THE EFFECT OF LAND USE/LAND COVER ON HEADWATER-SLOPE WETLANDS IN BALDWIN COUNTY, ALABAMA

by

Wales Flynt Barksdale Jr.

A thesis submitted to the Graduate Faculty of
Auburn University
in partial fulfillment of the
requirements for the Degree of
Master of Science

Auburn, Alabama
May 4, 2013

Keywords: Headwater wetlands, Hydrogeomorphic approach, Forested wetland structure,
Land use/land cover, Prevalence index, Soil carbon, Coastal Alabama

Approved by

Christopher J. Anderson, Assistant Professor, School of Forestry and Wildlife Sciences
B. Graeme Lockaby, Professor, School of Forestry and Wildlife Sciences
Latif Kalin, Associate Professor, School of Forestry and Wildlife Sciences

Abstract

Land use and land cover (LULC) have been shown to greatly influence aquatic environments although very few studies have described the effect on headwater wetlands. The overall goal of this research was to assess the impact of LULC on headwater-slope wetlands and their associated functions. The objectives were to (1) assess the influence of LULC on the forested structure, composition and function as modeled by the hydrogeomorphic approach, and (2) evaluate the effects of LULC on specific wetland functions (carbon cycling and water storage) through direct measurement. Using 30 wetlands across a gradient of surrounding LULC, tree density, tree diameter, species cover, and soil value/chroma were measured. Prevalence index (PI) was calculated as an indicator of potential species shift due to change of hydroperiod related to changes in LULC within corresponding watersheds. Using the collected wetland data, functional capacity indices (FCI) were calculated using the hydrogeomorphic approach (HGM) for wetland wildlife habitat, organic carbon cycling, water storage and characteristic vegetative community. The presence of exotic species was significantly related to LULC with increases in *Ligustrum sinense* (Chinese privet) and *Sapium sebiferum* (Chinese tallow tree) as forested land cover decreased. Prevalence index, as well as, soil chroma decreased with forested land cover, potentially describing longer saturated soil conditions within wetlands whose watersheds were dominated by forests. The same level of significance did not occur when the FCI scores were evaluated as response variables to LULC. The HGM may not be suitable to describe impacts of shifts in land use because of inherent issues with the wetland hydrology variable used to

calculate FCI scores. However, FCI describing wetland ability to provide characteristic wildlife habitat was significantly related to LULC. Although useful for regulatory purposes, HGM may require further development and validation before it should be utilized for scientific purposes. In the second part of this study, land use and soil data were utilized to calculate runoff curve numbers (CN) and used with site-specific precipitation data to calculate total runoff (P_e) produced within 15 wetland watersheds. Water level recorders were installed in each wetland and programmed to measure surface and ground water levels. Hydrologic metrics derived from these data included: Richard-Baker index (RB), water level rate of change (WLΔ), water level variability (standard deviation of measurements), and a measure of the percent of time the water table was within 20-cm of ground surface (WT_{20}). Richard-Baker index was the only hydrological metric to be significantly related to land use and decreased with increasing CN. Bimonthly monitoring of forest floor carbon (C_{FF}) and litter fall carbon (C_{LF}) were conducted within each plot to determine if watershed runoff influenced the storage of forest litter fall. A significant relationship was detected between decreasing C_{FF} with increased CN and P_e suggesting hydrologic export may be an important process in forest carbon dynamics where surrounding lands have been altered. Soil carbon was quantified in each wetland by collecting cores to a depth of 1m and processing 10-cm sections for carbon analysis. Increased duration of soil saturation resulted in decreasing bulk density, potentially describing the role of soil saturation in the storage of organic carbon.

Acknowledgments

I would like to thank my major professor and advisor, Dr. Chris Anderson for his patience and guidance throughout this research project. Much appreciation is deserved by my committee members, Drs. Graeme Lockaby and Latif Kalin for their insight and support. I thank the Center for Forest Sustainability (CFS) for providing funding for this research and offering me the opportunity to learn and grow as a scientist. Thanks goes out to the many private land owners, The Nature Conservancy, the City of Foley, and the City of Robertsdale for allowing property access to conduct my research. Finally, my sincerest gratitude to my friends who not only helped as field and lab assistance but also gave personal support during these times: Diane Alix, Joe D'Angelo, Thomas Hess, Dana McKnight, Brad Schneid, Amir Sharifi, and Madeline Wedge.

Table of Contents

List of Figures

Chapter II

Chapter III

List of Tables

CHAPTER I

THE EFFECT OF LAND USE/LAND COVER ON THE STRUCTURE, FUNCTION, AND BIOGEOCHEMICAL PROCESSES OF HEADWATER-SLOPE WETLANDS IN BALDWIN COUNTY, ALABAMA: AN INTRODUCTION

Land use and land cover (LULC) have been demonstrated to greatly influence aquatic environments. However, few studies have aimed to describe these effects on headwater wetlands. The conversion of forested land to urban areas and for agriculture production has been shown to significantly alter hydrological regimes and biogeochemical processes (DeLaney 1995; Faulkner 2005). These landscape alterations are significant to wetland environments as hydrology is the major driver of wetland ecosystems processes (Messina and Conner 1998). Agricultural practices tend to impact water quality by increasing sediment and nutrient loadings as a result of soil disturbance and various drainage techniques (DeLaney 1995; Zedler 2003). Similar and more devastating effects often result from urbanization. Increased impervious surface area (ISA) resulting from urban development alters hydrology, water chemistry, and stream morphology (Paul and Meyer 2001). Increased road density associated with urban areas can impact hydrologic processes by intercepting horizontal flows (Forman and Alexander 1998). These general concepts of how LULC alters aquatic ecosystems are understood however there are substantial gaps in our understanding of these effects on headwater systems which drive downstream processes.

Hydrology is a major driver of ecosystem processes within wetland environments including, vegetation, soils and nutrient cycling (Ehrenfeld et al. 2003). Headwater wetlands are located at the upper reaches of coastal creeks and based on their position in the landscape may be particularly susceptible to landscape alterations related to land use change. Processes within headwater wetlands are driven by a consistently high water table, which originates as subsurface flow in the surrounding uplands (Noble et al. 2007). Approximately 90% of flow from upland environments to aquatic environments is subsurface (Kaye et al. 2006). Based on this hydrologic link to the surrounding landscape, any alteration to hydrology could have significant impacts on wetland systems.

Wetland functions are closely linked to forest structure and composition, with hydrology driving these conditions. As these systems become hydrologically impacted shifts in vegetative community and structure can be expected. For instance, the prevalence of wetland species may change as hydroperiod shifts. Species composition may become more facultative with increased upland species as a result of drainage, or the system may shift to primarily obligate wetland species due to ponding. These shifts in community type can result in changes in forest productivity, which drives trophic webs (Wallace et al. 1999). Under natural conditions these systems provide stable water supplies to downstream systems due to their inherent ability to store water. The capacity to store water can be lessened by land use change, resulting in habitat alterations and influencing aquatic biodiversity (Baker 1992; Brinson 1996).

Biogeochemical processes, such as carbon cycling and decomposition are driven by many factors including: moisture, temperature, nutrient supply, and litter quality (Baker et al. 2001; Lecerf et al. 2007). However, hydroperiod is by far the most important (Mitsch and Gosselink 2000). Decomposition is often optimized when a cycle of wetting and drying exists within a wetland (Brinson et al. 1981). Wetland biogeochemical processes are critical for the global carbon cycle as greater than 50% of all the world’s stored soil carbon is within wetlands (Mitsch and Gosselink 2000). Any changes in hydroperiod resulting from land use alteration could have significant impacts on the carbon cycle, shifting wetland environments from carbon sinks to sources and vice versa. Linkages between upland and wetland environments are critical in maintaining wetland productivity as they receive nutrients and organic material from the surrounding landscape (Messina and Conner 1998; Albeho 2001). The cycling of this organic material in wetlands is the combination of physical breakdown (abrasion, fragmentation, and leaching), decomposition processes, and hydrologic export (Webster and Benfield 1986). Due to increased peak and surface water flows associated with land use change increased erosion of forest floor litter and faster decomposition due to physical disturbance can be expected (Xiong and Nillson 1997; Glazebrook and Robertson 1999). The ability of wetlands to function hydrologically and biogeochemically and the relation of these functions and water quality demonstrate a close link of upland, wetland, and aquatic ecosystems (Wallbridge 1993). Based on this close link we expect wetlands to function more as sinks for organic carbon where land use change within their immediate watershed has been minimized.

The main goal of this project was to improve the understanding and management of headwater wetland habitats within developing areas in coastal Alabama. This is critical for Baldwin County which was chosen as our study area. From 1990-2000 Baldwin County had a 43% increase in population which was the second highest population increase for a county in Alabama (BCPZD 2005). The following decade also saw a substantial increase in population, 24%. (USCB 2010). This area was originally impacted by agriculture but as population increases further disturbance will result from urbanization. We expect ecosystems processes within headwater wetlands to be altered and related to land use change within a wetland's immediate watershed.

Chapter 2 of this thesis focuses on changes in wetland characteristics (vegetation community and structure, soil color value, hydrology, etc.) due to land conversion from historically forested lands to agricultural and urban land types. The collection of data for this study was based on the methods outlined in the Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing the Functions of Headwater Slope Wetlands on the Mississippi and Alabama Coastal Plains (Noble et al. 2007). The objective of this chapter was to determine how land use change and urbanization influences the condition of headwater wetlands within Southern Mississippi and Alabama and the Southern Coastal Plain physiographic region.

A more detailed study of land use effects on wetland functions is provided in Chapter 3; specifically, how LULC impacts wetlands hydrology and biogeochemical processes. We selected a subset of 15 headwater wetlands based on a range of altered land-use in which we monitored groundwater hydrology and metrics of carbon cycling (leaf litter production, standing stock of litter, and soil carbon). In this study, the connection between upland land-use, hydrology, and shifts in the carbon cycle as a result of altered hydrology were explored.

Finally, Chapter 4 summarizes the above research and results. Explanation and synthesis of these results provide for an increased understanding of the role these environments on maintaining water quality, biodiversity, carbon cycling and other services these wetlands provide. These conclusions should contribute to the development of planning strategies and techniques to preserve these critical ecosystems.

1.1 References

Abelho, M. 2001. From litter fall to breakdown in streams: a review. The Scientific World. 1:656-80.

Baker, L.A. 1992. Introduction to nonpoint source pollution in the United States and prospects for wetland use. Ecological Engineering. 1:1-26.

Baker III, T.T., B.G. Lockaby, W.H. Conner, C.E. Meier, J.A. Stanturf, M.K. Burke. 2001. Leaf litter decomposition and nutrient dynamics in four southern forested floodplain communities. Soil Sci. Am. J. 65:1334-47.

Brinson, M.M., A.E., Lugo and S. Brown. 1981. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics. 12:123-.61

Brinson, M. 1996. Assessing wetland function using HGM. National Wetland Newsletter January-February: 10-16.

De Laney, T.A. 1995. Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. Journal of Soil and Water Conservation 50(6): 620-28.

Ehrenfeld, J.G., H.B. Cutway, R. Hamilton IV, and E. Stander. 2003. Hydrologic description of forested wetlands in Northeastern New Jersey, USA- an urban/suburban region. Wetlands 23(4): 685-700.

Faulkner, S. 2004. Urbanization impacts on the structure and function of forested wetlands. Urban Ecosystems. 7:89-106.

Forman, R.T.T. and L.E. Alexander. 1998. Roads and Their Major Ecological Effects. Annual Review of Ecology and Systematics: 207-31.

Glazebrook, H.S. and A.I. Robertson. 1999. The effect of flooding and flood timing on leaf litter breakdown rates and nutrient dynamics in a river red gum (*Eucalyptu camaldulensis*) forest. Australian Journal of Ecology. 24:625-35.

Kaye, J.P., P.M. Groffman, N.B. Grimm, L.A. Baker, and R.V. Pouyat. 2006. A distinct urban chemistry? TRENDS in Ecology and Evolution. 21(4):192-99.

Lecerf, A., G. Risnoveanu, C. Popescu, M.O. Gessner, and E. Chauvet. 2007. Decomposition of diverse litter mixture in streams. Ecology 88(1):219-27.

Messina, M.G. and W.H. Conner. 1998. Southern Forested Wetlands: Ecology and Management. Lewis Publishers. Boca Raton.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. 3rd ed., John Wiley and Sons, New York.

Noble, C.V., J.S. Wakeley, T.H. Robert and C. Henderson. 2007. Regional guidebook for applying the hydrogeomorphic approach to assessing the functions of headwater slope wetlands on the Mississippi and Alabama coastal plains. U.S. Army Corps of Engineers.

Paul, M.J. and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecological Systems 32:333-65.

Wallace, J.B., S.L. Eggert, J.L. Meyer, and J.R. Webster. 1999. Effects of resource limitation on detrital-based ecosystems. Ecological Monographs 69(4):409-42.

Wallbridge, M.R. 1993. Functions and values of forested wetlands in the southern United States. Journal of Forestry. May:15-19

Webster, J.R., and E.F. Benfield 1986. Vascular plant breakdown in freshwater ecosystems. Annual Review of Ecology and Systematics. 17:567-94.

Xiong, S. and C. Nilsson. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: a review. The Botanical Review 63(3):240-64.

Zedler, J.B. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecological Environments 1(2):65-72.

CHAPTER II

THE EFFECT OF LAND USE/LAND COVER ON FORESTED STRUCTURE AND COMPOSITION IN HEADWATER-SLOPE WETLANDS IN BALDWIN COUNTY, ALABAMA

2.1 Abstract

Aquatic and wetland environments have been shown to be greatly influenced by land use and land cover (LULC) but little research has focused on headwater wetlands. The objective of this study was to evaluate the effects of land use change on the structure, composition and function of headwater wetlands in coastal Alabama. A total of 30 headwater wetlands were selected across Baldwin County, Alabama that represented a range of surrounding LULC. Percent forest, agriculture and impervious surface area (ISA) were quantified in each of the 30 corresponding watersheds. Principal component analysis (PCA) was performed on LULC data to eliminate collinearity within land use data and produced orthogonal variables were produced, which represented forested cover and conversion of forested land to agricultural production. For each wetland tree density, tree diameter, species cover, species importance values, and soil value/chroma were measured. Prevalence index was calculated as an indicator of potential species shift due to change of hydroperiod resulting from LULC within corresponding watersheds. Using the collected data, functional capacity indices (FCI) were calculated for wetland wildlife habitat, carbon cycling, water storage and characteristic vegetative community using the hydrogeomorphic approach (HGM). The presence of exotic species was significantly related to LULC with increases in Chinese privet (*Ligustrum sinense*) and Chinese tallow tree (*Sapium sebiferum*) as forested land cover decreased. Prevalence index (PI), as well as, soil

chroma and value decreased with forested land cover, potentially describing longer saturated soil conditions within wetlands whose watersheds were dominated by forests. Fewer relationships were detected using FCI scores as response variables. The HGM may not be suitable to describe shifts in land use because of inherent issues with the wetland hydrology variable and its measurement in the field. However, the functional capacity index (FCI) describing the ability to provide characteristic wildlife habitat was significantly related to LULC. Although useful for regulatory purposes HGM may require further development and validation before it should be utilized for scientific purposes.

Keywords: Headwater slope wetland, forest structure, hydrogeomorphic approach, land use land cover, Magnolia virginiana, hydroperiod, coastal Alabama

2.2 Introduction

Land use change associated with urbanization and agricultural practices is a leading cause of environmental degradation of aquatic systems resulting in significant alterations to hydrologic regimes and biogeochemical processes (Faulkner 2004; Messina and Conner 1998; DeLaney 1995). Agricultural practices leading to soil disturbance and various drainage techniques can change hydrologic conditions on the landscape, increase sediments and nutrients in runoff, and affect downstream water quality (DeLaney 1995; Zedler 2003, Blann et al. 2009). As with agriculture, urbanization results in large areas of land being cleared, paved, and dramatically modified. Shifts in LULC for both urban expansion and agricultural production have been show to have significant effects on local hydrology, water chemistry, and morphology (Paul and Meyer 2001; Poff et al. 2006; Blann et al. 2009). It is generally understood that land use changes affect aquatic systems however there are still gaps in our understanding, particularly as it relates to wetlands and headwaters.

Hydrology is the primary factor influencing wetland vegetation, soils, and ecosystem dynamics (Ehrenfeld et al. 2003). Based on their position in the landscape, headwater wetlands may be particularly susceptible to land use change. Headwater wetland processes are characterized by a consistent high water table, which originates as subsurface flow from the surrounding landscape (Noble et al. 2007). This is critical as it has been estimated that up to 90% of flow from upland to aquatic environments is subsurface (Kaye et al. 2006). Any hydrologic alteration within the watershed could have major impacts on these wetland systems by changing drainage patterns. Ditching and the installation of drainage tiles may alter the hydrology of wetland systems by facilitating water drainage for improved agricultural production on adjacent lands (Blann et al. 2009). Shifts in hydroperiod and increased sedimentation and nutrients from agricultural runoff are often major pollutants in wetland environments (Gemborys and Hodgkins 1971;Baker 1992; Gleason and Euliss 1998, Herliby et al. 1998). These pollutants can have many direct and indirect effects on wetlands environments such as: shifts in vegetation, loss of wildlife biodiversity and alterations to aquatic food webs (Baker 1992; Robb 1992; Gleason and Euliss 1998; Zedler 2003). Similarly, decreases in ground water recharge, increased surface water flows and velocities, reduced infiltration of precipitation, lower base flows, decreased water storage, and increased erosion have all been associated with urban land use and impervious surface area (ISA) (Ehrenfeld et al. 2003; Walsh et al. 2005; Chadwick et al. 2006). Changes in these hydrologic conditions can have a major impact on wetland hydroperiod and connectivity of aquatic environments. Hydrologic connectivity is further disrupted by piping and conveyance of storm water which facilitates drainage of the landscape in order to lessen flooding in urban areas

(Walsh et al. 2005). As a result, urban streams and wetlands tend to have “flashier” hydrographs than that of natural environments with greater flow and faster ascending/descending limbs (Walsh et al. 2005). Increases in peak flows associated with increased ISA can change the natural disturbance regime, which can have significant impacts on native stream organisms and riparian vegetation (Jones et al. 2000).

Wetland functions are closely linked to forest structure and composition; with hydrology as a primary driver of these conditions. As these environments are impacted hydrologically, species composition and forest structure may change with shifting ecosystem processes. For instance, the prevalence of wetland obligate species may change as hydroperiod shifts. Species composition may become more wetland obligate due to ponding and increased flooding, or the system may shift to more facultative and upland species as a result of drainage. Shifts in community type also can result in changes in productivity and dependent trophic webs (Wallace et al. 1999). Alterations on the landscape can also decrease the ability for wetlands to store water due to excessive sedimentation (Baker 1992) and lower base flows (Chadwick et al. 2006). Under natural conditions, these wetlands could provide more stable water supplies downstream maintaining habitat and influencing aquatic biodiversity (Brinson 1996). Roads can directly and indirectly alter plant communities by providing preferential migration corridors and points of establishment for invasive plant species (von der Lippe and Kowarik 2007). Such non-native invasions have proven to be major threats to local diversity and other ecosystem functions (Lundgren et al. 2004).

Wetland carbon cycling and decomposition rates are important functions driven by many factors; however, hydroperiod is by far the most important (Mitsch and Gosselink 2000). As an example, decomposition is often optimized when a cycle of wetting and drying exist within a wetland (Brinson et al. 1981). Linkages between the upland environment and wetland ecosystems are vital in maintaining the productivity within small forested wetland systems, as they receive both organic matter and nutrients from surrounding upland landscapes (Messina and Conner 1998; Abelho 2001). Due to increased peak flows and surface water energy associated with land use change (Walsh et al. 2005; Chadwick et al. 2006; Nagy et al. 2011), increased erosion of forest floor litter and faster decomposition attributed to physical processes (i.e. fragmentation) can be expected (Xiong and Nillson 1997; Glazebrook and Robertson 1999). As a result, urban wetlands may function less as carbon sinks than wetlands within more forested environments.

The objective of this study was to examine how land use change may impact headwater wetland habitats within coastal Alabama. This is a crucial issue due to rapid land use change in the region. For instance, from 1990-2000 Baldwin County, Alabama had the second highest increase in population (43%) within the state of Alabama (BCPZD 2005). Similarly, from 2000-2010 Baldwin County had a 24% increase in population (USCB 2010). This area has traditionally been impacted by agriculture, but increases in population and urban land use will certainly result in further forest cover loss. There have been studies on how LULC can affect carbon cycling, hydrology, and vegetation, but very little has been done to demonstrate these effects throughout this region and within headwater environments. Examining wetlands along a gradient of increasing land use change, we expected that wetland structure, species composition, and the functional capacity of headwater wetlands will change as surrounding natural land cover in the watershed is lost.

2.3 Methods

2.3.1 Description of Study Sites

Headwater wetlands for this study were located in Baldwin County, AL, within the Southern Coastal Plain physiographic region (Fig. 2.1). The region is characterized by mild winters and hot and humid summers with mean annual temperatures ranging from 15 to 21˚C, and annual precipitation ranging from 125 to 180 cm which is fairly evenly distributed throughout the year (Noble et al. 2007). Headwater wetlands in coastal Alabama are groundwater driven and located at the headwater reaches of first order streams (Noble et al. 2007). These wetlands are typically comprised of alluvial soils that are classified as “wet loamy alluvial lands”, while the uplands are generally sandy soils derived from marine deposits (McBride and Burgess 1964). This study began in June 2009 with the identification of approximately 60 headwater slope wetlands as potential study sites. Wetlands were identified with the use of topographic maps, aerial imagery, National Wetlands Inventory maps, and the Soil Survey of Baldwin County, Alabama (McBride and Burgess 1964). Each wetland was then visited to conduct a preliminary assessment. From this initial assessment a total of 30 headwater slope wetlands were selected for further evaluation that displayed a range of surrounding LULC (e.g.: urban, agriculture, forest) (Fig. 2.2,Table 2.1). Wetlands selected were all forested (with evidence of tree species characteristic of headwater wetlands) and comparable in size (both wetland and watershed) to reduce confounding factors that may affect forest structure, composition, and function.

2.3.2 Watershed Assessment and LULC Classification

The associated watershed for each wetland was delineated using the ArcSWAT 2009.93.6 model for ESRI ArcMap 9.3 (ESRI 2009). Alterations in watershed area (increase or decrease) due to anthropogenic drainage features were considered from on-site observations and using current aerial photography. Weighted runoff scores were calculated with the use of soil drainage classes (hydrologic groups) found in the Soil Survey of Baldwin County (McBride and Burgess 1964) and the predominant land use on each soil type. Characteristics of LULC within corresponding wetland watersheds were evaluated with the use of ESRI ArcMap 9.3 (ESRI 2009). Aerial images from 2009 were used to manually delineate several characteristic land uses (Anderson et al. 1976). For each watershed, LULC features were classed into pervious and impervious surfaces, which were further divided in to subclasses to test between differing land use types. Pervious subclasses included: forested lands, agricultural lands, and unpaved roads and driveways; impervious subclasses included: building structures, roads, and other paved surfaces (driveways, patios, etc.). LULC classes and subclasses were then quantified for total percent forested land, agricultural land (row crops, pasture, etc.) and impervious surface area (ISA) to facilitate a better understanding and interpretation of land use data. Urban land use was represented by ISA as a more direct measure of influence rather than discerning broad categorical measures (e.g.: low, medium, and high density residential).

2.3.3 Wetland Structure and Composition

Field assessments were conducted on all selected headwater wetlands. Four 100 m^2 plots were established within each wetland to capture variability in vegetative community types. Within each plot, tree density (stems ha^{-1}) and tree diameter (DBH- cm) were measured for all canopy individuals (DBH > 10.2cm). Using these data, mean basal area (m^2 ha^{-1}) was calculated for each wetland. Sapling and shrub layer was defined as woody plants with a height greater than 1m and a DBH < 10.2cm. Understory included all herbaceous and woody vegetation less than 1m tall. Total cover for canopy, sapling/shrub, and understory species were visually estimated as

% cover per strata. Exotic species cover was calculated based on cover data within each forest strata. To better describe the canopy composition, a point-quadrat survey was conducted to quantify individual species abundance. One random point was assigned within each wetland plot and the four closet canopy species were measured for distance to point and basal area. These combined data were used to estimate species density (stems ha^{-1}), dominance (% of total basal area), and frequency (% of plots detected). Importance values (IV) were then calculated based on relative density, dominance, and frequency, resulting in a maximum possible IV score of 300 (IV_{300}).

To describe hydrologic alterations and resulting changes in the vegetative community, prevalence index (PI), an index of hydrophytic vegetation abundance, was calculated for all three strata within each wetland with use of species cover values and wetland indicator status (obligate [OBL], facultative wetland [FACW], facultative [FAC], facultative upland [FACU], and upland [UPL]) as listed in *The National List of Wetland Plants: Region 2* (Reed 1988). A hydrophytic index score was assigned to each wetland indicator status with OBL species receiving a value of 1, FACW species a 2, FAC species a 3, FACU species a 4, and UPL species a 5 (Wentworth et al. 1988). The vegetative cover per plot of each indicator status was multiplied by its hydrophytic index score to calculate wetland PI (weighted average per plot). Thus, wetlands with lower PI values were comprised of more hydrophytic species.

Four soil samples of the top 15 cm of soil profile were examined within each wetland plot. Soil chroma and value were averaged per wetland using a Munsell soil chart. A visual estimate of detritus cover was recorded for each plot and averaged per wetland. The greatest depth of apparent flooding (based on height of water marks, drift lines, berms, dams, etc.) was recorded as an indicator of maximum water depth within each wetland.

2.3.4 Wetland Functional Assessment

The data collected for the watershed and wetland assessments were used to conduct a functional assessment utilizing the hydrogeomorphic approach (HGM). The HGM is an assessment tool designed to assess the capacity of wetlands to perform important functions relative to reference wetlands in the region (Brinson 1996). We utilized a local HGM guidebook (The Regional Guidebook for Applying the Hydrogeomporhic Approach to Assessing the Functions of Headwater Slope Wetlands in the Mississippi and Alabama Coastal Plains, Noble et al. 2007) to examine the functional capacity of the study wetlands. Using data collected, variables were quantified, calculated in the assessment models, and a Functional Capacity Index (FCI) score was generated that ranged from 0.0-1.0. The FCI is a measure of performance for a particular wetland function where a score of 1.0 is representative of reference conditions. As the capacity for a wetland to perform a function decreases so does the score. HGM functional models included: 1) Water Storage, 2) Organic Carbon Cycling, 3) Maintenance of Characteristic Plant Community, and 4) Provide Characteristic Wildlife Habitat. Details regarding variables and functional assessment models are provided in Appendix A.

2.3.5 Statistical Analysis

Data were analyzed to test for the relation of land-use change on the structure, composition, and function of headwater wetlands within coastal Alabama using R 2.12.2 software (R Development Core Team 2010). Due to inherent collinearity associated with land use data (King et al. 2005), principal component analysis (PCA) was utilized to generate truly independent variables (principal components) that best explain the variance within percent forested land use, agricultural land use and ISA data. Based on the component loadings (correlation coefficients)

(Table 2.2), Comp. 1 explained 86.1 percent of the variation within land use data. Comp. 1 was highly correlated with forest cover while being negatively correlated with agricultural land use and no correlation to ISA. Comp. 2 explained 14.3 percent of the variation within land use data with a strong positive correlation to both agriculture and forested cover and a negative correlation to ISA describing a more mixed LULC of increasing forested land cover and agriculture use and decreasing urban land use. Comp. 3 was utilized in regression models but was not described as < 1 percent of variation within land use data was explained with Comp. 3 and greater than 99.4 percent was explained by Comp. 1 and Comp. 2. Pearson's correlation analysis was utilized to explore relationships between principal components and forest structure, composition, and FCI scores. Apparent relationships were tested through multivariate linear regression analysis to examine the relation between generated principal components and the metrics explaining wetland structure and composition (vegetation, soil, and hydrology), forest canopy species (per IV_{300}), and the FCI scores of each corresponding wetland. Assumptions for regression were tested with the use of the Shapiro – Wilk's test for normality and Bartlett's test of homoscedasticity with no need for transformations. Multivariate models with all three components were tested and statistical significance was set at á=0.05.

2.4 Results

2.4.1 Watershed Characteristics and LULC Classification

Based on digital elevation models, ArcSWAT delineated historic watershed boundaries for all 30 headwater wetlands producing a mean area of 89.5 ± 12.8 ha (Table 2.1). Manually adjusting watershed area (corrected for changes in hydrological movement due to landscape modifications, i.e. road construction) resulted in an adjusted mean area of 86.2 ± 13.0 ha. The resulting percent change in watershed area was 9.6 ± 2.6 ha. Mean wetland area was manually delineated for all study wetlands and was 3.2 ± 0.7 ha. Mean ratio of watershed to wetland area was of 8.2 ± 2.8.

LULC was manually delineated for each of the 30 corresponding watersheds (Fig. 2.2). Mean forested land cover was 44.9±5.4% and was evenly distributed between a range of 6 and 100%. Mean agricultural land cover was 27.0 ± 4.4% and ISA had a mean area within all watersheds of 3.4 ± 1.0% and a range of 0.0 and 24.0%.

2.4.2 Wetland Structure and Composition, and Function

2.4.2.1 Structure and Composition

Species composition varied across study sites with a total of 51 individual species detected throughout all wetlands (Table 2.3). Dominant canopy tree species included: *Magnolia virginiana, Nyssa biflora, Acer rubrum, Pinus elliottii* and *Sapium sebiferum.* Mean IV_{300} (Table 2.4) for these dominant canopy species were: 102.2 ± 9.4, 85.4 ± 11.9, 31.1 ± 6.7, 26.0 ± 7.6 and 25.4 ± 9.1 respectively. Regression indicated no statistically significant relationships between LULC and species IV_{300}. Based on species cover and importance data (Tables 2.3 and 2.4) the dominant shrub species were: *Cyrilla racemiflora, Myrica cerifera,* and non-native *Ligustrum sinense.* Understory species were dominated by various ferns including: *Woodwardia areolata, Osmunda cinnamomea*, and *Osmunda regalis.*

A range of measured forest structure and other environmental conditions were detected across the 30 headwater wetlands (Table 2.5). Mean canopy tree density was 864 ± 63 stems ha^{-1} and had a significant positive relationship with Comp. 2 ($p = 0.002$) (Fig. 2.3a). Canopy tree diameter was also significantly related to Comp. 2 ($p = 0.001$) with a mean of 22.1 ± 1.2 cm (Fig.

2.3b). Mean wetland basal area was 30.5 ± 2.1 m^2 ha^{-1} and with no significant relationships with any components. Canopy tree cover had a significant positive relationship with Comp. 2 (p = 0.006), while exotic canopy cover was negatively related to Comp. 1 (p = 0.039). Shrub cover was negetively related to Comp. 2 (p = 0.010) with exotic shrub cover (primarily *Ligustrum sinense*) being significantly related to Comp. 1 and Comp. 2 (p = 0.008, 0.008). Exotic ground cover had a significant negative relationship with Comp. 1 (p = 0.026). Mean PI for the overstory was 2.1 ± 0.1, mean PI midstory was 2.6 ± 0.1 and mean PI for understory was 2.0 ± 0.1. PI displayed no significant relationships with either component.

2.4.2.2 Soils and Hydrology

Mean soil chroma and value were 1.7 and 2.8 respectively across wetland sites (Table 2.5). Soils varied across all sites with 23% of sites not being characteristic of wetland soils (based on soil chroma > 2, Mitsch and Gosselink 2000). Soils were typically sandy loams with high organic content; however, we noticed shifts in soil texture and composition associated with LULC. Soil chroma displayed a significant negative relationship with Comp. 1 (p = 0.006) (Fig. 2.4). Detritus cover ranged from 100% to as little as 25% but was not found to be significantly related to LULC. Maximum apparent flooding depth was not found to be significant and had a mean of 4.7 ± 1.8 cm while ranging from 0 to 30 cm of surface water (Table 2.5).

2.4.2.3 Wetland Function

FCI scores estimating the capacity for a wetland to cycle organic carbon, store water, and provide wildlife habitat all had a maximum value of 0.99, while characteristic vegetation community had a maximum value of 0.97 (Table 2.5). These values are representative of reference conditions and coincided with a priori reference sites. The ability to maintain characteristic vegetation community was the most variable function ranging from 0.36 – 0.97. FCI scores representing wetland capacity to cycle organic carbon and provide water storage were much less variable with similar mean values of 0.80 ± 0.003. Providing wildlife habitat had a relatively large range (0.42 – 0.99) with a mean score of 0.68 ± 0.01 and a significant positive relationship to Comp. 1 ($p = 0.010$), while no significance was found with the other functions (Table 2.5).

2.5 Discussion

Results from this study indicated that changes in wetland forest structure, cover by exotic species, and soil chroma were related to land conversion. Based on PCA components detected and the variance explained by land classes, forest cover and agriculture seem to be the dominant landscape factors influencing these wetlands. For instance, results showed that wetland soil chroma was negatively related to Comp.1 (Fig 2.4) that could be related to altered hydrology, increased soil temperatures, and historical sediment input. Several studies have shown that wetlands within less forested watersheds tend to experience a greater number of peak flows and lower base flows, while storing water for shorter durations (Ehrenfeld et al. 2003; Poff et al. 2006; Blann et al. 2009; Nagy et al. 2011). Decreases in forested cover may lower wetland base flows which could reduce water level proximity to the soil surface. Lower water levels can increase soil oxidation, diminish soil organic matter, and increase soil color strength associated with mineral elements (Mitsch and Gosselink 2000). This scenario has been proposed by Groffman et al. (2003) to describe hydrological drought and the potential changes shown to occur in urban riparian soils. Organic matter decomposition and other soil microbial processes may be enhanced with increased soil temperature associated with less forest cover surrounding wetlands (Allan 2004).

Other alterations may contribute to changes in wetland soils indicated by chroma shifts. Increases in agricultural land use may be associated with historical increases in sediment input which when coupled with hydrologic alterations may contribute to changes in soil chroma and values. The role of sedimentation is uncertain given the gradual topography of the region and the prevalence of sandy soils however several wetlands with highly converted watersheds were observed to have much less microtopography (personal observation) suggesting potential sedimentation. Field observations and water level data from a subset of wetlands (see Chap. 3) indicated greater surface flows within wetlands whose watersheds have been altered which may reduce the amount of organic matter that can be accumulated in soil. Hydric soils are characterized by low redox conditions that maintain chroma of <2, a threshold that has been used for the classification and delineation of wetlands (Mitsch and Gosselink 2000). This designation of hydric soils is interesting as only 77% of our sites had surface soils indicative of hydric soil conditions (Fig 2.5). These results suggest that land conversion in this region may potentially alter hydric soils and their associated functions.

Increased exotic species cover increased at all strata with increased land conversion and showed different relationships with both components. Exotic shrub cover was primarily *Ligustrum sinense* which was, on average, 84% of midstory cover where it was established. Several studies have made the linkage between loss of forest cover (from increased agriculture and/or urban land use) and increased occurrence of exotic vegetation (Galatowitsch et al. 2000; Moffatt et al. 2004; Miller et al. 2006). Houlahan et al. (2006) found percent land use change adjacent (within 300m) to wetlands in Ontario, Canada predicted declines in species richness due to the increased presence of exotic species. The authors emphasized the correlation between land use change (loss of forest cover) and increases in number of dispersal routes for exotic species into wetlands. Increased exotic shrub cover may also influence recruitment of native woody vegetation and forest structure. Once established, *Ligustrum sinense* has been shown to form dense thickets prohibiting the recruitment of canopy tree species. Like exotic shrub cover, average wetland tree DBH decreased (and stem density increased) in relation to Comp. 2 (increased forest and agriculture, decreased ISA). (Fig 2.4). This was similar to Mitchell et al. (2011) who described forests in the lower piedmont region of Georgia with lower basal areas and fewer canopy trees in areas with greater abundance of *Ligustrum sinense*. Changes in forest canopy structure and abundance may be due to increased disturbance as forested land cover decreases. Mean canopy tree DBH (22.1 ± 1.21 cm) was less than designated reference standard (30 cm, Noble et al. 2007). Likewise, mean tree density (864 ± 63 stems ha^{-1}) was also outside the reference standard range for these wetlands (250-425 stems ha^{-1}, Noble et al. 2007).   A post-hoc correlation analysis showed tree density was negatively correlated (r=-0.47) with exotic shrub cover. This increase in exotic shrub cover is most likely a result of edge effects and points of invasion for non-native species (von der Lippe and Kowarik 2007, Oneal and Rotenberry 2008, Niggeman et al. 2009). As forested environments become fragmented, these areas are affected by increased light penetration enabling the recruitment of less shade tolerant species that would otherwise not become established in closed canopy forested systems.

No relationship was found between detectable maximum flooding depth and land use. Recent research has associated higher peak flows and altered hydrology with land use change in the southern Coastal Plain Physiographic region (Nagy et al. 2011, Chap. 3) and the lack of statistical significance is likely the result of a poor metric for hydrologic impact associated with HGM methodology (see below). PI values describe the abundance of characteristic wetland species and may explain how different strata are affected by hydrological shifts at different rates.

The potential for shifts in vegetation has previously been demonstrated in southwest Alabama forested wetlands. Gembroys and Hodgkins (1971) found increased importance of *Magnolia virginiana* and *Nyssa biflora* from more facultative species (*Liquidamber styraciflua, Nyssa sylvatica, Quercus nigra*) as soil moisture increased. In this study, the midstory showed a statistically insignificant (p = 0.07) but potentially biologically significant relationship between decreasing PI and forested watershed cover, describing a wetter environment supporting OBL and FACW plant species. The apparent shift in soil saturation between forested and agricultural environments may be attributed to less infiltration due to faster drainage of water across the landscape. Based on the range of PI values in the understory, it appears to be more responsive than both the midstory and overstory, which, could be expected as herbaceous vegetation are not as long lived. Due to the higher turnover rate of herbaceous vegetation, the understory is often a better indicator of current soil saturation conditions (Naiman et al. 2005).

Except for providing characteristic wildlife habitat, the hydrogeomorphic approach did not relate shifts in LULC to wetland functional capacity. This was contrary to our predictions because wetland functions are not solely driven by processes within the wetland boundary. Therefore, we predicted HGM models would indicate LULC to have a strong influence on wetland functions. Model results indicated three of the four functions (maintain characteristic plant community, water storage, and cycle organic carbon) were not related to land-use. Based on the model equations, the characteristic wildlife habitat FCI was the only model that was significantly related to any land use component. This function was strongly influenced by prominent model variable forest connectivity (*Vconn*) which emphasizes contiguous forest immediately adjacent to the wetland boundary. This was (as expected) highly correlated with forested land use (r=0.66). The lack of statistical relationships between the other functions and land use may be traced to the influence of other variables. Specifically, the model variable for describing alterations in hydrology, *Vhydroalt*, is a dominant variable in both the cycle organic carbon and water storage functional capacity indices (see Appendix B). However, there may be inherent issues with calculating *Vhydroalt*, which relies on the field indicators. The indicators (i.e. height of drift lines and/or watermarks) may not always be apparent and/or representative of current hydrology. For instance, watermarks may persist over time and not reflect current hydrological characteristics. Similarly, drift lines may not be detectable in all wetlands. Drift lines need either micro-topography or vegetation to dampen hydraulic energy and allow the accumulation of detrital material. If such conditions do not exist due to historical sedimentation or exclusion of understory vegetation, drift lines may not be apparent. We did not detect a significant correlation between flooding depth and LULC however results from concurrent research (see Chap. 3) suggest that wetlands with greater forested land conversion promotes hydrologic export of organic matter. Further investigation of headwater wetland hydrology may be needed to identify environmental indicators for shifts in hydrology that may be more conspicuous. Cole (2006) determined that no less than one year of hydrological monitoring is adequate for the development of assessment protocols. A more complete metric assessment for hydrology may be needed that includes belowground changes. Rooting depth or soil chroma/value may be more concrete.

Lack of relationships between the capacity of a wetland to maintain a characteristic plant community and land use was unexpected. Non-native and invasive species were significantly related with land use across all three strata, however, HGM calculates the model variable *Vcomp* based on the composition of the highest strata with greater than 20 percent cover utilizing the 50/20 rule for dominant species classification of that uppermost strata (Noble et al. 2007). This results in the abundance of *Ligustrum senense*, a non-native, highly invasive shrub, not being considered within our HGM models, and *Sapium sebifera*, an invasive tree species, only being considered when it is dominant and otherwise not accounting for its occurrence. Relationships between land use alteration, changes in hydrology, and wetland specific processes are complex and may not be fully represented by functional assessment models such as these. Stander and Ehrenfeld (2009) also noted that this complexity may not be definable with simple rapid assessment procedures, such as the HGM.

2.6 Conclusions

Hydrological processes are primary drivers of wetland environments and further explanation of shifts in hydrological regime may shed more light on how headwater wetland systems are affected by deforestation. Trends related to decreases in canopy tree density and increases in exotic shrub cover suggest these wetlands may fail to recruit characteristic species as surrounding land use changes. Increased presence and cover of *Ligustrum sinense* appears to be associated with increased forest fragmentation, edge effects, and disturbance. Decreases in forested land cover was related to higher soil chroma suggesting reduced soil saturation, higher soil temperatures, and/or increased sedimentation. Both reduced soil saturation and increased sediment input would suggest an increased importance of surface water inputs. We observed shifts in function related to changing LULC; however, in calculating a standard index for function (i.e. HGM) we seemed to lose some clarity in describing the system. While the HGM may produce consistent values, these values may not adequately explain how a specific function may shift due to LULC. We believe further investigation of particular functions and variable scores (especially hydrology) may provide necessary information to potentially improve HGM results. Further investigation may allow for greater understanding for improved management and conservation of these headwater systems.

2.7 References

Abelho, M. 2001. From litter fall to breakdown in streams: a review. The Scientific World. 1:656-80.

Allan, J.D. 2004. Landscapes and Riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics. 35:257-84.

Anderson, J.R., E.E. Hardy, J.T. Roach, R.E. Witmer. 1976. A land use and land cover classification system for use with remote sensor data. Geological Survey Professional Paper 964.

Baker, L.A. 1992. Introduction to nonpoint source pollution in the United States and prospects for wetland use. Ecological Engineering. 1:1-26.

Baldwin County Planning and Zoning Department. 2005 The Baldwin county conservation plan-final summary document. Bay Minette, Alabama.

Blann, K.L, J.L. Anderson, G.R. Sands, and B. Vondracek. 2009. Effects of Agricultural Drainage on Aquatic Ecosystems: A Review. Critical Reviews in Environmental Science and Technology 39:909-1001.

Brinson, M. 1996. Assessing wetland function using HGM. National Wetland Newsletter January-February: 10-16.

Brinson, M.M., A.E Lugo, and S. Brown. 1981. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics. 12:123-61.

Chadwick, M.A., D.R. Dooberfuhl, A.C. Benke, A.D. Huryn, K. Suberkropp, J.E. Thiele. 2006. Urbanization affects stream ecosystem function by altering hydrology, chemistry, and biotic richness. Ecological Applications 16(5):1796-1807.

Crim J.F. 2007. Water quality changes across an urban-rural gradient in streams of the west Georgia piedmont. M.S. Thesis, Auburn University, Auburn AL.

DeLaney, T.A. 1995. Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. Journal of Soil and Water Conservation 50(6):620-28.

Ehrenfeld, J.G., H.B. Cutway, R. Hamilton IV, and E. Stander. 2003. Hydrologic description of forested wetland in Northeastern New Jersey, USA- and urban/suburban region. Wetlands 23(4): 685-700.

ESRI. ArcGIS 9.3. Redlands, CA: Environmental Systems Research Institute (ESRI). 2009.

Faulkner, S. 2004. Urbanization impacts on the structure and function of forested wetlands. Urban Ecosystems. 7:89-106.

Galatowitsch, S.M., D.C. Whited, R. Lehtinen, J. Husveth, and K.Schik. 2000. The vegetation of wet meadows in relation to their land-use. Environmental Monitoring and Assessment 60:124-44.

Gemborys S.R. and E.J. Hodkins. 1971. Forests of small stream bottoms in t eh coastal plain of Southwestern Alabama. Ecology 52:1.

Glazebrook, H.S. and A.I. Robertson. 1999. The effect of flooding and flood timing on leaf litter breakdown rates and nutrient dynamics in a river red gum (*Eucalyptu camaldulensis*) forest. Austrailian Journal of Ecology. 24: 625-35.

Gleason, R.A. and N.H. Euliss Jr. 1998. Sedimentation of prairie wetlands. Great Plains Research 8:97-112.

Herlihy, A.T., J.T. Stoddard, and C.B. Johnson. 1998. The relationship between stream chemistry and watershed land cover data in the mid-Atlantic region, U.S. Water, Air, and Soil Pollution 105 (1-2):377-86.

Houlahan, J.E., P.A. Keddy, K. Makkay, and C.S. Findlay. 2006. The effects of adjacent land use on wetland species richness and community composition. Wetlands 26(1): 77-96.

Jones, J.A., F.J. Swanson, B.C. Wemple, and K.U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14(1): 76-85.

Kaye, J.P., P.M. Groffman, N.B. Grimm, L.A. Baker, and R.V. Pouyat. 2006. A distinct urban chemistry? Trends in Ecology and Evolution. 21(4):192-99.

King, R.S., M.E. Baker, D.F. Whigham, D.E. Weller, T.E. Jordan, P.F. Kazyak, and M.K. Hurd. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15(1):137-53.

Lundgren, M.R., C.J. Small, G.D. Dreyer. 2004. Influence of land use and sites characteristics on invasive plant abundance in the Quinebaug Highlands of Southern New England. Northeastern Naturalist 11(3):3313-32.

McBride, E.H. and L.H. Burgess. 1964. Soil survey of Baldwin County, Alabama. U.S. Department of Agriculture., Soil Conservation Service. Soil Survey Report, 12: 110.

Messina, M.G. and W.H. Conner. 1998. Southern Forested Wetlands: Ecology and Management. Lewis Publishers. Boca Raton.

Miller, S.J., D.H. Wardrop, W.M. Mahaney, and R.P. Brooks. 2006. A palnt-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecological Indicators 6(2):290-312.

Mitchell, J.D., B.G. Lockaby, E.F. Brantley. 2011. Influence of Chinese privet (*Ligustrum sinense*) on decomposition and nutrient availability in riparian forests. Invasive Plant Science and Management 4:437-47.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. 3rd ed., John Wiley and Sons, New York.

Moffatt, S.F., S.M. McLachlan, and N.C. Kenkel. 2004. Impacts of land use on riparian forest along an urban – rural gradient in southern Manitoba. Plant Ecology 174:119-35.

Nagy, R.C., B.G. Lockaby, L. Kalin, and C. Anderson. 2011. Effects of urbanization on stream hydrology and water quality: the Florida Gulf Coast. Hydrological Processes DOI: 10.1002/hyp.8336.

Naiman, R.J., H. Decamps, M.E. McClain. 2005. Riparia. Elsevier. Amsterdam.

Niggemann, M., J. Jetzkowitz, S. Brunzel, M.C. Wichmann, R.Bialozyt. 2009. Distribution of plants explained by human movement behavior. Ecological Modeling. 220:1339-46.

Noble, C.V., J.S. Wakeley, T.H. Roberts and C. Henderson. 2007. Regional guidebook for applying the hydrogeomorphic approach to assessing the functions of headwater slope wetlands on the Mississippi and Alabama coastal plains. U.S. Army Corps of Engineers.

Oneal, A.S. and J.T. Rotenberry. 2008. Riparian plant composition in an urbanizing landscape in southern California, U.S.A. Landscape Ecology 23:553-67.

Paul, M.J. and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology Systematics 32:333-65.

Poff, N.L., B.P. Bledsoe, C.O. Cuhaciayan. 2006. Hydrologic variation with land use across the contiguous United States: geomorphic and ecological consequences for stream ecosystem. Geomorphology: 264-85.

R Development Core Team 2010. R: A language and environment for statistical computing, reference index version 2.12.12. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Reed Jr., P.B. 1988. National list of plant species that occur in wetlands: national summary. Biological Report 88:24.

Robb, D.M. 1992. The role of wetland water quality standards in nonpoint source pollution control strategies. Ecological Engineering 1:143-48.

Schoonover, J.E., B.G. Lockaby, B.S. Helms. 2006. Impact of land cover on stream hydrology in the West Georgia piedmont, USA. Journal of Environmental Quality 35:2123-31.

U.S. Census Bureau. 2010. 2010 Census Data. http://2010.census.gov/2010census/data/.

von der Lippe, M. and I. Kawarik. 2007. Long-distance dispersal of plants by vehicles as a driver of plant invasions. Conservation Biology 21(4):986-96.

Wallace, J.B., S.L. Eggert, J.L. Meyer, and J.R. Webster. 1999. Effects of resource limitation on detrital-based ecosystems. Ecological Monographs 69(4):409-42.

Walsh, C.J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman, and R.P. Morgan II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society. 24(3):706-23.

Wentworth, T. R. and G.P. Johnson, and R.L. Kologiski. 1988. Designation of wetlands by weighted averages of vegetation data: a preliminary evaluation. American Water Resources Association. 24(2):389-96.

Xiong, S. and C. Nilsson. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: a review. The Botanical Review 63(3):240-64.

Zedler, J.B. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecological Environments 1(2):65-72.

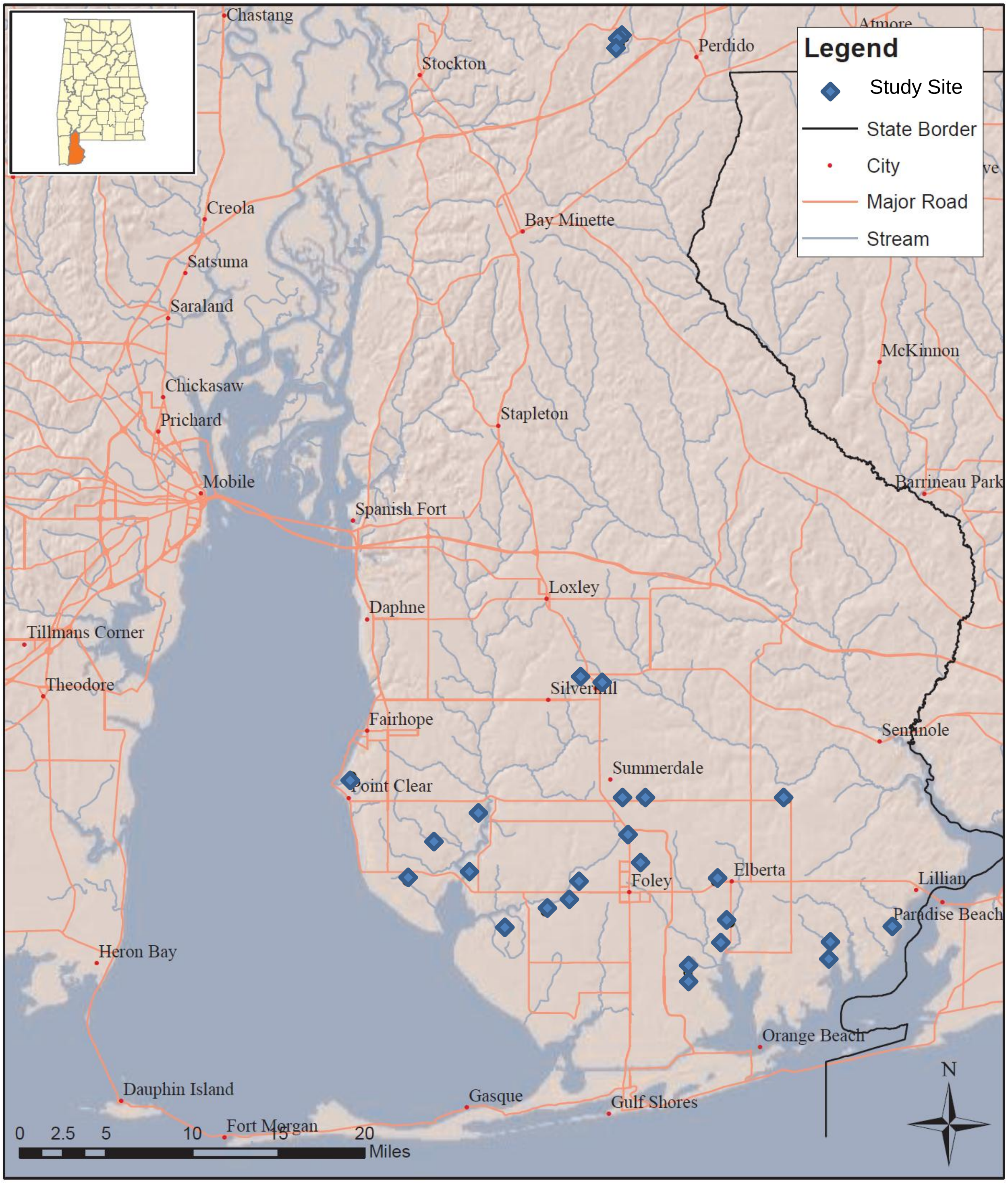

Figure 2.1: Map of Baldwin County, Alabama displaying the distribution of 30 headwater wetlands sampled summer 2009.

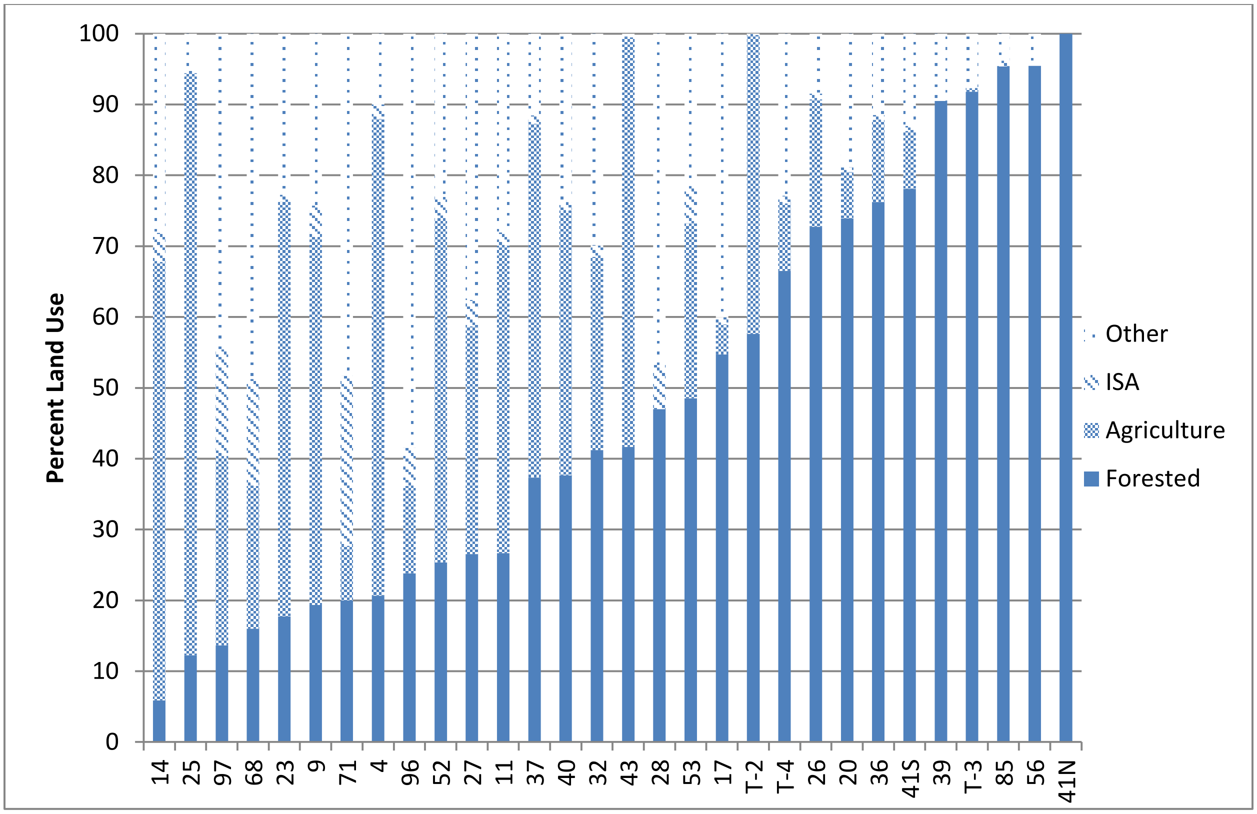

Figure 2.2: Relative land use classification of 30 study watersheds. Other classification is composed primarily of dirt road and open space, but includes all categories not defined as agricultural, forested or ISA classifications.

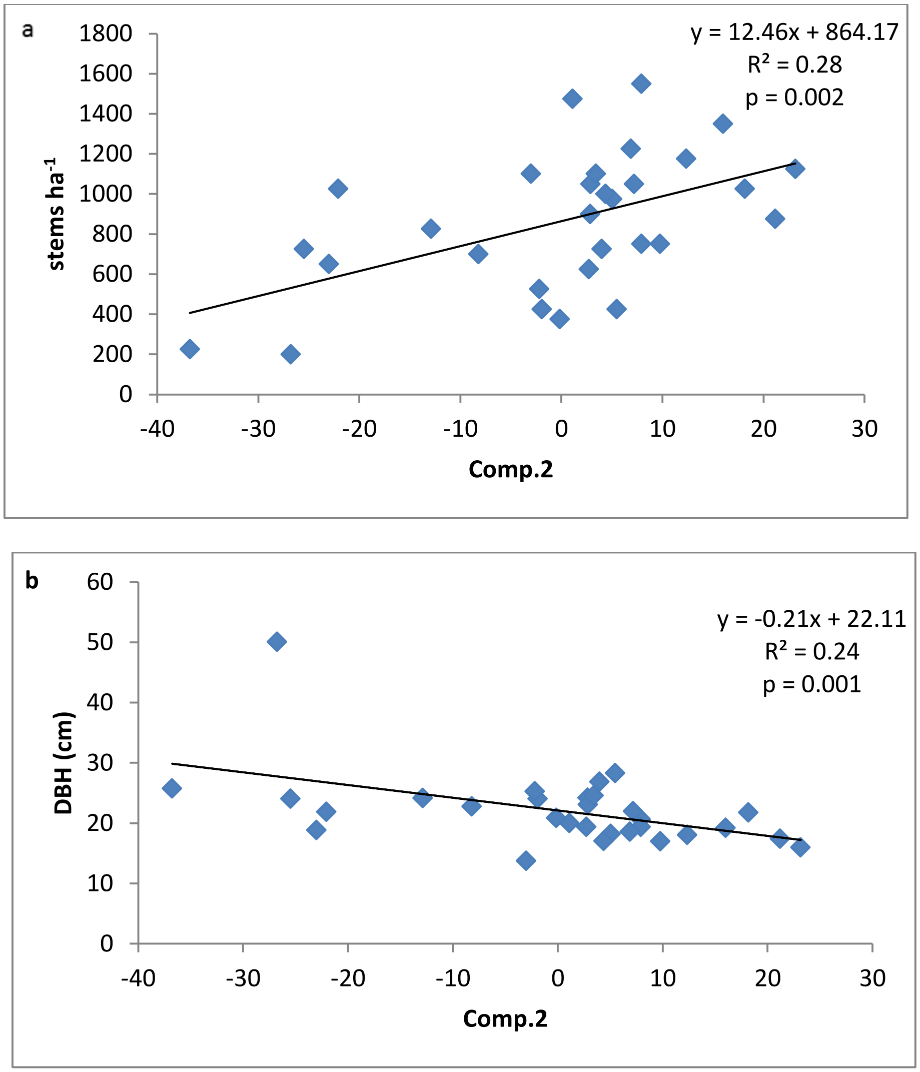

Figure 2.3: Linear regression relationships between a) mean canopy tree density and b) mean canopy tree diameter with Comp. 2 .

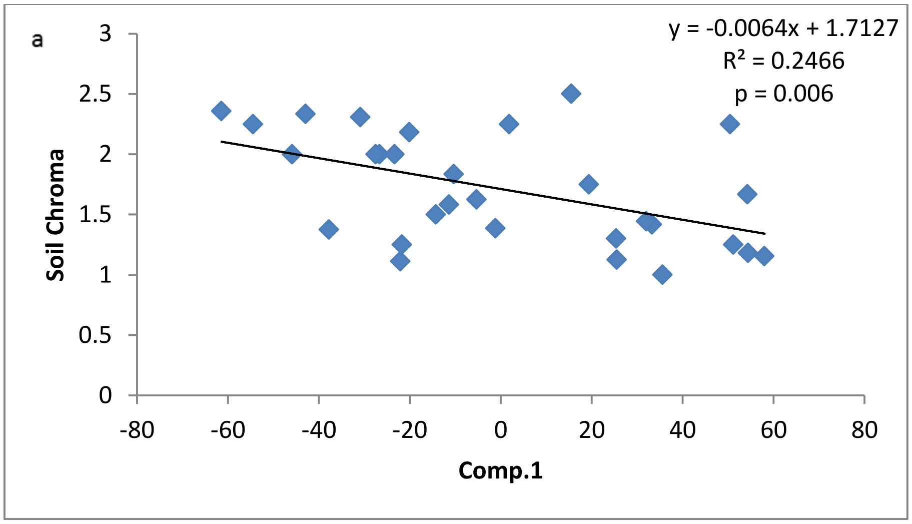

Figure 2.4: Linear regression relationship between soil chroma and Comp. 1.

Table 2.1: General characteristics of 30 headwater wetland study sites. Adjusted watershed area reflects hydrologic changes in landscape due to anthropogenic disturbances.

Wetland ID	Historic Watershed Area	Adjusted Watershed Area	Percent Change	Wetland Area	Percent Wetland:Watershed Area	Percent Land use within Watershed		
						ISA	Agriculture	Forest
4	143.5	139.7	2.7	1.4	0.98	2.0	67.3	20.7
9	83.2	83.2	0.0	4.4	5.26	4.4	52.0	19.3
11	66.9	55.0	17.8	5.4	9.84	1.8	43.4	26.7
14	294.7	278.0	5.7	4.6	1.67	4.1	61.9	5.9
17	81.5	78.4	3.8	0.7	0.91	1.0	4.2	54.7
20	28.5	28.5	0.0	0.5	1.68	0.7	6.6	73.9
23	79.0	76.8	2.8	3.3	4.25	1.0	58.6	17.7
25	30.7	29.4	4.0	0.3	1.21	0.3	82.2	12.3
26	87.9	87.9	0.0	1.9	2.15	0.8	18.0	72.7
27	229.6	293.9	28.0	2.6	0.87	3.7	32.2	26.5
28	57.8	62.9	8.7	0.8	1.29	6.2	0.0	47.0
32	147.0	127.8	13.1	0.5	0.40	1.6	27.3	41.2
36	48.2	48.1	0.2	0.1	0.22	0.7	11.6	76.2
37	43.7	41.1	6.1	2.3	5.65	1.2	50.0	37.3
39	22.0	22.0	0.0	0.4	1.74	0.0	0.0	90.5
40	109.9	97.9	10.9	1.4	1.45	1.2	37.4	37.6
43	1.9	1.9	0.0	1.5	79.78	0.0	57.8	41.7
52	147.9	132.5	10.4	1.4	1.08	3.0	48.6	25.3
53	184.0	113.0	38.6	0.6	0.56	5.2	24.7	48.5
56	31.3	11.5	63.3	1.3	10.88	0.0	0.0	95.5
68	141.6	151.1	6.7	9.4	6.24	15.1	20.2	16.0
71	36.8	39.5	7.4	2.7	6.81	24.0	7.7	20.0
85	60.3	60.3	0.0	21.0	34.89	0.8	0.0	95.4
96	31.7	30.5	3.8	2.1	6.83	5.4	12.3	23.8
97	106.1	140.1	32.1	5.2	3.72	15.0	26.8	13.7
41N	16.7	16.7	0.0	2.7	15.81	0.0	0.0	100.0

Table 2.1 continued: General characteristics of 30 headwater wetland study sites. Adjusted watershed area reflects hydrologic changes in landscape due to anthropogenic disturbances.

Wetland ID	Historic Watershed Area	Adjusted Watershed Area	Percent Change	Wetland Area	Percent Wetland:Watershed Area	Percent Land use within Watershed		
						ISA	Agriculture	Forest
41S	214.6	194.9	9.1	3.2	1.63	0.7	8.2	78.1
T2	16.4	16.4	0.0	3.1	18.99	0.2	42.2	57.6
T3	95.2	81.3	14.6	4.3	5.34	0.1	0.5	91.8
T4	46.6	46.6	0.0	6.5	13.99	1.1	9.5	66.5

Table 2.2: Loadings (correlation coefficients) of land use variables and principal components.

Land Use	Comp. 1	Comp. 2
Forested	0.79	0.57
Agriculture	-0.60	0.77
ISA	- -	-0.30

Table 2.3: Species list, indicator status, and percent of wetlands in which a species was present.

Species	Indicator status	Percent	Species	Indicator status	Percent
		Canopy Tree Species			
Magnolia virginiana	FACW+	97	*Fraxinus pennsylvanica*	FACW	20
Acer rubrum	FAC	77	*Nyssa sylvatica*	FAC	20
Nyssa biflora	OBL	52	*Magnolia grandiflora*	FAC+	10
Sapium sebifera	FAC	47	*Nyssa aquatica*	OBL	7
Quercus nigra	FAC	43	*Persea palustris*	OBL	7
Pinus elliotii	FACW	37	*Liquidambar styraciflua*	FAC+	7
Persea borbonia	FACW	30	*Prunus serotina*	FACU	3
Liriodendron tulipifera	FAC	23			
		Sapling/Shrub Species			
Ligustrum sinense	FAC	77	*Ilex opaca*	FAC-	7
Cyrilla racemiflora	FACW	37	*Calicarpa americana*	FACU-	7
Myrica cerifera	FAC+	37	*Myrica inodora*	OBL	3
Sambucus canadensis	FACW-	10	*Ilex vomitoria*	FAC	3
Itea virginica	FACW+	10	*Alnus serrulata*	FACW+	3
Ilex glabra	FACW	7			
		Herbaceous/vine species			
Woodwardia areolata	OBL	73	***Commelina cummunis***	FAC	3
Osmunda cinnamomea	FACW+	47	*Cyperus retrorsus*	FACU-	3
Arundinaria gigantea	FACW	30	*Decondon verticillatis*	OBL	3
Osmunda regalis	OBL	27	***Lygodium japonicum***	FAC	3
Rubus spp.	UPL	17	*Parthenocissus quenquefolia*	FAC	3
Sphagnum spp.	OBL	13	*Peltandra virginica*	OBL	3
Vitus rotundifolia	FAC	13	*Pilea pumila*	FACW	3
Juncus spp.	OBL	10	*Sagittaria lancifolia*	OBL	3
Pteridium aquilinum	FACU	10	*Smilax rotundifolia*	FAC	3
Smilax laurifolia	FACW+	10	*Thelypteris thelypteroides*	FACW+	3
Ptilimnium capillaceum	OBL	7	*Cinna arudinacea*	FACW	3
Sagittaria latifolia	OBL	7	*Toxicodendron radicans*	FAC	3
Vaccineum spp.	FAC	7			

Non-native species in bold text.

Table 2.4: Importance values for 15 canopy species and relation to PCA components. Pearson's correlation coefficients for independent variables explained by principal components (Comp. 1 and Comp.2).

Species	Maximum	Mean	Minimum	Standard Error	Comp.1	Comp.2
Magnolia virginiana	233.4	102.2	0.0	9.4	0.06	0.15
Nyssa. biflora	272.5	85.4	0.0	11.9	0.12	-0.09
Acer rubrum	119.1	31.1	0.0	6.7	-0.16	-0.25
Pinus elliottii	151.3	26.0	0.0	7.6	-0.25	0.22
Sapium sebifera	213.9	25.4	0.0	9.1	0.03	0.09
Lireodendron tulipefera	78.2	10.9	0.0	3.8	-0.06	-0.05
Quercus spp.	53.9	9.0	0.0	2.9	0.12	-0.32
Cinnamomum camphora	57.1	3.9	0.0	2.3	0.20	0.01
Ligustrum sinense	25.3	2.9	0.0	1.2	-0.07	-0.11
Gordonia lasianthus	30.5	1.0	0.0	1.0	-0.05	0.27
Cyrilla racemiflora	23.3	0.8	0.0	0.8	0.21	-0.05
Ilex americana	20.1	0.7	0.0	0.7	-0.06	-0.05
Magnolia grandiflora	17.4	0.6	0.0	0.6	0.20	-0.01
Persea palustris	15.6	0.5	0.0	0.5	-0.02	0.02
Sambucus canadenensis	15.2	0.5	0.0	0.5	0.15	-0.10

Table 2.5: General statistics for environmental variables and FCI scores. PI=Prevalence index. Pearson's correlation coefficients for independent variables explained by principal components (Comp. 1 and Comp.2).

Variable	Maximum	Mean	Minimum	Standard Error	Comp. 1	Comp.2
DBH(cm)	50.1	22.1	13.7	1.2	-0.05	-0.48**
Density(stems/ha)	1550	864	200	62.7	-0.17	0.53**
Basal Area (m^2/ha)	52.2	30.5	11.5	2.1	-0.17	0.11
Canopy Cover (%)	88.8	65.2	31.3	3.2	-0.01	0.49**
Exotic Canopy Cover (%)	65.0	5.0	0.0	2.5	-0.38*	0.02
Shrub Cover (%)	73.8	43.2	7.5	3.6	-0.13	-0.47**
Exotic Shrub Cover (%)	67.5	22.1	0.0	4.3	-0.44**	-0.44**
Ground Cover (%)	81.2	36.3	2.0	4.5	0.25	0.17
Exotic Ground Cover (%)	34.0	3.3	0.0	1.2	-0.44*	0.25
Detritus Cover (%)	100.0	78.3	25.0	4.1	0.30	0.01
Hydrology (cm)	30.0	4.7	0.0	1.8	0.19	0.10
Soil Value	4.0	2.8	2.0	0.1	-0.34	-0.29
Soil Chroma	2.5	1.7	1.0	0.1	-0.50**	-0.05
PI Overstory	2.8	2.1	1.3	0.1	- 0.01	-0.03
PI Midstory	3.0	2.6	1.9	0.1	-0.34	- 0.10
PI Understory	4.0	2.0	1.0	0.1	-0.30	-0.10
Wetland Functional Capacity Index						
Cycle Carbon	0.99	0.80	0.64	0.00	0.24	-0.28
Vegetative Community	0.97	0.61	0.36	0.01	0.19	- 0.21
Water Storage	0.99	0.80	0.62	0.00	0.23	- 0.23
Wildlife Habitat	0.99	0.68	0.42	0.01	0.48*	-0.09

Significance:'*'p<0.05, '**'p<0.01, '***'p<0.001.

CHAPTER III

SHIFTS IN CARBON CYCLING DUE TO HYDROLOGICAL ALTERATIONS AS THE RESULT OF LAND USE CHANGE

3.1 Abstract

The effects of land use and land cover (LULC) on aquatic systems has been extensively studied, however very little research has focused on headwater wetlands. The objective of this study was to evaluate the effects of watershed LULC on carbon cycling and hydrology of headwater wetlands in coastal Alabama. Although headwater wetlands are commonly groundwater driven, increased forest conversion by agriculture and urban land use may increase surface water flow and effectively export leaf litter from wetlands. A total of 15 headwater slope wetlands were selected to reflect a range of surrounding land use across coastal Alabama. Land use and soil data were utilized to calculate runoff curve numbers (CN) (influenced by LULC and soil characteristics) and used with precipitation data to calculate total runoff (P_e) from each watershed. Within each wetland, water level recorders were installed to monitor ground water for one year and generate hydrological measures of water level. Further hydrological metrics were generated to assess the flashiness of groundwater levels, including: Richard-Baker index (RB), water level rate of change (WLΔ), water level variability (WLV), percent of time the water table was below 50-cm of ground surface ($WT_{<-50}$) and percent of time the water table was within 20-cm of ground surface ($WT_{\pm 20}$). In addition to hydrologic measures, carbon dynamics associated with leaf litter were also monitored during this time period. Leaf litter traps were installed to determine litter fall carbon input (C_{LF}). Bimonthly monitoring of forest floor carbon (C_{FF}) was conducted within each wetland to evaluate its relationship to watershed runoff. Total soil carbon

was quantified in each wetland by collecting 1-m cores with a 5.8cm diameter corer and analyzing 10-cm sections. RB was the only hydrological metric significantly related to land use and decreased with increasing CN. Watershed runoff characteristics (CN and P_e) were significantly related to C_{FF}:C_{LF} and mean C_{FF} values suggesting that hydrologic export and/or decomposition of forest floor litter increased with increasing runoff. Soil carbon and bulk density (BD) were related to longer periods of high water tables, indicating the importance of soil saturation and the storage of organic carbon.

3.2 Introduction

Conversion of forest land for urban development and agricultural production can result in significant alterations to streams, wetlands, and other aquatic systems (DeLaney 1995; Messina and Conner 1998; Faulkner 2004). Decreasing forest cover on the landscape has commonly been shown to increase water yield due to reductions in evapotranspiration and canopy interception (Grace 2005). Other land alterations can change how water moves through a watershed. Soil disturbance and various drainage techniques associated with agriculture can alter hydrologic conditions and increase sediment and nutrient loading in runoff (De Laney 1995; Zedler 2003). Similarly, urbanization modifies the landscape through conversion of forested land to impervious surfaces. Due to increased impervious surface area (ISA), urban land use often impacts local hydrology, water chemistry and stream morphology (Paul and Meyer 2001). A general understanding of the effects of LULC on stream and wetland systems has been developed (Walsh et al. 2005; Nagy et al. 2011), however, there are gaps in our understanding related to headwater systems.

Wetland functions are closely linked to forest structure, composition, and adjacent land use. As these environments are impacted, species composition and forest structure may change. Hydrology is the key driver in wetland ecosystems, due to its influence on vegetation, soils, fauna, and biogeochemical cycles (Ehrenfeld et al. 2003). Changes in hydrology as a result of LULC can cause decreases in ground water recharge, increased surface water flow frequency and flow velocities, less infiltration of rain water, lower base flows, and greater amounts of erosion (Ehrenfeld et al. 2003; Walsh et al. 2005; Chadwick et al. 2006). These changes can influence ecosystem processes through shifts in vegetation, loss of wildlife biodiversity, alterations in aquatic food webs, and decreased water storage and flood abatement. (Baker 1992; Robb 1992; Gleason and Euliss 1998; Jones et al. 2000; Zedler 2003). Water storage within wetland environments provides stable water supplies to stream systems, which can be lessened due to alterations in the landscape influencing aquatic biodiversity (Baker 1992; Brinson 1996). These impacts can also have substantial effects on wetland forest productivity and carbon cycling. Litter fall production has been shown to be closely linked to the magnitude and extent of flooding as well as soil water content (Brinson 1990). This can be critical for forest floor dynamics as leaf litter fall can account for 72% of total litter production in broad-leaved riparian forests and 80% in needle-leaved systems (Xiong and Nilsson 1997). Forest floor stock also has the potential to be redistributed by flood waters. Cuffney (1988) reported that more frequently flooded riparian areas displaced greater amounts of litter material than less flooded systems. Based on the relationships between wetland hydrology, litter production and forest floor stocking, hydrological alterations normally have a substantial effect on biogeochemical cycling.

Biogeochemical processes such as carbon cycling and decomposition rates are driven by many factors including soil moisture, temperature, litter quality, pH and redox conditions (Baker et al. 2001; Lee and Bukaveckas 2002; Lecerf et al. 2007). Changes in hydroperiod have the ability to influence these factors and alters the capacity for wetlands to retain carbon. For example, decomposition is often optimized when a cycle of wetting and drying exist within a wetland (Brinson et al. 1981; Baker et al. 2001), implying that any hydrologic alteration that causes ponding or drying of a wetland will have significant effects on the carbon cycle. The loss of forest land cover can also influence the amount of carbon coming into wetlands. Linkages between upland and wetland ecosystems are often important to the productivity of wetlands, as they can receive substantial organic matter and nutrients from surrounding upland landscapes (Messina and Conner 1998; Abelho 2001). The cycling of this material in wetlands is normally the combination of physical breakdown (abrasion, fragmentation, and leaching), decomposition, and hydrological export, with the pace being strongly tied to hydroperiod (Webster and Benfield 1986; Baker et al. 2001). Based on consistent soil saturation reported for headwater wetlands (Noble et al. 2007), we expect these wetland environments would act as sinks for organic carbon. Based on the potential link between LULC, hydrology and the cycling of carbon we expect wetlands within converted watersheds to function less as carbon sinks than wetlands within more forested environments. The objective of this study was to examine if land use change and related hydrological alterations may influence carbon cycling within headwater wetlands in coastal Alabama. Examining wetlands along a gradient of increasing runoff potential, corresponding shifts in hydrology and carbon cycling (litter fall production, forest floor dynamics, and soil carbon) were expected. Specifically, I hypothesized that decreases in natural forest cover would increase runoff and decrease groundwater levels within these wetlands resulting in decreased litter layer stock and soil carbon.

3.3 Methods

3.3.1 Description of Study Sites

Headwater wetlands for this study were located in Baldwin County, AL, within the Southern Coastal Plain physiographic region (Fig. 1). The region is characterized by mild winters and hot and humid summers with mean annual temperatures ranging from 15 to 21˚C, and annual precipitation ranging from 125 to 180cm which is fairly evenly distributed throughout the year (Noble et al. 2007). Headwater wetlands in coastal Alabama are groundwater driven and located at the headwater reaches of first order streams (Noble et al. 2007). These wetlands are typically comprised of alluvial soils that are classified as “wet loamy alluvial lands”, while the uplands are generally sandy as soils derived from marine deposits (McBride and Burgess 1964). In June 2009, approximately 60 headwater slope wetlands were identified as potential study sites using aerial imagery, National Wetlands Inventory maps, topographic maps, and the Soil Survey of Baldwin County, Alabama (McBride and Burgess 1964). Each wetland was then visited to conduct a preliminary assessment. From these wetlands, a total of 15 headwater slope wetlands were selected that displayed a range of surrounding LULC (e.g.: urban, agriculture, forest) typical of the region (Table 3.1). Wetlands selected for complete assessment were forested (with evidence of characteristic species) and comparable in size (both wetland and watershed) to reduce confounding factors that may affect wetland hydrology and organic carbon cycling (subset of Chap. 2 wetlands).

3.3.2 Watershed Runoff Calculations

To describe the effect of both land use and soil conditions, we utilized the Soil Conservation Service (SCS) curve number method for abstraction to calculate excess precipitation (runoff) for each study watershed. The associated watershed for each wetland was delineated using the ArcSWAT 2009.93.6 model for ESRI ArcMap 9.3 (2009). Land use data developed with 2005 aerial images were used to determine watershed LULC (BCZPD 2005). Soil permeability was defined by hydrological soil groupings (HSG) based on soil texture (A= sandy, loamy sand, or sandy loam; B= silt loam or loam; C= sandy clay loam; D= clay loam, silty clay loam, sandy clay, silty clay, or clay) with A having the highest permeability and D having the lowest (USDA 2007; personal communication: Baldwin County, Alabama, NRCS). Weighted runoff curve numbers (CN) were calculated using percent LULC data and HSG to account for the combined influence of land use and soil permeability (SCS curve number method; [Mishra and Singh 2004, Hawkins et al. 2009]). A reference CN was calculated for each watershed and compared to current CN to demonstrate the effect of land use change on CN by assuming watersheds were completely forested prior to European settlement. A significant and positive relationship between forested land cover loss on CN was detected ($p=0.0004$), Fig. 3.3. Utilizing NEXRAD data to establish watershed specific estimates for daily precipitation (P) (SCONC 2012), potential maximum retention of rainfall after runoff (S), initial abstraction (retention of rainfall) (I_a), and excess precipitation (P_e) (cm per watershed) were calculated using the Soil Conservation Service (SCS) curve number method (Mishra and Singh 2004; Hawkins et al. 2009). Original assumptions regarding the relationship of I_a and S ($I_a=0.2S$; $\lambda=0.2$) have shown that the general relationship $I_a=0.05S$ ($\lambda=0.05$) is more realistic (Jiang 2001). Therefore, adjusted runoff curve numbers ($CN_{0.05}$, hence forth denoted CN) were calculated to represent these improvements (Jiang 2001; Hawkins et al. 2009) using the following equations:

For calculation of P_e, P must be $\geq \lambda * S$:

$$P_e = \frac{(P - I_a)^2}{P + I_a + S}$$

$$I_a = \lambda * S$$

$$S = \frac{2540}{CN} - 100 \ \ (mm)$$

$$CN_{0.05} = \frac{100}{1.879[\frac{100}{CN_{0.20}} - 1]^{1.15} + 1}$$

The SCS method for abstraction is not valid when CN < 40 (Hawkins et al. 2009). As a result any CN < 40 was set at 40 for use in the calculation of P_e.

3.3.3 Wetland Hydrology

Within each wetland an In-Situ Mini-Troll 500 pressure transducer was installed to monitor water table levels. Each transducer was installed within a shallow well using a 1.5m perforated PVC pipe located at the lowest point of elevation within each wetland. The transducers were programmed to measure water level at 15-min intervals and suspended within the well casing at a depth of 1.25m to measure changes in the groundwater levels and allow 0.25m of well casing for the deposition of detritus and soil particles which may have accumulated over the monitoring period. Groundwater hydrology was monitored from February 25, 2011 – March 23, 2012. Study wetlands experienced drought conditions throughout the summer of 2011 (June through November 2011) in which water tables fell below the detection depth of the pressure transducers. Water level median and standard deviation (representing water level variability [WLV]) were calculated as measurers of stage and changes in groundwater hydrology. The Richard-Baker index (RB) was adapted, as a measure of flashiness and hydrologic response of groundwater stage (y) within each wetland. The standard RB index is calculated as follows:

$$RB = \frac{\sum_{i=1}^{n} |q_i - q_{i-1}|}{\sum_{i=1}^{n} q_i}$$

where discharge (q) is utilized rather than stage. Due to the inherently low variability of ground water levels, RB was multiplied by 1000 for demonstration purposes. Our adapted RB index utilizing stage (y) was calculated as follows:

$$RB = \left(\frac{\sum_{i=1}^{n} |y_i - y_{i-1}|}{\sum_{i=1}^{n} y_i}\right) * 1000$$

Measurements were automatically recorded every 15 minutes and averaged to produce an hourly value which is represented by "i". Sample size (n) averaged approximately 9000 across 15 study wetlands. Similarly, an index representing the rate of change of groundwater levels (WLΔ) was introduced for each wetland:

$$\text{WL}\Delta = \left(\frac{\sum_{i=0}^{n} |y_i - y_{i-1}|}{n}\right) * 10$$

As with RB, values were adjusted in magnitude for demonstration purposes. Sample size (n) was consistent with RB, as was the time interval (i). Due to drought conditions, many water levels dropped below detection levels and hydrological metrics (WLV, RB and WLΔ) were not possible. Therefore, these measures were only compared among concurrent sampling periods with measurable water table depths. Water table depths collected between 04/24/2012-07/17/2012 were omitted.

To evaluate water table proximity to soil surface, the percent of time the water table was below 50 cm of the ground surface (WT_{-50}), below 20 cm (WT_{-20}), within 20 cm ($WT_{\pm 20}$), and above 20 cm (WT_{+20}) was calculated for the entire hydrologic monitoring period.

3.3.4 Leaf Litter Measurements

Four 100-m^2 plots were established within each wetland to be representative of the environment. To quantify carbon inputs from litter fall, a leaf litter trap was randomly established in each plot. Each trap was 0.5m x 0.5m x 0.15m and staked (~1 m above ground surface) to prevent movement during high water events. Traps were placed in February 2011, with first collection occurring in May 2011 and collected bimonthly for the duration of the study. To account for three months of litter fall collection at the beginning of the study, daily accrual rates were used to estimate 2 months (60 days) of collection. Two sites (68 and 71) were added in May with their first litter fall collection occurring in July. Litter fall daily accrual rates were calculated for both wetlands and used to estimate litter fall for May 2011. Leaves were air dried and weighed to measure litter fall dry weight. Litter fall carbon (C_{LF}) was calculated based on a subsample of dry leaf material analyzed for total C concentration using the dry combustion method utilizing an Elementar Vario Macro CNS analyzer (Kirsten 1979). Litter fall dry weight and C concentration were multiplied and g-C m^{-2} yr^{-1} were calculated based on annual totals.

To quantify carbon dynamics on the forest floor, changes in the standing stock of litter were measured to account for forest production (inputs) and decomposition/hydrologic export (outputs). We monitored changes in litter for one year by collecting 4 random samples within each wetland plot on a bimonthly basis using a 28-cm diameter metal ring, all recognizable litter within the sampling ring was collected, returned to the laboratory, and oven-dried at 65˚C for 72 hours. As with C_{LF}, samples were processed for estimation of total forest floor carbon concentration per the dry combustion method utilizing an Elementar Vario Macro CNS analyzer (Kirsten 1979) and average forest floor carbon content was calculated per wetland (C_{FF}) (g-C m^{-2} yr^{-1}). A ratio of average C_{FF} (per sampling event) to total C_{LF} (C_{FF}:C_{LF}) was calculated as an index of average forest floor carbon content, while accounting for litter fall inputs to aid in the detection of carbon import from adjacent uplands. Wetlands with C_{LF}:C_{FF} closer to 1 were interpreted as having a higher capacity to retain litter fall.

3.3.5 Soil carbon

To measure carbon stock in the soil (C_{SOIL}), 1-m cores were collected using a 5.8 cm diameter corer with subsamples collected at the following increments: 0-10cm, 30-40cm, 60-70cm, 80-90cm. For some wetlands, fully saturated soil conditions made intact coring impossible due to the unconsolidated nature of the soil profile. For these wetlands, a one meter PVC pipe, with one end sharpened, was inserted into the soil and suctioned to extract the sample. Compaction of cores was unavoidable yet accounted for by monitoring the depth of compaction and as the sample was collected. Compaction was minimal once certain depths were reached and it was assumed that the sample was representative of the first meter of the soil profile. A LECO CNS 2000 carbon analyzer was used to analyze soil samples for C concentration through the dry combustion method (Kirsten 1979) and utilized to calculate carbon content (g-C m^{-2}). Bulk density for each increment was measured as the dry weight of soil per volume (g cm^{-3}). Total soil carbon (C_{SOIL}) (g-C m^{-3}) was calculated for each increment with an average value calculated across depths to represent stored carbon within the first meter of the soil profile. Surface soil carbon (C_{SURF}) (g-C m^{-3}) was determined as total carbon within 0 – 10 cm of the soil profile, while subsurface soil carbon (C_{SSURF}) (g-C m^{-3}) was determined within 30-40 cm of the soil profile.

3.3.6 Statistical Analysis

Data were analyzed using R 2.12.2 software (R Development Core Team 2010) to test the effects of land-use change on groundwater hydrology and carbon content and cycling of headwater wetlands. Simple linear regression models were generated using independent variables associated with watershed runoff (CN and P_e) to test for effects on wetland hydrologic variables (Median, WLV, $WT_{<-50}$, $WT_{<-20}$, $WT_{\pm 20}$, $WT_{>+20}$, R-B, and WLΔ). Both watershed runoff and wetland hydrological variables were then used as independent variables to test for relationships with carbon cycle metrics (C_{LF}, C_{FF}, C_{FF}:C_{LF}, C_{SOIL}, $C_{SURFACE}$, $C_{SUBSURFACE}$, BD). The Shapiro-Wilk's test for normality and the Bartlett's test for homoscedasticity of variance were utilized to test for assumptions with no transformations required. Statistical significance was set at α=0.05. Graphics were developed using Minitab 16.0 (Minitab Inc. 2010) and Microsoft Excel (Microsoft Corporation 2010).

3.4 Results

3.4.1 Watershed and Runoff Characteristics

Based on digital elevation models, watershed boundaries for all 15 headwater wetlands produced a mean area of 82.0 ± 14.1 ha (Table 3.1). Precipitation across the study area had a mean of 129.1 ± 3.7 cm while ranging from 107.9 – 149.2 cm with northern sites (T-2, T-3, and T-4) receiving less precipitation than others. CN ranged from 37.8-74.6 and watersheds produced an average P_e of 11.3 ± 1.7 cm with a range of 2.3 – 23.7 cm over the duration of the study period (Table 3.2). Historic CN ranged from 30.1 – 43.9 producing a mean value of 36.8

compared to an average current CN of 51.1 (Table 3.2) which was negatively correlated with forested land cover (Fig. 3.3).

3.4.2 Wetland Hydrology

During the summer and fall (June through November) of 2011, drought conditions occurred across the study area with the driest period occurring May - July (Fig. 3.2) (NCDC. May 20, 2012). During this drought, 7 of the 15 wetlands had water levels fall below their water level recorders, causing these periods of the data to be omitted. Wetlands that maintained high water levels tended to be the result of ponded conditions due to fill roads being immediately downstream of the wetland or the presence of an agricultural pond immediately adjacent to the wetland. Trends related to decreased quartile range of groundwater depth and increases in outlying measurements seemed to be associated with increasing CN (Fig. 3.4). In general as CN increased, quartile ranges decreased and the number of high surface flow events increased (as indicated by the number of outlying water measurements, Fig. 3.4). Average groundwater depth median was -28.6 ± 5.7 cm, and WLV, an indicator of water table stability, generated a mean of 19.9 ± 2.6 cm. R-B and WLΔ produced mean values of 1.51 ± 1.04 and 1.05 ± 0.44 respectively.. R-B was found to be significantly related to both CN and P_e ($p = 0.043$, 0.032 respectively) (Fig. 3.5). Utilizing the entire data set, $WT_{<-50}$ had a mean of 35.1 ± 7.2%. $WT_{<-20}$ and $WT_{>+20}$ had means of 59.5 ± 8.3% and 0.5 ± 0.3% respectively, while $WT_{<>\pm 20}$ had a mean value of 40.5 ± 8.3%.

3.4.3 Organic Carbon Cycling

Litter fall production (C_{LF}) ranged from 251.2 to 489.3 g-C m^{-2} yr^{-1} with a mean of 329.9 ± 15.4 g-C m^{-2} yr^{-1} (Table 3.4). No significant relationship was detected between C_{LF} and watershed runoff. Forest floor carbon (C_{FF}) had a mean of 202.0 ± 15.2 g-C m^{-2}, ranged from 77.4 to 290.9 g-C m^{-2} (Table 3.5) and had significant negative relationships with CN ($p = 0.001$), and P_e ($p = 0.010$) (Fig. 3.6 a and b). C_{FF}:C_{LF} averaged 0.63 ± 0.05 (Table 3.5) and had a significantly negative relationship with CN and P_e, (p = 0.004, 0.038, respectively) (Fig. 3.6 c and d).

C_{SOIL} ranged from 12.2 – 58.7 g-C m^{-3} with a mean of 32.5 ± 3.3 g-C m^{-3} while displaying no significant relationships with any wetland hydrology metrics. Total carbon (g-C m^{-3}) was also evaluated for each incremental soil sample. Samples collected from 0-10 cm, 30-40 cm, 60-70 cm, and 90-100 cm of the soil profile averaged 29.7 ± 3.8, 33.5 ± 4.5, 34.6 ± 6.3, 23.9 ± 4.3 g-C m^{-3} respectively (Table 3.4). A significant positive relationship was found between C_{SSURF} and median water table depth ($p = 0.011$). Bulk density (BD) ranged from 0.11 – 0.99 g-cm^{-3} with a mean of 0.55 ± 0.07 g-cm^{-3} and had a significant and positive relation to $WT_{<-20}$ and $WT_{<-50}$ (p = 0.026, 0.027 respectively), while being negatively related to $WT_{\pm 20}$ ($p = 0.026$) (Fig. 3.8).

3.5 Discussion

Based on the tight link between upland environments, wetland hydrology, and ecosystems processes (Walbridge 1993; Messina and Conner 1998) strong relationships between watershed runoff, forest floor metrics and watershed runoff conditions were expected. Similarly, soil carbon was expected to display relationships with groundwater metrics. This study showed a significant negative relationship between the RB and CN, describing increased flashiness as forested land cover decreases. Watersheds with higher CN displayed shorter lag-times with water levels that peaked and returned to base flow more rapidly. These effects of LULC on stream hydrology have been repeatedly demonstrated (Paul and Meyer 2001, Blann et al. 2009, Hardison et al. 2009, Nagy et al. 2011) and were apparent in this study.

Results from this study were unable to quantify direct correlation with watershed runoff (CN and P_e) and wetland water level responses such as higher peak flows and lower base flows. This could be attributed to the importance of agriculture in the region. Although agriculture has been shown to elicit some of the same effects on runoff as urban land use (higher peak flows, lower base flows; Walsh et al. 2005) there is often greater variability associated with hydrologic responses to agriculture (Blann et al. 2009). The effects of LULC may also be tempered by the gradual topography in the study area (Coastal Plain) compared to other topographic regions where land use changes may be more severe (Utz et al. 2009). Detection of water level differences could also be contributed to the level of drought our study area experienced during the summer of 2011.The drought limited data collection as water tables dropped below our instruments in 10 of 15 study watersheds and restricted measurements to only wetter periods of the year. The depth of water table during summer of 2011 was intriguing as ground water driven wetlands commonly maintain water tables at or near the ground surface (Mitsch and Gosselink 2001, Noble et al. 2007).

Decreases in forested cover and landscape drainage have been shown to increase runoff and potential energy of overland flows (Fig. 3.3, Meyer et al. 2004, Poff et al. 2006). This study supports this as mean forest floor carbon (C_{FF}) was negatively related to watershed runoff potential (CN, P_e) describing wetlands with less forest floor leaf litter as watershed forested land cover decreases. The ratio of average forest floor carbon to total carbon from litter fall (C_{FF}:C_{LF}) was also negatively correlated to CN and P_e, which indicates C_{FF} was related to CN and P_e regardless of total wetland C_{LF}. Precipitation events which create overland flow have the ability to alter forest floor organic matter by importing upland material and redistributing litter within the wetland area. Xiong and Nilsson (1997) described this litter redistribution as the product of three related processes: 1) litter erosion, 2) transport, and 3) deposition. This study was consistent with Cuffney (1988) observed increases in forest floor erosion to be related to higher and more frequent surface water events in riparian forests.

C_{FF}:C_{LF} values can be interpreted as the ability for headwater wetlands to retain and cycle carbon. These results show all sites lost forest floor carbon relative to the amount of litter fall(C_{FF}:C_{LF} < 1) with the most altered sites (highest CN) having the lowest ratio values (Table 3.5). While statistical analysis was conducted using annual averages and totals, bimonthly values appear to follow the same trends with P_e. Mean C_{FF} often decreases in months with increases in P_e, (Table 3.5). Similar fluctuations exist within litter fall data as well (Table 3.4). There was a seasonal trend within litter fall data which peaked in the fall and was the lowest in the spring. However, some increases in the amount of litter fall collected could be the result of severe rain events between sampling periods (Table 3.2). For instance, a large proportion of total P_e came during the July and September sampling periods, which coincided with decreases in C_{FF} sampled in July (export) and an increases in September (accumulation). These data do not lend themselves to separation of decomposition and hydrological export yet increases in both are likely. Rates of decomposition have been shown to increase with conversion of lands from forested cover to urban uses (McDonnell et al. 1997) and often attributed to increased disturbance, soil and ambient temperature, surface hydrology and nutrient influx into urban forested environments (Kaye et al. 2006). An increase in the non-native invasive shrub *Ligustrum sinense* was significantly related to decreasing forest cover across a larger subset of headwater wetland in Baldwin County, Alabama (see Chap 2). Recent research has described increasing rates of decomposition as a result of higher litter quality of *L. sinense* than that of native species in the piedmont physiographic region of Georgia, USA (Mitchell et al. 2011). Like decomposition, increased surface flows capable of significant carbon export were observed and related to changes in land use.

Wetland hydrographic data (Fig. 3.4) showed a greater number of large outlying surface water events as CN increased. These surface events may have displaced considerable forest floor organic matter which may be exported from the headwater system further downstream. Although there is very little previous information on how increased runoff changes headwater wetland litter dynamics, there is evidence to show ramifications for stream environments. Meyer et al. (2005) found decreases in stream ecosystem function (nutrient cycling and stable hydroperiod) were related to urbanization and its effect on reduced standing stock of organic matter. Similar responses associated with increased peak flows and total runoff with increased agricultural production have also been detected (Magner et al. 2004, Blann et al. 2009). Allan et al. (1997) found that organic matter inputs and related habitat quality in southeastern Michigan streams were best predicted by agricultural land use in the catchment. Based on these attributes of altered systems, C_{FF}:C_{LF} may be a useful measure for evaluating the potential influence of hydrologic export on headwater carbon cycling. Organic matter transport downstream is an important wetland function and provides the foundation for aquatic trophic webs (Aldridge et al. 2009), macro-invertebrate assemblages (Huryn et al. 2002), nutrient turnover (Rheinhardt et al. 1999) and is critical for stream morphology (Richards et al. 1999), however it is uncertain what effect increased export of coarse organic material would have on downstream habitats.

Hydrologic export supported by these data may help interpret the relationships with C_{SOIL} and wetland hydrology. Average C_{SOIL} for the entire 1-m core was not significantly correlated with any watershed hydrological variables; however C_{SURF} did display significant relationships to several wetland hydrology metrics ($WT_{<-50}$, $WT_{<-20}$, $WT_{>+20}$). This may be due to greater variability of soil saturation within the top few cm of soil surface as a response to landscape alterations. Decreases in $WT_{\pm 20}$ and increases in $WT_{<-20}$ and $WT_{<-50}$ describe wetlands with lower water levels and/or shorter durations of inundation which would allow for increased mineralization of soil organic matter (McLatchey and Reddy 1998). Richard-Baker index (RB) described flashy systems that may be responsible for greater amounts of hydrologic export due to high peak flows. Figure 3.4 demonstrates the association with decreasing forested land use and soil permeability to higher flows and more extreme events. As CN increases, lessened permeability, quartile ranges appear to tighten and the number of outlying data points dramatically increases, providing a greater occurrence of overland flow events was detected and higher levels of hydrologic export of organic material. Such flows limit the residence time of forest floor material and the opportunity for that material to accumulate as soil organic matter. The relationships between C_{SURF} and water table measures are supported by a significant increase in BD (higher soil mineral content, lower organic content) as WLÄ and $WT_{<-20}$ increased and a decrease in BD as $WT_{\pm 20}$ increased (Fig 3.8).

3.6 Conclusion

Ecosystem processes within headwater wetlands are driven by hydrology that normally originates as groundwater. Hydrological alterations within study wetlands were apparent both visually and quantifiably. Wetlands within deforested watersheds had more surface water influence, as evident by ditching and soil accretion around the bases of individual trees. Results showed a shift from ground water to surface water driven wetlands as converted watersheds were characterized by more extreme measurements of water level depth. These changes to wetland hydrology had considerable effects on the cycling and storage of forest floor carbon. In reference sites, water levels slowly increased as subsurface water entered and became stored within the soil profile. Contrary to these sites, in more altered landscapes water entered the systems as surface flow from specific locations (ditches and storm water outlets) with higher energy and greater potential for export of detrital material. Such export lessened the ability for headwater wetlands to sequester carbon and reduced the available leaf litter that may influence nutrient cycling and soil organic matter. While this study provides an initial understanding of headwater wetland function that is currently lacking, further investigation could provide information that may improve management of these vital systems. Long term hydrological monitoring that can account for climatic variability would increase understanding of the ability for these wetlands to store water for long term release and supply to downstream systems. Although our results suggest that hydrological export is critical, additional studies focused on the role of decomposition within headwater wetlands would allow for separation of mass loss between export and decomposition (here data are a combination of the two). As carbon storage becomes more of a global issue, it is important to address the capacity of headwater systems to act as sinks for organic carbon and to understand the important relationship between land use conversion and carbon storage within these critical environments.

3.7 References

Abelho, M. 2001. From litter fall to breakdown in streams: a review. The Scientific World. 1:656-80.

Allan J.D., D.L. Ericson, J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology. 37:149-61.

Aldrige K.T., J.D. Brookes, and G.G. Ganf. 2009. Rehabilitation of stream ecosystem functions through the reintroduction of coarse particulate organic matter. Restoration Ecology. 17(1):97-106.

Baker, L.A. 1992. Introduction to nonpoint source pollution in the United States and prospects for wetland use. Ecological Engineering. 1:1-26.

Baker III, T.T., B.G. Lockaby, W.H. Conner, C.E. Meier, J.A. Stanturf, M.K. Burke. 2001. Leaf litter decomposition and nutrient dynamics in four southern forested floodplain communities. Soil Science Society of America Journal. 65:1334-47.

Baldwin County Planning and Zoning Department. 2005. The Baldwin county conservation plan-final summary document. Bay Minette, Alabama.

Blann, K.L, J.L. Anderson, G.R. Sands, and B. Vondracek. 2009. Effects of Agricultural Drainage on Aquatic Ecosystems: A Review. Critical Reviews in Environmental Science and Technology 39:909-1001.

Brinson, M.M., A.E., Lugo and S. Brown. 1981. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics. 12:123-61.

Brinson, M.M. 1990. Riverine forests. Chapter 5. Ecosystems of the World: Forested Wetlands. Elsevier, Amsterdam.

Chadwick, M.A., D.R. Dooberfuhl, A.C. Benke, A.D. Huryn, K. Suberkropp, J.E. Thiele. 2006. Urbanization affects stream ecosystem function by altering hydrology, chemistry , and biotic richness. Ecological Applications 16(5):1796-1807.

DeLaney, T.A. 1995. Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. Journal of Soil and Water Conservation 50(6):620-28.

Ehrenfeld, J.G., H.B. Cutway, R. Hamilton IV, and E. Stander. 2003. Hydrologic description of forested wetland in Northeastern New Jersey, USAWetlands 23(4): 685-700.

ESRI. ArcGIS 9.3. Redlands, CA: Environmental Systems Research Institute (ESRI); 2009.

Faulkner, S. 2004. Urbanization impacts on the structure and function of forested wetlands. Urban Ecosystems. 7:89-106.

Gleason, R.A. and N.H. Euliss Jr. 1998. Sedimentation of prairie wetlands. Great Plains Research 8:97-112.

Grace III, J.M. 2005. Forest operations and water quality in the South. American Society for Agricultural Engineers. 48(2):871-80.

Hardison, E.C., M.A. O'Driscoll, J.P. DeLoatch, R.J. Howard, and M.M. Brinson. 2009. Urban land use, channel incision, and water table decline along coastal plain streams, North Carolina. Journal of the American Water Resources Association 45(4):1032-46.

Hawkins, R.H., T.J. ward, D.E. Woodward, J. A. Van Mullem. 2009. Curve number hydrology: state of the practice. American Society of Civil Engineers. Virginia USA.

Huryn, A.D., V.M. Butz, C.J. Arbuckle, and L. Tsomides. 2002. Catchment land-use, macroinvertebrate and detritus processing in headwater streams: Taxonomic richness versus function. Freshwater Biology 47:401-15

Jiang, R. 2001. Investigation of runoff curve number initial abstraction ratio. MS thesis, Watershed Management, University of Arizona, Tucson, AZ. 120pp.

Jones, J.A., F.J. Swanson, B.C. Wemple, and K.U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14(1):76-85.

Kaye, J.P., P.M. Groffman, N.B. Grimm, L.A. Baker, and R.V. Pouyat. 2006. A distinct urban chemistry? Trends in Ecology and Evolution. 21(4):192-99.

Kirsten, W.J. 1979. Automated methods for the simultaneous determination of carbon, hydrogen, nitrogen, and sulfur, and sulfur alone in organic and inorganic materials. Analytical Chemistry 51:1173-79.

Lecerf, A., G. Risnoveanu, C. Popescu, M.O. Gessner, and E. Chauvet. 2007. Decomposition of diverse litter mixture in streams. Ecology 88(1):219-27.

Lee, A.A, P.A. Bukaveckas. 2002. Surface water nutrient concentrations and litter decomposition rates in wetlands impacted by agriculture and mining activities. Aquatic Botany 74: 273-85.

Lundgren, M.R., C.J. Small, G.D. Dreyer. 2004. Influence of land use and sites characteristics on invasive plant abundance in the Quinebaug Highlands of Southern New England. Northeastern Naturalist 11(3):3313-32.

Magner, J.A., G.A. Payne, and L.J. Steffen. 2004. Drainage effects on stream nitrate-n and hydrology in south-central Minnesota (USA). Environmental Monitoring and Assessments 91:183-98.

McBride, E.H. and L.H. Burgess. 1964. Soil survey of Baldwin County, Alabama. U.S. Department of Agriculture. Soil Conservation Service. Soil Survey Report, 12: 110.

McDonnell, M.J., S.T.A. Pickett, P. Groffman, P. Bohlen, R.V. Pouyat, W. Zipperer, R.W. Parmelee, M.M. Carreiro, and K. Medley. 1997. Ecosystem processes along an urban to rural gradient. Urban Ecosystems 1: 21-36.

McLatchey, G.P. and K.R. Reddy. 1998. Regulation of organic matter decomposition and nutrient release in a wetland soil. Journal of Environmental Quality 27: 1268-74.

Microsoft Corporation. 2010. Microsoft Excel [computer software]. Redmond, Washington: Microsoft.

Minitab 16 Statistical Software 2010. [Computer software]. State College, PA: Minitab, Inc.

Mitchell, J.D., B.G. Lockaby, and E.F. Brantley. 2011. Influence of Chinese privet (*Ligustrum sinense*) on decomposition and nutrient availability in riparian forests. Invasive Plant Science and Management. 4(4):437-47.

Messina, M.G. and W.H. Conner. 1998. Southern Forested Wetlands: Ecology and Management. Lewis Publishers. Boca Raton.

Meyer, J.L., M.J. Paul, and W.K. Taubee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society. 24: 602-12.

Mishra, S.K. and V.P. Singh. 2004. Validity and extension of the SCS-CN method for computing infiltration and rainfall-excess rates. Hydrological Processes 18: 3323-45.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. 3rd ed., John Wiley and Sons, New York.

Nagy, R.C., B.G. Lockaby, L. Kalin, and C. Anderson. 2011. Effects of urbanization on stream hydrology and water quality: the Florida Gulf Coast. Hydrological Processes DOI: 10.1002/hyp.8336.

National Climate Data Center. Data retrieved May 20, 2012.

Noble, C.V., J.S. Wakeley, T.H. Robert and C. Henderson. 2007. Regional guidebook for applying for applying the hydrogeomorphic approach to assessing the functions of headwater slope wetlands on the Mississippi and Alabama coastal plains. U.S. Army Corps of Engineers.

Paul, M.J. and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333-65.

Poff, N.L., B.P. Bledsoe, C.O. Cuhaciyan. 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. Geomorphology 79:264-85.

R Development Core Team 2010. R: A language and environment for statistical computing, reference index version 2.12.12. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Rheinhardt, R.D., M.C. Rheinhardt, M.M. Brinson, and K.E. Faser, Jr. 1999. Application of reference data for assessing and restoring headwater ecosystems. Restoration Ecology 7(3):241-51.

Richards, C., L.B. Johnson, and G.E. Host. 1996. Lanscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53(1):295-311.

Robb, D.M. 1992. The role of wetland water quality standards in nonpoint source pollution control strategies. Ecological Engineering 1:143-48.

Schoonover, J.E., B.G. Lockaby, B.S. Helms. 2006. Impact of land cover on stream hydrology in the West Georgia piedmont, USA. Journal of Environmental Quality 35:2123-31.

Stander, E.K., and J.G. Ehrenfeld. 2009. Rapid assessment of urban wetlands: functional assessment model development and evaluation. Wetlands. 29(1):262-76.

State Climate Office of North Carolina. Data retrieved May 15, 2012. http://www.nc-climate.ncsu.edu/mpe_departure/.

Swan, C.M., M.A. Gluth, and C.L. Horne. 2009. Leaf litter species evenness influences non-additive breakdown in a headwater stream. Ecology. 90(6):1650-58.

USCB. 2010. 2010 Census Data. <http://2010.census.gov/2010census/data/>.

USDA. Natural Resource Conservation Service. 2007. Chapter 7: Hydrological Soil Groups. Part 630 Hydrology National Engineering Handbook.

Wallace, J.B., S.L. Eggert, J.L. Meyer, and J.R. Webster. 1999. Effects of resource limitation on detrital-based ecosystems. Ecological Monographs 69(4):409-42.

Walbridge, M.R. 1993. Functions and values of forested wetlands in the southern United States. Journal of Forestry. May: 15-19

Walsh, C.J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman, and R.P. Morgan II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society. 24(3):706-23.

Webster, J.R., and E.F. Benfield 1986. Vascular plant breakdown in freshwater ecosystems. Annual Review of Ecology and Systematics. 17:567-94.

Xiong, S. and C. Nilsson. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: a review. The Botanical Review 63(3):240-64.

Zedler, J.B. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecological Environments 1(2):65-72.

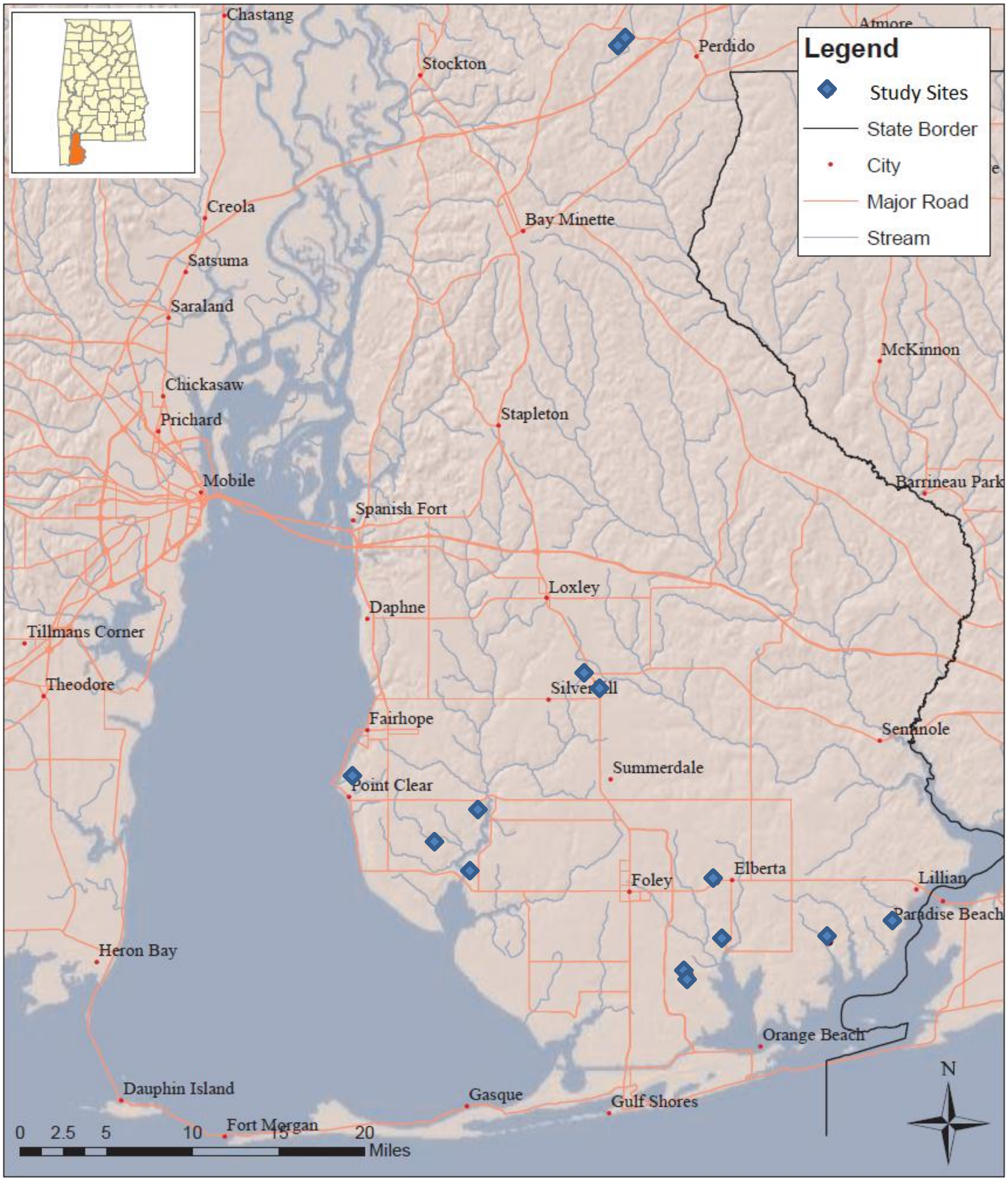

Figure 3.1: Map of Baldwin County, Alabama displaying the distribution of 15 headwater wetlands sampled February 2011 – March 2012.

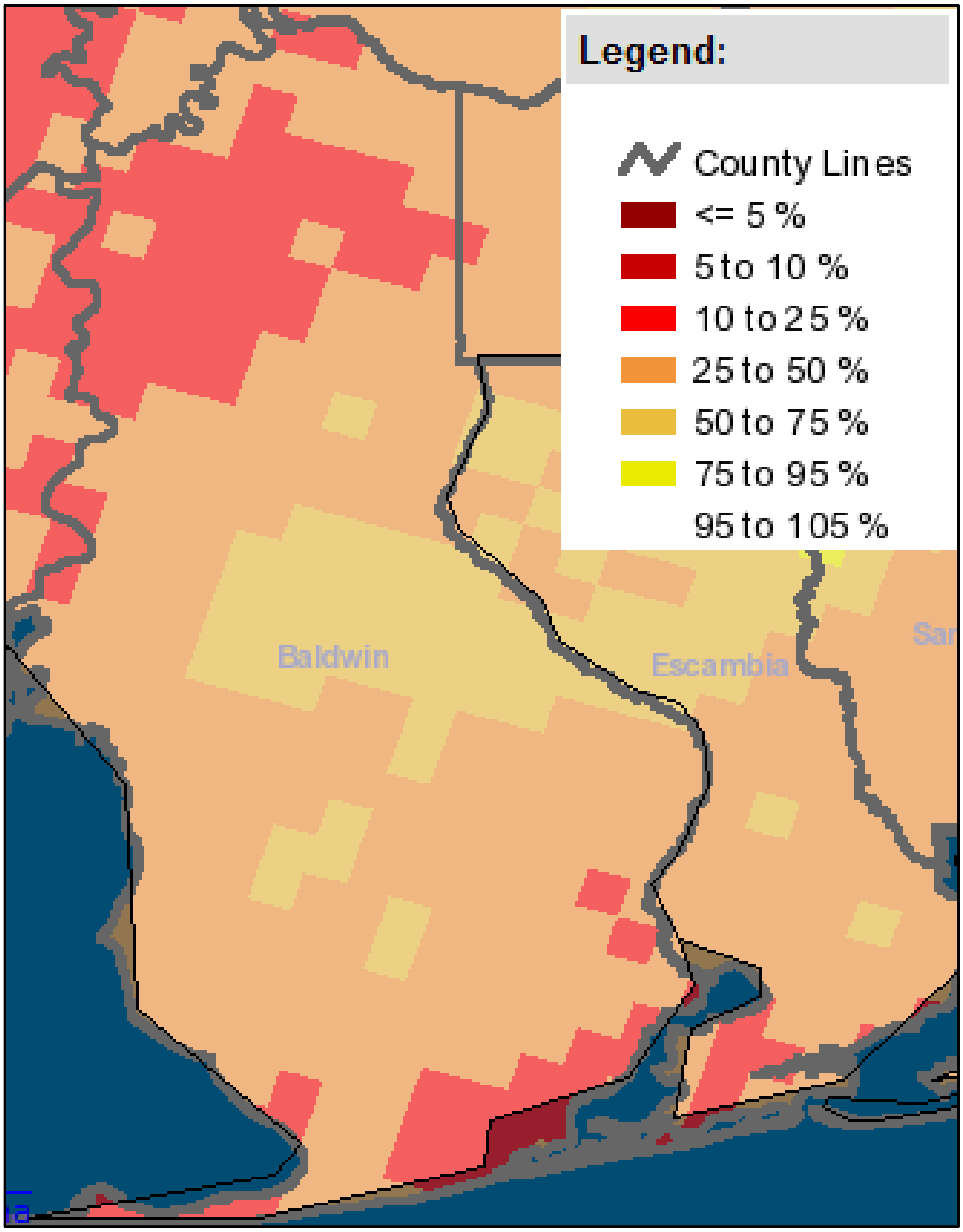

Figure 3.2: Map of Baldwin County, Alabama displaying the level of drought (May 2011-July 2011). Color gradation represents percent of normal rainfall. (SCONC)

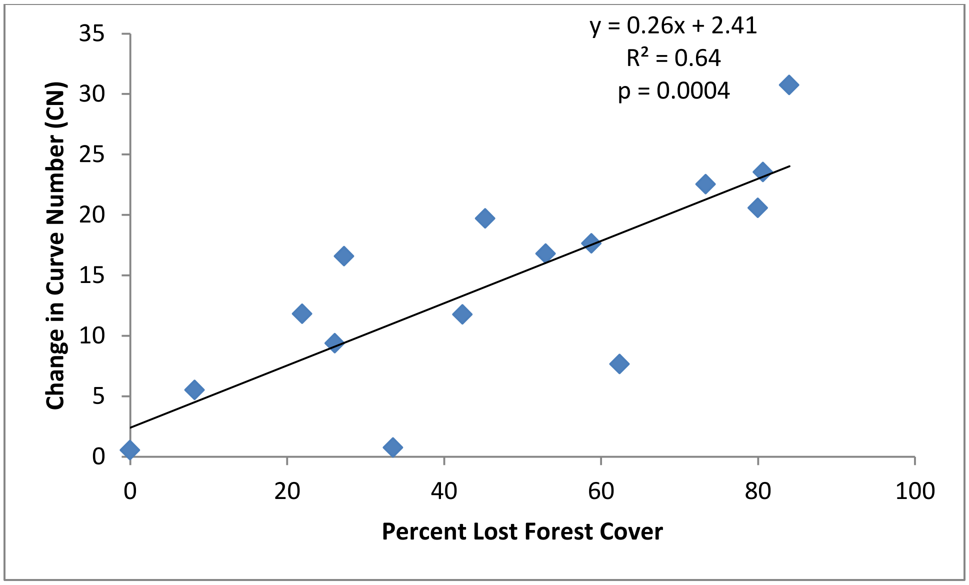

Figure 3.3: Influence of loss of forested land cover on maintaining watershed curve number (CN).

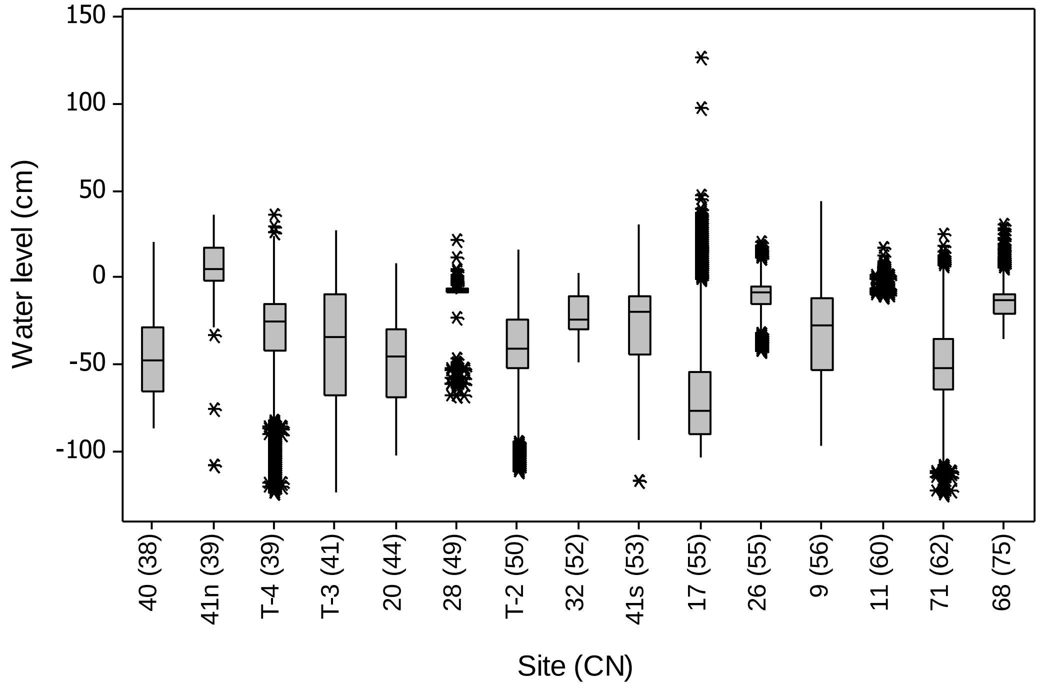

Figure 3.4: Boxplot of groundwater hydrology for 15 headwater wetlands arranged by increasing CN. Outlier measurements represented by asterisks (*).

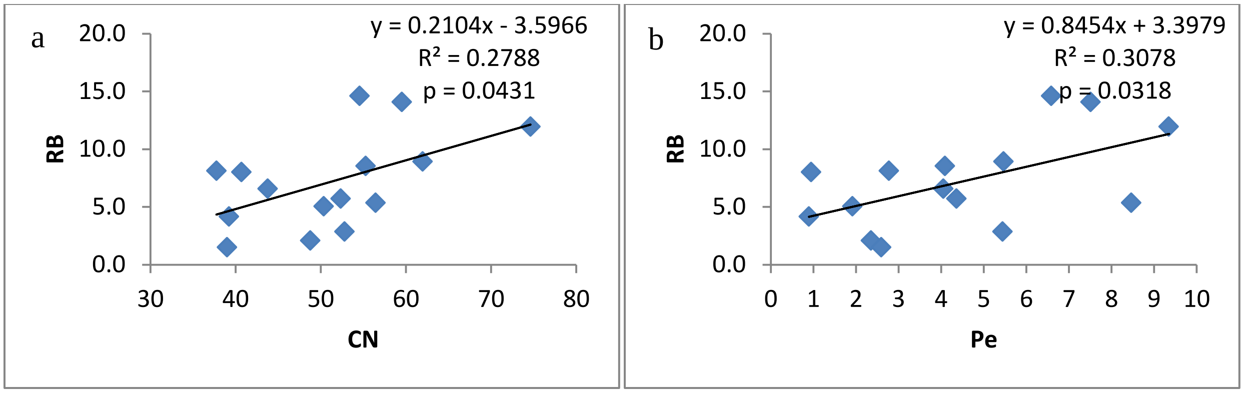

Figure 3.5: Linear regressions of Richard-Baker index (RB) to a) watershed runoff curve number (CN) b) and total runoff (P_e)

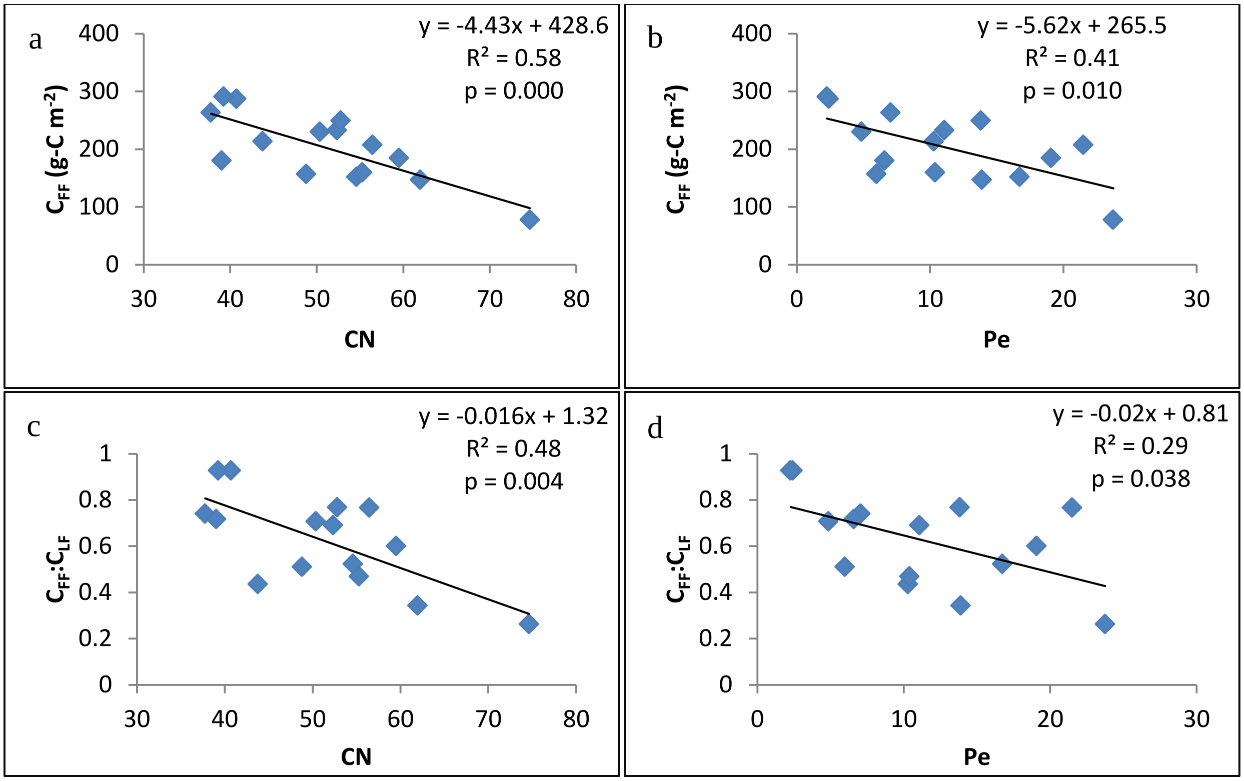

Figure 3.6: Linear regression of forest floor carbon (C_{FF}) and forest floor carbon: litter fall carbon ratio (C_{FF}:C_{LF}) against watershed runoff curve number (CN) and excess precipitation (P_e). a) C_{FF} regressed to CN, b) C_{FF} regressed to P_e, c) C_{FF}:C_{LF} regressed to CN, d) C_{FF}:C_{LF} regressed with P_e.

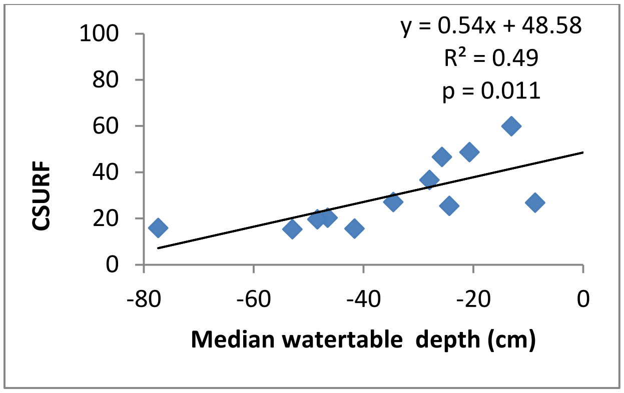

Figure 3.7: Linear regression total soil surface carbon (C_{SURF}) (0-10cm) and median water table depth.

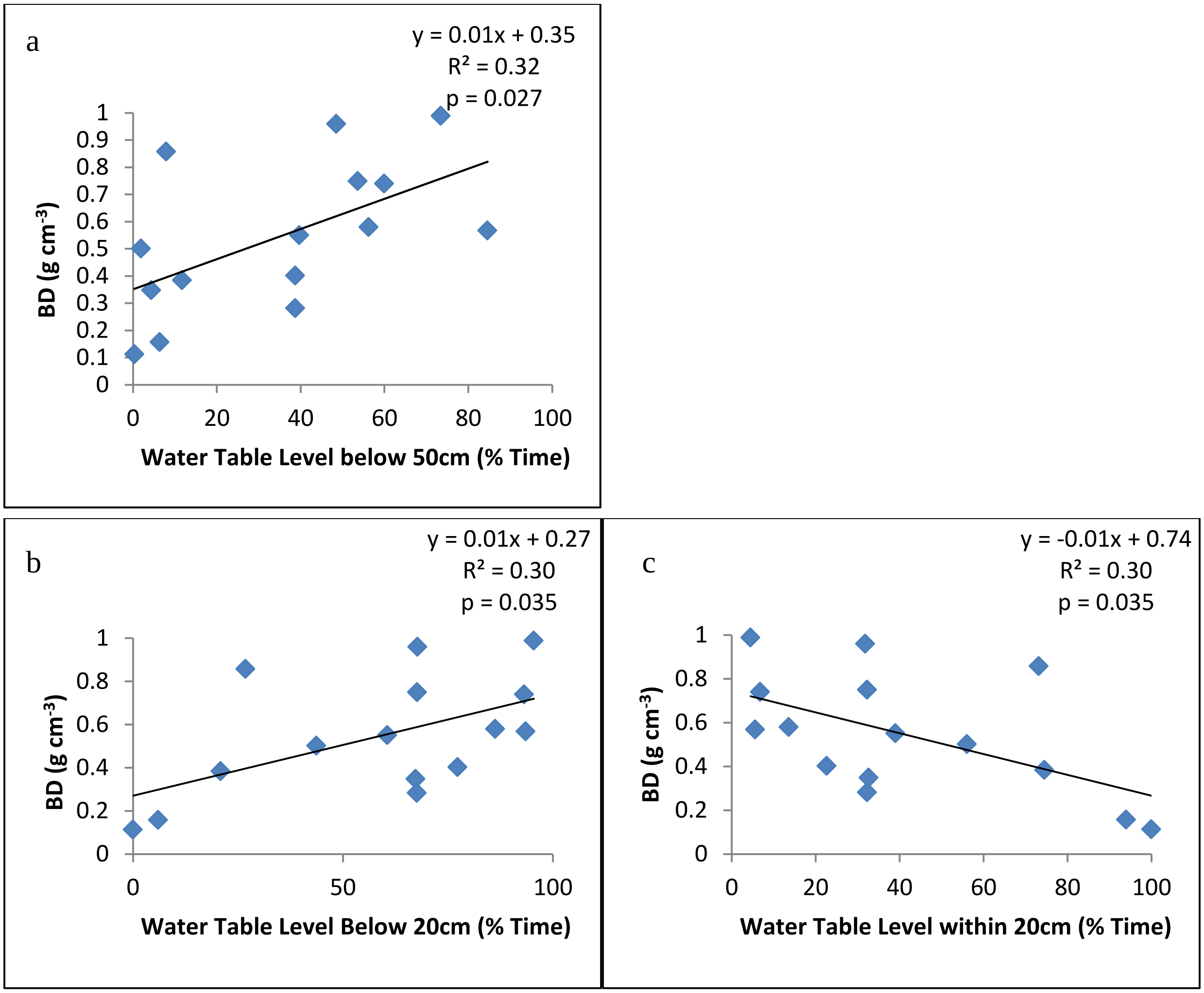

Figure 3.8: Linear regression of average bulk density (BD) of top meter of soil profile to a) Percent time water table was below 50cm, b) percent time water table was below 20cm, and c) percent time water table was within 20cm of ground surface.

Table 3.1: General characteristics of 15 headwater wetland study sites. Adjusted watershed area reflects hydrologic changes in landscape due to anthropogenic disturbances.

Wetland ID	Historic Watershed Area	Adjusted Watershed Area	Percent Change	Wetland Area	Percent Wetland:Watershed Area	Percent Land use within Watershed		
						ISA	Agriculture	Forest
9	83.2	83.2	0.0	4.4	5.26	4.4	52.0	19.3
11	66.9	55.0	17.8	5.4	9.84	1.8	43.4	26.7
17	81.5	78.4	3.8	0.7	0.91	1.0	4.2	54.7
20	28.5	28.5	0.0	0.5	1.68	0.7	6.6	73.9
26	87.9	87.9	0.0	1.9	2.15	0.8	18.0	72.7
28	57.8	62.9	8.7	0.8	1.29	6.2	0.0	47.0
32	147.0	127.8	13.1	0.5	0.40	1.6	27.3	41.2
40	109.9	97.9	10.9	1.4	1.45	1.2	37.4	37.6
41N	16.7	16.7	0.0	2.7	15.81	0.0	0.0	100.0
41S	214.6	194.9	9.1	3.2	1.63	0.7	8.2	78.1
68	141.6	151.1	6.7	9.4	6.24	15.1	20.2	16.0
71	36.8	39.5	7.4	2.7	6.81	24.0	7.7	20.0
T2	16.4	16.4	0.0	3.1	18.99	0.2	42.2	57.6
T3	95.2	81.3	14.6	4.3	5.34	0.1	0.5	91.8
T4	46.6	46.6	0.0	6.5	13.99	1.1	9.5	66.5

Table 3.2: Watershed runoff characteristics for 15 study watersheds. Historic and Current Curve number (CN), Total Precipitation (P), bimonthly runoff (P_e) and total runoff (Total P_e).

Wetland ID	Historic CN	Current CN	P (cm)	P_e (cm) Mar	May	Jul	Sep	Nov	Jan	Mar	Total P_e (cm)
9	32.9	56.4	149.2	2.49	0.14	4.61	7.73	0.81	2.78	2.94	21.5
11	37.0	59.5	144.7	0.94	0.25	2.76	10.30	1.34	1.60	1.88	19.1
17	34.9	54.6	138.0	3.21	0.03	7.85	4.33	0.29	0.05	0.97	16.7
20	34.4	43.8	138.0	2.09	0.00	5.20	2.50	0.08	0.01	0.43	10.3
26	38.7	55.3	131.6	3.34	0.03	4.10	0.86	0.13	0.06	1.88	10.4
28	32.0	48.8	113.5	1.39	0.06	0.72	2.79	0.31	0.14	0.59	6.0
32	34.7	52.3	123.3	0.89	0.06	0.88	6.87	0.49	0.67	1.22	11.1
40	30.1	37.8	126.8	0.25	0.00	0.27	5.65	0.23	0.17	0.48	7.1
41N	38.5	39.0	144.7	0.27	0.01	0.79	4.16	0.39	0.43	0.42	6.5
41S	41.0	52.8	144.7	0.65	0.12	1.92	7.90	0.94	1.08	1.23	13.8
68	43.9	74.6	129.0	1.78	0.23	9.04	10.01	0.49	0.27	1.93	23.8
71	41.4	62.0	129.0	1.08	0.13	5.21	6.14	0.26	0.13	0.94	13.9
T-2	38.6	50.4	107.9	0.41	0.02	0.54	0.85	0.12	0.24	2.83	5.0
T-3	35.2	40.7	107.9	0.18	0.00	0.24	0.38	0.02	0.06	1.53	2.4
T-4	38.5	39.2	107.9	0.16	0.00	0.22	0.36	0.02	0.05	1.46	2.3

Table 3.3: Hydrological variables for 15 study watersheds. Periods of drought resulting in water tables below water level recorder being omitted for analysis.

Wetland ID	Minimum (cm)	Median (cm)	Maximum (cm)	WLV (cm)	$WT_{<-50}$ (%)	$WT_{<-20}$ (%)	$WT_{\pm 20}$ (%)	$WT_{>+20}$ (%)	RB	WLΔ ($cm\ hr^{-1}$)
9	-97.5	-28.0	44.2	24.0	48.5	71.9	28.1	0.4	5.3	3.5
11	-11.3	-3.8	17.4	1.9	0.4	0.4	99.6	0.0	14.1	1.0
17	-103.9	-77.4	127.3	29.1	84.6	93.6	6.4	0.9	14.6	5.1
20	-103.0*	-46.5	7.6	24.1	73.4	97.8	2.2	0.0	6.6	3.4
26	-43.1*	-8.7	20.5	8.6	7.9	28.1	71.9	0.0	8.5	2.8
28	-69.0	-8.4	21.5	14.3	6.3	6.4	93.6	0.0	2.1	1.2
32	-49.8	-24.4	1.7	11.1	4.3	61.9	38.1	0.0	5.7	1.5
40	-87.5	-48.4	19.7	22.4	56.2	86.1	13.9	0.1	8.1	3.3
41N	-109.0*	5.0	35.4	12.4	11.7	21.3	78.7	4.6	1.5	1.7
41S	-118.0*	-20.7	30.6	22.5	39.7	64.8	35.2	0.4	2.8	2.6
68	-36.2	-13.0	30.0	8.3	1.9	44.3	55.7	0.2	11.9	2.5
71	-125.3*	-52.9	24.4	27.5	59.9	93.6	6.4	0.0	8.9	6.3
T-2	-112.5	-41.6	15.4	25.5	38.7	77.3	22.7	0.0	5.1	3.7
T-3	-125.0*	-34.5	27.0	37.7	53.6	73.3	26.7	0.0	8.0	6.6
T-4	-125.0*	-25.7	35.9	28.9	38.7	72.3	27.7	0.1	4.2	3.8

* denotes depth of water level recording device. Actual minimum value is unknown and less than reported value.

Table 3.4: Mean wetland litter fall production (C_{LF}) measured bimonthly (g-C m^{-2}).

Wetland ID	May*	Jul	Sep	Nov	Jan	Mar	Total
9	30.7	43.0	43.1	46.1	94.4	12.8	270.1
11	50.5	45.9	57.4	57.4	75.8	20.4	307.3
17	27.7	33.4	51.6	90.8	61.5	25.6	290.6
20	80.2	108.0	43.7	51.2	88.8	117.4	489.3
26	30.1	59.4	46.5	52.0	130.3	22.2	340.5
28	44.3	55.5	50.8	30.9	103.1	23.1	307.7
32	40.7	65.2	48.2	63.1	103.0	16.9	337.1
40	41.6	58.1	48.5	63.1	124.8	19.3	355.3
41N	23.4	49.2	50.1	64.7	52.5	11.2	251.2
41S	35.7	63.9	47.6	49.4	107.6	20.7	325.0
68	49.3	85.0	32.8	25.5	76.9	25.5	295.1
71	71.8	83.4	45.1	77.2	119.7	32.2	429.4
T-2	32.4	70.3	46.8	57.1	104.3	14.5	325.5
T-3	28.4	49.4	45.8	57.2	114.2	14.7	309.5
T-4	27.4	54.1	50.3	76.4	76.2	29.7	314.1

*May litter fall (g-C m^{-2} yr^{-1}) is adjusted based on timing of initial data collection – see text for details.

Table 3.5: Mean wetland standing stock of carbon in the forest floor (C_{FF}) measured bimonthly (g-C m^{-2}) and a ratio of average C_{FF} to total C_{LF} (C_{FF}:C_{LF}) indicator of forest floor stability.

Wetland ID	May	Jul	Sep	Nov	Jan	Mar	Annual Average	SD	C_{FF}:C_{LF}
9	162.5	165.3	352.7	197.1	219.4	145.1	207.0	76.2	0.77
11	204.3	123.4	143.0	196.4	241.2	198.8	184.5	43.4	0.60
17	150.4	16.1	104.0	208.8	204.4	226.6	151.7	80.4	0.52
20	249.7	147.6	176.2	171.8	227.5	305.6	213.1	59.1	0.44
26	134.3	36.8	128.9	200.5	225.0	231.9	159.5	74.6	0.47
28	167.0	132.7	146.7	144.1	188.5	161.3	156.7	19.9	0.51
32	298.2	138.1	211.9	257.1	270.4	220.4	232.7	56.3	0.69
40	282.0	243.0	279.6	257.1	289.8	226.8	263.0	24.9	0.74
41N	199.2	89.4	79.3	206.6	295.4	209.7	179.9	82.0	0.72
41S	224.8	254.9	177.1	248.2	281.4	310.0	249.4	46.0	0.77
68	194.5	6.8	38.5	98.7	64.1	62.0	77.4	64.9	0.26
71	216.6	95.2	113.7	151.7	156.7	146.2	146.7	41.8	0.34
T-2	304.4	164.4	127.0	258.5	245.5	279.6	229.9	69.2	0.71
T-3	336.3	230.8	289.1	294.7	328.1	242.7	287.0	43.1	0.93
T-4	288.1	208.9	370.9	336.5	319.9	220.9	290.9	64.7	0.93

Table 3.6: Soil carbon within first meter of soil profile for 15 head water wetlands (Total content g-C m^{-3}). Average carbon concentration (%TC) and average bulk density (BD g cm^{-3}) for entire profile reported.

Wetland ID	0-10cm	30-40cm	60-70cm	90-100cm	AVG %TC	AVG BD	Total Content
9	36.6	28.3	32.6	14.9	3.2	0.96	28.1
17	15.8	59.7	94.5	64.8	10.0	0.57	58.7
20	20.2	20.2	15.1	6.9	1.7	0.99	15.6
26	26.7	16.2	3.9	1.9	2.3	0.86	12.2
32	25.3	44.7	59.7	23.4	12.3	0.35	38.3
40	19.5	14.1	51.9	26.7	4.9	0.58	28.0
41s	48.7	28.0	39.1	39.9	7.7	0.55	38.9
68	59.8	72.0	28.1	16.4	10.9	0.50	44.0
71	15.2	30.5	22.0	23.9	3.3	0.74	22.9
T-2	15.6	19.6	19.1	12.2	4.4	0.40	16.6
T-3	27.0	32.2	29.6	29.0	4.6	0.75	29.5
T-4	46.6	37.1	20.2	26.4	12.0	0.28	32.5
11*	34.0		58.0		31.1	0.11	46.0
28*	33.7		55.8		16.9	0.16	44.7
41n*	28.7		34.8		5.8	0.38	31.7

* Due to soil saturation and unconsolidated soil material wetland ID 11, 28, and 41N samples were partitioned into 40 cm increments only (0-40 cm and 60-100 cm).

Chapter IV

SUMMARY AND CONCLUTIONS

Headwater-slope wetlands are critical ecosystems in the Southeastern Coastal Plain Physiographic region of Alabama, USA. These systems play important roles in maintaining both water quality and quantity to downstream environments. Headwater wetlands are located at the upper reaches of coastal creeks and normally driven by groundwater flow which originates in the surrounding upland landscape. Based on their location, anthropogenic alterations to the landscape can have significant impacts on the functions and services these important wetlands provide. This research focused on the effects of land use/land cover (LULC) change has on headwater wetlands. The goal of this study was to describe the impacts of LULC on headwater wetlands and detect possible trends related to the effect of degraded wetland function as a result of human induced landscape alterations. Specifically this project focused on: 1) examining the effect LULC has on wetland forest structure, composition, soils and hydrology, as well as basic ecosystem functions; and 2) studying alterations in hydrologic regime related to LULC and corresponding shifts in both carbon cycling and storage. This project consisted of two-part study examining wetlands over a range of surrounding LULC conditions typical of coastal Alabama. Some of the most relevant findings related to headwaters wetlands and LULC are provided below:

Forest Structure and Composition

- Canopy tree diameter, density, and cover were all related to decreases in forested land cover. These data suggest that as these systems are isolated from other forested environments and more dramatically impacted recruitment of tree species may decline.

This caused canopy individuals not to be replaced post-mortem and promote an older canopy (larger, fewer individuals).

- Ground cover of native canopy and herbaceous species decreased with increasing land use conversion, while shrub cover increased as a result of invasion of *Ligustrum sinense* (Chinese privet). This likely reflected the increased exposure to exotic species at wetlands surrounding by urban and agricultural lands.
- The presence of non-native plant species significantly increased in all three forest strata as natural forest cover declined within the watershed. This may be significant because as the systems become more heavily impacted, management of these non-native species will most likely become increasingly difficult.

Soils and Hydrology

- Soil chroma decrease with increasing levels of forested cover. This relationship may be due to increasing soil saturation as a result of increased forest cover influencing the accumulation of soil organic matter and driving soil chroma and value lower.
- The prevalence index of hydrophytic vegetation was lower within heavily forested watersheds. This also describes increased soil saturation as a result of more forest cover within immediate watersheds.

Wetland Function

- The functional capacity index (FCI) for providing for characteristic wildlife habitat was the only ecosystem function to be related to land use based on the hydrogeomorphic (HGM) approach. This function increased as watersheds became more forested as a result of a particular HGM variable that measured the extent of contiguous forested land cover immediately adjacent to the wetland boundary.

Hydrologic Regime

- Richard-Baker index (an index of hydrologic flashiness) was the only hydrological metric to be related to LULC. This lack of relationship with basic measures of hydrology (base flow, peak flow, mean flow, etc.) is most likely the result of omitting drought periods which skewed data to make wetlands appear wetter. Further monitoring of ground water levels within these headwater systems may shed more light on the influence of LULC on groundwater hydrology.

Carbon Cycle and Carbon Storage

- Mean forest floor carbon significantly decreased with increasing levels of forest conversion and runoff. The relationship most likely describes the effect of increased runoff on the hydrologic export of organic material, which is essential for ecosystem functions
- Wetlands within altered environments appeared to release forest floor carbon at a faster rate than wetlands within primarily forested watersheds. This is most likely the result of increased surface runoff due to increased impervious surfaces and storm water systems that discharge water directly into the wetland.
- Soil carbon for the top meter of the soil profile displayed no significant relationships with hydrological metrics. However, surface soil carbon increased with longer periods of soil saturation. This is may be due to lower rates of decomposition under saturated conditions.
- Bulk density decreased with longer periods of soil saturation, describing soils with greater organic matter when the water table is at or near the soil surface for longer durations.

This research provides a foundation of understanding on the effect of LULC on headwater wetlands that has not been previously described. Environmental variables that were not examined such as, topography, measures of absolute decomposition, and above ground net primary productivity may elucidate better understanding regarding the organic carbon cycle. Hydrological monitoring needs to be examined over a longer time period to account for annual variability and should be studied at greater depths to limit the loss of data resulting from drought conditions. This study was not established to develop management strategies but the knowledge gained should be utilized in a way that will limit future impacts on these critical ecosystems. In conclusion, this work identified important trends related to the impacts of LULC on forest structure and composition, wetland function, hydrologic regime, and organic carbon cycle and provides a previously unavailable understanding of the consequences of land alterations on headwater slope wetlands.

Appendix A

The water storage function is designed to assess the ability of a wetland to store and release stable water supplies throughout the year. The storage and stable release of water is important in maintaining baseflows. The water storage functional index is calculated with the use of five variables which include: hydrological alterations (*Vhydroalt*), canopy tree density (*Vctden*), canopy tree diameter (*Vctd*), change in catchment size (*Vcatch*), and upland land use (*Vupuse*). The organic carbon cycling function is utilized to describe the ability of the wetland to perform biogeochemical processes. This index is calculated with five variables: hydrological alterations (*Vhydroalt*), canopy tree density (*Vctden*), canopy tree diameter (*Vctd*), surface soil organic matter content (*Vssom*), and soil detritus (*Vdetritus*). The maintenance of characteristic plant communities' function utilizes variables including: canopy tree density (*Vctden*), canopy tree diameter (*Vctd*) and vegetation composition (*Vcomp*). This function is used to describe the structure and function of the plant community which is critical in supporting habitat for native wildlife and supplying organic material. Finally, the characteristic wildlife habitat function measures six variables which include: hydrological alterations (*Vhydroalt*), upland land use (*Vupuse*), change in catchment size (*Vcatch*), canopy tree density (*Vctden*), canopy tree diameter (*Vctd*), and habitat connections (*Vconnect*). Since all study sites have a dominant overstory, canopy tree diameter and canopy tree density are used in the functional indexes. If canopy cover was less than twenty percent, a variable for sapling/shrub cover (*Vssc*) would replace the two canopy variables (*Vctd* and *Vctden*). If both canopy cover and sapling/shrub cover (*Vssc*) were less than twenty percent a variable for ground vegetation cover (*Vgvc*) would be utilized. See table 1 for further explanation of measurement techniques used to calculate variable scores. Functional Capacity indexes are calculated as follows:

Water Storage

$$FCI = \left\{ Vhydroalt * \left[\frac{\frac{(Vcatch + Vupuse)}{2} + \frac{(Vctd + Vctden)}{2}}{2} \right] \right\}^{\frac{1}{2}}$$

Cycle Organic Carbon

$$FCI = \left\{ Vhydroalt * \left[\frac{\frac{(Vdetritus + Vssom)}{2} + \frac{(Vctd + Vctden)}{2}}{2} \right] \right\}^{\frac{1}{2}}$$

Maintain Characteristic Plant Community

$$FCI = \left[\frac{\frac{(Vctd + Vctden)}{2} + Vcomp}{2} \right]$$

Provide Characteristic Wildlife Habitat

$$FCI = \left[\left\{ Vhydroalt * \left[\frac{(Vcatch + Vupuse)}{2} \right] \right\}^{\frac{1}{2}} * \left\{ \frac{\frac{(Vctd + Vctden)}{2} + Vconnect}{2} \right\} \right]^{\frac{1}{2}}$$

Table A.1: List of HGM variables and measurement techniques.

Variable	Measurement technique	Subindex Score calculation
Canopy tree density (*Vctden*)	Number of trees per plot X100 = trees/hectare.	Use provided graph to assign the subindex score associated with a particular density value
Canopy tree diameter (*Vctd*)	Diameter at breast height (DBH).	Use provided graph to assign the subindex score associated with a particular diameter value
Sapling/shrub cover (*Vssc*)	Visual estimate of percent coverage by sapling/shrub layer. All woody vegetation >1.5m tall and <4" DBH.	Use provided graph to assign the subindex score associated with a particular cover value
Ground vegetation cover (*Vgvc*)	Visual estimate of percent coverage of all herbaceous cover and woody vegetation <1.5m tall.	Use provided graph to assign the subindex score associated with a particular cover value
Detritus cover (*Vdetritus*)	Visual estimate of percent coverage or detritus material	Use provided graph to assign the subindex score associated with a particular cover value
Vegetation community (*Vcomp*)	Rank of all vegetative species within plot. Apply 50/20 rule to assess dominate species.	Group and count species based on provided table. Groups 1 and 2 are for native species. Group 3 is reserved for non-native species and group 1 contains only reference species. Calculate a quality index and adjust the quality index for species richness with provided equations. Calculate the square root of the adjusted quality index to generate the subindex score.
Surface soil organic matter (*Vssom*)	Soil value measured with Munsell Soil Chart within the first six inches of soil.	Refer to provided table for subindex scores based on soil color value.
Hydrologic alterations (*Vhydroalt*)	Measurement of height of drift lines, moss collars, buttress etc. as an assessment of excessive surface water	Use provided graph to assign the subindex score associated with a particular depth of hydrologic alteration value
Habitat connections (*Vconnect*)	Utilize GIS to assigned buffers at 10m, 30m, and 150m. Measure percentage of wetland perimeter that is connected to suitable wildlife habitat extending to all 3 buffers.	Use provided graph to assign the subindex score associated with a particular density value
Upland land-use (*Vupuse*)	Define land-use and soil types within the associated watershed boundary. Utilize NRCS hydrologic soil group data and runoff curve numbers to asses surface runoff potential.	Use provided graph to assign the subindex score associated with a particular weighted runoff score value
Change in catchment size (*Vcatch*)	Utilize GIS to assess changes in watershed boundary based on human activities (road construction, ditches, etc.) to quantify percent change in catchment area.	Use provided graph to assign the subindex score associated with a particular percent change value

Appendix A.2: List of HGM variable scores for 30 headwater slope wetlands

	Variable Scores										
ID	Ctdia	Ctden	GVCover	HydAlt	SSCover	Detcover	SSOrganic	VegComp	Conn	Upuse	Catch
4	0.55	0.10	0.20	1.00	0.55	0.80	0.80	0.87	0.18	0.55	0.97
9	0.50	0.70	1.00	1.00	0.80	0.85	0.80	0.76	0.66	0.71	1.00
11	0.30	0.10	0.40	0.80	0.60	1.00	1.00	0.64	0.67	0.65	0.82
14	0.75	0.25	0.85	0.95	0.40	0.95	1.00	0.82	0.29	0.40	0.94
17	0.75	0.35	0.35	0.75	0.25	0.75	0.80	0.71	0.59	0.74	0.96
20	0.55	1.00	0.70	1.00	0.85	0.90	0.80	0.93	1.00	0.99	1.00
23	0.55	0.10	0.02	1.00	0.50	0.25	0.80	0.64	0.28	0.55	0.97
25	0.35	0.10	0.60	1.00	0.25	0.45	0.80	0.50	0.27	0.42	0.96
26	0.50	0.50	0.10	1.00	0.95	0.80	0.80	0.64	0.66	0.89	1.00
27	0.70	0.60	0.25	1.00	0.80	0.40	0.80	0.66	0.33	0.57	0.72
28	0.55	0.10	1.00	1.00	0.75	0.85	0.80	0.50	0.66	1.00	0.91
32	0.25	0.10	0.70	1.00	0.30	0.50	0.80	0.64	0.32	0.40	0.87
33	0.80	1.00	0.30	1.00	0.85	0.35	0.80	0.82	0.33	0.70	0.10
36	0.50	0.15	0.85	0.85	0.55	0.85	0.80	0.71	0.53	0.89	1.00
37	0.50	0.10	0.85	1.00	0.40	0.95	0.80	0.53	0.66	0.62	0.94
39	0.50	0.10	1.00	1.00	0.40	0.95	0.80	0.47	0.17	0.90	1.00
40	0.70	0.10	0.60	1.00	0.90	0.55	0.80	0.76	0.56	1.00	0.89
43	0.45	0.30	0.20	1.00	0.25	1.00	0.80	0.71	0.00	0.74	1.00
52	0.80	0.50	0.40	1.00	0.95	0.90	0.80	0.53	0.66	0.57	0.90
53	0.85	1.00	0.15	0.80	1.00	0.90	0.80	0.71	0.66	0.79	0.61
56	0.60	0.10	0.55	1.00	0.55	0.55	0.55	0.50	0.66	1.00	0.37
68	1.00	0.85	0.00	0.98	0.80	0.40	0.80	0.50	0.06	0.40	0.93
71	0.80	0.90	0.20	1.00	0.85	0.90	0.80	0.32	0.23	0.61	0.93
85	0.10	0.10	0.10	1.00	0.25	1.00	0.80	0.71	1.00	0.75	1.00
96	0.70	0.55	0.45	1.00	0.95	1.00	0.80	0.93	0.37	0.61	0.96
97	0.50	0.65	0.15	0.90	1.00	0.80	0.60	0.58	0.16	0.46	0.68
41N	0.35	0.40	1.00	1.00	0.75	0.95	1.00	0.86	0.66	1.00	1.00
41S	0.55	0.75	1.00	1.00	0.65	0.90	1.00	0.91	0.66	0.83	0.91
T-2	0.55	0.10	0.93	1.00	0.40	0.83	1.00	0.96	1.00	0.84	1.00
T-3	0.95	1.00	0.45	1.00	0.00	1.00	1.00	0.96	1.00	1.00	0.85
T-4	0.80	0.85	0.80	0.95	0.35	1.00	1.00	0.96	1.00	1.00	1.00

Appendix A.3: List of HGM Functional Capacity Index (FCI) scores for 30 headwater slope wetlands

	Functional Capacity Indexes			
ID	Carbon Cycling	Plant Community	Water Storage	Wildlife Habitat
4	0.75	0.60	0.74	0.47
9	0.84	0.68	0.86	0.77
11	0.69	0.42	0.62	0.58
14	0.84	0.66	0.82	0.61
17	0.70	0.63	0.73	0.68
20	0.90	0.85	0.94	0.94
23	0.65	0.48	0.74	0.51
25	0.65	0.36	0.67	0.45
26	0.81	0.57	0.73	0.66
27	0.79	0.65	0.85	0.66
28	0.76	0.41	0.70	0.63
32	0.64	0.41	0.71	0.48
33	0.86	0.86	0.87	0.69
36	0.76	0.52	0.80	0.65
37	0.77	0.41	0.73	0.65
39	0.77	0.47	0.79	0.48
40	0.73	0.53	0.83	0.69
43	0.80	0.54	0.78	0.42
52	0.87	0.59	0.84	0.76
53	0.84	0.82	0.82	0.79
56	0.72	0.43	0.74	0.66
68	0.86	0.71	0.88	0.63
71	0.92	0.59	0.90	0.69
85	0.71	0.40	0.70	0.72
96	0.87	0.78	0.84	0.66
97	0.76	0.58	0.72	0.51
41N	0.82	0.62	0.83	0.72
41S	0.89	0.78	0.87	0.78
T-2	0.79	0.64	0.79	0.80
T-3	0.99	0.97	0.99	0.99
T-4	0.93	0.89	0.91	0.93

Appendix A.4: Data required for calculation of HGM variables scores for 30 headwater slope wetland

ID	DBH	Density	Basal Area	Canopy Cover (exotic)	Shrub Cover (exotic)	Herb Cover (exotic)	Detritus Cover	Flooding Depth	Soil Value	Soil Chroma	PI Canopy	PI Shrub	PI Herb
4	19.2	1350	39.1	83.8 (2.5)	45.0 (43.8)	15.0 (11.8)	80.0	0.0	3.0	2.0	1.5	2.9	2.9
9	19.4	625	18.4	52.5 (2.5)	55.0 (45.0)	71.3 (12.5)	82.5	0.0	2.7	1.4	2.7	2.7	1.9
11	20.0	1475	46.3	85.0 (0.0)	16.3 (0.0)	6.3 (0.3)	100.0	27.0	2.0	2.0	1.5	1.9	1.9
14	24.2	900	41.5	46.3 (0.0)	32.5 (26.8)	62.5 (2.5)	92.5	18.0	2.5	2.3	1.5	2.8	1.5
17	24.2	825	37.9	83.8 (0.0)	32.5 (28.3)	26.3 (9.3)	73.8	0.0	3.0	1.8	1.8	2.9	0.8
20	20.9	375	12.8	57.5 (0.0)	58.8 (46.3)	52.5 (7.5)	88.8	0.0	2.6	1.4	2.7	2.9	1.6
23	20.7	1550	51.9	85.0 (0.0)	38.8 (14.0)	2.0 (0.5)	25.0	0.0	3.0	2.3	1.3	2.9	2.3
25	16.0	1125	22.6	81.3 (65.0)	18.8 (13.8)	41.3 (34.0)	42.5	0.0	3.0	2.4	2.8	2.9	2.7
26	19.4	750	22.1	52.5 (0.0)	66.3 (3.8)	9.5 (0.0)	80.0	0.0	2.8	1.3	2.5	2.3	1.7
27	22.8	700	28.5	85.0 (21.3)	57.5 (15.5)	16.3 (3.3)	38.8	0.0	3.0	2.2	1.9	2.9	1.7
28	21.9	1025	38.6	71.3 (0.0)	55.0 (1.3)	81.2 (0.3)	83.8	0.0	3.0	2.5	1.9	2.2	1.0
32	13.7	1100	16.3	66.3 (0.0)	22.5 (1.3)	50.0 (0.3)	50.0	0.0	3.0	1.6	2.3	2.7	1.5
36	18.2	975	25.4	71.3 (0.0)	40.0 (19.3)	60.0 (2.8)	82.5	0.0	2.6	1.4	1.7	2.9	1.4
37	18.0	1175	30.0	75.0 (15.0)	43.8 (37.5)	60.0 (1.6)	91.3	0.0	3.1	1.1	2.5	2.9	1.3
39	17.1	1000	22.9	82.5 (0.0)	30.5 (0.0)	77.5 (0.0)	92.5	0.0	2.9	2.3	--	--	--
40	23.1	1050	44.1	71.3 (0.0)	65.0 (12.3)	41.3 (5.0)	52.5	24.0	3.0	1.5	2.3	2.5	1.7
43	17.4	875	20.9	88.8 (0.0)	12.5 (0.0)	16.3 (0.0)	100.0	0.0	2.9	2.0	2.5	2.7	2.4
52	26.8	725	41.0	40.0 (1.3)	66.3 (62.5)	27.5 (0.8)	85.0	0.0	2.9	2.3	2.1	3.0	2.8
53	24.1	425	19.3	40.0 (0.0)	73.8 (66.3)	11.3 (2.8)	88.8	0.0	3.4	2.3	2.0	3.0	1.7
56	22.0	1050	39.8	76.3 (0.0)	40.0 (0.0)	36.3 (1.3)	37.5	30.0	2.5	1.2	2.1	2.2	1.5
68	50.1	200	39.4	31.3 (0.0)	55.0 (55.0)	2.5 (0.0)	38.8	0.0	2.9	1.3	2.0	3.0	4.0
71	25.7	225	11.7	51.3 (37.5)	60.0 (47.5)	15.0 (5.3)	87.5	0.0	2.9	1.6	2.8	2.8	2.2
85	18.6	1225	33.3	78.8 (0.0)	7.5 (0.0)	10.0 (0.0)	100.0	0.0	2.9	1.7	2.0	2.3	2.0
96	24.0	725	32.9	36.3 (0.0)	67.5 (67.5)	33.8 (0.5)	100.0	0.0	2.8	1.8	2.0	3.0	1.3
97	18.8	650	18.1	48.8 (3.8)	70.0 (47.5)	11.3 (5.5)	82.5	0.0	4.0	2.0	2.3	2.6	2.3
41N	17.0	750	17.0	43.8 (1.3)	57.3 (1.3)	77.5 (1.3)	91.5	0.0	2.4	1.2	1.9	2.0	1.3
41S	24.6	1100	52.2	58.8 (0.0)	42.5 (0.0)	23.8 (0.0)	100.0	0.0	2.2	1.0	1.9	2.1	2.0
T-2	21.8	1025	38.1	87.5 (0.0)	30.0 (7.5)	62.5 (5.5)	80.0	0.0	2.2	1.4	2.1	2.5	1.3
T-3	28.3	425	26.7	60.0 (0.0)	7.8 (0.0)	33.8 (0.0)	100.0	15.0	2.3	1.3	2.3	2.5	2.9
T-4	25.3	525	26.3	63.78 (0.0)	27.3 (0.0)	55.0 (0.0)	100.0	25.0	2.2	1.1	2.2	2.2	2.0