EVALUATING SOIL CHANGE AFTER TWENTY-YEARS OF INTENSIVE PINE PLANTATION MANAGEMENT IN THE SOUTHEASTERN UNITED STATES

by

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(Under the Direction of Daniel Markewitz)

ABSTRACT

The sustainability of soil productivity under pine plantation management remains uncertain due to limited long-term soil sampling. Resampling of the forest floor and soil from the Consortium of Accelerated Pine Production Studies (CAPPS) plots in Georgia USA after twenty years of plantation management has provided an opportunity to study long-term soil response to fertilization and vegetation control from mid- to end-rotation focusing on soil C, N, and acidity. Fertilization had no effect on soil C content but increased soil N content. Fertilization also induced soil acidification as reflected by decreased pH, reduced base cation concentrations, and elevated exchangeable acidity. Vegetation control reduced both soil C and N content with little effect on soil acidity. Regardless of management, soil under plantations generally saw reduced soil C and increased soil acidification from mid- to end-rotation. These trends suggest stresses on soil sustainability that require continued management.

INDEX WORDS: repeated measure, pine plantations, fertilization, understory vegetation control, soil change, carbon, nitrogen, base cations, acidity

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CHAPTER I

INTRODUCTION

Pine plantations (predominantly loblolly (*Pinus taeda*) and slash (*Pinus elliottii*) pine) have a large footprint on the landscape of the southeastern United States. Currently, 15% (~41 million ha) of forest land in the southeastern United States is managed pine plantation (Oswalt et al., 2019). These plantations utilize practices such as fertilization and competing vegetation control to increase site productivity. Previous work has demonstrated that fertilization and competing vegetation control in plantations can significantly alter soil properties such as total C and N. Alterations in soil properties can positively or negatively impact ecosystem services such as carbon sequestration, nutrient cycling, and water storage (Adhikari & Hartemink, 2016).

The purpose of this thesis is to evaluate the effects of pine plantation management on soil C, N, and exchange chemistry at the end of a three-decade rotation including a twenty-year resampling in 2001 (~age 10) and 2021 (~age 30). Treatments included repeated fertilization, largely with N and P but also some micro-nutrients, and complete understory vegetation control. Soil total C and N were analyzed in both mineral soil and forest floor as was soil pH; exchangeable acidity; and exchangeable bases ($Ca^{2+,}$ Mg^{2+}, and K^{+}). These properties were selected based on their value as soil health indicators (Bagnall et al, 2023). Soil health indicators provide insight into the capacity of soil to carry out ecosystem functions including carbon sequestration, nutrient cycling, and water storage. Though previous studies have evaluated the impact of fertilization and competing vegetation control on soil a limited number have looked specifically at the end of the rotation or have incorporated a 20-yr resampling.

The effect of pine plantation management on soil C, N, and exchange chemistry is evaluated in this thesis through both literature review and experimentation. In chapter two, relevant literature surrounding the pertinent soil properties (i.e., total C, total N, pH, acidity, and exchangeable cations) and the impacts of forest management are reviewed. In chapter three, experimental effects of management on soil C and N are presented. In chapter four, the effects of management under the same experiment on exchange chemistry are addressed through the soil pH, acidity, and exchangeable cations analysis. In chapter five, the major findings of this thesis are summarized.

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CHAPTER II

LITERATURE REVIEW

2.1 Carbon Cycling in Forest Soils

Soil C stocks make up a significant portion of the C budget globally and at the forest stand scale. Globally, soil contains 1700 Pg-C, which is approximately double the atmospheric stock (875 Pg-C) and quadruple (450-Pg-C) the vegetation stock (Friedlingstein et al., 2022). At the forest stand scale, 32-60% of total ecosystem carbon is stored within the soil (Pan et al., 2011). This includes coarse woody debris, forest litter, dead roots, and humified substances. Soil properties have a significant influence on C storage in forests. Clay content is positively correlated with soil carbon content as clays provide protection through mineral association (Gruba & Socha, 2019; Schimel et al., 1994). Tree species and diversity impact soil C storage through differences in litter decomposition and belowground C investment. Conifer dominated forests store more soil C compared to broadleaf systems as litter decomposition rates are slower under conifers (Dawud et al., 2016). Diverse forests also tend to store more C in the soil compared to monoculture systems as different rooting strategies of broadleaf and conifer species are complementary increasing total C inputs within the soil (Augusto & Boca, 2022; Dawud et al., 2016).

Soil C is both temporally and spatially dynamic at the forest stand scale. The residence time of soil C varies from a few years for intact plant material to centuries for highly humified substances (Gaudinski et al., 2000). Most C in soil moves through the system via litterfall, root turnover, autotrophic respiration and heterotopic respiration (Fahey et al., 2005). Climate plays a

major role in soil C movement, since it is directly tied to biological activity. Temperature and precipitation have been identified as primary factors for determining soil C stocks and fluxes (Guo et al., 2006; Luo et al., 2017). Increases in temperature and/or drought can result in increased loss of soil C (Crowther et al., 2016; Melillo et al., 2011; Noormets et al., 2010; Ziegler et al., 2017). The loss is potentially linked to increased heterotrophic respiration from soil warming and reduced photosynthetic inputs from slower plant growth during drought.

2.2 Nitrogen Cycling in Forest Soils

Soil N makes up a major portion of the active nitrogen budget globally and at the forest stand scale. Approximately, 81 Pg of N is actively cycled in the soil across the globe which is more than the16 Pg of N in vegetation (Zhang et al., 2020). Although the atmospheric N pool is the largest at 4,000,000 Pg of N it is biologically inactive and unavailable for uptake by organisms (Zhang et al., 2020). In forest stands, soil can hold anywhere between 2 to 22 Mg/ha of N within 100 cm of soil (Johnson & Turner, 2014). Across boreal, temperate, and tropical forests, soil N makes up anywhere between 58 to 88% of active ecosystem N (Cusack et al., 2010; Richter & Markewitz, 2000; Sponseller et al., 2016). Tropical and temperate forest ecosystems are N-limited as the systems display significant growth responses of 19-60% when N is added (LeBauer & Treseder, 2008). Ecosystems can remain productive in the face of N limitation by N cycling.

Due to the limited supply of N in forest soils, ecosystems depend heavily on cycling to maintain productivity. The internal cycling of nitrogen involves mineralization, nitrification, and immobilization. Mineralization is the oxidation of organic N to NH_4^+. Rates of mineralization in

forest systems are highly variable, ranging from 35-105 kg N $ha^{-1}yr^{-1}$, depending on other soil properties including moisture, temperature, initial soil N content, bulk density, pH, litter quality, and microbial diversity (Adams & Attiwill, 1986; Craine et al., 2007; Scott & Binkley, 1997). Nitrification or the oxidation of NH_4^+ to NO_3^- occurs at an average rate of 2.58 mg/kg*day in forests across boreal, temperate, and tropical climates, which is lower than the cropland average of 3.82 mg/kg*day (Li et al., 2020). The reason forest ecosystems exhibit reduced nitrification rates compared to the cropland average is because acidic forest soils, which typically have a pH <5.5, are unfavorable to the primary bacterial nitrifiers (Ste-Marie & Paré, 1999; Szukics et al., 2010). Although there are instances of heterotrophic nitrifiers handling extremely acidic forest soils with pH <3.0 (Li et al., 2018). Lastly, immobilization is the incorporation of inorganic NH_4^+ and NO_3^- to organic N forms mainly proteins in microbial cells. At the forest stand scale, the average immobilization rate was approximately 15.26 mg/kg*day, which is greater than the average immobilization rate of cropland at 5.55 mg/kg*day (Li et al., 2021). Immobilization rates are driven directly by microbial biomass which is generally higher in natural ecosystems like forests compared to cropland where intensive practices have been detrimental microbial function and productivity.

External fluxes are another important component of N cycling in forest soils. Fluxes include fixation, deposition, volatilization, denitrification, and leaching. N fixation is the main process to form biologically active N from inert N_2 with the exclusion of anthropogenic processes such as the Haber Bosch process. N fixation makes up the majority of biologically active N inputs to boreal, temperate and tropical forests at rates ranging from 2.1 to 18 kg/ha*yr (Bormann et al., 1977; Yu & Zhuang et al., 2020). The estimates of biological N fixation include contributions from free-living and symbiotic organisms. Rates of biological N fixation are

controlled by temperature, moisture, and initial N content as these all affect the success of symbiotic associations and biomass of free-living fixers. Deposition of organic and inorganic nitrogen species is the other significant N input to forests contributing on average 6 kg/ha*yr of N (Vishwakarma et al., 2023).

Potential fluxes out of forest soil include NH_4^+ conversion to NH_3 that can escape into the atmosphere through volatilization. Volatilization of NH_4^+ is limited in forests due to low soil pH and is not considered a major flux (Avnimelech & Laher, 1977; Mahendrappa & Ogden, 1973). Another pathway is NO_3^- that can be reduced in low oxygen conditions and lost as N_2, NO_2, or N_2O gas through the process of denitrification. Average rates of denitrification in well-drained forest soils tend to be quite low at ~1 kg/ha*yr due to unfavorable redox conditions (Barton et al., 1999; Szukics et al., 2010). There is evidence that intermittent flooding and low oxygen conditions in forest soils can increase denitrification up to weeks after the event (Tomasek et al., 2019). A final pathway is losses of N through leaching either as NO_3^-, NH_4^+, or dissolved organic nitrogen, although leaching losses of N are typically limited by nutrient retention through internal cycling (Davis, 2014; Gundersen et al., 2006). If forests ever became N saturated, theoretically N would leach more readily, however, N saturation in the absence of forest fertilization has not often been observed (Schrijver et al., 2008). It should be noted that any transformations that utilize redox reactions such as nitrification or denitrification in the N cycle require C as an energy source and therefore are coupled to the soil C cycle.

2.2 Soil Exchange Chemistry in Forests

Soil exchange chemistry involves the interaction of soil solution with exchange sites on soil mineral and organic surfaces. The status of the soil exchange chemistry can be assessed through pH, exchangeable acidity, and exchangeable bases. Soil pH is a measure of the activity of H^+ ions in soil. Most forest soils have acidic pHs between 4-6 depending on soil parent material and forest composition (Riha et al., 1986). Exchangeable acidity compliments pH as it provides the total concentration of acidic ions (i.e., H^+ and Al^{3+}). Counter to exchangeable acidity are the exchangeable bases, which are alkaline ions (i.e., Ca^{2+}, Mg^{2+}, K^+, and Na^+). Soil exchange properties provide insights into acidification which occurs through natural and anthropogenic processes.

Soil acidification occurs naturally during forest soil formation as inorganic (i.e., carbonic acid) and organic (i.e., fulvic acid) acids weather minerals resulting in displacement of base cations either through leaching or plant uptake. Natural acidification, which has been observed using proton budgeting, occurs largely through excess base cation accumulation in woody vegetation with minor contributions from the dissociation of organic acids produced by vegetation (Fujii et al., 2008). The biological processes that naturally acidify forest soils have also been observed after afforestation. Conversion of tropical and temperate grasslands to forests resulted in a decrease in soil pH from 0.2-1.0 units, decrease in Ca concentration by 29-40%, and increase in exchangeable acidity by 0.5-1.2 kmol+/ha*yr through 100 cm of mineral soil (Berthrong et al., 2009; Huang et al., 2022; Jobbagy & Jackson, 2003). Acidification from the conversion of grassland to forest was driven by redistribution of base cations into woody vegetation and not leaching. Afforestation has also been shown to neutralize acidity when initial pH was 4 or lower when compared to fallow, grassland, and cropland (Hong et al., 2018). It was

determined that for soils with pH lower than 4 acidity was neutralized by hydroxide ions released from vegetation from the excess uptake of anions.

The leading cause of human-driven soil acidification in forests is acid rain. Increased emissions from burning fossil fuels led to a drop in precipitation pH from 5.6 to 3.5-5.0 as well as an increase in NO_3^- and SO_4^- concentrations across North America, Europe, and Asia (Menz & Seip, 2004). Acid deposition leads to soil acidification through the leaching of NO_3^- and SO_4^-, which also move base cations to maintain ionic balance. Soils with low buffering capacity are at the greatest risk as they do not have the capacity to replace lost cations via mineral weathering. In the United States, hardwood forests in the northeast lost approximately 42 kmol+/ha of Ca over 43 years which was approximately double the amount taken up by biomass indicating increased leaching from acid deposition (Likens et al., 1996). Conifer forests in the southeastern United States also acidified as soil pH dropped by 1 unit, base cation concentration decreased by 1.57 kmol+/ha and total acidity increased by 3.28 kmol+/ha over thirty years, however, acid deposition only accounted for 38% of acid inputs (Markewitz et al., 1998). The issue of acid rain in North America and Europe has improved as N and S emissions have dropped by ~80% over the last 50 years (Grennfelt et al., 2019). Reduction of emissions has already led to recovery of soils in the northeastern United States with soil pH increasing from 4.26 to 4.43 though base cation levels have remained suppressed (Lawrence et al., 2015). Soil acidification is still an active concern in China as soil pH in tropical and temperate forests there has dropped by 0.36 units since the 1980's with 84% of acid input coming from acidic deposition (Zhu et al., 2016). China has not instituted any effective emissions control measures and is still experiencing elevated concentrations of N and S species in precipitation.

2.3 Fertilization and Forest Soils

Pine plantation management in the southeastern Unites States utilizes a myriad of practices to push productivity. Fertilization is one such practice done to overcome nutrient limitations on a given site, typically N and P (Fox et al., 2007b). Using fertilizers has increased productivity by ~5 Mg/ha*yr compared to naturally regenerated pine forest (Stanturf et al., 2003). Due to increased costs of fertilizer materials, the amount of plantation land fertilized has decreased from 650,000 ha in the year 2000 to 240,000 ha in 2016 (Albaugh et al., 2019). Although fertilization has benefited short-term productivity it has the potential to negatively impact soil properties and therefore, long-term soil productivity.

The impacts of fertilization on soil C and N in pine plantations have been previously studied in different aged systems on multiple soil types across the United States. Soil C in early-aged rotations did not respond to fertilization across several soil orders (Lee et al., 2003; McFarlone et al., 2009). However, Lee (2003) and McFarlone (2009) did find an increase in soil N by as much as 20% from fertilization in early-aged rotations. Resampling of pine plantations from early- to mid-rotation found no effect from fertilization on either soil C or N over time (Johnson et al., 2003; Leggett et al., 2006). At mid-rotation, fertilization had no effect on soil C and N (Mosier et al., 2019; Scott et al., 2014; Vogel et al., 2021). By end-rotation, fertilization had mixed effects on soil C and N. One study on an Inceptisol found early-rotation fertilization had no effect on soil C and N by end-rotation (Harding & Jokela, 1994). A second study on a Spodosol found early- to mid-rotation fertilization increased soil C by 6% and soil N by 6% (Vogel et al., 2011). The mixed response of soil C and N to fertilization is potentially a function of differences in treatment length and intensity.

Observed increases in soil C and N from fertilization are related to shifts in soil microbial communities and function. Fertilization with N has been shown to decrease heterotrophic respiration by 16-41% (Bowden et al., 2004; Phillips & Fahey, 2007; Wallenstein et al., 2006). Decreases in heterotrophic respiration were accompanied by a ~40% loss of microbial biomass (Demoling et al. 2008; Wallenstein et al., 2006). Oxidase enzyme activity which is responsible for the breakdown of recalcitrant C compounds was also suppressed by fertilization (Billings & Ziegler, 2008; Jian et al., 2016; Saiya-Cork et al., 2002). Changes in microbial communities and function were correlated with decreases in belowground investment by vegetation in the form of fine roots and mycorrhiza (Carrara et al., 2018; Phillips & Fahey, 2007). The suppression of microbial communities by fertilization would reduce decomposition rates in soil allowing for greater accumulation of C and N.

The effects of fertilization on soil exchange chemistry have not been directly addressed in pine plantations, however, there are studies on the effects of simulated N deposition in a variety of forest soils. Meta-analysis of simulated N deposition studies shows soil pH decreases by 0.2 units in forests after 40 years across a climatic gradient (Chen et al., 2023; Tian et al., 2015). For alpine forest soils, N deposition decreases soil pH by 0.2-1.0 units and reduces base cation concentrations by 24-40% after adding 60-150 kg/ha*yr of N over 3-11 years (Bowman et al., 2008; Lieb et al., 2011). Boreal forests have exhibited a 1.0 unit decrease in soil pH, 70% reduction in base cation concentration, and 222% increase in exchangeable Al^{3+} after 30 years of N additions though effects diminished below 45 cm (Hogberg et al., 2006). Tropical forest soils also exhibit some level of acidification from 6-8 years of N additions with a 0.2 unit decrease in soil pH and 45% reduction in base cation concentration (Lu et al., 2014; Mao et al., 2017). There are instances of tropical forest soils resisting acidification from N additions though the study

utilized a broadleaf plantation rather than native vegetation (Huang et al 2021). Lastly, temperate forests have seen a soil pH decrease of 0.7 units with significant mobilization of Al^{3+} (Bergkvist et al., 1991). Since simulated N deposition has had a clear acidifying effect on soils it is likely that fertilization in pine plantations will behave similarly. The acidifying effect of N additions is likely related to increases in vegetation growth. Many of these systems are N limited. Also, if N additions are large enough then the system can become N saturated and increase base cation reductions through leaching of NO_3^-.

2.3 Vegetation Control and Forest Soils

Competing vegetation control is another common practice done to improve crop tree productivity in pine plantations. Competition in plantations can be detrimental to crop tree growth early in rotation and typically requires removal by herbicides such as glyphosate, hexazinone, imazapyr, metsulfuron methyl, and triclopyr or other vegetation control strategies (Fox et al., 2007a). Controlling vegetation early in rotation has been shown to increase pine biomass as much as 12-fold demonstrating its value as a silvicultural practice (Ferreira et al., 2020; Nelson et al., 1981). However, the removal of vegetation can be detrimental to long-term productivity as it shifts above- and belowground nutrient dynamics impacting a variety of soil properties.

The response of soil C and N to vegetation control has been studied in pine plantations at varying stand ages. In early-rotation aged plantations (<10 yrs), soil C and N remain unaffected by vegetation control (Lee et al., 2003; McFarlone et al., 2009). An exception to this is in a study where vegetation control and fertilization were used together resulting in a 23% decrease

in soil C and no change in soil N (Sarkhot et al., 2007). Repeated measurement of soils from early- to mid-rotation (10-16 yrs) show vegetation control enhances soil C and N losses by 6-10% in pine plantations (Miller et al., 2006). Mid-rotation response of soil C and N to vegetation control is mixed. A set of studies on spodosols found a ~16% decrease in soil C from vegetation control at mid-rotation (Maillard et al., 2010; Shan et al., 2001). Conversely, another set of studies at mid-rotation on Ultisols and Alfisols saw no change in soil C from vegetation control at mid-rotation (Mosier et al., 2019; Scott et al., 2014; Vogel et al., 2021). By end-rotation, vegetation control has been shown to decrease soil C by 6% with no effect on soil N (Vogel et al., 2011). It should be noted that vegetation control includes a range of practices that use different herbicides, can be broadcast or banded, and are applied over different periods of time. Complete vegetation control was not maintained in studies that lacked significant response. In contrast, vegetation control has the capacity to negatively affect soil C and N when used throughout a rotation. The ability of vegetation control to reduce soil C and N is connected to the loss of belowground biomass. Understory vegetation has been estimated to make up anywhere from 27 to 80% of belowground biomass in temperate and subtropical pine forests (Gonzalez et al., 2013; Helmisaari et al., 2007; Jiang et al., 2018). When vegetation control is used in the case of pine plantations then there can be as much as a 49% net loss of fine root biomass when considering increased growth response from crop trees (Shan et al., 2001). Soil N losses increase after vegetation control because of increased uptake by crop trees as well as leaching from increased presence of inorganic N. Studies have demonstrated that vegetation control increases mineralization and nitrification producing leachable NO_3^- (Gurlevik et al., 2004; Munson & Timmer, 1995).

The response of exchange chemistry status in forest soils to understory vegetation removal is mixed. In a Mason pine (*Pinus massoniana*) plantation, there was no effect on soil pH from the removal of understory vegetation after three years of treatment (Shen et al., 2018). A lack of response in soil pH from understory vegetation removal was also seen in mixed species plantations after a year of treatment (Zhao et al., 2011). Conversely, understory vegetation removal under Scots pine (*Pinus sylvestris*) resulted in 0.6 unit decline in soil pH and 51% decline in total base cations after six years (Zhang et al., 2022). The difference in response could be related to nutrient status of the soils as the nutrient poor soils in Zhang (2022) likely have poor acid neutralizing capacity compared to the other studies such as Shen (2018) where the plantations were located on nutrient-rich soils with high acid neutralizing capacity. Therefore, the relationship between understory vegetation removal and the status of soil exchange appears to be based on soil type as supported by current literature.

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CHAPTER III

SOIL CARBON LOSS AND OCCULT NITROGEN GAINS AFTER THREE DECADES UNDER MANAGED PINE[1]

[1] Hoffman, S.A. and Markewtiz, D. To be submitted to *Forest Ecology and Management*

3.1. Abstract

Understanding the impacts of silviculture on soil C and N is critical for sustainable forest management. This study investigates the influence of fertilization and vegetation control on soil C and N concentrations and contents in *Pinus taeda* plantations of four well-replication locations in Georgia, USA considering soil depths of 0-10, 10-30, and 30-50 cm and stand ages of ~10 and ~30 years. It is hypothesized that vegetation control would reduce soil C and N due to the removal of belowground inputs from understory vegetation, while fertilization would enhance soil C and N by suppressing microbial decomposition. Vegetation control negatively impacted mineral soil C and N through 50 cm, but compensatory accumulation in the forest floor and aboveground biomass resulted in an overall gain to ecosystem C and N. Fertilization increased forest floor and mineral soil C and N through 50 cm, although gains in the mineral soil were modest compared to increases in forest floor and aboveground biomass. Stand age showed no clear effect on mineral soil C, while mineral soil N increased at one Piedmont location (Eatonton) and decreased at a lower coastal plain location (Waycross) over time. Increased soil N in Eatonton was related to an unexplained increase in ecosystem N. Despite not mirroring operational practices, this study demonstrates that intensive management practices can affect soil C and N dynamics in pine plantations.

3.2.Introduction

Pine plantations are estimated to currently make up 19% of forest cover (~41 million ha) in the southeastern United States (Oswalt et al., 2019). As such, understanding plantation management effects on forest soil is critical for sustaining productivity and ecosystem services such as carbon sequestration, nutrient cycling, and water quality and quantity (Adhikari & Hartemink, 2016). Plantation management in the Southeast typically involves a combination of genetic improvement, site preparation, understory vegetation control, and fertilization with N and P as a way of accelerating growth (Fox et al., 2007). Studies on the effects of plantation management on soils have focused on total organic C and total N as they are key indicators of the soil's capacity to provide ecosystem services and soil health (Lehmann et al., 2020). Soil organic C acts as the main energy source for microbial communities in the soil that are responsible for nutrient cycling and soil structure formation via aggregation. Soil N acts as an important N pool that plants and microbes rely on to function and grow.

The effects of pine plantation management, mainly fertilization and vegetation control, on soil C and N have been explored through a range of stand ages. In early-rotation aged pine plantations, for example, across several soil orders (i.e., Ultisol, Alfisol, and Inceptisol) soil C did not respond to fertilization or vegetation control, but soil N increased by as much as 20% from N fertilization (Lee et al., 2003; McFarlone et al., 2009). In similar aged pine plantations on Spodosols, combined vegetation control and fertilization resulted in soil C losses as high as 23% with no changes to soil N (Sarkhot et al., 2007). When considering how soil C and N

changed over time from planting to ages 11-18, fertilization and vegetation control applied shortly after planting had no effect on soil C or N if planted in Ultisols, Alfisols, and Inceptisols (Johnson et al., 2003; Leggett et al., 2006). In contrast, in some cases vegetation control early in rotation amplified soil C and N loses by 10 and 6%, respectively, across various soils in the southeastern United States (Miller et al., 2006). By mid-rotation, the effects of vegetation control and fertilization remain mixed. Pine plantations planted on Spodosols at mid-rotation lost soil C by as much as 16% from vegetation control (Maillard et al., 2010; Shan et al., 2001) but on other soil orders (Ultisol and Alfisol) there was no change in soil C from vegetation control or fertilization (Mosier et al., 2019; Scott et al., 2014; Vogel et al., 2021). Many of these mid-rotation studies that found no treatment effect only applied fertilizer or vegetation control treatments early in rotation rather than annually or biannually through mid-rotation. By the end of rotation, one study on Inceptisol with fertilization applied during the first two years of rotation had no effect on soil C or N (Harding & Jokela, 1994). In a second study at the end of rotation on a Spodosol that applied treatments up until age 17, vegetation control decreased soil C by 6 % with no change in soil N while fertilization increased soil C by 6 % with a 6 % increase in soil N (Vogel et al., 2011). Although the relationship between plantation management and soil C and N has been explored for various age groups many of the studies did not consistently apply treatments throughout rotation; only studied the response on a select soil type such as Spodosols; and mostly focused on stand ages up to mid-rotation. Therefore, a need exists to study the impacts of plantation management on soil C and N with consistent treatments on several soil types over the span of a rotation.

The goal of this study is to evaluate the effects of intensive pine plantation management (i.e., fertilization and vegetation control) on soil C and N from the first decade to the third decade

of the rotation. This is accomplished through a resampling of the Consortium of Accelerated Pine Production Studies (CAPPS) plots from 2001 when plots were 8 to 12-yrs-old to 2021 just prior to final harvest. The CAPPS experiment includes five installations that stretch over a 360 km distance from middle to southern Georgia so captures a soil and climatic gradient. Results from the 2001 sampling showed that soil C and N significantly decreased by 20 and 19% from vegetation control while the effect of fertilization was relatively small and location dependent (Echeverria et al., 2004; Satori et al., 2007). In a 2010 sampling of a subset of the CAPPS plots that focused on the surface 0-10 cm there was a 13 and 17% loss of soil C and N on average with herbicide control and an average soil C and N increase of ~10% from fertilization except for a single site where fertilization was discontinued (Rifai et al., 2010). Aboveground accumulations of C and N in biomass was enhanced by fertilization and vegetation control (Zhao et al., 2022). Given previous research and our own findings from these sites, it is hypothesized that vegetation control will have a persisting negative effect on soil C and N as belowground contributions of C and N from understory vegetation will still be absent. Decreases in soil C and N from vegetation control are typically explained by loss of belowground biomass from understory vegetation such as fine roots. In pine plantations where soil C decreased from vegetation control the loss of fine root biomass was as high as 49% (Shan et al., 2001). Understory vegetation has been shown to make up 27-80% of belowground root biomass in temperate and subtropical pine forests accounting for a large portion of belowground C that will be lost via vegetation control (Gonzalez et al., 2013; Helmisaari et al., 2007; Jiang et al., 2018). Additionally, the loss of belowground biomass from vegetation control could result in greater N leaching due to increased presence of inorganic N. Several studies have found that vegetation control has increased mineralized N while also enhancing nitrification which produces leachable NO_3 (Gurlevik et al.,

2004; Munson & Timmer, 1995). Fertilization is hypothesized to increase soil C and N over time as microbial biomass as well as enzymatic activity will be suppressed by N inputs. Gains in soil C and N from fertilization are linked to changes in soil microbial activity and composition. Long-term fertilization has decreased soil respiration rates by 16-41% which is an indicator that decomposition has slowed (Bowden et al., 2004; Phillips & Fahey, 2007; Wallenstein et al., 2006). Microbial biomass decreased in forest soil from fertilization by ~40%, which is correlated with decreased respiration (Demoling et al. 2008; Wallenstein et al., 2006). The reduction in microbial biomass in forest soils from fertilization has led to lower levels of oxidase enzymes which are responsible for the decomposition of recalcitrant C compounds which causes reduced respiration (Billings & Ziegler, 2008; Jian et al., 2016; Saiya-Cork et al., 2002). The response of the soil microbial community to fertilization is correlated with decreases in belowground biomass investment by trees in the form of fine roots and mycorrhizal associations (Carrara et al., 2018; Phillips & Fahey, 2007). By improving our current understanding of soil C and N responses to plantation management it will be possible to improve practices to better soil health and therefore long-term productivity.

3.3.Materials & Methods

3.3.1. Study Sites

The soils analyzed in this study were taken from five different installations in Georgia that are part of CAPPS. The locations include Whitehall Forest in Athens; B.F. Grant Forest in Eatonton (two installations), Tifton Research Farm in Tifton; and Dixon State Forest in Waycross (only Wet installation). These sites span three physiographic regions that represent much of the southeastern United States pine growing region: Piedmont, upper Atlantic coastal

plain, and lower Atlantic coastal plain. Piedmont sites in Athens and Eatonton are dominated by Kanhapludults mainly the Cecil series. Soils in the piedmont tend to be deep and well-drained formed in granite and gneiss with higher clay content in the subsurface and with underlying saprolite. The upper coastal plain site in Tifton is dominated by Kandiudults such as the Tifton series. Soils in the upper coastal plain are deep and well-drained formed from marine deposits with high sand content, around 85% in the surface, as well inclusions of ironstone and plinthite. The lower coastal plain site in Waycross is dominated by Paleaquults including the Pelham series with some inclusions of Alaquods like the Mascotte series. Soils in this part of the lower coastal plain are poorly drained with sandier textures formed from marine deposits with subsurface gleying. These soils share similarities with upper coastal plain soils but are poorly drained.

Stands at each site were hand planted with loblolly pine (*Pinus taeda*) seedlings on cutover forest land at a density of 1680 trees ha^{-1} between 1987 to 1995. Athens and Eatonton were flat planted with 1-0 half-sib, open pollinated seedlings from family 10-25 , and Tifton and Waycross sites were bedded and planted with half-sib seedlings from family 7-56. Each installation contains multiple blocks (complete and incomplete) with four treatments. Treatments include a control (C), fertilized (F), understory vegetation control (previously referred to as herbicide) (H) and fertilized with understory vegetation control (HF). Athens and Tifton both have 2 complete and 2 incomplete blocks (C and H only). Eatonton has a total of 10 complete blocks across two installations. Waycross consists of 6 complete blocks. The total number of individual plots across the whole study is 88. One plot from Athens and 3 from Eatonton were dropped from the study due to excessive understory competition and pine mortality with fertilization or pine beetle outbreaks.

Fertilization treatment consisted of broadcast NPK application via cyclone spreader with rates of 69 kg N/ha, 56 kg P/ha, and 56 kg K/ha applied annually as DAP, NH_4NO_3, and KCl in the first two growing seasons. In growing seasons 3-9, 57 kg N/ha was added annually in the form of NH_4NO_3. Growing season 10 involved the addition of 114 kg N/ha and 28 kg P/ha as NH_4NO_3 and Super Triple Phosphate. This was followed up by the addition 113 kg N/ha, 25 kg P/ha, and 47 kg K/ha as NH_4NO_3 and super rainbow (10-10-10) in growing season 11. Lastly, there was biannual application of 114 kg N/ha as NH_4NO_3 from growing seasons 12 to 28. Understory vegetation control treatments were achieved through broadcast application of sulfometuron methyl in early spring for the first three growing seasons and spot treatment with glyphosate in mid-summer every year as needed for the full rotation. A block from two installations (Eatonton and Waycross) was also thinned around mid-rotation.

3.3.2. Mineral Soil and Forest Floor Sampling

The study utilizes a twenty-year resampling of mineral soil and forest floor that were first sampled in 2001 at stand ages 6-14 and then again in 2021 at stand age 26-34. Stand ages varied within a given site to account for differences in planting year conditions such as precipitation or temperature. Forest floor samples were collected at eight randomly selected locations within a plot. At each location an in-row and between row pair of samples was collected totaling 16 samples per plot. Paired sampling was especially necessary to account for the effect of bedding employed in Tifton and Waycross. A 1225 cm^2 quadrat was used to harvest forest floors down to bare soil. After forest floor collection, mineral soil at three depths: 0-10, 10-30, and 30-50 cm was collected with a 2-cm diameter punch tube. Forest floor samples were dried at 65°C until a constant weight was reached and then ground through a Wiley mill using a 1 mm mesh screen.

Mineral soil samples were air-dried and passed through a 2 mm sieve. A subsample of forest floor was burned at 400ºC to correct for the mineral ash content (Schulte & Hopkins, 1996).

Bulk density was determined using the core method (Blake,1986). Cores were taken from two random locations in each plot with one in-row and one between-row at depths of 0-7.5, 15-22.5, and 35-42.5 cm utilizing a 7.5 cm diameter and 7.5 cm deep core and a 5 kg slide hammer. Each core was dried at 105°C until constant weight. Coarse fragments and roots were removed from each core with volume and weight measured to remove the effect of non-soil components on bulk density.

3.3.3. Carbon & Nitrogen Analyses

Mineral soil and forest floor samples as prepared above were further ground using a ball mill grinder to a fine, homogenized powder. Approximately 100 mg of ground mineral soil was then weighed and placed into 10x9 mm tin capsules while ~10 mg of ground forest floor was weighed and placed into 5x9 mm tin capsules. Encapsulated samples were run on a Flash 2000 NC Soil Analyzer, which utilizes a dry combustion method for total carbon and nitrogen concentration determination (Nelson & Sommers, 1996). Standard reference materials ERA CRM 542 (soil) and NBS SRM 1575 (pine needles) were used for quality assurance and had an error of 1.8 and 9.6 % absolute across both materials for C and N, respectively. In addition, a subsample of archived mineral soil and forest floor samples collected in 2001 were reanalyzed in 2021 using the above method to determine any analytical biases between the analytical years. Forest floor (Figure 3.1) and mineral soil (Figure 3.2) analysis in 2021 and 2001 were highly correlated (R^2 = 0.94-0.98) with a near perfect linear relationship between analytical years. Total carbon and nitrogen content (i.e., kg/ha) for mineral soil and forest floor were converted to per

area basis for all samples with soil bulk density and forest floor weight that were measured on the plot level. Total C and N content for soil and forest floor could not be calculated for 2001 Athens and 2001 Tifton as they do not have bulk density or forest floor data. As such, these sites were included in analyses of treatment effects in 2021 but not changes with sampling year.

Aboveground biomass estimates were considered for ecosystem budgets. Aboveground biomass estimates and nutrient contents were derived from Zhao (2022) and Albaugh (2008) where individual trees were measured for each plot in CAPPS (Table 3.1). Tree biomass estimations were done on a component basis (e.g., stem, bark, branch, and foliage). Note that belowground estimates are not included due to difficulty of collecting root data. C content of each biomass component was calculated with the assumption that biomass is 50% C. N content of each component was calculated with the average concentrations reported in Albaugh (2008). Total C and N content in biomass was estimated by summing all components across each plot. Tree mortality was also measured by Zhao (2022) providing a living and dead biomass pool. Total ecosystem pools were then constructed by combining living biomass, dead biomass, forest floor, and mineral soil (0-50 cm) estimates. Ecosystem budgets in this study underestimate the total C and N pools as belowground estimates were not included.

3.3.4. Statistical Analysis

The study was designed as a randomized, incomplete block with a repeated measure over time and depth. Data from this study was analyzed with a Type-III ANOVA using a linear mixed-effects model in R (Bates et al., 2015; R Core Team, 2022). Fertilizer treatment, vegetation control treatment, location, depth, and sampling year were assigned as fixed effects. Interactions between fertilizer treatment, vegetation control treatment, depths, and sampling year

were also assessed in this model. Block and plot ID were treated as random effects. Planting year was left out of the model as it was demonstrated to not influence the variance components of the model. Note location was not included in the interactions due to issues with model complexity and sufficient sample size. Interactions with location and treatments were assessed by running the model for individual sites. The results of these individual site models were only mentioned if their effects differed between locations. Significance was determined using an alpha value of 0.10. Soil and forest floor data were natural log transformed prior to analysis, as untransformed data was significantly different from normal based on a Shapiro-Wilkes test (P-value < 0.05). Back-transformed means are presented in the results while the original log-transformed means and standard deviations can be found in tables and figures.

3.4. Results

3.4.1. Forest Floor Mass

The average forest floor mass for CAPPS was 28,032 kg/ha across treatments and sampling years (Table 3.2). The minimum amount of forest floor was 8,285 kg/ha from 2001 control plots in Eatonton while the maximum forest floor was 106,603 kg/ha from 2021 fertilizer plots in Waycross. Forest floor mass varied significantly between locations ($p = <0.001$). Waycross and Tifton had the greatest forest floor masses on average at 39,816 and 40,090 kg/ha. Athens and Eatonton had slightly less mass at 35,936 and 20,213 kg/ha.

Both fertilization and vegetation control significantly increased forest floor mass (Table 3.3, Figure 3.3 and 3.4). In 2021, fertilized plots had an average mass of 58,219 kg/ha compared to 37,718 kg/ha in unfertilized plots ($p = <0.001$). Vegetation control plots had 49,346 kg/ha of mass compared to the 41,295 kg/ha found in non-vegetation control plots. Significant increases

in forest floor mass were found across all treatments over time going from 14,259 kg/ha in 2001 to 45,429 kg/ha in 2021 ($p = <0.001$; Figure 3.5). The effects of fertilization on forest floor mass were consistent through time. The same is true for vegetation control as its effect remained constant over time.

3.4.2. Forest Floor C & N

Average forest floor C concentration was 502 g/kg varying from 461 g/kg in 2021 vegetation control plots in Eatonton to 606 g/kg in 2021 vegetation control + fertilizer plots in Athens, although these changes are presumed to vary mostly as a function of mineral contamination (Table 3.2). There was no significant difference in forest floor C concentrations between locations. The average N concentration of forest floor across treatments and sampling year was 13.9 g/kg with a range from 9.0 g/kg in 2001 vegetation control plots in Waycross to 18.2 g/kg in 2021 fertilizer plots in Athens. N concentration of forest floor was significantly different across locations. Athens, Eatonton, and Tifton had similar N concentrations at 14.1, 14.9, and 16.4 g/kg, respectively. Waycross, on the other hand, had lower forest floor N concentration averaging 11.8 g/kg across treatments and sampling year.

Neither fertilization nor vegetation control consistently impacted forest floor C concentrations across the pooled data. Effects of sampling year for forest floor C concentrations were also not significant when evaluated across all locations. Any observed differences in forest floor C concentrations appear to be a function of mineral contamination during sampling.

For forest floor N concentration fertilization significantly increased forest floor N concentration while vegetation control significantly decreased N concentration (Table 3.3;

Figure 3.6 and 3.7). Forest floor N concentrations in 2021 went from 14.3 g/kg in unfertilized plots to 15.6 g/kg in fertilized plots ($p = <0.001$). On the other hand, vegetation control decreased forest floor N concentrations from 15.3 g/kg in plots without vegetation control to 14.4 g/kg in plots with vegetation control ($p = <0.001$). Forest floor N concentrations significantly increased from 12.6 g/kg in 2001 to 14.8 g/kg in 2021 ($p = <0.001$) across treatments (Figure 3.8). The effect of fertilization on forest floor N concentration lessened over time. The increase in forest floor N from fertilization went from a difference of 2.6 g/kg in 2001 to 1.3 g/kg in 2021 ($p = 0.049$). The decrease of forest floor N concentration from vegetation control also lessened over time. The decrease in forest floor N concentration from vegetation control went from a difference of -2.1 g/kg in 2001 to -0.9 g/kg in 2021 ($p = 0.075$).

3.4.3. Bulk Density

The average bulk density across locations, depths, treatments, and sampling year was 1.45 g/cm^3 (Table 3.4). The minimum bulk density across locations was 1.11 g/cm^3 in 0-10 cm of the 2021 vegetation control + fertilizer plots in Athens with a maximum of 1.77 g/cm^3 in 10-30 cm of the 2021 fertilizer plots in Tifton. Bulk density was significantly different across locations ($p = <0.001$). Coastal plain soils in Tifton and Waycross exhibited the highest bulk density across all depths and treatments with an average of 1.55 and 1.52 g/cm^3, respectively. Conversely, piedmont soils in Athens and Eatonton displayed the lowest bulk density across depths and treatments at 1.33 and 1.41 g/cm^3, respectively.

Fertilization decreased bulk density while vegetation control had no significant effect across all locations (Table 3.5; Figure 3.9 and 3.10). The average bulk density for fertilized plots

in 2021 was 1.47 g/cm^3 compared to 1.45 g/cm^3 in unfertilized plots (p = 0.061). Vegetation control had no significant effect on bulk density across sampling years. Bulk density did significantly increase over time going from 1.43 g/cm^3 in 2001 to 1.46 g/cm^3 in 2021 when averaged across treatments and depths (p = <0.001; Figure 3.11). The interaction between fertilization and depth was significant in 2021 (p = 0.077). Changes in bulk density from fertilization went from -0.04 g/cm^3 in 0-10 cm to 0.02 g/cm^3 in 30-50 cm. At the same time, the effect of vegetation control on bulk density significantly decreased with depth (p = 0.018) even though vegetation control did not have an overall effect. The effect of vegetation control on bulk density went from 0.05 g/cm^3 at 0-10 cm to -0.01 g/cm^3 at 30-50 cm across locations. Increases in bulk density over time were significantly higher at deeper depths (p = <0.001). Average changes in bulk density over time went from a decrease of -0.02 g/cm^3 at 0-10 cm to an increase of 0.09 g/cm^3 at 30-50 cm. Fertilization and vegetation control treatment effects remained consistent through time.

3.4.4. Soil Carbon & Nitrogen

The average soil C concentration for all locations, sampling years, depths, and treatments was 6.7 g/kg (Table 3.4). The soil C concentration ranged from 2.4 g/kg in 30-50 cm of 2021 vegetation control plots in Tifton to 18.7 g/kg in 0-10 cm of 2001 fertilizer plots in Waycross. Soil C concentration significantly differed across locations (p = <0.001). Waycross had the highest average soil C concentration across depths and sampling year with an average of 8.4 g/kg while Tifton had the lowest at 5.2 g/kg. Eatonton and Athens had similar soil C concentration at 6.9 and 6.1 g/kg, respectively. The average soil N across locations, sampling year, treatments, and depths was 0.36 g/kg. The range of average soil N across locations was 0.05 g/kg in 30-50

cm of 2021 fertilizer plots to 1.27 g/kg in 0-10 cm of 2021 fertilizer plots. Soil N varied significantly between locations ($p = 0.06$). Tifton and Waycross had the lowest soil N concentration when averaged across sampling years, treatments, and depth at 0.27 and 0.30 g/kg, respectively. N concentration of soil in Athens and Eatonton was slightly higher at 0.52 and 0.39 g/kg, respectively.

Soil C concentration decreased with vegetation control and increased with fertilization in 2021 (Table 3.5; Figure 3.12 and 3.13). Plots that received vegetation control had, on average, soil C concentration of 6.2 g/kg while plots without vegetation control had an average of 6.8 g/kg ($p = 0.012$). In contrast, fertilization significantly increased soil C from 6.2 g/kg in unfertilized to 6.9 g/kg in fertilized plots across both sampling years ($p = 0.064$). There was a significant decrease in soil C concentration over time regardless of treatment going from 7.1 g/kg in 2001 to 6.5 g/kg in 2021 ($p = 0.008$: Figure 3.14). The fertilization effect on soil C concentration did not significantly vary across sampling years or depths. Similarly, the vegetation control effect on soil C concentration did not differ by sampling year or depth.

Soil N concentration significantly increased with fertilization and decreased with vegetation control in 2021 (Table 3.5; Figure 3.15 and 3.16). Fertilized plots had an average soil N concentration of 0.36 g/kg versus 0.34 g/kg in unfertilized plots ($p = 0.086$). Vegetation control significantly decreased soil N concentration from 0.37 g/kg in non-vegetation control plots to 0.33 g/kg in vegetation control plots ($p = 0.078$). Soil N concentration did not change consistently from 2001 to 2021 across all locations (Figure 3.17). Rather, soil N concentration increased in Eatonton from 0.36 g/kg in 2001 to 0.43 g/kg in 2021 ($p = 0.009$) and decreased in Waycross from 0.38 g/kg in 2001 to 0.24 g/kg in 2021 ($p = <0.001$). The effects of fertilization on soil N concentration did not significantly change across depths or sampling years. The

vegetation control effect on soil N concentrations was also unchanged by depth or sampling year with the only exception being vegetation control plots in Tifton that had increasing soil N concentration at 30-50 cm while decreasing soil N concentration at 0-10 and 10-30 cm ($p = 0.089$). Changes in soil N concentration over time were constant through depth.

3.5.Discussion

The above responses in forest floor and mineral soil mass, bulk density, and C and N concentration to pine plantation management with fertilization and vegetation control combine to quantify gains and losses of C and N from the soil profile (Table 3.6 and 3.7; Figure 3.18 to 3.23). In this regard, complete understory vegetation control consistently reduced C in the mineral soil through the end of the rotation (Table 3.6; Figure 3.18). By the end of rotation, 4.1 Mg-C/ha or 9% of mineral soil C was lost from vegetation control. Studies that maintained vegetation control treatments past early rotation also found similar losses of mineral soil C contents ranging from 6 to 23% from early- to end-of-rotation (Maillard et al., 2010; Sarkhot et al., 2007; Shan et al., 2001; Vogel et al., 2011). Increases in forest floor from vegetation control did not result in increased mineral soil C content or reduced losses as might be expected. Forest floor C pools increased by 4.0 Mg-C/ha or 19% at the end of the rotation. The lack of a positive correlation between mineral soil C content and forest floor C content changes is consistent with findings from litter manipulation studies, which have shown that the soil C content does not respond to additions or exclusion of forest floor even over decadal time scales (Huang & Spohn, 2015; Man et al.,2022; Wang et al., 2017). Vegetation control also increased C accumulation in living aboveground biomass by 12.7 Mg-C/ha and dead aboveground biomass by 8.6 Mg-C/ha at end-of-rotation (Zhao et al., 2022). The lack of soil C content response to increased aboveground

biomass from vegetation control indicates a decoupling between aboveground and belowground carbon cycling processes. Changes in soil C contents appear more closely linked to changes in belowground biomass production (Lajtha et al., 2018 ;Shan et al., 2001). If this is the case, soil C loss from vegetation control may be diminished or eliminated by allowing the return of understory vegetation earlier in rotation, which is a more common operational practice (Fox et al, 2007).

The use of vegetation control resulted in losses of mineral soil N at early- and end-of-rotation (Table 3.7; Figure 3.19). In 2021, stands lost 357 kg-N/ha or 13% from vegetation control. Mineral soil N losses reported in this study are larger than in other studies that applied vegetation control throughout rotation, which found a 0 to 6% change from treatment (Sarkhot et al., 2007; Vogel et al., 2011). A potential explanation for this discrepancy is that other longer-term vegetation control studies were done on poorly-drained soils, which exhibit lower potential N losses, possibly from reduced mineralization and nitrification (Ullah et al., 2009). Reductions in mineralization and nitrification will lower concentrations of leachable NO_3^-. A portion of mineral soil N decline under vegetation control can be accounted for by uptake into forest floor and aboveground biomass. Forest floor accumulated an additional 38 kg-N/ha in 2021 under vegetation control. Living biomass accumulated an additional 29 kg-N/ha in 2021. This does not include the N accumulated in dead biomass which was estimated at 24 kg-N/ha in 2021. This means increases in tree productivity from vegetation control as measured by forest floor and biomass accounts for 90% of N content decline in the mineral soil. The remaining amount of N content decline under vegetation control may have resulted from N leaching lower in the soil profile or out of the soil system.

Fertilization led to only a slight accumulation of C in the mineral soil despite increases in the forest floor (Table 3.6; Figure 3.20). Stands across CAPPS gained 3.9 Mg-C/ha or 10% more mineral soil C content in 2021. The lack of soil C content change in response to fertilization was due to increases in soil C concentration being offset by lowered bulk density. The observed increases in mineral soil C content are similar to the 6% increases in Vogel (2011). Other fertilization studies in pine plantations found no effect on mineral soil C content, however, many of these studies lack persistent fertilization inputs through rotation (Johnson et al., 2003; Leggett et al., 2006; Mosier et al., 2019). Fertilization did result in a significant accumulation of C and N in forest floor similar to vegetation control. Forest floor accumulated 10.2 Mg-C/ha or 54% more forest floor C under fertilization in 2021. However, living biomass C accumulation declined under fertilization in 2021 by 20.1 Mg-C/ha with a concurrent increase in dead biomass C of 47.8 Mg-C/ha. The reduction in C accumulation in living biomass was driven by high tree mortality in fertilized plots which were not thinned as would be common in operationally managed stands. We did not quantify coarse woody debris that would represent the dead biomass, but this component of carbon sequestration, which was evident on the forest floor, did not transform to increased mineral soil C.

Increased N inputs through fertilization did not translate to increased mineral soil N but increased N in the forest floor and aboveground biomass (Table 3.7; Figure 3.21). Measured mineral soil N content under fertilization was greater by 105 kg-N/ha or 4% in 2021. Increases in mineral soil N from fertilization of 6% were reported at the end of the rotation (age 26) when applied through age 17 (Vogel et al., 2011). Mineral soil N gains up to 20% were observed at mid-rotation in plantations that fertilized early in rotation (McFarlone et al., 2009). Other plantation studies found no mineral soil N response from early-rotation fertilization when

measured at mid-rotation (Johnson et al., 2003). Assuming all additional N gains are from fertilizer input, only 2% of the added 1790 kg-N/ha made it to mineral soil in 2021. A large portion of fertilizer N ended up in the forest floor and aboveground biomass. Forest floor accumulated 370 kg-N/ha or 69% more N under fertilization by 2021 accounting for 21% of added N. Living biomass lost 57 kg-N/ha less N from fertilization by 2021, which was due to increased tree mortality, but if dead biomass is considered then 150 kg-N/ha more N was accumulated in dead biomass over the whole rotation. This means net growth of biomass accounts for 5% of added N. Therefore, the resulting increase in ecosystem N in 2021 of 568 kg-N/ha can only account for 31% of added N from fertilization. The remaining amount of N could have been lost to leaching or other N transformations such as denitrification though levels are quite low in forests due to acidity. This lack of N retention contrasts with earlier results for the Waycross location (Will et al 2006). At mid-rotation (age 13), Will et al (2006) reported a 90% retention of fertilizer N inputs. Recent work of Zhao et al (2022) that also includes the same CAPPS stands quantified annual net primary productivity (ANPP) over the rotation. Zhao et al (2022) showed a decline in ANPP after mid-rotation concluding that ANPP was not nutrient limited and may, in fact, have been N saturated with the high inputs of this study. The low recovery rate of N from fertilization is a concern both economically as N not recovered is revenue lost and ecologically as leached N has the potential to reduce water quality in the surrounding area. The combined results from Will et al (2006), Zhao et al (2022), and the current study suggest N dynamics through mid-rotation may well differ from those in the later decades of the rotation.

Mineral soil C contents remained largely stable from early to end of the rotation in pine plantations regardless of management (Table 3.6; Figure 3.22). Control plots across CAPPS saw

a loss in soil C of 2.4 Mg-C/ha or 0.1 Mg-C/ha/yr through a 50 cm depth between 2001 and 2021. Losses of soil C were overshadowed by the 12.6 Mg-C/ha or 0.6 Mg-C/ha/yr gain in forest floor C and 128.9 Mg-C/ha or 6.4 Mg-C/ha/yr in aboveground biomass C (both living and dead).This is in agreement with other long-term pine plantations studies which have found either no change or a slight increases in soil C of 1.5 Mg-C/ha or 0.04 Mg-C/ha/year over 14-40 years (Johnson et al., 2003; Markewtiz et al., 2002; Richter et al., 1999; Scott et al, 2014). Changes in soil C in these studies had a small effect on total ecosystem C, which was similar to CAPPS where significant C gains were mostly in forest floor and biomass.

Soil N did not change consistently across locations from early to end of the rotation in pine plantations. Control plots in Eatonton saw a significant gain in soil N of 643 kg-N/ha or 32 kg-N/ha/yr over the twenty-year period (Table 3.7; Figure 3.23). In those same plots 204 kg-N/ha or 10 kg-N/ha/yr of N was accumulated in forest floor with an additional 348 kg-N/ha or 17 kg-N/ha/yr in aboveground biomass, which includes both living and dead biomass, from 2001 to 2021. After accounting for potential external inputs from atmospheric deposition (Environmental Protection Agency 2023) of 8 kg-N/ha/yr (168 kg-N/ha) and biological N fixation (Schaefer et al., 2007) of 0.2 kg/ha/yr (4 kg/ha) the ecosystem gained 1023 kg-N/ha or 51 kg-N/ha/yr of unexplained N. Conversely, Waycross control plots had significant soil N losses of 889 kg-N/ha or 44 kg-N/ha/yr of soil N over the rotation. At the same time, 800 kg-N/ha or 40 kg-N/ha/yr of N went to forest floor with an additional 307 kg-N/ha or 15 kg-N/ha/yr in aboveground biomass. This means Waycross sites gained 386 kg-N/ha or 10 kg-N/ha/yr of N from an unknown source after accounting for atmospheric deposition and biological N fixation. Gains in ecosystem N over time have been reported in pine plantations between age 18 and 40 years old, although accretion rates were lower on average ranging from 5.9 to 31.1 kg-N/ha/yr (Johnson et al., 2003; Richter et

al.,2000). These same studies found either no change or a decrease in soil N of 20.6 kg-N/ha/yr. Changes in ecosystem N at these sites were accounted for by atmospheric deposition and N fixation, which was not the case for CAPPS. Significant accumulations of N were seen in mixed-hardwood stands that were adjacent to stands reported in Johnson (2003), which gained 43 to 70 kg-N/ha/yr of ecosystem N and 70 kg-N/ha/yr of soil N at stand age 15 years with no clear explanation (Johnson & Todd, 1998). Unexplained increases in soil N found in Johnson (1998) were also reported in 2013 after 33 years with an average increase of 51.3 kg-N/ha/yr (Johnson et al., 2016). Claims of "occult N" have previously been questioned due to poor experimental design, mainly focusing on quality assurance measures (Binkley et al., 2000). This study put forth great effort to avoid such concerns by utilizing archived samples, similar analytical techniques, and having an original investigator present from previous CAPPS studies. Future research should not discount these N findings, particularly given recent work (Zhao et al 2022) suggesting N limitation may decline late in pine rotations.

3.6.Conclusion

Pine plantation management has the potential to improve or degrade soil C and N pools depending on practices used through rotation. Vegetation control can result in soil C and N losses that remain significant through a whole rotation. Conversely, fertilization has resulted in a slight increase in soil C with no effect on N and in this case in the absence of thinning accelerated self-thinning and mortality of trees. Soil types also seem to play a role in how soil C and N change in response to management and time. The results of this study caution against intensifying vegetation control practices to the extent of complete vegetation control throughout the rotation or fertilizing with N biennially in pine plantations to improve soil C and N status or

site productivity. Clarifying the source of unexplained N accumulation (i.e., occult N) over three decades of pine plantation growth remains a worthy challenge.

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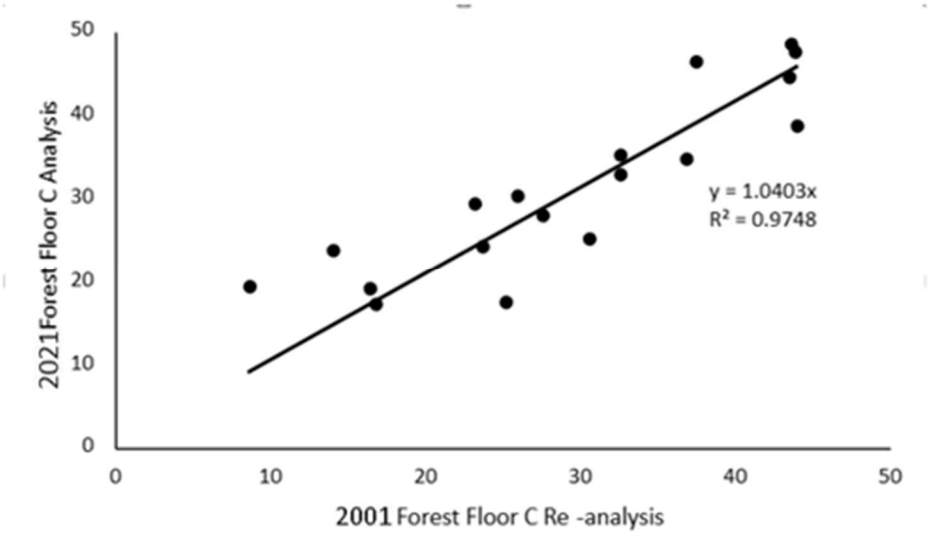
(a)



(b)

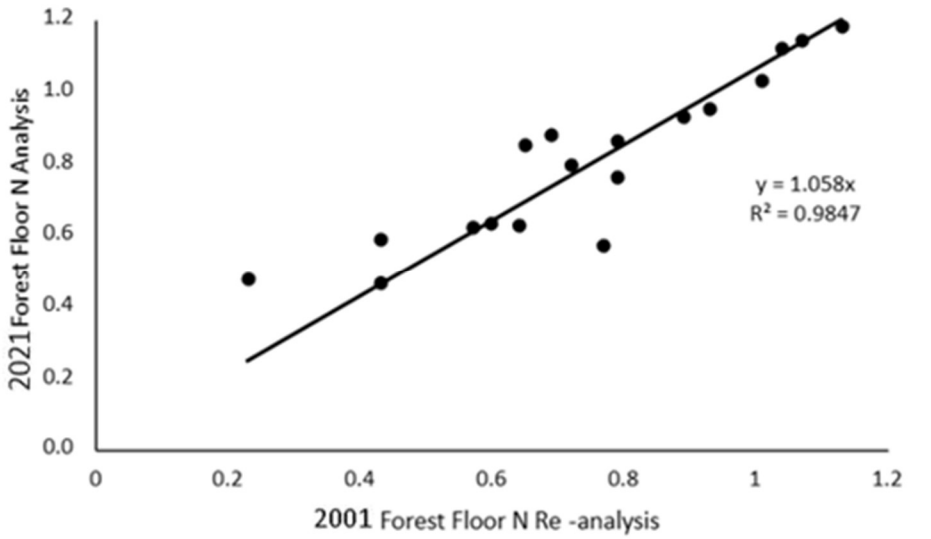
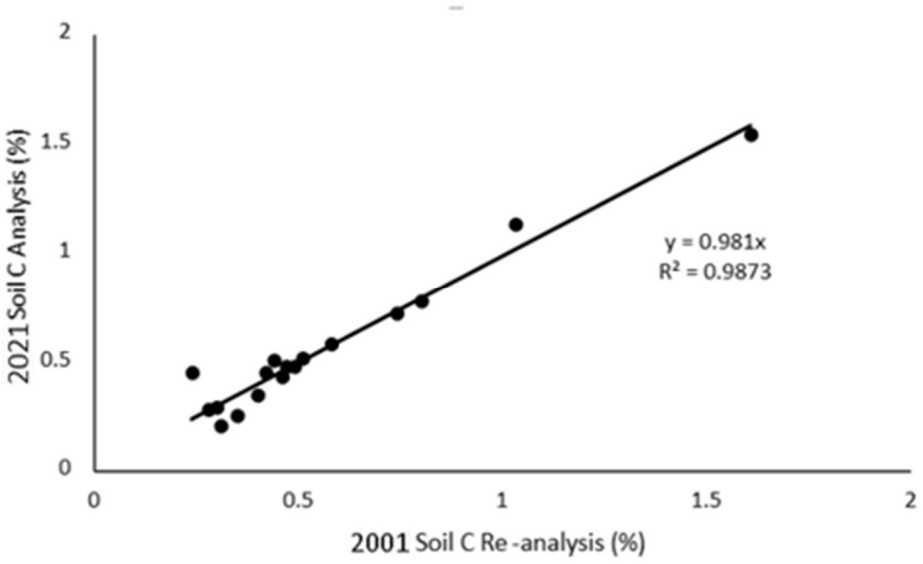


Figure 3.1 Comparison of 2001 analysis and 2021 re-analysis of a) forest floor C and b) forest floor N of a sub-sample of 2001 soil samples (n = 10).

(a)



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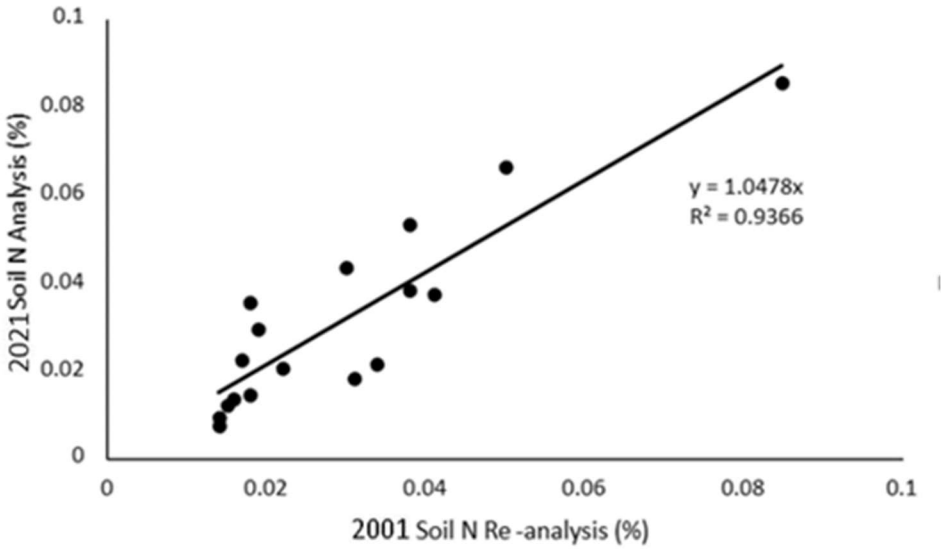

Figure 3.2 Comparison of 2001 analysis and 2021 re-analysis of a) soil C and b) soil N of a sub-sample of 2001 soil samples (n = 17).

Table 3.1 Mean tree biomass (Mg/ha (standard deviation)) and nutrient concentration estimates by component averaged across locations and treatments. Includes living and dead biomass pools for 2001 and 2021.

Component	Living Biomass[1]		Dead Biomass[1]		C	N[2]
	2001	2021	2001	2021	(%)	(%)
Stem	71 (45)	246 (56)	2 (4)	70 (50)	50	0.09
Bark	8 (4)	18 (4)	0 (0)	6 (4)	50	0.29
Branch	14 (6)	25 (5)	0 (1)	8 (6)	50	0.38
Foliage	5 (1)	5 (1)	0 (0)	2 (1)	50	1.08

[1] Biomass estimates for each component taken from Zhao (2022)

[2] Nitrogen concentration estimates for *Pinus taeda* taken from Albaugh (2008)

Table 3.2 Summary of forest floors mass, C concentrations, and N concentrations for CAPPS plots in Georgia USA. Included are back-transformed means and log-transformed means (standard deviation).

Location	Year	Trt	Mass (kg/ha)	log(Mass)	C (%)	log(C)	N (%)	log(N)
Athens	2021	C	27884	10.24 (0.23)	46.8	3.85 (0.33)	1.41	0.34 (0.08)
		F	25921	10.16 (NA)	51.1	3.93 (NA)	1.82	0.6 (NA)
		H	41954	10.64 (0.23)	54.3	4 (0.15)	1.28	0.25 (0.15)
		HF	51561	10.85 (0.35)	60.6	4.1 (0.21)	1.52	0.42 (0.01)
Eatonton	2001	C	8284	9.02 (0.51)	50.0	3.91 (0.00)	1.45	0.37 (0.22)
		F	13347	9.5 (0.75)	50.0	3.91 (0.00)	1.78	0.58 (0.14)
		H	12749	9.45 (0.54)	50.0	3.91 (0.00)	1.23	0.21 (0.19)
		HF	15441	9.64 (0.81)	50.0	3.91 (0.00)	1.40	0.34 (0.19)
	2021	C	21876	9.99 (0.26)	48.2	3.88 (0.09)	1.48	0.39 (0.14)
		F	37937	10.54 (0.31)	50.4	3.92 (0.1)	1.73	0.55 (0.13)
		H	28513	10.26 (0.45)	46.1	3.83 (0.1)	1.49	0.4 (0.1)
		HF	53001	10.88 (0.1)	50.8	3.93 (0.06)	1.57	0.45 (0.11)
Tifton	2021	C	32573	10.39 (0.19)	52.2	3.96 (0.11)	1.76	0.57 (0.03)
		F	41840	10.64 (0.3)	50.7	3.93 (0.004)	1.81	0.59 (0)
		H	42569	10.66 (0.32)	48.3	3.88 (0.04)	1.43	0.36 (0.13)
		HF	51606	10.85 (0.35)	48.9	3.89 (0.03)	1.71	0.54 (0.11)
Waycross	2001	C	14299	9.57 (0.29)	49.9	3.91 (0.00)	0.96	-0.04 (0.16)
		F	21285	9.97 (0.65)	50.0	3.91 (0.00)	1.29	0.25 (0.09)
		H	13646	9.52 (0.47)	50.0	3.91 (0.00)	0.90	-0.11 (0.15)
		HF	25409	10.14 (0.6)	50.0	3.91 (0.00)	1.13	0.12 (0.12)
	2021	C	76986	11.25 (0.26)	53.4	3.98 (0.2)	1.22	0.2 (0.26)
		F	106602	11.58 (0.25)	54.5	4 (0.14)	1.41	0.34 (0.13)
		H	77397	11.26 (0.18)	54.9	4.01 (0.15)	1.36	0.31 (0.14)
		HF	94868	11.46 (0.26)	47.7	3.87 (0.1)	1.33	0.28 (0.2)

Table 3.3 p-Values for ANOVA of forest floor mass, C concentrations, and N concentrations.

Effect	df	Mass	C%	N%
Fert	1	<0.001	0.451	<0.001
Herb	1	0.008	0.700	<0.001
Year	1	<0.001	0.843	<0.001
Location	3	0.001	0.133	<0.001
Fert:Herb	1	0.750	0.705	0.170
Fert:Year	1	0.859	0.386	0.049
Herb:Year	1	0.919	0.715	0.075
Fert:Herb:Year	1	0.910	0.684	0.766

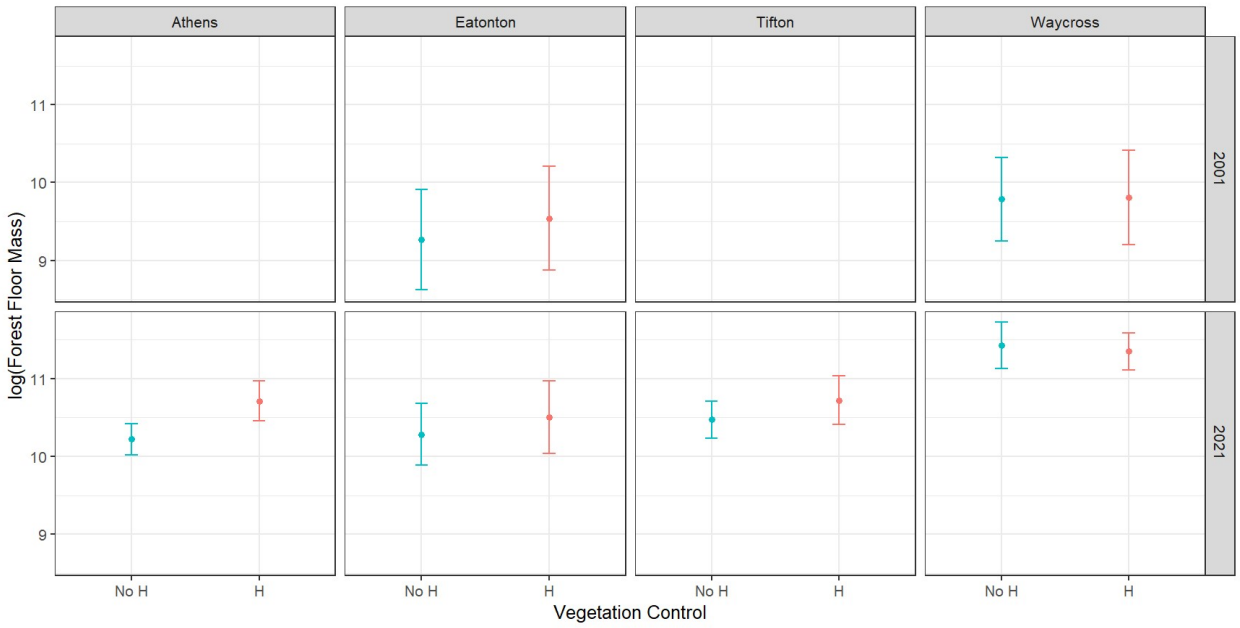

Figure 3.3 Comparison of log-transformed forest floor mass between plots with vegetation control treatment (H and HF) and without vegetation control treatment (C and F).

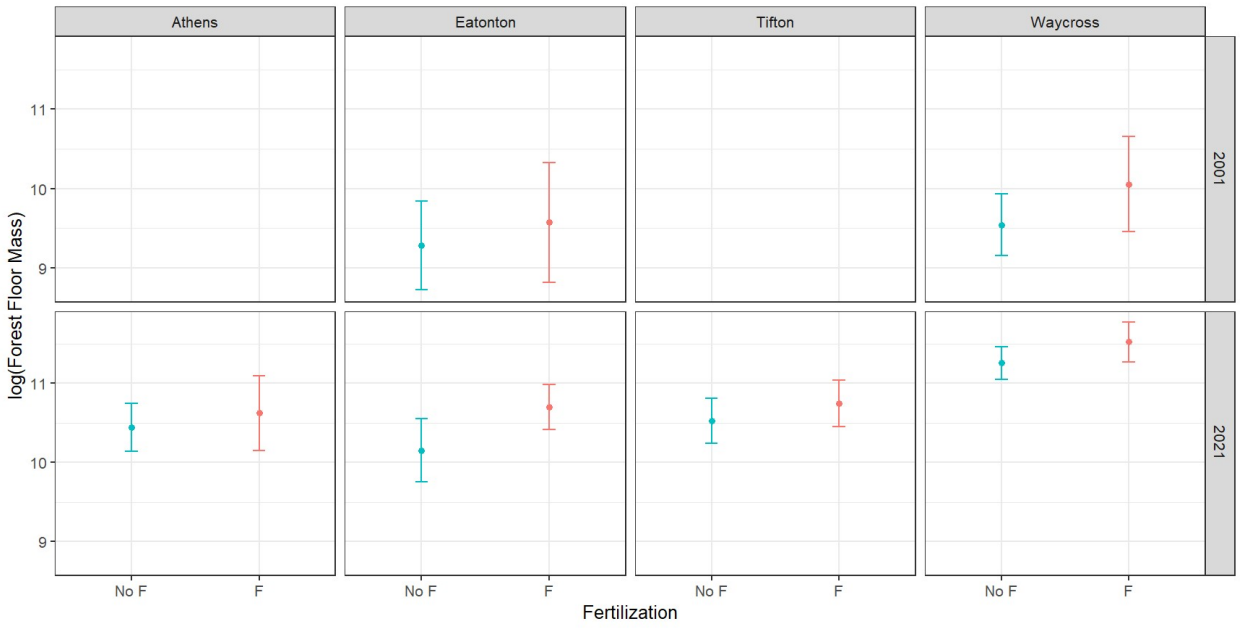

Figure 3.4 Comparison of log-transformed forest floor mass between plots with fertilization (F and HF) and without fertilization (C and H).

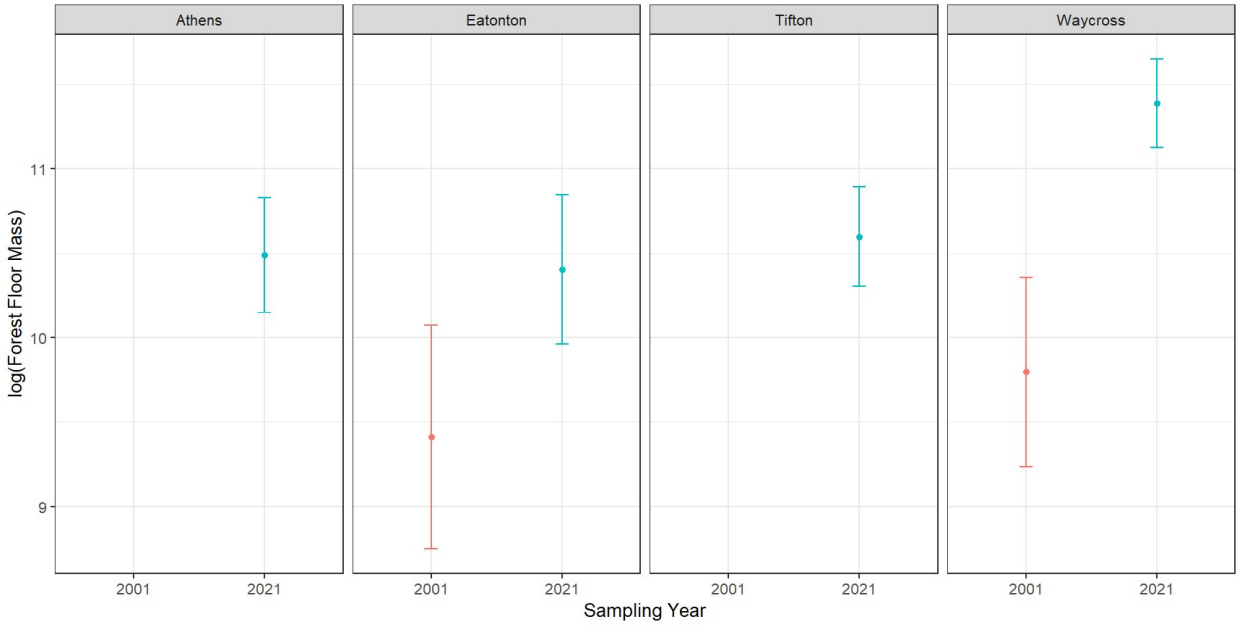

Figure 3.5 Comparison of log-transformed forest floor mass between sampling years (2001 vs 2021).

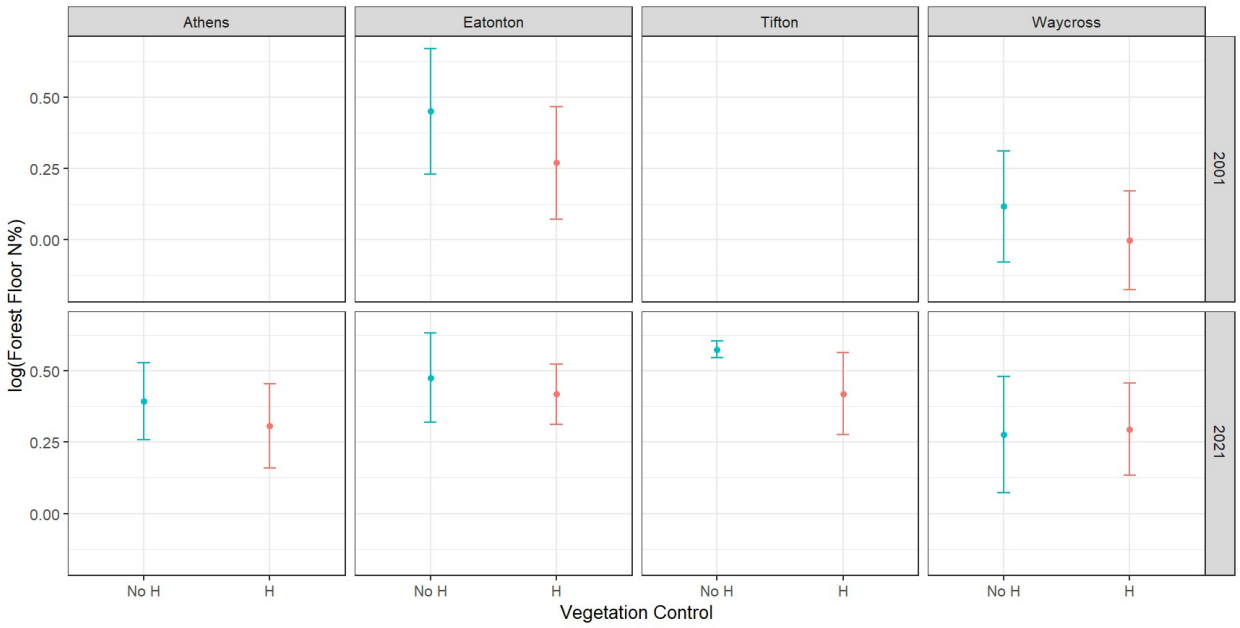

Figure 3.6 Comparison of log-transformed forest floor N concentrations between plots with vegetation control treatment (H and HF) and without vegetation control treatment (C and F).

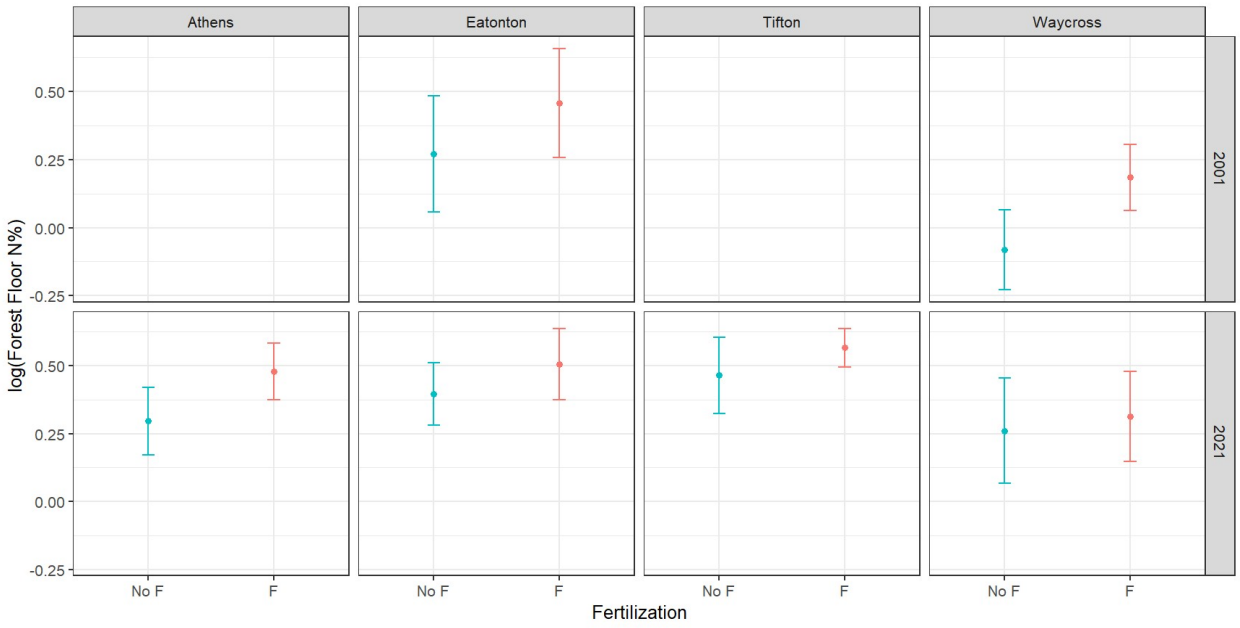

Figure 3.7 Comparison of log-transformed forest floor N concentrations between plots with fertilization (F and HF) and without fertilization (C and H).

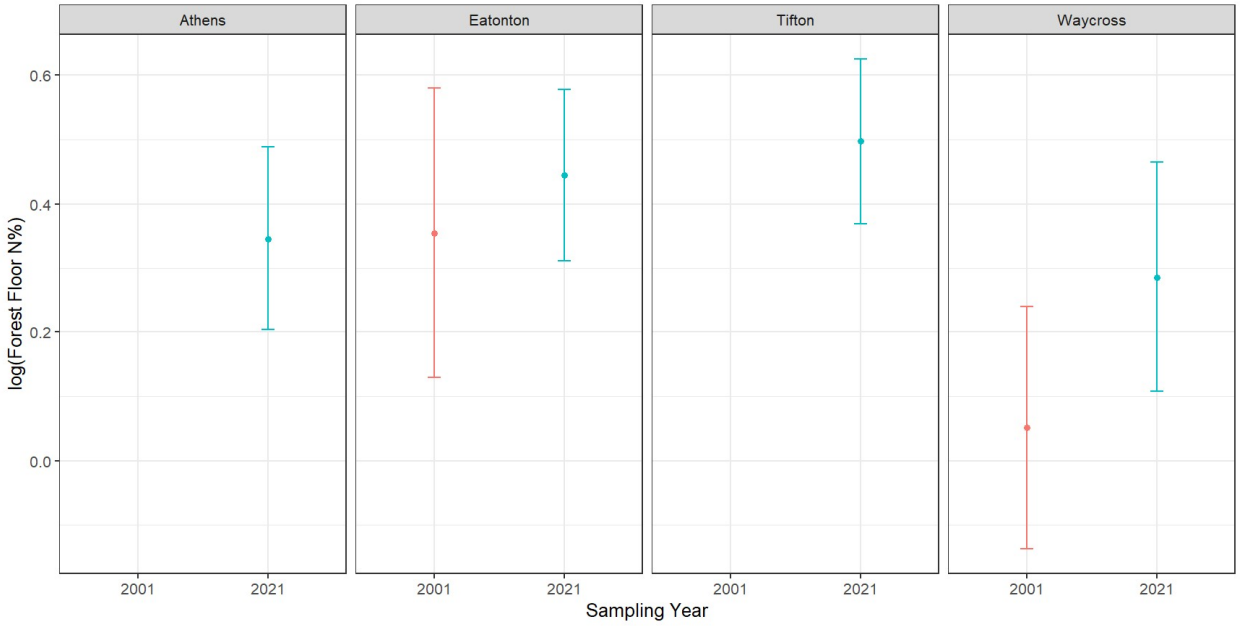

Figure 3.8 Comparison of log-transformed forest floor N concentrations between sampling years (2001 vs 2021).

Table 3.4 Summary of mineral soil bulk density, C concentrations, and N concentrations for CAPPS plots in Georgia USA. Included are back-transformed means and log-transformed means (standard deviation).

Location	Year	Trt	Depth (cm)	BD (g/cm3)	log(BD)	C (%)	log(C)	N (%)	log(N)
Athens	2021	C	0	1.24	0.21 (0.04)	1.66	0.5 (0.18)	0.10	-2.3 (0.09)
			10	1.44	0.36 (0.03)	0.65	-0.43 (0.32)	0.05	-3.09 (0.19)
			30	1.39	0.33 (0.06)	0.33	-1.11 (0.34)	0.03	-3.47 (0.16)
		F	0	1.16	0.14 (NA)	1.78	0.58 (NA)	0.11	-2.17 (NA)
			10	1.29	0.25 (NA)	0.66	-0.42 (NA)	0.05	-2.92 (NA)
			30	1.29	0.26 (NA)	0.34	-1.09 (NA)	0.04	-3.27 (NA)
		H	0	1.31	0.27 (0.03)	1.37	0.31 (0.15)	0.08	-2.5 (0.07)
			10	1.48	0.39 (0.05)	0.61	-0.5 (0.13)	0.04	-3.14 (0.18)
			30	1.36	0.31 (0.05)	0.29	-1.25 (0.12)	0.03	-3.52 (0.24)
		HF	0	1.11	0.1 (0.04)	1.66	0.51 (0.04)	0.10	-2.28 (0)
			10	1.22	0.2 (0.12)	0.74	-0.3 (0.08)	0.06	-2.89 (0.09)
			30	1.39	0.33 (0.02)	0.39	-0.93 (0.17)	0.04	-3.3 (0.21)
Eatonton	2001	C	0	1.36	0.31 (0.11)	1.56	0.44 (0.41)	0.06	-2.76 (0.46)
			10	1.44	0.37 (0.03)	0.67	-0.4 (0.58)	0.04	-3.3 (0.6)
			30	1.35	0.3 (0.02)	0.32	-1.13 (0.36)	0.02	-3.93 (0.31)
		F	0	1.32	0.28 (0.11)	1.49	0.4 (0.29)	0.08	-2.58 (0.35)
			10	1.45	0.37 (0.03)	0.64	-0.45 (0.26)	0.04	-3.26 (0.34)
			30	1.35	0.3 (0.02)	0.35	-1.04 (0.22)	0.02	-3.7 (0.27)
		H	0	1.39	0.33 (0.1)	1.25	0.22 (0.17)	0.06	-2.86 (0.25)
			10	1.44	0.37 (0.03)	0.53	-0.64 (0.31)	0.03	-3.49 (0.25)
			30	1.35	0.3 (0.02)	0.29	-1.25 (0.2)	0.02	-3.9 (0.29)
		HF	0	1.37	0.32 (0.06)	1.33	0.28 (0.2)	0.06	-2.74 (0.24)
			10	1.44	0.37 (0.03)	0.55	-0.61 (0.17)	0.03	-3.4 (0.14)
			30	1.35	0.3 (0.02)	0.28	-1.27 (0.12)	0.02	-3.82 (0.17)
	2021	C	0	1.35	0.3 (0.1)	1.55	0.44 (0.43)	0.09	-2.38 (0.35)
			10	1.59	0.46 (0.1)	0.56	-0.59 (0.45)	0.04	-3.2 (0.29)
			30	1.47	0.38 (0.11)	0.26	-1.33 (0.46)	0.02	-3.96 (0.84)
		F	0	1.28	0.24 (0.13)	1.45	0.37 (0.43)	0.13	-2.06 (0.56)
			10	1.49	0.4 (0.07)	0.57	-0.56 (0.2)	0.05	-3.09 (0.2)
			30	1.47	0.38 (0.1)	0.29	-1.24 (0.26)	0.03	-3.43 (0.56)
		H	0	1.38	0.32 (0.1)	1.38	0.32 (0.17)	0.08	-2.5 (0.15)
			10	1.44	0.36 (0.15)	0.49	-0.72 (0.21)	0.04	-3.2 (0.61)
			30	1.44	0.37 (0.04)	0.25	-1.38 (0.27)	0.02	-3.85 (0.65)
		HF	0	1.29	0.26 (0.08)	1.42	0.35 (0.21)	0.08	-2.52 (0.23)
			10	1.51	0.41 (0.07)	0.48	-0.73 (0.16)	0.03	-3.36 (0.19)

			30	1.48	0.39 (0.08)	0.27	-1.3 (0.28)	0.01	-4.27 (1.14)
Tifton	2021	C	0	1.28	0.25 (0.1)	0.97	-0.03 (0.26)	0.05	-2.93 (0.27)
			10	1.63	0.49 (0.05)	0.45	-0.8 (0.32)	0.03	-3.59 (0.28)
			30	1.63	0.49 (0.08)	0.29	-1.25 (0.16)	0.01	-4.58 (1.55)
		F	0	1.27	0.24 (0.07)	1.31	0.27 (0.31)	0.07	-2.67 (0.3)
			10	1.73	0.55 (0.01)	0.54	-0.62 (0.25)	0.03	-3.53 (0.17)
			30	1.77	0.57 (0.05)	0.38	-0.98 (0.01)	0.01	-5.28 (2.3)
		H	0	1.40	0.33 (0.04)	0.98	-0.02 (0.1)	0.05	-2.96 (0.1)
			10	1.66	0.51 (0.01)	0.39	-0.95 (0.23)	0.02	-3.78 (0.15)
			30	1.69	0.52 (0.08)	0.24	-1.43 (0.14)	0.02	-3.97 (0.13)
		HF	0	1.34	0.29 (0.09)	1.06	0.05 (0.37)	0.06	-2.9 (0.26)
			10	1.63	0.49 (0.02)	0.57	-0.57 (0.02)	0.03	-3.43 (0.15)
			30	1.75	0.56 (0.05)	0.36	-1.03 (0.02)	0.02	-3.7 (0.17)
Waycross	2001	C	0	1.34	0.29 (0.08)	1.71	0.54 (0.1)	0.08	-2.48 (0.2)
			10	1.57	0.45 (0.08)	0.73	-0.31 (0.31)	0.03	-3.43 (0.5)
			30	1.62	0.48 (0.03)	0.49	-0.72 (0.27)	0.02	-3.83 (0.22)
		F	0	1.25	0.22 (0.13)	1.87	0.62 (0.14)	0.09	-2.42 (0.12)
			10	1.55	0.44 (0.08)	0.89	-0.12 (0.25)	0.04	-3.29 (0.41)
			30	1.62	0.48 (0.03)	0.50	-0.69 (0.16)	0.02	-3.89 (0.19)
		H	0	1.35	0.3 (0.05)	1.52	0.42 (0.26)	0.07	-2.65 (0.19)
			10	1.53	0.43 (0.07)	0.76	-0.28 (0.16)	0.03	-3.41 (0.2)
			30	1.61	0.48 (0.02)	0.44	-0.81 (0.21)	0.02	-4.01 (0.23)
		HF	0	1.35	0.3 (0.08)	1.73	0.55 (0.14)	0.08	-2.5 (0.13)
			10	1.55	0.44 (0.08)	0.89	-0.12 (0.15)	0.04	-3.22 (0.2)
			30	1.62	0.48 (0.03)	0.44	-0.81 (0.12)	0.02	-3.97 (0.21)
	2021	C	0	1.33	0.29 (0.1)	1.43	0.36 (0.3)	0.05	-3.01 (0.43)
			10	1.62	0.48 (0.03)	0.78	-0.25 (0.61)	0.03	-3.61 (1.21)
			30	1.71	0.53 (0.03)	0.53	-0.64 (0.27)	0.01	-4.41 (0.45)
		F	0	1.33	0.29 (0.12)	1.41	0.35 (0.27)	0.05	-3.02 (0.35)
			10	1.60	0.47 (0.05)	0.86	-0.15 (0.12)	0.03	-3.65 (0.21)
			30	1.65	0.5 (0.03)	0.63	-0.47 (0.12)	0.01	-4.23 (0.35)
		H	0	1.37	0.32 (0.03)	1.22	0.2 (0.2)	0.04	-3.22 (0.29)
			10	1.64	0.49 (0.03)	0.73	-0.31 (0.21)	0.02	-4.06 (0.46)
			30	1.66	0.51 (0.01)	0.50	-0.69 (0.24)	0.01	-4.43 (0.47)
		HF	0	1.41	0.34 (0.04)	1.22	0.2 (0.3)	0.04	-3.2 (0.4)
			10	1.59	0.47 (0.05)	0.80	-0.23 (0.25)	0.02	-3.82 (0.31)
			30	1.63	0.49 (0.05)	0.51	-0.67 (0.14)	0.01	-4.2 (0.17)

Table 3.5 p-Values for ANOVA of mineral soil bulk density, C concentrations, and N concentrations.

Effect	df	BD	C%	N%
Fert	1	0.061	0.064	0.086
Herb	1	0.267	0.012	0.078
Depth	2	<0.001	<0.001	<0.001
Year	1	<0.001	0.008	0.111
Location	3	<0.001	<0.001	<0.001
Fert:Herb	1	0.551	0.874	0.476
Fert:Depth	2	0.077	0.528	0.874
Herb:Depth	2	0.018	0.965	0.428
Fert:Year	1	0.403	0.674	0.774
Herb:Year	1	0.598	0.587	0.584
Depth:Year	2	<0.001	0.538	0.348
Fert:Herb:Depth	2	0.989	0.731	0.528
Fert:Herb:Year	1	0.758	0.983	0.326
Fert:Depth:Year	2	0.808	0.439	0.987
Herb:Depth:Year	2	0.891	0.947	0.702
Fert:Herb:Depth:Year	2	0.556	0.827	0.870

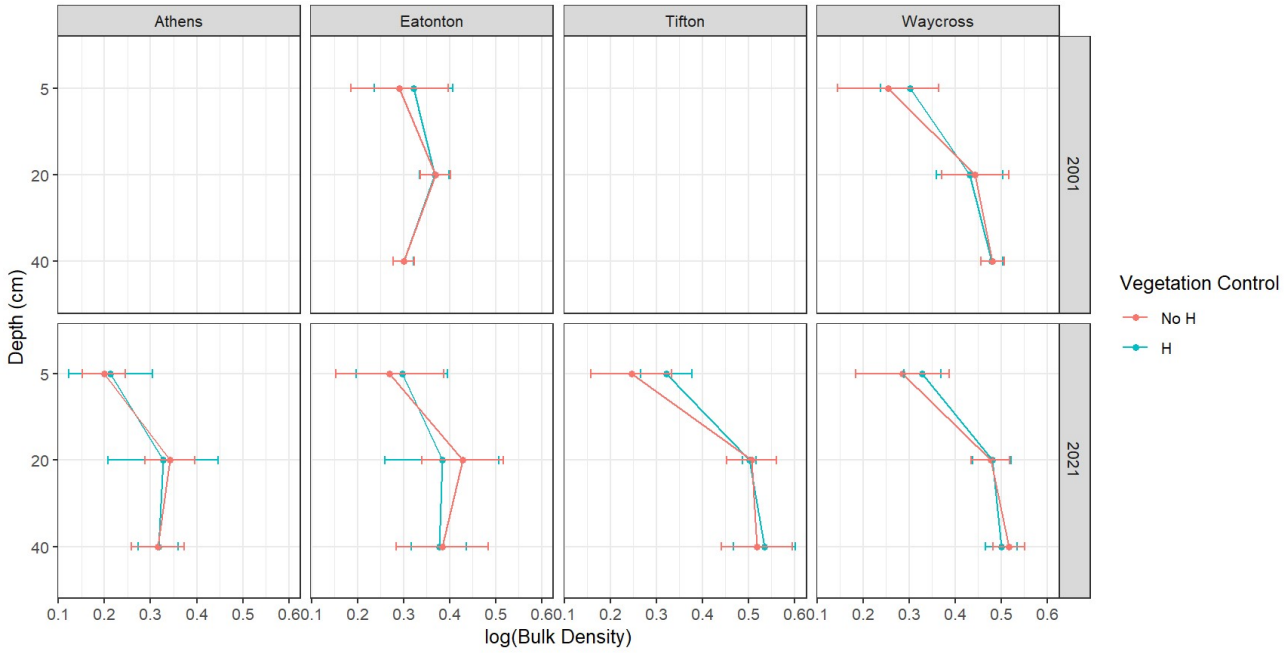

Figure 3.9 Comparison of log-transformed soil bulk density between plots with vegetation control treatment (H and HF) and without vegetation control treatment (C and F).

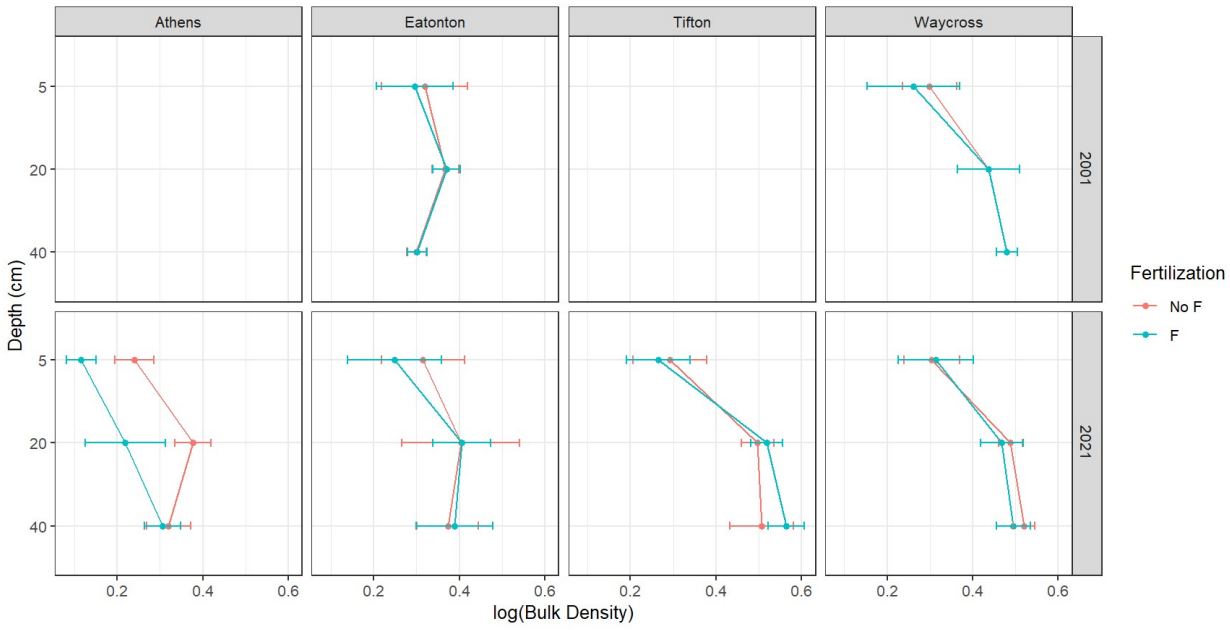

Figure 3.10 Comparison of log-transformed soil bulk density between plots with fertilization (F and HF) and without fertilization (C and H).

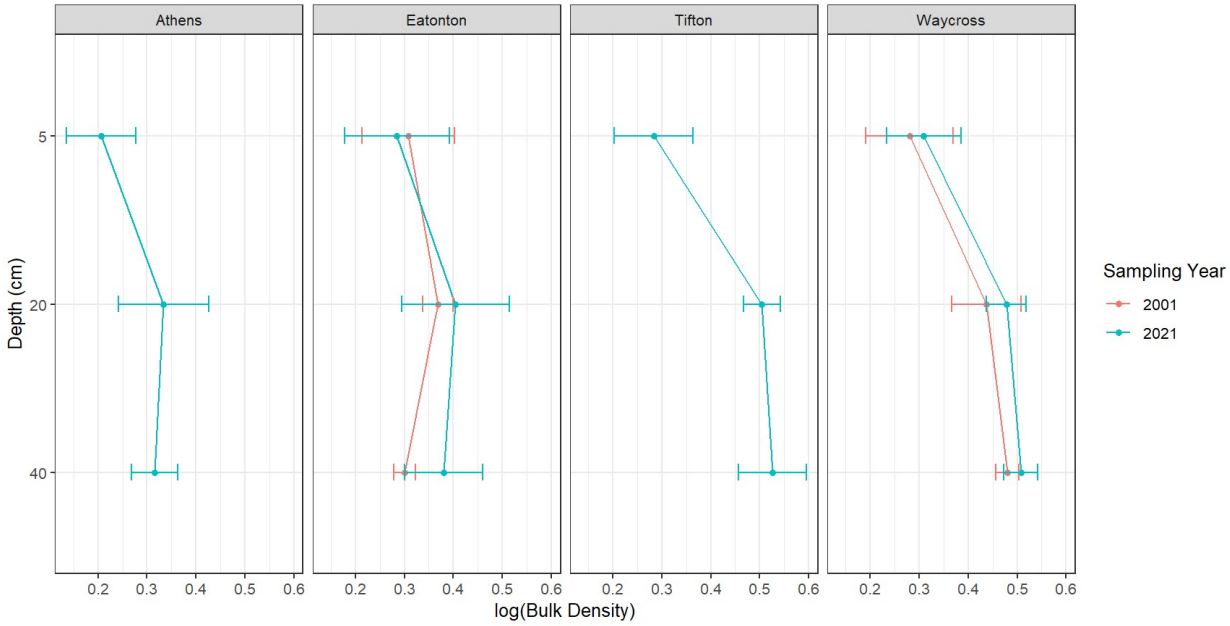

Figure 3.11 Comparison of log-transformed soil bulk density between sampling years (2001 vs 2021).

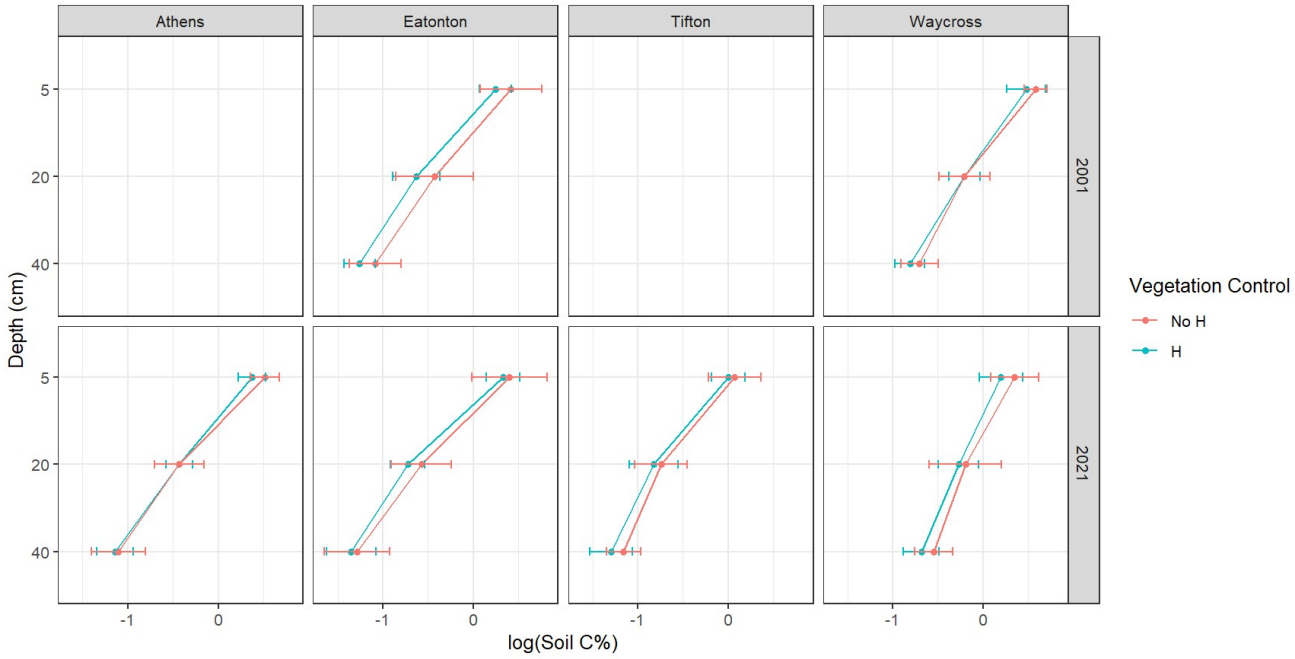

Figure 3.12 Comparison of log-transformed soil C concentrations between plots with vegetation control treatment (H and HF) and without vegetation control treatment (C and F).

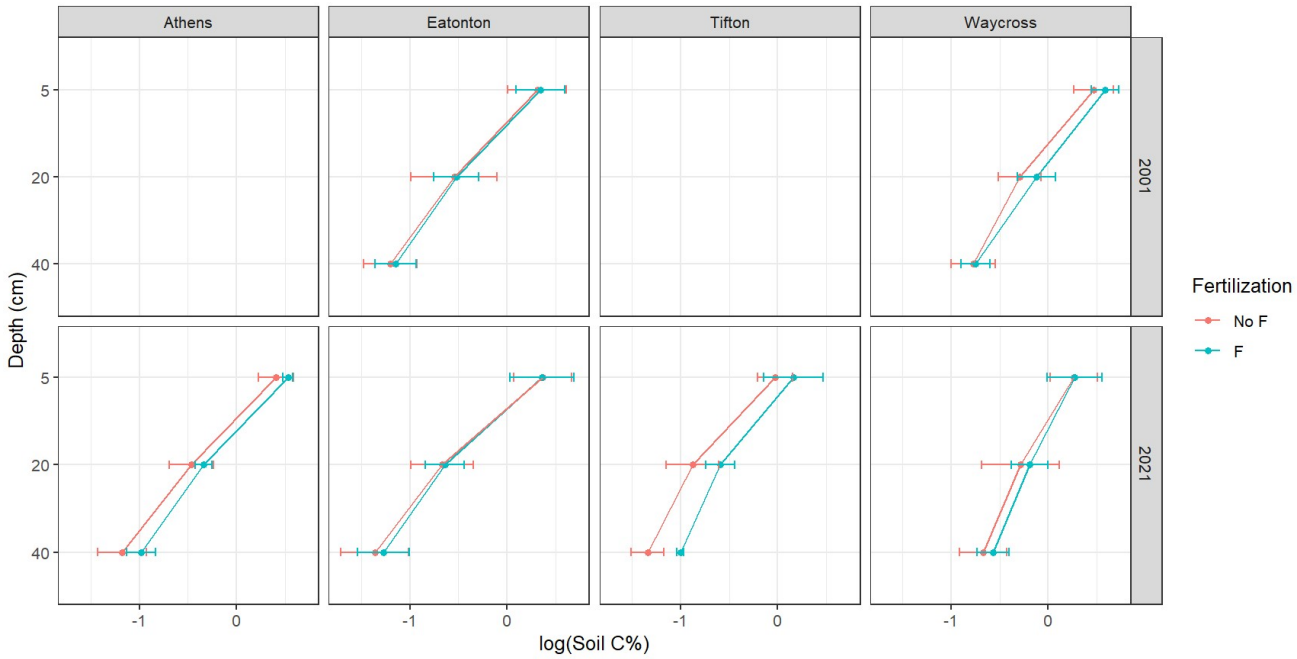

Figure 3.13 Comparison of log-transformed soil C concentrations between plots with fertilization (F and HF) and without fertilization (C and H).

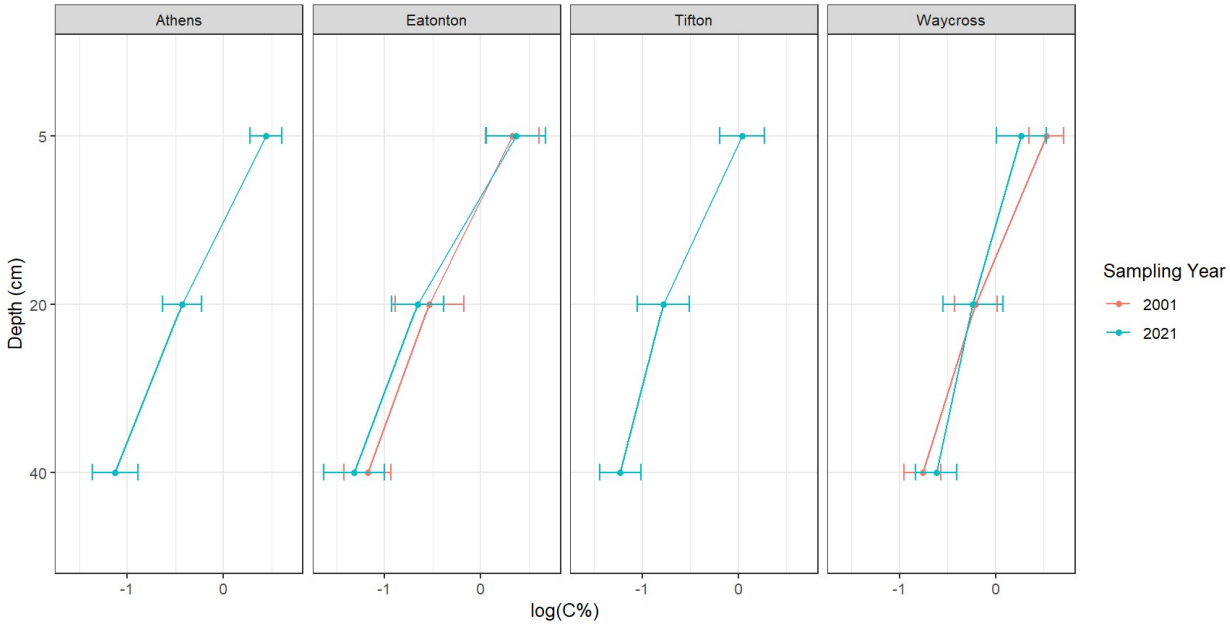

Figure 3.14 Comparison of log-transformed soil C concentrations between sampling years (2001 vs 2021).

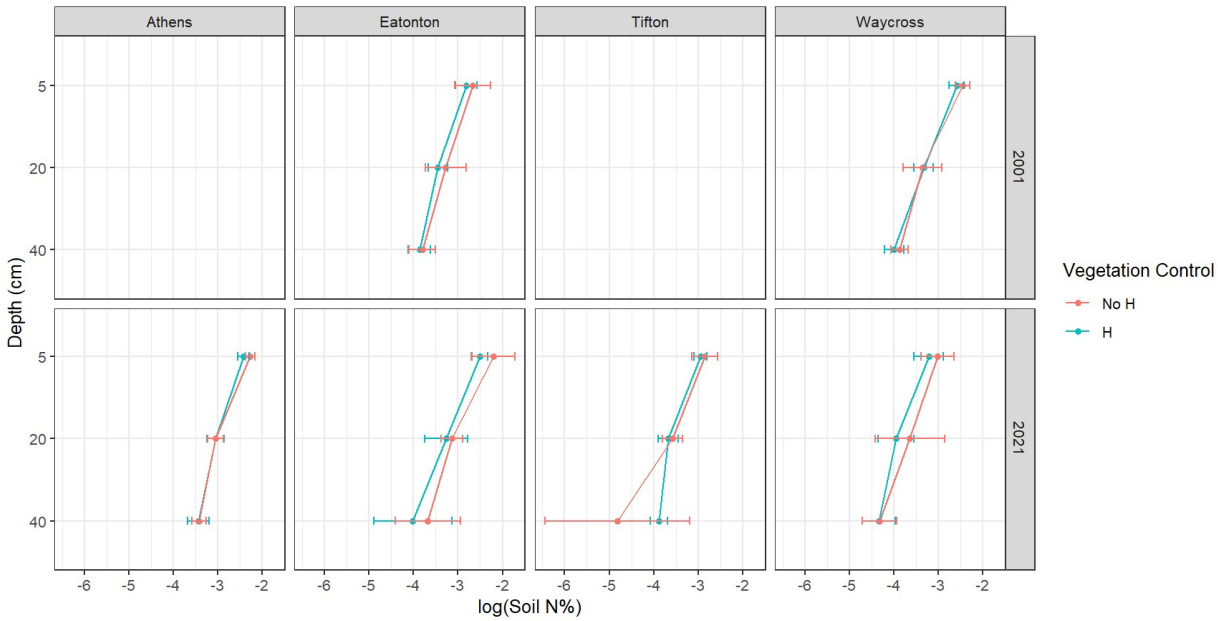

Figure 3.15 Comparison of log-transformed soil N concentrations between plots with vegetation control treatment (H and HF) and without vegetation control treatment (C and F).

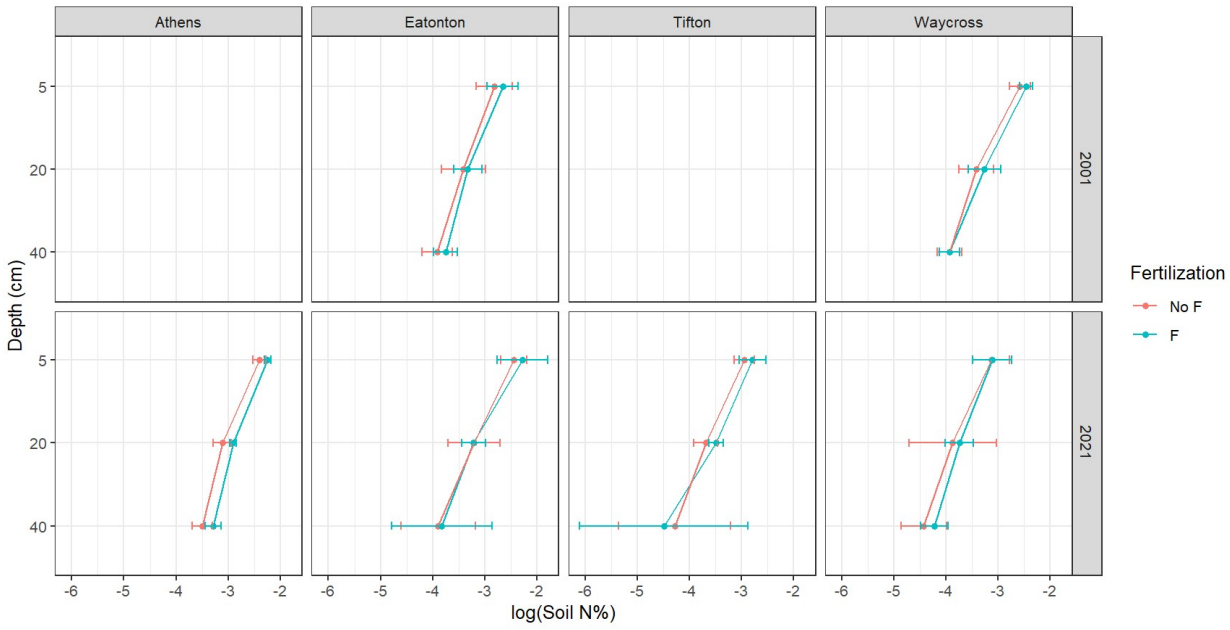

Figure 3.16 Comparison of log-transformed soil N concentrations between plots with fertilization (F and HF) and without fertilization (C and H).

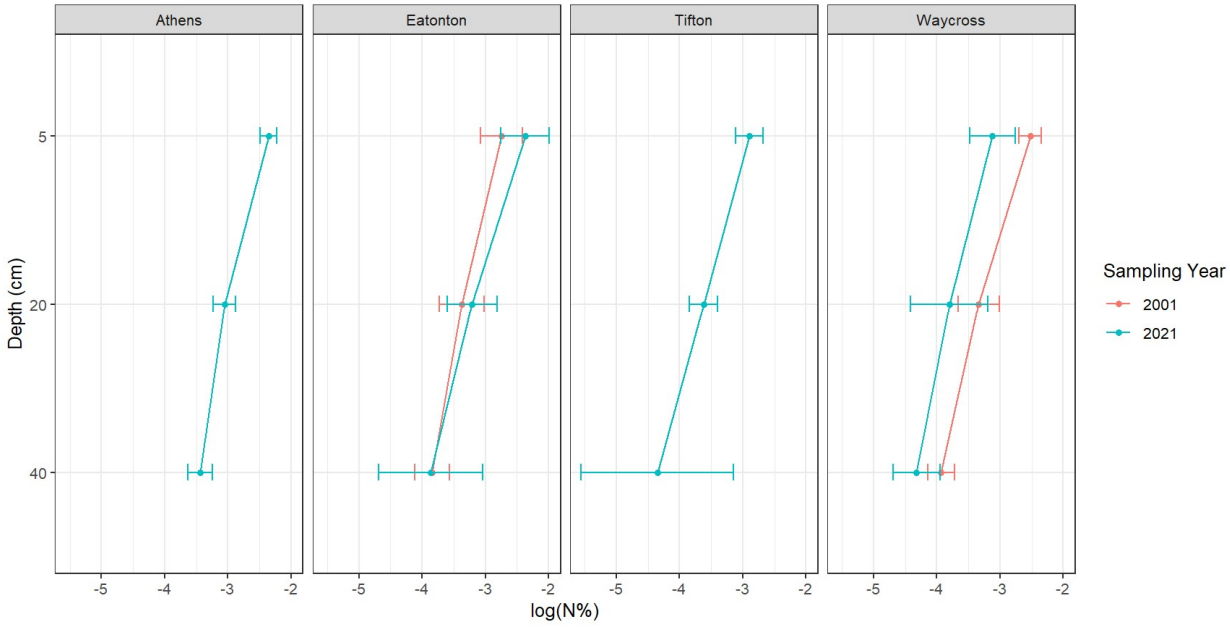

Figure 3.17 Comparison of log-transformed soil N concentrations between sampling years (2001 vs 2021).

Table 3.6 Ecosystem C pools including living biomass, dead biomass, forest floor, and mineral soil (interquartile range) across locations, sampling year, and treatment. Living and dead biomass means are untransformed. Forest floor and mineral soil means were back transformed from a log transformation.

Location	Year	Trt	Living Biomass (Mg/ha)	Dead Biomass (Mg/ha)	Forest Floor (Mg/ha)	Soil 0-10 (Mg/ha)	Soil 10-30 (Mg/ha)	Soil 30-50 (Mg/ha)	Total C (Mg/ha)
Athens	2021	C	155.5 (146.77-161.26)	18.74 (11.21-27.69)	13.94 (11.79-15.48)	20.48 (18.12-22.52)	18.64 (15.14-20.82)	9.15 (8.18-10.17)	236.45 (211.22-257.94)
		F	114.32 (114.32-114.32)	57.81 (57.81-57.81)	12.96 (12.96-12.96)	20.61 (20.61-20.61)	17.03 (17.03-17.03)	8.73 (8.73-8.73)	231.46 (231.46-231.46)
		H	167.92 (163.54-172.3)	24.45 (16.83-28.77)	20.98 (18.18-23.69)	17.9 (16.78-19.43)	17.96 (16.99-18.36)	7.84 (7.2-8.44)	257.05 (239.53-271)
		HF	115.54 (109.11-121.97)	92.27 (88.36-96.17)	25.78 (22.76-29.2)	18.38 (17.86-18.9)	18.12 (16.87-19.46)	10.95 (10.27-11.69)	281.04 (265.25-297.4)
Eatonton	2001	C	25.5 (17.83-36.3)	0.09 (0-0.01)	4.14 (3.2-5.46)	21.18 (16.26-27.39)	19.36 (13.99-24.33)	8.7 (6.77-9.46)	78.97 (58.04-102.95)
		F	35.76 (11.72-57.13)	0.51 (0.08-0.5)	6.67 (5.1-10.71)	19.64 (16.06-24.74)	18.55 (14.97-23.16)	9.53 (8.65-10.97)	90.66 (56.59-127.21)
		H	45.08 (29.28-59.94)	0.44 (0-0.7)	6.37 (5.48-8.71)	17.33 (15.29-19.86)	15.19 (12.27-19.74)	7.74 (6.93-8.79)	92.17 (69.26-117.75)
		HF	58.13 (29.88-82.01)	1.13 (0.02-1.32)	7.72 (6.92-12.91)	18.24 (16.37-20.38)	15.77 (13.76-18.04)	7.59 (6.84-8.41)	108.57 (73.79-143.08)
	2021	C	137.61 (126.22-147.79)	21.1 (10.17-31.37)	10.94 (9.16-13.8)	20.89 (16.82-26.4)	17.63 (14.29-21.94)	7.74 (6.89-9.44)	215.9 (183.55-250.73)
		F	105.9 (56.48-142.55)	75.12 (77.39-86.68)	18.97 (17.15-23.17)	18.53 (16.52-22.66)	17.03 (14.87-20.34)	8.47 (7.06-9.28)	244.01 (189.47-304.68)
		H	144.84 (126.6-158.11)	30.77 (20.21-38.41)	14.26 (12.32-18.23)	19.12 (16.78-20.26)	13.99 (11.87-16.04)	7.24 (6.3-8.05)	230.21 (194.07-259.09)
		HF	139.36 (107.6-175.41)	78.16 (42.31-107.01)	26.5 (25.83-28.47)	18.28 (16.98-20.88)	14.57 (13.28-16.75)	8.07 (6.24-9.5)	284.94 (212.23-358.01)

Tifton	2021	C	177.09 (175.91-178.72)	22.55 (19.49-29.33)	16.29 (14.04-19.06)	12.5 (10.86-13.03)	14.57 (12.91-16.19)	9.35 (8.78-9.71)	252.35 (241.99-266.04)
		F	122.51 (112.2-132.83)	56.43 (52.33-60.52)	20.92 (18.81-23.27)	16.61 (15.26-18.07)	18.71 (17.15-20.4)	13.34 (13.06-13.62)	248.51 (228.82-268.71)
		H	181.79 (169.67-190.25)	46.06 (34.1-60.5)	21.28 (18.63-22.97)	13.72 (13.33-14.44)	12.84 (12.33-14.3)	8.08 (7.35-8.38)	283.78 (255.41-310.84)
		HF	170.01 (157.25-182.76)	50.37 (49.19-51.55)	25.8 (22.78-29.23)	14.18 (12.84-15.66)	18.47 (18.27-18.68)	12.49 (12.21-12.77)	291.32 (272.53-310.65)
Waycross	2001	C	37.92 (36.07-43.32)	0.32 (0.02-0.12)	7.15 (6.11-8.78)	22.95 (20.91-24.29)	23.08 (21.8-28.78)	15.74 (13.44-20.63)	107.16 (98.34-125.92)
		F	76.54 (54.27-89.29)	4.35 (0.49-7.01)	10.64 (6.07-18.67)	23.28 (19.85-25.72)	27.42 (25.85-32.56)	16.25 (14.47-18.03)	158.49 (121.00-191.29)
		H	47.88 (43.86-49.84)	0.67 (0.01-0.75)	6.82 (4.67-8.7)	20.6 (17.89-23.09)	23.17 (22.37-24.8)	14.35 (13.21-15.94)	113.49 (102.01-123.13)
		HF	83 (56.49-108.25)	2.88 (0.38-4.36)	12.7 (8.79-16.57)	23.41 (22.09-24.41)	27.43 (26.38-27.58)	14.36 (13.36-14.94)	163.78 (127.5-196.11)
	2021	C	148.57 (136.79-166.71)	12.05 (5.24-8.61)	38.49 (30.74-47.18)	19.01 (17.67-21.43)	25.38 (19.76-28.98)	18.01 (15.16-21.2)	261.52 (225.37-294.12)
		F	148.44 (135.96-165.15)	68.84 (61.1-75.31)	53.3 (49.04-52.86)	18.79 (18.07-21.43)	27.45 (26.89-29.21)	20.75 (19.24-21.58)	337.57 (310.3-365.53)
		H	155.23 (145.25-168.96)	14.58 (11.96-20.99)	38.7 (35.32-42.21)	16.73 (14.87-17.57)	23.95 (20.97-27.15)	16.71 (14.04-17.67)	265.91 (242.41-294.55)
		HF	153.5 (149.1-167.5)	70.77 (64.05-70.71)	47.43 (38.3-58.51)	17.22 (14.67-20.86)	25.43 (22.71-25)	16.63 (16.47-17.65)	330.97 (305.3-360.23)

Table 3.7 Ecosystem N pools including living biomass, dead biomass, forest floor, and mineral soil (interquartile range) by location, sampling year, and treatment. Living and dead biomass means are untransformed. Forest floor and mineral soil means were back transformed from a log transformation.

Location	Year	Trt	Living Biomass (kg/ha)	Dead Biomass (kg/ha)	Forest Floor (kg/ha)	Soil 0-10 (kg/ha)	Soil 10-30 (kg/ha)	Soil 30-50 (kg/ha)	Total N (kg/ha)
Athens	2021	C	445 (426-458)	72 (43-108)	393 (359-410)	1239 (1151-1337)	1314 (1103-1518)	868 (831-937)	4332 (3913-4768)
		F	335 (335-335)	230 (230-230)	471 (471-471)	1317 (1317-1317)	1393 (1393-1393)	984 (984-984)	4730 (4730-4730)
		H	473 (463-490)	88 (61-104)	539 (501-580)	1077 (1049-1096)	1285 (1162-1402)	807 (671-939)	4270 (3907-4612)
		HF	328 (314-341)	335 (326-343)	782 (689-888)	1136 (1124-1149)	1353 (1254-1460)	1030 (962-1104)	4964 (4669-5285)
Eatonton	2001	C	115 (90-159)	0 (0-0)	120 (88-177)	862 (679-1111)	1068 (763-1373)	532 (464-513)	2697 (2083-3332)
		F	153 (70-229)	2 (0-0)	238 (182-354)	999 (846-1307)	1116 (790-1311)	672 (558-822)	3179 (2447-4023)
		H	185 (143-226)	1 (0-0)	157 (132-215)	795 (716-915)	885 (760-953)	544 (446-653)	2568 (2196-2962)
		HF	225 (147-295)	7 (0-10)	217 (185-347)	883 (726-1102)	964 (877-1060)	595 (517-664)	2890 (2452-3479)
	2021	C	398 (369-424)	65 (30-95)	324 (280-399)	1253 (1054-1544)	1291 (1034-1508)	561 (274-957)	3892 (3040-4926)
		F	310 (212-397)	238 (223-298)	658 (585-823)	1625 (1256-1584)	1360 (1167-1610)	954 (688-1265)	5144 (4130-5976)
		H	413 (366-447)	94 (68-115)	424 (351-542)	1141 (1000-1328)	1170 (843-1427)	614 (337-977)	3855 (2966-4835)
		HF	388 (293-492)	230 (128-315)	833 (776-884)	1036 (906-1194)	1054 (939-1214)	415 (393-843)	3956 (3434-4941)

Tifton	2021	C	511 (495-526)	79 (67-103)	574 (501-659)	683 (582-727)	899 (760-983)	334 (286-728)	3080 (2692-3726)
		F	381 (352-409)	228 (215-240)	756 (680-841)	873 (807-946)	1015 (958-1075)	181 (82-400)	3434 (3094-3911)
		H	511 (474-544)	161 (124-207)	610 (546-677)	724 (712-750)	759 (711-821)	638 (594-650)	3403 (3161-3649)
		HF	497 (467-526)	198 (187-208)	885 (812-964)	740 (698-784)	1052 (1002-1104)	867 (830-906)	4238 (3995-4493)
Waycross	2001	C	150 (134-164)	2 (0-0)	137 (119-158)	1123 (968-1165)	1020 (1027-1208)	702 (658-768)	3135 (2906-3464)
		F	272 (222-304)	16 (0-30)	274 (149-473)	1106 (1067-1200)	1149 (981-1478)	658 (590-717)	3475 (3008-4201)
		H	184 (172-190)	2 (0-0)	123 (96-137)	955 (863-1038)	1013 (881-1178)	588 (531-678)	2864 (2542-3220)
		HF	293 (231-348)	10 (0-20)	286 (182-387)	1107 (1068-1188)	1234 (1127-1286)	611 (544-688)	3541 (3153-3916)
	2021	C	419 (392-474)	40 (20-30)	937 (643-1381)	658 (552-770)	882 (726-1060)	417 (427-523)	3352 (2759-4238)
		F	412 (374-458)	204 (180-220)	1501 (1317-1719)	651 (534-782)	828 (727-914)	479 (376-574)	4075 (3508-4667)
		H	438 (413-467)	44 (40-60)	1050 (935-1132)	550 (475-688)	566 (427-746)	395 (384-482)	3042 (2674-3574)
		HF	429 (421-465)	210 (190-210)	1255 (1058-1454)	573 (469-770)	697 (569-764)	487 (438-490)	3653 (3145-4154)

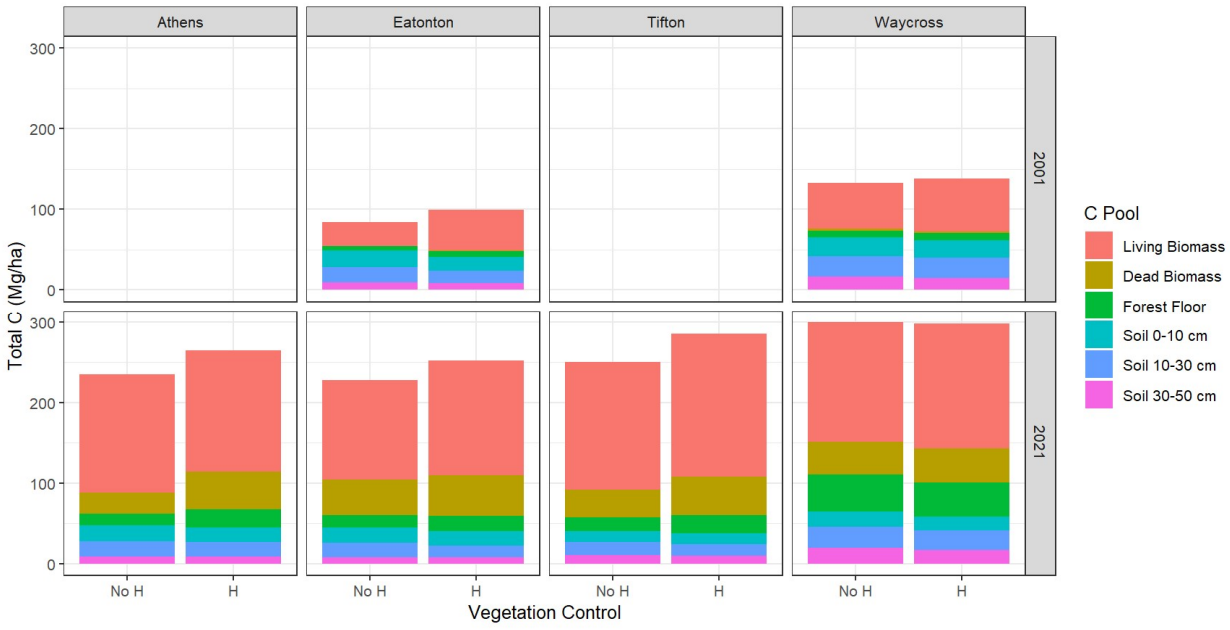

Figure 3.18 Comparison of ecosystem C contents (living biomass, dead biomass, forest floor, mineral soil) between vegetation control treatment (H and HF) and non-vegetation control treatment (C and F) plots.

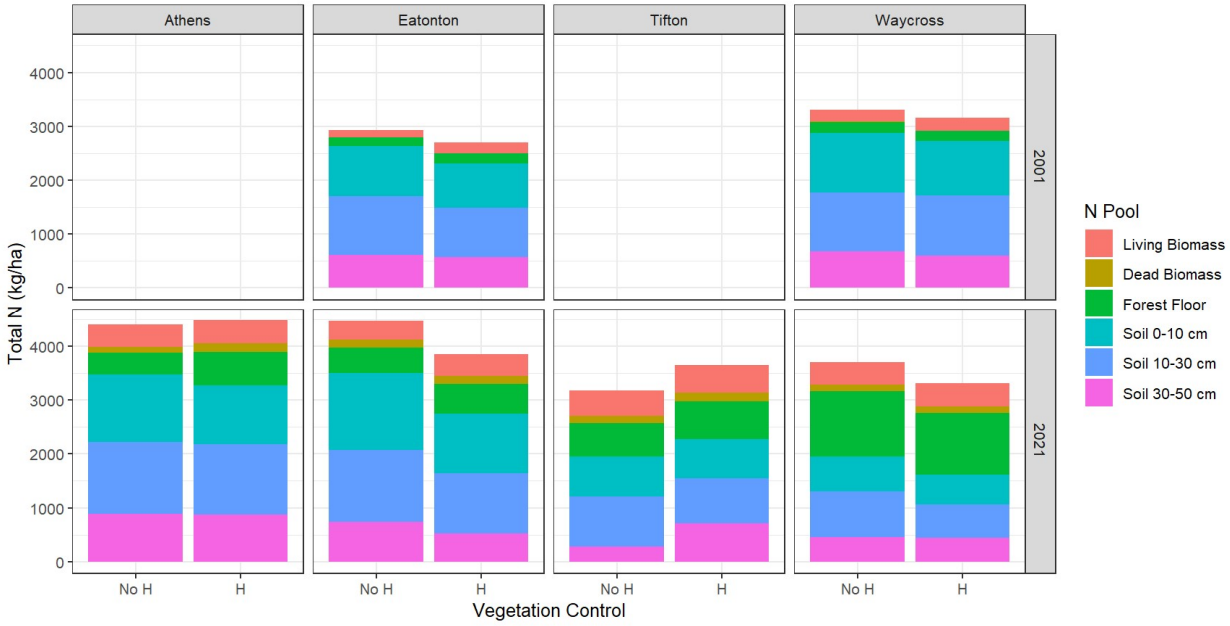

Figure 3.19 Comparison of ecosystem N contents (living biomass, dead biomass, forest floor, mineral soil) between vegetation control treatment (H and HF) and non-vegetation control treatment (C and F) plots.

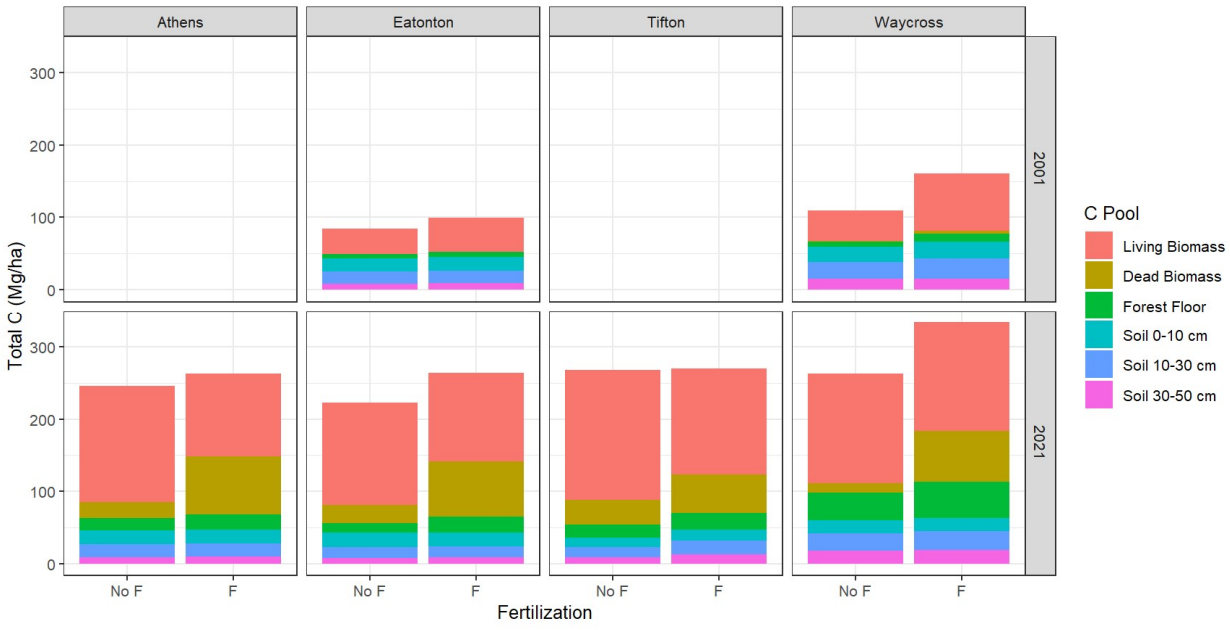

Figure 3.20 Comparison of ecosystem C contents (living biomass, dead biomass, forest floor, mineral soil) between fertilized (F and HF) and unfertilized (C and H) plots.

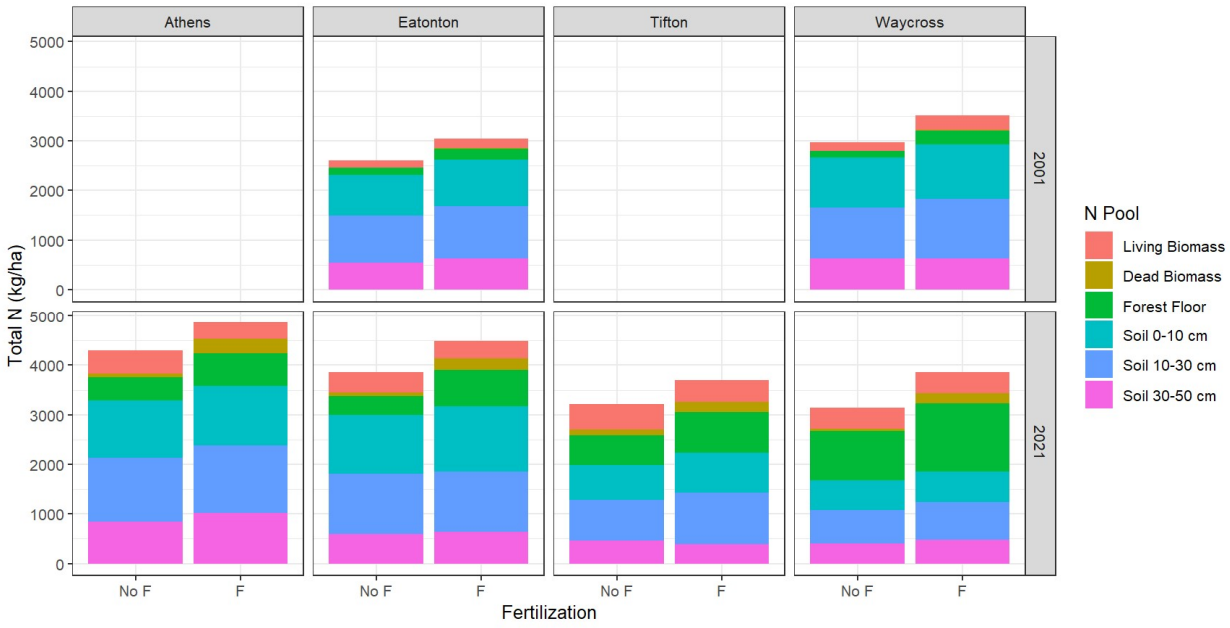

Figure 3.21 Comparison of ecosystem N contents (living biomass, dead biomass, forest floor, mineral soil) between fertilized (F and HF) and unfertilized (C and H) plots.

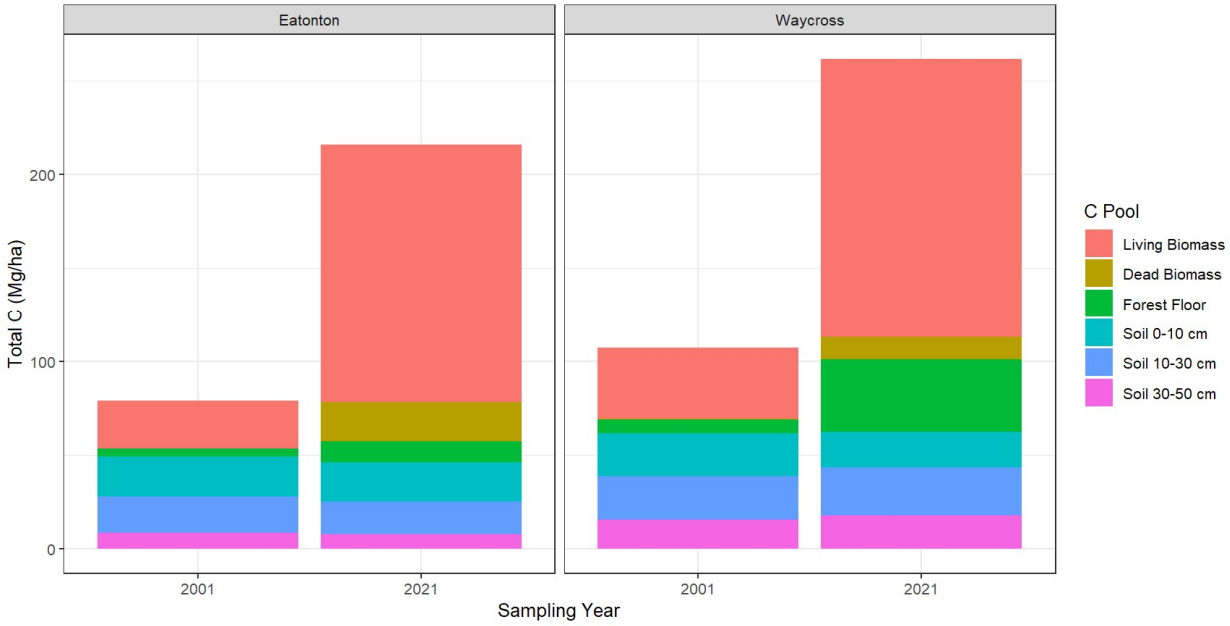

Figure 3.22 Comparison of ecosystem C contents (living biomass, dead biomass, forest floor, mineral soil) for control plots between sampling years (2001 vs 2021).

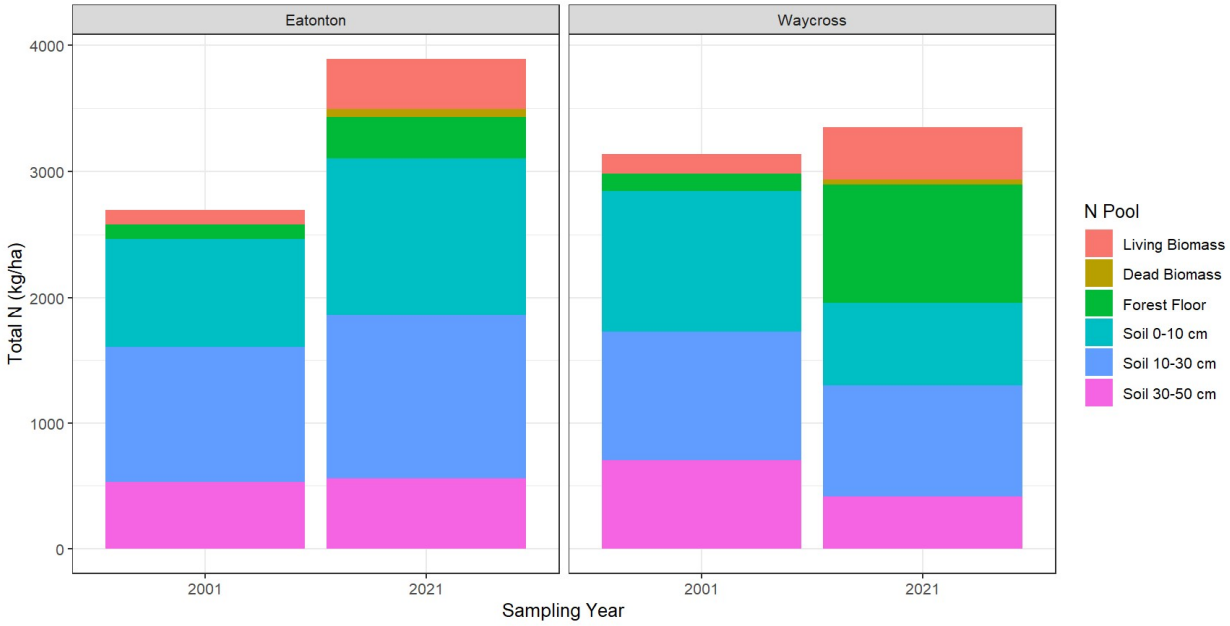

Figure 3.23 Comparison of ecosystem N contents (living biomass, dead biomass, forest floor, mineral soil) for control plots between sampling years (2001 vs 2021).

CHAPTER IV

SOIL ACIDIFICATION AFTER THREE DECADES UNDER PINE PLANTATION MANAGEMENT[2]

[2] Hoffman, S.A. and Markewtiz, D. To be submitted to *Soil Science Society of America Journal*

4.1. Abstract

Understanding the impacts of silviculture on soil acidification is critical for sustainable forest management. This study investigates changes in soil exchange chemistry from 0-50 cm in response to fertilization and vegetation control under *Pinus taeda* plantations in four well-replicated locations in Georgia, USA at stand ages ~10 and ~30 years. The hypothesis posits that both fertilization and vegetation control will induce soil acidification by enhancing productivity and thus base cation uptake. An additional acidifying effect from fertilization is theorized as N additions might lead to cation loss through leaching. By the stand age of 30 years, fertilization demonstrates a clear acidifying impact on the soil, reflected in decreased pH and base cation content, along with increased exchangeable acidity. Acidification from fertilization is only partially explained by increased biomass uptake, indicating the likelihood of leaching. Surprisingly, vegetation control did not acidify the system within the studied 50 cm depth even though treatment increased productivity. Regardless of treatment, soil acidification under pines becomes evident after two decades, marked by lower pH and base cations, and increased exchangeable acidity. Acidification over time appeared to be linked mainly to biomass uptake. The acidifying effect in this study raises concerns about the impact of plantation management on soils, particularly as systems enter multiple rotations and become more intensively managed.

4.2. Introduction

Soil acidification is a concern for forest ecosystems as it results in significant changes to soil properties, which includes decreased pH; decreased base cation concentrations (Ca^{2+}, Mg^{2+}, and K^{+}); and increased acidic cation concentrations (H^{+} and Al^{3+}), that can negatively affect ecosystem productivity and function. Base cation losses are predicted to reduce productivity via nutrient deficiencies in the long-term for less sensitive species such as *Pinus taeda* (Loblolly Pine) and in the short-term for sensitive species such as *Acer saccharum* (Sugar Maple) (Fenn et al., 2006). Also, increases in Al^{3+} concentrations and mobility due to acidification can be toxic to plant roots and microbes further reducing productivity and negatively affecting ecosystem functions such as nutrient cycling (Bache, 1985; Kunhikrishnan et al., 2016; Delhaize & Ryan, 1995).

Declines in soil productivity and function in response to acidification have been observed when assessed across multiple ecosystems with a 14.7% decrease in soil respiration, 19.1% decrease in fine root biomass, and 9.6% decrease in microbial biomass (Meng et al. 2019). In the meta-analysis of Meng et al (2019) soil pH dropped by 0.24 units across forest and non-forest ecosystems as well as seven different soil orders. Also, there has been observed decreases in tree growth and vigor of temperate hardwood species such as *Acer saccharum* and boreal conifer species such as *Pinus sylvestris* (Scots Pine) from experimental acidification in the form of nitrogen and/or sulfur additions. Tree impacts have been explained by reduced base cation availability for the trees (Hogberg et al., 2006; Ouimet et al., 2017; Sullivan et al., 2013). For

forest production systems, such as pine plantations, this could mean a long-term reduction in site productivity.

Pine plantation management, mainly through fertilization and understory vegetation control in the southeast US, has the potential to further acidify forest soils by effecting the processes responsible for acidification. Processes that drive acidification of forest soils include leaching from N and S deposition, accumulation of cations in biomass, oxidation of N and S compounds, release of organic acids from biomass, and formation of carbonic acid (Binkely & Richter, 1987). Previous studies have demonstrated the acidifying effect of N additions in forest ecosystems but only in secondary forests and grasslands with little work in plantations. Globally, N additions via atmospheric deposition or experimental addition have led to ~0.2 unit decrease in soil pH in 0-15 cm of soil over 40 years across desert, shrubland, cropland, wetland, grassland, forest, and tundra ecosystems (Chen et al., 2023; Tian et al., 2015). More locally in alpine soils under predominately forb and shrub cover, soil pH decreased between ~0.2-1.0 units with a ~24-40% decrease in base cation concentrations in the top 15 cm when 60-150 kg/ha/yr of N was applied over a period of 3-11 years (Bowman et al., 2008; Lieb et al., 2011). Under boreal forests, N additions of 60-90 kg/ha/yr over 30 years caused ~1.0 unit decrease in soil pH, up to 70% decrease in base cation concentrations, and up to a 222% increase in Al^{3+} in 0-55 cm of mineral soil (Hogberg et al., 2006). Temperate grasslands across limestone and acidic soils also display significant decrease in soil pH ~0.2-1.0 units and a 2-16% decrease in total base cation concentrations in the top 20 cm under N additions of 140 kg/ha/yr over 8 years (Horswill et al., 2008). Temperate forests similarly exhibited soil acidification from N additions of 198 kg/ha/yr over ~6 years with a 0.7 unit decrease in soil solution pH and significant mobilization of Al^{3+} in the top 30 cm of soil (Bergkvist et al., 1992). Tropical soils under natural forest exhibited a

similar pH decrease at ~0.2 units and reduction in base cation concentration of ~38-45% in the top 10 cm of mineral soil from ~6-8 years of adding N at a rate of 150 kg/ha/yr (Lu et al., 2014; Mao et al., 2017). Tropical soils under broadleaf plantations, however, displayed no significant decreases in soil pH or base cations in the top 10 cm of mineral soils after 7 years of adding 100 kg/ha/yr of N indicating a possible difference between the response of plantations and natural forest to N additions (Huang et al 2021). Since N additions have a acidifying effect in most forest ecosystems there is great incentive to understand if N additions in the form of fertilization in plantations will have a similar effect over the long-term.

Little effort has been made to characterize the acidifying effect of understory vegetation control in pine plantations over a whole rotation, but short-term plantation studies have found a mixed response. Acidification has been observed in a Scots pine plantation after six years of vegetation control with a 0.6 unit decrease in soil pH and 51% decrease in soil base cations (Zhang et al., 2022). Conversely, soil exchange chemistry in Mason pine (*Pinus massoniana*) and mixed plantations were unresponsive to vegetation control after short-term treatment of one to three years (Shen et al., 2018; Zhao et al., 2011). Although there are conflicting results on the impacts of understory vegetation control on soil acidification, there is a clear logic to the expectation that competing vegetation control leads to increased crop tree growth and thus base cation removal (i.e., Ca, Mg) in the form of harvest and should lead to greater soil acidification. Whether this expectation holds for deeply rooted forest ecosystems or specifically under pine plantations of the Southeast US with 25-year rotations and on a diversity of soil types warrants continued study.

The goal of this study is to assess acidification potential of pine plantation management (i.e., fertilization and vegetation control) by monitoring changes in soil pH, base cation content

and acidity from the first to third decade of the rotation. Specifically, a sampling of the Consortium of Accelerated Pine Production Studies (CAPPS) plots in 2001 at stand ages 8-12 years old and a resampling in 2021 at stand ages 29-33 years old is used to assess the effects of fertilization and understory vegetation control over time. CAPPS consists of five different installations covering a 360 km distance representing a climate and soil gradient from middle to south Georgia, United States. Although previous CAPPS studies largely focused on other soil properties (i.e., soil C and N) there is some indication of soil acidification from plantation management. In a more limited sampling of only 0-10 cm soil in 2007-2008 (14-18 yrs old), soil pH was 0.5 units lower in fertilized plots compared to unfertilized plots while soil pH remained unaffected by vegetation control (Rifai et al., 2010). Based on Rifai et al (2010) and previous N deposition studies, it is hypothesized that fertilization will continue to lower soil pH, decrease base cation concentration, and increase acidity through the end of the rotation as N loading will likely result in greater NO_3^- leaching and therefore base cation leaching. Conversely, the lack of effect from understory vegetation control on soil pH seen in Rifai (2010) as well as mixed responses of soil pH and base cations from short-term vegetation control it is unlikely a response will be observed at the end of the rotation.

4.3. Materials & Methods

4.3.1. Study Site

Soils were sampled from the Consortium for Accelerated Pine Productivity Studies (CAPPS) plots that were planted between 1987 to 1996. The CAPPS plots are made up of five installations at four locations: Athens, Eatonton-Monitor, Eatonton-Powerline, Tifton and Waycross-Wet. Athens, Eatonton-Monitor, and Eatonton-Powerline are all located in the

Piedmont, which consists of Kanhapludults such as the Cecil series. Kanhapludults in the Piedmont are well-drained, deep soils with high clay content formed in granite and gneiss bedrock. Tifton is located in the Upper Coastal Plain which consists of Kandiudults such as the Tifton series. Kandiudults in the Upper Coastal Plain are well-drained, deep soils with sandier surface textures formed from marine deposits. Waycross is located in the Atlantic Coast Flatwoods which consists of Paleaquults such as the Pelham series and Alaquods such as the Mascotte series. Paleaquults in the Atlantic Coast Flatwoods are poorly-drained soils formed from marine deposits with sandier textures that display sub-surface gleying. Alaquods in the Atlantic Coast Flatwoods contain similar properties as Paleaquults with the addition of a subsurface accumulation of carbon in a spodic horizon.

CAPPS is a randomized, incomplete block experimental design (Borders and Bailey, 2001). Each Location of CAPPS is made up of four to six blocks. Blocks were planted between 1987 and 1995 with 1680 trees per hectare with Waycross and Tifton being bedded and Athens and Eatonton being flat planted. These blocks then contain some combination of plots that receive fertilizer (F), understory vegetation control (H), fertilizer + understory vegetation control (HF), and/or no treatment (C). Originally there was n = 88 treatment plots, however, four plots were dropped due to excessive competition and pine beetle damage with fertilization from Athens (n=1) and Eatonton (n=3) bringing the final plot total to n = 84.

Fertilization treatment consisted of broadcast NPK application. Rates of fertilization application included 69 kg N/ha, 56 kg P/ha, and 56 kg K/ha applied annually as DAP, NH_4NO_3, and KCl in growing seasons 1-2. In growing seasons 3-9, 57 kg N/ha was added annually as NH_4NO_3. In growing season 10, 114 kg N/ha and 28 kg P/ha as NH_4NO_3 and Super Triple Phosphate were added. In growing season 11, 113 kg N/ha, 25 kg P/ha, and 47 kg K/ha as

NH_4NO_3 and super rainbow (10-10-10) that also includes some micronutrients was applied. In growing season 12-28, 114 kg N/ha was applied biennially as NH_4NO_3 with no more additions of P. Vegetation control treatments involved complete understory vegetation control using applications of sulfometuron methyl for the first three growing seasons and glyphosate (or manual removal) every year as needed for the full rotation. A block at Eatonton-Monitor, Eatonton-Powerline, and Waycross-Wet was also thinned at mid-rotation. Forest floor and mineral soils have been previously sampled in the CAPPS plots (Echeverria et al., 2004; Satori et al., 2007) with a majority of plots sampled at age 6 to14 years old in 2001. As such, current sampling in 2021 is a twenty-year resampling as individual plots have grown anywhere from 6 to 26 or 14 to 34 years old.

4.3.2. Forest Floor & Mineral Soil Sampling

Forest floor and mineral soil samples were taken in accordance with the original sampling done in 2001. A sample was composed of collections taken around eight randomly selected trees from an interior measurement plot with 81 trees (9 x 9 rows). At each tree, samples were paired with an on-bed and off-bed sample (or in-row and between-row if not bedded) with sample points based on randomly selected orientations and x-y distances from the tree for a total of 16 collection points. Forest floor was collected using a 35 x 35 cm quadrat. At each quadrat all organic material was removed down to the mineral soil excluding woody material >1 cm. Soil cores were taken beneath each forest floor collection point using ~3 cm (0-30 cm) and ~2 cm (30-50 cm) diameter punch tubes. Resulting soil cores were separated into three depths before being composited (i.e., 0-10, 10-30, and 30-50 cm). Separate mineral soil

samples were taken to measure bulk density. Bulk density cores were taken at two of the sampled points in each plot with a 4.5 kg slide hammer and a 7.5 cm diameter x 7.5 cm tall core.

The resulting forest floor samples were oven-dried at 65ºC and homogenized by plot. Homogenized forest floor samples were put through a Wiley mill with 1mm screen for digestion. Measured forest floor mass was adjusted via loss on ignition to remove the influence of mineral soil contamination. Composited soil cores were air-dried and put through a 2 mm sieve prior to analysis. Bulk density samples were dried at 105ºC and corrected for rocks and roots (Blake, 1965).

4.3.3. Soil pH

Soil pH was measured using a pH electrode in a 2:1 soil slurry (Thomas, 1996). Twenty grams of mineral soil was mixed with 40 ml of DI water for the water pH and 0.01 M $CaCl_2$ solution for the salt pH. Only the salt pH is discussed in this study as it is considered the most representative given fertilization treatments. All pH measurements were made using the Orion™ ROSS Ultra™ Low Maintenance pH/ATC Triode™ Combination Electrode with an Orion™ 710A Benchtop pH Meter. A subset of 2001 samples (n = 18) were re-run in 2021 to assess analytical bias (Figure 4.1). Measured pH values in 2021 were approximately 8% lower compared to initial measurements in 2001. No adjustments were made to pH values as analytical bias is potentially an artifact of sample storage (Falkengren-Grerup & Ursula, 1995). Additionally, statistical tests done on adjusted and unadjusted data were not different.

4.3.4. Base Cations & Acidity

Base cations were measured for forest floor samples via digestion of 0.1 g of oven-dried material using a combination of HNO_3 and H_2O_2, and heating on a Westco AD 4020 Heating Block (Environmental Protection Agency, 1996). Digested samples were diluted to 75 mL using DI water and decanted for measurement on an ICP-OES to measure cation concentrations in solution. A subsample of 2001 forest floor samples (n = 18) were re-run in 2021 to address the potential for bias between analytical years. Measured cations in 2021 were highly correlated with 2001 measurements with a 3% difference (Figure 4.1). No adjustments were made to forest floor cation concentrations.

Exchangeable base cations, (i.e., Ca, Mg, and K) were extracted from mineral soil samples using a Melich-I solution in a ratio of 20 mL solution to 5 g of air-dried mineral soil (Helmke & Sparks, 1996). The mixture was shaken for 5 minutes at 180 rpm and filtered through a Whatman 42 ashless filter. The resulting solution was also analyzed on an ICP-OES.

Exchangeable acidity was measured for mineral soil using a KCl extraction following Thomas (1982). In brief, 5 g of air-dried mineral soil was extracted in 50 mL of 1 M KCl by shaking for 30 minutes, filtered through a Whatman 42 ashless filter, and the extractant titrated to pH 8.2 using 0.02 M NaOH. All titrations were done using a Metrohm 751 GPD Titrino. Reanalysis of exchangeable base cations and acidity was done on all mineral soil and forest floor samples from 2001 to avoid potential analytical bias between sampling years.

For mass balance calculations, cation contents in soil and forest floor were calculated from soil and forest floor concentration and soil bulk density and forest floor mass. To estimate cation contents in aboveground biomass, biomass estimates from the CAPPS sites were taken from Zhao et al. (2022) while concentrations of cations in biomass were taken from other

locations in Albaugh et al. (2008) (Table 4.1). National Atmospheric Deposition Program (NADP) data on base cation deposition was considered to account for additional inputs (National Atmospheric Deposition Program, 2022). Complete cation budgets were formulated only for plots with complete soil, forest floor, and biomass data for 2001 and 2021, n = 48.

4.3.5. Statistical Analysis

The study follows a randomized, incomplete block design with a repeated measure over sampling year and depth. Planting year of the different blocks, which ranged from 1987 to 1995, was tested but was never significant so was dropped from the analysis. Treatment effects were evaluated with a Type-III ANOVA using a linear mixed-effects model in R (Bates et al., 2015; R Core Team, 2022). Fertilizer treatment, herbicide treatment, location, depth, and sampling year were assigned as fixed effects with interactions between fertilizer, herbicide, depth, and sampling year. The block and plot were treated as random effects. Forest floor and soil data were log transformed prior to analysis, as untransformed data were significantly different from normal based on a Shapiro-Wilkes test (P-value < 0.05). The only exception to log-transformations is soil pH as it is already log transformed. Back-transformed means are presented in the results with log-transformed means and standard deviations provided in tables. Figures utilize log-transformed means with standard deviations with the exception of soil cation concentrations.

4.4. Results

4.4.1. Forest Floor Mass

The average forest floor mass within CAPPS was 38.0 Mg/ha when averaged across treatments and sampling years (see Table 3.2). The range of forest floor mass was anywhere

between 9.2 Mg/ha in 2001 Eatonton C plots to 109.5 Mg/ha in 2021 Waycross F plots. Forest floor mass was significantly different between locations ($p < 0.001$). Waycross had 56.1 Mg/ha of forest floor mass which is substantially more than amounts sampled from Athens, Eatonton, and Tifton which had 37.3, 25.4, and 41.8 Mg/ha, respectively.

Fertilization and vegetation control significantly increased forest floor mass. Forest floor mass also significantly increased from 2001 to 2021 (see Table 3.3, Figure 3.3 and 3.4). When averaged across locations, 2021 fertilized plots had significantly more forest floor mass ($p < 0.001$) at 65.2 Mg/ha in 2021 while unfertilized plots had 43.6 Mg/ha. Plots that received vegetation control also had significantly more forest floor mass ($p < 0.001$) at 55.4 Mg/ha while reference plots had 49.9 Mg/ha in 2021 when averaged across locations. Forest floor mass significantly increased from 17.2 Mg/ha to 53.2 Mg/ha in 2021 when averaged across locations and treatments (see Figure 3.5). There was no additive effect when both fertilization and vegetation control were employed. Also, the effects of fertilization and vegetation control on forest floor mass did not change from 2001 to 2021.

4.4.2. Forest Floor Base Cations

Base cation concentrations in forest floor averaged around 2,199 mg/kg for Ca, 519.4 mg/kg for Mg, and 451.2 mg/kg for K (Table 4.2). Cation concentration in the forest floor did vary significantly between locations ($p < 0.001$). Base cations concentrations were higher in Athens and Eatonton such that Ca concentrations were 2,409 mg/kg and 2,904 mg/kg compared to Tifton and Waycross which had 1,697 mg/kg and 1,504 mg/kg when averaged across treatments and sampling year.

Forest floor Ca was significantly reduced by fertilization and vegetation control while also decreasing over time regardless of treatment (Table 4.3; Figure 4.2 and 4.3). Fertilized plots had significantly reduced Ca concentration in the forest floor ($p < 0.001$) at 1,562 mg/kg compared to unfertilized plots at 1,950 mg/kg in 2021. Vegetation control plots also had significantly reduced concentrations of Ca in the forest floor ($p < 0.001$), however, the reduction lessened over time ($p = 0.079$) with treated plots containing 1,591 mg/kg in 2021 compared to reference plots that contained 2,010 mg/kg in 2021. Forest floor Mg concentrations saw a significant reduction from vegetation control ($p < 0.001$) similar to Ca, however, the effect of fertilization was not apparent (Table 4.3; Figure 4.2 and 4.3). K concentrations in forest floor were unaffected by both fertilization and vegetation control (Table 4.3; Figure 4.2 and 4.3). Lastly, concentrations of Ca ($p < 0.001$), Mg ($p < 0.001$), and K ($p < 0.001$), were significantly reduced from 2001 to 2021 (Table 4.3; Figure 4.4). The largest change in forest floor concentration was Ca which went from 2,972 mg/kg in 2001 to 1,773 mg/kg in 2021 when averaged across locations and treatments.

4.4.3. Bulk Density

The average bulk density across CAPPS was 1.46 g/cm^3 with a range of 1.11 g/cm^3 in the 0-10 cm layer of the 2021 Athens HF plots to 1.77 g/cm^3 in the 30-50 cm layer of the 2021 Tifton F plots (see Table 3.4). Bulk density varied significantly across depth ($p < 0.001$) and location ($p < 0.001$). The 0-10 cm layer of mineral soil had the lowest bulk density of 1.34 g/ cm^3 while the 10-30 cm and 30-50 cm layers had bulk densities of 1.52 and 1.51 g/cm^3, respectively, when averaged across year, treatment, and location. Athens has the lowest bulk density of all locations at 1.34 g/cm^3 while Tifton had the highest at 1.56 g/cm^3 when averaged across year,

treatment, and depth. The average bulk densities of Eatonton and Waycross were 1.41 and 1.52 g/cm^3, respectively.

Across all locations, fertilization significantly decreased bulk density while vegetation control had no significant effect (see Table 3.5; Figure 3.9 and 3.10). There was also a significant increase in bulk density over time. Fertilized plots had a bulk density of 1.45 g/cm^3 while unfertilized plots had a bulk density of 1.47 g/cm^3 in 2021 across all locations. Bulk density of plots with vegetation control were not significantly different from plots without vegetation control. Bulk density increased over time going from 1.43 g/cm^3 in 2001 to 1.46 g/cm^3 in 2021 ($p < 0.001$) when averaged across location and depth (see Figure 3.11). The effects of fertilization ($p = 0.077$), vegetation control ($p = 0.018$), and sampling year ($p < 0.001$) all varied significantly through depth. The effect of fertilization on bulk density went from a difference of -0.04 g/cm^3 in the 0-10 cm layer to a difference of -0.008 g/cm^3 in the 10-30 cm layer and 0.016 g/cm^3 in the 30-50 cm layer when averaged across years. The effect of vegetation control on bulk density went from a difference of -0.046 g/cm^3 in the 0-10 cm layer to a difference of 0.021 g/cm^3 in the 10-30 cm layer, and 0.008 g/cm^3 in the 30-50 cm layer. The change in bulk density from 2001 to 2021 varied through depth going from a difference of -0.02 g/cm^3 in the 0-10 cm layer to a difference of 0.058 g/cm^3 in the 10-30 cm layer and 0.088 g/cm^3 in the 30-50 cm layer. There was no interaction between the effects of fertilization, vegetation control, or time on bulk density.

4.4.4. Soil pH

CAPPS soils were generally acidic with an average pH of 3.92 with a range of 2.75 in the 0-10 cm layer of the 2021 Waycross F plots to 4.83 measured in the 30-50 cm of the 2001

Eatonton H plots (Table 4.4). Soil pH differed significantly between locations ($p < 0.001$) and through depths ($p < 0.001$). Waycross and Tifton had lower pH values on average of around 3.40 and 3.39 while Eatonton and Athens had higher pH values of 4.30 and 4.22, respectively. Measured pH was also lower in the 0-10 cm layer at 3.69 while being slightly higher at the 10-30 cm layer at around 3.93 and 30-50 cm layer at around 4.15 when averaged across treatments and sampling year.

Fertilization plots had lower soil pH while vegetation control had no significant effect (Table 4.5; Figure 4.5 and 4.6). Soil pH was significantly lower in fertilized plots ($p < 0.001$) with average measurements of 3.51 in 2021 compared to unfertilized plots with average measurements of 3.87. Vegetation control had no significant effect on soil pH when averaged across depth, location, and sampling year. There was a decrease in soil pH over time ($p < 0.001$; Figure 4.7). Soil pH went from 4.20 in 2001 to 3.72 in 2021. Depth had a significant interaction with fertilization ($p < 0.001$) and vegetation control ($p = 0.058$). Changes in soil pH from fertilization went from a difference of -0.43 in 0-10 cm to a difference of -0.27 in 10-30 cm and -0.14 in 30-50 cm when averaged across sampling year and location. The effect of vegetation control on soil pH went from a difference of -0.09 in 0-10 cm to a difference of -0.03 in 10-30 and then 0.03 in the 30-50 cm, also averaged across sampling year and location. The effects of fertilization and vegetation control did not change over time. There was also no significant interaction between fertilization and vegetation control treatments.

4.4.5. Soil Base Cations

Exchangeable Ca, Mg, and K levels in soil were relatively low across CAPPS with average concentrations of 0.68, 0.40, and 0.074 cmol+/kg through 50 cm (Table 4.4). Base cation

concentrations significantly differ between locations and depths. Waycross and Tifton had lower exchangeable base cation concentrations compared to Athens and Eatonton. For example, Waycross and Tifton soils had an average Ca concentration of 0.081 and 0.29 cmol+/kg, respectively, while Athens and Eatonton had 0.68 and 1.14 cmol+/kg, respectively. Additionally, exchangeable cation concentrations decreased with depth. For example, Ca decreased from 0.79 cmol+/kg at 0-10 cm to 0.61 cmol+/kg at 10-30 cm to 0.65 cmol+/kg at 30-50 cm.

Ca concentrations in the mineral soil were significantly reduced by fertilization while remaining unaffected by vegetation control (Table 4.5; Figure 4.8 and 4.9). Fertilized plots had significantly lower Ca concentrations ($p < 0.001$) at 0.76 cmolc+/kg in 2001 and 0.35 cmolc+/kg in 2021 compared to unfertilized plots at 0.95 cmolc+/kg in 2001 and 0.71 cmolc+/kg in 2021. Vegetation control did not have a significant effect on Ca. Ca concentration in soil significantly decreased over time ($p < 0.001$) from 0.86 to 0.56 cmol+/kg (Table 4.5; Figure 4.10). Depth has a significant interaction on the effect of fertilization ($p < 0.001$) and vegetation control ($p = 0.056$) on Ca concentrations. Changes in soil Ca from fertilization go from a difference of -0.57 cmolc+/kg in 0-10 cm to a difference of -0.25 cmolc+/kg in 10-30 cm and then 0.006 cmolc+/kg in 30-50 cm. The effect of vegetation control on Ca concentrations ranged from a difference of -0.29 cmolc+/kg in 0-10 cm to a difference of -0.053 cmolc+/kg in 10-30 cm to 0.065 cmolc+/kg in 30-50 cm. Soil Ca changes over time varied significantly through soil depth ($p < 0.001$) going from a difference of -0.37 cmolc+/kg in 0-10 cm to a difference of -0.38 cmolc+/kg in 10-30 cm to -0.16 cmolc+/kg in 30-50 cm. There is an additional interaction between fertilization and time on soil Ca concentration ($p < 0.001$) as the difference between non-Fert and Fert increased from -0.11 cmolc+/kg in 2001 to -0.16 cmolc+/kg in 2021 while vegetation control remained

unaffected. There was no significant interaction between fertilization and vegetation control for the treatment effects on soil Ca. Mg follows a similar pattern as Ca, however, vegetation control significantly reduced Mg concentrations ($p = 0.001$). Plots that received vegetation control had Mg concentrations of 0.46 cmolc+/kg in 2001 and 0.28 cmolc+/kg in 2021 while plots with no vegetation control had Mg concentrations of 0.54 cmolc+/kg in 2001 and 0.38 cmolc+/kg in 2021 (Figure 4.8). Also, the effect of fertilization on Mg concentrations did not differ through depth as with Ca concentrations. K concentrations did not respond to fertilization or vegetation control.

4.4.6. Exchangeable Acidity

Average exchangeable acidity through 50 cm was 1.00 $cmol^{+}/kg$ across all locations (Table 4.4). There was a significant difference in exchangeable acidity across locations ($p = 0.003$) and depths ($p < 0.001$). The Athens and Waycross plots had slightly higher levels of acidity at 1.16 and 1.19 $cmol^{+}/kg$ through 50 cm. Eatonton and Tifton, on the other hand, had slightly less acidity at 0.83 and 1.11 $cmol^{+}/kg$. Acidity also decreased with depth across all locations going from 1.26 $cmol^{+}/kg$ at 0-10 cm to 0.97 cmol+/kg at 10-30 cm and then 0.76 $cmol^{+}/kg$ at 30-50 cm.

Soil acidity significantly increased from the use of fertilization but remained unaffected by vegetation control (Table 4.5; Figure 4.8 and 4.9). Fertilized plots had a significantly higher acidity ($p < 0.001$) with 0.82 cmol+/kg in 2001 and 1.67 cmol+/kg in 2021 compared to unfertilized plots with 0.63 cmol+/kg in 2001 and 0.85 cmol+/kg in 2021. Conversely, vegetation control had no effect on soil acidity. Soil acidity also significantly increased over time ($p < 0.001$) going from 0.72 cmol+/kg in 2001 to 1.20 cmol+/kg in 2021 (Figure 4.10). The effect of

fertilization on soil acidity grew over time ($p < 0.001$) going from a difference of 0.19 cmol+/kg in 2001 to 0.82 cmol+/kg in 2021. Increases in soil acidity from fertilization also decreased significantly through depth ($p = 0.004$) going from an increase of 0.67 cmol+/kg at 0-10 cm to 0.62 cmol+/kg at 10-30 cm and then declining to 0.31 cmol+/kg at 30-50 cm. Although the effect of vegetation control on soil acidity was insignificant across all locations there was a significant change from treatment at depth ($p = 0.016$) going from a gain of 0.05 cmol+/kg at 0-10 cm to a loss of -0.10 cmol+/kg at 10-30 cm, and then -0.17 cmol+/kg at 30-50.

4.5. Discussion

Fertilization amplified soil acidification under pine plantations. By 2021, fertilization lowered pH by 0.4 units, reduced Ca and Mg content by 24 kmol+/ha and increased exchangeable acidity by 56 kmol+/ha in the top 50 cm of soil when treatment effects were averaged across locations. The acidifying effect of fertilization is akin to the results of N addition studies in forests across climatic regimes from tropical to boreal (Bergkvist et al., 1991; Bowman et al., 2008; Chen et al., 2023; Hogberg et al., 2006; Horswill et al., 2008; Lieb et al., 2011; Lu et al., 2014; Mao et al., 2017; Tian et al., 2015). There are contradictory findings on the effects of N fertilization on soil acidity such as a tropical plantation study which found no effect from fertilization; however, this study was in the minority (Huang et al 2021). The loss of base cations in N fertilized soils has been correlated with an increase in NO_3- concentration which can leach Ca, Mg, and K (Gundersen et al., 2006; Ingerslev, 1997; Lucase et al., 2011). Another potential explanation of the effect of fertilization on acidity is increased nutrient uptake as fertilized trees tend to grow better. In 2021, the contribution to acidification from base cation uptake in living and dead biomass was 21 kmol+/ha higher in fertilized plots when averaged across locations

(Albaugh et al., 2008; Zhao et al., 2022). There is also an additional 1 kmol+/ha of acidification from base cation accumulation in the forest floor by 2021. This means 39% of the increase in acidity from fertilization can be explained by biomass uptake with the remaining caused by leaching. For example, if all NO_3^- from fertilization (~895 kg-N/ha) leached it could contribute up to 66 kmol+/ha of acidification. The actual contribution of leaching from fertilization is lower as some portion of NO_3^- is taken up by the trees. Regardless, it is likely a large majority of acidification from fertilization is from NO_3^- leaching.

Vegetation control did not enhance or alleviate soil acidification. Soil pH, base cations, and exchangeable acidity were relatively consistent across plots with and without vegetation control. This is consistent with short-term studies that have shown no effect from vegetation control in plantation forests (Shen et al., 2018; Zhao et al., 2011). In contrast, there are examples of acidification being induced from vegetation control in plantations (Zhang et al., 2022). What is surprising is vegetation control has caused an increase in cation uptake via increases in biomass growth that did not affect soil cation content. Furthermore, there is not sufficient evidence to indicate vegetation control is causing acidification in pine plantations.

Significant soil acidification occurred under the pine plantation management of this study. Decreases in soil pH, declines in exchangeable base cations, and increase in exchangeable acidity were observed over the twenty-year sampling period. Across all locations, soil pH decreased by ~0.5 units while Ca and Mg content decreased by ~10 kmol+/ha and exchangeable acidity increased by ~35 kmol+/ha over 0 to 50 cm of soil. Patterns of long-term acidification have been previously observed in coniferous forests in North America with pH decreases as high as 1.3 units, reduction in Ca and Mg content of 20-44 kmol+/ha, and increases in exchangeable acidity of 37 kmol+/ha down to 60 cm over 20-46 years (Brand et al., 1986; Knoepp et al., 1994;

Richter et al., 1994). This study, however, is the first recorded instance in intensively managed loblolly pine plantations. Hardwood forests in North America have also exhibited long-term acidification with a decrease in pH up to 0.9 units and a reduction in Ca and Mg concentration up to 4.06 cmol+/kg after 30 years (Bailey et al., 2005). Conversely, there are examples of hardwood forests resisting soil acidification in the mineral horizons (Courchesne et al., 2005; Trettin et al., 1999). Previously, much of the change seen in forest soil acidity in North America has been attributed to natural processes, mainly biomass accumulation of nutrients. The contribution of natural processes to acid accumulation in forests has been as high as 62% with the remaining percentage being attributed to acid deposition (Markewitz et al., 1998). For CAPPS, based on regional biomass cation concentrations (Albaugh et al., 2008; Zhao et al., 2022), average uptake of base cations into living and dead biomass accounted for 19 kmol+/ha or 54% of accumulated acidification over time across all locations and treatments. Base cation accumulation in forest floor accounted for an additional 5 kmol+/ha or 14% of accumulated acidification after twenty years. This means up to 68% of soil acidification gained over time might be explained by internal uptake of base cations by trees which substantially exceeds acid inputs from deposition which equated to 1.4 kmol+/ha or 4% of accumulated acidity over twenty years (NADP, 2022). The unaccounted for acid inputs could be related to uptake of other cations such as NH_4^+ or other organic acids released by vegetation over time.

4.6.Conclusion

There is clear evidence that pine plantation management in this study, which could be considered intensive, acidified a range of soil types across Georgia. High levels of fertilization in this study (1790 kg-N/ha over ~30 years) in the form of NH_4NO_3 caused significantly lower

pH, reduced base cation contents, and increased exchangeable soil acidity, which are all indicative of soil acidification. A large portion of accumulated soil acidification in fertilized plots can be related to enhanced growth of biomass, however, there appears to be another process causing a substantial portion of acidification, potentially leaching. In contrast, vegetation control, which also enhanced stand level growth, caused little to no acidification so was not additive with fertilization. Pine plantations, like forest growth in general, appears to have a general acidifying effect on soil over the span of a whole rotation. A majority of this acidification can be explained by uptake of nutrient cations in biomass that is enhanced by high rates of N fertilization but N leaching as NO_3 also can play a role. How the observed acidification across multiple soil types from repeated N fertilization and complete competing vegetation control in the pine plantations of this study translates into more typical operational management (lower N fertilizer inputs and limited vegetation control) for the southeastern US region over multiple future rotations is a critical question. We need to better understand if base cation deficiencies will develop into potential productivity issues in the future or are there biogeochemical forest processes (i.e., mineral weathering or deep root uptake) that will buffer these deficiencies.

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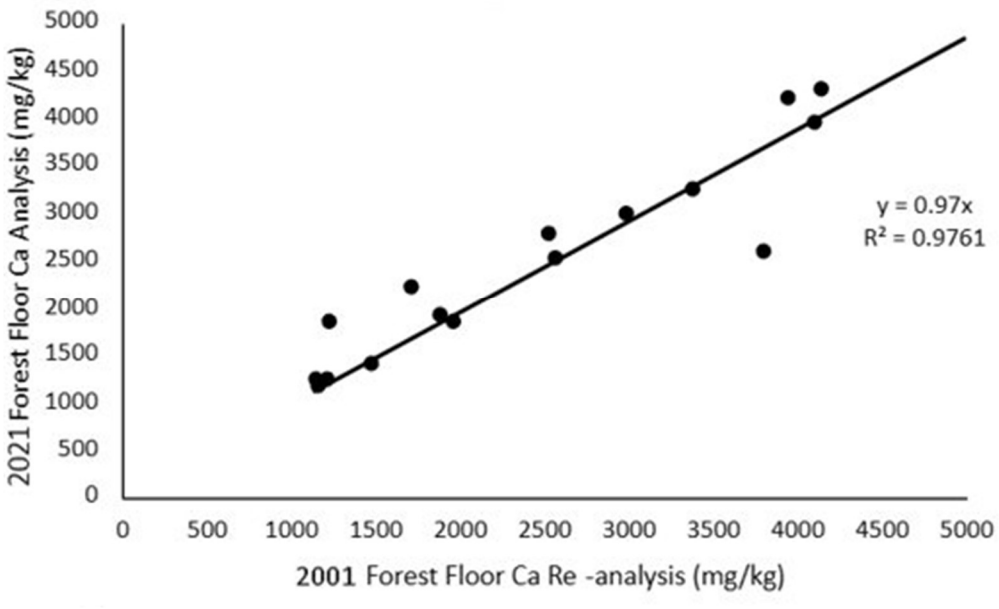
(a)



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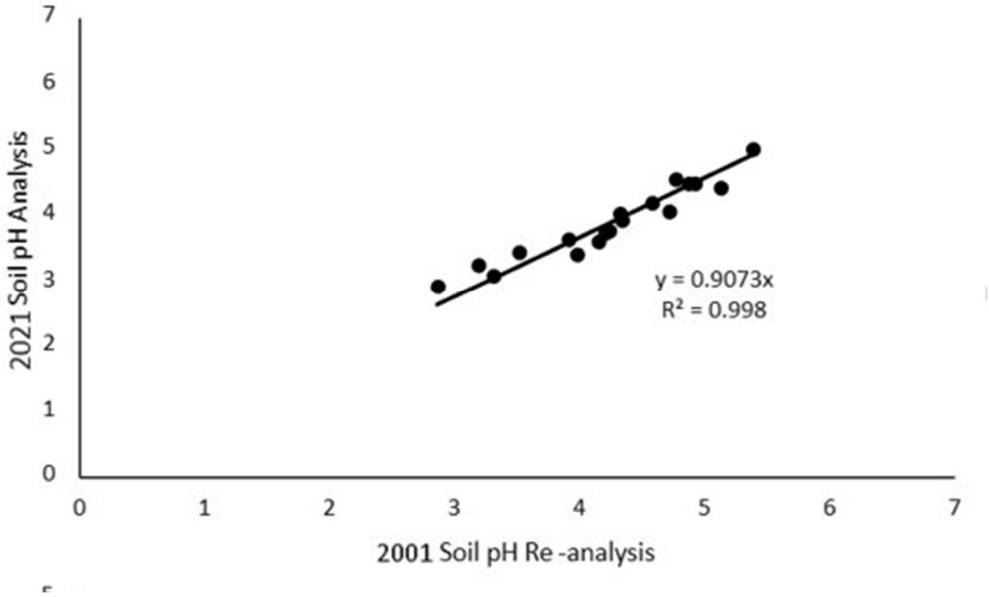

Figure 4.1 Comparison of 2001 analysis and 2021 re-analysis of (a) forest floor cations (e.g., Ca) and (b) soil pH_{salt} of a sub-sample 2001 soil samples (n = 18).

Table 4.1 Mean tree biomass (Mg/ha (standard deviation)) and nutrient concentration estimates by component averaged across locations and treatments. Includes living and dead biomass pools for 2001 and 2021.

Component	Living Biomass[1]		Dead Biomass[1]		Ca[2]	Mg[2]	K[2]
	2001	2021	2001	2021	(%)	(%)	(%)
Stem	71 (45)	246 (56)	2 (4)	70 (50)	0.07	0.03	0.08
Bark	8 (4)	18 (4)	0 (0)	6 (4)	0.19	0.04	0.08
Branch	14 (6)	25 (5)	0 (1)	8 (6)	0.21	0.06	0.18
Foliage	5 (1)	5 (1)	0 (0)	2 (1)	0.17	0.09	0.32

[1] Biomass estimates for each component taken from Zhao (2022)

[2] Base cation concentration estimates for *Pinus taeda* taken from Albaugh (2008)

Table 4.2 Summary of forest floors base cation concentrations (Ca, Mg, and K) for CAPPS plots in Georgia USA. Included are back-transformed means and log-transformed means (standard deviation)

Location	Year	Trt	Ca (mg/kg)	log(Ca)	Mg (mg/kg)	log(Mg)	K (mg/kg)	log(K)
Athens	2021	C	2783.10	7.93 (0.28)	1452.45	7.28 (0.43)	1323.96	7.19 (0.51)
		F	2053.50	7.63 (NA)	2102.25	7.65 (NA)	1713.75	7.45 (NA)
		H	2261.56	7.72 (0.3)	1059.59	6.97 (0.48)	878.35	6.78 (0.64)
		HF	2218.57	7.7 (0.09)	1224.86	7.11 (0.33)	1159.87	7.06 (0.26)
Eatonton	2001	C	4676.41	8.45 (0.3)	771.04	6.65 (0.29)	693.46	6.54 (0.39)
		F	4616.70	8.44 (0.32)	849.10	6.74 (0.26)	863.42	6.76 (0.22)
		H	2994.96	8 (0.38)	568.35	6.34 (0.37)	638.76	6.46 (0.32)
		HF	2729.36	7.91 (0.26)	594.70	6.39 (0.14)	659.22	6.49 (0.29)
	2021	C	3832.90	8.25 (0.09)	779.58	6.66 (0.15)	795.41	6.68 (0.32)
		F	2819.86	7.94 (0.3)	731.28	6.59 (0.38)	706.65	6.56 (0.14)
		H	2001.65	7.6 (0.21)	535.70	6.28 (0.24)	700.31	6.55 (0.18)
		HF	1897.93	7.55 (0.19)	481.05	6.18 (0.14)	667.82	6.5 (0.41)
Tifton	2021	C	2168.56	7.68 (0.18)	278.20	5.63 (0.42)	425.58	6.05 (0.16)
		F	1503.94	7.32 (0.2)	303.46	5.72 (0.33)	382.54	5.95 (0.06)
		H	1602.42	7.38 (0.28)	188.88	5.24 (0.73)	353.42	5.87 (0.22)
		HF	1313.64	7.18 (0.05)	184.68	5.22 (0.4)	299.14	5.7 (0.04)
Waycross	2001	C	3357.97	8.12 (0.28)	745.55	6.61 (0.2)	411.73	6.02 (0.21)
		F	2027.45	7.61 (0.53)	522.68	6.26 (0.29)	454.75	6.12 (0.27)
		H	2575.08	7.85 (0.46)	575.69	6.36 (0.27)	360.44	5.89 (0.24)
		HF	1557.71	7.35 (0.49)	419.24	6.04 (0.27)	391.99	5.97 (0.21)
	2021	C	1242.44	7.12 (0.32)	340.80	5.83 (0.14)	111.87	4.72 (0.52)
		F	847.95	6.74 (0.38)	255.36	5.54 (0.21)	37.87	3.63 (0.9)
		H	1198.57	7.09 (0.31)	285.25	5.65 (0.24)	95.01	4.55 (0.2)
		HF	690.35	6.54 (0.33)	231.18	5.44 (0.34)	36.63	3.6 (0.36)

Table 4.3 p-Values for ANOVA of forest floor base cation concentrations (Ca, Mg, and K).

	Ca	Mg	K
Fert	<0.001	0.143	0.508
Herb	<0.001	<0.001	0.274
Year	<0.001	<0.001	<0.001
Location	<0.001	<0.001	<0.001
Fert:Herb	0.301	0.696	0.953
Fert:Year	0.928	0.819	0.030
Herb:Year	0.079	0.897	0.526
Fert:Herb:Year	0.192	0.714	0.494

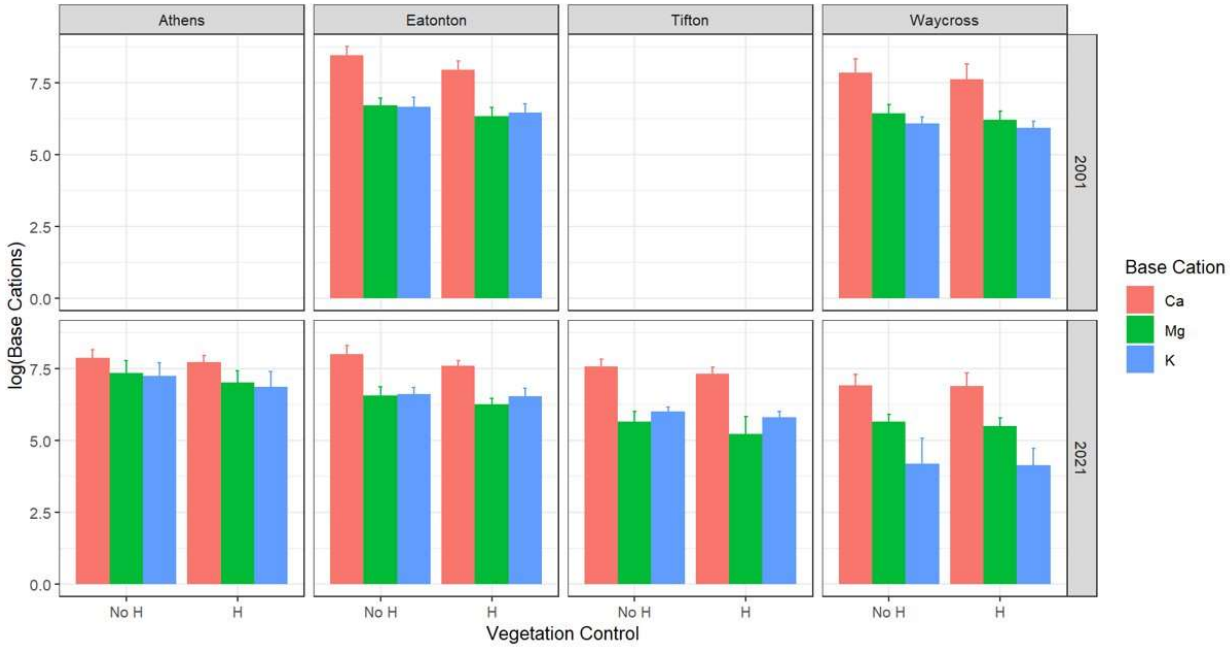

Figure 4.2 Log-transformed cation concentrations in forest floor (means±1SD) under vegetation control (H and HF) and non-vegetation control treatments (C and F).

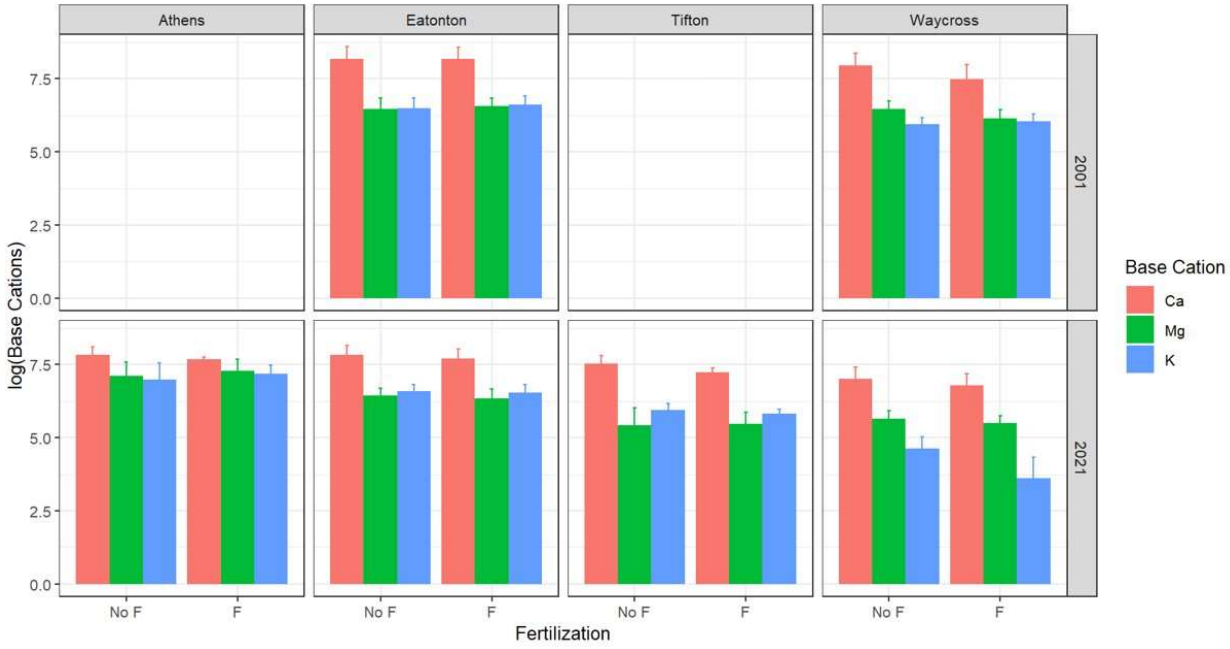

Figure 4.3 Log-transformed cation concentrations in forest floor (means±1 SD) under fertilization (F and HF) and non-fertilization treatments (C and H).

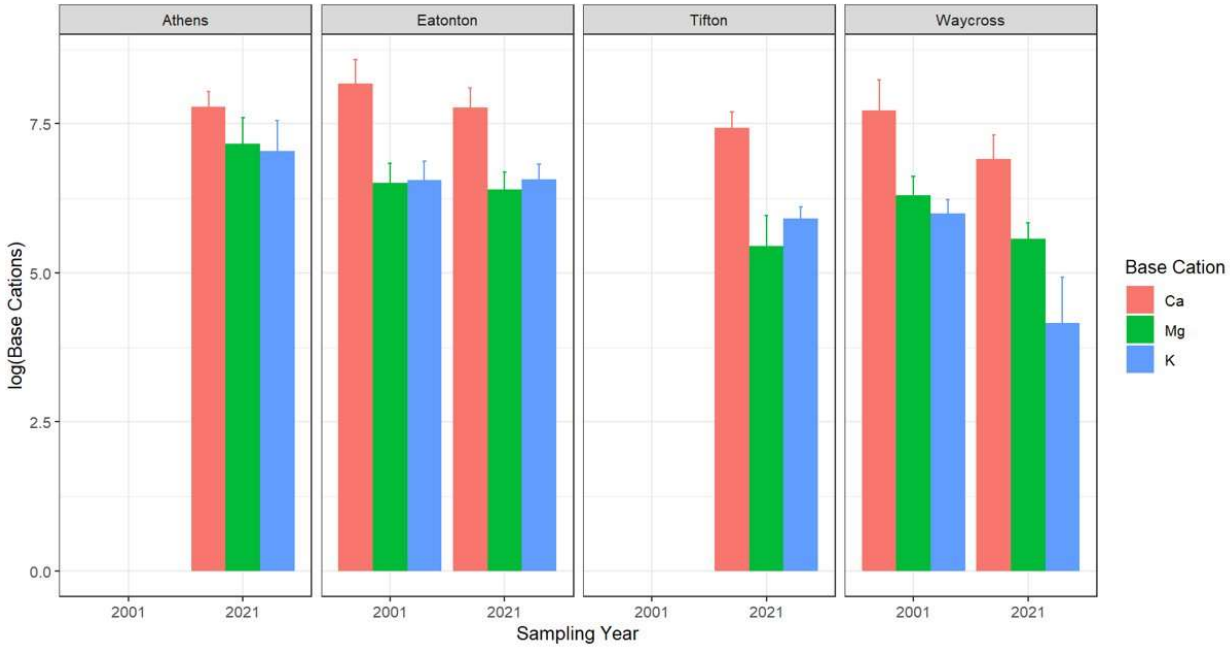

Figure 4.4 Log-transformed cation concentrations in forest floor (means±1 SD) between sampling years averaged across treatments.

Table 4.4 Summary of soil pH, exchangeable acidity, and base cation concentrations (Ca, Mg, and K) for CAPPS plots in Georgia USA. Included are back-transformed means and log-transformed means (standard deviation).

Location	Year	Trt	Depth	pH_{Salt}	Exc. Acid (cmol+/kg)	log(Exc. Acid)
Athens	2021	C	0	4.32 (0.1)	0.69	-0.36 (0.24)
			10	4.26 (0.14)	0.71	-0.35 (0.68)
			30	4.34 (0.17)	0.72	-0.32 (0.69)
		F	0	3.51 (NA)	2.85	1.05 (NA)
			10	4.68 (NA)	2.75	1.01 (NA)
			30	3.89 (NA)	2.13	0.75 (NA)
		H	0	4.37 (0.22)	0.76	-0.27 (0.11)
			10	4.47 (0.34)	0.81	-0.21 (0.22)
			30	4.37 (0.2)	0.59	-0.53 (0.55)
		HF	0	3.52 (0.26)	2.94	1.08 (0.2)
			10	3.76 (0.11)	2.02	0.7 (0.14)
			30	4.09 (0.08)	1.06	0.06 (0.29)
Eatonton	2001	C	0	4.82 (0.27)	0.35	-1.04 (0.53)
			10	4.78 (0.4)	0.26	-1.33 (1.2)
			30	4.7 (0.4)	0.38	-0.97 (1.21)
		F	0	4.4 (0.25)	0.37	-1 (1.2)
			10	4.52 (0.33)	0.40	-0.91 (0.84)
			30	4.65 (0.35)	0.29	-1.24 (1.4)
		H	0	4.69 (0.18)	0.34	-1.08 (0.27)
			10	4.66 (0.17)	0.27	-1.3 (0.56)
			30	4.83 (0.29)	0.26	-1.36 (0.67)
		HF	0	4.26 (0.22)	0.59	-0.53 (0.63)
			10	4.53 (0.25)	0.28	-1.27 (0.91)
			30	4.64 (0.24)	0.32	-1.13 (0.88)
	2021	C	0	4.35 (0.27)	0.35	-1.06 (0.43)
			10	4.25 (0.36)	0.52	-0.65 (0.75)
			30	4.24 (0.33)	0.47	-0.75 (0.89)
		F	0	3.66 (0.37)	1.81	0.59 (0.72)
			10	3.81 (0.23)	1.97	0.68 (0.56)
			30	4.02 (0.18)	1.03	0.03 (0.79)
		H	0	3.97 (0.45)	0.83	-0.19 (0.76)
			10	4.04 (0.35)	0.55	-0.61 (0.6)
			30	4.2 (0.28)	0.41	-0.89 (0.52)
		HF	0	3.41 (0.17)	2.00	0.69 (0.4)
			10	3.72 (0.15)	1.99	0.69 (0.24)

			30	3.92 (0.15)	1.10	0.09 (0.46)
Tifton	2021	C	0	3.4 (0.23)	0.81	-0.21 (0.23)
			10	3.64 (0.12)	0.70	-0.36 (0.24)
			30	3.57 (0.08)	0.87	-0.14 (0.13)
		F	0	2.93 (0.16)	1.62	0.48 (0.52)
			10	3.15 (0.16)	1.00	0 (0.45)
			30	3.3 (0.06)	1.09	0.09 (0.84)
		H	0	3.35 (0.22)	0.98	-0.02 (0.25)
			10	3.68 (0.22)	0.71	-0.34 (0.46)
			30	3.59 (0.16)	0.92	-0.09 (0.41)
		HF	0	2.9 (0.1)	1.55	0.44 (0.21)
			10	3.11 (0.01)	1.76	0.57 (0.39)
			30	3.24 (0.04)	1.92	0.65 (0.59)
Waycross	2001	C	0	3.31 (0.57)	1.40	0.34 (0.47)
			10	3.63 (0.33)	1.01	0.01 (0.21)
			30	4.08 (0.14)	0.67	-0.4 (0.17)
		F	0	2.9 (0.23)	1.81	0.59 (0.17)
			10	3.41 (0.2)	1.15	0.14 (0.19)
			30	4.05 (0.14)	0.64	-0.45 (0.09)
		H	0	3.03 (0.36)	1.43	0.36 (0.3)
			10	3.56 (0.29)	0.92	-0.09 (0.26)
			30	4 (0.2)	0.60	-0.52 (0.2)
		HF	0	3.06 (0.25)	1.95	0.67 (0.35)
			10	3.52 (0.34)	0.99	-0.01 (0.06)
			30	4.12 (0.13)	0.57	-0.56 (0.1)
	2021	C	0	2.92 (0.3)	2.14	0.76 (0.31)
			10	3.36 (0.2)	0.89	-0.12 (0.24)
			30	3.76 (0.11)	0.72	-0.33 (0.42)
		F	0	2.75 (0.12)	1.62	0.48 (0.34)
			10	3.15 (0.15)	1.29	0.25 (0.17)
			30	3.59 (0.05)	0.96	-0.04 (0.19)
		H	0	2.84 (0.14)	1.59	0.46 (0.19)
			10	3.32 (0.32)	0.98	-0.02 (0.16)
			30	3.8 (0.14)	0.66	-0.42 (0.26)
		HF	0	2.76 (0.14)	1.74	0.55 (0.14)
			10	3.23 (0.15)	1.28	0.25 (0.18)
			30	3.71 (0.1)	0.80	-0.22 (0.09)

Table 4.4 continued. Summary of soil pH, exchangeable acidity, and base cation concentrations (Ca, Mg, and K) for CAPPS plots in Georgia USA. Included are back-transformed means and log-transformed means (standard deviation).

Location	Year	Trt	Depth (cm)	Ca (cmol+/kg)	log(Ca)	Mg (cmol+/kg)	log(Mg)	K (cmol+/kg)	log(K)
Athens	2021	C	0	1.32	0.28 (0.31)	0.61	-0.5 (0.29)	0.15	-1.89 (0.3)
			10	0.61	-0.5 (0.33)	0.55	-0.59 (0.43)	0.13	-2.04 (0.48)
			30	0.50	-0.7 (0.35)	0.75	-0.29 (0.48)	0.12	-2.09 (0.59)
		F	0	0.37	-0.99 (NA)	0.21	-1.56 (NA)	0.12	-2.14 (NA)
			10	0.19	-1.64 (NA)	0.16	-1.86 (NA)	0.12	-2.1 (NA)
			30	0.47	-0.76 (NA)	0.55	-0.59 (NA)	0.17	-1.76 (NA)
		H	0	0.84	-0.18 (0.28)	0.37	-0.99 (0.37)	0.13	-2.06 (0.26)
			10	0.71	-0.34 (0.41)	0.58	-0.54 (0.36)	0.13	-2.06 (0.51)
			30	0.60	-0.52 (0.48)	0.90	-0.11 (0.25)	0.17	-1.79 (0.45)
		HF	0	0.17	-1.78 (0.14)	0.10	-2.28 (0.32)	0.07	-2.65 (0.06)
			10	0.18	-1.71 (0.25)	0.10	-2.31 (0.45)	0.09	-2.44 (0.14)
			30	0.83	-0.19 (0.07)	0.38	-0.96 (0.35)	0.20	-1.6 (0.39)
Eatonton	2001	C	0	1.86	0.62 (0.57)	0.71	-0.34 (0.31)	0.16	-1.82 (0.61)
			10	1.14	0.13 (0.63)	0.75	-0.28 (0.25)	0.09	-2.41 (0.78)
			30	0.88	-0.13 (0.6)	0.98	-0.02 (0.25)	0.07	-2.65 (0.73)
		F	0	1.28	0.25 (0.34)	0.54	-0.62 (0.37)	0.16	-1.81 (0.38)
			10	1.17	0.16 (0.45)	0.77	-0.26 (0.41)	0.10	-2.35 (0.5)
			30	1.22	0.2 (0.67)	1.07	0.07 (0.47)	0.08	-2.48 (0.5)
		H	0	1.45	0.37 (0.52)	0.55	-0.61 (0.36)	0.14	-1.99 (0.51)
			10	1.20	0.18 (0.45)	0.74	-0.3 (0.21)	0.09	-2.4 (0.54)
			30	1.05	0.05 (0.49)	0.95	-0.05 (0.27)	0.09	-2.44 (0.53)
		HF	0	0.92	-0.09 (0.54)	0.34	-1.08 (0.55)	0.13	-2.03 (0.42)
			10	1.13	0.12 (0.39)	0.69	-0.38 (0.29)	0.11	-2.22 (0.61)
			30	1.03	0.03 (0.31)	0.98	-0.02 (0.16)	0.10	-2.32 (0.65)
	2021	C	0	1.71	0.54 (0.53)	0.68	-0.39 (0.26)	0.12	-2.14 (0.53)
			10	0.93	-0.08 (0.59)	0.66	-0.42 (0.16)	0.08	-2.47 (0.76)
			30	0.86	-0.15 (0.75)	0.91	-0.09 (0.23)	0.06	-2.81 (0.86)
		F	0	0.45	-0.79 (0.85)	0.21	-1.56 (0.9)	0.09	-2.43 (0.43)
			10	0.36	-1.03 (0.96)	0.19	-1.65 (1.08)	0.07	-2.69 (0.44)
			30	0.98	-0.02 (0.77)	0.58	-0.55 (0.82)	0.06	-2.75 (0.59)
		H	0	0.83	-0.18 (0.87)	0.33	-1.1 (0.82)	0.10	-2.3 (0.34)
			10	0.85	-0.17 (0.7)	0.48	-0.73 (0.65)	0.08	-2.59 (0.68)
			30	0.98	-0.03 (0.39)	0.82	-0.19 (0.34)	0.06	-2.77 (0.52)
		HF	0	0.19	-1.65 (0.21)	0.09	-2.44 (0.21)	0.07	-2.63 (0.28)
			10	0.19	-1.66 (0.64)	0.09	-2.42 (0.52)	0.07	-2.69 (0.43)

			30	0.75	-0.29 (0.41)	0.40	-0.93 (0.42)	0.07	-2.64 (0.49)
Tifton	2021	C	0	0.82	-0.2 (0.63)	0.23	-1.49 (0.34)	0.06	-2.77 (0.3)
			10	0.22	-1.53 (0.92)	0.08	-2.49 (0.46)	0.02	-3.97 (0.34)
			30	0.25	-1.41 (0.99)	0.11	-2.23 (0.69)	0.02	-3.88 (0.29)
		F	0	0.11	-2.19 (0.69)	0.07	-2.69 (0.58)	0.03	-3.42 (0.44)
			10	0.03	-3.4 (0.75)	0.03	-3.65 (1.05)	0.01	-4.36 (0.46)
			30	0.03	-3.5 (1.04)	0.03	-3.52 (1.58)	0.01	-4.45 (0.73)
		H	0	0.38	-0.96 (0.41)	0.12	-2.09 (0.34)	0.04	-3.33 (0.15)
			10	0.15	-1.87 (0.85)	0.06	-2.74 (0.67)	0.02	-4.03 (0.27)
			30	0.17	-1.74 (1.05)	0.09	-2.41 (0.95)	0.02	-3.84 (0.48)
		HF	0	0.10	-2.27 (0.67)	0.05	-2.99 (0.29)	0.03	-3.44 (0.05)
			10	0.04	-3.31 (0.8)	0.02	-3.72 (0.68)	0.02	-3.83 (0.44)
			30	0.04	-3.13 (0.81)	0.02	-3.71 (0.93)	0.05	-2.92 (0.62)
Waycross	2001	C	0	0.18	-1.7 (0.78)	0.08	-2.59 (0.6)	0.04	-3.28 (0.18)
			10	0.06	-2.85 (0.92)	0.02	-3.85 (0.55)	0.01	-4.36 (0.5)
			30	0.04	-3.25 (0.73)	0.02	-4.01 (0.52)	0.01	-4.49 (0.44)
		F	0	0.10	-2.33 (0.97)	0.03	-3.42 (0.71)	0.02	-3.82 (0.81)
			10	0.10	-2.31 (0.59)	0.03	-3.38 (0.53)	0.03	-3.65 (0.38)
			30	0.02	-3.7 (0.51)	0.01	-4.49 (0.37)	0.01	-4.78 (0.33)
		H	0	0.10	-2.34 (1.46)	0.04	-3.3 (1.06)	0.02	-3.8 (0.94)
			10	0.08	-2.5 (0.82)	0.02	-3.73 (0.44)	0.01	-4.35 (0.35)
			30	0.04	-3.17 (0.72)	0.02	-4.11 (0.4)	0.01	-4.6 (0.5)
		HF	0	0.07	-2.7 (1.05)	0.02	-3.74 (0.69)	0.02	-3.91 (0.75)
			10	0.05	-3.05 (0.92)	0.02	-3.91 (0.46)	0.02	-4.03 (0.49)
			30	0.02	-3.7 (0.37)	0.01	-4.48 (0.29)	0.01	-4.75 (0.36)
	2021	C	0	0.09	-2.36 (0.63)	0.06	-2.75 (0.53)	0.04	-3.24 (0.21)
			10	0.05	-3.01 (1.16)	0.02	-3.78 (0.46)	0.02	-4.03 (0.46)
			30	0.03	-3.38 (0.91)	0.02	-4.12 (0.49)	0.01	-4.65 (0.47)
		F	0	0.05	-2.92 (0.51)	0.03	-3.37 (0.45)	0.02	-3.76 (0.48)
			10	0.04	-3.22 (1.02)	0.02	-3.87 (0.89)	0.01	-4.27 (0.65)
			30	0.03	-3.37 (0.56)	0.01	-4.29 (0.43)	0.01	-4.73 (0.39)
		H	0	0.08	-2.49 (0.92)	0.04	-3.29 (0.69)	0.02	-3.79 (0.59)
			10	0.07	-2.69 (1.04)	0.02	-3.79 (0.69)	0.01	-4.28 (0.64)
			30	0.03	-3.44 (0.88)	0.02	-4.18 (0.5)	0.01	-4.64 (0.53)
		HF	0	0.05	-3.04 (0.67)	0.02	-3.7 (0.31)	0.01	-4.31 (0.21)
			10	0.03	-3.46 (0.33)	0.02	-4.13 (0.23)	0.01	-4.6 (0.26)
			30	0.02	-3.89 (0.19)	0.01	-4.54 (0.11)	0.01	-4.87 (0.3)

Table 4.5 p-Values for ANOVA of soil pH, base cation concentrations (Ca, Mg, and K), and exchangeable acidity.

	pH	Ca	Mg	K	Exc. Acid
Fert	<0.001	<0.001	<0.001	0.194	<0.001
Herb	0.267	0.105	0.001	0.217	0.977
Depth	<0.001	<0.001	0.099	<0.001	<0.001
Year	<0.001	<0.001	<0.001	<0.001	<0.001
Location	<0.001	<0.001	<0.001	<0.001	0.003
Fert:Herb	0.585	0.322	0.234	0.823	0.932
Fert:Depth	<0.001	<0.001	0.120	0.006	0.004
Herb:Depth	0.058	0.056	0.070	<0.001	0.016
Fert:Year	0.063	<0.001	<0.001	0.007	<0.001
Herb:Year	0.471	0.175	0.087	0.412	0.541
Depth:Year	0.503	<0.001	0.009	0.025	0.136
Fert:Herb:Depth	0.679	0.265	0.552	0.809	0.263
Fert:Herb:Year	0.639	0.986	0.878	0.786	0.490
Fert:Depth:Year	0.981	0.019	0.125	0.045	0.474
Herb:Depth:Year	0.917	0.668	0.822	0.664	0.714
Fert:Herb:Depth:Year	0.789	0.995	0.947	0.748	0.083

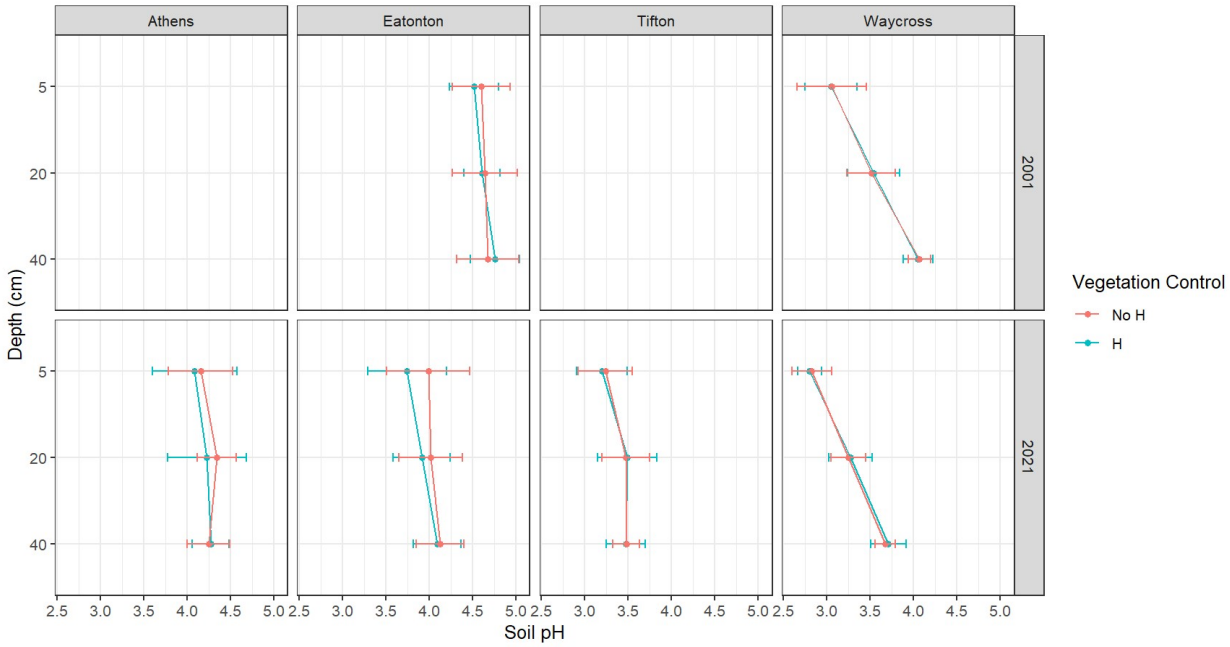

Figure 4.5 Soil pH_{salt} (means±1 SD) under vegetation control (H and HF) and non-vegetation control treatments (C and F).

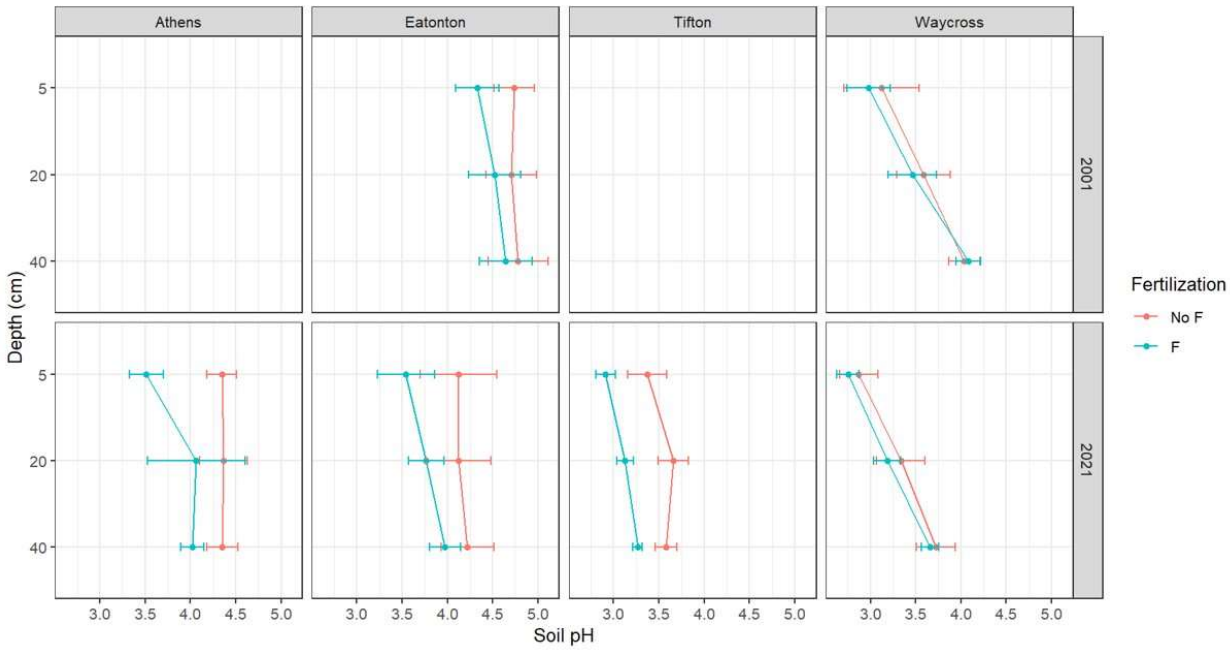

Figure 4.6 Soil pH_{salt} (means±1 SD) under fertilization (F and HF) and non-fertilization treatments (C and H).

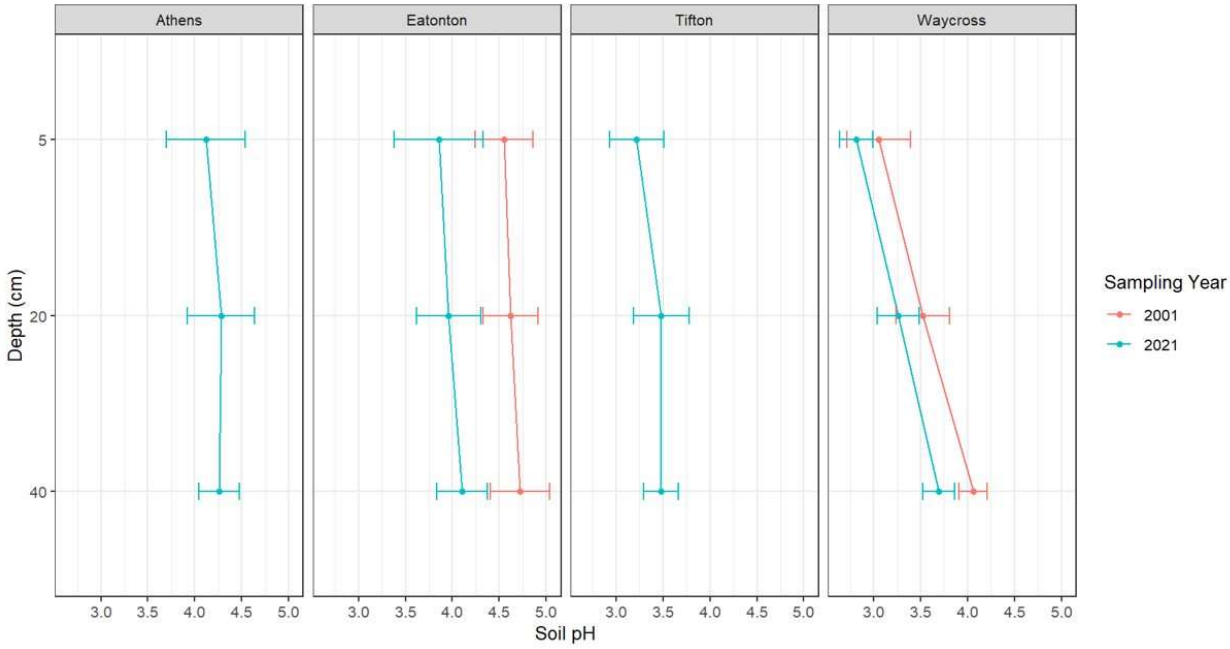

Figure 4.7 Soil pH_{salt} (means±1 SD) between sampling years averaged across treatments.

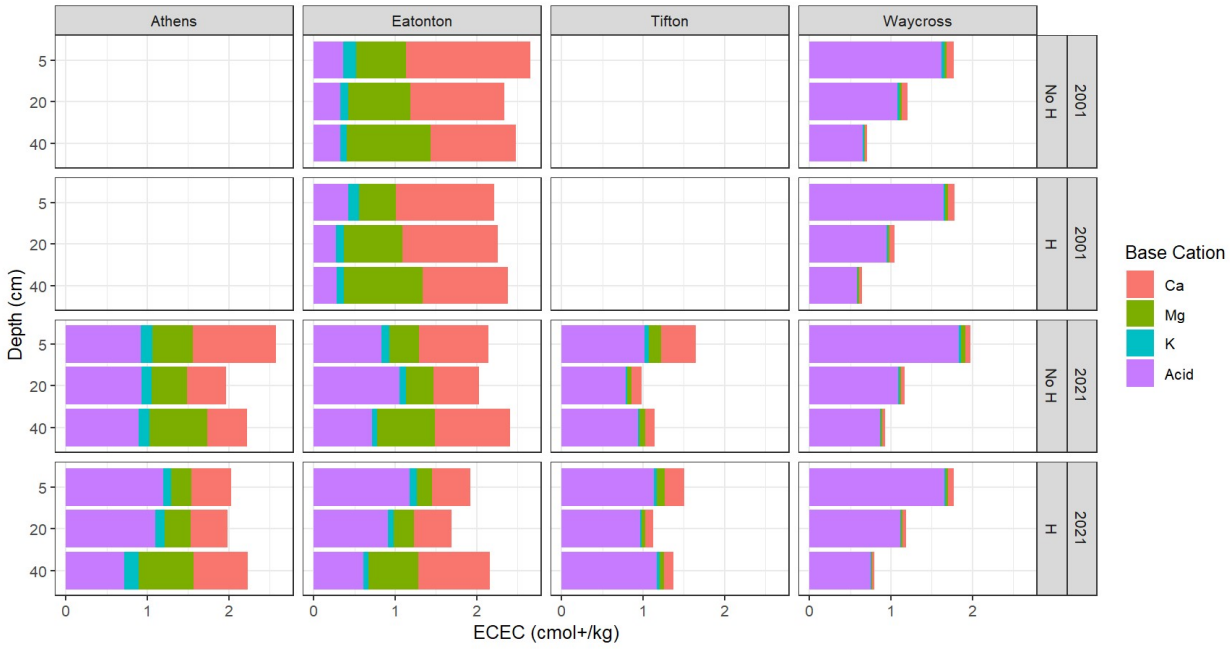

Figure 4.8 Back-transformed soil cation concentrations (means) under vegetation control (H and HF) and non-vegetation control treatments (C and F).

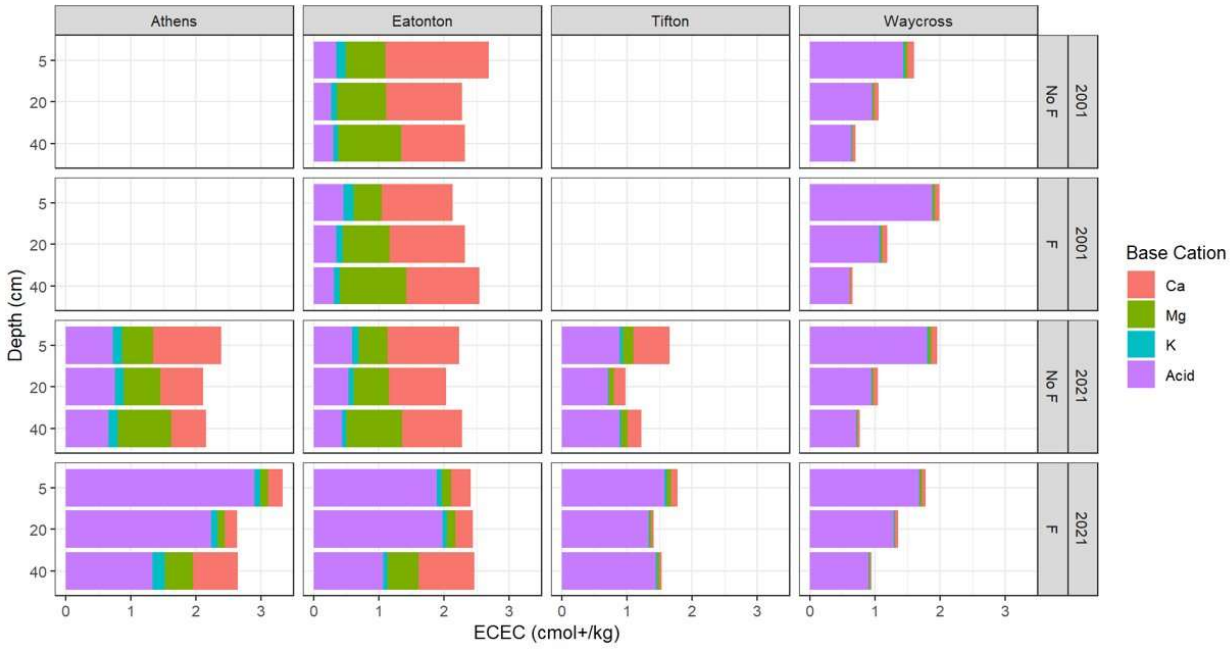

Figure 4.9 Back-transformed soil cation concentrations (means) under fertilization (F and HF) and non-fertilization treatments (C and H).

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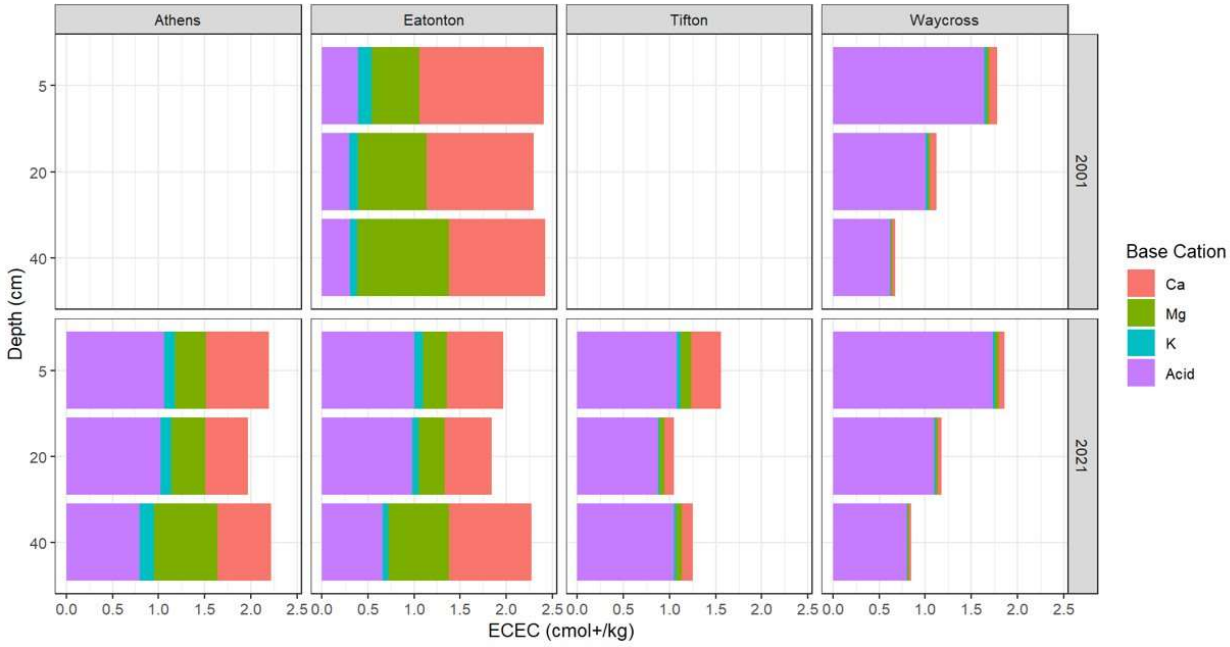

Figure 4.10 Soil cation concentration (means) across sampling years averaged across treatments

CHAPTER V

CONCLUSIONS

The results of this thesis support the hypothesis that plantation management can impact long-term productivity by effecting important soil properties such as C and N content as well as exchange chemistry. In 2021, fertilization did not affect mineral soil C and N content. Instead, the use of fertilizer increased C and N pools in forest floor and biomass. These findings agree with previous CAPPS samplings in 2001 which found mixed effects of fertilization on soil C and N. Conversely, vegetation control resulted in a reduction in soil C and N content while still increasing forest floor and biomass pools. The reduction in soil C and N is likely a function of decreased fine root biomass from loss of understory vegetation. The findings surrounding vegetation control were also in agreement with the 2001 CAPPS samplings which found a reduction in soil C and N from treatment. Lastly, there was no significant change in soil C from mid- to end-rotation. Though there were significant gains in forest floor and biomass C pools resulting in a net gain to ecosystem C. The same cannot be said for soil N, which saw significant gains in Eatonton and losses in Waycross. Though gains in forest floor and biomass N pools result in a net gain in ecosystem N that cannot be explained through estimated N fluxes. This highlights a need to develop more intensive budgets to fully understand the effect of plantation management on belowground processes.

The greater concern over the effect of plantation management on soils seems to be acidification. By end of rotation, fertilization caused a significant decline in soil pH and base cation content while increasing exchangeable acidity. Nutrient budgets show that increased

uptake of base cations in biomass are responsible for the acidifying effect of fertilization on plantation soil. Vegetation control, on the other hand, did not influence soil exchange chemistry as soil pH, base cation content, and exchangeable acidity remain unaffected. Over time plantation soils appear to acidify regardless of treatment. Again, the main acidifying process in plantations appears to be biomass growth. Although there is little evidence of acidification negatively affecting pine productivity, this study highlights a need to continue observing these systems as continued acidification might be detrimental to long-term productivity of these systems.