating and 40-199 had significantly greater ($P = 0.02$ soil P (Mdns: 2.25 and 3.01 mg kg^{-1}, respectively) than samples 200 m or more from hay (Mdn: 1.79 mg kg^{-1}) (Figure 3.11).

In Watkinsville, differences based on distance to hay were seen in the 0-5 and 5-10 cm depths, but not in the 10-20 cm samples (Figure 3.12). The 0-5 and 5-10 cm samples showed the highest soil P values in the samples occurring 100-199 m and 200 m or greater zones from the nearest hay source ($P = 0.0101$ and $P < 0.0001$; Mdns: 3.96 and 5.09 mg kg^{-1}, respectively) (Figure 3.12).

When measuring the effect of distance to the nearest water source on the median soil P concentrations, differences existed regardless of depth for each location ($P < 0.0001$, Figure 3.13). When samples were divided by depth, soil P concentrations 40 at to 199 m were also almost always greater than the distance zone adjacent to waterers or greater than 199 m from the waterers ($P = 0.004$, 0.09, 0.007 for 0-5, 5-10, and 10-20 cm, respectively). Additionally, when samples were divided by both depth and location, Eatonton contributed most to the differences (Figure 3.14). Unlike the hay sources, soil P did not decrease progressively from nearest to farthest among the designated distances. This may be a result of cattle activity near water troughs as animals drink, but do not camp in the area immediately surrounding the troughs.

Distribution of soil P concentrations varied significantly depending on distance of a sampling point to the closest source of natural shade in both Eatonton and Watkinsville (Figure 3.16). When sampling points were divided into the four distance zones, soil P was significantly different regardless of which depths were compared (Figure 3.16, 3.17, and 3.18). In Eatonton, soil P in all soil depths were influence by nearness to shade but in Watkinsville only the two deeper depths sample showed significant differences. Unlike the effect of either distance to hay or distance to water, soil P was significantly less when sample points were closer (either less than 40 or less than 100 m) to a shade source.

In Eatonton, no sampling points existed further than 200 m from the nearest shade source. Therefore, for Eatonton shade analysis, the distance categories were: 0-39 m, 40-99 m, and 100-199 m. When all sampling depths were analyzed together, samples occurring 0-39 m from a shade were significantly lower ($P < 0.0001$) in soil P than samples 40-99 m from a shade and both were less than samples taken at least 100 m from a shade source (Figure 3.17).

In Watkinsville, however, no differences were seen in the 0-5 cm sampling depth, regardless of distance to shade (Figure 3.18). In both the 5-10 and 10-20 cm sampling depths, the highest soil P values were seen in the samples occurring 200 m or greater from the nearest shade (Mdns: 5.45 and 4.13 $\mu g\ g^{-1}$ for 5-10 and 10-20 cm, respectively). Additionally, the samples occurring from 0-39 m from shade had the lowest soil P in the 10-20 cm samples and 40-99 m from a shade source had the lowest soil P in the 5-10 cm samples (Figure 3.18).

Because sample soil P is lower in samples taken closer to shade sources, the effect of cattle activity on the extractable soil P is questionable. Additionally, because shade in Watkinsville is most often located in downhill positions that are vulnerable to erosion, P from animal deposition may not have ever be incorporated into the soil but rather wash away in rainfall events.

Relationship of Soil Phosphorus to VNIR Spectral Scans

In order to determine the potential for using VNIR spectra to make nutrient estimates and recommendations, the relationship of various wavelengths in VNIR spectra were determined. Full spectra were analyzed against the measured soil P concentrations using PLSR and returned no results (RMPRESS < 0.5) even when samples were divided by geographic location, sample depth, or soil type. For this reason, spectra were analyzed based on target wavelengths that were determined through recursive partitioning techniques. Wavelengths that showed potential relationships to extractable soil P were (in order of response during recursive partitioning): 671 nm, 524 nm, 1394 nm, 1107 nm, 523 nm, 505 nm, 1979 nm, 966 nm, 691 nm, and 457 nm.

In almost every wavelength range (and at most depths), the Watkinsville samples had stronger responses to VNIR spectral analyses, although neither location provided relationships strong enough to make nutrient estimates or recommendations. It may be suggested that because the Watkinsville samples had generally higher soil P values, that VNIR response to P may be stronger because they are at a level that is more easily detected using VNIR. Because of the limited range of soil P values in conjunction with highly variable soil types, Eatonton samples may not have shown enough distinction for strong VNIR relationships to form.

Runoff

Runoff concentration values of both DOC and DRP did show relationships to soil C or P in their respective field-scale watersheds, however these relationships were not what was expected. Correlations for Soil C and DOC in runoff were strongest with linear regressions. When all data were combined (locations and soils depths), only a weak linear relationship between DOC in runoff and LOI soil carbon (r^2 = 0.35) (Figure 3.19) was shown, which was inadequate to accurately predict DOC concentrations from LOI values alone. Regressions showed slight improvement when runoff and soils data were split by geographic location (Eatonton and Watkinsville) (Figures 3.20 and 3.21). Improved correlations in Watkinsville DOC-LOI relationships (r^2 = 0.56) (Figure 3.21) may be due to a more consistent clay mineralogy of the soils (Cecil sandy loam,15 to 20% clay) and Pacolet sandy clay loam,20 to 35% clay) compared with Eatonton (r^2 = 0.32) (Davidson loam, 8 to 28 % clay) which would increase the amount of organic matter in the soil. Relationships were likely improved because high LOI areas also showed very high DOC values; there were Watkinsville AOIs with very low DOC values (min=8.98 mg L^{-1} for Watkinsville).

There was no detectable (r^2 > 0.15) trend between DOC and soil POXC concentrations (r^2 = 0.0015) (Figure 3.22). This was unexpected because the POXC soil analysis is designed to detect labile soil C, which has been shown as a major contributor to DOC concentrations in runoff, particularly during low-intensity runoff events (Jacinthe et al., 2004). Although trends improve when runoff sites were separated into their respective geographic locations, neither site showed a trend that would be strong enough to predict DOC concentrations. Relationships also improved when outliers (DOC > 25.00 mg L^{-1}) from each geographic location were removed from the regression (r^2 = 0.31 and 0.002 for Eatonton and Watkinsville, respectively) (Figures 3.23 and 3.24). These geographic differences in correlation strength may again be attributed to the soil properties in the 0-5 cm samples.

Overall, as POXC soil concentrations increased, the variability of DOC samples also increased. Therefore, the Watkinsville soils that demonstrated a much higher POXC range (1200-1800 mg kg^{-1} for AOI) and had higher variability in runoff DOC values and therefore a lower r^2 (0.0024) than the Eatonton soils (r^2 = 0.31) that had generally lower POXC values (350-1250 mg kg^{-1} for AOIs).

Relationships between soil P and DRP concentrations also showed slight trends when locations were analyzed together, but improved drastically when outliers were removed (Figure 3.25). Similarly to the trend seen in DOC-POXC relationships, as soil P concentrations increased, DRP values became more variable. When runoff collectors that had a contributing Mehlich-1 soil P of 10.5 mg L^{-1} or greater were removed from the analysis, the correlation improved (r^2 = 0.12 to 0.43). This was the strongest correlation of the three relationships when samples were not divided by geography (Figure 3.26).

When separated solely by geographic location, relationships were not necessarily improved. Eatonton samples, which are characterized by a lower contributing soil P value (2.25-6.79 mg kg^{-1}), had a stronger relationship (r^2 = 0.57) (Figure 3.27) than the Watkinsville soils which had higher (1.58-20.59 mg kg^{-1}) and more variable soil P values (r^2 = 0.37) (Figure 3.28), however, when sites with contributing soil P greater than 10.5 mg kg^{-1} were eliminated the correlation of both sites improved (r^2 = 0.43) (Figure 3.26).

Other factors that may have contributed to DOC and DRP concentrations in runoff samples were analyzed using multivariate analyses including PCA. Possible factors include: rainfall amount per event; mean air temperature of event; mean AOI slope; mean AOI aspect; and concentration of the contributing nutrient (LOI, POXC, or soil P) in the 0-5, 5-10, and 10-20 cm sampling depths. Initial multivariate analyses showed a 0.37 correlation between DOC and LOI and a 0.11 correlation between DOC and POXC in the 0-5 cm sample depths, which supported the regression analyses. The strongest correlation for any runoff nutrient existed between 0-5 cm AOI soil P and DRP (0.44) when PCA analyses were run. This also supported the regression analyses that demonstrated the strongest relationship between DRP and AOI soil P (0-5 cm sample depth) in both Eatonton and Watkinsville (r^2 = 0.57 and 0.37, respectively).

Multivariate analyses included analyzing nutrient relationships to each depth (0-5, 5-10, and 10-20 cm) independently. DOC to LOI, DOC to POXC, and DRP to soil P all showed a decreasing relationship with depth (0.35, 0.19, and 0.06 for DOC-LOI; 0.09, -0.03, and -0.09 for DOC-POXC; and 0.44, 0.42, and 0.18 for DRP-soil P at the 0-5, 5-10, and 10-20 cm depths, respectively). Additionally, a stronger correlation (0.46) was found between POXC concentration and slope then POXC and DOC. Surprisingly, rainfall amount per event showed little to no correlation with either DOC or DRP (0.003, 0.008, and 0.079 for DOC-LOI, DOC-POXC, and DRP-soil P, respectively). Further analyses to determine the effect of rainfall intensity, rather than overall rainfall may show stronger relationship trends.

Conclusions

Mehlich-1 P showed significant differences between Watkinsville and Eatonton locations. This was likely due to differences in the primary soil classifications between the two locations. The North and West unit pastures in Watkinsville, Georgia are primarily Cecil sandy loam and Pacolet sandy clay loam while the Eatonton research pastures were primarily Davidson loam. Recursive partitioning showed that a bulk density of 1.2 g cm^{-3} may be related to a difference in M1 soil P; this bulk density is similar to the bulk density (1.26 g cm^{-3}) that was previously related to LOI and POXC carbon analyses.

When distribution of soil P concentrations were evaluated with distance from hay feeding locations, soil P concentrations were significantly greater in areas within 0 to 39 m of hay feeding locations and decreased as distance to hay increased. This trend occurred at each sampling depth in Eatonton and in the 0-5 and 5-10 cm samples in Watkinsville. The relationship of soil P to distance to waterers existed in the 0-5 and 10-20 cm sampling zone at either location; 5-10 cm samples were not affected. Unlike distance to hay, when soil P was compared to distance to water, the soils samples taken 40-99 m from the nearest water source had the greatest soil P. This was likely due to cattle activity near water troughs as animals drank, but did not camp in the immediate vicinity. Relationships of soil P concentrations and shade were inverse of soil P and hay feeding locations: soil P was lower in the areas closest to shade sources and increased as distance to shade decreased.

Overall the relationships between soil P and various spectral wavelengths were only moderately responsive. Full spectral data produced almost no response to soil P data. Moderate, but variable, responses were given when sample data was compared to specific wavelength ranges. Relationships improved when samples were characterized by soil type.

While runoff concentrations showed mild relationships to the soil C and P data, correlation values were not strong enough to make accurate DOC or DRP predictions from soils data alone. Each of the runoff relationships (DOC-LOI, DOC-POXC, and DRP-soil P) improved when runoff collection sites were separated by geographic location. This suggests that soils may contribute to differences in runoff concentrations. Because runoff measurements, like many agricultural measurements, are complex and pasture relationships are interdependent, it may be valuable to explore additional factors, such as rainfall intensity and vegetative cover contributors for determining runoff quality. In the future, runoff nutrient concentrations may be predicted with highly developed calibration equations for various pasture sites. This additional information has the potential to have major land management effects.

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Table 3.1. Location, surface area, and mean and median soil P (Mehlich-1) values in three soil depths for each soil type sampled. E, ME, and SE represent erosion classification according to NRCS soil survey classes "eroded," "moderately eroded," and "severely eroded," respectively.

			Mean Mehlich-1 P ($ug\ g^{-1}$)		
Soil Type	**Location**	**Area (ha)**	0-5 cm	5-10 cm	10-20 cm
Cecil soils, (Slope 0-2); overwash Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	0.5	2.3/2.2	2.7/2.3	2.0/1.4
Cecil Sandy Loam (Slope 2-6); E Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	21.9	4.2/3.6	5.1/4.8	4.3/3.9
Cecil Sandy Clay Loam; (Slope 2-6); SE Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	28.1	3.9/3.0	5.8/4.9	4.6/4.0
Conagree and Toccoa Soils Loamy, mixed, active, nonacid, thermic, Typic Udifluvents	Eatonton	1.2	1.7/1.7	1.9/1.9	1.6/1.6
Davidson Loam (Slope 2-6); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	23.0	1.6/1.9	1.4/1.6	1.3/1.6
Davidson Clay Loam (Slope 6-10); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	20.0	2.5/2.2	1.7/1.8	1.8/1.7
Davidson Clay Loam (Slope 10-25); ME Fine, kaolinitic, thermic Rhodic Kandiudults	Eatonton	6.6	1.9/2.1	1.5/1.6	1.5/1.7
Enon soils; E Fine, mixed, active, thermic Ultic Hapludalfs	Eatonton	9.0	1.8/1.8	0.6/0.6	0.6/0.6
Iredell Loam (Slope 2-6) Fine, mixed, active, thermic Oxyaquic Vertic Hapludalfs	Eatonton	10.1	2.3/2.2	2.3/2.1	1.9/1.9
Pacolet Sandy Clay Loam (Slope 6-10); SE Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	30.6	4.9/3.7	5.0/4.2	4.0/3.3
Pacolet Sandy Clay Loam (Slope 10-15); SE Fine, kaolinitic, thermic Typic Kanhapludults	Watkinsville	3.7	5.9/6.0	5.1/4.6	4.6/3.0
Wilkes Sandy Loam (Slope 2-10); E Loamy, mixed, active, thermic shallow Typic Hapludalfs	Eatonton	14.0	3.8/3.0	3.1/2.1	2.6/2.1

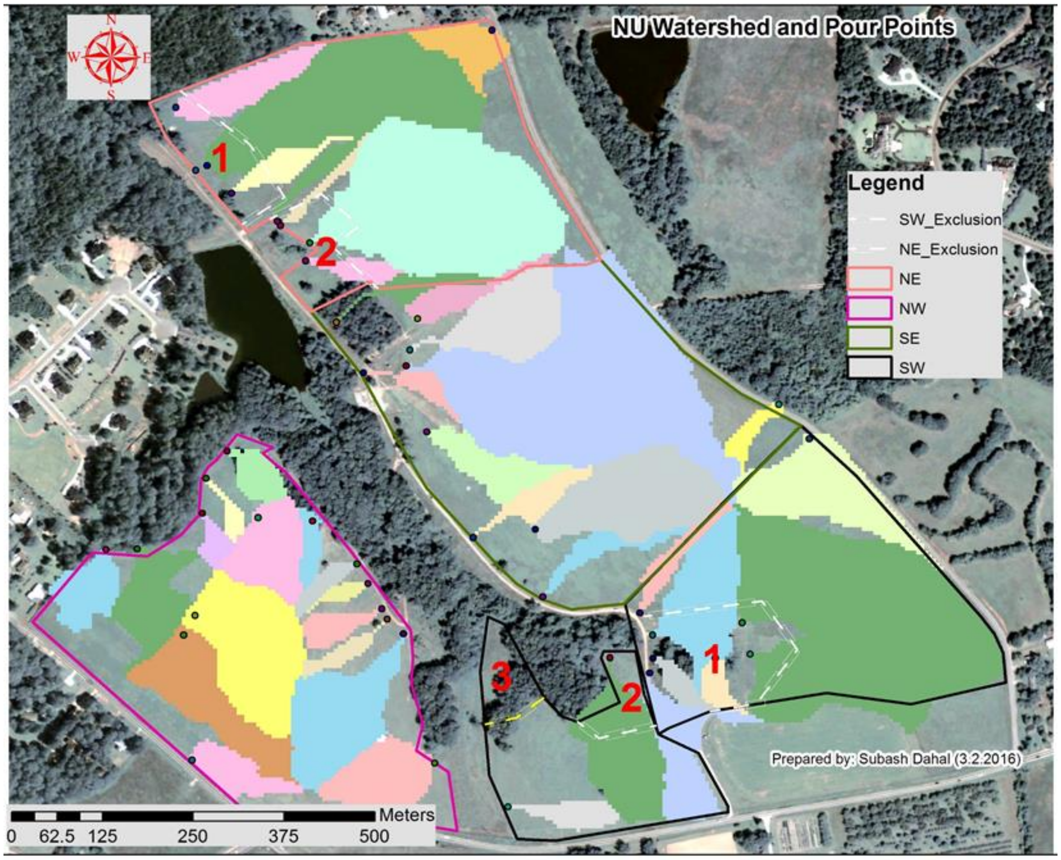

Figure 3.1 Map of North Unit watersheds created using ArcGIS 10.2.1 used to determine runoff collector locations. Circles represent pour points and colored blocks represent individual watersheds. Image and watershed delineations courtesy of Subash Dahal.

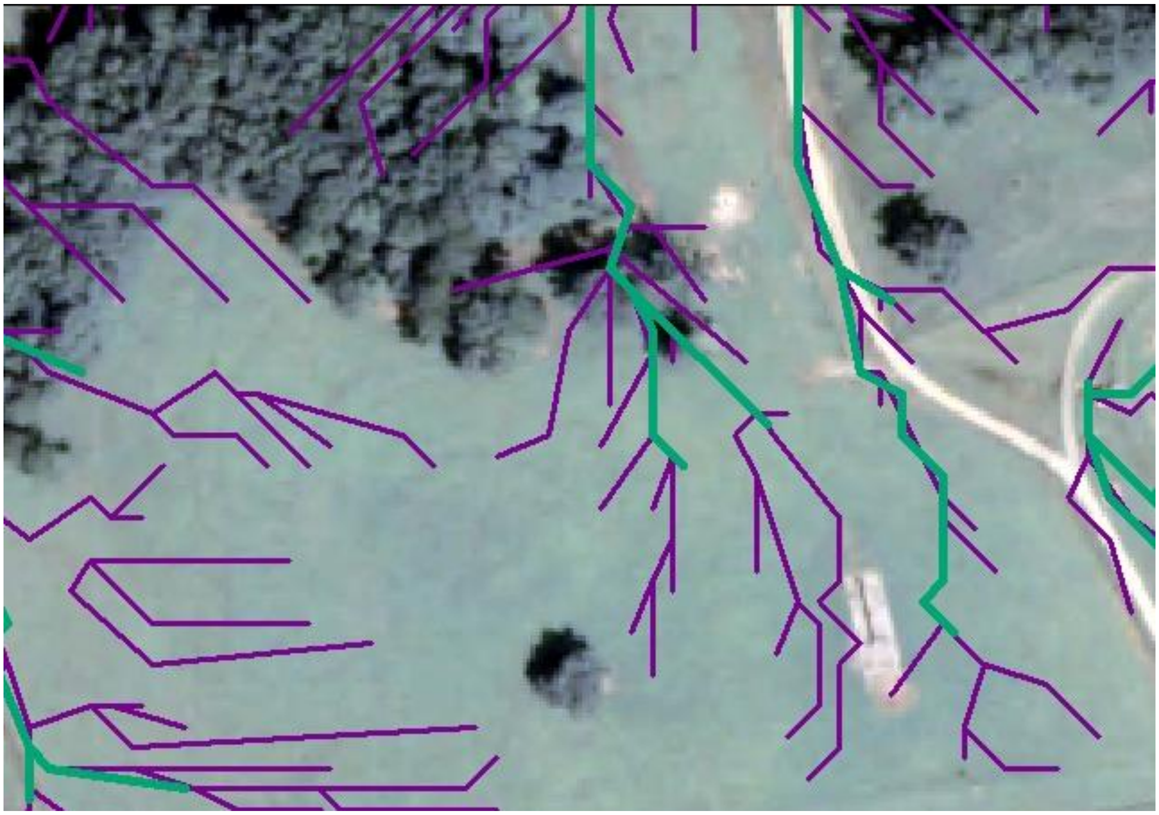

Figure 3.2. Aerial image of North Unit pasture with flow paths created using ArcGIS 10.2.1. Image and flow paths courtesy of Subash Dahal.

Figure 3.3. Map of North Unit runoff collector locations.

Figure 3.4. Map of West Unit runoff collector locations.

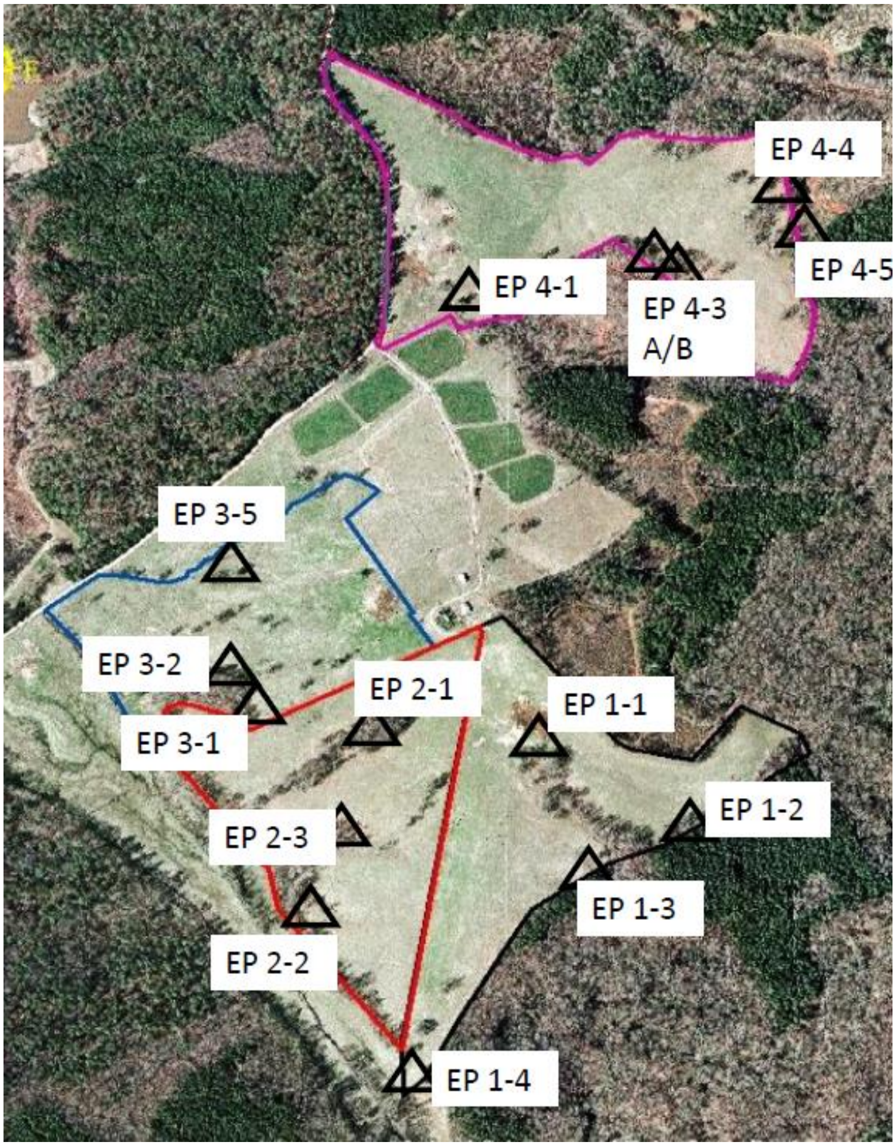

Figure 3.5. Map of Eatonton runoff collector locations.

Figure 3.6. The photo on the left shows the set up for the Poore-point runoff collectors placed at the edge-of-field in order to collect runoff. The tubing creates a vacuum system that slowly collects water at different runoff heights during and after a runoff-producing rainfall event. The photo on the right demonstrates the placement of the runoff collector within the pasture. The blue arrows demonstrate a concentrated flow pathway.

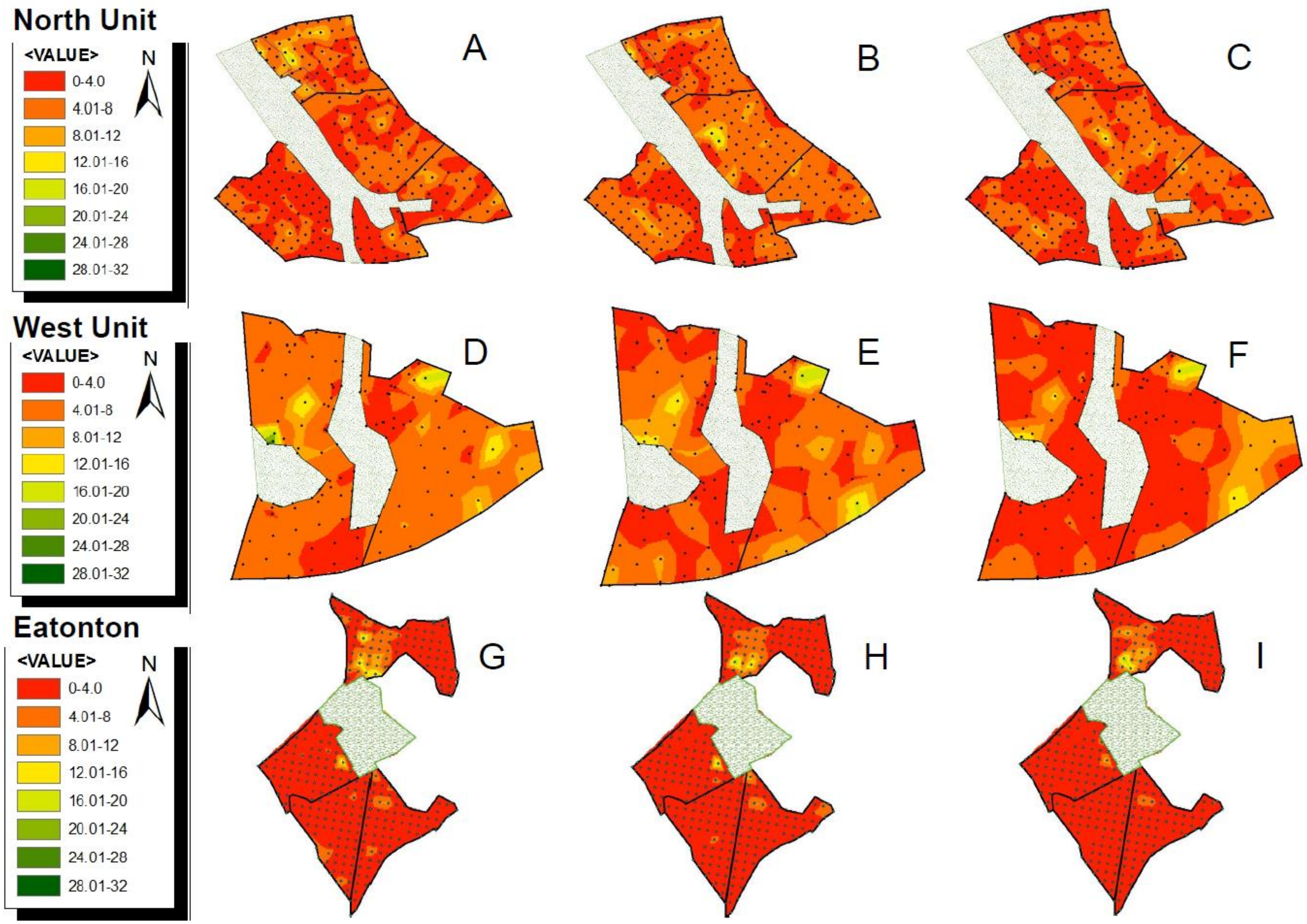

Figure 3.7. The Mehlich-1 extractable P values for the Watkinsville North (A-C) and West Unit (D-F) and Eatonton Beef Research Unit (G-I) top to bottom and for each sampling depth (0-5, 5-10, and 10-20 cm) are shown from left to right. Mehlich-1 extractable P is displayed in (mg P kg^{-1}), with low bulk densities displayed in red (0.0 – 4.0) and high Mehlich-1 extractable P values (28-32) shown in green.

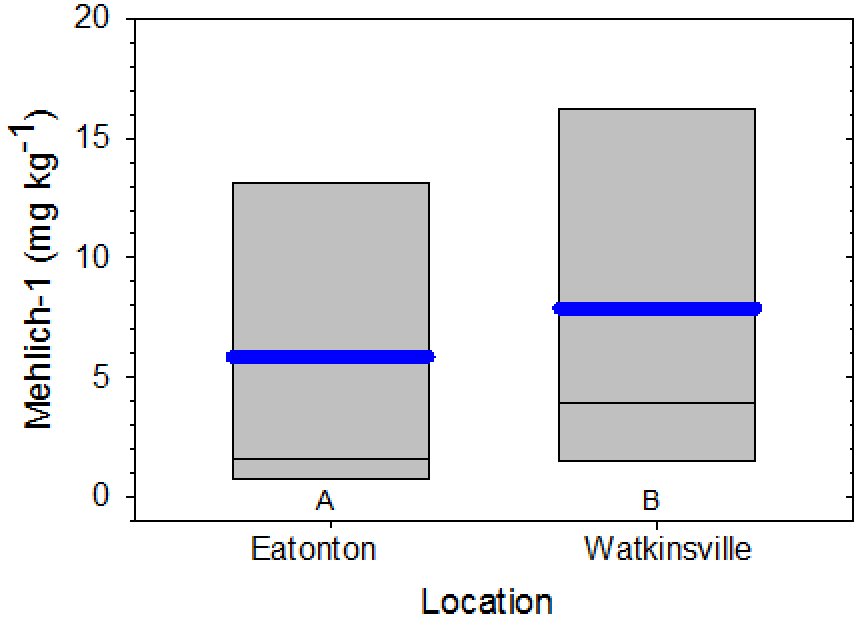

Figure 3.8. Box plots for Mehlich-1 soil P at all three depths (0-5, 5-10, and 10-20 cm) for the Eatonton and Watkinsville sites. Blue lines are means and thin black lines are medians. Different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

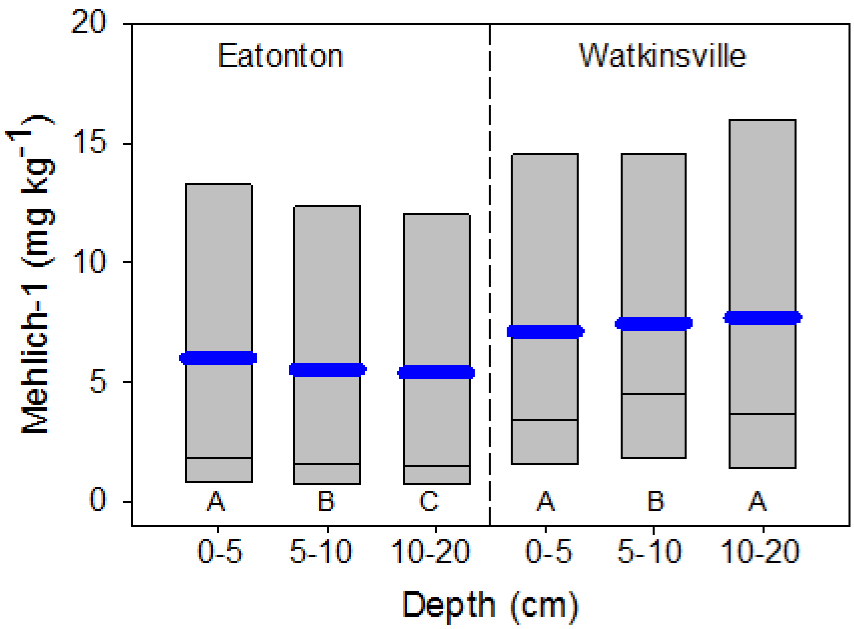

Figure 3.9. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus for each sampling depth in Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

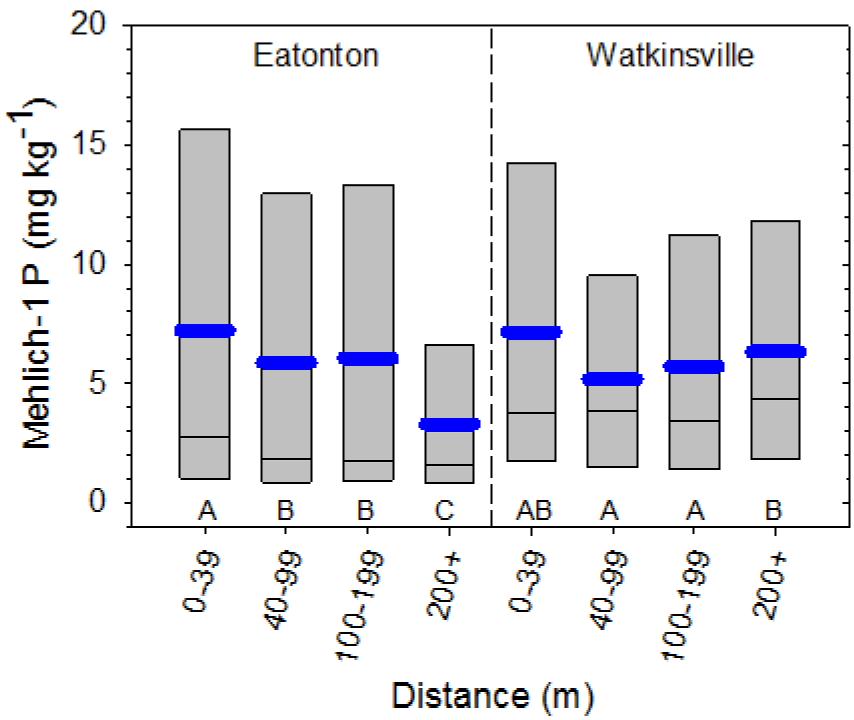

Figure 3.10. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

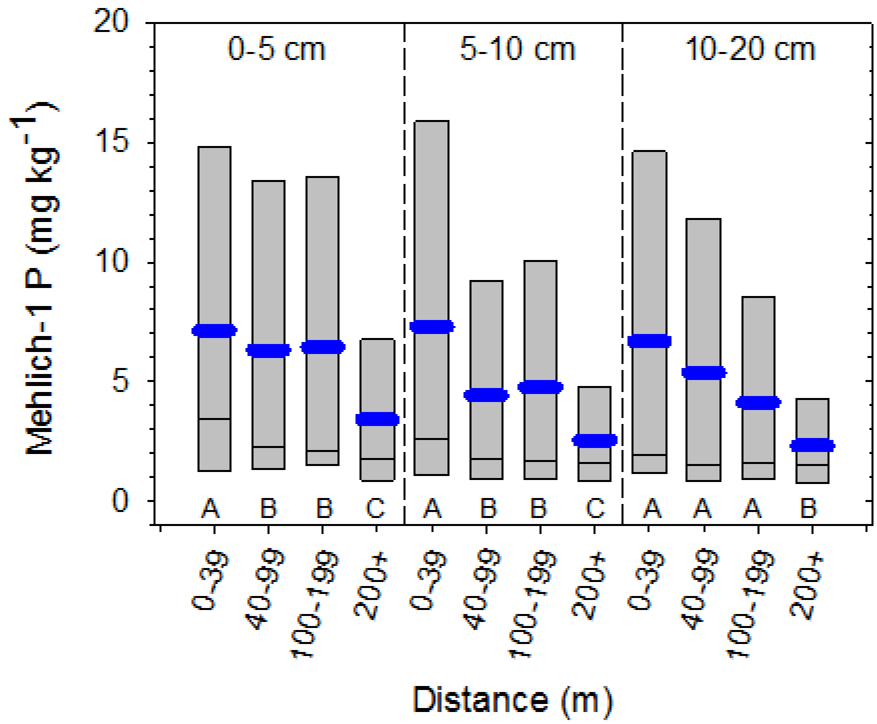

Figure 3.11. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus for each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

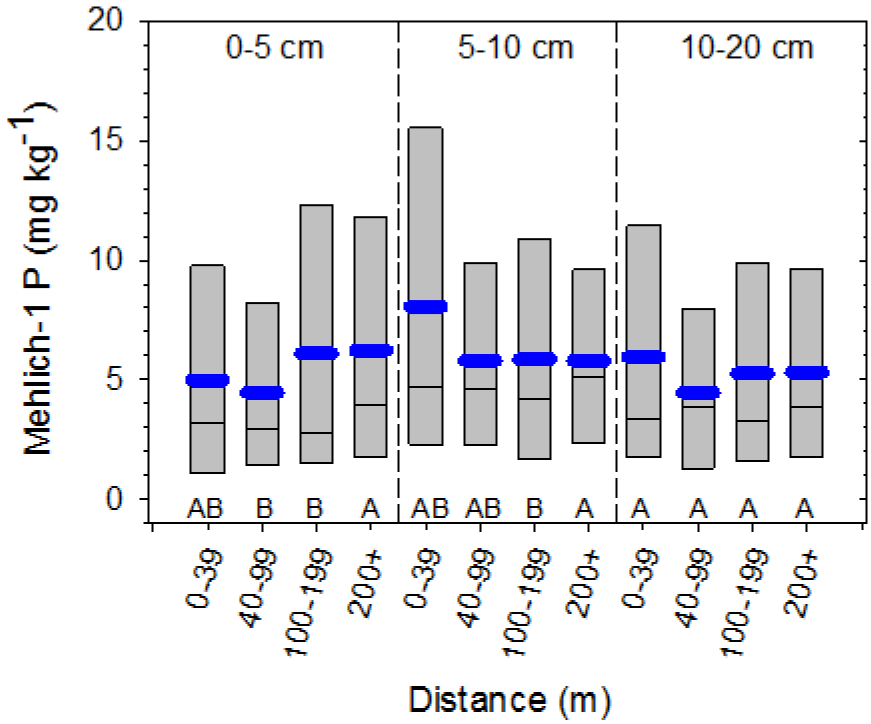

Figure 3.12. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus for each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest hay source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

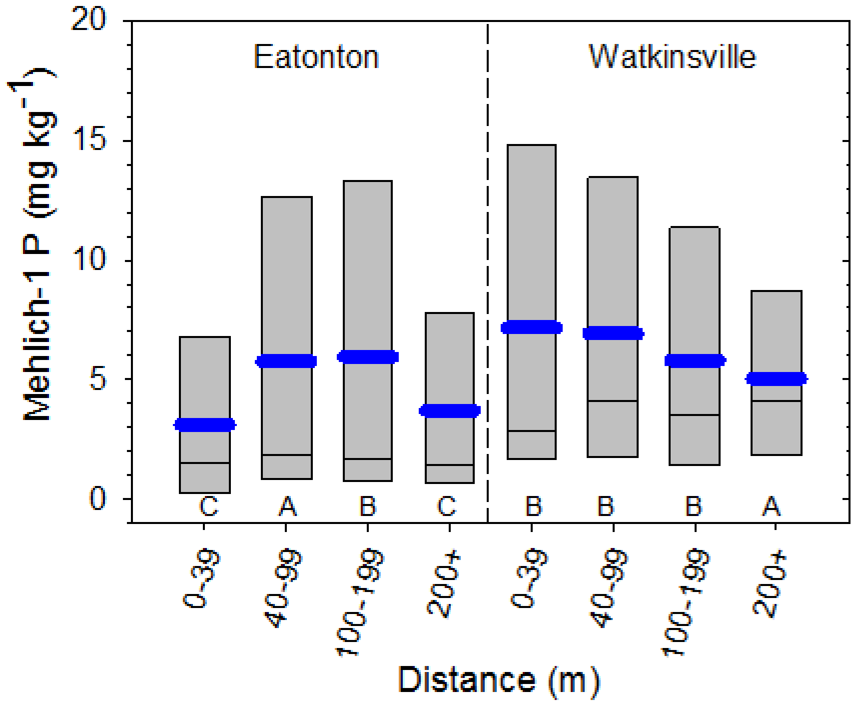

Figure 3.13. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations Mehlich-1 soil phosphorus in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

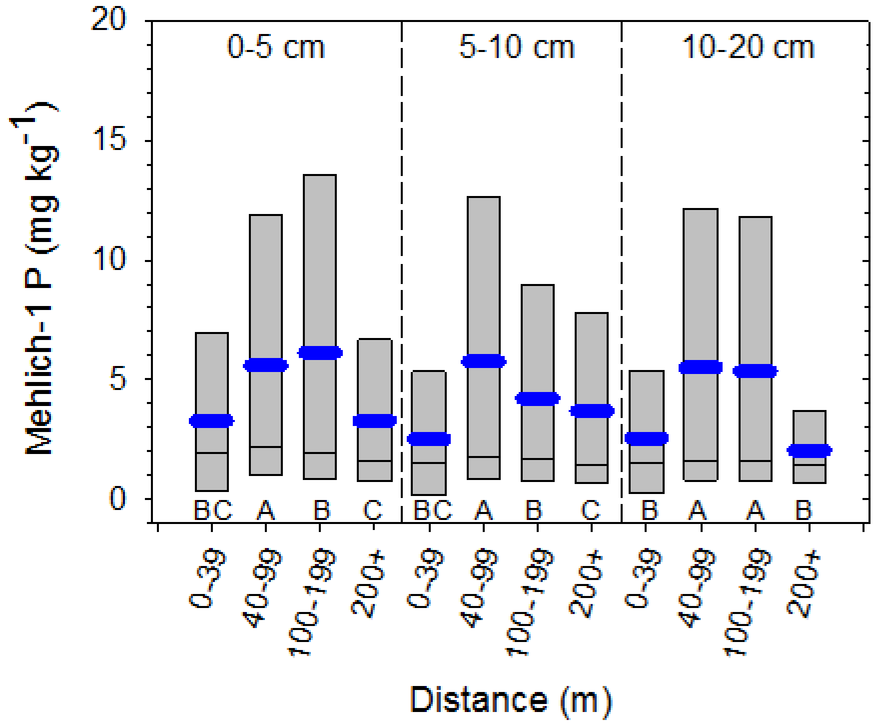

Figure 3.14. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations Mehlich-1 soil phosphorus at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

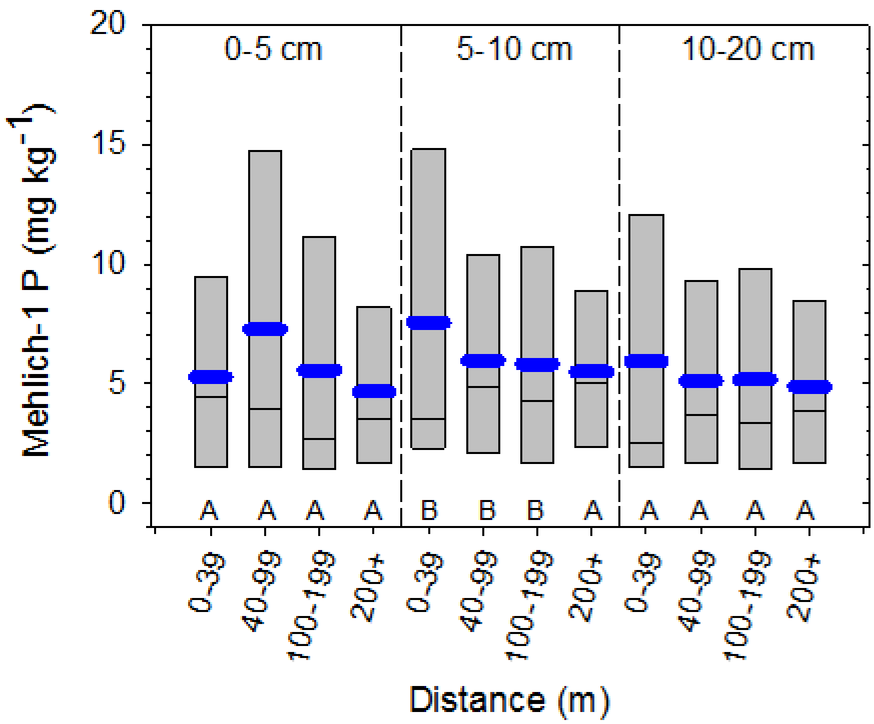

Figure 3.15. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus at each sampling depth in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest water source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

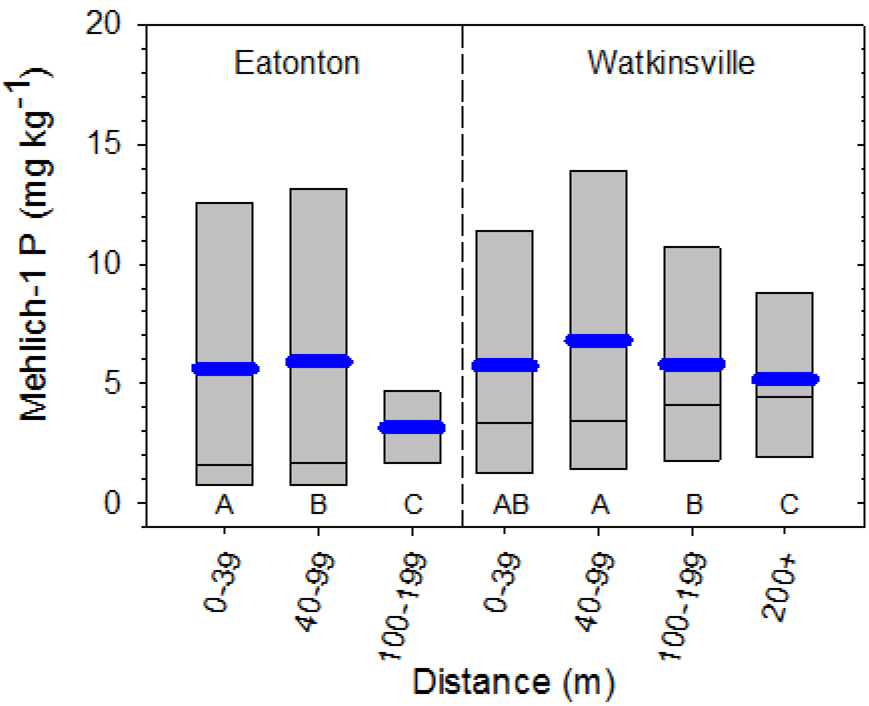

Figure 3.16. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus in geometric distances 0-39, 40-99, 100-199, and > 200 m from the nearest shade source in the Eatonton and Watkinsville sites. Within each site, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

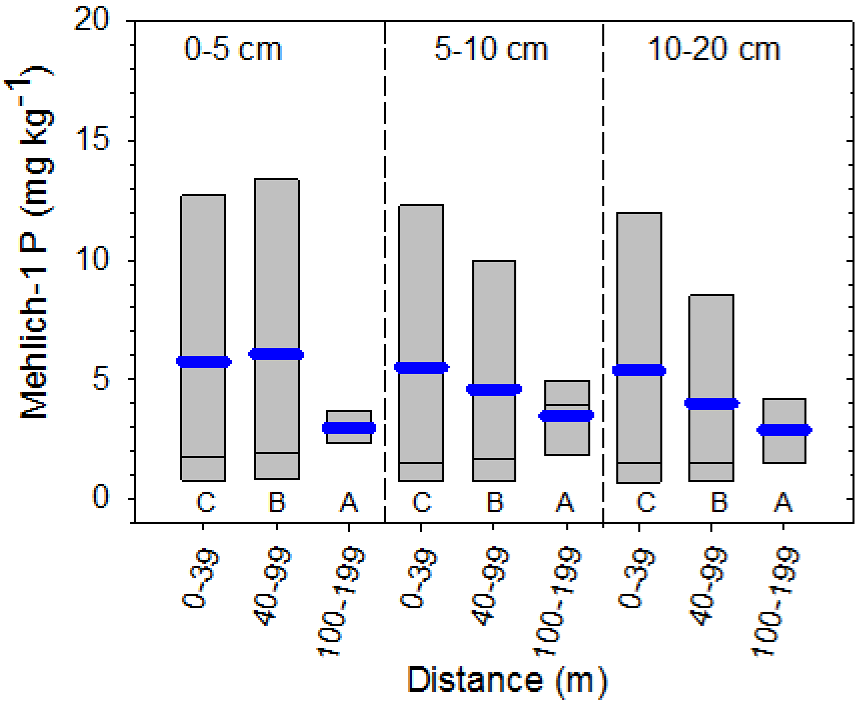

Figure 3.17. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus at each sampling depth in geometric distances 0-39, 40-99, and 100-199 m from the nearest shade source in the Eatonton site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

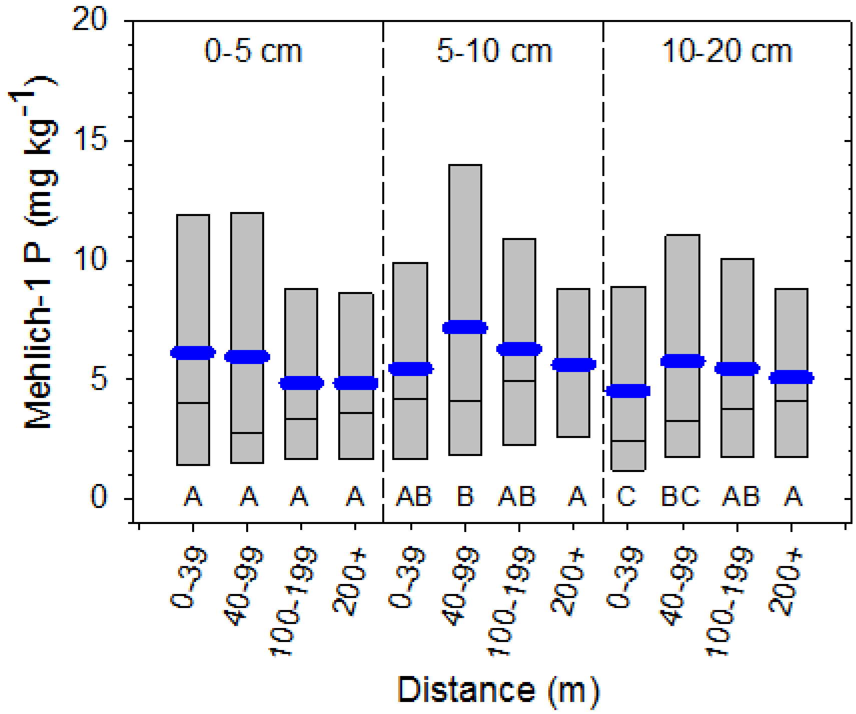

Figure 3.18. The first to third quartiles (gray range), mean (bold blue line), and median (thin black line) for concentrations of Mehlich-1 soil phosphorus at each sampling depth in distances 0-39, 40-99, 100-199, and > 200 m from the nearest shade source in the Watkinsville site. Within each depth, different letters indicate significant differences between medians according to pairwise comparisons using Wilcoxon Rank Sum test at p=0.05.

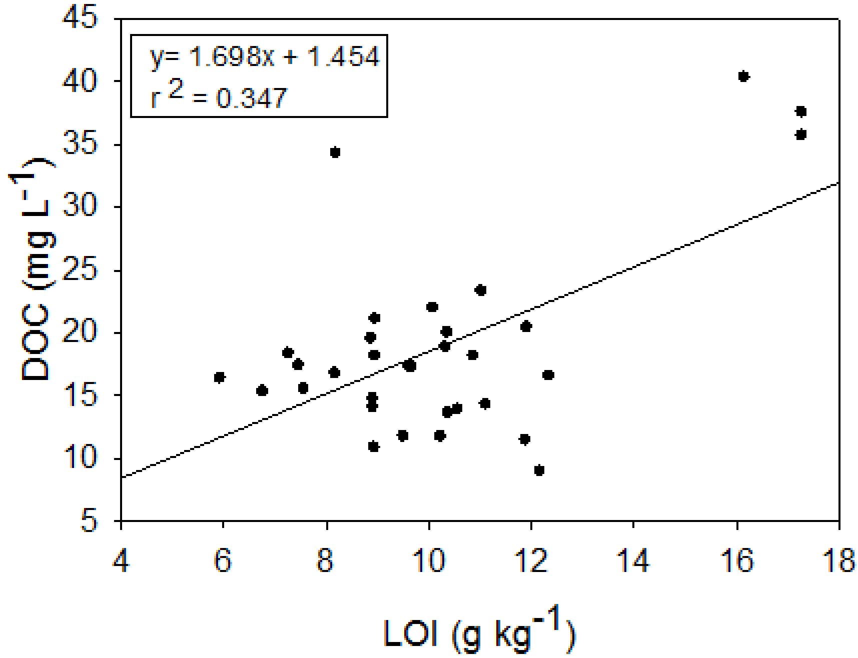

Figure 3.19. Relationship between mean dissolved organic carbon runoff per runoff collector and soil loss on ignition carbon (LOI) of the contributing area of interest (AOI).

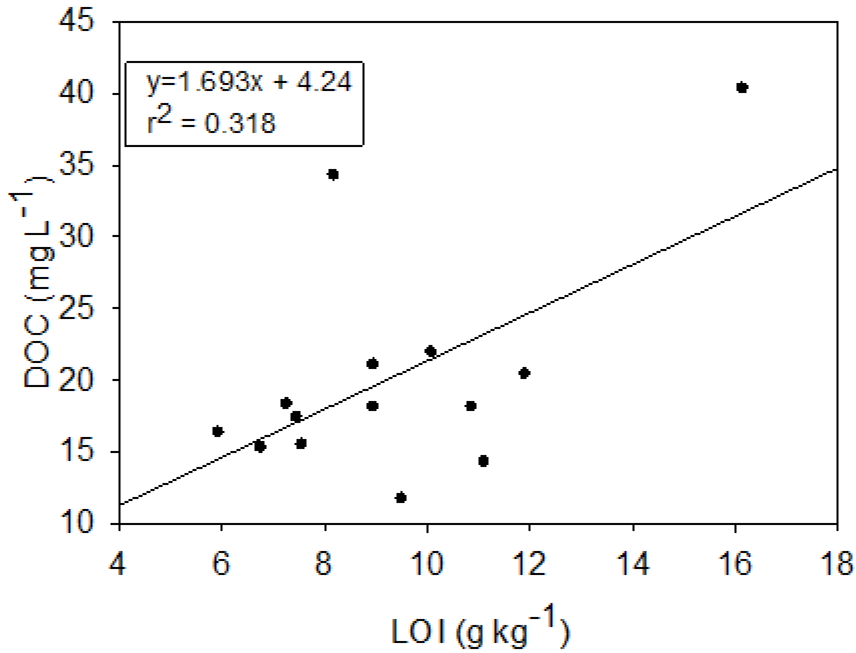

Figure 3.20. Relationship of mean dissolved organic carbon runoff per runoff collector and soil loss on ignition (LOI) carbon of the contributing area of interest (AOI) for Eatonton.

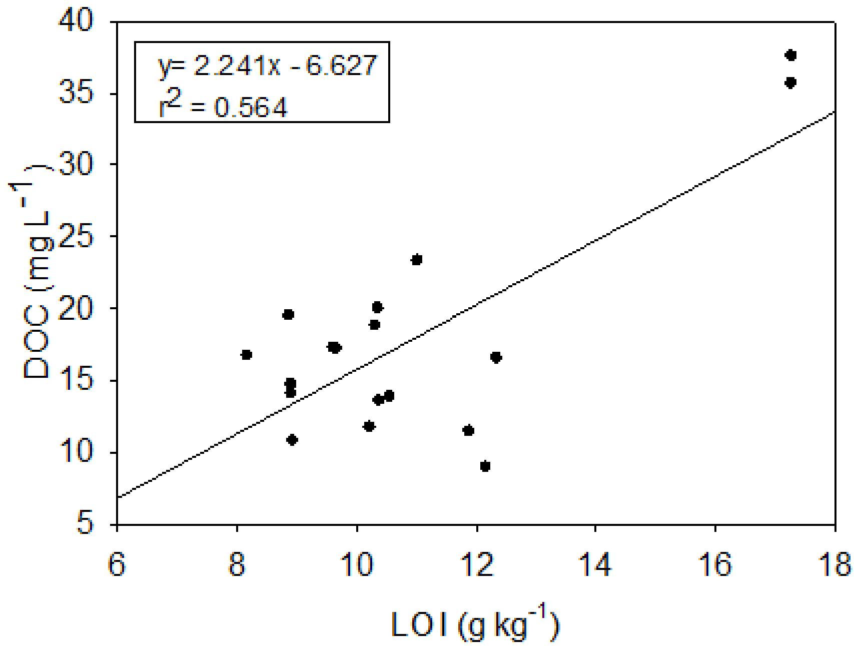

Figure 3.21. Relationship of mean dissolved organic carbon runoff per runoff collector and soil loss on ignition carbon of the contributing area of interest (AOI) Watkinsville.

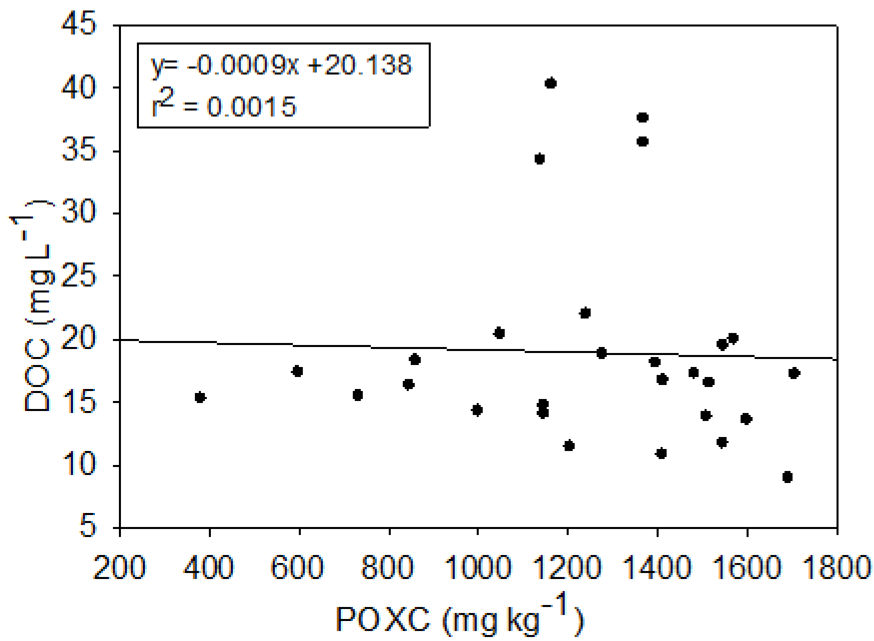

Figure 3.22. Relationship of mean dissolved organic carbon runoff per runoff collector and soil permanganate oxidizable carbon (POXC) of the contributing area of interest (AOI) for both locations.

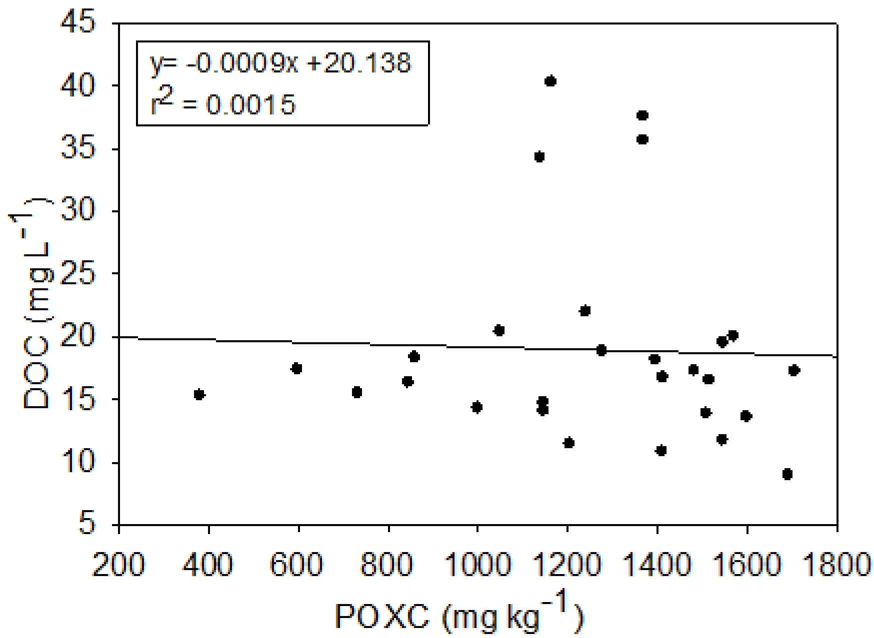

Figure 3.23. Relationship of mean dissolved organic carbon runoff per runoff collector and soil permanganate oxidizable carbon (POXC) of the contributing area of interest (AOI) for Watkinsville.

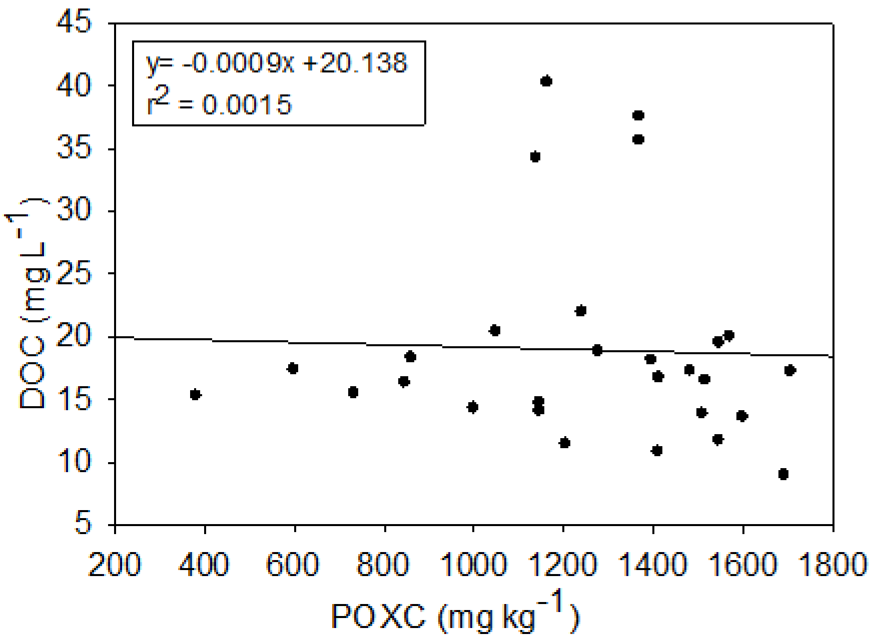

Figure 3.24. Relationship of mean dissolved organic carbon runoff per runoff collector and soil permanganate oxidizable carbon (POXC) of the contributing area of interest (AOI) for Eatonton.

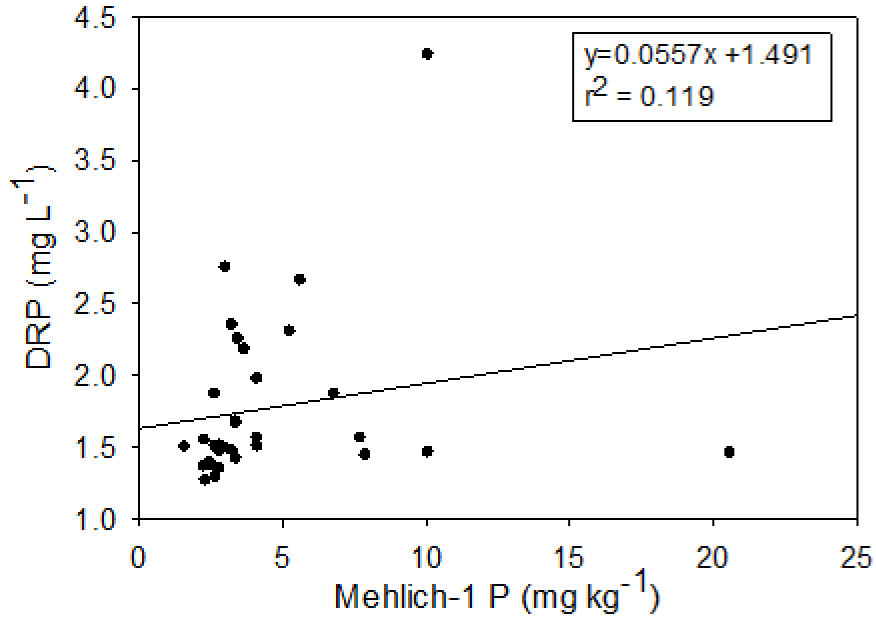

Figure 3.25. Relationship of mean runoff dissolved reactive phosphorus (DRP) per runoff collector to Mehlich-1 soil phosphorus of the contributing AOI is shown Eatonton and Watkinsville.

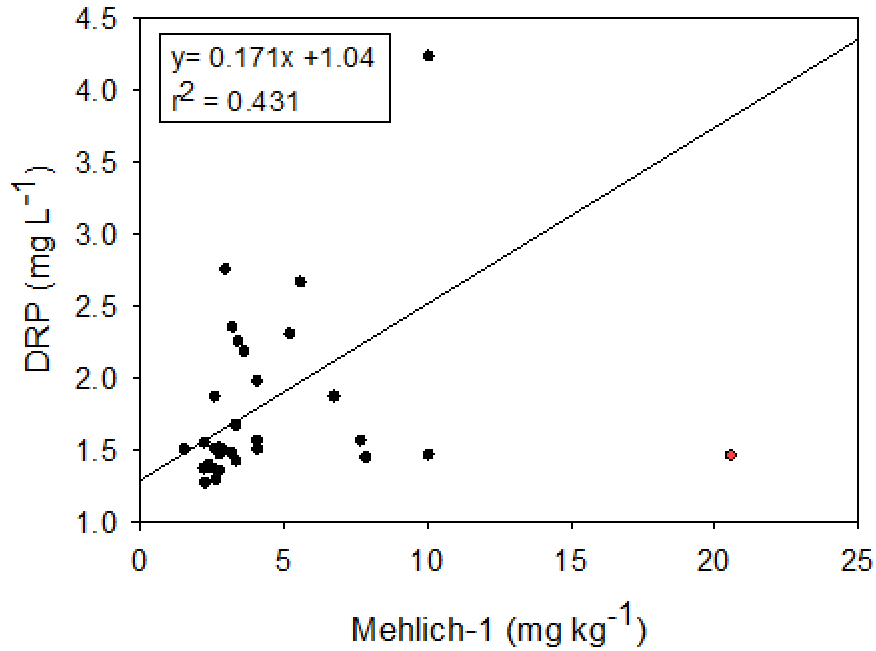

Figure 3.26. Relationship of mean runoff dissolved reactive phosphorus (DRP) per runoff collector to Mehlich-1 soil phosphorus of the contributing AOI is displayed without outliers (in red) for Eatonton and Watkinsville.

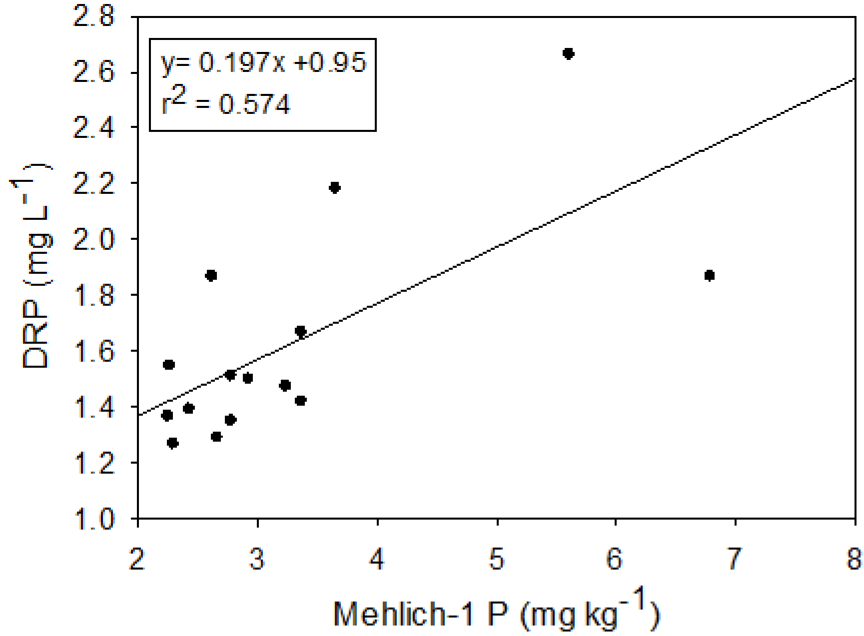

Figure 3.27. Relationship of mean runoff dissolved reactive phosphorus (DRP) per runoff collector to Mehlich-1 soil phosphorus of the contributing AOI for Eatonton.

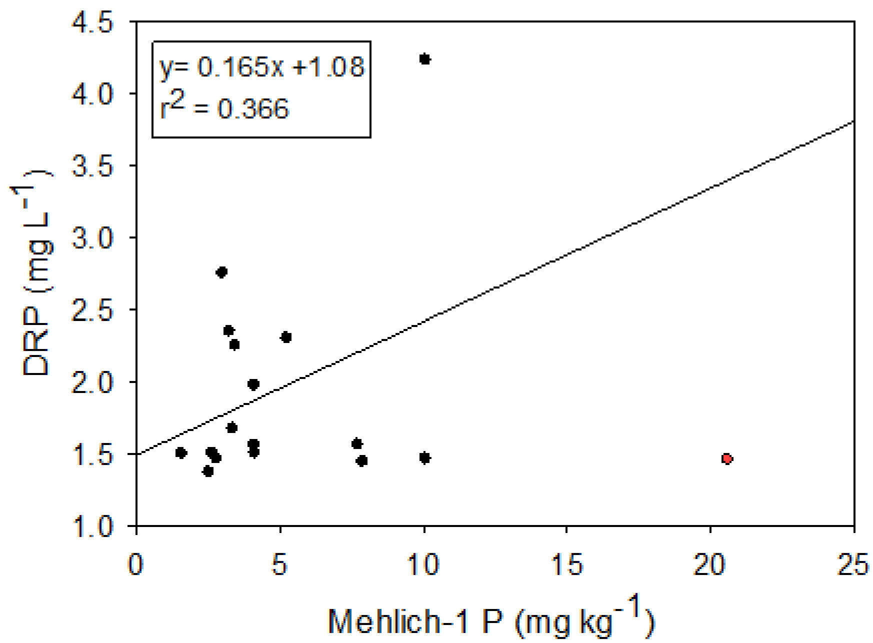

Figure 3.28. Relationship of mean runoff dissolved reactive phosphorus (DRP) per runoff collector to Mehlich-1 soil phosphorus of the contributing AOI without outliers (in red) for Watkinsville.

CHAPTER 4

CONCLUSIONS

The objectives of this study were to 1) determine the spatial distribution of C and P in southern Piedmont pastures and the potential of *in situ*, visible and near-infrared (VNIR) spectroscopy to rapidly determine bulk density (BD), soil C measured as loss-on-ignition (LOI) or permanganate oxidizable C (POXC), and 2) determine the relationship of soil C and P to concentrations of dissolved organic C (DOC) and dissolved reactive phosphorus (DRP) in runoff.

It was determined that there was a strong linear relationship ($y= 0.0469x - 0.629$; $r^2 = 0.8971$) between the measured LOI and third-party analyzed SOC concentrations. Additionally, POXC analysis methods exhibited more spatial variability than the LOI carbon analysis method. Because of the strong correlation between LOI and SOC, LOI is an effective and inexpensive means to monitor pasture soil carbon, however, to determine changes in soil C resulting from management changes, POXC is more effective. This analysis strategy is rapid, inexpensive, and rapidly (9 to 12 months) shows changes in soil C pools.

When soil variables (LOI, POXC, BD, and P) were measured at different distances from pasture equipage (water, hay, shade), various trends were found. Overall, LOI and POXC carbon showed the largest differences between 0-199 m and 200 m or greater from a hay, water, or shade source. Bulk densities were often lower when samples were taken close to a hay or water source because of increased hoof action in these areas. Conversely, bulk densities showed an increase in areas close to or within a shade source because of heavy cattle loafing. In many cases, bulk density values were inverse to carbon measurements. The highest values of carbon often occurred in uphill cattle camping areas where there were lower quantities of overland flow of runoff during a precipitation event. POXC values did not show many differences among the three sampling depths. This is likely because of soil heterogeneity and variability of POXC values. When soil P was compared to pasture equipage, no definitive trend was seen.

Soil P values were significantly higher in areas that were closest to hay sources and decreased as distance increased. Conversely, soil P was lowest in the areas closest to or in shaded areas and increased as the distance to shade increased. Finally, only the 0-5 cm sampling depth showed any relationship to the distance to the nearest waterer. These values were highest in samples taken between 40 and 99 m from a waterer, suggesting high activity near by a water source, though not in the immediate vicinity.

The use of VNIR reflectance spectra (from 450-2400 nm) to estimate soil C and soil P will need to be investigated further. Although these parameters have been shown to be accurately determined through spectroscopy on processed samples, the variable results make the use of this technology a poor indicator of nutrient concentrations for *in situ* sampling. However, improved relationships for each of the measured parameters (LOI, POXC, and soil P) when soil type and sampling depth are taken into account make the future use of *in situ* VNIR nutrient estimates a possibility.

Runoff concentrations of DOC and DRP showed linear trends to LOI and soil P, respectively; however, POXC showed almost no relationship (r-square=0.0015) to DOC values. Relationships improved when runoff samples were divided by sample location, suggesting that some differences may be due to soil parameters present at each location. Additionally, correlations were strongest when soil P values were 10 mg P kg-1 or lower; high soil POXC values also showed greater runoff variability. In order to improve the ability to predict runoff concentrations of DOC and DRP, additional factors need to be taken into account. Initial multivariate analyses indicated that overall rainfall total for the event did not contribute to DOC or DRP concentrations, however, additional factors of interest may include the slope; air temperature; soil type or percentages of sand, silt, and clay; and rainfall intensity.

Because of the large scale of the agriculture industry as well as increased pressure for grow more using fewer resources, innovative methods for agricultural management are crucial. Such innovative techniques will not only make production systems as efficient as possible, but also promote sustainability. Developing not only a better grazing system, but also the most effective monitoring system can have huge positive impacts on the cattle production industry and the surrounding environment.

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