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Agricultural conservation practices and wetland ecosystem services in the wetland-rich Piedmont-Coastal Plain region

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Abstract. In the eastern U.S. Coastal Plain and Piedmont region, diverse inland wetlands (riverine, depressional, wet flats) have been impacted by or converted to agriculture. Farm Bill conservation practices that restore or enhance wetlands can return their ecological functions and services to the agricultural landscape. We review the extent of regional knowledge regarding the effectiveness of these conservation practices. Riparian buffers and wetland habitat management have been the most commonly applied wetland-related practices across the region. Riparian Forest Buffers (RFB) have been most studied as a practice. Water quality functions including pollutant removal, provision of aquatic habitat, and enhanced instream chemical processing have been documented from either installed RFBs or natural riparian forests; forest buffers also serve wildlife habitat functions that depend in part on buffer width and connectivity. Wetland restoration/creation and habitat management practices have been less studied on regional agricultural lands; however, research on mitigation wetlands suggests that functional hydrology, vegetation, and faunal communities can be restored in depressional wetlands, and the wetland habitat management practices represent techniques adapted from those used successfully on wildlife refuges. Other conservation practices can also support wetland services. Drainage management on converted wetland flats restores some water storage functions, and viable wetlands can persist within grazed flats if livestock access and grazing are managed appropriately. Because wetland hydrogeomorphic type influences functions, ecosystem services from conservation wetlands will depend on the specifics of how practices are implemented. In a region of diverse wetlands, evaluation of ecological benefits could be improved with more information on the wetland types restored, created, and managed.

Key words: Coastal Plain; conservation practices; ecosystem services; Piedmont; restoration; riparian buffers; water quality; wetlands; wildlife.

Introduction

With a humid climate and topography favoring poorly drained soils, the U.S. region represented by the Piedmont and Coastal Plain physiographic provinces (Fig. 1) is a wetland-rich landscape. Despite a long history of Native American occupation and impacts from early European settlement, the region still has approximately half of all freshwater wetlands and 95% of all estuarine wetlands (by area) in the conterminous United States (Tiner 1987, Hefner et al. 1994, Moulton et al. 1997). Colonial settlement of the eastern coasts was well established by the early 1700s, and lowland wetlands were drained to support farming and grazing. However, by the early 1800s, trends in population growth and territorial expansion shifted agricultural development and major wetland impacts to states west of the Appalachian Mountains and to the Mississippi Valley (Dahl and Allord 1996). In the east, eventual collapse of upland agriculture by the early 20th century led to widespread land abandonment, natural forest regrowth, and active reforestation (Kauppi et al. 2006), even as wetland drainage continued in some areas. Today, over 60% of the Piedmont and Coastal Plain east of the Mississippi River is forested, and only ~20% is in some form of agriculture (USDA 2006). Over 70% of nonfederal regional wetlands are on lands classed as forest, while <10% are on agricultural lands (USDA NRCS 2009).

Conservation practices implemented by the U.S. Department of Agriculture (USDA) under Food Security Act (Farm Bill) programs can improve the maintenance and delivery of wetland ecosystem services on privately owned agricultural lands. However, the region's distinctive land use history has shaped current interactions between agriculture and wetland conservation practices. The Natural Resources Conservation Service (NRCS) initiated the Conservation Effects Assessment Project (CEAP) to develop methods for quantifying environmental benefits derived from Farm Bill programs and practices, and a major component is assessing benefits to wetland services (see Eckles 2011). This paper forms part of a multi-region information

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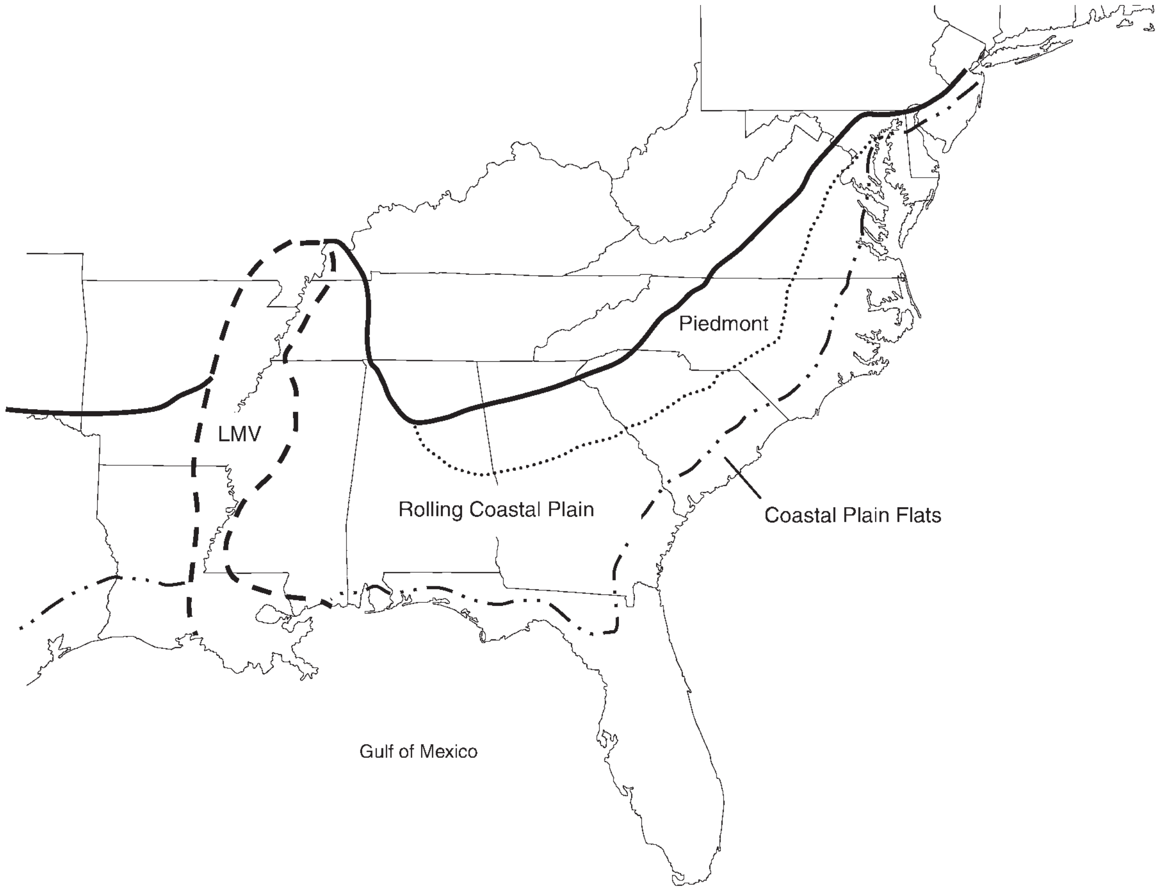

FIG. 1. Subregions of the Gulf–Atlantic Coastal Plain states, USA. The thick solid line indicates the approximate northern extent of Hammond's (1970) Gulf–Atlantic Division landform; dotted and dotted-dashed lines delineate the Piedmont, Rolling Coastal Plain, and Coastal Flats; and the thick dashed line shows the Lower Mississippi Alluvial Valley (LMV) region. Hammond's system is the basis for the USDA Land Resource Regions (LRR), where Piedmont plus Rolling Coastal Plain is the Atlantic–Gulf Slope LRR, and the Coastal Flats equals the Atlantic–Gulf and Florida Lowlands LRRs.

synthesis to summarize current knowledge of the ecosystem services provided by wetland-related conservation practices and to identify knowledge gaps and emerging issues (Brinson and Eckles 2011). Our review focuses on the eastern Piedmont–Coastal Plain, which spans the Atlantic and Gulf Coast states from New Jersey to Mississippi (Fig. 1). We describe key features of regional wetlands, summarize trends in land use and wetland change, and review the available research on the effectiveness of wetland-related practices on regional agricultural lands. The Lower Mississippi Alluvial Valley (LMV) and its associated wetlands are reviewed separately (Faulkner et al. 2011); both the Florida Everglades region and the western Coastal Plain differ sufficiently in ecoregional and agricultural character that we will generally not include them in our treatment.

Piedmont–Coastal Plain Wetlands: Ecological Functions and Services

The Piedmont–Coastal Plain region (Fig. 1) is traversed by many river systems that originate either in the Appalachian Highlands or within the region and discharge to Gulf–Atlantic coastal waters. Wetlands comprise ~16% of regional land area, but subregional percentages vary from low (<5%) to high (~30%) along the seaward gradient from the dissected Piedmont to the poorly drained Coastal Flats (Tiner 1987, Hefner et al. 1994). Wetland diversity is notable and contributes functional complexity to the landscape. All wetland hydrogeomorphic classes (riverine, flat, depressional, estuarine, slope, and lacustrine; Smith et al. 1995) occur in the region, but the first four predominate. The major inland freshwater classes (riverine, flat, depressional) are embedded within uplands and thus are directly affected by agricultural activities. Apart from some localized salt hay farming (see Philipp 2005), agricultural production affects estuarine (saltwater) wetlands mainly indirectly through impacts on the quantity and quality of inland waters reaching the coasts. Consequently, we focus our review on freshwater wetlands.

These regional wetlands have hydrologic, biogeochemical, and biotic functions that provide the "services" to maintain sustainable ecosystems and provide human benefits. All wetland types function in nutrient cycling and transformations, the specifics of which depend upon the organisms present, the substrates, and system hydrology. Likewise, all wetland types generate biological productivity and habitats for plant and animal biodiversity. Economically, forested wetlands provide an important timber resource, and seasonally dry herbaceous wetlands can be grazed. However, wetland types differ in some ecosystem functions and services because of differences in landscape position, water sources, and hydrodynamics (Brinson 1993), as summarized from pertinent reviews cited in the following paragraphs.

Riverine wetlands vary from narrow riparian corridors along small streams to large river floodplains with complex microtopography. These wetlands typically receive water as inflows from adjacent uplands or by periodic overbank flooding. Seasonal flooding dynamics influence substrates, biotic communities, and wetland functions. Floodplain wetlands function uniquely in detaining high-energy floodwaters, attenuating peak flows, and maintaining channel base flows. As part of a landscape drainage network intercepting sediments, nutrients, and other pollutants, riverine and riparian wetlands also play a critical role in regulating the quality of regional surface waters. Nutrients are retained and cycled internally, lost in gaseous forms through denitrification (for nitrogen), and incorporated into organic materials for downstream export to detritus-based estuarine food webs. Riverine wetland soils have more organic matter than upland soils; nutrient and sediment inputs contribute to high biological productivity where soils are periodically aerated. Physiography influences wetland properties, as Piedmont-origin (red- or brown-water) rivers have more inorganic nutrients and sediment loads than Coastal Plain-origin (blackwater) rivers that are nutrient-dilute but high in organic acids. Riverine wetlands are largely forested systems, with diverse forest types shaped by local interactions between hydrology and microtopography. Biological productivity, structural complexity, and adjacency to uplands make riverine wetlands some of the most ecologically and economically valuable wildlife and fisheries habitats in the United States (from Sharitz and Mitsch 1993, Hodges 1998, Kellison et al. 1998).

Wetland flats are common on coastal terraces where low land relief and shallow subsurface confining layers result in saturated soils with poor lateral and vertical drainage. Seasonal changes in evapotranspiration (ET) result in large water table fluctuations, which provide rainwater storage after dry periods and release water slowly from saturated soils by diffuse flow to headwater streams or other shorelines. Adjacent to coastal areas, such freshwater releases are important for regulating salinity conditions in estuarine habitats. Because flats are mainly rain-fed and do not receive upland inflows, they tend to be nutrient-limited, although this varies somewhat with the degree of saturation and contact with mineral soil. Water outflows export dissolved organic carbon and organically bound nutrients, but are low in inorganic nutrients. Interactions among hydroperiod, soil properties, and fire determine the ecological character of wet flats. Mineral-soil flats exhibit an inverse hydroperiod–fire frequency continuum from drier pine savannas to wet evergreen bay forests or deciduous hardwood swamps. Nutrient pulses from fires are rapidly resequestered in recovering vegetation. In areas of prolonged saturation or shallow flooding, peat accretion results in organic-soil flats with evergreen shrub–bog (pocosin) vegetation that burns infrequently. Large expanses of pocosin, as on the North Carolina Coastal Flats, sequester carbon in organic soils and function in maintaining land surface (peat accretion) in response to sea level rise. Wetland flats generally support fauna requiring interspersed terrestrial and wet habitats, or fire-maintained vegetation (from Richardson and Gibbons 1993, Harms et al. 1998, Rheinhardt et al. 2002).

Depressional wetlands include large Carolina bays and smaller wetlands (e.g., Delmarva bays, Citronelle ponds, cypress domes) that are especially numerous across the Rolling Coastal Plain and some parts of the Coastal Flats. Found in various topographic positions, they develop in hollows with a subsurface confining layer that promotes surface water ponding to depths of $\geq$1 meter. Outlets may occasionally be present, but water levels mainly fluctuate vertically with seasonal and annual changes in rainfall and ET. Some groundwater exchanges may occur, depending upon topographic position, underlying substrates, and seasonal shifts in ET and subsurface fluxes. Water storage may be small on a unit basis, but the cumulative effect of many depressions may be substantial at a watershed scale (Brown and Sullivan 1988). In addition to storing rainwater, depressions may retain added nutrients if they receive water inflows and have limited outflows. Depending on size and location, depressional wetlands exhibit hydroperiod diversity from semipermanently ponded to frequently dry; soil organic content and fire susceptibility vary accordingly. These properties shape plant communities that range structurally from open-water ponds to emergent marshes and swamp forests; hydroperiod and vegetation diversity in turn shape the faunal communities. Because periodic drying restricts the presence of permanent fish populations, depressional wetlands have a distinctive habitat function as breeding refugia for many aquatic invertebrates and amphibians (from Richardson and Gibbons 1993, Sharitz 2003, De Steven and Toner 2004).

Wetland Impacts from Piedmont–Coastal Plain Agriculture

The dominant regional soils are highly weathered acidic and sandy Ultisols, with better-drained Udults on the Piedmont and Rolling Coastal Plain, and poorly drained Aquults on the Coastal Flats. Soils of the geologically younger Florida peninsula are mainly sandy Entisols or poorly drained Spodosols (Aquods) (Foth and Schafer 1980). Thus, natural soil infertility or soil wetness have strongly influenced agricultural land use, particularly after European settlement. For over 200 years, farming generally took the form of extensive shifting cultivation. Forestland was cleared, cropped for several years, then abandoned to open-range grazing and forest regrowth while other land was cleared (or re-cleared) for new crops (Otto 1994). Colonists first settled along fertile river valleys of the Coastal Flats, and lowland wetlands were drained and cleared where possible (Lilly 1981, Dahl and Allord 1996). However, the land area needed for shifting agriculture prompted large population migrations inland to the better drained Rolling Coastal Plain and Piedmont. An extensive agriculture of profitable cash and food crops (cotton, tobacco, rice, corn) dominated until the economic upheavals of the Civil War period, after which federal drainage incentives and mechanized technologies began shifting agricultural production farther westward (Rasmussen 1960, Otto 1994). Following widespread farmland abandonment during the economic depression of the 1930s, much of the retired land reverted to natural woodland or active plantation forestry (Allen et al. 1996, Carmichael 1997).

A diversified agriculture currently comprises ~20% of regional land area; concentrated livestock-feeding operations are also common. On highly erodible Piedmont soils, poor historic farming practices resulted in severe topsoil loss and gully erosion into waterways; consequently, federal soil conservation programs promoted a land use shift to pine silviculture (Allen et al. 1996). The Rolling Coastal Plain and Coastal Flats (where drainage allows) remain a mix of pine forestry and agriculture. Pastureland is concentrated in the Piedmont and Rolling Coastal Plain, whereas the open flatwoods of south-central Florida support the only substantial rangeland-based grazing east of the Mississippi River. The more populated mid-Atlantic states of the Chesapeake Bay watershed are more urbanized; however, a recent trend is rapid urbanization across the Piedmont and in coastal areas at the expense of both agricultural and forestland, and at rates higher than national averages (Wear 2002, USDA NRCS 2009).

Agricultural use of Piedmont and Rolling Coastal Plain uplands requires minimal artificial drainage, thus, impacts to adjacent wetlands typically involve upland runoffs or marginal drainage to accommodate field expansion. However, where landscape or wetland internal drainage is poor (as on the Coastal Flats, or in wet flats or depressions generally), agriculture often resulted in larger direct wetland losses because artificial drainage is needed to bring lands into production. For the 200 years between the 1780s and 1980s, estimated losses of original wetland area range from 25% to 55% for most states of the region (Dahl 1990). However, it is unclear what proportion was historical loss vs. accelerated loss in the mid-20th century from intensified agriculture. Apart from some localized areas with large conversions, proportionally less wetland area was drained in the Piedmont–Coastal Plain region compared to the Upper Midwest and Lower Mississippi Valley (where proportional losses often exceeded 70%; Dahl 1990). Reflecting the dominant land use, managed timber harvest from forested wetlands is also a major activity across the Coastal Plain (Kellison and Young 1997).

Because Piedmont–Coastal Plain wetlands comprise high proportions of national wetland area, recent nationwide changes have tended to reflect regional trends (Table 1). Annual rates of net wetland loss have declined substantially since the mid-1950s. Agriculture was the main cause of wetland loss until the mid-1980s, with high regional losses in the bottomland forests of the Lower Mississippi Valley, the wet flats of coastal North Carolina, and the freshwater marshes of the Everglades (Frayer et al. 1983, Hefner and Brown 1985). However, by 2004, urban and rural development was the major cause of wetland loss (Dahl 2006). Declining rates of net loss have been attributed to the introduction of wetland regulation in the mid-1980s; this began a shift from unregulated impacts to either "regulated and permitted" losses under Clean Water Act Section 404 (on nonagricultural lands), or to disincentives against wetland drainage under Farm Bill "Swampbuster" provisions (on agricultural lands). Gains from wetland mitigation and restoration have also offset ongoing losses (Dahl 2006). Another notable trend is a sustained increase in freshwater ponds (open-water areas <8 ha in size), with the result that the latest inventory recorded a net wetland "gain" even as loss of vegetated wetlands continues (Table 1). Half or more of the increase represented created ponds on agricultural or developed lands (Dahl 2000, 2006). New ponds in the Southeast comprised a rising proportion of nationwide increases, from 27% to 54% between the 1950s and 1990s (Table 1); qualitative data (Dahl 2006) suggest that this trend is continuing.

Wetland Conservation Practices in the Piedmont–Coastal Plain

Under Farm Bill programs, conservation "practices" are applied to reduce soil erosion, protect water quality, and provide wildlife habitat or other environmental benefits on agricultural lands. Among numerous practices with defined implementation standards, those most related to wetland ecosystem services are "wetland" and "conservation buffer" practices. Wetland practices involve restoring, creating, or managing wetland habitats. Pond construction may also be a wetland practice if it creates habitat meeting wetland definitions or is designed to support wetland vegetation and fauna. Buffer practices are planted or protected vegetated areas designed to reduce upland impacts on adjacent wetland

TABLE 1. Trends for annual net change in wetland area for selected wetland categories in the conterminous United States, and approximate percentage of the changes occurring in the Southeastern states.

		Freshwater wetlands†		Estuarine wetlands		Freshwater ponds	
Survey period	No. years	Annual net change (10^3 ha/yr)	Percentage in Southeast	Annual net change (10^3 ha/yr)	Percentage in Southeast	Annual net change (10^3 ha/yr)	Percentage in Southeast
1950s–1970s	20	−177.9	84	−7.4	85	+42.0	27
1970s–1880s	9	−114.8	89	−2.7	84	+33.9	50
1986–1997	11	−23.3	45	−0.4	‡	+23.2	54
1998–2004	6	+14.9§	n.a.	−1.9	n.a.	+46.9	n.a.

Notes: Southeastern states are North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Arkansas, Louisiana, Tennessee, and Kentucky. Data not available are indicated by n.a. Data sources are: Frayer et al. (1983), Hefner and Brown (1985), Dahl and Johnson (1991), Hefner et al. (1994), Dahl (2000, 2006), and T. E. Dahl, *unpublished data*.

† Includes open-water ponds.

‡ The Southeast had a net gain of 0.8×10^3 ha/yr.

§ Net gain resulted from ponds, whereas vegetated wetlands had a net loss of -33.4×10^3 ha/yr.

or aquatic ecosystems, or to provide ecotonal habitat. Other conservation practices that are not strictly wetland-related may be used to support wetland and buffer practices. For example, tree planting is a commonly applied practice on uplands, but planting bottomland tree species may be used in restoring a forested wetland or establishing a riparian buffer. Drainage-water management can be applied to protect water quality in adjacent wetlands and waters, and this would require use of specific water-control practices (dike, water-control structure). Practices that manage grazing intensity or livestock access can protect sensitive wetlands and riparian habitats.

Among recent applications of wetland-related practices in the Piedmont–Coastal Plain (Table 2), conservation buffers were the most commonly used. Among wetland practices specifically, managing for wildlife habitat was more frequent than restoration/creation, particularly in the Coastal Flats, where managed public waterfowl impoundments are also common (Gordon et al. 1989). These practices have been supported mainly by the incentives-based Conservation Reserve (CRP) and Wetland Reserve (WRP) Programs, and the advisory Conservation Technical Assistance Program (CTA). Programs can shape practice implementation; for example, most conservation buffers were applied under CRP (77% of buffers and buffer area), whereas wetland practices were associated with WRP and CTA (76% of practice area). Other ancillary practices (Table 2) may be directed in part at wetland services, but the extent cannot be determined from the available data. Of these, water management is used most often in the Coastal Flats, whereas grazing and access management predominate on the Piedmont and Rolling Coastal Plain.

For the two main financial-incentive programs, cumulative regional activities in wetland-related practices comprise relatively small proportions of national totals. Of wetland and buffer practices installed under the Conservation Reserve Program, the eastern Coastal Plain states (excluding Mississippi as part of the LMV) had only 5% of the national acreage in those practices (data from FSA 2006). Of cumulative land enrollments in the Wetland Reserve Program (where presumably most applied practices support wetland functions), the same eastern Coastal Plain states had 12% of the nationally enrolled WRP acreage, in contrast to 32% in the LMV region (data from USDA NRCS 2006). In part, these proportions may reflect the relatively small regional percentage of agricultural land compared to forested land.

Given that the eastern Piedmont–Coastal Plain is predominantly forested, most regional research on how management practices affect wetlands has been conducted within upland forestry lands or in harvested bottomland forests (e.g., Wigley and Roberts 1994, Lockaby et al. 1997, Sun et al. 2001). In contrast, there has been little regional study of most USDA wetland conservation practices on Farm Bill program lands, or on agricultural lands in general. However, research from similar systems on nonprogram lands can provide some indication of the likely benefits. Here, we synthesize the available regional literature on implemented conservation practices and also draw from studies of wetland-related practices in other contexts that may be relevant to providing wetland services on agricultural lands. Resource issues associated with the major inland wetland classes are highlighted and discussed in relation to relevant conservation practices.

Riverine wetlands and riparian buffer practices

Agriculture historically degraded regional riverine systems not only by forest clearing, but also by severe soil erosion and sedimentation into waterways, particularly in the Piedmont and Rolling Coastal Plain. Widespread upland reforestation has contributed to reversing these trends (Hodges 1998), although historic sediment deposits still affect stream floodplain dynamics (Meade 1982, Walter and Merritts 2008). Ongoing concerns are protecting surface water quality from pollutants (excess nutrients, sediment, and chemicals) associated with intensive agriculture, and continued threats to streamside forest habitat from farm field expansion or pond construction. Small streams (first- to third-order) are most in need of riparian protection and

TABLE 2. Frequency of selected wetland, buffer, and ancillary (potentially wetland-associated) NRCS conservation practices applied in recent Farm Bill program enrollments (2000–2006) across the Piedmont and eastern Coastal Plain, by subregions.

	Practice frequency (counts)			
Conservation practice and codes†	Piedmont	Rolling Coastal Plain	Coastal Flats	Region total
Wetland				
Wetland Restoration, Creation (657, 658)	51	176	661	888
Wetland Habitat Management (644, 646, 659)	221	512	1666	2399
Total	272	688	2327	3287
Ponds (378, 399)	143	1367	115	1625
Conservation buffer				
Filter Strip (393)	548	2128	5216	7892
Riparian Buffer (mainly 391, Forested)	1896	4376	2627	8899
Total	2444	6504	7843	16 791
Water management				
Dike (356)	20	74	69	163
Structure for Water Control (587)	73	85	331	489
Drainage Water Management (554)	0	1	179	180
Total	93	160	579	832
Grazing and access management				
Prescribed Grazing (528)	10 463	19 101	3309	32 873
Fencing, Use Exclusion (382, 472)	5101	13 251	2095	20 447
Total	15 564	32 352	5404	53 320

Notes: Data from NRCS program records for the Piedmont–Coastal Plain portions of New Jersey, Delaware, Maryland, Virginia, North Carolina, South Carolina, Georgia, Florida, Alabama, and Mississippi (Mississippi data do not include the Lower Mississippi Alluvial Valley [LMV] region).

† Practice descriptions are available online at ⟨http://www.nrcs.usda.gov/technical/standards/⟩ and ⟨http://www.nrcs.usda.gov/technical/efotg⟩.

represent ~99% of all streams on Piedmont–Coastal Plain landscapes (Rheinhardt et al. 1999, Sweeney et al. 2002). Riparian buffer practices address these concerns and were the most commonly implemented wetland-related regional practice (Table 2). Buffer practices may be installed either as wetlands or adjacent to wetlands, streams, or other water bodies. Riparian Forest Buffers (RFB; practice 391) were used more frequently on the Piedmont and Rolling Coastal Plain, whereas Grass Filter Strips (practice 393) were more prevalent on the Coastal Flats, mainly in states of the Chesapeake Bay watershed. Grass buffers can function in sediment trapping and pollutant removal (e.g., Rankins et al. 2001), but forested riparian zones provide greater structural complexity and ecosystem services than do herbaceous buffers alone (e.g., Stauffer and Best 1980, Sweeney et al. 2004). For example, a rare comparative study on the northern Piedmont (Sweeney et al. 2004) found that forested stream reaches had wider channels and more streambed habitat than grass-bordered streams; consequently, the forested reaches had significantly more macro-invertebrates, organic-matter processing, and nitrogen uptake. Given that forested riparian zones represent the natural condition for regional streams, functions of RFBs have been a focus of regional research studies.

Water quality functions.—The regional RFB practice was largely based on research in existing riparian forests within Coastal Plain agricultural watersheds, which showed that forested riparian zones removed sediment from surface runoff and nitrate from shallow groundwater (Lowrance et al. 1984, Peterjohn and Correll 1984, Jacobs and Gilliam 1985). These early studies provided the model and recommendations for installing and managing the RFB practice (Lowrance et al. 1985, Welsch 1991). The practice standard originally consisted of a three-zone buffer with a sediment-trapping grass filter strip adjacent to the crop field (Zone 3), a permanent hardwood forest along the stream (Zone 1), and a potentially harvestable pine forest (Zone 2) between the two. Currently, the Grass Filter Strip is considered a separate practice (practice 393) that can be installed with RFBs or in other locations.

There is a growing body of research on the regional RFB practice. Installed RFBs removed incoming nitrogen (N) and phosphorus (P), and filtered herbicide movement in the first 10 years after establishment (Vellidis et al. 2002, 2003). A hydrogeomorphic model of biogeochemical functions suggested that restoring 20 m wide (10 m per side) forested buffers on headwater streams could improve water quality without substantially reducing arable land (Rheinhardt et al. 1999).

Field studies confirmed that buffers of this size do provide measurable water quality benefits; for example, narrow (8 m and 15 m) three-zone buffers established under CRP on the North Carolina Coastal Plain consistently removed nitrate from shallow groundwater in two of three cases (Hyatt et al. 2006). Modeling also showed that a minimum RFB width of ~11 m on a farm field with water quality problems could achieve at least a 50% reduction in N, P, and sediment loads (Lowrance et al. 2001).

RFBs provide in situ and downstream water quality services by: (1) filtering and retaining pollutants in surface runoff and subsurface flow from source areas, and (2) enhancing instream chemical processes (Sweeney et al. 2004). Pollutant filtering depends on sediment trapping, water infiltration and soil particle co-deposition, vegetation uptake, and denitrification, all of which are provided by both forest and perennial herbaceous vegetation. Instream effects depend upon benthic habitat area and microbial communities, both of which are enhanced by forest buffers. Improved instream function is inferred from study of existing riparian forests (Sweeney et al. 2004), but the RFB practice is modeled after these forests and thus extrapolation to the practice is reasonable.

The specific processes vary with soil and hydrogeologic conditions (Angier et al. 2002, Spruill 2004). Deep sandy soils have high water infiltration rates, whereas finer textured or organic soils provide slower travel times for wetland-associated subsurface processes such as groundwater denitrification and recharge. Sediment and chemical removal is less effective when surface flow bypasses the RFB through ditches, or if subsurface flow moves below the RFB root zone (e.g., in tile drainage) or passes too rapidly through the surface aquifer. For example, a Coastal Plain riparian forest with high nitrate inputs (25 mg/L) but a high velocity of groundwater movement was inefficient in nitrate removal, allowing 56–68% of nitrate (14–17 mg/L) to reach the stream channel (Correll et al. 1997). An efficient forest buffer would have a configuration that converts surface runoff to subsurface flow, such as a sandy soil with high infiltration rates adjacent to the source area and a downslope organic-rich (wetland) soil with high denitrification rates adjacent to the stream (Casey and Klaine 2001, Casey et al. 2001). The three-zone RFB achieves this configuration because the grass filter strip promotes infiltration and flow spreading as well as sediment trapping. Retention of P depends on soil geochemical properties and vegetation storage, so riparian buffers may be less efficient in P removal because there are no atmospheric sinks for P analogous to gaseous denitrification of N (Walbridge and Lockaby 1994). Soils in Coastal Plain riparian wetlands had less P sorption capacity than upland soils (Axt and Walbridge 1999). In wetland soils converted to agriculture, added calcium and magnesium from past liming may increase soil P retention capacity; this capacity may persist if wetlands are later restored (e.g., Hogan et al. 2004). This suggests a particular need to combine upland and wetland/riparian conservation practices for controlling excess P.

Restoring RFBs requires successfully establishing woody vegetation, typically by planting bare-root tree seedlings. Inadequate seedling survival and growth can limit full attainment of water quality benefits until canopy closure (which may take up to 15 years), but protecting seedlings from herbivory and competing vegetation improves survival (Sweeney et al. 2002). Planting a single fast-growing early-succession species might achieve faster canopy closure, but afforestation with tree monocultures is less desirable ecologically and is generally not allowed in the RFB practice. Instream benefits of restored forest buffers can develop relatively quickly. Newly established (one to four years) RFBs across the Virginia Piedmont–Coastal Plain had positive effects on aquatic ecosystems based on an Index of Biotic Integrity (IBI) and a Stream Visual Assessment Protocol (SVAP), with the most percentage improvement in small streams (Teels et al. 2006). Notably, these buffers were restored by excluding livestock access, which led to rapid growth of planted and natural riparian vegetation. The improved IBI and SVAP indices suggested that gains in riparian functions are possible within the first five years. Established RFBs can also be managed as an agroforestry practice (Schultz et al. 2000). A Coastal Plain RFB where the Zone 2 pine forest was either clear-cut or thinned performed similar water quality functions as an unmanaged RFB (Lowrance et al. 2000, Lowrance and Sheridan 2005). With suitable harvesting practices (see Sun et al. 2001), management of forested riparian zones and wetlands should be possible, although it may take >15 years for some soil properties to return to preharvest conditions (Maul et al. 1999).

Wildlife habitat functions.—In addition to their water quality functions, riparian buffers serve as local wildlife habitat and as movement corridors if they connect larger habitat blocks. The structural complexity and litter layer of forest buffers provide greater habitat value than herbaceous (or no) buffers (Stauffer and Best 1980, Howard and Allen 1989, Sweeney et al. 2004). Habitat quality for instream aquatic fauna is enhanced through increased stream width (more habitat area), canopy shading (moderating water temperatures), and inputs of litter and woody debris (providing a food base, structural cover, and channel complexity). Buffer width is a critical variable affecting habitat functions (Clark and Reeder 2007). Where land relief is low and riparian slopes shallow, RFB widths that protect water quality and aquatic habitat may be narrower than those needed to support terrestrial riparian fauna.

There is a substantial literature on RFBs in forestry applications, where they are termed "streamside management zones" (e.g., Lee et al. 2004). However, silviculture generates less chemical runoff than agriculture, and ecotonal vegetation typically changes less sharply between RFBs and managed stands than between RFBs and farm fields. The wildlife benefits of agricultural RFBs have received less regional study, but results are generally consistent with the forestry studies. Studies in harvested pine plantations on the Texas Coastal Plain (Rudolf and Dickson 1990, Dickson et al. 1995) and in mid-Atlantic agricultural landscapes (Keller et al. 1993) have shown positive effects of RFB width on the abundance and diversity of vertebrate taxa. Narrow buffers and corridors ($\leq$30 m) were less favorable for herpetofauna and some mammals, but supported more bird species of edge and early-succession habitats. In contrast, forest interior/Neotropical migratory birds and arboreal mammals such as squirrels were favored by wider corridors (50–100 m or more). If RFBs are managed as an agroforestry practice, forest thinning will temporarily revert vegetative structure to an earlier successional stage and thus alter the types of riparian fauna using the habitat (Howard and Allen 1989, Lee et al. 2004). Watershed land use may also influence RFB efficacy. In first-order Piedmont streams, indirect evidence suggested that narrow buffer widths would not protect habitat quality for stream salamanders in highly disturbed (urbanized) watersheds (Willson and Dorcas 2003).

Drainage and grazing management on wetland flats

Because of prolonged soil wetness, using wetland flats for crop production or plantation forestry requires wildfire control and artificial drainage via an extensive network of canals and ditches. Drier mineral-soil flats are especially susceptible to conversion because regulatory wetland definitions become problematic (Harms et al. 1998), but technological advances also promoted large-scale drainage of wet pocosin flats in North Carolina (Sharitz and Gresham 1998). Drainage systems increase total and peak discharges, thus altering water export from slower diffuse outflow to faster channelized runoff (Daniel 1981). Productive cropping also requires additional inputs of fertilizer and lime. Higher water and nutrient runoffs degrade water quality in receiving streams and estuaries via altered salinities and eutrophication (Daniel 1981, Sharitz and Gresham 1998).

Drainage management.—Drainage Water Management (DWM; practice 554) can potentially reduce the water quality impacts of agricultural drainage on wet flats. Water-control structures are used to hold back drainage system outflows, mainly during the dormant (nongrowing) season. Retaining dormant-season water in the soil profile mimics the historic hydroperiod and reduces chemical outflows; growing-season water releases can also be regulated to provide moisture for crops (Gilliam et al. 1979). By managing drained flats as temporarily saturated systems, DWM can be used with minimal detriment to crop productivity while restoring some wetland hydrologic and biogeochemical function. The practice is in widespread use on the North Carolina Coastal Flats, where it is generally applied on lands with <1% slope. It has proved successful in reducing nitrate-N exports from drained and farmed wetlands (Gilliam et al. 1979, Skaggs and Gilliam 1981). There is potential for application on steeper lands (up to 2% slope), but installation and maintenance costs are greater. Routing runoffs through buffers or constructed wetlands can also improve outflow water quality (Chescheir et al. 1992, Poe et al. 2003).

Controlling P exports from drained wetland flats may be more complex than for N exports. For example, impaired water quality is a major concern on south Florida wet flats converted to "improved" (drained, fertilized, planted) pasturelands. Long-term pasture fertilization has resulted in elevated soil P levels and eutrophication of wetlands and surface waters, particularly in the ecologically sensitive Lake Okeechobee watershed (Flaig and Reddy 1995, Gathumbi et al. 2005). Outflow water quality may be partially managed by DWM (Tanner et al. 1984, Flaig and Reddy 1995), and a recent pilot project (Bohlen et al. 2009) is applying the practice under a "pay-for-services" approach to provide both water storage and nutrient retention on grazing lands. However, if nutrient-enriched wetland areas are restored hydrologically for purposes of water quality improvement, they may become P sources rather than sinks if excess soil P is re-solubilized (Pant and Reddy 2003).

Grazing management.—In the central Florida Coastal Flats, large expanses of cut-over wet flatwoods were drained with surface ditches and converted to grazing use (Kalmbacher et al. 1984, Long et al. 1986). These areas include semi-native rangeland and improved tame pastures with interspersed wetland marshes, many connected by the ditch systems. In contrast to flats converted for crop production, rangeland flats may be managed with fire to improve the forage resource (Sievers 1985). Livestock can degrade wetland biodiversity and water quality by trampling soils, reducing vegetation biomass, altering plant composition, disturbing wildlife, and depositing excreta (Whyte and Cain 1981, Tanner et al. 1984). Impacts are mitigated through conservation practices that exclude livestock access (practices 382 and 472) or that manage grazing through control of stocking rates and grazing periods (practice 528). Regional application of these practices is common (Table 2), though not all may be associated with wetland or riparian protection.

There has been limited regional study of grazing management practices, perhaps because previous research in western rangelands has demonstrated the likely benefits. However, the presence of functional wetland plant and faunal communities within central Florida rangelands (Babbitt and Tanner 2000, Steinman et al. 2003) indicates that viable wetlands can persist if grazing intensity and season are managed appropriately. Faunal use depends on the vegetative structure of wetlands and the adjacent uplands; for example, wetlands near forested hammocks supported arboreal herpetofauna not seen in wetlands of more open pastures (Babbitt and Tanner 2000). Whether water quality benefits will result from improved grazing management is less certain. Excluding livestock reduces fecal contamination, but studies have suggested that persistent effects of prior pasture fertilization now influence wetland water quality more strongly than short-term changes in stocking rates (Tanner and Terry 1991, Steinman et al. 2003).

Depressional wetlands and restoration/creation practices

Across the Coastal Plain, many depressional wetlands were ditched and drained for agriculture or forestry. Short-hydroperiod wetlands were especially likely to be altered, and small depressions could be completely obliterated by land smoothing for cultivation. As perhaps <10% of these wetlands remain intact, they are sites of high value for conservation and restoration (Sharitz 2003). Owing to recent changes to regulation of nonagricultural wetlands under the Clean Water Act, such "isolated" depressional wetlands are highly vulnerable to development loss on urbanizing lands (see Sharitz 2003). In contrast, there may still be some limited protection afforded on agricultural lands because Farm Bill Swampbuster provisions create financial disincentives to wetland drainage. Loss of depressional wetlands particularly affects habitat functions for characteristic biota such as pond-breeding amphibians that require sites with seasonal or variable hydroperiods to eliminate fish predators.

Depressions are potentially restorable if drainage is blocked to reestablish historic water levels and hydroperiods (De Steven et al. 2006); for example, some self-recovery has been observed in abandoned depression wetlands where surface drainage ditches were not maintained (Kirkman et al. 1996). The relative ease of establishing basin hydrology is one reason that wetlands created to mitigate nonagricultural wetland losses are frequently depressional (e.g., Spieles 2005). On agricultural lands, the conservation practices of Wetland Restoration and Creation (practices 657 and 658) aim to establish functional wetlands similar to the historic condition or to other natural wetlands. We found no literature on how these practices are actually implemented in the region; however, it is probable that many applications are depressional in character owing to relative ease of establishment. Hydrology is restored by drainage cessation and ditch plugging, or created by excavation plus water level control structures or dikes. Revegetation may be accomplished by encouraging natural plant succession, adding wetland topsoil, or planting desired species. Upland buffers around the wetlands may also be established. Building ponds (practices 378 and 399) can create depressional wetland habitats, although it favors permanent open-water sites rather than seasonal or temporary vegetated wetlands. Wetland restoration/creation was relatively uncommon compared to other regional practices (Table 2), with restoration more prevalent than creation. Pond creation was more frequent. Landscape drainage characteristics likely influenced practice application, as restoration/creation was most common on the Coastal Flats, while most ponds were installed on the Rolling Coastal Plain (Table 2).

There has been little direct study of these wetland practices on Southeastern agricultural lands. However, studies of depressional-type restored or created mitigation wetlands on other lands provide some indication of the expected benefits. Reversal of artificial drainage by ditch plugging appears to reestablish hydrologic functions readily, and improvements in water quality have also been observed (e.g., Whigham et al. 2002, Bruland et al. 2003). The region's abundant rainfall, long growing seasons, and diverse flora favor rapid development of functional wetland plant communities from seed banks and dispersal (Whigham et al. 2002, Zampella and Laidig 2003, De Steven et al. 2006). Vegetation composition will depend upon the restored hydroperiod; however, if former wetlands under prolonged drainage have inadequate seed banks, supplemental planting may be needed. Restoring hydrology and vegetation will promote recovery of soil biogeochemical functions, but these may take longer to develop (e.g., Bruland et al. 2003, Hogan et al. 2004). Invasive plant species have not been reported as a specific problem in restored Coastal Plain depression wetlands (De Steven et al. 2006), but could be of more concern in created wetlands that lack seed banks, or in other wetland types such as river floodplains and coastal marshes (e.g., Ward 2002, Battaglia et al. 2009).

With respect to wildlife habitat functions, regional studies in forested and mixed forest–agricultural landscapes have documented successful use of restored or created mitigation wetlands by herpetofauna (Pechmann et al. 2001, Wetland Science Institute 2001, Touré and Middendorf 2002), wetland-dependent birds (Muir Hotaling et al. 2002, Snell-Rood and Cristol 2003), aquatic invertebrates (Streever et al. 1996, Taylor and DeBiase 2005), fish (Streever and Crisman 1993, Langston and Kent 1997), and bats (Menzel et al. 2005). The variable hydroperiods of restored depressional wetlands can be difficult to predict; thus, some fauna will be benefited whereas others may be less favored. On agricultural lands, greater distances from source habitats or inhospitable intervening habitat could potentially hinder colonization by less mobile fauna. Nonetheless, as successful faunal response to wetland restoration has been demonstrated in agriculture-dominated regions such as the Northern Plains (see Haufler 2005, Rewa 2007), undoubtedly the practice would yield positive wildlife benefits in the forest-dominated Piedmont–Coastal Plain region.

Compared to restored wetlands, created sites are more likely to lag in achieving functions equivalent to natural wetlands. Created ponds and wetlands may be engineered to hold water more permanently, rather than function as the seasonal or temporary wetlands important to many depressional wetland fauna. Excavation of steep-sided basins hinders development of ecotonal vegetation typical of natural wetlands (Zampella and Laidig 2003). Created wetlands may achieve wetland soil functions slowly if they are excavated down to a subsoil devoid of organic matter, nutrients, and soil microbes (NRC 2001). These limitations can be overcome in part by importing organic soil amendments or wetland topsoil, which can also introduce desirable wetland plant propagules (e.g., Erwin 1990, Anderson and Cowell 2004, Bruland and Richardson 2004).

Achieving similarity to reference wetlands has been a critical issue for nonagricultural mitigation wetlands because the mitigation is compensating for a permitted wetland loss. On agricultural lands, restoration is not necessarily compensatory; thus, successful wetland establishment can provide beneficial services irrespective of close matching to reference systems. Success depends fundamentally upon establishing wetland hydrology, but also on project goals, the plant or faunal community targeted, and ability to overcome legacies of past disturbance (e.g., residual soil contaminants or exotic species). Restoring a wetland in place provides former functions in the historic location; however, the services recovered may depend on current land uses in the broader landscape (NRC 2001). Wetland hydrogeomorphic type is an important variable for assessing the benefits of restored or created wetlands, but few studies have explicitly accounted for this, and even fewer have examined abiotic and biotic functions simultaneously (Barton et al. 2004).

Other wetland practices

Wetland habitat management, the most common regional wetland "practice" (Table 2), encompasses a suite of related practices that partly reflect variations in management setting: Wetland Enhancement (practice 659) is applied to existing wetlands, Wetland Wildlife Habitat Management (practice 644) is conducted on or adjacent to wetlands or other waters, and Shallow Water Management for Wildlife (practice 646) may be used on fallow agricultural fields or other moist-soil areas. The targeted species are generally waterfowl and wading birds, but other wildlife (e.g., nongame birds, furbearers, herpetofauna) can benefit directly or indirectly. Diverse techniques may be used, but the practices generally consist of managing hydrology and vegetation to provide suitable water, cover, and foods for desired waterbirds. The principal systems (moist-soil management, marsh management, greentree reservoirs) all involve cycles of flooding and dewatering to influence plant succession, aquatic invertebrate production, and waterbird use (Weller 1990, Payne 1992). Small ponds or larger impoundments may be constructed in uplands or by modifying existing wetlands; previously restored/created conservation wetlands can also be managed. Hydrology is manipulated with water-control structures and by reshaping topography to provide varied water depths. Vegetation management can include promoting seed bank emergence, adding plant species, removing exotics, or using prescribed burning and grazing. Sites may be further enhanced by adding coarse woody debris, or by establishing vegetated buffers and corridors to other habitat blocks. This practice suite can also include rehabilitating a degraded wetland or protecting an intact wetland with recognized wildlife value.

We could not identify published studies that specifically examined in what forms, or with what results, these wetland management practices have been implemented on Farm Bill conservation program lands across the Piedmont–Coastal Plain region. However, the practices are intended to mimic techniques widely used on federal and state wildlife areas, based on decades of waterbird-management research elsewhere (see Smith et al. 1989, Weller 1990, Payne 1992). Existing reviews clearly document the benefits of managed wetland habitats for waterbirds and other wildlife (Haufler 2005, Kaminski et al. 2006). Over half of the wetland habitat management practices applied in the region (Table 2) were reported under advisory rather than incentive-based Farm Bill programs, and perhaps involved partnering with wildlife agencies that can supply information on successful techniques from their own refuge monitoring (e.g., Strader and Stinson 2005).

These practices may involve trade-offs with other wetland functions. Managing sites for waterbirds may reduce habitat quality for nontarget species; for example, if conversion to semipermanent wetlands lessens the availability of seasonal or temporary wetlands important to other fauna (Kruczynski 1990). Traditional methods may affect wetland floristic quality; for example, early moist-soil management guidelines encouraged sowing nonnative plant varieties with high seed production for waterfowl food (Weller 1990). Use of native plants or natural regeneration is now encouraged, but nonnatives may still be used if not considered invasive (Strader and Stinson 2005). Finally, how wetland habitat management practices provide for nontargeted services (e.g., soil or water quality) is unstudied. Those functions would likely vary with the landscape setting and how the managed wetland behaves as a hydrogeomorphic type.

Evaluating the benefits of regional habitat management practices is potentially complex. Because management systems and wetland types are diverse, practices may be implemented differently across subregions or even locally. Depending upon the extent of diligence by landowners, complex water-management techniques may not be replicated to the same degree on agricultural lands as on wildlife refuges. Practices may also vary with choice of Farm Bill program. For example, the regional practice data indicated that WRP implemented larger average project area for habitat management than did other programs. Because WRP projects are long term or permanent, they may be coordinated better with federal or state agencies to enhance adjacent protected easements or public wildlife areas (Gray and Teels 2006).

Conclusions

In the eastern Piedmont–Coastal Plain region, ecosystem services provided by the riparian buffer practices have been well documented at farm-field and stream-reach scales. These services include water quality improvement, greater wildlife habitat value, and enhanced aquatic ecosystem functions. In contrast, wetland practices (restoration/creation and habitat management) have not been well studied on regional agricultural lands. Substantial benefits in wildlife habitat and water quality services are likely, based on regional studies of similar practices in other contexts.

Whereas conservation buffer practices are comprised of defined techniques (e.g., riparian forest buffer, herbaceous buffer, filter strip, etc.), the wetland practices are broader in scope. Field implementations may be quite varied depending upon the wetland type being established or managed, landscape location, the nature of past alterations, or landowner objectives. Thus, it is harder to generalize about results of wetland practices because the techniques are diverse, the approaches under each practice overlap considerably, and there is less information on what is installed in the field. Wetland hydrogeomorphic type (depressional, riverine, wet flat) influences wetland functions and the resulting ecological services. In regions dominated by one wetland type (e.g., prairie potholes in the Northern Plains), the services from wetland practices will be evident. However, in a region of diverse wetlands such as the Piedmont–Coastal Plain, the outcomes of practice application are less certain because the wetland types or management systems are not known. Ecological benefits will be provided regardless, but a fuller knowledge of what is implemented (whether through improved practice definitions, monitoring, or other mechanisms) would improve understanding of the range of ecosystem services provided in wetland-diverse regions. For example, more detailed standards referencing the wetland type restored or the habitat-management technique used would bring wetland practices more into line with practice suites that have defined subclasses, such as conservation buffers.

Emerging issues

A regional trend over the past several decades has been a significant increase in open-water ponds, which are defined as wetlands in wetland status surveys (Dahl 2006). In offsetting losses of vegetated wetlands, created ponds contribute to meeting wetland "quantity" goals, but may not provide comparable ecosystem services to vegetated wetlands. However, the prospects for doing so may be greater on agricultural lands than in urban settings. Ponds as a conservation practice are unstudied in the region and merit further attention. Understanding farm pond ecological functions, especially over time if they are impacted by sedimentation or other agricultural activities, will be important for assessing gains in wetland "quality" as well as quantity.

The Southeastern United States also experienced a rapid growth of concentrated swine-feeding operations in the 1990s, raising concerns about water quality impacts from inadvertent discharges of waste effluents to surface waters (Stone et al. 2004). Effluent management typically involves storage in settlement and anaerobic lagoons, followed by land application. Focused research has indicated that constructed treatment wetlands can reduce nutrient loads in these effluents before they are reapplied to crop fields (Cronk 1996, Knight et al. 2000), but questions regarding effectiveness remain. Most such treatment wetlands have been installed in the Southeastern states (Cronk 1996), as pilot or operational projects (e.g., Stone et al. 2004). If improved designs contribute to greater feasibility, the practice may come into more use on regional agricultural lands. As these constructed wetlands may be recorded in wetland status surveys, it could be of future interest to determine if functions other than pollutant reduction can develop in these sites.

Ongoing research is documenting that wetland and buffer practices on agricultural lands can provide ecological services at local scales. However, there has been little attention to evaluation at the watershed or landscape scale. For example, the water quality benefits of buffer practices have been demonstrated at stream-reach scales, but land use changes in the larger watershed may offset such local improvements. Many fauna readily use restored/created wetlands, but whether populations are enhanced region-wide is a question that cannot be addressed at the single-farm scale. Because installing these practices for experimental study even in small watersheds will be beyond the scope of most research agencies, assessing larger scale effects will necessarily require the integration of research monitoring with targeted program implementation of practices in targeted watersheds.

Finally, with rapid urbanization occurring across the Southeast (Wear 2002), a longer term question is how the wetland practices and ecosystem services installed on agricultural lands may be affected by future land use change. We can hypothesize that outcomes could vary with program and practice type. Practices installed under 30-year or perpetual easement programs (e.g., Wetland Reserve Program) could be more persistent than those established under shorter term (10–15 year) contract programs (e.g., Conservation Reserve Program). Depending upon the intensity of rural-to-urban conversion, retained riparian buffers could still provide water quality services in the altered landscape, whereas conservation wetlands established for habitat services might be more likely to experience impairment or loss as the landscape matrix becomes more urbanized.

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