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Effects of agriculture and wetland restoration on hydrology, soils, and water quality of a Carolina bay complex

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Abstract

We compared hydrology, soils, and water quality of an agricultural field (AG), a two-year-old restored wetland (RW), and two reference ecosystems (a non-riverine swamp forest (NRSF) and a high pocosin forest (POC)) located at the Barra Farms Regional Wetland Mitigation Bank, a Carolina bay complex in Cumberland County, North Carolina. Our main objectives were to: 1) determine if the RW exhibited hydrology comparable to a reference ecosystem, 2) characterize the soils of the AG, RW, and reference ecosystems, and 3) assess differences in water quality in the surface outflow from the AG, RW, and reference ecosystems. Water table data indicated that the hydrology of the RW has been successfully reestablished as the hydroperiod of the RW closely matched that of the NRSF in 1998 and 1999. Jurisdictional hydrologic success criterion was also met by the RW in both years. To characterize soil properties, soil cores from each ecosystem were analyzed for bulk density (D_b), total carbon (C_t), nitrogen (N_t), and phosphorus (P_t), extractable phosphate (PO_{4w}), nitrogen (N_{ex}), and cations (Ca_{ex}, Mg_{ex}, K_{ex}, Na_{ex}), as well as pH. Bulk density, P_t, Ca_{ex}, Mg_{ex}, and pH were greatly elevated in the AG and RW compared to the reference ecosystems. Water quality monitoring consisted of measuring soluble reactive phosphorus (SRP), total phosphorus (TP), nitrate + nitrite (NOX), and total nitrogen (TN) concentrations in surface water from the AG, RW, and reference outflows. Outflow concentrations of SRP, TP, and NOX were highest and most variable in the AG, while TN was highest in the reference. This study suggested that while restoration of wetland hydrology has been successful in the short term, alteration of wetland soil properties by agriculture was so intense, that changes due to restoration were not apparent for most soil parameters. Restoration also appeared to provide water quality benefits, as outflow concentrations of SRP, TP, NOX, and TN were lower in the RW than the AG.

Introduction

Wetland restoration is a promising strategy for alleviating water quality problems in watersheds dominated by agriculture. The effectiveness of the wetland at improving water quality will depend on the flow of the water through the system (hydrology), as well as the forms and amounts of nutrients in the soil (soil properties). We report here the early results of a study investigating the effects of agriculture and wetland restoration on hydrology, soil properties, and water quality of the Barra Farms Regional Wetland Mitigation Bank, a Carolina bay complex in Cumberland County, North Carolina.

The conversion of a wetland to agricultural production has implications for all components of the ecosystem. In terms of hydrology, the establishment of networks of drainage ditches lowers the water table, promotes rapid drainage during and after precipitation, and creates conditions of continuous surface water flow; in contrast, prior to ditching, water tables would be higher, drainage would be slower, and only intermittent flow would occur (Sharitz and Gibbons 1982; Richardson and Gibbons 1993). Reversal of these effects, to reestablish wetland hydrology is often cited as the most critical component to wetland restoration success (Kusler and Kentula 1990); as hydrology has been considered the master variable controlling redox status, pH, nutrient cycling, community composition, and wetland development (Bridgham and Richardson 1993). Thus, the first objective of our paper was to determine if the restored wetland (RW) exhibited representative wetland hydrology and met jurisdictional hydrologic success criteria. This was accomplished by comparing the seasonal pattern of water table depths of the RW to that of the reference non-riverine swamp forest (NRSF).

Upon conversion to agriculture, wetland soils that were once subjected to reducing conditions and low rates of decomposition are subjected to oxidizing conditions and high rates of decomposition (Armentano and Menges 1986; Schlesinger 1986). Artificial drainage leads to the loss of organic matter and subsequent soil subsidence (Lilly 1981). Following the initial impacts of ditching and clearing, come the secondary impacts that result from tillage, liming and fertilization. Tillage has been shown to increase compaction of wetland soils (Brady and Weil 1999; Braekke 1999). Liming increases soil pH and elevates base cation content (Simmons et al. 1996; Braekke 1999). This, in turn, can further increase decomposition (Lilly 1981; Compton and Boone 2000). Fertilization often leads to over-saturation of agricultural soils with inorganic nutrients such as nitrate and phosphate. This occurs as more nutrients are applied as fertilizer than are taken up by crops during the growing season. Like hydrology, the restoration of wetland soil properties is another important factor in restoration, as soils are the physical foundation of wetland ecosystems (Stolt et al. 2000). Our second objective was to characterize and compare the soils of the agricultural (AG), restored wetland (RW), and two reference ecosystems to assess the impacts of land-use on wetland soil properties. Unlike hydrology, soil properties are more difficult to restore, less often considered in restoration plans, and rarely monitored in years following restoration (Shaffer and Ernst 1999).

Conversion of wetland to agriculture in the North Carolina coastal plain has also been shown to affect surface water quality (Sharitz and Gibbons 1982; Ash et al. 1983; Richardson 1981; Richardson and Gibbons 1993). Specifically, conductivity, pH, sediment, phosphorous and nitrogen concentrations have been shown to be much higher in agricultural ditches draining converted wetlands compared to streams draining unaltered wetlands in the coastal plain (Kuenzler et al. 1977; Kirby-Smith and Barber 1979; Sharitz and Gibbons 1982; Ash et al. 1983). Channelized streams are also more likely to be located in heavily managed areas that tend to export large amounts of nutrients as a result of fertilization and liming (Sharitz and Gibbons 1982). The third objective of our paper was to assess the differences in water quality in the outflow from the AG, RW, and reference ecosystems at Barra Farms.

Characteristics of unaltered Carolina bays

Carolina bays are elliptical depressions found on the southeastern coastal plain that are consistently oriented in a northwest-southeast direction (Sharitz and Gibbons 1982). Common features of these bays include an ovate shape with the large end at the northwest, a sand rim prevalent at the southeast margin, the presence of shrub bog communities, and a water table dependent on precipitation and evapotranspiration (Sharitz and Gibbons 1982). According the United States Fish and Wildlife Service system, Carolina bays are classified as either forested or scrub-shrub palustrine wetlands (Cowardin et al. 1979). However, due to their variability in size, depth, and substrate conditions, Carolina bays do not have a single characteristic cover type and may contain woody, shrub-scrub, herbaceous, and even aquatic vegetation (Sharitz and Gibbons 1982).

Although the general features and vegetative communities of Carolina bays have been characterized, few in-depth studies have been conducted on these ecosystems (Richardson and Gibbons 1993) and thus, despite their abundance in the southeast, relatively little is known about their hydrology, community structure, succession, trophic dynamics, and mineral cycling (Schalles and Shure 1989). When land-use effects are overlaid upon the complex pattern of natural succession in Carolina bays, the ecosystem structure, function, and successional patterns become difficult to predict or quantify. For instance, over the last 300 years, Carolina bays have been frequently burned by Native Americans (Wells and Boyce 1953), and more recently drained for agriculture, forestry, industry, and other land management activities (Kirkman and Sharitz 1994).

Study site

The four ecosystems chosen for our research were all part of a 975 hectare (ha) Carolina bay complex located in Cumberland County, North Carolina (Figure 1). Clearing and ditching beginning in the 1960's converted the natural vegetative communities into a large-scale farm operation (Land Management Group Inc. 2000a). A system of primary and secondary ditches was established with tertiary ditches added later in the 1970s (Land Management Group Inc. 2000a). During the 1970s and early 80s, Barra Farms was one of the largest farming operations in the North Carolina coastal plain (J. Bullard, personal communication). For the last 10 years, the site has been farmed much less intensively.

During a four-month period, from October 1997 to January 1998, 250 ha at the southern end of the site were restored from agriculture to wetland, creating the Barra Farms Cape Fear Regional Mitigation Bank (Figure 1). The restoration was conducted by Ecosystems Land Mitigation Bank Corporation (ELMBC), a mitigation banking firm based in Winter Park, Florida, USA. The restoration process consisted of two main components: 1) filling 3,300 m of linear ditches to reestablish surface and groundwater flow through the restored wetland, and, 2) planting 192,000 individual woody seedlings (see below).

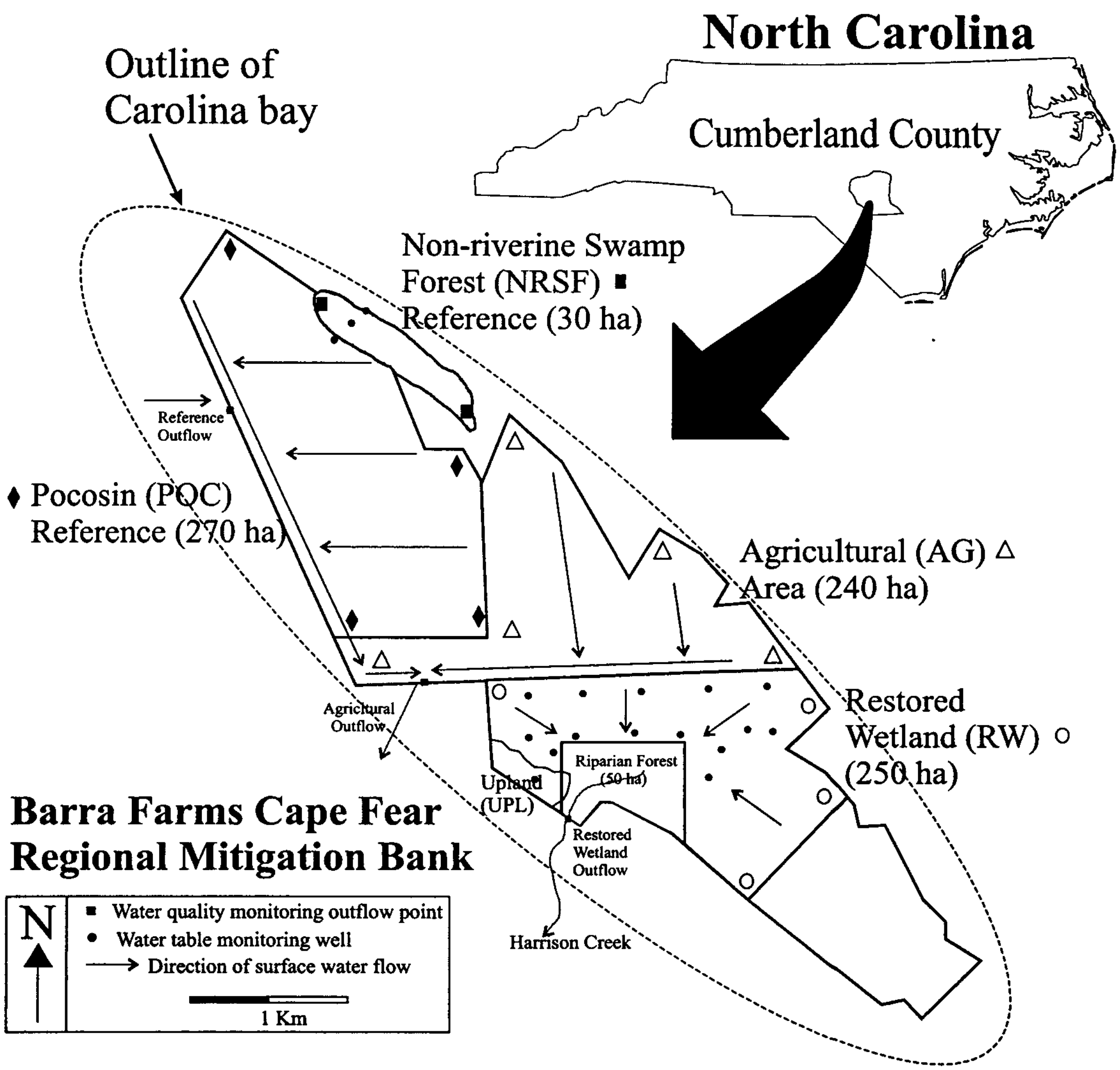

Figure 1. Carolina bay study area in Cumberland County, North Carolina.

Secondary activities included stream restoration in Harrison Creek and supplemental planting in an adjacent riparian forest.

Prior to agricultural activity, the bay complex comprised the majority of the 2,500 ha drainage area for Harrison Creek. However, the network of agricultural ditches used to drain the site reduced the drainage area of Harrison Creek to 130 ha (Environmental Services 1997). After restoration, the drainage area of Harrison Creek was increased from 130 ha to 380 ha. Surface runoff and subsurface flow from the RW pass through the riparian forest before exiting the system. Improvements to water quality most likely result from the filling of ditches in the restored wetland and the redirection of flow through the riparian forest. Thus, in terms of water quality, we considered the restored wetland and the riparian forest a unit and will refer to them as the RW.

Seedlings planted in the RW are typically found in these bay complexes and included bald cypress (*Taxodium distichum* [L.] Richard), Tupelo (*Nyssa ssp.*), Atlantic White Cedar (*Chamaecyparis thyoides* [L.] BSP.), water ash (*Fraxinus caroliniana* Miller), red bay (*Persea borbonia* [L.] Sprengel), tulip poplar (*Liriodendron tulipifera*, L.), cherrybark oak (*Quercus pagoda* Raf.), overcup oak (*Quercus lyrata* Walter), willow oak (*Quercus phellos* L.), swamp chestnut oak (*Quercus michauxii* Nuttall), and laurel oak (*Quercus laurifolia* Michaux). Interestingly, while *T. distichum* might have been found in the near-stream area or the floodplain of the riparian forest, it probably would not have occurred in the central portions of the bay removed from the stream (Rheinhardt and Brinson 2000). In these areas, it may have been more appropriate to plant *Taxodium ascendens*, a more fire-tolerant species. To protect seedlings from flood stress during the fall of 1999, ELMBC installed a series of culverts to drain standing water from the RW. Supplemental planting of an additional 43,300 seedlings was performed February 8–11, 2000 to replace dead seedlings and maintain a tree density of 320 per ha (Land Management Group Inc. 2000b).

Very little information is available to characterize the original status of the bay complex. A historic timber map of the site identified areas of hardwood in central portion of the bay complex, and areas of pine and juniper at the periphery (Flowers 1924). The original Soil Survey of Cumberland County classified the soils of the bay complex as Portsmouth loam, a Typic Umbraquult (Perkins et al. 1925). Soils of this series are described as poorly drained and range from dark-gray loam to muck. They are underlain by silty to sandy clay, and, in many places, have accumulated large quantities of organic matter (Perkins et al. 1925). The survey states that Portsmouth loam soils in Cumberland County supported forests of cypress, gum, and maple with an understory of gallberry, huckleberry, and bay bushes (Perkins et al. 1925).

The most recent County Soil Survey of Cumberland and Hoke Counties reclassified the soils of the bay complex as Croatan muck, a Terric Haplosaprist (Hudson 1984). This is a very poorly drained, organic soil that is formed of highly decomposed organic material and underlain by loamy textured marine and fluvial sediment. Intensive agricultural activities at the study site have caused massive changes in the Croatan muck soils. Much of the organic matter has been lost, and mineral subhorizons have been brought to the surface by plowing. For these reasons, when choosing a reference soil, both a true Croatan muck and an organic-rich mineral soil might be considered appropriate.

It the absence of other more specific historical information about the vegetation of the site, the nature of the vegetative communities that existed prior to cultivation must be inferred from the surrounding ecosystems. However, due to the heterogeneity of vegetative communities within Carolina bays, the restoration site may not have been originally identical to the adjacent agricultural area. Therefore, sampling several local reference wetlands can provide information about the range of values characteristic of regional reference conditions. Thus, we included in this study two reference communities, each with different soil and vegetative characteristics. Both reference areas are part of the bay complex, and are typical of communities observed in other unaltered bays of this region (Shafale and Weakley 1990). Following the classification scheme of Schafale and Weakley, we refer to them as the high pocosin (POC) and the non-riverine swamp forest (NRSF).

The vegetation of the POC was characterized by a thick understory of *Lyonia lucida* (fetterbush), *Ilex glabra* (low gallberry), *Smilax laurifolia* (green briar), with emergent tree species such as *Pinus serotina* (pond pine), *Magnolia virgiana* (sweetbay), and *Gordonia lasianthus* (loblolly bay). The vegetation of the NRSF consisted of a much more open shrub layer of *L. lucida*, and *S. laurifolia*, with a closed canopy of tree species such as *Acer rubrum* (red maple), *T. distichum, M. virgiana, Liriodendron tulipifera* (tulip poplar), and *Pinus taeda* (loblolly pine).

Importantly, recent management practices in the study area have involved long-term fire suppression. Thus, our control sites may not represent truly unaltered conditions. However, at the time of the study, no other reference areas were available that had been subjected to any other type of fire regime.

Methods

Hydrology

Total monthly precipitation (PPT) and average monthly temperature data were obtained from a weather station in Fayeteville, North Carolina (NOAA 1998–1999), located about 15 km from the site. The temperature data was used to calculate potential evapotranspiration (PET) with the Thornthwaite equation (Dunne and Leopold 1978).

Slotted PVC pipes were installed to monitor water table depth in accordance with US Army Corps of Engineers' specifications (Sprecher 1993). Wells were sunk in a pre-augured hole to a depth of 80 cm below the soil surface. Our study included the 15 wells in the RW located on former agriculture land, the 3 wells from the NRSF, and an on-site upland (UPL) well.

Water table depths were compared in the RW and the NRSF, with the UPL included to contrast natural and restored wetland hydrology to upland hydrology. The NRSF was selected as the hydrologic reference since soils in this area more closely represented the RW and AG soils as compared to the more organic POC soils. No water table data was available for the POC or the AG ecosystems. However, throughout the study period, the AG ecosystem was drained by a series of tiles and ditches so that the water table was less than 30 cm from the soil surface during the growing season except during extreme flooding events.

Depth to the water table was measured on a weekly basis during the jurisdictional growing seasons of 1998 and 1999. The jurisdictional requirements placed by the Army Corps of Engineers on ELMBC mandated that water table depths of the RW must be within 30 cm of the soil surface during the growing season for at least 50% of the time in which the water table of the NRSF is within 30 cm (EcoScience 1998). The average length of the jurisdictional growing season for the region was determined to be 242 days, starting on March 18 and ending November 15 (Natural Resources Conservation Service. 2000). To determine if the RW met jurisdictional success criteria, we calculated the number of consecutive days the water tables of the RW and NRSF were within 30 cm, the percent of the growing season for which these intervals occurred, and the percent of the time the water tables in the RW were within 30 cm compared to those of the NRSF. We assumed that two consecutive readings showing saturated conditions represented 7 continuous days of saturation. While this sampling interval may not be able to totally capture fluctuations in water table in response to storm events, it has been shown that such a frequency is adequate for defining average system behavior (Shaffer et al. 2000).

Soil sampling

We took soil samples from five depth intervals (0–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, and 80–100 cm) in the AG, RW, NRSF, and POC to compare soil characteristics among ecosystems. A Dutch corer was used for collecting samples for chemical analyses on August 12–13, 1999 and May 18, 2000. Eight cores (40 samples) were taken from the RW and 3 cores (15 samples) were taken from the other three ecosystems. Additional surface soil samples were taken from the AG (8 samples) and reference ecosystems (3 samples each). Samples were placed into sealable bags and stored in a cooler under field moist conditions until analysis or further sample preparation. Bulk density sampling (3 replicate cores per ecosystem) was conducted with a piston corer for mineral soil areas of the RW, AG, and NRSF. The organic soils of the POC and NRSF were sampled with a box corer at the upper two depths and a Russian peat corer for the lower three depths. Bulk density sampling was conducted on November 30, and December 1 and 5, 2000.

Laboratory analyses of soil properties

Soil bulk density was determined by a process of oven-drying the soil cores at 70 °C for 3 days, weighing the dried soil, and dividing by the volume of the corer. Soil pH was determined with a Beckman pH meter on field moist soils with 2:1 and 10:1 water to soil ratios for mineral and organic soils respectively (Hendershot et al. 1993). For the total carbon, nitrogen, and phosphorus analyses, soils were air dried and homogenized with a mortar and pestle. Total carbon and nitrogen concentration were determined by dry combustion with a Perkin Elmer 2400 CHNS autoanalyzer. Total phosphorus concentration was measured on HNO_3-$HClO_4$ digests by automated ascorbic acid reduction on a Lachat autoanalyzer (O'Halloran

1993). Exchangeable ammonium was determined on field moist soil by extracting with 2 M KCl followed by filtration though Whatman 42 filter paper. Extracts were analyzed on a TRAACS autoanalyzer (Keeney and Nelson 1982). An identical procedure was used for nitrate/nitrite, except filtration was followed by automated Cd reduction on a Lachat autoanalyzer (Keeney and Nelson 1982). Water extractable phosphate (PO_{4w}) content was determined on oven-dried, 2 mm sieved soils. Soils were shaken in DI water at 160 rpm for 1 hour (Pote et al. 1999). After reduction with ascorbic acid, samples were analyzed for molybdate reactive phosphorus with a Beckman UV-Vis Spectrophotometer (Murphy and Riley 1962). Analysis of exchangeable cations (Ca_{ex}, Mg_{ex}, K_{ex}, and Na_{ex}) were conducted on oven dried, 2 mm sieved soils, by extracting with 1 N ammonium acetate at pH 7.0 (Jackson 1958). Samples were analyzed with a Perkin Elmer AA Spectrophotometer.

Water quality monitoring and laboratory analyses

A water quality-monitoring program was established in 1999 to measure surface water P and N concentrations in outflows from the reference ecosystems, the agricultural ecosystem (AG), and the restored wetland (RW) (Figure 1). There was only one reference outflow because it consisted of a ditch that drained both the pocosin and non-riverine swamp forest. The AG outflow drained a section of Barra Farms that was actively farmed during the study period. The RW outflow consisted of the stream that forms the headwaters of Harrison Creek. It is an ephemeral stream that typically flows only after heavy rainfall during the winter and early spring. The REF and AG outflows, on the other hand, support continuous flow throughout the year.

Grab samples of surface water were taken from each of the three sites discussed above. Sample preservation for soluble reactive phosphorus (SRP) analysis consisted of filtration with a 0.45-micron filter and storage on ice. The filtered samples were subsequently analyzed colorimetrically within 48 hours of the collection time on a Beckman UV-Vis spectrophotometer (Murphy and Riley 1962). Sample preservation for total phosphorus (TP), nitrate + nitrite (NOX) and total nitrogen (TN) consisted of preservation with sulfuric acid. Total phosphorus was determined by persulfate digestion and measuring SRP in the digests (Vaithiyanathan and Richardson 1997). Nitrate + nitrite was determined by automated cadmium reduction on a Bran and Lubbe TRAACS autoanalyzer and TN was determined by persulfate digestion and measuring NOX in the digests (Vaithiyanathan and Richardson 1997).

Data analysis

The mean concentration for each soil parameter was calculated for each ecosystem at the five depth intervals. Due to large differences in bulk density (D_b) between ecosystems, and with depth within an ecosystem, we presented our nutrient data on a mass per volume basis (Bridgham and Richardson 1993). To do this, we determined the mean bulk density for each ecosystem at each depth. We then multiplied the nutrient concentration data by the appropriate bulk density value. For parameters in which depth profiles did not reveal clear trends, data were pooled into two depth classes: 0–40 cm and 40–100 cm. To do this, we added all the points within a depth class and express data as $mg \cdot cm^{-3}$ from 0–40 cm and $mg \cdot cm^{-3}$ from 40–100 cm. This was done for C_t, N_t, P_t, N_{ex}, and PO_{4w}. Due to concerns about pseudoreplication, we did not conduct parametric statistical analyses on the soil profile or pooled soil data.

To further assess differences in surface soils, we conducted a principal components analysis (PCA) on the volumetric surface soil data (0–20 cm) with PC-ORD software (McCune and Mefford 1999). PCA was used to summarize correlations among environmental variables. Biplots relating the four ecosystem types with measured soil properties were created for visual interpretation of the PCA.

S-Plus was used to conduct Classification and Regression Tree (CART) analysis on the volumetric 0–20 cm surface soil data to determine where soil chemistry divisions occurred among the four ecosystem types. CART is a multivariate regression tool that recursively divides groups into two groups based on the environmental variable that best reduces variability within the two groups (Venables and Ripley 1997).

Differences in mean concentration of SRP, TP, NOX, and TN in the AG, RW and outflows from reference sites were summarized with boxplots. The amount of water quality data available for statistical analysis was confined to those dates in which surface water outflow was observed from the AG, RW, and reference ecosystems.

Results

Hydrology

The seasonal patterns of water table depth, or hydroperiods, of the RW and NRSF were very similar in the first two years of monitoring (Figure 2) and appeared to be controlled by PPT and PET (Figure 2i). For example, in 1998 and 1999, both the RW and NRSF had water tables above the soil surface from March to May, during the period when PPT exceeded PET. This was followed by sharp decreases in water table depths from May through August, when PET exceeded PPT. In 1998, water tables of the RW and NRSF rose slightly in August in response to precipitation events, but then dropped in September. In, 1999, on the other hand, water tables in the RW, NRSF, and UPL rose sharply in late August to above the soil surface in response to heavy precipitation from hurricanes Dennis, Floyd, and Irene. In fact, some of the monitoring wells were overtopped by standing water during this period. Water tables remained above or near the soil surface in the RW and NRSF through December 1999. In contrast to the RW and NRSF ecosystems, the water table in the upland site was deeper, less variable, and tended to recede more rapidly after flooding (Figure 2ii).

In terms of jurisdictional hydrologic success criteria, water tables in the RW were within 30 cm of the soil surface for a mean of 98 consecutive days (mid-March-late June) in 1998. The 95% confidence level (CL) associated with this mean value (n = 15 wells) was 8 days. Ninety eight days corresponded to 40% of the jurisdictional growing season and to 110% of the time that water table depths in the NRSF were within 30 cm. By comparison, mean water table

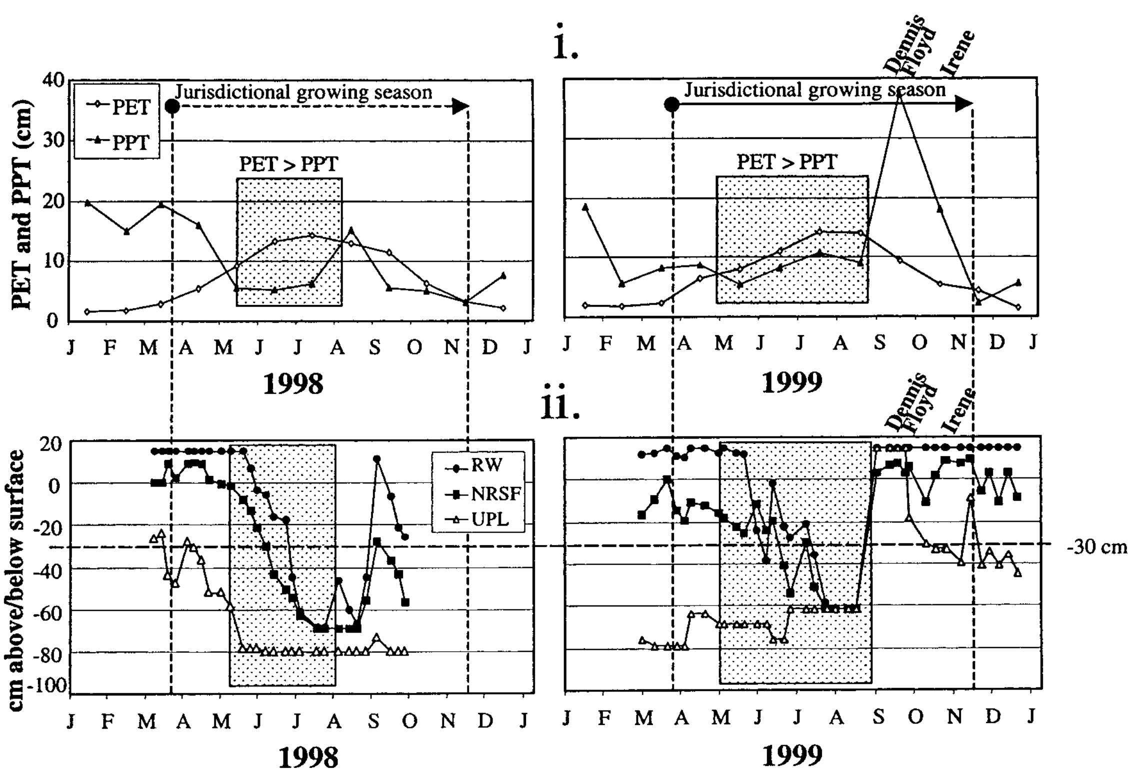

Figure 2. i. Total monthly potential evapotranspiration (PET) and precipitation (PPT) for 1998 and 1999. Shaded parts of the figure show when PET > PPT. The dashed vertical lines indicate the start and end of the jurisdictional growing season. 2ii. Mean water table depths for the restored wetland (RW), non-riverine swamp forest (NRSF), and the upland (UPL) in 1998 and 1999. The dashed horizontal line indicates the minimum depth above which water tables in the RW must occur to meet jurisdictional hydrologic success criteria. Dates of 3 hurricanes (Dennis, Floyd, and Irene) that passed through North Carolina in the fall of 1999 have been included. From September-December 1999, water levels in the RW and NRSF overtopped monitoring wells and are reported as + 20 cm.

depths of the NRSF were within 30 cm of the surface for an average of 89 consecutive days (95% CL = 15 days, n = 3) or 36.6% of the growing season in 1998. In early September, water tables of both the RW and NRSF rose slightly, with those in the RW being within 30 cm of the surface for the rest of the month, or another 11% of the growing season. In 1999, water tables in the RW were within 30 cm for a mean of 89 consecutive days (mid-March-mid-June). The 95% CL associated with this mean value was 12 days. This corresponded to 36.8% of the growing season, or 92% of the NRSF. By comparison, mean water table depths of the NRSF were within 30 CM for 96 consecutive days (95% CL = 0 days, n = 3) or 39.7% of the growing season in 1999. Water tables in the RW again rose to within 30 cm from mid-June to mid-July in 1999 (32 days in total), for another 13% of the growing season. Furthermore, in response to heavy precipitation from the 3 hurricanes, water tables of the RW and NRSF rose to at or above the soil surface in September 1999 and remained there another 76 days or 31.4% of the growing season. Thus, water tables were within 30 cm of the surface in the RW for 51% of the total growing season in 1998 and for 81% of the total growing season in 1999.

Soil properties

Bulk densities (D_bs) of the AG and RW ecosystems ranged from 0.31 to 1.67 $g \cdot cm^{-3}$. By comparison, D_bs of the NRSF and POC ranged from 0.10 to 1.37 $g \cdot cm^{-3}$ and 0.11 to 0.23 $g \cdot cm^{-3}$, respectively. Bulk density in the POC soils was almost uniform with depth, and showed much lower variability in comparison with the AG, RW, and NRSF (not shown). The D_b of the NRSF was similar to the POC in the upper 20 cm, but not at greater depths.

The four ecosystems all had relatively similar total carbon (C_t) in the upper soil pools as listed in the following order: AG (173 $mg \cdot cm^{-3}$) > NRSF (131 $mg \cdot cm^{-3}$) > RW (111 $mg \cdot cm^{-3}$) > POC (91.6 $mg \cdot cm^{-3}$) (Table 1). The AG and NRSF had slightly higher C_t than the RW and the NRSF. The RW contained about 36% less C_t than did the AG. The NRSF had the highest C_t in the lower 40–100 cm pool, followed by the AG, RW, and POC (Table 1). The AG and RW had higher C_t in upper than lower soil pools, whereas C_t in reference ecosystems did not appear to change as much with depth. The standard errors of C_t were greatest in the AG and NRSF, intermediate in the RW, and smallest in the POC.

The pattern of total nitrogen (N_t) in the upper 40 cm was similar to that of C_t: AG (5.29 $mg \cdot cm^{-3}$) > NRSF (4.38 $mg \cdot cm^{-3}$) > RW (3.31 $mg \cdot cm^{-3}$) > POC (2.61 $mg \cdot cm^{-2}$) (Table 1). As with C_t, the AG and RW had higher N_t than the POC and similar N_t to that of the NRSF. The RW had about 37% less N_t than did the AG. Total nitrogen in the upper soil pools of the AG and RW was approximately double that of the lower soil pools (Table 1). Reference ecosystems also had higher N_t in the upper soil pools although this difference was not as pronounced as with the AG and RW. Again, standard errors of N_t were greatest in the AG and NRSF, intermediate in the RW, and smallest in the POC.

Total phosphorus in the upper soil pool occurred in the following order: AG (0.29 $mg \cdot cm^{-3}$) > RW (0.25 $mg \cdot cm^{-3}$) > NRSF (0.12 $mg \cdot cm^{-3}$) > POC (0.07 $mg \cdot cm^{-2}$) (Table 1). The AG and RW had approximately twice the P_t in the upper 0–40 cm than that of the NRSF and four times that of the POC. Total phosphorus was about 16% less in the RW than in the AG. Additionally, P_t was greater in the upper 0–40 cm than in the lower 40–100 cm for all the ecosystems. Again, standard errors were highest for the AG and NRSF.

Table 1. Total soil carbon (C_t), nitrogen (N_t), and phosphorus (P_t) content in the upper and lower pools of the agricultural (AG), restored wetland (RW), non-riverine swamp forest (NRSF), and pocosin (POC). Data presented are means ± 1 SE.

Total nutrient content in the upper (0–40 cm) and lower (40–100 cm) soil pools ($mg \cdot cm^{-3}$)				
Ecosystem (n)	Depth (cm)	C_t	N_t	P_t
AG (6)	0–40	173 (58.5)	5.29 (1.29)	0.29 (0.08)
AG (9)	40–100	105 (48.8)	3.04 (0.98)	0.17 (0.06)
RW (16)	0–40	111 (16.6)	3.31 (0.50)	0.25 (0.05)
RW (24)	40–100	51.1 (7.77)	1.37 (0.24)	0.12 (0.01)
NRSF (6)	0–40	131 (34.3)	4.38 (1.22)	0.12 (0.03)
NRSF (9)	40–100	130 (47.5)	3.50 (1.28)	0.07 (0.02)
POC (6)	0–40	91.6 (2.86)	2.61 (0.05)	0.07 (0.00)
POC (9)	40–100	78.4 (0.65)	1.83 (0.06)	0.03 (0.00)

Water extractable phosphate was highest in the upper 0–40 cm of the AG. Levels of PO_{4w} in the AG were about twice as much as those in the NRSF and POC, and were about three times that of the RW (Figure 3i). Similar to PO_{4w}, N_{ex} was highest in the surface soil of the AG. Extractable inorganic nitrogen was at least 4 times higher in the AG, RW, and NRSF than in the POC. Furthermore, N_{ex} was higher at the surface than at depth for all land-uses except for the POC (Figure 3ii). The POC had very low N_{ex} in both the upper and lower soil pools (Figure 3ii).

Mean Ca_{ex} content at the 0–20 cm depth in the AG and RW was thirty times greater than that of the reference systems (Figure 4i). Despite the large differences in Ca_{ex} among ecosystems in the upper 0–20 cm, all appeared to have relatively similar Ca_{ex} at depths of 80–100 cm. Exchangeable calcium also appeared to be slightly higher at the surface of the RW than in the AG, but this trend disappeared below the depth of 60 cm.

Like Ca_{ex}, Mg_{ex} was greatly elevated in the surface of the AG and RW compared to the reference ecosystems (Figure 4ii). Exchangeable magnesium concentrations were lower than Ca_{ex} concentrations for

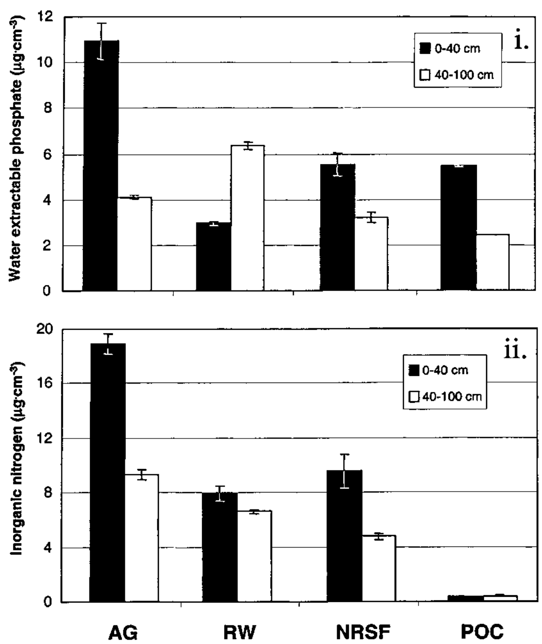

Figure 3. Pooled upper (0–40 cm) and lower (40–100 cm) soil pools of water extractable phosphate (i) and inorganic nitrogen (ii) from the four ecosystems. Data presented are means ± 1 standard error.

the AG and RW but not for the reference ecosystems. Variability of Mg_{ex} in the AG and RW also appeared to be greater than variability of Ca_{ex}. Depth profiles of K_{ex} and Na_{ex} (not shown) were more difficult to interpret. Exchangeable potassium was generally higher in the AG and RW systems (20–90 $\mu g\ cm^{-3}$) than in the reference ecosystems (4–60 $\mu g\ cm^{-3}$). Na_{ex}, on the other hand, was generally higher in the reference systems (7–50 $\mu g\ cm^{-3}$) than in the AG and RW (3–20 $\mu g\ cm^{-3}$). Soil pH was higher in the AG and RW than in the reference ecosystems throughout the soil profile, although these differences were most apparent in the upper 0–40 cm (Figure 4iii).

To compare the overall nutrient status of the four ecosystems a PCA was conducted on the upper 0–20 cm data for all the soil chemistry parameters. The first three principal components axes were significant according to the broken-stick eigenvalue test (Legendre and Legendre 1988). Axis 1, 2, and 3 recovered 47%, 19%, and 14% of the total variance in soil data (Figure 5). Parameters with relatively strong positive loadings on axis 1 were C_t (0.41), N_t (0.40), Mg_{ex} (0.38), K_{ex} (0.36), Ca_{ex} (0.34), and NO_3-NO_2 (0.31). There were no negative loadings on axis 1. Strong positive loadings on axis 2 occurred for pH (0.57) and P_t (0.45), while strong negative loadings were found for Na_{ex} (−0.45). Ammonium (0.45), NO_3-NO_2 (0.39), and P_t (0.36) all had strong positive loadings and Mg_{ex} (−0.36) and Ca_{ex} (−0.31) has strong negative loading on axis 3. Axis 1 represented a gradient from high to low C_t, N_t, and base cations, axis 2 represented a pH and P_t gradient, and axis 3 was most strongly related to exchangeable N fractions. The two reference communities were grouped together at the low ends of axis 1 and axis 2. The POC soils were positioned at slightly higher values on axis 1, indicating higher C_t, and lower values on axis 2, indicating lower pHs. The NRSF soils, on the other hand, tended to have lower C_t and higher pHs. In this ordination, the NRSF sites were similar, if not identical, to the POC, as the POC and NRSF cores were tightly clustered on the PCA diagram. The AG and RW soils, on the other hand, were well mixed and positioned over a wide range of values on both axes. One point from the AG was positioned at an extremely high value of axis 1 and did not group with the other points.

To determine which soil variables were the best determinants of among-ecosystem differences, we also conducted a CART analysis. Results of the

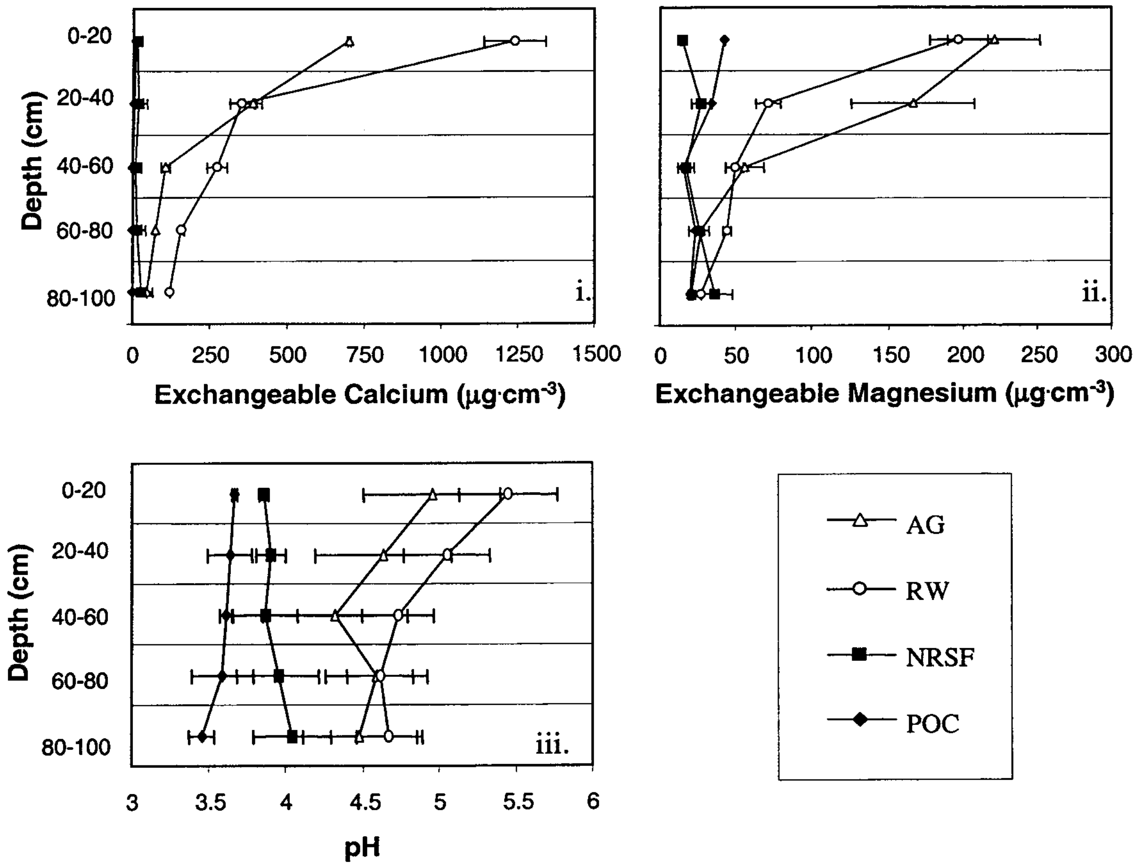

Figure 4. Depth profiles of exchangeable calcium (i), exchangeable magnesium (ii), and pH (iii), in the agricultural (AG), restored wetland (RW), non-riverine swamp forest (NRSF), and pocosin (POC). Data presented are means ± 1 standard error.

CART for surface soils suggested that Ca_{ex} was the most important variable separating the AG and RW from the reference ecosystems, with a breakpoint of 152 μg cm^{-3}(Figure 6). Water extractable phosphate concentration was the most significant division between the AG and RW, with a breakpoint of 7.18 μg cm^{-3} and higher values occurring in the AG. The most significant parameter separating the reference systems was Mg_{ex}. The breakpoint for Mg_{ex} occurred at 19.0 μg cm^{-3} with the POC exhibiting higher values than the NRSF. There were no further significant divisions in the tree. Residual mean deviance of the CART model was 0.4527, and the misclassification error rate was 2/30 or 0.0667.

Outflow water quality

Over the course of our initial water quality monitoring (1998–2000), the concentrations of nutrients in surface water outflows were measured from the AG and reference ecosystems on all 30 sampling dates to compare water quality differences. Outflow from the RW, however, was observed only on 6 out of the 30 dates. Thus, statistical analyses were limited to those six dates when outflow concentrations could be compared for all three ecosystems. During the monitoring period, mean concentrations of soluble reactive phosphorus (SRP) in the AG (110 μg·L^{-1}) were over three times that of the reference (30 μg·L^{-1}), while the RW exhibited an intermediate value (80 μg·L^{-1}). A similar trend was observed for total phosphorus (TP) for which mean concentrations were over three times higher in the AG (170 μg·L^{-1}) than in the reference (50 μg·L^{-1}), with the RW exhibiting an intermediate value (120 μg·L^{-1}) (Figure 7i). In outflows from all three ecosystems, SRP comprised over

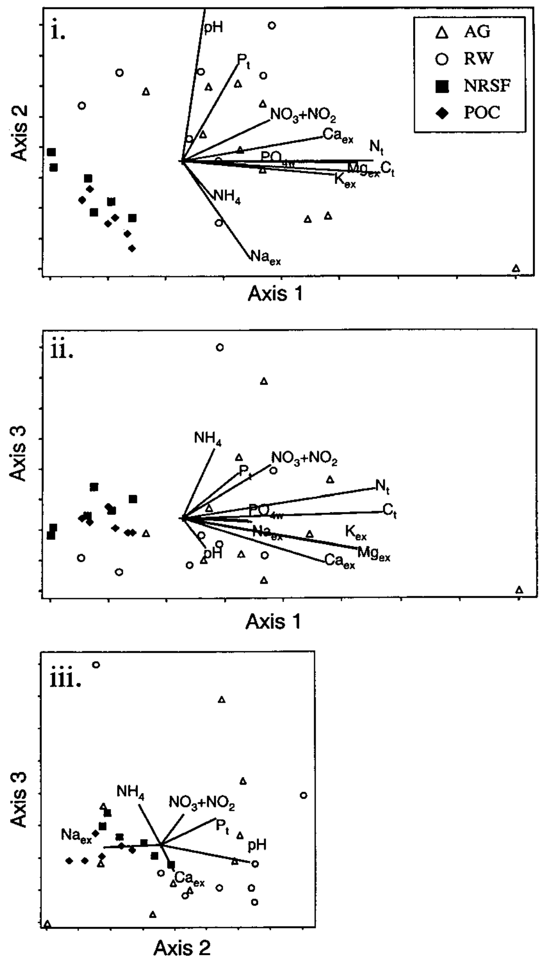

Figure 5. Principal components analysis (PCA) of the soil properties of the agricultural (AG), restored wetland (RW), non-riverine swamp forest (NRSF), and pocosin (POC). Vectors representing the magnitude of correlation with axis 1 and 2 (i), 1 and 3 (ii), and 2 and 3 (iii), for the individual soil variables were overlaid on the PCA. Axes 1, 2, and 3 accounted for 47%, 19%, and 13% of the total variance, respectively. All 11 soil parameters were used for the PCA, but only 6 appeared in figure 5iii due to low loadings for the other variables on axes 2 and 3.

60% of the total phosphorus (TP). When compared to the AG, the outflow of the RW was 27% lower in SRP and 30% lower in TP.

Mean nitrate + nitrite (NOX) was highest in the AG (216 $\mu g \cdot L^{-1}$) and much lower in the RW (7.4 $\mu g \cdot L^{-1}$) and reference (8.2 $\mu g \cdot L^{-1}$). Total nitrogen (TN) concentrations (1000–2000 $\mu g \cdot L^{-1}$ range) were much higher than TP concentrations (30–300 $\mu g \cdot L^{-1}$ range) in all three outflows. In contrast to TP, TN concentrations were highest in the reference (1730 $\mu g \cdot L^{-1}$) and lowest in the RW (1290 $\mu g \cdot L^{-1}$) with intermediate values in the AG (1590 $\mu g \cdot L^{-1}$) (Figure 7ii). Nitrate + nitrite accounted for very little (< 1%) of the TN in the RW and reference outflows, whereas it accounted for about 14% of the TN in the AG outflow. Similar to the results for P, the outflow of the RW was 97% lower in NOX and 19% lower in TN when compared to that of the AG.

Discussion

Hydrology

The seasonal pattern of hydrology of the RW and NRSF, arising from the balance between precipitation and evapotranspiration, was similar to those reported in previous studies of Carolina bays (Schalles and Shure 1989; Kirkman and Sharitz 1994; Lide et al. 1995). It appeared that, despite 30 years of artificial drainage and alteration of natural hydrology, the filling of ditches in the RW almost immediately reestablished wetland hydrology. The fact that wetland hydrology has been reestablished bodes well for Barra Farms, as a previous study suggested that Carolina bays are resilient, and if proper hydrologic regimes are restored, they can be expected to recover toward functioning wetland conditions (Kirkman et al. 1996). Though not specifically measured, the sporadic flow from the RW, as opposed to continuous flow from the AG, suggested that the RW was better at retaining precipitation inputs. This indicated that the RW may also provide flood control benefits in the watershed.

As hydrologic restoration appeared to be successful ecologically, it was also successful jurisdictionally. Mean water table depths in the RW were within 30 cm for 110% of the period in which the water table depths in the NRSF were within 30 cm in 1998 and 92.8% of this period in 1999. Thus, in both years the RW well exceeded the minimum hydrologic success criteria. While hydrologic success criteria for the RW was based on the hydroperiod of a reference wetland, hydrologic success criteria usually involves continuously maintaining water tables within 30 cm of the soil surface for at least 12.5% of the jurisdictional growing season (Rheinhardt and Brinson 2000). Had this criterion been used for the RW, water table depths would have been well above 12.5% in both years (one continuous period of 40% in 1998 and three periods of

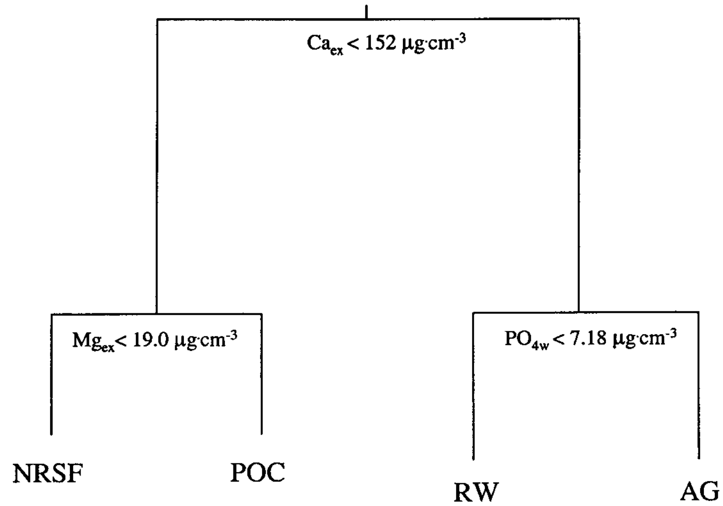

Figure 6. Results of CART analysis for surface soils (0–20 cm) determined that exchangeable calcium (Ca_{ex}) was the most important variable separating reference from AG and RW ecosystems. Exchangeable magnesium (Mg_{ex}) was the best soil parameter to be used in differentiating the reference ecosystems, whereas water extractable phosphate (PO_{4w}) best explained the differences between the AG and RW.

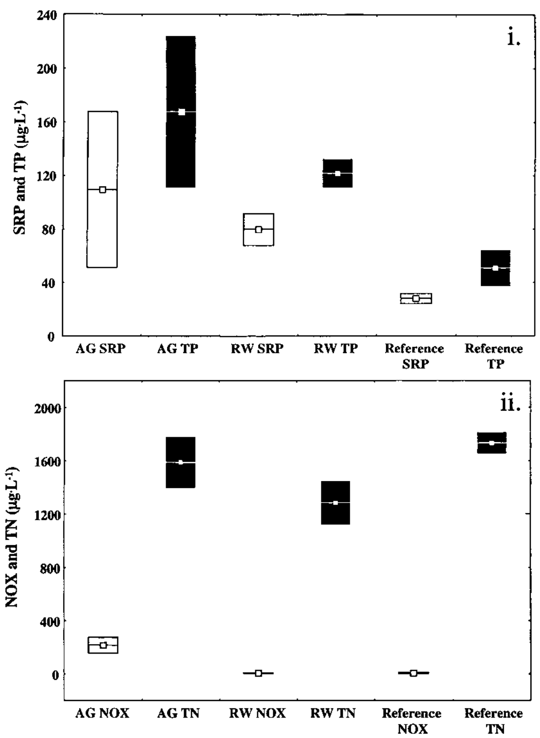

Figure 7. i. Box plots of soluble reactive phosphorus (SRP) and total phosphorous (TP) in the outflow from the agricultural (AG) restored wetland (RW), and reference forests. 7ii. Boxplots of nitrate+nitrite (NOX) and total nitrogen (TN). Center point = mean, box = ± 1 standard error of the mean.

37%, 13%, and 31% in 1999). From these results, we concluded that the RW has successfully met both ecological and jurisdictional hydrology requirements in the first two years of monitoring.

Soil properties

Our data contained trends that revealed important differences in the soil properties of the four ecosystems. We attributed the higher bulk densities in the RW and AG fields to a history of clearing, draining, and plowing. Bulk density results were consistent with, though slightly higher than, a previous study that reported bulk densities in the plow layer (0–40 cm) of cultivated wetland soils in the North Carolina coastal plain to range from 0.35 to 1.23 $g \cdot cm^{-3}$(Eiumnoh 1977).

The AG and RW generally had higher C_t and N_t than the POC and similar C_t and N_t to that of the NRSF. Our values for N_t in the upper 40 cm were similar to those reported for the top 30 cm of unaltered, ditched, and agricultural peatlands in North Carolina (Bridgham et al. 1991). Our results for C_t, did not agree with our expectations based upon previous studies that reported losses of C_t upon entering cultivation (Schlesinger 1986; Richter et al. 1999; Verheyen et al. 1999; Compton and Boone 2000). Certain management activities, such as manure amendment or conservation tillage, can increase soil carbon (Buyanovsky and Wagner 1998; Schlesinger

1999); however, it is not known whether these practices were used here. Alternatively, compaction in the surface soils of the farmed ecosystems may have increased the mass per unit volume of C_t in the soil.

It was surprising that the RW had 36% less C_t in the upper 40 cm than did the AG, given their similar land-use histories. It is possible that C_t was lost due the grading and scraping of surface soil horizons that occurred during the course of restoration to fill ditches and create microtopography. However, despite any possible negative effects on C_t that might have resulted from the restoration process, the RW has retained a relatively high amount of C_t. Thus, our study indicated that agricultural activity in hydric soils does not always result in a massive depletion of soil carbon. The idea that mitigation wetlands require long periods of time to accrue soil carbon comparable to reference wetlands is derived from studies of created wetlands, which usually involved the deposit of dredged material or excavation into upland subsoil; such created wetlands often have very low soil organic matter and associated C_t(Bishel-Machung et al. 1996; Shaffer and Ernst 1999; Stolt et al. 2000). In contrast, the RW at Barra Farms began with a pool of C_t comparable to those of the reference wetlands. Thus, we expect that the residual C_t will provide functional benefits that would not have been present if mitigation had involved creation rather than restoration.

Fertilization of agricultural lands leads to an accumulation of P_t (Compton and Boone 2000), and in this study the ecosystems that have been under agriculture were enriched in P throughout the profile. Besides C_t and N_t, the only parameter for which the AG and RW did not exhibit similar values was PO_{4w}. It was interesting to note that the unfertilized POC had a higher PO_{4w} content than did the RW that had been fertilized until 1997. In addition, the RW was the only ecosystem in which PO_{4w} was greater at depth than it was at the surface (Figure 4i). The depletion of PO_{4w} levels in the upper 0–40 cm pool of the RW might be explained by a variety of factors including the uptake of the rapidly growing wetland vegetation, increased microbial immobilization, the physical process of restoration that disturbed and removed the surface soil horizons, or the cessation of fertilization of the RW with inorganic P fertilizers. Fertilization also left a strong signature of N_{ex} in the upper 0–40 cm of the AG system, while values in the RW and NRSF were comparable, and values of N_{ex} in the POC were very low, similar to values reported in a previous pocosin study (Bridgham and Richardson 1993). Elevated levels of PO_{4w} and N_{ex} may increase the growth of plants in the RW as well as shift the composition of the plant community from what would otherwise occur in the Bay complex.

Of all the soil parameters measured in this study, Ca_{ex}, and Mg_{ex}, and pH were most affected by agricultural land-use. Each of these parameters was greatly elevated in the upper 0–20 cm of the AG and RW in comparison to the upper 0–20 cm of the reference ecosystems. However, at depths of 80–100 cm, land-use differences for these parameters were much less apparent. This suggested that agricultural activity at Barra Farms had the greatest impact upon the upper soil profile. Low soil pH values reported for the reference ecosystems in our study are typical of highly organic soils of pocosins and non-riverine swamp forests of North Carolina (Walbridge 1991; Bridgham and Richardson 1993). While the reference ecosystems had mean pHs between 3.3 and 4.1, the farmed ecosystems had mean pHs between 4.3 and 5.8. This enrichment in exchangeable Ca and Mg and increase in pH was most likely a result of heavy liming of the farmed ecosystems with compounds such as calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$). Our results agreed with those of a study in Norway that investigated the effects of liming on organic soils (Braekke 1999). As with the increased PO_{4w} and N_{ex}, elevated Ca_{ex}, and Mg_{ex}, and pH in the RW may stimulate growth and shift the composition of the plant community.

What we observed in a univariate context with the individual soil variables was also corroborated in the multivariate context of the PCA and CART. In the PCA, the majority of the points of the AG and RW systems were positioned in a space correlated with high nutrient concentrations and pH, while the POC and NRSF were positioned at low nutrient concentrations and pH. The PCA ordination of the soil properties revealed a tight clustering of the reference ecosystems, in contrast to the AG and RW that had shifted their position in the ordination and showed higher variability. While agricultural practices tend to homogenize soil properties within a single field (Paz-Gonzáles et al. 2000), our AG and RW samples included multiple fields that may have experienced different rotations and may not have been managed with the same intensity. This may have lead to higher variability among the AG and RW soil properties. The CART analysis indicated that measurements of Ca_{ex} in wetland soils of the Coastal Plain might serve as a useful indicator of agricultural land-use impacts. The multivariate CART model also indicated that PO_{4w} was the best variable at separating the AG and RW.

Outflow water quality

Water quality from the AG, RW, and reference outflows varied according to the land-use of each system. The AG ecosystem, subject to alteration of hydrology in the form of drainage and soil properties in the forms of plowing and fertilization, had the highest and most variable SRP, TP, and NOX concentrations. The RW, in which wetland hydrology was reestablished and which had not been actively managed in the previous two years, tended to export lower concentrations of nutrients of all forms than did the AG ecosystem. The reference ecosystems exported the smallest amounts of SRP, TP, and NOX, but the largest amounts of TN. Values for the reference outflow fall within or close to ranges reported by Walbridge and Richardson (1991) for SRP (0–26 $\mu g \cdot L^{-1}$), TP (20–140 $\mu g \cdot L^{-1}$), NOX (0–1364 $\mu g \cdot L^{-1}$) and TN (1070–1940 $\mu g \cdot L^{-1}$) in undisturbed pocosins. Our results also agreed with other studies that reported increases in inorganic N and P in outflow from pocosins as a result of ditching (Williams and Askew 1988) and threefold increases in N export and increases in P export by an order of magnitude as a result of agricultural activity (Skaggs et al. 1980). These effects arise from a combination of increased hydrologic export and inability of these wetlands to serve as sinks for fertilizer applied in excess of crop uptake (Walbridge and Richardson 1991). Continued fertilization of the agricultural area at Barra Farms will result in high N and P export in the drainage ditches. As the system becomes more saturated with N and especially P, export of these nutrients in surface water outflows may even increase. By comparison, restoration at Barra Farms, which involved a cessation of fertilization and increased hydraulic retention time, seemed to have mitigated these effects. Our data indicates that SRP, TP, and NOX concentrations in outflow waters from former agricultural fields can be brought closer to those of undisturbed forested wetlands through restoration as shown by reductions in SRP, TP, NOX, and TN concentrations of 30, 27, and 97, and 19% respectively, in the RW as compared to the AG. However, due to the accumulation of phosphorus in the soil of the RW, it may be some time before TP values are as low in the RW as they are in the reference outflow.

Conclusions

Despite 30 years of intensive drainage and agricultural land-use at Barra Farms, wetland hydrology was quickly and effectively reestablished in the RW by filling in ditches. Both the seasonal pattern of water table depths and duration of time that water table depths were within 30 cm of the soil surface were very similar in the restored and reference monitoring wells. The major effects of agricultural land-use on wetland soil properties were to increase D_b, P_t, Ca_{ex}, Mg_{ex}, and pH of the AG and RW soils. Upon comparing the soils of the AG to RW, the main differences between the two ecosystems were in C_t, N_t, and PO_{4w}, which were higher in the upper 0–40 cm of the AG. Interestingly, the soils of the RW had relatively high amounts of C_t that may enhance ecosystem function in the early years of development. Even though large changes in most soil properties weren't observed following restoration, it appeared that restoration activities did provide water quality benefits as shown by the 30, 27, 97, and 19% lower concentrations of TP, SRP, NOX, and TN respectively, in the RW as compared to the AG. Further process-level research is necessary to determine which mechanisms are responsible for providing these water quality benefits.

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