INTRODUCTION

It has been stated that the definitive test of success for restored and created wetlands (RWs and CWs) is how closely they function like natural wetlands (NWs) (Galatowitsch and van der Valk 1994). Currently, the ability of RWs and CWs to replace NW functions is a topic of considerable debate (Zedler and Calloway 1999, Stolt et al. 2000, Hunter and Faulkner 2001, Spencer et al. 2001). Soil properties of RWs and CWs have almost always been shown to differ from NWs (Verhoeven et al. 2001). For example, RWs and CWs usually have lower SOM content than NWs (Confer and Niering 1992, Galatowisch and van der Valk 1996, Bishel-Machung et al. 1996, Shaffer and Ernst 1999, Whittecar and Daniels 1999, Campbell et al. 2002). Such soil conditions can lead to low growth and survival of planted and colonizing species. Litter layers in RWs and CWs are often poorly developed or absent in comparison to that of NWs (Hunter and Faulkner 2001, Verhoeven et al. 2001). As a result of the low SOM content and paucity of litter, it has been speculated that the microbial communities in the soil of RWs and CWs are much less viable than those of NWs (Duncan and Groffman 1994). As very few restored or created wetland projects take any consideration of soil characteristics (Shaffer and Ernst 1999), there is currently a need for additional data on the development of soil properties in RWs and CWs to indicate whether they are actually proceeding toward natural wetland ecosystems and if they will be able replace the functions performed by the impacted wetlands (Spencer et al. 2001). Furthermore, soil characteristics are useful parameters for comparison across sites as some can be easily measured in the field or laboratory (Bishel-Machung et al. 1996).
For this investigation, I chose to measure soil organic matter (SOM) content as it has been shown to be: (1) an excellent indicator of soil quality; (2) well correlated with a number of important wetland processes such as respiration (Howard and Howard 1993), denitrification (Brettar and Höfle 2002), and P sorption (Axt and Walbridge 1999); (3) significantly different in RWs and NWs (Confer and Niering 1992, Bishel-Machung et al. 1996, Shaffer and Ernst 1999, Stolt et al. 2000, Campbell et al. 2002); and (4) a limiting factor in the development of RWs (Shaffer and Ernst 1999). This investigation was limited to non-tidal forested RWs because this class of wetlands has been shown to perform important nutrient transformation and retention functions (Lowrance 1992, Walbridge and Struthers 1993, Axt and Walbridge 1999). To control for differences in geology, soils, vegetation, and climate, site pairs were located in areas of similar topographic settings, identical mapped USDA soil series when possible, or in a closely related soil series.

Thus, the objective of this study was to compare the differences in SOM content of CW/RWs to those of paired NWs across a range of four hydrogeomorphic (HGM) settings (Brinson 1993). Previous research has established that wetlands should be grouped by HGM setting when comparing ecosystem properties and functions (Brinson & Rheinhardt 1996, Rheinhardt et al. 1998, Verhoeven et al. 2001). The following null hypotheses were tested: (1) there will be no difference in the SOM content of paired CW/RWs and NWs; (2) there will be no difference in the SOM content of wetlands across different HGM subclasses; (3) there will be no significant interactions between
wetland status and hydrogeomorphic subclass, or in other words, the differences between CW/RWs and NWs will be consistent across all subclasses.

METHODS

Study Sites

The sites selected for this study were all compensatory wetland mitigation sites and were located in the Coastal Plain physiographic region of North Carolina (Table 2.1, Figure 2.1). Nine of the sites belonged to the North Carolina Department of Transportation (NCDOT), while the other two sites, Bailey Creek and Parker Farm belonged to the PCS Phosphate Company. The sites were grouped into four different hydrogeomorphic subclasses common in the southeastern Coastal Plain including: (1) headwater riverine (stream order ≤ 2); (2) mainstem riverine (stream order > 2); (3) non-riverine mineral soil flat; and (4) non-riverine organic soil flat (Brinson 1993, Cole et al. 1997). Paired plots were used to control for differences in geology, soils, and vegetation that occur across the Coastal Plain. Sites were selected for sampling that contained both CW/RW and NW areas located in similar HGM settings, with closely-related or identical mapped USDA soil series. This helped to narrow the causes of edaphic differences between the CW/RW and NW plots to prior land-use and mitigation activities. All the CW/RWs sampled in this study were relatively young, between three and nine years since creation or restoration.

Due to their location adjacent to headwater streams, the Rowel Branch, Wiggins Mill, and Bailey Creek sites were classified by the Hydrogeomorphic (HGM)
Table 2.1. Site Characteristics of the 11 paired sites sampled in this study including name, hydrogeomorphic subclass, ssize, construction dates, and mapped soil series present. The prior land use (PLU) and mitigation activities are described briefly in the final column.

<table>
<thead>
<tr>
<th>Site Name</th>
<th>HGM Subclass</th>
<th>Size (ha)</th>
<th>Constructed</th>
<th>Soil series present</th>
<th>Prior land use and mitigation activities</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rowel Branch</td>
<td>Headwater Riverine</td>
<td>6.5</td>
<td>2000-2001</td>
<td>Foreston, Muckalee, Rains</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Wiggins Mill</td>
<td>Headwater Riverine</td>
<td>36</td>
<td>2000-2001</td>
<td>Bibb, Rains, Tomotely</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Bailey Creek</td>
<td>Headwater Riverine</td>
<td>1.2</td>
<td>1995</td>
<td>Dorovan, Muckalee</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Grimesland</td>
<td>Mainstem Riverine</td>
<td>3.1</td>
<td>1999-2000</td>
<td>Lakeland, Portsmouth, Swamp</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Haws Run ABC</td>
<td>Mainstem Riverine</td>
<td>11</td>
<td>1999</td>
<td>Muckalee, Foreston Leaf</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Balance Farm</td>
<td>Nonriverine Mineral Soil Flat</td>
<td>163</td>
<td>1998</td>
<td>Roanoke</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Mildred Woods</td>
<td>Nonriverine Mineral Soil Flat</td>
<td>240</td>
<td>1995</td>
<td>Altavista, Roanoke</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Barra Farm</td>
<td>Nonriverine Organic Soil Flat</td>
<td>240</td>
<td>1997-1998</td>
<td>Croatan</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Parker Farm</td>
<td>Nonriverine Organic Soil Flat</td>
<td>160</td>
<td>1995</td>
<td>Ponzer</td>
<td>PLU: MA:</td>
</tr>
<tr>
<td>Dismal Swamp</td>
<td>Nonriverine Organic Soil Flat</td>
<td>250</td>
<td>1996-1997</td>
<td>Scuppernong</td>
<td>PLU: MA:</td>
</tr>
</tbody>
</table>

FP = floodplain; AG = agriculture, SILV = aIlviculture, and MT = microtopography.
Figure 2.1. Map of the locations of the 11 paired created/restored and natural wetland sites sampled in this study.
Legend
- Study Sites
- Fall Line dividing Piedmont and Coastal Plain
Classification System (Brinson 1993a, Cole et al. 1997) as headwater riverine wetlands. The Rowel Branch site, located to the southeast of Wilmington, includes a former streambed and floodplain that had been bypassed by a diversion canal during the 1970s. Mitigation activities consisted of removing fill from floodplain, placing it back into the diversion canal, re-routing stream flow back into the relic channel, and planting woody seedlings. The 6.5 ha mitigation site was constructed from spring 2000 through summer 2001. The NW was located a few kilometers upstream of the RW and was similar in size to the RW. Soil present in the RW and NW areas have been mapped as the Foreston (coarse-loamy, siliceous, semiactive, thermic Aquic Paleudults), Muckalee (coarse-loamy, siliceous, superactive, nonacid, thermic Typic Fluvaquents), and Rains series (fine-loamy, siliceous, semiactive, thermic Typic Paleaquults).

The 36 ha Wiggins Mill site is located near the city of Wilson. Originally, this forested headwater wetland site contained a number of first and second order streams and associated riparian wetlands. These areas were subjected to logging, clearing, and excavation in the 1950s. This resulted in a mosaic of land-use types including forest, abandoned fields, pasture, and open water. Mitigation activities at this site occurred during 2000 and 2001 and consisted of filling ditches, grading, creating sinuous first and second order stream channels, and planting wetland tree seedlings. The NW was located adjacent to RW in a forested area that had not been converted to agriculture or pasture. Soils present in the RW and NW sections of this site have been mapped as the Bibb (coarse-loamy, siliceous, active, acid, thermic Typic Fluvaquents), Rains, and Tomotley series (fine-loamy, mixed, semiactive, thermic Typic Endoaquults).
The 1.2 ha Bailey Creek site, located near the town of Aurora, is owned by the PCS Phosphate Corporation. Originally, this area formed the headwaters of Bailey Creek. During phosphate mining operations, the area was cleared and excavated. Mitigation activities in 1995 recreated a reach of headwater stream and floodplain where the original stream had been located. The site was planted with a variety of wetland tree seedlings. The NW area was located in a nearby forested floodplain that had not been impacted by phosphate mining or agriculture. Soil present at CW and the NW have been mapped as the Muckalee and Dorovan series (dysic, thermic Typic Haplosaprists).

Due to their location in the floodplain of larger-order (stream order > 2) streams, the Grimesland and Haws Run sites were classified by the HGM system as mainstem riverine wetlands. The 220 ha Grimesland site, located to the east of Greenville, includes a variety of natural Taxodium-Nyssa (cypress-gum) swamp, bottomland hardwood forest, upland forest, and borrow pit lakes. The borrow pit lakes were the result of sand mining conducted at the site since the 1960s. From August 1999 to January 2000, mitigation efforts resulted in the creation of 3.1 hectares of wetland. Creation included filling some of the borrow pit lakes, grading to the floodplain elevation, and planting wetland tree seedlings. The mitigation area was located between Grindle Creek, a third-order tributary of the Tar River that flows through the site, and one of the borrow pit lakes. The NW was located adjacent to the CW. Soils present in the MIW and NW areas at the site have been mapped as Lakeland (thermic, coated Typic Quartzipsamments), Portsmouth (fine-loamy over sandy or sandy-skeletal, mixed, semiactive, thermic Typic Umbraquults), and swamp deposits (USDA 1974).
The 240 ha Haws Run site, located northeast of Willmington, once consisted of a third-order riverine swamp and mineral soil flat (Rheinhardt and Brinson 2002). In the 1970s, the *Taxodium distichum* (L.) L.C. Rich. (bald cypress) was cut from the riverine swamps and the flat was converted to a pasture for *Bison bison* (bison) (Rheinhardt and Brinson 2002). Mitigation activities at this site occurred during the summer of 1998 and the spring of 1999. To create 11 ha of riverine wetlands, the northern end of the site was excavated to the elevation of the adjacent swamp forest and planted with a variety of wetland tree seedlings. Soils present in this section of the site have been mapped as the Foreston series (Barnhill 1990, Barnhill 1992). Ditches were filled in the remaining central portion of the site in an attempt to restore a wet pine savanna community. A reference area exists at the southern end of the site adjacent to the third-order Shelter Swamp Creek. The soils in the NW areas of the site have been mapped as the Muckalee series (Barnhill 1990, Barnhill 1992).

Due to their mineral soil substrates and location in interstream divides, the ABC, Balance Farm, and Mildred Woods sites were classified by the HGM system as non-riverine mineral soil flats. The 75 ha ABC site is located to the northeast of the town of Washington. Much of the interstream area of this site was cleared, ditched, and converted to agricultural fields in the 1960s. The former agricultural field, approximately 37 ha in size, was restored to wetland during September 2000 through January 2001. Mitigation activities consisted of constructing impervious ditch plugs, backfilling ditches, recreating surface microtopography with hummocks and hollows, adding topsoil, and planting wetland tree seedlings. The NW was located in part of the forested interstream
area that was not converted to agriculture. Soils from both the RW and NW areas have been mapped as the Leaf series (fine, mixed, active, thermic Typic Albaquults).

The 163 ha Balance Farms site is located on the outer coastal plain, just south of the Virginia Border. Before conversion to agriculture in 1975, the site contained areas of wet hardwood forest (Rheinhardt and Brinson 2002). The site was extensively ditched in order to provide adequate drainage for crop production. Agricultural activities continued until 1994. In 1998, mitigation activities attempted to restore 95 ha to wet hardwood flat by plugging and filling the internal ditches with material from the inter ditch areas, placing flashboard risers on remaining perimeter ditches, and planting a variety of wetland tree seedlings. The NW transect was located in an onsite forested area that was not converted to agriculture. The soils of the RW and NW areas have been mapped as the Roanoke series (fine, mixed, semiactive, thermic Typic Endoaquults).

The 240 ha Mildred Woods site is located adjacent to US Highway 64 approximately five km east of Tarboro. The site consists of prior converted agricultural fields as well as forested areas that have not been cleared. The site was extensively ditched in order to provide adequate drainage for agriculture. In the summer of 1995 mitigation activities attempted to restore wetland hydrology by plugging and filling the ditch network. Then in the winter of 1996, 86 ha of the site were planted with wet hardwood forest species. This section of the site was sampled in this study. Other areas of the site were planted with *Chamaecyparis thyoides* L. (Atlantic white cedar), *Quercus-Carya* (oak-hickory) forest, and *Pinus palustris* P. Mill. (long leaf pine). The NW area was located in an onsite wet hardwood forested area that had not been cleared or
converted to agriculture. Soils present in the RW and NW areas have been mapped as the Roanoke and Altavista series (fine-loamy, mixed, semiactive, thermic Aquic Hapludults).

Due to their organic soil substrates and location in interstream divides, the Barra Farm, Parker Farm, and Dismal Swamp sites were classified by the HGM system as nonriverine organic soil flats. Unlike the mineral soils of the nonriverine sites, the soils of these sites were Histosols that can have organic matter contents of up to 90%. The Barra Farm site was part of 975 ha Carolina bay complex located to the southeast of the city of Fayetteville. Clearing and ditching beginning in the 1960’s converted the natural wetland communities into a large-scale agricultural operation. From fall 1997 to winter 1998, 250 ha at the southern end of the site were restored from agriculture to wetland (Bruland et al. 2003). The restoration process consisted of filling ditches and planting wetland tree seedlings. The soils in the RW and NW areas of the site have been mapped as the Croatan series (loamy, siliceous, dysic, thermic haplosaprist) (Hudson 1984).

The 160 ha Parker Farm site, also located near the town of Aurora, is the second site owned by PCS Phosphate Corporation. Clearing and ditching beginning in the 1960’s converted the natural nonriverine forested wetland into an agricultural field. In 1995, the site was restored by filling ditches and planting wetland tree seedlings. The soils in the RW and NW areas sampled at this site were mapped as the Ponzer series (loamy, mixed, dysic, thermic Terric Haplosaprist).

The 250 ha Dismal Swamp site is located along the boundaries of Gates and Perquimans countries in northeastern North Carolina. The site was originally a *C. thyoides* swamp that has been cleared, ditched, and drained to facilitate silvicultural and
agricultural activities. It was restored from summer 1996 to winter 1997. Restoration consisted of filling ditches, recontouring soil to establish surface microtopography, and planting of tree seedlings. The soils in both the RW and NW areas sampled at this site have been mapped as the Scuppernong series (loamy, mixed, dysic, thermic Terric Haplosapristes).

**Soil Sampling**

At each site, two transects were established, one in the CW/RW and the other in the NW. At the riverine sites (Rowel Branch, Wiggins Mill, Bailey Creek, Grimesland, and Haws Run), transects ran parallel to the direction of water flow, while at the non-riverine sites (ABC, Balance Farm, Mildred Woods, Barra Farms, Parker Farms, and Dismal Swamp), transects were oriented in a random direction. Along each transect, three to six soil cores were collected. The cores were collected, during the spring and summer of 2002, from the upper 20 cm of the soil profile in plastic sleeves with a piston corer of 5 cm diameter. Upon arrival, the cores were extruded from the sleeves and oven dried at 105°C for 24 hours. Once dried, the cores were ground and passed through a two mm sieve to remove rock fragments and other large organic debris. The sieved soil was then used for the determination of the soil organic matter (SOM) content by loss on ignition at 450 °C for 4 hours (Campbell et al. 2002).
**Statistical Analyses**

A general linearized model (PROC GLM, SAS for Windows Version 8.2, SAS Institute, Cary, North Carolina) was utilized to analyze the soils data. The GLM approach allowed for an unbalanced sampling design (five riverine and six nonriverine sites) as well as the different number of cores (from 3-6) collected at each site. The GLM was used to test the effects of wetland status (mitigation versus natural), HGM setting (headwater riverine, mainstem riverine, nonriverine mineral soil flat, nonriverine organic soil flat), individual sites within HGM settings (Site[HGM] effects for Rowel Branch, Wiggins Mill, Bailey Creek, Grimesland, Haws Run, ABC, Balance Farm, Mildred Woods, Barra Farm, Parker Farm, and Dismal Swamp), and the status by site within HGM setting interaction (Status*Site[HGM]). Differences between mean values of mitigation and natural wetlands, the four HGM subclasses, individual sites, and their interactions were analyzed with a least squares test in SAS. A log-transformation was performed prior to the analysis so that the data would better conform to the GLM assumptions of normality and homogeneous variances. Data in the figures was presented in the normal rather than the log scale for ease of interpretation.

**RESULTS**

*Plot means*

The mean SOM content for all 11 CW/RWs sampled in this study was 11.8 %. This was significantly lower ($p<0.001$) than the mean value for SOM content in the paired NWs at 28.9 % (Figure 2a, Table 2.2). When the data from both CW/RW and NW
plots was pooled together by HGM subclass, the headwater riverine (8.1 %), mainstem riverine (11.1 %), and nonriverine mineral soil (10.3 %) subclasses had significantly lower \((p<0.001)\) mean SOM contents than the nonriverine organic soil subclass (31.0 %) (Table 2.2). After stratifying the data from the HGM subclasses by wetland status, the SOM was significantly lower in the CW/RWs across all four HGM subclasses (Figure 2b, Table 2.2). There were also significant differences in mean SOM content of the individual sites within the subclasses (Table 2.2). For example, in the nonriverine organic subclass, mean SOM at Dismal Swamp was significantly higher than that of Parker Farm and Barra Farm, and mean SOM at Parker Farm was significantly higher than that of Barra Farm. Finally, the effects of the status by site within subclass interaction were also significant (Table 2.2), indicating that differences in SOM of CW/RWs and NWs were not always consistent across the subclasses (Figure 2c). For instance, at ten sites, mean SOM content was significantly lower in CW/RWs than in their paired NWs, but at Bailey Creek, there was no significant difference between SOM content of the CW and NW.

The created wetland with the lowest mean SOM content was Haws Run at 1.5 %. The restored wetlands at Mildred Woods (2.2 %), Rowel Branch (2.9), and the created wetland at Grimesland (3.7 %) also had quite low SOM contents. In contrast, the restored wetlands with the highest SOM content were Parker Farm (24.2 %), Barra Farm (25.7 %), and Dismal Swamp (44.9 %). The NWs with the lowest mean SOM content were Wiggins Mill (8.6 %), Bailey Creek (8.9 %), and Balance Farm (10.1 %). In contrast, the NWs with the highest mean SOM content were Barra Farm (43.7 %), Parker
Table 2.2. Results of the analysis of variance testing for significance of wetland status (created/restored versus natural), hydrogeomorphic subclass (headwater riverine, mainstem riverine, nonriverine mineral soil flat, nonriverine organic soil flat), the status by HGM subclass interaction, the site within HGM subclass effect, and the status by site within subclass effect for soil organic matter (SOM). All significant effects ($p<0.05$) are indicated by bold type.

<table>
<thead>
<tr>
<th>Soil Property</th>
<th>Source of Variation</th>
<th>df‡</th>
<th>F</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>SOM† (%)</td>
<td>Status</td>
<td>1</td>
<td><strong>218.5</strong></td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td></td>
<td>HGM Subclass</td>
<td>3</td>
<td><strong>152.4</strong></td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td></td>
<td>Status * HGM</td>
<td>3</td>
<td><strong>12.1</strong></td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td></td>
<td>Site(HGM)</td>
<td>7</td>
<td>4.7</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td></td>
<td>Status * Site(HGM)</td>
<td>7</td>
<td><strong>7.3</strong></td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>

† SOM was log-transformed to better meet the GLM assumptions.
‡ Error degrees of freedom = 66.
Figure 2.2. Comparison of soil organic matter (SOM) in created/restored wetlands (CW/RWs) and natural wetlands (NWs) (A), of SOM in CW/RWs and NWs across the four hydrogeomorphic subclasses (B), and of SOM across the 11 paired CW/RWs and NWs (C).
Farm (77.4 %), and Dismal Swamp (80.2 %). When compared to their paired CW/RWs, mean SOM values in the NWs were three times higher at Grimesland, ABC, and Parker Farm, five times higher for Rowel Branch, six times higher for Mildred Woods, and 18 times higher for Haws Run. Finally after excluding the nonriverine organic soil flats, there was no significant difference between the SOM content of the CWs and the RWs ($p = 0.39$)

DISCUSSION

While there are some problems with the loss of ignition technique for measuring SOM content (Axt and Walbridge 1999), it has been widely used in studies of wetland soils, and is especially common in studies that compare soil properties in CW/RWs to those of NWs because it is a simple and quick method that does not require the use of expensive laboratory equipment. Similar to previous studies that compared SOM content in CW/RWs and NWs (Bishel-Machung et al. 1996, Gallatowitsch and Van der Valk 1996, Shaffer and Ernst 1999, Campbell et al. 2002), this study reported lower SOM in the CW/RWs than in NWs. The lower SOM in the CW/RWs may be due to increased decomposition during prior land-uses, or to the removal of organic-rich topsoil during excavation into subsurface horizons that often occurs in wetland creation. Low levels of SOM in CW/RWs may hinder the development of microbial communities that are critical to wetland function (Bruland, unpublished data). Interestingly, the mean SOM content of 11.8 % reported for the 11 CW/RWs sampled in this study was nearly double the mean SOM contents of 6.2 % reported for a study of 44 mitigation wetlands in Pennsylvania
(Bishel-Machung et al. 1996) and 6.7 % reported in a study of nine created wetlands in Oregon (Shaffer and Ernst 1999). As this study used the same method to quantify SOM (loss on ignition at 450°C) as the studies from Pennsylvania and Oregon, the differences in SOM content among the studies were not due to differences in analytical techniques. The main reason for the higher mean SOM content in the North Carolina CW/RWs was that, unlike Pennsylvania and Oregon, North Carolina has a number of created and restored wetland sites located on former agricultural land with histosol substrates. These types of sites have very high SOM content, and certain more recalcitrant fractions of it persist even after decades of agricultural activity (Bridgham et al. 1991).

Mean SOM content in the NWs from North Carolina (28.9 %) was also higher than mean SOM content in the NWs from Pennsylvania (21.7 %) and Oregon (8.0 %) (Bishel-Machung et al. 1996, Shaffer and Ernst 1999). Unlike this study, the studies in Pennsylvania and Oregon did not use a paired site approach, but rather randomly sampled CW/RW and NW from larger identified populations. With random sampling, it becomes much more difficult to argue that differences in SOM content of CW/RWs and NWs were due to land-use or mitigation techniques and as a number of other confounding factors may explain the differences between CW/RWs and NWs such as differences in geology, soil type, and vegetative communities.

Wetland status was not the only factor controlling SOM content in the sites sampled in this study. The HGM setting in which the site was located was also an important factor. When CW/RW and NW areas were pooled by HGM subclass, the nonriverine organic soil subclass had significantly higher mean SOM content than the
other three subclasses. This indicated that studies of forested wetlands in the southeastern Coastal Plain should differentiate between sites with organic and mineral soils. Including the organic soil flat subclass with the riverine, or nonriverine mineral soil sites would be comparable to grouping apples with oranges. While this study did not show significant differences in SOM content between headwater and mainstem riverine sites, this was most likely due to the limited sample size and high within subclass variability of the sites that were sampled. When mean SOM content of CW/RWs and NWs were compared within subclasses, SOM was lower in the CW/RWs than the NWs across every subclass, suggesting that the effects of prior-land use and mitigation were at least consistent across these Coastal Plain forested wetland types. The study of SOM content of wetlands in Oregon also reported significant differences in SOM content of CW/RWs and NW across HGM subclasses, with CW/RWs having lower within subclass mean SOM content in four out of five subclasses for which this comparison could be made (Shaffer and Ernst 1999).

Despite the efforts to group the sites sampled in this study into similar HGM subclasses, there was considerable within subclass variation. In the riverine subclasses, certain CW/RWs had significantly lower mean SOM content than their NW pairs, while other CW/RWs had SOM contents that were closer to, if not comparable, to the SOM contents of their NW pairs. For the nonriverine sites, this was also especially evident for the NW areas in the organic soil subclass, which showed significantly higher mean SOM content at Parker Farm and Dismal Swamp than at Barra Farm. It was also interesting to note that after excluding the nonriverine organic flats, there were no significant
differences in SOM content of the CWs and RWs. However, the small and unbalanced sample sizes (n=3 for CWs, n=5 for RWs) used for this comparison were not ideal. Future studies that sample additional sites may be able to detect difference between SOM content of CWs and RWs.

The sites from the nonriverine organic soil subclass deserve additional attention and discussion. While sites with these organic soil substrates are most likely to be found in the southeastern United States in states such as North Carolina, Florida, and Louisiana, there are considerable areas of histosols in other states such as Minnesota, Wisconsin, Michigan, and New York. If there is a choice in mitigation options between restoration sites of different substrates, sites located on histosols should be targeted for restoration for a number of reasons. First, these sites contain pools of SOM, which despite their recalcitrance, or differences in organic matter from unaltered NWs, provide an energy source to microbes that may serve to “jump-start” wetland function once hydrology is restored. Second, under agriculture or other anthropogenically-influenced land-uses, these pools of SOM would be subject to oxidation and loss to the atmosphere as CO₂, contributing to the greenhouse effect and global climate change (Richardson 1983, Richardson and Gibbons 1993). Third, by taking these areas out of agricultural production, they appear to provide not only flood control but also water quality benefits (Bruland et al. 2003). Without ditches, these sites can retain enormous quantities of water in microtopographic depressions and surface soil horizons. Residual nutrients in the soils and porewater from liming and fertilization may stimulate rapid microbial and vegetative development. Fourth, from an economic perspective these sites require
intensive drainage, tillage, liming, and fertilization in order to remain productive agricultural land (Lilly 1981, Brady and Weil 1999). While they many be productive in the first few years after conversion, long-term sustainability of these types of sites may be economically unrealistic. Consequently, such areas may be more readily purchased from landowners than other types of agricultural land that requires less maintenance.

In the state of North Carolina, since this study began, two other sites located on histosols have been purchased by NCDOT for restoration, the 300 ha Juniper Bay site near Lumberton and the 1630 ha Croatan Mitigation Bank near Jacksonville. Both sites are large tracks of land that were originally wetland, converted to agriculture or silvicultural production, and are now in the process of being restored by plugging and filling in ditches, and planting with wetland tree species. Future research is needed to assess the development of wetland functions in these sites to determine if these types of sites provide optimal areas for wetland restoration.

In addition to measuring SOM, at a subset of eight sites, bulk density (BD), percent sand content, microbial biomass carbon (MBC), the phosphorus sorption index (PSI), and denitrification potential were also measured. While this data was not presented here, the trends in the MBC, PSI, and denitrification data were similar those observed with SOM. For example, MBC, PSI, and denitrification potential were lower in the CW/RW sites than in the NW sites across all HGM subclasses and across all but one of the site pairs. As BD and sand were negatively correlated to SOM, CW/RWs had higher BD and sand content than NWs across all HGM subclasses and site pairs.
Furthermore, the nonriverine organic soil class appeared to be significantly different from the other subclasses for these additional soil properties.