12-2021

Longleaf Pine Restoration in the South Carolina Sandhills Wiregrass Gap

Jacob W. Murray
jwm4@g.clemson.edu

LONGLEAF PINE RESTORATION IN THE SOUTH CAROLINA SANDHILLS
WIREGRASS GAP

A Thesis
Presented to
the Graduate School of
Clemson University

In Partial Fulfillment
of the Requirements for the Degree
Master of Science
Forest Resources

---

by
Jacob Murray
December 2021

Accepted by:
Dr. Donald Hagan, Committee Chair
Dr. Patrick Hiesl
Dr. Robert Baldwin

ABSTRACT

Longleaf pine restoration has been a topic of great concern and intrigue in the southeast and has taken on new fervor in recent decades as restoration methods continue to develop. Many landowners and forest managers are now pursuing ecological forestry and restoration ecology in great numbers as a new form of land management. As a result, longleaf pine restoration has been extensively investigated through research and applied forest ecology. However, niche regions can often be overlooked, as is the case with the Carolina Sandhills Wiregrass Gap, a region devoid of wiregrass and one that is on the outskirt regions of the historical longleaf pine range. Field studies were conducted pre- and post-harvest during two growing seasons in dense loblolly pine stands, which were actively being converted to longleaf pine habitat through restoration timber harvesting. Located in Camden, South Carolina, the study site was positioned directly in the heart of the Carolina Sandhills Wiregrass Gap. Management operations that manipulate logging slash can have considerable effects on resource availability in this region. In the first study (Chapter 1), we conducted an abiotic environmental inventory for pre- and post-harvest site conditions to identify the effects of logging slash manipulation and soil moisture gradients on ecological trends, including woody fuel loads, water retention rates, and nutrient availability. Soil nutrient availability was considerably low across the site, but discrepancies in how logging slash was manipulated significantly influenced water availability. Slash treatments that resulted in masticated fuel beds retained soil moisture at significantly higher rates than those that removed large amounts of biomass. However, methods that removed biomass did not negatively influence nutrient capital

either, easing concerns about its sustainability as a restoration practice. Based on some of these characteristics, we developed a series of habitat suitability models for longleaf pine restoration across the entirety of the South Carolina Sandhills Wiregrass Gap using spatial modeling. Driven by parcellation and ecological criteria consistent with longleaf-sandhill ecology, models with differently weighted criteria were developed to test for sensitivity and restoration suitability (Chapter 2). The model outputs confirmed that a large portion of the Carolina Sandhills Wiregrass Gap is still suitable for longleaf pine restoration. As a result, the expansion of longleaf pine habitat in this region is not only plausible but is likely if forest initiatives continue to trend towards applied forest ecology. Models, such as those developed in this investigation that can identify potential restoration sites at the parcel level are essential in identifying sites with the highest likelihood of restoration success and productivity. To test for applicability, harvesting productivity was investigated to fill a contemporary gap in empirical knowledge regarding restoration timber harvesting in South Carolina (Chapter 3). In the past century, logging businesses have increased in size, complexity, and computerization, making productivity and industry trends important components to quantify regarding the feasibility of restoration harvesting. Our findings indicate that machine productivity in our study was high on average, but also that the implementation of a two-person logging crew considerably decreased the utilization rates for the knuckle-boom loader. Overall costs were below average, indicating that this type of restoration harvest would be financially feasible for both landowners and logging companies.

ACKNOWLEDGMENTS

There are countless individuals and organizations that deserve acknowledgment regarding the success of this research. First and foremost, I would like to thank my committee members, Dr. Donald Hagan, Dr. Patrick Hiesl, and Dr. Robert Baldwin, for their tireless efforts in providing insight and guidance in these endeavors. Each committee member was able to provide input and direction tailored to their specialized fields, all of which I deemed critical to the success of this research as a whole. I want to acknowledge my committee advisor, Dr. Donald Hagan, specifically, as he was an extremely influential individual in my pursuit of higher education and my growth as a scientist and educator. As somebody who also obtained an undergraduate degree at Clemson University, I cannot stress the level of professionalism and care that all three of my committee members exhibit for both their graduate and undergraduate students.

I am also very grateful for all of the support provided by Clemson's Department of Forestry and Environmental Conservation. Many of the professors who helped me along the way are not mentioned by name in this section due to a lack of time and space. Nonetheless, they were highly influential and should know that their direction and insight helped mold my direction in this field. Dr. Wayne Chao was one of those people. His experience and expertise in dealing with equipment and interpreting our findings was essential and I am grateful for all of the help. I should also acknowledge several other graduate students who helped me understand the graduate school process and navigate my way through it. Ellie Johnson, Manisha Parajuli, Keith Phelps, Emily Oakman, Chip Byrd, and many of my other peers helped me manage and gauge my progress along the way. They were continued sources of support and motivation. From peer-reviewing papers to helping me conduct fieldwork, I appreciate all of the help that they provided.

I would also like to thank the Margaret Lloyd Foundation, especially the current chairman of the foundation and one of my committee members, Dr. Baldwin, as a source of funding for this project and a source of insight into the long-term goals of the property. Additionally, as the education specialist on the property, Leoncia Cruz was instrumental in helping me schedule my fieldwork, continually provided me access to equipment and various other resources in Camden. Moreover, I would like to thank Coy Myers and everybody at Forest Land Management Inc., who were vital at helping me set the stage for this project and field my numerous questions. They helped prepare the property and scheduled all of our silvicultural operations, meaning that without their help, many of our research questions would have gone unanswered. Our logging crew also deserves recognition for going above and beyond what any average timber harvest would require. With their participation in the project, they helped add a great deal of information to the study. Not only were they accommodating and diligent in getting our treatments correct, but they also allowed us to film various aspects of the operation, which was in many ways an inconvenience to how they usually operated.

Finally, I owe a great deal of my success to my fiancée, Mackenzie, who, without her continued support, none of this would be possible. She acts as a constant source of encouragement and inspiration and is unwavering in her dedication to our success. I would also like to note that much of this project occurred during the COVID-19 outbreak, which provided many new challenges. However, all of the aforementioned people and

many of the unnamed contributors were able to overcome difficult circumstances and were steadfast in their dedication to the success of this project.

TABLE OF CONTENTS

Table of Contents (Continued)

Page

Table of Contents (Continued)

Page

LIST OF TABLES

List of Tables (Continued)

Page

LIST OF FIGURES

List of Figures (Continued)

Page

List of Figures (Continued)

Page

List of Figures (Continued)

Page

Page

CHAPTER ONE

ABIOTIC RESPONSES TO LONGLEAF PINE RESTORATION METHODS IN THE SOUTH CAROLINA SANDHILLS

Abstract

Silvicultural treatments that mimic natural disturbance regimes are commonly used to restore early successional plant communities. In longleaf pine systems, silvicultural applications that manipulate woody fuel loading and the structural composition of the landscape can have both short and long-term effects on the functionality of an ecosystem. Longleaf pine environments developed over a wide range of habitats, and the abiotic responses to different restoration approaches can differ significantly depending on the ecological characteristics present. Longleaf-wiregrass communities have been the historically dominant community type throughout much of the southeast, but the restoration methodology used to reestablish these communities has been widespread. This variation has led to skepticism regarding what restoration practices are appropriate and sustainable. This study aimed to understand how overstory manipulation and slash management affect woody fuel loading, water retention rates, and nutrient availability post-harvest, all of which are essential components to long-term restoration success. Our findings indicate that fuel reduction methods that resulted in masticated fuel beds significantly increased the amount of fine woody debris, subsequently increasing surface fuel compaction and soil moisture retention rates. Masticated fuel beds increased water retention by 37% and 41% on average compared to conventional harvesting and biomass harvesting, for example. Correspondingly, drier

sites exacerbated these findings with a 45% increase in soil water tension from our mesic to our xeric sites. However, there were no consistent trends for nutrient stocks across treatments, indicating that the intensity to which harvesting slash is manipulated may not have any short-term impact on nutrient availability for vegetation. These findings will potentially act as a baseline for future studies that will evaluate long-term ecological responses to restoration disturbances in the region.

1.1 Introduction

In South Carolina, the Coastal Plain encompasses approximately two-thirds of the state and is characterized by low elevations and a slight but consistent slope towards the coast (Griffith et al., 2002; Murphy, 2016; Van Lear and Jones, 1987). The Southeastern plains were historically dominated by longleaf pine (*Pinus palustris* Mill.) stands with small patches of oak-hickory-pine and a diverse assortment of age classes, vertical structure, and plant and animal species (Griffith et al., 2002; Van Lear and Jones, 1987). Wetter sites exhibited a southern mixed-forest type with various other species such as American beech (*Fagus grandifolia* Ehrh.), sweetgum (*Liquidambar styraciflua* L.), southern magnolia (*Magnolia grandiflora* L.), Carolina cherry laurel (*Prunus caroliniana* Mill.), various oaks (*Quercus spp*. Mill.), and a mix of southern yellow pines (*Pinus spp.* Mill.) (Griffith et al., 2002; Quarterman and Keever, 1962). The South Carolina Sandhills is a level IV ecoregion located in the uppermost portion of the Southeastern Plains, southernly adjacent to the Fall Line and the Piedmont (Griffith et al., 2002). Whereas the Piedmont and the Blue Ridge are composed mainly of metamorphic and igneous rock from the Precambrian and Paleozoic eras, the Southeastern Plains, an ecoregion that includes the Sandhills, the Atlantic Southern Loam Plains, and the Southeastern Floodplains, has a stark contrast in geology with sands, silts, and clays acting as a byproduct of the Cretaceous and Tertiary periods (Griffith et al., 2002; Murphy, 2016).

The Sandhills, approximately half of the size of the southeastern plains (12% of the state), contain primarily marine sands and clays overlain by the erosion and deposition of crystalline and metamorphic rocks exuding from the Piedmont (Griffith et al., 2002; Murphy, 2016; USDA-NRCS, 2006). It is dominated by sandy Ultisols and Entisols with a thermic soil temperature regime (mean annual soil temperature between 15-22 degrees C), a udic soil moisture regime, and a deep, well-drained loamy sand composition (Series, n.d.; Murphy, 2016; USDA-NRCS, 2006). The high permeability in these soils causes rapid rainwater drainage, leaving an acidic unproductive topsoil layer that is often nutrient deficient (Murphy, 2016).

Of the 17 essential nutrients and the six primary nutrients (C, H, O, N, P, K) required for plant growth, Nitrogen (N), Sulfur (S), and Phosphorus (P) are highly influenced by the presence of organic matter and are often limited in sandhill communities (Donovan et al., 2000; Mahler, 2004). Moreover, following disturbances, subsurface pools of N and P are key indicators for ecosystem recovery and nutrient availability (Klimas, 2020). Inorganic nitrogen (NO_3^- and NH_4^+) is the most limiting nutrient due to the low litter quality (high C:N ratio) and high soil acidity found in longleaf pine sandhill sites. Excessively high C:N ratios result in ammonium immobilization, while nitrification is inhibited at low pH values (Sahrawat, 2008). Inorganic phosphorus ($H_2PO_4^-$, HPO_4^{2-}, and PO_4^{3-}) is essential for plant growth, energy transfer (ADP and ATP), reproduction, photosynthesis, and several other plant functions, frequently showing deficiencies in soils with pH values below 5.5 or above 6.5 (Mahler, 2004). However, increases in soil organic matter have been found to reduce the strength of P adsorption and the maximum phosphate buffering capacity (Yang et al., 2019). Similarly, inorganic sulfur (SO_4^{2-}) is also considerably influenced by soil organic matter and is most important in the formation of chlorophyll. Sulfate is often deficient in coarse sandy soils with good drainage, typical of those found in the Carolina Sandhills (Mahler, 2004; Stewart, n.d.; Walker and Adams, 1958).

These dry, desolate circumstances make plant growth difficult in the sandhills and consequently promote hardy xerophytic vegetation. These vegetation types can be categorized as different combinations of pine-oak vegetation, with longleaf pine acting as the dominant species in a complex of turkey oaks (*Quercus laevis* Walter), blackjack oaks (*Quercus marilandica,* Muenchh.), bluejack oaks (*Quercus incana* Bartram), sand live oaks (*Quercus geminata* Small), bluestems (*Andropogon spp.* and *Schizachyrium scoparium* Michx.), panicums (*Panicum spp.* L.), wiregrass (*Aristida spp.* Michx.), and other sandhill vegetation (USDA-NRCS, 2006). In fact, the Sandhills support one of the last remaining strongholds of longleaf pine populations across the entire southeastern United States (University of Georgia, 2021). Longleaf pine habitats can support over 900 plant species in total throughout their range and can provide resources for at least 29 threatened and endangered species and hundreds of bird, mammal, and herpetofauna species, most of which forage on or near the understory (Engstrom, 1993; Harrington et al., 2013; U.S, Fish and Wildlife Service, 2009; Van Lear et al., 2005). Over 65% of mammal and 35% of bird species forage almost exclusively on or in the herbaceous understory, including the red-cockaded woodpecker (*Leuconotopicus borealis* Vieillot) and the gopher tortoise (*Gopherus polyphemus* Daudin), both of which are endangered keystone species and essential to the survival of at least 27 other wildlife species (Engstrom, 1993; Westerhold, 2013). Unfortunately, due to resource exploitation, agricultural development, and fire suppression, longleaf pine habitat has seen a 97% decline in its historical range, amounting to only 1.2 million hectares of original habitat (Frost, 1993; Wahlenberg, 1946; U.S. Fish and Wildlife Service, 2003; Outcalt and Sheffield, 1996; Van Lear et al., 2005). Today, much of the historical range of longleaf pine habitat has either been developed or has transitioned into dense hardwood, loblolly pine, and slash pine forests (Sherrard, 1903; Frost, 1993).

Longleaf pine ecosystems developed over various landscapes, including flatwoods, sandhills, and clayhills, and are often classified based on soil moisture characteristics (Abrahamson and Hartnett 1990; Myers 1990; Van Lear and Jones, 1987). Longleaf pine trees thrive in well-drained sandy soils and are both fire-adapted and shade-intolerant, meaning that, within just a few years of fire exclusion on dry soils, reproduction halts as a result of successional development (Frost, 1993; U.S. Fish and Wildlife Service, 2003). Fire is essential for maintaining the rich and diverse herbaceous layer found in grassland savannas. Walker and Peet (1983) found that on mesic sites burned annually, above-ground production was as high as 375 g m^{-2} with 42 species per 0.25 m^2. High frequency, low intensity growing season fires (1-3 years) that spread through grass-dominated ground cover have been the most common historical disturbance regime for most, if not all, of the longleaf pine ecosystems in the southeastern U.S. (Christensen, 1981; Cox et al., 2004; Engstrom, 1993; Stambaugh et al., 2011). Fire reduces woody fuel loading, controls invasive species, and increases species richness (Reinhart and Menges, 2004; Wade and Lundsford, 1990; Walker and Peet, 1983). Furthermore, it can reduce litter and duff accumulation, a phenomenon correlated with decreases in bunchgrass cover as it reduces germination and seed contact with bare soil. Hiers et al. (2007) found a negative linear relationship between duff depth and species richness in xeric longleaf systems in Florida, with a 60-75% decline in bunchgrass species across plots with an average duff depth of 2 cm or more.

Therefore, it is noted that herbaceous fuels should be restored in conjunction with longleaf pine seedling establishment (Walker and Silletti, 2007). Grasses act as recurrent fuel sources for fires which decrease competition, facilitate seedling establishment, and are essential for emergence out of the 'grass stage,' ultimately promoting longleaf pine habitat and regeneration (Brockway, 2005; Wade and Lundsford, 1990; Walker and Silletti, 2007). Most, if not all, longleaf restoration projects include reintroducing fire into the ecosystem and controlling woody competition. In a study that measured the vegetation responses of midstory mulching and prescribed burning in longleaf pine ecosystems, it was found that shrub and vine cover, grass cover, and forb cover all increased (9.9%, 9.5%, and 3.3%, respectively) across 13 months following the combination of treatments (Brockway et al., 2009). Plant establishment and restoration are often limited by light availability when there is a densely established overstory, so treatments that decrease the basal area and create gaps in the canopy will increase light penetration and encourage the growth of early successional species.

Consequently, many of the longleaf pine restoration efforts that take place focus on transitioning loblolly pine, slash pine, and hardwood stands to longleaf habitat using a combination of prescribed burning, herbicide, and mechanical treatments (Brockway et al., 2009; Brockway et al., 2009; Outcalt, 1992; Outcalt and Lewis, 1990; Provencher et al., 2001; Walker et al., 2004). For example, while measuring understory responses to canopy treatments in Aiken, SC, it was found that forbs and grasses increased 13% and 8%, respectively, following a timber thinning and 7% and 9% following an herbicide treatment (Harrington and Edwards, 1999). Brockway and Outcalt (2000) noted that a 2.2 kg ha^{-1} hexazinone application in the Florida Sandhills successfully reduced scrub oak competition by 80%, consistently increasing forb, graminoid, and longleaf cover over time. Traditionally, longleaf pine restoration has been conducted using conventional harvesting applications which include clearcutting the existing canopy, planting longleaf pine seedlings, and employing release treatments (Knapp et al., 2006; Knapp et al., 2014). However, in the absence of a closed canopy, such as in a savanna or after a timber harvest, water or nutrient deficiencies could potentially limit plant establishment. Van Eerden (1997) found that in the Carolina Sandhills, bunchgrass seedling establishment was lower on xeric sites compared to sites with higher quantities of loam and silt, indicating that moisture retention was a limiting factor. Similarly, alternative restoration methods such as regenerative seed tree harvesting have been found to promote natural regeneration and promote early successional habitat, but still often employ conventional harvesting methods that are not without their limitations (Boyer and Peterson, 1983; Croker, 1976).

Biomass harvesting and mastication are two alternatives to conventional harvesting that have been found to influence ecological responses differently. Biomass harvesting has intensified due to the domestic and international market demand for wood-based bioenergy (Aguilar et al., 2020; Costanza et al., 2015; North and Piennar, 2021).

However, this newfound interest and increased use of forest biomass as an energy resource have raised concerns regarding long-term forest sustainability (Richardson et al., 2006; Vance et al., 2014). Increased demands for woody biomass could have detrimental effects, including significant forestland reductions and the degradation of forest structure, composition, and nutrient cycles (Aguilar et al., 2020; Janowiak and Webster, 2010; North and Pienaar, 2021; Walker et al., 2010). Soil quality, for example, is directly correlated to soil organic matter, which is subsequently affected by the quality and quantity of input materials (Janowiak and Webster, 2010; Walker et al., 2010). Soil nutrient stocks are essential for plant growth, and the removal of tree components with higher nutrient contents than tree wood (e.g., leaves, cambium, and root tips) can cause declines in long-term site productivity (Janowiak and Webster, 2010). Moreover, the intensified removal of large quantities of woody material can reduce the amount of soil organic matter produced over time (Janowiak and Webster, 2010). For sites that experience inherently low soil qualities, such as the Carolina Sandhills, nutritional deficiencies could intensify following this type of application (Janowiak and Webster, 2010; Richardson et al., 2006). Additionally, the removal of logging slash and increased amount of machine disturbance has raised questions about soil erosion, disturbance, and compaction, although these effects typically do not increase after two machine passes, are greatest immediately following a harvest, and generally recover within 2-5 years (Aust and Blinn, 2004; Janowiak and Webster, 2010; Wang et al., 2005). Consequently, many states, regions, and private organizations have employed biomass harvesting guidelines

(BHG) which aim to mitigate negative impacts associated with biomass removal (Evans et al., 2013; North and Pienaar, 2021).

The same concerns are not associated with fuel mastication, for example, because, even in conjunction with a timber harvest, the removal of woody biomass is not as intense. However, the spatial arrangement of residual fuels does have the ability to impact ecological responses following its implementation, and the effects have been wide-ranging (Kreye et al., 2012; Kreye and Varner, 2007; Overby and Gottfries, 2009). Mastication has been extensively applied in fire-prone ecosystems as a method of controlling woody fuel loading and reducing fire hazards (Kane et al., 2009; Kreye et al., 2012; Kreye et al., 2014; Hood and Wu, 2006). Mastication operations have been found to effectively control competition, promote longleaf habitat, and influence fire behavior in a number of circumstances (Brockway, 2005; Burns and Hebb, 1972). Walker et al. (2004) noted that mastication treatments were significant in reducing hardwood competition and promoting longleaf pine, a similar finding to Tanner et al. (1988), where mechanical drum chopping reduced saw palmetto and runner oak cover by 25% and 50%, respectively. Additionally, Kreye et al. (2014) noted that understory fuels were quick to respond even with increases in surface-fuel compaction following mastication treatments. Other studies have found that masticated fuels produce increases in inorganic nitrogen and moisture retention which can greatly benefit plant communities (Kreye et al., 2011; Kreye et al., 2012; Rhoades et al., 2012; Young et al., 2013). This is interesting considering masticated fuels have the ability to increase C:N ratios through deposition of woody material, ultimately slowing decomposition rates and limiting available N needed for long-term productivity (Overby and Gottfries, 2009). As a result, long-term studies should be conducted with an emphasis on documenting fire behavior, moisture retention, and vegetation responses with increases in masticated fuel decomposition over time.

How restoration is achieved varies greatly, and the resulting ecological impacts can be widespread. This study intends to provide additional information on longleaf pine restoration methods in the Carolina Sandhills based on alternative forms of timber harvesting. By quantifying the abiotic environmental conditions following different manipulations of logging slash, we hope to identify optimal restoration methods while simultaneously setting the stage for future studies within the region. Our specific objectives were to:

1. Quantify changes to woody fuel loading and soil bulk density following multiple silvicultural treatment simulations (i.e., conventional harvesting, biomass harvesting, fuel mastication) that result in slash configuration differences.
2. Determine how logging slash manipulation affects soil moisture retention and nutrient availability along a soil moisture gradient.

We predict that fluctuating harvesting intensities will result in varying levels of restoration success, allowing us to identify the optimal method for longleaf understory reestablishment in the region. The soils in this area are deep, very well drained, and extremely sandy, making plant establishment particularly difficult due to a lack of available moisture. We expect to find that significant changes to the spatial arrangement of woody material following our treatments will yield varying levels of moisture retention and nutrient availability. However, it is certainly possible that mesic areas will accumulate a more significant concentration of nutrients due to the spatial arrangement of the landscape. Additionally, treatments that remove a large amount of woody debris, such as our biomass harvest treatment, could yield our most noteworthy results given the removal of excess carbon and the increase in bare soil.

Nevertheless, we believe that masticated fuels will yield greater moisture retention rates due to the compressed nature of fine woody fuels, and that these differences will be most evident in xeric conditions. Mastication significantly increases the surface area to volume ratio of woody material, which can accelerate the rate of decomposition and influence nitrogen dynamics. High C:N ratios in deposited woody materials stimulate soil microbial activity, which will likely intensify through mastication due to its spatial arrangement, resulting in short but intense periods of nitrogen immobilization followed by rapid decomposition rates. Over time, this could increase mineralization and positively influence the vegetation response. Treatments that do not manipulate slash in this way will likely decompose more slowly and retain less soil water due to an increase in bare soil.

1.2 Methods

1.2.1 Study Site

The "Hardscramble" property is located in the city of Camden in Kershaw County, SC and exhibits a climate typical of the Carolina Sandhills region (Coordinates: 34.26732, -80.65668) (Figure 1.1). According to the SC state climatology office, Kershaw county has a humid, subtropical climate with year-round rainfall and hot summers with average high temperatures reaching 31.1 degrees C between June and August. Average precipitation drops in September from 117.3 mm to 85.1 mm. In the last 30 years, Camden, SC has averaged 97 cm of annual precipitation with a minimum average annual temperature of 10.3 degrees C and a maximum average annual temperature of 24 degrees C. The property is approximately 305 hectares (753 acres), retains fourteen different soil types, and is characterized by various habitats ranging from xeric uplands to low-lying wetlands. The Congaree Land Trust classified the property into four different forest types with six different timber types (Table 1.1). This ecological heterogeneity can support an abundance of wildlife species, including white-tailed deer (*Odocoileus virginianus*), eastern wild turkeys (*Meleagris gallopavo*), eastern gray squirrels (*Sciurus carolinensis*), fox squirrels (*Sciurus niger*), wood ducks (*Aix sponsa*), and numerous bird species, including a healthy population of woodpeckers, songbirds, and birds of prey.

Table 1.1: Congaree Land Trust classifications for forest and timber types located on Hardscramble in Camden, SC.

Forest Types	Timber Types
Mixed Pine and Oak Forest	Mature Longleaf Pine
Longleaf Pine and Scrub Oak forest	Mature Loblolly Pine
Floodplain Forest	Mixed Pine
Pocosin Forest	Upland Hardwood
	Bottomland Hardwood
	Planted Loblolly Pine

The study area is a 28.7 hectare (71 acres) subsection of Hardscramble, located in the Northeast corner of the property. The majority of the site was classified as a longleaf pine and scrub oak forest comprised mostly of mature longleaf pine. However, the area would more accurately be described as a series of uneven-aged loblolly stands with a distinct scrub oak midstory emblematic of the sandhills region. There were isolated pockets of longleaf in the most xeric portions of the property and more minor, scattered pockets throughout the study area, but the majority of the overstory was occupied by loblolly pines. Many of these longleaf pine trees were mature and sawtimber-sized, some estimated to be over 150-years old. However, most were confined to approximately 7.3 hectares (18 acres) along xeric ridges in the northern reaches of the property. The majority of the study area was a mixed pine and oak forest with a loblolly overstory and a densely stratified midstory of hardwood species, including red and white oaks, hickories (*Carya spp.*), sweetgums, sparkleberry (*Vaccinium arboretum*), and American holly (*Ilex opaca*). A timber cruise conducted in June 2020 revealed the study area to have 266.7 tonnes (294 tons) of pine chip-n-saw, 717.6 tonnes (791 tons) of pine sawtimber, 727.6 tonnes (802 tons) of pine pulpwood, 75.3 tonnes (83 tons) of grade hardwood logs, 22.7 tonnes (25 tons) of gum logs, and 1215.6 tonnes (1,340 tons) of hardwood pulpwood.

Intense shading and a high average basal area caused by a prolonged lack of forest management inhibited understory development and longleaf pine regeneration for much of the study area (Figure 1.5).

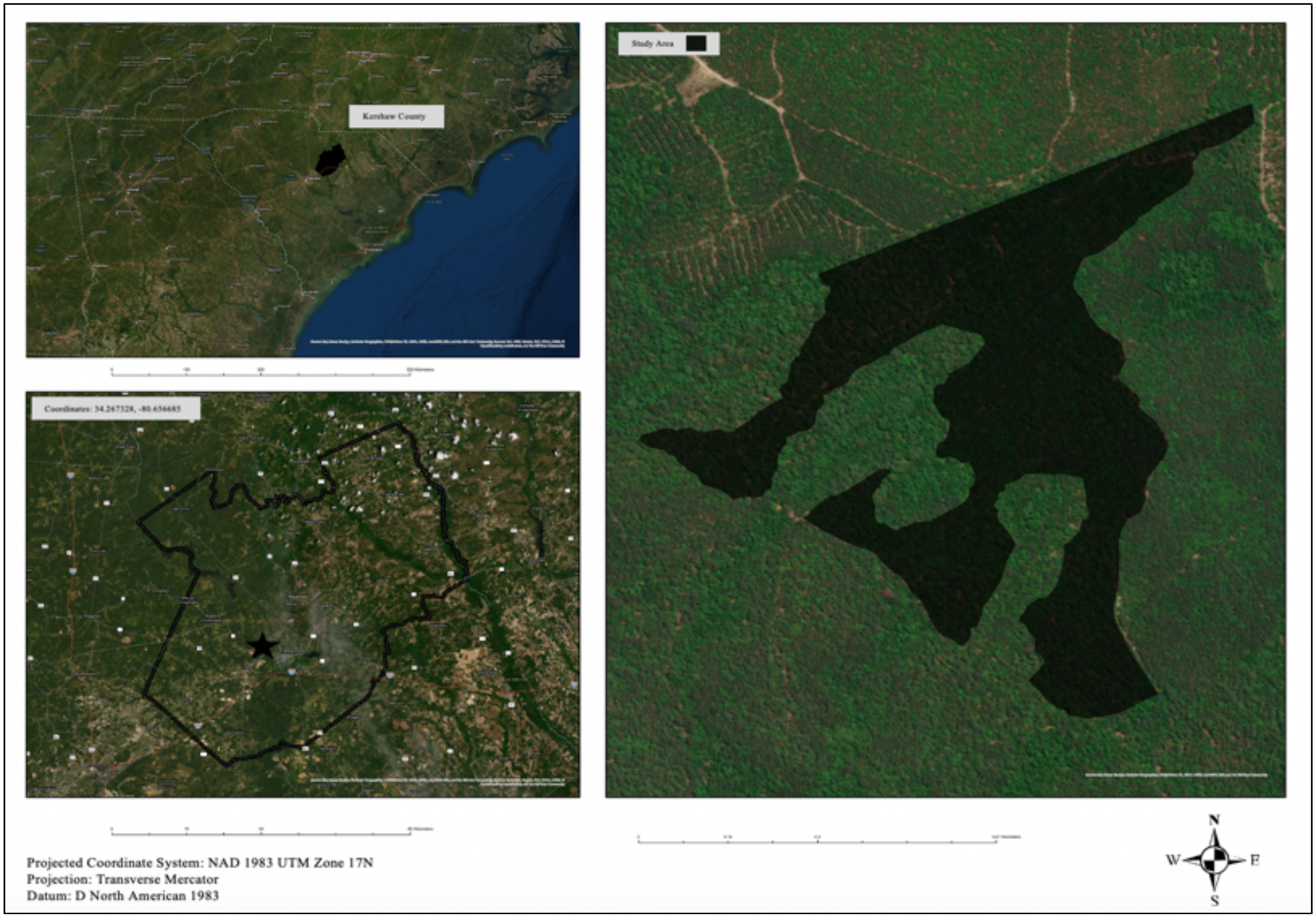

Figure 1.1: Map of 28.7 hectare seed-tree harvest study site located on Hardscramble in Camden, SC.

1.2.2 Prior Research

Before the study, a Level 3&4 Carolina Vegetation Survey (CVS) was conducted on the property in Summer 2018 to quantify the composition and structure of stand 6, a large portion of our study area that was described as a longleaf pine-scrub oak forest by the Congaree Land Trust. The vegetation cover percentage was determined at breast height and ground level to understand the midstory and overstory effect on understory growth. Thirty 10 m^2 plots were randomly set up in a portion of the study area, and each species, as well as the abundance of each species, was recorded along with the approximate cover percentages.

Across all samples, the average litter and duff depth was 2.50 cm, with an average of 4.5 species (21 total) and 27.3 plants (834 total) per CVS plot for all vegetation. Based on the diameter class occurrence data, the structure of the forest indicated an uneven-aged stand that follows a reverse J-shaped distribution as the diameter class increases (Figure 1.2). The most significant number of plants were found in the 1-2.5 cm and the 0-1 cm diameter classes, respectively. Across thirty plots, 29 seedlings or herbaceous plants (3%), 294 trees (35%), and 511 (61%) saplings were recorded, indicating a densely populated midstory and a relatively nonexistent understory. There is also a notable trend between the moisture conditions and the plant community structure. In the most xeric areas, there was a lower amount of overstory and midstory cover and a higher amount of understory herbaceous cover (Figure 1.3). Wetter sites saw an increase in overstory and midstory cover with a decrease in understory richness. Additionally, 17% of all saplings and trees were longleaf pines clustered near the samples' northeast portion.

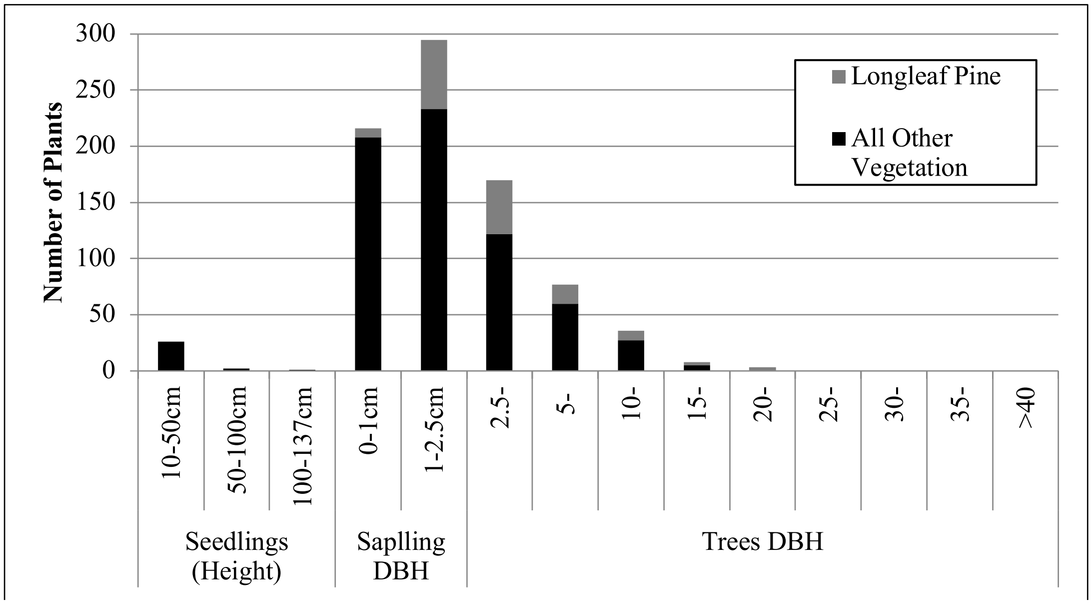

Figure 1.2: Diameter class distribution for all vegetation measured in 30 CVS plots located in a portion of our study area in 2018.

In Fall 2019, a landscape ecological classification (LEC) system developed for the hilly coastal plain province of South Carolina was used to classify sites along a soil moisture gradient with four classifications ranging from the wettest sites to the driest sites: Mesic, Submesic, Subxeric, Xeric (Van Lear and Jones, 1987). The classification system uses the depth to the clay layer as an indication of soil moisture content, with deeper clay layers and more extensive sandy soil layers acting as an indication of a drier site. A depth of 0-50.8 cm (0-20 in), 50.8-101.6 cm (20-40 in), 101.6-203.2 cm (40-80 in), and >203.2 cm (>80 in) indicates a mesic, submesic, subxeric, and xeric site, respectively.

One-hundred depth samples were taken at random across the same area as the CVS conducted in 2018 (Figure 1.3). Of the sampled points, 16% were mesic, 42% were submesic, 39% were subxeric, and 3% were xeric. Each moisture class correlates with a suitable plant community and suggests several species based on significant plant traits.

For example, longleaf pine trees, which thrive in well-drained sandy soils and readily inhabit xeric sites, are suitable for growth in submesic, subxeric, and xeric sites accounting for 84% of the samples. Drier sites seem to have more longleaf trees and fewer loblolly trees, but this trend reverses with an increase in soil moisture. Several of these points were located in pocosins which were excluded from the study.

Xeric sites and longleaf pine occurrences were located primarily in the northeastern corner of the study area along a prominent ridge, while the number of drains, loblolly pines, and submesic sites increased towards the southern portion of the samples. As a result, we extended and reshaped our study area to follow a moisture gradient across 28.7 hectares that overlapped the original sampling sites. This designation became the area used for our study (Figure 1.4). Sites became drier as they went up in elevation and followed a general trend across the study area, arcing from the south to the northwest and then to the northeast, essentially getting drier as one travels south to north across the site (Figure 1.4).

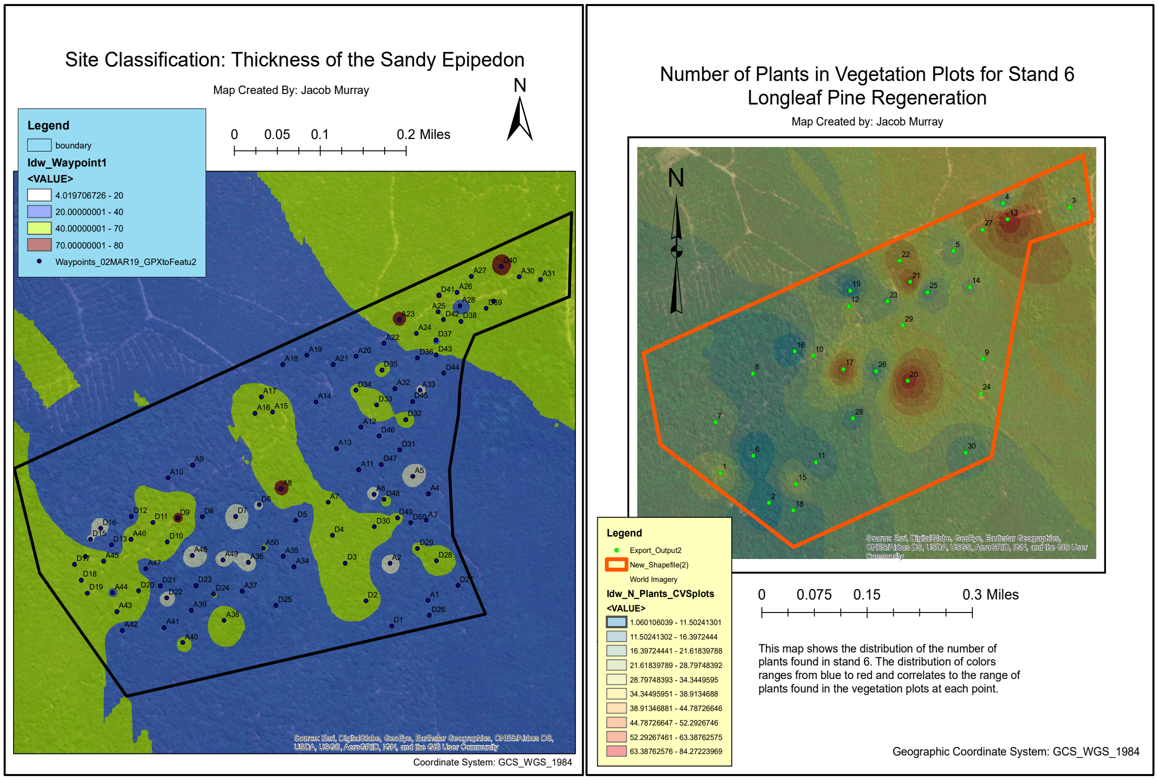

Figure 1.3: Map of 100 LEC sampling points and interpolated values ranging from mesic to xeric (left). Map of interpolated values for the number of plants recorded across 30 CVS plots (right).

1.2.3 Project Design

A seed-tree timber harvest was conducted from November to December of 2020 over the entire study area. As required by the conservation easement, 30.48 m (100 ft) buffers were placed around sensitive habitats, including wetlands and drainages. The harvesting operation and any additional management actions complied with the rules and regulations set out by the SCBMP instruction manual. Pre- and post-harvest site conditions were identified for soils, vegetation, and downed woody material (DWM) between different silvicultural treatments along an established moisture gradient (Figure 1.4). All longleaf pine trees were marked to avoid, all loblolly pine trees were removed, and all hardwood trees with a <30.48 cm (<12 in) diameter were removed. These leftover hardwood seed trees act as a gene source for future generations and an essential resource for wildlife. A majority of the area following the harvest exemplified a seed-tree harvest, while small patches along the northern ridge of the study area showed a stand composition similar to that of a shelterwood cut.

The study employed a split-block design that organized the 28.7 hectares (71 acres) into four blocks along a soil moisture gradient derived from the sandhills ecological classification findings in 2019. Each soil moisture block was subdivided into three treatment zones, and three random sampling points were placed within each zone using ArcGIS mapping applications (Figure 1.4). Our treatments simulated different manipulations of logging slash resulting from three different silvicultural practices: Biomass Harvesting, Fuel Mastication, and Conventional Harvesting (Figure 1.5). In the conventional harvesting treatment, logging slash was evenly distributed across the site, acting as the standard for slash management and post-harvest site conditions. For the biomass harvest treatment, all of the slash in the treatment zones was removed to simulate a biomass harvest by exposing as much bare ground as possible and removing large quantities of woody debris from the site. The masticated fuel treatment took place in April 2021. The slash was evenly distributed across the study area during the timber harvest and then chipped so that there was a resulting layer of masticated fuels coating the now exposed forest floor.

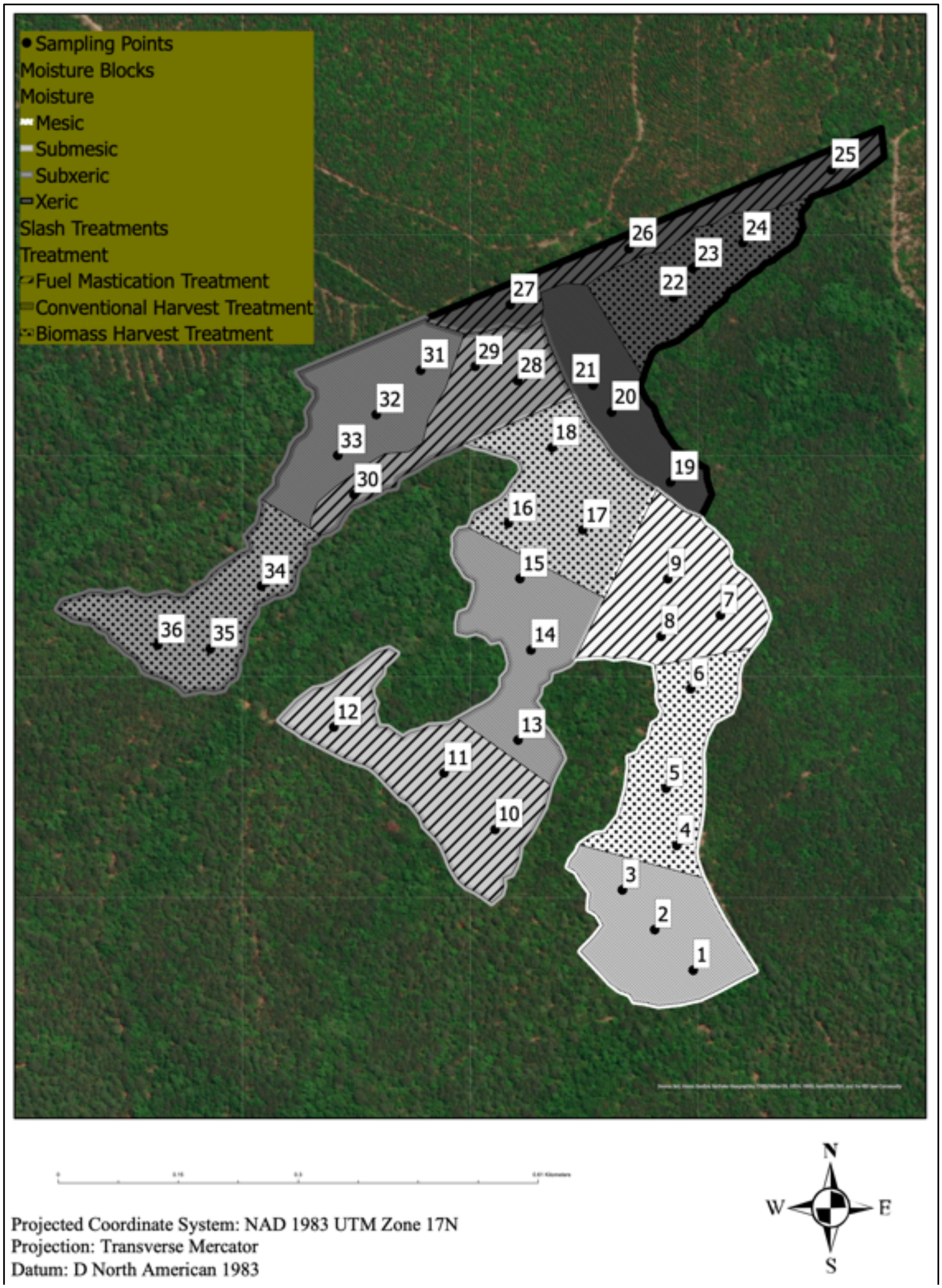

Figure 1.4: Study area map for project moisture blocks (4) with embedded treatments (3) and sampling point (36) locations in Camden, SC.

Figure 1.5: Comparison of preharvest site conditions (A), our masticated fuels treatment (B), our conventional harvest treatment (C), and our biomass harvest treatment (D).

1.2.4 Pre-treatment Sampling

1.2.4.1 Downed Woody Material

For pre-harvest fuel loading measurements, downed woody material (DWM) was distinguished as either coarse woody debris (CWD) or fine woody debris (FWD) using the line intersect method (Benjamin et al., 2013; Briedis et al., 2011). One 30.48 m long transect was established at a random azimuth from each sampling point (36), totaling 1,097 m (3,600 ft) across the study area. Any woody material crossing the transect line was measured according to the sampling criteria developed by Brown (1974). CWD was classified as any woody material that had a small end diameter ≥7.62 cm (≥ 3 in) and a length ≥0.91 m (≥ 3 ft), while FWD was classified as any material with a small end diameter ≥1.27 cm (≥ 0.5 in) and ≤7.62 cm (≤ 3 in) and a length ≥0.3 m (≥ 1 foot). The midpoint diameter and length were recorded for FWD, while the large and small end diameters, midpoint diameter, intersecting diameter, and length were recorded for CWD. Additionally, a decay class reduction factor was assigned to each piece of DWM using a five-stage classification scheme developed by Waddell (2002).

Both CWD and FWD volumes were calculated using volume formulas derived from their unique shapes. Fraver et al. (2007) found that of the six standard formulas used to estimate the volume of downed woody material, actual CWD volumes were closest to the conical frustum formula, which underestimated the actual volume of woody debris, and the second-order paraboloid formula, which overestimated the volume. As a result, CWD volume was calculated using the conic paraboloid equation, which combines the two formulas with the assumption that each piece of CWD is between the shape of a second-order paraboloid and a cone (A_a= cross-sectional area at the upper end; A_b= the cross-sectional area at the lower end; L= length) (Fraver et al., 2007).

$$\text{Conic-paraboloid Volume} = (L/12) * (5A_a + 5A_b + (A_aA_b)^{1/2}) \quad (1.1)$$

FWD volume was estimated using Huber's volume formula and assumes that the downed material has a paraboloid frustum shape (A_m= cross-sectional area at the midpoint; L= length) (Benjamin et al., 2013; Fraver et al., 2007; Husch et al., 2003).

$$\text{Huber's Volume} = L * A_m \quad (1.2)$$

We then estimated per unit values following Waddell's (2002) equations and DeVries' (1973) formula, converting to kilograms per hectare using specific gravity and the decay class reduction factor for CWD (Benjamin et al., 2013). Per-unit values of Carbon were also estimated using specific gravity and a carbon conversion factor (Waddell, 2002). However, for FWD biomass estimates, volumes were multiplied by the average bulk density for *Quercus spp.* (579.7 kg m^{-3}) and an assumed decay class reduction factor of 0.80 (Woodall and Monleon, 2008). We used oak bulk density because it is higher than average pine bulk densities and FWD is often underestimated (Woodall and Monleon, 2008).

1.2.4.2 Soil Water Tension

Soil water potential (kPa) was measured using thirty-six 15.24 cm (6 in) tensiometers (Soil Measurement Systems, Tucson, Arizona, USA), one placed at each sampling point using a 2.54 cm diameter soil probe (Figure 1.6). Biweekly measurements were taken from 20 July 2020 to 27 August 2020 using a digital vacuum sensor (tensimeter). The tensiometers were refilled if they dropped below -85 kPa and were replaced if any damage was sustained through falling debris or wildlife interference.

Figure 1.6: Tensiometer (15.24 cm) placement for one sampling point during pre-harvest measurements on Hardscramble in Camden, SC.

1.2.5 Post-treatment Sampling

1.2.5.1 Sampling Design

After the timber harvest, bulk density, soil, and FWD samples were collected along with weekly tensiometer readings, CWD surveys, and bare ground assessments at each sampling point. One tensiometer was placed at each sampling point post-harvest, and a 4 x 20-meter CWD strip plot was placed approximately 2.5 meters from that tensiometer at the same azimuth as the pre-harvest woody debris transects, indicated by a piece of protruding rebar (Figure 1.7). Four destructive FWD samples were taken at each plot corner, four soil samples were taken at the midpoints of each plot boundary line, and three bulk density samples were taken at 5-meter intervals in the center of each CWD plot (Figure 1.7). Additionally, one 20.3 cm (8 in) lysimeter was placed randomly in each treatment zone using ArcGIS mapping applications to supplement our soil sample findings (Figure 1.8).

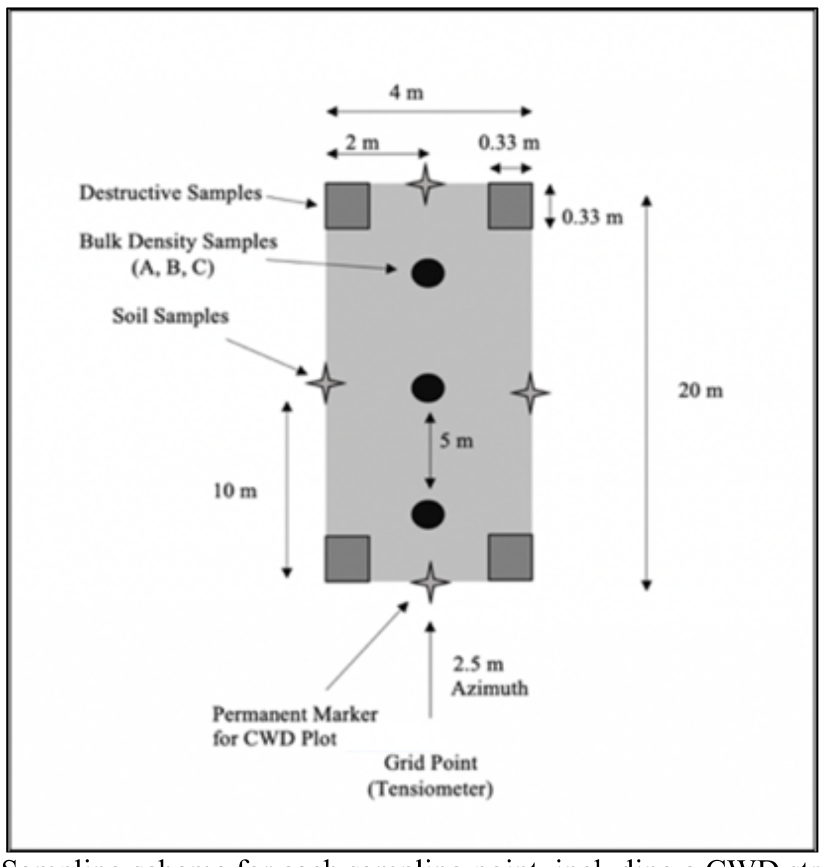

Figure 1.7: Sampling scheme for each sampling point, including a CWD strip plot, bulk density samples, soil samples, and destructive FWD subplots.

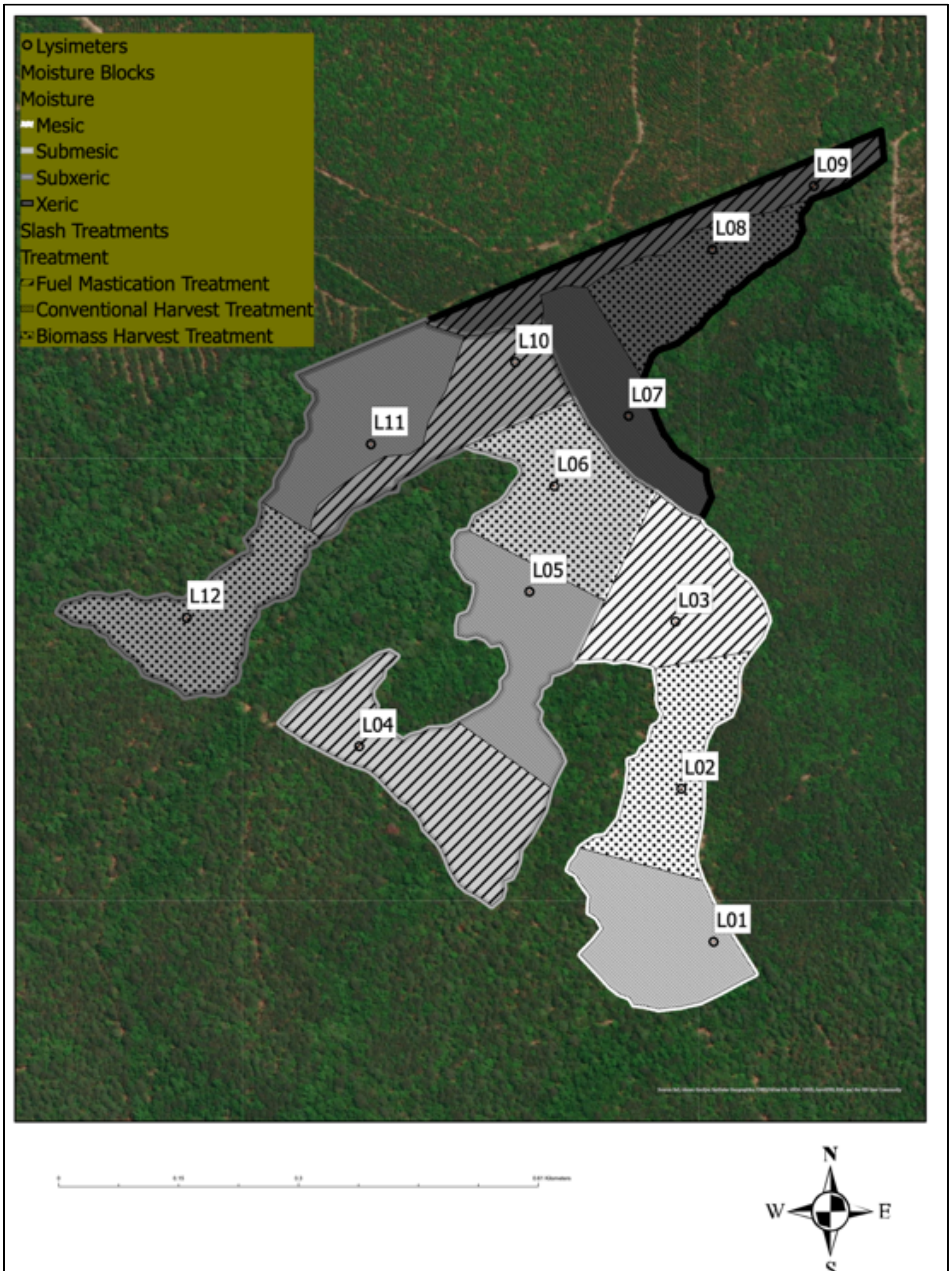

Figure 1.8: Study area map for project moisture blocks with embedded treatments and lysimeter locations.

1.2.5.2 Soil Characteristics

In order to understand the total amount of nutrients present, four soil samples were collected from each CWD plot post-treatment using a soil probe at a depth of 20 cm, yielding approximately 0.5 liters (2 cups) of soil when combined. The subsamples were collected at the midpoints of each plot boundary line, mixed, and air-dried overnight before being bagged and sent to the Clemson Agricultural Service Laboratory for analysis (Figure 1.7). The samples were labeled as sandy, course-textured soils predominantly comprised of sand, where the depth to the clay layer is greater than 101.6 cm (40 in) (Code 1). The soils were tested for pH, buffer pH, extractable elements (P, K, Ca, Mg, Zn, Mn, Cu, B, Na, S), organic matter, nitrate-nitrogen (NO_3^-N), and soluble salts. Calculations for the cation exchange capacity (CEC), acidity, and percent base saturation were also included in the analysis.

In order to supplement our soil sample findings, weekly soil solution samples were obtained from twelve 30-cm porous cup suction lysimeters (Hanna Instruments HI89300-30) from June 2021 to August 2021, totaling 36 samples across three complete sampling sets. Soil solutions are able to help identify nutrient availability at a specific point in time. One lysimeter was placed 6 inches into the soil in each treatment block (3 lysimeters per moisture block). Using spectrophotometry ammonium, nitrate, and phosphate concentrations were measured using the Environmental Protection Agency (EPA) compliant FIA-012 method, the FIA-026 Cadmium reduction method (EPA 353.2, SM 4500-NO3), and the FIA-073 sequential flow injection method (EPA 365.1), respectively.

1.2.5.3 Soil Bulk Density

Using the cylindrical core method, three bulk density samples (A, B, C) were collected at 5-meter intervals in the center of each CWD plot (Figure 1.7) (USDA-NRCS, 1999). A soil ring (D= 12.1 cm; H=19.1 cm) was driven into the ground using a 20.3 cm slide hammer and carefully removed with the excavated soil core intact. Any soil that extended past the dimensions of the ring was removed along with any plant material using a pocketknife. The soil core was then removed from the ring and placed in a sample bag where it was weighed and dried for 48 hours at 105 degrees Celsius. The dried samples were then reweighed and used to calculate bulk density (g cm^{-3}) and soil porosity (%).

Calculations:

Bulk Density (g cm^{-3}) = Dry Weight of Sample (g) / Volume of Soil Core (cm^3) ....(1.3)

Soil porosity (%) = 1 – (Bulk Density (g cm^{-3}) / 2.65 (BD of a rock)) ....(1.4)

1.2.5.4 Downed Woody Material

After the harvest, DWM was inventoried again as FWD and CWD but sampled differently due to the difficulty of measuring masticated fuels. CWD was sampled using a 4 x 20-meter strip plot about 2.5 meters away from the sampling point at the same azimuth as the pre-harvest transects (Figure 1.7) (Kane et al., 2009). The length, large end diameter, and small end diameter were measured for logs with a length ≥1 m, a large end diameter ≥15 cm, and a small end diameter ≥7.62 cm (Waldrop et al., 2010). The weight of each piece of material was obtained using sampling estimations and the average bulk densities per species provided by the FIA program, which were then converted to metric units (Woodall and Monleon, 2008). Equation 1.5 was used for pieces in decay classes 1-4 and equation 1.6 was used for a decay class of 5. Per-unit values (kg ha^{-1}) were obtained by scaling up our strip plots using the combined weight of the woody material inventoried within the boundaries of each plot. Per-unit values of Carbon were also estimated using specific gravity and a carbon conversion factor (Waddell, 2002).

Calculations:

$$V\ (ft^3) = \frac{(\pi/8) * (((\text{End Diameter 1 (in)})^2 + (\text{End Diameter 2 (in)})^2) * \text{Length (ft)}}{144} \quad \text{...(1.5)}$$

$$V\ (ft^3) = \frac{(\pi/4) * (\text{Midpoint Diameter (in)})^2 * \text{Length (ft)}}{144} \quad \text{...(1.6)}$$

$$\text{Biomass (lb)} = \text{Bulk Density (lb ft}^{-3}) * \text{Decay-reduction Factor} * \text{Volume (ft}^3) \quad \text{.......(1.7)}$$

Surface fuels, FWD, litter, and duff were destructively sampled using four nested 0.1 m^2 subplots at the corners of each strip plot yielding 0.4 m^2 per sampling point and 13.4 m^2 across the study area (Figure 1.7). FWD that extended outside the subplot was removed using loppers or a handsaw, and all samples were sorted. Litter and duff were combined due to the difficulty of separating the two, and FWD was separated into time-lag classes (1, 10, and 100 h) based on diameter size (0-0.635 cm, 0.635-2.54 cm, and 2.54-7.62 cm, respectively) (Figure 1.10). Pinecones were incorporated as part of the litter and duff samples because they act as critical ignition sources for fires in the duff layer (Kreye et al., 2013). Fractured particles, pieces of FWD that were more than 50% physically altered from chipping, were sorted into size classes based on the width of the chip that exhibited the most significant percentage of the chips form (Figure 1.9) (Kane et al., 2009). This measurement method is different from other methods of fuels classification that use the average of chip dimensions, the theory being that under uniform conditions, such as the uniform application of heat in an oven, moisture changes would be quickest through its thinnest parts (Kreye et al., 2014). All samples were rinsed to remove any mineral soil, which can overestimate the dry weight of the material, and then air-dried for 48-hours before being weighed, dried in an air-dry oven at 105 degrees C for 72 hours, and finally reweighed. Dry weights were converted to per-unit values (kg ha^{-1}) and compared to pre-harvest values across treatments.

Figure 1.9: Example of measuring guidelines for classifying masticated fuels in destructive sampling for the 1-hr time lag category. With 60% of the woody material measuring below 0.635 cm, this piece of woody material is classified as a 1-hr fuel.

Figure 1.10: An example of 1-hr (A), 10-hr (B), and 100-hr (C) fuel classes being sorted, washed, and air-dried with the inclusion of masticated fuels.

1.2.5.5 Soil Water Tension

Soil water potential (kPa) measurements were repeated post-harvest using the same methodology as pre-treatment sampling. Weekly measurements were taken from May 2021 to August 2021, totaling 321 readings.

1.2.6 Data Analysis

All data transformations and analyses were conducted using Microsoft Excel (Microsoft Corporation, 2018), R (R Core Team, 2020), and RStudio (RStudio Team, 2020), and all maps and spatial investigations were completed using ArcGIS Pro (Esri Inc., 2020). In order to better understand the effects of our treatments and soil moisture characteristics in the Sandhills Wiregrass Gap on environmental conditions, we used the data compiled from 2020 to 2021 to investigate different trends between our treatments. Soil nutrient availability, bulk density, and downed woody material were analyzed using a two-way analysis of variance (ANOVA), and time-integrated measures (matric water potential) were analyzed with a repeated-measures ANOVA using a Bonferroni correction. The 'Treatment' and 'Moisture Block' variables were converted into factor data, measures of central tendency and dispersion for our dependent variables were determined, and finally we tested for normal distribution using Normal Q-Q plots and the Shapiro-Wilk test on the ANOVA residuals. We also employed a Levene's Test to measure for the homogeneity of variances. Data that were not normally distributed were log-transformed to meet the assumptions of normality. Tukey's HSD post hoc tests were performed for pairwise comparisons using a Bonferroni correction and, for any significant interactions, we developed an interaction plot using least-Squares Means tests for different factor combinations.

1.3 Results

1.3.1 Soil Characteristics

Across the total study area, soil pH was strongly acidic (M= 4.9, SD= 0.2) and did not exhibit a significant difference in means across treatments (F= 0.510, P=0.607) or between moisture blocks (F=1.182, P=0.338). We see a similar trend in extractable potassium (K), where we have low potassium levels across the entire study area (M= 20.8 ppm, SD= 8.7 ppm) but no significant effects across treatments or moisture blocks (F= 1.544, P= 0.234; F= 0.646, P= 0.593; respectively). Nitrate nitrogen (NO3-N) was insufficient across all sites, with 80.6% of all samples having 0 ppm, 16.7% having 1 ppm, and only one sample showing 5 ppm.

The only elements that exhibited sufficient levels of availability were manganese (Mn) in 25% of samples and magnesium (Mg) in 5.6% of samples. Extractable phosphorus (P) was available in insufficient amounts across the entire study area (M= 4.3 ppm, SD= 3.3 ppm), exhibiting low amounts of P (0-15 ppm) in 97% of our samples. However, there was a significant difference in the means between treatments (F= 6.572, P= 0.005) and moisture blocks (F= 5.438, P= 0.005), with our submesic biomass harvest treatment exhibiting the greatest amount of P per unit area. Pairwise comparisons found that, on average, P levels were 113% greater in biomass harvest treatments compared to our conventional treatments, and 210% higher in submesic sites compared to subxeric sites (Table 1.2, 1.3, respectively) (Figure 1.11).

Soil organic matter was relatively low across the study area, averaging 1.92% of the soil composition by weight across all sites. We did not find significant mean differences between treatments (F= 1.935, P= 0.166) or moisture blocks (F=2.740, P= 0.066). However, we did find significant interaction effects between our independent variables (F= 3.519, P= 0.012), with our biomass harvest treatment in our subxeric moisture block exhibiting the greatest amount of soil organic matter and significantly higher amount of organic matter compared to conventional harvest treatments in submesic (+1.55%) and subxeric sites (+1.41%), and biomass harvest treatments in submesic (+1.87%) and xeric sites (1.61%) (Figure 1.12).

For our soil solutions, we found significant NO_3^- concentration differences across means for our moisture groups (F= 5.496, P= 0.002) with pairwise comparisons showing higher average concentrations for subxeric sites (Figure 1.13). Across all sites, our highest maximum concentration was 6.3 ppm and was located in our subxeric masticated fuel treatment. However, there were no discernable trends across slash manipulations, with mean differences between treatments (F= 0.012, P= 0.988) and interaction effects between independent variables (F= 1.204, P= 0.317) exhibiting no statistical significance. Additionally, we did not find significant mean differences in Gaussian peak predictions for our treatments (F= 0.266, P= 0.769) or moisture blocks (F= 2.774, P= 0.063).

Comparatively, mean NH_4^+ concentrations were extremely low but were significantly different between moisture groups (F= 4.905, P= 0.003) and treatments (F= 8.634, P= <0.001). On average, our biomass treatment produced significantly lower concentrations compared to all other treatments, and our mesic site produced significantly higher concentrations than all other moisture blocks. We also found significant interaction effects between our independent variables, with conventional harvest sites on mesic soil producing very high concentrations (F= 3.717, P= 0.002) (Figure 1.14). However, there were no significant findings for Gaussian peak predictions across or between independent variables (Table 1.4). PO_4^{3-} concentrations showed no significant differences between the means or between the mean Gaussian peak predictions across or between independent variables (Table 1.5, 1.6). Maximum concentrations ranged from 1.34 to 44.94 ppm and the highest predicted mean Gaussian peak prediction was 1.14 ppm with a majority of the results having less than 0.5 ppm (Table 1.7; Figure 1.15).

Table 1.2: Tukey contrasts showing multiple comparisons of means for mean differences in extractable P (ppm) between samples obtained from three logging slash treatments in 2021.

Linear Hypotheses	Mean Difference	Lower	Upper	P-value
Biomass - Mastication == 0	0.2055	-0.0194	0.4304	0.078
Conventional - Mastication == 0	-0.1170	-0.3419	0.1079	0.409
Conventional - Biomass == 0	-0.3225	-0.5474	-0.0976	0.004

Table 1.3: Tukey contrasts showing multiple comparisons of means for differences in extractable P (ppm) between samples obtained across a soil moisture gradient in 2021.

Linear Hypotheses	Mean Difference	Lower	Upper	P-value
Submesic - Mesic == 0	0.2445	-0.0423	0.5314	0.1144
Subxeric - Mesic == 0	-0.1700	-0.4569	0.1169	0.3791
Xeric - Mesic == 0	0.0678	-0.2191	0.3547	0.9139
Subxeric - Submesic == 0	-0.4145	-0.7014	-0.1276	0.0029
Xeric - Submesic == 0	-0.1767	-0.4636	0.1101	0.3458
Xeric - Subxeric == 0	0.2378	-0.0491	0.5246	0.1295

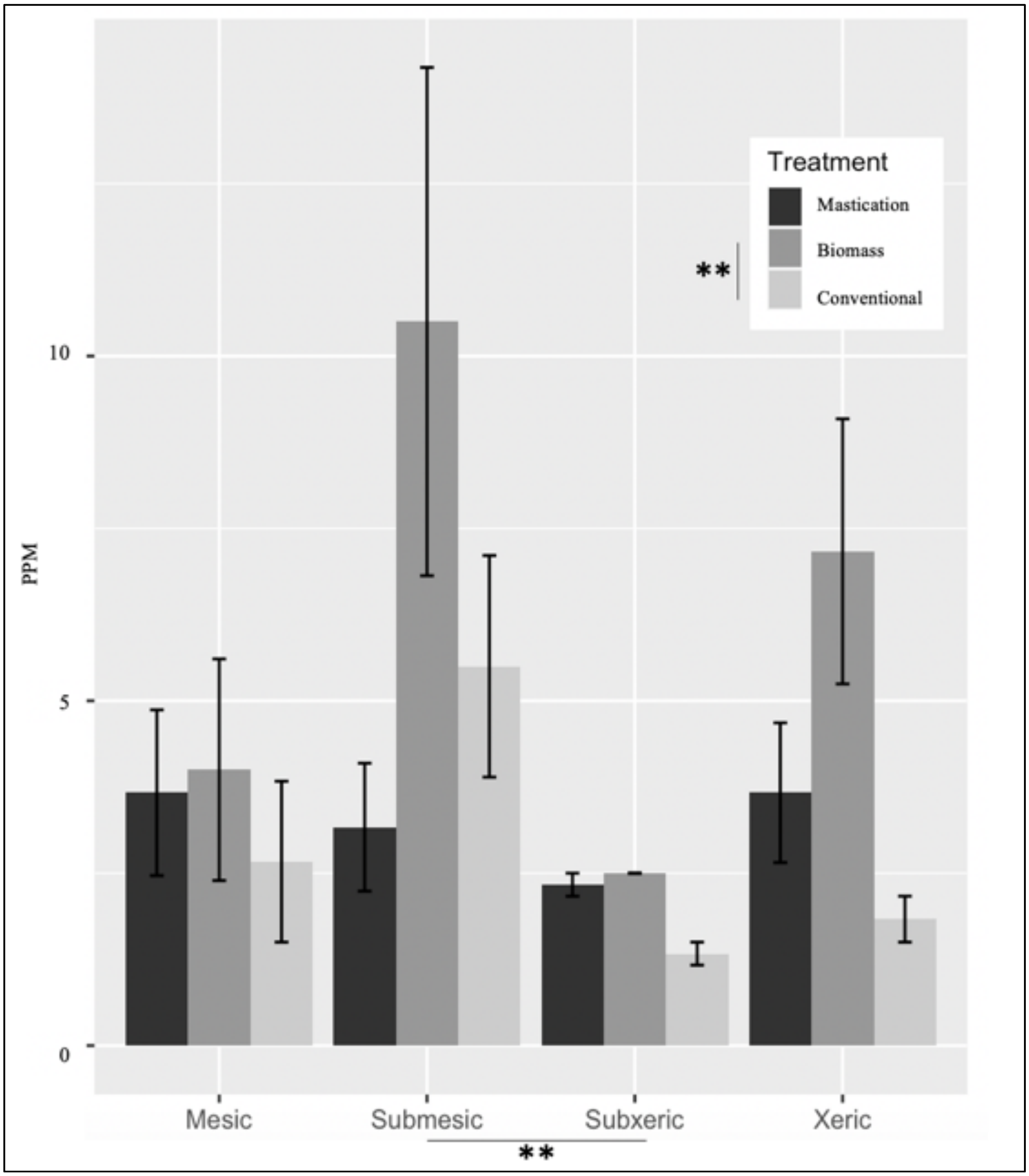

Figure 1.11: Bar plot of mean phosphorus availability (ppm) with standard error of mean bars for treatments across our soil moisture gradient in the Carolina Wiregrass Gap in Camden, SC. Statistically different means are represented by asterisks located by the legend for between-treatment differences and between labels for between-group differences (α < 0.05).

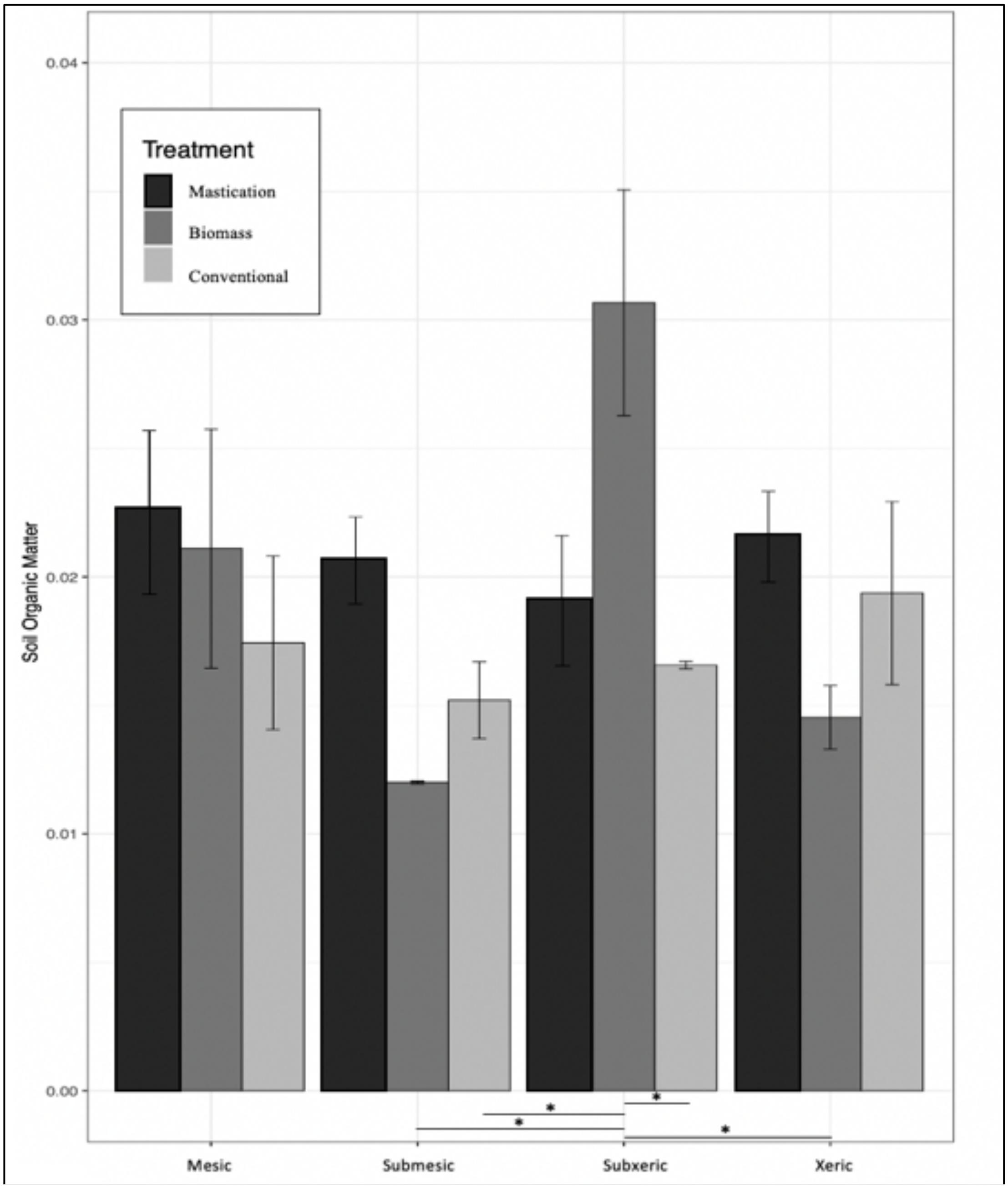

Figure 1.12: Bar plot with 95% confidence intervals of mean soil organic matter (%) for treatments across our soil moisture gradient in the Carolina Wiregrass Gap in Camden, SC. Statistically different means for pairwise comparisons are represented by an asterisk ($\alpha < 0.05$). For example, the submesic biomass harvest treatment is significantly greater than the conventional harvest treatment located in the submesic moisture group.

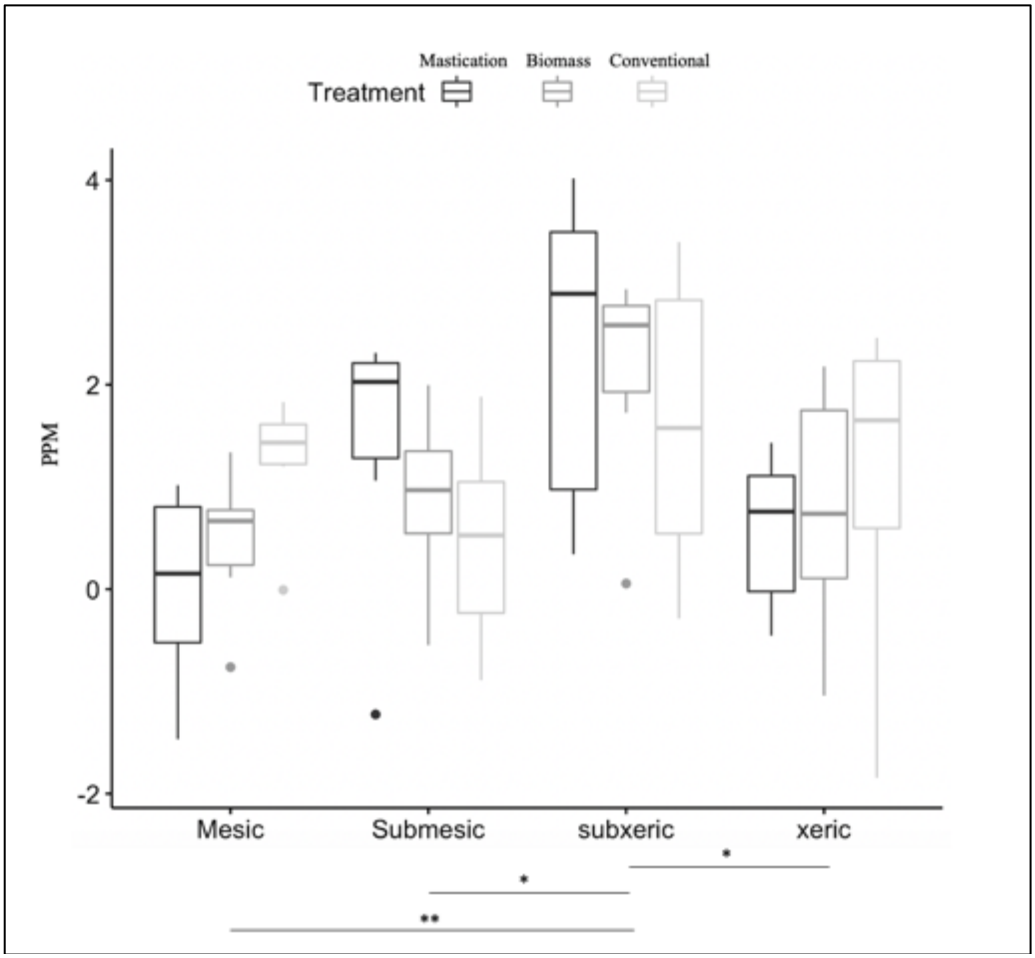

Figure 1.13: Boxplot of mean comparisons for NO_3^- concentrations (ppm) across moisture groups and treatments in Camden, SC. Asterisks above lines connecting pairwise comparisons are significantly different from each other ($\alpha < 0.05$).

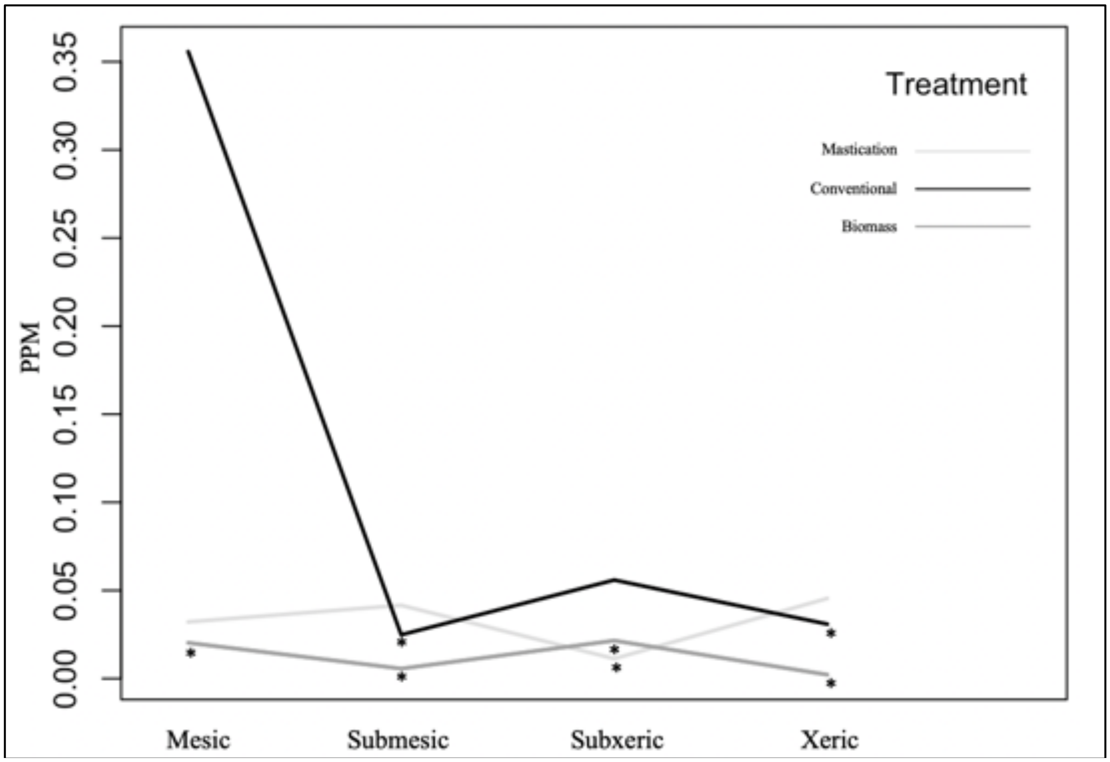

Figure 1.14: Interaction plot for significant mean concentration (ppm) differences for NH_4^{3-} across moisture groups and treatments in Camden, SC. Pairwise comparisons for mean NH_4^{3-} concentrations are significantly different from conventional harvests in mesic sites if they have an asterisk attached ($\alpha < 0.05$).

Table 1.4: Mean parameters of soil solution variable nutrient concentrations (ppm) at a 15.24 cm soil depth between slash manipulation treatments and moisture groups in Camden, SC. Parenthetical values are standard deviation.

Moisture Group	Treatment	NO_3^-			NH_4^+			PO_4^{3-}		
		Predicted Peak	Average	Max	Predicted Peak	Average	Max	Predicted Peak	Average	Max
Mesic	Mastication	0.56 (0.03)	0.43 (0.17)	0.58	0.21 (0.34)	0.17 (0.24)	0.61	0.24 (0.16)	1.11 (1.84)	6.35
	Biomass	0.56 (0.01)	0.52 (0.10)	0.62	0.03 (0.02)	0.04 (0.06)	0.19	1.14 (1.57)	5.51 (13.40)	44.94
	Conventional	0.74 (0.05)	0.65 (0.12)	0.79	0.19 (0.18)	0.29 (0.18)	0.66	0.23 (0.04)	0.57 (1.06)	2.83
Submesic	Mastication	0.91 (0.35)	0.87 (0.40)	1.27	0.04 (0.02)	0.05 (0.03)	0.13	0.63 (0.75)	3.68 (9.75)	33.55
	Biomass	0.67 (0.19)	0.59 (0.18)	0.89	0.01 (0.01)	0.01 (0.04)	0.14	0.38 (0.35)	1.20 (2.84)	10.01
	Conventional	0.65 (0.13)	0.52 (0.18)	0.8	0.01 (0.01)	0.03 (0.05)	0.16	0.17 (0.06)	0.33 (0.36)	1.34
Subxeric	Mastication	3.35 (2.88)	3.33 (2.59)	6.32	0.01 (0.01)	0.03 (0.06)	0.19	0.26 (0.14)	0.77 (1.97)	5.01
	Biomass	2.13 (1.23)	2.03 (1.15)	3.16	0.01 (0.01)	0.04 (0.07)	0.24	0.18 (0.08)	0.94 (2.12)	7.48
	Conventional	2.17 (2.56)	2.07 (2.35)	5.12	0.05 (0.04)	0.05 (0.04)	0.12	0.41 (0.46)	0.92 (2.36)	8.08
Xeric	Mastication	0.57 (0.02)	0.52 (0.11)	0.64	0.07 (0.08)	0.08 (0.1)	0.35	0.27 (0.21)	1.71 (4.47)	15.74
	Biomass	0.75 (0.32)	0.65 (0.31)	1.11	0.03 (0.04)	0.01 (0.04)	0.08	0.19 (0.08)	0.39 (1.50)	4.57
	Conventional	0.96 (0.56)	0.85 (0.58)	1.59	0.06 (0.09)	0.05 (0.06)	0.16	0.60 (0.77)	1.23 (1.95)	7.01

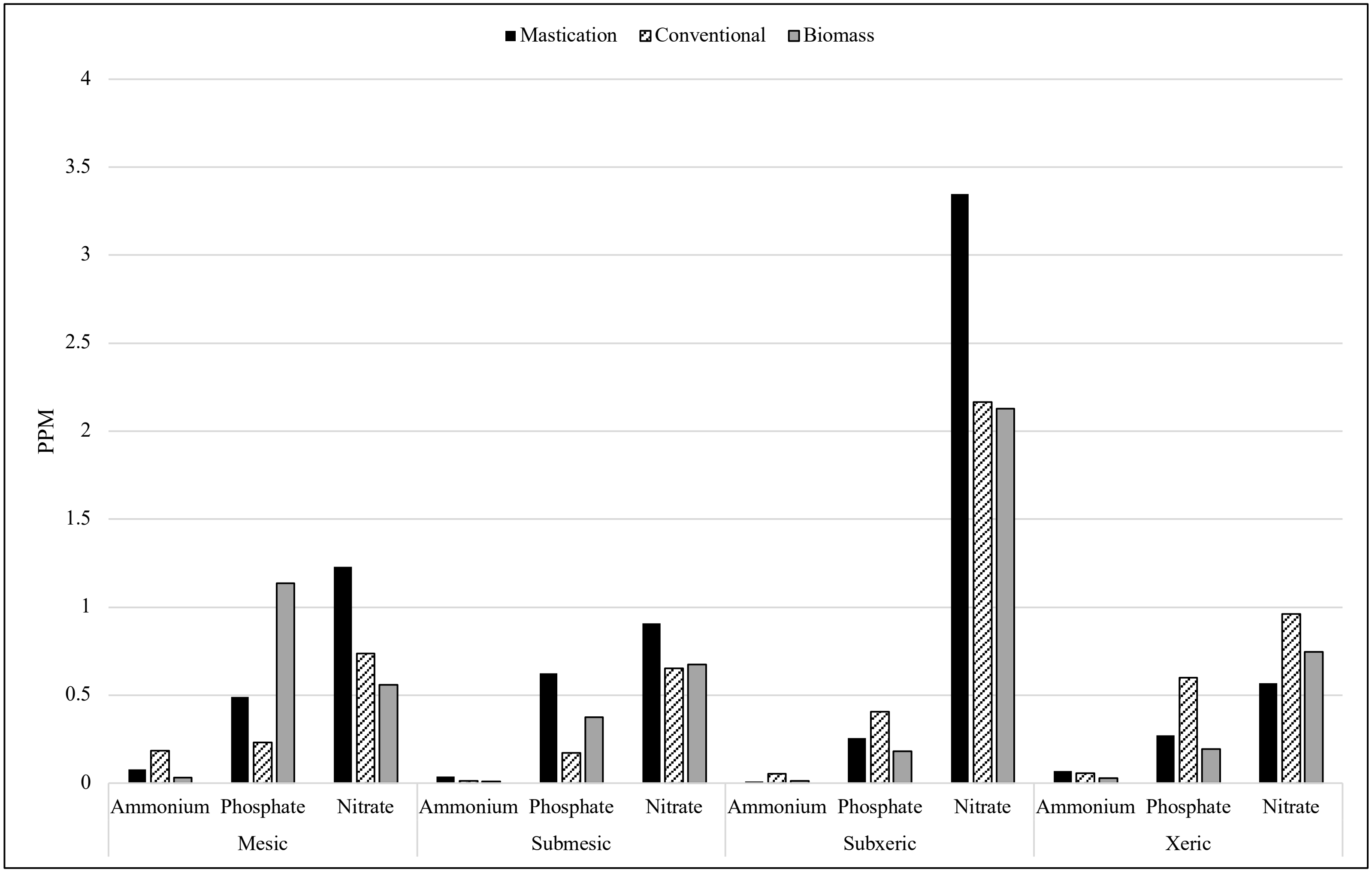

Figure 1.15: Average predicted Gaussian peaks for nutrient concentrations (ppm) across moisture groups and slash treatments in Camden, SC.

1.3.2 Soil Bulk Density

Soil bulk density (g cm^{-3}) was positively correlated with xeric conditions, increasing on average as the soil moisture gradient became drier and sandier (Table 1.8). Significant differences were found between moisture blocks (F= 5.910, P=0.001) but not between treatments (F= 0.106, P= 0.900), and there were no significant interaction effects (F= 0.572, P= 0.752). Xeric and subxeric soil bulk densities were 15.6% and 13.3% higher, respectively, compared to mesic sites (Figure 1.16). Soil porosity tended to decrease as sites became drier (54% to 43%), indicating a tighter hold on soil moisture with xerification.

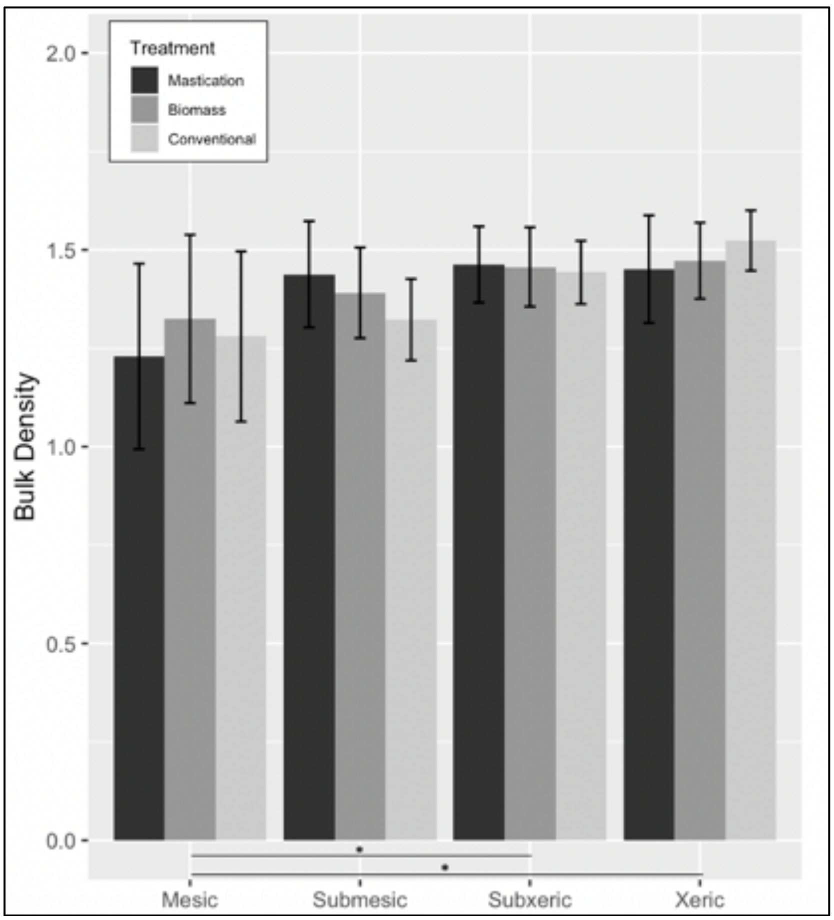

Figure 1.16: Bar plot with 95% confidence intervals for mean soil bulk density (g cm^{-3}) for treatments across soil moisture groups in the Carolina Wiregrass Gap in Camden, SC. Statistically different means are represented by an asterisk located next to the legend for between-treatment differences and above x-axis categories for between-group differences ($\alpha < 0.05$).

1.3.3 Downed Woody Material

1.3.3.1 Coarse Woody Debris (CWD)

A preharvest inventory of CWD found that there were no significant differences in the mean amounts of biomass (kg ha^{-1}) or the CWD predicted weight of carbon (kg ha^{-1}) across the moisture blocks or the treatment zones (Table 1.9, 1.10, respectively). There was an average of 168.3 kg ha^{-1} (Min = 11.6 kg ha^{-1}; Max = 3,303.9 kg ha^{-1}) across the entire study area, with only two samples indicating more than 1,000 kg ha^{-1}. The average weight of Carbon for CWD was low across the study site as well, resulting in only one sample over 1,000 kg ha^{-1} and an average of 84.7 kg ha^{-1} (Min=6.1 kg ha^{-1}; Max= 1,622.2 kg ha^{-1}). Biomass and carbon averages were not significantly different between CWD decay classes either. However, they did increase from decay class 2-5, while decay class 1 exhibited the greatest amount of biomass, 178% more on average compared to the next highest average displayed by decay class 5 (Table 1.11). Over 50% of the CWD pieces were classified with a decay class reduction factor of 4 (Figure 1.19).

Post-treatment CWD analysis found that there were significant differences in the means between treatments for CWD biomass ($F=4.091$, $P= 0.030$) and CWD carbon estimates ($F=4.147$, $P=0.028$). However, for both woody biomass and carbon, there were no significant findings between moisture groups ($F=1.727$, $P=0.188$; $F=4.147$, $P=0.284$; respectively) and there were no interaction effects between independent variables ($F=2.434$, $P=0.056$; $F=2.486$, $P=0.052$; respectively). Predictably, the amount of CWD biomass increased across treatments from our masticated sites (lowest) to our conventional harvest sites (highest). Pairwise comparisons found that, on average, conventional harvesting produced 7,349 kg ha^{-1} (142%) more CWD and 3,645.8 kg ha^{-1} (141%) more CWD carbon compared to our mastication treatment (Figure 1.17). Furthermore, there was a significant increase in the average amount of

CWD post-treatment compared to our pre-treatment findings (F=82.58, P= <0.001) (Figure 1.18). Across the entire study site, our post-harvest estimates for CWD biomass and carbon were 8,239.1 kg ha^{-1} (SD=7,964.5 kg ha^{-1}) and 4,110.3 kg ha^{-1} (SD= 3,950.8 kg ha^{-1}), respectively. Additionally, while decay class analysis was limited due to our method of collection, the number of stems was negatively correlated with an increase in decay class, with class 1 accounting for 38% of the measured trees, our largest representation (Figure 1.19).

Table 1.5: Mean pre-treatment CWD biomass (kg ha^{-1}) and carbon (kg ha^{-1}) estimates compared across woody decay classes in Camden, SC. The effects of decomposition increase with an increase in decay class (Waddell, 2002).

Decay Class	Average CWD Biomass (kg ha^{-1})	Average CWD Carbon Weight (kg ha^{-1})	Percentage of Woody Material (%)
1	837.42±1385.53	678.47±415.40	7%
2	74.49±57.41	37.77±30.04	8%
3	93.54±135.20	46.57±66.17	22%
4	125.60±171.87	64.23±89.20	53%
5	300.76±403.16	152.03±203.29	11%

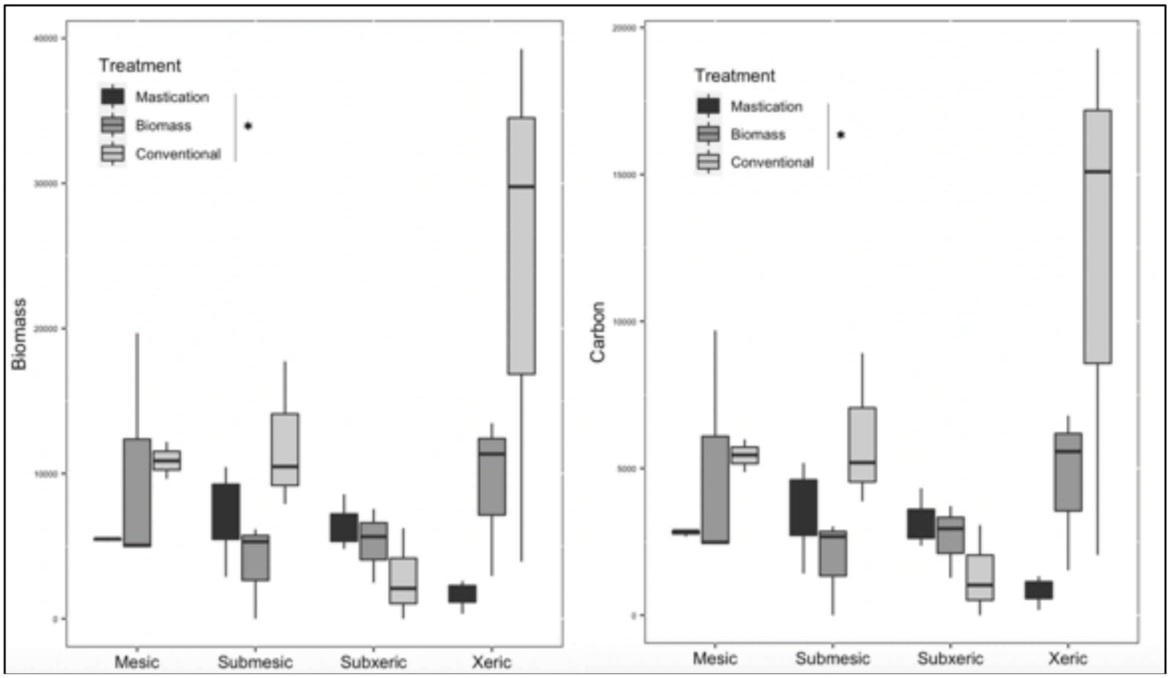

Figure 1.17: Post-treatment boxplots of CWD biomass (kg ha^{-1}) (left) and CWD carbon estimates (kg ha^{-1}) (right) for treatments across soil moisture groups in the Carolina Wiregrass Gap in Camden, SC. Statistically different means are represented by an asterisk located next to the legend for between-treatment differences (α < 0.05).

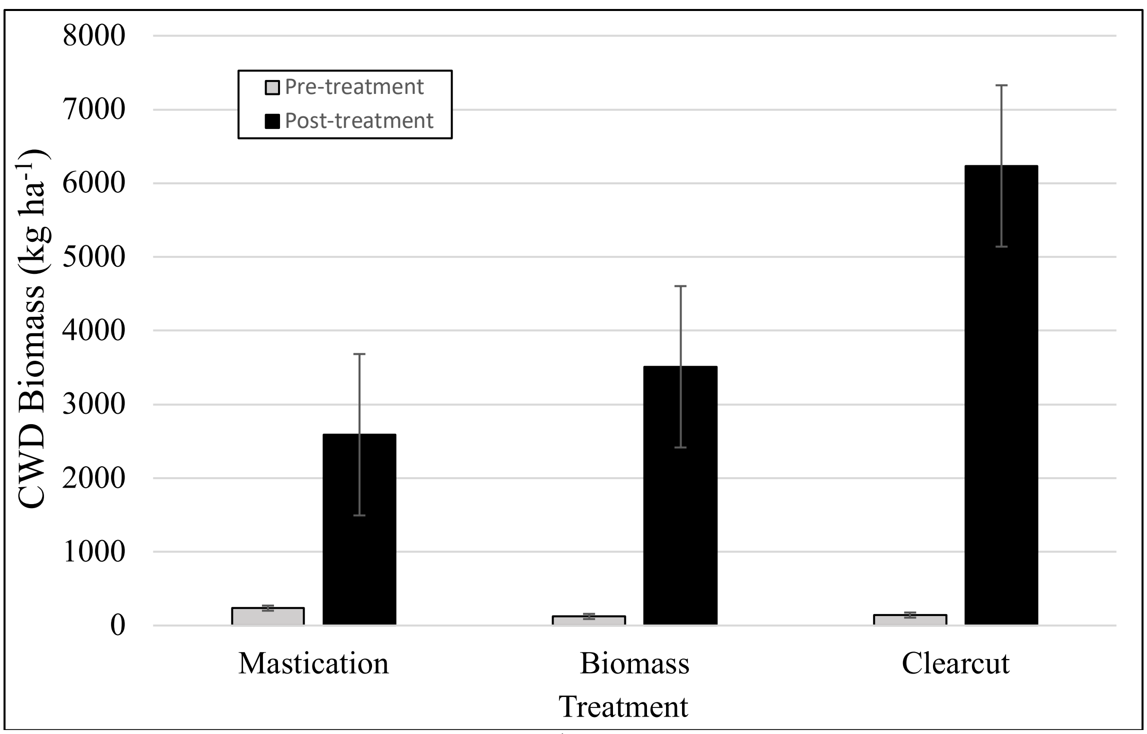

Figure 1.18: Bar plot of mean CWD (kg ha^{-1}) comparison between pre- and post-treatment findings across treatments in Camden, SC, with significant mean differences between-dates indicated by the presence of an asterisk above the legend (α < 0.05).

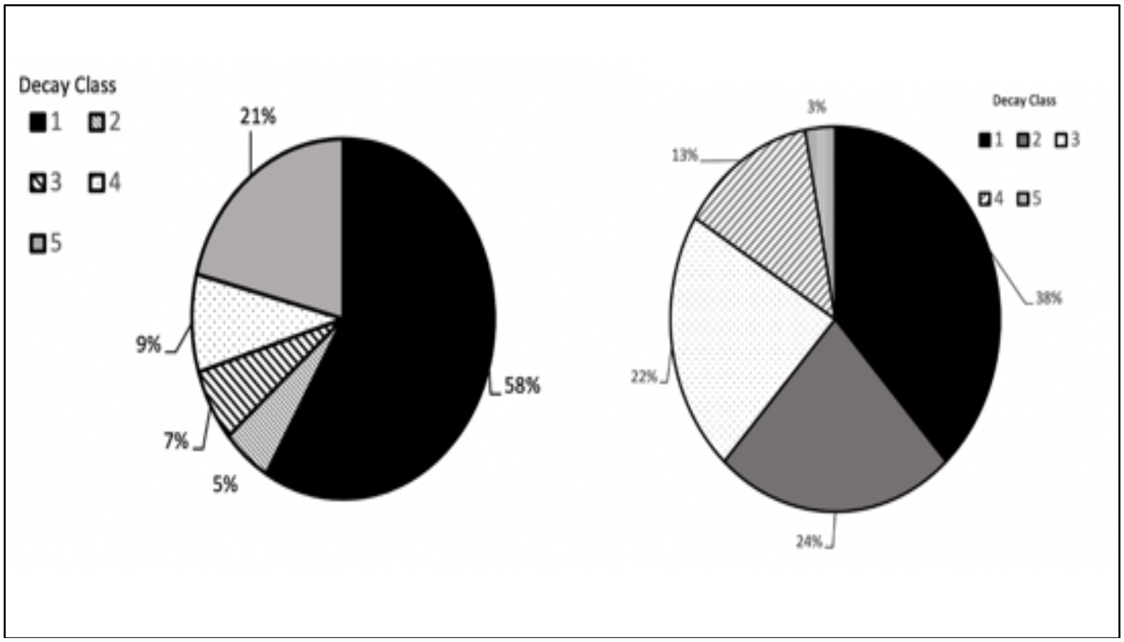

Figure 1.19: Pre-treatment (left) and post-treatment (right) comparison of CWD stem representation for all sites based on decay class in Camden, SC. Increasing decay classes correlate with increases in decomposition (Waddell, 2002).

1.3.3.2 Fine Woody Debris (FWD)

The average pre-treatment FWD across all sites was 2,802.4 kg ha^{-1} and there were no significant mean differences across moisture groups (F= 2.548, P= 0.080) or across treatment zones (F= 0.258, P= 0.775). However, post-treatment measurements did find significant differences between our treatments for the 1, 10, and 100-hr fuel classes (F= 22.618, P= <0.05; F= 5.090, P= 0.014; F= 5.785, P= 0.009; respectively). On average, the mastication treatment produced 6,642.4 kg ha^{-1} and 6,116.8 kg ha^{-1} more woody material in the 1-hr fuel class and 7,316.9 kg ha^{-1} and 7,236.0 kg ha^{-1} in the 10-hr fuel class compared to the biomass and conventional harvest treatments respectively (Figure 1.20). This trend reverses with our 100-hr fuel class, producing the least amount of fuel per unit area in masticated fuel treatments (Figure

1.20). The conventional harvest treatment produced the largest number of 100-hr fuels, and significantly more than the mastication treatment (75% more kg ha^{-1}) (Figure 1.20). Across all sites, the harvest generated an average of 5,328.5 kg ha^{-1}, 11,978.5 kg ha^{-1}, and 16,398.5 kg ha^{-1} for our 1, 10, and 100-hr fuel classes, respectively (Figure 1.20).

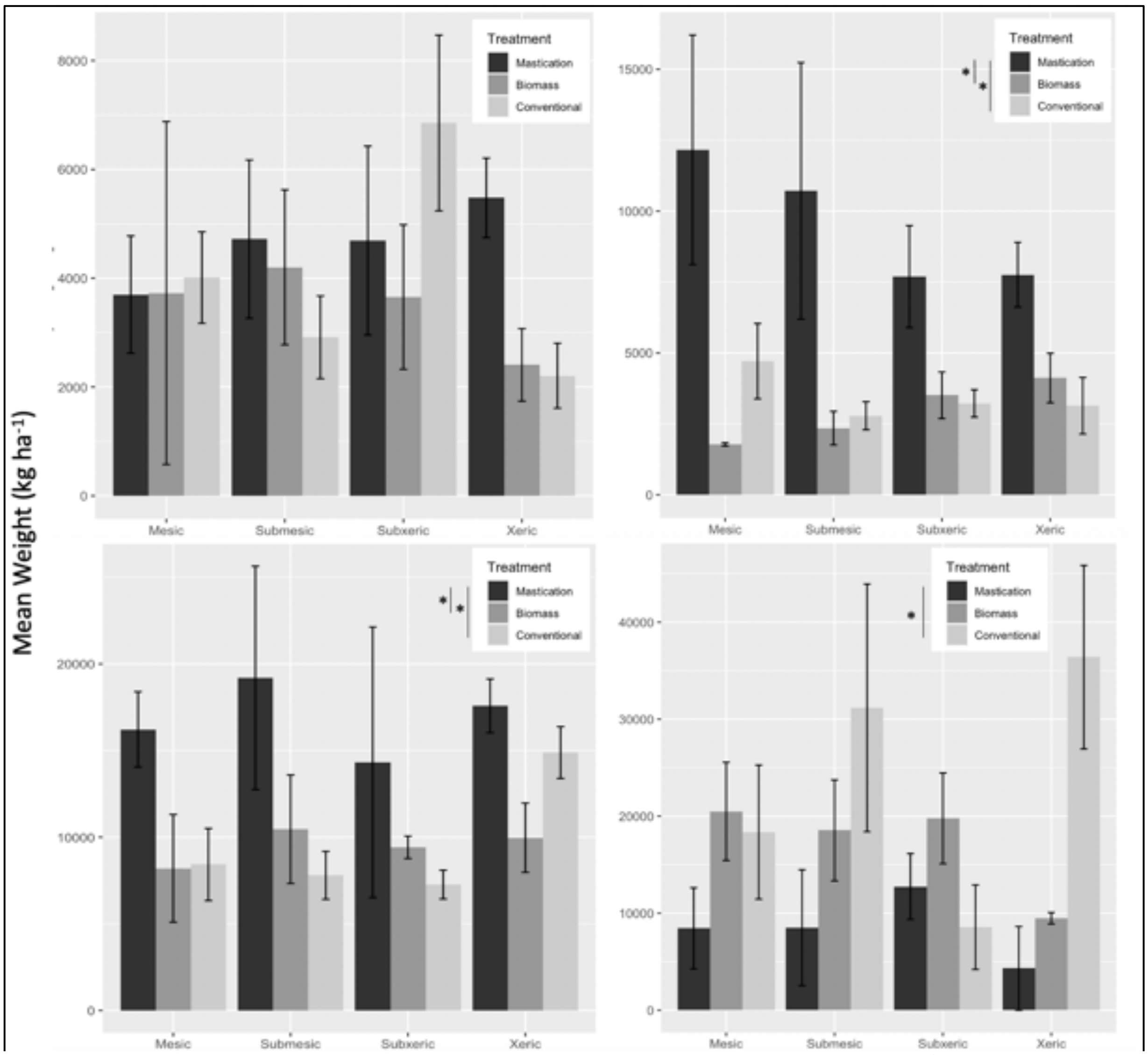

Figure 1.20: Post-treatment bar plots with error bars for mean weights (kg ha^{-1}) of litter and duff (upper left), 1-hr fuels (upper right), 10-hr fuels (lower left), and 100-hr fuels (lower right) across moisture groups and treatments. Statistically different means are represented by an asterisk located next to the legend for between-treatment differences ($\alpha < 0.05$).

1.3.3.3 Litter and Duff

Pre-treatment destructive samples for litter and duff were not taken. However, a CVS conducted in 2018 found that the average depth was 2.50 cm across the entire site and primarily consisted of low-quality needle litter. Post-treatment destructive sampling found that the mean dry weight for litter and duff was 4,046.7 kg ha^{-1} (SD= 2432.6 kg ha^{-1}) across the entire study area and that the means across treatments (F= 0.631, P= 0.541) and moisture groups (F= 0.747, P= 0.535) were not significantly different. Using average pre-treatment duff and litter depths, we estimated that post-treatment bulk density averaged 16.2 kg m^{-3}. However, because these estimates were obtained from the combination of post- and pre-treatment values, these predictions are subject to skepticism.

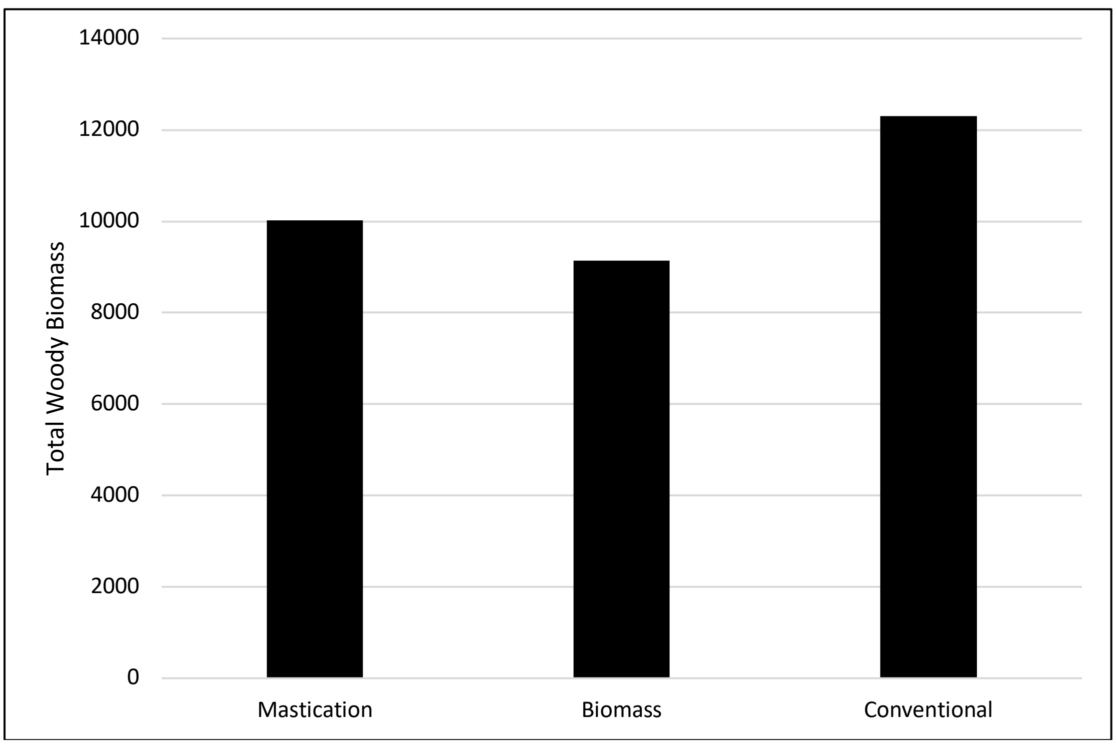

Figure 1.21: Average post-treatment woody biomass (kg ha^{-1}) across all sites for a seed-tree timber harvest in Camden, SC.

1.3.4 Soil Water Tension

Pre-harvest measures of soil water matric potential yielded significant differences in the means across dates (F= 20.103, P= <0.05), but not across our moisture gradient (F= 0.912, P= 0.438) or treatment zones (F= 1.763, P= 0.177). However, following the implementation of our treatments, we found significant differences between moisture blocks (F= 14.481, P= <0.05), treatments (F= 20.748, P= <0.05), and dates (F= 42.615, P= <0.05), indicating that, in addition to sporadic rainfall events, slash manipulation was influential in determining soil water availability for vegetation. Our statistical analysis produced significant interaction effects between our moisture gradient and our treatments (F= 6.631, P= <0.05), as well as between our treatments and recording dates (F=11.212, P= <0.05) (Figure 1.22). Our masticated fuel treatment remained consistently high in water availability across dates, and the greatest differences between treatments were observed in our xeric sites during dry periods. Soil tension increased along our moisture gradient from mesic to xeric but did not differ significantly between biomass and conventional harvest treatments. However, for five sampling periods during the growing season, moisture became limiting (<-30 kPa) or near-limiting for the biomass and conventional harvest treatment, while soil water in masticated fuel treatments never dropped below -20 kPa. For these five sampling dates, soil moisture was significantly higher than other treatments (Figure 1.22).

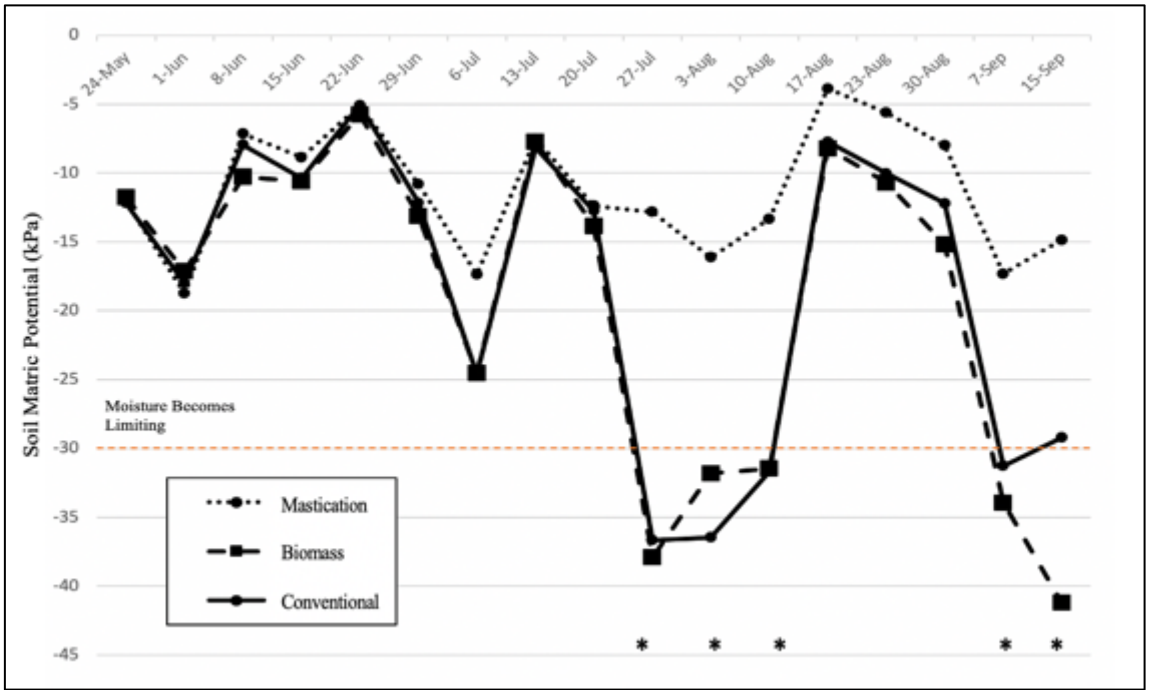

Figure 1.22: Average soil matric water potential (kPa) readings across slash manipulation treatments taken weekly from May 24, 2021 to September 15, 2021 in Camden, SC. Dates marked by an asterisk have significantly lower soil water tension in the mastication treatment compared to both other treatments ($\alpha < 0.05$).

1.4 Discussion

1.4.1 Soil Characteristics

Overall, our soil composition was characteristic of Carolina sandhill ecosystems, exhibiting high acidity with extremely well-drained sandy soils across the entire site. We found similar findings to Brendemuehl (1967) and Faust (1976), who noted that Sandhills fall line habitat averages less than 2% organic matter and retains a soil pH range from about 4.0-5.6. Organic matter, located on soil surfaces in this region, oxidizes quickly and releases nutrients that are readily leached from upper soil levels by frequent rainfall events (Burns and Hebb, 1949; Faust, 1976; Murphy, 2016). While this is not an uncommon finding, it is an important reference to understanding the effects of treatments on nutrient availability and long-term community development in the region. Low pH values can cause extreme deficiencies in soil nutrients, especially nitrogen and phosphorus. For example, nitrification is highly sensitive to changes in soil pH, litter composition, and soil water content, essentially ceasing at values less than 5.0 pH (Linn and Doran, 1984; McCaskill et al., 2018; Wilson et al., 2002). Additionally, inorganic phosphorus is frequently deficient at pH values below 5.5, and these effects are often exacerbated by a lack of organic matter (Mahler, 2004). However, significant differences found across our study area could be a result of legacy phosphorus resulting from previous fertilization of agricultural fields. Craft and Chiang (2002) discovered that C:N ratios increased from wetlands (17:1) to uplands (43:1) and that organic compounds accounted for 97% of N and 82% of P in xeric soils for longleaf pine habitats. This combination could explain the insufficient levels of NO_3^-, NH_4^+, and PO_4^{3-} concentrations found in our study.

This was interesting considering other longleaf restoration studies found that NH_4^+ concentrations can supplement longleaf growth during growth-limiting environmental periods exhibiting low NO_3^- concentrations (McCaskill et al., 2017). In fact, though, we observed that NO_3^- concentrations were considerably higher than NH_4^+ concentrations but were still low (<1 ppm) in most of our soil tests and solutions. However, we did observe a spike in NO_3^-concentrations in our subxeric site, but there were no discernable trends for nutrient availability across our treatments. Wilson et al. (2002) found that soil temperature increases were the principal influence in producing large pools of inorganic nitrogen in xeric sites, but this does not explain why there was not a similar pulse of nitrogen compared to our subxeric results. We also noted a parallel spike in soil organic matter in our subxeric site, but it did not seem to influence NH_4^+ or PO_4^{3-}.

These results lead us to believe that the study area was not only nitrogen-poor but was probably nutrient deficient as a result of natural processes and low litter quality. As a result, it could be argued that biomass harvesting is more sustainable than originally predicted considering nutrient stocks were not affected by our treatments. While we observed significant differences in NO_3^- and NH_4^+ concentrations, none of these differences were found in masticated fuel treatments, and none of the differences showed consistent trends between moisture groups. Young et al. (2014), Kobziar et al. (2013), and Overby and Gottfried (2017) found similar results to our findings, showing that the addition of masticated fuels did not result in short-term changes to nutrient availability, including nitrogen. Correspondingly, our findings indicate that microsite differences were the primary factors in determining concentrations and that our treatments do not initially result in changes to nutrient availability but could play a part in short-term immobilization and long-term mineralization. Woody plant residue, leftover as a byproduct of mastication and timber harvesting, has a high C:N ratio, resulting in short-term periods of immobilization (Rhodes et al., 2012; Sahrawat, 2008; Young et al., 2014). Unfortunately, because longleaf ecosystems in sandhill regions are already nutrient deficient, many restorative methods that remove soil carbon, such as prescribed burning and biomass harvesting, result in short-term changes to nutrient availability but have little long-term effect on soil quality (Lavoie et al., 2014; Raison et al., 2009). For example, after only one year following a prescribed fire in a longleaf pine stand in the Florida Sandhills, C and N pools recovered to 67% and 76%, respectively, of pre-treatment levels (Lavoie et al., 2010). Due to the fragmented and compacted nature of masticated fuels, decomposition typically happens at faster rates than coarse woody debris, decreasing the period of immobilization and resulting in mineralization (Rhodes et al., 2012; Young et al., 2013). Due to an increase in the surface area to volume ratio, this could potentially influence long-term nitrogen dynamics differently. For example, Rhodes et al. (2012) noted an initial decline in plant-available nitrogen following mastication, but then a rapid rate of mineralization, showing an elevated level of available nitrogen (+32%) three to five years following the treatment. Young et al. (2013) found similar results and concluded that masticated fuels could improve pine seedling growth with an increase in available nitrogen and moisture retention. It is also very possible that we are exhibiting a period of immobilization compared to pre-harvest concentration levels and that, because the C:N ratio would increase similarly across all treatments just in different spatial arrangements, our initial findings do not exhibit any differences between treatments. Pre-treatment concentrations were not evaluated but long-term studies on mineralization could still indicate significant fluctuations in soil organic matter and nutrient availability. In an already nitrogen-limited environment, immobilization may not be as apparent as an influx of nitrogen, meaning that long-term studies should continue to monitor changes in soil chemistry.

1.4.2 Soil Bulk Density

Average soil bulk density readings increased with xerification ranging from 1.23 g cm^{-3} in mesic sites to 1.52 g cm^{-3} in xeric sites, an unsurprising result considering total soil porosity decreased with xerification. For non-compacted sites, the total volume of surface soil is, on average, 45% soil particles, 5% organic matter, and 50% pore space occupied by either air or water (Cavigelli et al., 1998). As water-filled pore space exceeds 60%, the level at which maximum aerobic microbial activity occurs, ammonification, nitrification, and soil respiration all decrease significantly due to aeration limitations, while water-filled pore space exceeding 80% results in large amounts of denitrification (Linn and Doran, 1984; USDA-NRCS). None of our sites exhibited this level of saturation, and our soil porosity remained consistent, between 43-54%, but our organic matter was considerably lower. Soil bulk density varies greatly by soil type and soil depth, with porous soils containing large amounts of organic matter, silt, or clay having lower bulk densities (0.5-1.6 g cm^{-3}) compared to sandy soils (1.3-1.7 g cm^{-3}) with less total pore space and deeper soil layers with high rates of compaction (> 2.0 g cm^{-3}) (McKenzie et al., 2002; Brown and Wherrett, 2021). Compaction decreases total pore volume and reduces available water accessible by plants, with densities greater than 1.6 g cm^{-3}, negatively affecting root growth (Brown and Wherrett, 2021; Hunt and Gilkes, 1992; McKenzie et al., 2004). For sands and loamy sands, like those present in the Carolina Sandhills, the USDA-NRCS identifies ideal bulk densities for plant growth as <1.60 g cm^{-3} with bulk densities >1.80 g cm^{-3} completely restricting root growth. This could potentially affect our long-term vegetation response in xeric sites, especially if soil water is scarce.

These findings were contrary to our initial expectations, which assumed that mastication would increase bulk density due to increases in compaction or decrease due to increases in woody material. For example, soil bulk density can increase considerably following a timber harvest due to the influence of heavy machinery. As a result, seedling establishment and site productivity can decrease (Reisinger et al., 1988). Wang et al. (2005) confirmed this during a timber harvest, with bulk density increasing by 126.4 kg m^{-3} along skid trails, with 57% of the change occurring after two loaded passes. Moreover, Lockaby and Vidrine (1984) found that soil compaction was 12% higher in areas that experienced high levels of equipment traffic, resulting in significant declines in the number of trees per acre (88-91%) and height growth (39-59%). However, we found no significant differences between treatments, indicating that the only significant trend was an increase in bulk density with an increase in xeric soils. This could likely result from the already established internal road system, which redirected machine traffic, reducing its effects across the site. Long-term monitoring should be conducted across the study area to quantify the effects of decomposition on bulk density, especially in masticated sites that increase the surface area of biomass on the forest floor.

1.4.3 Downed Woody Material

Pre-treatment measurements of downed woody material found that CWD and FWD fuel loading were similar across the study area, confirming that our site conditions before the harvest were relatively uniform. Post-treatment biomass (FWD and CWD) was, on average, different across treatments. Biomass harvest treatments exhibited the lowest amount of downed woody material, which was expected, but masticated fuels should have exhibited the same amount of woody material compared to our conventional harvest. Conventional harvesting was estimated to produce over 2,000 more kg ha^{-1} compared to mastication. This finding could be a result of our sampling methods which assumed a consistent shape for CWD logs. Additionally, we used a plot-based method for measuring all fuels, which is less consistent than a hybrid methodology (Kane et al., 2009). Due to the irregular shape of masticated fuels, it is recommended that quantification of woody material should come in the form of hybrid methodology in which masticated fuelbeds are measured using plot-based measurements and larger, coarse woody debris is measured using the standard planar intercept method outlined by Brown (1974) (Kane et al., 2009).

Predictably, post-treatment measurements of DWM found noticeably more CWD in conventional harvest sites than our masticated fuel site, but essentially no difference between our biomass and conventional harvests. Biomass harvesting typically utilizes in-woods chippers that consume downed and dead woody material, logging slash, and brush, reducing the total amount of biomass on a site. Nevertheless, because our treatments only simulated a biomass harvest, there were limitations with how accurately we could clear our sites and, thus, resulted in minimal differences for woody fuel loading between treatments. By creating slash piles in lieu of chipping leftover slash, much of the woody debris was lost in transit, generating more CWD than our masticated fuels operation, but not significantly less than our conventional harvest treatment. Therefore, we expected that our treatment effects would be most apparent in our masticated fuel sites as the composition of residual fuels was physically altered with the addition of mastication. Masticated treatments had significantly more 1-hr and 10-hr fuels than other treatments, which showed no difference. This is similar to other studies that investigated mechanical chipping effects on ecological dynamics in which there was a concentration of woody material in lower time-lag classes (Kane et al., 2009; Kreye et al., 2012; Kreye et al., 2014). Based on the nature of mastication operations, a concentration of smaller fuels is expected. However, total fuel loading across masticated sites is highly variable, with one study finding 15,300-63,400 kg ha^{-1} across ten different treatments (Kane et al., 2009). Fuel loading configuration for mechanical treatments is dependent on the material, the amount of material, and the type of fragment produced, all of which influence post-disturbance ecology (Jain et al., 2018).

1.4.4 Soil Water Tension

Soil water matric potential was consistent across all sites for pre-treatment findings, with our largest fluctuations seen between dates that did and did not experience precipitation events close to our observation period. Expectedly, an increase in matted fuel beds produced by our mastication treatment developed higher and much more consistent moisture retention rates even during moisture-limiting environmental periods. Compacted woody fuelbeds can inhibit vegetation responses, flammability, and can significantly increase moisture retention (Jain et al., 2018; Kreye et al., 2012; Kreye et al., 2014). Kreye et al. (2012) found that fuelbeds composed of 1-hr and 10-hr fuel particles increased water retention in lower layers, with response drying times ranging from 40 to 69 hours. This can potentially result in positive vegetation responses and pulses of available nutrients, known as an Assart effect, but can also limit prescribed fire capabilities and immobilize nitrogen in the short-term (Kreye et al., 2012; Kreye et al., 2014; Overby and Gottfies, 2009; Young et al., 2013). Mastication was the only treatment that exhibited significantly less moisture tension than other treatments and was also the only treatment that did not become limiting.

Additionally, our differences were exacerbated in xeric sites, showing an increase in matric water tension with the removal of biomass and an increase in xeric conditions. Mesic, submesic, subxeric, and xeric sites produced averages of -12.5 kPa, -13.7 kPa, -15.3 kPa, and -22.7 kPa, respectively, representing a 45% increase in soil moisture tension from our mesic to our xeric sites. Five readings resulted in average soil matric water potentials dropping below or nearing -30 kPa for the biomass and conventional harvest treatment. However, for all of these points, our masticated fuel treatment remained consistent. Masticated fuel beds produced water tension rates 37-41% lower than other treatments even during the height of the growing season when the average annual precipitation drops by over 30mm. Our findings indicated that masticated fuel beds can significantly increase the amount of available water and are most influential in xeric sites during periods with low amounts of rainfall. Loose sandy soils can produce additional heat through reradiation, which subsequently increases the evapotranspiration of surface moisture (Faust, 1976). Masticated fuels have the ability to reduce the rate of surface water evaporation, stabilize soil temperature, and slow the rate of infiltration (Jain et al., 2018; Qin et al., 2016). These effects can positively influence vegetation responses but can also sporadically influence nutrient availability depending on varying environmental conditions (Jain et al., 2018).

1.5 Conclusions

Our study indicates that nutrient availability was not affected by our treatments, and the intense removal of biomass did not result in a decrease in soil nutrients. This could indicate that the practice of biomass harvesting could be sustained in the region. However, mastication was found to reduce woody fuel loading for larger time-lag classes while simultaneously retaining soil moisture at significantly higher rates than other treatments. By exhibiting consistent soil retention rates, even during the driest parts of the growing season, masticated fuels could potentially result in a much different successional trajectory compared to other treatments. Responses to mastication are inconsistent, and these effects are both positive and negative depending on the nature of the operation and the environmental conditions (Jain et al., 2018). In the sandhills region, an area characterized by deep, well-drained soils, this can promote longleaf pine habitat but increase compaction. More than likely, this will result in an initial period of immobilization but will likely result in an influx of nitrogen and soil organic matter once decomposition reaches its peak. However, masticated fuel beds often smolder when prescribed fire is employed, meaning that if there is no positive vegetation response, fire applications could be limited. The absence of wiregrass could potentially add another critical aspect to the success of this restoration and the influence of masticated fuels. Without a robust herbaceous understory present to facilitate burning, the area could quickly return to a mix of loblolly and longleaf with a scrub oak midstory. It is possible that restoration treatments that manipulate logging slash differently could influence understory and overstory restoration success. To quantify these changes and understand the influence of different restoration approaches in the region, additional management applications and long-term observational studies need to be applied to understand ecological trends.

CHAPTER TWO

SPATIAL SUITABILITY MODELING FOR LONGLEAF PINE RESTORATION IN THE SOUTH CAROLINA SANDHILLS WIREGRASS GAP

Abstract

Currently representing less than 3% of its natural range, longleaf pine habitat restoration has become an important topic for forest management in recent decades due to its inherent ecological value. As a result, spatial investigations into changing forest characteristics and potential restoration sites are important components in increasing longleaf pine occurrences and range expansions. Technological advances have also given researchers the ability to remotely analyze niche regions as subsets of larger complexes. This study developed multiple habitat suitability models for longleaf pine restoration in the South Carolina Sandhills Wiregrass Gap, a subregion of the Carolina Sandhills devoid of wiregrass, an important ecological component to a majority of longleaf pine ecosystems. Using advanced spatial modeling and analysis tools, weighted overlays of ecological raster data were employed across parcels within the region to predict potential sites for longleaf restoration. We found that only 1.4% of parcels within this region were potentially appropriate for restoration (>30 hectares) but still accounted for almost a third of the total area. Our models predicted that, across the entire Wiregrass Gap, 26.8-29.8% of the area was located within parcels larger than 30 hectares was appropriate for restoration. Our findings demonstrated that about 70% of the study area was either developed, was too small, or had ecological characteristics that were inconsistent with restoration initiatives. Within parcels larger than 30 hectares, 83-91% of the area was appropriate for restoration. This study is intended to provide a working model for

researchers and forest managers by ranking sites based on applicability. However, because of the variability associated with spatial modeling and identifying ecological restoration criteria, further testing needs to be applied to evaluate the accuracy of these models.

2.1 Introduction

2.1.1 History of Longleaf Pine Ecosystems

Longleaf pine (*Pinus palustris* Mill.) dominance came to fruition in the Atlantic Coastal Plain between 7500-5000 years ago, during the mid-Holocene period (Brockway, 2005; Landers et al., 1995). During this time, climatic conditions progressed through several warming periods and, as the glaciers began to retreat, plant species migrated from the south to previously glaciated areas, fire occurrences became more common, and environmental conditions began to favor longleaf pine ecosystems (Brockway, 2005; Landers et al., 1995). While the last glacial maximum did not overlap with the historical range of longleaf pine ecosystems, the changing climatic conditions were undoubtedly significant. Following their establishment, longleaf pine habitats continued to spread and dominate the landscape for thousands of years. Extensive anthropogenic influences and regular lightning strikes created a recurrent disturbance regime denoted by frequent, low-intensity fires that helped create one of the most diverse and species-rich ecosystems in North America (Brockway, 2005).

Before European development, the southeastern United States comprised more than 30 million hectares of pure longleaf pine forest, with an additional 7 million hectares of longleaf pine being represented in mixed stands (Frost, 1993; Van Lear et al., 2005). While the exact extent of longleaf pine ecosystems in pre-settlement forests remains unknown, pre-contact range models and predictions have been generated using historical records, eyewitness accounts, and past disturbance indicators dating back to 1896 (Frost, 1993; Outcalt, 2000). These “pre-European” ecosystems thrived over many landscapes, ranging from eastern Virginia to Texas, typically described as longleaf pine savannas (Landers et al., 1995). A longleaf pine savanna can be depicted as a grassland ecosystem dominated by a rich and diverse assortment of grasses and forbs, but with a monoculture of low-density longleaf pine trees in the overstory. Historically, these ecosystems were maintained by a pyric herbivory system supported by frequent fire occurrences primarily caused by anthropogenic influences combined with lightning events and large-ungulate grazing (Brockway et al., 2005; Van Lear et al., 2005). Native American cultures regularly burned the landscape to provide various benefits, including fuel reduction, wildlife habitat enhancement, ecosystem manipulation, and parasite and insect reduction (Brockway, 2005).

Longleaf pine exploitation first manifested itself in the 17th century through pitch, tar, and turpentine accumulation as part of an effort to bolster British naval stores during a time of intense European expansion and development (Outland, 2004). These naval stores were highly sought after from the early 1600s through the mid-1800s, where, by 1850, North Carolina had become the world's leading supplier of pitch and tar, based primarily on their exploitation of longleaf pine forests (Frost, 1993; Outland, 2004). Longleaf pine ecosystems began their most significant decline during the late 18th and early 19th century, when the turpentine and logging industries began to exploit these ecosystems, with essentially no restoration plan (Outland, 2004). According to the U.S. Census of Agriculture (1902), by 1900, 27% of all longleaf pine upland habitats had been converted for agricultural use (U.S. Census of Agriculture, 1902; Frost, 1993). Colonists would often clear forests by girdling trees to make way for grazing and cropland establishment. Between 1750 and 1850, almost all longleaf communities located on fertile soils in the Piedmont were converted to either farmland or pastureland (Williams 1989; Frost, 1993). In a study published by researchers at Clemson University in 2005 about the history and restoration of longleaf pine ecosystems, it was noted that, due to logging and improper management techniques applied in the mid 19th century, longleaf pine forests became one of the most endangered ecosystems in the United States, accounting for less than 3% of its original range (Van Lear et al., 2005).

The logging industry, which was relatively slow-paced during the colonial period, saw some minor success with the introduction of water-powered mills and then steam-powered machinery in the late 19th century (Frost, 1993; Jose et al., 2003). Until the mid-1800s, commercial logging efforts were focused mainly on the Northeast and then on the Midwest, where vast white pine (*Pinus strobus* L.) forests were exploited. However, following the Civil War, commercial logging efforts began to focus on southern forests. Steam-powered logging equipment had become more advanced, and the turpentine industry made a resurgence in both North and South Carolina, all of which acted as a precondition to the devastation of longleaf pine ecosystems (Frost, 1993). Between 1870 and 1930, there was an intense period of logging which saw the removal of nearly all of the remaining virgin forest in the southeast (Frost, 1993; Jose et al., 2003).

Additionally, a severe and widespread fire suppression policy was implemented in the early 1900s, accelerating the loss and degradation of longleaf pine forests. In 1943, costs associated with fire suppression efforts constituted 20% of the total budget for the U.S. public treasury (Loveridge, 1944). Foresters in the 20th century found that fire exclusion in longleaf forests resulted in dense hardwood, loblolly, and slash pine understories (Sherrard, 1903; Frost, 1993). Without fire or a reasonable substitute to reset the environment to an earlier seral stage, mesophytic pines, hardwoods, and shrubs outcompete longleaf pine seedlings by quickly overtopping them and shading out the understory (Brockway, 2005). This succession often leads to a xeric hardwood or mixed pine community (Abrahamson, 1984; Stout and Marion, 1993). By 1900, longleaf pine ecosystems had been largely destroyed in both Virginia and North Carolina. Ecosystems that did get reforested were often replaced with loblolly pines (*Pinus taeda*) and other plantation-style pine forests which promoted fast-paced timber growth with less management and a higher economic yield (Frost, 1993; Jose et al., 2003). Today, there are approximately 1.2 million hectares of original longleaf pine habitat left, representing a 97% overall decline in its historical range (Frost, 1993; Wahlenberg, 1946; U.S. Fish and Wildlife Service, 2003; Outcalt and Sheffield, 1996; Van Lear et al., 2005).

2.1.2 Restoration and Spatial Modeling in the SC Sandhills Wiregrass Gap

Within this range, longleaf pine habitats can be subdivided into many ecosystem types that range from clayhills to sandhills to low-lying flatwoods. One of the areas of particular importance is the Carolina Sandhills, which has received attention in recent decades as a prime ecoregion for longleaf pine restoration (Aschenbach, 2010; Dagley et al., 2002; Harrington et al., 2021). The South Carolina Sandhills are located just below the Fall Line, run southwest to northeast across the state, and are primarily comprised of marine sands and clays deposited through erosion and weathering in the Piedmont region of SC (Griffith et al., 2002; Murphy, 2016, USDA-NRCS, 2006). These soils are deep and highly permeable, resulting in rapid rainwater drainage and a nutrient deficient topsoil layer that is often unproductive (Murphy, 2016).

Historically, longleaf pine-wiregrass communities have been the dominant vegetation type in these systems (Brockway and Lewis, 1997; Peet, 1993; Wall et al., 2012). Wiregrass, a warm-season, perennial bunchgrass native to the southeast, acts as a critical foundation species and a source of structure and high-quality fuel for understory fire facilitation in pine-wiregrass ecosystems (Hardin and White, 1989; Mulligan et al., 2002; Noss, 1989; Seamon et al., 1989). Wiregrass acts as the dominant ground species within its range, often exhibiting >75% cover with overstory basal areas between 9.2-11.5 m^2 ha^{-1} (40-50 ft^2 $acre^{-1}$) (Outcalt et al., 1999). However, wiregrass is absent in a relatively large area of the Carolina Sandhills, commonly referred to as the 'Carolina Wiregrass Gap'. There has been very little information on why this gap occurs, only that it separates two species of wiregrass, *Aristida stricta* L. to the north and *Aristida beyrichiana* Trin. & Rupr. to the south (Peet, 1993; USDA-NRCS, 2014). As a result, this area is distinct in its ecological makeup and requires a different management approach than other longleaf systems in similar environments. For example, in the absence of wiregrass, other ruderal bunch grasses often fill that gap as the dominant vegetation type and act as understory fuel for fire implementation, ultimately changing the ecosystem dynamics (Brockway et al., 2005). This is true not only for management but also for restoration, and we should therefore investigate potential restoration areas using ecological attributes specific to the area.

Longleaf pine restoration efforts began in the mid-1900s with the development of the Stoddard-Neel method as a system to restore quail populations (Neel et al., 2011). However, restoration goals have developed within the last three decades as the ecological value of these habitats has been widely recognized. While game management remains an important aspect of longleaf pine restoration, the focus has shifted towards the protection of threatened and endangered species such as the red-cockaded woodpecker and the gopher tortoise (Condon and Putz, 2007; Moser et al., 2002; Neel et al., 2011; Shaw and Long, 2007). Public concerns seem to have changed as conservation and sustainability act as two of the main driving components in current research and restoration efforts (Brockway and Outcalt 1998, Condon and Putz, 2007; Cox; et al., 2004; Gordon and Rice 1998, Hattenbach et al. 1998, Mulligan et al. 2002). While there have been a substantial number of projects conducted in the sandhills, especially in the Florida and Georgia Sandhills, the information reported on restoration efforts in the Wiregrass Gap region of the Carolina Sandhills is almost nonexistent (Mcgee and Scott, 1965). Although, it remains a critical area to longleaf habitat restoration with a study conducted in the Clemson Sandhill Research and Education Center finding a total of 328 species, 237 genera, and 100 families inventoried over 215 hectares of greenspace (Jenkins and McMillan, 2009).

Consequently, spatially explicit suitability modeling can act as a valuable tool in examining changes in species distributions and predicting appropriate sites for criteria-driven investigations. Overlay weighted models can apply a standard measurement scale based on dissimilar inputs at varying levels of importance and have been used extensively worldwide as a baseline for critical decision making (Kalirai et al., 2015; Riad et al., 2011; Roslee et al., 2017). These types of models have been successfully applied in several research studies, including the prioritization of longleaf pine conservation areas and the analysis of longleaf pine resilience in North Carolina. (The Nature Conservancy, 2016; Vorhees, 2015). However, while there have been some spatial models detailing aspects of longleaf pine habitat in SC, such as the isolation of RCW groups and the quantification of vegetation, we do not see a similar modeling approach in South Carolina comparatively with regards to longleaf pine restoration modeling (Moseley, 2019). Moreover, there have been few spatial investigations into niche regions, such as the Carolina Wiregrass Gap, especially at the parcel level. By identifying parcels ranked in the most appropriate sites, you add another level of applicability to your study, which can then be tested (Tiwari and Ajemra, 2021).

The purpose of this study was to use ecological raster data and the capabilities of ArcGIS Pro Spatial Analyst to explore potentially viable sites for longleaf pine restoration operations in the SC Sandhills Wiregrass Gap. The ecological requirements for longleaf pine growth are specific to the region, making site determination challenging. In the sandhills, the depth to the impervious clay layer is one of the most influential factors in determining the appropriateness for longleaf pines because it slows the infiltration of rainwater, making sites with high clay content near the soil's surface wetter than sites with deeper layers of clay. Longleaf pines thrive in well-drained sandy soils from submesic to xeric sites, usually including arenic paleudults, grossarenic paleudults, or quartzipsamments soil types (Van Lear and Jones, 1987). By using weighted overlay as a method of modeling suitability, we expect to be able to identify parcels within our study area that could also be restored to a historical longleaf pine habitat based on the site's ecological attributes. Considering the historical distribution of longleaf pine encompasses the entirety of the Carolina Sandhills Wiregrass Gap, the region itself would theoretically be appropriate for such a restoration. However, increases in urbanization and the fragmentation of landscapes make longleaf pine restoration difficult due to the pyrogenic nature of the management requirements. Densely populated areas often restrict prescribed fire and large-scale logging operations through city ordinances and diameter cut limitations. By applying rasterized data that includes levels of development, soil characteristics, and vegetation types, we hope to provide a helpful model for researchers, managers, and ecologists alike. This model will avoid urbanization and wetlands while targeting parcels in the study area that are appropriately sized and have the ecological characteristics that are desired.

2.2 Methods

2.2.1 Study Area

The study area includes the historical range (potential habitat) for longleaf pines in the South Carolina Sandhills Wiregrass Gap, an area that covers approximately 552,441 hectares across six counties in the middle of the state (Figure 2.1). The study area was identified by selecting the level IV ecoregion within the counties devoid of wiregrass in the longleaf pine historic range outlined by Little's "Atlas of United States Trees" series (1971) and digitized by Thompson et al. (1999) (Figure 2.1). The Wiregrass Gap distribution map was identified through the USDA PLANTS Database (2014), and the Level IV ecoregion image layer was retrieved from the US EPA Office of Environmental Information and the US EPA Office of Research and Development (2021). USA wetland boundaries were recovered from the National Wetlands Inventory produced by the US Fish and Wildlife Service (2021).

For this project, I used geospatial analysis and a weighted overlay to develop suitability models for longleaf pine restoration in the Carolina Sandhills Wiregrass Gap based on ecological raster data obtained from 8 sources and parcel data obtained from 6 sources. Different weighted combinations of ecological rasters were tested to identify sites appropriate for longleaf pine restoration on a ranking system from 1-5, 1 being the most appropriate, and 5 being the least appropriate. Additionally, individual parcels were identified within the area that could potentially be targeted for such a project. Raster data was extracted to fit the study area and reclassed based on size.

Longleaf pine habitats are appropriate for restoration in submesic, subxeric, and xeric sites in the Carolina Sandhills, meaning that restoration success is greatly influenced by soil composition and hydrology (Van Lear and Jones, 1987). To measure suitability, I employed five soil rasters, one vegetation raster, and two land-use rasters at a unanimous cell size of 30 meters. All layers had the same geographic coordinate system (WGS 1984), datum (D WGS 1984), projection (Mercator Auxiliary Sphere), and projected coordinate system (WGS 1984 Web Mercator). A majority of the data found was accessed through the Esri ArcGIS Living Atlas of the World. Model outputs had cell sizes of 250 meters. Four combinations of weighted overlays were produced, weighting each raster category (soil, vegetation, and land use) higher than the other two and then weighting them all the same. The first overlay weighted each category evenly (33%, 33%, 33%), the second overlay weighted soil rasters higher (50%, 25%, 25%), the third overlay weighted vegetation rasters higher (25%, 50%, 25%), and the final overlay weighted land cover higher (25%, 25%, 50%). Land cover categorized areas based on development and use, vegetation layers identified areas with specific forest types, such as areas with longleaf pine, and soil layers classified areas based on composition, stability, and permeability. Finally, to test our models' accuracy, we investigated nine sites within our study area known to have longleaf pine present and compared them to our model outputs (Figure 2.2).

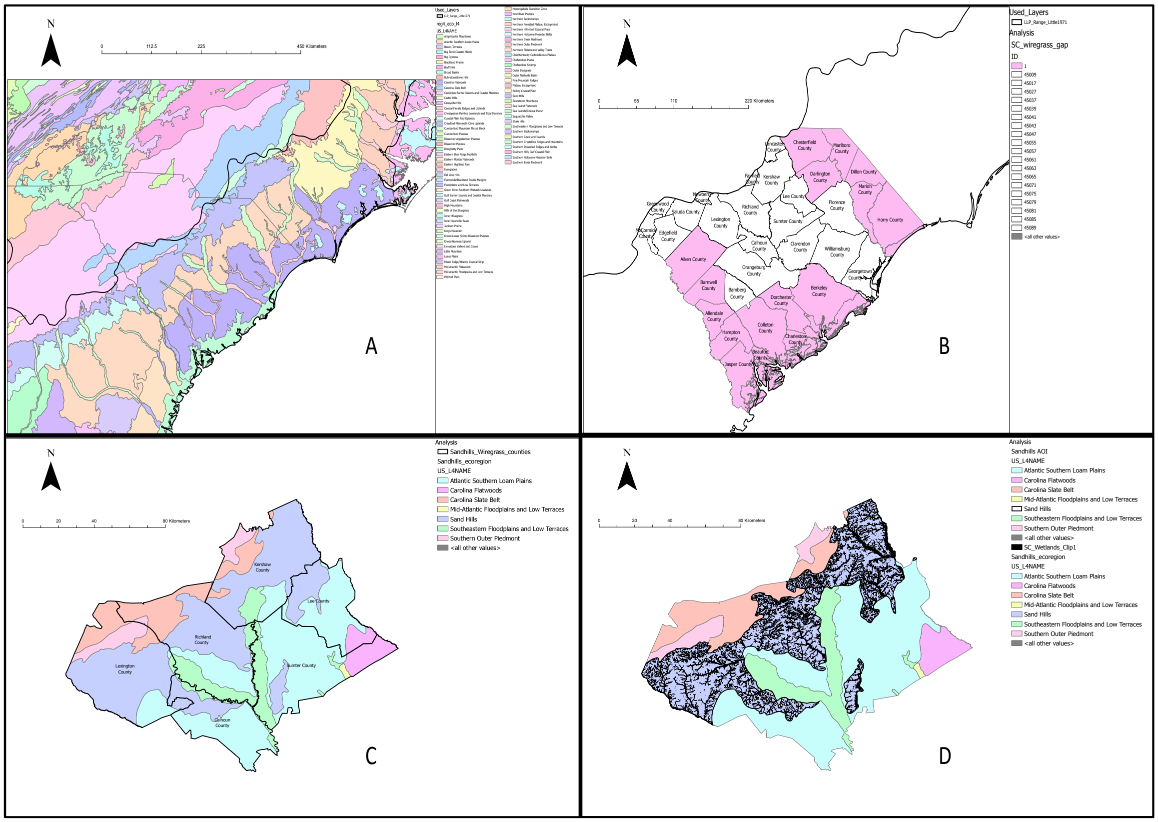

Figure 2.1: Map showing the study area outlined with wetlands (D) located within the Wiregrass Gap distribution (B) and level IV Sandhills ecoregion (C). Map A illustrates the range of ecoregions in South Carolina separated by the historic longleaf pine range, map B shows the South Carolina Wiregrass Gap distribution by county with colored counties having wiregrass present, map C shows the ecoregion distribution within SC counties, and map D illustrates the study area selection with wetland delineation.

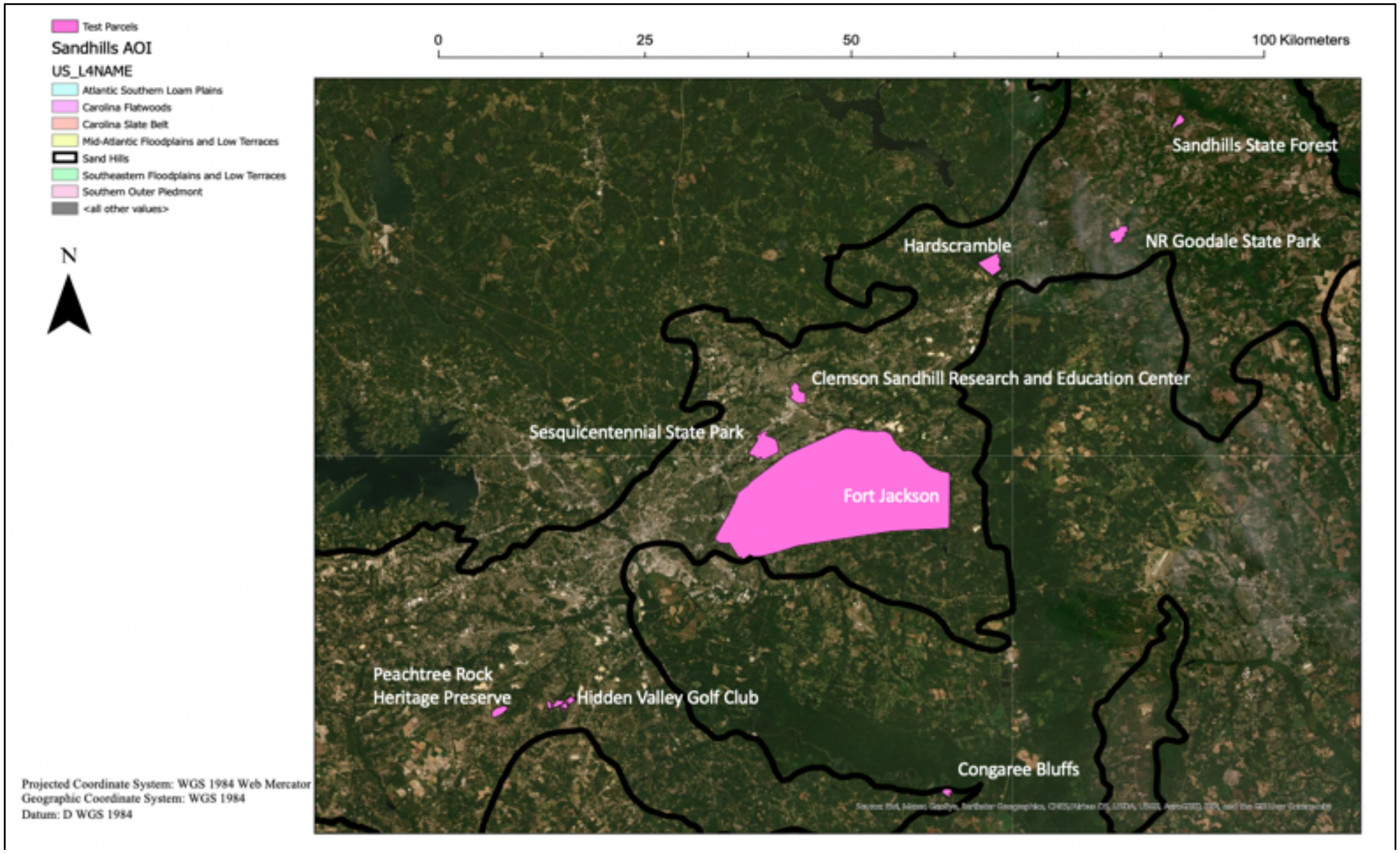

Figure 2.2: Map of nine test parcels within our study area known to have longleaf pine present on the property and are larger that 30 hectares.

2.2.2 Data Analysis Layers

I. USA SSURGO – Soil Hydrologic Group (2020) (USDA NRCS, Esri)

This raster layer, provided by the USDA Natural Resources Conservation Service (NRCS), was derived from 30m cell rasters with hydrologic groups defined from the gSSURGO map unit aggregated attribute table. It illustrates seven hydrologic soil classes that correspond to the rate of rainfall absorption into the soil. This raster was reclassed to prioritize prime longleaf pine habitat, including naturally well-drained sandy soils with high rates of infiltration, low runoff rates, and xeric conditions (Boyer and Peterson, 1983; Van Lear and Jones, 1987) (Table 2.1).

Table 2.1: Reclassed USDA-NRCS hydrologic soil class raster with definitions.

Reclass Rankings	Hydrologic Soil Group	Description
1	A	Deep, well-drained sands/gravelly sands with high infiltration and low runoff.
2	B	Deep well-drained soils with a moderately fine-coarse texture and a moderate rate of infiltration and runoff.
3	C	Soils with at least one layer that impedes the downward movement of water and fine textured soils. It results in a slow rate of infiltration and high amounts of runoff.
4	D	Very slow infiltration rate and high runoff potential. Composed of clays that have a high shrink-swell potential, soils that have a high-water table, soils that have a clay pan or clay layer at or near the surface, and soils that are shallow over impervious material.
	A/D	Very slow infiltration rate due to a high-water table. Would have high infiltration and low runoff rates if drained.
5	C/D	Very slow infiltration rate due to a high-water table. Would have a slow rate of infiltration if drained.
	B/D	Very slow infiltration rate due to a high-water table. Would have a moderate rate of infiltration and runoff if drained.

II. <u>USA SSURGO – Soil Hydric Class (2020) (USDA NRCS, Esri)</u>

This 30m cell raster layer was produced by the USDA NRCS and illustrates the percentage of hydric soils that produce anaerobic conditions in the top layers of the soil. It was created using the gNATSO database and spatially identifies poorly drained soils saturated for prolonged periods, generating wetland vegetation (A-F). The raster was reclassed to prioritize non-hydric soils as longleaf pines grow best in well-drained, xeric, and unsaturated soils (Boyer and Peterson, 1983; Van Lear and Jones, 1987) (Table 2.2).

Table 2.2: Reclassed USDA-NRCS soil hydric class raster with definitions.

Reclass Rankings	Soil Hydric Class	Description
1	A	Not Hydric (0%)
2	B	Partially Hydric (1 – 25%)
3	C	Partially Hydric (26 – 50%)
4	D	Partially Hydric (51 – 75%)
5	E/F	Partially Hydric (76 – 95%)/Fully Hydric (100%)

III. <u>USA SSURGO – Drainage Class (2020) (USDA NRCS, Esri)</u>

This 30m cell raster layer was produced by the USDA NRCS and illustrates soil classifications based on eight drainage classes that rank soils based on soil moisture characteristics (Table 2.3). The layer was created using data from the gNATSO database and was reclassed (1-5) to rank well-drained soils, a characteristic of longleaf pine habitats, higher than poorly drained, mesic soils (Boyer and Peterson, 1983) (Table 2.4). However, excessively drained soils were ranked lower due to water and nutrient limitations in extremely xeric conditions.

Table 2.3: USDA-NRCS soil drainage class raster with definitions.

Drainage Class	Description
0	Excessively drained – Water is removed rapidly. Coarse-textured soils with high hydraulic conductivity. Can be very shallow.
1	Slightly excessively drained – Water is removed rapidly. Very similar features to drainage class 0 but is more likely to have free internal water.
2	Well drained: Water is removed readily. Plants have access to water for most of the growing season. Wetness does not inhibit growth of roots.
3	Moderately well drained – Water is removed from the soil slowly and remain wet for short periods during the growing season, affecting mesophytic crops.
4	Somewhat poorly drained – Water is removed slowly so that the soil is wet at shallow depths for significant periods of time.
5	Poorly drained – Water is removed very slowly and is wet at shallow depths remaining wet for long periods of time.
6	Very poorly drained – Water is not removed from the soil and free water stays near the ground surface during the growing season.
7	Subaqueous Soils – Under water.

Table 2.4: Reclassed USDA-NRCS hydrologic soil class raster with definitions.

Reclass Rankings	Drainage Class Rank	Description
1	2, 3	Well drained and moderately well drained soils.
2	0, 1	Excessively drained and Slightly excessively drained soils.
3	4	Somewhat poorly drained soils.
4	5	Poorly drained soils.
5	6, 7	Very poorly drained and subaqueous soils.

IV. USA SSURGO – Erodibility Factor (2020) (USDA NRCS, Esri)

This 30m cell raster layer was produced by the USDA NRCS and spatially illustrates soil erodibility, the quantification of soil particle susceptibility to detachment by water flux, using six erodibility factor categories. The raster layer was reclassed to place low soil erosion predictions as a priority over higher ones as erodibility highly influences germination (Barnet et al., 1990) (Table 2.5).

Table 2.5: Reclassed USDA-NRCS soil erodibility raster with definitions.

Reclass Rankings	Erodibility Factor	Description
1	0 – 0.10	Low Erodibility
2	0.11 – 0.20	Slight Erodibility
3	0.21 – 0.30	Moderate Erodibility
4	0.31 – 0.40	High Erodibility
5	0.41 – 0.50/0.51 – 0.64	Excessive Erodibility/Extreme Erodibility

V. USA SSURGO – Available Water Storage 0-150cm (2020) (USDA NRCS, Esri)

This 30m cell raster displays the maximum amount of water (cm) in the upper 150cm of soil available to plants and is based on rainfall amounts, soil infiltration, and soil storage capacity. The layer was created from data acquired from the gNATSGO database and is influential in predicting soil drought susceptibility, hydrologic models, and plant productivity. Even though water availability is vital for plant growth, the raster data was reclassified to prioritize xeric conditions and low water availability based on environmental conditions associated with longleaf pine growth (Van Lear and Jones, 1987) (Table 2.6).

Table 2.6: Reclassed USDA-NRCS available water storage (0-150 cm) raster with definitions.

Reclass Rankings	Available Water	Description
1	0 – 20 cm	Xeric
2	20.1 – 30 cm	Subxeric
3	30.1 – 40 cm	Submesic
4	40.1 – 50 cm	Mesic
5	50.1 – 90 cm	Wetland

VI. USA Forest Type (2019) (USDA – FIA Program, RSAC, Esri)

This 250m cell raster illustrates 141 forest types in the US and was created from MODIS images taken during the 2002-2003 growing seasons combined with ~100 other raster layers. Forest types were reclassed, and cell sizes were changed to 30m. Areas that already had longleaf pine present were given priority over pure pine and hardwood stands, mixed pine/oak stands, and wetland vegetation (Table 2.7). Areas with longleaf already present would theoretically be easier to restore, conifer stands would probably exhibit similar characteristics to restored longleaf pine stands, and pure stands would be easier to harvest over mixed stands due to increased amounts of sorting during the logging operation. Finally, wetland vegetation would indicate environmental conditions that were unsuited for such a project.

Table 2.7: Reclassed USDA-FIA forest vegetation raster with definitions.

Reclass Rankings	Forest Type
1	Longleaf pine, longleaf/oak
2	Slash pine, loblolly pine, southern scrub oak
3	Shortleaf pine/oak, loblolly pine/hardwood
4	Post oak, blackjack oak, white oak, red oak, hickory, yellow-poplar, white oak/northern red oak, sweetgum, mixed upland hardwoods
5	Pond pine, baldcypress, water tupelo, sweetbay, swamp tupelo, red maple river birch, sycamore

VII. USA NLCD Land Cover (2021) (NLCD, USGS, Esri)

This raster layer displays a time series of land cover for the contiguous US based on data obtained from the National Land Cover Database. Raster data is grouped into 20 land cover classes that include vegetation type, development density, agricultural use, water, and barren lands. The latest time series layer (2016) was used in our analysis, and the cover types were reclassed based on restoration viability (Table 2.8). Undeveloped vacant land with an established herbaceous understory was prioritized over forested stands, cultivated fields, wetlands, and developed areas. Open water and developed areas are unavailable for such a restoration, wetlands are ecologically inappropriate for longleaf pine growth, cultivated fields could potentially be converted for restoration but are currently being occupied, and a forested stand would have to go through a restoration timber harvest before conversion to longleaf. Therefore, vacant land capable of fire facilitation, or at least land with the potential to support herbaceous vegetation, would be most appropriate for restoration.

Table 2.8: Reclassed NLCD land cover raster with definitions.

Reclass Rankings	Land Cover
1	Barren land, shrubland, scrubland, grassland, herbaceous grasslands
2	Deciduous forest, evergreen forest, mixed forest
3	Pastureland, hay, cultivated crops
4	Woody wetlands, emergent herbaceous wetlands
5	Open water, low development, medium development, high development, developed open space

VIII. USGS GAP Analysis Land Cover (2019) (USGS GAP Analysis Project)

This raster layer displays 30+ terrestrial ecosystems by state, territory and landscape based on national imagery data obtained in 2011. The raster data was reclassed to grant priority to vacant sites that most closely mimic longleaf pine habitat (Table 2.9). Urban development and wetlands were ranked last, followed by mesic vegetation types, thickets, and pine woodlands/mixed forest. The most appropriate sites were designated as areas with sandy bare soil, an open canopy, a recently cleared forest, or an established grassland.

Table 2.9: Reclassed USGS-GAP analysis land cover raster with definitions.

Reclass Rankings	Land Cover
1	Sandy bare soil, open canopy, cleared forest, grassland, pastureland
2	Dry scrub, shrub thicket, closed canopy evergreen forest, woodland, needle-leaved evergreen mixed forest, woodland, pine-woodlands
3	Wet scrub, shrub thicket, mesic mixed forest, cultivated land
4	Pocosin, mesic deciduous forest, mesic evergreen forest
5	Wetlands, wet soil, rock outcrop, aquatic vegetation, urban development

IX. Carolina Wiregrass Gap Parcel Data

County-level parcel data was obtained from each county GIS office and clipped to the study area (Table 2.10). The shapefiles were combined, converted to raster data, and reclassed to prioritize larger parcels over smaller ones. Since longleaf pine habitat and restoration efforts are highly correlated with red-cockaded woodpeckers (RCWs), we prioritized large parcels to effectively conduct a restoration that would eventually create enough habitat to sustain an RCW population (Craig et al., 2010; Crowder et al., 1998; Shaw and Long, 2007). According to the US Fish and Wildlife Service (2020), the typical territory required to support one group of RCWs ranges, on average, from about 50.6-80.9 hectares, but can be as low as 24.3 hectares and as high as 242.8 hectares (USDA, 2009). Similarly, according to a local forestry consulting agency in Kershaw County, RCW relocations and artificial cavity nests were only applied in suitable habitats larger than 30.4 hectares, which we used as our stoppage measurement for restoration parcel size (Forest Land Management Inc., 2021). Of the 227,614 parcels in our study area, only 1.4% (3,146) were larger than 30 hectares.

Table 2.10: Parcel data sources for Carolina Sandhills Wiregrass Gap counties.

County	Parcel Source
Kershaw County	Kershaw County Addressing, GIS Mapping, Information Technology Services – Kershaw County Government Center
Lee County	Lee County GIS/Mapping and E911 Addressing – Lee County Assessor's Office
Richland County	Clemson Center for Geospatial Technologies – Richland County GIS Office
Lexington County	Lexington County Department of Planning and GIS – County Administration Building
Calhoun County	Clemson Center for Geospatial Technologies – Calhoun County GIS Systems – Assessor's Office
Sumter County	Sumter County GIS Mapping Services – Sumter City-County Planning Department

2.2.3 Data Analysis

All data transformations and analyses were conducted using Microsoft Excel (Microsoft Corporation, 2018), R (R Core Team, 2020), and RStudio (RStudio Team, 2020). All maps and spatial investigations were completed using ArcGIS Pro (Esri Inc., 2020). Using a two-way analysis of variance (ANOVA), we compared mean size predictions across models and rankings. The 'model' and 'rank' variables were converted into factor data and measures of central tendency and dispersion for our dependent variable was determined. We tested for normal distribution using Normal Q-Q plots and the Shapiro-Wilk test on ANOVA residuals. Data that did not meet the assumptions of normality were normalized and Tukey HSD post hoc tests were performed for pairwise comparisons.

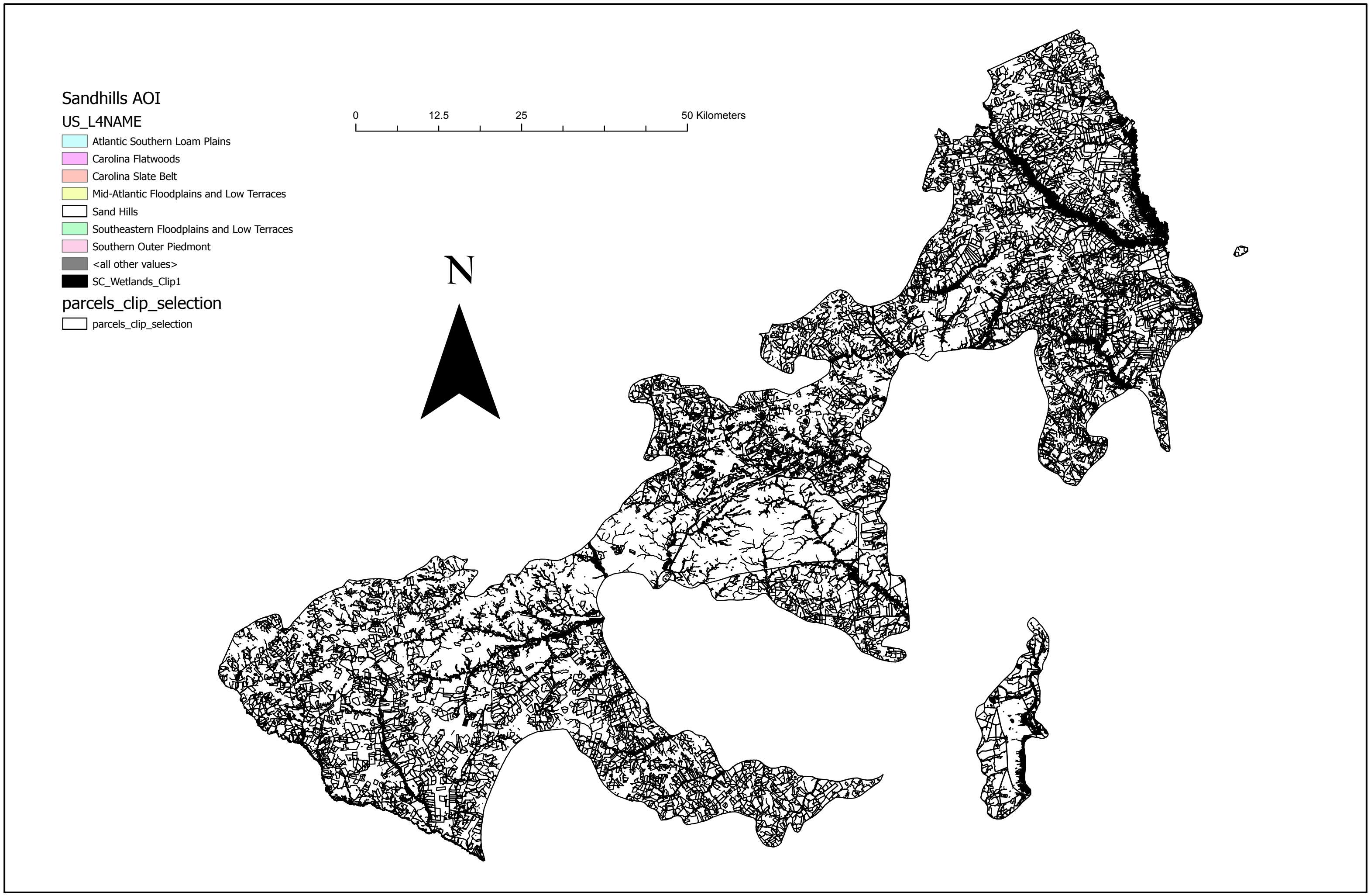

Figure 2.3: Map of parcellation for sites larger than 30 hectares in the South Carolina Sandhills Wiregrass Gap.

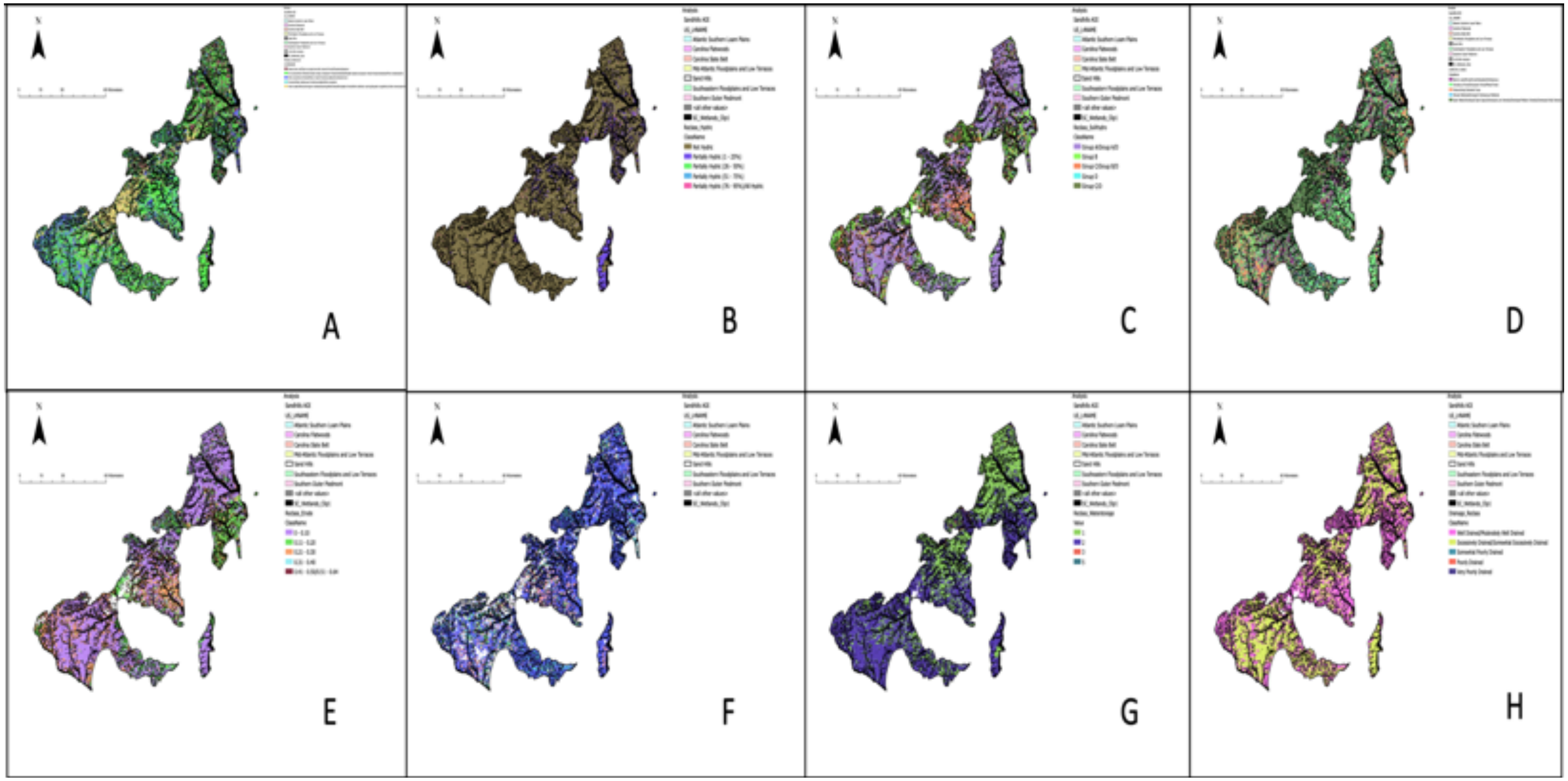

Figure 2.4: Maps of rasterized data layers located in the Carolina Wiregrass Gap, including land cover (A), soil hydric class (B), soil hydrologic group (C), land use (D), erodibility (E), forest type (F), water storage percentage (G), and drainage class (H).

2.3 Results

2.3.1 Longleaf Pine Habitat Suitability Models

While only 1.4% of parcels were identified as large enough for longleaf pine habitat restoration, because of their larger size, they accounted for 32.47% of the entire Carolina Wiregrass Gap, or 179,368.75 hectares (Figure 2.3). Across all models, we produced 3-4 categories of suitability with ranks 1-2 acting as appropriate sites for restoration. Across all models and ranks, we only found significant differences between rankings ($P<0.05$), with no significant differences between models ($P=0.929$). Pairwise comparisons found that our second ranking had a more extensive representation, having significantly more hectares than category 1 ($P<0.05$) and 3 ($P<0.05$). All models produced similar results, yielding appropriate rankings for 26.8-29.8% of the Carolina Wiregrass Gap and 83-91% of our targeted parcel area. Only one model produced more than three categories, and it was combined in the analysis as any site with a rank higher than 2 was unlikely to support this type of restoration.

For our equally weighted longleaf pine restoration model, our weighted overlay produced three categories of suitability within our targeted parcels (1-3) (Figure 2.5). On a scale of 1 to 3, cells were defined as ideal, appropriate, and inappropriate for longleaf pine restoration in the Carolina Sandhills Wiregrass Gap. Over 67% of our identified parcel area was deemed appropriate for longleaf pine restoration, approximately 42,156 hectares (Figure 2.5). Comparatively, only 23.5% of this area was deemed ideal for restoration and most likely already had longleaf pine present within the vicinity (Figure 2.5). Inappropriate sites were our most underrepresented category, totaling just 9.5% of our target area (Figure 2.5). As a result, our model predicts that longleaf pine restoration projects, excluding wetlands, development, and inappropriate site conditions, would be appropriate for 90.5% of the selected parcel area and 29.6% (163,306.25 ha) of the entire Carolina Sandhills Wiregrass Gap (Figure 2.5).

Our model that weighted soil characteristics higher than other ecological criteria yielded the greatest results, with 21% of our identified parcel area ideal, 70% appropriate, and 9% inappropriate for longleaf restoration (Figure 2.8). In total, this resulted in 29.8% (164,625 ha) of the Carolina Wiregrass Gap being predicted as suitable for restoration, meaning that ~70% of the original study area is already being utilized for other purposes or does not have the capabilities of supporting this type of habitat. These results were similar to our model that weighted existing vegetation higher than other model criteria as it predicted that just 29% (160,481 ha) of the Carolina Wiregrass Gap, and ~89% of our selected parcel area was appropriate for longleaf pine establishment (Figure 2.7). Finally, our heavily weighted land cover model produced the lowest representation of suitable habitat, yielding just 26.8% (148,087.5 ha) of the Carolina Wiregrass Gap and 83% of our selected parcels (Figure 2.6). Of the suitable parcel area, only 6% and 8% were predicted to already have longleaf pine present for the land cover and forest models, respectively (Figure 2.6).

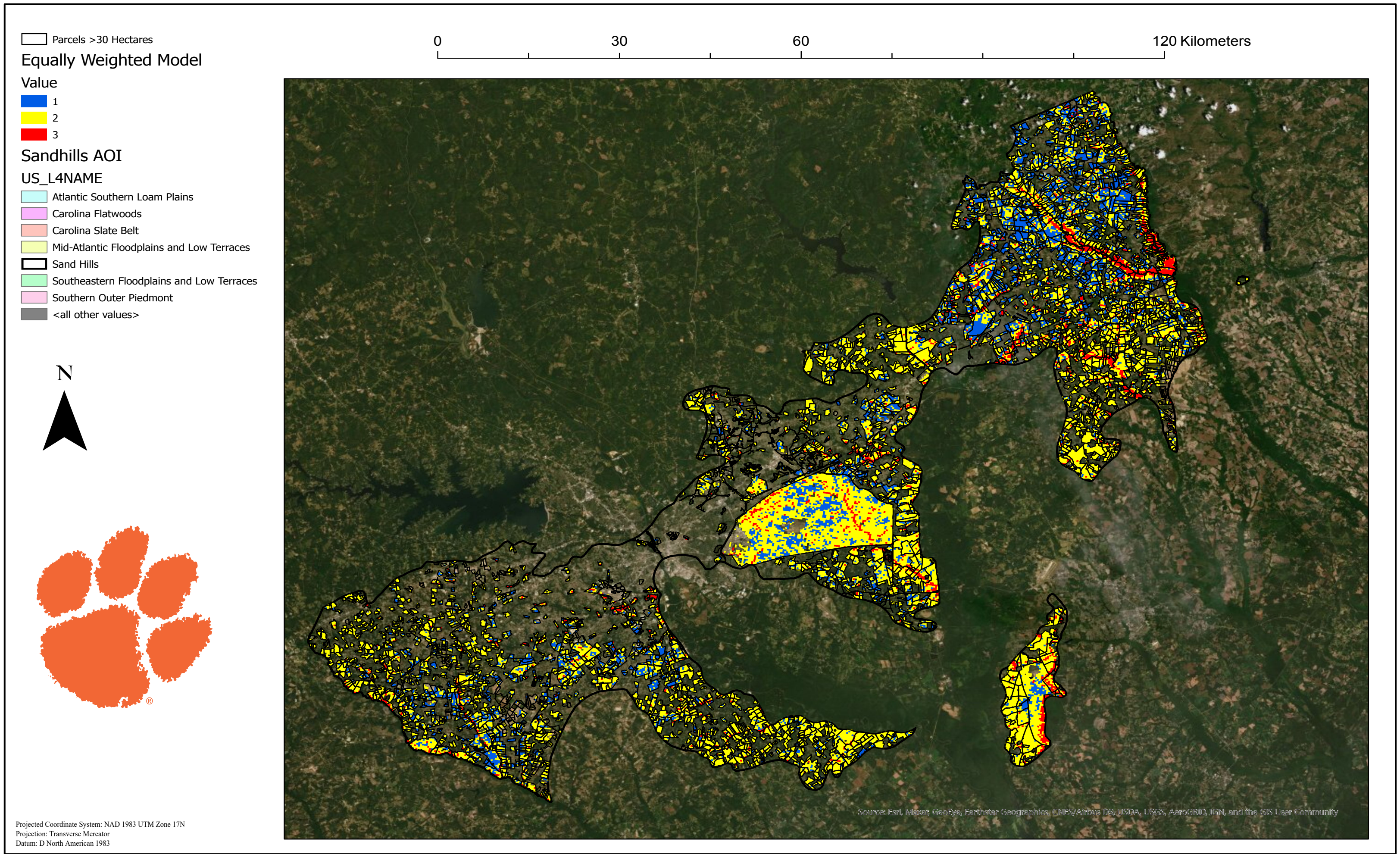

Figure 2.5: Map of longleaf pine restoration suitability model located in the South Carolina Sandhills Wiregrass Gap that weight ecological criteria equally across parcels larger than 30 hectares.

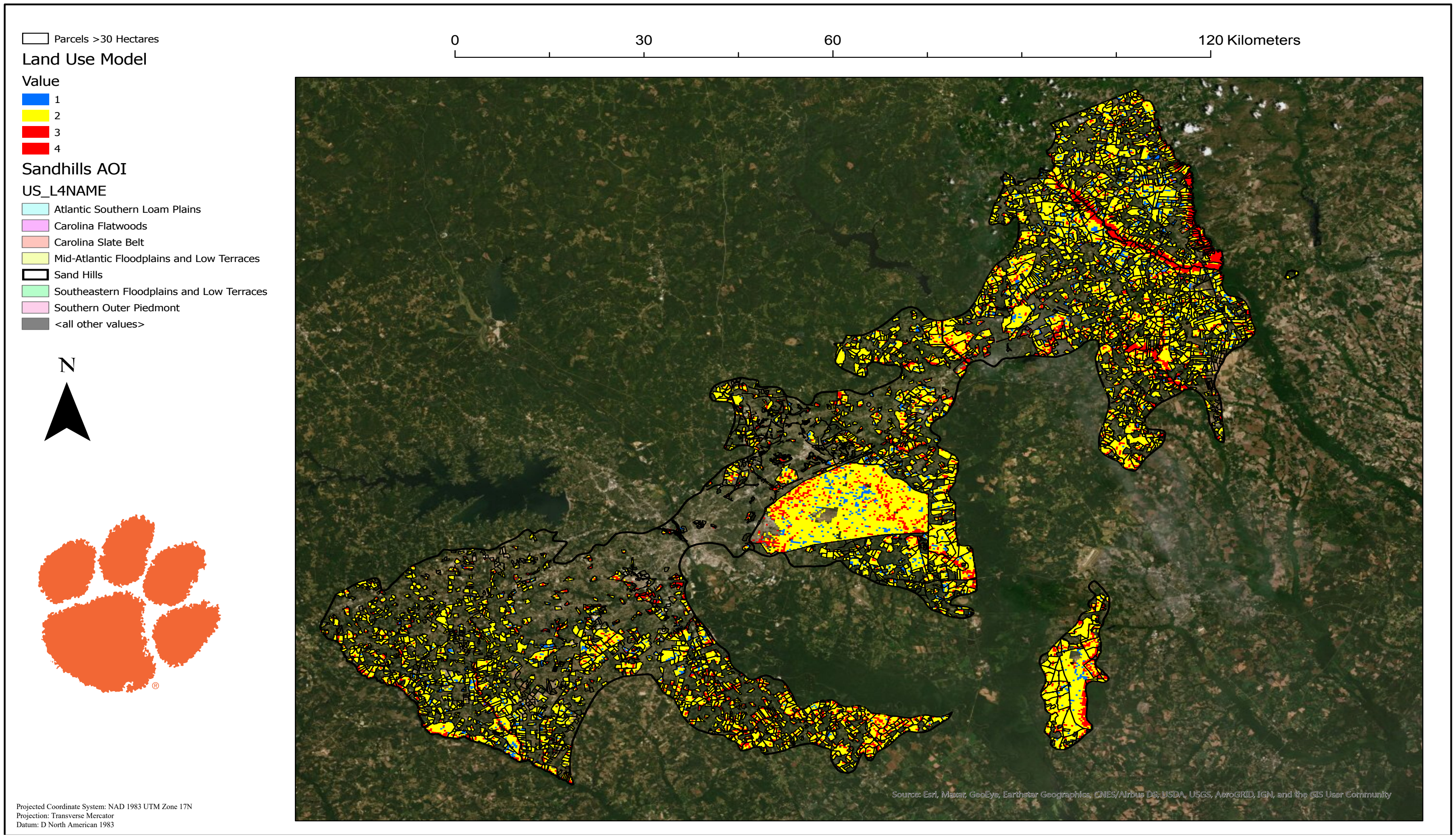

Figure 2.6: Map of longleaf pine restoration suitability model located in the Carolina Sandhills Wiregrass Gap that weights land usage higher across parcels larger than 30 hectares.

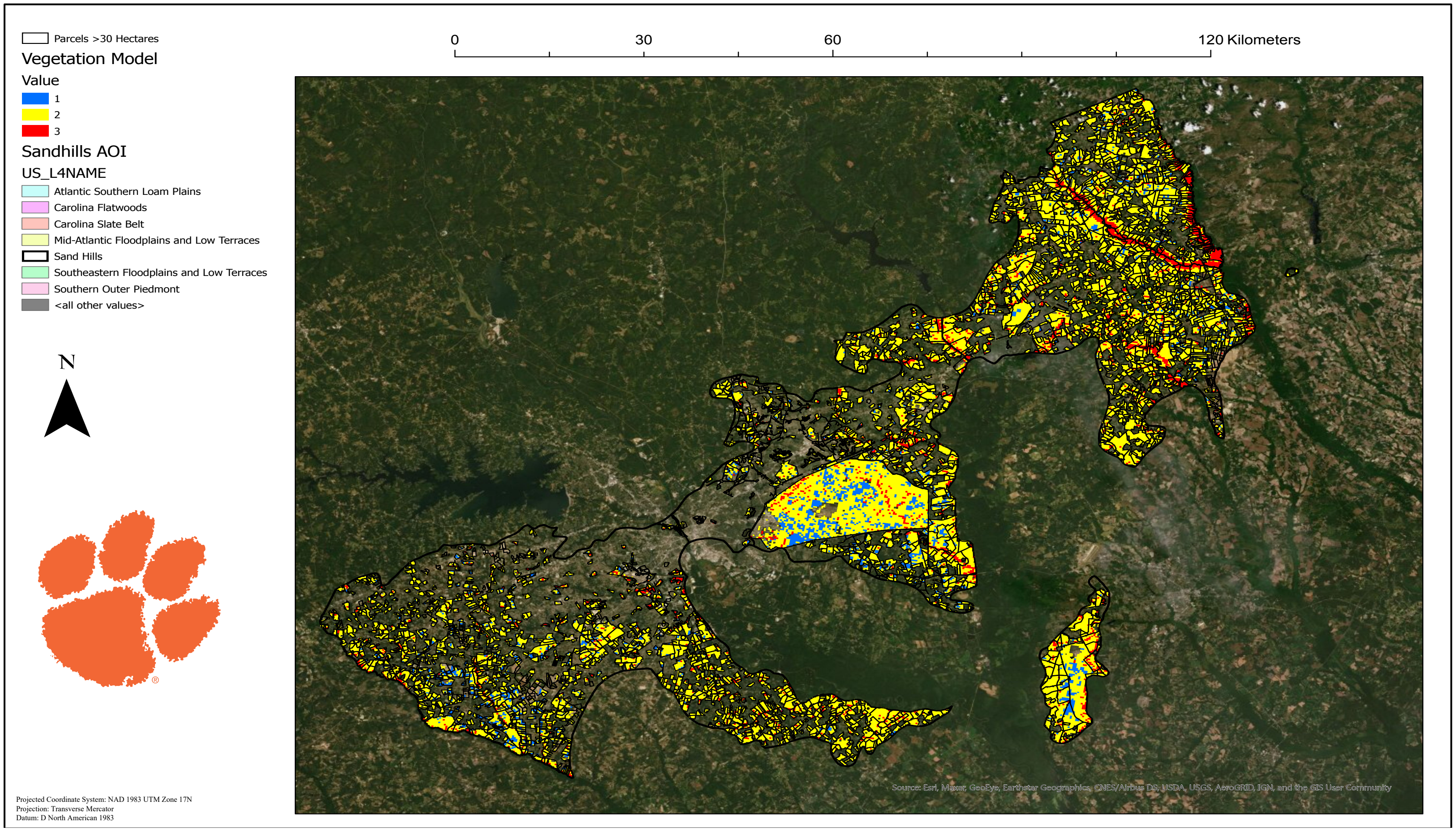

Figure 2.7: Map of longleaf pine restoration suitability model located in the Carolina Sandhills Wiregrass Gap that weights current vegetation higher across parcels larger than 30 hectares.

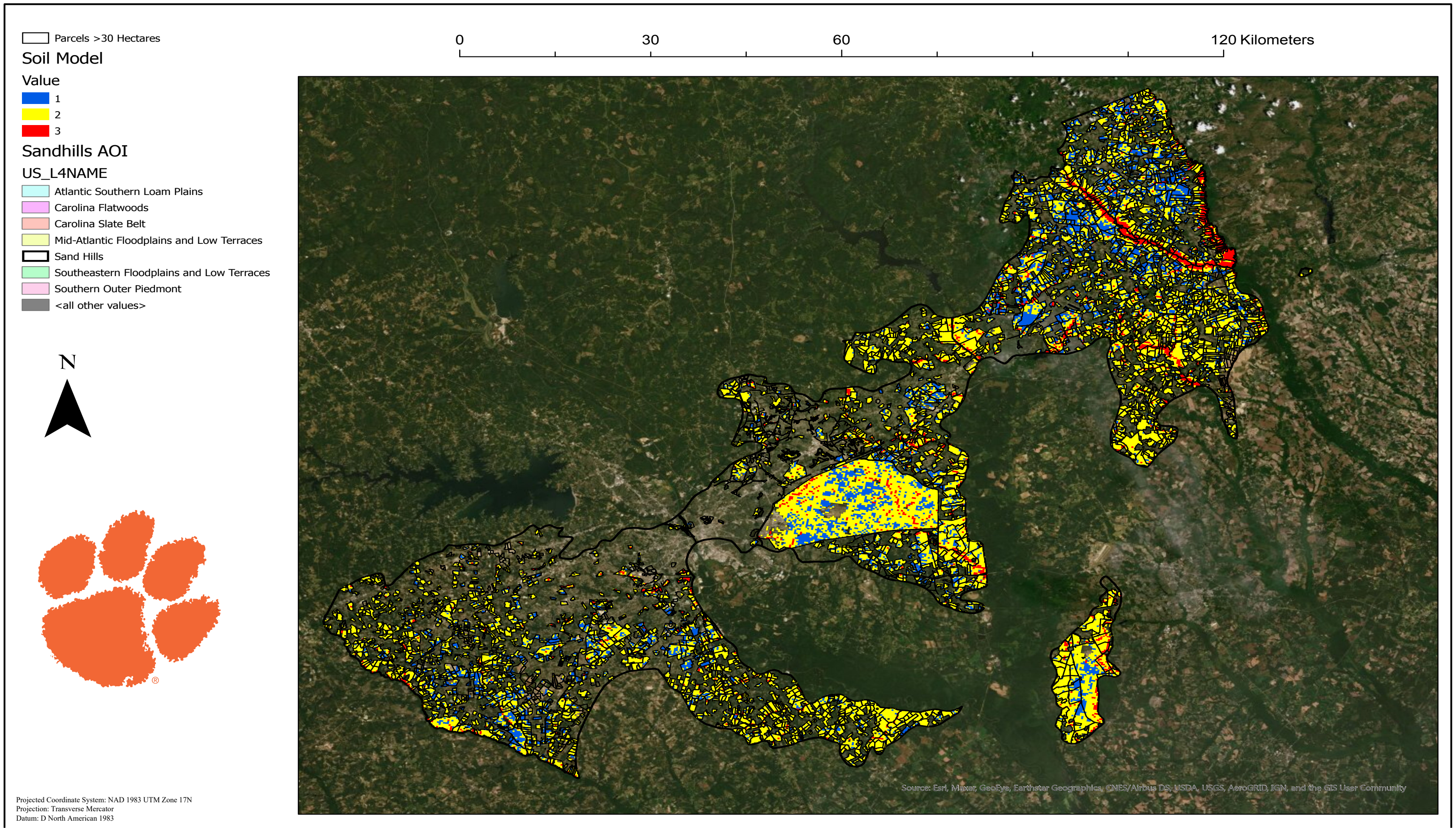

Figure 2.8: Map of longleaf pine restoration suitability model located in the Carolina Sandhills Wiregrass Gap that weights soil characteristics higher across parcels larger than 30 hectares.

2.3.2 Test Parcels

Our nine test parcels known to contain longleaf pine encompassed a total of 22,761 hectares, with Fort Jackson acting as the largest parcel at ~20,993 hectares. Fort Jackson is known to have over half of its area classified as longleaf pine habitat, and many of the other test parcels have significant portions of their area labeled as longleaf pine habitat as well. Approximately 92% of the 22,761 test hectares were classified, and the percentage of area ranked in the 1 category was predictably higher than our original models because of the assurance of the presence of longleaf.

For our equally weighted raster, only one test parcel did not indicate the presence of ideal restoration habitat (Hardscramble), and every parcel was, to some degree, appropriate for restoration. However, as indicated by our original model, longleaf is ideal for restoration in parcels adjacent to the Hardscramble property. Within our test parcels, 22.8% of the area was classified as ideal, 69.7% was deemed appropriate for restoration, and 7.5% was deemed inappropriate for our equally weighted model (Figure 2.9).

Our models that ranked soil characteristics and existing vegetation as higher than other criteria showed similar results compared to our test sites in the equally weighted model (Figure 2.11, 2.12). Respectively, these models predicted 25.3% and 19.7% of the test parcel area being ideal for restoration, 68.6% and 73.5% as appropriate for restoration, and only 6% and 6.8% as inappropriate. Again, Hardscramble was the only parcel that was not predicted to have any ideal restoration sites on the property for our soil weighted model, with every parcel being appropriate for restoration in some capacity.

However, while every parcel was appropriate for restoration, our model that focused on existing vegetation only predicted five of nine parcels to have ideal habitat present.

Finally, our land cover weighted model yielded our most conservative predictions, with just 6.6% of the test parcel area acting as ideal longleaf pine habitat, 78.6% as potential restoration sites, and 14.8% as inappropriate habitat. Of the nine test parcels, only six were predicted to have ideal longleaf pine habitat present on the property (Figure 2.10).

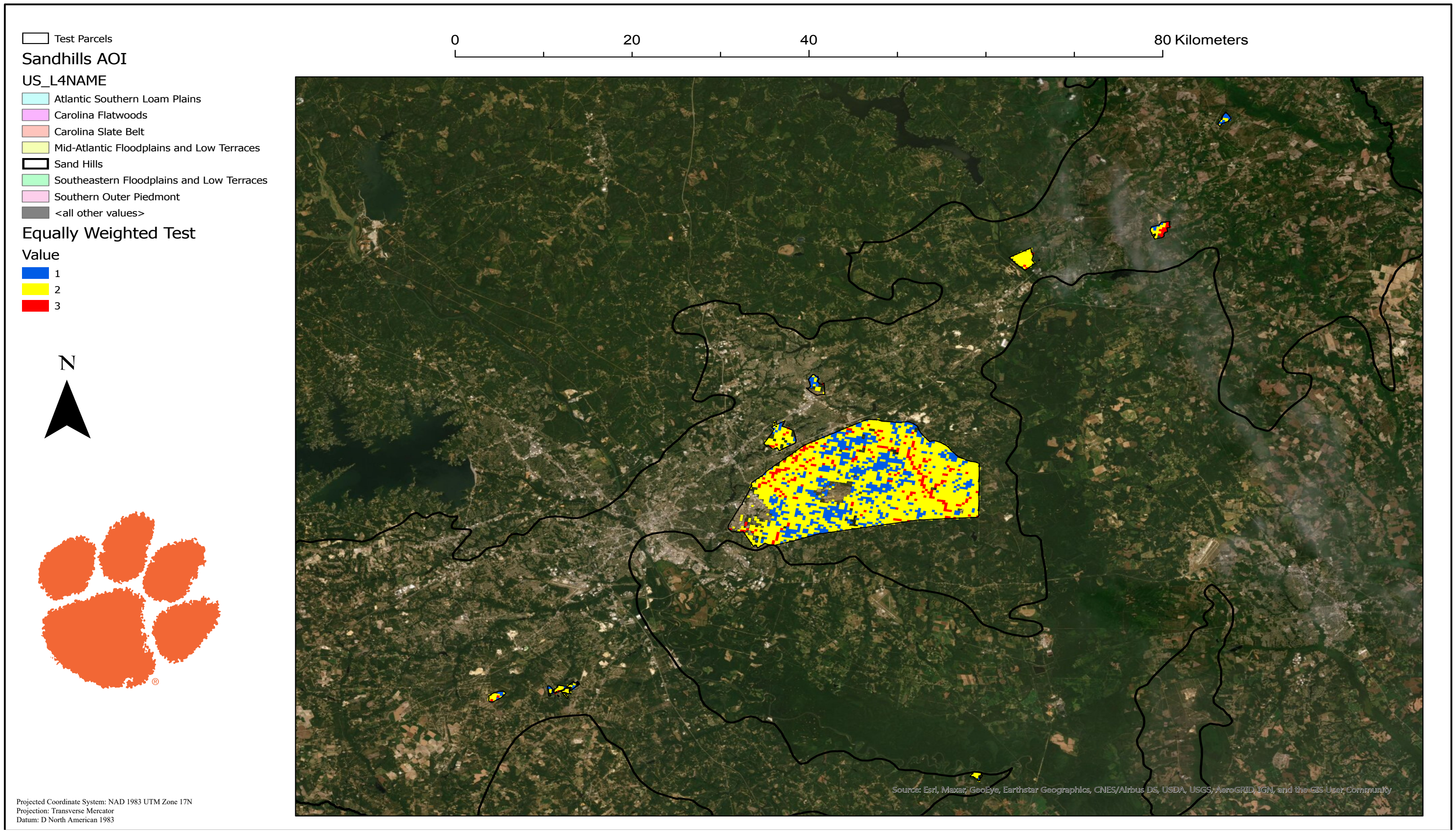

Figure 2.9: Map of longleaf pine restoration suitability model located in 9 test parcels (>30 hectares) in the Carolina Sandhills Wiregrass Gap that weights ecological criteria equally.

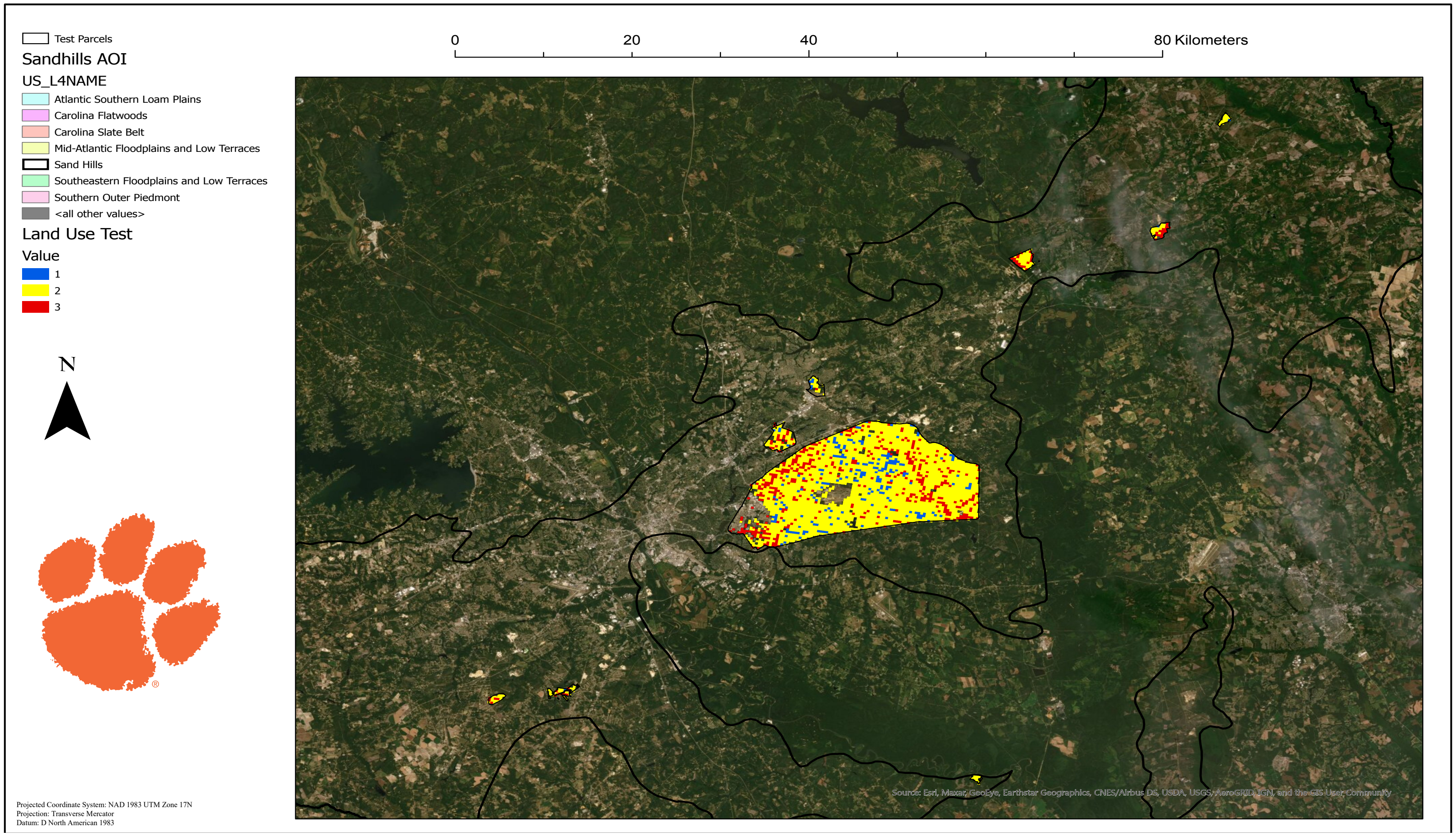

Figure 2.10: Map of longleaf pine restoration suitability model located in 9 test parcels (>30 hectares) in the Carolina Sandhills Wiregrass Gap that weights land usage higher than other criteria.

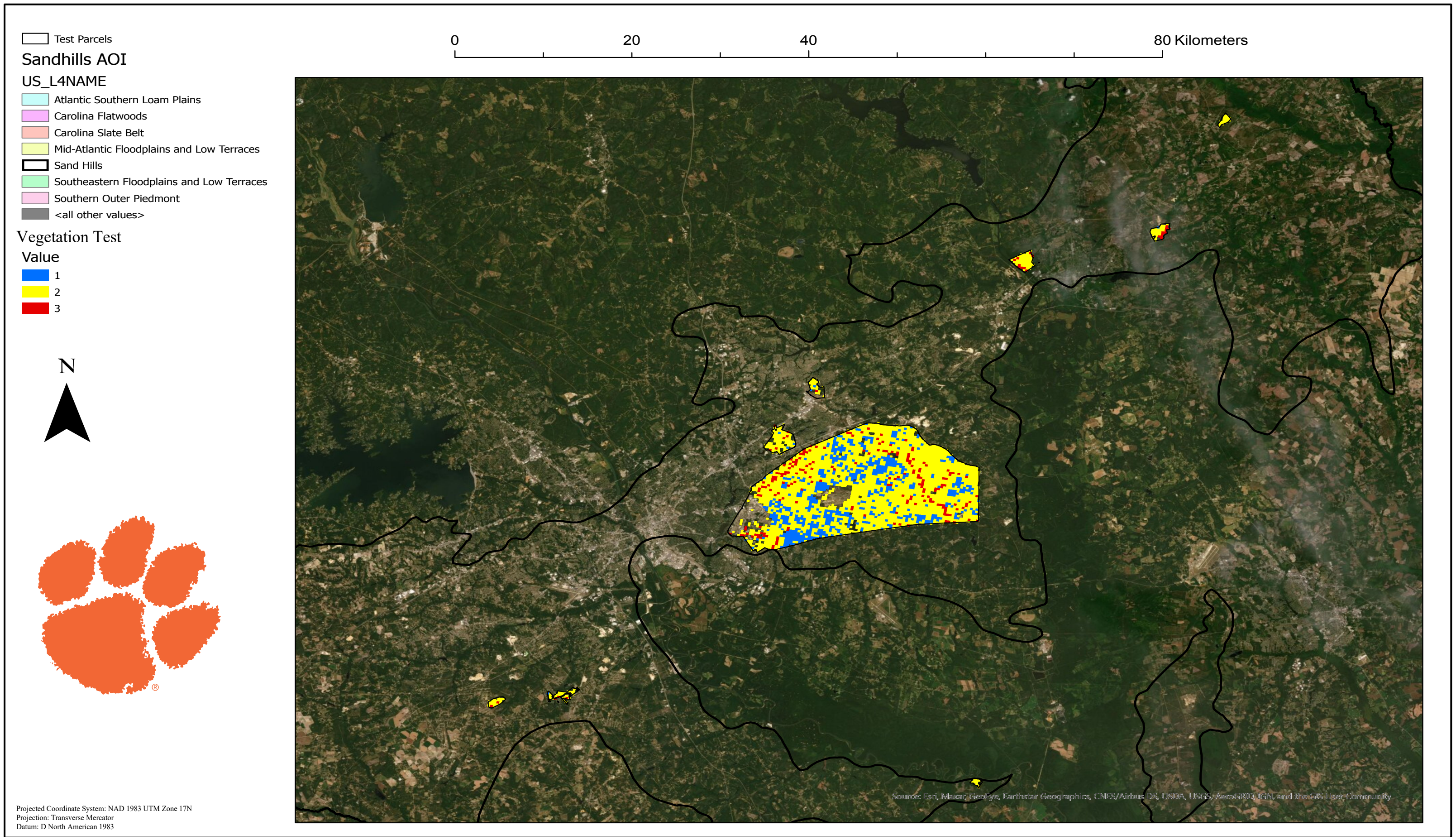

Figure 2.11: Map of longleaf pine restoration suitability model located in 9 test parcels (>30 hectares) in the Carolina Sandhills Wiregrass Gap that weights current vegetation higher than other criteria.

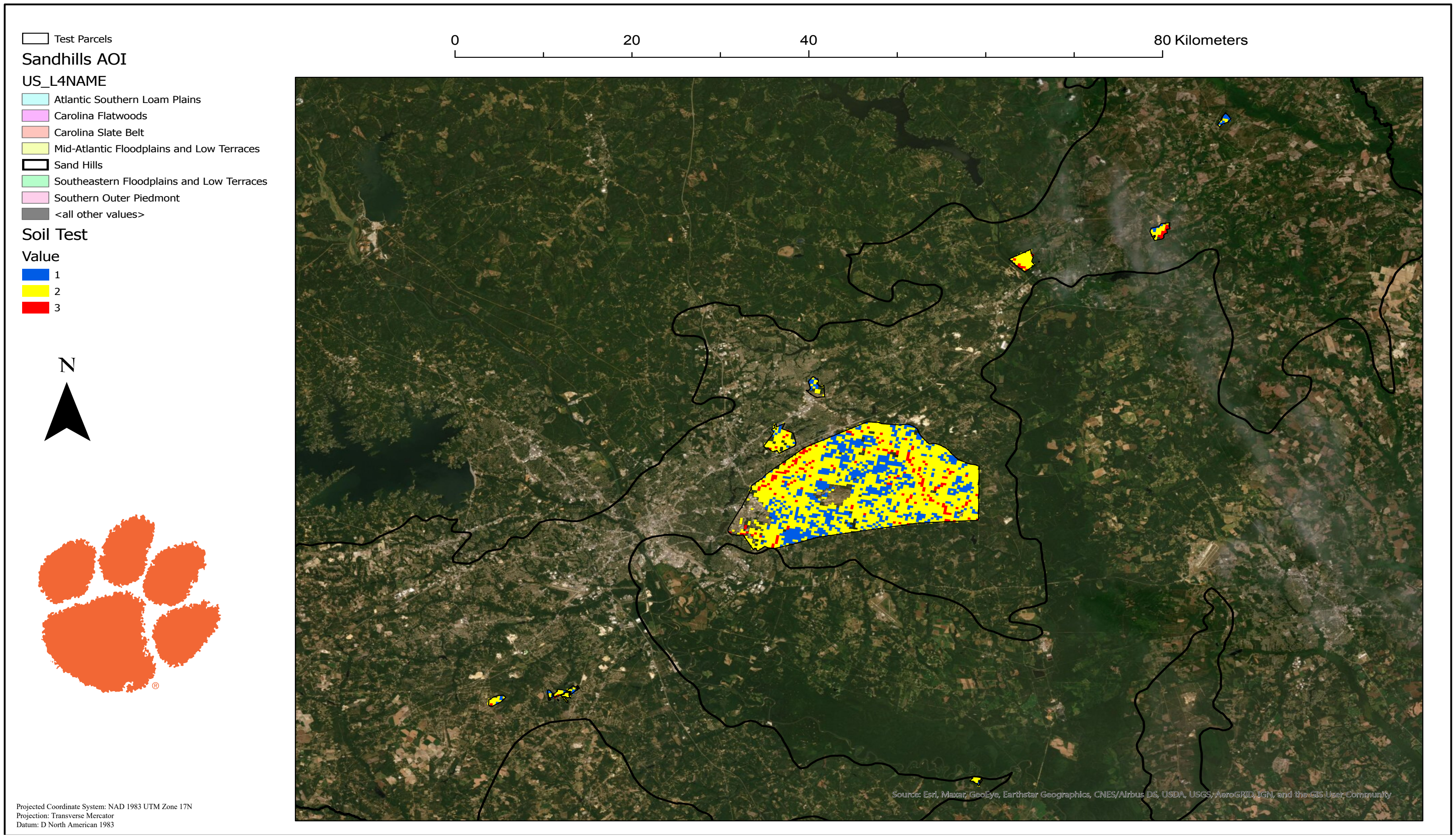

Figure 2.12: Map of longleaf pine restoration suitability model located in 9 test parcels (>30 hectares) in the Carolina Sandhills Wiregrass Gap that weights soil characteristics higher than other criteria.

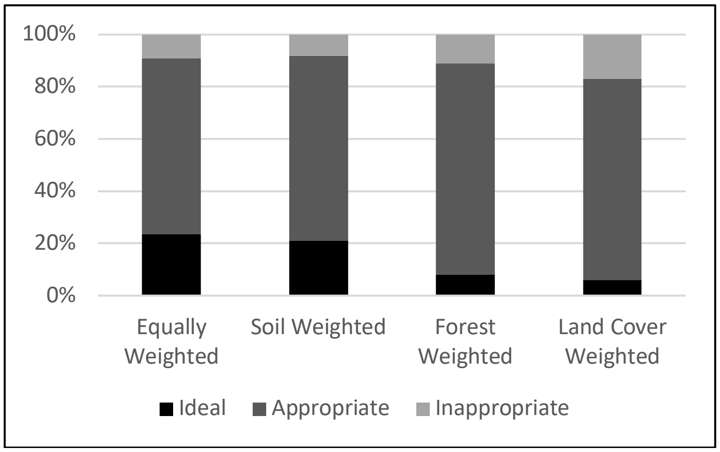

Figure 2.13: A 100% stacked column for parcel areas >30 hectares ranked in three categories from ideal to inappropriate across four habitat suitability models in the Carolina Sandhills Wiregrass Gap.

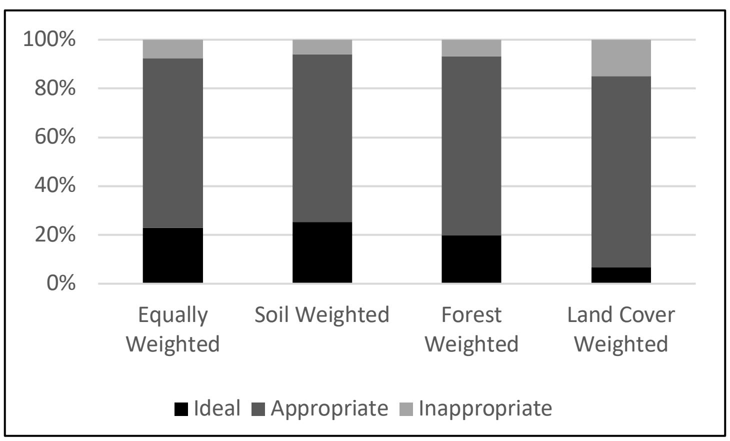

Figure 2.14: A 100% stacked column for test parcel areas ranked in three categories from ideal to inappropriate across four habitat suitability models in the Carolina Sandhills Wiregrass Gap.

2.4 Discussion

Most of the longleaf pine ecosystems that require restoration have been degraded, damaged, or transformed to the point that restorative measures are impossible (Walker et al., 2006). Ecosystems near shorelines, such as the southeastern coastal plain, are overly susceptible to natural disasters and the influence of climate change (Cartwright, 2016; Ojha et al., 2021). However, anthropogenic influences are primarily to blame for the decline in longleaf pine habitat and have exacerbated natural declines in many vulnerable ecosystems (Walker et al., 2006). As a result, a return to a 'historical' state is not only unlikely but may be inappropriate given the current circumstances, even though this has been the traditional reference ecosystem for restoration ecologists in longleaf systems (Walker et al., 2006). Still, while longleaf restoration acts as a formidable task, because of its presence and resiliency to environmental pressures in its natural range, it is feasible that longleaf pine abundancy could gradually increase by expanding existing pockets of longleaf (Landers et al., 1995; Van Lear et al., 2005; Walker et al., 2006). Once these areas and potential expansion zones are identified, restoration and adaptive management can be employed through proven silvicultural practices (Van Lear et al., 2005; Walker et al., 2006).

As of 2013, South Carolina remains 88% privately owned and, according to the SC Forestry Commission (2014), retains 13.1 million acres of forestland, of which 47% is softwood timber (Rose, 2015). The FIA reports that of these 13 million forested acres, which account for 68% of the state, longleaf pine was present on approximately 8%, accounting for about 50% of the basal area in half of its area (Rose, 2015; Zoë, 2015). Estimates indicate that this means there are about 207.5 million longleaf pine trees in South Carolina as of 2013, a 3.5% increase between 2011-2013 and a 98.8% increase between 2001-2013 (Rose, 2015; Zoë, 2015).

Additionally, within the same period, there was a 0.7% increase in forest land in the Southern Coastal Plain, indicating that private landowner and restoration initiatives have contributed significantly to this steady increase in longleaf pine habitat (Rose, 2015; Zoë, 2015). Yet, while planting efforts have observably increased, this is still a tiny fraction of historical range estimates. Only about half of the existing longleaf pine habitat is represented on public lands, meaning private ownership and habitat manipulation remain vital components in long-term sustainability (Rose, 2015; Zoë, 2015). The USDA-NRCS, The Longleaf Alliance, the SC Wildlife Federation, and several other organizations have worked with private landowners in restoring habitats on an incremental scale, increasing awareness and restoration success over time.

Ojha et al. (2021) used inverse distance weighted (IDW) spatial interpolation to illustrate spatial variations in forest attributes across the longleaf pine historical range for two similar FIA datasets (Figure 2.15). Overall, they found that longleaf pine basal area and aboveground biomass generally increased throughout its range, but species richness and diversity tended to decrease from 1997-2018 (Ojha et al., 2021). Moreover, longleaf density decreased substantially in some SC areas as of 2011. These results were highly variable and require further investigation but can still provide critical information that helps in long-term restoration and management objectives. Changes in the spatial distribution of forest attributes and growth trends can help identify suitable restoration sites. Comparing the spatial distribution of forest types in the Ojha et al. (2021) findings to our results in the SC Sandhills Wiregrass Gap, we see a concentration of longleaf pine habitats not only directly below the Fall Line where our study area is located but also radiating outward from the middle of the state where there is a large amount of urbanization. This indicates that our results were, at least, to some degree accurate with regards to predicting where longleaf pine habitats were likely located and confirms that this niche region in SC should continue to be an area of focus regarding longleaf restoration efforts.

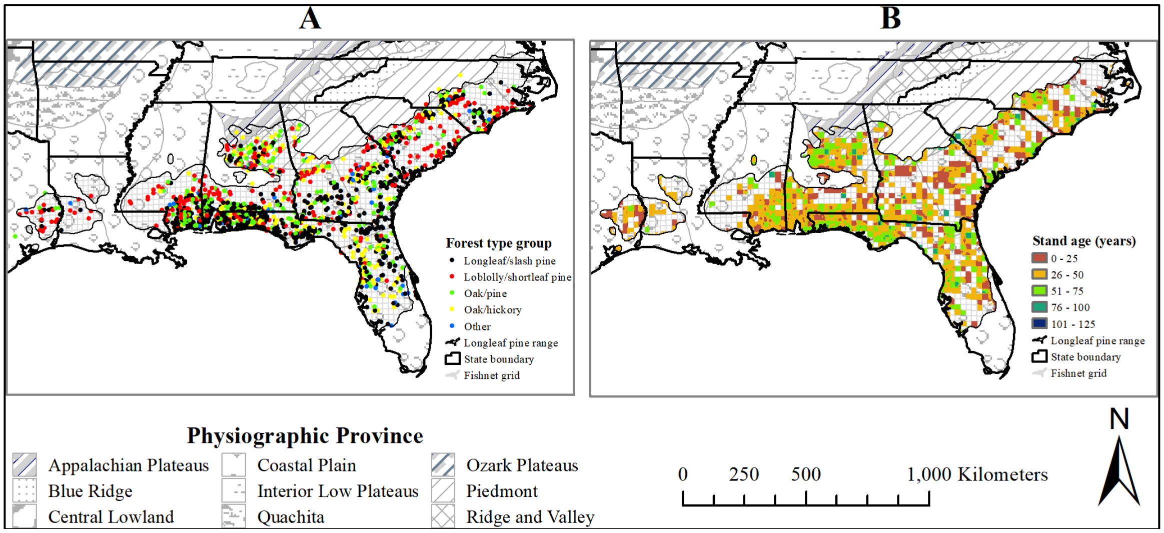

Figure 2.15: Spatial distribution of plots with dominant forest groups (A) and stand age classes derived from the FIA 2004 dataset (B) obtained from Ojha et al. (2021).

The longleaf pine habitat analyses that we developed were based on ecological data that focused on soil, vegetation, and land use criteria associated with longleaf pine habitats in the Carolina Sandhills Wiregrass Gap region. While only 1.4% of our parcels were potentially appropriate for restoration, this accounted for almost a third of the total area within our study site, acting as a testament to the degree of urbanization and habitat fragmentation in the region but also to the potential for forest restoration. Large privately owned parcels often indicate that at least part of the property is undeveloped and, given that our models produced large prediction percentages for appropriate longleaf pine habitat and that the entire region was once dominated by longleaf pines, we have to assume that parcels that cover large areas of land have the greatest potential to support silvicultural treatments that restore these habitats. If smaller parcels would have been included in the analysis, the percentage of area classified as ideal restoration habitat would undoubtedly decrease. However, it would still be higher than the percentage of suitable habitat across the entirety of our original study area (26.8-29.8%) because appropriate ecological characteristics would be present in some parcels between 0-30 hectares.

Based on our test parcels, we know that there is some variability in each model's accuracy and thus requires additional testing and on-ground proofing through additional fieldwork and research efforts. However, the addition of parcellation allows future researchers not only to test the accuracy of each model but also to employ sample restoration efforts. The Hardscramble property, for example, is a parcel that has employed a similar restoration in the last two years and has been extensively tested for its suitability (Murray, Chapter 1). Only part of the property is being restored, but that area was classified with a rank of 2, which indicates plausible suitability. However, we know that longleaf is present on the property, questioning the accuracy of habitat identification. Nevertheless, because longleaf pine ecosystems once dominated this entire area, the only habitat that would not be suitable would theoretically be wetlands and developed areas. As indicated by our models, suitability increases as the spatial distance from the city of Columbia increases. Moreover, wetlands were almost exclusively deemed inappropriate for restoration, illustrating serious declines in appropriateness along major waterways such as the Wateree river. This leads us to believe that at least one, if not all of our models, could be applied in critical decision making for potential restorations with some additional testing.

Overall, each model was similar in spatial classification (Figure 2.13). However, models that weighted forest type and land use higher than other categories were more conservative with regards to predicting the presence of longleaf pine (Figure 2.14). This variability leads us to believe that our equally weighted model and our soil model most likely overrepresented the number of longleaf pines within the region but were also very similar in spatial prediction, yielding the most optimistic results. Comparatively, it could also be argued that these were our most accurate models as well, considering they both predicted the presence of longleaf in 8 of 9 of our test parcels which were not identified through the other two illustrations. This is promising, considering they both predicted that over 90% of the area for parcels larger than 30 hectares were either appropriate for longleaf pine restoration or already had longleaf present. However, the presence of longleaf is not seemingly as significant, considering it only serves as an indication of restoration capabilities. Across all models, there was a visual concentration of potential restoration sites in the northeast portion of the study area, which coincided with an increase in the number of large-sized parcels. This is another point of contention as much of this area is privately owned, meaning that surveys and data collection efforts need to be extensively applied to make accurate restoration and management predictions.

However, other conservation efforts, such as the ACE Basin Project which currently protects over 160,000 acres of wetlands and wildlife habitat from commercial and residential development, have been highly successful in protecting and restoring sensitive habitats. The ACE Basin Project has been able to protect over 70,000 acres of privately owned land in rapidly growing metropolitan areas through conservation easements and educational outreach. A similar conservation approach could theoretically be applied in the Northeast portion of the SC Sandhills Wiregrass Gap due to its abundance of large parcels and appropriate habitat outlined by our model. Urbanization is an ever-encroaching phenomenon that can be curbed through conservation initiatives. However, the first step in protecting vital ecosystems is inventorying what is available and then identifying what is possible. A collection of conservation easements in areas outlined by our model could potentially increase the amount of longleaf pine habitat over time, reducing fragmentation, preventing future losses, and increasing awareness.

2.5 Conclusions

Longleaf pine habitat is a vital ecosystem type and remains historically and ecologically significant in the South despite its current availability compared to other species abundances. However, spatial modeling approaches can accelerate the rate at which longleaf pine habitats are restored by predicting the suitability of a location based on observed ecological attributes. This study employed one method of suitability modeling to identify appropriate sites for longleaf pine restoration in the South Carolina Sandhills Wiregrass Gap and did so with the addition of parcellation. By providing multiple models, we have created an opportunity for future researchers to test their accuracy and build off our criteria while targeting areas of interest for future restoration projects. While urbanization is undoubtedly high, there is still a large percentage of the region that is rural and retains the ability to support silvicultural treatments and other restorative methods. Conservation initiatives are possible in this region, and spatial modeling that can illustrate that to the public can acceleration the overall restoration of longleaf pine ecosystems. By investigating our models, a landowner could realize their property's capability, conservationists could target specific sites, and researchers could refine the target area based on additional criteria. Further testing and spatial investigation need to be applied in this region and in similar niche environments to advance the support and increase the scope of longleaf pine restoration in South Carolina.

CHAPTER THREE

ASSESSING THE PRODUCTIVITY AND COST OF TIMBER HARVESTING DURING A SOUTH CAROLINA LONGLEAF PINE RESTORATION

Abstract

As a function of silvicultural treatments, timber harvesting operations are helpful tools in restoring sensitive habitats such as longleaf pine ecosystems. However, logging companies are often functionally limited by operational cost and site-specific productivity potential. By employing a time and motion study, this project aimed to quantify the cost and productivity of conventional logging equipment in a South Carolina southern pine timber harvest used to promote longleaf pine. Our study found that the feller-buncher, grapple skidder, and knuckle-boom loader produced, on average, 60.7, 23.6, and 45.2 tonnes per Productive Machine Hour (PMH), respectively. However, when the knuckle-boom loader was solely loading log trucks, productivity increased to 48.4 tonnes per PMH. Additionally, a machine rates analysis found that the total cost for each machine was \$106.60, \$119.09, and \$119.35 per PMH, respectively, with a total per tonne cost of cutting, skidding, and loading of \$9.43.

3.1 Introduction

Due to the ecological value of longleaf pine (*Pinus palustris* Mill.) ecosystems and their widely recognized decline in past centuries, restoration efforts have been a forest management focus for over 70+ years (Condon and Putz, 2007; Engstrom, 1993; Moser et al., 2002; Neel et al., 2011; Westerhold, 2013). Longleaf pine habitats across the Southeastern US support over 29 threatened and endangered wildlife species, 900 plant species, and hundreds of bird, mammal, and herpetofauna species, a majority of which forage on or near the herbaceous understory (Engstrom, 1993; Harrington et al., 2013; U.S, Fish and Wildlife Service, 2009; Van Lear et al., 2005; Westerhold, 2013). Reestablishing herbaceous vegetation is an essential aspect of longleaf pine restoration and should be completed before or during longleaf pine planting (Walker and Silletti, 2007). Besides wildlife forage, this vegetation acts as an important fuel source for prescribed burning, an essential element in promoting longleaf pine growth and maintaining the rich and diverse herbaceous layer found in longleaf pine savannas (Brockway, 2005; Walker and Silletti, 2007). Restoration methods that re-instate these ecological processes accelerate the recovery, health, and sustainability of these systems.

Timber thinning, selective cutting, and clear-cutting are three common examples of harvesting applications that have been found to promote longleaf pine habitat and restore herbaceous understory fuels used for sustainable forest management (Brockway, 2005; Brockway et al., 2007; Harrington and Edwards, 1999; Walker et al., 2004). By reducing the canopy cover and removing a portion, or all, of the basal area, light can penetrate the understory and promote early successional habitat. The most common method of longleaf pine restoration is the application of a clear-cut followed by planting and the employment of one or more release treatments (Knapp et al., 2006; Knapp et al., 2014). However, seed tree cuts can act as alternative forms of restoration harvesting that promote natural regeneration when there is an established population of longleaf pines (Boyer and Peterson, 1983; Croker, 1976). This restoration methodology mimics natural disturbance regimes and creates early successional habitat suitable for longleaf pine understory planting while promoting natural regeneration (Walker and Silletti, 2007). With the forest industry being a highly cost-intensive business with extreme regional variability (Hiesl and Benjamin, 2013), restoration is contingent on several factors, including economic viability. Successful logging businesses are essential to the forest products industry and forest management as a whole, significantly influencing the capabilities of foresters and landowners who aim to employ silvicultural treatments to meet management objectives.

Cost data comparisons from past and current logging operations are essential for estimating future operational costs and increasing profitability. Average total costs for timber harvesting are dependent on productivity and the individual factors of production (e.g., type of equipment, harvesting conditions, management structure, innovation, labor, rules and regulations) (Mac Donagh et al., 2019). A significant aspect of production costs are machine rates, defined as the summation of fixed, variable, and labor costs per machine while in use (Heinrich, 1992). Machine rates are influenced, among other things, by machine life, insurance, depreciation, taxes, fuel costs, repair, maintenance, and labor costs. Life expectancy and the initial purchasing price of machines remain the most critical factors regarding machine costs, with a one-year change in machine life resulting in a 10-15% change in operating costs per Productive Machine Hour (PMH) (Brinker et al., 2002). Conventional ground-based harvesting efforts can cost as much as 40-50% of the delivered cost of wood, and additional costs can make profitability difficult in the absence of large volumes of valuable material (Wood-Energy, 2019). Predictably, this is influenced by the type of equipment used as well as the harvesting conditions, which can range from the type of stand to the season in which the operation occurs. Therefore, studies that are able to evaluate productivity and cost over a large range of variables can help increase productivity, profitability, and our general understanding of industry trends over time (Koŝir et al., 2015; Szewczyk and Sowa, 2017; Stampfer and Lexer, 2001).

The primary purpose of work studies in the logging industry is performance evaluation, and time and motion studies capable of analyzing the time consumption of elemental work tasks are the most appropriate forms of investigation (Spinelli and Visser, 2008). Manual time studies remain the most common form of exploration and are crucial in promoting innovation and advances in the logging industry (Koŝir et al., 2015). This type of methodology can quantify machine and system productivity, which, when coupled with machine rates, can be used to calculate the per-unit cost of production (Adebayo et al., 2007; Contreras et al., 2017; Koŝir et al., 2015). Productivity is often measured in weight per hour of work or volume per work cycle, and variations in a myriad of harvesting conditions can result in fluctuating levels of production and cost (Conrad et al., 2018; Hiesl and Benjamin, 2013; Spinelli et al., 2010). For example, harvesting volume per area and owner/operator experience are significant factors in predicting per unit logging costs (Germaine et al., 2019).

The purpose of this study was to evaluate the productivity and cost of timber harvesting equipment in South Carolina during a seed-tree timber harvest designed to promote longleaf pine habitat. Our specific study objectives were to:

I. Evaluate time consumption for elemental work tasks performed by each piece of harvesting equipment (feller-buncher, grapple skidder, knuckle-boom loader) during a conventional seed-tree timber harvest.

II. Quantify the productivity of each machine.

III. Estimate the cost of production for each machine and from stump to truck.

3.2 Methods

3.2.1 Study Site

The study area was a 28.7 hectare (71 acres) subsection of a 304.7 hectare (753 acres) property called Hardscramble located in Camden, SC, in Kershaw County, USA (Coordinates: 34.267328, -80.656685). The site exhibited ecological conditions typical of those found in the Carolina sandhills and retained an established internal forest road system (Figure 3.1). Kershaw County has a humid, subtropical climate with year-round rainfall and hot summers (SC Climatology Office, n.d.). Camden averages 97 cm of precipitation annually with a minimum average annual temperature of 10.3 degrees C and a maximum average annual temperature of 24 degrees C. The study site retains a udic soil moisture regime and consists of well-drained sandy soils with gentle to moderate slopes (<8% on average), making it conducive for wet-weather logging. Before our harvest, the study area was comprised of dense uneven-aged sets of loblolly pine (*Pinus taeda* L.) stands with scrubby hardwood midstories and isolated pockets of longleaf pine. Large timber (>30.5 cm DBH) was present, with some pines estimated to be over 150 years old, but the majority of timber was pulpwood-sized with low heights (approx. 18 m) and small diameters (approx. 25 cm and smaller). A timber cruise conducted in June 2020 by a consulting forestry company estimated the study area to have 266.7 tonnes (294 tons) of pine chip-n-saw, 717.6 tonnes (791 tons) of pine sawtimber, 727.6 tonnes (802 tons) of pine pulpwood, 75.3 tonnes (83 tons) of grade hardwood logs, 22.7 tonnes (25 tons) of gum logs, and 1215.6 tonnes (1,340 tons) of hardwood pulpwood.

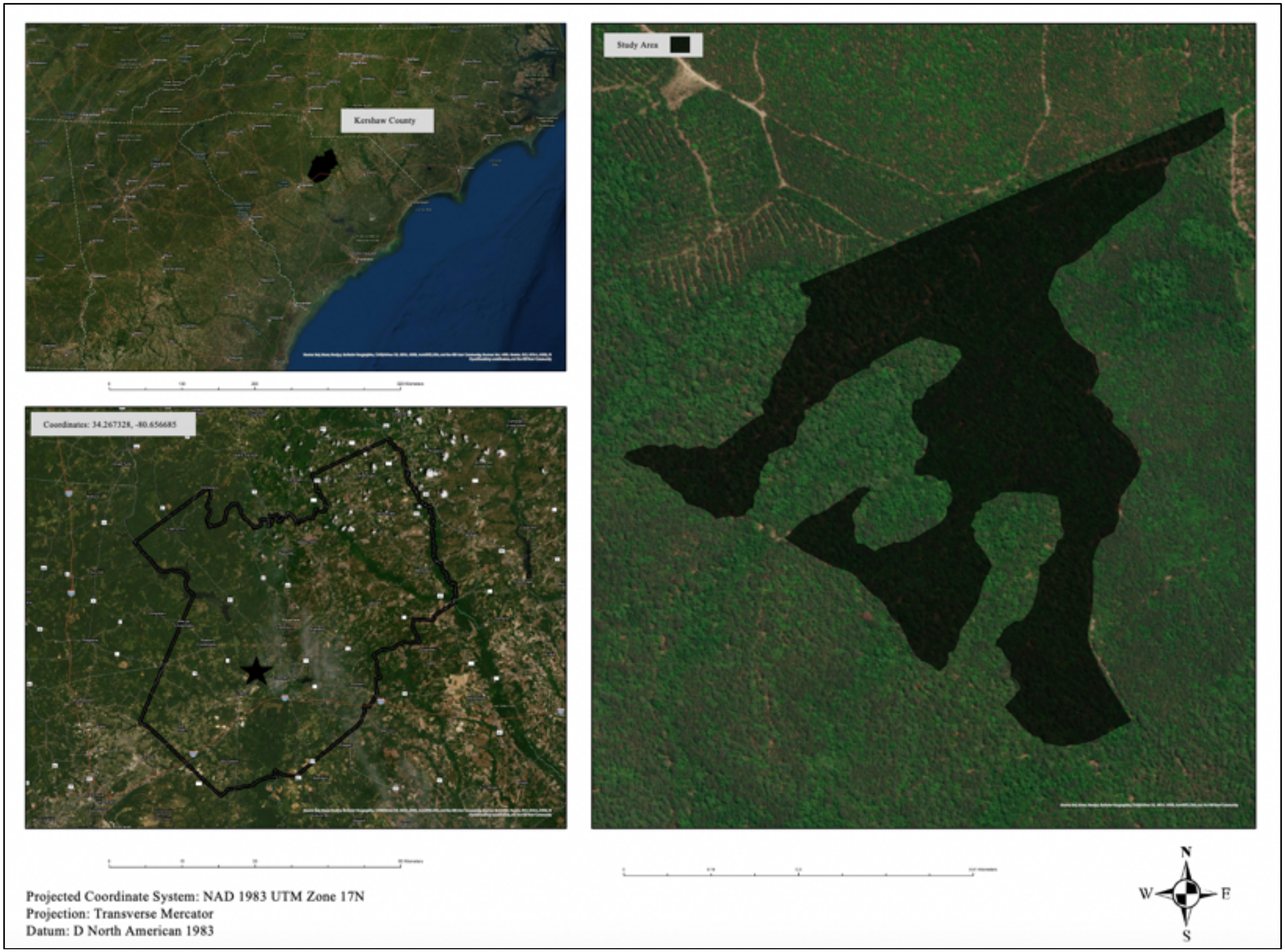

Figure 3.1: Map of 28.7 hectare seed-tree harvest study site located on Hardscramble in Camden, SC.

3.2.2 Harvesting Equipment and Operation

The study area was harvested during November and December of 2020 using a seed-tree harvest designed to promote longleaf pine habitat. Using conventional whole-tree timber harvesting, our study aimed to remove all loblolly pine trees and all hardwoods less than 30.5 cm DBH, leaving only longleaf pine trees and a few remaining hardwood seed trees evenly spread across the site. The resulting stand conditions represented a typical seed-tree harvest in the majority of the site, with about 7.3 hectares (18 acres) of the study area illustrating a shelterwood harvest due to the abundant clustering of longleaf pine trees in xeric sandy ridges along its northern boundaries. The logging deck location was moved once to decrease the skidding distance, resulting in an average maximum skidding distance of 461.9 meters. The operation consisted of a two-person crew operating a Tigercat 720G rubber-tired feller-buncher, a John Deere 748-II series, dual-function rubber-tired grapple skidder with the largest grapple capacity (1.77 m^2), and a Tigercat 234B trailer-mounted knuckle-boom loader with a pull through delimber (Figure 3.2). Infrequently, if both operators were working away from the logging deck when the log truck arrived, the truck driver would operate the loader to load the truck.

Operator 1 (>45 years old) was the senior operator and had over 25 years of experience running equipment. He was the feller-buncher operator, the main loading operator, and acted as the project foreman. Operator 2 (27 years old) was the other equipment operator on the site and had less than 10 years of experience. He was the main skidder operator and would split his time between skidding and loading. Operator 3 (>45 years old) was the truck driver and would infrequently load his own log truck but did not use any other machinery.

Figure 3.2: Tigercat 720G drive-to-tree feller-buncher (A), John Deere 748-II series, dual-function rubber-tired grapple skidder (B), 234B Tigercat trailer-mounted knuckle-boom grapple loader and pull through delimber (C) in a Southern pine seed-tree harvest in Camden, SC.

3.2.3 Time and Motion Study

Time and motion data was compiled through video footage and then analyzed to compare the time consumption of elements in a work cycle for each machine. GoPro Hero cameras were attached to the inside cab window facing the felling head in the feller-buncher and facing the grapple in the grapple skidder. A tripod mounted digital camera was placed at the logging deck to capture loading operations, including log truck interactions across multiple operators. Time measurements were recorded for each work element within a cycle (Table 3.1), using the time study software UMT+ (Laubrass Inc.). Three productive and two unproductive work tasks were identified as elements in work cycles for each machine, with PMH excluding all delays (e.g. mechanical delays and nonmechanical delays). For each cycle, the number of stems processed was recorded. Machine utilization was calculated using the ratio of PMH to observed machine hours (OMH) in lieu of Scheduled Machine Hours (SMH). OMH were used to calculate utilization due to the nature of the study, which filmed over several partial days, providing a good representation of time consumption and delays, but did not provide PMH and SMH for a complete day. Machine productivity was estimated using the average green weight of merchantable stems to a 10.2 cm DOB for loblolly pines (0.24 tonnes) which were visually assessed to have a 20.3 cm DBH and an 18.3 m total height (Saucier et al., 1981).

Table 3.1: Machine cycle and element description for timber harvesting equipment for a Southern pine seed-tree restoration harvest in Camden, SC.

Machine	Work Element	Definition
Feller-Buncher	Empty Felling Head Movement	Empty-head travel time between trees and cutting sites. Initiated immediately after a new cycle and stops with the first cut.
	Cutting Trees	The act of sawing, accumulating timber, and moving loaded. Initiated after the first cut and stops when the bunching head begins to lean forward to drop a load.
	Bunching	Drop and placement of bunched timber. New cycles were initiated at the beginning of the study and once a load was dropped and the bunching head was erect and empty. Leaning forward to empty the bunching head started the task.
	Mechanical Delays	Machine delays that require machine repair or maintenance. Non-productive work element.
	Nonmechanical Delays	Elapsed idling time for machines that are running but not performing any tasks. Non-productive work element.
Grapple Skidder	Empty Travel	Empty-grapple travel time between the logging deck and bunched stems. Once the load was placed at the logging deck and the machine began to move empty a new cycle started. All elapsed time, including miscellaneous activity, was included in empty travel time until the skidder began to back-up to grab a load.
	Loaded Travel	Loaded travel initiated when the skidder backed towards a pile and stopped when the skidder returned to the deck.
	Load Placement	The act of backing up a load to the logging deck and dropping it off. New cycles were initiated once the load was dropped and the skidder began to move unloaded.
	Mechanical Delays	Machine delays that require machine repair or maintenance. Non-productive work element.
	Nonmechanical Delays	Elapsed idling time for machines that are running but not performing any tasks. Non-productive work element.
Knuckle-boom Loader	Empty Movement	All unloaded movement while processing and sorting stems. New cycles were initiated after a load was placed and new unloaded movement began. The elapsed time for this element stopped once the loader picked up one or more stems.
	Loaded Movement	Any task that was completed with stems in the grapple. Productive work element that was initiated with the grab of one or more stems and ended with the initiation of unloaded movement.
	Cleaning Deck	Removing excess logging slash and sweeping the deck with a horizontal stem. Initiated once the loader grabbed debris and stopped once the boom moved unloaded towards merchantable stems.
	Mechanical Delays	Machine delays that require machine repair or maintenance. Non-productive work element.
	Nonmechanical Delays	Elapsed idling time for machines that are running but not performing any tasks. Non-productive work element.
Log Truck	Empty Grapple Movement	All unloaded movement while processing and sorting stems. New cycles were initiated once the log truck arrived and after a load was placed and new unloaded movement began. The elapsed time for this element stopped once the loader picked up one or more stems.
	Loaded Grapple Movement	Any task that was completed with stems in the grapple. Productive work element that was initiated with the grab of one or more stems and ended with the initiation of unloaded movement.
	Cleaning Deck	Removing excess logging slash and sweeping the deck with a horizontal stem. Initiated once the loader grabbed debris and stopped once the boom moved unloaded towards merchantable stems.
	Mechanical Delays	Machine delays that require machine repair or maintenance. Non-productive work element.
	Nonmechanical Delays	Elapsed idling time for machines that are running but not performing any tasks. Non-productive work element.

3.2.4 Machine Rates and Operational Costs

Machine rates for harvesting equipment were calculated using the machine rate worksheet provided by the Alabama Agricultural Experiment Station (Brinker et al., 2002). Machine rate input values were obtained from the logging company, previous studies, and southeastern logging reports. The logging company provided the average purchasing price of their machinery ($270,000) and the machine life for each machine (3-5 years). The logging company also provided the scheduled machine hours for each machine, estimated to be about 2,500 hours per year. The average operator wage ($16.74/hr) and benefit rate (30.8%) was identified from occupational employment and wage statistics for logging operators provided by the US Bureau of Labor Statistics (2021) in combination with compensation indices for logging and trucking occupations provided by Baker and Brooks (2016). Off-road fuel costs were acquired through 3rd quarter averages provided by TimberMart South, Inc. (2020), while machine horsepower ratings were compiled from machine brochures provided by Tigercat International Inc. (2019, 2020) and Deere and Company (n.d.). All other input variables, including salvage values (%), repair and maintenance values (%), interest rates (%), insurance and tax rates (%), fuel consumption rates (gal/hp-hr), and lube and oil rates (%) were obtained from a compilation of surveys and literature reviews outlined by Brinker et al. (2002). We calculated ownership, operating, and total costs for each piece of logging equipment using these values. The unit cost of production was then calculated dividing machine rates by machine productivity.

3.2.5 Data Analysis

All data analyses used to quantify time and motion, productivity, and cost were conducted using Microsoft Excel (Microsoft Corporation, 2018), R (R Core Team, 2020), and RStudio (RStudio Team, 2020). Due to the small number of samples, a Kruskal-Wallis test ($\alpha < 0.05$) was employed as a non-parametric alternative to an ANOVA test used to compare probability distributions for utilization rates between machines and different aspects of log truck loading between operators. For significant findings, we calculated the effect size and employed a post-hoc pairwise comparison using a Dunn test.

3.3 Results

3.3.1 Feller-buncher Time and Productivity

We observed approximately 10 hours of machine activity for the rubber-tired feller-buncher used in our seed-tree harvest. Per cycle, cutting trees was the most observed work task (51.3%), followed by empty felling head movement (26.1%), and bunching (20.0%). Mechanical delays and non-mechanical delays accounted for 0.4% and 2.1% of observation time per cycle. The average cycle time was 55±76 seconds with an average delay-free cycle time of 47±31 seconds. The average time consumption for the productive work elements of cutting, empty felling head movement, and bunching was 26±23, 13±19, and 8±4 seconds, respectively. The average time consumption for mechanical and nonmechanical delays was 7±58 and 1±17 seconds per cycle, respectively. Across all observations, the feller-buncher harvested a total of 2,159 trees over 649 cycles, approximately 3.3±2.4 trees per cycle. For the 649 observed cycles, the majority (63%) were cut in accumulations of 1-3 trees, with 16% of the bunches accumulating six or more (Figure 3.3). The largest bunch accumulation was 14 trees, primarily attributed to high amounts of small diameter timber. From these observations we estimated that feller-buncher productivity averaged 0.8±0.6 tonnes per cycle and 60.7 tonnes per PMH.

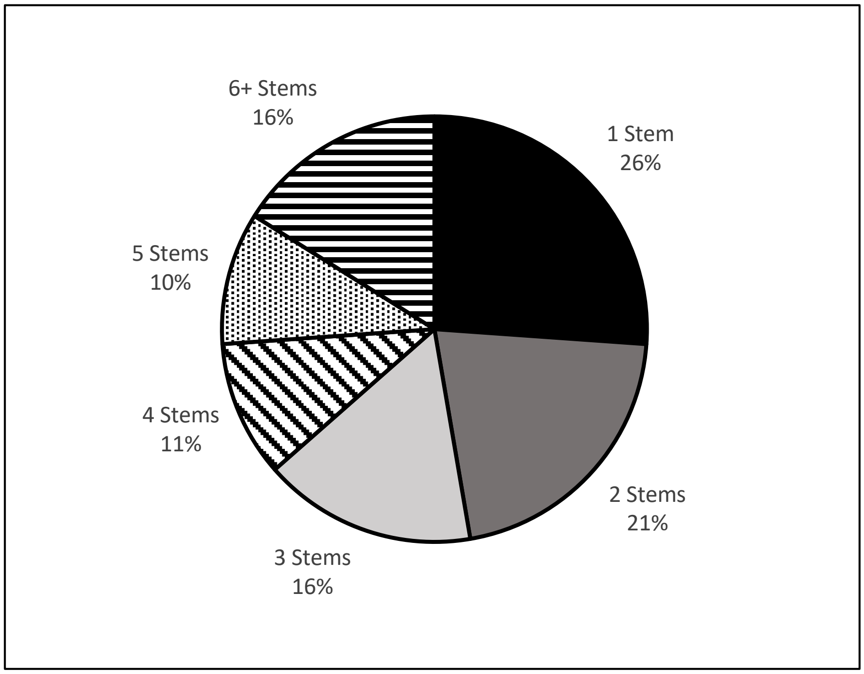

Figure 3.3: Frequency of tree count in feller-buncher head accumulations in a Southern pine seed-tree harvest in Camden, SC (n= 649).

3.3.2 Grapple Skidder Time and Productivity

Across 19.1 hours of total observation time, the most observed task per cycle for the rubber-tired grapple skidder was empty travel (45.1%), followed by loaded travel (38.0%) and load placement (4%), with nonmechanical delays and mechanical delays accounting for 12.5% and 0.4%, respectively. The average cycle time was 11.0±12.5 minutes with an average delay-free cycle time of 7.4±4.3 minutes. The average time consumption for the productive work elements of traveling empty, traveling loaded, and placing a load was 4.1±3.4, 3.2±2.2, and 0.2±0.2 minutes, respectively. The average time consumption for mechanical and nonmechanical delays was 0.1±0.6 and 3.6±10.7 minutes per cycle, respectively. Across all observations, the skidder accumulated 1,397 trees over 104 cycles, approximately 12.2±8.2 trees per cycle. From these observations we estimated that grapple skidder productivity averaged 2.9±2.0 tonnes per cycle and 23.6 tonnes per PMH.

3.3.3 Knuckle-boom Loader Time and Productivity

We observed 23.7 hours of machine activity for our trailer-mounted knuckle-boom loader used in our seed-tree harvest. For productive work tasks per cycle, loaded grapple movement (58.0%) consumed the most time, followed by empty grapple movement (35.6%) and deck cleaning (0.5%). Mechanical delays and nonmechanical delays accounted for 0.5% and 4.7% of observation time per cycle. The average cycle time was 1.5±4.4 minutes with an average delay-free cycle time of 0.9±0.4 minutes. The average time consumption for the productive work elements of loaded grapple movement, empty grapple movement, and deck cleaning took, on average, about 31±22, 19±14, and 1±5 seconds per cycle, respectively. The average time consumption for mechanical and nonmechanical delays was 2±36 and 39±255 seconds per cycle, respectively. Across all observations, the knuckle-boom loader was able to process or load 2,039 stems over 945 cycles, approximately 2.4±1.7 trees per cycle. For 945 observed cycles, the majority (81%) were cut in accumulations of 1-3 trees, with 19% of cycles processing four of more (Figure 3.4). The largest grapple accumulation was 12 trees, indicating that there was an abundance of small diameter timber and that a majority of trees were processed before loading. From these observations we estimated that knuckle-boom loader productivity averaged 0.6±0.4 tonnes per cycle and 45.2 tonnes per PMH.

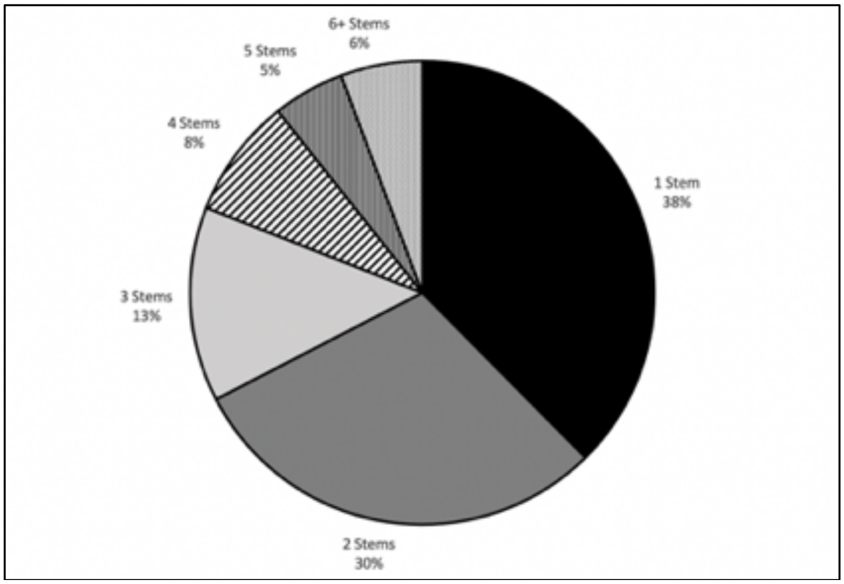

Figure 3.4: Frequency of stem count in grapple accumulations for the trailer-mounted knuckle-boom loader in a Southern pine seed-tree harvest in Camden, SC (n= 945).

3.3.4 Log Truck Time and Productivity

During the observed machine time for our trailer-mounted knuckle-boom loader, we were able to capture 18 truck loadings which totaled approximately 6.0 hours. This indicates that a log truck was present for 25.3% of the total observed time of the knuckle-boom loader. On average, each truck retained 62.8±28.8 trees and took 20±7.9 minutes to load. Each truck was loaded with average of 22.3±4.8 grapple cycles.

For productive work elements per cycle, loaded grapple movement consumed the most time (56.7%), followed by empty grapple movement (41.4%) and deck cleaning (0.5%). Mechanical and nonmechanical delays accounted for 0.2% and 1.2% of observation time per cycle, respectively. The average cycle time was 53±46 seconds with an average delay-free cycle time of 50±25 seconds. The average time consumption for the productive work elements of loaded grapple movement, empty grapple movement, and deck cleaning was 30±20, 20±13, and 0±2 seconds per cycle, respectively. The average time consumption for mechanical and nonmechanical delays was 0±3 and 3±41 seconds per cycle, respectively. Across all observations, the knuckle-boom loader loaded 1,131 trees over 419 cycles, approximately 2.8±1.2 trees per cycle. For 419 cycles, the majority of grapples (70.8%) had 1-3 trees, with two stems acting as the most common accumulation size, and a maximum number of 12 stems (Figure 3.5). From these observations we estimated that loading productivity averaged 0.7±0.3 tonnes per cycle and 48.4 tonnes per PMH.

Three different operators were involved in loading log trucks, with the most experienced operator acting as the main loader operator and loading 61.1% of the observed log trucks. Average loading time varied as much as 17% between operators but there were no significant differences for the loading duration ($p= 0.515$) or the number of cycles ($p= 0.725$) needed to load a truck. There were, however, significant differences and small effects observed between operators for the average time per cycle ($p= 0.028$) and the average number of trees per cycle ($p= 0.027$), with pairwise comparisons finding that Operator 1 had significantly shorter cycle times ($p= 0.036$) and loaded less trees per cycle ($p= 0.034$) than Operator 3 (Table 3.2). As a result, Operator 2 was the least productive, followed by Operator 3 and Operator 1, with an 8.2% increase in productivity from Operator 1 to Operator 2 (Table 3.2).

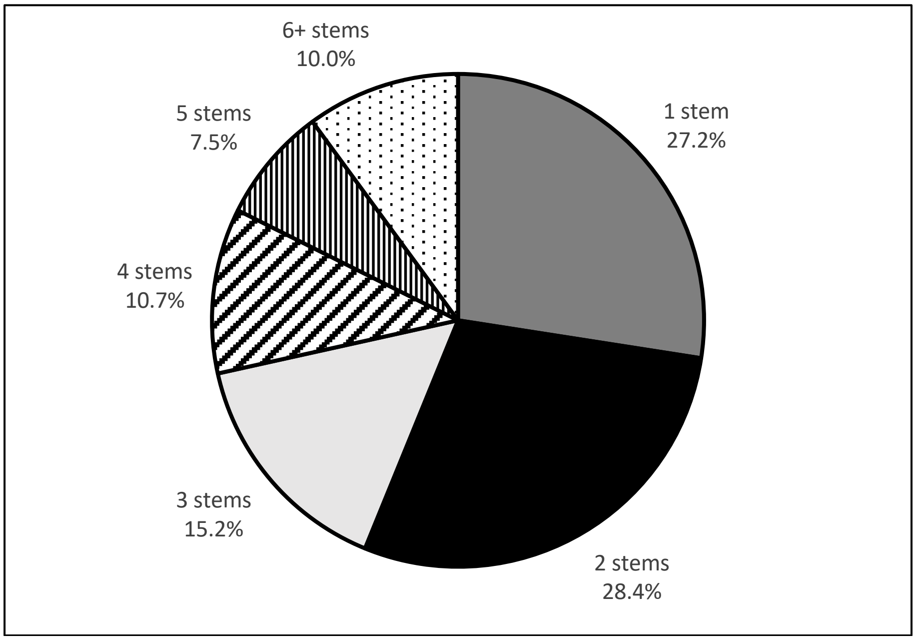

Figure 3.5: Frequency of stem count in grapple accumulations for trailer-mounted knuckle-boom log truck loading in a Southern pine seed-tree harvest in Camden, SC (n= 419).

Table 3.2: Operator differences for log truck loading in a Southern pine seed-tree timber harvest in Camden, SC. Values that are labeled with double asterisks are significantly different from each other (α < 0.05).

Calculations	Operator 1 (Feller-buncher Operator)	Operator 2 (Skidder Operator)	Operator 3 (Truck Operator)
Trucks Loaded	61.1%	16.7%	22.2%
Average Duration	0:18:40 ± 0:07:14	0:21:36 ± 0:12:04	0:22:26 ± 0:08:11
Average Time/Cycle	0:00:50 ± 0:00:40*	0:00:52 ± 0:00:27	0:01:02 ± 0:01:07*
Average Stems/Cycle	2.6 ± 2.0*	2.5 ± 1.8	3.1 ± 2.1*
Average Tonnes/PMH	44.9	41.5	43.2

3.3.5 Machine Utilization

Machine utilization varied greatly between machines, ranging from 55-84% of observed machine time (Table 3.3). The feller-buncher exhibited the highest utilization rates followed by the grapple skidder and the knuckle-boom loader. There were significant differences and large effects seen between the utilization rates for each machine ($p = 0.002$) with pairwise comparisons indicating that feller-buncher utilization was significantly higher than the knuckle-boom loader utilization ($p = 0.001$) (Figure 3.6). This variation is largely attributed to increases in non-mechanical delays as mechanical delays were uncommon during our period of observation. For over 52 hours of total observed machine hours, mechanical delays accounted for just 52.2 minutes producing between 98-99% availability for all machines. Over time we see a similar trend where feller-buncher utilization is consistently high compared to the knuckle-boom loader and the grapple skidder (Figure 3.7). However, this is all dependent on logging crew dynamics as there was a two-man crew working three machines. For example, there is a sharp uptick in grapple skidder utilization towards the end of the harvest, likely because of a decrease in feller-buncher use.

Table 3.3: Summary of utilization rates (%) based on productive, observed, and available machine hours across timber harvesting equipment in a Southern pine seed-tree harvest in Camden, SC.

Machine Type	Productive Machine Hours (hh:mm:ss)	Observed Machine Hours (hh:mm:ss)	Available Machine Hours (hh:mm:ss)	Machine Utilization Rates (%)
Feller-buncher	8:25:14	9:58:12	9:45:30	84%
Grapple Skidder	12:48:00	19:05:53	18:59:24	67%
Knuckle-boom Loader	13:09:39	23:44:35	23:11:33	55%

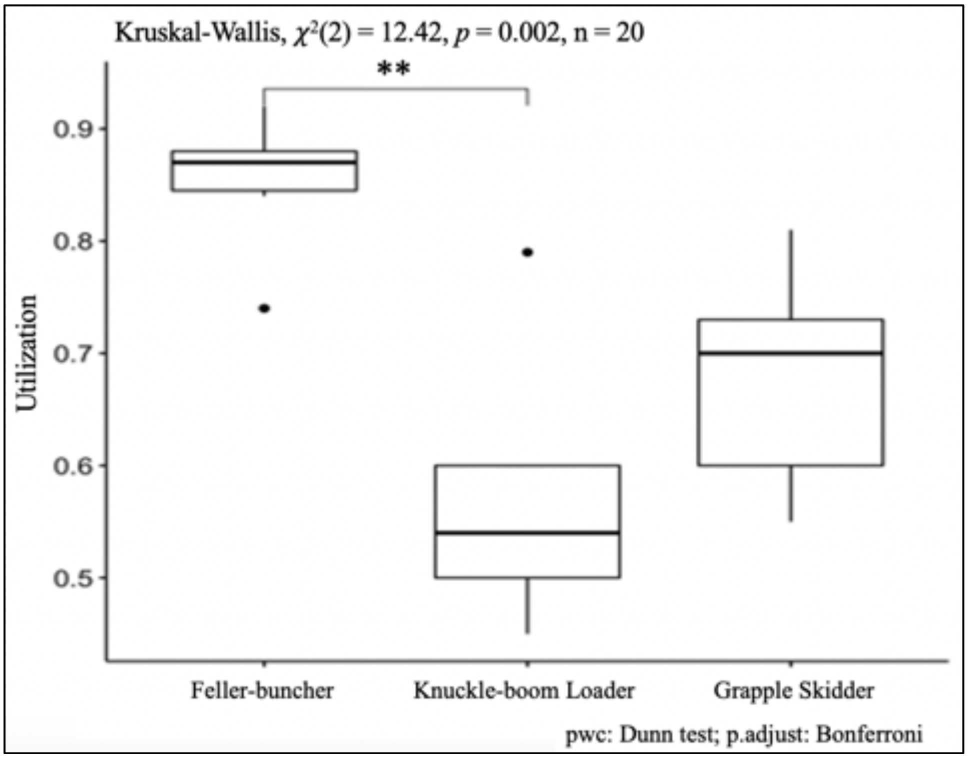

Figure 3.6: Boxplot of utilization rates for timber harvesting equipment in a Southern pine seed-tree harvest in Camden, SC. Significant values are identified by asterisks ($\alpha < 0.05$).

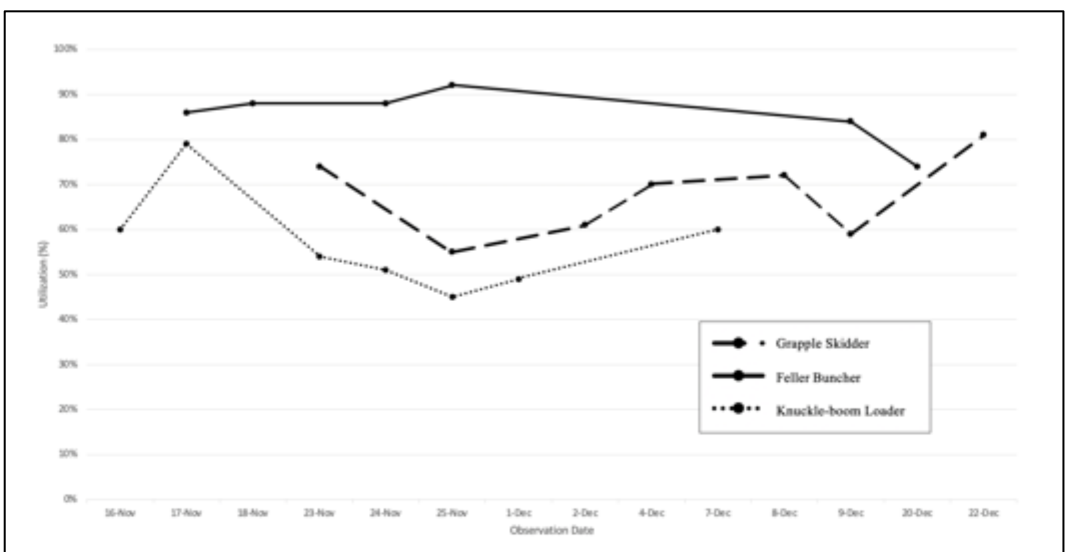

Figure 3.7: Timber harvesting equipment utilization rates (%) across sampling dates in a Southern pine seed-tree harvest in Camden, SC.

3.3.6 Machine Rates and Cost

The logging company provided information on several machines, with incomplete information being substituted with information from the literature (Table 3.4). The purchasing price for each machine averaged 275,000, and the machine life ranged from 4 to 5 years. The logging company further reported that their scheduled machine hours amounted to 2,500 hours per year for each machine, or 50 hours per week, excluding two weeks per year for time off. From these input values, we calculated that the feller-buncher used in our harvest had the highest cost per scheduled machine hour ($89.09/SMH) but the lowest cost per productive machine hour ($106.06/PMH) across all machines. In comparison, the rubber-tired grapple skidder exhibited the second highest cost per scheduled machine hour ($79.79/SMH) and per productive machine hour (119.09/PMH). However, while the knuckle-boom loader showed a considerable decrease in the cost per scheduled machine hour ($65.64/SMH), the total cost per productive machine hour was similar to the findings for the rubber-tired grapple skidder ($119.35/PMH), differing only by $0.26/PMH. Using these values, the unit cost of production for felling, skidding, and loading was $1.75, $5.05, and $2.64 per tonne, respectively. Total unit cost of production for cutting, skidding, and loading was $9.43/tonne.

Table 3.4: Machine rate calculation input values and total costs per productive and scheduled machine hours for timber harvesting equipment used in a Southern pine seed-tree restoration harvest in Camden, SC.

Machine Rates and Cost	Rubber-tired Feller-buncher	Rubber-tired Grapple Skidder	Knuckle-boom Grapple Loader
Purchase Price ($)	275,000	275,000	275,000
Machine Horsepower Rating (HP)	203	263	168
Machine Life (years)	4	5	5
Salvage Value (%)	20	20	30
Utilization (%)	84	67	55
Repair and Maintenance (%)	100	90	90
Interest Rate (%)	10	10	10
Insurance and Tax Rate (%)	4.5	5	1.5
Fuel Consumption Rate (gal/hp-hr)	0.0263	0.028	0.0217
Fuel Cost ($/gal)	2.34	2.34	2.34
Lube and Oil Cost (%)	36.8	36.8	36.8
Operator Wage and Benefits ($/hr)	21.9	21.9	21.9
SMH (hrs/year)	2500	2500	2500
Interest Cost ($/yr)	19250.00	18700.00	19800.00
Insurance and Tax Cost ($/yr)	8662.50	9350.00	2970.00
Yearly Ownership Cost ($/yr)	82912.50	72050.00	61270.00
Ownership Cost per SMH ($/hr)	33.17	28.82	24.51
Ownership Cost per PMH ($/hr)	39.48	43.01	44.56
Fuel Cost ($/hr)	10.46	14.43	7.15
Lube Cost ($/hr)	3.85	5.31	2.63
Repair and Maintenance Cost ($/hr)	26.19	23.64	25.20
Operator Labor Cost ($/hr)	26.07	32.69	39.82
Operative Cost per PMH ($/hr)	66.58	76.07	74.79
Operating Cost per SMH ($/hr)	55.92	50.97	41.14
Total Cost per SMH ($/hr)	89.09	79.79	65.64
Total Cost per PMH ($/hr)	106.06	119.09	119.35

3.4 Discussion

3.4.1 Feller-buncher Time and Productivity

Feller-buncher cycle times are primarily influenced by disparities in terrain, stand densities, and the distance between trees, mainly because cutting time does not change significantly between diameter sizes (Akay et al., 2004; Hiesl and Benjamin, 2013). We were only able to observe around 10 hours of machine activity for the rubber-tired feller-buncher in our study because of scheduling and equipment limitations, but the act of cutting accounted for more than half of all observations per cycle. While empty felling head travel was the second most observed element, it only amounted to about half the cutting time. This could be explained by the high density of stems located in our study area (Akay et al., 2004; Green et al., 1987). Green et al. (1987) noted that feller-buncher productivity was greatest when working in closely spaced corridors, minimizing overall travel time. These conditions enable operators to maneuver with bunched stems, which was a common in our study and was classified as cutting time based on our work cycle definition. This likely increased the number of stems per cycle and resulted in a reduced empty felling head movement rate compared to other studies (Bilici, 2021; Lanford and Sirois, 1983; Wang et al., 2004). Bolstering this argument is the similar bunching size configurations found in our harvest to those with smaller tree sizes. About 74% of all feller-buncher head accumulations had two or more trees, similar to accumulation sizes found in small-diameter stands (Hiesl and Benjamin, 2013). Spinelli et al. (2002) found similar results in fast-growing Eucalypt stands where the drive-to-tree 4-wheeled feller-buncher averaged 4.1 trees per cycle, and the act of cutting was proportionately much higher than empty felling head movement. However, their productivity was much lower compared to our findings, most likely due to an abundance of small-diameter timber.

Feller-buncher productivity variability is widespread across different studies depending on a multitude of factors that range from the composition of a stand to the equipment used in a harvest. It has been noted that harvesting equipment designed to cut trees is especially susceptible to productivity fluctuations (Hiesl and Benjamin, 2013). Our study found similar productivity values compared to Andersson and Evans (1996) who reported feller-buncher productivity rates between 27.0-84.0 tonnes/PMH (converted to tonnes using average green weight of *Populus tremuloides* per cubic meter, 714 kg/m^3). However, this is unexpected considering their study retained large amounts of high-volume timber (0.23-0.69 tonnes/tree) found in overmature aspen stands. Our study site did retain some large-diameter timber (>35.6 cm DBH), but heights were relatively low (15.2-18.3 meters) on average, due to the stunting nature of the nutrient-poor, dry soils found in the region. Additionally, a majority of the timber harvested had small diameters, which would indicate a reduction in productivity. Conrad and Dahlen (2019) also reported high average productivity rates for feller-bunchers in conventional systems (98.0 tonnes/PMH per machine), but this was mostly attributed to large tree estimates (0.46-0.75 tonnes/tree). Productivity typically increases with an increase in stem size due to larger volume accumulations over the same period (Andersson and Evans, 1996; Wang et al., 2004; Li et al., 2006; Kluender et al., 1997). We expected our results to be more consistent with small-diameter harvesting studies, such as those found by Hiesl and Benjamin (2013) who reported productivity rates between 22.8-62.3 m^3/PMH (20.5-56.1

tonnes/PMH for a hardwood conversion factor of 0.9 tonnes/m^3). While our results did not exhibit extreme overestimation compared to this study, it should be noted that their study compared differences between swing-to tree feller-bunchers, which could have an impact. Hiesl and Benjamin (2013) also reported in a literature review that, of nine studies that researched feller-buncher productivity, a majority produced 20-30 m^3/PMH. Yet, many of those studies either had different stand structures, machinery, terrain, or harvesting systems (Gingras, 1989; Lanford and Stokes,1996; Wang et al., 2004).

Our large productivity estimates coupled with our relatively small average weight estimates could also be attributed to the inclusion of nonmerchantable timber during our cutting cycles, which drastically increased the number of stems per cycle. Our tonnage estimates were based on merchantable timber extracted by the grapple skidder, meaning that there was a large number of stems that were cut and included into our felling cycles that were not accounted for in the estimation. This resulted in fast and abundant cutting cycles with productivity rates near those in large-diameter stands. Additionally, the study site is located within the Coastal Plain, an area known to have high productivity due to its flat, gently sloping terrain (Conrad et al., 2018; Barrett et al., 2017). Combined with high utilization rates, this made our feller-buncher the most productive and least costly of all of the machines used in the study.

3.4.2 Grapple Skidder Time and Productivity

For the rubber-tired grapple skidder in our study, the three most observed time consumption elements, in descending order, were empty travel, nonmechanical delays and loaded travel. This is surprising, considering other studies have found that loaded travel typically accounts for a more significant percentage of productive time (Contreras et al., 2017; Kulak et al., 2017; Mousavi et al., 2013). However, both Kluender and Stokes (1994) and Wang et al. (2004) found similar results to ours where the average empty travel time was higher than loaded travel time. Study discrepancies could be attributed to the slash removal techniques incorporated into our study. Not only do BMPs require loggers to spread leftover slash across logging sites as a method of erosion control, but we also required loggers to pile slash in a third of the study area, which was an extraneous task required for another study being conducted on the same site. Extraneous tasks mixed in with typical logging operations can decrease overall productivity and cause time consumption fluctuations. Additionally, we incorporated slash removal as an empty travel time component due to the distinction made between dragging loads and carrying deck slash.

Our study found that grapple skidder cycle times took about 11 minutes per cycle, which was similar to other previous studies with comparable skidding distances (Akay et al., 2004; Kulak et al., 2017). During the harvest, our logging deck was moved to decrease skidding distance and increase overall productivity, resulting in a maximum average skidding distance of 461.9 meters, and an estimated average skidding distance of 221.6 meters. Kulak et al. (2017) found skidding cycles averaged between 12 and 16 minutes depending on the type of harvest and the skidding distance, with maximum distances ranging between 246 to 581 meters and average distances ranging between 124-246 meters. Akay et al. (2004) noted that each cycle only took about 10.63 minutes at an average distance of 300 meters, but with an average slope of 35%, which was surprising considering cycle times typically decline with an increase in distance and slope. Kluender and Stokes (1994) found that, for average skidding distances between 165.2 and 206.7 meters, the average cycle time ranged from 7.33 to 10.24 productive machine minutes for three different harvesting methods, with the shelterwood cut and the clear-cut being similar in harvesting intensity and average cycle time comparatively. Additionally, our skidding cycle times were similar to skidding times per productive hour for findings reported by Kluender et al. (1997). However, Santos de Freitas (2019) found that only one out of five skidders experienced a lower average cycle time than our study, which had an average skidding distance of 246.6 meters. However, a majority of skidders experienced cycle times greater than 18 minutes on average, with skidding distances ranging between 347.2 and 1,138.4 meters. Yet, the observed number of stems per cycle in our study was about three to four times greater than harvesting techniques investigated by both Kluender and Stokes (1994) and Santos de Freitas (2019), most likely due to the large number of small diameter trees found in our site.

Overall productivity was consistent with Andersson and Evans (1996), which was again, very high due to the high volume of large-diameter timber. The average load size was consistent with the range reported by Wang et al (2004), but skidding distance varied wildly for their study, resulting in a wide range of values. On average, Wang et al. (2004) reported average skidder productivity at about 13.1 tonnes/PMH. On the other hand, Conrad and Dahlen (2019) reported 31.8 tonnes/PMH for their skidder productivity, which was consistent with findings from Andersson and Evans (1996), indicating that large timber and exceptionally short skidding distances (average of 50 meters) increased overall skidding productivity. As a result, our increased productivity rate was most likely a result of our short cycle time, large bunch sizes, and short skidding distance on flat terrain (Conrad et al., 2018). In small-diameter stands, however, Bolding et al. (2009) noted that delay free skidding cycles took about 3.5 minutes, less than half of our delay free cycles, but only produced 14.7 tonnes/PMH for each machine.

While our findings indicate that the grapple skidder in our study was highly productive, it was still the least productive of all harvesting machines in our study and resulted in a majority of the observed bottlenecks. This finding is not uncommon and many operations employ two skidders to combat this. High-efficiency skidding is directly correlated to optimal bunch size and skidding distance, which changes depending on machine specifications and stand composition (Bradley, 1984; Winsauer et al., 1984). This could be the case in our findings, as machine capabilities have undoubtedly increased over time (Conrad et al., 2018). For example, the engine power (hp) exhibited by our grapple skidder was twice the maximum horsepower exhibited by similar equipment studies 20 years ago (Kluender et al., 1997). Cubbage et al. (1989) confirmed that, for mechanized grapple skidders, there is a distinct negative relationship between tree volume, tract size, tree density, and the average logging costs for an operation (Egan and Baumgras, 2003). While our study did not take stand composition into account, rubber-tired skidder production is largely influenced by haul size, which likely increased productivity compared to other studies because of the large grapple capacity utilized by our skidder (1.77 m^2) (Hiesl, 2013; Li et al., 2006; Kluender et al., 1997; Lanford and

Stokes, 1996). It should be noted that this is the largest grapple size for this make and model.

3.4.3 Knuckle-boom Loader and Truck Loading

For the knuckle-boom loader, loaded movement accounted for the majority of cycle times. Loaded movement predictably took the longest amount of time because of the added processing time for each tree, which includes delimbing and sorting. This also attributed to the decrease in accumulation size compared to loading already processed stems onto log trucks. The pull-through delimber used in the harvest was much more efficient when processing only 1-2 stems, but operators could often stack more stems onto log trucks once already processed. Overall, our productivity was high but still realistic compared to the findings from Conrad and Dahlen (2019), which processed 67.1 tonnes per productive machine hour.

The loading process for log truck observations showed similar trends but with some notable distinctions. There were three different operators that loaded trucks, and, for all machines, operator experience has shown to be one of the chief influential factors regarding harvesting efficiency, varying by as much as 55% between experience levels (Egan and Baumgras, 2003; Hiesl, 2013; Hiesl and Benjamin; 2013; Ovaskainen et al., 2004). Our study observed a 17% average increase in loading duration when truck operators loaded their trucks compared to the primary operator. However, the operator that experienced the longest loading duration also loaded the greatest number of stems per cycle, offsetting any productivity declines that would result from the duration increase. As a result, Operator 2 was the least productive of all of the operators, unsurprising considering he had the least amount of overall experience. Even with similar experience levels running machinery over one's career, repetition and an increase in operating hours would result in increases in efficiency. Still, productivity remained high, and the average loading time per truck was only about 20 minutes. Cass et al. (2009) found that operators could load log trucks in 60% less time when products were processed and sorted prior to arrival. This more than made up for an increase in loading time, which was only about 3.8 minutes longer, on average, for the truck operator.

3.4.4 Machine Utilization

Machine utilization has increased over time as logging operations have become more advanced and technologically adept (Bilici, 2021). On average, our machine utilization rates varied by as much as 29% between machines but were not inconsistent with the other studies (Brinker et al., 2002; Daniel et al., 2019; Holzleitner et al., 2011; Tepylo, 2017; Thompson, 2001). A majority of our findings were attributed to fluctuations in idling time as we observed very few mechanical delays. Correspondingly, each machine, throughout our observations, experienced 98-99% availability. Delays are significant components in determining productivity and should, therefore, continue to be incorporated into time and motion studies to improve forest operations. However, delays can be categorized differently based on the nature of the delay, and they can be unpredictable in the extent and timing of their manifestation (Spinelli and Visser, 2008).

This makes utilization trends among harvesting operations challenging to quantify given the relatively short period of observation (Spinelli and Visser, 2008).

Another major non-productive component to harvesting operations is idling time, which can be incorporated as a biproduct of productive activity or categorized as a non-productive work element, which we chose to do in our study based on the limitations set forth by a two-person operating crew. Many of the operator-caused delays would result in one or more machines idling for an extended period, causing significant bottlenecks in the operation, usually caused by delays in wood delivery via the skidder. For example, the knuckle-boom loader, was greatly influenced by the rate at which wood is acquired at the deck. When investigating stroke delimbers, Hiesl et al. (2015) found that an idle time of 40% or more is often unavoidable under average harvesting conditions when processing is dependent on additional equipment.

A major component of this study was the influence of a two-person crew running three machines compared to a conventional ground-based harvesting approach that would utilize three people running three machines with a truck operator. Typically, the feller-buncher operator would cut large swaths of timber, and while the skidder operator was accumulating the loads, the feller-buncher operator would switch to the knuckle-boom loader. While one truck operator in our study would intermittently run the knuckle-boom loader if the main operators were indisposed, the majority of observations consisted of just two operators running three machines. Hence, if all three machines were running, which they frequently were, there would be a considerable increase in the amount of idling time for at least one machine. Kelly and Germain (2016) noted that overall system productivity increased by 77-145% with the introduction of one to two additional loggers for one-person logging simulations, decreasing costs by as much as 28% and increasing machine utilization (Kelly and Germain, 2016). However, the addition of a third logger in a two-person crew did not always result in improved performance (Kelly and Germain, 2016).

We found a significant decline in the utilization rate of our knuckle-boom loader because machine operations were only observed when machines were on, and only one operator would run the feller-buncher. Therefore, if the feller-buncher was active, the skidder operator would frequently switch between loading and skidding depending on wood availability and trucking demands. This increased the idling time for the grapple skidder but was more influential on the loader. If the primary operator was operating the knuckle-boom loader, the feller-buncher was most likely not running, and the skidder operator was exclusively operating that machine. Thus, the feller-buncher experienced the least amount of idling time because if it was not in use, it was off, and the knuckle-boom loader experienced the greatest amount of idling time because it was always either in use, was potentially needed for loading, or was waiting on the skidder to arrive.

Consequently, we see an increasing trend in utilization from the knuckle-boom loader to the feller-buncher based on the operation dynamics. These results were seemingly consistent with Brinker et al. (2002) analysis of machine rates. Feller-bunchers typically exhibit high rates of utilization. For example, Tepylo (2017) found that the feller-buncher used in their study had a 77.4% utilization rate. Skidders and loaders often show similar utilization rates due to the reliance of loaders on skidder productivity.

Holzleitner et al. (2011) found that when investigating long-term machine data for 19 different skidders, the average annual machine utilization rate was about 70%, only slightly different from our findings, but consistent with Brinker et al. (2002) and Daniel et al. (2019), who estimated utilization between 65-70%. Additionally, Thompson (2001) noted that the utilization rate for rubber-tired grapple skidders was between 64.75-76.52% across four machines in Alabama. We predict that utilization rates for our knuckle-boom loader were primarily caused by increases in nonmechanical delays resulting from a two-person operation.

3.4.5 Machine Rates and Costs

Machine rates and production costs are major concerns for logging businesses when deciding when and how to harvest a site. Additionally, costs have continued to increase over time at an accelerated rate compared to financial returns. For example, the costs of logging equipment have increased at considerably higher rates than the prices received by loggers making entry into the industry significantly more difficult (Cubbage et al., 1988). Interactions between productivity, mechanization, and the size of a logging business create a positive feedback loop in which, if one of these factors is improved, the other determinants improve as well. For example, by increasing the number of employees, there will likely be an increase in productivity, allowing owners to reinvest in their machinery, ultimately increasing the scope and scale of the entire operation and subsequently increasing profitability and productive potential.

Machine costs for the feller-buncher, skidder, and loader amounted to $89.09, $79.79, and $65.64 per scheduled machine hour, which was within the range of values found in previous studies from the 1990s and early 2000s (Brinker et al., 2002; Kluender and Stokes, 1994). For 26 feller-bunchers, 26 skidders, and 19 loaders studied, Brinker et al. (2002) found that the average cost per scheduled machine hour ranged from $56.25-$133.71, $49.36-$95.54, and $26.55-$67.21 (using a conversion factor of $1.52 in 2021 for every $1 in 2002). Our findings indicate that the high production rate resulted in low production costs with the act of felling, skidding, and loading amounting to only $1.75, $5.05, and $2.64 per tonne, respectively. This is consistent with cost estimations calculated by Conrad and Dahlen (2019), who noted that felling, skidding, and processing/loading costs ranged from $1.77-$2.69, $3.65-$4.42, and $1.96-$3.22 per tonne, respectively. However, production cost trends remained consistent in which there were higher costs associated with skidding compared to loading and felling, respectively (Conrad and Dahlen, 2019; Kluender and Stokes, 1994). Our estimates indicate that this manner of restoration harvesting is profitable enough to conduct. For example, the cut and haul rate for eastern North Carolina in 2014-2015 was about $14.70/tonne and, according to a US logging index in the South during 2011, was about $13.78/tonne across the entire Southeast (Baker et al., 2014; Hahn, 2015; Mac Donagh et al., 2019). This leaves a reasonable profit margin for logging companies who choose to take on logging jobs in similar circumstances.

3.5 Conclusions

With large capital investments and increasing machine costs, logging is an extremely cost intensive industry that requires continual monitoring and evaluation across a multitude of harvesting conditions. Time and motion studies remain the most common and informative way of evaluating productivity, utilization, and costs for timber harvesting operations. With increased restoration initiatives, applying these studies in nontraditional settings is essential to broadening our understanding of logging capabilities and making informative management decisions under specified limitations. This study was applied to fill a contemporary gap in empirical knowledge on restoration harvesting cost and productivity when promoting longleaf pine habitat in South Carolina. We provided insight into systematic variations of logging crew dynamics by evaluating productivity and costs for a two-person crew running a conventional operation. However, this study was not without its limitations and, with a lack of current scientific literature on productivity and costs for harvesting operations in South Carolina, additional studies need to be applied to evaluate long-term trends and improve our understanding of the industry as a whole.

APPENDIX A

Table A.1: Mean bulk density and soil porosity for three treatments across four moisture groups for a longleaf restoration project in the Carolina wiregrass gap in Camden, SC.

Moisture Group	Treatment	Bulk Density (g cm^{-3})	Soil Porosity (%)
Mesic	Mastication	1.23±0.307	54%
	Conventional	1.28±0.281	52%
	Biomass	1.32±0.278	50%
Submesic	Mastication	1.44±0.176	46%
	Conventional	1.32±0.135	50%
	Biomass	1.39±0.150	48%
Subxeric	Mastication	1.46±0.126	45%
	Conventional	1.44±0.104	46%
	Biomass	1.46±0.131	45%
Xeric	Mastication	1.45±0.178	45%
	Conventional	1.52±0.100	43%
	Biomass	1.47±0.126	44%

Table A.2: Average biomass (kg ha^{-1}) for FWD and CWD between treatments and across moisture groups for pre- and post-treatment sampling in Camden, SC.

Moisture Group	Treatment	Pre-treatment		Post-treatment				
		FWD	CWD	Litter	1-HR	10-HR	100-HR	CWD
Mesic	Mastication	2664.1	119.2	3699.4	12162.3	16220.5	8440.2	5459.9
	Biomass	1676	114.5	3727.3	1779.5	8202.3	20494.1	9885.9
	Conventional	1526.2	50.1	4011.2	4711.3	8425.9	18361.8	10898.2
Submesic	Mastication	2086.9	63.8	4717.8	10713.9	19198.7	8505.6	7134.5
	Biomass	2966.5	134.7	4201.1	2350.3	10460.7	18542.7	3820.3
	Conventional	3037.1	324.9	2913.7	2788.2	7799.2	31166.1	12037.8
Subxeric	Mastication	4218.7	41.3	4691.3	7691.4	14322.4	12748	6413.1
	Biomass	2241.5	47.2	3654.7	3509.8	9415.4	19778.5	5233.6
	Conventional	2552.7	55.3	6853.3	3221.3	7266.3	8557.3	2776.1
Xeric	Mastication	2875.8	0	5477.2	7758.7	17576.4	4312	1629.6
	Biomass	3521	36	2406.3	4117.2	9971.8	9482.3	9257.3
	Conventional	4262	56	2207.4	3138.2	14882.4	36394	24321.7
Average		2802.4	168.3	4046.7	5328.5	11978.5	16398.5	8239

REFERENCES

Abrahamson, W. G. (1984). Post-fire recovery of Florida Lake Wales ridge vegetation. *American Journal of Botany*, *71*(1), 9-21.

Abrahamson, W. G., & Hartnett, D. C. (1990). Pine flatwoods and dry prairies. *Ecosystems of Florida*, 103-149.

Adebayo, A. B., Han, H. S., & Johnson, L. (2007). Productivity and cost of cut-to-length and whole-tree harvesting in a mixed-conifer stand. *Forest Products Journal*, *57*(6), 59.

Aguilar, F. X., Mirzaee, A., McGarvey, R. G., Shifley, S. R., & Burtraw, D. (2020). Expansion of US wood pellet industry points to positive trends but the need for continued monitoring. *Scientific reports*, *10*(1), 1-17.

Akay, A. E. (1998). Estimating machine rates and production for selected forest harvesting machines operating in the western United States and determining the most economical machine combinations under representative conditions in Turkey.

Akay, A. E., Erdas, O., & Sessions, J. (2004). Determining productivity of mechanized harvesting machines. *Journal of applied Sciences*, *4*(1), 100-105.

Andersson, B., & Evans, C. M. (1996). Harvesting overmature aspen stands in central Alberta. *Special Report Forest Engineering Research Institute of Canada*, (SR 112).

Aschenbach, T. A., Foster, B. L., & Imm, D. W. (2010). The initial phase of a longleaf pine-wiregrass savanna restoration: species establishment and community responses. *Restoration Ecology*, *18*(5), 762-771.

Aust, W. M., & Blinn, C. R. (2004). Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). *Water, Air and Soil Pollution: Focus*, *4*(1), 5-36.

Baker, S. A., Mei, B., Harris, T. G., & Greene, W. D. (2014). An index for logging cost changes across the US South. *Journal of Forestry*, *112*(3), 296-301.

Baker, Shawn, & Mendell, Brooks (2016). Compensation indices for logging and trucking occupations. *Wood Supply Research Institute: Valley Head, WV, USA*.

Barnett, J. P., Lauer, D. K., & Brissette, J. C. (1990). Regenerating longleaf pine with artificial methods.

Barrett, S. M., Bolding, M. C., & Munsell, J. F. (2017). Characteristics of logging businesses across Virginia's diverse physiographic regions. *Forests*, *8*(12), 468.

Benjamin, J. G., Seymour, R. S., Meacham, E., & Wilson, J. (2013). Impact of whole-tree and cut-to-length harvesting on postharvest condition and logging costs for early commercial thinning in Maine. *Northern Journal of Applied Forestry*, *30*(4), 149-155.

Bilici, E. (2021). Investigation of Feller-Buncher Performance Using Weibull Distribution. *Forests*, *12*(3), 284.

Bilskie, J., & Scientific, C. (2001). Soil water status: content and potential. *Campbell Scientific, Inc. App. Note: 2S-1 http://s. campbellsci. com/documents/ca/technical-papers/soilh20c. pdf (2 de Abril de 2014)*.

Blinn, C. R., O'Hara, T. J., Chura, D. T., & Russell, M. B. (2015). Minnesota's logging businesses: An assessment of the health and viability of the sector. *Forest Science*, *61*(2), 381-387.

Bolding, M. C., Kellogg, L. D., & Davis, C. T. (2009). Productivity and costs of an integrated mechanical forest fuel reduction operation in southwest Oregon. *Forest Products Journal*, *59*(3).

Boyer, W. D., & Peterson, D. W. (1983). Longleaf pine. *Burns, RM (Tech Comp.), Silvicultural systems for the major forest types of the United States. USDA Forest Service Agric. Handb*, *445*, 153-156.

Bradley, G. A. (Ed.). (1984). *Land use and forest resources in a changing environment: the urban/forest interface* (No. 49). University of Washington Press.

Brendemuehl, R. H. (1968). Research progress in the use of fertilizers to increase pine growth on the Florida Sandhills. *Forest Fertilization-Theory and Practice. TVA, Knoxville, Tenn*, 191-196.

Briedis, J. I., Wilson, J. S., Benjamin, J. G., & Wagner, R. G. (2011). Logging residue volumes and characteristics following integrated roundwood and energy-wood whole-tree harvesting in central Maine. *Northern Journal of Applied Forestry*, *28*(2), 66-71.

Brinker, R. W., Kinard, J., Rummer, R., & Lanford, B. (2002). Machine rates for selected forest harvesting machines. *In: Machine Rates for Selected Forest Harvesting Maines, 32 p.*

Brockway, D. G. (2005). *Restoration of longleaf pine ecosystems* (Vol. 83). USDA Forest Service, Southern Research Station.

Brockway, D. G., & Lewis, C. E. (1997). Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. *Forest Ecology and Management*, *96*(1-2), 167-183.

Brockway, D. G., & Outcalt, K. W. (1998). Gap-phase regeneration in longleaf pine wiregrass ecosystems. *Forest Ecology and Management*, *106*(2-3), 125-139.

Brockway, D. G., & Outcalt, K. W. (2000). Restoring longleaf pine wiregrass ecosystems: Hexazinone application enhances effects of prescribed fire. *Forest Ecology and Management*, *137*(1-3), 121-138.

Brockway, D. G., Outcalt, K. W., & Boyer, W. D. (2007). Longleaf pine regeneration ecology and methods. In *The longleaf pine ecosystem* (pp. 95-133). Springer, New York, NY.

Brockway, D. G., Outcalt, K. W., Estes, B. L., & Rummer, R. B. (2009). Vegetation response to midstorey mulching and prescribed burning for wildfire hazard reduction and longleaf pine (Pinus palustris Mill.) ecosystem restoration. *Forestry*, *82*(3), 299-314.

Brockway, D. G., Outcalt, K. W., Tomczak, D. J., & Johnson, E. E. (2005). Restoring longleaf pine forest ecosystems in the southern US. *In: Restoration of Boreal and Temperate Forests, Stanturf, John A.; Madsen, Palle; eds., Chapter thirty-two, CRC Press, Boca Raton, 2005, p. 501-519*.

Brown, J. K. (1974). Handbook for inventorying downed woody material. *Gen. Tech. Rep. INT-16. Ogden, UT: US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 24 p.*, *16*.

Brown, K., & Wherrett, A. (2021). *Bulk density – measurement*. Bulk Density - Measurement | Fact Sheets. Retrieved September 14, 2021, from https://soilquality.org.au/factsheets/bulk-density-measurement.

Bureau of Labor Statistics (BLS). (2017a). *Quarterly census of employment and wages. US Department of Labor*. Available online at http://www.bls. gov/cew/datatoc.htm; last accessed 2021.

Burns, R. M. (1983). *Silvicultural systems for the major forest types of the United States* (No. 445). US Department of Agriculture, Forest Service.

Burns, R. M., & Hebb, E. A. (1949). *Site preparation and reforestation of droughty, acid sands* (No. 426). US Department of Agriculture.

Camden, Sc Climate Camden, Sc Temperatures Camden, Sc Weather Averages. (n.d.). *Camden, SC climate & temperature.* Retrieved September 19, 2021, from https://www.camden.climatemps.com/.

Cartwright, J. M., & Costanza, J. (2016). *Ecosystem vulnerability to climate change in the southeastern United States* (No. 2016-3052). US Geological Survey.

Cass, R. D., Baker, S. A., & Greene, W. D. (2009). Cost and productivity impacts of product sorting on conventional ground-based timber harvesting operations. *Forest Products Journal*, *59*(11-12), 108-114.

Cavigelli, M. A., Deming, S. R., Probyn, L. K., & Harwood, R. R. (1998). Michigan field crop ecology. *Michigan State University Extension Bulletin E-2624, East Lansing*.

Christensen, N. L. (1977). Fire and soil-plant nutrient relations in a pine-wiregrass savanna on the coastal plain of North Carolina. *Oecologia*, *31*(1), 27-44.

Christensen, N. L. (1981). Fire regimes in southeastern ecosystems. *Fire regimes and ecosystem properties*, *112*, 136.

Christensen, N. L. (1988). Vegetation of the southeastern coastal plain. *North American terrestrial vegetation*, 317-363.

Condon, B., & Putz, F. E. (2007). Countering the broadleaf invasion: financial and carbon consequences of removing hardwoods during longleaf pine savanna restoration. *Restoration Ecology*, *15*(2), 296-303.

Conrad IV, J. L., & Dahlen, J. (2019). Productivity and cost of processors in whole-tree harvesting systems in southern pine stands. *Forest Science*, *65*(6), 767-775.

Conrad IV, J. L., Greene, W. D., & Hiesl, P. (2018). The evolution of logging businesses in Georgia 1987–2017 and South Carolina 2012–2017. *Forest Science*, *64*(6), 671-681.

Contreras, M., Freitas, R., Ribeiro, L., Stringer, J., & Clark, C. (2017). Multi-camera surveillance systems for time and motion studies of timber harvesting equipment. *Computers and Electronics in Agriculture*, *135*, 208-215.

Costanza, J. K., Abt, R. C., McKerrow, A. J., & Collazo, J. A. (2015). Linking state-and-transition simulation and timber supply models for forest biomass production scenarios. *AIMS Environmental Science*, *2*(2), 180-202.

Cox, Anne, Gordon, Doria, Slapcinsky, Jodi, & Seamon, Greg (2004). Understory restoration in longleaf pine Sandhills. *Natural Areas Journal*, *24*, 4-14.

Craft, C. B., & Chiang, C. (2002). Forms and amounts of soil nitrogen and phosphorus across a longleaf pine–depressional wetland landscape. *Soil Science Society of America Journal*, *66*(5), 1713-1721.

Craig, J.B., Lanham, J.D., Costa, Ralph, & Baldwin, Robert. (2010). Use of GIS to Assess the Demographic Isolation of Red-cockaded Woodpecker Groups in SC. *Department of Forestry and Natural Resources, Clemson University, SC, 29634 USA*.

Croker, T. C. (1976). *Regenerating longleaf pine naturally*. US Department of Agriculture, Forest Service, Southern Forest Experiment Station.

Crowder, L. B., Priddy, J. A., & Walters, J. R. (1998). Demographic isolation of red-cockaded woodpecker groups: a model analysis. *USFWS Project Final Report. Duke University Marine Laboratory, Beaufort, NC. and Virginia Polytechnic Institute and State University, Blacksburg VA*.

Cubbage, F. W., Greene, W. D., & Lyon, J. P. (1989). Tree size and species, stand volume, and tract size: effects on southern harvesting costs. *Southern Journal of Applied Forestry*, *13*(3), 145-152.

Cubbage, F. W., Stokes, B. J., & Granskog, J. E. (1988). Trends in southern forest harvesting equipment and logging costs. *Forest Products Journal Vol. 32 (2): 6-10*.

Dagley, C. M., Harrington, T. B., & Edwards, M. B. (2002). Understory restoration in longleaf pine plantations: Overstory effects of competition and needlefall. In *Eleventh Biennial Southern Silvicultural Research Conference* (pp. 487-489).

Daniel, M. J., Gallagher, T., Mitchell, D., McDonald, T., & Via, B. (2019). Productivity and cost estimates for incorporating tracked processors into conventional loblolly pine harvesting regimes in the Southeastern United States. *International Journal of Forest Engineering*, *30*(2), 155-162.

Davis, M. (1990). Old-growth in the East: a preliminary overview. *Earth First*.

de Vries, P. G. (1973). *A general theory on line intersect sampling: with application to logging residue inventory* (No. 73-11). Veeman.

Deere and Company. (n.d.). *L-II Series Skidders Brochure*. Retrieved 2021, from https://www.deere.com/assets/pdfs/common/products/skidders/skidder-L-SeriesII-DKDLSKDR.pdf.

Donovan, L. A., West, J. B., & McLeod, K. W. (2000). Quercus species differ in water and nutrient characteristics in a resource-limited fall-line sandhill habitat. *Tree Physiology*, *20*(14), 929-936.

Egan, A. F., & Baumgras, J. E. (2003). Ground skidding and harvested stand attributes in Appalachian hardwood stands in West Virginia. *Forest Products Journal. 53 (9): 59-63.*

Engstrom, R. T. (1993, May). Characteristic mammals and birds of longleaf pine forests. In *Proceedings of the Tall Timbers Fire Ecology Conference* (Vol. 18, pp. 127-138).

Esri Inc. (2020). *ArcGIS Pro* (Version 2.5). Esri Inc. https://www.esri.com/en-us/arcgis/products/arcgis-pro/overview

Evans, A. M., Perschel, R. T., & Kittler, B. A. (2013). Overview of forest biomass harvesting guidelines. *Journal of sustainable forestry*, *32*(1-2), 89-107.

Faust, W. Z. (1976). A vegetation analysis of the Georgia fall-line Sandhills. *Rhodora*, *78*(815), 525-531.

Fernow, B. E. (1893). *Effect of Turpentine Gathering on the Timber of Longleaf Pine* (No. 9). US Government Printing Office.

Fill, J. M., Welch, S. M., Waldron, J. L., & Mousseau, T. A. (2012). The reproductive response of an endemic bunchgrass indicates historical timing of a keystone process. *Ecosphere*, *3*(7), 1-12.

Forest Land Management Inc. (2021). Personal Interview. [Personal Interview].

Fraver, S., Ringvall, A., & Jonsson, B. G. (2007). Refining volume estimates of down woody debris. *Canadian Journal of Forest Research*, *37*(3), 627-633.

Frost, C. C. (1993, May). Four centuries of changing landscape patterns in the longleaf pine ecosystem. In *Proceedings of the Tall Timbers fire ecology conference* (Vol. 18, pp. 17-43).

Germain, R., Regula, J., Bick, S., & Zhang, L. (2019). Factors impacting logging costs: A case study in the Northeast, US. *The Forestry Chronicle*, *95*(1), 16-23.

Gordon, D. R., & Rice, K. J. (1998). Patterns of differentiation in wiregrass (Aristida beyrichiana): implications for restoration efforts. *Restoration Ecology*, *6*(2), 166-174.

Grattan, S. R., & Oster, J. (2003). Field Use of Tensiometers. *Field use of Tensiometers :: Department of Land, air and water resources - UC Davis.* Retrieved from: https://lawr.ucdavis.edu/cooperative-extension/irrigation/drought-tips/field-use-tensiometers.

Greene, W. D., Lanford, B. L., & Mykytka, E. F. (1987). Stand and operating effects on feller-buncher productivity in second thinnings of southern pine. *For. Prod. J.;(United States)*, *37*(3).

Greene, W. D., Marchman, S. C., & Baker, S. A. (2013, July). Changes in logging firm demographics and logging capacity in the US South. In *Proceedings of the 36th Annual Council on Forest Engineering Meeting* (p. 7).

Griffith, G. E., Omernik, J. M., Comstock, J. A., Schafale, M. P., McNab, W. H., Lenat, D. R., & MacPherson, T. F. (2002). Ecoregions of South Carolina US Environmental Protection Agency. *Corvallis, OR (map scale 1: 1,500,000)*.

Hahn, G. (2015). *Wood procurement and harvesting trends in North Carolina*.

Hardin, E. D., & White, D. L. (1989). Rare vascular plant taxa associated with wiregrass(Aristida stricta) in the southeastern United States. *Natural Areas Journal*, *9*(4), 234-245.

Harper, KM. (1913). Economic botany of Alabama. Part 1. Geological Survey of Alabama, Monograph 8. University, Alabama. 233 p.

Harrington, T. B. (2006). Restoring the Savanna to the Savannah River Site *(No. na). USDA Forest Service, Savannah River, New Ellenton, SC.*

Harrington, T. B., & Edwards, M. B. (1999). Understory vegetation, resource availability, and litterfall responses to pine thinning and woody vegetation control in longleaf pine plantations. *Canadian Journal of Forest Research*, *29*(7), 1055-1064.

Harrington, T. B., Dagley, C. M., & Edwards, M. B. (2003). Above-and belowground competition from longleaf pine plantations limits performance of reintroduced herbaceous species. *Forest Science*, *49*(5), 681-695.

Harrington, T. B., Miller, K. V., & Park, N. (2013). *Restoring a disappearing ecosystem: the Longleaf Pine Savanna* (No. 13-04-R). USDA Forest Service-Savannah River, New Ellenton, SC.

Harrington, T. B., Minogue, P. J., Sutton, R. A., Peter, D. H., & Blake, J. I. (2021). Initial plant community responses to hardwood control treatments in restoration of remnant longleaf pine (Pinus palustris) woodlands. *Forest Ecology and Management*, *491*, 119197.

Hattenbach, M. J., Gordon, D. R., Seamon, G. S., & Studenmund, R. G. (1998). Development of directseeding techniques to restore native groundcover in a sandhill ecosystem. In *Proceedings of the longleaf pine ecosystem restoration symposium: ecological restoration and regional conservation strategies. Longleaf Alliance Report* (No. 3, pp. 64-70).

Heinrich, R. (1992). Cost Control in Forest Harvesting and Road Construction. *Food and Agriculture Organization of the United Nations Publications, Rome, Italy*.

Hiers, J. K., O'Brien, J. J., Will, R. E., & Mitchell, R. J. (2007). Forest floor depth mediates understory vigor in xeric Pinus palustris ecosystems. *Ecological Applications*, *17*(3), 806-814.

Hiesl, P. (2013). Productivity standards for whole-tree and cut-to-length harvesting systems in Maine.

Hiesl, P. (2020). A Brief Summary Of The 2017 South Carolina Logging Business Survey.

Hiesl, P., & Benjamin, J. G. (2013). A multi-stem feller-buncher cycle-time model for partial harvest of small-diameter wood stands. *International journal of forest engineering*, *24*(2), 101-108.

Hiesl, P., & Benjamin, J. G. (2013). Applicability of international harvesting equipment productivity studies in Maine, USA: A literature review. *Forests*, *4*(4), 898-921.

Holzleitner, F., Stampfer, K., & Visser, R. (2011). Utilization rates and cost factors in timber harvesting based on long-term machine data. *Croatian Journal of Forest Engineering: Journal for Theory and Application of Forestry Engineering*, *32*(2), 501-508.

Hood, S., & Wu, R. (2006). Estimating fuel bed loadings in masticated areas. In *In: Andrews, Patricia L.; Butler, Bret W., comps. 2006. Fuels Management-How to Measure Success: Conference Proceedings. 28-30 March 2006; Portland, OR. Proceedings RMRS-P-41. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. 333-340* (Vol. 41).

Hu, H., Wang, G. G., Walker, J. L., & Knapp, B. O. (2012). Silvicultural treatments for converting loblolly pine to longleaf pine dominance: Effects on resource availability and their relationships with planted longleaf pine seedlings. *Forest ecology and management*, *282*, 115-123.

Huffman, J. M. (2006). *Historical fire regimes in southeastern pine savannas*. Louisiana State University and Agricultural & Mechanical College.

Hunt, N., & Gilkes, R. (1992). Farm Monitoring Handbook–A practical down-toearth manual for farmers and other land users. *University of Western Australia: Nedlands, WA, and Land Management Society: Como, WA*.

Husch, B., T.W. Beers, and J.A. Kershaw Jr. (2003). Forest mensuration. John Wiley & In *Proceedings of the Tall Timbers Fire Ecology Conference* (Vol. 18, pp. 127-138).

Jain, T., Sikkink, P., Keefe, R., & Byrne, J. (2018). To masticate or not: useful tips for treating forest, woodland, and shrubland vegetation. *Gen. Tech. Rep. RMRS-GTR-381. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 55 p.*, *381*.

Janowiak, M. K., & Webster, C. R. (2010). Promoting ecological sustainability in woody biomass harvesting. *Journal of forestry*, *108*(1), 16-23.

Jenkins, R. A., & McMillan, P. D. (2009). Vascular flora of Sandhill Research and Education Center, Richland County, South Carolina. *castanea*, *74*(2), 168-180.

Jose, S., Jokela, E. J., & Miller, D. L. (2007). The longleaf pine ecosystem. In *The longleaf pine ecosystem* (pp. 3-8). Springer, New York, NY.

Jose, S., Merritt, S., & Ramsey, C. L. (2003). Growth, nutrition, photosynthesis and transpiration responses of longleaf pine seedlings to light, water and nitrogen. *Forest Ecology and Management*, *180*(1-3), 335-344.

Jose, S., Ranasinghe, S., & Ramsey, C. L. (2010). Longleaf pine (Pinus palustris P. Mill.) restoration using herbicides: Overstory and understory vegetation responses on a coastal plain flatwoods site in Florida, USA. *Restoration Ecology*, *18*(2), 244-251.

Kaeser, M. J., & Kirkman, L. K. (2012). Seed longevity of 12 native herbaceous species in a fire-maintained pine savanna after 8 years of burial. *Forest Ecology and Management*, *281*, 68-74.

Kaliraj, S., Chandrasekar, N., & Magesh, N. S. (2015). Evaluation of multiple environmental factors for site-specific groundwater recharge structures in the Vaigai River upper basin, Tamil Nadu, India, using GIS-based weighted overlay analysis. *Environmental earth sciences*, *74*(5), 4355-4380.

Kane, J. M., Varner, J. M., & Knapp, E. E. (2009). Novel fuelbed characteristics associated with mechanical mastication treatments in northern California and south-western Oregon, USA. *International Journal of Wildland Fire*, *18*(6), 686-697.

Kelly, M. C., & Germain, R. H. (2016). Is it efficient to single-handedly run a multi-machine harvesting operation? A case study from the Northeast United States. *International Journal of Forest Engineering*, *27*(3), 140-150.

Klimas, K. B. (2020). *Prescribed Fire Effects on Water Quality Variables in the Southern Appalachian Region* (Doctoral dissertation, Clemson University).

Kluender, R. A., & Stokes, B. J. (1994). Productivity and costs of three harvesting methods. *Southern Journal of Applied Forestry*, *18*(4), 168-174.

Kluender, R., Lortz, D., McCoy, W., Stokes, B., & Klepac, J. (1997). Productivity of rubber-tired skidders in southern pine forests. *Forest Products Journal. 47 (11/12): 53-58.*

Knapp, B. O., Walker, J. L., Wang, G. G., Hu, H., & Addington, R. N. (2014). Effects of overstory retention, herbicides, and fertilization on sub-canopy vegetation structure and functional group composition in loblolly pine forests restored to longleaf pine. *Forest ecology and management*, *320*, 149-160.

Knapp, B. O., Wang, G. G., Walker, J. L., & Cohen, S. (2006). Effects of site preparation treatments on early growth and survival of planted longleaf pine (Pinus palustris Mill.) seedlings in North Carolina. *Forest Ecology and Management*, *226*(1-3), 122-128.

Kobziar, L. N., Long, A. J., Zipperer, W. C., & Kreye, J. K. (2013). Characterization of Masticated Fuelbeds and Fuel Treatment Effectiveness in Southeastern US Pine Ecosystems.

Komarek, E. V. (1967, November). The nature of lightning fires. In *Tall Timbers Fire Ecology Conference* (Vol. 7, pp. 5-41).

Koŝir, B., Magagnotti, N., & Spinelli, R. (2015). The role of work studies in forest engineering: status and perspectives. *International journal of forest engineering*, *26*(3), 160-170.

Kreye, J. K., Kobziar, L. N., & Camp, J. M. (2014). Immediate and short-term response of understory fuels following mechanical mastication in a pine flatwoods site of Florida, USA. *Forest Ecology and Management*, *313*, 340-354.

Kreye, J. K., Varner, J. M., & Knapp, E. E. (2011). Effects of particle fracturing and moisture content on fire behaviour in masticated fuelbeds burned in a laboratory. *International Journal of Wildland Fire*, *20*(2), 308-317.

Kreye, J. K., Varner, J. M., & Knapp, E. E. (2012). Moisture desorption in mechanically masticated fuels: effects of particle fracturing and fuelbed compaction. *International Journal of Wildland Fire*, *21*(7), 894-904.

Kreye, J. K., Varner, J. M., Dugaw, C. J., Cao, J., Szecsei, J., & Engber, E. A. (2013). Pine cones facilitate ignition of forest floor duff. *Canadian Journal of Forest Research*, *43*(5), 512-516.

Kreye, J., & Varner, J. M. (2007). Moisture dynamics in masticated fuelbeds: A preliminary analysis. In *In: Butler, Bret W.; Cook, Wayne, comps. The fire environment--innovations, management, and policy; conference proceedings. 26-30 March 2007; Destin, FL. Proceedings RMRS-P-46CD. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. CD-ROM. p. 173-186* (Vol. 46).

Kulak, D., Stańczykiewicz, A., & Szewczyk, G. (2017). Productivity and time consumption of timber extraction with a grapple skidder in selected pine stands. *Croatian Journal of Forest Engineering: Journal for Theory and Application of Forestry Engineering*, *38*(1), 55-63.

Landers, J. L., Van Lear, D. H., & Boyer, W. D. (1995). The longleaf pine forests of the southeast: requiem or renaissance?. *Journal of Forestry 93 (11): 39-44.*

Lanford, B. L., & Sirois, D. L. (1983). Drive-to-tree, rubber-tired feller buncher production studies [Statistics on mechanized harvesting of southern pine stands in Georgia, Florida, and Mississippi]. *Forest Service General Technical Report SO (USA). no. 45.*

Lanford, B., & Stokes, B. J. (1996). Comparison of two thinning systems. Part 2. Productivity and costs. *Forest Products Journal. 46 (11/12): 47-53.*

Lavoie, M., Mack, M. C., Hiers, J. K., Pokswinski, S., Barnett, A., & Provencher, L. (2014). Effects of restoration techniques on soil carbon and nitrogen dynamics in Florida longleaf pine (Pinus palustris) sandhill forests. *Forests*, *5*(3), 498-517.

Lavoie, M., Starr, G., Mack, M. C., Martin, T. A., & Gholz, H. L. (2010). Effects of a prescribed fire on understory vegetation, carbon pools, and soil nutrients in a longleaf pine-slash pine forest in Florida. *Natural Areas Journal*, *30*(1), 82-94.

Lee, M. T., Peet, R. K., Roberts, S. D., & Wentworth, T. R. (2006). CVS-EEP protocol for recording vegetation. *Carolina Vegetation Survey. Retrieved August*, *17*, 2008.

Leon, B. H., & Benjamin, J. G. (2012). A survey of business attributes, harvest capacity and equipment infrastructure of logging businesses in the northern forest. *School of Forest Resources, University of Maine, Orono*.

Li, Y., Wang, J., Miller, G., & McNeel, J. (2006). Production economics of harvesting small-diameter hardwood stands in central Appalachia. *Forest Products Journal 56 (3): 81-86*, *56*(3).

Linn, D. M., & Doran, J. W. (1984). Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. *Soil Science Society of America Journal*, *48*(6), 1267-1272.

Little Jr, E. L. (1971). Atlas of United States trees. Volume 1. Conifers and important hardwoods. Miscellaneous publication 1146. *US Department of Agriculture, Forest Service, Washington, DC*.

Lockaby, B. G., & Vidrine, C. G. (1984). Effect of logging equipment traffic on soil density and growth and survival of young loblolly pine. *Southern journal of applied forestry*, *8*(2), 109-112.

Longleaf Alliance. (1998). Proceedings of the Longleaf Pine Ecosystem Restoration Symposium: Presented at the Society for Ecological Restoration 9th Annual International Conference. *Ecological Restoration and Regional Conservation Strategies": November 12-15, 1997, Fort Lauderdale, Florida USA*. No. 3.

Loveridge, E. W. (1944). The fire suppression policy of the US Forest Service. *Journal of Forestry*, *42*(8), 549-554.

Mac Donagh, P., Roll, J., Hahn, G., & Cubbage, F. (2019). Timber Harvesting Production, Costs, Innovation, and Capacity in the Southern Cone and the US South. In *Timber Buildings and Sustainability*. IntechOpen.

Mahler, Robert. (2004). Nutrients Plants Require for Growth. *University of Idaho College of Agricultural and Life Sciences. https://www.extension.uidaho.edu/publishing/pdf/CIS/CIS1124.pdf.* (2021).

McCaskill, G. L., Jose, S., Chauhan, A., & Ogram, A. V. (2018). Soil nitrogen dynamics as an indicator for longleaf pine restoration. *Restoration Ecology*, *26*(2), 264-274.

McGee, C. E., & Scott, H. R. (1965). Planting longleaf pine in the Carolina Sandhills. *Tree Planters' Notes No. 70*, *1965*, 4-5.

McKenzie, N., Coughlan, K., & Cresswell, H. (2002). *Soil physical measurement and interpretation for land evaluation* (Vol. 5). Csiro Publishing.

Microsoft Corporation. (2018). *Microsoft Excel*. Retrieved from https://office.microsoft.com/excel

Mohr, C. T., & Roth, F. (1896). The timber pines of the southern United States. Bulletin 13.

Moldenhauer, M. C., & Bolding, M. C. (2009). Parcelization of South Carolina's private forestland: Loggers' reactions to a growing threat. *Forest Products Journal*, *59*(6), 37.

Moseley, J. (2019). *Site suitability analysis of endangered species habitat areas for pesticide risk assessment: A case study of the red-cockaded woodpecker in the coastal plain counties of North Carolina*.

Moser, W. K., & Yu, C. K. (2003). Effects of overstory structure and fire regime upon diversity and abundance of selected understory species in longleaf pine (Pinus palustris Mill.) forests in southeastern Georgia. *J. Forest Sci*, *49*, 395-402.

Moser, W. K., Jackson, S. M., Podrazsky, V., & Larsen, D. R. (2002). Examination of stand structure on quail plantations in the Red Hills region of Georgia and Florida managed by the Stoddard–Neel system: an example for forest managers. *Forestry*, *75*(4), 443-449.

Mousavi, R., Nikoye Seyahkal, M., & Naghdi, R. (2013). Comparison of timber skidding using two ground-based skidding systems: grapple skidding vs. cable skidding. *International Journal of Forest, Soil and Erosion (IJFSE)*, *3*(3), 79-86.

Mulligan, M. K., Kirkman, L. K., & Mitchell, R. J. (2002). Aristida beyrichiana (wiregrass) establishment and recruitment: implications for restoration. *Restoration Ecology*, *10*(1), 68-76.

Murphy, C. H. (2016). *Sandhills*. South Carolina Encyclopedia. Retrieved from: https://www.scencyclopedia.org/sce/entries/Sandhills/. (2021).

Myers, R. L. (1990). Scrub and high pine. *Ecosystems of Florida*, *765*, 150-193.

Neel, L., Sutter, P. S., & Way, A. G. (2011). *The art of managing longleaf: a personal history of the Stoddard-Neel approach*. University of Georgia Press. Netherlands. 23 p.

North, B. W., & Pienaar, E. F. (2021). Continued obstacles to wood-based biomass production in the southeastern United States. *GCB Bioenergy*, *13*(7), 1043-1053.

Noss, R. F. (1989). Longleaf pine and wiregrass: keystone components of an endangered ecosystem. *Natural Areas Journal*, *9*(4), 211-213.

Ojha, S. K., Dimov, L. D., & Tadesse, W. (2021). Growth, proportion, and distribution pattern of longleaf pine across southeastern forests and disturbance types: A change assessment for the period 1997-2018. *Plos one*, *16*(1), e0245218.

Oswalt, C. M., Cooper, J. A., Brockway, D. G., Brooks, H. W., Walker, J. L., Connor, K. F., & Conner, R. C. (2012). History and current condition of longleaf pine in the southern United States. *General Technical Report-Southern Research Station, USDA Forest Service*.

Oswalt, S. N., Smith, W. B., Miles, P. D., & Pugh, S. A. (2014). Forest Resources of the United States, 2012: a technical document supporting the Forest Service 2010 update of the RPA Assessment. *Gen. Tech. Rep. WO-91. Washington, DC: US Department of Agriculture, Forest Service, Washington Office. 218 p.*, *91*.

Outcalt, K. W. (1992). Factors affecting wiregrass (Aristida stricta Michx.) cover on uncut and site prepared Sandhills areas in Central Florida. *Ecological Engineering*, *1*(3), 245-251.

Outcalt, K. W., & Lewis, C. E. (1990). Response of wiregrass (Aristida stricta) to mechanical site preparation. In *Wiregrass biology and management symposium proceedings. KBN Engineering and Applied Sciences, Gainesville, FL*.

Outcalt, K. W., & Sheffield, R. M. (1996). *Longleaf pine forest: Trends and current conditions. Forest Service resource bulletin* (No. PB-97-122022/XAB; FSRB-SRS-9). Forest Service, Asheville, NC (United States). Southeastern Forest Experiment Station.

Outcalt, K. W., Williams, M. E., & Onokpise, O. (1999). Restoring Aristida stricta to Pinus palustris ecosystems on the Atlantic Coastal Plain, USA. *Restoration Ecology*, *7*(3), 262-270.

Outland III, R. B. (2004). *Tapping the pines: The naval stores industry in the American South*. LSU Press.

Ovaskainen, H., Uusitalo, J., & Väätäinen, K. (2004). Characteristics and significance of a harvester operators' working technique in thinnings. *International journal of forest engineering*, *15*(2), 67-77.

Overby, S. T., & Gottfried, G. J. (2017). Microbial and nitrogen pool response to fuel treatments in Pinyon-Juniper woodlands of the southwestern USA. *Forest Ecology and Management*, *406*, 138-146.

Overby, S., & Gottfries, G. (2009). Does mastication residue alter soil nitrogen dynamics in woodlands of Southwest Colorado?.

Peet, R. K. (1993). A taxonomic study of Aristida stricta and A. beyrichiana. *Rhodora*, 25-37.

Peet, R. K. (2007). Ecological classification of longleaf pine woodlands. In *The longleaf pine ecosystem* (pp. 51-93). Springer, New York, NY.

Platt, W.J., J.S. Glitzenstein, and D.R. Streng. (1991). Evaluating pyrogenicity and its effects on vegetation in longleaf pine savannas, in: Proceedings of the 17th Tall Timbers Fire Ecology Conference: High Intensity Fire in Wildlands: Management Challenges and Options, Tall Timbers Research Station, Tallahassee, Florida.

Provencher, L., Herring, B. J., Gordon, D. R., Rodgers, H. L., Galley, K. E., Tanner, G. W., ... & Brennan, L. A. (2001). Effects of hardwood reduction techniques on longleaf pine sandhill vegetation in northwest Florida. *Restoration Ecology*, *9*(1), 13-27.

Qin, S., Li, S., Kang, S., Du, T., Tong, L., & Ding, R. (2016). Can the drip irrigation under film mulch reduce crop evapotranspiration and save water under the sufficient irrigation condition?. *Agricultural Water Management*, *177*, 128-137.

Quarterman, E., & Keever, C. (1962). Southern mixed hardwood forest: climax in the southeastern coastal plain, USA. *Ecological Monographs*, *32*(2), 167-185.

R Core Team (2020). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Raison, R. J., Khanna, P. K., & Jacobsen, K. L. (2009). Effects of Fire on Forest Nutrient Cycles Joan Romanya and Isabel Serrasolses. In *Fire effects on soils and restoration strategies* (pp. 241-272). CRC Press.

Reinhart, K. O., & Menges, E. S. (2004). Effects of re-introducing fire to a central Florida sandhill community. *Applied Vegetation Science*, *7*(1), 141-150.

Reisinger, T. W., Simmons, G. L., & Pope, P. E. (1988). The impact of timber harvesting on soil properties and seedling growth in the south. *Southern Journal of Applied Forestry*, *12*(1), 58-67.

Rhoades, C. C., Battaglia, M. A., Rocca, M. E., & Ryan, M. G. (2012). Short-and medium-term effects of fuel reduction mulch treatments on soil nitrogen availability in Colorado conifer forests. *Forest Ecology and Management*, *276*, 231-238.

Riad, P. H., Billib, M., Hassan, A. A., Salam, M. A., & El Din, M. N. (2011). Application of the overlay weighted model and Boolean logic to determine the best locations for artificial recharge of groundwater. *Journal of Urban and Environmental Engineering*, *5*(2), 57-66.

Richardson, J., Björheden, R., Hakkila, P., Lowe, A. T., & Smith, C. T. (Eds.). (2006). *Bioenergy from sustainable forestry: guiding principles and practice* (Vol. 71). Springer Science & Business Media.

Rodgers, H. L., & Provencher, L. (1999). Analysis of longleaf pine sandhill vegetation in northwest Florida. *Castanea*, 138-162.

Rose, A. K. (2015). Forests of South Carolina, 2013. *Resource Update FS-36. Asheville, NC: US Department of Agriculture Forest Service, Southern Research Station. 4 p.*, *36*, 1-4.

Roslee, R., Mickey, A. C., Simon, N., & Norhisham, M. N. (2017). Landslide susceptibility analysis (LSA) using weighted overlay method (WOM) along the Genting Sempah to Bentong Highway, Pahang. *Malaysian Journal of Geosciences (MJG)*, *1*(2), 13-19.

RStudio Team (2020). RStudio: Integrated Development Environment for R. *RStudio, PBC, Boston, MA* URL http://www.rstudio.com/.

Sahrawat, K. L. (2008). Factors affecting nitrification in soils. *Communications in Soil Science and Plant Analysis*, *39*(9-10), 1436-1446.

Santos de Freitas, R. L. (2019). Multi-Camera Surveillance System for Time and Motion Studies of Timber Harvesting Operations. (Doctoral Dissertation).

Saucier, J. R., Phillips, D. R., & Williams, J. G. (1981). Green weight, volume, board-foot, and cord tables for the major southern pine species. *Georgia forest research paper (USA)*.

SCFC SC Forest Products Mill Directory. (2021). *Forest products Mill directory for South Carolina*. Retrieved from https://www.state.sc.us/forest/refind.htm.

Seamon, P. A., Myers, R. L., Robbins, L. E., & Seamon, G. S. (1989). Wiregrass reproduction and community restoration. *Natural Areas Journal*, *9*(4), 264-265.

Series, B. (n.d.) MLRA 137: Carolina and Georgia Sand Hills.

Sharma, A., Jose, S., Bohn, K. K., & Andreu, M. G. (2012). Effects of reproduction methods and overstory species composition on understory light availability in longleaf pine–slash pine ecosystems. *Forest Ecology and Management*, *284*, 23–33.

Shaw, J. D., & Long, J. N. (2007). A density management diagram for longleaf pine stands with application to red-cockaded woodpecker habitat. *Southern Journal of Applied Forestry*, *31*(1), 28-38.

Sherrard, T. H. (1903). *A working plan for forest lands in Hampton and Beaufort counties, South Carolina* (No. 43). US Government Printing Office. Sons, Inc., Hoboken, NJ. 443 p.

South Carolina Forestry Commission. (2014). South Carolina Forest Industry Facts. *South Carolina's Forest Resource Assessment and Strategy.*

South Carolina Wildlife Federation. (n.d.). *Longleaf pine program*. Retrieved 2021, from http://www.scwf.org/longleaf.

Spies, T. A., & Cline, S. P. (1988). Coarse woody debris in forests and plantations of coastal Oregon. *From the forest to the sea: a story of fallen trees. Gen. Tech. Rep. PNW-GTR-229. Portland, OR: Pacific Northwest Research Station, Forest Service, US Department of Agriculture*, 5-23.

Spinelli, R., & Visser, R. (2008). Analyzing and estimating delays in harvester operations. *International journal of forest engineering*, *19*(1), 36-41.

Spinelli, R., Hartsough, B. R., Owende, P. M., & Ward, S. M. (2002). Productivity and cost of mechanized whole-tree harvesting of fast-growing eucalypt stands. *International Journal of Forest Engineering*, *13*(2), 49-60.

Spinelli, R., Magagnotti, N., & Nati, C. (2010). Harvesting vineyard pruning residues for energy use. *Biosystems Engineering*, *105*(3), 316-322.

Stambaugh, M. C., Guyette, R. P., & Marschall, J. M. (2011). Longleaf pine (Pinus palustris Mill.) fire scars reveal new details of a frequent fire regime. *Journal of Vegetation Science*, *22*(6), 1094-1104.

Stampfer, K., & Lexer, M. J. (2001). *Multicriteria evaluation of thinning operations on steep terrain*. na.

Stewart, W.M. (n.d.). Sulfur--The 4th Major Nutrient. *IPNI Plant Nutrition TODAY.* Spring, 2010, No. 7,. Retrieved From: www.ipni.net/pnt

Stout, I. J., & Marion, W. R. (1993). Pine flatwoods and xeric pine forests of the southern (lower) coastal plain. *Biodiversity of the southeastern United States: lowland terrestrial communities*, 373-446.

Streng, D. R., Glitzenstein, J. S., & Platt, W. J. (1993, May). Evaluating effects of season of burn in longleaf pine forests: a critical literature review and some results from an ongoing long-term study. In *Proceedings of the tall timbers fire ecology conference* (Vol. 18, pp. 227-263). Tall Timbers Research Station Tallahassee, FL.

Szewczyk, G., & Sowa, J. M. (2017). The accuracy of measurements in a time study of harvester operations. *New Zealand Journal of Forestry Science*, *47*(1), 1-10.

Tanner, G. W., Wood, J. M., Kalmbacher, R. S., & Martin, F. G. (1988). Mechanical shrub control on flatwoods range in south Florida. *Rangeland Ecology & Management/Journal of Range Management Archives*, *41*(3), 245-248.

Taylor, F. W. (1895). A piece-rate system, being a step toward partial solution of the labor problem. *Transactions of the American Society of Mechanical Engineers*, *16*, 856-903.

Tepylo, S. A. (2017). *Monitoring productivity and utilization of a feller-buncher using FPDat system* (Doctoral dissertation).

The Nature Conservancy. (2016). *Landscape-Scale Spatial Analysis of Longleaf Pine Ecosystem Resilience in North Carolina*. Retrieved August 13, 2021, from https://www.natureserve.org/sites/default/files/presentations/2018-bio-conference/McIver_2018SEBCF.pdf.

The South Carolina State Climatology Office. (n.d.). South Carolina State climatology office. Retrieved September 19, 2021, from https://www.dnr.sc.gov/climate/sco/ClimateData/countyData/county_kershaw.phd

Thompson, J. D. (2001). Calculating utilization rates for rubber tired grapple skidders in the southern United States. In *Proceedings of the 24th Annual COFE Meeting, Showshoe, West Virginia, July 15-19, 2001, p. 29-31*.

Thompson, R. S., Anderson, K. H., & Bartlein, P. J. (1999). Digital representations of tree species range maps from "Atlas of United States trees" by Elbert L. Little, Jr.(and other publications). *Atlas of relations between climatic parameters and distributions of important trees and shrubs in North America*.

Tigercat International Inc. (2019). 234B/T234B Loader Brochure. Retrieved 2021, from https://www.tigercat.com/wp-content/uploads/2014/12/234B-T234B-1.2-0919-hi-res.pdf.

Tigercat International Inc. (2020). Drive-to-Tree Feller Bunchers Brochure. Retrieved 2021, from https://www.tigercat.com/wp-content/uploads/2014/12/WFB-1.3-CV731-EN-1020-hi-res.pdf.

TimberMart-South. (2020). 4th Quarter 2020 Logging Rates Report. Retrieved from file:///Users/jacobmurray/Downloads/4Q2020%20Logging%20Rate%20Report.pdf.

TimberMart-South. (2020). The Journal of Southern Timber Prices. Retrieved from file:///Users/jacobmurray/Downloads/4Q2020%20Logging%20Rate%20Report.pdf.

Tiwari, A., & Ajmera, S. (2021). Land Suitability Assessment for Agriculture Using Analytical Hierarchy Process and Weighted Overlay Analysis in ArcGIS ModelBuilder. In *Recent Trends in Civil Engineering* (pp. 735-762). Springer, Singapore.

Tuller, M., Or, D., & Hillel, D. (2004). Retention of water in soil and the soil water characteristic curve. *Encyclopedia of Soils in the Environment*, *4*, 278-289.

U.S. Bureau of Labor Statistics. (2021). *Occupational Employment and Wage Statistics*. U.S. Bureau of Labor Statistics. Retrieved 2021, from https://data.bls.gov/oes/#/geoOcc/Multiple%20occupations%20for%20one%20geographical%20area.

U.S. Census Off. (1902). Twelfth census of the U.S. Vol. 5. Agriculture. Part 1. U.S. Government Printing Office, Washington, DC 767 p.

U.S. Census Off. 1841, 1853, 1866, 1872, 1883, 1895. Censuses of Manufactures. U.S. Government Printing Office, Washington.

U.S. Fish and Wildlife Service (2003). Longleaf pine ecosystem fact sheet. Retrieved from: http://www.southeast.fws.gov/pfwpine.html (accessed 4 November 2003).

U.S. Fish and Wildlife Service. (2003). Recovery plan for the red-cockaded woodpecker (Picoides borealis): second revision. *U.S. Fish and Wildlife Service, Atlanta, GA.*

University of Georgia. (2021). *Sandhills Fact Sheet*. Savanna River Ecology Laboratory. Retrieved 2021, from https://archive-srel.uga.edu/outreach/factsheet/sandhills.html.

US Bureau of the Census. (2017). Housing starts: Total new privately owned housing units started. Available online from *the Federal Reserve Bank of St. Louis at: https://fred.stlouisfed.org/series/HOUST;* last accessed 2021.

US Department of Agriculture (USDA) Forest Service. (2009). Red-cockaded Woodpecker habitat restoration and improvement projects in the Sweetwater and Coleman Lake area: Environmental assessment.

US Fish and Wildlife Service (2020). *U.S. fish and Wildlife service - Red-cockaded Woodpecker recovery*. Official Web page of the U S Fish and Wildlife Service. Retrieved September 14, 2021, from https://www.fws.gov/rcwrecovery/rcw.html.

US Fish and Wildlife Service. (2010). Longleaf pine range-wide conservation initiative. US Department of the Interior, Fish and Wildlife Service.

USDA-NRCS (1999). Soil Quality Test Kit Guide. *USDA Agricultural Research Service. National Consevation Service. Soil*.

USDA-NRCS. (2014). The PLANTS Database (http://plants.usda.gov, 14 January 2014). National Plant Data Team, Greensboro, NC 27401-4901 USA.

USDA-NRCS. (n.d.). *Inherent Factors Affecting Bulk Density and Available Water Capacity*. Soil Quality Kit- Guide for Educators. Retrieved 2021, from https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_053260.pdf.

USDA-NRCS. (2006). Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin. *US Department of Agriculture Handbook 296*.

USDA, NRCS. (2014). The PLANTS Database. *National Plant Data Team, Greensboro, NC 27401-4901 USA*. Retrieved from: http://plants.usda.gov

USDA. (1984). Interpretation of Tensiometer Readings. *Irrigation Water Management*. Retrieved 2021, from: https://efotg.sc.egov.usda.gov/references/public/NE/NIG_Interp_Tensiometer_Read_Table686-2_pg9-228.pdf.

Van Eerden, B. P. (1997). *Studies on the reproductive biology of wiregrass (Aristida stricta Michaux) in the Carolina Sandhills* (Doctoral dissertation, University of Georgia).

Van Lear, D. H., & Jones, S. M. (1987). An Example of Site Classification in the Southeastern Coastal Plain Based on Vegetation and Land Type[1]. *Southern Journal of Applied Forestry*, *11*(1), 23-28.

Van Lear, D. H., Carroll, W. D., Kapeluck, P. R., & Johnson, R. (2005). History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. *Forest ecology and Management*, *211*(1-2), 150-165. 11(1):23-28.

Vance, E. D., Aust, W. M., Strahm, B. D., Froese, R. E., Harrison, R. B., & Morris, L. A. (2014). Biomass harvesting and soil productivity: is the science meeting our policy needs?. *Soil Science Society of America Journal*, *78*(S1), S95-S104.

Vorhees, L. (2015). *PRIORITIZING CONSERVATION AREAS FOR LONGLEAF PINE FORESTS IN NORTH CAROLINA* (Doctoral dissertation, Duke University).

Waddell, K. L. (2002). Sampling coarse woody debris for multiple attributes in extensive resource inventories. *Ecological indicators*, *1*(3), 139-153.

Wade, D. D., & Lundsford, J. (1990). Fire as a forest management tool: prescribed burning in the southern United States. *Unasylva*, *41*(3), 28-38.

Wahlenberg, W. G. (1946). Longleaf pine: Its use, ecology, regeneration, protection, growth, and management. *Longleaf Pine: its use, ecology, regeneration, protection, growth, and management.*

Waldrop, T. A., White, D. L., & Jones, S. M. (1992). Fire regimes for pine-grassland communities in the southeastern United States. *Forest Ecology and Management*, *47*(1-4), 195-210.

Waldrop, T., Phillips, R. A., & Simon, D. A. (2010). Fuels and predicted fire behavior in the southern Appalachian Mountains after fire and fire surrogate treatments. *Forest Science*, *56*(1), 32-45.

Walker, J. L., & Silletti, A. M. (2007). Restoring the ground layer of longleaf pine ecosystems. In *The Longleaf Pine Ecosystem* (pp. 297-333). Springer, New York, NY.

Walker, J. L., van Eerden, B. P., Robinson, D., & Hausch, M. (2004). Burning and chopping for woodpeckers and wiregrass?. *In: Costa, Ralph; Daniels, Susan J., eds. Red-cockaded woodpecker: Road to recovery. Blaine, WA: Hancock House Publishers: 683-686.*

Walker, J., & Peet, R. K. (1984). Composition and species diversity of pine-wiregrass savannas of the Green Swamp, North Carolina. *Vegetation*, *55*(3), 163-179.

Walker, L. R., Jose, S., Jokela, E. J., & Howarth, R. W. (2006). *The longleaf pine ecosystem: ecology, silviculture, and restoration*. Springer Science & Business Media.

Walker, T. W., & Adams, A. F. R. (1958). Studies on soil organic matter: I. Influence of phosphorus content of parent materials on accumulations of carbon, nitrogen, sulfur, and organic phosphorus in grassland soils. *Soil science*, *85*(6), 307-318.

Walker, T., Cardellichio, P., Colnes, A., Gunn, J., Kittler, B., Perschel, B., ... & Initiative, N. C. (2010). Biomass sustainability and carbon policy study. *Manomet Center for Conservation Sciences*.

Wall, W. A., Hoffmann, W. A., Wentworth, T. R., Gray, J. B., & Hohmann, M. G. (2012). Demographic effects of fire on two endemic plant species in the longleaf pine-wiregrass ecosystem. *Plant Ecology*, *213*(7), 1093-1104.

Wang, J., & LeDoux, C. B. (2003). Estimating and validating ground-based timber harvesting production through computer simulation. *Forest Science*, *49*(1), 64-76.

Wang, J., LeDoux, C. B., Edwards, P., & Jones, M. (2005). Soil bulk density changes caused by mechanized harvesting: A case study in central Appalachia. *Forest Products Journal. 55 (11): 37-40.*, *55*(11).

Wang, J., Long, C., & McNeel, J. (2004). Production and cost analysis of a feller-buncher and grapple skidder in central Appalachian hardwood forests. *Forest products journal*, *54*(12), 159.

Westerhold, J. (2013). The Long Road to Recovery for the Longleaf Pine. Retrieved from https://www.americanforests.org/magazine/article/the-long-road-to-recovery-for-thelongleaf-pine/ (2021).

Williams, M. (1992). *Americans and their forests: a historical geography*. Cambridge University Press.

Winsauer, S. A. (1984). *Feller/bunchers in plantation thinnings: Factors affecting productivity* (Vol. 254). US Department of Agriculture, Forest Service, North Central Forest Experiment Station.

Wood-Energy. (2019, September 5). *Cost factors in harvesting and transporting woody biomass*. Wood Energy. Retrieved 2021, from https://wood-energy.extension.org/cost-factors-in-harvesting-and-transporting-woody-biomass/.

Woodall, C. W., & Monleon, V. J. (2008). Sampling protocol, estimation, and analysis procedures for the down woody materials indicator of the FIA program. *Gen. Tech. Rep. NRS-22. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 68 p.*, *22*.

Woodall, C. W., Ince, P. J., Skog, K. E., Aguilar, F. X., Keegan, C. E., Sorenson, C. B., ... & Smith, W. B. (2011). An overview of the forest products sector downturn in the United States. *Forest Products Journal*, *61*(8), 595-603.

Worley, D. P. (1965). *Gross Job Time Studies: An Efficient Method for Analyzing Forestry Costs* (Vol. 54). Rocky Mountain Forest and Range Experiment Station, Forest Service, US Department of Agriculture.

Yang, X., Chen, X., & Yang, X. (2019). Effect of organic matter on phosphorus adsorption and desorption in a black soil from Northeast China. *Soil and Tillage Research*, *187*, 85-91.

Young, K. R., Roundy, B. A., & Eggett, D. L. (2013). Plant establishment in masticated Utah juniper woodlands. *Rangeland Ecology & Management*, *66*(5), 597-607.

Zoë Hoyle, Z. (2015, October 20). *Longleaf pine at a 50-Year high in South Carolina*. CompassLive. Retrieved 2021, from https://www.srs.fs.usda.gov/compass/2015/10/20/longleaf-pine-at-a-50-year-high-in-south-carolina/.