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COMPARISON OF VEGETATIVE COMMUNITY AND SOIL ORGANIC MATTER
DEPTH AMONG REFERENCE SITES AND TWO RESTORED WETLANDS IN THE
COASTAL PLAIN OF SOUTH CAROLINA

A Thesis
Presented to
the Graduate School of
Clemson University

In Partial Fulfillment
of the Requirements for the Degree
Master of Science
Wildlife and Fisheries Biology

by
Jessica N. Clark
August 2023

Accepted by:
Dr. Althea Hagan, Committee Chair
Dr. James T. Anderson
Dr. Monica Folk

ABSTRACT

Wetland loss and degradation from agriculture, urbanization, forestry, and mining is a global issue. South Carolina alone has lost over 27% of its wetlands. This historical wetland loss and climatic changes and impacts make restoring wetlands critical for the state. In restoration, understanding the difference in restored and reference wetland's vegetation and soil organic matter depth can be crucial in assessing the recovery rate and determining environmental functions and services. The main objectives for our research were to determine differences in soil organic matter depth and vegetative community between the restored Brosnan Forest wetlands, the headwater flats and headwater slopes, and reference sites at Francis Marion National Forest. Our results indicate no significant differences in soil organic matter depth between the two restored wetlands. However, the reference site's soil organic matter depth was twice that of the Brosnan wetlands. Additionally, there are differences in vegetative community between the two restored wetlands; different species dominated each wetland. However, compared to the reference sites, the Brosnan wetlands were more similar, with only a few quadrats reaching the reference sites.

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CHAPTER ONE

LITERATURE REVIEW ON WETLAND ECOLOGY, HISTORIC WETLAND LOSS, WET SAVANNAS, AND SLOPED FORESTED HEADWATER WETLANDS

Wetland Ecology

Wetland Ecology and Importance

The definition of a wetland has changed throughout the years and does so on an agency-to-agency basis. The most agreed-upon features that define a wetland are an area inundated by water for a portion of the year, with saturated soils, and hydrophytes, flood-tolerant biota (Keddy, 2010) (Mitsch & Gosselink, 2007). Wetlands comprise 5 to 8 percent of the world's landscape, approximately 7 to 10 million km² (Mitsch & Gosselink, 2007). The United States, excluding Hawaii and Alaska, has an estimated 43.6 million hectares of wetlands. Although they make up a small portion of the landscape, wetlands are home to a disproportionately large number of species (Mitsch & Gosselink, 2007) (Keddy, 2010) (Willaims & Dodd, 1978).

As previously mentioned, there are distinct and interacting features that create and influence wetland ecosystems. Those features include hydrology, physiochemical factors, biota, geomorphology, and climate. Wetlands can be inundated through one or many different hydrological processes, each affecting the biota and physiochemical environment of the system. Those hydrological processes are precipitation, surface flow, groundwater, tides, and flooding. Understanding the hydroperiod of a system is essential not only in classifying the wetland but also in the creation or maintenance of a wetland

(Mitsch & Gosselink, 2007). Climate is a crucial factor in the hydrology of a wetland ecosystem. Wetlands, primarily those reliant on precipitation, are found in regions with high precipitation (i.e., tropics) or cooler areas with low evapotranspiration rates (i.e., boreal region). The type of hydrological process that wets a wetland can affect physiochemical processes and the type of biota observed. The physiochemical processes directly tied to hydrology are oxygen availability, nutrient availability, toxicity, and pH levels. The geomorphological setting of a wetland (e.g., headwaters, coastal lagoons, river floodplains, intertidal zones, inland lakes, and inland depressions) can also be important in determining wetland type and what biota will be observed (Little, 2013).

Wetlands provide a wide array of environmental services. At the individual level, wetlands supply refugia to many biota hunted or fished for recreation. Additionally, wetlands are useful in carbon storage (Villa & Bernal, 2018; Mikhailova et al., 2021), flood control (Tang & Kavvas, 2020), water storage (Hubbard & Linder, 1986; Lane and D'Amico, 2010), wildlife habitat (Keddy, 2010; Williams and Dodd, 1978), and water filtration (Burkett & Kusler, 2007; Dordio et al., 2008). On a global scale, wetlands are vital in carbon sequestration and stabilizing nitrogen, carbon dioxide, and methane (Mitsch and Gosselink, 2007). Wetlands are not only valuable for their anthropogenic value and function; they also play an important role in biodiversity all over the globe (e.g., 25% of plant species in Malaysia occur in peat swamps, 10% of the world's fish fauna occur in the Amazon basin) (Mitsch & Gosselink, 2007; Gopal, 2009). Many species rely solely on wetlands for habitats; it is estimated that in 1991 43% of 595 plant

and animal species in the U.S. Fish and Wildlife Service's federally threatened or endangered list are dependent on wetland systems (Murdock, 1994).

Wetland Loss and History

Wetlands are historically one of America's most threatened and vulnerable ecosystems. Today's largest cities have been built on wetlands (e.g., Boston, Washington, D.C., Chicago, and Paris, France) (Mitsch and Gosselink, 2007). In the past, wetlands were defined only by their resources or agricultural potential and were seen as places that should be avoided or altered (Mitsch and Gosselink, 2007). Shaler (1890) defined wetlands as soil too wet for agriculture. Therefore, at that time, wetlands were seemingly defined in terms of what they could provide, and the only management policy was their drainage. Culturally, wetlands were and are still seen as wild or eerie. By looking at the names of a few historical swamps, the attitude towards them is evident (e.g., the Great Dismal Swamp in North Carolina and Tate's Hell Swamp in Florida). Additionally, the view of wetlands is clear in comics such as Swamp Thing and the endless use of wetlands in films to invoke a feeling of unease.

From 1780 to 1980, the lower 48 of the United States lost approximately 53% of its natural wetlands, with 56 to 65% of loss due to agriculture (Dahl, 1990; Mitsch & Gosselink, 2007). For inland wetlands, this loss can be attributed to filling for urban development and draining for agriculture, forestry, and mining. (Dahl, 1990; Mitsch and Gosselink, 2007). The destruction of wetland ecosystems was accepted and encouraged, and because of this, states like Ohio and California have an estimated 90% loss (Dahl,

1990). South Carolina's wetland loss is lower than the national average, with an approximate loss of 27% (Dahl, 1999).

Mitigation and Restoration in Wetlands

To mitigate historic wetland loss and to minimize further loss, the U.S. Clean Water Act of 1972, section 404, requires permits through the U.S. Army Corp of Engineers before draining, damaging, or destroying a wetland, making the land developer financially responsible and obligated to mitigate for the loss at a 1:1 ratio, if impact is unavoidable (Mitsch & Gosselink, 2007; Stefanik & Mitsch, 2012). This 1:1 ratio is intended to support the Environmental Protection Agency's (EPA) set policy of "no net loss" (Mitsch & Gosselink, 2007; Stefanik & Mitsch, 2012). One of the methods used to provide wetland compensation is mitigation banking. Mitigation banking is restoring a large wetland area before compensation for impact is needed. The wetlands restored by a mitigation banker are used as a bank of credits that land developers can purchase to satisfy mitigation requirements and permits (Bendor, 2009). Bendor (2009) found that this method can experience a smaller lag between wetland loss and wetland regain of function since the mitigation is conducted before the initial impact.

Wetland restoration is intended to bring back biodiversity and environmental services that have historically been lost due to draining, ditching, and filling (Zedler et al. 2000). Restoration projects typically use reference ecosystems or species communities as a starting point and guide for a restoration (Matthews and Sypyreas, 2010). Reference sites are areas that are close in proximity and closely resemble the community type of the

restoration site. Generally, the success or progress of the restoration is estimated by the similarity of communities to reference ecosystem sites (Matthews and Sypreas, 2010).

Novel Community

The idea of a restoration reaching a reference or pre-disturbance site conditions may be outdated in the restoration field, as the reference ecosystems may never be achieved. Instead, novel communities are established (Stefanik & Mitsch, 2012). A novel ecosystem is one with species that do not share an evolutionary history and environmental conditions caused directly or indirectly by anthropogenic actions (Hobbs et al., 2006; Morse et al., 2014). These ecosystems are commonly observed in wetland restorations and creations (Hobbs et al., 2006; Moreno-Mateos et al., 2012; Stefanik & Mitsch, 2012). A novel ecosystem is a deviation from the natural trajectory of an ecosystem; however, these sites can still be self-sufficient, biodiverse, and offer ecosystem services (Morse et al., 2014; Perring et al., 2013). Additionally, novel ecosystems make up between 35–40% of the global land surface, and some projections in the literature suggest that there are more novel ecosystems than natural (Perring et al., 2013; Perring & Ellis, 2013; Ellis et al., 2010).

Wetland Plants and Communities

Role of Vegetation in Wetland Ecosystems

There are many definitions for wetland plants; the most common definition is a macrophyte that grows in water or on a substrate that periodically is oxygen deficient due

to excessive water content (US. Army Corp of Engineers 1987). Wetland macrophytes can differ from aquatic macrophytes in that they do not only grow in water but instead can grow in both terrestrial and anaerobic, inundated environments. Wetland and aquatic macrophytes can be classified as submergent, emergent, or floating. Emergent macrophytes are characterized by their root system growing in the soil or beneath the water surface and their reproductive and photosynthetic organs growing aurally (Cronk and Fennessy 2001).

Wetland vegetation is essential to the functionality of a wetland. Wetland vegetation has been used to delineate and classify wetlands for many years (U.S. EPA, 2002; Little, 2013). Wetland vegetation can also influence water chemistry (Ehrenfeld & Schneider, 1993), hydrology (Ehrenfeld & Schneider, 1993), sediment, habitat structure, and productivity of wetland ecosystems (Cronk and Fennessy, 2001). Wetland plants are distributed through the same mechanisms as any plant; those include migratory birds, wind, water, and animal/human transport. Wetland plants are listed as cosmopolitan, or endemic based on a plant's ability to distribute. Cosmopolitan species have a wide distribution range, whereas endemic species have a small range, whether from ecological barriers or environmental factors (Cronk and Fennessy 2001).

Forested Wetlands

A forested wetland is an area that is inundated by water for a portion of the year, with hydric soils, a canopy of woody species, and sometimes herbaceous species in the understory. The Southeast has 54% of all wooded palustrine wetlands in the United

States, and approximately 75% of all Southeastern wetlands are wooded (Messina and Connor, 1997). Forested headwater wetlands are found in the Southeastern United States. Headwaters are part of a basin that contributes to creating and maintaining downstream streams, rivers, lakes, and oceans (FEMA 1993; AFS 2008). Headwater wetlands are essential for downstream water quality, water supply, flood control, and aquifer replenishment; they provide habitat to many species and contribute to forest biodiversity through ecological corridors. When these wetlands are disturbed, it can cause loss of habitat, wetland function, and environmental services.

Bottomland Hardwoods

In South Carolina, there is a focus on restoring forested wetlands, more specifically, bottomland hardwoods. Bottomland hardwoods reside in riparian ecosystems throughout the Piedmont and coastal plain region of the Southeastern United States. These floodplain forests have essential functions of water quality improvement and biochemical processes (i.e., nitrification, denitrification, movement of carbon, phosphorus uptake, and decomposition) (Kellison and Young, 1997) (Hunter et al., 2008). Hydrology is the determining factor for bottomland hardwood ecosystems (Faulkner, 1992). The common wetland tree species found in bottomland hardwoods and forest wetlands are *Taxodium distichum* (bald cypress), *Nyssa aquatica* (water tupelo), *Acer rubrum* (red maple), and members of the *Fraxinus*, *Quercus*, *Salix*, and *Populus* genera (Cronk and Fennessy 2001). Hunter et al. (2008) observed that it is not enough to

plant woody species in the restoration of bottomland hardwood, and only when proper hydrology was restored did function return.

Headwater Wetlands

Forested headwater wetlands are found in the Southeastern United States. Headwaters are defined as a part of the river basin that contributes to the creation and maintenance of downstream rivers, lakes, and oceans (FEMA 1993; AFS 2008). Headwater wetlands are essential for downstream water quality, flood control, and aquifer replenishment; they provide habitat to many species and contribute to forest biodiversity through ecological corridors. Therefore, when these wetlands are disturbed, it can cause a loss of habitats, wetland functions, and environmental services.

The vegetation in natural headwater wetlands is usually comprised of flood-tolerant hardwoods. The most common woody species found are red maple (*Acer rubrum*), tulip poplar (*Liriodendron tulipifera*), swamp tupelo (*Nyssa biflora*), sweet bay (*Magnolia virginica*), loblolly pine (*Pinus taeda*), water tupelo (*Nyssa aquatica*), cherry bark oak (*Quercus pagoda*), willow oak (*Quercus phellos*), pond cypress (*Taxodium ascendens*), and bald cypress (*Taxodium distichum*) (NC Division of Water Resources, 2021). In some regions, headwater wetlands can have diverse shrub and herbaceous communities. Like many wetland systems, headwater wetlands are inundated for a portion of the year, and they typically remain dry throughout the growing season due to evapotranspiration. However, inundation and saturation frequency are dependent on the region. For example, most wetlands in Florida are wet from May to November. Still,

wetlands in South Carolina can experience two peaks (August to November from summer rains and January to February due to frontal rains and low evapotranspiration).

Pocosins

A Pocosin is a peatland found throughout the Atlantic coastal plain, from northern Virginia to northern Florida, and is typically on poorly drained, inter stream flats (Schafale & Weakley, 1990). Pocosins are naturally nutrient-poor and (depending on the region) can be saturated throughout the year. These peatland communities can range from High Pocosins, characterized by trees and dense shrubbery, to Low Pocosins, a rarer pocosin community type and includes shrubs, herbs, and widely scattered trees (Schafale & Weakley, 1990). Because Low Pocosins are more scarce than High Pocosins, the two types are typically lumped together in the literature.

Low Pocosins

Low pocosins plant community consists of a dense shrub layer measuring less than 1.5 meters tall and widely scattered trees, the most common being *Pinus serotina*, *Persea palustris*, *Gordonia lasianthus*, and *Magnolia virginiana* (Schafale and Weakley, 1990). Pocosin openings are usually small patches but can be hundreds of acres within low pocosins dominated by herbaceous species or dwarf shrubs. Dominant herbs in pocosin openings are *Anchistea virginica*, *Carex striata*, *Sarracenia flava*, and *Sarracenia purpurea*. Less common herbs include *Drosera intermedia*, *Rhynchospora*

chalarocephala, *Rhynchospora fascicularis*, *Xyris fimbriata*, *Utricularia subulata*, *Utricularia* spp., *Peltandra sagittifolia*, or *Lysimachia asperulifolia* (Otte, 1981

Wet Savannas

Wet savannas are any plant community with scattered trees, herbaceous vegetation, and hydric soils (Fowler & Beckage, 2020). Wet savannas are similar to wet prairies and are sometimes used synonymously in the literature when discussing soil, topography, and vegetation; however, wet savannas have a higher cover of trees (approximately less than 30%) (Ford, 2011). In the coastal plains of Florida, wet prairies and savannas have continuously moist soils; however, they are never inundated. The soils are typically acidic, nutrient-deficient, and flat or slightly sloped (FNAI, 2010). These ecosystems can be found between higher mesic or dry Flatwoods, dry prairies, shrub bogs, dome swamps, or low-lying depression marshes (FNAI, 2010). Fires are natural components of these ecosystems at an interval of every 2—3 years; in the absence of fire, woody species can invade, shading out herbaceous species (FNAI, 2010). Fire is also required to develop some of the present herbaceous species (FNAI, 2010).

In the drier areas of wet prairies and savannas of the coastal plain, the dominant species is *Aristida stricta* var. *beyrichiana* (FNAI, 2010). *Lycopodiella alopecuroides*, *Muhlenbergia expansa*, *Pinguicula lutea*, and *Rhexia alifanus* can also be found in these drier portions (FNAI, 2010). In the wetter areas, wiregrass may still be present, or it is commonly replaced by sedges, such as *Rhynchospora plumosa*, *R. oligantha*, *Scleria baldwinii*, *S. georgiana*, or *Aristida palustris*. Carnivorous plants can also be present in

wetter portions, including *Sarracenia* spp., *Drosera* spp., *Pinguicula* spp., *Utricularia* spp. Other common species include *Bigelowia nudata*, *Eriocaulon compressum*, *Oxypolis filifolia*, and *Xyris ambigua* (FNAI, 2010). Some common flowering graminoids and forbs are *Agalinis* spp., *Calopogon* spp., *Platanthera* spp., *Polygala* spp., *Rhexia* spp., *Sabatia* spp., *Xyris* spp., *Rhynchospora latifolia* (FNAI, 2010). Common composites, flowers made up of inflorescences, found in this ecosystem are in the genera *Balduina*, *Carphephorus*, *Coreopsis*, *Eupatorium*, *Eurybia*, *Helenium*, *Helianthus*, *Rudbeckia*, *Solidago*, and *Symphyotrichum* (FNAI, 2010). Woody species present in low abundance are *Hypericum brachyphyllum*, *H. myrtifolium*, *Myrica caroliniensis*, *Pinus elliottii*, *Taxodium ascendens*, *Nyssa sylvatica* var. *biflora*, and in some cases *Cyrilla racemiflora*, *Clethrea alnifolia*, *Ilex cassine* var. *myrtifolia*, and *Ilex coriacea* (FNAI, 2010).

Pond-cypress/pond pines are an uncommon vegetative community found in the coastal plain of the Southeastern United States (Steven & Harrison, 2022; Leblond & Grant, 2007). The loss of this wetland vegetation community can be attributed to these flat, shallow, herbaceous coastal plain wetlands being historically lost and nearly extinct vegetation communities from fire suppression and agricultural use (Platt, 1999). This community is semi-forested, characterized by the presence of *Pinus serotina* and *Taxodium ascendens* with a species-rich graminoid and forb ground cover, consisting of *Andropogon* spp., *Panicum* spp., *Amphicarpum* spp., *Rhynchospora* spp., *Iris* spp., *Lachnanthes* spp., *Dichanthelium* spp., *Rhexia* spp., *Ludwigia* spp., *Eleocharis* spp., *Xyris* spp., and *Eriocaulon* spp. (Steven & Harrison, 2022; Leblond & Grant, 2007). Low, evergreen shrub species can also be found in the genera *Ilex*, *Lyonia*, and *Persea* (Steven

& Harrison, 2022). These community types are also habitats for many rare plant species such as *Astragalus michauxii*, *Eupatorium resinosum*, *Lindera melissifolia*, *Litsea aestivalis*, *Rhexia aristosa*, and *Scleria georgiana* (Leblond & Grant, 2007). These systems typically have deep, sandy soils with a clay layer, and the cypress savannas tend to have longer hydroperiods, at 40% duration over a year (Steven & Harrison, 2022; Leblond & Grant, 2007).

Soils Background

Wetland Soils

Wetland soils, known as hydric soils, are formed during anaerobic conditions due to the presence of water, thus causing a reduction in soil oxidation potential, followed by denitrification and reduction of iron and sulfate (NRCS, 1998; Vasilas & Vasilas, 2013; Pezeshki & Delaune, 2012; USDA & NRCS, 2018) The U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS,1998) defines hydric soils as "a soil that forms under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part" (Mitsch & Gosselink, 2007). Hydric soils are essential in maintaining many vital functions of a wetland (e.g., water storage, groundwater recharge, groundwater discharge, water quality, and wildlife habitat) (Baker et al., 2008). There are two types of hydric soils: mineral and organic.

Hydric Organic Soils

Organic soil is the accumulation of organic plant material in various stages of decomposition due to anoxic conditions from stagnant or poorly drained conditions

(Mitsch and Gosselink, 2007). Organic soils are generally classified as soils with a high organic matter content (greater than 20 to 35%) in the upper 35.56 to 40.64 cm (Mitsch and Gosselink, 2007; Baker et al., 2008). Additionally, several physiochemical features of hydric organic soils distinguish them from mineral soils. These features include bulk density and porosity, hydraulic conductivity, nutrient availability, and cation exchange capacity (Mitsch and Gosselink, 2007). Organic soils have high porosity and water-holding capacity; therefore, their bulk density is much lower than mineral soils. Hydraulic conductivity depends on the degree of organic matter decomposition; although organic soils have higher porosity and water-holding capacity, it does not mean that water can pass through readily. Nutrient availability is often low in hydric organic soils, with most nutrients being organic and thus unavailable to plants. Hydric organic soils have a high cation exchange capacity. Organic content tends to be positively correlated with the number of hydrogen ions present, meaning as the organic content increases, so does the number of exchangeable hydrogen ions (Mitsch and Gosselink, 2007). Another distinguishing feature of organic soils is their color. Organic soils are typically dark in color, ranging from black to dark brown (Warren and Pearson, 2000; Mitsch and Gosselink, 2007). The color of organic soil is determined by the degree of decomposition, which can be broken into four categories: Fibrists, Folists, Hemists, and Saprist, or muck, are when the organic material is highly decomposed (two-thirds or more of the material is decomposed), resulting in dark black soils (Mitsch and Gosselink, 2007). When less than one-third of the organic material is decomposed, it is known as Fibrists, commonly peat. Peat tends to be slightly lighter in color than muck, ranging from brown

to dark brown. Hemists, also known as mucky peat or peaty muck, when the soil falls between fibrist and saprists (Mitsch and Gosselink, 2007). Finally, Folists are organic soils that are moderately decomposed and are made up primarily of leaf litter and other organic materials. This organic soil type can frequently be found in forested wetlands, they are dark in color and typically have a spongy texture, similar to Hemists.

Hydric Mineral Soils

The characteristics used to identify and define hydric mineral soils are redoximorphic features. This term refers to the features formed due to redox reactions and translocation (Mitsch and Gosselink, 2007). One characteristic of hydric mineral soils is mottling or gleying. Gleization is when saturation or inundation of the soil causes microbes to reduce iron and manganese into their soluble form, ferrous and manganous (Mitsch and Gosselink, 2007; Baker et al., 2008), Thus, making it possible for them to leach out of the soil causing a depletion, otherwise known as redox depletions (Mitsch and Gosselink, 2007; Baker et al., 2008). These redox depletions make the soil appear gray, black, or sometimes a green or blue-gray. Conversely, mottling occurs when soils are dry and iron is oxidized, causing a reddish, orange color. In the case of hydric mineral soil's physiochemical features, they typically have a high bulk density, low porosity (45-55%), high hydraulic conductivity, low water holding capacity, and low cation exchange capacity (due to the high amount of major metal cations) (Mitsch & Gosselink, 2007).

Riverine wetlands can range from organic to mineral soils, whereas most slope wetland soils are mineral (NCFS, 2009). The accumulation of black muck forms pocosins; therefore, their soils are characteristically deep and acidic hydric organic soil

(NCDEQ, 2019). The organic matter in pocosins can range from several inches to several feet in some instances and can be at or below the surface (NCFS, 2009). Low Pocosins typically have a deeper organic matter layer than High Pocosins, with plant roots never reaching mineral soils (NCNHP, 2018).

Restrictive Feature

A restrictive feature, also known as a restrictive layer or limiting layer, is a layer that impedes the flow of water and air and restricts root growth chemically, physically, or thermally (USDA, 2019). Some examples of restrictive features are bedrock, cemented layers, frozen layers, and dense layers (USDA, 2019). The restrictive feature is important because its depth affects vegetation, water-holding capacity, and nutrient availability (Rajakaruna & Boyd, 2008). According to the USDA's National Soil Survey Handbook, soils are assigned a rating class based on their limitations on water holding capacity and root zone (USDA, 2017). The soil is considered limited if the depth to the restrictive feature is within 50 cm; any value greater than 50 cm is deemed non-restrictive (USDA, 2017).

Soil Organic Matter in Wetland Restoration

The difference between soil organic matter (SOM) in created or restored wetlands and natural wetlands has been well documented in the literature (Bishel-Machung et al. 1996; Burland and Richardson, 2006; Campbell et al. 2002; Gallatowitsch & van der Valk, 1996; Shaffer and Ernst 1999). Therefore, it is known that created and restored wetlands tend to have lower SOM than natural wetlands of the same or similar wetland type. It can take decades to build organic matter (Anderson et al., 2005; Ballentine &

Schneider, 2009; Anderson & Mitsch, 2006) if they ever do (Bruland & Richardson, 2006; Schaffer and Ernst, 1999).

Brosnan Forest and Reference Sites

Brosnan Forest

The Brosnan Forest is a 5,827.47-hectare (ha) preserve in upper Dorchester County, SC, located north and south of Hwy 78 (Figure 1). The site is owned by Norfolk Southern Railway Company and used for recreation and as a conference facility for their employees. In the 1950s, several onsite wetlands were drained, ditched, and replaced with a loblolly pine plantation. The Coldwater Branch Mitigation Bank is restoring two impacted wetlands and a stream system. This research focuses on the two restored wetlands located south of Hwy 78 (Figure 1: sloped headwater wetland = green, flat headwater wetland = tan). The headwater flats wetland is considered an herbaceous-dominated wetland, while the slope wetland is forested. This is supported by historical aerial photographs of the site (pre-disturbance), this basin's unusual topographic and soil profile, and the presence of diverse freshwater herbaceous wetland vegetation less than a year after clearing, which naturally recruited to the site or developed quickly from a pre-existing seed bank. It is speculated that the wetland is historically a wet savanna. However, from the topography, it is also possible that the wetland could have traditionally been a pocosin, although it is less likely from looking at the historical aerial photographs (Folk & Ecosystem Planning and Restoration, 2020).

The two wetlands being restored are below a restored wetland that historically was a perched pocosin (Figure 1.1). Water from the pocosin feeds into the sloped headwater wetland, draining into the headwater flat wetland (the historically herbaceous wetland). The sloped headwater wetland is, as the name implies, sloped with longleaf pine upland on either side. The headwater flat wetland is slightly depressed in the landscape and flat. Stream construction is underway in the center of both wetlands to restore the natural Coldwater Branch stream (Folk & Ecosystem Planning and Restoration, 2020).

Francis Marion National Forest Reference Sites and Site Selection

The Francis Marion National Forest (FMNF) comprises approximately 104,813.58 ha and is located in the Coastal Plain region of South Carolina. The FMNF resides in the counties of Berkeley and Charleston and is 64.37 km north of the metropolitan area of Charleston, SC, and 48.28 km south of Myrtle Beach, SC (Figure 1.2). The climate for this area is humid subtropical and experiences an average annual precipitation of approximately 123.57 cm, with over half of the yearly precipitation occurring June-October (National Oceanic and Atmospheric Administration; Steven & Harrison, 2022). The first FMNF site is north of Steed Creek Rd., a main, paved road that runs through the FMNF; a powerline corridor runs parallel to the wetland slope (Figure 1.2). The cypress zone starts near a stream system, Cropnel Dam Creek, then transitions into a pond pine savanna as you near Steed Creek Rd. The second FMNF SOM site is also north of Steed Creek Rd (Figure 1.2). The cypress savanna zone is a small area on

the north edge of the road, and the pond pine zone expands to the west on either side of an unpaved forest road. A pocosin resides to the Northwest of the site. No topographical or soil data was found for either site in the United States Division of Agriculture's (USDA; 2023) Web Soil Survey database.

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Figures



Figure 1.1. Aerial imagery of the Brosnan Forest and wetland sites in relation to highway 78 and Dorchester, SC, USA.

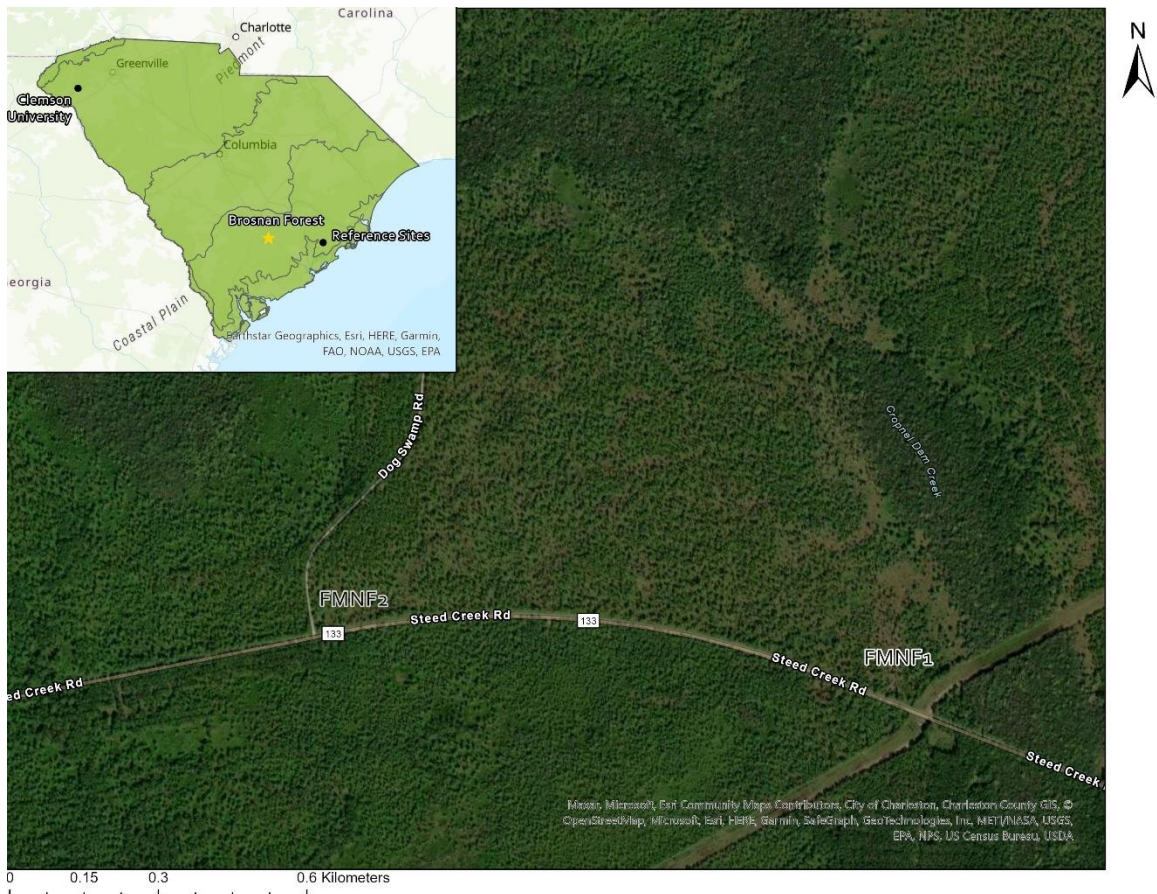


Figure 1.2. The inset map helps to locate Francis Marion National Forest (FMNF) reference sites and Broxton forests within South Carolina, USA. The main map shows the proximity of the FMNF sites to each other, a main road that runs through FMNF, Steed Creek Rd., as well as other possible disturbances, such as the powerline corridor parallel near the first site and the unpaved forest road near the second site.

CHAPTER TWO

VEGETATIVE COMPARISON BETWEEN REFERENCE SITES AND TWO RESTORED WETLANDS IN THE COASTAL PLAIN OF SOUTH CAROLINA

Abstract

Since European colonization, wetlands have been lost and degraded in the United States at an extreme rate from agriculture, urbanization, forestry, and mining, with over 2.6 million ha (27—32%) of wetlands lost in South Carolina alone. This historic wetland loss and climatic changes and impacts make restoring wetlands much more critical for the state. Our study aimed to determine the differences in vegetation community between two restored wetlands in the coastal plain of South Carolina. Additionally, we sought to determine if one of the restored wetlands, which is thought to be historically herbaceous, had a closer vegetative composition to the two selected reference sites. We used 36 randomly stratified transects established for mitigation monitoring and a systematic method for vegetation sampling, using a 1 m² quadrat to record abundance, density, and biomass. We analyzed the vegetation data using importance values by species, an indicator species analysis (ISA), an ANOVA to examine differences in biomass, and non-metric multidimensional scaling (NMDS) ordinations to visually interpret vegetation quadrats' dissimilarity visually. The ISA indicated that different species dominate the restored wetlands, and the ANOVA results suggested no significant difference in biomass between the restored and reference wetlands. The NMDS showed some separation between the restored wetlands and between the restored wetlands and the reference sites. Knowing the difference between restored and reference wetland vegetation can be

beneficial in understanding the recovery rate of the restored wetland and determining the return of some environmental functions and services.

Introduction

Wetlands are historically one of the most threatened and vulnerable ecosystems (Mitsch & Gosselink, 2007). From 1780 to 1980, the lower 48 of the United States lost 53% of its natural wetlands, with 56 to 65% due to agriculture (Dahl, 1990; Mitsch & Gosselink, 2007). For inland wetlands, this loss can be attributed to filling and draining for urban development, agriculture, forestry, and mining (Mitsch & Gosselink, 2007). South Carolina has lost over 27% of its wetlands (Dahl, 1999). Additionally, some regions' future climatic and environmental conditions trending towards wetter and warmer conditions could impact wetland structure and functions (Burkett & Kusler, 2007). This historical loss and current loss due to climate change makes restoring wetlands even more critical. Wetlands provide essential ecosystem services, such as carbon sequestration (Villa & Bernal, 2018; Mikhailova et al., 2021), flood control (Tang & Kavvas, 2020), water storage (Hubbard & Linder, 1986; Lane & D'Amico, 2010), wildlife habitat (Keddy, 2010; Williams & Dodd, 1978), and water filtration (Burkett & Kusler, 2007; Dordio et al., 2008).

To mitigate historic wetland loss and to minimize further loss, the U.S. Clean Water Act of 1972, section 404, requires permits through the U.S. Army Corp of Engineers before draining, damaging, or destroying a wetland, making the land developer financially responsible and obligated to mitigate for the loss at a 1:1 ratio, if impact is

unavoidable (Mitsch & Gosselink, 2007; Stefanik & Mitsch, 2012). This 1:1 ratio is intended to support the Environmental Protection Agency's (EPA) set policy of "no net loss" by assessing the wetland type, location or watershed, size, hydrology, biodiversity, and ecosystem services of the proposed impacted wetland and restoring or creating a wetland of equal size, wetland type, and ecological value (Mitsch & Gosselink, 2007; Stefanik & Mitsch, 2012). However, restoring wetlands can be a long, tedious process, and no net loss can be seemingly unattainable (Bendor, 2009; U.S. EPA, 2014), especially when restoring or creating wetland types that historically had deep organic soils (Ahn & Jones, 2013; Anderson et al., 2005; Anderson & Mitsch, 2006; Ballentine & Schneider, 2009; Bruland & Richardson, 2006; Shaffer & Ernst, 1999), which impact hydrological regime and vegetation community (Campbell et al., 2002; Bruland & Richardson, 2005; Bantilan-Smith et al., 2009; Ahn & Dee, 2011; Stefanik & Mitsch, 2012). Additionally, recovery rates and the trajectory of a restored or created wetland to a natural reference or pre-disturbance conditions can be impacted based on climate, ecosystem size, physical characteristics, site history, and potential impacts (Moreno-Mateos et al., 2012). Therefore, it is a complicated and slow process for wetland restorations to successfully achieve pre-disturbance or reference site conditions, if at all (Moreno-Mateos et al., 2012; Stefanik & Mitsch, 2012; Zedler and Callaway, 2002), and can often result in the establishment of a novel ecosystem instead.

A novel ecosystem is one with species that do not share the evolutionary history and environmental conditions caused directly or indirectly by anthropogenic actions (Hobbs et al., 2006; Morse et al., 2014). These ecosystems are commonly observed in

wetland restorations and creations (Hobbs et al., 2006; Moreno-Mateos et al., 2012; Stefanik & Mitsch, 2012). A novel ecosystem is a deviation from the natural trajectory of an ecosystem. However, these sites can still be self-sufficient, biodiverse, and offer ecosystem services (Morse et al., 2014; Perring et al., 2013). Additionally, novel ecosystems make up between 35–40% of the global land surface, and some projections in the literature suggest that there are more novel ecosystems than natural (Perring et al., 2013; Perring & Ellis, 2013; Ellis et al., 2010). Although a site may never reach a historical or reference state, having a reference site or pre-disturbance site conditions is a valuable tool for restoration (Stefanik & Mitsch, 2012).

When designing the plan for a created or restored wetland site, it is preferable to have historic, pre-disturbance site characteristics and conditions for the area you are restoring to replicate wetland type and processes (Stefanik & Mitsch, 2012). However, in wetland mitigation, this information isn't always available. Therefore, reference sites are valuable tools for grasping a starting point and guide for the restoration (Mitsch & Gosselink, 2007; Stefanik & Mitsch, 2012). Reference sites are areas that are close in proximity and closely resemble the community type of the restoration site. At our research site, pre-disturbance vegetation data isn't known, therefore understanding typical vegetation found in the two different wetland types, pond-cypress/pond pine, and forested headwater wetland, is vital information for comparing the vegetative community of the two wetlands at our research site and finding pond-cypress/pond pine reference sites.

Pond-cypress/pond pines are an uncommon vegetative community found in the coastal plain of the Southeastern United States (Steven & Harrison, 2022; Leblond & Grant, 2007). This community is semi-forested, characterized by the presence of *Pinus serotina* and *Taxodium ascendens* with a species-rich graminoid and forb ground cover, consisting of *Andropogon* spp., *Panicum* spp., *Amphicarpum* spp., *Rhynchospora* spp., *Iris* spp., *Lachnanthes* spp., *Dichantheium* spp., *Rhexia* spp., *Ludwigia* spp., *Eleocharis* spp., *Xyris* spp., and *Eriocaulon* spp. (Steven & Harrison, 2022; Leblond & Grant, 2007). Low, evergreen shrub species from the genera *Ilex*, *Lyonia*, and *Persea* can also be found (Steven & Harrison, 2022).

Forested headwater wetlands are found in the southeastern United States. Headwaters are defined as a part of the river basin that contributes to the creation and maintenance of downstream rivers, lakes, and oceans (FEMA 1993; AFS 2008). Headwater wetlands are essential for downstream water quality (Bullock & Acreman, 2003; Alexander et al., 2007; Colvin et al., 2019; Yeo et al., 2019), flood control (Bullock & Acreman, 2003), and aquifer replenishment (Bullock & Acreman, 2003; Roulet, 1990; Yeo et al., 2019; Ramesh et al., 2020); they provide habitat to a vast amount of species (Clipp and Anderson, 2014; Colvin et al., 2019) and contribute to forest biodiversity through ecological corridors (Colvin et al., 2019). The vegetation in headwater wetlands is usually comprised of mostly flood-tolerant hardwoods. The most common woody species found are red maple (*Acer rubrum*), tulip poplar (*Liriodendron tulipifera*), swamp tupelo (*Nyssa biflora*), sweet bay (*Magnolia virginiana*), loblolly pine (*Pinus taeda*), water tupelo (*Nyssa aquatica*), cherry bark oak (*Quercus pagoda*), willow oak (*Quercus*

phellos), pond cypress (*Taxodium ascendens*), and bald cypress (*Taxodium distichum*) (NC Division of Water Resources, 2021). In some regions, headwater wetlands can have diverse shrub and herbaceous communities.

Our research aimed to determine if there was a significant difference between the restored wetlands and reference sites in South Carolina's Coastal Plain. We were specifically interested in determining if there was a difference in the vegetative community between the two restored wetlands after 70 years of disturbance and if one of the restored wetlands that are historically herbaceous had a closer vegetative composition to the reference wetlands. We hypothesize that the historically herbaceous wetland would have more herbaceous species that naturally recruited or developed from an existing seed bank than the restored, historically forested wetland. Furthermore, we hypothesized that the reference wetland would have more biomass than the two restored wetlands.

Methods

Site Description

The Francis Marion National Forest (FMNF) comprises approximately 104,813.58 hectares (ha) and is located in the Coastal Plain region of South Carolina. The FMNF resides in the counties of Berkeley and Charleston and is 64.3738 km north of the metropolitan area of Charleston, SC, and 48.2803 km south of Myrtle Beach, SC (Figure 2.1). The climate for this area is humid subtropical and experiences an average annual precipitation of approximately 123.6 cm, with over half of the yearly precipitation occurring June-October (National Oceanic and Atmospheric Administration; Steven &

Harrison, 2022). The first FMNF site is north of Steed Creek Rd., a main, paved road that runs through the FMNF; a powerline corridor runs parallel to the wetland slope (Figure 2.1). The cypress zone starts near a stream system, Cropnel Dam Creek, then transitions into a pond pine savanna as you near Steed Creek Rd. The second FMNF SOM site is also north of Steed Creek Rd (Figure 2.1). The cypress savanna zone is a small area on the north edge of the road, and the pond pine zone expands to the west on either side of an unpaved forest road. A pocosin resides to the Northwest of the site. No topographical or soil data was found for either site in the United States Division of Agriculture's (USDA; 2023) Web Soil Survey database.

The Brosnan Forest is a 5,827.5-ha preserve in upper Dorchester County, SC, located north and south of Hwy 78 (Figure 2.2). The site is owned by Norfolk Southern Railway Company and is used for recreation and as a conference facility for their employees. In the 1950s, several onsite wetlands were drained, ditched, and replaced with a loblolly pine plantation. The Coldwater Branch Mitigation Bank is restoring two impacted wetlands and a stream system. This research focuses on the two headwater wetlands, comprising 115.34 ha, being restored above the stream project (Figure 2.3: sloped headwater wetland = green, flat headwater wetland = tan). The headwater flats wetland is considered a herbaceous-dominated wetland, while the slope wetland is forested. This is supported by historical aerial photographs of the site (pre-disturbance), the flats basin's unusual topographic and soil profile, and the presence of diverse freshwater herbaceous wetland vegetation less than a year after clearing, which naturally

recruited to the site or developed quickly from a pre-existing seed bank. It is speculated that the wetland was historically a wet savanna. It is also possible that the wetland could have historically been a pocosin, although it is less likely from looking at the historical aerial photographs.

To give more detail on the topography and arrangement of the site (Figure 2.3), the two wetlands being restored are below a partially restored wetland that historically was a perched pocosin. Water from the pocosin feeds into the headwater slope wetland, draining into the headwater flats wetland (the historically herbaceous wetland). The headwater slope wetland is, as the name implies, sloped, and the headwater flats wetland is slightly depressed in the landscape and flat, with both surrounded by managed longleaf pine (*Pinus palustris*) uplands.

Reference Site Selection

Data were collected from the two restored wetlands at the Brosnan Forest and two reference sites at the Francis Marion National Forest. The Francis Marion National Forest reference sites were chosen from the suggestion and expertise of Chick Gaddy, a consultant and rare wetland community expert for the Brosnan Forest Coldwater Branch Mitigation Bank (Folk & Ecosystem Planning and Restoration, 2020). The suggested sites were confirmed through qualitative data using satellite imagery from Google Maps (2022) and site visits. To find a potential site at the Francis Marion National Forest, we looked for an open canopy, pond pine (*Pinus serotina*), and a pond cypress (*Taxodium ascendens*) savanna zone. Investigators believe a pond pine, pond cypress (pine-cypress)

savanna is the community type most closely resembling the flat headwater wetland. Once a few possible reference sites were picked, we traveled to each location to ensure it was a true pine-cypress savanna and not the result of recent logging, was large enough for 1 to 2 transects, and had an understory of mostly wetland grasses and forbs, that would be indicative of a savanna. When two reference sites that fit the criteria were found, two 50 m transects were set up in each.

Vegetation Sampling

The 36 randomly stratified transects established for mitigation vegetation monitoring the establishment and success of planted and naturally recruited woody vegetation were used for our herbaceous vegetation sampling. The transects run 50 m and cover approximately 1% of the restoration area to ensure the response of the mitigation units is appropriately represented (Krebs, 1999). Additionally, four transects, two per site, were established in the Francis Marion National Forest. Transects were established in the pond pine savanna zone for mitigation monitoring in proximity to a hydrological monitoring well.

I used the systematic vegetation sampling method, as Elzinga et al. (1998) recommended. We conducted the vegetation survey over 4 weeks in the Summer of 2022 (June 17th- July 15th). Following the systematic method for vegetation surveying, I, with the assistance of two technicians, placed a 1 m² quadrat 5 m to the left of the 50 m transect and estimated abundance, density, biomass, and identification of the vegetation to

species, or genus when unable to identify to species, within the quadrat. A quadrat was placed every 10 m along the transect, starting at zero, totaling six quadrats per transect.

For plant identification, if an unknown was encountered, a sample was collected and run through an AI identification application, iNaturalist, for general identification and recording in the field. Once out of the field, the Google slides of species expected to be encountered were looked through. If there was still no identification, *The Guide to Common Wetland Plants of North Carolina* (Gianopulos et al., 2021) and the *Southern Wetland Flora Field Office Guide to Plant Species* (Mohlenbrock, 1987) were used to achieve identification.

To estimate abundance in the quadrats, I used percent cover, as suggested by Little (2013). The percent cover was visually assessed in the field following the Carolina Vegetative Survey (CVS) cover scale (Lee et al., 2008). Each cover code, the cover scale unit, is assigned 1 through 10, with 1 being 0.1% or very few individuals and 10 being 95–100% (Table 2.1). The purpose of the cover scale is to alleviate the difficulty of perceiving cover percentages on a linear scale. Each plant in the quadrat was written down on a data sheet and given a cover code. The same person assigned the cover codes to limit observer variation objectivity further. After the vegetation was identified and assigned a cover code, stem counts were used to measure density. Stem counts were recorded for each species within a quadrat.

The Robel Pole method was the third measurement we used in surveying vegetation. The Robel Pole method is a non-destructive method of recording aboveground biomass for herbaceous vegetation. This method is purported to be as effective and accurate as clipping methods for estimating biomass in herbaceous ecosystems and was used because destructive methods were not permitted since the research site is a monitored mitigation project. I followed Uresk and Benzon's (2007) modified Robel pole method, using a pole with alternating 1.27 cm (0.5 inches) bands of white and brown. The bands were numbered starting from 0 at the bottom of the pole. To ensure the appropriate distance and height from the ground was achieved, a 4 m string was attached to the pole with a meter stick to complete a 4 m distance and a viewpoint of 1 m above the ground for each recording. The Robel pole was placed in the center of each quadrat, and four visual obstruction recordings (VOR) were taken in each cardinal direction. These VOR measurements were recorded at each quadrat along the transect. The four VORs were then averaged for each pole station (Uresk & Benzon, 2007).

Data Analysis

All vegetation and quadrat data were input into Google Sheets before transferring it to Microsoft Excel for further data cleaning and analysis. Importance values were calculated for each species in each quadrat in Microsoft Excel (2023). The importance values were calculated from the relative density (stem count) and relative cover using the midpoint percentage of cover classification codes (McCune & Grace, 2002; Little, 2013; Lee et al., 2008). Additionally, any species considered "trace" or covered less than 0.5% of a quadrat was removed from the analysis, because these species occurred in low

abundance and our hypotheses mainly focused on the species that were dominate and the main drivers of the vegetation composition in the Brosnan wetlands and reference sites.

An Indicator Species Analysis (ISA) was conducted in R using the *vegan* and *indicspecies* programs to identify indicators in species composition differences between the headwater flats and slope wetlands (De Caceres & Legendre, 2009). An ISA is a statistical method of discovering species indicative of a particular group or site (Dufrêne & Legendre, 1997). The importance values were used to calculate the indicator values and p-value using the *multipatt* function in R. Indicator values are different from importance values in that importance values gives a species a value based on the relative cover and density across all the sites. However, the indicator values use specificity, exclusivity of a species to a particular site, and fidelity, how abundant that species is in a particular site compared to the other sites, to rank a species 0 to 1, with a high indicator value suggesting a strong association with that species and site or habitat (Dufrêne & Legendre, 1997).

To visualize whether there was a difference between the vegetative quadrats and species observed in the two wetlands at the Brosnan Forest, a Non-metric Multidimensional Scaling (NMDS) analysis was performed. A Bray-Curtis dissimilarity matrix was constructed using the importance values calculated for species by quadrat. The data were then analyzed in RStudio, using the *metaMDS* function in the *vegan* package (Version 1.4.1106; RStudio Team, 2021). Over 100 randomized runs were performed to find the best solution. The final solution was used for several plots.

Since VORs were collected per quadrat and not by species, the data could not be used in the importance value calculations and analyzed with the NMDS and ISA.

Therefore, an ANOVA was used to examine differences in estimated biomass between the Brosnan wetlands and between the reference sites and the Brosnan wetlands. Each VOR average calculated per quadrat was used to run an ANOVA in JMP Pro, with a significance value of 0.05.

Results

A total of 91 species were identified in Brosnan Forest and FMNF quadrats. There were 37 species that are classified as woody, and 56 herbaceous species. Out of 91 species 8 species were found only in FMNF (*Bignonia capreolata*, *Ruellia caroliniensis*, *Eryngium yuccifolium*, *Euthamia caroliniana*, *Monarda* spp. *Morella cerifera*, *Nekemias arborea*, *Sonchis arvensis*), and 39 species of those species found only in the Brosnan Forest (*Alnus serrulate*, *Carex glaucescens*, *Cephalanthus* spp. *Cyperus eragrostis*, *Dichanthelium* spp., *Dulichium arundinaceum*, *Echinochloa crus-galli*, *Equisetum arvense*, *Erechtites hieraciifolius*, *Erigeron* spp., *Eriocaulon aquaticum*, *Eupatorium capillifolium*, *Fimbristylis* spp., *Galium* spp., *Gaylussacia frondosa*, *Juncus acuminatus*, *Juncus scirpoides*, *Leersia* spp., *Lespedeza* spp., *Nyssa aquatica*, *Osmunda cinnamomeum*, *Persea borbonia*, *Phalaris* spp., *Pinus serotina*, *Plantago* spp., *Pluchea* spp., *Polygala lutea*, *Pterocaulon obtusifolium*, *Quercus lyrata*, *Quercus michauxii*, *Quercus phellos*, *Rhus* spp., *Saccharum* spp., *Scirpus cyperinus*, *Scleria* spp., *Solidago* spp., *Sorghum* spp., *Taxodium distichum*, *Viola* spp.). Additionally, 40 species were found to be significant from the NMDS, p value > 0.05 (Table 2.2)

Non-metric Multidimensional Scaling (NMDS)

The NMDS for the headwater flats and slope wetlands was resolved with two axes, and after performing multiple runs, the stress values ranged from 0.284 to 0.290. The best solution was achieved in one of the runs, 0.284. However, it was not repeated after a maximum run of 100. The resulting NMDS plots showed some overlap of vegetation quadrats (Figure 2.4.1; Figure 2.4.2) for the headwater slope and headwater flat wetlands. The NMDS for the headwater flats and the FMNF sites was resolved with two axes, with a stress value of 0.257 (Figure 2.5.1, 2.5.2).

Indicator Species Analysis (ISA)

The ISA identified 12 indicator species in the headwater flats and slope wetlands out of 84 species (Table 2.3). There were 7 indicator species for the headwater flats (*Juncus spp.*, *Chasmanthium laxum*, *Polygala lutea*, *Galactia volubilis*, *Magnolia virginiana*, *Carex glaucescens*, and *Eutrochium purpureum*) and 5 indicator species for the headwater slope (*Vaccinium corymbosum*, *Persea borbonia*, *Magnolia virginiana*, and *Toxicodendron radicans*). Additionally, 11 indicator species were identified for the Francis Marion National Forest reference wetlands (*Arundinaria gigantea*, *Centella erecta*, *Eutrochium purpureum*, *Ilex glabra*, *Amphicarpum spp.*, *Euthamia graminifolia*, *Nekemias arborea*, *Vitis rotundifolia*, *Andropogon glomeratus*, *Monarda spp.*, and *Aristida stricta*) (Table 2.3)

Visual Obstruction Recordings (VOR)

There was a significant difference ($p = 0.0495$) in VOR found between the headwater flats (mean = 20.88 cm; SE = 0.88), headwater slopes (mean = 21.17 cm; SE = 1.79), and the FMNF reference sites (mean = 25 cm; SE = 1.12) (Figure 2.6)

Discussion

From the results of our NMDS, it appears that the headwater slope and headwater flats are dissimilar in species composition. Although the headwater flats' wetland quadrats are nested within the headwater slope quadrats, the color points of the headwater flats seem to cluster, with few points close to the headwater slope. This result suggests that the vegetation quadrats in the headwater flats wetland differ from those within the headwater slope wetland. However, there are some similarities in species composition among certain quadrats in both Brosnan wetlands, but additional species may be present in the headwater slope wetland. The NMDS comparing the FMNF reference sites and headwater flats wetlands showed that the headwater flats quadrats seem to be more similar in species composition, and the FMNF quadrats were more similar to other FMNF quadrats. This outcome is suggested by the proximity of many headwater flats points. However, a few headwater flat quadrats are approaching the reference site quadrats.

The ISA shows a difference in species dominating the two Brosnan wetlands. Woody species (*Vaccinium corymbosum*, *Persea borbonia*, *Magnolia virginiana*, and *Toxicodendron radicans*) seem to be establishing and dominating more quickly in the headwater slope wetland, than in the headwater flat wetland, which is primarily

herbaceous (*Juncus spp.*, *Chasmanthium laxum*, *Polygala lutea*, *Galactia volubilis*, *Carex glaucescens*, *Eutrochium purpureum*).

The headwater slope wetland is differentiated mainly by shade-tolerant, mesic shrubs (*Vaccinium corymbosum*, *Persea borbonia*, *Magnolia virginiana*) that typically are found in forested wetlands and a vine (*Toxicodendron radicans*) that can grow in a wide range of environments, tolerating xeric and mesic conditions, but is common in forested and scrub-shrub wetlands. Conversely, the headwater flats wetland is differentiated by mostly mesic, herbaceous species that prefer open canopy, but can be shade tolerant (*Juncus spp.*, *Carex glaucescens*, *Chasmanthium laxum*, *Polygala lutea*, and *Eutrochium purpureum*). The headwater flats and slope wetlands share one indicator species, *Magnolia virginiana*. This finding is not entirely surprising as this species can quickly recolonize after a disturbance and tolerates full sun to shade (Jones et al. 2000). Additionally, this species can be found in forested wetlands and savannas (Jones et al. 2000).

The species that dominate the reference sites in Francis Marion National Forest (*Arundinaria gigantea*, *Centella erecta*, *Eutrochium purpureum*, *Ilex glabra*, *Amphicarpum spp.*, *Euthamia graminifolia*, *Nekemias arborea*, *Vitis rotundifolia*, *Andropogon glomeratus*, *Monarda spp.*, *Aristida stricta*), indicate primarily herbaceous species, with some shrubs and woody vines, that mostly prefer open canopy, mesic conditions, but can be shade tolerant. Some of these species are indicative of a pine-cypress savanna (*Ilex glabra* and *Amphicarpum spp.*). Other facultative wetland species are present (*Arundinaria gigantea*, *Centella erecta*, *Eutrochium purpureum*, *Euthamia*

graminifolia, *Andropogon glomeratus*, *Monarda* spp.), as well as a facultative species that is tolerant of a range of moisture (*Aristida stricta*). Although the headwater flats and reference sites only shared one indicator species (*Eutrochium purpureum*), the conditions and the species that dominate them are potentially more similar than those indicative of the headwater slope wetland.

Potential limitations of our research could be the subjectivity of our recordings for VOR estimating biomass and percent cover. However, since our research site is still being monitored for mitigation purposes, a non-destructive recording of biomass was needed. Although percent cover can be subjective in the field, only one person recorded percent cover to attempt to keep percent cover recordings consistent in the field. Additionally, the NMDS did not repeat the best solution before reaching the stopping criteria of 100, and both stress value ranges are on the high end of generally acceptable values (Clarke, 1993). However, this could be due to not having covariates to assist the NMDS ordinate directionality.

Recovery rates and the trajectory at which a created or restored wetland reaches a natural reference or pre-disturbance conditions can be challenging to determine, can vary based upon a multitude of factors (Moreno-Mateos et al., 2012), and could result in a novel ecosystem (Hobbs et al., 2006; Moreno-Mateos et al., 2012; Stefanik & Mitsch, 2012). Therefore, it can take years to restore ecosystem functions and services if they ever do. Our research indicates that the recovery rate can vary within a wetland. The results of our NMDS suggest that a few of the vegetation quadrats in the headwater flats are reaching the species composition of the quadrats in the FMNF reference sites.

Although the headwater slope wetland is considered a different wetland type, a few quadrats are also reaching the FMNF reference. However, most of the vegetation quadrats in the Brosnan wetlands are more similar to one another than the reference sites. This finding could suggest variability in the disturbance and other physical conditions of the wetland could impact the recovery rate in some areas of the headwater flats and headwater slope wetlands. Additionally, there was no statistically significant difference for biomass between the restored and reference sites. The lack of a difference in VORs between the restored and reference sites could potentially show that some environmental services, such as biomass production, can return quicker than others, such as those associated with soil organic matter.

Conclusions

From our NMDS and ISA, we observed that the vegetative community differs between headwater slope and headwater flats. However, when comparing the Brosnan wetlands and the FMNF reference sites, there seems to be little difference in species composition between the Brosnan wetlands, aside from a few headwater flats quadrat points, which seemed to be more dissimilar, based on the proximity to other quadrat points. A few quadrat points from both the headwater slopes and headwater flats were approaching the species composition of the FMNF reference sites. Furthermore, the species that dominate each wetland are different. Herbaceous species are more prevalent and indicative of the headwater flats and reference sites than woody species. These differences could be due to hydrological and topographical differences between the wetlands. Therefore, learning the hydrological differences and taking soil organic matter

depths at each quadrat could be valuable in further understanding differences in vegetation between the two Brosnan wetlands and possibly assisting the NMDS in ordinating. Additionally, further research should be conducted on the progression of the vegetation communities among the sites and sampling in reference sites for the headwater slope to assess the differences.

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Tables

Table 2.1. Cover codes with each scale unit used to analyze percent cover as recommended by the Carolina Vegetation Survey (CVS), with “trace” being a cover code of 1. The third column is the median used for analysis.

Cover Code	Percentage Range	Median used for Analysis
1	(0.1%)	0
2	0—1%	0.5
3	1—2%	1.5
4	2—5%	3.5
5	5—10%	7.5
6	10—25%	17.5
7	25—50%	37.5
8	50—75%	62.5
9	75—95%	85
10	95—100%	97.5

Table 2.2. Significant species found from the results of the NMDS using species importance values calculated using cover and density of a species per quadrat, from the data collected in the Summer (July-June) 2022, at the Brosnan Forest, Dorchester, SC.

Species	p-value
<i>Acer rubrum</i>	0.005
<i>Amphicarpum amphicarpon</i>	0.003
<i>Andropogon glomeratus</i>	0.007
<i>Arundinaria gigantea</i>	0.001
<i>Centella erecta</i>	0.009
<i>Chasmanthium laxum</i>	0.032
<i>Clethra alnifolia</i>	0.001
<i>Cyperus eragrostis</i>	0.028
<i>Cyperus</i> spp.	0.001
<i>Dulichium arundinaceum</i>	0.002
<i>Equisetum arvense</i>	0.005
<i>Erechtites hieraciifolius</i>	0.001
<i>Eriocaulon purpureum</i>	0.012
<i>Eriocaulon</i> spp.	0.025
<i>Euthamia caroliniana</i>	0.045
<i>Euthamia graminifolia</i>	0.011
<i>Gaylussacia frondosa</i>	0.009
<i>Juncus</i> spp.	0.001
<i>Liquidambar styraciflua</i>	0.029
<i>Lyonia lucida</i>	0.009
<i>Morella cerifera</i>	0.003
<i>Myrica caroliniensis</i>	0.031
<i>Nekemias arborea</i>	0.017
<i>Panicum</i> spp.	0.001
<i>Persea borbonia</i>	0.026
<i>Pteridium aquilinum</i>	0.001
<i>Quercus michauxii</i>	0.043
<i>Quercus</i> spp.	0.04
<i>Rhexia</i> spp.	0.025
<i>Rhynchospora</i> spp.	0.001
<i>Rubus pensilvanicus</i>	0.001
<i>Ruellia carolinesis</i>	0.045
<i>Scirpus cyperinus</i>	0.007
<i>Smilax</i> spp.	0.02

<i>Sonchis.arvensis</i>	0.035
<i>Toxicodendron.radicans</i>	0.042
<i>Vaccinium.corymbosum</i>	0.012
<i>Vitis.rotundifolia</i>	0.026
<i>Woodwardia.virginica</i>	0.001
<i>Xyris spp.</i>	0.001

Table 2.3. Wetland type, species, indicator value (IV), and results from the ISA (indicator species analysis) using species importance values, calculated using cover and density of a species per quadrat, from the data collected in the Summer (July-June) 2022, at the Brosnan Forest, Dorchester, SC. To determine if there is a difference in vegetation between the two wetlands, headwater slope and flats.

Wetland	Species	IV	p
Headwater Slope			
	<i>Vaccinium corymbosum</i>	0.426	0.005**
	<i>Persea borbonia</i>	0.371	0.01**
	<i>Magnolia virginiana</i>	0.299	0.01**
	<i>Toxicodendron radicans</i>	0.232	0.02*
Headwater Flats			
	<i>Juncus spp.</i>	0.452	0.005**
	<i>Chasmanthium laxum</i>	0.41	0.01**
	<i>Polygala lutea</i>	0.306	0.035*
	<i>Galatica volubilis</i>	0.251	0.005**
	<i>Magnolia virginiana</i>	0.251	0.01**
	<i>Carex glaucescens</i>	0.246	0.045*
	<i>Eutrochium purpureum</i>	0.232	0.005**
Francis Marion National Forest Reference Sites			

<i>Arundinaria gigantea</i>	0.700	0.005**
<i>Centella erecta</i>	0.618	0.005**
<i>Eutrochium purpureum</i>	0.508	0.005**
<i>Ilex glabra</i>	0.504	0.005*
<i>Amphicarpum spp.</i>	0.356	0.035*
<i>Euthamia graminifolia</i>	0.408	0.005**
<i>Nekemias arborea</i>	0.408	0.005**
<i>Vitis rotundifolia</i>	0.408	0.005**
<i>Andropogon glomeratus</i>	0.333	0.010**
<i>Xyris spp.</i>	0.332	0.050*
<i>Monarda spp.</i>	0.289	0.025*
<i>Aristida stricta</i>	0.224	0.045 *

Figures



Figure 2.1. The inset map helps to locate Francis Marion National Forest (FMNF) reference sites and Broonan forests within South Carolina, USA. The main map shows the proximity of the FMNF sites to each other, a main road that runs through FMNF, Steed Creek Rd., as well as other possible disturbances, such as the powerline corridor parallel near the first site and the unpaved forest road near the second site. Additionally, the orange circular points indicate the established vegetation transects, two transects at the first site and one at the second site.

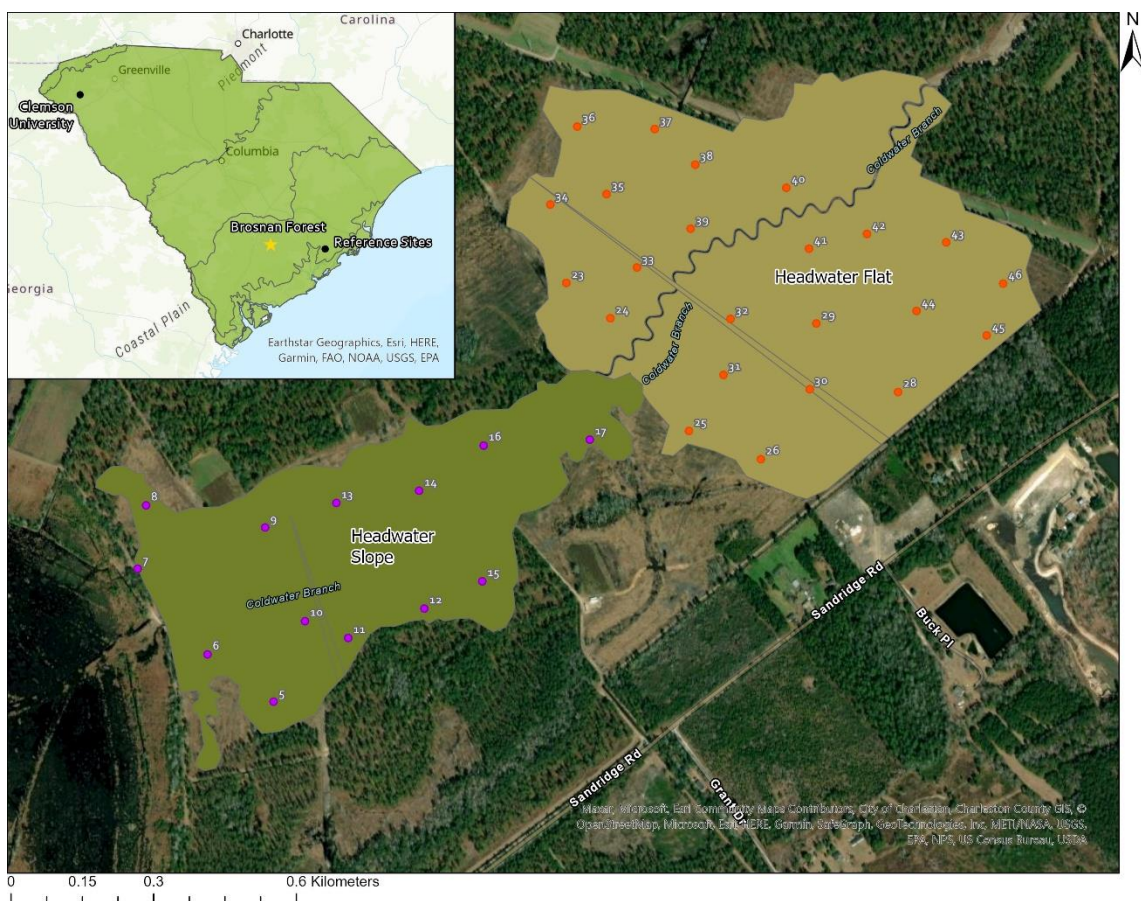


Figure 2.3. The inset helps to locate Brosnan Forest and the reference sites within South Carolina. The locations of the 36 (labeled 5–17, 23–46) vegetation transects at Brosnan Forest are reflected in the main map, with each point representing the start point of the transect in June and July of 2022, and February of 2023. Vegetation transects 1–4 and 18–22 were removed from the data analysis, as these are located within a different wetland type from the headwater flats and slope wetlands.

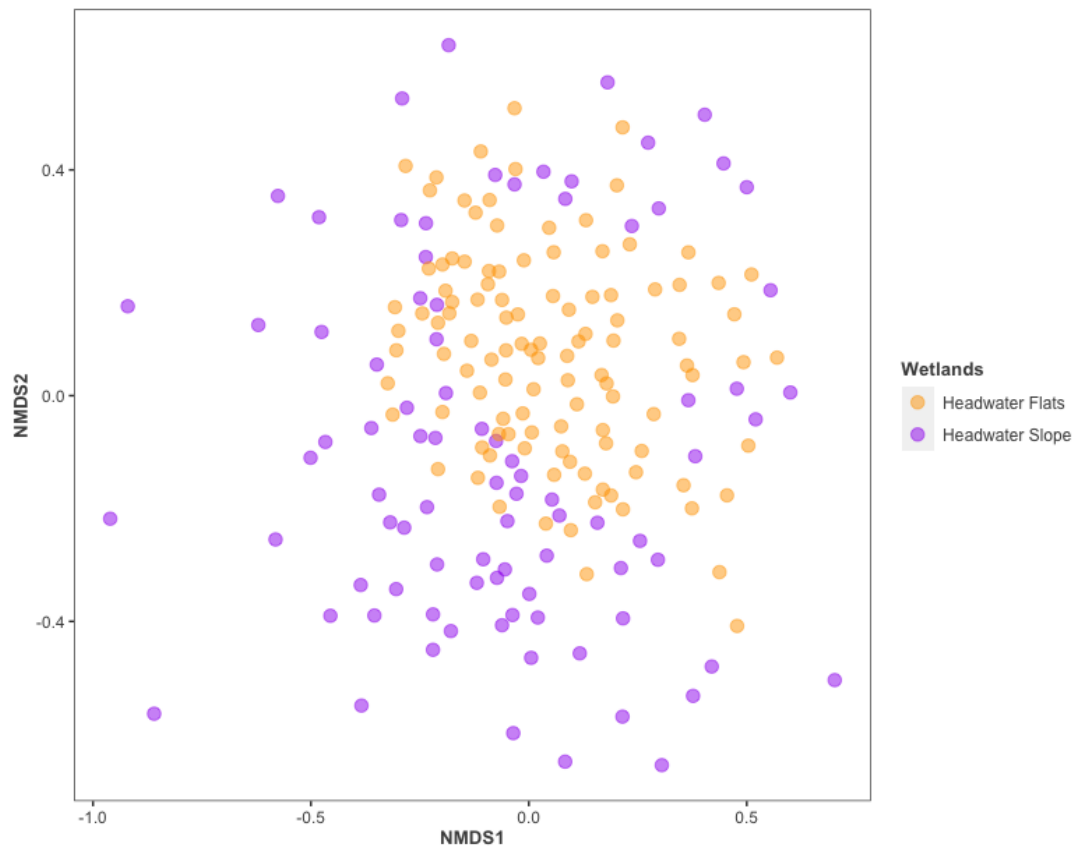


Figure 2.4.1. Non-metric Multidimensional Analysis (NMDS) showing dissimilarity among vegetation found headwater slope (purple) and the headwater flat wetlands (orange) in the Brosnan Forest, Dorchester, County, South Carolina, USA, and the Francis Marion National Forest, Huger, SC, USA indicated by point color overlap. Data was collected in the months of June and July of 2022. Species that were in less than 1% of the plot were excluded from the analysis.

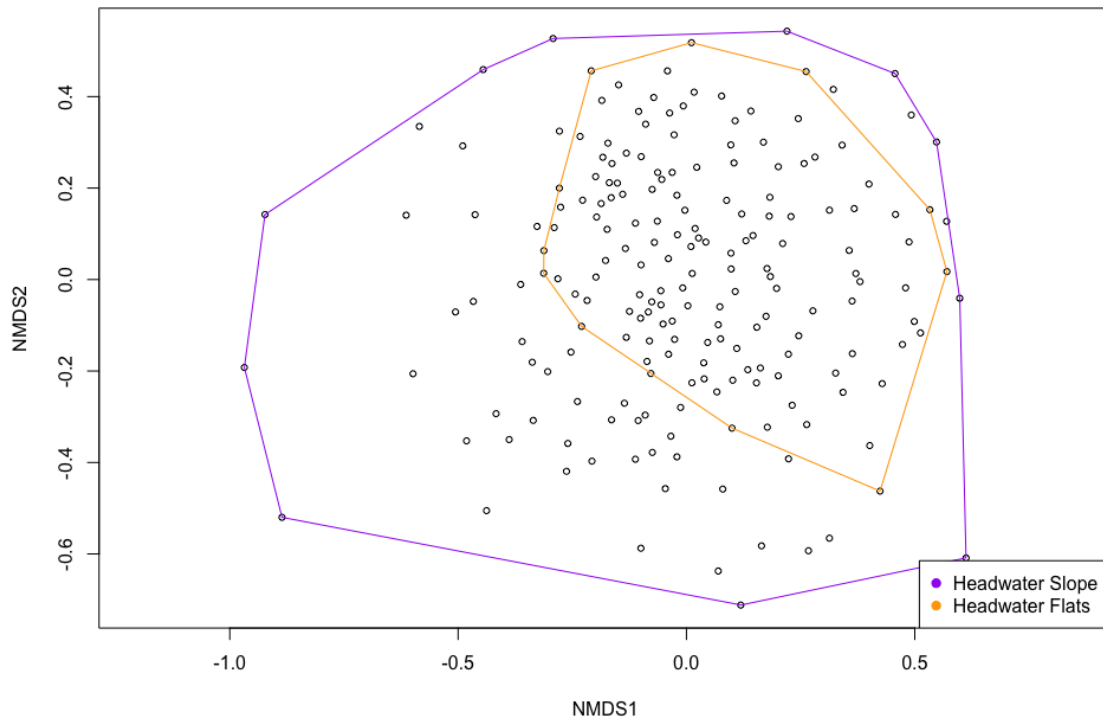


Figure 2.4.2. Non-metric Multidimensional Analysis (NMDS) showing similarity among vegetation found in the headwater slope (blue) and the headwater flat wetlands (orange) in the Brosnan Forest, Dorchester, County, South Carolina, USA, indicated by polygon overlap. Data were collected in June and July of 2022. Species in less than 1% of the plot were excluded from the analysis.

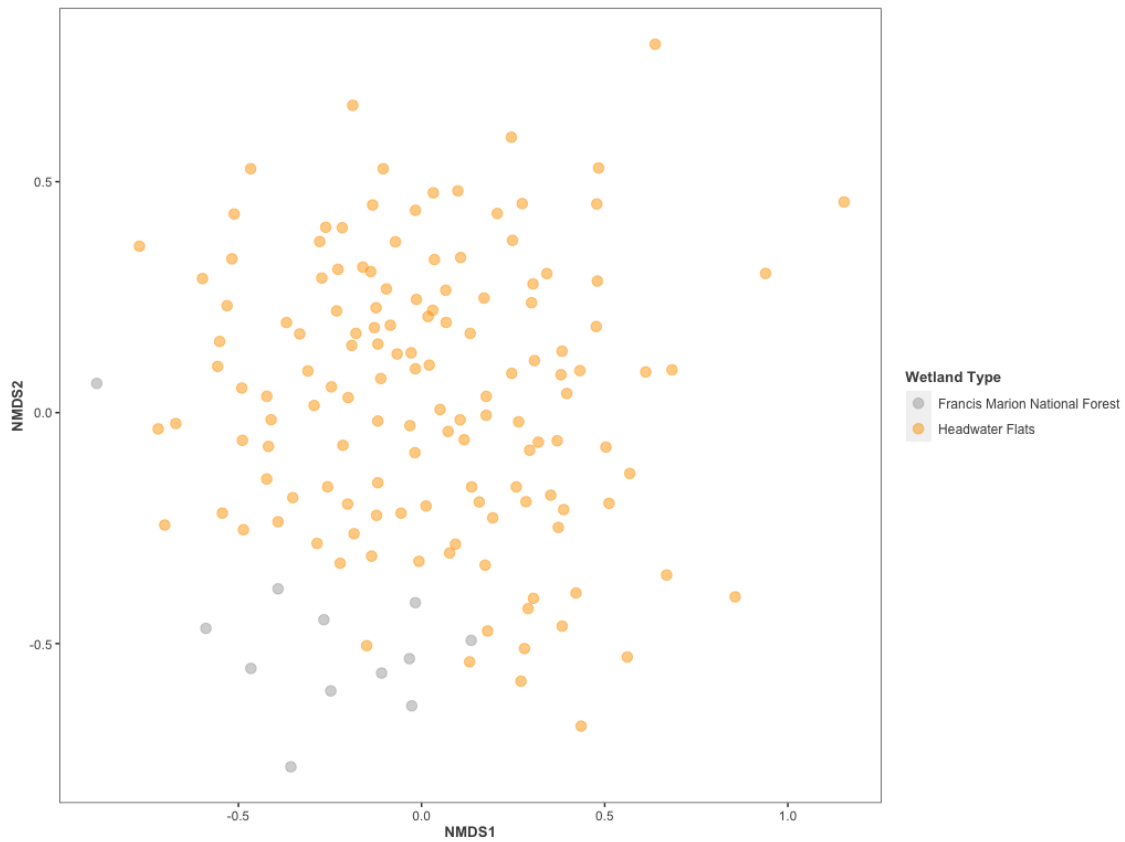


Figure 2.5.1. Non-metric Multidimensional Analysis (NMDS) showing similarity among some of the quadrats found in the Francis Marion National Forest reference sites (grey) and the headwater flat wetlands (orange) in the Brosnan Forest, Dorchester, County, South Carolina, USA, and the Francis Marion National Forest, Huger, SC, USA indicated by point color overlap. Data were collected in June and July of 2022. Species in less than 1% of the plot were excluded from the analysis.

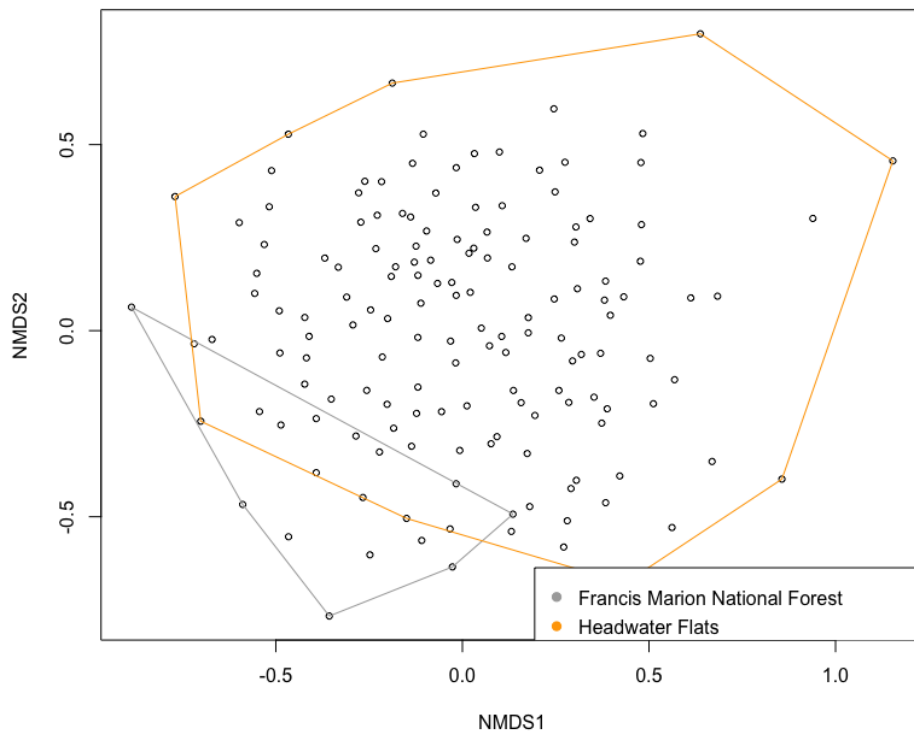


Figure 2.5.2 Non-metric Multidimensional Analysis (NMDS) shows similarity among a few vegetation quadrats the Headwater Flats (orange) and the Francis Marion National Forest reference sites (grey) in the Brosnan Forest, Dorchester, County, South Carolina, USA, and the Francis Marion National Forest, Huger, SC, USA indicated by polygon overlap. Data were collected in June and July of 2022. Species in less than 1% of the plot were excluded from the analysis.

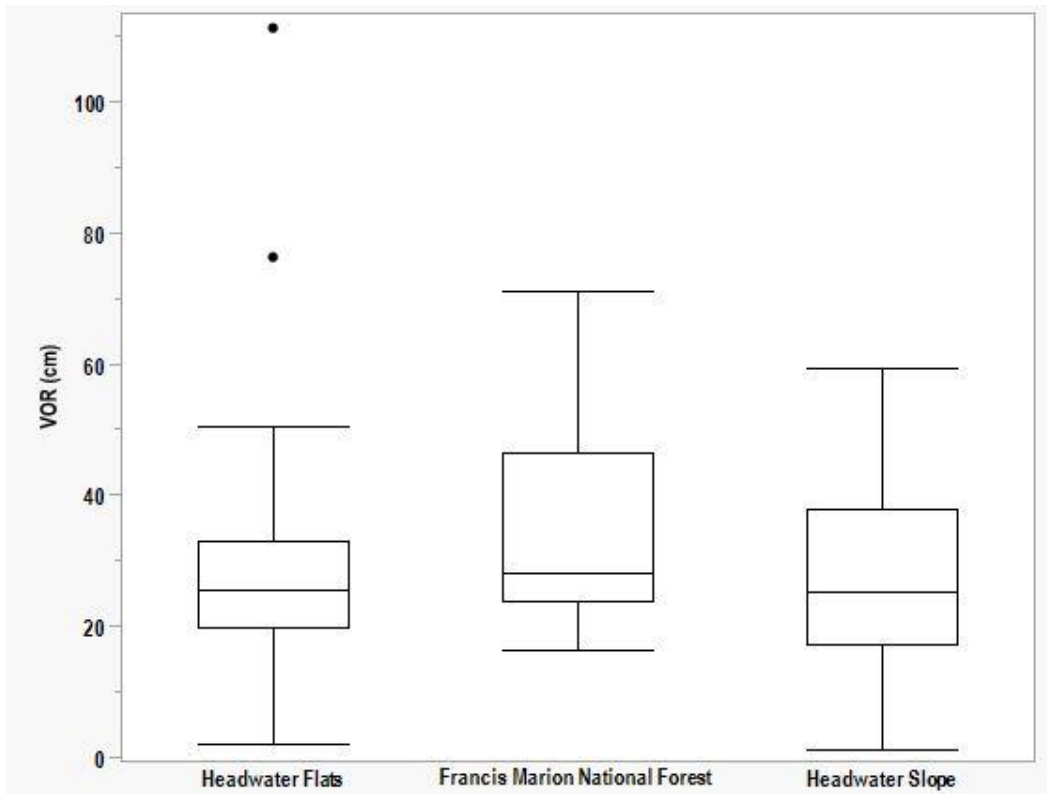


Figure 2.6. Distribution of variance using the average Robel pole visual obstruction recordings (VOR), showing a significant difference ($p = 0.495$) in VOR between the Brosnan wetlands and the reference sites at the Francis Marion National Forest.

CHAPTER THREE

COMPARISON OF SOIL ORGANIC MATTER DEPTH AMONG REFERENCE SITES AND TWO RESTORED WETLANDS IN THE COASTAL PLAIN OF SOUTH CAROLINA

Abstract

Wetland loss and degradation from agriculture, urbanization, forestry, and mining is a global issue. South Carolina has lost over 27% of its wetlands. This historical wetland loss and accelerated loss of coastal wetlands due to climatic changes make restoring wetlands a critical issue for the state. However, wetland restoration can be tedious, especially with a site that historically had deep organic matter. Our study aimed to determine the differences in soil organic matter depth variability between reference sites and restoration sites in the coastal plain of South Carolina. Additionally, we sought to determine whether historic wetland type impacted soil organic matter depth between restoration sites after 70 years of the same disturbance. We created cross-wetland transects using Google Maps and current GIS Satellite Data and recorded soil organic matter depth. We analyzed the soil organic matter depth point recordings using ANOVAs and Welch's t-test and created soil profile graphs to visually represent the soil organic matter profiles of each wetland. The restored wetland's organic soil depths were more variable than the reference wetlands. The reference site's soil organic matter depth was two times greater than the restored wetlands, and the restored wetland's organic soil depths were not significantly different. Knowing a wetland's soil organic matter profile

can be important when planning a restoration, as it can give insights into how the plant community may react to returned hydrology.

Introduction

In the United States, wetlands have historically been threatened and vulnerable ecosystems, with over half of natural wetlands being lost post European settlement (Dahl, 1990; Mitsch & Gosselink, 2007). Much of this historic loss for inland wetlands can be attributed to agriculture, as well as urban development, forestry, and mining (Dahl, 1990; Mitsch & Gosselink, 2007). South Carolina alone has lost over 27% of its wetlands, with some estimates closer to over 30% (Dahl, 1990; Dahl, 1999). This historic loss can be detrimental to some areas, because of the essential ecosystem services wetlands can provide such as flood control (Tang & Kavvas, 2020), water storage (Hubbard and Linder, 1986; Lane and D'Amico, 2010), wildlife habitat (Keddy, 2010; Williams and Dodd, 1978), and water filtration (Burkett & Kusler, 2007; Dordio et al., 2008). Additionally, with current projections from the Intergovernmental Panel on Climate Change (IPCC, 2022) predicting future wetter and warmer conditions for some regions, restoring lost or impacted wetlands can be essential to combating the intense effects of climate change. However, restoring wetlands can be a long, tedious process, especially wetlands that historically had deep organic soils, as it can take decades to build up only a few centimeters (Anderson et al., 2005; Ballentine & Schneider, 2009; Anderson & Mitsch, 2006), and some wetlands never recover their organic soils (Bruland & Richardson, 2006; Ahn & Jones, 2013; Shaffer & Ernst, 1999).

Wetland soils, known as hydric soils, are formed during anaerobic conditions due to saturation near the surface, causing a reduction in soil oxidation-reduction potential, followed by denitrification, reduction of iron, and sulfate (NRCS, 1998; Vasilas & Vasilas, 2013; Pezeshki & DeLaune, 2012; USDA & NRCS, 2018). Hydric soils are essential for maintaining many vital functions of a wetland (e.g., water storage, groundwater recharge, groundwater discharge, water quality, and wildlife habitat) (Baker et al., 2008). There are two types of hydric soils: hydric mineral and organic (Vepraskas & Craft, 2016). In this study, we will focus on organic hydric soils. Organic soil is formed by the accumulation of organic plant material in various stages of decomposition due to anoxic conditions from stagnant or poorly drained conditions (Mitsch & Gosselink, 2007; Vepraskas and Craft, 2016). Organic soils are generally classified as soils with organic matter content greater than 20 to 35% from 0 to 35.56 – 40.64 cm (Mitsch & Gosselink, 2007; Baker et al., 2008; Vepraskas & Craft, 2016).

Many wetlands are underlain by a restrictive feature which is a layer that impedes the flow of water and air and restricts root growth chemically, physically, or thermally (USDA, 2019). The depth of the restrictive feature affects vegetation, water-holding capacity, and nutrient availability (Rajakaruna & Boyd, 2008). Some examples of restrictive features are bedrock, cemented layers, frozen layers, and dense layers (USDA, 2019). Created and restored wetlands tend to have lower SOM than natural wetlands of the same or similar wetland type (Bishel-Machung et al., 1996; Burland & Richardson, 2006; Campbell et al., 2002; Galatowitsch & Van der Valk, 1996; Shaffer & Ernst, 1999). However, there has been little focus on comparing SOM in wetlands with the

same disturbance and age post-restoration but different historic wetland types and topography to see how these differences impact the remaining SOM. Investigating how the SOM varies through space as you cross through a restored wetland is particularly interesting.

Our research aimed to determine if there was a significant difference between the restored wetlands and reference sites in South Carolina's Coastal Plain. We were specifically interested in determining if wetland type affected the degree of organic soil loss, percent organic matter (OM%), and percent nitrogen (N%) after 70 years of disturbance and could be used in determining and differentiating them. Furthermore, we hypothesize that the historical type of wetland might impact how variable the organic matter would be after disturbance.

Methods

Soil Organic Matter Depth/ Site Description

Soil organic matter depth data were collected in June and July 2022. Before data collection, cross-sections of the two wetlands at Brosnan Forest were mapped using Google Maps, starting at the edge of the wetland (where the wetland met the longleaf pine (*Pinus palustris*) upland) and ending 10–20 m in the upland portion or, in some cases, at the road or property edge (Figure 3.2). The Francis Marion National Forest (FMNF) soil transects were placed with the start point beginning in the cypress savanna, going through the pond pine savanna, and ending 10–20 m into the upland (Figure 3.3). A

detailed site description of both the Brosnan Forest wetlands, the headwater flats and slopes, and the FMNF reference sites can be found in Chapters 1 and 2.

The GPS application, Avenza, was used to locate the coordinates of the starting point and to record the coordinates of each soil depth location. Every 10 m along the transect, coordinates and organic soil depth were recorded. Soil depths were measured by inserting a 1.5 m soil probe in the soil until it stopped, either by reaching a mineral layer or restrictive feature, then placing a finger flush with the ground and measuring the length of the pole to the nearest 5 cm (Young & Duever, 1997). When needed, a 1.5 m extension was used. This methodology was repeated along the length of the transect.

Soil Collection

Soil samples were collected in the last week of July 2022 at the Brosnan Forest wetland sites to examine the differences in organic matter and nitrogen percentage between the headwater slope and flats wetland. The soil samples were collected near established randomly located vegetation mitigation monitoring transects, with 10 samples collected in the flats and 10 in the sloped headwater wetlands. For each of the 20 samples, a trowel was used to collect 12 soil subsamples (Clemson University Ag Service Lab, n.d.). at the start and end points of the mitigation vegetation monitoring transects to ensure that we were not digging within the boundaries of the study area used for mitigation monitoring. Each soil subsample was taken at approximately 10-20 cm from the surface. The 12 subsamples were placed in a clean plastic bucket and mixed thoroughly, then 1 to 2 cups were placed in a plastic bag to create one sample for that

transect. This process was then repeated for a total of 20 samples. Samples were allowed to air dry before transporting them to the Clemson Agricultural Service Soil Lab for nitrogen and organic matter percentage analysis.

Data Analysis

Soil depth data were analyzed using Microsoft Excel, the statistical software JMP Pro 16 (JMP Pro 16, 2023). Prior to analysis in JMP Pro, depths taken in logging roads, uplands, and stream construction zones were removed using field notes and aerial photos with the soil depth data points overlaid. Soil depth profile graphs were created of each transect with distance (m) along a transect as the X-intercept and soil depth (m) as the Y-intercept to visualize variability in soil depths between the flat and sloped headwater wetland and the reference sites at the Francis Marion National Forest. To determine whether this apparent optical variability was statistically significant, we divided the data points from the transects in the headwater flats (214 data points), headwater slope wetlands (307 data points), and Francis Marion National Forest wetlands (63 data points) and tested for variance over the entire dataset, and divided the data into percentiles to test for variance using JMP Pro. We repeated the analysis for the transects from the flats and sloped headwater wetlands. JMP Pro automatically runs an O'Brien, Brown-Forsythe, Levene, Bartlett, and 2-sided F Test to test for equal variances. If variances were unequal, we used a Welch Analysis of Variance (ANOVA) test to analyze if a difference was present between sites. If variances were equal, we used an ANOVA and t-test, using an alpha level of 0.05 to test for significance.

To further analyze whether there was statistical variability in soil depth between the flats and sloped headwater wetland, we divided the entire data set (excluding the transects from the reference sites and the data points taken in logging and stream construction zones) into percentiles: 10th (including the minimum value), 25th, 50th, 75th, and 90th (including the maximum value). We then ran the same analyses for equal variances in JMP and, depending on the results, analyzed for variance with a Welch ANOVA test, ANOVA, or t-test. The percentiles were used to see if there was a difference between the Brosnan wetlands at different depth ranges, to identify whether one wetland had a higher number of lower depths and vice versa.

Results

The soil depth graphs showed visual variability in organic soil depth along most transects within the headwater slope and flats wetlands (Figures 3.4.1 and 3.4.2). However, the soil organic matter in the headwater flats wetland was significantly deeper ($p = 0.0318$) than in the headwater slope (Figure 3.4.2). There was noticeably less variation in organic soil depth at the reference sites in the Francis Marion National Forest, with deeper soils in the cypress zone and somewhat shallower, but still consistently deep, soil organic matter through the pond pine savanna zone, then a gradual decrease in soil organic matter closer to the upland (Figures 3.4.3).

Organic matter soil depth at Francis Marion National Forest (mean depth = 72.3 cm; SE = 0.046) was significantly deeper ($p = <0.0001$) than at Brosnan, Headwater Flats (mean depth = 37.5 cm; SE = 0.023) and Headwater Slope (mean depth = 32 cm; SE =

0.022) (Figure 3.5.1). There was also a significant difference ($p = 0.0318$) overall between the organic matter soil depths of the headwater slope and headwater flats wetlands at Brosnan, with the headwater flats wetland organic matter depth being significantly deeper (Figure 3.5.2). However, when the data points from the soil transects in Brosnan were divided into percentiles and tested for variance, there was no significant difference between the two wetlands at all percentiles: 10th ($p = 0.239$), 25th ($p = 0.201$), 50th ($p = 0.489$), 75th ($p = 0.8629$), and 90th ($p = 0.316$). Additionally, after removing the soil organic matter depths that were taken in more heavily disturbed areas (logging decks, logging roads, and stream construction zones) there was no significant difference ($p = 0.344$) between the headwater slope (mean depth = 36.5 cm; SE = 0.0232) and headwater flats (mean depth = 29.5 cm; SE = 0.0196).

A significant difference was not found for the percent of organic matter ($p = 0.86$) from the soil samples collected in the headwater slope (mean = 12.67; SE = 1.86) and the headwater flats (mean = 12.18; SE = 2.11) wetlands (Figures 3.6.3). Additionally, no significant difference was found for the percent of nitrogen ($p = 0.23$) in the headwater slope (mean = 0.39; SE = 0.077) and the headwater flat (mean = 0.54; SE = 0.088) wetlands (Figure 3.5.4).

Discussion

Based on the results, the reference wetlands in the Francis Marion National Forest have significantly deeper and less variable organic matter soil depth than the restored wetlands in the Brosnan Forest, as seen by Figure 6.1. This can be observed by the

number of outliers in Figure 3.5.1, and the SOM depth graphs (Figures 3.4.1—3.5.3). The average organic matter soil depth in Francis Marion National Forest is almost double that of the transects at the Brosnan Forest. This is to be expected since the restored wetlands were heavily disturbed due to draining, ditching, and establishment and management of a loblolly plantation for over 70 years.

The methodology used for determining soil organic matter depth is not widely discussed in the literature, and the use of transects to create a cross-section (profile) of the wetland's soil depths appears fairly novel. The same results (determining organic soil depths) may have been achieved using a statistically significant number of random points across reference and restored wetlands. However, using transects allowed us to visualize the variability in SOM across the wetland. There is also some subjectivity about when the mineral or restrictive layer was reached. Another possible limitation was insufficient reference sites for the flat headwater wetland. The shortage of references for the flat headwater wetland can be attributed to these flat, shallow, herbaceous coastal plain wetlands being historically lost and nearly extinct ecosystem types from fire suppression and agricultural use (Platt, 1999), as was done at the Brosnan Forest for a pine plantation. Additionally, none of the OM% recordings we took were in the typical range of 20% to 35% of organic hydric soils, which could indicate that much of the soil in the Brosnan wetlands is unconsolidated. Despite these limitations, the methodology proved effective in determining soil organic matter depth in the reference sites and demonstrating a significant difference between reference and restored wetlands. This methodology can

still prove to be effective in areas with known organic hydric soils to create a baseline for continued monitoring and to visualize the soil profile of a wetland.

It is known that during wetland impacts, there can be dramatic losses of soil carbon, which can further degrade associated ecosystem services (Mikhailova et al., 2021), and mitigating these losses can be a long, tedious process (Anderson et al., 2005; Ballentine & Schneider, 2009; Anderson and Mitsch, 2006;) and the site may never reach the historical state (Bruland & Richardson, 2006; Ahn & Jones, 2013; Shaffer & Ernst, 1999). However, the variability and distribution of that loss have not been readily discussed in the literature. Nor how the uneven distribution of loss can impact restoration success. It is evident that disturbance associated with draining, ditching, and planting loblolly plantations at the Brosnan Forest has resulted in SOM loss compared to the reference sites at Francis Marion National Forest and that SOM loss is not uniform across the Brosnan wetlands, with highly variable SOM depths seen from the soil profile figures (Figures 3.4.1 & 3.4.2) and the number of outliers from the boxplot in Figure 3.5.1. Therefore, even with the associated difficulties of soil restoration, the organic matter depth and soil profile should be considered in restoration success assessment. A reduction in soil organic matter can impact restoration or mitigation success by affecting the ability to reach vegetation or hydrological goals without the soils matching that of historical systems. The methodology used for this study identified the variability in soil organic matter loss that can occur after long-term disturbance and how the variability differs from the reference sites. Thus, when carbon sequestration is a mitigation goal, the

methodology used here could help identify a pre-restoration SOM profile that could be tracked through time to see how much SOM accumulates in the wetland.

Further research could compare the results of using SOM depth transects across the wetland versus random SOM depth points. Additionally, comparing the vegetation or hydrology where the SOM depths are taken could further our knowledge of how SOM losses due to disturbances can impact the hydrology and vegetation on a spatial and temporal scale. It could also be interesting and beneficial for mitigation regulators and managers to assess any changes in SOM depth and variability through time to see if there were any significant changes or differences between the Brosnan wetlands (Shaffer & Ernst, 1999; Anderson et al., 2005; Anderson & Mitsch, 2006; Bruland & Richardson, 2006; Ballentine & Schneider, 2009; Ahn & Jones, 2013).

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Figures



Figure 3.1. Aerial imagery of the Brosnan Forest and wetland sites in relation to highway 78 and Dorchester, SC, USA.

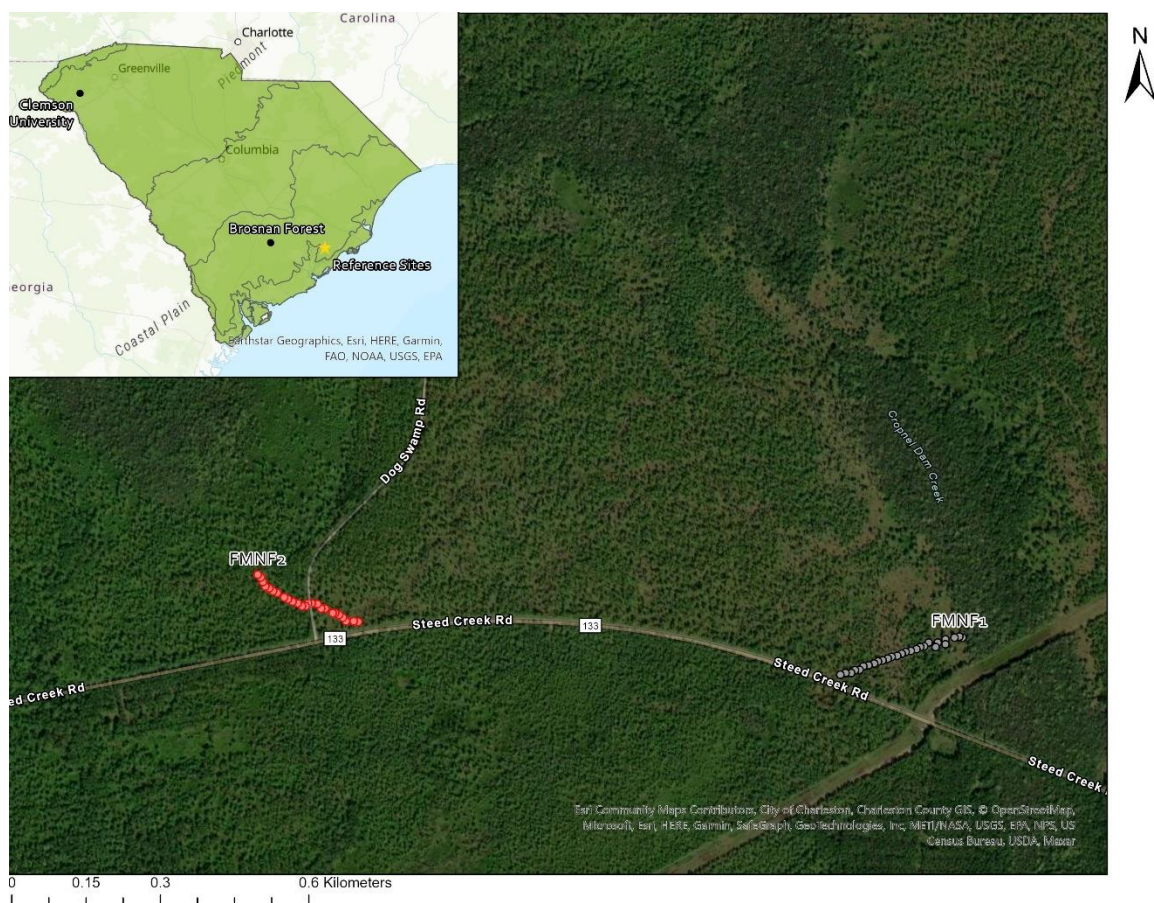


Figure 3.2. The inset map shows the proximity of the FMNF sites to each other, the city of Huger, SC, a main road that runs through FMNF, Steed Creek Rd., as well as other possible disturbances, such as the unpaved forest road in the first site and the powerline corridor parallel to the second site. The two main maps reflect the two soil organic matter transect locations, with each point representing a depth taken along the transect in June and July of 2022 and February of 2023.

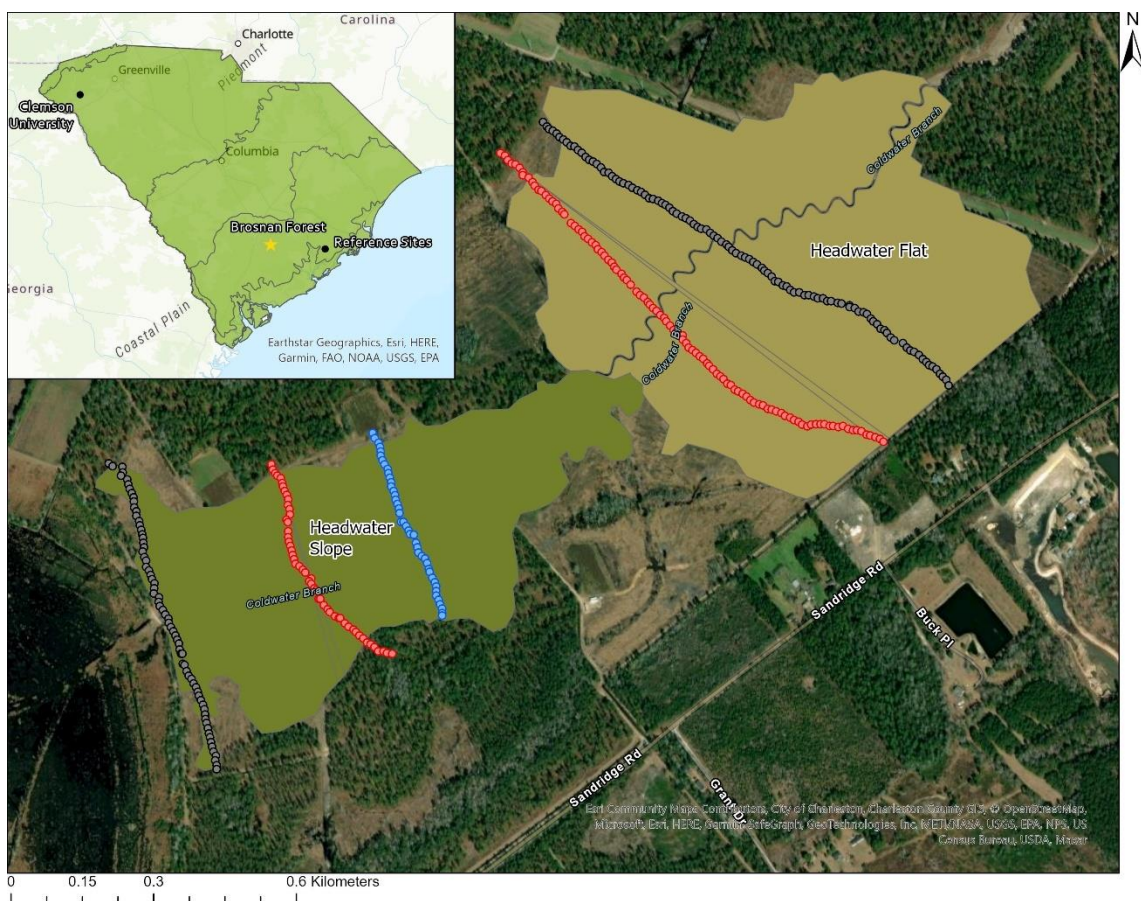


Figure 3.3. The inset helps to locate Brosnan Forest and the reference sites within South Carolina. The locations of the five soil organic matter transects at Brosnan Forest are reflected in the main map, with each point representing a depth taken along the transect in June and July of 2022 and February of 2023.

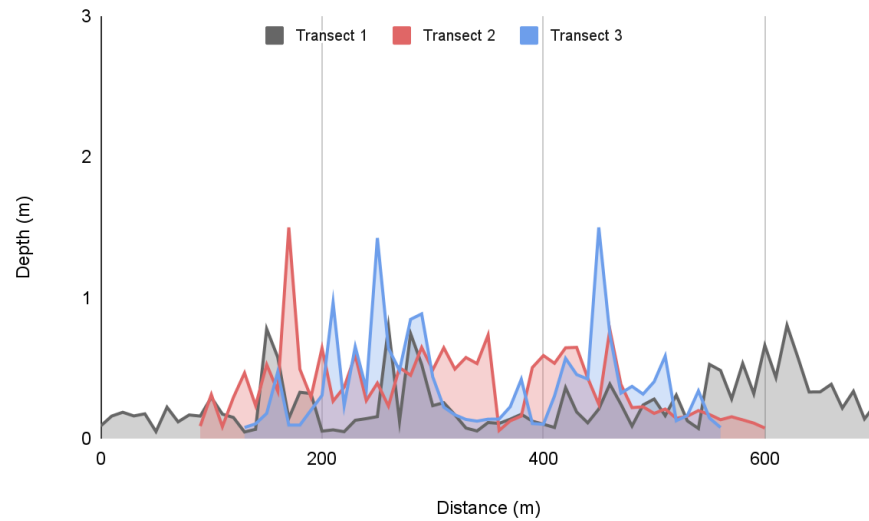


Figure 3.4.1. The Brosnan Forest headwater slope wetland SOM transects displaying a variable SOM profile, with deep SOM recordings of over 1.5 m and shallow readings of less than 0.5m along the transect.

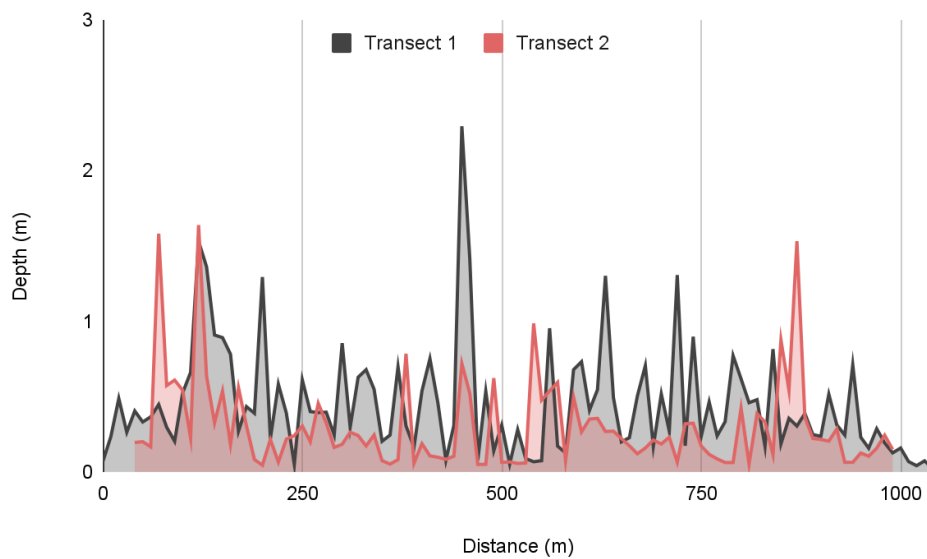


Figure 3.4.2. The Brosnan Forest headwater flat wetland SOM transects displaying a variable SOM profile, with deep SOM recordings of over 1.5 m and shallow readings of less than 0.5m along the transect.

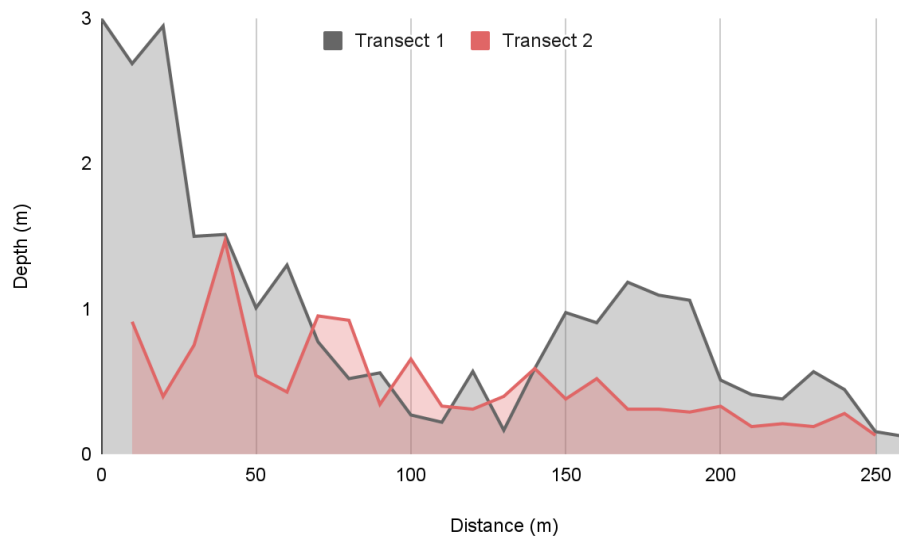


Figure 3.4.3. The two Francis Marion National Forest (FMNF) reference SOM transects, displaying deep SOM in the cypress savanna zone at the beginning of the transect and a gradual decline in SOM through the pond savanna until reaching the upland.

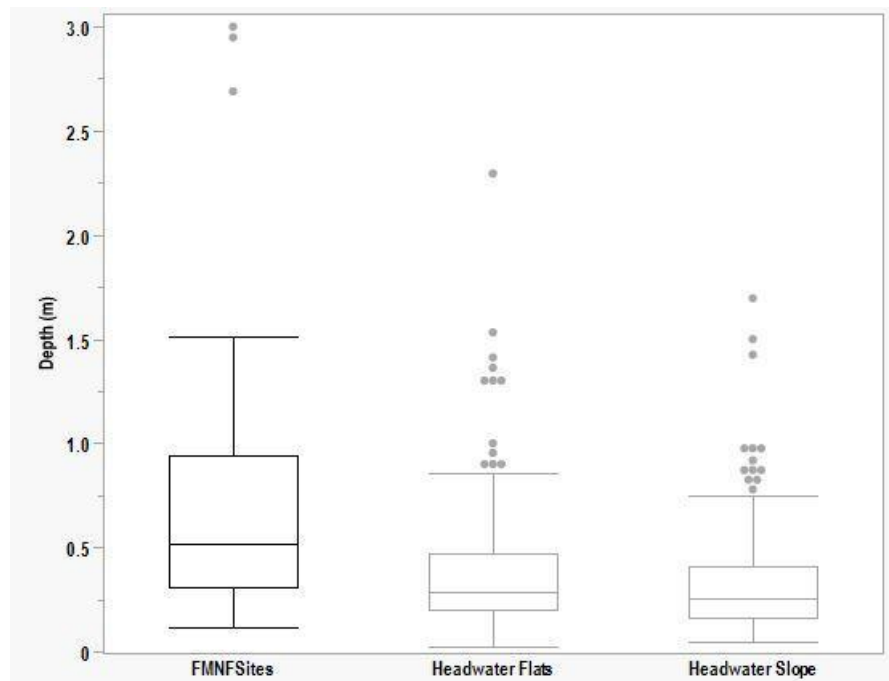


Figure 3.5.1. Distribution of variance showing significantly ($p = <0.0001$) deeper organic depths for soil transects at Francis Marion National Forest than the ones at the Brosnan Forest.

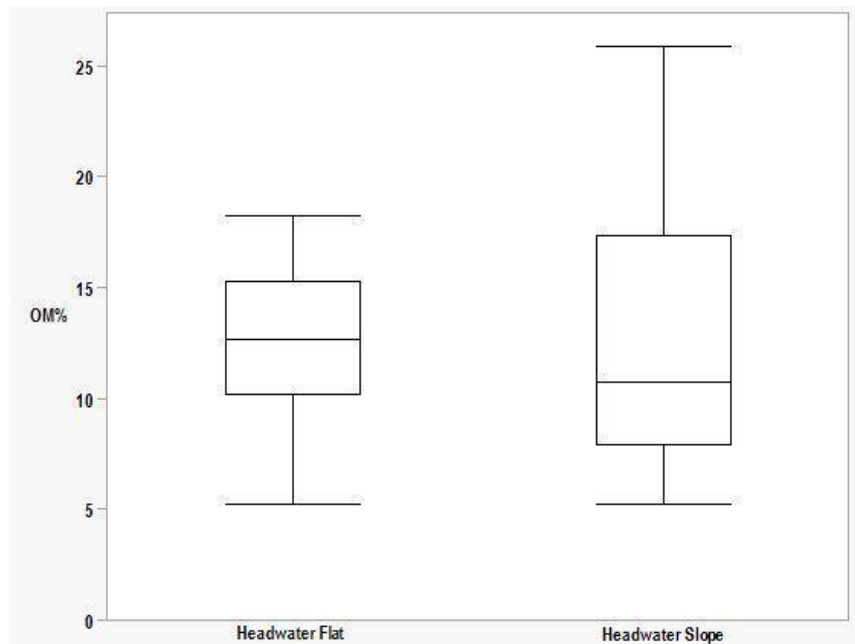


Figure 3.5.2. Distribution of variance for the percentage of organic matter, after removing soil depths taken from heavily disturbed sites from the soil samples collected at the headwater flat and headwater slope wetland, showing an insignificant difference between the two wetlands.

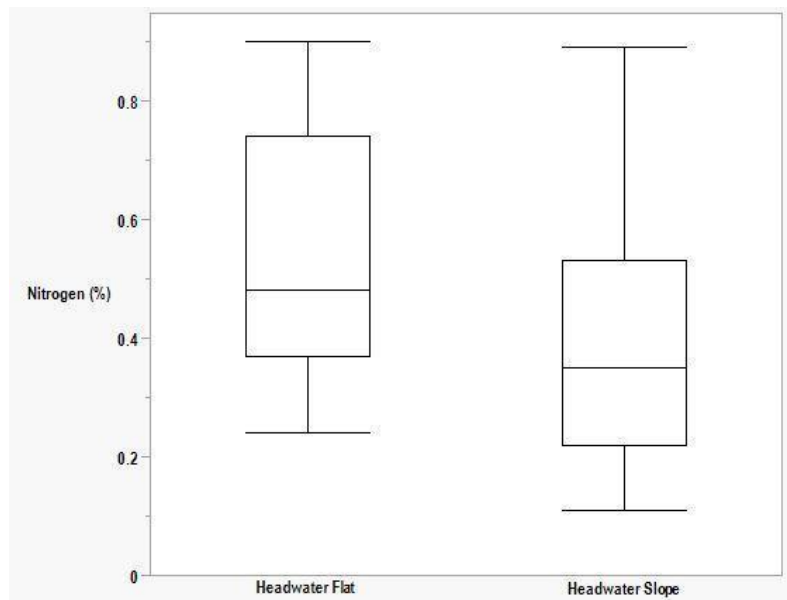


Figure 3.5.3. Distribution of variance for the percentage of nitrogen from the soil samples collected at the headwater flat and headwater slope wetland insignificant difference between the two wetlands.

APPENDICES

Appendix A

ROBEL POLE



Figure A-1. Image of Robel pole methodology taken at each quadrat. A technician would hold the Robel pole in the center of the quadrat, and the observer would stand 4 m away and 1 m above the ground while another technician recorded the visual obstruction observation for that cardinal direction.

Appendix B

SOIL PROBE IMAGES

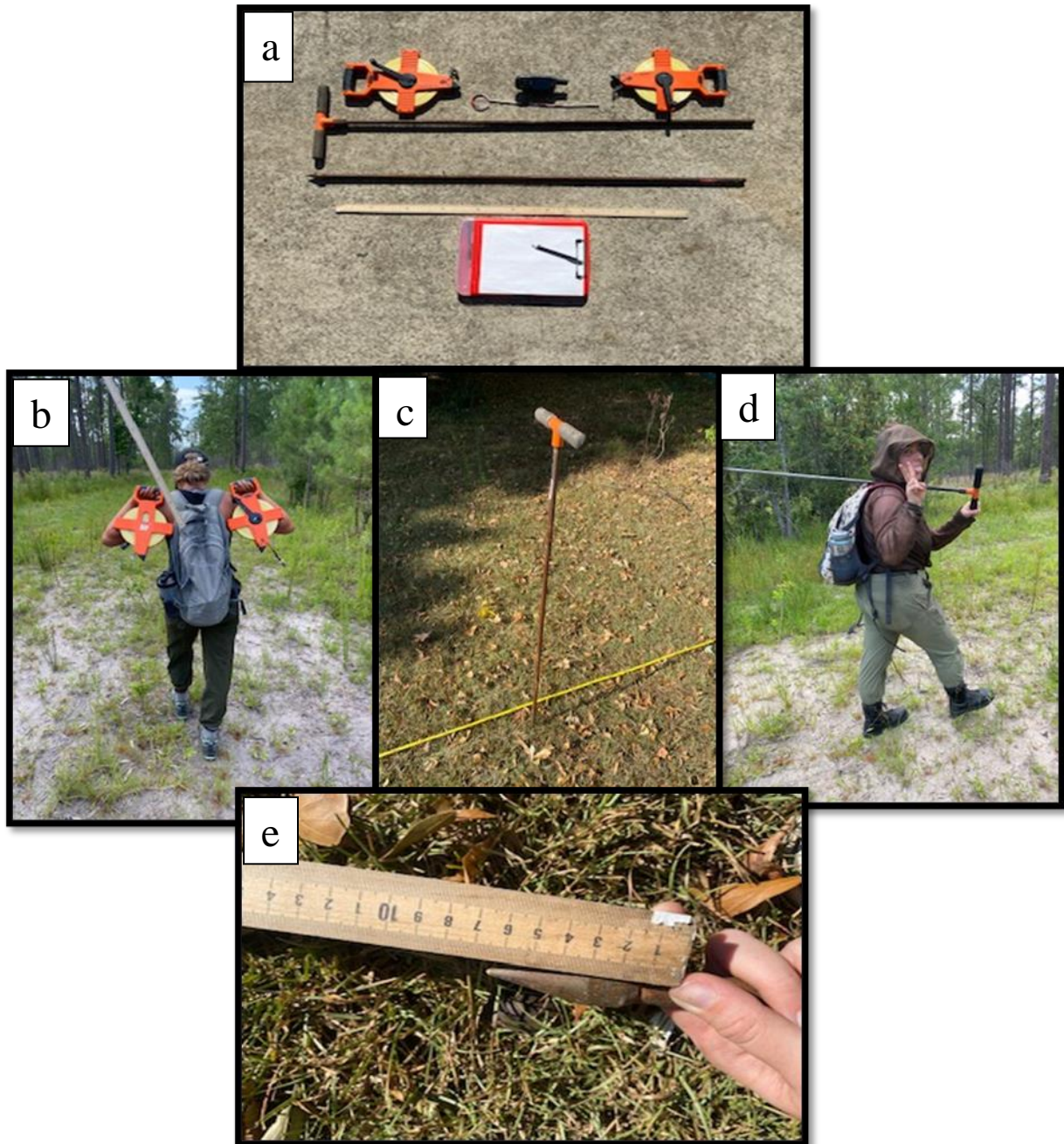


Figure B-1. Images of the soil probe equipment and methodology. Image a display the equipment used to record soil organic matter depth. Images b and d depict how the equipment was transported from each transect in the field. Images c and e exhibit how each measurement was recorded.