Aqua-2 : Forested Wetlands

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What are the history, status, and likely future of forested wetlands in the South?

1 Key Findings

- Approximately half of U.S. wetlands present in colonial times have been lost primarily due to agriculture. The South had approximately 35 million acres of forested wetland remaining by 1996, 91 percent of which were riverine wetland.

- Rates of loss--change from wetland to nonwetland--were greatest from the 1950s to the 1970s. Since then the rates have slowed but losses are still occurring due to agriculture, urban and rural development, and silviculture

- According to the National Wetland Inventory, 3.5 million acres of southern forested wetland underwent changes between 1986-1997. Ninety percent of the changes were conversions to another wetland or aquatic habitat type. Of these conversions 95 percent were to scrub-shrub or emergent wetlands. During this same time period approximately 119,000 acres of forested wetland went into urban and rural development, 112,000 acres were converted to agriculture, and 102,00 acres underwent intensive silviculture.

- As of 1997, Georgia, Florida, and Louisiana have the greatest amount of forested wetland in the South followed, in descending order by Mississippi, South Carolina, North Carolina, Arkansas, Texas, Alabama, Virginia, Tennessee, and Kentucky.

- Restoration has been attempted primarily in riverine wetlands in the Lower Mississippi Valley, but success in restoring wetland acreage and function has been limited. Restoration of other forested wetlands, like mineral soil pine flats, would have to include the reintroduction of fire.

- Offsetting losses of wetland functions through the Section 404 permitting process has not been well documented but appears to have had limited success.

2 Introduction

This Chapter describes the history, status, and likely future of forested wetlands in the South. Key issues include: (1)the quantity of forested wetlands in the South, (2)the quality of forested wetlands in the South (3)how function is affected by impacts associated with development and agricultural and silvicultural conversions; (4)restoration of these wetland systems to replace lost functions; and (5)public policies designed to protect and restore forested wetlands. All these issues are discussed. Due to public concerns about the effects of silvicultural operations on forested wetlands and their surrounding landscapes, special attention is given to changes in condition of forested wetlands caused by silviculture.

2.1 History

Southern forested wetlands have undergone natural and human-induced disturbances for thousands of years. These disturbances have led to the species rich flora and fauna found in these ecosystems today. Even before prehistoric man arrived in the South geologic changes due to plate tectonics, Appalachian Mountain uplift and subsequent erosion, rising sea-levels, and the advance and retreat of glaciers, resulted in ecological changes, species migrations, and shifts in community composition. Warmer climates, beginning about 16,000 years ago caused southern forests to shift from predominantly northern softwood forests to forests dominated by oaks and hickories (Delcourt and others 1993). These climate changes and concomitant sea level rise caused many wetlands to form due to rises in water tables, which often inundated river valleys. Pre-European settlement forests were diverse, with varying tree ages interspersed with openings providing habitat for a diverse range of wildlife (Dickson 1991). Fire, ice storms, tornadoes, hurricanes, insects and diseases disturbed these ecosystems and influenced forest composition (Askins 2001).

In addition to the long-term geologic and climatic changes and the frequent natural disturbances (primarily storms and fire), Native Americans impacted southern forested wetlands by settling and farming the fertile and tillable floodplains from the Little Tennessee River to the Mississippi River (Delcourt and others 1993). Forests were cleared not only for agriculture but also for firewood and stockades. Cleared areas were also burned regularly to prepare themfor planting (Wigley and Roberts 1997). In the 16th and 17th centuries, 80 percent of Native Americans in the south died due to diseases brought by early European explorers. One result was a decline of the Native American agricultural system. Agricultural fields were abandoned and tree growth became established on many acres of forested wetland and upland (HISTORY AND BACKGROUND SECTION). Consequently, the forest vegetation encountered by southern colonists in the mid-1700s was the result of thousands of years of geologic, climatic, and human influence. Growth of forest stands that regenerated after climatic and biologic disturbances, and Native American abandonment affected forest composition and age at the time of European settlement. For instance, in the Coastal Plain, abandoned agricultural fields probably supported extensive tracts of pure pine (Allen and others 1996). The forests encountered in the 1700s were not the vast, unbroken expanses of giant trees romantically portrayed early in the 19th century (Wigley and Roberts 1997, Delcourt and others 1993). Many were young stands resulting from natural and man-induced disturbances. The flora and fauna of these ecosystems were and are adapted to disturbance. In the case of mineral soil pine flats, they require fire to maintain them. Therefore, disturbance is a natural and often forgotten component of forested wetland systems that is necessary in considering their restoration.

2.2 Definitions

What is a wetland? Current definitions include three main components: (1) the presence of water at the surface or within the root zone, (2) unique soil conditions that differ from adjacent uplands, and (3) vegetation adapted to the wet conditions (Mitsch and Gosselink 2000). Precise wetland definitions are needed by wetland managers and regulators as well as wetland scientists (Mitsch and Gosselink 2000). The wetland regulatory definition used to establish Fderal jurisdiction for the wetland permitting program under Section 404 of the Clean Water Act is:

"those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas (33 CFR 328.3(b); 1984)".

The wetland definition adopted by scientists in the U.S. Fish and Wildlife Service for the purposes of inventorying wetland resources in the United States is:

"Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water…. Wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes, (2) the substrate is predominantly undrained hydric soil, and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year (Cowardin and others 1979)."

Once a wetland-upland boundary is defined and delineated, the quality or capability of the wetland to function, becomes a concern. There is great diversity in the types of wetlands in the South, the functions they perform, and the goods and services they provide society. To deal with this diversity, wetlands are grouped according to factors that substantially contribute to wetland functioning. Hydrogeomorphic Classification (HGM) (Brinson 1993) groups wetlands based upon their landscape position, water source, and hydrodynamics. By grouping or classifying wetlands using the Hydrogeomorphic Classification the presumption is that wetlands with similar landscape position, water source and hydrodynamics will function similarly. In the Southern United States, most forested wetlands are classed as riverine, flat, and depression wetland. Much of the following discussion deals with these three classes.

3 Methods

The status of and trends in southern forested wetlands were derived primarily from National Wetland Inventory (NWI) reports (Dahl 1990 Hefner and Brown 1985; Hefner and others 1994, Dahl 2000). Information from these reports was used to develop a composite picture of the acreage and loss of forested wetlands in the South from the 1780s to the present. Acreages were taken directly from the U.S. Fish and Wildlife Service Wetland Status and Trend reports. For the 10 Southeastern States of Kentucky, Tennessee, North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Arkansas. Data for the 1986-1997 time period, generated for this report by the Fish and Wildlife Service, were also used directly. The NWI Status and Trends reports represent the most comprehensive and consistent source of information on forested wetland conversions and losses over the last 200 years.

Information from the National Resources Inventory (NRI) prepared by the Natural Resources Conservation Service (NRCS) and the Forest Inventory and Analysis (FIA) units of the USDA Forest Service were used to fill gaps in information about impact and restoration acreages, and changes in forest type and ownership. National Wetland Inventory and NRI data have similar geographic coverage but are not directly comparable because NRI does not classify wetlands in the same manner as NWI and does not include Federal land or coastal areas in its estimates. The FIA forested wetland data cover only five States -- Virginia, North Carolina, South Carolina, Georgia, and Florida. To date FIA has collected wetland data at only one point in time for each state. Thus, data does not represent changes in forested wetland acres over time. Since NRI and FIA data are limited geographically and temporally, NWI data are the primary basis for the status and trend numbers reported herein.

Literature, including hydrogeomorphic approach models for low-gradient riverine wetlands, pine flatwood wetlands, hardwood flat wetlands, and forested depressions were reviewed to develop hypotheses about the effects of alteration on the structure and function of forested wetlands. Hypothesized impacts were then checked against scientific studies done in similar wetlands where available. Predominant forested wetland types in the South (Messina and Conner 1998) were placed in HGM classes. Functional assessment models for those classes and/or subclasses were then reviewed to hypothesize, based upon structural alterations to the wetland, the impacts of alterations by silviculture, agriculture, or development. Due to the large geographic area encompassed by the Southern Forest Assessment (13 States) and the large variability in on-site wetland and surrounding landscape conditions, the estimated impacts are generic. The specific projects must be individually assessed. The generic assessments of impacts described here do provide useful insights into the ecological ramifications of these activities, the fate of wetlands which have been modified, and potential hypotheses for additional research. Wetland restoration literature was reviewed, as were ongoing studies on the extent and success of wetland restoration. NRI and data from the Wetland Reserve Program (WRP) administered by NRCS was also used to estimate the number of acres where wetland restoration have been attempted. The assumption with WRP data is that acres enrolled in this program result in a gain in forested wetland.

4 Data Sources

Status and trends of southern forested wetlands were derived from National Wetland Inventory (NWI) reports for the United States and the Southeast (Dahl 1990, Dahl 2000, Hefner and Brown 1985; Hefner and others 1994). These reports also provided information on the causes of forested wetland loss. The NWI was undertaken by the U.S. Fish and Wildlife Service to provide a comprehensive inventory of the Nation's wetlands. The NWI is conducted at 10-year intervals. Gains and losses of wetlands are estimated using aerial photographs, soil surveys, topographic maps, and field work on a permanent set of randomly selected points (Shepard and others 1998, Dahl 2000). These photos are analyzed for a selected 10-year interval to detect changes in wetlands. Quality control is included throughout the data collection and analysis stages, and 21 percent of the plots are field verified (Dahl 2000). Studies have been completed for the 1950s to 1970s, 1970s to 1980s, and 1980s to 1990s.

Since NWI is used as the primary source of status and trends data for this chapter, terminology used by NWI in reporting changes in forested wetlands (Dahl 2000) is important to understand. Terms regarding wetland types and land-use definitions can be found in Dahl (2000). However, two pivotal terms are defined here. "Conversion" is a change in vegetative cover on an area that is still a wetland. In other words, when a forested wetland is "converted" it remains a wetland (i.e., soils and hydrology remain intact) but the dominant vegetation is changed. Wetland "loss" is a change in which an area no longer has the hydrologic characteristics of a wetland. “Losses” involve the detection on high resolution aerial photographs of: (1) significant hydrologic alterations such as large ditches and levees, (2) soil alterations such as filling or leveling, and (3) upland vegetation indicating the wetland character of a site has been removed.

The National Resources Inventory (NRI), prepared by the Natural Resources Conservation Service, is an inventory of multiple natural resource conditions on non-Federal land in the United States (Shepard and others 1998). The purpose of the NRI is to provide information for policymaking in natural resource conservation programs at State and Federal levels. The NRI is based upon stratified random samples distributed throughout the country. Data are collected using aerial photographs and ancillary data and by making select field visits.

Forest Inventory and Analysis (FIA) data gathered by the USDA Forest Service also were used in this report. The purpose of FIA is to provide information on forest resources at the local, State and national levels. The evaluations are State-by State multiple resource inventories of land use, timber, wildlife, range, recreation, water and soils completed on a 7- to 10-year cycle. Data in this report were collected between 1989 and 1998 during the forest surveys in Virgina, North and South Carolina, Georgia, and Florida from field plots that met Federal wetland criteria (areas having wetland soils, plants and hydrology) (Brown and others in press).

Scientific literature including HGM models for low gradient riverine wetlands (Ainslie and others 1999, Smith and Klimas in press), pine flatwood wetlands (Rheinhardt and others 2001), hardwood flat wetlands (Smith and Klimas in press) and forested depressions (Smith and Klimas in press), were reviewed as a means to hypothesize the effects of conversion on the structure and function of forested wetlands. Information on land ownership and timber harvests came from FIA data and Brown and others (in press). Wetland restoration literature and university studies on the extent and success of wetland restoration also were reviewed.

5 Results and Discussion

5.1 Status of Forested Wetlands

In colonial times (circa 1780) the Conterminous United States had approximately 221 million acres of wetlands (Dahl 1990). These wetlands had been, and would continue to be, affected by natural and anthropogenic disturbances. Over the next 200 years (circa 1980) the total wetland area in the country was reduced by over 50 percent to 104 million acres (Table 1). Losses are primarily attributable to clearing and draining for agriculture. Frayer and others (1983) suggest that the greatest losses between the 1950s and the 1980s were in freshwater forested wetlands. Abernethy and Turner (1987) estimated losses of forested wetlands were up to 5 times greater than those of nonforested wetland between 1940 and 1980. Almost 7 million forested wetland acres were lost in the Lower Mississippi Valley alone.

Hefner and Brown (1985) reported that 47 percent (48.9 million acres) of the wetlands in the Conterminous United States occur in 10 Southeastern States (Kentucky, Tennessee, North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Arkansas). In addition, 65 percent of all the forested wetlands in the Conterminous United States occurred in these 10 Southern States. Table 2 provides an estimate of total wetland acres, forested wetland acres and forested wetland change in Southern States. Hefner and Brown (1985) reported that for the period between the 1950s and 1970s the South sustained the greatest wetland losses in the country. Forested wetland losses were attributed to massive clearing and drainage projects designed to bring wetlands into agricultural production. As of the 1970s Hefner and Brown (1985) reported that 80 percent of the 25 million acres of forested wetland in the Lower Mississippi River Valley had been lost to agriculture. Major losses of pocosins and Carolina Bays in North Carolina were attributed to agriculture and peat mining. Overall, forested wetland acres in the South declined by 16 percent between the 1950s and 1970s. (Table 1).

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Hefner and others (1994) reported that approximately 3.1 million acres (9 percent) of forested wetlands in the South were lost or converted in the 1970s and 1980s (Table 1). Almost 69 percent of the South's forested wetland losses were recorded in the Gulf-Atlantic Coastal Flats and Lower Mississippi Alluvial Plain(Figure 1). The Gulf-Atlantic Coastal Flats of North Carolina and the Lower Mississippi Alluvial Plain of Louisiana suffered the greatest losses during this time period. Nearly 1.2 million acres were lost in North Carolina presumably to silviculture and agriculture, and nearly 1 million acres of forested riverine wetlands (bottomland hardwood wetland) were severely affected primarily by agriculture in the Lower Mississippi Alluvial Plain. Although the net rate of wetland loss declined from 386,000 acres per year from the 1950s to 1970s, to 259,000 acres per year from the 1970s to 1980s the rate at which forested wetlands declined accelerated (Hefner and others 1994). Forested wetlands in these 10 Southeastern States were lost or converted at an average rate of 276,000 acres per year from the 1950s to 1970s but lost at an average rate of 345,000 acres per year from the 1970s to 1980s (Hefner and others 1994).

The drop in overall wetland loss rate resumed between 1986 and 1993 declining 80 percent to

58,500 acres per year for the Conterminous United States (Dahl 2000). The change in forested wetland acres during this time period was approximately 3 percent (Table 1). Dahl (2000) estimated that nationally 4 million acres of forested wetland underwent some change in condition between 1986 and 1997. Most were converted to freshwater shrub wetlands by timber harvesting or other processes that removed the tree canopy but retained the wetland character. Table 3 shows a breakdown of the number of palustrine (freshwater) forested wetland acres lost or converted by activity and by State for the period of 1986-1997, recorded by NWI, for the 13 Southern States included in the Southern Forest Resource Assessment. Georgia, North Carolina, Mississippi, South Carolina, and Alabama showed the greatest change in forested wetland area -- over 300,000 acres/State. In each of the above cases over 80 percent of the change in wetland type resulted from a "conversion" from forested wetland to shrub-scrub or emergent wetland. Overall, 90 percent of the change in forested wetland acres in the 13 Southern States resulted from these types of conversions. Ninety-five percent of the conversions of forested wetland were to shrub-scrub or emergent wetland types.

According to NWI, "losses" (changes from wetland to nonwetland) accounted for 10 percent of the change in forested wetlands in the South or 356,000 acres between 1986 and 1997. Thirty-three percent of the losses were due to urban/rural development, 31 percent to agriculture and 29 percent to silviculture. The remaining 7 percent of losses of forested wetland were attributed to "other land uses". The NWI attributes losses to silviculture, if drainage occurs on any forested site (including those in agricultural or urban landscapes) such that a shift from wetland vegetation to upland vegetation is apparent (C. Storrs, pers comm.) The three States with the greatest reported losses due to silviculture were Louisiana, Georgia and Arkansas. The three States with the greatest loss due to agriculture are Mississippi, Georgia, and Tennessee. The three States with the greatest losses to development were Florida, Mississippi, and Georgia.

Direct comparisons of various wetland inventories is difficult due to the dynamic nature of wetlands, differences in the time period in which the inventories are made, differences in geographic cover, and differences in sampling and delineation protocols (Shepard and others 1998). However, indirect comparison of the NWI and NRI results are interesting. From 1982-1987 the National Resources Inventory data indicated that urban, industrial, and residential land uses caused 48 percent of the wetland losses in the Conterminous United States. Agriculture was responsible for 37 percent of wetland losses, while the remaining 15 percent were converted to barren land, open water, or forest (Brady and Flather 1994). For this time period the NRI data suggest a shift from agriculture to urban development as the major cause of wetland conversion. From 1982-1992 NRI data indicate that 55 percent of the total wetland loss in the Nation occurred in the 12 Southern States. During this period, wooded wetlands showed the lowest loss rate in recent decades. According to NRI, 75 percent of the losses from 1982-1992 were due to development (Shepard and others 1998). The updated 1997 NRI report shows that 12.5 percent of the losses of wetlands in the South are attributable to silviculture, 18.4 percent to agriculture, 58 percent to development, and 10.1 percent to miscellaneous climatic and hydrologic changes (Figure 2). Differences in definitions for attributing loss are a primary reason for discrepancies in wetland loss and conversion estimates between NWI and NRI (C. Storrs pers. comm.).

Land ownership patterns of forested wetlands have been summarized for 5 of the 13 Southern States by Brown and others (in press). Abput 60 percent of the wetland timberland in Virginia, North and South Carolina, Georgia and Florida is privately owned. Forest industry owns 28 percent of the land, and the public owns 12 percent (Brown and others in press). Data from the other 8 Southern States is unavailable. Of the wetland timberland in the five Southern States for which data are available, 62 percent is covered with bottomland hardwoods, 25 percent with pine plantations and natural pine stands, and 10 percent oak-pine stands. Most of these forest types are in private nonindustrial ownership except for pine plantations, which are largely owned by forest industry (68 percent)(Brown and others in press). The percentage of timberland in wetland and the expected increase in timber harvest in the South (CHAPTER TIMBER-1) indicate the likelihood of additional wetland modifications due to silvicultural activities.

5.1.1 Likely Future of Forested Wetlands in the South

Projecting changes in forested wetlands in the South is difficult, if not impossible, because of the wide variety of scientific, societal, and economic factors that affect the forested wetland resource. Science has provided a great deal of information on how wetlands function and how man's activities affect those functions. However, much information is not known and is difficult to discern. The values that people associate with forested wetlands vary greatly. They range from valuing old-growth forest to the exclusion of timber harvesting to valuing forested wetlands as merchantable timber or nothing more than potential development sites. Economic factors are important because, ultimately, wetlands are lost to development, agriculture, or converted to intensive silviculture based upon economics.

This Section of the Chapter addresses changes in wetland condition, with particular emphasis on silviculture, current policies, and the efficacy of current forested wetland restoration efforts in the South. Additional information about forces of change in southern forests can be gained from other Chapters in this Assessment.

Forested wetland types in the South are highly variable, ranging from baldcypress swamps to scrub-shrub bogs that undergo cycles of wildfire. Due to these differences in vegetation, hydrology, landscape position and degree of alteration wetlands differ in the functions they perform and their ability to perform those functions (Brinson and Rheinhardt 1998). Wetland functions can be simply described as the things that wetlands do. Many of these functions, such as surface and groundwater conveyance and storage, nutrient cycling, and organic carbon export provide societal benefits, goods, and services, (such as floodwater storage, water quality enhancement, and wildlife habitat). Because of the large geographic area encompassed in this study (13 States) generalizations about forested wetlands must be made. The hydrogeomorphic classification (Brinson 1993) and functional assessment approach (Smith and others 1995) provide a means to make these broad generalizations about similar forested wetland types, the functions they perform, and the effects of certain activities on those functions.

The predominant forested wetlands in the South can be classified into four hydrogeomorphic

(HGM) classes: (1) riverine, (2) organic soil flats, (3) mineral soil flats and (4) depressions (Brinson 1993). Wetlands in each class occupy similar landscape positions and havesimilar hydrology. The presumption in HGM is that if wetlands occupy similar landscape positions so that the water, which drives wetland functions, comes from similar sources and flows into and out of wetlands in similar ways, the ecological processes (functions) that make wetlands important will be similar. This is a logical simplification that facilitates the discussion of wetland ecological characteristics and processes and human impacts.

In general, southern deepwater swamps, major alluvial floodplains and minor alluvial floodplains (Messina and Connor 1998) can be combined into the riverine class. Carolina Bays, Pondcypress swamps, and mountain fens can all be classified as depressions with similar depressional geomorphology and low energy surface runoff or groundwater hydrodynamics. Wet pine flatwoods are classified as mineral soil pine flats due to their soil composition, flat topography, and the predominance of rainfall for their hydrology. Pocosins are classified as organic soil flats. Their topography and hydrology are similar to those of mineral soil flats, but soil composition is dominated by peat. The flats class encompasses areas dominated by pines and by hardwoods. However, mineral soil pine flats will be the predominant flats class discussed in this Chapter due to their extent, fire ecology, and vulnerability to alteration. Based upon the acreage estimates in Table 4, riverine is the predominant HGM class in the South, followed by flatwoods and depressions.

In general, the hydrologic regime is one of the main factors controlling ecosystem functions in all wetlands and differentiating wetland types. The timing, duration, depth, and fluctuations in water level affect biogeochemical processes and plant distribution patterns. The rate, magnitude, and timing of biogeochemical processes are determined by hydrology and the living components of an ecosystem. For instance, primary producers (plants) assimilate nutrients and elements in soil, and use energy from sunlight to fix carbon. When they die, they depend upon microbial organisms in soil to transform carbon and nutrients such as nitrogen and phosphorous to forms that are available to other plants. Therefore, wetland conditions that maintain plants and soil microbial populations are those that drive characteristic biogeochemical processes. These processes help to sustain the wetland plant community, which provides much of the structure required by wildlife. The integrated combination of water, soils, and plants sustains the ecosystem and provides many of the values attributed to wetlands.

5.1.2 Riverine Wetlands

Riverine wetlands occur in floodplains and riparian corridors in association with stream channels (Brinson 1993). The dominant water source for these wetlands is from the stream channel via overbank flooding or through subsurface connections between the stream channel and the wetland. Riverine wetlands lose surface water in four ways: (1) surface flow of floodwater to the channel, (2) subsurface water flow to the channel, (3) percolation to deeper groundwater, and (4) evapotranspiration. Evapotranspiration includes evaporation from soil and water surfaces and movement of water through plants to the atmosphere. Unimpacted southern forested riverine wetlands typically extend perpendicularly from a stream channel to the edge of the stream's floodplain. They have unaltered soils and a mature tree canopy, and they range from narrow riparian strips in low-order streams to broad alluvial valleys several miles wide (Sharitz and Mitsch 1993). This wetland ecosystem occurs in the Lower Mississippi River Valley as far north as southern Illinois and along many streams that drain the South Atlantic Coastal Plain into the Atlantic Ocean.

The functions of riverine wetlands are closely tied to flooding of adjacent streams and the soil and vegetation which result. Flooding is important both ecologically and societally because floodwaters move sediments and nutrients into and out of the wetlands. Wetlands detain floodwaters and prevent or minimize flood damages downstream (Sharitz and Mitsch 1993, Kellison and others 1998, Mitsch and Gosselink 2000). Riverine wetlands enhances water quality by intercepting sediments, elements, and compounds from upland or aquatic nonpoint sources of pollution. They permanently remove or temporarily immobilize nutrients, metals and other toxic compounds (Ainslie and others 1999). Hydrologic, soil, and biological factors determine the ability of a riverine wetland to sustain a characteristic plant community. The vegetation of low gradient alluvial riverine wetlands is extremely diverse (Sharitz and Mitsch 1993). The ability to maintain a characteristic plant community is important because of the intrinsic value of the plants themselves, and the many attributes and processes of riverine wetlands influenced by the plant community. For example, plants influence primary productivity, nutrient cycling, and the ability to provide a variety of habitats necessary to maintain local and regional diversity of animals (Brinson 1990, Gosselink and others 1990, Harris and Gosselink 1990). Riverine wetlands provide habitats for a diversity of terrestrial, semiaquatic, and aquatic organisms. They provide access to and from uplands for completion of aquatic species' life cycles, provide refuges and habitat for birds, and act as conduits for dispersal of species to other areas. Most wildlife and fish species in riverine wetlands depend on the amount and timing of flooding, the variable topography which allows different plants and animals to become established, forest tree composition and structure, and proximity to other habitats. Riverine wetlands also must be viewed in their landscape context or in relation to the other land uses around them. Generally, the continuity of vegetation, the connection between specific vegetation types, the presence and size of corridors between upland and wetland habitats, and corridors among wetlands all have direct bearing on the movement and behavior of animals that use wetlands.

5.1.3 Depression wetlands

These wetlands occur in topographic depressions that allow the accumulation of surface water(Brinson 1993). Depression wetlands may have a combination of inlets and outlets or lack them completely. Potential water sources are precipitation, overland flow, streams, or groundwater/interflow from adjacent uplands. Water typically flows from the outside of the depression to the center. Upward and downward movement of the water table may vary daily to seasonally. Cypress domes and Carolina Bays are typical regional forested wetland types (Messina and Conner 1998) that occur in depressions. Pondcypress domes are poorly drained to permanently wet depressional wetlands that occur in the southeastern Coastal Plain and are abundant in Florida (Ewel 1990). Cypress domes are shallow, circular, nutrient-poor swamps located in depressions on low relief landscapes. They often have an underlying impervious layer of soil that inhibits downward movement of water. These wetlands are called "domes" because the tallest trees are in the center and the smaller trees near the edge give the appearance of a dome. Domes have long-standing, nutrient poor water which is often dominated by precipitation and surface inflow (Mitsch and Goselink 2000). Limited plant growth rates are related to both low flow and lack of nutrient availability.

Carolina bays occur on the Atlantic Coastal Plain from New Jersey to Florida. The water source for Carolina bays ranges from predominantly precipitation to predominantly groundwater. These bays occur in clusters, are commonly elliptical in shape, and are often oriented in a northwesterly to southeasterly direction. Larger, deeper Carolina bays contain lakes, but the majority of them are wetlands with diverse plant communities ranging from shrub-bog pocosins to marshes to hardwood- or cypress-dominated swamp forests. Many bays may become blanketed by an overgrowth of bog vegetation, which compresses lower layers of peat, making them relatively impervious to water movement. The result is a ponding of water, making the depression saturated for long periods of time. Bays are critical breeding sites for amphibians and habitat for birds and other wildlife. They often host rare or endangered plants.

Detention of runoff water is an important depressional wetland function because runoff, or occasional overbank flooding in riparian depressions, alters flood timing, duration, and magnitude. The result is reduced flood flow downstream. Water storage or detention has significant effects on biogeochemical cycling; plant distribution, composition and abundance; and on wildlife populations. Just as in riverine wetlands, nutrient cycling is mediated primarily by two processes: (1) nutrient uptake by plants (primary production), and (2) nutrient release from dead plants for renewed uptake by plants (detrital turnover). Because of their location on the landscape, depressional wetlands, particularly those in lower portions of watersheds, are strategically located to remove and sequester sediments, imported nutrients, contaminants, and other elements and compounds before they can contribute to groundwater and surface water pollution downstream. These contaminants are removed from incoming water by the interaction of water, wetland vegetation, wetland microbes, detrital material, and soil. The primary benefit of this function is that the removal, conversion, and sequestration of compounds by depressional wetlands reduces the load of nutrients and pollutants in groundwater and in any surface water leaving the depressional wetland. Not all depressions are positioned or capable of removing these sediments, compounds, and contaminants. For instance, depressions at the “top” of drainage basins, or those in flat topography, may not receive pollutants from upstream.

Depressional wetlands support many animal populations. They provide habitats within the actual wetland and in conjunction with the surrounding landscape. They maintain regional biodiversity by providing open water, nesting cavities, cover and food chain support for a variety of animals (Ewel 1998). In some regions, Carolina bays are major and critical focal points for breeding and feeding of a large variety of nonaquatic vertebrate and invertebrate animal species. The biomass of animals in these Carolina bays is extremely high compared to adjacent terrestrial habitats or more permanent aquatic habitats (Richardson and Gibbons 1993).

5.1.4 Forested wet flats

In the Southern United States, wet flats occur on poorly drained mineral or organic soils in lowland areas (Harms and others 1998, Rheinhardt and others 2001). Wet flats on organic, or peaty, soils are called pocosins. Pocosins differ from mineral soil flats in both geomorphology and vegetation. Pocosins are located on topographic highs, are dominated by evergreen shrubs, and most burn every 15-30 years (Rheinhardt and others 2001, Richardson 1981). The hydrologic regime of pocosins is driven by precipitation, but water flows outward from the center and eventually forms headwater streams near the wetland's outer boundaries (Brinson 1993). The organic soils of pocosins tend to hold water longer than mineral soil flats. As a result, frequency of fire is less than in mineral soil flats.

Mineral soil flats are most common on areas between rivers, extensive lake bottoms, or large floodplain terraces where the main source of water is abundant precipitation and slow drainage associated with a landscape of low relief (Brinson 1993, Rheinhardt and others 2001). This class predominantly occurs on the Atlantic Coastal Plain from Virginia to Texas (Fig 1). There are two subclasses of mineral soil flats: those dominated by a closed canopy of hardwoods; and those characterized by open savanna with widely scattered pines (Rheinhardt and others 2001). Mineral soil hardwood flats in the Yazoo Basin of Mississippi occur on former and current floodplains created by the Mississippi River and its tributaries (Smith and Klimas in press). Mineral soil flats receive virtually no groundwater discharge. This characteristic distinguishes them from depressions. The dominant direction of water movement is downward through infiltration. These wetlands lose water by evapotranspiration, surface runoff, and seepage to underlying groundwater. They are distinguished from flat upland areas by their poor drainage due to impermeable layers (hardpans), and slow lateral drainage. Mineral soil pine flats will be the focus of the following discussion due to the millions of acres that still exist and their susceptibility to alteration due to fire exclusion, development, and silvicultural conversion to pine plantation.

The pre-European landscape was largely maintained by fires resulting from lightning strikes and Native American burning. However, with the colonization and subsequent management by Europeans less than 2 percent of the fire-maintained character of mineral soil pine flats remained by the 1990s. In their least altered condition, wet pine flats have very few trees. When trees are present, longleaf, pond and occasionally slash and loblolly pines are naturally associated with this wetland type. All four pines can tolerate ground fires by the time they reach 6-9 feet in height, but longleaf is the only pine whose seedlings are adapted to tolerate fire. The combined stresses of fire and wetness led to the evolution of an unusually rich flora on many wet pine flats (Rheinhardt and others 2001).

Wet pine flats differ from other wetlands due to a combination of factors that do not occur together in any other wetland type. These factors combine to control the biogeochemical processes characteristic of wet pine flats:

(1) The source of water, dominated by precipitation and vertical fluctuations in water

level driven by evapotranspiration,are generally low in nutrients.

(2) When flooding occurs, it is shallow (10-20 cm) and flows slowly.

(3) The number of pits and mounds on the ground surface is high, and provides a diverse array of aerated and anoxic conditions for soil microbial organisms.

(4) Nutrient recycling occurs in pulses following fires, which recur on a frequent basis, thus enabling a rapid turnover of nutrients. These four attributes enable wet pine flats to tightly and rapidly cycle nutrients. As a result, wet pine flats rapidly recover their characteristic biomass and structure after fires (Rheinhardt and others 2001).

Plant communities characteristic of unaltered wet pine flats are maintained by an appropriate hydrologic regime, fire regime, and biogeochemical processes that require intact soil conditions. Under relatively unaltered conditions, these three parameters combine to maintain a grassy savanna with few or no trees. On some sites, the herbaceous plant community is extremely rich. In fact, the herbaceous species richness is the highest recorded in the Western Hemisphere (Walker and Peet 1983). This herbaceous assemblage is extremely sensitive to alteration and, as a consequence, many species associated with this ecosystem are rare or threatened with extinction. Because the herbaceous community of wet pine flats is so sensitive to alteration (fire exclusion, hydrologic alteration, and soil disturbance), its condition provides information on habitat quality. Plant populations in wet pine flats have evolved to both withstand and require frequent fire. Fire stimulates flowering and seed set in many wet savanna species, such as toothache grass and wiregrass. As a result, species composition and spatial habitat structure reflect fire frequency. In the absence of fire, wet pine flat vegetative composition becomes dominated by shrubs or hardwood trees. This is a degraded condition when compared to a fire maintained wet pine flat.

Animals that use unaltered wet pine flats for all or part of their lives are adapted to habitats maintained by frequent fire. Frequent fire maintains open savanna, which is important to some animal species using wet pine flats. For animal species that utilize both unaltered wet pine flats and other similar fire-maintained landscapes, the total area of fire-maintained landscape (both wetland and upland) is critical. Because fire frequency has been drastically reduced in most areas of the Southeast, many animal species that require habitat maintained by frequent fire are threatened or endangered over most of their historic range. Maintenance of a characteristic animal assemblage depends upon: (a) habitat quality within the site (on-site quality) and (b) the quality of the surrounding landscape that provides supplemental resources (landscape quality). On-site habitat quality can be inferred from the structure and composition of the plant community.

A number of species rely on fire-maintained pine ecosystems of which wet flats are a part. For example, birds and other wide-ranging animals that rely on fire-maintained systems do not appear to differentiate wet pine flats from uplands, as long as both are fire-maintained. Thus, fire-maintained uplands supplement resources available in fire-maintained wet flats and vice versa.

5.2 Alterations to Forested Wetlands due to Development, Agriculture, and Silviculture

Functions of forested wetlands and the concomitant goods and services they provide can be degraded or destroyed by human activities. Activities that affect forested wetlands fit into four broad categories: (1) urban development, (2) rural development, (3) agriculture, and (4) silviculture. Since each wetland impact carries a unique set of circumstances and responses, these categories are rather gross. Their use, however, helps to describe wetland status, trends, and impacts in the South.

NWI defines urban development as intensive use in which much of the land is covered by structures including, buildings, roads, commercial developments, power and communication facilities, city parks, ball fields, and golf courses. In rural development, land use is less intensive and the density of structures is more sparse. Agriculture is defined as land use primarily for the production of food and fiber including horticultural, row and close-grown crops as well as animal forage. Silviculture is defined here as management of land for production of wood (Dahl 2000).

The replacement of forested wetlands with urban and/or rural development constitutes an irreversible loss, since the wetland is replaced by upland. Developed areas lack wetland hydrology, soils, and vegetation, either singly or in any combination. Changing a forested wetland to an agricultural field typically changes its hydrology and vegetation and disturbs its soil. However, some of these agricultural activities , such as drainage and removal of native vegetation, can be reversed and wetlands restored. Silvicultural activities typically do not lead to a loss of wetland status but may temporarily affect wetland functions. In forested riverine wetlands, for example, the overstory vegetation is removed but hydrology is left largely intact. Like some agricutural effects, silvicultural effects can be reversed and the wetland functions restored. More specific aspects of these activities will be discussed below.

5.2.1 Urban and rural development

The effects of urban and rural development on riverine, flat, and depressional wetlands in the South are similar. Forest vegetation is cleared, areas are drained or filled to escape flooding, structures are built, and wetland vegetation is replaced. These activities eliminate the ability of forested wetlands to store and convey surface water and groundwater. Water runs off these developed surfaces faster, reaching streams quicker and contributing to larger floods downstream. Development also eliminates the water-quality enhancement of forested wetlands. Development alters the hydrology and replaces the soils and vegetation with man-made structures which are not able to take up excess nutrients and other pollutants. The structures may actually contribute pollutants to adjacent aquatic ecosystems. Basnyat and others (1999) reported that urban land is the strongest contributor of nitrate to adjacent streams in Alabama. Alteration of hydrology and replacement of vegetation and soils with man-made structures also eliminate the forested wetland plant community and the wildlife associated with these areas. In other words, urban and rural development typically replace the wetland with upland and developed land with none of the functions of wetlands and little chance of restoration.

5.2.2 Agriculture

Generally, agricultural activities in forested wetlands manipulate hydrology, remove native vegetation, and disturb the soils for the purpose of crop production. Drainage, channelization, and levee construction impact the flow of water to and from a wetland site in an effort to dry-out the area. When wetlands are drained for agricultural use, they no longer function as wetlands (Mitsch and Gosselink 2000).

In riverine wetlands, hydrology is the principal force for maintaining ecological processes and vegetation structure (Gosselink and others 1990). Drainage and channelization, allowed water to reach the wetland but removed it from the site and/or watershed more quickly. Levees prevent flood waters from reaching the wetland at natural intervals (once to several times per year). Thus, drainage, channelization and levee construction result in changes in the timing of delivery of water (frequency), the amount of water delivered (magnitude), and the length of time the water remains in the wetland (duration). Duration of inundation is important in nutrient cycling, removal of pollutants and sediments, and export of organic carbon. Changes in hydroperiod also change the plant community, which alters the living and dead plant biomass components of nutrient cycling and organic carbon export. Construction of drainage ditches and channelization can affect the flow of subsurface water in a riverine wetland by changing the gradient of subsurface flow. Typically the result is a lower water table in the vicinity of the ditch or deepened channel. A shallower water table affects the ability of the riverine wetland to gradually contribute to stream flows during dry periods. Lowering the water table also affects biogeochemical processes and plant and animal communities that depend on the maintenance of a stable groundwater table (Ainslie and others 1999).

By impairing the ability of overbank flows to reach riverine wetland sites, levees prevent elements and compounds and sediments from reaching the wetland where they are deposited or removed. Levees prevent flood flows from transporting organic carbon to downstream aquatic ecosystems. They also act as barriers to aquatic species that use the floodplains for spawning and rearing (Lambou 1990, Baker and Kilgore 1994).

Clearing the native vegetation of a forested riverine wetland and replacing it with a crop dramatically reduces the site's structural diversity, wildlife food producing capacity, and nesting and escape cover (Gosselink and others 1990). Clearing also affects forest patch dynamics by decreasing forest patch size, interrupting forest continuity, decreasing the percentage of regional forested wetland, and increasing edge between community types. Soil tilling is likely to decrease the amount of organic matter in the soil due to oxidation. It also reduces water infiltration by creating a plow pan (Drees and others 1994). Therefore, clearing of native vegetation and forest structure and repeated plowing and tilling have the aggregate effect of causing more water to run off farm fields contributing greater flows and nonpoint-source pollutants (Basnyat and others 1999).

Many Carolina bays (Richardson and Gibbons 1993) have been significantly altered by agricultural practices, and some are being used for wastewater treatment. Managing forested depressions for agriculture involves clearing existing vegetation, installing drainage ditches through the rim of the Carolina Bay, tilling the soil, and planting the site in the desired crop species. Draining the depression alters the duration of ponding and the amount of water in the wetland. Plants, animals and the biogeochemistry of the wetland are affected. Disrupting the surface of the soil by tilling affects the amount of organic material in the soil. As water is drained from the depression, soil organic material is exposed to the air, speeding its removal through oxidation. As soils are disturbed, more organic carbon is exposed from deeper in the soil and more is oxidized as a result, the balances among water, carbon, and other elements like nitrogen and phosphorous are disrupted. Accumulation of too much sediment in depressional wetlands, from erosion in nearby uplands, decreases wetland water storage volume, decreases the duration of water retention in wetlands, and changes plant community structure by burial of seed banks. As with riverine wetlands clearing the existing vegetation in Carolina bays alters the composition and structure of the native plant community and affects wildlife species that utilize the depression.

Sharitz and Gresham (1998) report that 97 percent of the Carolina bays in South Carolina have been disturbed by agriculture (71 percent), logging (34 percent), or both. Agriculture is the oldest and predominant use of bays having started in th 1940s. Soils in Carolina bays are highly organic and have a high nutrient holding capacity. They are attractive to farmers if drainage is accomplished; soil pH is raised by liming; minor nutrients tied up by the highly organic soils are supplied to the crops with spray; and weeds are controlled, primarily with herbicides. If these activities are completed, Carolina bays are 10-15 percent more productive than upland soils, but these activities alter the structure and function of the Carolina bay.

Organic soil flats were cleared and drained for agriculture as early as the 1780s. Several large pocosins have been impacted by corporate agricultural operations, which have drained, limed, and fertilized these wetlands for corn and soybean production. Off-site effects of draining pocosins for agriculture included decreased salinity in adjacent estuaries, increased turbidity in adjacent streams immediately after development, and increased phosphate, nitrate, and ammonia inputs into adjacent streams and estuaries, particularly when runoff volumes are high (Sharitz and Gresham 1998). These problems can be minimized by managing the water levels in the drainage ditches with flashboard risers, which maintain water tables and slow the delivery of water to adjacent streams and estuaries. In 1989 14 percent of pocosins in North Carolina were owned by corporate agriculture and 36 percent by major timber companies (Richardson and Gibbons 1993). Originally pocosins covered 2,244,000 acres in North Carolina but by 1980 this had been reduced by 739,000 acres due to agriculture, silviculture and development (Richardson and Gibbons 1993). Clearing pocosins for agriculture is no longer practiced due to restrictions placed on landowners by the Food Security Act and Section 404 of the Clean Water Act.

5.2.3 Silviculture

Silvicultural activities in forested riverine wetlands typically consist of clearcutting overstory vegetation and allowing natural regeneration from sprouts (Walbridge and Lockaby 1994, Kellison and Young 1997, Lockaby and others 1997a). The stand then progresses from a thicket dominated by briars, vines and tree seedlings and sprouts, to a sapling stage after 10-20 years, to a pole timber stage after 20-30 years, to a small sawlog stage at 30-50 years, and finally to a mature forest stage beyond age 50 (Kellison and Young 1997). Hydrologic responses to this silvicultural regime typically are short-term elevations in the water table due to a reduction in evapotranspiration (Lockaby and others 1997a, Sun and others 2001). Removing the trees reduces the amount of the soil water transpired by plants and the water then fills more soil pores resulting in a water-table rise. However, this reduction in evapotranspiration is typically negated by the sprouting vegetation on the clearcut site within 2 years (Lockaby and others 1997b). Another hydrologic effect of harvesting riverine wetlands is soil compaction which interferes with the movement of water through the soil. Lockaby and others (1997) determined the hydraulic conductivity of the saturated soil was reduced 50-90 percent in the ruts caused by skidding of logs. This effect can be temporary, depending on the soil type and hydrology of the wetland (Rapp and others 2001, Perison and others 1997).

There is concern that harvesting and site-preparation in wetlands cause or contribute to the generation of nonpoint-source pollutants, particularly sediment. Ensign and Mallin (2001) found that when compared to an upstream reference site, a stream in the Coastal Plain of North Carolina experienced higher levels of nutrients (nitrogen and phosphorous), higher fecal coliform levels, and recurrent algal blooms for up to 15 months after clearcut harvesting of adjacent forested wetlands. The authors speculated that these effects were due to the inability of the clearcut wetland site to retain and transform upstream agricultural pollutants. However, other studies indicate the magnitude of these effects is small and the longevity is brief (Shepard 1994, Walbridge and Lockaby 1994, Lockaby and others 1997a, Messina and others 1997). Studies indicate that after revegetation sediment deposition in wetlands is actually greater on harvested sites because the amount of vegetation is greater, thus slowing floodwaters to a greater degree and allowing more sediment to drop from the water column (Aust and others 1997, Perison and others 1997).

The capacity of forested riverine wetlands to act as sinks, sources or transformers of nutrients and carbon depends upon landscape position, the amounts of nutrients entering the wetland, and the time since disturbance. The degree to which silviculture affects a riverine wetland's capacity to transform nutrients and sequester other pollutants is uncertain (Lockaby and others 1997a). Conceptually, riverine wetlands serve as sinks when they receive high inputs of nutrients. They may serve as sources when disturbed to the point where active oxidation of soil organic matter or export of mineral sediment is occurring and they may serve as transformers in relatively undisturbed situations. However, Lockaby and others (1999) point out that few generalizations can be made about biogeochemical cycling and nutrient retention functions because of the variable nature of responses of riverine wetlands to harvests, and the inability of current scientific methods to detect subtle biogeochemical changes due to silvicultural activities. Thus, they conclude that the ability to predict whether long-term shifts in biogeochemical transformations occur due to silviculture is minimal and that there is a critical need to understand how silviculture affects the enhancement of water quality in riverine wetlands.

Perhaps the most apparent effect of silvicultural operations on forested riverine wetlands is the removal of the tree canopy. The ability of the forested wetland to recover from harvesting is of interest to both forest industry and conservation interests. Generalizations about the productivity of forested riverine wetlands and their ability to recover from harvests are difficult due to the diversity of forested wetlands. Different moisture regimes, hydrologic conditions, and soil types have resulted in the diversity of wetland types (Conner 1994). Comparisons between harvested sites and reference sites require long-term study. A study conducted 1 year after harvesting in a Texas riverine wetland showed little difference in the composition of tree species regenerating on the harvested site and the presence of those species on an unharvested site (Messina and others 1997). Another study conducted 7 years after harvest in a tupelo-cypress riverine wetland indicated that harvested stands were stocked with tree species similar to the reference. The stand harvested by helicopter had an even distribution of overstory species, while the stand harvested with ground-based methods was dominated by tupelo gum (Aust and others 1997). In a study conducted 8 years after harvesting a riverine wetland in South Carolina, no difference between the species composition of the overstory of harvested and unharvested stands was detected. However, midstory and understory vegetation differed between the two treatments (Rapp and others 2001). These authors concluded that the effects of harvesting are short-lived and that these stands will return to pretreatment species composition. Additional long-term research is needed to continue to track the development of the plant community and ecological functions in harvested stands compared with unharvested stands.

Wildlife species have a variety of ecological roles that contribute to the maintenance of the forested riverine wetland. Wildlife contribute to the dispersal of plants by caching and transporting seeds, they alter forest structure and composition by eating vegetation and creating impoundments. They alter soil and forest productivity by burrowing and preying on macroinvertebrates. They support food webs, transport energy to surrounding ecosystems, and recolonize of adjacent habitats (Wigley and Lancia 1998). Biotic and abiotic factors determine the inherent capacity of a forested wetland to support a community of wildlife species. Soils, topography, hydrology, disturbance, climate, stand vegetation, landscape pattern of habitats and land uses, wildlife community interactions, and human-related alteration of forest structure and composition affect the abundance of wildlife (Wigley and Lancia 1998). The contribution of wildlife to ecological processes and the factors influencing wildlife presence are complex. As a result, evaluating the effects of clearcutting with natural regeneration on riverine wetlands is difficult.

At the stand scale, the vertical and horizontal dimensions of forest structure are important because the taller and more layers present, from the forest floor to the canopy, the more opportunities for foraging, nesting, and escaping from predators (Wigley and Lancia 1998). As plant succession proceeds in forested wetlands structural diversity tends to increase, but the frequency and duration of flooding may reduce the mid- and understory vegetation. Thus, some animals needing lower layers of the forest, such as the wood thrush, hooded warbler, Swainson's warbler, may not be present in natural forest stands (Howard and Allen 1989). However, flooding may contribute to vertical diversity by creating snags, which are important to some species like the prothonatary warbler, wood ducks, woodpeckers, and bats (Wigley and Lancia 1998). Horizontal diversity refers to the distribution of vegetation or other structural features in patches throughout the stand. This horizontal diversity can provide habitat for early successional species in a mature stand or mature stand species in an early successional stand. Diversity of mast producing species can also ensure a consistent food supply. When production of one tree species is low, that of another species may be high.

Edges occur between wetland forest types, wetland and upland forest types, or between land uses. The effects of these edges vary. Edges can increase species diversity by providing habitat for the species in the abutting habitats plus those species that prefer edges. On the other hand edges can increase predation and brood parasitism by brown headed cowbirds and add exotic species (Wigley and Lancia 1998). Riverine wetlands can serve as regional migration corridors for black bear, neotropical songbirds, and waterfowl (Gosselink and others 1990). However, these corridors can aid in the conveyance of species from one habitat to another or, as with edges, can convey predators, diseases, and parasites. Forested wetlands also fit into a landscape mosaic of habitat types that may be important to species needing several habitats to fulfill life requirements. Species presence and productivity are sometimes viewed as functions of the size and shape of a wetland habitat patch, amount of edge, distance from patches of similar habitat (isolation), amount of time since isolation, and immigration and dispersal of animals from habitats (Wigley and Roberts 1997). However, much of the landscape-scale information on the effect of these wildlife habitat functions on the presence and productivity of wildlife populations is based on theory. Little data exist for managed forest landcapes to validate these theories (Wigley and Roberts,1994 and 1997, Wigley and Lancia 1998).

Riverine forested wetlands have an abundance of detritus, hard and soft mast, snags, cavity trees and large woody debris on the ground as well as multilayered vegetation, these typically support conditions rich and diverse wildlife communities (Wigley and Lancia 1998, Ainslie and others 1999, Gosselink and others 1990). Forest management activities potentially influence wildlife habitat at site-specific and landscape scales. Clearcuts with natural regeneration temporarily reduce availability of hard mast and canopy and cavity trees (Wigley and Roberts 1994, Wigley and Roberts 1997). However, regeneration of woody vegetation and ground vegetation growth typically increase after harvest, downed woody debris often increases due to harvesting (assuming it is not windrowed and burned), and early successional wildlife species may increase. Clawson and others 1997 found that amphibian population diversity and abundance were only temporarily affected by harvesting. Thus, many habitat alterations due to forest management are temporary.

From a landscape perspective there is a growing recognition that the lack of early successional forest, including but not exclusive to forested wetland, is limiting biodiversity in the eastern United States (Trani and others 2001, Thompson and Degraaf 2001, Hunter and others 2001, Litvaitis 2001, Wigley and Roberts 1997). Thompson and Degraaf (2001) suggest that silvicultural operations can contribute to landscape diversity by creating early successional habitats in forested landscapes. Several studies have suggested that in largely forested landscapes early successional patches increase wildlife diversity (Thompson and others 1992, Welsh and Healy 1993). However, as previously pointed out, little is known of the effects of forest management in landscapes permanently fragmented by conversion to agriculture or urban development.

5.2.3.1 Depressions

Sharitz and Gresham (1998) note that managing Carolina bays for timber requires clearing the existing vegetation, installing drainage ditches within the bay and through the rim, bedding the bay soil, and planting trees. Any of these activities greatly alters the structure and function of the bay ecosystem.

Pondcypress swamps are harvested for sawtimber and increasingly for landscape mulch. Typically, they are harvested by clearcutting. Clearcuts regenerate well (Ewel and others 1989), but leaving some mature trees to produce seed is advocated due to uncertainty of resprouting and seed production (Ewel 1998). After harvesting, water levels in pondcypress swamps typically rise and amphibian and wading bird usage of the post-harvest swamp increases. Mammal useage also changes, with fewer nest and den sites but more prey available (Ewel 1998).

5.2.3.2 Mineral Soil Pine Flats

On mineral soil flats, three parameters stand out as being essential for determining the degree to which ecosystem processes are altered by a given impact: (1) the alterations in the hydrologic regime, (2) alterations in fire regime, and (3) alterations in the soil. These changes in ecosystem processes on mineral soil flats, alter plant and animal habitats. Hydrologic fluctuations determine the composition of fire-tolerant vegetation, and soil conditions control the dynamics of biogeochemical transformations by soil microbes. Fires maintain open, sometimes treeless, savannas by precluding species that would otherwise shade out characteristic savanna plants and provide nutrients in discrete pulses utilized by savanna plants (Rheinhardt and others 2001).

Silvicultural impacts on flat wetlands typically include surface and subsurface drainage, ditching, harvest and mechanical reduction of native vegetation, bedding, which alters microtopographic relief, and the construction of roads (Harms and others 1998). The objective of intensive management on these mineral soil flat wetlands is to produce pine plantations. Most biogeochemical processes in wetlands depend on the distribution and timing of flooded and dry conditions. Draining a mineral soil flat eliminates flooding and soil saturation, which in turn alters processes that depend on flooded conditions, including fermentation, and denitrification.

With the exception of artificial drainage, most alterations to hydrologic regime are localized in their effect on biogeochemical processes and habitat quality. For example, a dam (even a low one such as a road fill) can impede surface flow and back water up over a large area. One result is a longer period of inundation. Input of excess water from off site can likewise increase the duration and depth of water levels. Alterations to water balance change the duration and timing of flooding and the saturation of soil in the upper horizons. In contrast, artificial drainage reduces inundation periods. Artificial drains transport water, nutrients, and dissolved organic matter into streams downstream, altering the water flow and chimstry for a period of 2-3 years. (Beasley and Granillo 1988, Amatya and others 1997, Lebo and Herrmann 1998). However these studies also indicate that the hydrologic effects of ditches can be ameliorated with water control structures such as flash board risers (Sun and others 2001).

Soil condition on mineral soil flats also can be affected by intensive silvicultural activities (Miwa and others 1997, Miwa and others 1999). Microbial organisms and plants are adapted to characteristic microtopographic structure, soil texture, and nutrient regime. Alterations to soils affect these conditions upon which soil microbes and plants depend. The result may be a change in biogeochemical cycling processes. For example, harvesting can affect water holding capacity and available water for plant growth and slow internal soil drainage, causing higher water tables and slower site drainage (Miwa and others 1997). Bedding is currently the best available technique to ameliorate these effects. However, bedding also may affect soil bulk density both on the beds and in the trenches between, thus altering interstitial pore space and substrate conditions on which soil microbes and plants depend. In addition, microtopographic variation is changed by a regular distribution of small, low (10-20 cm high), regularly distributed hummocks to a parallel array of trenches and high ridges (15-30+ cm high). On bedded sites, duration and frequency of flooding are increased in trenches and decreased on beds relative to unaltered conditions. Which results in altered rates, timing, and magnitudes of biogeochemical processes (Rheinhardt and others 2001).

Mechanical treatment of native vegetation and bedding a mineral soil flat to produce pine plantations affects fire-maintained wildlife habitat of wet pine flats. For example, several amphibian species are associated with fire-maintained landscapes and travel across wet flats to breeding ponds in cypress depressions. There is evidence that intensive silviculture may detrimentally affect amphibian and reptile populations (Rheinhardt and others 2001) because intensive silviculture relies on a series of raised parallel-aligned beds on which pine seedlings are planted. Standing water in the troughs between beds may cue amphibians to lay their eggs in these troughs, where water sits for too short a time to support larval development, rather than in deeper, more permanent cypress depressions which are commonly scattered throughout wet pine flats.

5.3 Policy

Development, agriculture, and silviculture are regulated primarily by two Federal laws: the Food Security Act, Public Law 104-127) (FSA) and the Clean Water Act (CWA). The objective of the "Swampbuster" provision of the FSA is to discourage alteration of wetland hydrology, vegetation and soils to facilitate production of commodity crops (Strand 1997). FSA penalizes landowners who alter wetlands for this purpose by removing their eligibility for Federal subsidies. However, agricultural landowners may retain their eligibility for benefits by restoring, enhancing, or creating wetlands to compensate for lost wetland functions and values.

Development, agriculture and silviculture are also regulated under Section 404 of the CWA. Section 404 requires that anyone proposing to place fill material into waters of the United States, including wetlands, must obtain a permit from the U.S Army Corps of Engineers. In order to obtain a permit the "applicant" must show: (1) why the project cannot be located somewhere besides a wetland, (2) why the project will not adversely harm the wetland, and (3) what the applicant will do (if granted the permit) to offset the loss of wetland functions and values. Replacement of lost wetland functions and values is typically accomplished through "mitigation" -the restoration, enhancement, or creation of wetlands in another location. For a more in depth discussion of these laws see SOCIO-3.

Under Section 404 (f) of the CWA, normal silvicultural and agricultural activities, such as plowing, seeding, cultivating, minor drainage, and harvesting for the production of food, fiber and forest products, are exempt from the permitting requirements. However, these activities must be part of an ongoing agricultural or silvicultural operation and may not change a wetland to an upland. In addition, construction of forest roads is exempt under Section 404(f) as long as 15 Federally prescribed best management practices (BMPs) are implemented. The issues surrounding forest road construction, and the BMPs used to ameliorate water quality impacts of roads are discussed further in Chapter AQUA-4.

5.4 Restoration

Approximately half of the South's forested wetlands have been lost in the last 200 years. Along with this loss in acreage has been the loss of wetland functions and societal benefits, goods and services described in the last section. In an attempt to ameliorate the environmental damage of wetland loss, restoration of former forested wetlands is being attempted throughout the South. Wetland restoration is defined by the Society of Wetland Scientists as, "actions taken in a converted or degraded natural wetland that result in the establishment of ecological processes, functions, and biotic/abiotic linkages and lead to a persistent, resilient system integrated within its landscape." The goal of restoration of wetland ecosystems was expressed by the National Research Council (1992) as, "returning the system to a close approximation of the predisturbance ecosystem that is persistent and self-sustaining (although dynamic in its composition and functioning)." Therefore, since much of the forested wetland loss in the past has been due to agriculture, any national or regional program designed to restore millions of acres of former wetlands will have to focus primarily on wetlands converted to agricultural use (National Research Council 1992). Presumably these agricultural lands would still occupy the same landscape position and have the same or similar hydrology as the original wetlands prior to conversion. An exception to this is in areas where extensive levee systems like those in the Lower Mississippi Valley have restricted flooding on a broad scale.

Although forested wetlands have been lost throughout the South, perhaps the most acute losses have been in the Lower Mississippi Alluvial Valley (LMAV). There, approximately 18 million acres of wetland were lost to agricultural conversions ((King and Keeland 1999)). Such conversions have involved clearing the natural forested wetland vegetation, drainage, and flood control. In the LMAV, the estimated original 25 million acres were reduced to approximately 5 million acres by 1978 (Hefner and Brown 1985). Ninety-six percent of the forested wetland losses in the LMAV were due to agriculture; the remaining losses were due to construction of flood control structures, surface mining, and urbanization (Schoenholtz and others in prep).

In the 1970s and 1980s the U.S. Fish and Wildlife Service recognized the trend in forested wetland loss and associated habitat impacts in the LMAV and began a campaign to reestablish forested wetlands in the LMAV (King and Keeland 1999). The development of the Wetland Reserve Program (WRP) by NRCS as well as smaller projects undertaken by the U.S. Army Corps of Engineers and State Fish and Game agencies has intensified reforestation/restoration in the LMAV making this area the largest reforestation/restoration effort in the South. Figure 3, derived from NRI data from 1982-1992 indicates that 17.5 percent of the watersheds in the South experienced a gain of forested wetland, 31.2 percent experienced a loss, and 51.3 percent experienced no change. However, it is uncertain if the acres reported in the NRI represent actual acres restored versus acres enrolled in WRP.

The Wetland Reserve Program of the 1990 Farm Bill is directed at wetland systems and provides for conservation easements for 10-30 years. The 1990 Farm Bill, which was reauthorized in 1996, established that up to 1 million of the 6 million acres of cropland eligible for the Conservation Reserve Program (CRP) may be wetlands. This program, unlike most others, has the potential to restore large acreages of forested wetlands in the South.

King and Keeland (1999) reported that approximately 195,000 acres have been reforested in the LMAV. Restoration of forested wetland systems in the LMAV involves restoration of the geomorphic, hydrological, and ecological processes that drive these wetland systems. Massive forest clearing, construction of thousands of miles of drainage ditches, broad-scale channelization of streams and rivers, flood prevention, and farming practices have changed hydrology, topography and soils. Restoration of wetland functions is extremely difficult there. Table 5 shows that 64 percent of the WRP acres are in the States of Mississippi, Louisiana, and Arkansas. Presumably, all or a major portion are in the LMAV. Figure 4 shows the number of WRP acres by State in the South. Once again Mississippi, Louisiana, and Arkansas have the greatest number of farmers enrolled. In addition to WRP acres, the U.S. Fish and Wildlife

Service has planted approximately 59,000 acres and State Wildlife Management Areas have planted 28,000 acres (Schoenholtz and others in prep). Information could not be found to document restoration efforts in other parts of the South. Programmatic success of restoration is determined by the number of trees surviving (>125/acres) on a WRP site after 3 years. Ecological success is difficult to determine and, due to the protracted nature of forested wetland restoration, will continue to be difficult to determine in the future.

Currently, restoration has attempted to re-establish forested wetland hydrology and vegetation on sites where these two characteristics have been removed. Thus, much of the restoration effort has been directed toward agricultural land. However, some wetland ecosystems, namely mineral soil pine flats, have been ecologically degraded by exclusion of natural disturbances like fire. Restoration of wetland ecologic processes, functions and biotic/abiotic linkages could be achieved if the disturbance regime were re-established. Lorimer (2001) points out the important role fire has historically played in maintaining plant species composition and structure in the South and its effects on wildlife abundance and distribution. Thompson and DeGraaf (2001) suggest that historic disturbance regimes can provide effective models for silviculture by substituting harvesting for fire. In largely forested regions like the Northeastern and Mid-Atlantic United States, harvesting can promote early successional growth and increase biodiversity (Thompson and others 1992, Welsh and Healy 1993, Hagan and others 1997). However, restoration of mineral soil pine flat wetlands can best be achieved by re-establishing frequent fire into these ecosystems.

Section 404 of the Clean Water Act regulations establishes procedures for permitting the discharge of solid fill material into wetlands. This program is administered primarily by the U.S. Army Corps of Engineers with oversight from the Environmental Protection Agency (EPA). If impacts due to these permitted activities are considered to be unavoidable, restoration of former wetlands is typically required to offset losses. Restoration of forested wetlands is a typical requirement of the Section 404 permitting program. Although many small-scale wetland restoration projects have been required in the history of the Section 404 program, the Corps and EPA maintain no systematic accounting of these projects or their success.

Little consistent data are available to track the amount of forested wetland mitigation that has been required or the amount that has actually been completed. It is even more difficult to ascribe success to many of the mitigation efforts that have been undertaken. Two studies in the South found that many of the mitigation projects proposed and carried out under the Section 404 program did not replace the wetlands originally impacted (Pfeifer and Kaiser 1995; Morgan and Roberts 1999). The National Research Council (1992) listed the following as reasons for unsuccessful mitigation in a regulatory context:

(1) Poor design of mitigation projects by individuals lacking sufficient expertise to address the complexities of wetland ecosystems.

(2) Landowners often prepare the least expensive and least time consuming plan acceptable to the regulatory agencies leading to "half-hearted" attempts to restore

wetlands.

(3) Wetlands restored in the regulatory context are often small in size, widely separated from other wetlands, and threatened by adjacent landuses.

(4) After initial restoration wetland mitigation sites receive very little management.

For these reasons wetlands restored in the regulatory context may be less likely to achieve restoration goals. A recent report on compensating for wetland losses under the CWA concluded that the goal of no net loss of wetlands is not being met for wetland functions by the Section 404 mitigation program, despite progress over the last 20 years (National Research Council 2001).

6 Conclusions

Forested wetlands provide a variety of hydrologic, biogeochemical and habitat functions unique to these ecosystems. Landscape position, water, soils and plants all contribute to the structure and function of forested wetlands in the South. All these contributions can be degraded by human impacts. Status and trends indicate that the rates of wetland losses in general are down to 356,000 acres (2.3 percent) for the period of 1986-1997. According to NWI approximately, 119,000 acres of forested wetland have been lost to urban/rural development, 112,000 acres to agriculture and 102,000 acres to silviculture. Approximately 3 million acres of forested wetland were converted by silvicultural operations to different (forest) wetland types. Timber harvests in the South are expected to increase over the next 20 years. Since almost one quarter of the timberland in the South is forested wetland, it is likely that impacts to forested wetlands as a result of intensified silviculture will continue and perhaps additional acreage will be affected in the future. Silvicultural operations affect the hydrologic and structural characteristics of wetlands. However, when hydrology is not permanently altered and sites are allowed to regenerate naturally, indications are that, in time, they function similarly to unaltered wetlands. Sites converted to intensive pine plantation culture experience longer term changes to their structural and biotic diversity.

There is a great deal of potential for restoration of forested wetlands on former agricultural land in the South. The Wetland Reserve Program and the Section 404 program provide opportunities to restore these former wetlands. However, forested wetland restoration is a complex undertaking, and must be done carefully to recreate the lost functions and values of forested wetlands in the south.

7 Needs for Additional Research

(1) Landscape level studies are needed to determine the causal mechanisms for wildlife and water quality response to landscape configurations and features such as corridors. We need to know how forest treatments affect wildlife and plant communities and stream water quality in the various types of wetlands in landscapes predominated by: riverine forests, a mix of riverine

and upland forests, a variety of wetland types (e.g., Coastal Plain where riverine, depression and flat classes occur together in close proximity), and a variety of landuses (agriculture, urban/rural, etc.). Information from this type of research should be integrated with research from site specific scales.

(2) Research is needed on the water quality enhancement functions of forested wetlands and the impacts of forest practices on those processes in different wetland classes.

(3) At present, three Federal agencies -the Fish and Wildlife Service, the Natural Resources Conservation Service, and Forest Service collect landscape-scale wetlands data. However, due to different data objectives and agency missions much of this data is incompatible for tracking status and trends of forested wetlands. A unified database of this information is needed.

(4) Cause and effect research is needed by hydrogeomorphic class, at the site specific and landscape scale on representative sites across Region.

(5) Long-term monitoring of restoration and mitigation is needed by hydrogeomorphic class at representative sites across the South.

8 Acknowledgments

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10 Tables and Figures

Table 1--Composite of National Wetland Inventory (NW) wetland status and trend information for the Conterminous and Southeastern United States

Time period	Geographic extent of estimate	Total wetland (Ac)	Forested wetland (Ac)	Source
1780	Conterminous U.S	221,000,000	No Estimate	Dahl (1990)
1980	Conterminous U.S	104,000,000	No Estimate	
% change		47%		
1950	Southeast U.S. (10 states)	54,257,000	38,000,000	Hefner and Brown (1985)
1970	Southeast U.S. (10 states)	46,500,000	32,000,000	
% change		15%	16%	
1970	Southeast U.S. (10 states)	51,200,000	35,300,000	Hefner and others (1994)
1980	Southeast U.S. (10 states)	48,900,000	33,004,000	
% change		5%	7%	
1986	Conterminous U.S	106,135,700	51,929,600	Dahl (2000)
1997	Conterminous U.S	105,500,000	50,728,500	
% change		1%	3%	
1986	Southeast U.S. (10 states)*	49,883,779	33,735,000	
1997	Southeast U.S. (10 states)*	49,585,000	32,643,000	
% change		1%	3%	

* estimated from percentages , specific to the South, from Hefner and others (1994) applied to National data from Dahl (2000). Wetland acreages derived from NWI reports and/or calculated from reported percentages.

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Table 2--Comparison of total wetland and forested wetland acres and the predominant cause of change

State	Total wetland (Ac)	Percent of State land surface in wetland	Forested wetland (Ac)	Forested wetland change (Ac)	Predominant cause of change
Alabama	2.7 million	8%	2.2 million	97,000	Agriculture
Arkansas	3.6 million	10%	2.8 million	210,000	Agriculture
Florida	11 million	30%	5.5 million	184,100	Other wetland types and Urbanization
Georgia	7.7 million	20%	6.1 million	500,000	Other wetland types
Kentucky	388,000	1%	274,000	9884	Agriculture and mining
Louisiana	8.8 million	28%	4.9 million	628,000	Agriculture
Mississippi	4.4 million	14%	3.7 million	365,000	Agriculture
North Carolina	5 million	15%	3.4 million	1.2 million	Other
South Carolina	4.7 million	24%	3.6 million	125,000	Agriculture, Urban, Forestry

Tennessee	632,000	2%	630,000	25,000	Agriculture
Texas	6.4 million		2.5 million	60,540	Agriculture, reservoirs
Virginia			683,000	20,000	

Data abstracted from Hefner and others (1994), Frayer and Hefner (1991) and Shepard and others (1998).

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Table 3--Estimated acres of palustrine forested wetland conversion and loss by state and by activity in the South

	Estimated acres of palustrine forested converted to:				Estimated acres of palustrine forested lost to:					Total estimated acres					
State	Palus-trine Emer-gent	Palus-trine Shrub	Palus-trine Other	Deep-water	Agricul-ture	Urban develop	Rural develop	Silvi-culture	Other	Conversion		Loss		Change	
AL	53,717	289,103	3,317		2,686	18	6,301		512	346,137	(27)	9,517	(43)	355,654	(26)
AR	25,411	100,813	13,262	5,259	1,404	149	3,619	28,958	3,213	144,745	(22)	37,343	(37)	182,088	(23)
FL	41,654	187,284	3,366	492	6,789	11,487	26,424	582	3,892	232,796	(21)	49,174	(23)	281,970	(18)
GA	108,778	677,994	18,593	2,554	24,049	12,422	7,330	30,745	6,075	807,919	(11)	80,621	(28)	888,540	(11)
KY				174	9,629				868	174	(*)	10,497	(60)	10,671	(59)
LA	21,834	83,700	25,447	13,357	9,015	1,311	4,921	40,319	4,742	144,388	(21)	60,308	(31)	204,646	(18)
MS	32,963	386,429	11,250	6,587	34,841	4,951	15,508		1,102	437,229	(29)	56,402	(45)	493,631	(26)
NC	56,393	472,116	6,235	86	2,888	1,245	2,677		246	534,830	(16)	7,056	(22)	541,886	(16)
OK	18,352	5,340	15,536	92	2,628	3,679				39,320	(42)	6,307	(69)	45,627	(38)
SC	60,368	294,246	5,298	33	1,184	9,293	3,445	630	2,185	359,945	(19)	16,737	(35)	376,682	(18)
TN		21	42		16,882	174	94			63	(*)	17,150	(64)	17,213	(64)

TX	30,357	70,262	755	118			3,040	1,005	59	101,492	(54)	4,104	(50)	105,596	(53)
VA	9,697	1,279	1,177			1,177				12,153	(64)	1,177	(*)	13,330	(59)
TOTAL	459,524	2,568,587	104,278	28,752	111,995	45,906	73,359	102,239	22,894	3,161,141		356,393		3,517,534	
	(10)	(8)			(25)	(19)	(21)	(29)	(18)	(7)		(13)		(7)	

Note: Values in parentheses are percent coefficient of variation with an asterix (*) indicating it's statistically unreliable.

Estimate based on U.S. Fish and Wildlife Service, National Wetland Inventory data from 1986-1997(Dahl 2000).

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Table 4--Comparison of forested wetland community types and extents with hydrogeomorphic class

Forested community type	Predominant HGM class	Extent in the South (acres)	Source
Southern deepwater swamp	Riverine	Included in major and minor alluvial floodplain estimates	Conner and Buford (1998)
Major alluvial floodplain	Riverine	11.8 million	Kellison and others. (1998)
Minor alluvial floodplain	Riverine	20 million	Hodges (1998)
Carolina bays	Depressions		Sharitz and Gresham (1998)
Southern mountain fens	Depression/slope	6200	Moorhead and Russell (1998)
Pondcypress swamps	Depression		Ewel (1998)
Pocosins	Organic flats	695,000	Sharitz and Gresham (1998)
Wet flatwoods (pine)	Mineral flats	2.5 million	Harms and others. (1998)

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Table 5--Wetland Reserve Program acres by State in the South

State	Total WRP acres	% of national total WRP acres	% of southern total WRP acres
Virginia	1063	0.12	0.219
North Carolina	18216	1.99	3.751
South Carolina	13507	1.48	2.781
Georgia	7374	0.81	1.518
Florida	45225	4.94	9.312
Kentucky	7613	0.83	1.568
Tennessee	13976	1.53	2.878
Alabama	1410	0.15	0.290
Mississippi	92107	10.06	18.965
Arkansas	87664	9.58	18.050
Louisiana	132319	14.46	27.245
Oklahoma	30304	3.31	6.240
Texas	34892	3.81	7.184
TOTAL	485670	53.07	100.000

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Figure 1--Physiographic regions of the southern United States (Source: Hammond 1970).

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Figure 2--Change in forested wetlands in South based on Natural Resources Inventory 1982-1987.

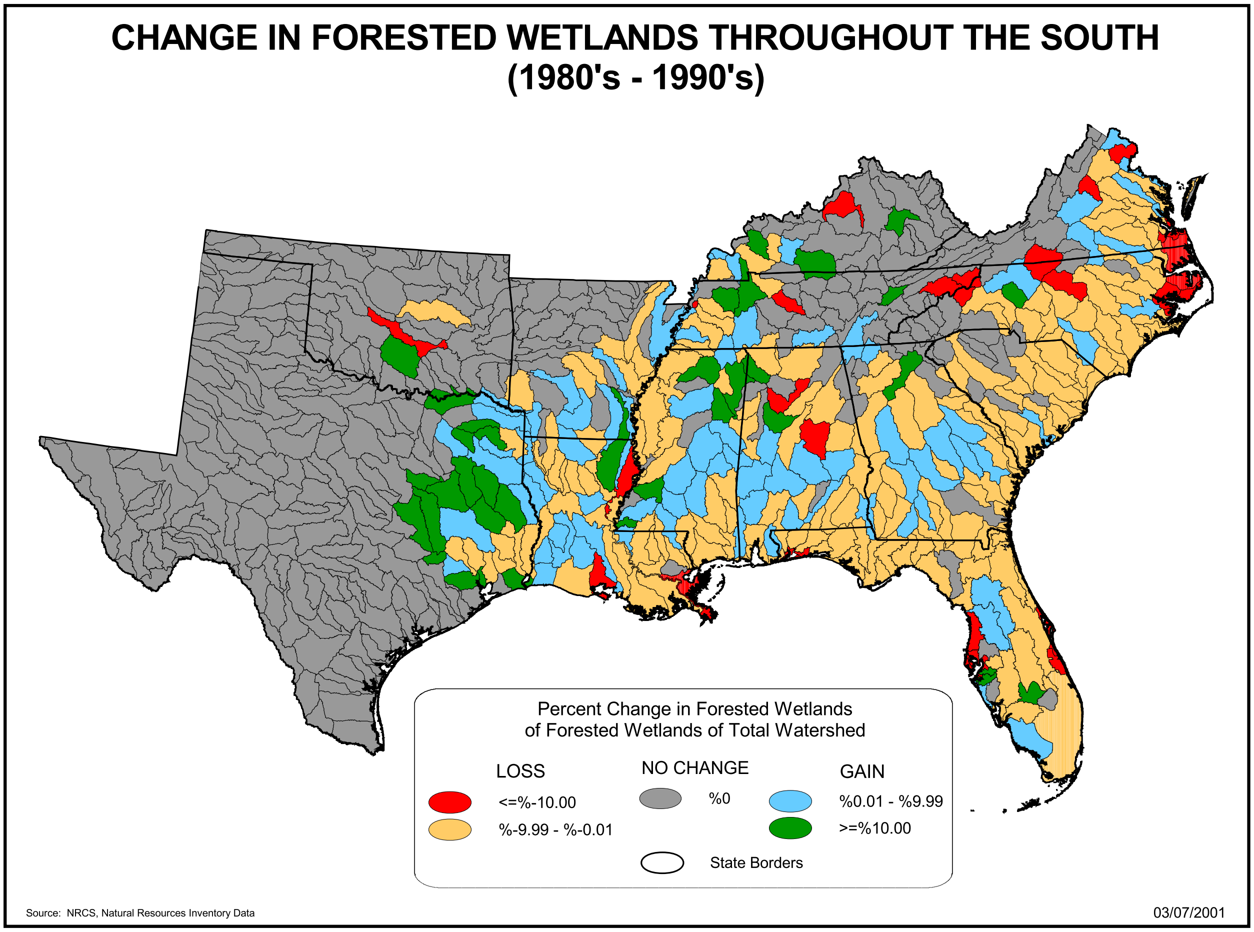

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Figure 3--Causes of palustrine and estuarine wetland losses based on 1992-1997 National Resources Inventory Data.

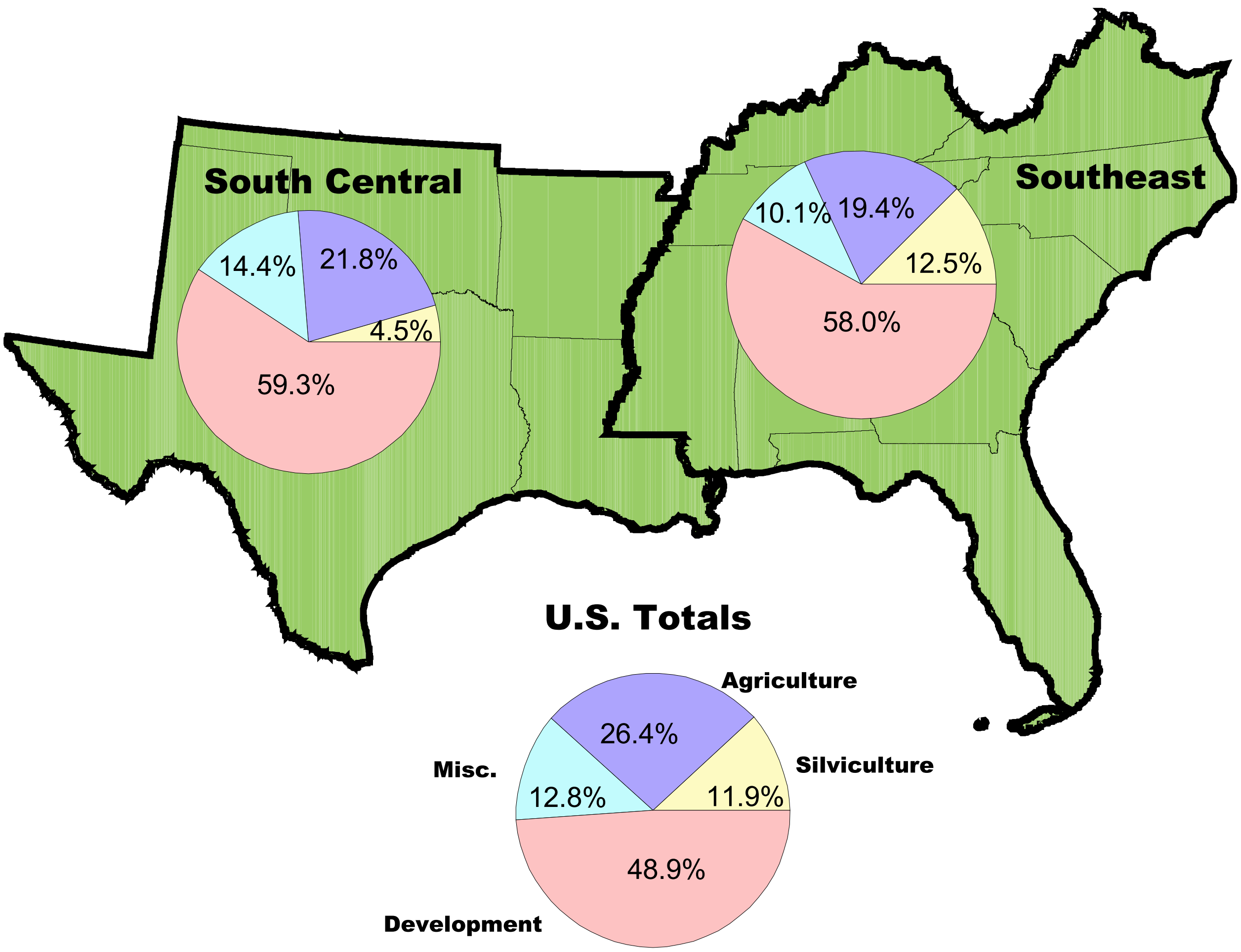

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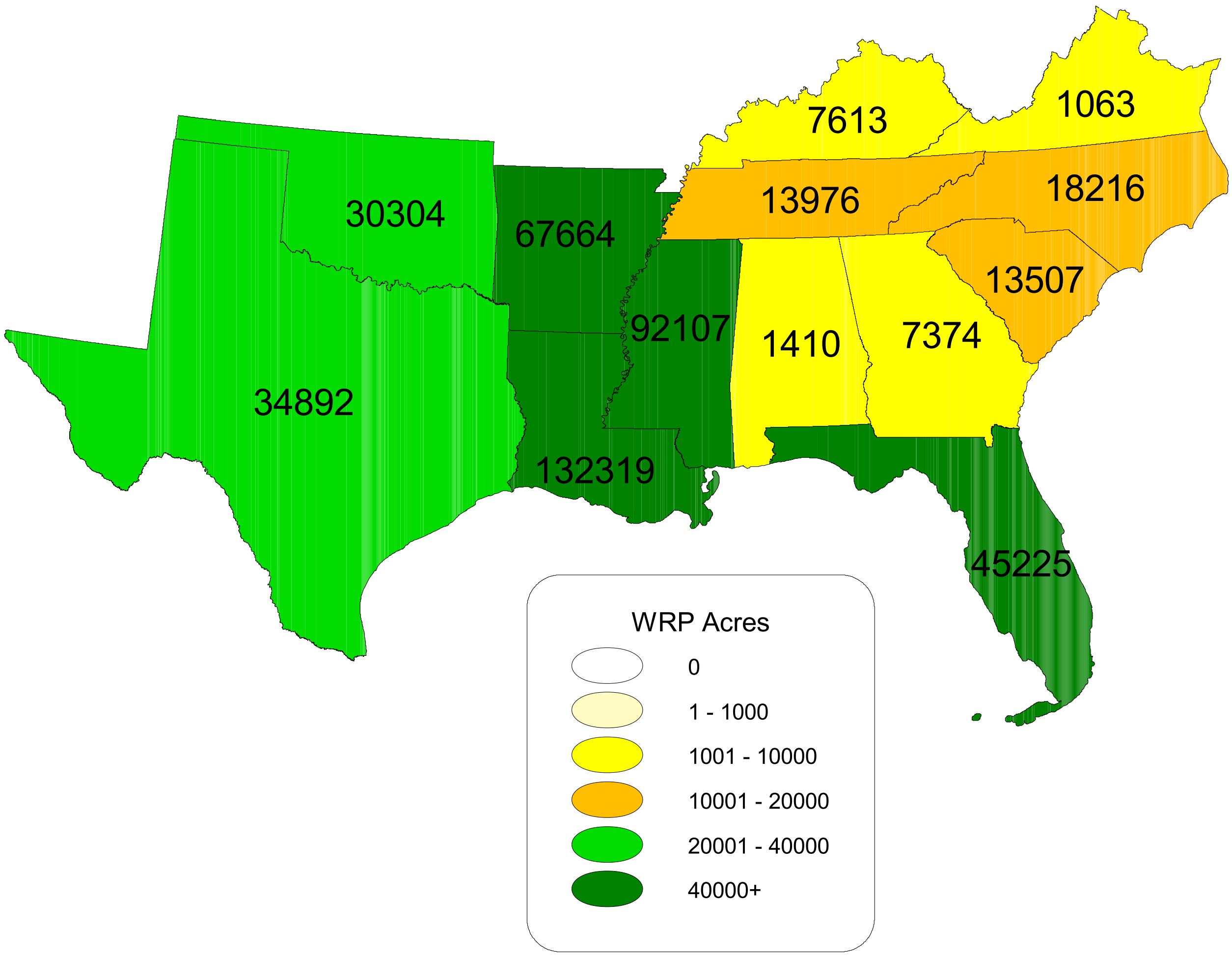
Figure 4--Causes of palustrine and estuarine wetland losses based on 1992-1997 Natural Resources Inventory data.



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