

EVALUATING THE LOCATION, EXTENT, AND CONDITION OF ISOLATED
WETLANDS IN THE DOUGHERTY PLAIN, GEORGIA, USA

by

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(Under the Direction of Jeffrey Hepinstall-Cymerman)

ABSTRACT

Geographically isolated wetlands are common throughout Georgia's Dougherty Plain physiographic district, and provide many valuable ecosystem services. However, they are currently excluded from federal and state protection. To better understand the isolated wetland resource, we created a spatially-explicit timeline of land use and land cover within Dougherty Plain, developed a spatial map of isolated wetlands, and constructed a multi-metric framework to assess the ecological condition of 40 isolated wetlands at multiple scales. We found drastic shifts in land use and land cover between 1948 and 2007. We estimate that there are approximately 11,620 isolated wetlands covering 42,431 ha within the Dougherty Plain. The ecological assessment showed strong correlation between assessment methods and scales, and indicates that a remote assessment may be suitable for future studies. Future research can use this information to identify isolated wetlands in the Dougherty Plain, explore their land cover trajectories, and assess their ecological condition.

INDEX WORDS: Dougherty Plain, Georgia, ecological assessment, isolated wetland, land cover, land use,

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CHAPTER 1

INTRODUCTION

PROJECT OVERVIEW

Geographically isolated wetlands are common features of the landscape throughout the southeastern Coastal Plain and particularly within the karst topography of the Dougherty Plain in southwestern Georgia (Beck & Arden 1983; Tiner 2003a). A geographically isolated wetland refers to an individual wetland that is completely surrounded by upland of a greater elevation and thereby isolated from other bodies of surface water (Tiner 2003b). The specific wetlands of interest in this study are known locally by many names including: lime-sink ponds, lime-sink depressions, Grady ponds, Citronelle ponds, gum ponds, and Carolina bay-like depressions. These wetlands contribute to regional biodiversity and produce an array of valuable ecosystem services (Pechmann et al. 1989; Edwards & Weakley 2001; Leibowitz 2003; Whigham & Jordan 2003).

Isolated wetlands are vulnerable to anthropogenic disturbance and historically have been subjected to widespread modification (Bennett & Nelson 1991; Brinson & Malvarez 2002; Tiner 2003a). The conversion of these wetlands and the forested uplands in which they are embedded to other land use types poses a critical threat to the ecological integrity of these systems. Currently, geographically isolated wetlands are not subject to federal regulation under the Clean Water Act (*Rapanos v United States* 2006; Leibowitz et al. 2008; Sponberg 2009), nor are they regulated by the state of Georgia. Basic information such as the number, spatial extent, spatial

configuration, and condition of geographically isolated wetlands throughout Georgia is lacking or extremely limited and available data is often contradictory (Environmental Protection Division 1999; Brown 2002). In the absence of such information, state and federal regulators are unlikely to prioritize the legal protection of isolated wetlands. To improve understanding of the isolated wetland resource in the Dougherty Plain, this study quantifies the changing landscape of the Dougherty Plain, estimates the number and location of isolated wetlands, and develops a multi-scale assessment framework to characterize the condition of isolated wetlands.

The following sections review pertinent literature on the characteristics of geographically isolated wetlands, threats facing isolated wetlands in a changing landscape, and methods of assessing wetland condition.

LITERATURE REVIEW

Characteristics of Geographically Isolated Wetlands

Geographically isolated wetlands are found throughout North America, although there is regional variation in density, size, geomorphology, hydrology, function, vegetation type, name, and other characteristics (Tiner et al. 2002; Tiner 2003a). Such wetlands are common features throughout the Dougherty Plain of southwestern Georgia and are thought to originate from the dissolution of underlying limestone deposits and the subsequent subsidence of the residual overburden (Beck & Arden 1983). These wetlands vary in size from large (hundreds of hectares), shallow, flat areas to small (several m²), steep sided holes (Kirkman et al. 2000). Based on vegetation and soils, minimally disturbed isolated wetlands in the Dougherty Plain can be classified into 3 distinct types: cypress-gum swamps, cypress savannas and herbaceous marshes (Kirkman et al. 2000). These differences have been attributed to multiple factors including the geologic age of individual wetlands (Hendricks & Goodwin 1952), hydroperiod,

fire frequency, and disturbance history (Kirkman 1995; Kirkman et al. 1996; Kirkman et al. 2000).

The pre-settlement Dougherty Plain was dominated by fire-dependent longleaf pine (*Pinus palustris*) or slash pine (*P. elliottii*) communities (Kirkman et al. 2000). Isolated wetlands embedded within this landscape are (or were) frequently subjected to lightning- and human-ignited fire as it spread from upland areas during dry periods (Ewel 1990; Sutter & Kral 1994; Kirkman 1995; Sharitz & Gresham 1998; Kirkman et al. 2000). As a consequence of recurring fire, this frequent disturbance regime can be a major driver in the development of the structure and diversity observed in the unique floral and faunal communities inhabiting these wetlands (Wharton 1978; Sutter & Kral 1994; Kirkman et al. 2000). The intensity and timing of fire has been shown to affect the regeneration of wetland plants (Kirkman & Sharitz 1993; Laubhan 1995; Kirkman et al. 1998a). The relative fire tolerances of plants may play a major role in the structure of plant communities in fire-prone wetlands, and the absence of fire may alter long-term successional pathways and result in distinct plant communities (Kirkman et al. 2000).

Isolated wetlands support species-rich plant communities that include many rare or endemic plants (Kirkman & Shartz 1994; Sutter & Kral 1994). On the privately owned property of the J.W. Jones Ecological Research Center in southwestern Georgia (Ichauway), depressional wetlands make up <3% of the land area, but harbor ~30% of vascular plant diversity (Drew et al. 1998). Depressional wetlands also provide vital wildlife habitat, especially for pond-breeding amphibians (Pechmann et al. 1989; Kirkman & Shartz 1994; Sutter & Kral 1994; Semlitsch & Bodie 1998; Snodgrass et al. 2000; Edwards & Weakley 2001; Smith et al. 2006). Specifically, minimally disturbed wetlands located at Ichauway have been shown to support 28 species of

amphibians, 13 of which breed almost exclusively in such wetlands (Moler & Franz 1987; Smith et al. 2006).

Ecosystem Services

Ecosystem services refer to those benefits that society obtains from ecosystems and may include provisioning services (e.g. fresh water, food, fiber, and fuel), supporting services (e.g. nutrient cycling and soil development), regulating services (e.g. storm/floodwater attenuation or regulation of disease vectors) and cultural services (e.g. religious, educational, or recreational opportunities) (Millinium Ecosystem Assessment 2000). However, ecosystem services are often non-market or marketless products, meaning that they are not actively bought and sold, and thus are frequently viewed as lacking economic value. Nonetheless, attempts have been made to quantify the fiscal value of such services and the ecosystems that provide them (Turner et al. 1988; Costanza et al. 1989; Mitchell & Carson 1989; Dixon & Sherman 1990; Barde & Pearce 1991; Aylward & Barbier 1992; Pearce 1993; Larsen et al. 1994; Costanza & Folke 1997; Goulder & Kennedy 1997). A synthesis of relevant literature found that terrestrial (land-based) wetlands, as opposed to marine or coastal wetlands, benefit society by providing gas regulation, disturbance regulation, water regulation, water supply, waste treatment, habitat/refugia, food production, raw materials, recreation, and cultural services (Costanza et al. 1997). Based on services provided, terrestrial wetland systems were valued at \$14,785 (1994 USD) per hectare per year (Costanza et al. 1997). According to this valuation, terrestrial wetlands are by far the most valuable ecological systems on earth from a dollar per hectare perspective. The second most valuable system was determined to be Lakes/rivers, and was valued at a distant \$8,498 (1994 USD) per hectare per year. While such studies are fraught with limitations and

confounding variables, the point is clear: wetland systems provide multiple services of significant value to humanity.

Although a thorough analysis of the particular ecosystem functions/services associated with isolated wetlands of the Dougherty Plain is lacking, broad conclusions may be drawn from research in similar wetland systems. Isolated wetlands potentially reduce flood peaks by attenuating surface flows and increasing evapotranspiration (Leibowitz 2003), and improve water quality by acting as nutrient sinks for non-point sources of pollution within the wetland catchment basin (Leibowitz 2003; Whigham & Jordan 2003). Isolated wetlands support regional biodiversity by providing unique habitats that maintain meta-populations of upland and wetland flora and fauna, including endemic, threatened, or endangered species (De Steven & Toner 1997; Semlitsch & Bodie 1998; Kirkman et al. 1999; Snodgrass et al. 2000; Edwards & Weakley 2001; Sharitz 2003; De Steven & Toner 2004; Smith et al. 2006). Although not well documented, these wetlands may also have important roles in carbon sequestration (Euliss et al. 2006) and the maintenance of human health (Coluzzi 1994; Wilcox & Colwell 2005; Chipps et al. 2006; PonÇOn et al. 2007; Kirkman et al. 2010).

Functional properties of wetlands vary as a result of complex relationships between landscape position, hydrologic regime, and disturbance history. Thus, the capacity of an individual wetland to produce beneficial services to society also varies according to differences or fluctuations in these variables. Additionally, the ability of a wetland to function properly or provide benefits and services to society can be impaired or eliminated by naturally- or anthropogenically-induced alterations to normal processes or disturbance regimes (e.g. hydrodynamics, fire frequency/intensity, nutrient accretion) (U.S. Environmental Protection Agency 2001).

Legal Status

The current legal status of isolated wetlands is ambiguous and has been described elsewhere (Christie & Hausmann 2003; Leibowitz et al. 2008; Sponberg 2009). Briefly, because isolated wetlands lack connections to other bodies of surface water they were not subject to regulation under the Federal Water Pollution Control Act of 1972, or subsequent amendments (AKA: Clean Water Act [CWA]). The preamble of the 1986 reissuance of the definition of jurisdictional waters referred to the use of isolated wetlands by migratory birds as an example of links to interstate commerce, effectively extending federal regulation authorized by the CWA to isolated wetlands (51 C.F.R. 41206, 41217). However, in 2001 the Supreme Court ruled that isolated wetlands were not “waters of the United States,” and therefore were not subject to federal regulation under the CWA (*Solid Waste Agency of Cook County v United States Army Corps of Engineers* 2001). In *Rapanos v United States* (2006) the Supreme Court ruled that non-navigable streams and adjacent wetlands (NNSAWs), which could include isolated wetlands, may be subject to regulation under the CWA if they display a demonstrable “significant nexus” with navigable waters (Leibowitz et al. 2008). This ruling has generated much confusion within the regulatory and scientific communities (Sponberg 2009).

Given the current lack of clear federal regulation, some states have enacted legislation designed to regulate isolated wetlands (Christie & Hausmann 2003); however, no such policies have been enacted in Georgia. Indeed, basic information on which to base a regulatory framework, such as trends in the number, extent, spatial configuration and condition of isolated wetlands throughout Georgia is lacking, limited, or frequently contradictory (Environmental Protection Division 1999; Brown 2002).

Threats Facing Isolated Wetlands in a Changing Landscape

The forests and associated communities of the southeastern U.S. (FL, GA, SC, NC, VA, KY, TN, AL, MS, LA, AR, eastern TX and eastern OK) have changed dramatically since European colonization. In pre-settlement times the forests were generally more open and this structure was maintained by frequent fires from lightning strikes and anthropogenic ignition (Carroll et al. 2002; Owen 2002; Leininger & Reams 2004). This is particularly true of the longleaf pine forest, which once dominated the southeastern Coastal Plain, and extended into other physiographic provinces (Christensen 1981; Frost 1993; Ware et al. 1993). The introduction of European diseases and subsequent reduction of Native American populations reduced the frequency of anthropogenic ignition sources, which altered the fire regime and resulted in forests that are more dense and stratified (Carroll et al. 2002; Owen 2002; Leininger & Reams 2004; Rauscher 2004). Southeastern forests have been continually altered since European settlement, directly through timber harvesting and conversion to agriculture or other land uses, and indirectly through the introduction of exotic plants, animals, and microbes (Owen 2002; Leininger & Reams 2004; Rauscher 2004). The current condition of southeastern forests is the product of a long history of land use.

Throughout North America, as well as the Southeast, isolated wetlands have been, and currently remain, susceptible to anthropogenic disturbance and modification (Leibowitz 2003; Tiner 2003b). The most common types of disturbance are associated with agriculture, plantation forestry, and urban land uses (Leibowitz 2003). Regionally, approximately 97% of Carolina Bays in South Carolina have been disturbed by agricultural or silvicultural practices (Bennett & Nelson 1991). Additionally, 25% of the area covered by palustrine emergent wetlands within the Ichauwaynochaway Creek Basin of southwest GA was lost, primarily to agriculture, between the

mid-1980's and mid-1990's (Houhoulis & Michener 2000). Such disturbances alter the hydrogeomorphic variables and associated natural processes that create and maintain these unique environments and thus curtail the ability of affected landscapes to provide ecosystem services.

The history of the Southeast is intimately intertwined with the expansion of the timber and forest products industries (Rauscher & Johnsen 2004; Izlar 2006). Currently, the Southeast is the largest global supplier of softwood timber (Leininger & Reams 2004). Earlier research and development in areas such as silviculture, fire suppression, nurseries, insect and disease control, forest survey techniques, mensuration, statistics, tree genetics, and forest products paved the way for the broad scale industrial planting of southern pines in the mid-twentieth century (Barnett 2004; Leininger & Reams 2004; Izlar 2006). Although there are instances of southern pine plantations being established as early as 1913 and large tracts were planted by the Forest Service and Civilian Conservation Corp during the 1920's and 1930's, the effort did not begin intensively until the 1950's (Barnett 2004; Leininger & Reams 2004; Izlar 2006).

In 1953 planted pine covered approximately 809,370 ha across the Southeast and increased to 12,949,940 ha by 1999 (Wear & Greis 2004). These plantations were established on land which previously supported mixed pine or hardwood forest (47%), natural pine forests (28%), or agricultural fields (25%) (Wear & Greis 2004). A study of landscape changes in Georgia found that forested acreage generally increased throughout the state between 1935 and 1985, and that the proportion of area occupied by forests was lowest in the agricultural counties in the southwestern corner of the state (Odum & Turner 1989). In 1985, pine plantations comprised approximately 15% of total forested acreage in the state (Odum & Turner 1989).

The conversion or drainage of wetlands for agricultural use has been identified as the primary driver of inland wetland loss in the U.S. and around the world (U.S. Congress Office of Technology Assessment 1993; Millinium Ecosystem Assessment 2000; Heimlich 2003; Wiebe & Gollehon 2006). In 2002, agricultural croplands represented the third largest land use in the contiguous 48 states, and covered ~23% of total land area (Wiebe & Gollehon 2006). The face of agriculture was profoundly changed with the advent of center pivot irrigation in 1948 (Harrison 2008). Center pivot irrigation expanded rapidly, covering ~7,891,370 ha in 1998, and accounting for ~31% of all irrigated land in the U.S. (Martin 1999). In 1970, Georgia farmers irrigated ~58,679 ha with center pivot systems (Haire 2005). The largest increase in irrigated acreage in Georgia occurred between 1977 and 1980, when irrigated acreage grew from ~239,574 ha to ~404,686 ha (Haire 2005). By 2004, ~602,982 ha of cropland were under irrigation in Georgia, with corn, cotton, and peanuts accounting for ~76% of the irrigated area (Haire 2005).

Farmers quickly embraced center pivot irrigation in the Dougherty Plain; where soils are typically sandy and well drained (Harrison 2008). Five counties (Baker, Mitchell, Seminole, Miller and Decatur) within the Dougherty Plain, accounted for 66% of the total groundwater withdrawals within the Flint River Basin in 1990 (Couch et al. 1996). The largest proportion of total water withdrawals within the Lower Flint River Basin are for agricultural use and averaged 178 million gallons per day in 1990, and this demand is predicted to double by 2020 (Marella et al. 1993).

Like agriculture, the area of urban land in the contiguous 48 states increased rapidly, from ~6,070,284 ha to ~25,899,881 ha, between 1945 and 1997 (U.S. Government Accounting Office 2001; Heimlich 2003). Paved road mileage increased by ~278 percent over the same time

period (U.S. Environmental Protection Agency 2000b; U.S Government Accounting Office 2001). Up to 50 percent of the land area within U.S. cities consists of transportation-related infrastructure, such as parking lots and roads (Natural Resources Defense Council 1999; U.S. Government Accounting Office 2001). The Dougherty Plain is primarily rural. Albany is the largest city, and is located in Dougherty County in the northeast portion of the Dougherty Plain. The surrounding 5-county metropolitan area has a population of ~165,000 (U. S. Census Bureau 2009).

Hydrologic Alterations

The function and vegetative structure of depressional wetlands is intimately tied to hydroperiod (Torak et al. 1993; Kirkman et al. 2000); however, the primary variables driving hydroperiod and seasonal hydrologic variation within these wetlands in the Dougherty Plain are debated (Phillips & Shedlock 1993; Sutter & Kral 1994; Crownover et al. 1995; Lide et al. 1995). One assertion is that recently formed depressions drain into the underlying residuum, and over time the bottom becomes sealed by an impermeable layer of clay and silt (Hendricks & Goodwin 1952; Hayes et al. 1983), thereby rendering the hydrodynamics of the wetland independent of underlying aquifer variation (Blood et al. 1997). Previous studies in the Dougherty Plain suggest that evapotranspiration and precipitation are the dominant forces influencing wetland hydrologic regimes and that interaction with surficial groundwater occurs only in times of high water table elevation (Hendricks & Goodwin 1952; Hendricks 1954). Conversely, a simulation model of the water elevation and flow paths of the upper Floridan aquifer, which underlies the Dougherty Plain, identified depressional wetlands that were operating as recharge and discharge points of the aquifer within the Dougherty Plain (Torak et al. 1993). The limestone bearing the upper Floridan aquifer is highly fractured within the

Dougherty Plain, and the confining layer that separates the aquifer from the overburden is variable and discontinuous (Hayes et al. 1983). Thus, hydrologic linkages between the local residuum and the aquifer may occur.

Certain locations in the Dougherty Plain have been identified as vertical discharge areas where the upper Floridan aquifer discharges into the unconsolidated residuum (Torak et al. 1996). Research in one such location identified an assemblage of hydrologic controls that may be correlated with wetland hydrodynamics (Blood et al. 1997). The findings of this study conflict with the long held belief that these wetlands are independent systems responding only to rainfall and evapotranspiration. While consensus is lacking as to the primary drivers of wetland hydrodynamics, it is evident that wetland hydrology is attributable to multiple factors including wetland morphology, rainfall, evapotranspiration and subsurface stratigraphy, thus, dynamics vary throughout the region depending on the local environment and conditions at each site.

Minimally disturbed wetlands display wide variation in the frequency, length, and depth of inundation (Hendricks & Goodwin 1952; Blood et al. 1997; Kirkman et al. 2000). Generally, these wetlands are filled by precipitation during late winter storms and dry down during early summer (Battle & Golladay 2001a), but the length and depth of inundation in any one wetland is ultimately controlled by the aforementioned factors. While hydrology is often the dominant factor in the development of wetland vegetation types, landscape position, soils, and natural disturbances, such as fire, frequently interact with the hydrologic regime to influence vegetation (Cypert 1961; De Steven & Toner 1997; Kirkman et al. 2000; De Steven & Toner 2004; Euliss et al. 2004; Sharitz & Pennings 2006). The isolated and often shallow profile of these systems makes them particularly vulnerable to the alteration of hydrologic regimes from anthropogenic

disturbances such as draining, filling, or conversion to permanent ponds for recreational or agricultural uses.

Alterations to wetland hydrology are a concern because of the importance hydrology plays in controlling other natural processes and driving the development of floral and faunal communities (Kirkman & Sharitz 1993; Kirkman 1995; Kirkman et al. 2000; Euliss et al. 2004). Altered hydrologic regimes likely illicit the most direct and radical changes within wetlands, particularly in the short term, as compared to sedimentation or nutrient enrichment (Reinelt et al. 1998). Changes in hydrologic inputs may alter the depth, duration, and frequency of inundation periods and alter water level response times within affected wetlands (U.S. Environmental Protection Agency 1993; Euliss & Mushet 1996; Reinelt et al. 1998).

Large scale landscape alteration, such as agriculture, silvicultural operations, and urban expansion, often begin with the ditching of wetlands. Wetland drainage decreases hydroperiod length, and may result in subsequent invasion by terrestrial vegetation and the loss of valuable amphibian breeding grounds. Conversely, a ditch may be used to channel water into a wetland, and often one wetland is ditched and drained into a neighboring wetland (McAllister et al. 2000; Leibowitz 2003). This may result in an increase in the duration, frequency, or depth of inundation. Permanently inundated wetlands are more likely to support fish and other top predators (Snodgrass et al. 1996, which may negatively impact the success of many amphibian species and lead to changes in the structure of macroinvertebrate communities (Brooks 2000; Acosta & Perry 2001; Battle et al. 2001; De Meester et al. 2005).

Inevitably, the conversion of natural forest lands to other land use types involves the removal of forest canopy. Forest canopy removal decreases the leaf area index and the interception rate of precipitation, and results in a greater amount of precipitation reaching the

ground and loss of evapotranspiration. In the case of forestry, canopy removal has been shown to elevate water tables and increase the depth and length of wetland inundation for several years following harvest activities (Lockaby et al. 1997; Sun et al. 2000; Bliss & Comerford 2002; Jackson 2006). However, the hydrologic alterations associated with tree harvest are probably small and short-lived, relative to the natural variation associated with these systems, and have seldom been identified as primary drivers of biotic change in wetlands (Batzner et al. 2000; Hutchens et al. 2004; Sun et al. 2004).

Relative to silvicultural activities, the hydrologic impacts associated with agricultural and urban alterations are generally longer-lived. These activities often involve the denuding and compaction of soils by heavy machinery that reduces infiltration rates (Jackson 2006). The combined effect is an increase in the amount of precipitation contacting the soil surface and a decrease in the infiltration rates of the soil. When the precipitation rate exceeds the infiltration rate overland flow occurs. In the Dougherty Plain, where topographic relief is limited and natural surface drainages are scarce, overland flow may terminate in topographic depressions where isolated wetlands occur. Overland flow may influence receiving wetlands by altering the volume and frequency of hydrologic inputs, transporting contaminants, and scouring or depositing sediments. Thus, overland flow has been identified as a primary driver of change in wetlands and may affect hydrology, water quality, soils, and biota (Watson et al. 1981; U.S. Environmental Protection Agency 1993; Guntenspergen & Dunn 1998; Reinelt et al. 1998; Jackson 2006).

The volume of runoff generated within urban areas is probably greater than in agricultural areas due to the percentage of impervious land cover (U.S. Environmental Protection Agency 1993; U.S. Government Accounting Office 2001). Additionally, wetlands in urban areas

are often prime candidates for storm water storage and runoff is intentionally directed to them through storm sewer networks or ditches (Livingston 1988; McArthur 1989; U.S. Environmental Protection Agency 1993; Reinelt et al. 1998). Thus the impacts to a wetland's hydrologic regime may be more pronounced. For example, in depressional wetlands in Washington State, plant species richness was significantly and negatively correlated with increasing impervious area within the watershed and water level fluctuations (Azous 1991; Reinelt et al. 1998). Furthermore, native plant species were less tolerant of water level fluctuations than were exotic species (Cook & Azous 1997; Reinelt et al. 1998). Concurrently, amphibian species richness was negatively correlated with increasing watershed development and associated water level fluctuations (Richter & Azous 1995; Reinelt et al. 1998).

In agricultural landscapes where field irrigation is widespread, such as the Dougherty Plain, other impacts may occur. Depending on the source of irrigation water (surficial aquifer, confined aquifer, or surface water) and the source of hydrological inputs for a given wetland, inundation periods of receiving wetlands have been shown to decrease or increase (Bolen et al. 1989; Smith & Haukos 2002; Jackson 2006). However, these findings may not be relevant in wetlands where hydrodynamics are driven by precipitation and evapotranspiration. Irrigated agriculture may impact wetland hydroperiods directly through supplemental water inputs or withdrawing water from a wetland for irrigation, or indirectly by lowering local groundwater levels.

Physical Disturbance

Physical disturbance refers to the direct alteration of natural sites; often through mechanized means. In forestry, many early pine plantations were established on abandoned agricultural lands and such plantations were much more productive than their counterparts

located on lands without a history of agriculture (Leininger & Reams 2004). Therefore, foresters developed mechanical site preparation techniques designed to create conditions similar to those found in old fields (Leininger & Reams 2004). In the southeastern Piedmont and upper Coastal Plain mechanical site preparation most frequently involves shearing with a KG blade, followed by windrowing and broadcast disking (Leininger & Reams 2004). Windrows and slash piles are often burned before replanting (Leininger & Reams 2004). In the Coastal Plain, soil saturation can be problematic even on drained sites and bedding techniques were developed to create elevated, well-drained microsites which improve seedling survival and growth (Leininger & Reams 2004).

The intensity of mechanized site preparation generally increased during the 1960's and 1970's (Leininger & Reams 2004). Such practices left sites with bare soil that resulted in increased erosion, increased sedimentation rates, and turbidity in receiving streams and wetlands (Leininger & Reams 2004). Environmental concerns led to the regulation of forestry activities as nonpoint-sources of pollution under the Clean Water Act (Leininger & Reams 2004). Subsequently, voluntary Best Management Practices (BMPs) for forestry have been established in all southeastern states (Leininger & Reams 2004). Environmental concerns, federal legislation, BMPs, and concerns over long-term site productivity led foresters to look for less intensive methods, such as chemical site preparation (Leininger & Reams 2004).

While the impacts of silvicultural practices on biotic assemblages within streams have received much attention, relatively little research has focused on similar impacts in wetland systems. Hutchens et al. (2004) reviewed relevant research on the impacts of silviculture within both stream and wetland systems of the eastern United States. Initially following canopy removal, previously forested wetlands may be invaded by upland species or they may develop

characteristics more commonly associated with marshes (Mitchell et al. 1995; Aust et al. 1997; Batzer et al. 2000; Roy et al. 2000; Hutchens et al. 2004). Corresponding changes have been observed in faunal communities with species composition shifting toward that more commonly observed in upland or marsh conditions (Mitchell et al. 1995; Clawson et al. 1997; Perison et al. 1997; Batzer et al. 2000; Hutchens et al. 2004). Loss of canopy can also lead to elevated levels of solar radiation, which can affect soil and water temperatures and may impact amphibian communities (Clawson et al. 1997). Many of the changes described above are presumed to be related to increased solar radiation following canopy removal and vegetation may revert back toward original communities as the site successional matures. For example, a vegetation analysis of recently clear-cut cypress swamps in Florida found that 9 of 10 swamps sampled were likely to regain normal species composition and stem densities (Ewel et al. 1989).

Agricultural practices have evolved in similar ways to forestry in recent decades, with animal- and man-power giving way to more intensive and mechanized management practices driven by advancing technology (Gardner 2002). Common practices include draining, plowing, disking, harrowing, planting, terracing, or leveling of agricultural fields. Wetlands incorporated within agricultural fields are often physically disturbed by such processes, especially during drier periods (Bolen et al. 1989). In the Dougherty Plain, many such wetlands are dredged with the fill-dirt being used to construct “ramps” of higher elevation to support center-pivot irrigation towers. Physical changes may affect the hydrologic regime within a wetland by concentrating water in dredged areas, creating both permanently inundated and permanently dry microsites. Additionally, small or shallow wetlands may be completely filled-in to create additional land for agriculture or urban development, or excavated to create ponds for livestock, irrigation water, or recreational uses.

Water Quality Impairment

The effects of timber harvesting and associated silvicultural practices on water quality in streams and wetlands have been well described and a review of the current literature is available (Sun et al. 2004). The specific impacts of forestry operations depend upon the characteristics of the site and the management and intensity of the specific techniques employed. Considerably more research has been conducted in streams and riparian areas than in wetlands. Generally, intensive forest operations in riparian areas tend to raise stream temperatures and increase sediment and nutrient loads (Sun et al. 2004). The construction and erosion of forest roads has been implicated as the primary cause of increased sedimentation and turbidity associated with forestry practices (Sun et al. 2004). Industrial forest lands are commonly fertilized with nitrogen (N) and phosphorus (P) and such activities frequently result in elevated concentrations of N and P in receiving waters (Sun et al. 2004). However, the observed concentrations were usually one-tenth of those observed in agriculture and elevated nutrient conditions were relatively short lived (Sun et al. 2004). Current BMPs have successfully reduced or eliminated risks to water quality associated with forestry activities (Sun et al. 2004).

In watersheds or wetland basins dominated by agricultural or urban land uses, overland flow can be a significant source of non-point pollution to receiving waters. In agricultural areas, overland flow typically transports sediment, nutrients, pathogens, pesticides, metals, and salts (U.S. Environmental Protection Agency 2005; Jackson 2006). In urban areas overland flow typically transports all of these contaminants plus organics, hydrocarbons, and trash (U.S. Government Accounting Office 2001; Jackson 2006). It is difficult to isolate the effects that these individual constituents have on wetland systems. The precise concentrations of contaminants are determined by many factors and vary by location, but generally these

constituents have the potential to impair water quality, degrade aquatic ecosystems, and pose risks to human health (U.S. Government Accounting Office 2001; U.S. Environmental Protection Agency 2005; Jackson 2006).

Agricultural and urban runoff increase sedimentation rates and turbidity in receiving wetlands (Martin & Hartmann 1987; Dieter 1991; Gleason & Euliss 1998; Jackson 2006). Erosion rates are greatest within watersheds undergoing development, significantly lower in agricultural watersheds, and lowest in watersheds composed of undisturbed vegetation (Novotny & Olem 1994). Increased sedimentation reduces germination, emergence, and species richness of wetland macrophytes (Jurik et al. 1994; Gleason & Euliss 1998; Gleason et al. 2003), and increased turbidity leads to decreased vegetative production (Robel 1961). As a result, elevated levels of turbidity and sedimentation may lead to a reduction in primary productivity within wetlands. Additionally, sedimentation and subsequent burial have been shown to directly reduce total invertebrate emergence (Gleason et al. 2003). Thus, elevated sedimentation rates have the potential to alter food webs within aquatic systems and impair wetland functions related to water quality, nutrient cycling, and pollutant sequestration and transformation (Gleason & Euliss 1998). Further, extreme sedimentation may fill wetland basins and result in a loss of all wetland functions and ecosystem services (Gleason & Euliss 1998)

Nutrients, especially N and P, are of particular concern because of their ability to impair aquatic systems (Carpenter et al. 1998). In the U.S., eutrophication associated with N and P enrichment is the most common impairment of surface waters (U.S. Environmental Protection Agency Carpenter et al. 1998; 2000a). Agricultural and urban land-uses are the primary sources of non-point nutrient pollution (National Research Council 1992; Novotny & Olem 1994; Sharpley et al. 1994; U.S. Environmental Protection Agency 1996; Carpenter et al. 1998). In the

Dougherty Plain, significantly higher concentrations of $\text{PO}_4\text{-P}$ were detected in agriculturally disturbed depressional wetlands as opposed to those in reference condition (Battle et al. 2001). Nutrient enrichment and eutrophication of aquatic systems may produce deleterious effects: algal blooms, anoxic conditions, death of fish or other biota, altered floral and faunal communities including the invasion of exotic species, loss of biodiversity, human and animal health issues, and loss of usable water sources for agricultural, industrial, or other human uses (Carpenter et al. 1998). Additionally, eutrophication is associated with increasing larval populations of disease transmitting mosquitoes (Shaman et al. 2002; Pope et al. 2005; Kirkman et al. in press).

In irrigated agricultural areas, the rotation of center-pivot systems requires the periodic chemical or mechanical removal of woody vegetation in uncultivated areas of the field, such as wetlands. Additionally, irrigation systems allow for chemigation, or the direct application of agricultural chemicals (i.e., fertilizers, herbicides, pesticides, fungicides, and nematicides) through irrigation water (New & Fipps Date Unknown). Thus, embedded wetlands may receive direct nutrient and chemical inputs from irrigation systems passing above them.

Landscape Fragmentation

Currently, the longleaf pine ecosystem is ranked as the third most endangered ecosystem in the U.S. and occurs on approximately 3% of its historical range (Noss et al. 1995; Outcalt & Sheffield 1996; Van Lear et al. 2005). Overviews of this ecosystem and the factors contributing to this decline are available (Frost 1993; Landers et al. 1995; Engstrom et al. 2001; Van Lear et al. 2005). Conversion of natural forest land to other land use types has resulted in a fragmented landscape. Landscape fragmentation and fire suppression efforts associated with current land uses have reduced the fire conductivity of the landscape (Grumbine 1994; Leach &

Givnish 1996; Collins et al. 1998; Hiers et al. 2000). Consequently, the maintenance of fire-prone ecosystems now requires intensive management with prescribed fire that mimics presumed natural fire frequency.

Continued landscape conversion and fire suppression efforts threaten the survival of many floral and faunal species associated with the longleaf pine ecosystem (Frost 1993; Noss et al. 1995; Van Lear et al. 2005) and concomitantly encourage the invasion of exotic species (Engstrom 1993; Van Lear et al. 2005). The longleaf pine ecosystem supports one of the most floristically diverse communities in the U.S. (Peet & Allard 1993; Walker 1993; Van Lear et al. 2005). Currently, 27 plant species listed as federally threatened or endangered and 187 species considered as rare depend on the remaining tracts of longleaf pine habitat (Walker 1993; 1998; Van Lear et al. 2005). Additionally, this ecosystem provides habitat for diverse and unique communities of mammals, birds, and herpetofauna, many of which are of special conservation concern (Engstrom 1993; Van Lear et al. 2005). Much of this diversity is associated with isolated wetlands embedded within the ecosystem (Peet & Allard 1993; Walker 1993; Drew et al. 1998; Van Lear et al. 2005; Smith et al. 2006).

Ecotones, or zones of transition between marine, terrestrial, and freshwater habitats, are often particularly important components of the landscape (Kirkman et al. 1998b; Ewel et al. 2001). Ecotones physically link adjacent ecosystems and landscape processes by mediating the transfer of materials, nutrients, and energy. For example, the area of transition between isolated wetlands and fire-maintained longleaf pine uplands supports species rich floral and faunal communities and facilitates fire transfer between the respective systems (Kirkman et al. 1998b; Gibbons 2003). However, conversion of longleaf pine uplands to other land use types and active efforts to suppress fire across the landscape, specifically within wetlands, may result in structural

and compositional changes within ecotones and wetlands (Kirkman 1995). Fire suppression may enhance the ability of fire-intolerant species to encroach upon isolated wetlands and associated ecotones (Kirkman 1995). As ecotones shift toward hardwood dominance, their ability to conduct fire is reduced. Eventually, such areas may become barriers to fire, rather than conduits (Kirkman 1995).

One of the previously identified ecosystem services associated with isolated wetlands is the maintenance of metapopulations of unique biotic communities; including many endemic and rare species (De Steven & Toner 1997; Semlitsch & Bodie 1998; Kirkman et al. 1999; Snodgrass et al. 2000; Edwards & Weakley 2001; Sharitz 2003; De Steven & Toner 2004; Smith et al. 2006). However, faunal communities associated with isolated wetlands, particularly herpetofauna, are often equally dependent on the forested upland matrix surrounding the wetland habitat and on corridors of suitable habitat connecting isolated wetlands (Gibbons 1970; Gibbons & Bennett 1974; Morreale et al. 1984; Lovich 1990; Burke & Gibbons 1995; Semlitsch 1998; Gibbons 2003). As individual wetlands are converted to other land use types and the upland matrix becomes progressively more fragmented, isolated wetlands (and associated biotic communities) are increasingly cut off from landscape level processes such as fire and from one another. Thus, the ability of isolated wetlands to support and maintain metapopulations becomes increasingly impaired (MacArthur & Wilson 1967; Levins 1970; McCullough 1996; Semlitsch & Bodie 1998; Leibowitz 2003).

Assessing Wetland Condition

The previously described threats are commonly associated with the conversion of a wetland or surrounding area to a more intensive land use. However, because the threats often occur in concert it is difficult to observe the effect of any one threat in isolation. Floral and

faunal communities may be impacted by an individual threat or by the cumulative effects of many threats associated with a particular land use type. Thus, interpreting the effects of land use change on the overall condition or integrity of individual wetlands has been the subject of much research. Due to local and regional heterogeneity of wetlands, such studies are generally conducted by comparing characteristics or metrics of a wetland of interest to metrics observed in a similar reference or minimally disturbed wetland.

The goal of reference-based studies is to directly compare the biological integrity of the impacted system to the biological integrity of the reference system (Karr 1993). Generally, such studies seek to observe differences or changes in the diversity, abundance, or structure of a target species or assemblage as compared to values recorded in reference conditions. Various assemblages have been used to assess wetland integrity; including vegetation (Lopez & Fennessy 2002; Smith & Haukos 2002; Miller & Wardrop 2006; Mack 2007), macroinvertebrates (Euliss & Mushet 1999; Battle et al. 2001), wildlife (Fairbairn & Dinsmore 2001), and diatoms (Lane & Brown 2007). Other research has compared physical parameters, such as soil characteristics or water quality (Battle et al. 2001; Whigham & Jordan 2003; Jordan et al. 2007).

The United States Environmental Protection Agency has outlined a framework for wetland monitoring and assessment with three levels that vary in scale and intensity (Fennessy et al. 2004). The three assessment levels are: Landscape Scale Assessment (Level 1), Rapid Field Methods (Level 2), and Intensive Biological and Physico-Chemical Measures (Level 3) (Fennessy et al. 2004). This assessment procedure has been successfully employed in some regions (Mack 2006; Reiss & Brown 2007), but the accurate evaluation of specific wetland types requires development and calibration of wetland or region specific assessment procedures at each of the three levels.

Level 1 or landscape assessments typically integrate available GIS data to remotely quantify the landscape context of an individual wetland. Research has shown that surrounding land use or land cover affects physical, chemical, and biological components of aquatic systems (Johnes et al. 1996; Zacharias et al. 2004; Houlahan et al. 2006). A recently developed Level 1 procedure, the Landscape Development Intensity (LDI) index, uses land cover maps to calculate the intensity of human activity surrounding an area of interest (Brown & Vivas 2005). It ranks land cover classes according to the amount of non-renewable energy necessary for their maintenance and assigns a LDI coefficient to each land cover class (Odum 1996; Brown & Vivas 2005). An area weighted mean is calculated from LDI coefficients in the area immediately surrounding a wetland and this value represents the wetland's LDI index score. LDI scores can be developed with preexisting land cover maps or with land cover digitized from aerial photographs (Reiss & Brown 2007). This technique was developed and calibrated in Florida but has been applied elsewhere (Mack 2006; Vivas & Brown 2006; Stein et al. 2009).

Level 2 or rapid assessments include quantitative or qualitative field based measurements of wetland condition that are simple, rapid, and easy to apply (U.S. Environmental Protection Agency 2002; Fennessy et al. 2004). These methods typically rely on best professional judgment to evaluate readily observable wetland features or stressors that are sensitive to human impacts on wetland systems (e.g., Van Dam et al. 1998; Mack et al. 2000). Fennessy et al. (2004) evaluated 16 rapid assessment procedures to identify the pros and cons of each method and highlight the most common and useful indicators of wetland condition. Most rapid assessments evaluate properties of wetland hydrology, soils, vegetation, and landscape setting (Fennessy et al. 2004).

Level 3 or intensive assessments involve the field collection of detailed and quantified information on physical, chemical, or biological wetland parameters (U.S. Environmental Protection Agency 2002). These methods include characteristics such as sedimentation rates, water quality, or biotic assemblages (e.g., vegetation or macroinvertebrates). Level 3 assessments often seek to quantify overall community condition by employing multi-metric indices, such as an index of biotic integrity. These indices aggregate individual metrics representing biological characteristics that respond predictably to a human disturbance gradient (Karr & Chu 1999).

Increasing interest in assessing and monitoring wetland condition has led to the development of several methods targeting each of the three levels described above (e.g., Van Dam et al. 1998; U.S. Environmental Protection Agency 2002; Fennessy et al. 2004; Brown & Vivas 2005). Established methods are generally constructed for specific wetland types or regions and transferability is often limited. In the Dougherty Plain physiographic district of southwest Georgia, ongoing research on isolated wetlands necessitates a multi-scale, flexible, and standardized assessment protocol. However, no existing methods have been constructed, calibrated, or evaluated for these systems.

OBJECTIVES

The overall objective of this study is to determine the prevalence and condition of isolated wetlands within the Dougherty Plain physiographic district of southwestern Georgia, USA. The second chapter characterizes the historical context of land use in the region by quantifying land use and land cover (LULC) for a 10% sample of the total area of Dougherty Plain in each of 4 study years (1948, 1968, 1993 and 2007). The third chapter combines readily accessible Geographic Information System (GIS) data to estimate the number, extent, and spatial

configuration of isolated wetlands in the Dougherty Plain. The fourth chapter develops a multi-metric wetland condition assessment framework that incorporates qualitative and quantitative measurements at multiple spatial scales. Finally, the fifth chapter presents a summary of the conclusions and implications of each chapter and suggests future directions of research.

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CHAPTER 2

SIX DECADES (1948-2007) OF LANDSCAPE CHANGE IN THE DOUGHERTY PLAIN OF SOUTHWEST GEORGIA, USA¹

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ABSTRACT

Patterns of land use and land cover (LULC) are driven by biophysical, socioeconomic, and technological variables, and affect landscape structure, functions, and processes. To quantify temporal changes in LULC patterns within a 10% sample of the Dougherty Plain physiographic district in southwestern Georgia, we constructed a spatially-explicit historical timeline of LULC using aerial photographs from 1948, 1968, 1993, and 2007. Our results show major declines of natural forest and unirrigated agriculture, and increases in the area of planted and irrigated agriculture. Generally, the landscape became more homogeneous, with more, smaller, and increasingly uniform patches. Attempts to corroborate our estimates with those of other sources were confounded by differences in the definition of LULC classes. Additionally, LULC were quite variable with the Dougherty Plain; indicating that a more local scale analysis may be appropriate. The number and area of isolated wetlands were strongly correlated with climatic conditions, and thus not suitable for tracking changes over time.

Key Words: Dougherty Plain, Georgia, isolated wetlands, land cover, land use

INTRODUCTION

Spatial and temporal patterns of land use and land cover (LULC) result from interactions between biophysical, socioeconomic, and technological drivers (Domon & Bouchard 2007). Anthropogenic alteration of LULC increased rapidly during the 20th century, and 39-50% of the earth's land surface has been transformed by human activity (Vitousek et al. 1997). The resulting pattern of LULC affects landscape structure, functions, and processes. Locally, LULC affects ecological phenomena, such as erosion rates and water quality (Bolstad & Swank 1997; Gergel et al. 2002; Houlihan & Findlay 2004), the distribution and movement of floral and faunal populations (Harding et al. 1998; Galatowitsch et al. 2000; Piha et al. 2007; Attum et al. 2008), and the spread of disturbance (Turner 1987; Frost 1993). Globally, LULC transformation is the main cause of biological diversity loss and is a primary driver of climate change (Vitousek et al. 1997).

While much research has focused on the influence of current LULC patterns on physical, chemical, and biological properties of landscapes, evidence suggests that historical LULC patterns have legacies that also shape these properties (Eberhardt et al. 2003; Foster et al. 2003). For example, fish and invertebrate community composition in southern Appalachian streams were best predicted with LULC data from the 1950's, while current LULC was a comparatively poor indicator (Harding et al. 1998). Similarly, Käyhkö and Skånes (2008) identified LULC trajectories as primary drivers of the local abundance and distribution patterns of oak trees in southwestern Finland. Thus, an understanding of both current and historic LULC is a valuable starting point for research and/or management activities focusing on the conservation or restoration of specific landscapes or ecosystems, or the services they provide (Eberhardt et al. 2003; Foster et al. 2003; Domon & Bouchard 2007; Käyhkö & Skånes 2008).

Measurements of LULC pattern typically fall into one of two categories (Turner 1989; Danielson et al. 1992): composition (e.g., the presence, absence, or proportion of a specific LULC class or patch type) or configuration (e.g., the spatial positioning and physical characteristics of individual LULC patches). A patch is a contiguous area of a single LULC class surrounded by different LULC classes. A landscape consists of all of the patches and LULC classes occurring within a defined area. Standard metrics have been developed to quantify aspects of LULC composition and configuration at both the patch and landscape scale (Turner 1989; McGarigal & Marks 1995). These metrics allow researchers to quantify variability within and among sites, describe changes over time, and explain ecological processes and phenomena (Turner & Rushcer 1988; Griffith et al. 2003; Moreno-Mateos et al. 2008; Ruiz & Domon 2009).

In this study, we examine changes in LULC composition and configuration occurring between 1948 and 2007 in the Dougherty Plain physiographic district of southwestern Georgia. Additionally, we track changes in the number and total area of isolated wetlands in the Dougherty Plain. The objective of this study was to determine how landscape pattern has changed over time and relate these changes to prevailing biophysical, socioeconomic, and technological drivers, and to quantify the loss of isolated wetlands in the Dougherty Plain.

METHODS

Study Area

The Dougherty Plain is a karstic physiographic district covering 6,690 km² and parts of 14 counties in southwest Georgia (Figure 1.1). It borders the Pelham Escarpment to the south and east and the Fall-line Hills and Chattahoochee River to the north and west. It has low topographic relief and ranges in elevation from 93 m in the northeast to 24 m in the southeast

(Clark & Zisa 1976). The region is characterized by relatively broad, flat uplands, numerous isolated wetlands, and few, but deeply incised streams. The Dougherty Plain is located within the historic range of the longleaf pine (*Pinus palustris*) and wiregrass (*Aristida stricta*) ecosystem, which once dominated the southeastern Coastal Plain and extended into other physiographic provinces (Christensen 1981; Frost 1993; Ware et al. 1993). Currently, the longleaf pine ecosystem ranks among the most endangered ecosystems in the U.S. and occurs on approximately 3% of its historical range (Noss et al. 1995; Outcalt & Sheffield 1996; Van Lear et al. 2005). The Dougherty Plain is one of Georgia's most important agricultural areas, and agricultural irrigation is widespread (Li & Jackson 2005; Harrison 2008), as are natural forests and pine timber plantations.

Data Collection and Preparation

We manually digitized LULC data and isolated wetland polygons within 39 randomly selected sample areas ranging from 13.13 km² to 20.10 km² and covering a combined 675 km² or 10% of the total area of the Dougherty Plain. We digitized data from aerial photographs acquired in 1948, 1968, 1993, and 2007. We downloaded photographs for 1948 and 1968 from the Digital Library of Georgia (University System of Georgia 2010), or else manually scanned them from paper photographs at the University of Georgia Map Library. We geo-rectified photographs in ArcMAP 9.3 (Environmental Systems Research Institute, Redlands, CA, USA), and enforced a maximum root mean square error of 5. For the 1993 and 2007 sample years, we acquired digital aerial imagery from the National Agriculture Imagery Program. For each sample area and year combination, we screen digitized LULC into 1 of 11 potential classes (Table 2.1) at a scale of 1:10,000. The minimum sampling unit was 0.001 ha. A single individual conducted all digitizing to minimize operator error.

Analysis

We used Patch Analyst (Elkie et al. 1999) to calculate six metrics of LULC pattern (Table 2.2). When appropriate we calculated metrics at the landscape level (considering every patch in the landscape) and at the patch level (considering patches of each LULC class individually). We calculated all metrics for each of the 39 sample areas. Preliminary data analysis revealed non-normality and heteroscedasticity within the data; indicating that the population mean is an inappropriate measure of central tendency. Thus we use median values as a measure of central tendency (Snedecor & Cochran 1989). We used a non-parametric statistical procedure (Friedman's repeated measures analysis of variance on ranks) to test the null hypothesis that, for each metric, there would be no difference in mean ranks between each sample year (Conover 1999; Systat Software 2008). We included sample areas as blocking factors to control for geographic variation. Where appropriate, we used *post hoc* Tukey's pairwise comparisons, with family-wise error rates controlled at 95%, to isolate significant differences between sample years (Systat Software 2008). We conducted all analyses using SigmaPlot 11 (Systat Software Inc., San Jose, CA, USA).

Next, we conducted a bootstrap resampling technique with 2,000 iterations to estimate means and 95% confidence intervals (95% CI) for the proportion of each LULC class in each of our sample years (Wang 2001). To estimate the area occupied by each LULC class in each time step for the entire Dougherty Plain, we multiplied the bootstrapped means and 95 % CIs by 6,689.52 km² (the area of the Dougherty Plain). Next we compared these estimates with those of other sources. Estimates of the area of harvested cropland, cropland under irrigation, and land in orchards were available by county for various years (United States Census Bureau 1950-1992; United States Department of Agriculture 2007). Additionally, data on water and urban areas

were available for 1991 (Natural Resources Spatial Analysis Laboratory 2007a) and 2005 (Natural Resources Spatial Analysis Laboratory 2007b). To qualitatively assess the local differences within the Dougherty Plain we examined the proportions of 6 major LULC classes forest, herbaceous, planted pine, unirrigated agriculture, irrigated agriculture, and developed areas) for each individual sample area and sample period.

We conducted a preliminary analysis of the isolated wetland number and total area data to determine the reliability of identification of wetlands from aerial photointerpretation between sample periods. We used Spearman's rank correlation (SigmaPlot 11, Systat Software Inc., San Jose, CA, USA) to assess the relationship between the number and total area of isolated wetlands observed in each sample year and the Palmer Drought Severity Index (PDSI). The PDSI uses temperature and rainfall data to determine relative dryness, and PDSI values for southwestern Georgia were obtained from the National Climate Data Center (Accessed September 2009). Spearman's rank correlation coefficient (ρ) is a non-parametric measure of statistical dependence between two variables. ρ ranges between -1 and +1. A value of -1 indicates a perfect negative Spearman correlation between variables. A value of +1 indicates a perfect positive Spearman correlation between the variables. A value of 0 indicates no Spearman correlation between the variables. Both the number and total area of isolated wetlands displayed a strong, positive correlation with the PDSI ($\rho > 0.90$, $p < 0.001$). Based on this assessment, we concluded that the delineation of wetland polygons reflected relatively short-term climatic conditions, and did not adequately represent stable wetlands existing through time; thus, this land cover type was deleted from further analyses.

RESULTS

The proportion of the landscape in each LULC class varied throughout the study period (Table 2.3, Figure 2.2). The proportion of the Dougherty Plain in developed, road, planted pine, and irrigated agriculture classes increased between 1948 and 2007 ($p < 0.05$), while declines occurred in forest, unirrigated agriculture, and water ($p < 0.05$). In contrast, the orchard and herbaceous classes remained proportionally stable. The most prevalent changes were the steep declines in unirrigated agriculture and forest, and the rapid increase of planted pine and irrigated agriculture.

Change in fragmentation and uniformity in LULC was one of the most notable patterns that emerged during the 60 year study period occurred with more, smaller, and increasingly uniform sizes of patches present over time (Table 2.3, Figure 2.2). The overall trends toward increased patch density were reflected in most of the individual LULC classes. Increasing patch density occurred at the same time that mean patch size decreased, indicating many more, smaller patches, and appears to be driven primarily by declines in mean patch size of the forest and unirrigated agriculture classes.

We observed increases in forest patch density between 1948 and 2007, and 1993 and 2007 ($p < 0.05$) (Table 2.3, Figure 2.2), while the mean size of forest patches decreased over the same time periods ($p < 0.05$). The patch density of unirrigated agriculture remained stable among sample years, but mean patch size decreased between all sample years ($p < 0.05$), except between 1948 and 1968. The stability of unirrigated agriculture patch density is somewhat counterintuitive, given the decrease in the proportion of the landscape it covers, but it is attributable to the current placement of unirrigated agriculture fields in the dry corners of irrigated fields and the increasing use of wildlife foodplots.

Following initial detection in 1993, the patch density of irrigated agriculture increased between 1993 and 2007 ($p < 0.05$), while mean patch size remained stable. Planted pine patch density increased between all sample years ($p < 0.05$), and mean patch size increased between 1948 and 1968 ($p < 0.05$) and then remained relatively stable through 2007 (Table 2.3, Figure 2.2). The patch densities of the herbaceous and developed classes were significantly higher in 2007, as compared to every other sample year ($p < 0.05$). The herbaceous mean patch size trended lower, and decreased between 1948 and 2007 ($p < 0.05$), while developed patches increased between 1948 and 1993, 1948-2007 and 1968-2007 ($p < 0.05$). Open water patch density fluctuated between sample years and was lower in 1968 and 2007, as compared to 1948 and 1993 ($p < 0.05$), but mean patch size remained stable. The patch density of orchards and roads were stable among sample years. The mean patch size of orchards remained stable, but mean patch size of roads increased after 1948 ($p < 0.05$).

We calculated Shannon's diversity index as a measure of relative LULC diversity. It increases as the number of LULC classes increases or as their proportional distribution becomes more even (McGarigal & Marks 1995; Elkie et al. 1999). This index increased from 1948 to 1993 but remained stable from 1993 to 2007 (Figure 2.3). This pattern reflects the loss of area in formerly dominant LULC classes (i.e., forest and un-irrigated agriculture), the addition of novel LULC types (i.e., irrigated agriculture and unknown), and increasing evenness in the proportional distribution of LULC classes. Similarly, Shannon's evenness index measures the distribution and abundance of LULC classes and approaches 1 as the distribution of class types becomes more even (Figure 2.3) (McGarigal & Marks 1995; Elkie et al. 1999). The evenness index increased from 1948 to 1993 and then dropped slightly between 1993 and 2007. The

decrease between 1993 and 2007 reflects the continued growth in planted pine and irrigated agriculture at the expense of other classes.

The bootstrap estimated means and 95% CIs estimate the area covered by each LULC class during each sample year for the entire Dougherty Plain (Table 2.4). When comparable sources of land cover data were available, we estimated the area of various LULC classes within the Dougherty Plain and compared these estimates to the bootstrapped means and 95 CIs (Figure 2.4). Of 10 comparisons made, only 5 showed agreement between the bootstrapped estimated 95% CIs and the comparable data sources.

The qualitative-visual assessment of LULC proportions across the 39 sample areas revealed a large degree of variation across the Dougherty Plain (Appendix A). Locating individual sample areas on the map provided in Appendix A allows for a visual-qualitative assessment of LULC trends across the Dougherty Plain. The forest LULC class generally decreased throughout the Dougherty Plain, but the decrease was more pronounced in the southern portion, particularly in Seminole, Decatur, and Miller Counties. Herbaceous areas also decreased throughout the Dougherty Plain, but the steepest declines were observed in the northern counties of Dougherty, Worth, and Lee. Planted pine increased throughout the Dougherty Plain, but the increase was highly variable among sample areas and no geographic pattern was observed. Unirrigated agriculture decreased sharply in all areas of the Dougherty Plain. Decreases were sharpest in the southern half of the Dougherty Plain, because this area contained the greatest amount of unirrigated agriculture in 1948. Irrigated agriculture generally increased throughout the Dougherty Plain, but the increase was less pronounced in the northern counties of Calhoun, Dougherty, Worth, and Lee. Developed areas were minimal throughout the

Dougherty Plain, but were most concentrated in Decatur and Dougherty counties; likely reflecting urban expansion around the cities of Bainbridge and Albany.

DISCUSSION

This research documents dramatic changes in the area of natural forests, planted pine forests, unirrigated agriculture and irrigated agriculture in the Dougherty Plain during a 60 year period. Although numerous state-wide studies also report spatial and temporal LULC patterns (Bederman 1960; 1970; Brasswell 1972; Knight 1973; Turner & Rushcer 1988; Brown 2002), differences in sample dates, the extent and location of sample areas, variation in LULC trends across the state, technical differences in the definition of LULC classes, and the statistical methods used make direct comparison somewhat imprecise.

It is clear that the total area of row crop agriculture in Georgia declined steadily throughout most of the 20th century after peaking between 1910 and 1920 (Turner & Rushcer 1988; Brown 2002) for much of the state; however, the Coastal Plain (including the Dougherty Plain) saw minimal change relative to other physiographic regions (Turner & Rushcer 1988; Brown 2002). Turner and Rushcer (1988) found that agricultural areas declined in the Piedmont and Mountain physiographic regions of Georgia between the 1930's and 1980's, but increased within the Coastal Plain. We found that total agricultural area (unirrigated and irrigated) remained relatively stable, even though irrigated agriculture steadily increased after 1968.

Georgia's forests, both natural and planted, increased by 18% between the mid-1930's and 1997 (Brown 2002). This increase is tied to the statewide decline in agriculture during the same time period. Throughout the state, abandoned agricultural lands succeeded into coniferous forest (Turner & Rushcer 1988; Brown 2002; Rauscher 2004). The replanting of Georgia's forests, primarily into southern pines, began in earnest during the mid-1940's or 1950's and

continued throughout the rest of the century (Brown 2002; Rauscher 2004). Turner and Ruscher (1988) reported that while the area of deciduous forest remained stable or declined throughout Georgia, the area of coniferous forest increased in all physiographic regions of the state between the 1930's and 1980's. In the Dougherty Plain, natural forests declined between 1948 and 2007, but the addition of planted pine led to an increase in total forested lands (forest and planted pine). Although Turner and Ruscher (1988) did not distinguish between natural and planted coniferous forests, their results concur with our findings and indicate that trends in forestland composition observed in the Dougherty Plain may be similar to trends in other physiographic regions of the state.

The landscape metrics of patch density, mean patch size, patch size standard deviation, Shannon's diversity index, and Shannon's evenness index are more difficult to compare with other sources, because they are particularly sensitive to spatial scale and categorical resolution (Buyantuyev & Wu 2007). Similar trends toward landscape fragmentation have been reported for the southeast (Griffith et al. 2003) and other regions of the U.S. (Reed et al. 1996; Hawbaker et al. 2006). Our findings contrast with those of Turner and Ruscher (1988), who report decreasing fragmentation in the Georgia landscape between the 1930's and the 1980's. The discrepancy is probably due to differences in categorical resolution; our study used 11 LULC classes while they used 8.

Comparing our bootstrap estimated means and 95% confidence intervals with other available data returned mixed results. Of 10 comparisons made, only 5 were within the 95% CIs for the designated LULC class and year. Lack of comparability is largely methodological, and not surprising. Differences in data collection techniques, the definition of LULC categories, and dates of the studies confound the comparisons. For example, the census data was acquired

through stratified random mail-out/mail-back questionnaires, and the technical definition of the LULC categories differs from ours (United States Census Bureau 1950-1992; United States Department of Agriculture 2007). Also, the definitions of individual categories reported in the census have changed over time, and in some cases are not directly comparable to the same category reported in a previous year's census. Comparisons were further hindered by converting county level Census data into an estimate of LULC within the Dougherty Plain.

The open water and urban area data from 1991 and 2005 (Natural Resources Spatial Analysis Laboratory 2007a; b) are based on LANDSAT data and are probably more comparable to our data even though the years assessed are not congruent with those in our study. Differences in LULC patterns of change across the Dougherty Plain revealed the importance of local geographic drivers of LULC change. This variation suggests analyses to link local LULC trends to local geographic variables, such as topography or geologic stratigraphy.

Drivers of Change

LULC patterns over space and time are related to biophysical, socioeconomic, and technological factors (Domon & Bouchard 2007), and identifying the drivers of LULC change is a complicated task (Lambin et al. 2001). This study does not attempt to quantify the influence of specific factors on the LULC changes that occurred in the Dougherty Plain. However, we do briefly highlight major technological and socioeconomic trends, which likely affected the changes we observed.

The agricultural and forest industries benefited from continual increases in mechanization over the sample time period (Gardner 2002; Izlar 2006). Horses and mules provided the majority of farm traction power until the mid-1940's (Gardner 2002). Thereafter, the use of tractors and related equipment increased and eventually replaced both animal and human sources of labor

(Gardner 2002). Tools such as selective breeding, fertilizers, and pesticides were widely adopted during the study period and their use increased rapidly (Gardner 2002). The market and demand for southern pines grew rapidly as they were increasingly used in forest products, e.g., kraft paper, newsprint, plywood, and oriented strand board. These technological advances allowed for the intensification of the agricultural and forest industries during the 20th century. However, the most visible and direct impact of technological innovation was the introduction, and rapid adoption, of center-pivot irrigation during the 1970s, which increased crop yields and provided a form of insurance in times of drought (Harrison 2008).

Prevailing socio-economic conditions drive, and are driven by, technological innovation. Many of the aforementioned technological advances benefited from major public and private investment in the forestry and agricultural industries (Rauscher & Johnsen 2004; Siry 2004; Izlar 2006). Additionally, the federal government has a long and complex history of influencing, directly and indirectly, these industries through price supports, subsidies, acreage quotas, and import/export legislation. Perhaps the most noteworthy government program affecting LULC change during the sample period is the Conservation Reserve Program. The first Conservation Reserve Program was created as part of the Soil Bank Act of 1956 and was designed to encourage the diversion of cropland into other uses, e.g., pine plantations, through direct payments to the landowner (Brown 2002). Since that time, similar programs have been in place, albeit with different names. The Food Security Act of 1985 created the Conservation Reserve Program of today. The effects of the Conservation Reserve Program(s) are visible as notable spikes in the acreage of planted forest lands throughout Georgia in the late 1950's to early 1960's and throughout the 1980's (Brown 2002). We suggest that this policy may have

contributed to the decline in unirrigated agriculture and the increase in planted pine observed in this study.

Confounding Variables

Creating LULC data from aerial imagery products involves inherent limitations and pitfalls, particularly when historical imagery is involved. Geo-rectification proved challenging for photographs from the 1948 and 1968, especially in areas that subsequently underwent substantial landscape changes. Small errors in the geo-rectification process can lead to large discrepancies in LULC between years, so minimizing the root mean square errors proved crucial to our success. Additionally, we observed large variations in the quality of imagery both between and within sample years. The data presented here represent a single individual's interpretation of aerial imagery, and while we hope this minimized operator error in the interpretation and digitization process, we recognize the potential for increased operator bias in certain situations. Finally, our LULC classification scheme included only 11 classes, which represents a simplified model of reality but is appropriate for the specific objectives and constraints of this research.

CONCLUSIONS

An important contribution of this study is that it provides the foundation for development of predictive models of current land use, trajectories of landscape change, and ecological condition of wetlands when coupled with intensive ecological assessment. Changing LULC patterns have implications for ecological processes occurring throughout the Dougherty Plain landscape and are likely drivers of the distribution and abundance of wildlife of concern, such as the Northern Bobwhite, gopher tortoises, Bachman's sparrow, amphibians (Twedt et al. 2007).

Similarly, temporal changes in LULC patterns drive sedimentation and nutrient accumulation rates in wetlands throughout the Dougherty Plain (Craft & Casey 2000).

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Table 2.1. Name and description of land use and land cover classes.

Name	Description
Bare	At least 80% of the surface lacking observable cover. Exposed soil, sand, rock, etc. Excluding bare soil associated with agriculture.
Unknown	Areas obscured by cloud cover.
Developed	All residential, agricultural, commercial, and industrial buildings.
Forest	All naturally occurring forestlands, and planted forestlands of undetectable successional stage.
Herbaceous	Herbaceous vegetation, including non-planted pasturelands.
Irrigated Agriculture	Irrigated, row-crop agricultural fields. Includes irrigated, planted pasturelands.
Orchard	Tree crops (fruits and nuts).
Planted Pine	Planted pine plantations.
Road	Automobile transportation routes. Includes all mediums/substrates (sand, gravel, concrete, asphalt, etc.).
Unirrigated Agriculture	Non-irrigated, row-crop agricultural fields. Includes non-irrigated, planted pasturelands.
Water	Areas of standing water with at least 80% of the surface lacking other observable cover.

Table 2.2. Description of land use and land cover metrics. PR, PD, MPS, and PSSD were calculated separately for each of 39 sample areas. SDI and SEI were calculated for the entire sample landscape.

Metric	Description
PR	Proportion - The proportion of each sample area represented by each LULC class.
PD	Patch Density - The total number of patches divided by the area (km ²) of the sample area.
MPS	Mean Patch Size - the mean patch size in hectares of given LULC classes and sample area.
PSSD	Patch Size Standard Deviation - The standard deviation among patches of given LULC classes and sample areas.
SDI	Shannon's Diversity Index - A measure of the number and proportional distribution of LULC classes.
SEI	Shannon's Evenness Index - A measure of the distribution and abundance of LULC classes.

Table 2.3. Median values of land use and land cover pattern metrics for the total sample landscape (Landscape) and individual LULC classes in the Dougherty Plain, Georgia, USA. Median values are reported because of non-normality and heteroscedasticity. Values are rounded to the nearest tenth. Arrows indicate direction of significant ($p < 0.05$) change in mean ranks.

LULC	1948	1968	1993	2007	Overall
Proportion of Sample Landscape					
Landscape	N/A	N/A	N/A	N/A	
Developed	0.3	0.3	0.4	0.7	↑
Forest	39.3	38.9 ↓	29.6	29.2	↓
Herbaceous	14.9	15.0	12.1	13.2	
Irrigated Ag	0.00	0.00 ↑	6.7 ↑	17.2	↑
Orchard	0.2	0.0	0.0	0.0	
Planted Pine	0.00 ↑	3.3 ↑	8.4	12.8	↑
Road	0.9 ↑	1.5	1.5	1.5	↑
Unirrigated Ag	26.3	26.8 ↓	15.6 ↓	8.1	↓
Water	1.7 ↓	0.1 ↑	0.7 ↓	0.2	↓
Patch Density per km ²					
Landscape	7.3	7.6 ↑	8.4 ↑	9.8	↑
Developed	0.9	1.1	1.2 ↑	1.4	↑
Forest	1.5	1.6	1.6 ↑	1.8	↑
Herbaceous	2.2	2.0	2.1 ↑	2.6	↑
Irrigated Ag	0.0	0.0 ↑	0.2 ↑	0.4	↑
Orchard	0.1	0.1	0.1	0.1	
Planted Pine	0.0 ↑	0.4	0.6 ↑	1.2	↑
Road	0.1	0.1	0.1	0.1	
Unirrigated Ag	1.2	1.4	1.4	1.2	
Water	1.1	0.1 ↑	0.5 ↓	0.2	↓

Table 2.3. Continued.

LULC	1948	1968	1993	2007	Overall
Mean Patch Size (ha)					
Landscape	13.7	13.2 ↓	12.0 ↓	10.2	↓
Developed	0.3	0.4	0.5	0.6	↑
Forest	28.6	22.1	20.2	17.0	↓
Herbaceous	7.9	6.5	5.6	5.1	↓
Irrigated Ag	0.0	0.0 ↑	32.6	30.7	↑
Orchard	2.3	0.7	1.2	0.8	
Planted Pine	0.0 ↑	7.5	14.2	11.6	↑
Road	8.0 ↑	13.9	13.8	13.3	↑
Unirrigated Ag	23.4	21.5 ↓	12.9 ↓	6.3	↓
Water	1.9	0.5	1.2	0.9	
Patch Size Standard Deviation (ha)					
Landscape	43.2	39.0 ↓	32.9	31.9	↓
Developed	0.2	0.3 ↑	0.7	0.7	↑
Forest	72.9	62.2	41.5	36.1	↓
Herbaceous	16.0	10.5	10.0	7.6	
Irrigated Ag	0.0	0.0 ↑	19.3	23.8	↑
Orchard	0.0	0.0	0.0	0.0	
Planted Pine	0.0 ↑	6.8	14.5	14.8	↑
Road	3.8 ↑	5.5	3.0	5.9	
Unirrigated Ag	28.1	24.2	15.7 ↓	7.3	↓
Water	2.4 ↓	0.0 ↑	1.1 ↓	0.4	↓

Table 2.4. Estimate of mean area (km²) and 95% confidence intervals around these means for each land use and land cover class throughout the entire Dougherty Plain in each sample year. Estimates were generated through bootstrap analysis with 2,000 iterations.

LULC Class	1948		1968		1993		2007	
	Mean	95% CI	Mean	95% CI	Mean	95% CI	Mean	95% CI
Developed	23	17 - 30	38	27 - 52	123	64 - 200	183	88 - 315
Forest	2,787	2,384 - 3,202	2,767	2,381 - 3,160	2,130	1,785 - 2,491	2,050	1,712 - 2,396
Herbaceous	1,202	971 - 1,440	1,110	902 - 1,334	924	771 - 1,090	1,013	832 - 1,203
Irrigated Agriculture	0		0		764	548 - 1,010	1,215	917 - 1,509
Orchard	300	119 - 536	257	88 - 476	317	119 - 571	271	95 - 498
Planted Pine	20	8 - 35	398	245 - 578	892	627 - 1,167	1,126	867 - 1,418
Road	76	61 - 94	120	102 - 141	114	96 - 137	115	96 - 136
Unirrigated Agriculture	2,048	1,757 - 2,366	1,916	1,622 - 2,216	1,263	1,010 - 1,492	613	480 - 768
Water	241	164 - 334	85	23 - 168	150	82 - 243	75	35 - 130

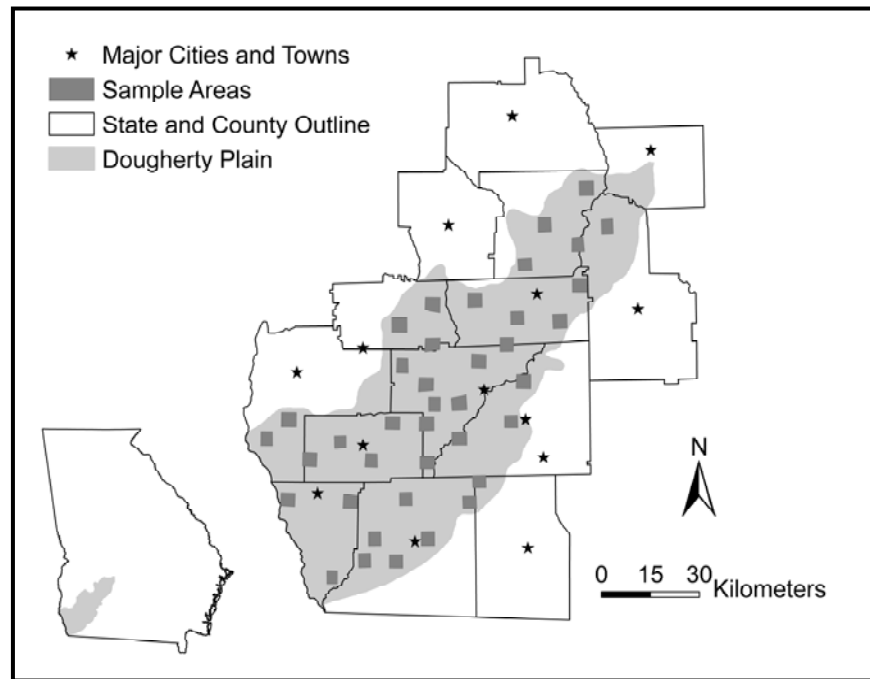


Figure 2.1. Map of study area depicting the Dougherty Plain physiographic district, county boundaries, major cities and towns, and 39 sample areas.

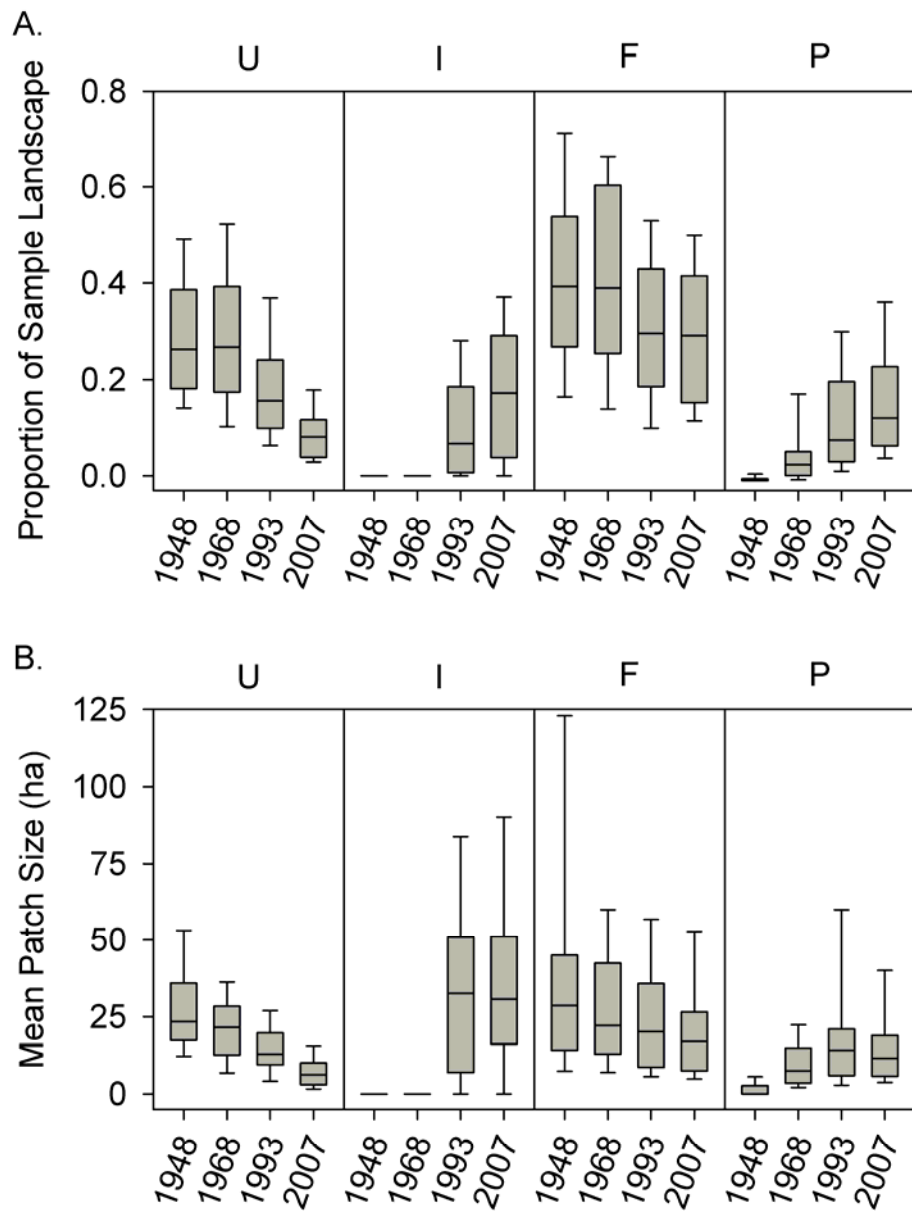


Figure 2.2. Box plots depict median and interquartile range, and error bars depict the 10th and 90th percentiles of proportion (A), mean patch size (B), patch density (C), and patch size standard deviation (D) for four LULC classes: U = Unirrigated Agriculture, I = Irrigated Agriculture, F = Natural Forest, and P = Planted Pine. Median and percentile values are reported due to non-normality and heteroscedasticity present in the data.

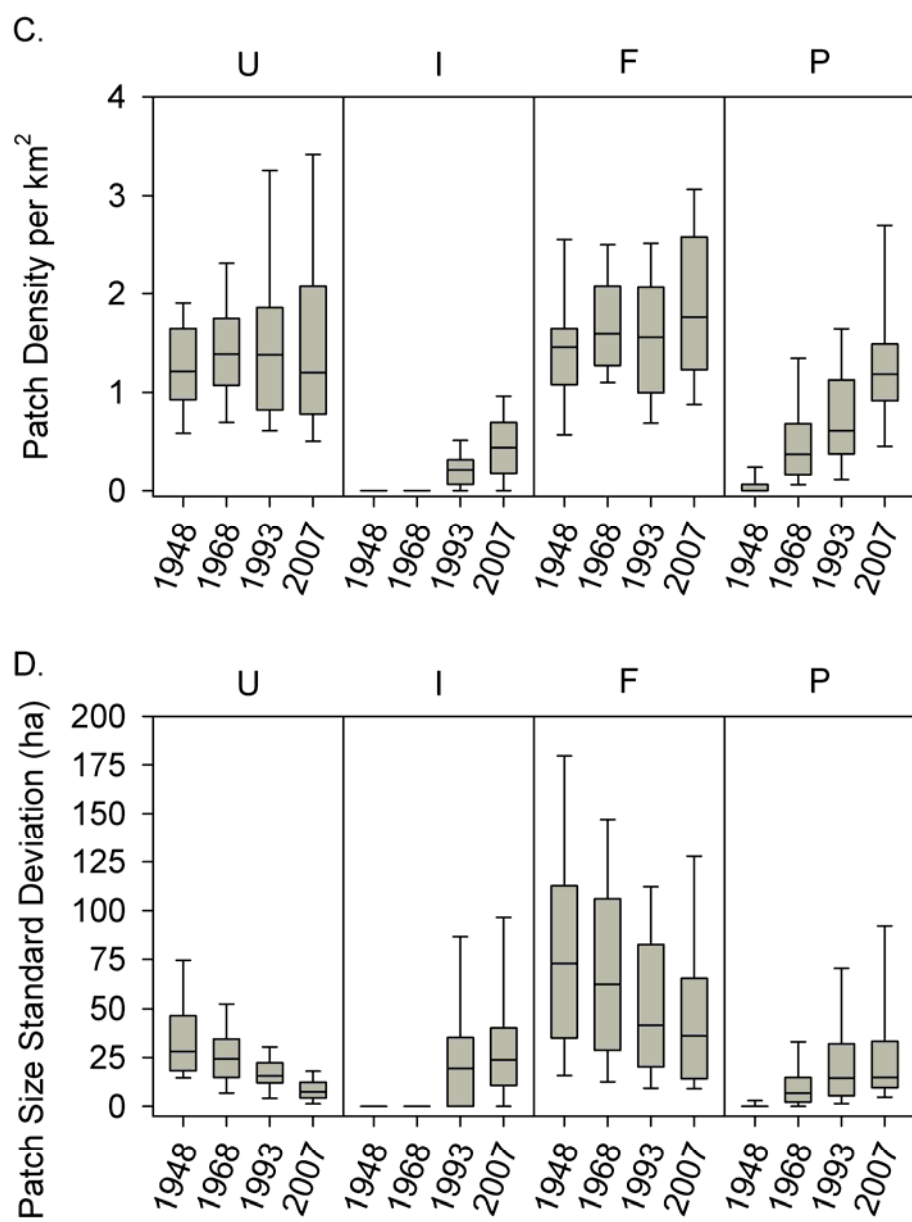


Figure 2.2. (Continued).

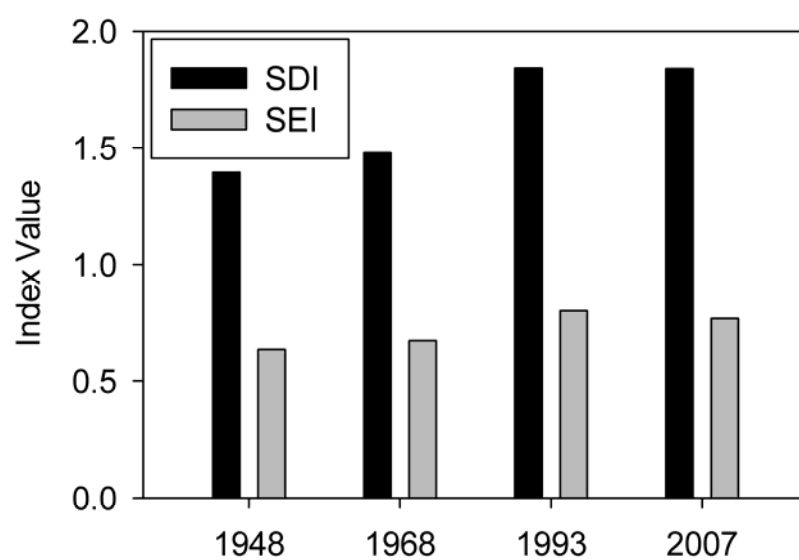


Figure 2.3. Shannon's diversity index (SDI) and Shannon's evenness index (SEI) scores for land use/land cover of the Dougherty Plain, Georgia (1948-2007).

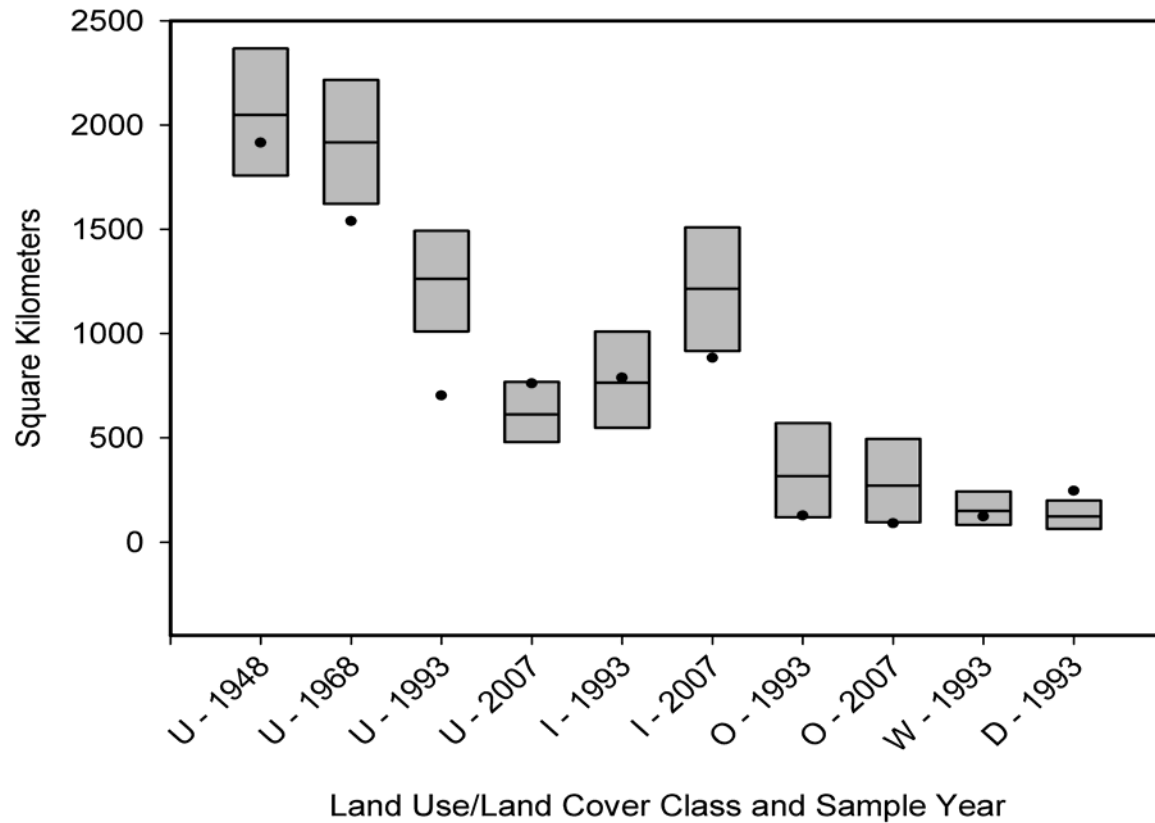
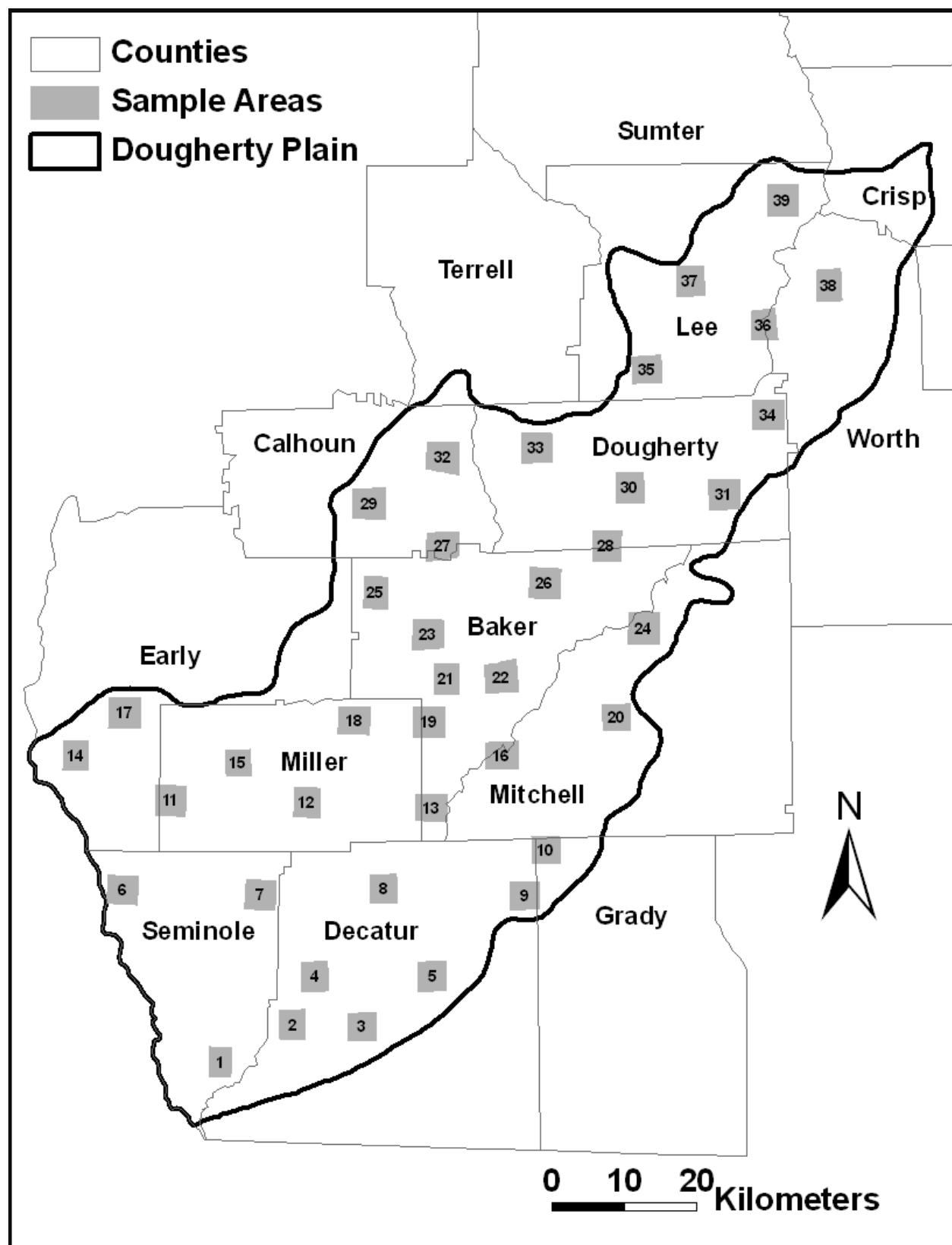
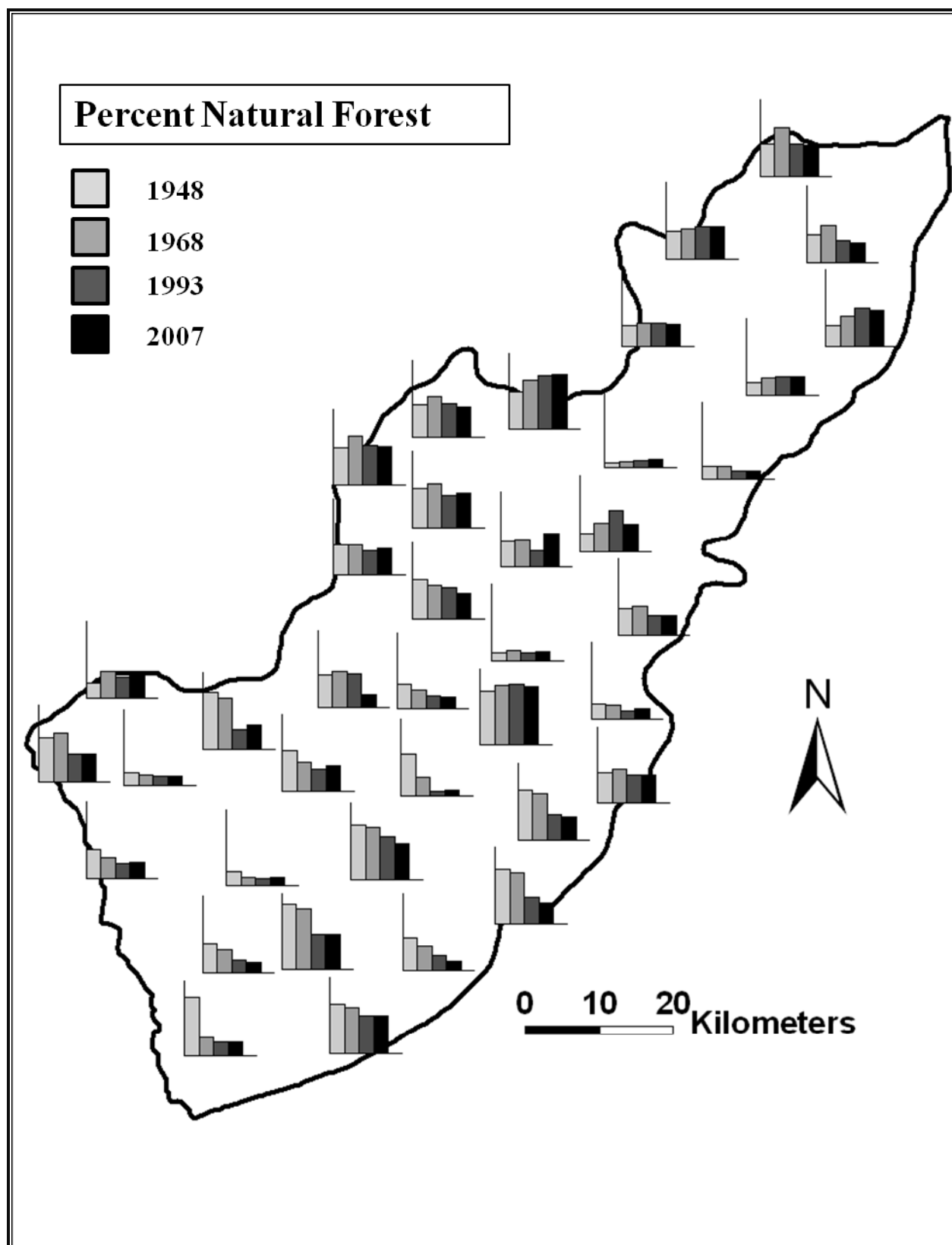


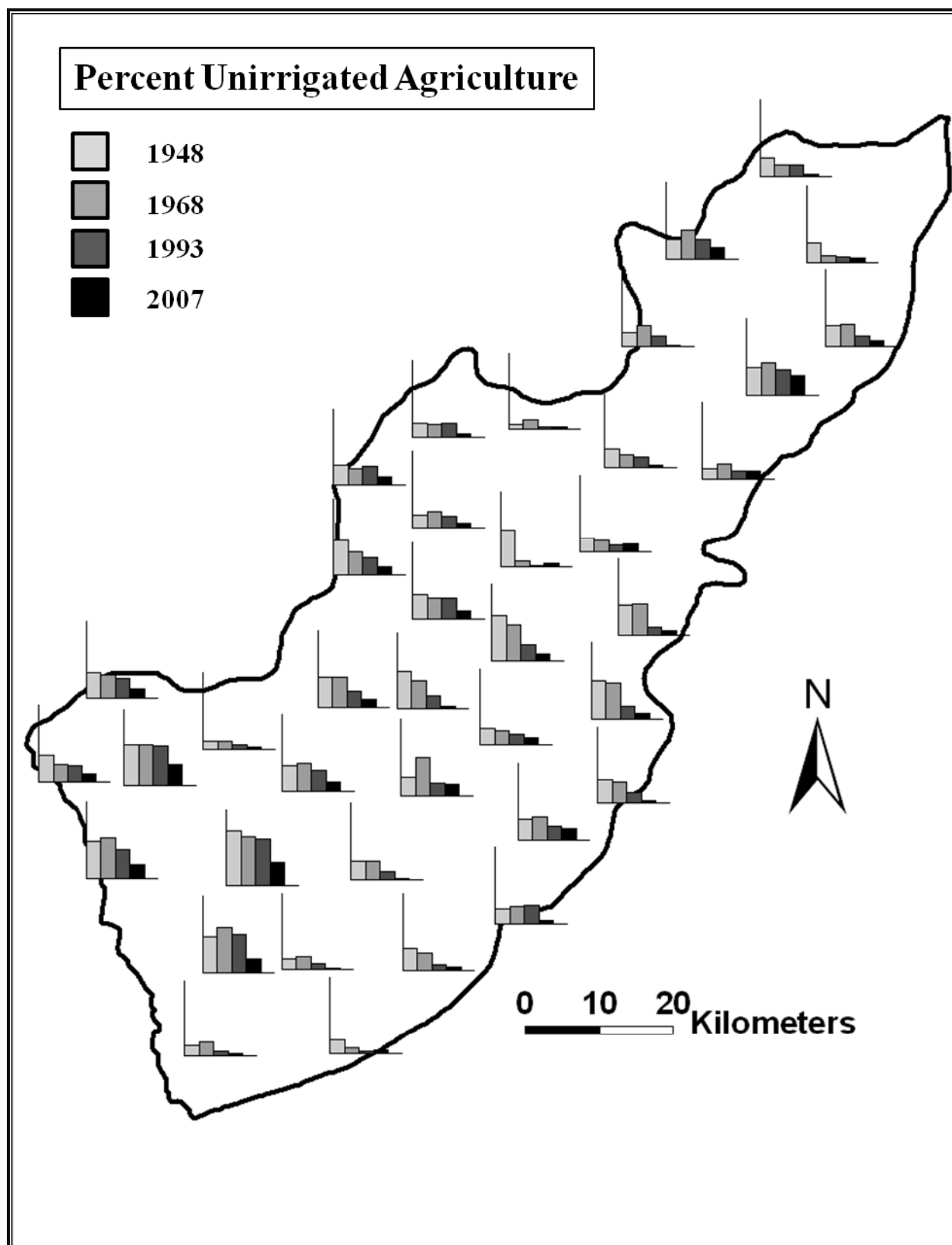
Figure 2.4. Comparison of observed area (km²) extrapolated to entire Dougherty Plain for selected land use/land cover (LULC) classes in specific sample years. Box plots depict 95% confidence intervals and means of the area covered by each LULC class (Unirrigated Agriculture [U], Irrigated Agriculture [I], Orchard [O], Water [W], and Developed [D]) and were calculated through bootstrap analysis with 2,000 iterations. Black circles represent estimates of area covered by various LULC classes calculated from various sources including census data and satellite based land cover data. Comparable data were often not available for the exact years sampled in this study, so these estimates are generated from the closest available year.

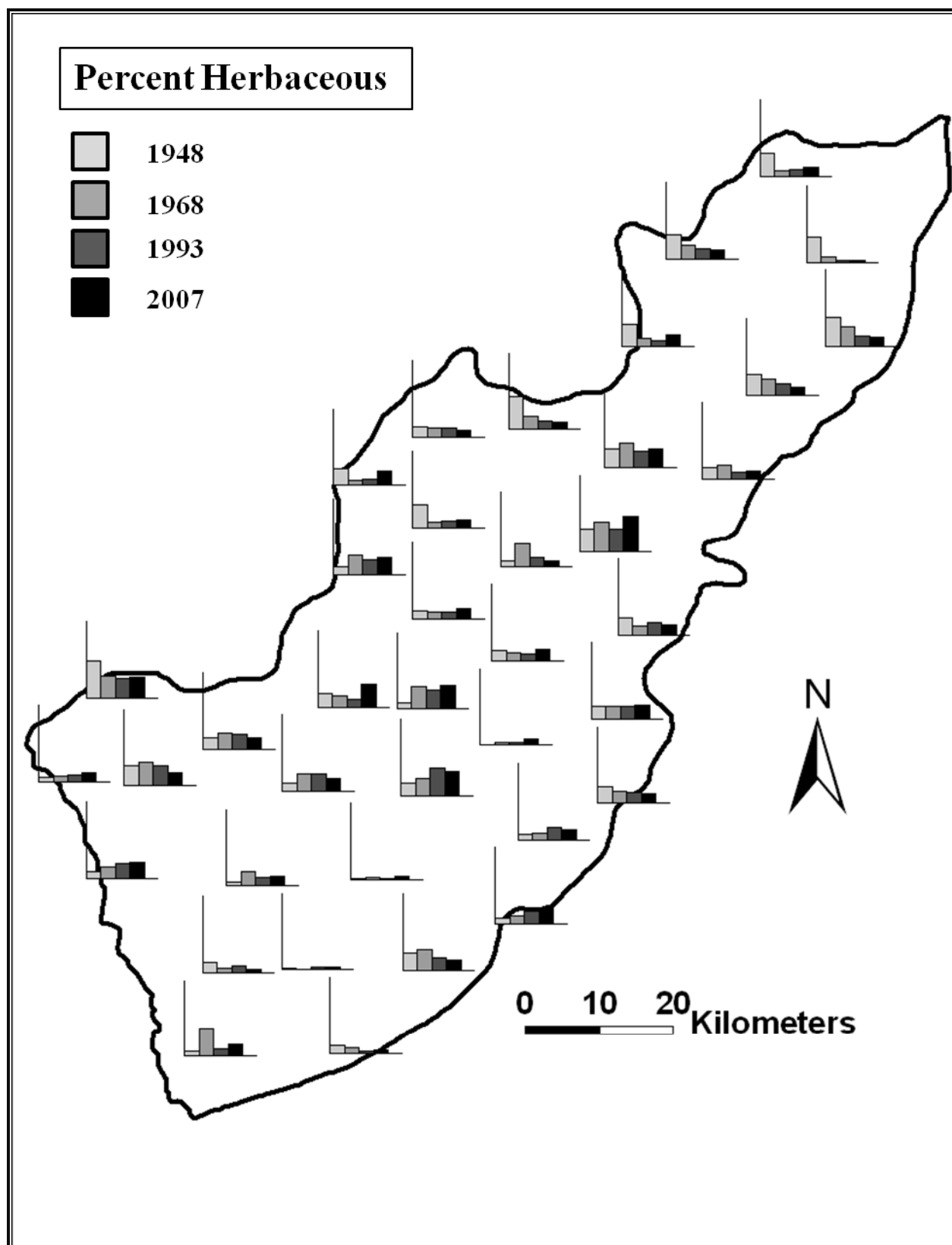
APPENDIX A

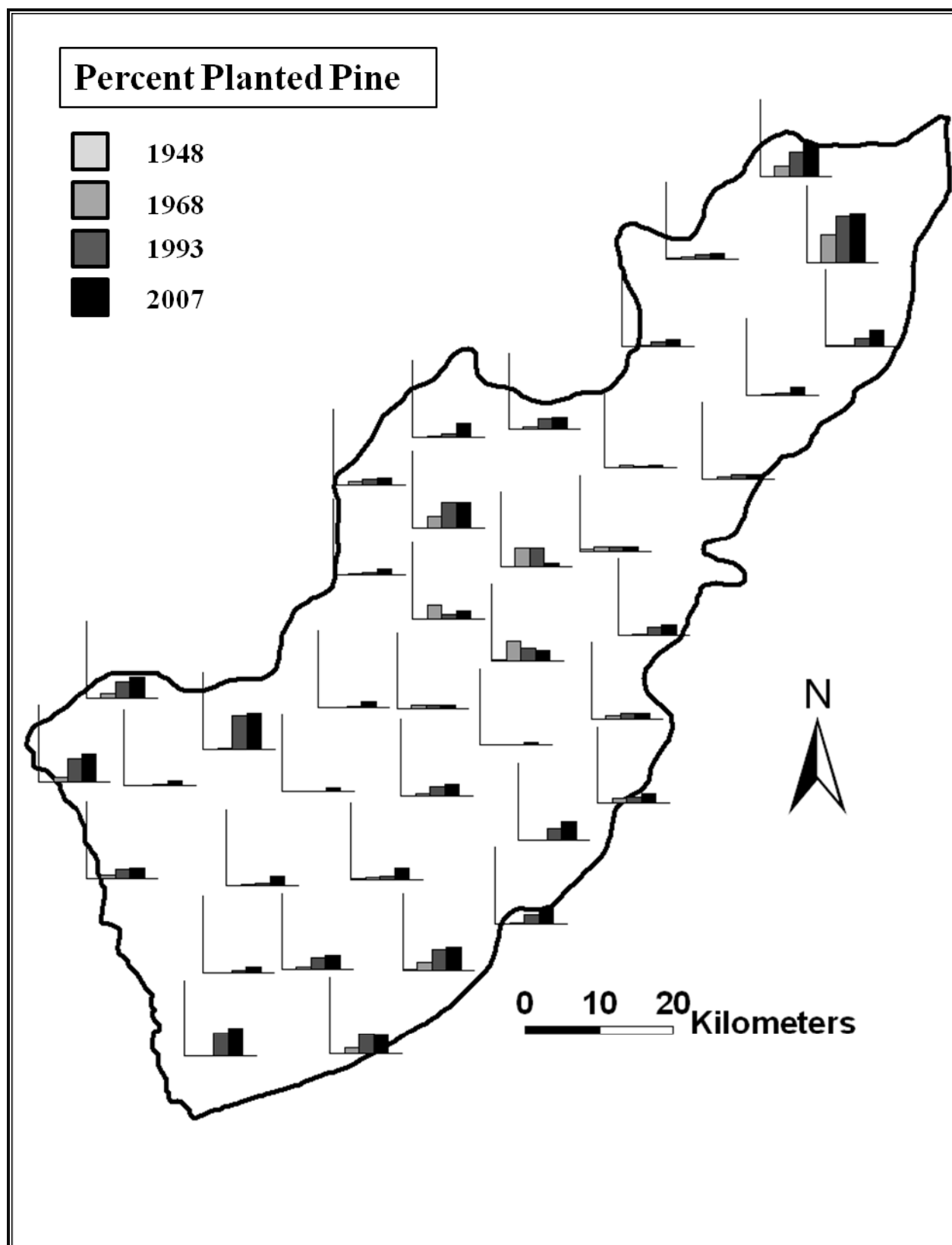
Appendix A highlights major trends in land use and land cover (LULC) for each of the 39 sample areas. The map on the following page displays the exact location of each sample area within the Dougherty Plain. In subsequent maps, sample areas are depicted by bar charts. The bar charts are positioned in the approximate location of the represented sample area, but may be slightly offset to avoid overlapping with neighboring bar charts. The bar charts display the percentage of each sample area represented by specified LULC classes in each of the four sample time periods. In each bar chart, the full length of the y-axis bar represents 100% of the sample area. Only LULC classes covering more than 10% of the sample landscape or considered to be of specific interest (i.e., Developed) are presented. These maps can be used to visually assess LULC composition trends within the Dougherty Plain.

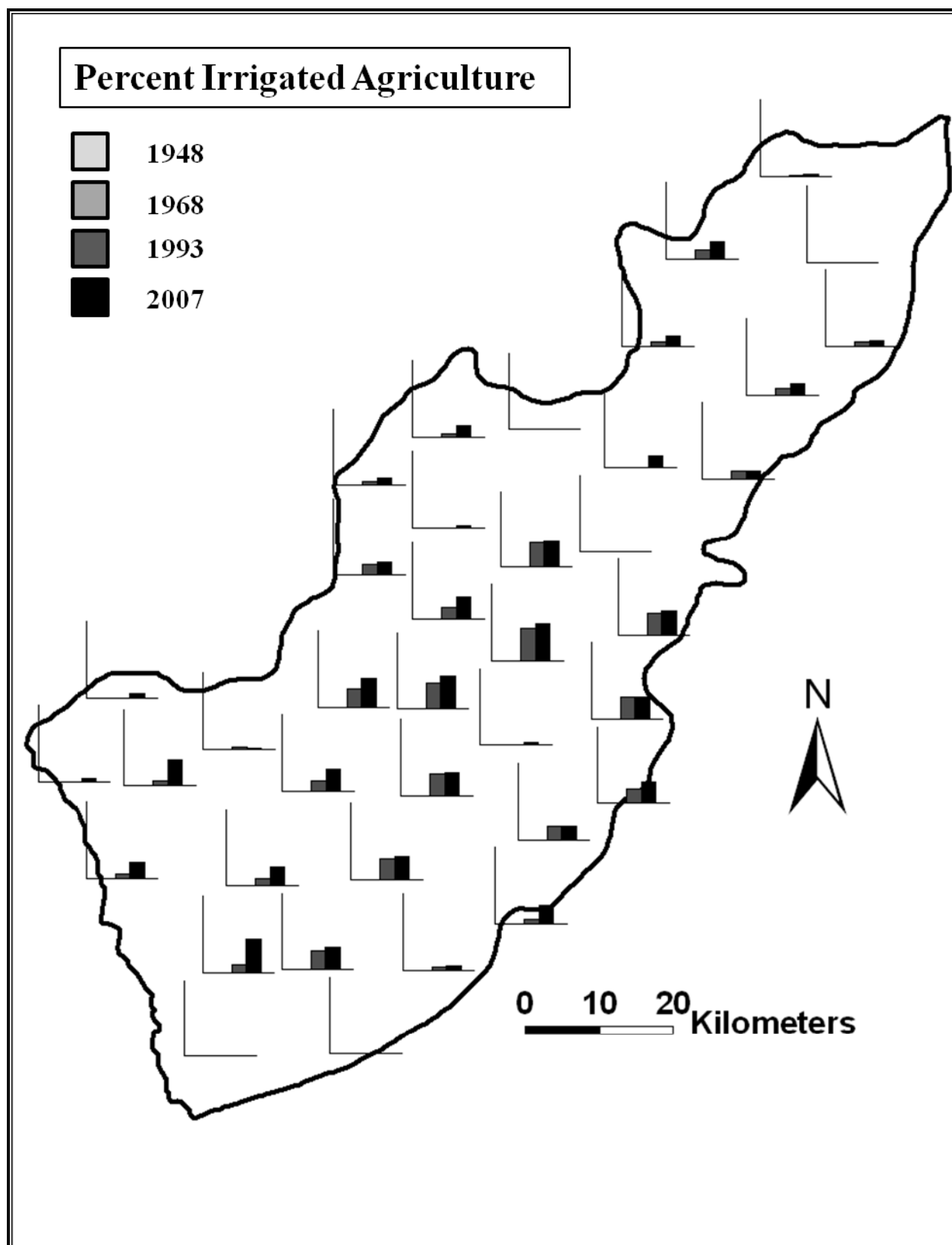


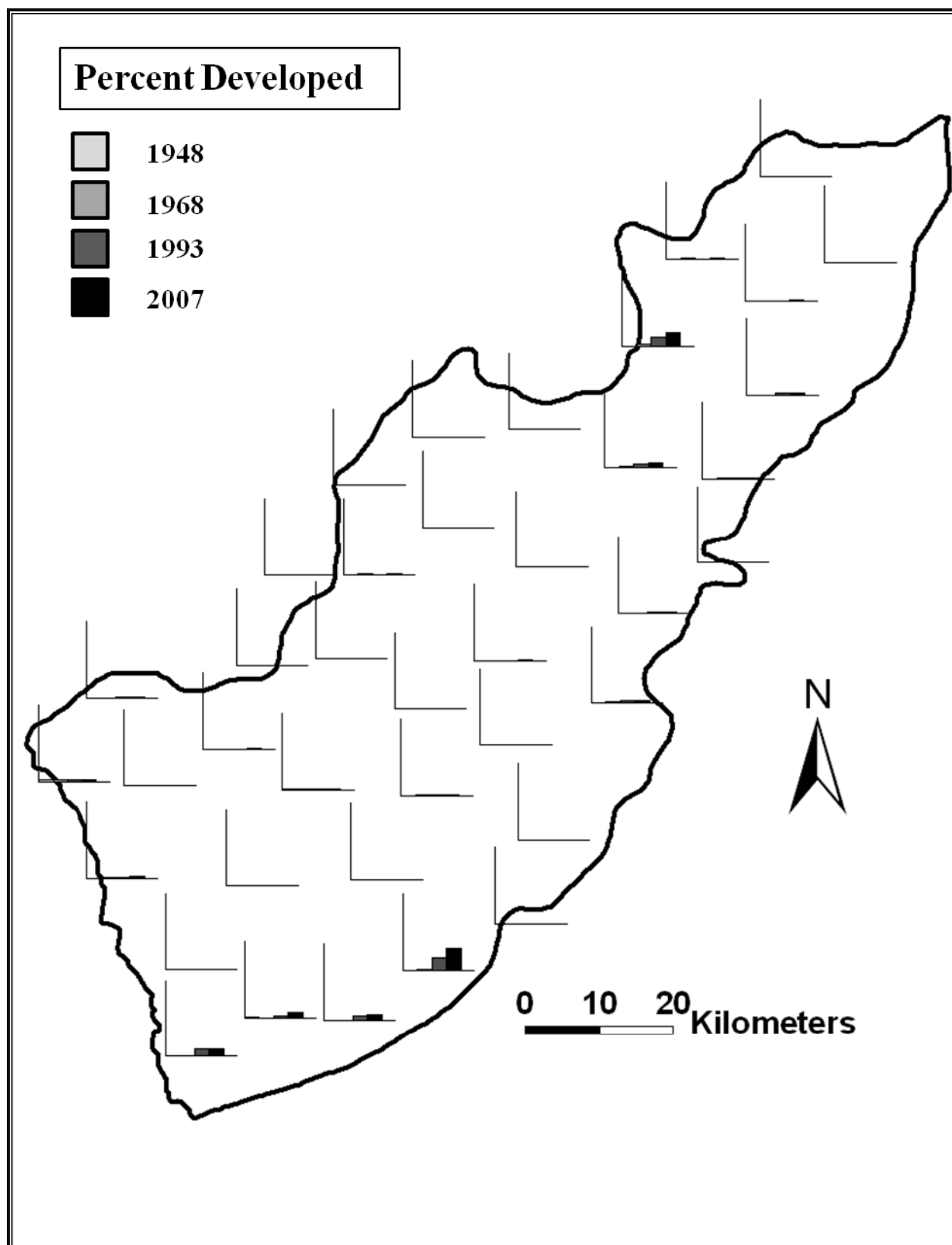












CHAPTER 3
MAPPING GEOGRAPHICALLY ISOLATED WETLANDS IN THE DOUGHERTY PLAIN,
GEORGIA, USA¹

¹ Marting, G.I., Kirkman, L.K., and J. Hepinstall-Cymerman. To be submitted to *Wetlands*.

ABSTRACT

Increasing interest in wetland assessment and monitoring requires an understanding of the location, spatial extent, and configuration of wetlands across the landscape. The National Wetlands Inventory (NWI) is the most commonly used data source for this information, but suffers from accuracy limitations, particularly in agricultural landscapes, and for seasonally-ponded, or forested wetlands. This study focuses on geographically isolated wetlands in an agricultural landscape, and seeks to improve wetland mapping accuracy by integrating currently available NWI maps with other geospatial data sources (i.e., soils and elevation data). These new wetland models predicted isolated wetlands not detected by NWI, and suggest that NWI represents a conservative estimate of existing isolated wetlands. A remote accuracy assessment indicates major improvements in accuracy, as compared to NWI maps. The isolated wetland population in the study area is dominated by small (< 4 ha) wetlands, and isolated wetlands are clustered. This study highlights a framework for improving the accuracy of wetland maps, and serves as a starting point for ongoing wetland assessment and monitoring efforts within the Dougherty Plain physiographic district of Georgia.

Key Words: Dougherty Plain, geographically isolated wetlands, Georgia, National Wetlands Inventory, wetland mapping.

INTRODUCTION

The National Wetland Inventory (NWI) is the most available and commonly used data on the location and spatial extent of wetlands throughout the United States. However, farmed, seasonally-ponded, and forested wetlands are often conservatively mapped or missing altogether (Tiner 1999). These limitations are particularly acute in areas such as the Dougherty Plain physiographic district of southwestern Georgia, where the karst topography is dotted with geographically isolated wetlands (Hendricks & Goodwin 1952; Beck & Arden 1983) and intensive agriculture is widespread (Harrison 2008). The inaccurate representation of wetlands poses a problem for researchers, regulators, and planners requiring a comprehensive accounting of wetland resources. We propose that the remote identification of geographically isolated wetlands can be improved by combining currently available NWI maps with other readily accessible geospatial data (i.e., Geographic Information System [GIS] data). We test this by using additional data sources to develop and compare the accuracy of multiple maps of geographically isolated wetlands within the Dougherty Plain.

NWI maps were developed by the U.S. Fish and Wildlife Service to depict the location, shape, and type of wetlands and deepwater habitats in many portions of the United States. The maps are created through photo-interpretation of mid- to high-altitude aerial photographs, and the acquisition date and scale of photos used varies across the country (Tiner 1999). NWI maps use a scientific, as opposed to a regulatory, definition of wetlands, because photo-interpreters cannot remotely assess the soil, vegetation, and hydrology parameters required for wetland field delineations (Tiner 1999). Tiner (1999) provides a detailed overview of NWI strengths and limitations as well as information on relevant studies assessing the accuracy of NWI maps. Despite limitations, NWI maps are often the only digital wetland maps available and are

frequently employed in scientific research including the mapping of geographically isolated wetlands (Tiner 2003).

Soil survey maps, created by the U.S. Department of Agriculture, Natural Resources Conservation Service, are another potentially useful tool in the remote identification and mapping of wetlands. The Soil Survey Geographic (SSURGO) data base is the most detailed geospatial soil data available and is designed for use at local scales (National Soil Survey Center 1995). The SSURGO digital soil maps are created from soil field surveys and aerial photo interpretation and are produced at varying scales (National Soil Survey Center 1995). The SSURGO database contains spatial data, e.g., soil map unit polygons, and attribute data describing physical and chemical properties as well as suitable uses for each soil map unit (National Soil Survey Center 1995). By querying the attribute data, users can identify soil map units consisting of hydric soils or containing hydric inclusions, as well as polygons identified as water or swamp. As with the NWI maps, the SSURGO data are useful in identifying potential wetlands from a scientific, but not necessarily a regulatory, perspective (Tiner 1999).

Generally, NWI maps tend to underestimate wetland extent and the hydric soil map units of digital soil survey maps tend to overestimate wetland extent (Tiner 1999). NWI and soil survey maps have been used separately, together, and in combination with ancillary data sources, e.g., elevation data and hydrology, to estimate wetland location and extent. Specifically, Tiner (2003) used NWI maps in conjunction with digital line graphs (DLGs), digital raster graphics (DRGs), Topologically Integrated Geographic Encoding and Referencing (TIGER) data, and aerial photographs to estimate the extent of geographically isolated wetlands in 72 study sites across the United States. McCauley and Jenkins (2005) used soil survey maps, DRGs, and digital elevation models (DEMS) to estimate the pre-settlement extent of depressional wetlands

in Champaign County, IL, and then compared these estimates with NWI maps to calculate wetland loss since settlement. Other research has used bare-ground Light Detection and Ranging (LIDAR) elevation data to identify depressional wetlands (Gritzner 2006), but LIDAR data is not yet widely available. Tiner (1999) suggests that NWI and soil survey maps could be combined to develop a more accurate depiction of existing and potential wetland areas. In this study, we draw on the work of Tiner (2003) and McCauley and Jenkins (2005) to identify and estimate the extent of geographically isolated wetlands in the Dougherty Plain.

Our definition of a geographically isolated wetland follows Tiner (2003) and refers to an individual wetland, which is completely surrounded by upland of a greater elevation and lacks visible surface-water connections to perennial streams or rivers, estuaries, or the ocean. We emphasize that this definition is distinct from a jurisdictional definition, which refers to those waters or wetlands that are subject to federal regulation under the Clean Water Act (CWA) (33 U.S.C. §§ 1251-1387), but see 33 U.S.C. § 502[7], 33 C.F.R. § 328.3[a] and 40 C.F.R. § 230.3[s] for specific definitions. Currently, geographically isolated wetlands are not subject to federal regulation under the Clean Water Act unless they are adjacent or connected to jurisdictional waters via a “significant nexus” (*Rapanos v United States* 2006; Leibowitz et al. 2008; Sponberg 2009). This study identifies geographically isolated wetlands based on available GIS data sources and is not an indication of the regulatory status of individual wetlands as established by field delineations and jurisdictional determinations (Tiner 2003).

Geographically isolated wetlands in the Dougherty Plain range in size from large (hundreds of hectares) shallow flat areas down to small (several m²) steep sided holes (Kirkman et al. 2000). Typically, they are inundated by precipitation in late fall and winter and dry down in spring and summer (Kirkman et al. 1998). Geographically isolated wetlands provide

ecosystem functions and services by reducing peak flood flows (Leibowitz 2003), improving water quality (Leibowitz 2003; Whigham & Jordan 2003), and supporting regional biodiversity (Semlitsch & Bodie 1998; Kirkman et al. 1999; Sharitz 2003). Geographically isolated wetlands support species-rich communities of flora (Kaeser & Kirkman 2009) and fauna (Battle & Golladay 2002; Smith et al. 2006) including state and federally listed species. Such wetlands also support many species at the meta-population level (Joyal et al. 2001; Fortuna et al. 2006). Small isolated wetlands in particular have historically been undervalued, but their loss or alteration has potential consequences for inter-wetland species dispersal and meta-population survival (Gibbs 1993; Semlitsch & Bodie 1998; McCauley & Jenkins 2005).

Given the current lack of federal regulations protecting geographically isolated wetlands (Leibowitz et al. 2008; Sponberg 2009), some states have enacted legislation to regulate impacts (Christie & Hausmann 2003), but no such policies have been implemented in Georgia. Indeed, basic information such as the number, spatial extent, and spatial configuration of geographically isolated wetlands throughout the state is lacking or the limited data is often inconsistent (Environmental Protection Division 1999; Brown 2002). Such basic spatial information is prerequisite to assessing wetland condition and function or initiating regulatory or incentive policies at the state or regional level.

In this study, we combine soils and elevation data with existing NWI maps to develop an improved spatial model of geographically isolated wetlands in the Dougherty Plain, and then quantify the number, spatial extent, and spatial configuration in the landscape. First, we used several sources of GIS data to develop multiple spatial models predicting the location and spatial extent of geographically isolated wetlands. We independently validated each model's output and calculated several accuracy measures to identify the most accurate model or combination of

models. We also compared the predicted number, total area, and mean area $\pm$ SD of geographically isolated wetlands among models. Finally, for the most accurate model, we assessed the size class distribution, nearest neighbor distances, and spatial configuration of predicted geographically isolated wetlands to inform upcoming research efforts as well as future regulatory decisions.

METHODS

Study Area

The Dougherty Plain is a karstic physiographic district located within the Coastal Plain of southwestern Georgia and covers 6,690 km² and parts of 14 counties (Figure 3.1). It is positioned between the Pelham Escarpment to the south and east and the Fall-line Hills and Chattahoochee River to the north and west (Beck & Arden 1983). It has low topographic relief and ranges in elevation from 93 m in the northeast to 24 m in the southeast (Clark & Zisa 1976). The region is characterized by broad, flat uplands, numerous geographically isolated wetlands, and few but deeply incised streams. Geographically isolated wetlands are common features of the landscape due to the dissolution of underlying limestone and the subsequent collapse of the overburden (Hendricks & Goodwin 1952; Beck & Arden 1983).

National Wetlands Inventory

NWI data for the counties of the Dougherty Plain were created from 1:58,000 color infrared photographs acquired in March of 1983 (U.S. Fish and Wildlife Service 1993). The target mapping unit for the Dougherty Plain was 0.41-1.21 ha, i.e., more than half of the wetlands within this size class were mapped and nearly all larger wetlands were mapped (Tiner 1999). NWI data for the Dougherty Plain were downloaded (<http://data.geocomm.com>) as 1:24,000 7.5" quadrangles and imported into ArcGIS 9.2 (Environmental Systems Research

Institute, Redlands, CA, USA) as ArcGIS “shapefiles.” These shapefiles were merged and then dissolved, so that each individual wetland was identified by a single polygon. Finally, this shapefile was clipped to the extent of the Dougherty Plain. NWI linear and point data were not used.

Hydric Soils and the Hydric Soils Model

SSURGO data were obtained (<http://data.georgiaspatial.org>) for each county within the Dougherty Plain and were merged and clipped to the extent of the Dougherty Plain. We used SSURGO attribute data (specifically, the “hydcomp” relational table) to identify soil map units consisting of hydric soils and soils with hydric inclusions, and we also included soil map units described as water or swamp (National Soil Survey Center 1995). We used these soil map units to develop a potential hydric soils data layer for use in subsequent models. For the hydric soils model (HSM), we used the same process to select and uniquely identify those map units consisting entirely of hydric soils, i.e., no non-hydric inclusions were identified in the soil map unit description (National Soil Survey Center 1995). The HSM also included soil map units identified as water and swamp. SSURGO linear and point data were not used.

Digital Raster Graphic (DRG) Model

DRGs were acquired (<http://data.georgiaspatial.org>) as 1:24,000 countywide mosaics for each county within the Dougherty Plain. DRGs are scanned images of U.S. Geological Survey 7.5” topographic maps produced between the 1970s and 1990s. Topographic maps are created through a combination of photogrammetry and field surveys, and they depict elevation contour lines and prominent natural and man-made features of the landscape (U. S. Geological Survey 2008). Although the topographic maps depict wetland areas, we did not use these in our analysis. Rather, we used the contour lines to identify topographic depressions (i.e., areas of

internal drainage), which are landforms where geographically isolated wetlands are likely to occur. DRGs were manually scanned and each topographic depression was identified and digitized as a polygon in ArcGIS 9.2 (McCauley & Jenkins 2005). Digitized depressions and the hydric soils data layer described above were intersected to create the DRG model of isolated wetlands (McCauley & Jenkins 2005).

Digital Elevation Model (DEM) Model

Digital elevation models (DEMs) are digital representations of ground surface topography. DEMs are classified into one of three levels (1, 2, or 3) according to the quality of the data, where level 1 represents the poorest quality and least accurate, and level 3 represents the highest quality and most accurate (U.S. Geological Survey 1998). Level 2 DEMs are generated by digitizing hypsographic and hydrographic data either through photogrammetry or from existing maps (U.S. Geological Survey 1998), and represent the best available data for the Dougherty Plain. The level 2 DEMs were acquired (<http://www.geocomm.com>) with 10 m resolution (each cell represents a 10 X 10 m area on the ground) for each quadrangle within the Dougherty Plain. The quadrangles were combined in ArcGIS 9.2 to create a continuous DEM coverage for the study area. Depressions in elevation were identified using ArcGIS Spatial Analyst “Flow Direction” and “Sink” commands. The “Flow Direction” command identifies the potential direction of water flow from each cell to its steepest down slope neighbor. The “Sink” command uses the output from “Flow Direction” to identify endorheic cells (areas of internal drainage), which are synonymous with our definition of depressional wetlands. The resulting sinks or areas of internal drainage were converted to polygons and intersected with the potential hydric soils data layer described above to create the DEM model of isolated wetlands (McCauley & Jenkins 2005).

Identifying Geographically Isolated Wetlands

The preceding methods identify probable locations of wetlands within the Dougherty Plain. The next step was to identify and remove non-isolated wetlands, i.e., wetlands with obvious surface connections to perennial streams or rivers. In general, the Savannah District of the United States Army Corps of Engineers (which is responsible for regulating wetlands in Georgia) considers wetlands located within 30.48 m (100 ft) of navigable waters or within the 100-year flood zone to be non-isolated (D. Crosby, personal communication, as cited in Kramer & Carpendedo 2009), and we followed this criteria in identifying non-isolated wetlands. Considering wetlands within 30.48 m of a navigable waterway to be non-isolated ensures that riparian or headwater wetland polygons are not disconnected from their associated streams during the GIS analysis (Tiner 2003). Additionally, the 30.48 m distance we use in this study is intermediate between the 20 m and 40 m stream buffers Tiner (2003) used to identify geographically isolated wetlands in selected areas of the United States. Excluding wetlands located within the 100-year floodplain further reduces the possibility of including wetlands with intermittent connections to navigable waterways. We recognize that these assumptions do not necessarily guarantee the absence of connectivity to surface drainages or could result in the exclusion of some isolated wetlands located within the floodplain but lacking connectivity. This approach provides a more conservative alternative to that of considering all wetlands not intersecting a mapped stream line to be isolated.

The National Hydrography Dataset (NHD) is a digital spatial dataset representing the surface waters of the United States. High-resolution NHD stream data for the Dougherty Plain were downloaded (<http://nationalmap.gov>) as 1:24,000 county mosaics. Non-isolated streams were identified by selecting and assigning a unique identifier to those streams that drain to Lake

Seminole (the eventual outfall for all non-isolated waters within the Dougherty Plain). Next, Federal Emergency Management Agency (FEMA) Q3 Flood Data maps were downloaded (<http://data.georgiaspatial.org>) for each county within the Dougherty Plain. The 100-year flood zones were selected and those intersecting non-isolated NHD streams were uniquely identified. Finally, those wetland polygons located or partially located within 30.48 m of non-isolated NHD streams or within 100-year flood zones associated with non-isolated streams were identified as non-isolated wetlands and removed from the models.

Model Validation

The NWI, HSM, DRG, and DEM models were evaluated for accuracy by comparing them with geographically isolated wetland polygons independently digitized at 1:10,000 scale from National Agriculture Imagery Program (NAIP) 2009 aerial imagery (Figure 3.2). Thirty-nine sample areas, representing a combined 10% of the total area of the Dougherty Plain, were randomly selected throughout the study area. NAIP 2009 aerial imagery was downloaded (<http://data.georgiaspatial.org>) for each county within the Dougherty Plain. Within each sample area, wetlands not located within 30.48 m of non-isolated NHD streams or within their associated 100-year flood zones were manually digitized in ArcGIS 9.2, and considered to represent geographically isolated wetlands (Figure 3.2). Five hundred validation points were created using Hawth's Tools (Beyer 2004) in ArcGIS 9.2. Two-hundred and fifty points were randomly generated within the geographically isolated wetland polygons digitized from 2009 NAIP photographs, and 250 points were randomly generated outside of these polygons but within the 39 sample areas. A minimum distance of 100 m was enforced between all validation points. For each model, validation points located within predicted geographically isolated wetlands were uniquely identified. Finally, each validation point was overlaid on 2009 NAIP

imagery and identified as being located either inside or outside of a geographically isolated wetland.

The validation point data was used to construct a confusion matrix and generate accuracy statistics for each isolated wetland model and each possible combination of models (Fielding & Bell 1997). Accuracy statistics were calculated for 10 parameters, but here we report on the following three (Fielding & Bell 1997):

Kappa – the proportion of positive and negative agreement penalized for random agreement.

Sensitivity – the percent of correctly classified validation points located within actual geographically isolated wetlands.

Specificity – the percent of correctly classified validation points located outside of geographically isolated wetlands.

Analysis

We calculated total number, total area, and mean area $\pm$ SD, for predicted geographically isolated wetlands and compared results among selected models. The most accurate model was determined to be the combination of the NWI, HSM, and DRG models (hereafter the Combined Model [CM]). Next, we used the CM to calculate several composition and configuration metrics. We grouped predicted geographically isolated wetlands into 1 ha size classes to determine the size class distribution. Using a custom Visual Basic script, we calculated nearest neighbor distances (edge to edge) for all predicted polygons. To assess the potential importance of smaller wetlands from a meta-population species dispersal perspective, we recalculated the nearest neighbor distance sequentially; excluding progressively larger 1 ha size classes. A subset of the nearest neighbor results were checked against distances measured manually.

Spatial clustering of predicted geographically isolated wetlands was evaluated using Spatial Statistics tools in ArcMAP 9.2. The degree of spatial clustering was assessed using the Average Nearest Neighbor Analysis which calculates a nearest neighbor index by dividing the observed distance between polygon centroids by the average expected distance in a hypothetical random distribution. A nearest neighbor index value greater than 1 indicates dispersion, and a value less than 1 indicates clustering. Statistical significance is based on Z scores, and at the 0.05 significance level a Z score between -1.96 and 1.96 indicates that the observed pattern is likely random. We used Spatial Autocorrelation (Moran's I) to discern if the pattern of predicted wetland polygons was related to wetland size. A Moran's I value near 1.00 indicates that wetlands of similar size tend to cluster together, and a value near -1.00 indicates dispersion between wetlands of similar size. The significance of the Moran's I value is interpreted with a Z score, as described above. High/Low Clustering Analysis was used to determine if spatial clustering is more pronounced among larger or smaller wetlands. This tool calculates the General G statistic and associated Z score; higher values indicate clustering of larger wetlands and small values indicate clustering of smaller wetlands. Finally, Cluster/Outlier Analysis with Rendering was used to identify and map wetland clusters. This analysis calculates and spatially displays Morans's I values and Z scores for each wetland polygon, and identifies wetlands surrounded by neighbors of similar size (clusters) and those surrounded by neighbors of dissimilar size (outliers).

RESULTS

Of the 15 models tested (NWI, HSM, DEM, DRG, and all possible 2-, 3-, and 4-way combinations), the model combining NWI, HSM, and DRG (CM) and the model combining NWI, HSM, DRG, and DEM (hereafter the Total Model [TM]) generated the most accurate

predictions of geographically isolated wetlands (Table 3.1). These models represent major improvements in Kappa and sensitivity scores as compared to all other models (Table 3.1). Although the CM and TM do have slightly lower specificity scores as compared to other models; indicating that some non-wetland areas were included within predicted wetland polygons (Table 3.1). The accuracy statistics between the CM and the TM were identical; indicating that the DEM model provides no additional benefit to accuracy. Although the TM identified 154 more wetlands than the CM, these were generally quite small (mean = 0.09 ± 0.18 ha [mean $\pm$ SD]) and irregular in shape. The CM represents the best compromise between Kappa, sensitivity, specificity, and data requirements and the geographically isolated wetlands it predicts are displayed in Figure 3.3.

The models varied in the number, total area, mean area, and variability in area of predicted geographically isolated wetlands (Table 3.1). Individually the HSM, DEM, and DRG models predicted many fewer wetlands as compared to the NWI model (Figure 3.4), because, unlike the NWI, these models can only predict wetlands in specific landscape settings, i.e., topographic depressions or areas with listed hydric soils. Additionally, the NWI model tended to predict wetlands that were smaller in area than those predicted by the HSM and DRG models (Table 3.1), and this reflects the minimum mapping units of the respective data sources (Tiner 1999). In many cases, the NWI model predicted multiple wetlands that were encompassed within a single wetland boundary predicted by the HSM or DRG models, and in this situation the CM counts only one wetland (Figures 3.4 & 3.5). For example, of the 11,620 geographically isolated wetlands predicted by the CM, 9,743 overlapped, at least in part, with NWI wetlands. In all, the CM predicted 1,874 additional wetlands and 19,737 additional wetland hectares as

compared to NWI. These additional wetlands are generally small (mean = 2.04 ± 3.19 ha [mean $\pm$ SD]); with 45% measuring less than 1 ha in size.

The CM estimates a density of 1.70 geographically isolated wetlands per km² in the Dougherty Plain, and the population of isolated wetlands is dominated by smaller size classes (Figure 3.6). Fifty percent of wetlands are less than 1 ha in size, and 81% are less than 4 ha in size (Figure 3.6). Only 7% of wetlands are 10 ha or greater in size (Figure 3.6). The mean nearest neighbor distance (edge to edge) between wetlands is 180 m when all wetlands are considered (Figure 3.7); however, this distance increases rapidly as the smaller size classes are removed from the population (Figure 3.7).

Average Nearest Neighbor Analysis indicates that geographically isolated wetlands predicted by the CM are spatially clustered across the Dougherty Plain ($Z = -10.42$, $P < 0.01$). Spatial Autocorrelation (Moran's I) analysis indicates that wetlands of similar size tend to cluster together ($Z = 12.83$, $P < 0.01$). Additionally, High/Low Clustering Analysis shows that larger wetlands are significantly clustered across the landscape ($Z = 7.75$, $P < 0.01$). Cluster/Outlier Analysis with Rendering shows the location of wetland clusters that are statistically significant ($P < 0.05$) (Figure 3.8).

DISCUSSION

We found that the representation of geographically isolated wetlands was greatly improved when wetlands predicted by HSM and DRG models were added to NWI wetlands. Individually, each of these models accurately predicted unique isolated wetlands in specific landscape positions but failed to predict many other isolated wetlands. In combining data sources, our models complemented each other and predicted isolated wetlands that may have been missed by any individual model. The CM achieved greater accuracy by uniting the

advantages of all three models. These predictions of number, total area, mean area $\pm$ SD, and the related information on spatial configuration represent the only spatially explicit estimates reported for geographically isolated wetlands in the Dougherty Plain to date.

The density of predicted isolated wetlands ($1.74/\text{km}^2$) in the Dougherty Plain is likely a result of karst topography, which promotes the formation of depression wetlands through the dissolution of subsurface limestone and subsequent collapse of overburden (Hendricks & Goodwin 1952; Beck & Arden 1983). Our results reiterate the regional abundance of small (< 4.0 ha) geographically isolated wetlands highlighted by Semlitsch and Bodie (1998). Similarly high proportions of small to large wetlands have been reported from isolated wetland populations in South Carolina (Semlitsch & Bodie 1998) and Maine (Gibbs 1993). The abundance of smaller wetlands reduces nearest-neighbor distances and maintains the source-sink dynamics of species at the meta-population level (Gibbs 1993; Semlitsch & Bodie 1998; Joyal et al. 2001).

Small isolated wetlands historically have been, and remain, more vulnerable to alteration and degradation, because they are easier to ditch, drain, or fill and have been specifically excluded from federal protection (Bennett & Nelson 1991; Semlitsch & Bodie 1998; Leibowitz & Nadeau 2003). For example, until 1996 the United States Army Corps of Engineer's Nationwide General Permit (NWP) 26 authorized the filling of "headwaters and isolated waters" up to 4.0 ha in size with minimal review (National Research Council 1995; Kaiser 1998). The 1996 re-issuance of NWP 26 reduced the maximum size threshold to 1.2 ha, and NWP 26 expired in 2000 (Copeland 2008). Eighty-one percent of the wetlands predicted by the CM are less than 4.0 ha and 54% are less than 1.2 ha in size. Thus even when geographically isolated wetlands were protected under the CWA, NWP 26 potentially allowed for the discharge of

dredged or fill material into the majority of the isolated wetlands in the Dougherty Plain with minimal review.

Geographically isolated wetlands are distributed throughout the Dougherty Plain, but wetland clusters appear to be located in the central Dougherty Plain; an area that lacks prevalent surface drainage networks (Figure 3.9). The significant spatial clustering of geographically isolated wetlands in portions of the Dougherty Plain is comparable to clustered patterns of depression wetlands in Illinois (McCauley & Jenkins 2005). The factors driving wetland clustering in the Dougherty Plain are poorly understood, but may be linked to local geology, topography, and hydrology (Hendricks & Goodwin 1952; Beck & Arden 1983).

Limitations

While the increased accuracy of the CM supports its use as a wetland mapping tool, we recognize that our predicted isolated wetland polygons reflect the limitations and assumptions of the individual data sources. Consequently, the map product does not replace the need for field-based wetland delineations and may differ from jurisdictional determinations (Tiner 2003). Wetlands characterized as isolated in this analysis could be connected to non-isolated waters via groundwater connections, small channels, ditches, or sub-surface drain tiles that cannot be detected with currently available data. The accuracy of our results may be further improved as new and finer resolution data (e.g., elevation data derived from Light Detection Ranging [LIDAR]) become available.

Future Directions

Our approach and results are broadly relevant to the field of spatial ecology, e.g., meta-populations (Hanski 1999; Loreau 2003) and specifically pertinent to wetland research and management in the Dougherty Plain; including the conservation of wetland flora and fauna

(Golladay et al. 1997; Drew et al. 1998; Semlitsch & Bodie 1998; Gibbs 2000; Edwards & Weakley 2001; Smith et al. 2006). Ongoing research in the Dougherty Plain focuses on developing a multi-metric framework to assess the ecological condition of geographically isolated wetlands at local wetland and landscape scales. Such goals necessitate an accurate depiction of the number, location, and spatial configuration of geographically isolated wetlands within the Dougherty Plain, and our model provides a much-improved tool to obtain that information. This work reveals the abundance and distribution of isolated wetlands; information that should be useful in regional land use planning and provides quantification of the resource that is of emerging interest to wetland regulatory agencies. Finally, we demonstrate an efficient method for mapping isolated wetlands that would be readily applied to other regions, particularly those dominated by agricultural land use.

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Table 3.1. Accuracy and wetland statistics of GIS-based model results of isolated wetlands in the Dougherty Plain, Georgia, USA. Models include the National Wetlands Inventory Model (NWI), Hydric Soils Model (HSM), Digital Raster Graphics Model (DRG), Total Model (TM), and Combined Model (CM). TM is an aggregation of the NWI, HSM, DRG, and DEM models. CM is an aggregation of the NWI, HSM, and DRG models.

Model	Accuracy Statistics			Wetland Statistics		
	Kappa	Sensitivity	Specificity	Number	Total Area (ha)	Mean Area $\pm$ SD (ha)
NWI	0.70	68%	99%	11,239	22,694	2.02 $\pm$ 6.57
HSM	0.62	60%	99%	5,219	26,366	5.05 $\pm$ 14.99
DRG	0.21	41%	99%	3,882	16,443	4.24 $\pm$ 14.10
DEM	0.04	9%	99%	4,238	2,358	0.56 $\pm$ 3.33
TM	0.88	90%	98%	11,774	42,515	3.61 $\pm$ 13.00
CM	0.88	90%	98%	11,620	42,431	3.65 $\pm$ 13.07

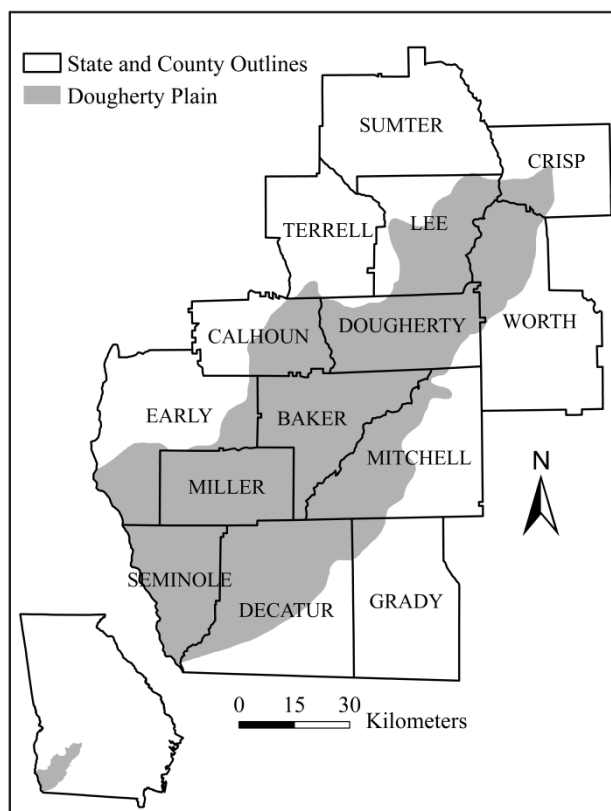


Figure 3.1. Map of the Dougherty Plain study area, neighboring counties and its location in the state of Georgia, USA.

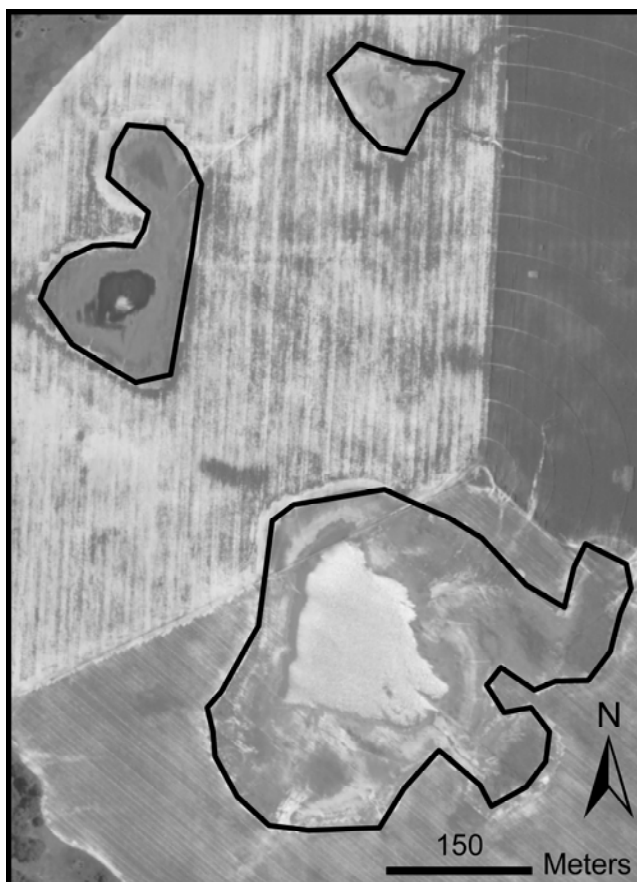


Figure 3.2. Example of isolated wetland polygons manually digitized at 1:10,000 from 2009 National Aerial Imagery Program aerial photographs (shown here in grayscale; photographs were taken in true color). Wetland polygons were digitized within 39 randomly selected areas within Dougherty Plain and were used in the model validation process.

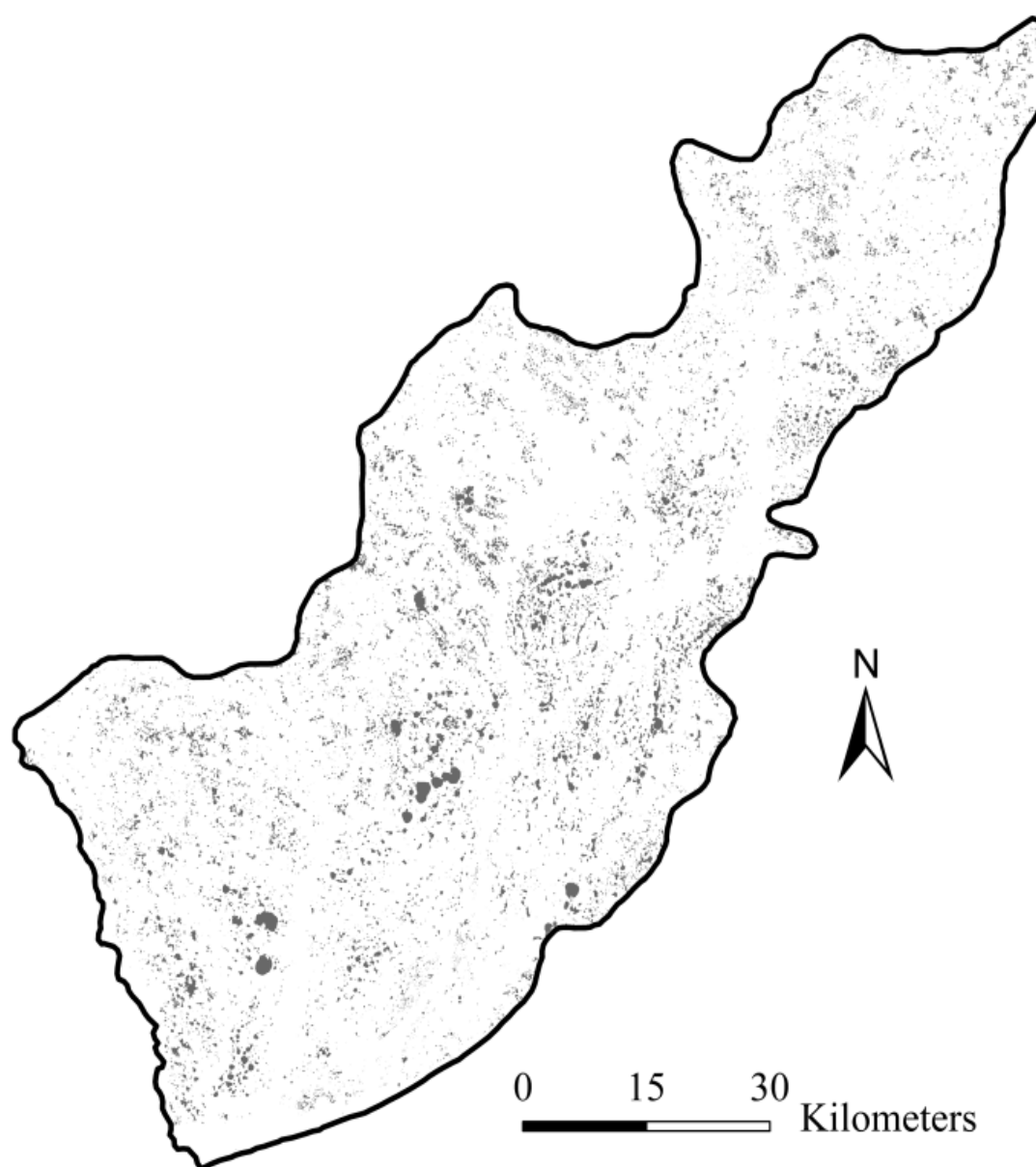


Figure 3.3. The location and spatial extent of isolated wetlands predicted by the Combined Model within the Dougherty Plain, Georgia, USA.

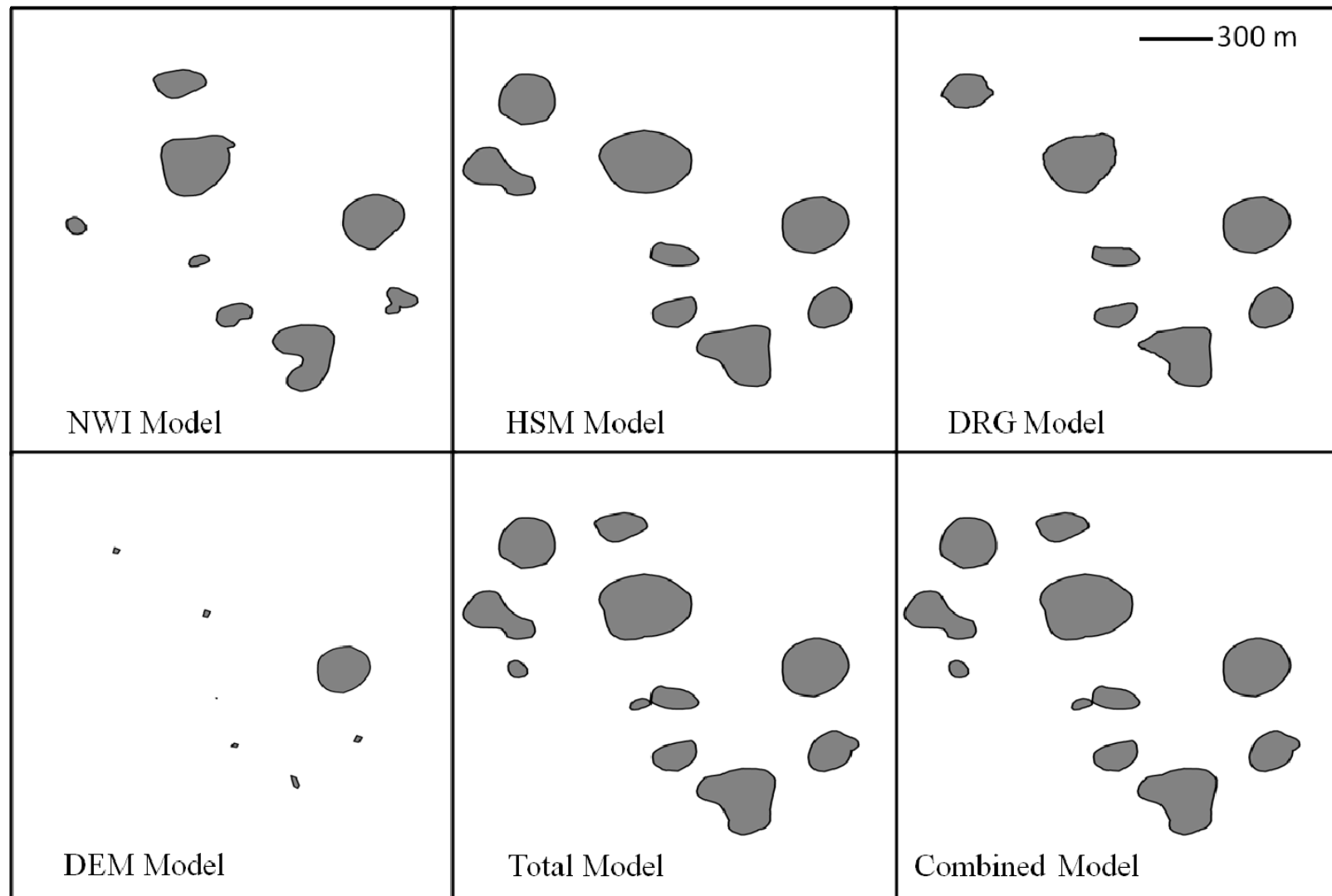


Figure 3.4. Example of differences in predicted wetland polygons between models for a selected area of the Dougherty Plain, Georgia, USA. The Total model is the aggregation of the NWI, HSM, DRG, and DEM models. The Combined Model is the aggregation of the NWI, HSM, and DRG models. In this example the predictions of the Total and Combined models are identical.

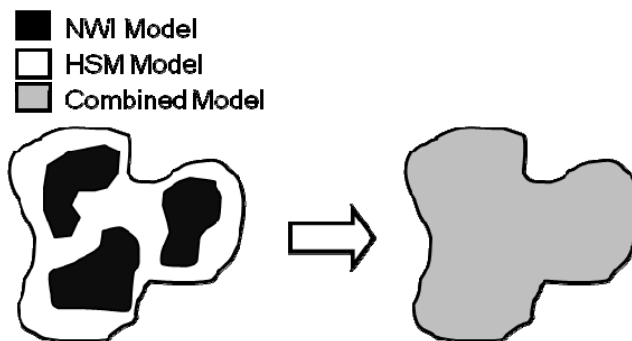


Figure 3.5. Illustration of how wetlands detected by different models are incorporated into the Combined model. Here the NWI Model detected 3 small wetlands contained within 1 larger wetland detected by the HSM Model. The NWI and HSM wetlands are depicted as a single wetland in the Combined Model.

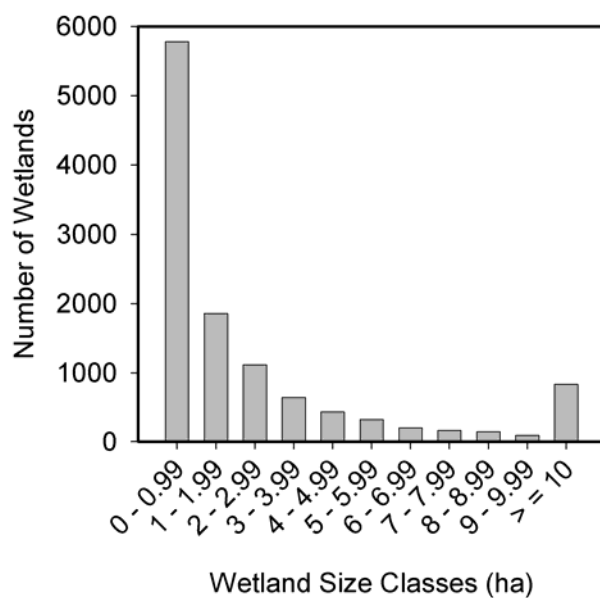


Figure 3.6. The size class distribution of isolated wetlands predicted by the Combined Model in the Dougherty Plain, GA, USA. Classes are approximate. For example, the 0-1 class ranges from 0.00 ha up to 0.99 ha.

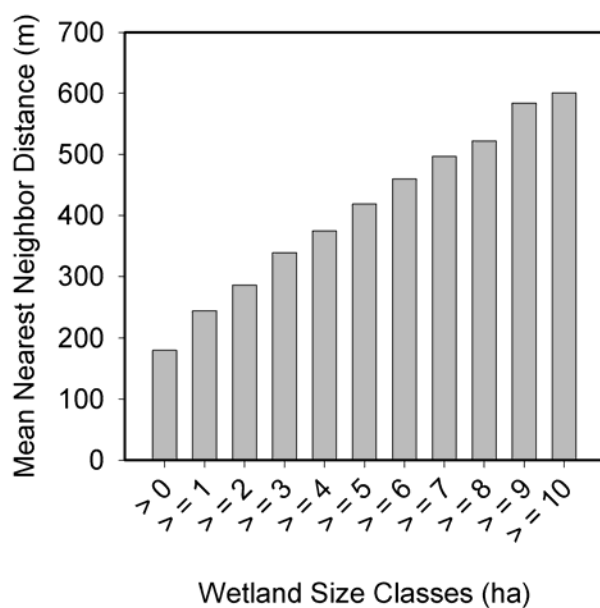


Figure 3.7. Mean nearest neighbor distances (edge to edge) between isolated wetlands predicted by the Combined Model in the Dougherty Plain, GA, USA. Distances were calculated sequentially with all previous size classes excluded.

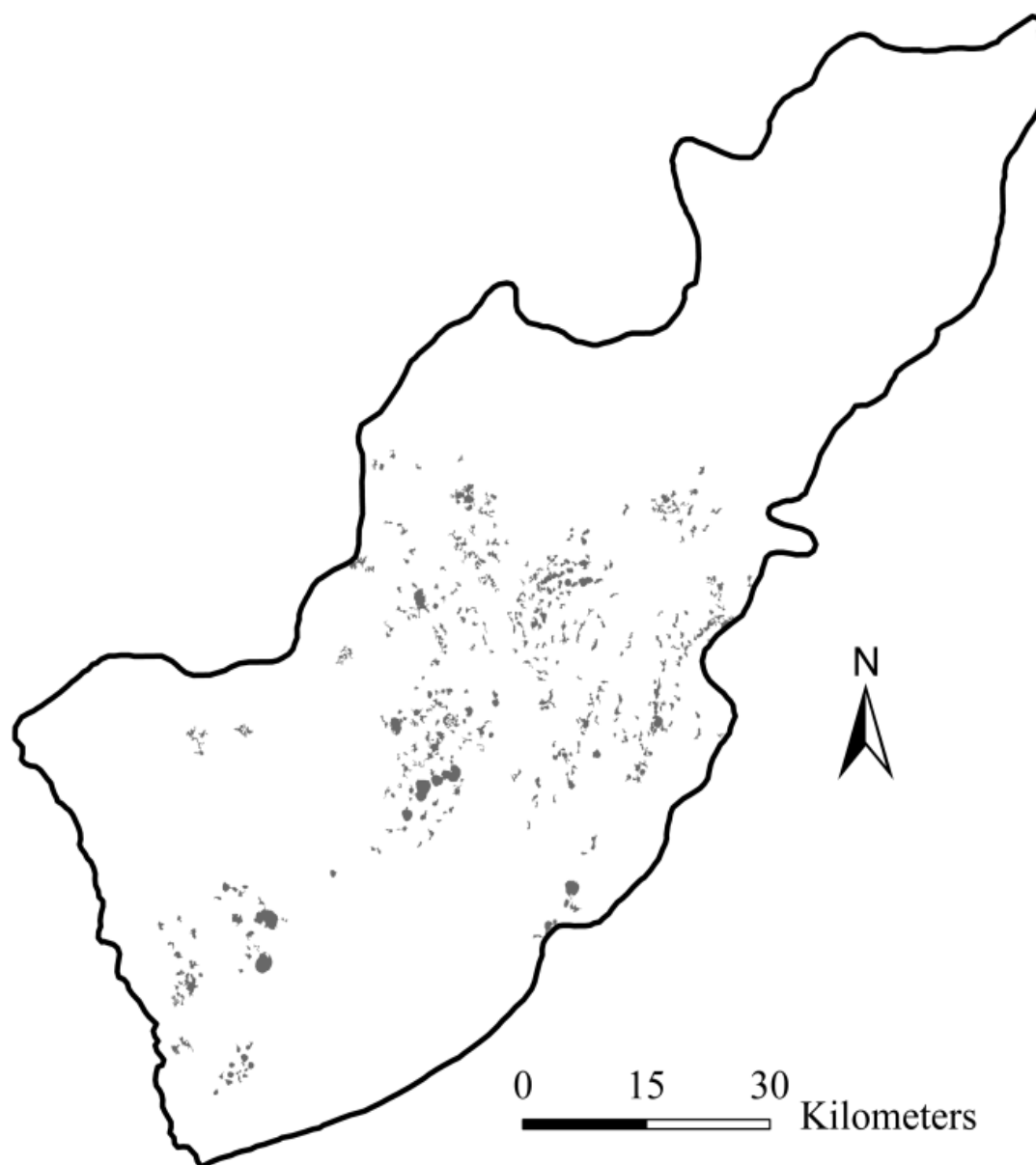


Figure 3.8. Cluster/Outlier Analysis with Rendering depicting spatially autocorrelated clusters of larger wetlands ($P < 0.05$) within the Dougherty Plain, Georgia, USA.

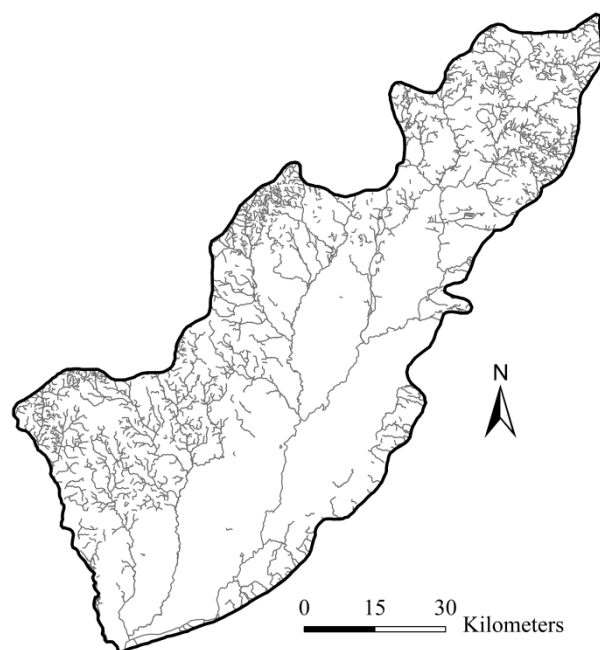


Figure 3.9. Drainage patterns from the National Hydrography Dataset (<http://nationalmap.gov>) for the Dougherty Plain, Georgia, USA.

CHAPTER 4

DEVELOPING A WETLAND ASSESSMENT FRAMEWORK FOR ISOLATED WETLANDS
IN THE DOUGHERTY PLAIN, GEORGIA, USA¹

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ABSTRACT

Increasing interest in assessing and monitoring wetland condition has led to the development of a 3-level wetland assessment method combining remote landscape assessment (Level 1), rapid field methods (Level 2), and intensive field methods (Level 3). This study integrates these assessment levels to evaluate 40 isolated wetlands located in Georgia's Dougherty Plain physiographic district. We used the Landscape Development Intensity index (LDI) as a Level 1 assessment tool, and calculated LDI values from 5 data sources, including manually digitized land cover data and 4 sources of preexisting satellite derived land cover data. We also conducted rapid assessments, macrophyte surveys, and water quality analyses at each wetland, and developed Wetland Condition Indices (WCIs) for assessment Levels 2 and 3. Using Spearman's correlation coefficient, we found strong and significant correlations among each of the three assessment levels; indicating agreement between the assessment levels and suggesting that (following regional calibration) the time-intensive Level 3 assessment could be estimated or predicted by the less intensive Level 1 or Level 2 assessments. We identified two preexisting satellite based land cover datasets as suitable surrogates for manually digitized land cover data, and we suggest that these datasets be used to calculate the Level 1 LDI assessment method in future regional or state level wetland assessment efforts.

Key Words: isolated wetland, landscape development intensity, wetland assessment

INTRODUCTION

Interest in assessing and monitoring wetland condition has increased recently and is currently the focus of much research (e.g., Van Dam et al. 1998; U.S. Environmental Protection Agency 2002; Fennessy et al. 2004; Brown & Vivas 2005; Whigham et al. 2009; Sifneos et al. 2010). To develop a more effective wetland monitoring program, the United States Environmental Protection Agency outlined a three level framework for wetland monitoring and assessment (Fennessy et al. 2004). This framework splits wetland assessment procedures into integrated levels that vary in scale and intensity. The three assessment levels are: Landscape Scale Assessment (Level 1), Rapid Field Methods (Level 2), and Intensive Field Methods (Level 3) (Fennessy et al. 2004). This framework has been successfully applied in multiple wetland assessment efforts (e.g., Mack 2006; Reiss & Brown 2007), but the accurate evaluation of individual wetland types generally requires development and calibration of wetland- or region-specific assessment procedures at each of the three levels.

Level 1 or landscape assessments typically integrate available GIS data to remotely quantify the landscape context of an individual wetland. Research has shown that surrounding land use or land cover affects physical, chemical, and biological components of aquatic systems (Johnes et al. 1996; Zacharias et al. 2004; Houlahan et al. 2006). A recently developed Level 1 procedure, the Landscape Development Intensity (LDI) index, uses land cover maps to quantify the intensity of human activity surrounding an area of interest (Brown & Vivas 2005). The LDI ranks land cover classes according to the amount of non-renewable energy (emergy) necessary for their maintenance and assigns a LDI coefficient to each land cover class (Odum 1996; Brown & Vivas 2005). For an individual wetland, an area weighted mean is calculated from LDI coefficients of land cover polygons surrounding the wetland boundary and extending out to a

specified buffer distance; this value represents the LDI index score for that specific wetland. LDI scores can be developed with preexisting land cover maps or with land cover digitized from aerial photographs (Reiss & Brown 2007). Digitizing land cover from aerial photography may result in greater accuracy, but is impractical for regional scale assessments. The LDI was developed and calibrated in Florida, but has been applied elsewhere (Mack 2006; Vivas & Brown 2006; Stein et al. 2009).

Level 2 or rapid assessments include quantitative or qualitative field based measurements of wetland condition that are simple, rapid, and easy to apply (U.S. Environmental Protection Agency 2002; Fennessy et al. 2004). These methods typically rely on best professional judgment to evaluate readily observable wetland features or stressors that are sensitive to human impacts on wetland systems (Van Dam et al. 1998; Mack et al. 2000). Fennessy et al. (2004) evaluated 16 rapid assessment procedures to identify the strengths and weaknesses of each method and highlight the most common and useful indicators of wetland condition. Rapid assessments most commonly evaluate properties of wetland hydrology, soils, vegetation, and landscape setting (Fennessy et al. 2004).

Level 3 or intensive assessments involve field collection of detailed, quantitative information on physical, chemical, and/or biological wetland parameters (U.S. Environmental Protection Agency 2002). These methods measure characteristics such as sedimentation rates, water quality, or biotic assemblages (e.g., vegetation or macroinvertebrates). Level 3 assessments often seek to quantify overall community condition by employing multi-metric indices, such as an index of biotic integrity. These indices aggregate individual metrics representing biological characteristics that respond predictably to a human disturbance gradient (Karr & Chu 1999).

Specific methods are generally constructed for particular wetland types or regions and transferability to other wetland systems or geographic areas often is limited. In the Dougherty Plain physiographic district of southwest Georgia, no existing methods have been constructed, calibrated, or evaluated for geographically isolated wetlands. A multi-scale, flexible, and standardized assessment protocol is needed for current and future assessments of wetland condition and quantification of ecological services relative to land use. The goals of the research presented here are to: 1) develop a preliminary three level ecological assessment framework for isolated depressional wetlands in the Dougherty Plain physiographic district of southwest Georgia; 2) determine the relationships among the three assessment levels; and 3) identify the most appropriate source of existing land cover data for calculating the Level 1 (LDI) assessment at a regional scale.

METHODS

Study Area

The Dougherty Plain is a karstic physiographic district located within the Coastal Plain of southwestern Georgia and covers 6,690 km² and parts of 14 counties (Fig. 4.1). It is positioned between the Pelham Escarpment to the south and east and the Fall-line Hills and Chattahoochee River to the north and west (Beck & Arden 1983). It has low topographic relief and ranges in elevation from 93 m in the northeast to 24 m in the southeast (Clark & Zisa 1976). The region is characterized by relatively broad, flat uplands, numerous isolated wetlands, and few but deeply incised streams. Isolated wetlands are common features of the landscape due to the dissolution of underlying limestone and the subsequent collapse of the overburden (Beck & Arden 1983). There are an estimated 11,620 isolated wetlands within the Dougherty Plain; ranging in size from <0.01 ha to 488.51 ha (Chapter 3, this volume). The isolated wetland population is dominated

by small wetlands with 50% of the of the 11,620 estimated wetlands measuring less than 1 ha and 81% measuring less than 4 ha in size (Chapter 3, this volume). The Dougherty Plain is a rural area and land cover is dominated by agriculture, natural forests, and pine plantations (Chapter 2, this volume).

Wetland Delineation

We selected forty isolated depressional wetlands within the Dougherty Plain for ecological assessment. Study sites were selected from a pool of isolated wetlands identified from aerial photographs and a recently developed map of isolated wetlands in the region (Chapter 3, this volume). Wetland selection was not random, but rather focused on selecting wetlands that spanned the regional disturbance gradient and were accessible. Additionally, the isolated wetland population in the Dougherty Plain is dominated by small (< 4-ha) wetlands (Chapter 3, this volume), and therefore we emphasized the selection of smaller wetlands. Wetland boundaries were delineated in the field during July and August of 2009 based on the presence of hydric soil indicators (Natural Resources Conservation Service 2006). We relied on hydric soils criteria to delineate wetlands because hydrophytic vegetation and indicators of wetland hydrology were frequently obscured in wetlands subject to frequent disturbance (e.g., wetlands embedded in an agricultural matrix). We recorded the boundary of each wetland with a TDS Nomad GPS unit (Tripod Data Systems, Westminster, CO, USA) operating with a Crescent A100 Smart Antenna (Hemisphere GPS, Calgary, AB, CAN) with estimated sub-meter horizontal accuracy. Wetlands ranged in area from 0.30 to 7.38 ha.

Level 1: Landscape Development Intensity (LDI) Index

Field-delineated wetland polygons were added as a vector layer to ArcMAP 9.2 (Environmental Systems Research Institute, Redlands, CA, USA) and a 100-m buffer was

created around each wetland polygon. Other studies have demonstrated 100 m as the optimal buffer distance for calculating LDI scores of isolated wetlands in Florida (Lane et al. 2003; Brown & Vivas 2005). Within the 100-m buffer, land cover was manually digitized from 2009 National Agriculture Imagery Program aerial photography at a constant 1:5,000 scale and with a 0.1-hectare minimum mapping unit. The classification scheme and LDI coefficients for manually digitized land cover data follows Reiss and Brown (2007) (Table 4.1). Additionally, we calculated LDI using existing land cover data from four sources: 2001 National Land Cover Data (NLCD) (1:24,000 scale) (U.S. Geological Survey 2007), 2005 Georgia Land Use Trends (GLUT) (1:100,000 scale) (Natural Resources Spatial Analysis Laboratory 2007), 1998 Georgia 18-class Landsat Landcover (GA18) (1:24,000 scale) (Natural Resource Spatial Analysis Laboratory 1998a), and 1998 Georgia 44-class Landsat Landcover (GA44) (1:24,000 scale) (Natural Resource Spatial Analysis Laboratory 1998b). We converted datasets from raster to polygon data and clipped them to the extent of the 100-m wetland buffers. We developed a crosswalk to match the land cover classes in each dataset with the most similar land cover class and LDI coefficient as reported by Reiss and Brown (2007) (Table 4.1). LDI index scores were calculated as follows:

$$LDI_{total} = \sum (\%LU_i * LDI_i)$$

where LDI_{total} is the LDI index value for an individual wetland, $\%LU_i$ is the percent of a land cover class i within the 100-m buffer zone, and LDI_i is the LDI coefficient for land cover class i (Table 4.1) (Brown & Vivas 2005; Reiss & Brown 2007).

Level 2: Rapid Wetland Assessment

We developed a qualitative rapid assessment procedure designed to assess the level of disturbance within and surrounding a wetland. The procedure was adapted from Fennessy et al.

(2004) and Miller and Gunsalus (1997), and consists of metrics representing five disturbance categories (hydrology, soil, vegetation, upland/wetland buffer, and presence of livestock). Each metric is rated on a scale of 0-1, where 1 indicates no evidence of alteration or degradation and 0 indicates extreme alteration or degradation. At each wetland, disturbance categories were assigned scores of 0.00, 0.33, 0.66, or 1.00.

Level 3: Intensive Macrophyte and Water Chemistry Sampling

Macrophytes were surveyed at randomly selected points within each wetland between August and November 2009. Random points were generated within each delineated wetland polygon using Hawth's Analysis Tools (Beyer 2004) in ArcGIS 9.2, and a minimum distance of 20 m was enforced between points. Each wetland was assigned a minimum of 5 sample points. Beyond this minimum the number of sample points was based on the area of the wetland polygon. For example the smallest wetland (0.29 ha) received 6 vegetation sample points, and the largest wetland (7.38 ha) received 24 vegetation sample points. A 1-m² sampling frame was centered on each point and the presence and abundance of all macrophyte species rooted within the sampling frame were recorded.

Eight macrophyte metrics were calculated for each wetland and included: species richness; % introduced species; % weedy species; % wetland status (obligate and facultative wet) species; annual to perennial ratio; % native perennial species; % sensitive species; and % tolerant species (Appendix A). Sensitive and tolerant indicator species were identified using Indicator Species Analysis (ISA) (PC-ORD 5, MjM Software Design, Gleneden Beach, Oregon, USA). ISA evaluates the abundance and faithfulness of occurrence of species in a defined group. Our methods specifically follow those described by Reiss (2004), but also see Dufrene & Legendre (1997) and McCune and Grace (2002). Sample wetlands were categorized based on consecutive

0.25-increments in Photo-LDI index values from 1.0 to 4.5 (e.g., 1.00, 1.25, 1.50, 1.75, etc.). We conducted ISA at each break using the presence and abundance of macrophyte species at each wetland. At each break, species categorized as tolerant were associated with wetlands having Photo-LDI index values greater than the break value, and species categorized as sensitive were associated with wetlands having Photo-LDI index values less than the break value.

We calculated the percent sensitive and tolerant indicator species at each wetland, and used Spearman's rank correlation (SigmaPlot 11, Systat Software Inc., San Jose, CA, USA) to correlate percentages with Photo-LDI values (Reiss 2004). Spearman's rank correlation coefficient (ρ) is a non-parametric measure of statistical dependence between two variables. ρ ranges between -1 and +1. A value of -1 indicates a perfect negative correlation of ranks between variables. A value of +1 indicates a perfect positive correlation of ranks between the variables. A value of 0 indicates no correlation in ranks between the variables. Based on the strength of correlation and the number of indicator species identified, we selected a single Photo-LDI break to represent sensitive species and another to represent tolerant species (Reiss 2004). The majority of the species encountered in the wetlands were neither sensitive nor tolerant indicators.

Water quality samples were collected from each wetland on 3-4 February, 29-30 March, and 24-25 May. Due to wetland dry-down, water samples were not collected for three wetlands in March and seven wetlands in May. On each sampling date, three 500-ml water samples were collected in acid washed and rinsed polyethylene bottles at each site and stored on ice until processing. Blanks (i.e., collections of laboratory DI water) were collected and processed as described for field collected samples. We measured six water quality parameters that we expected to respond to landscape disturbance. Within 24-48 hrs of sample collection, alkalinity

and pH were determined with a Mettler DL12 titrator using unfiltered, room temperature subsamples. Additional subsamples were filtered and $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and $\text{PO}_4\text{-P}$ were determined with a Lachat Quickchem 8000. The weight of organic content (ash free dry mass [AFDM]) was calculated from the filters. All measurements were taken in triplicates for each sample date, and the arithmetic mean of the triplicates was calculated. Because wetlands dried down over time, we calculated the arithmetic mean of all sample dates for each measurement at each wetland and used this value for analysis.

Wetland Condition Indices

Level 2 and Level 3 Wetland Condition Indices (WCIs) were constructed from metrics satisfying the three criteria listed below derived from Reiss & Brown (2007):

1. Display strong and significant correlation ($\rho > 0.50$, $P < 0.001$) with LDI using Spearman's rank correlation (SigmaPlot 11, Systat Software Inc., San Jose, CA, USA)
2. Correlations with LDI are visually distinguishable in scatter plots
3. Show a significant difference ($P < 0.001$) between low ($\text{LDI} < 2.0$) ($n = 20$) and high ($\text{LDI} \geq 2.00$) ($n = 20$) LDI wetland groups tested with the Mann-Whitney Rank Sum test (SigmaPlot 11, Systat Software Inc., San Jose, CA, USA).

We used the Photo-LDI for metric selection because it represents the finest spatial resolution of the available data sources, and its land cover categories are identical to those reported by Reiss and Brown (2007). For the Level 2 Rapid WCI (RWCI), each of the five disturbance metrics were tested with the three criteria listed above. Those disturbance metrics satisfying all of the above criteria were summed together and divided by the number of included metrics. The result was a single value ranging from 0.0 to 1.0; where 1.0 indicates that no disturbance was observed

and a score of 0.0 indicates an extreme level of disturbance. For the Level 3 Macrophyte WCI (MWCI) and Water WCI (WWCI), the values of each acceptable metric were normalized on a scale of 0.0 to 1.0 (with 1.0 representing minimally impacted wetland condition), and then these values were summed and divided by the number of acceptable metrics (Fore 2004). The correlation between the Photo-LDI index and each WCI was evaluated. To examine the relationship between assessment levels, we determined the correlation between the Level 2 RWCI and each Level 3 WCIs.

Comparison of Alternate LDI Source Data

To identify land cover data sources that are acceptable surrogates for the Photo-LDI at regional scales, we conducted Spearman's rank correlations between the Photo-LDI and all other calculated LDI indices (NLCD, GLUT, GA18, and GA44). Additionally, relationships between the additional LDI indices (NLCD, GLUT, GA18, and GA44) and WCIs from Level 2 and Level 3 were evaluated with Spearman's rank correlations.

RESULTS

Photo-LDI index values displayed strong and significant correlation with four of five Level 2 Rapid Wetland Assessment metrics (Table 4.2), and the same four metrics distinguished between the low ($LDI < 2$) and high ($LDI \geq 2$) LDI wetland groups ($p < 0.001$) (Table 4.3). The hydrology, soil, vegetation, and upland/wetland buffer metrics were used to construct the RWCI. Photo-LDI index values displayed strong and significant correlation with seven of nine Level 3 Macrophyte metrics (Table 4.2), and the same seven metrics distinguished between the low and high LDI wetland groups ($p < 0.001$) (Table 4.3). However, due to redundancy among some of the metrics we constructed the MWCI using the % Introduced, % Weedy, Annual:Perennial Ratio, Native Perennial, % Sensitive, and % Tolerant metrics (Appendix A). The Photo-LDI

index displayed strong and significant correlation with three of five water quality measurements (Table 4.2) and these measurements differentiated between low and high LDI wetland groups ($p < 0.001$) (Table 4.3). The WWCI was constructed using pH, alkalinity, and PO₄-P measurements.

The RWCI, MWCI, and WWCI displayed strong and significant negative correlation with the Level 1 Photo-LDI index (Table 4.4) (Fig. 4.2). Negative correlation values indicate inverse scaling of the assessment procedures. Possible LDI values range from 1 to 10, with 1 representing a minimally developed wetland buffer. Possible WCI values range from 0 to 1, with 1 representing a minimally disturbed wetland. The RWCI (Mann-Whitney U-test Statistic (U) = 23, $P < 0.001$), MWCI (U = 15, $P < 0.001$), and WWCI (U = 29, $P < 0.001$) were able to distinguish between the Low (< 2.00) LDI and High (≥ 2.00) LDI wetland groups (Fig. 4.3). Additionally, the Level 2 RWCI displayed strong and significant correlation with the Level 3 MWCI ($\rho = 0.87$, $P < 0.001$), and WWCI ($\rho = 0.75$, $P < 0.001$).

Each of the additional LDI indices were strongly and significantly correlated with the Photo-LDI: NLCD-LDI ($\rho = 0.77$, $P < 0.001$), GLUT-LDI ($\rho = 0.56$, $P = 0.001$), GA18-LDI ($\rho = 0.89$, $P < 0.001$), and GA44-LDI ($\rho = 0.90$, $P < 0.001$). The additional LDI indices were correlated with Level 2 and Level 3 WCIs to varying degrees (Table 4.4). These results indicate that the GA18-LDI and GA44-LDI are the best surrogates for the Photo-LDI index and currently the most suitable for use in future regional wetland assessments.

DISCUSSION

This study joins a growing body of research in demonstrating the applicability of a three-level wetland assessment procedures and in highlighting the suitability of the LDI index as a Level 1 measure of landscape disturbance and wetland condition (e.g., Brown & Vivas 2005;

Mack 2006; Vivas & Brown 2006; Reiss & Brown 2007). Further, this research establishes the framework necessary to pursue long term wetland assessment and monitoring goals in Georgia's Dougherty Plain physiographic district and serves as an example of how such procedures can be established in other areas.

Although we demonstrated that the Level 1 LDI index can be calculated from land cover data that are manually interpreted and digitized from aerial photographs, the effort required may not be practical for application at broader scales (e.g., physiographic districts, states, regions). Our comparison of LDI indices generated from multiple sources of existing land cover data identified two data sources, GA18-LDI and GA44-LDI, that while dated (1998), are suitable for broad scale isolated wetland condition assessments in the Dougherty Plain and which may prove to be useful in other regions of the state. Although the NLCD-LDI was not as strongly correlated with the other measurements, it is available at the national level.

Even though the GLUT data represents the most recent land cover data for the area, these data had the weakest correlation with other measurements; probably due to its coarser categorical resolution (13 land cover classes) and spatial resolution (1:100,000 scale). Although the older land cover data (NLCD, GA18 and GA44) performed effectively in the relatively rural study area, similarly aged data may be a poor choice in other areas undergoing rapid land cover changes.

We chose to use the land cover classification scheme and LDI coefficients developed for use in Florida by Brown & Vivas (2005). Calculating LDI coefficients for land cover classes is a time intensive endeavor requiring collection and analysis of energy consumption data (see Brown [1980], Whitfield [1994], Brandt-Williams [2002], and Brown & Vivas [2005]), and using preexisting coefficients increased the expediency of this research. Given our close

proximity to the state of Florida and because the LDI index is normalized between the most intensive and least intensive land uses, land cover types and LDI coefficients likely are transferable. Other researchers have used Brown and Vivas' (2005) LDI coefficients in various parts of the country, e.g., California (Stein et al. 2009), Indiana (Brittenham 2009), Ohio (Mack 2006; Mack et al. 2008). We caution that land cover classes and LDI coefficients calculated by Brown and Vivas (2005) may not be appropriate for use in landscapes characterized by land use patterns that are dissimilar from those found in Florida. We also note that emergy analysis, which forms the basis for the LDI technique, has drawn both criticism and praise (see Hau & Bakshi [2004] for a review), and suggest that additional research is necessary to fully assess the LDI as a Level 1 assessment procedure.

Further research in the Dougherty Plain will be necessary to apply the assessment framework developed in this study. Initial efforts should focus on verifying the multi-level agreement that we observed by using Level 1 or Level 2 index scores to develop prediction intervals for more time intensive measurements (i.e., Level 3). Additionally, any predictions will need to be verified through additional field sampling. Long term efforts should focus on refining and calibrating metrics and indices at each level of the assessment framework, with the eventual goal of using the Level 1 LDI assessment to estimate the condition of isolated wetlands throughout the Dougherty Plain. Finally, future studies should seek to identify and quantify relationships between Level 1-3 assessments and wetland functions, processes, or ecosystem services (Reiss & Brown 2007). Information linking land use with wetland condition information will help provide guidance in land use planning, prioritization of wetlands for habitat conservation, and development of best management practices to enhance ecosystem services and regulatory policies at local, state, and national levels.

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Table 4.1. Land cover classification scheme. 2009 Aerial Photo classes and LDI values follow Reiss and Brown (2007). Land cover classes from National Land Cover Dataset (NLCD), Georgia Land Use Trends (GLUT), Georgia 18-class Landsat (GA18), and Georgia 44-class Landsat (GA44) were crosswalked to the most equivalent 2009 aerial photo land cover class and LDI value. Land cover classes present in the 2009 aerial photo dataset did not necessarily have analogues in each of the satellite-derived datasets.

2009 Aerial Photo	NLCD	GLUT	GA18	GA44	LDI
Natural systems	Ice/Snow; Barren Land; Scrub/Shrub; Grassland/Herbaceous; Woody Wetlands; Emergent Herbaceous Wetlands; All Forest classes	Beach/Dune/Mud; Clearcut/Sparse; Quarries/Stripmines/Rock Outcrop; All Forest classes; All Wetland classes	Beaches/Dunes/Mud; Clearcut/Sparse; Rock Outcrop; All Forest classes; All Wetland/Marsh classes	Beaches/Dune/Mud; Coastal Dune; Clearcut/Sparse; Rock Outcrop; All Forest classes; All Wetland/Swamp/Marsh classes	1.00
Open water	Open Water	Open Water	Open Water	Open Water	1.00
Pine plantations					1.58
Recreational/open space - low					1.83
Woodland pasture					2.02
Improved pasture (without livestock)					2.77
Improved pasture - low (with livestock)	Pasture/Hay		Pasture	Pasture/Hay	3.41
Citrus/orchard					3.68
Improved pasture - high (with livestock)					3.74
Recreational/open space - medium			Utility Swath	Utility Swaths; Parks/Recreation	4.38
Row crops	Cultivated Crops	Row Crop/Pasture	Row Crop	Row Crop	4.54

Table 4.1. Continued.

2009 Aerial Photo	NLCD	GLUT	GA18	GA44	LDI
Single family - low	Developed - low	Urban - low	Urban - low	All low intensity Urban classes	6.79
Recreational/open space - high	Developed open space		Golf course	Golf Course	6.92
Single family - high	Developed - medium				7.55
Highway (≥ 2 lane)			Transportation	Transportation	7.81
Industrial			Quarries/Stripmines	Quarries/Stripmines	8.32
Business district (≥ 2 stories)	Developed - high	Urban - high	Urban - high	All high intensity Urban classes	9.42

Table 4.2. Correlations of metrics from assessment Levels 2 and 3 with LDI index values calculated from 2009 aerial photographs. Spearman's rank correlation coefficient (ρ) is a non-parametric measure of statistical dependence between two variables. ρ ranges between -1 and +1. A value of -1 indicates a perfect negative correlation of ranks between variables. A value of +1 indicates a perfect positive correlation of ranks between the variables. A value of 0 indicates no correlation in ranks between the variables.

Level	Metric	ρ	p-value
2	Hydrology	-0.634	<0.001
	Soil	-0.784	<0.001
	Vegetation	-0.695	<0.001
	Buffer	-0.775	<0.001
	Grazing	-0.318	=0.005
3	Species Richness	-0.413	=0.008
	% Introduced	0.822	<0.001
	% Native	-0.797	<0.001
	% Weedy	0.744	<0.001
	% Wetland Status ¹	-0.348	=0.028
	Ratio Annual:Perennial	0.705	<0.001
	% Native Perennial	-0.749	<0.001
	% Sensitive	-0.735	<0.001
	% Tolerant	0.792	<0.001
	pH	0.738	<0.001
	Alkalinity	0.796	<0.001
	NH ₄ -N	0.170	=0.291
	NO ₃ -N	0.313	=0.049
	PO ₄ -P	0.656	<0.001
	Ash Free Dry Mass	0.366	=0.020

¹ % Wetland Status is the percentage of macrophyte species characterized as obligate wetland or facultative wetland.

Table 4.3. Testing the ability of individual Level 2 and Level 3 metrics to differentiate between “low” ($\text{LDI} < 2.00$) ($n = 20$) and “high” ($\text{LDI} \geq 2.00$) ($n = 20$) wetland groups using Mann-Whitney U Tests. The “low” wetland group is considered to be minimally disturbed, and the “high” wetland group is considered to be impacted by disturbance in the surrounding landscape.

Level	Metric	Mean $\pm$ Standard Deviation		U ¹	p-value
		Low LDI	High LDI		
2	Hydrology	0.90 $\pm$ 0.24	0.52 $\pm$ 0.35	72.50	<0.001
	Soil	0.78 $\pm$ 0.25	0.38 $\pm$ 0.31	69.50	<0.001
	Vegetation	0.97 $\pm$ 0.15	0.37 $\pm$ 0.34	29.00	<0.001
	Buffer	0.65 $\pm$ 0.35	0.17 $\pm$ 0.28	61.00	<0.001
	Livestock	0.98 $\pm$ 0.07	0.75 $\pm$ 0.42	147.50	=0.033
3	Species Richness	26.95 $\pm$ 12.84	21.65 $\pm$ 8.05	149.50	=0.176
	% Introduced	0.02 $\pm$ 0.03	0.16 $\pm$ 0.16	22.50	<0.001
	% Native	0.96 $\pm$ 0.04	0.80 $\pm$ 0.15	31.00	<0.001
	% Weedy	0.14 $\pm$ 0.14	0.50 $\pm$ 0.25	37.50	<0.001
	Wetland Status ²	0.56 $\pm$ 0.23	0.44 $\pm$ 0.15	117.00	=0.026
	Annual:Perennial	0.13 $\pm$ 0.14	0.90 $\pm$ 0.96	47.00	<0.001
	% Native Perennial	0.84 $\pm$ 0.13	0.47 $\pm$ 0.24	28.00	<0.001
	% Sensitive	0.21 $\pm$ 0.14	0.03 $\pm$ 0.03	17.00	<0.001
	% Tolerant	0.01 $\pm$ 0.03	0.20 $\pm$ 0.20	29.00	<0.001
	pH	5.03 $\pm$ 0.53	6.12 $\pm$ 0.47	39.00	<0.001
	Alkalinity	5.40 $\pm$ 4.65	28.56 $\pm$ 26.19	16.00	<0.001
	NH ₄ -N	29.47 $\pm$ 56.02	81.52 $\pm$ 123.04	146.00	=0.148
	NO ₃ -N	19.77 $\pm$ 54.21	155.27 $\pm$ 597.81	160.50	=0.291
	PO ₄ -P	14.00 $\pm$ 41.85	95.92 $\pm$ 170.39	59.00	<0.001
	AFDM	6.01 $\pm$ 4.81	21.46 $\pm$ 34.82	127.00	=0.050

¹ Mann-Whitney U-Test statistic

² % Wetland Status is the percentage of macrophyte species characterized as obligate wetland or facultative wetland.

Table 4.4. Spearman's rank correlations of Wetland Condition Indices (WCIs) with LDI index values calculated from 2009 aerial photographs, 2005 Georgia Land Use Trends (GLUT) data, 2001 National Land Cover Data (NLCD), 1998 18-class Georgia land cover data (GA18), and 1998 44-class Georgia land cover data (GA44). Spearman's rank correlation coefficient (rho) is a non-parametric measure of statistical dependence between two variables. Rho ranges between -1 and +1. A value of -1 indicates a perfect negative Spearman correlation between variables. A value of +1 indicates a perfect positive Spearman correlation between the variables. A value of 0 indicates no Spearman correlation between the variables.

Level	Index	Photo LDI		GLUT LDI		NLCD LDI		GA18 LDI		GA44 LDI	
		rho	p	rho	p	rho	p	rho	p	rho	p
2	RWCI ^a	-0.850	<0.001	-0.567	<0.001	-0.751	<0.001	-0.830	<0.001	-0.833	<0.001
3	MWCI ^b	-0.854	<0.001	-0.554	<0.001	-0.717	<0.001	-0.848	<0.001	-0.854	<0.001
	WWCI ^c	-0.777	<0.001	-0.454	=0.003	-0.600	<0.001	-0.803	<0.001	-0.810	<0.001

a = Rapid Wetland Condition Index

b = Macrophyte Wetland Condition Index

c = Water Wetland Condition Index

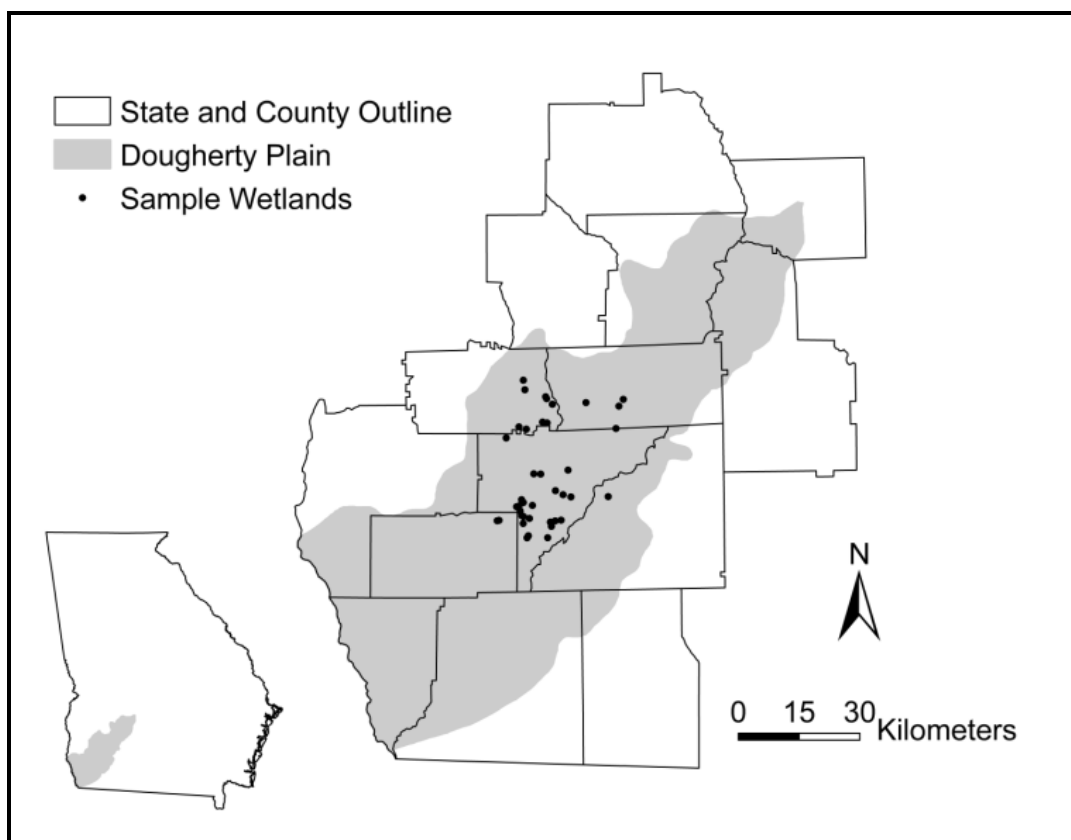


Figure 4.1. Map of the Dougherty Plain physiographic district in southwest Georgia and locations of wetlands study sites.

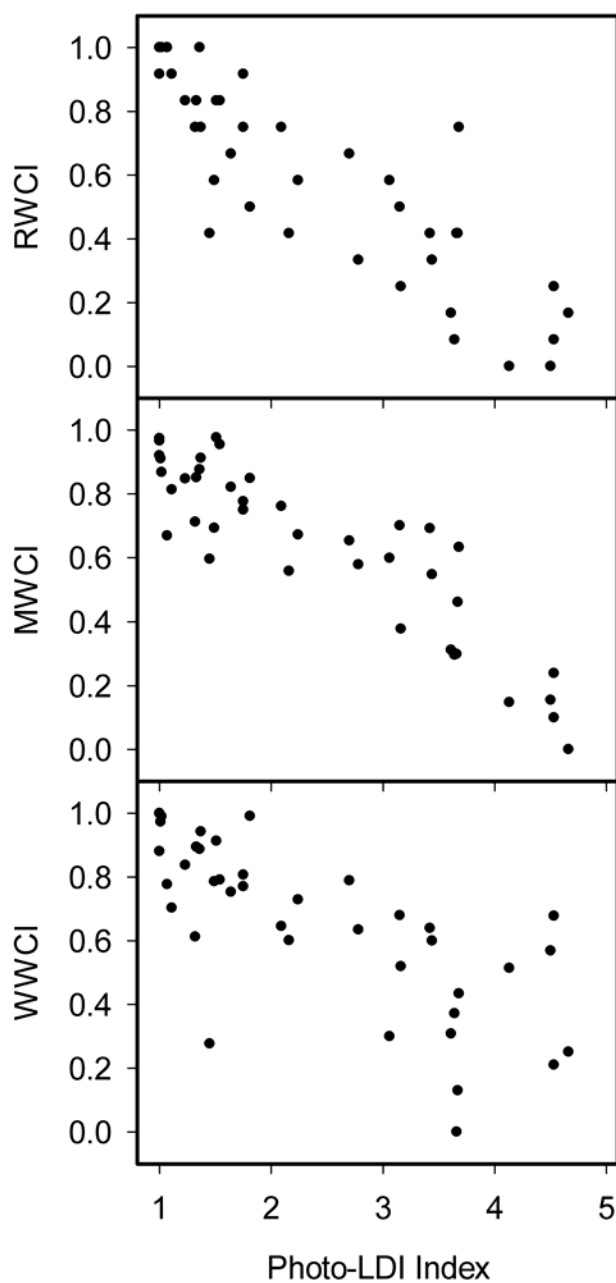


Figure 4.2. Relationship between Photo-LDI index, Rapid Wetland Condition Index (RWCI), Macrophyte Wetland Condition Index (MWCI), and Water Wetland Condition Index (WWCI) values of 40 study wetlands.

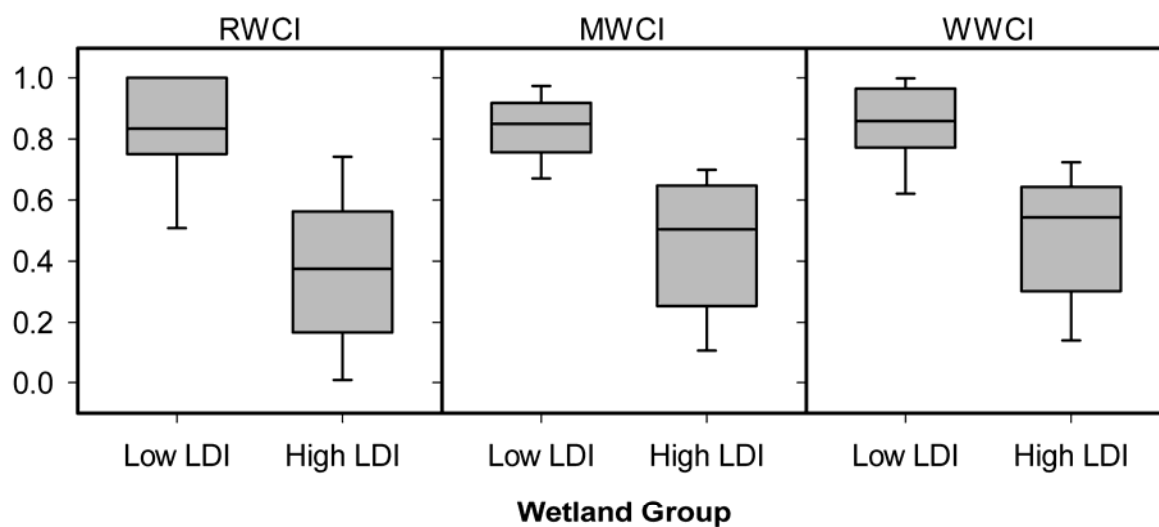


Figure 4.3. Median values and interquartile range for Rapid Wetland Condition Index (RWCI), Macrophyte Wetland Condition Index (MWCI), and Water Wetland Condition Index (WWCI) in Low (< 2.00) LDI and High (≥ 2.00) LDI wetland groups. Whisker bars represent the 10th and 90th percentiles.

APPENDIX A

Appendix A contains a series of tables detailing data collected and analyzed for each of the 40 sample wetlands.

Table A.1. Field delineated areas (hectare) of the 40 sample wetlands.

Table A-1. Wetland Area	
Wetland ID	Area (Hectares)
1	1.29
2	1.53
3	0.44
4	2.27
5	7.38
6	2.27
7	2.06
8	2.48
9	0.29
10	3.75
11	1.37
12	0.45
13	1.07
14	2.45
15	3.63
16	2.69
17	5.81
18	1.02
19	1.50
20	3.37
21	1.16
22	0.63
23	3.34
24	3.74
25	0.37
26	4.06
27	1.61
28	3.81
29	1.67
30	6.22
31	0.67
32	2.85
33	3.41
34	0.68
35	1.57
36	3.21
37	1.73

Table A-1. Wetland Area, continued	
Wetland ID	Area (Hectares)
38	0.60
39	1.22
40	1.00

Table A.2. Level 1 – Remote Assessment scores for each sample wetland. Landscape Development Intensity (LDI) index values were calculated from five different sources of land cover data: manually digitized from 2009 National Agriculture Imagery Program (NAIP) aerial photographs, 2005 Georgia Land Use Trends , 1998 Georgia 18-class Landsat Landcover (1998 GA18), 1998 Georgia 44-class Landsat Landcover (1998 GA44), and 2001 National Land Cover Data (2001 NLCD). See Chapter 4 text for citation information.

Table A-2. Level 1 - LDI Values					
Wetland ID	2009 NAIP	2005 GLUT	1998 GA18	1998 GA44	2001 NLCD
1	3.42	3.86	3.91	3.94	2.92
2	3.15	3.45	3.4	3.38	2.89
3	3.16	4.54	2.33	2.31	3.14
4	4.53	4.54	4.53	4.39	4.49
5	2.78	3.07	2.82	2.72	2.79
6	3.44	4.45	4.29	4.34	3.97
7	4.50	4.67	4.5	4.5	4.57
8	4.13	4.49	4.33	4.14	4.19
9	4.66	5.14	6.28	5.08	5.14
10	2.16	2.86	2.21	2.21	2.67
11	3.67	1.00	3.90	3.87	1.13
12	3.66	1.79	3.47	3.55	1.46
13	3.61	1.00	4.55	4.54	2.18
14	1.90	2.78	3.50	3.56	2.38
15	1.54	1.00	1.10	1.07	1.00
16	1.51	1.31	1.49	1.41	1.16
17	1.23	2.26	1.82	1.71	1.38
18	3.64	1.00	4.54	4.54	3.37
19	3.68	1.48	3.81	3.81	2.48
20	1.75	1.38	1.67	1.74	1.34
21	1.37	1.00	1.00	1.00	1.00
22	2.24	1.00	1.99	1.97	1.00
23	2.7	1.55	2.48	2.51	1.42
24	1.81	1.83	1.99	1.90	1.72
25	1.45	2.35	3.41	3.32	1.23
26	1.01	1.00	1.00	1.00	1.00
27	1.49	2.75	2.8	2.66	1.98
28	1.36	2.51	2.09	2.05	1.78
29	1.07	1.26	1.09	1.13	1.04
30	1.33	2.61	2.32	2.22	1.55
31	1.02	1.00	1.00	1.00	1.00

Table A-2. Level 1 - LDI Values, continued					
Wetland ID	2009 NAIP	2005 GLUT	1998 GA18	1998 GA44	2001 NLCD
32	1.00	1.00	1.00	1.00	1.00
33	1.00	1.00	1.00	1.00	1.00
34	1.00	1.00	1.00	1.00	1.00
35	4.53	4.74	4.57	4.44	2.82
36	3.06	4.26	4.11	3.85	1.53
37	1.64	2.95	2.36	2.29	2.17
38	2.09	2.94	2.77	2.68	2.71
39	1.11	1.55	1.18	1.21	1.14
40	1.32	2.07	1.69	1.74	1.52

Table A.3. Level 2 – Rapid Assessment scores for each sample wetland.

Table A-3. Level 2 - Rapid Assessment Scores					
Wetland ID	Hydrology	Soils	Vegetation	Buffer	Livestock
1	0.67	0.67	0.33	0.00	0.00
2	0.67	0.67	0.67	0.00	0.00
3	0.33	0.33	0.33	0.00	0.00
4	1.00	0.00	0.00	0.00	0.33
5	0.33	0.33	0.33	0.33	1.00
6	0.33	0.33	0.67	0.00	1.00
7	0.00	0.00	0.00	0.00	1.00
8	0.00	0.00	0.00	0.00	1.00
9	0.67	0.00	0.00	0.00	0.00
10	0.33	0.33	0.33	0.67	1.00
11	0.67	0.33	0.67	0.00	1.00
12	0.67	0.33	0.67	0.00	1.00
13	0.00	0.33	0.33	0.00	1.00
14	1.00	0.67	1.00	1.00	1.00
15	1.00	0.67	1.00	0.67	1.00
16	1.00	0.67	1.00	0.67	1.00
17	1.00	1.00	1.00	0.33	0.67
18	0.00	0.33	0.00	0.00	1.00
19	1.00	1.00	1.00	0.00	1.00
20	1.00	0.67	1.00	0.33	1.00
21	1.00	0.67	1.00	0.33	1.00
22	1.00	0.33	0.33	0.67	1.00
23	0.67	0.67	0.67	0.67	1.00
24	0.00	0.67	1.00	0.33	1.00
25	1.00	0.33	0.33	0.00	1.00
26	1.00	1.00	1.00	1.00	1.00
27	1.00	0.33	1.00	0.00	1.00
28	1.00	1.00	1.00	1.00	1.00
29	1.00	1.00	1.00	1.00	1.00
30	0.67	1.00	1.00	0.67	1.00
31	1.00	1.00	1.00	1.00	1.00
32	1.00	1.00	1.00	1.00	1.00
33	1.00	0.67	1.00	1.00	1.00
34	1.00	1.00	1.00	1.00	1.00
35	0.33	0.00	0.00	0.00	1.00
36	1.00	1.00	0.00	0.33	0.67

Table A-3. Level 2 - Rapid Assessment Scores, continued

Wetland ID	Hydrology	Soils	Vegetation	Buffer	Livestock
37	0.67	0.33	1.00	0.67	1.00
38	0.67	0.67	1.00	0.67	1.00
39	1.00	1.00	1.00	0.67	1.00
40	0.67	1.00	1.00	0.33	1.00

Table A.4. Species list and characteristics of macrophytes encountered in sample wetlands. The “Status” field indicates the Region 2 (Southeastern U.S.) wetland indicator status of each species (obligate wetland (OBL); facultative wetland = FACW; facultative = FAC; facultative upland = FACU; obligate upland = UPL; and no available Region 2 wetland status = X). The + and – modifiers indicate species more frequently found in wetlands (+) and less frequently found in wetlands (-). The “Native” field indicates whether a species is native or introduced. The “Weedy” field indicates whether a species is listed as a weed by the Southern Weed Science Society. The “Duration” field indicates the life span of a species: “A” = annual, “P” = perennial, “B” = biennial. The “Sensitive” field indicates those species we determined to be sensitive indicators at a Photo-LDI threshold of 1.75 using Indicator Species Analysis (ISA) (PC-ORD 5, MjM Software Design, Gleneden Beach, Oregon, USA). The “Tolerant” field indicates those species we determined to be tolerant indicators at a Photo-LDI threshold of 3.50 using ISA. Information for the “Status”, “Native”, “Weedy”, and “Duration” fields was obtained from <http://plants.usda.gov>. This appendix excludes 24 macrophytes that we were not able to identify to species.

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Acalypha gracilens</i>	X	Yes	No	A	No	No
<i>Acer rubrum</i>	FAC	Yes	No	P	No	No
<i>Agalinis harperi</i>	FAC+	Yes	No	A	No	No
<i>Allium canadense</i>	FACU-	Yes	Yes	P	No	No
<i>Amaranthus blitum</i>	X	No	No	A	No	No
<i>Amaranthus hybridus</i>	X	Yes	Yes	A	No	No
<i>Amaranthus spinosus</i>	FACU-	Yes	Yes	A	No	No
<i>Ambrosia artemisiifolia</i>	FACU	Yes	Yes	A	No	No
<i>Ammannia coccinea</i>	FACW+	Yes	Yes	A	No	Yes
<i>Ampelamus laevis</i>	FAC	Yes	No	P	No	No
<i>Ampelopsis arborea</i>	FAC+	Yes	Yes	P	No	No
<i>Amphicarpum muehlenbergianum</i>	FACW	Yes	No	P	No	No
<i>Anagallis arvensis</i>	FACU+	No	No	A/B	No	No
<i>Andropogon virginicus</i>	X	Yes	No	P	No	No
<i>Apios americana</i>	FACW	Yes	No	P	No	No
<i>Aristida palustris</i>	OBL	Yes	No	P	No	No
<i>Axonopus fissifolius</i>	FACW-	Yes	No	P	No	No
<i>Axonopus furcatus</i>	OBL	Yes	No	P	No	No
<i>Azolla caroliniana</i>	OBL	Yes	No	A	No	No
<i>Baccharis halimifolia</i>	FAC	Yes	No	P	No	No
<i>Bacopa caroliniana</i>	OBL	Yes	No	P	No	No
<i>Balduina uniflora</i>	FACW	Yes	No	P	No	No
<i>Bigelowia nudata</i>	FACW	Yes	No	P	No	No
<i>Boehmeria cylindrica</i>	FACW+	Yes	No	P	No	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Bulbostylis ciliatifolia</i>	FACU	Yes	No	A/P	No	No
<i>Callicarpa americana</i>	FACU-	Yes	No	P	No	No
<i>Campsis radican</i>	FAC	Yes	Yes	P	No	No
<i>Carex abscondita</i>	FACW	Yes	No	P	No	No
<i>Carex albolutescens</i>	FAC+	Yes	No	P	No	No
<i>Carex gigantea</i>	OBL	Yes	No	P	No	No
<i>Carex glaucescens</i>	OBL	Yes	No	P	Yes	No
<i>Carex verrucosa</i>	OBL	Yes	No	P	No	No
<i>Carya illinoensis</i>	FAC+	Yes	No	P	No	No
<i>Celtis laevigata</i>	FACW	Yes	No	P	No	No
<i>Centella asiatica</i>	FACW	Yes	No	P	No	No
<i>Cephalanthus occidentalis</i>	OBL	Yes	No	P	No	No
<i>Chamaesyce hirta</i>	X	Yes	No	A	No	No
<i>Cocculus carolinus</i>	FAC	Yes	Yes	P	No	No
<i>Coelorachis rugosa</i>	OBL	Yes	No	P	Yes	No
<i>Commelina communis</i>	FAC	No	Yes	A	No	No
<i>Cretagus aestivalis</i>	OBL	Yes	No	P	No	No
<i>Croton elliotii</i>	FACW+	Yes	No	A	Yes	No
<i>Ctenium aromaticum</i>	FACW	Yes	No	P	No	No
<i>Cuphea carthagenesis</i>	FACW	No	No	A/P	No	No
<i>Cyndon dactylon</i>	FACU	No	No	P	No	No
<i>Cyperus compressus</i>	FACW	Yes	No	A/P	No	Yes
<i>Cyperus croceus</i>	FAC	Yes	No	P	No	No
<i>Cyperus esculentus</i>	FAC	Yes	Yes	P	No	Yes
<i>Cyperus haspan</i>	OBL	Yes	No	P	No	No
<i>Cyperus iria</i>	FACW	No	Yes	A	No	Yes
<i>Cyperus lanceolatus</i>	FACW	Yes	No	P	No	No
<i>Cyperus polystachyos</i>	FACW	Yes	No	A/P	No	No
<i>Cyperus pseudovegetus</i>	FACW	Yes	No	P	No	No
<i>Cyperus retrorsus</i>	FACU+	Yes	No	P	No	No
<i>Cyperus rotundus</i>	FAC-	No	Yes	P	No	No
<i>Cyperus strigosus</i>	FACW	Yes	Yes	P	No	No
<i>Cyperus virens</i>	FACW	Yes	No	P	No	No
<i>Dactyloctenium aegyptium</i>	X	No	Yes	A	No	No
<i>Dichanthelium aciculare</i>	FACU	Yes	No	P	No	No
<i>Dichanthelium acuminatum</i>	FAC	Yes	No	P	No	No
<i>Dichanthelium erectifolium</i>	OBL	Yes	No	P	No	No
<i>Dichanthelium leucothrix</i>	X	Yes	No	P	Yes	No
<i>Dichanthelium ovale</i>	FACU	Yes	No	P	No	No
<i>Dichanthelium sphaerocarpon</i>	FACU	Yes	No	P	No	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Dichanthelium strigosum</i>	FAC	Yes	No	P	No	No
<i>Digitaria bicornis</i>	X	Yes	Yes	P	No	Yes
<i>Digitaria ciliaris</i>	X	Yes	Yes	A	No	Yes
<i>Digitaria serotina</i>	FAC	Yes	No	A	No	No
<i>Diodia teres</i>	FACU-	Yes	Yes	A/P	No	No
<i>Diodia virginiana</i>	FACW	Yes	Yes	A/P	No	No
<i>Diospyros virginiana</i>	FAC	Yes	No	P	Yes	No
<i>Drosera brevifolia</i>	OBL	Yes	No	A/P	No	No
<i>Dyschoriste oblongifolia</i>	X	Yes	No	P	No	No
<i>Echinochloa colona</i>	FACW	No	Yes	A	No	Yes
<i>Echinodorus tenellus</i>	OBL	Yes	No	P	No	No
<i>Eclipta prostrata</i>	FACW-	Yes	Yes	A/P	No	No
<i>Eleocharis acicularis</i>	OBL	Yes	No	A/P	No	No
<i>Eleocharis equisetoides</i>	OBL	Yes	No	P	No	No
<i>Eleocharis geniculata</i>	FACW+	Yes	No	A	No	No
<i>Eleocharis melanocarpa</i>	FACW	Yes	No	P	No	No
<i>Eleocharis minima</i>	OBL	Yes	No	A	No	No
<i>Eleocharis obtusa</i>	OBL	Yes	No	A/P	No	No
<i>Eleocharis tricostata</i>	FACW+	Yes	No	P	No	No
<i>Eleusine indica</i>	FACU	No	Yes	A	No	No
<i>Eragrostis hirsuta</i>	FACU	Yes	No	P	No	No
<i>Eragrostis virginica</i>	FACW	Yes	No	P	No	No
<i>Erechtites hieracifolius</i>	FAC-	Yes	No	A	No	No
<i>Erigeron vernus</i>	OBL	Yes	No	P	No	No
<i>Eriocaulon compressum</i>	OBL	Yes	No	P	No	No
<i>Eryngium prostratum</i>	FACW	Yes	No	P	No	No
<i>Eupatorium capillifolium</i>	FACU	Yes	Yes	P	No	No
<i>Eupatorium compositifolium</i>	FAC-	Yes	No	P	No	No
<i>Eupatorium hyssopifolium</i>	X	Yes	No	P	No	No
<i>Eupatorium leptophyllum</i>	FAC+	Yes	No	P	No	No
<i>Eupatorium leucolepis</i>	FACW+	Yes	No	P	No	No
<i>Eupatorium mohrii</i>	FACW-	Yes	No	P	No	No
<i>Eupatorium pilosum</i>	FACW	Yes	No	P	No	No
<i>Euthamia caroliniana</i>	FACU	Yes	No	P	No	No
<i>Fimbristylis autumnalis</i>	OBL	Yes	No	A	No	No
<i>Fimbristylis puberula</i>	OBL	Yes	No	P	No	No
<i>Fimbristylis vahlii</i>	OBL	Yes	No	A	No	No
<i>Fraxinus pennsylvanica</i>	FACW	Yes	No	P	No	No
<i>Fuirena squarrosa</i>	OBL	Yes	No	P	No	No
<i>Gamochaeta purpurea</i>	UPL	Yes	No	A/B	No	No
<i>Gelsemium sempervirens</i>	FAC	Yes	No	P	No	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Geranium carolinianum</i>	X	Yes	Yes	A/B	No	No
<i>Glycine max</i>	X	No	Yes	A/B	No	No
<i>Gratiola pilosa</i>	FACW-	Yes	No	P	No	No
<i>Gratiola ramosa</i>	FACW	Yes	No	P	No	No
<i>Hamamelis virginiana</i>	FACU	Yes	No	P	No	No
<i>Helenium pinnatifidum</i>	OBL	Yes	No	P	No	No
<i>Helianthus angustifolius</i>	FAC+	Yes	No	P	No	No
<i>Hypericum brachyphyllum</i>	FACW	Yes	No	P	No	No
<i>Hypericum crux-andreae</i>	FACW-	Yes	No	P	No	No
<i>Hypericum harperi</i>	X	Yes	No	P	No	No
<i>Hypoxis juncea</i>	FACW-	Yes	No	P	No	No
<i>Hyptis alata</i>	OBL	Yes	No	P	No	No
<i>Ilex cassine</i>	FACW	Yes	No	P	No	No
<i>Ilex coriacea</i>	FACW	Yes	No	P	No	No
<i>Ilex glabra</i>	FACW	Yes	No	P	No	No
<i>Ilex opaca</i>	FAC-	Yes	No	P	No	No
<i>Ipomea hederacea</i>	FAC-	No	Yes	A	No	No
<i>Ipomea quamoclit</i>	FACU+	No	Yes	A	No	No
<i>Iva microcephala</i>	FACW	Yes	No	A	No	No
<i>Jacquemontia tamnifolia</i>	FACU-	Yes	Yes	A	No	No
<i>Juncus effusus</i>	FACW+	Yes	No	P	No	No
<i>Juncus elliotii</i>	OBL	Yes	No	P	No	No
<i>Juncus repens</i>	OBL	Yes	No	A/P	No	No
<i>Juncus tenuis</i>	FAC	Yes	No	P	No	No
<i>Justicia ovata</i>	OBL	Yes	No	P	No	No
<i>Lachnanthes caroliniana</i>	OBL	Yes	No	P	No	No
<i>Lachnocaulon anceps</i>	OBL	Yes	No	P	No	No
<i>Leersia hexandra</i>	OBL	Yes	No	P	Yes	No
<i>Lemna minor</i>	OBL	Yes	No	P	No	No
<i>Leucothoe racemosa</i>	FACW	Yes	No	P	No	No
<i>Ligustrum sinense</i>	FAC	No	Yes	P	No	No
<i>Liquidambar styraciflua</i>	FAC+	Yes	No	P	No	No
<i>Ludwigia decurrens</i>	OBL	Yes	Yes	P	No	No
<i>Ludwigia glandulosa</i>	OBL	Yes	No	P	No	No
<i>Ludwigia hirtella</i>	FACW+	Yes	No	P	No	No
<i>Ludwigia linearis</i>	OBL	Yes	No	P	No	No
<i>Ludwigia linifolia</i>	OBL	Yes	No	P	No	No
<i>Ludwigia microcarpa</i>	OBL	Yes	No	P	No	No
<i>Ludwigia palustris</i>	OBL	Yes	No	P	No	No
<i>Ludwigia pilosa</i>	OBL	Yes	No	P	No	No
<i>Ludwigia repens</i>	OBL	Yes	No	P	No	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Ludwigia suffruticosa</i>	OBL	Yes	No	P	No	No
<i>Ludwigia virgata</i>	OBL	Yes	No	P	No	No
<i>Lycopus rubellus</i>	OBL	Yes	No	P	No	No
<i>Lygodium japonicum</i>	FAC	No	Yes	P	No	No
<i>Mecardonia acuminata</i>	OBL	Yes	No	P	No	No
<i>Melochia corchorifolia</i>	FAC	No	Yes	A	No	No
<i>Mitreola angustifolia</i>	FACW+	Yes	No	A	No	No
<i>Modiola caroliniana</i>	FACU+	Yes	No	A/B/P	No	No
<i>Mollugo verticillata</i>	FAC	Yes	Yes	A	No	Yes
<i>Myrica cerifera</i>	FAC+	Yes	No	P	No	No
<i>Nyssa sylvatica</i>	FAC	Yes	No	P	No	No
<i>Oldenlandia uniflora</i>	FACW-	Yes	No	A	No	No
<i>Oxalis corniculata</i>	FACU	Yes	Yes	A/P	No	No
<i>Oxalis rubra</i>	X	No	No	P	No	No
<i>Panicum anceps</i>	FAC-	Yes	No	P	No	No
<i>Panicum dichotomiflorum</i>	FACW	Yes	Yes	A	No	No
<i>Panicum hemitomom</i>	OBL	Yes	No	P	Yes	No
<i>Panicum hians</i>	OBL	Yes	No	P	No	No
<i>Panicum rigidulum</i>	FACW	Yes	No	P	No	No
<i>Panicum tenerum</i>	FACW	Yes	No	P	No	No
<i>Panicum verrucosum</i>	FACW	Yes	No	A	No	No
<i>Panicum virgatum</i>	FAC+	Yes	Yes	P	No	No
<i>Paspalum boscianum</i>	FACW	Yes	No	A	No	No
<i>Paspalum floridanum</i>	FACW-	Yes	No	P	No	No
<i>Paspalum notatum</i>	FACU+	Yes	Yes	P	No	No
<i>Paspalum plicatulum</i>	FAC	Yes	No	P	No	No
<i>Paspalum setaceum</i>	FAC	Yes	No	P	No	No
<i>Paspalum urvillei</i>	FAC	No	Yes	P	No	No
<i>Passiflora incarnata</i>	X	Yes	Yes	P	No	No
<i>Phyllanthus caroliniensis</i>	FAC+	Yes	No	A	No	No
<i>Phyllanthus urinaria</i>	FAC	No	Yes	A	No	No
<i>Physalis angulata</i>	FAC	Yes	Yes	A	No	No
<i>Phytolacca americana</i>	FACU+	Yes	Yes	P	No	No
<i>Pinus elliotii</i>	FACW	Yes	No	P	No	No
<i>Pinus palustris</i>	FACU+	Yes	No	P	No	No
<i>Pinus taeda</i>	FAC	Yes	No	P	No	No
<i>Piriqueta cistoides</i>	X	Yes	No	P	No	No
<i>Pityopsis aspera</i>	X	Yes	No	P	No	No
<i>Pityopsis graminifolia</i>	UPL	Yes	No	P	No	No
<i>Pluchea camphorata</i>	FACW	Yes	No	A/P	No	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Pluchea rosea</i>	FACW	Yes	No	P	No	No
<i>Polygala cymosa</i>	OBL	Yes	No	B	No	No
<i>Polygala lutea</i>	FACW+	Yes	No	B	No	No
<i>Polygala nana</i>	FAC+	Yes	No	A/P	No	No
<i>Polygonum hydropiperoides</i>	OBL	Yes	No	P	No	No
<i>Polygonum lapathifolium</i>	FACW	Yes	Yes	A	No	Yes
<i>Polygonum pensylvanicum</i>	FACW	Yes	Yes	A	No	No
<i>Polypremum procumbens</i>	FACU-	Yes	No	A/P	No	No
<i>Proserpinaca palustris</i>	OBL	Yes	No	P	No	No
<i>Proserpinaca pectinata</i>	OBL	Yes	No	P	No	No
<i>Quercus laurifolia</i>	FACW	Yes	No	P	Yes	No
<i>Quercus nigra</i>	FAC	Yes	No	P	No	No
<i>Quercus phellos</i>	FACW-	Yes	No	P	No	No
<i>Quercus virginiana</i>	FACU+	Yes	No	P	No	No
<i>Rhexia alifanus</i>	FACW	Yes	No	P	No	No
<i>Rhexia aristosa</i>	OBL	Yes	No	P	No	No
<i>Rhexia cubensis</i>	FACW+	Yes	No	P	No	No
<i>Rhexia mariana</i>	FACW+	Yes	No	P	No	No
<i>Rhexia nuttallii</i>	FACW+	Yes	No	P	No	No
<i>Rhexia petiolata</i>	FACW+	Yes	No	P	No	No
<i>Rhexia virginica</i>	FACW+	Yes	No	P	Yes	No
<i>Rhus copallinum</i>	X	Yes	No	P	No	No
<i>Rhynchospora cephalantha</i>	OBL	Yes	No	P	No	No
<i>Rhynchospora corniculata</i>	OBL	Yes	Yes	P	Yes	No
<i>Rhynchospora fascicularis</i>	FACW+	Yes	No	P	No	No
<i>Rhynchospora filifolia</i>	FACW-	Yes	No	P	Yes	No
<i>Rhynchospora globularis</i>	FACW	Yes	No	A/P	No	No
<i>Rhynchospora inundata</i>	OBL	Yes	No	P	No	No
<i>Rhynchospora microcarpa</i>	FACW+	Yes	No	P	No	No
<i>Rhynchospora perplexa</i>	OBL	Yes	No	P	No	No
<i>Rhynchospora tracyi</i>	OBL	Yes	No	P	No	No
<i>Rosa multiflora</i>	UPL	No	Yes	P	No	No
<i>Rotala ramosior</i>	OBL	Yes	No	A	No	No
<i>Rubus argutus</i>	FACU+	Yes	Yes	P	No	No
<i>Rubus cuneifolius</i>	FACU	Yes	No	P	Yes	No
<i>Rubus trivialis</i>	FAC	Yes	No	P	No	No
<i>Rudbeckia mohrii</i>	FACW+	Yes	No	P	No	No
<i>Rumex crispus</i>	FAC	No	Yes	P	No	No
<i>Saccharum coarctatum</i>	X	Yes	No	P	No	No
<i>Saccharum giganteum</i>	FACW	Yes	No	P	Yes	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Sagittaria graminea</i>	OBL	Yes	Yes	P	No	No
<i>Salix nigra</i>	OBL	Yes	No	P	No	No
<i>Sassafras albidum</i>	FACU	Yes	No	P	No	No
<i>Scirpus cyperinus</i>	OBL	Yes	No	P	No	No
<i>Scleria ciliata</i>	FAC	Yes	No	P	No	No
<i>Scleria georgiana</i>	FACW	Yes	No	P	No	No
<i>Scleria reticularis</i>	OBL	Yes	No	P	No	No
<i>Sclerolepis uniflora</i>	OBL	Yes	No	P	No	No
<i>Senna obtusifolia</i>	X	Yes	Yes	A/P	No	No
<i>Sesbania herbacea</i>	FACW	Yes	Yes	A/P	No	Yes
<i>Setaria parvifolia</i>	FAC	Yes	Yes	P	No	No
<i>Sicyos angulatus</i>	FACW-	Yes	Yes	A	No	No
<i>Sida rhombifolia</i>	FACU	Yes	Yes	A/P	No	No
<i>Sida spinosa</i>	FACU	Yes	Yes	A/P	No	No
<i>Smilax bona-nox</i>	FAC	Yes	No	P	No	No
<i>Smilax glauca</i>	FAC	Yes	No	P	No	No
<i>Smilax rotundifolia</i>	FAC	Yes	No	P	No	No
<i>Smilax tamnoides</i>	FAC+	Yes	No	P	No	No
<i>Solanum carolinense</i>	FACU	Yes	Yes	P	No	No
<i>Solidago canadensis</i>	FACU+	Yes	Yes	P	No	No
<i>Solidago leavenworthii</i>	FAC+	Yes	No	P	No	No
<i>Solidago stricta</i>	OBL	Yes	No	P	No	No
<i>Sporobolus floridanus</i>	FAC	Yes	No	P	Yes	No
<i>Stellaria media</i>	FACU	No	Yes	A/P	No	No
<i>Stylisma aquatica</i>	FACW+	Yes	No	P	No	No
<i>Styrax americanus</i>	FACW	Yes	No	P	No	No
<i>Symphotrichum dumosum</i>	FAC	Yes	No	P	No	No
<i>Syngonanthus flavidulus</i>	FACW+	Yes	No	P	No	No
<i>Taxodium ascendens</i>	X	Yes	No	P	Yes	No
<i>Toxicodendron radicans</i>	FAC	Yes	Yes	P	No	No
<i>Triadenum walteri</i>	OBL	Yes	No	P	No	No
<i>Tridens ambiguus</i>	FACW+	Yes	No	P	No	No
<i>Ulmus americana</i>	FACW	Yes	No	P	No	No
<i>Urochloa platyphylla</i>	FAC+	Yes	Yes	A	No	No
<i>Utricularia cornuta</i>	OBL	Yes	No	A/P	No	No
<i>Vaccinium arboreum</i>	FACU	Yes	No	P	No	No
<i>Verbena bonariensis</i>	FAC+	No	No	A/B/P	No	No
<i>Verbena brasiliensis</i>	FAC-	No	No	A	No	Yes
<i>Vicia acutifolia</i>	FACW+	Yes	No	P	No	No
<i>Viola arvensis</i>	X	No	Yes	A	No	No

Table A-4. Species Characteristics, continued

Species	Status	Native	Weedy	Duration	Sensitive	Tolerant
<i>Viola lanceolata</i>	OBL	Yes	No	P	Yes	No
<i>Vitis rotundifolia</i>	FAC	Yes	Yes	P	No	No
<i>Woodwardia virginica</i>	OBL	Yes	No	P	No	No
<i>Xanthium strumarium</i>	FAC	Yes	Yes	P	No	No
<i>Xyris ambigua</i>	OBL	Yes	No	P	No	No
<i>Xyris jupicai</i>	OBL	Yes	No	P	No	No

Table A.5. Level 3 – Macrophyte metric values for each sample wetland. Values were calculated for the following metrics: species richness, % introduced species, % weedy species, % wetland status species (obligate and facultative wet), annual to perennial ratio (A:P), % native perennial species (% Native P.) % sensitive species, and % tolerant species. See Table A-3 for additional information.

Wetland ID	Sp. Richness	% Introduced	% Weedy	% Wetland Status	A:P	% Native P.	% Sensitive	% Tolerant
1	28	7.14%	17.86%	53.57%	22.22%	64.29%	3.57%	0.00%
2	27	7.41%	18.52%	62.96%	33.33%	66.67%	11.11%	3.70%
3	22	27.27%	59.09%	45.45%	41.67%	50.00%	0.00%	13.64%
4	8	12.50%	100.00%	62.50%	267.00%	0.00%	0.00%	75.00%
5	34	8.82%	35.29%	50.00%	35.00%	55.88%	5.88%	11.76%
6	27	7.41%	48.15%	29.63%	28.57%	51.85%	3.70%	11.11%
7	16	6.25%	68.75%	37.50%	266.67%	12.50%	0.00%	37.50%
8	19	21.05%	73.68%	47.37%	180.00%	26.32%	0.00%	36.84%
9	3	66.67%	100.00%	66.67%	300.00%	0.00%	0.00%	66.67%
10	33	6.06%	42.42%	30.30%	38.89%	54.55%	6.06%	15.15%
11	20	20.00%	45.00%	50.00%	41.67%	60.00%	0.00%	15.00%
12	15	13.33%	66.67%	46.67%	120.00%	33.33%	0.00%	26.67%
13	23	30.43%	56.52%	43.48%	53.85%	43.48%	0.00%	21.74%
14	33	3.03%	18.18%	60.61%	19.23%	78.79%	12.12%	0.00%
15	26	0.00%	15.38%	69.23%	0.00%	92.31%	42.31%	0.00%
16	8	0.00%	12.50%	50.00%	0.00%	100.00%	50.00%	0.00%
17	32	0.00%	0.00%	75.00%	20.83%	75.00%	18.75%	0.00%
18	14	42.86%	57.14%	42.86%	50.00%	35.71%	0.00%	21.43%
19	22	13.64%	40.91%	31.82%	4.76%	81.82%	4.55%	4.55%
20	11	0.00%	36.36%	18.18%	11.11%	81.82%	9.09%	0.00%
21	5	0.00%	0.00%	80.00%	0.00%	100.00%	20.00%	0.00%
22	23	4.35%	30.43%	39.13%	23.53%	69.57%	4.35%	4.35%

Table A-5. Level 3 - Macrophyte Metric Values, continued

Wetland ID	Sp. Richness	% Introduced	% Weedy	% Wetland Status	A:P	% Native P.	% Sensitive	% Tolerant
23	22	4.55%	27.27%	40.91%	42.86%	63.64%	9.09%	9.09%
24	29	0.00%	3.45%	58.62%	16.00%	86.21%	13.79%	0.00%
25	14	7.14%	57.14%	7.14%	44.44%	92.86%	7.14%	14.29%
26	33	0.00%	3.03%	51.52%	6.90%	100.00%	21.21%	0.00%
27	21	9.52%	19.05%	52.38%	41.67%	57.14%	14.29%	0.00%
28	48	0.00%	2.08%	83.33%	4.65%	89.58%	16.67%	0.00%
29	27	3.70%	25.93%	29.63%	33.33%	55.56%	3.70%	0.00%
30	41	2.44%	2.44%	82.93%	5.88%	82.93%	17.07%	0.00%
31	45	0.00%	4.44%	66.67%	7.69%	86.67%	17.78%	0.00%
32	44	0.00%	2.27%	68.18%	5.00%	90.91%	27.27%	0.00%
33	21	0.00%	9.52%	61.90%	0.00%	95.24%	42.86%	0.00%
34	27	0.00%	3.70%	81.48%	4.17%	88.89%	40.74%	0.00%
35	17	17.65%	58.82%	5.88%	200.00%	23.53%	0.00%	23.53%
36	28	3.57%	46.43%	28.57%	26.32%	64.29%	0.00%	7.14%
37	39	5.13%	17.95%	58.97%	9.09%	82.05%	23.08%	0.00%
38	32	3.13%	9.38%	65.63%	15.38%	78.13%	3.13%	0.00%
39	10	0.00%	20.00%	10.00%	0.00%	90.00%	10.00%	0.00%
40	25	0.00%	20.00%	44.00%	33.33%	60.00%	4.00%	0.00%

Table A-6. Level 3 – Water quality measurement values for each sample wetland. TDM = total dry mass. AFDM = ash free dry mass.

Table A-6. Water Quality Measurements						
Wetland ID	pH	Alkalinity	PO4-P	NH4-N	NO3-N	AFDM
1	5.72	19.19	25.93	16.21	17.92	14.24
2	5.68	15.28	23.62	4.08	20.63	11.56
3	5.93	23.49	57.22	28.24	12.97	56.25
4	6.45	41.09	110.78	17.39	15.55	31.56
5	5.80	12.44	50.30	10.79	2.57	38.03
6	5.84	14.94	55.25	12.61	10.24	4.56
7	5.99	19.83	38.71	14.99	3.11	6.44
8	5.95	15.56	91.28	13.55	2.33	6.58
9	7.47	22.28	146.36	16.39	10.41	150.89
10	5.93	21.58	18.64	20.28	247.87	3.40
11	6.44	44.92	141.93	13.51	6.40	2.21
12	6.93	121.83	221.92	265.37	26.18	10.13
13	6.65	65.30	16.40	16.01	5.41	5.09
14	5.78	6.33	0.79	10.90	0.00	2.71
15	5.63	6.11	2.86	19.37	2.45	4.41
16	5.04	2.74	2.61	22.14	0.78	8.99
17	5.48	3.88	0.38	16.80	1.39	3.73
18	6.34	43.02	21.80	40.93	7.42	5.64
19	5.97	24.51	98.11	13.79	0.90	9.77
20	5.56	6.06	1.24	15.98	0.87	2.43
21	4.89	2.22	2.21	25.61	2.44	2.29
22	5.82	10.94	1.20	151.65	8.23	3.15
23	5.62	7.22	0.84	446.44	12.25	1.52
24	4.68	0.54	0.82	10.93	66.51	5.36
25	6.25	17.85	188.28	354.66	53.34	3.33
26	4.63	3.62	3.14	12.71	34.39	2.66
27	5.56	8.41	3.27	16.80	9.02	3.95
28	5.20	3.19	0.44	28.11	18.25	5.37
29	5.62	5.66	14.41	131.03	12.38	24.30
30	5.15	2.62	3.41	22.04	0.00	6.60
31	4.69	0.52	0.72	1.66	5.11	4.19
32	4.56	0.60	0.91	7.23	154.64	5.84
33	4.46	0.26	1.33	100.01	2,690.50	3.90
34	5.18	4.53	0.89	21.39	19.84	6.76
35	5.85	12.49	20.11	199.44	3.77	55.33

Table A-6. Water Quality Measurements, continued

Wetland ID	pH	Alkalinity	PO4-P	NH4-N	NO3-N	AFDM
36	6.09	17.92	775.56	13.85	5.43	5.00
37	5.71	7.90	9.69	15.73	3.55	7.36
38	6.03	17.33	2.40	27.51	0.74	4.93
39	5.92	11.04	5.74	34.76	1.37	5.82
40	6.01	13.89	36.89	8.76	3.59	10.11

CHAPTER 5

SUMMARY

Geographically isolated wetlands are common and important features in the landscape of the Dougherty Plain physiographic district of southwestern Georgia. Isolated wetlands are particularly vulnerable to anthropogenic disturbance and historically have been subjected to widespread modification (Bennett & Nelson 1991; Brinson & Malvarez 2002; Tiner 2003). Currently, geographically isolated wetlands are not subject to federal regulation under the Clean Water Act unless they are adjacent or connected to jurisdictional waters via a “significant nexus” (Leibowitz et al. 2008; Sponberg 2009). In the absence of federal regulation, isolated wetland policy is established by individual states, which are often unprepared or unable to manage these wetlands (Christie & Hausmann 2003). The inability or unwillingness of states to protect this valuable resource may be due, in part, to a lack of information. In Georgia, basic information on which to base a regulatory framework (e.g., the number, location, spatial extent, and condition of isolated wetlands) is generally lacking, and the available data are often contradictory (Georgia Department of Natural Resources 1999; Brown 2002). This thesis has taken a first step toward filling this information gap, at least in the Dougherty Plain.

In Chapter 2, I demonstrated that land use and land cover (LULC) patterns changed drastically within the Dougherty Plain over the past 60 years. Specifically, natural forests and unirrigated agriculture sharply declined in area while planted pine forests and irrigated agriculture increased. Additionally, I found that the landscape became more fragmented (a greater number of patches) and uniform (patches became more similar in size) over time. In

Chapter 3, I created an improved map of isolated wetlands by combining existing wetland maps with readily available spatial data sources (i.e., soils and elevation) to accurately locate and estimate the extent of isolated wetlands within the Dougherty Plain. In Chapter 4, I developed a multi-metric framework to assess the condition of isolated wetlands, and applied it to 40 isolated wetlands. I demonstrated strong correlation between assessment methods at different scales, which suggests that the Landscape Development Intensity (LDI) index (the Level 1, remote assessment, calculated from remotely sensed data) may be appropriate for the regional assessment of isolated wetland condition. Additionally, I identified existing LULC data which, when used to calculate the LDI, were highly correlated with finer-detail land use maps interpreted from aerial photography. Using existing LULC data will reduce the effort and cost of future remote wetland assessments in the region.

The results of the preceding chapters, and the spatial datasets created in the process, form the foundation for more intensive and applied research on the isolated wetland resource within the Dougherty Plain. For starters, I suggest combining the isolated wetland data layer created in Chapter 3 with the remote LDI assessment procedure used in Chapter 4. This, in conjunction with existing land cover data, will allow rapid quantification of the landscape development intensity surrounding each mapped wetland across the entire Dougherty Plain. The LDI values could be used to predict the result of more intensive wetland assessments (e.g., vegetation or water quality analyses), and the predictions could be tested at selected wetlands throughout the Dougherty Plain.

In Chapter 4 I demonstrated that current LULC surrounding a wetland was strongly correlated with both rapid and intensive measurements of wetland condition. However, past

LULC may also contribute to the current condition of ecosystems. The concept of LULC history or legacy suggests that previous land use activities influence the current and future structure and functions of ecosystems, and this concept has been applied in multiple ecological and environmental history studies (Foster et al. 2003; Franklin et al. 2004; Ruiz & Domon 2009). However, relatively little research has explored the importance of past LULC on the condition of aquatic or wetland systems.

Limited research indicates that historical LULC may be an important influence in aquatic ecosystems, including depressional wetlands. For example, invertebrate and fish communities in southern Appalachian streams were best predicted by 1950's LULC, while LULC in the 1990's was a comparatively poor indicator (Harding et al. 1998). A study of historical land use and vegetation patterns in 299 Carolina bays and bay-like depressions found that depressions subjected to agricultural disturbances in 1951 primarily developed into mixed hardwoods/pine communities or became pine plantations, as opposed to depressions not subjected to agricultural disturbances where vegetation structure/composition remained stable (Kirkman et al. 1996). Similarly, a study of lime-sink wetlands suggests that alterations in fire or hydrological regimes influences vegetative succession and may lead to distinct vegetation types (Kirkman et al. 2000). Research comparing lime-sinks with riparian wetlands in Baker County, GA suggests that sediment and nutrient retention is correlated with anthropogenic LULC within surrounding catchments or watersheds (Craft & Casey 2000). Data revealed significantly higher rates of sediment and nutrient accretion over the past 100 years, as opposed to the past 30 years, suggesting that past anthropogenic activities continue to impact the sediment and nutrient content within the studied wetlands (Craft & Casey 2000). However, most LULC studies targeting wetland or aquatic systems have only incorporated two time-steps, current LULC and one

measure of historical LULC. Increasing the temporal resolution may increase our understanding of how temporal patterns of land use and land use change affect the current condition of ecological systems.

I suggest combining the LULC database developed in Chapter 2 with the isolated wetland map developed in Chapter 3 to explore the long-term LULC histories or trajectories of isolated wetlands in the Dougherty Plain. Recently developed methods capable of tracking land use over multiple time steps, such as Landscape Change Trajectory Analysis (LCTA), may be helpful in identifying and understanding forces driving present wetland condition (Hietel et al. 2004; Käyhkö & Skånes 2006; 2008; Ruiz & Domon 2009). This technique has shown promise in identifying and quantifying ecosystem services associated with specific landscapes. For example, LCTA has been used to define the potential capacity of landscapes to maintain biodiversity (Käyhkö & Skånes 2006; 2008). If past LULC is determined to be a driver of isolated wetland condition, then it could be incorporated into the remote LDI assessment of isolated wetland condition developed in Chapter 4.

Isolated wetlands are not likely to regain federal protection without an act of Congress, and state governments are unlikely to implement wetland protection programs in the absence of scientific data presenting a clear need for legislative or regulatory action. Future research should focus on developing and/or refining maps depicting the accurate location and extent of historic and current isolated wetlands, and developing and/or calibrating an assessment framework to quantify wetland condition. Wetland assessment methods should seek to refine the relationship between remote and on-site assessments; with the eventual goal of conducting remote wetland assessments at the regional scale. This thesis provides the methods and data necessary to pursue such applied research goals in the Dougherty Plain, and elsewhere.

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