ABSTRACT

Title of Dissertation: A SPATIAL-TEMPORAL ANALYSIS OF WETLAND LOSS AND SECTION 404 PERMITTING ON THE DELMARVA PENINSULA FROM 1980 TO 2010

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Geospatial approaches for wetland change analyses have emphasized the evaluation of landscape change on a local level, but have often neglected to examine and integrate regional trends and patterns of land use and land cover change as well as the impacts of wetland management policies. This study attempts to bridge the gaps by integrating a geospatial assessment of land cover change and a geostatistical analysis of the physical and anthropogenic drivers of wetland change. The aim is to demonstrate how urban development, conservation, and climate change policy decisions influenced wetland change trends and patterns on the Delmarva Peninsula from 1980 to 2010.

Historical data on the nine counties on the Delmarva Peninsula illustrated the dynamism of population growth, sprawl, and different wetland management strategies. Data sets from the National Oceanic and Atmospheric Administration, the Chesapeake Bay Program, the U.S. Army Corps of Engineers, the U.S. Fish and Wildlife Service, and the U.S. Census Bureau, and other sources were gathered and assessed.

A land cover database was developed and analyzed using geospatial techniques, such as cross tabulation matrices and hot spot density analyses, in order to quantify and locate land cover change between 1984 and 2010. The results highlighted that anthropogenic drivers such as urbanization and agriculture were associated with the loss of wetlands in coastal areas as well as in upland, forested, suburban areas that were at low risk to flooding, but required deforestation in order to expand residential and commercial development. The greatest quantity and percentage of loss occurred between 1992 and 2001, and it was likely the result of increases in tourism and suburban sprawl (*e.g.*, the Housing Boom and roadway expansion). The majority of wetland loss tapered off in 2000, except on coastal areas suffering from sea level rise and shoreline erosion. The results also reinforced the need to address the negative impacts from certain activities related to agriculture and silviculture, which are exempt from Section 404 of the Clean Water Act, have on wetlands. Physical drivers and processes like inundation from sea level rise and soil erosion from surface runoff force communities to simultaneously adapt to multiple drivers of wetland loss and alteration. This study supports the hypothesis that an increase in development and wetland permitting indicates an increased a risk of wetland loss. In the end, the study demonstrates that geostatistical modelling techniques can be used to predict wetland loss, and that model performance and accuracy can be improved by reducing the multicollinearity of independent variables. Planners and policymakers can use these models to better understand the wetland locations that are at greatest risk to loss, as well as the drivers and landscape conditions that have the greatest influence on the probability of wetland loss.

A SPATIAL-TEMPORAL ANALYSIS OF WETLAND LOSS AND
SECTION 404 PERMITTING ON THE DELMARVA PENINSULA:
30 YEARS OF IMPACT FROM PHYSICAL AND ANTHROPOGENIC DRIVERS

By

Quentin Stubbs

Dissertation submitted to the Faculty of the Graduate School of the
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1 CHAPTER 1: INTRODUCTION

1.1 Background

1.1.1 Wetlands on the Delmarva Peninsula

Wetlands across the United States are vulnerable to disturbance, degradation, and loss (Tiner, 1999). According the U.S. Fish and Wildlife Service study of historical wetlands between the 1780's and the 1980's, the conterminous U.S. experienced a 53% loss of wetlands and each state that intersected the Chesapeake Bay Watershed experienced an average of loss of 50% wetland coverage (Dahl, 1990; NOAA, 2013). Today, wetlands represent approximately 28% of the Chesapeake Bay Watershed (NOAA, 2012; Tiner, 1987). Tidal and non-tidal wetlands fill a crucial role. According to the U.S. Department of Interior (DOI), Delmarva's inland wetlands intercept, and filter sediments and nutrient loads, detain and store flood water, serve as groundwater discharge areas, and provide food and habitat for endangered species like the Delmarva fox squirrel (Chase *et al*., 2003; Hartmann & Goldstein 1994; Tablante *et al*., 2002; Wilson *et al*., 2007). Over the past 200 years, the Peninsula has lost over 2.1 million acres of wetlands (open water and inland) due to dredging, channelization, draining and filling, and ponding (Dahl, 1990, 2000; Tiner & Finn, 1986).

The U.S. Clean Water Act (CWA) defines wetlands as "those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas (Clean Water Act of 1972, 40 CFR 232.2(r)). Wetlands on the Delmarva Peninsula consist of tidal and non-tidal wetlands. Tidal wetlands sit on the coasts of the Atlantic Ocean and the Chesapeake Bay, and have been shown to be responsive to changes in inundation frequency and magnitude (Baldwin *et al.*, 2012; Boesch, 2006; Sharpe & Baldwin, 2009). Non-tidal wetlands generally include those that sit on the floodplains of rivers and streams, geographically isolated depressions, and other low-lying inlands and are susceptible to draining, filling, nutrient runoff, groundwater interception, and soil saturation (Bachman *et al*., 1998; Rogers & McCarty, 2000).

Historically, wetlands in the United States have been viewed as a natural resource that is resilient to changes in climate, land use, and land cover. Until the mid-20th century, wetlands in the U.S. were viewed as having a negative, derogatory connotation. Thus wetlands were overtly drained, filled, and dredged to complement demands for agriculture, nautical transportation, and commercialization, without joint, regional, comprehensive plans from federal, state, and local governments to prevent and mitigate wetland loss with the implementation of policies related to wetland preservation and restoration (Hartmann & Goldstein, 1994). Over time, intense urbanization agricultural cultivation, intensification in the magnitude of hurricane/storm events, sea level rise, and subsequent pollution and sediment discharges have greatly decreased the area and functionality of wetlands in the Bay watershed. Because of wetland loss, federal and state agencies collaborated to create methods of evaluating wetland health and functionality, tracking the anthropogenic activities that alter wetlands, and forecasting areas (on multiple spatial scales) that are vulnerable to wetland loss (Boesch, 2006; Daniels *et al*., 2010).

1.1.2 Physical Processes Linked to Wetland Loss

Physical drivers of wetland loss, like climate change, pose an imminent threat to wetlands (Klemas, 2007; Makkeasorn *et al*., 2009). Sea level rise and hurricanes have led to long term inundation and extended exposure of non-adapted wetland vegetation to saline conditions, which have resulted in wetland loss (Carlisle *et al*., 2006; Najjar *et al*., 2000). One mechanism by which wetlands become vulnerable and eventually converted to open water is inundation, which results from a net loss of surface elevation and rates of sediment deposition relative to rates of sea level rise (Morris, 2002; Cahoon *et al*., 2011). Tidal inundation levels and storm surge levels have been monitored to diagnose the impact of sea level rise and storms on the distribution of wetlands on the Delmarva Peninsula. Dieback of estuarine, palustrine and riverine wetland vegetation and subsequent wetland loss have been linked to increases in tidal inundation, increases in anaerobic conditions, and increases in soil salinity from droughts and long periods of high temperatures (Crowell *et al*., 2011; Hicks *et al*., 1983; Kearney *et al*., 2002; Kearney & Riter, 2011; Kearney & Stevenson, 1991; Pilkey, 2002; Stevenson *et al*., 2002). Non-tidal and inland wetland losses have occurred primarily due to agricultural cultivation, deforestation, human-induced freshwater pond and lake construction, channelization, and urban development (Hartmann & Goldstein, 1994; Tiner, 1987). The quality and sustainability of inland forested and shrub wetlands has been degraded due to discharge of nutrients, herbicides and pesticides, chemicals from industries, and domestic and animal waste from sewage and septic systems (Tiner, 1987; Kearney *et al*., 2011; NRC, 2001; Stanhope *et al*., 2009).

1.1.3 Anthropogenic Drivers of Wetland Loss

A consensus exists among the scientific community, academia, and policymakers that there is a clear linkage between human-induced land use and land cover change and wetland loss in the Chesapeake Bay region (Tiner, 1995; Weller *et al*., 2007). Stressors in the Delmarva Peninsula have included agriculture, urbanization, and transportation infrastructure. Numerous studies support the concept that land cover/land use change directly affect wetland loss in the Chesapeake Bay as well as on the Delmarva Peninsula (Brooks *et al*., 2004; Daniels & Cumming, 2008; Kearney & Rogers, 2010; Mayer & Lopez, 2011; Weller *et al*., 2007).

The Delmarva Peninsula economy thrives on agriculture, poultry, the pulp industry, and tourism. Primary crops include corn and soy beans. Research has shown that the majority of wetlands on the Delmarva Peninsula are adjacent to riparian buffers, croplands, poultry farms, or pine plantations (Allen, 2009; Denver, 2004; Tablante, 2002). Conversion of hardwood forests to loblolly pine plantations as well as row crops led to a decrease in forested wetland cover and an increase in nutrient runoff, subsequently negatively impacting water quality and storm water retention (Hartmann & Goldstein, 1994; Stanhope *et al*., 2009; Whigham *et al*., 2007).

Population growth has increased the demand for housing throughout the Peninsula, especially on the Eastern Shore of Maryland and on uplands of major riverine systems like the Choptank and Chincoteague watersheds (CBP, 2004; Grant *et al*., 2011). As a result, wetlands have been fragmented by residential construction and the insertion of impervious surfaces (*e.g.,* roadways and bridges). Urban development has also increased recreational activities and has resulted in alteration of wetland connectivity and landscapes to cater to activities requiring pathways, ponds, and golf courses. With increases in population and the usage of land cover for residential and commercial purposes, wetlands have also become more vulnerable to pollution in the form of waste dumping, nutrient runoff, and sediments from dredging and filling, storm water runoff and waste discharges. Impacts of wetland loss include degradation in water quality, the alteration of surface water flows, loss of habitat for wetland dependent plant and animal species, and the introduction of invasive species (Baldwin *et al.*, 2012; Rogers & McCarty, 2000).

1.1.4 Landscape Indicators of Wetland Loss

In addition to anthropogenic drivers of wetland conversion, natural landscape characteristics and processes play on important role in wetland functionality and resilience. These concepts have also been referred to as landscape position, landform, water flow path, and water body types (LLWW indicators) (Dvorett *et al.*, 2012; Tiner, 2005). Topography, soil types, hydro-patterns (*e.g.*, tidal inundation and soil moisture), and vegetation cover directly influence spatial and temporal wetland extent and distribution (Lang & Kasischke, 2008; Tiner, 1995, 1999). Flat topography and shallow shorelines reinforce the sustainability of estuarine wetlands in the tidal saline regions, while palustrine wetlands dominate the non-tidal freshwater communities of the Delmarva Peninsula (Baldwin *et al*., 2012). Wetland soils, also referred to as hydric soils, are defined by the U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS, 2007) as "soil that formed under conditions of saturation, flooding or ponding long enough during the growing season to develop anaerobic conditions in the upper part." Alteration in the hydric soil horizon or removal of hydric soils can kill native plant species as well as alter the drainage and flow of storm water and agricultural runoff (Stammermann & Piasecki, 2012). Hydropatterns, a depiction of the spatial and temporal patterns of flooding (*e.g.*, tidal inundation and soil moisture) are the primary determinant of the extent and distribution of wetlands (Lang & Kasischke, 2008). Changes in biogeochemistry/water quality (*e.g.*, salinity, pathogens, metals, and pH) can also influence wetland function (Tiner, 1999; Baldwin *et al*., 2012).

1.1.5 Wetland Management and Policies

In an attempt to improve the management of wetlands on an international level, policymakers have composed an international treaty called the Ramsar Convention (Gulnihal, 2012). On a domestic level, the federal government has implemented legislation and programs like the PL-566 agricultural program, the US Army Corps of Engineers (USACE) wetland delineation manual, and the Federal Crop Insurance Corporation and the Intermodal Surface Transportation Efficiency Act, and Executive Order 13508 for Chesapeake Bay Protection and Restoration (Hartmann & Goldstein, 1994). In order to prevent, minimize and mitigate wetland loss on a nationwide scale, in 1975 the US Congress created Section 404 of the Clean Water Act, which authorizes the US Army Corps of Engineers to design, manage, review, administer and enforce permits activities that involve the discharge of dredged or fill material. Examples of such activities include, but are not limited to, residential development projects that require filling materials, the revitalizing of transportation infrastructure, and the construction of water resources like levees. In 1990, an Executive Order introduced a national goal of

"No Net Wetland Loss" (Copeland, 2010). While planners and policymakers have implemented these policies on national, local and state scales, multi-jurisdictional, regional impacts of land use change and permitting on wetland change have varied.

Despite governmental intervention, the Delmarva Peninsula has continued to experience agricultural conversion of wetlands, the expansion of impervious surfaces, and channelization of streams (Hartmann & Goldstein, 1994). Tiner and Finn (1986) examined the impacts of anthropogenic drivers of wetland loss by calculating statewide percentages of wetland loss between the 1950s and the 1970s. Agricultural cultivation accounted for 21% in DE, 38% in MD, and 45% in VA. Channelization accounted for losses of 54% in DE, 33% in MD, and 27% in VA. Ponding due to the implementation of stormwater treatment projects, the creation of reservoirs for livestock, or peat removal altered the natural hydrologic processes (e.g., the frequency of inundation) and accounted for approximately 20% of wetland loss in MD and VA, and urbanization led to an estimated 12% loss in DE. Wetland permitting on the Delmarva Peninsula is a complex process that would benefit from more cohesive information management between the municipalities, state environmental agencies and the US Army Corps of Engineers. A clear gap of knowledge exists between the geo-statistical relationship between the existing wetland permitting programs and changes in wetland cover identified by national land cover datasets (Brody *et al.*, 2008; Brody *et al.*, 2007).

The majority of studies on wetland change and permitting in the United States have focused on wetland permitting and the compensatory mitigation component of Section 404, rather than the relationships between the geographic distribution of permits, locations and trends of wetland loss, and the drivers of wetland loss (Brody *et al.*, 2008;

Cole & Shafer, 2002; Robb, 2002; Sifneos *et al*., 1992). According to the National Research Council, there is a need for time series research on wetland change in order to establish a baseline to calculate mitigation ratios and the spatial and biochemical efficiency of wetland restoration (NRC, 2001). Geographically, the majority of studies have focused on wetlands in Gulf of Mexico, the West Coast, and the Chesapeake Bay (Brody *et al*., 2008; Hartmann & Goldstein, 1994).

1.1.6 Methods for Assessing Drivers of Wetland Change and Permitting

The scientific, economic, and political communities are in need of targeted research that can quantify changes in wetland permitting and impacts on wetland spatial location, geochemical characteristics, functionality, and vulnerability (Tiner, 2005; Weller *et al*., 2007; Whigham *et al*., 2007). During the 1980s and 1990s, wetland researchers primarily performed wetland conditional and to some degree functional assessments using one or more of the following methods: The Wetland Evaluation Technique (WET) created by the Federal Highway Administration, The Environmental Monitoring Assessment Program (EMAP) created by the EPA, and the Hydrogeomorphic (HGM) approach created by the US Army Corps of Engineers (Daniels *et al.*, 2010; Dvorett *et al*., 2012; Hartmann & Goldstein, 1994; Stein *et al*., 2009). These models focused on evaluating the ecological conditions and functions of a specific wetland site in a given region. Though beneficial to ecologists and biologists, there is still a demand for a comprehensive method that can locate and quantify wetland loss on a regional level, and can explore the statistical relationship between physical and social drivers of wetland loss (Smith & Tran, 2003).

Historically, scientists have used active and passive remote sensing instruments to classify and quantify land cover and land use change, to diagnose wetland health, and identify areas of concern and vulnerability (Kearney *et al*., 1998, 2002, 2009; Kearney & Riter; 2011; Klemas 2007, 2011). Established scientists like Ralph Tiner, Don Cahoon, and John Day have stressed the importance of ground work to validate inferences on physical and biogeochemical processes made by remote sensing imagery classification and analysis (Cahoon & Turner, 1989; Day *et al*., 2008; Tiner, 2005; Turner & Rao, 1990).

Current initiatives that are relevant to the assessment of wetland change include time series analyses of aerial and satellite imagery, landscape and rapid assessment methods, and probability based sample surveys (Huang *et al.,* 2009; Klemas, 2007; Stevens *et al*., 2007;Thomas *et al*., 2011; Wardrop *et al*., 2007). Remote sensing products have been combined with finer resolution datasets (*e.g.,* the U.S. Fish and Wildlife Service (FWS) National Wetland Inventory) to improve the accuracy of wetland classes within land cover datasets (*e.g.,* the National Land Cover Dataset and the NOAA – Coastal Change Analysis Program). For example, Elijah Ramsey of the USGS utilizes remote sensing to classify and analyze changes in land cover to help track trends of wetland loss and recovery in the Gulf of Mexico (Klemas, 2011; Ramsey *et al.*, 2009, 2011). Land use and land cover change models (*e.g.,* the Chesapeake Bay Land Change Model) have helped predict population growth and subsequent land cover change trends (*e.g.*, wetland loss) in the Chesapeake Bay (Jantz *et al.,* 2010, 2011; Neilsen & Prince, 2008). Time series analyses, in situ groundwork, and GIS data regarding socioeconomic conditions can be powerful tools to assess wetland loss, if they are maintained and applicable to policies and regulations (Klemas, 2007).

The applicability of wetland time series analyses and vulnerability assessments can be improved by integrating landscape context descriptors (e.g., the spatial patterns of habitat, water levels, and buffers) with land use and land cover change datasets and socioeconomic indicators (Daniels & Cumming, 2008; Gutzwiller & Flather, 2011; Hollister *et al*., 2004; Phillips, 2004). However, most existing studies on the status of wetlands on Delmarva Peninsula either focus solely on local (micro scale) non-tidal systems, generic landscape indicators, geological properties, hydrogeomorphic functions, or agricultural development (Hussein & Rabenhorst, 2001; Nagler *et al*., 2009; Rabenhorst *et al*., 2001; Tiner, 2005; Whigham *et al*., 2007). On the other hand, policymakers have primarily relied on the FWS, state agencies, and think-tanks for research on the status and trends of wetlands on regional to national scales (USACE, 2010; USFWS, 2011; Mayer & Lopez, 2011; Tiner *et al.*, 2011), whereas the Congressional Research Service (CRS), the U.S. Army Corps of Engineers (USACE), and the Environmental Law Institute (ELI) have led research on wetland law and policies (Connolly *et al*., 2005; Connolly, 2006; ELI, 2008, 2010, 2015). There has been a recent increase in research on how wetland change and policies relate to one another on a variety of spatial (local and regional) and temporal (annual versus decadal) scales (Thomas & Lamb, 2005).

1.2 **Proposed Research**

1.2.1 Problem Statement

Despite existing policies and regulations addressing wetlands, net wetland loss on the Delmarva Peninsula continues (Brody *et al*., 2008). A comprehensive understanding of physical, socioeconomic, and policy drivers of wetland alteration and loss on a regional scale is needed to reduce future net wetland losses (EPA, 2008, Klemas, 2011; Tiner, 2005; Ward *et al*., 2012; Whigham *et al*., 2007).

This research is important because it provides policymakers and planners geospatial data and analyses that can be used to identify physical and anthropogenic drivers of wetland loss. Researchers need integrative methods of assessing the vulnerability of ecosystems and communities to wetland loss. Third, statistical and geospatial analyses must repeatable on multiple spatial and temporal scales in order to provide a foundation for effective wetland conservation and restoration policies on a regional level.

<u>Research Questions</u>

The overarching research questions that this study seeks to answer are:

1. What are the cumulative impacts of physical and anthropogenic drivers of wetland change, including the wetland permitting system, on wetland extent and distribution from 1980 to 2010?
2. And, which wetlands are most vulnerable to future alterations and loss assuming current drivers?

The overarching research questions are further divided into key questions:

1. What is the area of wetland change [How much wetland change has occurred] over the last 30 years on the Delmarva Peninsula measured by existing geospatial data sets?
2. What physical and anthropogenic drivers of land use and land cover change are correlated with wetland loss on the Delmarva Peninsula?
3. What information does the spatial and temporal distribution of wetland permits and wetland loss patterns provide regarding the influence of wetland change drivers and the impacts of the wetland permitting system?
4. What wetlands, watersheds, and counties are most vulnerable to wetland loss due to physical, socioeconomic and policies that drive wetland change?

1.2.2 Objectives

My dissertation will focus on wetland loss on the Delmarva Peninsula from 1980 until 2010. My objectives are (1) to locate and quantify wetland loss, (2) to investigate the relationship between wetland loss and the trends and pattern in wetland permitting, (3) to explore the statistical relationship between wetland loss, physical and anthropogenic drivers of wetland loss, and landscape indicators, and (4) to test the predictability of wetland change using predictor variables derived from principal component analysis. The overall schematic for my proposed research design is laid out in Figure 1.1. Schematics for each objective will be included in each respective section of the methodology.

My ultimate goals are to create a geo-statistical tool and recommendations that will:

1) Identify key drivers of wetland change;
2) Identify physical and political areas that are most vulnerable to wetland loss;
3) Provide information that can be used to improve wetland conservation, restoration and permitting strategies.

Objective 1: I will use existing land cover datasets to analyze how land use and land cover on the Delmarva Peninsula have changed over the last 30 years (Refer to Figure 1.2). My hypotheses are (1) that wetland change trends in the coverage and location of wetlands can be quantified using existing land cover maps, and (2) there has been a net loss of wetland acreage on the Delmarva Peninsula from 1980 to 2010.

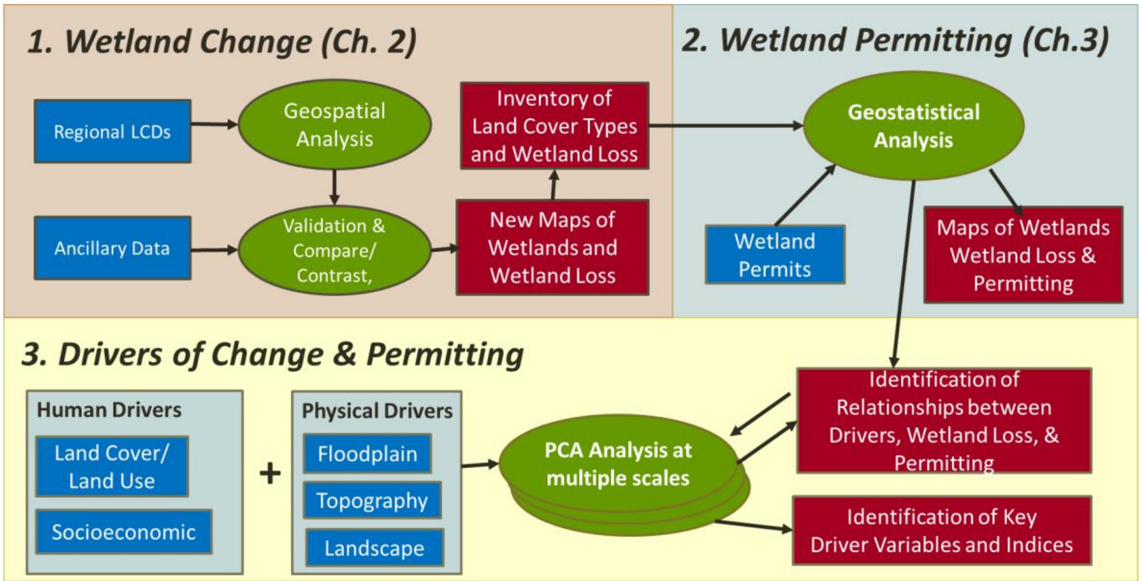

Figure 1.1 Overall Schematic for the Research Design

Note - For this framework diagram, BLUE boxes represent source data and indicators, GREEN circles represent objectives/processes and RED boxes represent deliverables.

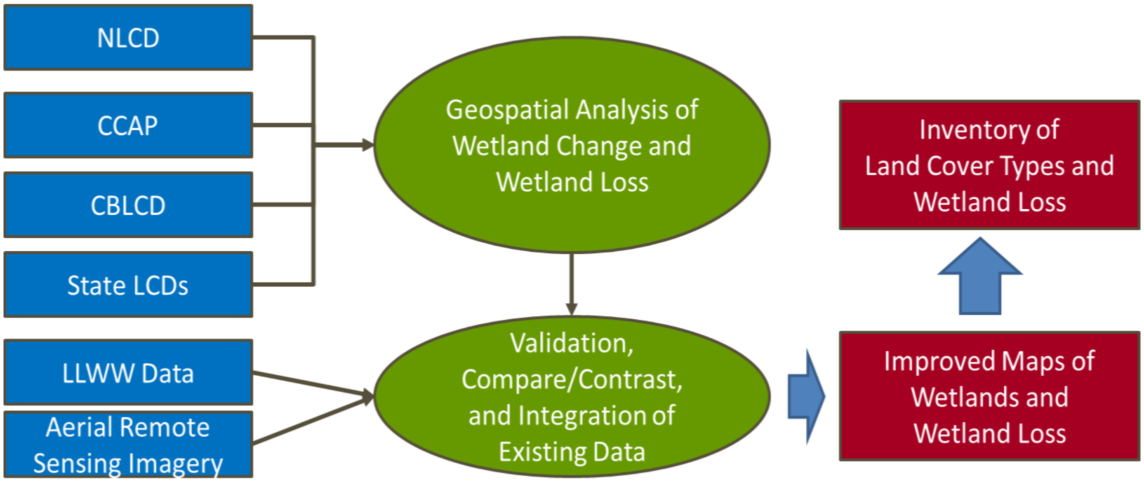

Figure 1.2 The Framework Design for Objective 1

Note - For this framework diagram, BLUE boxes represent source data and indicators, GREEN circles represent objectives/processes and RED boxes represent deliverables.

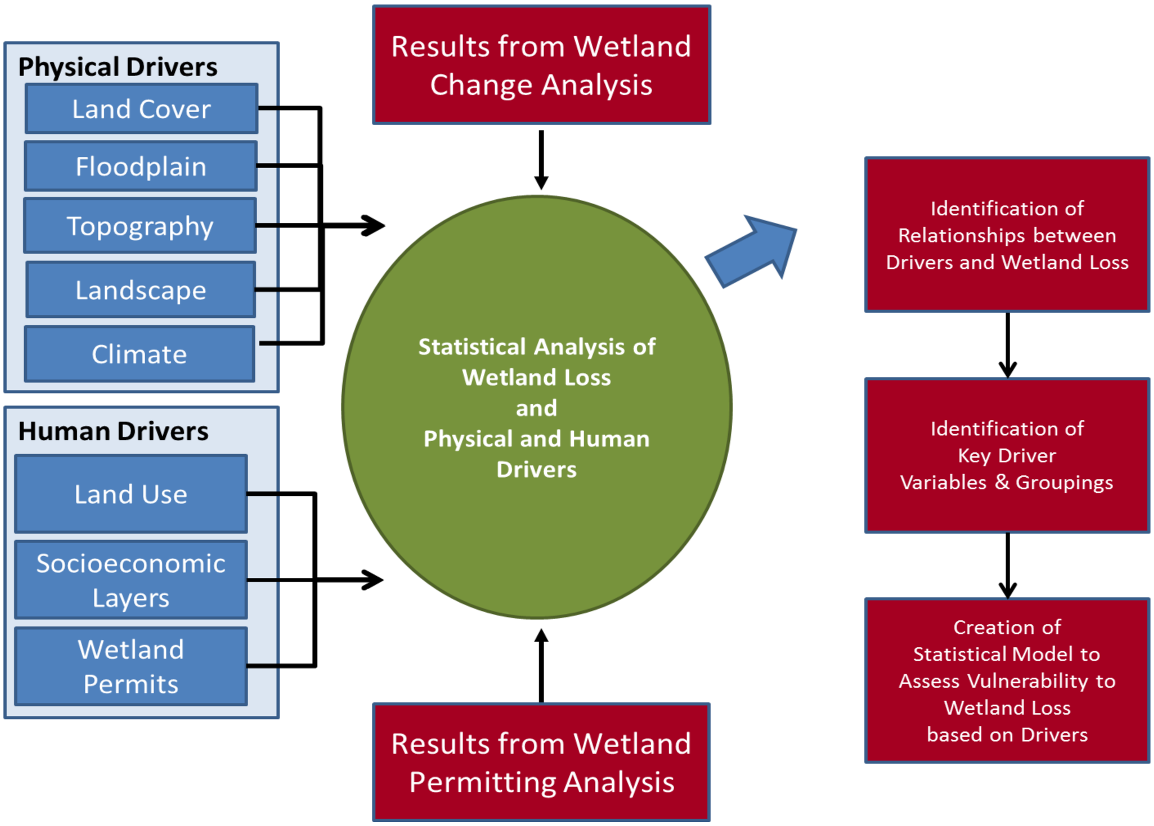

Figure 1.3 Framework Design for Objectives 2, 3, and 4

Note - For this framework diagram, BLUE boxes represent source data and indicators, GREEN circles represent objectives/processes and RED boxes represent deliverables.

Objective 2: I will locate and quantify the spatial and temporal distribution of wetland permits in relation to wetland loss patterns (Refer to Figure 1.3). My hypotheses are as follows: (1) Wetland loss is positively correlated with wetland permit density, (2) Permit density is positively correlated with the acreage of roadways and urban/suburban land cover, (3) Permit density is negatively correlated with the acreage of undeveloped land cover, and (4) Temporally, permit density is positively correlated with increases in the acreage of developed land cover.

Objective 3: I will explore the statistical relationship between the physical and anthropogenic drivers of land use and land cover change with wetland loss on the Delmarva Peninsula (Refer to Figure 1.3). My hypothesis is that wetland loss is positively correlated with the acreage of impervious surfaces (roadways), agriculture, urban and suburban residential and commercial land cover, and human population, and housing density.

I also identify the socioeconomic, political, and land scape context variables that have the strongest associations with wetland loss and wetland permitting (Refer to Figure 1.3). My hypothesis is that areas with the highest steepest slopes, highest soil erosion factors, densities of impervious surfaces, housing growth, agricultural expansion, wetland cover, and the shortest distance from the urban centers, roadways, and floodplains will have the highest associations with wetland loss and permitting.

My analysis will integrate physical and anthropogenic indicators, landscape characteristics, and proximity measurements (within a given buffer) to create a logistic model to test the predictability of wetland change. The project will combine Kentula *et al.*, (2004), Nosakhare *et al.*, (2012), Gutzwiller and Flather (2011), and Daniels and

Cumming (2008) research interdependencies between wetland habitat loss landscape characteristics, land cover change, and climate change with studies like that of Brody *et al.*, (2008) that explore the spatial-temporal distribution of wetland permits.

1.3 **Outline of Dissertation**

This dissertation is organized into five chapters (including this introduction). The research is presented as a set of stand-alone manuscripts (Chapter 2 to 4) that will be submitted to journals for peer-review. The first chapter (the introduction) introduces the problem statement, the objectives, and organization of the dissertation. The next three chapters include: 2) Assessment of Wetland Change on the Delmarva Peninsula from 1984 to 2010 using the Regional Land Cover Data Sets, 3) A Spatial-Temporal Analysis of Section 404 Permitting on the Delmarva Peninsula: Tracking Thirty Years of Wetland Change and Management, and 4) A The predictability of wetland change based on a principal component analysis of the primary drivers of wetland loss on the Delmarva Peninsula from 1980 to 2010. The final chapter (5) summarizes the major conclusions and discusses the significance of the research.

2 <u>CHAPTER 2: ASSESSMENT OF WETLAND CHANGE ON THE DELMARVA PENINSULA FROM 1984 TO 2010</u>

ABSTRACT

The decline in wetland extent and condition emphasizes the need for sound wetland restoration and conservation policies, which require baseline information on wetland status, change and change drivers. Multiple wetland maps are available but they can be quite inconsistent, due to different input and generation techniques, dates, and objectives. Moderate resolution ($30m^2$) regional land cover data sets were analyzed to: 1) quantify historical wetland changes on the Delmarva Peninsula at multiple spatial scales between 1984 and 2010, 2) identify differences in the spatial area of wetland change and discuss the source of and implications for these differences, and 3) investigate the extent to which drivers of wetland change can be identified using existing land cover data sets (LCDs). The following regional LCDs were considered: the National Oceanic and Atmospheric Administration (NOAA) Coastal Change Analysis Program (C-CAP), the U.S. Geological Survey (USGS) Chesapeake Bay Land Cover Data Series (CBLCD), and the USGS National Land Cover Database (NLCD). The C-CAP and CBLCD had the highest spatial agreement at 97%, and an average of 76% spatial agreement with the U.S. Fish and Wildlife Service (FWS) National Wetland Inventory (NWI). The highest percentages of net wetland loss occurred between 1992 and 2001, while net wetland gain occurred 2001 to 2010. Wetlands were predominantly converted (*e.g.,* lost) to croplands/grass/shrubs (67%) and water (11%), which could be linked to drivers like agriculture and sea level rise.

2.1 **Introduction**

The distribution, function, and value of wetlands are spatially and temporally dynamic. Wetlands mitigate sediment and nutrient loads, regulate stormwater, recharge groundwater, and provide food and habitat for coastal and migratory species (Chase, Musser, & Gardner, 2003; Hartmann & Goldstein, 1994; Tablante *et al.*, 2002; Wilson, Watts, & Brinker, 2007). Wetland functions and services are often influenced by their location within the greater landscape (Strayer *et al.,* 2003; Wu, 2004; Wu, Li, & Chen, 2011). For example, hydrologic connectivity between wetlands and upstream land cover directly impacts wetland function, and vulnerability. Natural and anthropogenic drivers of land use and land cover (LULC) change, like urbanization and climate change have been linked to the loss and fragmentation of wetlands. Drivers of wetland change operate on different temporal and spatial scales, which increases the need for cumulative assessments of LULC change, more specifically wetland change.

2.1.1 Background

Wetland change continues to threaten water quality and ecosystems in the Chesapeake Bay (CB) and the Delaware Bay (DB). The Delmarva Peninsula located in the Coastal Plain portion of CB has the highest concentration of wetlands in the CB watershed. According to the 2011 National Land Cover Database (NLCD), approximately 36% of wetlands in the CB watershed are located on the Delmarva Peninsula (Homer *et al*., 2015). Over the past 200 years, the Delmarva Peninsula has lost over 2.1 million acres of wetlands (including coastal and inland wetlands) due to dredging, channelization, draining and filling, and ponding (Dahl, 1990; Dahl 2000; Tiner & Finn, 1986; Tiner & Finn, 2012). Population growth has increased the demand for housing throughout the peninsula, especially on the Eastern Shore of Maryland and on uplands (*e.g.,* terrestrial lands that are commonly dry) of major riverine systems like the Choptank and Chincoteague watersheds (Moglen *et al.*, 2011; Nosakhare *et al.*, 2012; Chesapeake Bay Office, 2012). As a result, wetlands have been lost or fragmented by residential and recreational construction and related infrastructure (*e.g.,* roadways and bridges). Wetland loss on the peninsula is correlated with degradation in water quality, the alteration of surface water flows, loss of habitat for wetland dependent plant and animal species, and the proliferation of invasive species (Batzer & Baldwin, 2012; Denver *et al*, 2004; Rogers & McCarty, 2000). Despite governmental intervention, the Delmarva Peninsula has continued to experience agricultural conversion of wetlands, the expansion of impervious surfaces, and channelization (Dahl, 1990; Hartmann & Goldstein, 1994). Since the late 1700's, over 50% of wetlands in DE, approximately 60% of wetlands in MD, and 40% of wetlands in VA have been converted to non-wetlands (Dahl, 1990; Fretwell, Williams, & Redman, 1996). Between 1950 and 1970, agricultural cultivation accounted for 21% of wetland loss in DE, 38% in MD, and 45% in VA. Channelization that make streams more navigable for larger ships or diverted water away from agricultural land accounted for wetland losses of 54% in DE, 33% in MD, and 27% in VA. Ponding that resulted in permanently inundated wetlands due to peat removal or from the creation reservoirs for livestock accounted for approximately 20% of wetland loss in MD and VA, and urbanization led to an estimated 12% loss in DE (Tiner & Finn, 1986). There is a need for further research on wetland change, the drivers of wetland conversion, and the impacts of wetland change and policies on the biogeographical composition and function of wetlands on larger scales.

Previous studies have demonstrated that stressors like agriculture, urbanization, and sea level rise directly impact wetlands on the Delmarva Peninsula (Brooks, Wardrop, & Bishop, 2004; Daniels & Cumming, 2008; Kearney and Rogers, 2010; Mayer & Lopez, 2011; Weller *et al*., 2007). Although local studies can reveal important information regarding wetland change drivers and impacts, they have the potential to be too fine scale (*e.g.*, tax parcel and site specific) to accurately support [be the base of] public policies on issues that cross multiple spatial scales (*e.g.*, floodplain regions or transportation corridors). In-situ or short term research can be exclusive of socioeconomic (*e.g.*, migration) or landscape characteristics (*e.g.*, topography and streamflow) that may cross local boundaries or geophysical conditions, which can lead to the implementation of policies like infrastructure management that fail due to being unsustainable or inconsiderate of episodic drivers like hurricanes or continuous disturbances like sea level rise.

In order to create and implement efficient economic and environmental policies, policy and decision makers need a clearer understanding of: 1) where wetlands are located, 2) how wetlands respond to disturbances and 3) how the conversion of wetlands to other land cover types can impact ecosystem habitats and services (Boesch, 2006; Daniels *et al*., 2010). According to the National Research Council (NRC), there is also a need for time series research on wetland change in order to establish baseline indicators to target areas for mitigation and to evaluate the efficiency and effectiveness of wetland restoration projects (NRC, 2001). Natural resource managers use wetland maps to identify critical areas in need of sustainable conservation and restoration policies. In order to identify priority areas, scientists and policymakers must first understand the health, functionality, and connectivity of wetlands. Wetland mapping enables managers to aggregate/integrate and analyze multiple conditional and function-based wetlands assessment in order to understand how wetlands are responding to disturbances in a variety of hydrogeomorphic (HGM) classes(*e.g.*, upland, isolated, headwater, non-tidal, and estuarine). The integration of wetland maps with other GIS tools like floodplain mapping are used to inform the management of nutrient and sediment loading rates by comparing the spatial distribution of wetlands with recorded nutrient and sediment levels. The wetland maps help scientists understand how and where wetlands effect nutrient levels, which helps policymakers reduce negative impacts like algal blooms and eutrophication. Wetland maps are also vital in building scenarios and forecasting the impacts of climate change like sea level rise and increases in the size and duration of storm surges.

2.1.2 Existing Research

Historically, scientists have used passive (*e.g.*, Landsat and aerial photography) remote sensing instruments to classify, locate, and quantify land use and land cover change and to identify critical areas of concern and vulnerability (Barras, Bernier, and Morton, 2008; Couvillion *et al.*, 2011; Hartmann & Goldstein, 1994; Kearney & Riter, 2011; Klemas, 2004; Klemas, 2011). The most accurate, finest spatial resolution, national wetland mapping product is the National Wetland Inventory (NWI), a vector data set, created by the U.S. Fish and Wildlife Service (FWS) to establish an inventory of wetlands. However, the NWI dataset is not updated on a regular time step and population growth, anthropogenic development, climate change , and other dynamic drivers of wetland change necessitate up-to-date land cover products that can be used recurrently to calculate and interpret land cover change trends and patterns between natural and developed land cover. Thus, moderate spatial resolution remote sensing products have been combined with ancillary GIS datasets (*e.g.*, the National Wetland Inventory) to create land cover datasets that do not solely rely on spectral signatures or spatial resolutions too coarse to accurately identify and quantify wetland change. The most comprehensive, national land cover change data sets in the United States, applicable to this study, include the U.S. Geological Survey (USGS) NLCD, the National Oceanic and Atmospheric Administration (NOAA) – Coastal Change Analysis Program (C-CAP), and the USGS Chesapeake Bay Land Cover Database (CBLCD). These data sets utilize techniques like the Classification and Regression Tree (CART) analysis and the Cross Correlation Analysis (CCA) to detect wetland change. CART and CCA use decision trees to decipher the best classification of a pixel by process of elimination, based on the pixel's relationship to a series of land cover classes, their spectral signatures and the respective land cover classes within a set number of pixels surrounding the target pixel (SEGAP, 2014; Wang, 2010). These regional LCDs cannot only help us quantify and locate wetland change, but they may also be helpful for identifying potential drivers of wetland change. For example, a wetland change analysis of aerial photography from 1982 to 1989 on the Lower Eastern Shore of the peninsula found that agriculture, ditching, and the timber industry (*e.g.*, loblolly pine) were the primary drivers of palustrine forested wetland loss. Wetland stressors are often related to not only wetland loss but also degradation (Nielsen, Prince, & Koeln, 2008).

2.1.3 Objective

The objective of this study was to: 1) quantify historical wetland change trends and patterns at multiple spatial scales between 1984 and 2010 using regional LCDs, 2) identify differences in wetland related land cover change as determined using multiple LCDs and discuss the source of and implications for these differences, and 3) investigate the extent to which drivers of wetland change can be identified using existing LCDs.

2.2 **Data and Methods**

The methods section begins with a description of the study area, the Delmarva Peninsula. Second, the following source data sets are summarized: the regional land cover data sets, and the ancillary data used to test the spatial agreement of the regional LCDs. Third, the spatial accuracy of the wetland classifications is discussed in accordance to existing metadata and literature. The methods section concludes with a summary of the five step methodology.

2.2.1 **Study Area**

As illustrated in Figure 2.1, the Delmarva Peninsula is 14,130 km^2 in area and covers portions of Eastern Shore of Maryland, the majority of Delaware and two counties of northeastern Virginia. Located in the Coastal Plain Physiographic Province, the northwestern edge of the peninsula sits on a fall line that serves as a transitional zone between the Piedmont and Coastal Plain regions (Hartmann & Goldstein, 1994). The peninsula falls between three major water bodies: the Delaware Bay to the north, the Atlantic Ocean to the east, and the Chesapeake Bay to the south and west. The Delmarva Peninsula economy thrives on agriculture, poultry, the pulp industry, and tourism. The Delmarva Peninsula was used for this study because of its variation in land cover types, heavy concentration of wetland cover, trends of population growth, and its unique, position on the Mid-Atlantic seaboard that is susceptible to natural hazards like storm surges and sea level rise. The peninsula complex possesses a high percentage of undisturbed barrier islands and shorelines of the mainlands that are experiencing continuous pressures from seasonal tourism, coastal development, and sea level rise.

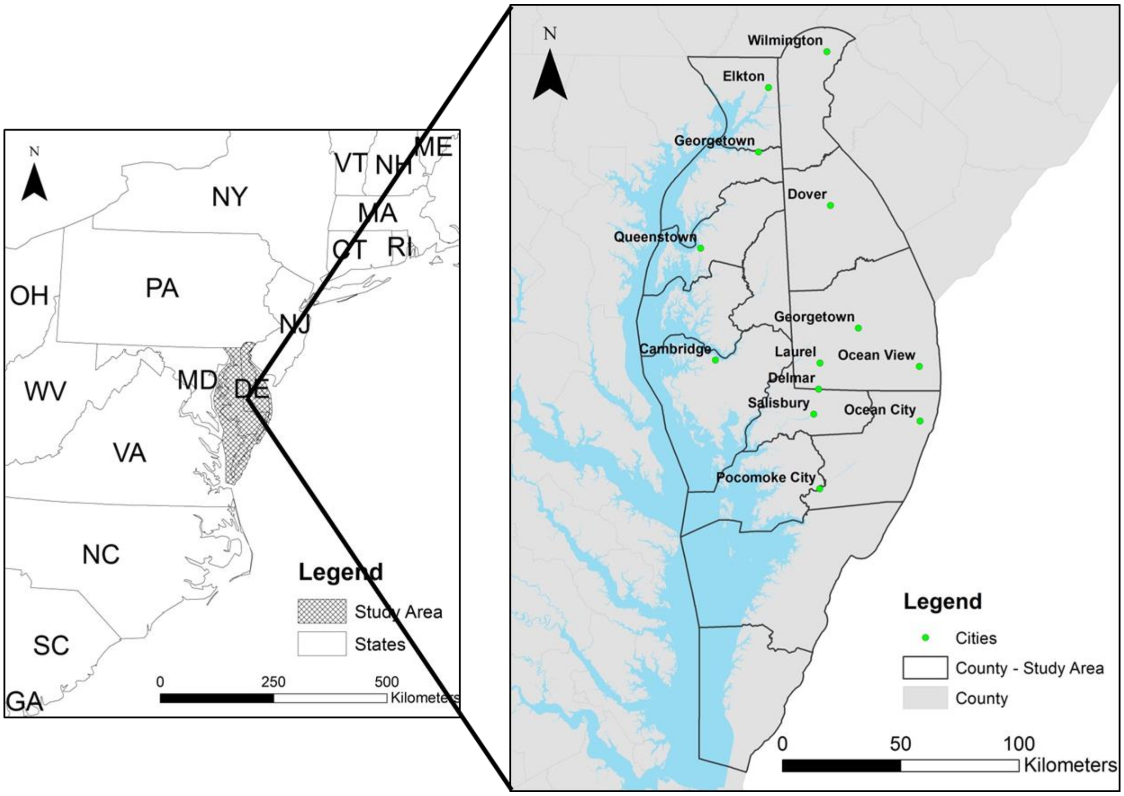

Figure 2.1 The Study Area – The Delmarva Peninsula subdivided counties

(Sources: US. Census Bureau)

2.2.2 Source Data

Regional Land Cover Data Sets. This study used the NLCD, C-CAP, the CBLCD, state LCDs, and the NWI to quantify and locate wetland change on the Delmarva Peninsula. Each LCD was compared to the most recent NWI and state land cover datasets from Delaware and Maryland to test the spatial accuracy of the regional LCDs on a pixel to pixel basis (See Tables 2.1 and 2.2). Table 2.1 provides a list of the moderate resolution regional datasets and Table 2.2 provides a descriptive list of the ancillary, vectorised data sets. Detailed information on each dataset is discussed in the following section. Table 2.3 provides a conversion matrix of the wetland classes from each regional LCD to Anderson Level-I and Level-II land cover classes.

Data Source	Wetland Land Cover Categories	Spatial Data Format And Resolution and MMU	Temporal Scale	Source Data: Classification Methodology	Imagery Year/ Quantity/Season	Accuracy Assessment of Wetland Classification User/Producer %	References
National Land Cover Database (NLCD) (USGS)	Anderson Level-II: Woody Emergent	Raster: 30 m^2 MMU: .5 pixel/1 acre	2001 2006	Landsat: Classification and Regression Tree (CART)	2001: 20 Leaf Off/ 19 Leaf On 2006: 3 Leaf Off/ 3 Leaf On	Anderson Level-I: 1992: 48%/64% 2001: 38%/78% 2006: 38%/39% 2006:	Wickham et al., 2013 USGS, 2012 Fry et al., 2011 Wickham et al., 2010 Fry et al., 2009 Homer et al., 2007 Hollister et al., 2004
The Coastal Change Analysis Program (C-CAP) (NOAA)	Anderson Level-II: Palustrine Forested Palustrine Scrub Shrub Estuarine Forested Estuarine Scrub Shrub Emergent Herbaceous Palustrine Emergent Estuarine Emergent	Raster: 30 m^2 MMU: .22 acres .88 acres	1992 1996 2001 2006 2010	Landsat: CART	1992: 5 Leaf Off/ 2 Leaf On 1996: 6 Leaf Off/ 3 Leaf On 2001: 9 Leaf Off/ 10 Leaf On 2006: 6 Leaf Off/ 6 Leaf On	Zone 60 (Mid Atlantic): 2012: 71% Wetland Class Specific: Palustrine woody/shrub-scrub: 82%/82% Palustrine emergent: 83%/83% Estuarine shrub-scrub: 50%/50% Estuarine emergent: 90%/100%	NOAA, 2013 McCombs et. al, 2016 Burkhalter et al., 2005 Dobson, 2005 NOAA, 1995
The Chesapeake Bay Land Cover Data Series (CBLCD) (USGS – CBP)	Anderson Level-I: Woody Emergent	Raster: 30 m^2	1984 1992 2001 2006 2011	NLCD 2001, Landsat: CART	NLCD 2001, 2001 Landsat – 3 seasonal dates (spring, leaf-off, leaf-on) Additional leaf-off imagery	Nationwide: N/A Wetland Class Specific: N/A	MDA, 2009

Table 2.1 Summary of Available Moderate Resolution (30 meter by 30 meter) Regional Land Cover Databases: the National Land Cover Database (NLCD), the Coastal Change Analysis Program (C-CAP), and the Chesapeake Bay Land Cover Data Series (CBLCD).

Data Source	Wetland Land Cover Categories	Spatial Data Format and Scale	Temporal Scale	Source Data: Classification Methodology	Imagery by Year/Season	Accuracy Assessment	References
National Wetland Inventory (NWI) (US FWS) NWI Archive (2002) (CBP)	Systems: Estuarine Palustrine Riverine Lacustrine (Cowardin Classification)	Vector State Scale: DE, MD, VA 1:58,000 1:40,000	DE: 1981-2009 MD: 1977-2009 VA: 1974-2010	color-infrared aerial photography	1977 -1999: Spring 2007: Leaf-On/Leaf Off	Nationwide: N/A Wetland Class Specific: Site Specific (e.g. MA and VA)	NWI Metadata Lunetta *et al*., 1991 Swartwout *et al*., 1981 Stolt and Baker, 1995
National Wetland Inventory Archive (2002) (CBP)	Systems: Estuarine Palustrine Riverine Lacustrine (Cowardin Classification)	Vector State Scale: DE, MD, VA	DE: 1975-1989 MD: 1977-1989 VA: 1980-1989	Aerial photography	1975 – 1989: Leaf On/Leaf Off	N/A	CBP, 2002
Land Use/ Land Cover (Delaware Office of State Planning Coordination)	Emergent – Tidal/Non-tidal Forested – Tidal/Non-tidal Scrub/Shrub Tidal/Non-tidal	Vector Minimum Mapping Unit: 0.1 to 0.5 acres 1:50,000	2002 2007 2012	Aerial Photography, Digital Orthophotography, False Color Infrared (FCIR), Leaf-Off Photography, Land-Use Land-Cover	2002: Spring 2007: Summer 2012	Nationwide: N/A	FGDC, 2009 Tiner *et al*., 2011
Land Use/ Land Cover (Maryland Department of Planning)	Anderson Level-I Classification: Wetlands	Vector: 1:63,360	2002 2010	SPOT imagery, Aerial photography MdProperty View (urban parcel data)	N/A	Nationwide: N/A Wetland Specific: N/A	MDP, 2003 MDP, 2010

Table 2.2 Geospatial Data Sets Integrated to Test the Spatial Agreement and Spatial Area of Wetland Coverage.

Note - Unlike the primary moderate resolution raster data sets, the ancillary data used in this study includes the vector data sets like the National Wetland Inventory and state land cover data sets from Delaware and Maryland. For each data set, the table lists the wetland land cover classification scheme, the file type, spatial resolution, and temporal scale. The table also lists each data set's methodology, source imagery, and existing accuracy assessments.

Anderson Level-I Category	Anderson Level-II Category NLCD and CBLCD	C-CAP Category
Urban or Built-up Land (1)	Developed, High Intensity (24) Developed, Medium Intensity (23) Developed, Low Intensity (22) Developed, Open Space (21)	High Intensity Developed (2) Medium Intensity Developed (3) Low Intensity Developed (4) Open Space Developed (5)
Agricultural Land (2)	Cultivated Crops (82) Pasture/Hay (81)	Cultivated Land (6) Pasture/Hay (7)
Rangeland (3)	Grassland / Herbaceous (71) Scrub / Shrub (52)	Grassland (8) Scrub Shrub (12)
Forest (4)	Deciduous Forest (41) Evergreen Forest (42) Mixed Forest (43)	Deciduous Forest (9) Evergreen Forest (10) Mixed Forest (11)
Wetlands (6)	Woody Wetlands (90)	Palustrine Forested Wetlands (13) Palustrine Scrub Shrub Wetlands (14) Estuarine Forested Wetlands (15) Estuarine Scrub Shrub Wetlands (16)
	Emergent Herbaceous Wetlands (95)	Palustrine Emergent Wetlands (17) Estuarine Emergent Wetlands (18)
Open Water (5)	Open Water (11)	Open Water (21) Palustrine Aquatic Bed (22) Estuarine Aquatic Bed (23)
Barren Land (7)	Barren Land (31)	Unconsolidated Shore (19) Barren Land (20)

Table 2.3 A land cover of classification scheme for the regional land cover data sets from 1984 to 2010. This matrix was used to normalize the NLCD, CBLCD, and C-CAP data sets to two basic scales: 1) Wetlands and 2) forested versus emergent herbaceous wetlands.

The National Land Cover Database (NLCD). With respect to the time period of this study, the Multi-Resolution Land Characteristics Consortium (MRLC) has produced NLCD datasets (NLCD) for 1992, 2001 and 2006 (Fry *et al*., 2011; Homer *et al*., 2007; Vogelmann *et al*., 2001) (See Table 2.1), but the 1992 dataset was not used as part of this study due to an unacceptable error level (Fry *et al.*, 2011; MDA, 2009). As shown in Table 2.3, this nationwide dataset consists of 16 land cover classes, which include 2 wetland classes (*i.e.* woody and emergent), four natural classes applicable to the study area *(i.e.* barren, forest, rangeland, and water), two agricultural classes (cultivated crops and pasture/hay), and four urban classes (developed: high intensity, medium intensity, low intensity, and open-space). Unlike the 1992 database, the 2001 and 2006 NLCD data sets were produced using a combination of supervised classification and geometric-correction of Landsat imagery (Fry *et al.*, 2011; Homer *et al.*, 2007; Vogelmann *et al.*, 2001). In order to perform the most accurate classification of wetland and impervious surface cover, analysts gathered Landsat-5 and Landsat-7 images (3 for 2001 and 2 for 2006) from leaf on and leaf off seasons for each path/row (Fry *et al.*, 2011; C. Homer *et al.*, 2007; Vogelmann *et al.*, 2001). The seasonal variation of imagery improved the accuracy of classifications of land covers that may be located under foliage (*e.g.,* forested wetlands) or under temporary hydrologic conditions (*e.g.,* ponding or episodic flooding) (Lunetta & Balogh, 1999; Lunetta *et al.*; 1991; Vogelmann *et al.*, 2001). The 2001 and 2006 datasets used automated Landsat imagery selection and CART analyses, that resulted in user's (producer's) accuracy of wetland Anderson Level-I classification of 38% (78%) for 2001 and user's accuracy of 38% (39%) for 2006. The NLCD uses substantial ancillary data in the mapping of urban land cover classes, which increases the accuracy of urban land cover, and the classification of road networks and infrastructure as urban land cover. The texture of the NLCD is smoothed through the application of a minimum mapping unit, which aggregates clusters of pixels of the same class (Fry *et al.*, 2011; Homer *et al.*, 2007). The NWI served as the primary ancillary source data, while, GAP analyses, USGS land use/land cover, hydrography, and digital elevation models were also used to increase wetland classification accuracy as well as to serve as baseline data for locations without NWI data (Vogelmann *et al*., 1998a; Vogelmann *et al*., 1998b).

The Coastal Change Analysis Program (C-CAP). The C-CAP is produced for coastal regions and adjacent uplands, which encapsulates the entire study area (*i.e.* the Delmarva Peninsula). The C-CAP data series used for this analysis came from 1992, 1996, 2001, 2006, and 2010 (See Table 2.1). The 1996, 2001, 2006, and 2010 updates were completed using a CART analysis, ad-hoc interpretations of aerial imagery, and field data collection to correct land cover classifications in locations with known errors of classification (Burkhalter, Herold, & Robinson, 2005; NOAA, 2015a,b). The 2006 C-CAP data was created by applying the existing classification methodology along with the CCA technique, which assisted in accounting for land cover changes between 2001 and 2006 (MDA, 2009). CCA identifies spectral signatures for each land cover class by comparing spectral values between the 2006 image and the 2001 base layer (Irani and Claggett, 2010). The C-CAP incorporates six wetland classes (See Table 2.3) unlike NLCD and the CBLCD, which use the Anderson-Level I classification system (Burkhalter *et al.*, 2005; Dobson *et al.*, 1995).

The Chesapeake Bay Land Cover Dataset (CBLCD). The CBLCD datasets of 1984, 1992, 2001, 2006, and 2011 were based on an integration of the C-CAP and NLCD (Irani & Claggett, 2010). For 2001, the C-CAP and NLCD were amalgamated to create a baseline map. The map was then updated for 2006, and retroactively for 1992 and 1984 using CCA and a CART (Irani & Claggett, 2010). An algorithm was also applied to the classification model to reduce over-classification of coastal emergent wetlands (MDA, 2009).

2.2.3 Ancillary Data for Comparison of Spatial and Temporal Agreement

In addition, the change patterns and trends from each regional land cover data set (*e.g.* C-CAP, CBLCD, and NLCD) were compared with NWI classifications and state land use and land cover datasets produced by reference data from aerial photography and other GIS layers. These data sets were chosen because they possessed a higher level of spatial resolution, accuracy percentages in wetland classification, and pre-existing wetland delineation and validation.

National Wetland Inventory (NWI). The U.S. FWS produces the NWI, the most comprehensive national wetland mapping initiative. The NWI produces wetland maps using moderate to fine resolution aerial photographs and satellite imagery along with a primarily non-automated photointerpretation mapping technique (See Table 2.2) (FGDC, 1998; FGDC, 2009). The NWI maps tend to have lower errors of commission than omission, meaning that pixels classified as wetlands on NWI maps have a high probability of being a wetland at the time that the aerial imagery was gathered (Nichols, 1994; Stolt & Baker, 1995). When incorporating the NWI's aerial imagery approach, forested wetlands tend to be more difficult to map than emergent wetlands, and have a wide variation in errors of omission (Kudray & Gale, 2000; Rolband, 1995; Tiner, 1990; Tiner, 2012). The majority of aerial photographs used to create NWI maps for the western portion of Delmarva Peninsula were between 10 and 30 years old (*e.g.,* gathered between the 197's and 2000's). Though NWI maps can be outdated or omit certain forested wetlands, and the NWI mapping process results in the most accurate and detailed dataset available for the US. NWI data have been used to support the monitoring and regulation of wetlands on national, state, and local scales. The archived 2002 NWI data set for the Chesapeake Bay Watershed was acquired from the Chesapeake Bay Program Office. The most recent NWI data were acquired by downloading seamless wetland data in the form of geodatabases on a state by state basis.

State Land Cover Datasets (State LCDs). The Maryland 2002 land use and land cover data sets were created by photographic interpretation of aerial photography and Landsat satellite imagery (See Table 2.2). The 2010 datasets were updated and validated by overlaying the 2002 base layer with high resolution (1m) 2007 National Agriculture Imagery Program (NAIP) aerial imagery and parcel information from the 2008 parcel level data from the 2008 edition of MDProperty View (MDP, 2010a,b; MDP, 2014). The Maryland land use/land cover datasets incorporated the Anderson Level-II land cover classification system, which has two categories of wetlands, woody or emergent.

The Delaware vector LCDs from 2002, 2007, and 2012 were produced using digital orthophotography, and consisted of one aggregated wetland class (See Table 2.2). The wetland layer was created by overlaying aerial photography onto the state NWI layer and implementing the Federal Geographic Data Committee's (FGDC) wetland classification and mapping standards (FGDC, 2009; Tiner *et al*., 2011). The State Wetlands Mapping Program (SWMP) data sets were updated in 2007 for Sussex, Kent, and New Castle Counties using color-infrared aerial photographs and supplemental airborne GPS data (Delaware Geospatial Data Exchange, 2014), in order to increase the accuracy of the land cover classification and decrease underestimates of wetlands due to foliage illustrated in imagery obtained during leaf-on seasons. Wetlands were also classified by using the following external data sets: 1992 color infrared imagery, NWI, the National Hydrography Dataset (NHD), the National Elevation Dataset (NED), and USGS Topographic maps (Delaware Geospatial Data Exchange, 2014; Tiner *et al.*, 2003).

Spatial Accuracy Assessment of Wetland Classifications. According to national scale accuracy assessments, the 1992 NLCD had 80% accuracy for Anderson-I classes, and 58% accurate for Anderson Level-I classes (See Table 2.4) (Dobson *et al.*, 1995; Homer *et al.*, 2007). For 2001, the NLCD was accurate for 85% for Anderson Level-I classes and approximately 79% accurate for Anderson Level-II classes (Dobson *et al.*, 1995; Homer *et al.*, 2007). No specific accuracy assessment was conducted for detecting wetlands in the 1992 or 2001 NLCD that fell in the study area. For C-CAP, accuracy assessments were completed on the 2001 C-CAP Zone 60, which includes parts of North Carolina, Virginia, Maryland, New Jersey, Delaware, Pennsylvania, and New York. The overall accuracy for land cover in Zone 60 is approximately 71% (Dobson *et al.*, 1995; Homer *et al.*, 2007). The spatial accuracy assessment of the 2010 C-CAP classifications in the Mid-Atlantic region found an average producer's accuracy (by wetland type) of approximately 82% for palustrine woody/shrub-scrub, 83% for palustrine emergent, 50% for estuarine shrub-scrub, and 100% for estuarine emergent. Percentages of user's accuracy by wetland type were all similar to producer's accuracy, except for estuarine emergent wetlands, which had a 10% reduction in accuracy (90%). It is important to note that the producer's and user's accuracy percentages for estuarine shrub-scrub wetlands may be skewed due to the assessment only including two sites for validation, while the other wetland classes had between 27 and 57 validation sites (NOAA, 2015a,b). The CBLCD is a product of two national LCDs (*e.g.*, C-CAP and NLCD) that have already undergone official accuracy assessments.

Existing accuracy assessments of NWI mapping have focused on comparison with field data (Kudray and Gale, 2000). For example NWI maps of wetlands in rural Massachusetts communities had a 95% accuracy of the quantity of wetlands correctly classified and delineated (Swartout, MacConnell, & Finn, 1982). The study incorporated extensive fieldwork to validate the delineation and classification of emergent, shrub/scrub palustrine, emergent and shrub/scrub estuarine wetlands. Other research has found that at least 91% of wetlands classified as palustrine wetlands on Virginia NWI maps met the field validation criteria as jurisdictional wetlands (Stolt & Baker, 1995). No other accuracy assessments of wetland classifications in regional land cover data sets have been conducted on wetland detections on the Delmarva Peninsula.

2.2.4 Methodology

The methodology consisted of five major steps: 1) pre-processing, 2) spatial matching of the regional and ancillary GIS data, 3) calculating the quantity, percentage and frequency of wetland change, 4) identifying hot spot locations of wetland change, and 5) analyzing the types of wetland conversion (*e.g.*, loss and gain) to explore potential natural and anthropogenic drivers of wetland change (See Figure 2.2).

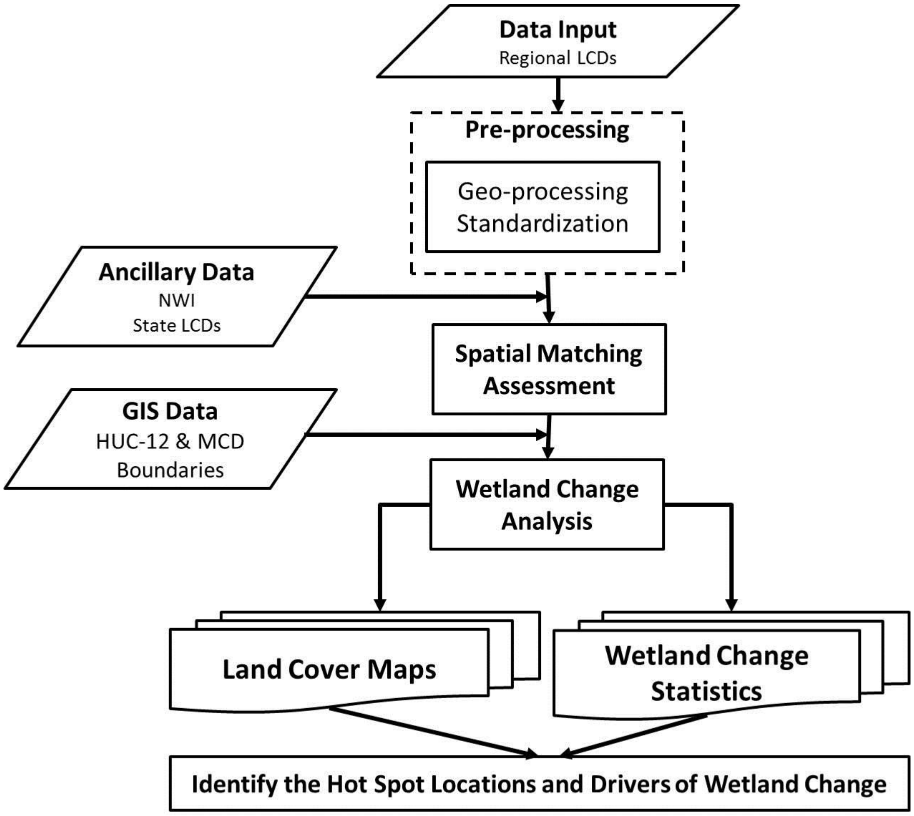

Figure 2.2 Flow Diagram of the Methodology

Each LCD and ancillary vector data set was converted into a 30-m resolution raster at the Albers conical equal area projection in ArcMap. A crosswalk was used to normalize the NLCD, CBLCD, and C-CAP data sets to two basic classifications: 1) wetlands versus non-wetlands and 2) forested versus emergent herbaceous wetlands. (See Table 2.3).

Datasets were aggregated to the following spatial scales: Hydrological Unit Code 12 (HUC-12; n = 246), Minor Civil Division (MCD; n = 135), county (n = 14), and state (n = 3). The HUC-12 scale was selected as the physical scale because it is the finest hydrological unit code that still preserves the boundaries of the respective surface drainage basins and the topographical gradients of adjacent units. The MCD scale was selected as the socioeconomic scale because of its medium resolution and its agreement with the political and Census boundaries most likely to stay constant during the period of study. Temporally, the study stretches from 1984, the year of the earliest LCD, to 2010, the year of the most recent LCD (NOAA, 2015a,b) (See Table 2.1).

Second, the LCDs were compared for spatial and temporal agreement, by spatial matching of data sets with corresponding years (*e.g.,* overlaying the 2001 NLCD on top of 2001 C-CAP). The same process was repeated using state level LCDs and the NWI. Percentages of agreement were calculated by dividing the count of the number pixels classified as wetlands (1) in LCD_1 also classified as wetlands (1) in LCD_0 by the total number of pixels classified as wetlands (1) in LCD_0. Third, the magnitude, frequency and hot spots of wetland change (including loss and gain) were identified by calculating the quantity (total area in hectares) and percentage (of total area of wetlands) of wetland change at each spatial scale (*e.g.,* HUC-12 and MCD), complementary time periods (*e.g.,* 2001 to 2006), and class (*e.g.,* Anderson Level-I and II). Finally, a time series analysis of wetland conversion was conducted to identify the potential drivers of wetland change.

In order to compare the agreement of the different land cover data sets, the locations and quantities of wetland loss and gain from each land cover dataset from 2001 to 2006 were compared, as this was the only time period that was constant across all three regional land datasets. The absolute quantity (Equation 1) and the relative quantity (Equation 2) of wetland changes were calculated.

$$(Wetland\ Area_1 - Wetland\ Area_0) \tag{1}$$

$$\left(\frac{Wetland\ Area_1 - Wetland\ Area_0}{Wetland\ Area_0}\right) \tag{2}$$

The percentage of wetland coverage (area) converted from wetland to another land cover type and vice-versa for each time period was calculated for each HUC-12 and MCD using the CBLCD and C-CAP LCDs. The difference of wetland changes for each time period was statistically tested using the Wilcoxon Signed-ranks test. The Wilcoxon Signed-ranks test is a non-parametric statistical test to assess the difference between two conditions where the samples, in this case wetland change, are correlated. The data sets can be compared repeatedly over consistent periods of time (*e.g.,* between $Year_0$ and $Year_1$), with a null hypothesis of the medians of the two samples being equal to 0.

Next, multiple data sources were aggregated to create a new, multi-source LCD time series to mirror the time series used in the CBLCD/-C-CAP analysis. As shown in Table 2.4, the state LCDs and an archived NWI data set for the Chesapeake Bay Watershed were matched to the closest CBLCD/C-CAP time period. For example, the 2007 DE LCD was placed under the 2006 time period. With the earliest CBLCD layer (1984) not having a woody wetland class, the 1992 C-CAP layer was burned into the

1984 layer along with the archived NWI data set, which was primarily based on aerial imagery collected between the mid 1970's to the early 1990's. Due to incorporating multiple sources with differing methodologies, the percentage of wetland coverage (area) converted from wetland to non-wetland cover types and vice-versa for each time period was calculated for each HUC-12 at a binary Anderson-I level (e.g., wetland versus non-wetland). For the remaining portion of the study, which focused on identifying hot spot locations and drivers of wetland change, only the CBLCD/C-CAP time series was used in order to provide results based on source data with a constant methodology,

Data Source	Primary Year					
	(I) 1984	(II) 1992	(III) 1996	(IV) 2001	(V) 2006	(VI) 2010
CBLCD	1984	1992	-	2001	2006	2011
C-CAP	1992	-	-	-	-	-
DE LCD	-	-	-	2002	2007	2012
MD LCD				2002		2010
NWI – CBW	X	-	-	-	-	-

Table 2.4 A breakdown of the multi-source time series by benchmark years and the respective years of the national and state LCDs along with the archived NWI data set.

After quantifying wetland change and testing for statistical significance, hotspots of wetland change (loss and gain) for each LCD were identified. Hotspots were identified by comparing the distribution of the percentages of wetland loss and gain at HUC-12 and MCD scales. Hot spots were visually defined as geographic units (*e.g.*, HUC-12, MCD, river system, city, transportation corridor, and park reserve) with the highest quantities or percentages of wetland change. The frequency of wetland change was analyzed by identifying pixels in the study area which experienced no change, permanent change, or temporary changes over the study period of 1984-2010. Permanent change was defined as a wetland conversion from wetland to non-wetland or vice versa that occurred in only one phase of the study period. Temporary change referred to wetlands that experienced multiple conversions (i.e., back and forth from a wetland to a different land cover.

The potential drivers of wetland change were identified by interpreting the conversion of pixel classifications from and to wetlands in acreage and percentage. The end goal was to see if the regional LCDs can be used to identify what land cover types wetlands were being converted into to help identify the drivers of wetland loss and the land cover types that were being converted into wetlands to help identify the drivers of wetland gain. For example, a HUC-12 that experience high percentages of wetland loss to water may be experiencing impacts related to sea level rise, while MCDs that experience increases in wetland acreage from agriculture could be benefiting from wetland restoration policies.

2.3 **Results**

The results section consists of a summary of the spatial agreement of the regional LCDS compared to the NWI, and state level land cover datasets at Anderson Level-I and Anderson Level-II classification schemes. Second, the quantities and percentages of wetland change and the frequency of wetland change are reported at the Anderson Level-I and Level-II classification schemes for the entire study area at HUC-12 and MCD spatial scales. Third, hotspots of wetland change are also identified at the same two spatial scales. Fourth, potential drivers of wetland change across the entire study area are discussed based on the C-CAP and CBLCD at the Anderson Level-I class scheme.

2.3.1 Spatial Agreement of the LCDs

With respect to the spatial agreement of wetland classifications between the regional LCDs between 2001 and 2006, two major findings stood out: (1) Regional LCDs were generally agreeable with average agreements of 97% in 2001 and 95% in 2006; and (2) CBLCD and C-CAP were even more similar at Anderson Level-I (See Table 2.5). Similarly, classification agreement at Anderson Level-II was highest (approximately 95% for woody and 93% for emergent wetland) between the 2006 C-CAP and CBLCD data sets. For woody wetlands, C-CAP had the highest overall percentage compatibility between all three LCDs, with an average of 97%. Overall, the NLCD had the lowest average agreement at 91% (see Table 2.5).

Year		2001		2002			2006		2007
LCD	C-CAP	CBLCD	NLCD	DE*	MD*	C-CAP	CBLCD	NLCD	DE*
C-CAP	-	100%	96%	69%	82%	-	97%	92%	69%
CBLCD	100%	-	95%	71%	81%	95%	-	95%	71%
NLCD	96%	96%	-	71%	82%	92%	97%	-	71%

Table 2.5 The spatial agreement (by percentage) between the regional land cover data sets and the state land cover data sets over 2001/2-2006/7 at Anderson Level-I (e.g. one general class of wetlands). Note - The regional LCDs were obtained one year before the state LCDs.

Woody (2006)				
LCD	C-CAP	CBLCD	NLCD	NWI
C-CAP	-	97%	91%	78%
CBLCD	98%	-	93%	77%
NLCD	93%	99%	-	77%
Emergent (2006)				
LCD	C-CAP	CBLCD	NLCD	NWI
C-CAP	-	93%	95%	76%
CBLCD	94%	-	94%	75%
NLCD	91%	89%	-	76%

Table 2.6 The spatial agreement (by percentage) between the 2006 regional land cover data sets and the NWI at the Anderson Level-II class scheme with two wetland classes

With respect to emergent wetlands, the C-CAP and CBLCD data sets had the highest percentage of total agreement, with an average of 93% (see Table 2.6). C-CAP had the highest overall percentage of compatibility between all three LCDs with an average of 95%. NLCD had the lowest compatibility with an average of 92%. With respect to the NWI data set, regional LCD woody wetlands and emergent wetlands were matched about 76%, though woody was slightly better matched. When testing the spatial agreement between the regional LCDs and the state level LCDs, regional data sets had a 71% agreement with the DE LCD and an 82% agreement with the MD LCD (see Table 2.5). When visually comparing the wetland coverage of the LCDs, there appeared to be two distinct regions of difference/disagreement. First, the DE LCDs appeared to show a higher concentration of wetlands in the uplands south of the Cedar Swamp Wildlife Area, which is located toward the central areas of the DE coastline. The second concentration was located north of the Nanticoke River, near the southwestern state border of DE and MD.

2.3.2 Wetland Change on the Delmarva Peninsula

Overall, the peninsula experienced a 2% (-9,255 ha) net wetland loss between 1984 and 2010. The overall trend of wetland change was consistent between C-CAP and CBLCD. According to CBLCD and C-CAP, the peninsula experienced a wetland loss of approximately 3% (-9,610 ha) between 1984 and 2001 with the largest surge in loss between 1992 and 2001 at 2%. The trend of loss decreased to 1% (-5,074 ha) between 2001 and 2006, and ended with a net gain of 0.2% (1,063 ha) between 2006 and 2010. This agrees with historical population change estimates from the Census Bureau. For example, between 1970 and 2010, the peninsula experienced the highest quantity (est. 176,000) and percentage (17%) of decadal growth between 1990 and 2000. The peninsula's population growth appeared to slow down to 14% from 2000 to 2010 (US Census Bureau, 2012). With respect to wetland gain at the C-CAP scale, the study area experienced a 0.5% (2,367 ha) gain of wetlands between 1992 and 2001, and an est. 0.1% (380 ha) gain between 2001 and 2006. According to CBLCD, the peninsula also experienced an est. 0.1% (303 ha) gain between 1984 and 1992, a 0.3% (1,148 ha) gain between 1992 and 2001, and an est. 0.05% (240 ha) gain between 2001 and 2006. The NLCD indicated a 0.6% (2,805) gain between 2001 and 2006 (see Table 2.7).

The trends of wetland change were consistent between all three data sets, but the quantities and conversions were different. For example, the CBLCD always showed the lowest quantities of wetland loss, almost 2,000 hectares less than C-CAP and NLCD (see Table 2.7 and Figure 2.3). NLCD showed the largest quantity and percentage of wetland gain between 2001 and 2006, at almost 2,500 hectares more than C-CAP and CBLCD. If NLCD would have had the same quantity of gain as C-CAP and CBLCD, approximately 300 hectares, its net wetland change would have almost doubled in quantity and percentage.

Wetland change at Various Spatial Scales. The spatial and temporal analyses of wetland change were conducted on two spatial scales: HUC-12 and MCD. The locations of wetland change at both scales agreed through each time period. However, the quantities and percentages of wetland change were higher at the HUC-12 scale. Wetland loss on the Delmarva Peninsula HUC-12 sub-watersheds from 1992 to 2010, according to C-CAP, is illustrated in Table 2.7. During this period, wetland cover decreased by 3.3%

(see Table 2.7). According to the spatial analysis of NOAA - C-CAP land cover data from 1992 and 2010, the peninsula experienced a net loss of 1.93% (9,078 ha) of wetland cover. The highest statistically significant ($p < 0.05$) percentage of loss occurred between 1996 and 2001, and the highest gain occurred between 2001 and 2010. Woody wetlands had the highest quantity of loss in all four periods.

The period by period quantity and percentage of wetland change in the study area from 1984 to 2010 according to C-CAP is illustrated in Table 2.8. During this period, wetland cover increased by 0.84% (1,685 ha) (see Figure 2.3, Figure 2.4 and Table 2.8). From 1992 and 2010, the peninsula experienced a net gain of 26,183 ha of wetland cover. All of the time periods had statistically significant ($p < .05$) percentage of change. The highest percentage of loss occurred between 1984 and 1992 (-4.9%, -24,498 ha). In contrast to C-CAP's one period of net gain, the multi-source data reflected two periods of gain: almost 5% (22,992 ha) from 1996 to 2001 and almost 6% (27,915 ha) from 2006 to 2010. The source data for $Year_1$ for both of these periods included state LCDS from Delaware and Maryland, suggesting that the results reflect an improvement due to finer resolution data rather than only natural and/or anthropogenic processes.

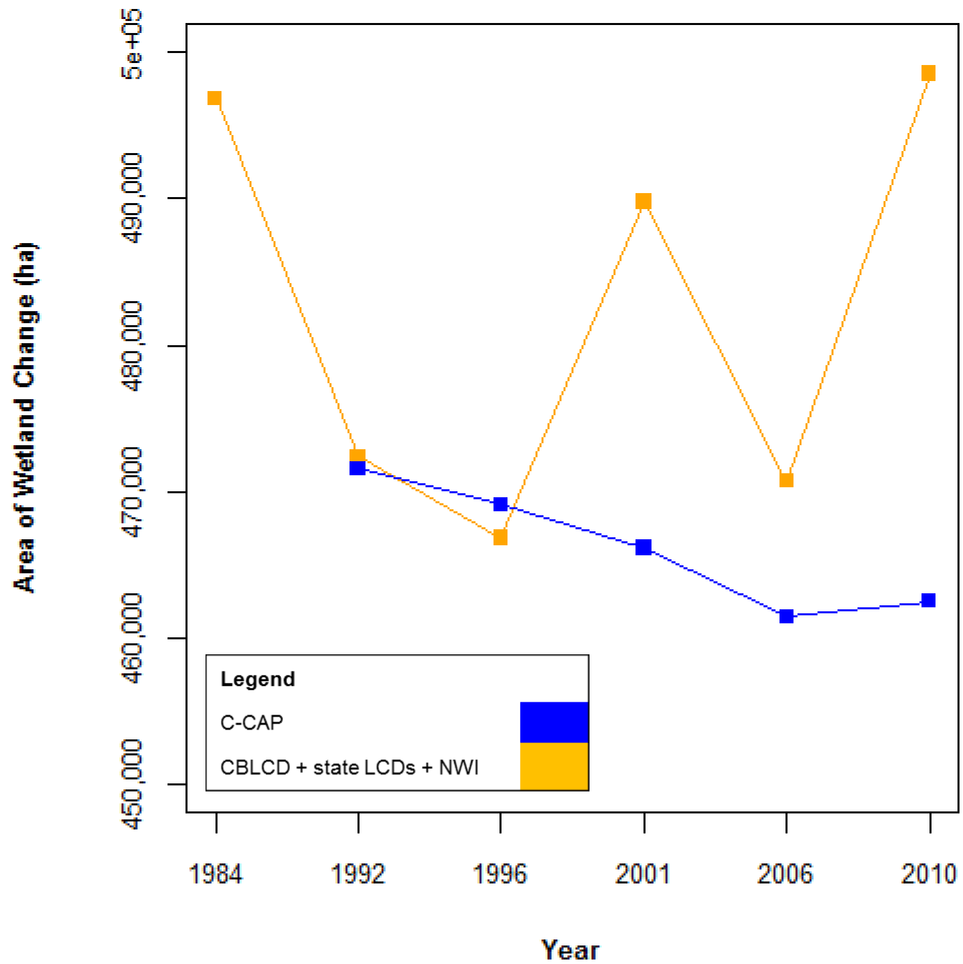

Figure 2.3 Temporal trends of the quantities of wetland change in the study area using the C-CAP, CBLCD, state LCDs, and archived NWI from 1984 to 2012.

Land Use	C-CAP		NLCD		CBLCD	
Total Wetland Area (ha) (2001)	466,112		435,384		429,529	
Total Wetland Area (ha) (2006)	461,418		432,528		425,981	
Net Change (ha)	-4,693		-2,856		-3,548	
Net Percent Change (%)	-1.01%		-0.66%		-0.83%	
Wetland Loss	(ha)	(%)	(ha)	(%)	(ha)	(%)
Development	640	12.6	690	12.2	262	7.3
Agriculture	2,447	48.2	1,059	18.7	2,212	61.5
Grass/Scrub	479	9.4	1,278	22.6	103	2.9
Forest	399	7.9	1,739	30.7	112	3.1
Water	749	14.8	240	4.2	539	15.0
Barren	358	7.1	654	11.5	371	10.3
Total Loss	5,074	100.0	5,660	100.0	3,599	100.0
Total Percentage Loss	-	1.10	-	1.30	-	0.80
Wetland Gain	(ha)	(%)	(ha)	(%)	(ha)	(%)
Development	12	3.1	0	0.0	0	0.0
Agriculture	26	6.9	1,129	40.3	25	10.5
Grass/Scrub	62	16.3	483	17.2	69	28.9
Forest	218	57.3	747	26.6	129	53.7
Water	34	9.0	24	0.9	14	5.8
Barren	28	7.3	421	15.0	3	1.1
Total Gain	380	100.0	2,805	100.0	240	100.0
Total Percentage Gain	-	0.10	-	0.60	-	0.10
Net Change	(ha)	(%)	(ha)	(%)	(ha)	(%)
Development	-628	-0.13	-690	-0.16	-262	-0.06
Agriculture	-2,421	-0.52	71	0.02	-2,187	-0.51
Forest	-417	-0.09	-795	-0.18	-33	-0.01
Grass/Scrub	-181	-0.04	-992	-0.23	17	0.00
Water	-715	-0.15	-216	-0.05	-525	-0.12
Barren	-331	-0.07	-232	-0.05	-368	-0.09
Total	-4,693	-	-2,856	-	-3,359	-

Table 2.7 A comparison of statistics of wetland change between 2001 and 2006 between the C-CAP, NLCD, and CBLCD regional land cover data sets.

Land Cover Change	1992 - 1996		1996 - 2001		2001 - 2006		2006 - 2010		1992 – 2010	
Total Wetland Area (Period I) (ha)		471,559		469,064		466,112		461,418		471,559
Total Wetland Area (Period II) (ha)		469,064		466,112		461,418		462,481		462,481
Net Change (ha)		-2,495		-2,952		-4,693		1,063		-9,078
Percent Change (%)		-0.53		-0.63		-1.01		0.23		-1.93
Wetland Loss	Quantity (ha)	Percent (%)	Quantity (ha)	Percent (%)	Quantity (ha)	Percent (%)	Quantity (ha)	Percent (%)	Quantity (ha)	Percent (%)
Development	68	1.9	130	2.3	640	12.6	171	12.6	1,009	6.5
Agriculture	347	9.9	2,671	47.4	2,447	48.2	354	26.1	5,820	37.4
Grass/Scrub	1,490	42.6	2,237	39.7	479	9.4	514	37.9	4,720	30.3
Forest	952	27.2	160	2.8	399	7.9	27	2.0	1,538	9.9
Water	579	16.5	190	3.4	749	14.8	143	10.6	1,662	10.7
Bare	60	1.7	245	4.3	358	7.1	146	10.8	809	5.2
Total	3,496	100	5,633	100	5,074	100.00	1,355	100.00	15,558	100
Wetland Gain	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
Development	49	4.9	1	0.1	12	3.1	75	3.1	137	2.1
Agriculture	361	36.1	215	8.0	26	6.9	975	40.3	1,577	24.3
Grass/Scrub	386	38.6	2,080	77.6	62	16.4	248	10.3	2,777	42.9
Forest	103	10.3	142	5.3	218	57.3	680	28.1	1,144	17.7
Water	0	0.03	180	6.7	34	9.0	242	10.0	457	7.1
Bare	101	10.1	62	2.3	28	7.3	198	8.2	389	6.0
Total	1,001	100	2,681	100	380	100	2,418	100	6,480	100
Net Change	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
Development	-19	-0.8	-129	-4.4	-628	-13.4	-96	-9.1	-872	9.6
Agriculture	14	0.6	-2,456	-83.2	-2,421	-51.6	621	58.4	-4,242	46.7
Grass/Scrub	-1,104	-44.2	-157	-5.3	-417	-8.9	-266	-25.0	-1,944	21.4
Forest	-849	-34.0	-17	-0.6	-181	-3.9	653	61.5	-394	4.3
Water	-579	-23.2	-10	-0.4	-715	-15.2	99	9.3	-1,206	13.3
Bare	41	1.6	-183	-6.2	-331	-7.0	52	4.9	-420	4.6
Total	-2,495	-	-2,952	-	-4,693	-	1,063	-	-9,078	-

Table 2.8 Wetland loss, gain and net wetland change by quantity in hectares and percentage between 1992 and 2010 using the Anderson Level-I classification system.

Wetland Change by Type (woody versus emergent). According to the results of the comparison of wetland change in woody and emergent wetlands, the majority of wetland change has been occurring in woody wetlands. As illustrated in Figure 2.4, woody wetlands experienced over 75% of the wetland loss and 85% of the wetland gain in each of the three C-CAP time periods from 1992 to 2006. However, there was a 13% increase in the percentage of wetland gain from emergent wetlands from 1992 to 2006.

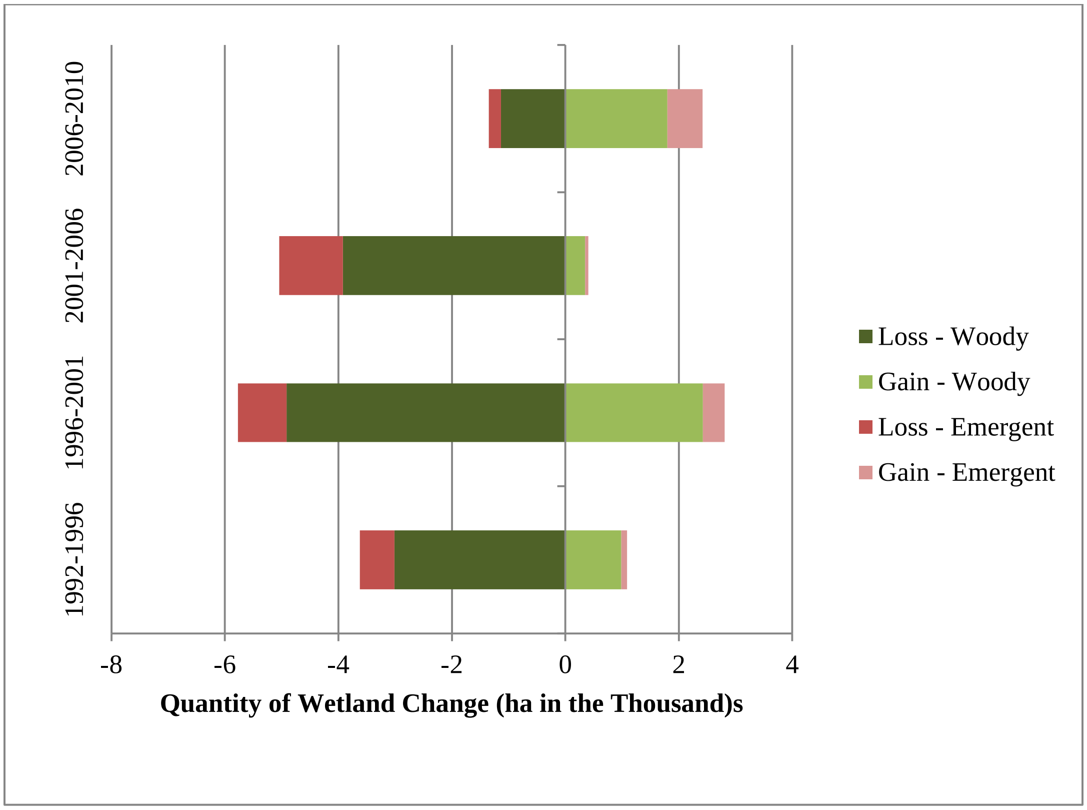

Figure 2.4 A bar graph of the quantities of wetland change in hectares (at the scale of the thousands) by wetland type at Anderson Level-II (e.g., wood versus emergent) from 1992 to 2010 according to the C-CAP land cover data sets.

2.3.3 Frequency of Wetland Change

Table 2.9 illustrates the breakdown of the amount of hectares in the entire study area that either experienced: a) only one phase of land type conversion from wetland to non-wetland or vice versa (*e.g.,* permanent change), b) multiple conversions (*e.g.*, temporary change) , or c) no change. Permanent wetland loss accounted for a little over 2.5% (12,576 ha) of the area of wetland change according C-CAP and 2% (9,091 ha) of wetland change in CBLCD. Most of these wetlands were classified as emergent wetlands or were adjacent to emergent wetlands, open water, or urbanized areas, especially areas in the largest patches of wetland coverage. Only 0.01% (48 ha) of wetland areas in C-CAP and 0.01% (33 ha) in CBLCD experienced temporary loss. Wetland areas that experienced temporary/fluctuating change (loss or gain) were either poorly classified wetlands or were located on or adjacent to areas of cultivation (*e.g.,* cultivated land cover), properties ripe for development, or coastal conditions vulnerable to tidal variation, inundation, and sea level rise. With respect to wetland gain, roughly 0.5% (199 ha) of the wetland coverage in C-CAP and 0.3% (119 ha) in CBLCD experienced permanent wetland gain. Further investigation is needed to determine whether sites of wetland gain were the result of specific wetland restoration and conservation projects.

LCD	C-CAP		CBLCD	
Type of Change	Quantity (ha)	Percent (%)	Quantity (ha)	Percent (%)
Permanent Loss	12,576	2.66	9,091	2.09
Permanent Gain	2486	0.53	1,237	0.28
Temporary Loss	48	0.01	33	0.01
Temporary Gain	199	0.04	119	0.03
No Change	456,662	96.8	424,771	97.6
Total	471,971	100	435,251	100

Table 2.9 A comparison of the frequency of wetland change according to C-CAP and the CBLCD. Permanent loss represents pixels that changed in classification from wetland to a non-wetland land cover, and remained a non-wetland cover for the remaining portion of the study period, and permanent gain vice versa. Temporary loss represents pixels that changed in classification from wetland to non-wetland land cover, but did not maintain the wetland classification for the remaining portion of the study period, and temporary gain vice versa.

2.3.4 Hot Spots of Wetland Change

The C-CAP and CBLCD had the best agreement in identifying hotspots, due to CBLCD incorporating C-CAP into its methodology. Although the NLCD map of wetland loss from 2001 to 2006 agreed with the CBLCD and C-CAP maps at both scales, it still overestimates the percentages of wetland loss. The two spatial scales of the wetland change analysis also had similar hotspots (*e.g.*, locations) of wetland change (from 6% to 75%) occurring in the following regions (labeled in Figure 2.5). Region A is located on the northern edges of the peninsula. Region B consists of urban corridors and riverine wetland systems towards the center of the peninsula stretching from the southern portion of Delaware to the Maryland, Virginia border. Region C focuses national wildlife refuges on the Delaware coastline. Region D encompasses barrier islands and in bay water bodies adjacent to the Atlantic Ocean stretching from Rehobeth Bay to south to Assateague Island. Region E includes the southeastern, Atlantic coast of the peninsula. And, Region F stretches from the Blackwater National Wildlife Refuge to the Wicomico River.

At the HUC-12 scale, barrier islands on the southern end of the peninsula experienced the highest percentages and quantities of wetland loss. The Parramore and Metompkin Islands (Region E) each lost a little more than 70% of their wetlands, according to C-CAP and CBLCD. They lost 92% and 82%, respectively, in NLCD (refer to Figure 2.5). The second tier areas were Assateague Island along the Atlantic Ocean and Breakwater Harbor in Delaware Bay (Region C), which lost 19% in C-CAP, 10% in CBLCD and 7% in NLCD. Third tier locations, included coastal areas that lost 3-8% of their wetland coverage and were concentrated on the Lower Eastern Shore (Region D) and Isle of Wight Bay (Region D) that is adjacent to the barrier islands of Ocean City, MD. The NLCD had substantially higher quantities of loss, but still had areas with large quantities of wetland cover, such as Isle of Wight (Region D) and Blackwater (Region F) on the central/western coast of the peninsula. NLCD had only 4 of the top 10 HUC-12s that were in CBLCD and C-CAP.

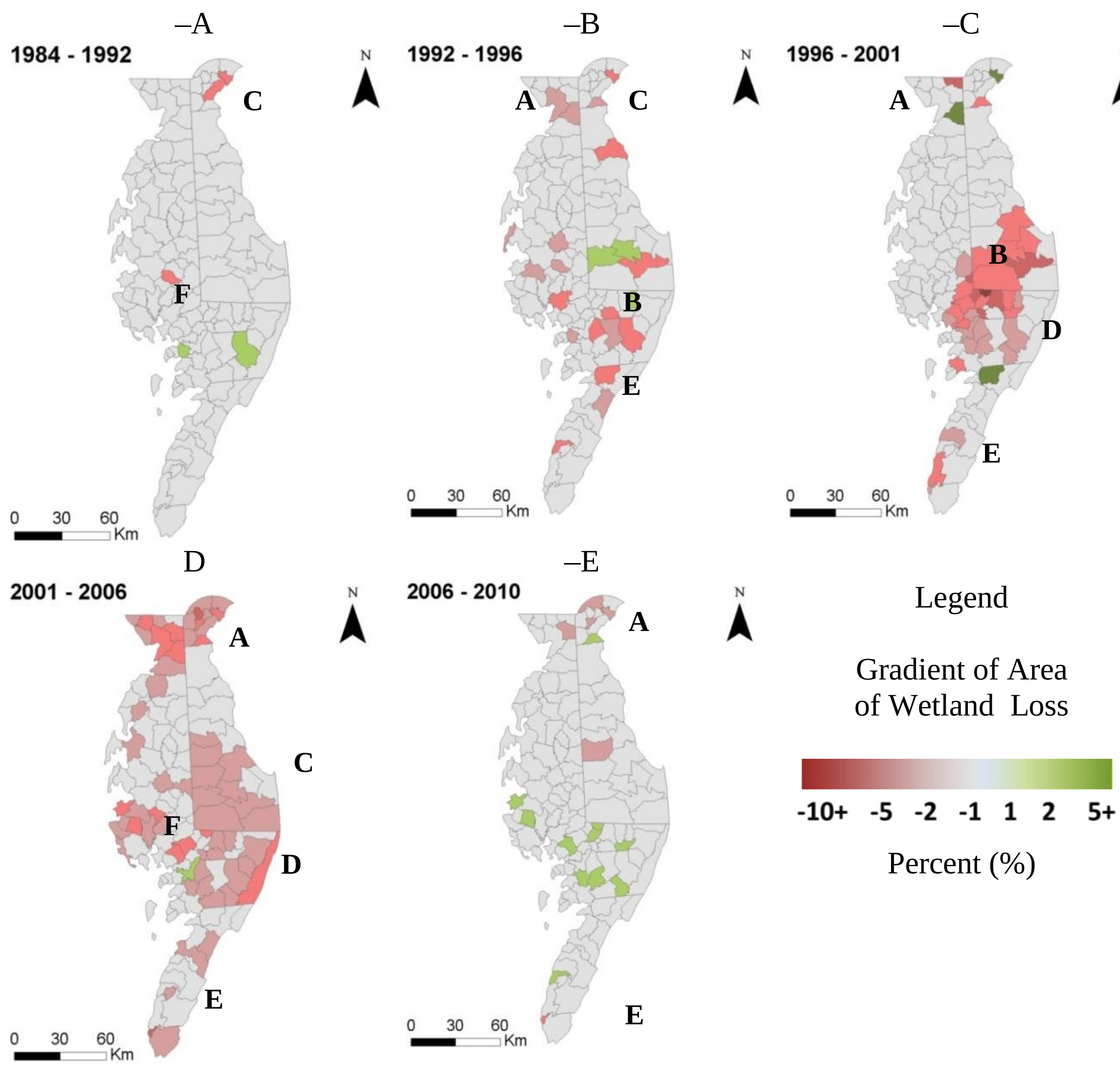

Figure 2.5 Maps illustrating wetland loss by percentage at the HUC -12 scale from 1984 to 2010: (A) CBLCD from 1984 to 1992 and (B thru D) C-CAP from 1992 to 2010.

At the MCD scale, C-CAP and CBLCD complemented one another, as the top 5 MCDs with the highest percentages of wetland loss, ranging from 7% to 12% were either located in the northeastern portion of the peninsula in urban corridors like Elkton, MD and Wilmington, DE (East Region A), its adjacent suburbs like Pike Creek, DE (Central Region A), in urban and preservation areas along U.S. Routes 13 and 113 in DE (Region B), or barrier islands like Ocean City, MD (Region D). Overall, the top 20 MCDs are interchangeable between the C-CAP and CBLCD, and included developed areas towards the center and western DE. NLCD has 4 of the top 10 from C-CAP and CBLCD, but has two to three times higher percentages of loss than C-CAP and CBLCD. Second tier MCDs had experienced an average 3% loss and were either located slightly northwest of the Blackwater National Wildlife Reserve, adjacent to the Nanticoke River, or on the Chesapeake and Delaware Canal (see Figure 2.5).

2.3.5 Indicators of Wetland Change

The non-wetland land cover classes in the national and state LCDs represent indicators of wetland change that can be linked to physical and/or anthropogenic drivers of wetland change. For example, the change in classification of a pixel from wetland to water can be used to track the spatial and temporal trends of wetland loss. However, the change in classification to water did not the drive the change. Potential drivers of the wetland loss to water include sea level rise, coastal erosion, and channelization.

Overall, C-CAP and CBLCD agreed regarding the ratios of the indicators of wetland change and the spatial hotspots of wetland change. In both data sets, agriculture accounted for 50% of wetland loss followed by water at 15%. The greatest disagreement between the two data sets was (net) wetland loss attributed to forest cover, with C-CAP loosing 384 more hectares of wetlands than CBLCD. Water accounted for almost two-thirds of wetland loss, whereas forest and grass/scrub cover accounted for almost three-fourths of wetland gain. When comparing temporal patterns of wetland change, the data sets showed agreement in the percentage of pixels that did not change in classification from 1992 to 2006 (est. 98%).

The NLCD did not agree as well with the other two data sets in wetland accuracy and wetland change. When looking at wetland change from 2001 to 2006, NLCD had approximately 50% (est. 1,300 ha) less wetland loss attributed to agriculture and almost 3 times (est. 1300 ha) more loss attributed to forest cover than C-CAP and CBLCD (See Table 2.6). With respect to wetland gain, NLCD had an average of 31% gain attributed to agriculture and an average of 30% less gain attributed to forest cover in C-CAP and CBLCD.

When examining the temporal trends and indicators of wetland change using the C-CAP LCD from 1992 to 1996, the top three land cover classes associated with net wetland change were grass/scrub (-43%, -1,490 ha), forest (-27%, -952 ha), and water (-17%, -579 ha). Agriculture accounted for almost -83% (-2,456 ha) of the net wetland change from 1996 to 2001 and -52% (-2,421 ha) from 2001 to 2006. From 2006 to 2010, the two major land covers associated with net wetland loss were agriculture at -25% (-266 ha) and development at -9% (-96), and the two major land covers associated with net wetland gain were grass/scrub (61%, 653 ha) and agriculture (58%, 621 ha).

According to the CBLCD, the top three land covers associated with wetland loss from 2001 to 2006 were agriculture/cultivation (61%, 2,212 ha), open water (15%, 539 ha), and barren land (10%, 371 ha). It is important to note that barren land cover is often considered a land cover that represents a “transition” or “clearing” of a natural land cover to an anthropogenic land cover, which can indirectly increase wetland loss driven by development. Unlike C-CAP and the CBLCD, NLCD showed that the highest quantity of wetland loss was attributable to forest cover (31%, 1,739 ha) followed by grass/shrub cover (23%, 1,278 ha), and agriculture (19%, 1,059 ha).

According to the C-CAP 2001-2006 data, the top three land covers associated with wetland gain were forest cover (57%, 218 ha), grass/shrub (16%, 62 ha), and open water (9%, 34 ha) (see Table 2.7). According to the CBLCD, the top three land covers associated with wetland gain were agriculture/cultivation (61%), open water (15%), and barren land (10%). Out of the three national LCDs, the NLCD indicated the largest total area (2,805 ha) of wetland gain. Its primary indicators of wetland loss differed slightly from those of C-CAP and the CBLCD, with agriculture accounting for 40% (1,129 ha), forest cover 27% (747 ha), and grass/shrub 17% (483 ha) of wetland gain.

2.4 **Discussion**

The discussion section will begin with a comparison of the results of wetland change according to the regional LCDs. Second, the discussion will shift to compare and contrast the quantities and hot spots of wetland change at the two spatial scales: HUC-12 and MCD. Third, the limitations and challenges of the study will be examined. Finally, the discussion will end with discourse on how time series analyses of wetland change can be applied to future research and policies as well as to practices that can strengthen the accuracy and applicability of future wetland change assessments.

2.4.1 **Wetland Change Agreement by regional Land Cover Dataset**

Overall, C-CAP and CBLCD had the highest statistical agreement in percentages and quantities of wetland change due to LCDs having similar methods, and the CBLCD incorporating C-CAP as source data. As previously stated, the most wetland loss occurred between 1992 and 2001 compared with 2001 and 2010. The NLCD did not agree as well with the other two data sets in wetland accuracy and wetland change. The NLCD's minimum mapping unit aggregation methods contributed to the smoothing of pixels with similar land cover classifications (Homer *et al.*, 2007). These methods resulted in the removal of isolated or small clusters of wetland pixels that were adjacent to or surrounded by large patches of non-wetlands, as well as isolated or small clusters of non-wetland pixels that were adjacent to or surrounded by large patches of wetlands. The classification of impervious surfaces (*e.g.*, roads) was also a major reason for classification differences. The NLCD used high- resolution ancillary data to increase the accuracy of mapping roads as developed areas (including rural/low-density areas), whereas in CBLCD and C-CAP collector and local roads often appear fragmented, and can be difficult to decipher. As a result, developed pixels from roads passing through forests/tree-cover and wetlands are often vulnerable to errors of omission and commission.

2.4.2 **Wetland Change Agreement by Spatial Scale**

When comparing the LCDs, the spatial scales and methodologies impacted the spatial agreement of wetland change and hotspots. C-CAP and CBLCD had the strongest agreement, because C-CAP focused on mapping land cover in coastal regions around the United States, which includes sub-watersheds in the Chesapeake Bay. On the other hand, NLCD was produced as part a national “wall to wall” mapping initiative to illustrate land-cover dynamics of the conterminous United State. Subsequently, different data acquisition, geo-processing, and classification methods were instituted, causing NLCD to disagree with the other two data sets. <Elaborate on limitations of NLCD.

2.4.3 **Limitations and Challenges**

The primary limitations to these regional LCD sets include, but are not limited to, seasonal variations of source imagery, the moderate spatial scales of the source imagery and LCD products, the coarse temporal distribution of the LCD products, and contrasts in methodologies. Seasonal variation of data collection directly influences the accuracy of the land cover classification. For example, data sets based on imagery that is collected strictly during leaf-on seasons are more likely inaccurately classify wetlands with deciduous plant canopy as well as other land cover classes that may be obscured by a leaf-on plant canopy (*e.g.*,, wetlands, impervious surfaces, waterways).

This moderate scale study is based on GIS layers created from Landsat imagery, which is considered by the scientific community to be medium resolution imagery at 30m. The imagery does not drilldown to a local, site-specific level that allows for consideration of landscape conditions that can vary at high spatial resolutions (*e.g.*, resolutions greater than 10m) (*e.g.*, topography, soil types, and hydrology). The medium resolution imagery also increases the potential for mixing of the spectral signatures of wetlands with those of hydric soils, shrub/scrub vegetation, and waterbodies containing sediments. Any localized wetland change captured in aerial photography has the potential to be immediately lost or diluted when the raster of the aerial photography has to be converted into a raster that is at the same scale of the regional LCD (*e.g.,* 30-meter resolution). It is also important to note that the 1 – 2% loss of wetland coverage represents the net change for the entire study area. Examining the results on HUC-12 and MCD scales on a period to period interval revealed a variation in percentage change from 36% loss to 17% gain. These percentages are also influenced by the baseline area of wetland cover in Period0. For example, a highly developed, upland HUC-12 or MCD near Wilmington, DE may have a lower quantity of wetland cover than an undisturbed, estuarine HUC-12 or MCD near Ocean City, MD. Subsequently, an equal area of wetland loss will result in different percentages of wetland loss (*e.g.,* the urban area having a higher percentage of loss than the estuarine area).

Temporal limitations include land cover datasets that are normally distributed on a 4 to 8 year basis. Although regional natural processes can occur on temporal scales ranging from instantaneous to centurial scales, episodic events can alter land use and land cover on a substantially shorter temporal scale (*e.g.,* months and days). Examples include but are not limited to hurricanes, flooding events, and construction/developmental projects. Natural hazards like hurricanes can also have long term impacts on the presence and health of wetland ecosystems. For example, hurricanes with high sustained wind speeds and strong storm surges can push saltwater upstream to freshwater wetlands with vegetation that cannot stand long term exposure to saline conditions. Also infrastructure, development projects like dams and canals often permanently interrupt the natural process of transportation valuable nutrient sediments downstream to coastal wetlands.

Both of these examples result in vegetation loss and wetland inundation that are identified by remote sensing instruments and mapping as wetland conversion from wetland to open water.

Contrasts in methodologies also limit the ability of the data sets to be temporally and spatially interchangeable. Not only does each of the land cover data sets have a different methodology, but there are also variations in methodologies within each land cover series. This may influence the ability to identify statistically significant trends and patterns of land cover change. For example, the CBLCD incorporated an algorithm to improve the classification of forested wetlands. Although it helps in accounting for forest wetland cover, the algorithm could decrease the quantity and percentage of pixels classified as emergent, which could impact the quantity and percentage of emergent wetland change. With respect to wetland conversion to and from developed land cover classes, the use of smoothing technique on the NLCD, along with data sets with conflicting scales, helps reduce spotty coverage of developed land cover classes and improve the connectivity of like pixels. However, the technique can decrease the calculated quantities of wetland cover lost to urban cover and vice versa.

Analyzing wetland change on one study area at two different spatial scales or boundary systems (*e.g.*, HUC-12 for physical and MCD for socioeconomic) also magnified how much visual ability to interpret wetland change detection depends on the lenses chosen by the user. The analysis must also take into account the spatial dimension of natural processes (*e.g.,* fluvial geomorphology). For example, hotspots of wetland change that appeared at the HUC-12 scale were diluted and harder to identify at the MCD scale. Often, only hotspots in areas with higher populations, development, or exposure to open water appeared at both HUC-12 and MCD scales.

2.4.4 Application of Wetland Change Analyses and Best management practices for future wetland assessments

A combination of techniques that integrate the use of remote sensing imagery with various landscape indicators are being used to map wetlands and track trends and patterns of wetland change (Tiner, 2004; Weller *et al.*, 2007). Overall, these regional LCDs can be utilized as a guide for wetland managers and planners to create wetland change or vulnerability indicators applicable to nationally recommended water quality criteria and wetland permitting regulations. However, a clear gap in knowledge exists between the geo-statistical relationship between the existing wetland permitting programs and changes in wetland cover identified by national land cover datasets (Brody *et al.*, 2007; Brody *et al*., 2008). For example, wetland permitting on the Delmarva Peninsula is a complex process that needs monitoring on regional, state, and local levels in order quantify the positive and negative impacts of permitted activities. The majority of studies on wetland change and permitting in the United States have focused on wetland permitting and the compensatory mitigation component of Section 404. In order to analyze the impacts of the potential drivers of permitting and wetland change, further research is needed on the relationships between the geographic distribution of permits, locations and trends of wetland loss, and the drivers of wetland loss on a regional level (Brody *et al.*, 2008; Cole and Shafer, 2002; Robb, 2002; Sifneos, Cake, and Kentula, 1992).

To enhance the utility of regional LCDs to natural resource managers, a few best management practices should be incorporated into the creation, quality control, and analysis of the LCDs. First, wetland classification accuracy will be increased if the source imagery (*e.g.,* aerial photography) is acquired more frequently during leaf-off season and on cloud-free days. This will decrease errors of omission and commission routed from wetlands covered by tree canopy, vegetation present in leaf-on season, and cloud cover as well as abnormal wetland spectral signatures during climate events like droughts. Also, an increase in the examination of time series data sets will increase the ability of researchers to quantify long-term trends in wetland hydrology (*e.g.,* inundation) and the impacts of climate and land use change on wetland ecosystem health, connectivity, and vulnerability. Second, site verification/validation should be incorporated in methodologies to validate the accuracy of wetland classifications, especially in areas where wetland change is not permanent. An improvement in multi-date classification methods and ancillary data will improve the accuracy of the frequency of change in classification by accounting for seasonal differences in spectral responses to climate conditions, changes in land uses (*e.g.,* ditching), isolated topographical conditions (Tiner *et al.*, 2003), and plant phenology (*e.g.,* transitions from dominant upland to lowland species or woody to emergent species).

Fieldwork or analyses of high-resolution imagery should also be conducted to verify wetland edges and boundaries with other land covers (*e.g.,* forest cover, riparian buffers, and water with subaquatic vegetation). Scientists have stressed the importance of ground work to validate inferences on physical and biogeochemical processes made by remote sensing imagery classification and analysis (Cahoon and Turner, 1989; Day *et al.*, 2008; Tiner, 2005; Turner and Rao, 1990). Third, time series analyses of wetland change should weight each regional LCD, based on the strength and specificity of each data set's source and ancillary data. Although the C-CAP, CBLCD, and NLCD spatially agree in identifying the site of wetland change, each excels in identifying different driver(s) and trends. For example, C-CAP has a stronger base with undeveloped land covers, whereas CBLCD and NLCD are frequently cited for their in-depth methodologies and ancillary data for classifying developed land covers. Fourth, researchers should increase the quantity and frequency of accuracy assessments for wetland classes in regional LCDs. An integration of site validation, wetland delineation, and high resolution imagery like aerial orthophotography would improve the accuracy of wetland maps and the ability to interpret time series analyses of wetland change. It would also allow GIS analysts to create regional wetland maps that cross different landscape, topographical, geological, climate and geopolitical conditions.

2.5 **Conclusion**

First, this study examined the spatial agreement of three national land cover data sets, the quantity and percentages of wetland change, and the drivers of the wetland change for the case of the Delmarva Peninsula in the Chesapeake and Delaware Bay watersheds. The C-CAP and CBLD have the highest overall spatial agreement at 97%. The 2001 regional LCDs had an average of 70% agreement with wetland coverage in the 2002/2007 DE LCDs, an 80% agreement with the 2002 MD LCD, and a 76% agreement with NWI in 2006. Depending on the data sources, likely dominant drivers of wetland loss were agriculture/cultivation followed by water (C-CAP/CBLCD).or forest and grass/shrub (NLCD). Similarly, forest cover and grass/shrub cover dominated wetland gain for C-CAP and CBLCD, whereas agriculture and forest cover dominated gain for NLCD. The differences in the potential drivers of wetland change were linked to methodological differences between the NLCD databases as well as the integration of C-CAP LCDs into the CBLCDs. C-CAP was chosen as the best regional LCD for time series analyses because of its temporal swath of coverage, its consistency in methodology, and its classification accuracy with respect to natural land coverage. Over the period 1984-2010, the period with highest percentages of wetland loss was between 1992 and 2001, and a decrease in wetland loss occurred from 2001 to 2010.

The surge in wetland loss is likely attributable to population growth and residential development and agriculture, while wetland gain could be linked to wetland restoration strategies, natural processes, and improvements in remote sensing and wetland classification. The primary hotspots of wetland loss occurred in wetland ecosystems in Delaware in or adjacent to preservation areas, along riverine systems like the Nanticoke River, bay systems that feed into the Atlantic Ocean, anthropogenic waterways like the Chesapeake Delaware Canal, previously degraded wetlands near agricultural land cover, and urban corridors like Ocean City, MD and suburban corridors in the center of peninsula. The high levels of wetland loss occurred along the Atlantic coast, which could be attributable to the consequences of sea level variability.

The applicability and efficiency of these data sets would be magnified with improvements, such as the prioritization of continuous usage of leaf off, color infrared seasonal data as the primary source of imagery to reduce the misclassification of wetlands as forest cover, acquisition of the highest resolution and lowest minimum mapping unit, and site verification/validation to validate wetland edges and boundaries with other land covers. In the end, these regional data sets can serve as geospatial guides to locate hotspots of wetland change and identify trends of change and critical areas vulnerable to over-development, sea level rise, excessive wetland permitting, and unsustainable stream and wetland restoration.

3 CHAPTER 3: A SPATIAL-TEMPORAL ANALYSIS OF SECTION 404 PERMITTING ON THE DELMARVA PENINSULA: TRACKING THIRTY YEARS OF WETLAND CHANGE AND MANAGEMENT

ABSTRACT

Since the 1780's, an average of almost 52% of the original wetlands in the Chesapeake Bay states have been degraded or removed, mainly for anthropogenic activities like dredging for shipping, ditching for agriculture, and extraction for urban development (Dahl, 1990). In order to quantify the impacts of wetland management and land use policies, managers and policymakers need a clearer understanding of the spatial and temporal trends and patterns of wetland change. This study was conducted to identify spatial and temporal trends and patterns of wetland change and Section 404 permitting on the Delmarva Peninsula between 1980 and to 2010. Results show that estuarine wetland loss was concentrated along the Atlantic Coast, as well as on the southwestern and northeastern corners of the Peninsula. Palustrine wetland loss was concentrated on the fringe of metropolitan areas, and extended into suburban areas. Temporally, wetland loss and permitting increased until 2006, except in VA where permitting continued to increase as a response to potential natural drivers of wetland loss like sea level rise. The majority of permits were distributed in MD (91%) for urban development in the tourist center of Worcester County. Permits were mainly distributed outside of urban areas in suburbia until the early 1990's, and did not begin targeting the suburbs again until 2004. The Corps mostly issued nationwide permits (56%) followed by state program permits (19%). Even though the majority of permits were issued for estuarine wetland projects (76%), the majority of wetland loss occurred in palustrine emergent and forested wetlands

(78%). This finding highlighted the issue that permitted activities were not only potentially driving wetland change, but were also being issued as a response to hazards like shoreline erosion. The results supported the hypothesis that wetland permitting and loss had statistically significant correlations ($p < .05$) with agriculture, forest cover, residential development, impervious roads, and landscape conditions.

3.1 **Introduction**

Wetlands support ecosystem health, water quality, and flood mitigation across the United States. Ecosystem services provided by wetlands include balancing of the water table by storing precipitation and controlling the movement of water through a watershed, erosion control, filtering and recycling nutrients from surface water runoff, and preventing stormwater from flooding developed areas (Brooks, Wardrop, and Bishop, 2004; Costanza *et al.,* 1997; Ingram and Foster 2008; Moglen *et al.,* 2011; Nichols and Strobel, 1991; Phillips *et al.,* 2007; Weller *et al.*, 2007). All of these services are vulnerable to diminishing returns when wetlands are lost, degraded, or impaired (*e.g.*, altering of landscape position and nutrient absorption due to inundation). Due in part to their substantial value to society, scientists and policymakers need a better understanding of the long term impacts of anthropogenic development and climate change on wetland conversion. Even with federal and state policies (*e.g.,* Section 404 of the Clean Water Act) to decrease wetland disturbance and loss, not much is known about how these policies impact wetland change on regional spatial and decadal temporal scales (Brody *et al.*, 2008). The primary goals of the Clean Water Act (CWA) are: 1) to reduce the release of pollutants into waters of the U.S., and 2) to regulate surface water quality. Section 404 of the CWA outlines the permitting program for activities that could result in the release of dredged and fill materials into waters of the U.S., which includes wetlands. Unlike most existing research on wetland permitting that focuses on federal permits, this research aggregates state with federal permitting records to identify spatial and temporal trends, patterns, indicators, and impacts of wetland permitting on the tristate Delmarva Peninsula between 1980 and 2010.

3.1.1 Wetland Change in the Chesapeake Bay Watershed

Over time, the health and functions of wetlands in the Chesapeake Bay and Delaware Bay watersheds have been stressed, degraded, or permanently altered due to anthropogenic and natural causes. Scientists and policymakers agree that there is a clear linkage between human-induced land uses, land cover change and wetland loss in the Chesapeake Bay region (Tiner 1995; Weller *et al.*, 2007). The primary stressors in the Delmarva Peninsula are agriculture, silviculture, urbanization, and transportation. Numerous studies support the concept that land cover/land use change directly affects wetland loss in the Chesapeake Bay as well as on the Delmarva Peninsula (Brooks *et al*., 2004; Daniels & Cumming, 2008; Kearney & Rogers, 2010; Mayer & Lopez, 2011; Weller *et al*., 2007).

For centuries, wetlands on the Delmarva Peninsula have been drained, filled, and dredged without sufficient governmental regulations to overcome wetland loss due to the increasing demands for agriculture, residential development, recreation and tourism, shipping, and transportation (Hartmann & Goldstein, 1994; Phillips *et al*., 2007). Coastal development has magnified the negative impacts of natural hazards like storm events and sea level rise on wetland habitats, agricultural soils, and coastal cities. Intense agriculture along with rising household density in coastal areas has increased the demand for transportation infrastructure (*e.g.*, roadways and railroads). Subsequent increases in impervious surfaces have required the alteration and removal of wetland, which has weakened the ability of wetlands to absorb surface-water runoff, resulting in an increase in vulnerability to flooding. Continuing to intensify agriculture activities like the poultry industry without maintaining a healthy, connected wetland ecosystem to compensate for the effects of nutrient runoff will likely lead to decreases in water quality.

3.1.2 Wetland Management Policies

While federal policies like Swampbuster, a provision of the Food Security Act of 1985 that discourages farmers from converting wetlands to croplands, have succeeded in reducing the conversion of wetlands to agriculture, wetlands continue to be converted into developed land covers through urbanization. Subsequently, federal, state, and local agencies collaborated to design and implement policies like Section 404 of the Clean Water Act (CWA) to protect, preserve, and restore wetlands nationwide, especially in regions experiencing heavy residential and commercial development like the Gulf Coast and the Chesapeake Bay Watershed (Taylor, 2014). The CWA authorizes the US Army Corps of Engineers as well as state and local agencies to design, manage, review, administer and enforce permits for development. In 1990, an Executive Order introduced a national goal of “No Net Wetland Loss” in order to help states and counties control the alteration, fragmentation, and degradation of their wetland habitats on national, regional, and local scales (Connolly *et al.*, 2005; Copeland, 2010). Temporally and spatially intact databases of national and state wetland permitting records as well as restoration programs exist that can be monitored, processed, and analyzed. However, there is still a limited understanding of how these policies have impacted wetlands on a regional spatial scale and decadal temporal scale (Brody *et al*., 2008; Peyre *et al*., 2001).

There is a demand for research that crosses political and physical boundaries, and that uses time series data to identify trends, patterns, and drivers of wetland change, because trends, patters, drivers, and impacts of wetland change are not always short term, and site specific. Geographic assessments of wetland change and permitting assessments on multiple spatial scales (*e.g*., national, regional, state, and (sub)watershed) are important because they aid policymakers in identifying: 1) locations that need to be zoned as critical areas or banking areas in order to prevent wetland loss or degradation, 2) wetland ecosystems that have experienced so much loss that total destruction is eminent, even if policies or regulations will be implemented, and 3) locations that are suitable for compensatory mitigation (*e.g.*, highway cloverleaf interchanges). A clearer picture of the cumulative impacts of wetland change drivers will help decision makers identify the most vulnerable wetland areas. Policymakers also need interdisciplinary mapping techniques that examine physical and socioeconomic dimensions of wetland change in order to determine the benefits of compensatory measures, restoration efforts, and best management practices (BMPs).

3.1.3 Review of the dimensions of Section 404 Wetland Permitting of the CWA

Wetland Permitting Data Sets. The goals of federal and state wetland permitting programs and regulations are generally: 1) to maintain a no net loss of wetland coverage nationwide and in each state, 2) to require mitigation of degraded or lost wetlands, and 3) to minimize the impacts of permitted activities on wetland health and ecosystem services (Connolly *et al*., 2005; Copeland 2010). This section explains the types, purposes, duration and jurisdiction of each permit under Section 404.

Federal – General and Individual Permits. The goal of general permits is to reduce the complexity, time span, and cost of redundant individual permit processes. General permits can be issued on nationwide or regional scales, and reissued every five years. The majority of these permits address projects involving discharge of dredged or filled materials and have specific conditions to minimize negative environmental impacts (Brody *et al*., 2007; Brody *et al.*, 2008; U.S. Army Corps of Engineers, 2015). Nationwide general permits are issued by USACE, and generally apply to activities that commonly occur across the nation like residential development, utilities, and wetland restoration projects (Brody *et al*., 2008; U.S. Army Corps of Engineers, 2015). In order to ensure minimal impacts, they also include regulatory conditions like compensatory mitigation measures for the wetlands altered or removed and adherence to the Coastal Zone Management Act (ELI, 2008; ELI, 2010). Each specified state agency, like the MDE, has the authority to review and enforce additional conditions on permits for tidal and/or non-tidal wetlands (MDE, 2008; MDE, 2009). In accordance with Section 401 to protect water quality, each permit must pass Section 401 Water Quality Certification regulations. In cooperation with reviewers like the Environmental Protection Agency and the U.S. Fish and Wildlife Service, the USACE has the authority to suspend, modify, revoke, or authorize nationwide or regional permits on a case-by-case basis. Regional general permits are also designed, reviewed, and enforced by the USACE, but can be issued for a larger area (*e.g.,* region) like a full state or watershed (*e.g.,* the Chesapeake Bay) (U.S. Army Corps of Engineers, 2015; Taylor, 2014). Programmatic general permits are submitted on behalf of a pre-existing state, local, or other federal agency's permitting program. The goal of programmatic permits is to reduce the redundancy of existing permitting types and application processes (U.S. Army Corps of Engineers, 2015). For example, MD SPGPs have replaced regional general permits. Multiple permits can be issued for one project, but one permit cannot be issued for multiple projects (MDE, 2009; U.S. Army Corps of Engineers, 2015).

All general Section 404 wetland permits must adhere to federal regulations like the Endangered Species Act, the National Historic Preservation Act, the National Environmental Policy Act, Section 401 of the CWA, and the Coastal Zone Management Act (Copeland, 2010). The permits must also adhere to state and local permitting, environmental, water quality, and zoning regulations. All approved general permits must also address activities related to issues like soil erosion, aquatic life movement, water quality, natural resources, waterfowl breeding, resource waters, and wild/scenic rivers (ELI, 2008; ELI, 2010). As previously mentioned, a permit can be denied due to the violation of any of the statutory regulations, adverse environmental effects, or site specific conditions. If so, the applicant will be directed towards a revision process, an individual permit application, or letter of permission.

Individual permits tend to have work plans that are more complex and outside of the scope of general permit, thus requiring a formal agency and public review (Copeland, 2010; ELI, 2008; ELI, 2010). Two types of individual permits can be obtained through the USACE or voluntarily by an individual state: standard permits or letters of permission. If the permit application is submitted to the state, the USACE and the state often offer a joint application and review process that follows the Corps application procedures (ASWM, 2016; Copeland, 2010; ELI, 2008; ELI, 2015). Each permit can be issued, denied, amended, suspended, or revoked through cooperative work between the USACE and state agencies. Letters of permission are created to streamline and abbreviate the individual permit application process. In order to gain approval of a letter of permission, the project must have minimal impacts on ecosystems and habitats (ELI, 2008).

State Program Permits (STPs). In Delaware, the state regulates tidal wetlands with state programmatic general permits (SPGPs) in accordance to the Delaware Wetlands Act of 1973 and the Subaqueous Lands Act of 1986, which regulates activities like construction, filling, dredging, and bulk heading. The Subaqueous Lands Act of 1986 regulates activities like the extraction, removal, or deposit of materials as well as the construction and renovation of structures (ELI, 2010; ELI, 2015). Activities like agriculture, grazing gaming, specified pest control, and navigational infrastructure are exempt from state regulated permitting, but must still have minimal impacts to existing wetlands (ELI, 2010; The Delaware General Assembly, 2014). The state also has the authority to review nationwide permits submitted to the U.S. Army Corps of Engineers. Though not officially regulated by state wetland permits, non-tidal and freshwater wetlands can be indirectly conserved, preserved, or protected through county and local planning and zoning strategies as well as agricultural conservation easements.

The State of Maryland regulates wetland permits in coastal and freshwater wetlands with SPGPs in accordance to the following state regulations: the Tidal Wetland

Act, the Nontidal Wetlands Act, the Chesapeake Bay Critical Area Act, and the Section 401 Water Quality Certification Program (ELI, 2008; MDE, 2016). These regulations require the submission of permit applications for activities like filling, dredging, and construction that fall within a specified buffer a particular wetland type, habitat protection areas, critical areas, and waterbodies like the Chesapeake Bay. The SPGP was designed to have an impact area of up to 1 acre. Activities that are exempt from regulation include activities related to agriculture, forestry, mosquito control, and the restoration of restorable infrastructure used to control erosion (MDE, 2016). The MDE has the authority to review joint state/federal applications with the U.S. Army Corps of Engineers.

The State of Virginia has the authority to regulate permits applicable to vegetated tidal wetlands and non-vegetated shorelines (ASWM, 2016; ELI, 2008). The permitting decision tree and delegation of duties is primarily based on the following regulations and policies: the Section 401 Certification of the Clean Water Act for Section 404 permits, the Virginia Water Protection Permit Program, and the VA Nontidal Wetlands Act. State permits are required for activities like excavating, filling, dumping or any activity that may alter the functionality of a wetland or associated hydro-geomorphological processes like drainage and flooding. The Virginia Marine Resource Commission (VMRC) also has a joint review process for tidal and non-tidal wetlands with the VA Department of Environmental Quality (DEQ), localities in the tidewater region of VA, and the USACE.

When attempting to address wetland change and restoration on a regional scale, policymakers must account for numerous differences in state wetland management policies, regulations, implementation strategies, and mapping (ASWM, 2016). For example, only MD and VA implement “No Net Loss” polices, and offer incentives for “Net Gain.” On the other hand, DE is the only state pursuing the power to manage the Section 401/404 permitting program with respect to state and federal wetlands (ASWM, 2016). With each state possessing different powers and responsibilities, approval and enforcement of permitted activities have the potential to be based on different political, economic, agricultural, and technical parameters. For example, the Corps’ compensatory mitigation strategies in VA and MD may differ in scope and temporality from DE, which requires a cooperative effort in managing wetlands on streams that cross state boundaries, like the Nanticoke River Watershed (ASWM, 2016).

Differences in wetland definitions and scientific methods complicate the ability to monitor and temporally assess trends, patterns, and drivers of wetland change. For example, state wetland delineation methods can differ from federal methods when delineating wetlands that do not fall under federal jurisdiction. MD is in the process of creating an EPA-approved State Wetland Plan to accompany DE and VA (ASWM, 2016) In order to regulate and preserve the distribution of wetland habitats, only MD and DE have implemented wetland and critical area buffering into their development and comprehensive plans.

Through state legislation and local regulations, officials can wetland management policies according to certain activities (*e.g.*, excavation for building a residential structure) and/or wetland type (*e.g.*, tidal or non-tidal) (ELI, 2008). Individual permits with activities that could have significant environmental impacts often avoid denial by containing compensatory mitigation measures to prevent environmental degradation (*e.g.*, wetland banking, 1 to 1 acre replacement, the purchase of existing wetland cover for preservation, etc.), subsequent revisions to address concerns noted by reviewers, the addition of specific conditions to the application (*e.g.*, long term monitoring requirements by the reviewer, or pre-application meetings with the Corps, federal, state, or local agencies (ELI, 2010).

3.1.4 Existing Research on Wetland Change and Permitting

The majority of research on wetland change and permitting in the North America has focused on wetland permitting and the compensatory mitigation component of Section 404, rather than the relationships between the geographic distribution of permits, locations and trends of wetland loss, and the drivers of wetland loss (Brody *et al*,. 2008; Cole & Shafer, 2002; Robb, 2002; Sifneos *et al*., 1992). The research targets mitigation measures and restoration efforts after a wetland or its ecosystem has been altered (Bendor, 2009; Cole & Shafer, 2002; Gardner *et al*., 2009; Kentula *et al*., 1992, Sifneos *et al*., 1992). Compensatory measures include activities like the creation of new wetlands to complement the number of acres lost from a specific activity or the insertion of depleted vegetation in degraded wetlands (Doyle *et al*., 2012; Goldman *et al*., 2015; Perry *et al*., 2001). With the majority of wetland permits being approved, compensatory mitigation stands as one of the primary application requirements regulators can use to force the applicants to reduce the short and long term impacts of wetland loss or degradation. The effectiveness and sustainability of compensatory measures can be tracked, and this increases the ability of regulators to enforce codes and penalties. The majority of pre and post permit research focuses on the performance of wetlands, impacts to landscape conditions, cumulative impacts, and potential drivers of wetland loss (Kelly, 2001; Stein & Ambrose, 1998).

Most existing research on wetland change or permitting on the study region has only focused on wetland change on short term, local scales, or on the functionality of specific wetland types in watersheds like the Nanticoke and Chincoteague (Cohen *et al*., 2016; Moglen *et al*., 2011; NOAA, 2012; Nosakhare *et al*., 2012; Tiner, 2004; Tiner, 2005). The closest approach to capturing trends of wetland change has come from a series of reports from the FWS (Dahl, 1990, 2000; Tiner *et al*., 1986; Tiner *et al*., 2011; Tiner *et al*., 2012).

Studies that have focused on spatial and temporal trends and patterns of wetland permitting have targeted wetlands in the Gulf of Mexico, the West Coast, the Mid-Atlantic, and Canada (Brody *et al.*, 2008; Hartmann & Goldstein, 1994). For example, Brody *et al.,* (2008) examined the Section 404 permitting program in the Gulf Coast region. Results for the Florida study area found that 57% of permits were distributed inside urban areas and 51% outside of 100-year floodplains between 1993 and 2003 (Brody *et al*., 2008). For the Texas portion of the study area, 78% fell inside urban areas and 61% outside the 100 year floodplain. The wetland types most impacted by permits in Texas were estuarine and palustrine in Florida. Clare and Creed (2014) examined the relationship between wetland management policies and change in a watershed in Alberta, Canada from 1999 to 2009They found that 80% of the area of wetland loss occurred in wetlands that were not being managed under a permitting system. The study also highlighted challenges in the government’s information management and data sharing strategies, which has led to regulatory oversight. Their wetlands inventory and land use analysis discovered that the majority of the wetlands in the study area were being converted in to agriculture or urban/industrial land uses.

Geographic assessments improve the understanding of the degree to which permitted activities have impacted wetland change on a long-term, regional scale. The results provide a clearer picture of how development, climate change, and landscape characteristics impact the spatial distribution of wetlands. Without wetland mapping and time series geospatial analyses, scientists, planners, and policymakers have limited ability to design sustainable wetland, floodplain, water quality policies that must protect wetlands on the Delmarva peninsula that is forecasted to simultaneously experience an increase in population growth and sea level rise. Before policymakers can decide on the most efficient restoration programs to fund, scientists must identify the wetlands most vulnerable to degradation or loss, as well as the wetlands that have passed the threshold of sustainability.

3.1.5 Objective

This study was conducted to identify spatial and temporal trends, patterns, indicators, and impacts of wetland permitting on the Delmarva Peninsula between 1980 and 2010. First, I used national LCDs to investigate spatial and temporal trends of wetland change between 1984 and 2010. Second, I aggregated and analyzed federal and state Section 404 permit records to identify spatial and temporal trends and patterns of wetland permitting. Third, I identified the wetland types with the highest percentages of permitted activities and the locations that experienced the highest concentrations of wetland loss and permitting. Fourth, I tested the correlation between wetland permit density and acreages of 3 land cover groupings that wetlands were converted into:

developed, natural, and agriculture. Finally, I examined the correlations between wetland permitting, wetland loss, and the following drivers of wetland change: urban development, landscape conditions (*e.g.,* floodplains), residential development, and road density.

3.2 **Data and Methods**

3.2.1 Study Area

As illustrated in Figure 3.1, the Delmarva Peninsula covers portions of the Eastern Shore of Maryland, the majority of Delaware and two counties of northeastern Virginia. Located in the Coastal Plain Physiographic Province, the northwestern edge of the peninsula sits on a fall line that serves as a transitional zone between the Piedmont and Coastal Plain regions (Goldstein & Hartman, 1994). The Peninsula sits between three primary waterbodies, the Atlantic Ocean to the East, the Delaware Bay to the northeast, and the Chesapeake Bay to the west. The Delmarva Peninsula will be used for this study because of its variation in land cover types, heavy concentration of wetland cover, trends of population growth, active participation in the Clean Water Act - Section 404 wetland permitting system, and its unique position on the Mid-Atlantic seaboard that is susceptible to sea level rise and hurricanes. The Delmarva Peninsula economy depends on tourism, shipping and agricultural activities like poultry farming and the pulp industry (American Farmland Trust, 2005; BEACON, 2010; Southwick Associates, 2012). The peninsula hydrogeomorphic complex consists of a high percentage of undisturbed barrier islands, shorelines, and sub-watersheds that continue to experience disturbances and ecological stress from urbanization, seasonal tourism, and sea level rise.

Since the late 1700's, the Delmarva Peninsula has lost an estimated 2 million acres of wetlands due to activities like ditching and ponding to improve hydrological conditions for farming, draining and filling to develop impervious surfaces, and channelization and dredging to improve waterway navigation (Dahl, 1990, 2000; Tiner & Finn, 1986). Over the past 200 years, each state on the Delmarva has over 40% of wetlands converted to non-wetlands (Dahl, 1990; Fretwell *et al*., 1996). Between the 1950 and 1970, agricultural cultivation accounted for est. 20% of wetland loss in DE, almost 40% in MD, and almost 50% in VA. Channelization accounted for wetland losses of est. 50% in DE, 30% in MD, and 25% in VA. DE suffered an est. 12% loss due to urban development, while 20% of the wetland loss in MD and VA were attributed to ponding (Tiner & Finn, 1986). The peninsula has experienced population growth due to sprawling communities from the metropolitan hubs like especially on the Eastern Shore of Maryland and on uplands (*e.g.,* terrestrial lands that are commonly dry) of major riverine systems like the Choptank, Nanticoke, and Pocomoke watersheds (Moglen *et al*., 2011; Nosakhare *et al*., 2012). As a result, wetlands have been fragmented or converted into developed land uses like roadways and neighborhoods (Dahl, 1990; Hartmann and Goldstein, 1984).

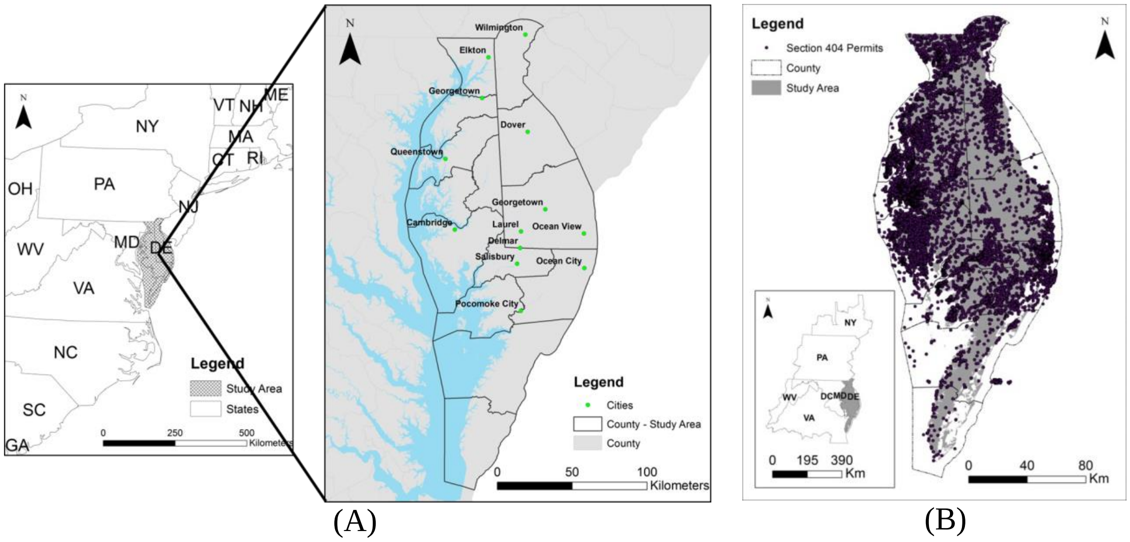

Figure 3.1 Map of the Delmarva Peninsula with the study areas highlighted. Expanded maps show Section 404 permit locations from 1980 to 2010 (a) The mid-Atlantic region and the Study Area (left) and The Delmarva Peninsula with bold county outlines (right); (b) The study area according to county boundaries from the U.S. Census Bureau and points representing Clean Water Act - the Section 404 as well as state programmatic permits issued between 1980 and 2010 .

3.2.2 Source Data

This study used regional land cover data sets to quantify and locate wetland change. Second, it used federal and state wetland permitting records to analyze spatial and temporal trends and patterns of changes in wetland permitting quantity and density. Detailed information on each data set is discussed in the following section.

Regional Land Cover Data Sets. This study used the National Oceanic and Atmospheric Administration (NOAA) Coastal Change Analysis Program (C-CAP) and the U.S. Geological Survey Chesapeake Bay Land Cover Database (CBLCD) 30-meter resolution data sets to quantify and locate wetland change on the Delmarva Peninsula. These validated data sets provide the ability to analyze wetland change across multiple state boundaries and time periods. The U.S. Fish and Wildlife Service (FWS) National Wetland Inventory (NWI) data set was used to assign the following wetland types to the Section 404 wetland permits: emergent, lacustrine, marine, palustrine, and riverine.

National Wetland Inventory (NWI). The NWI is the finest spatial resolution and most comprehensive national wetland mapping initiative. Furthermore, it also produces more accurate data than other national mapping programs. The NWI is a comprehensive database of wetland maps and GIS layers produced using moderate to fine resolution aerial photographs and satellite imagery combined with field verification and ancillary data (FGDC, 1998; Nichols, 1994). The majority of aerial photographs used to create NWI maps for the western portion of Delmarva Peninsula were as old as 30 years old (*e.g.,* gathered during the 1970's), and as old as 10 years (*e.g.,* gathered during the 2000's) for NWI maps of the eastern and southern sections of the peninsula. Though NWI maps are often outdated and are known to omit some forested wetlands and other wetlands that are difficult to detect, they are still one of the most comprehensive sources of wetland mapping in the U.S. They are frequently used to support the monitoring and regulation of wetlands on national, state, and local scales. NWI data were acquired by downloading seamless wetland data in the form of geodatabases on a state by state basis from the U.S. FWS Wetlands Mapper interface. This study could not analyze the relationship between specific wetlands and permits on a case-by-case basis, because neither the NWI nor the regional LCDs represented regulatory boundaries.

Wetland Permitting Data Sets. The goals of federal and state wetland permitting programs and regulations are generally to maintain a no net loss of wetland coverage nationwide and in each state, to encourage restoration of degraded or lost wetlands, and to minimize the impacts of permitted activities on wetland health and ecosystem services. The permitting records were provided by the U.S. Army Corps of Engineers (USACE), the Delaware Department of Natural Resources and Environmental Control (DNREC), the Maryland Department of the Environment (MDE), and the Virginia Institute of Marine Science (VIMS).

The federal permits (n = 21,634) used for this analysis were issued under Section 404 of the Clean Water Act from 1980 to 2010. The study area intersected the Baltimore, Philadelphia and Norfolk districts of the North Atlantic Division of the USACE. Each permit record included the permit type (*e.g.*, general, individual, nationwide, letter of permission, or state programmatic general), the year issued, and the geographic location of the permit according to latitude and longitude. The original state managed permitting records were provided by DNREC (n = 521 from 1997 to 2013), MDE (n = 3,981 from 1996 to 2011), and VIMS (n = 2,036 from 1997 to 2013).

3.2.3 **Methods**

The methodology consisted of five major steps: 1) pre-processing and spatial matching of the regional and ancillary GIS data, 2) calculating the quantity, percentage and density of wetland cover and change using regional land cover data sets 3) calculating the quantity, percentage and density of the distribution and change of wetland permitting at the state, county, and minor civil division (MCD) scales and wetland types at the wetland system scale, 4) identifying clusters of minor civil divisions with high permit density and wetland loss, and 5) analyzing the correlations between wetland density, density of wetland change, and acreages and densities of potential natural and anthropogenic drivers of wetland change (*e.g.,* developed, natural, and agricultural land cover classes).

Geo-processing and Standardization. The permitting records and pixels from each LCD were assigned the following socioeconomic spatial units by overlaying the following U.S. Census GIS layers in ArcGIS: state, county, and minor civil division (MCD; n = 135), and urban areas. The MCD scale was selected as the socioeconomic scale because of its moderate resolution and agreement with the political and Census county and block group boundaries, which were most likely to stay constant during the entire study period. Temporally, the study stretches from 1980, the year of the earliest permitting records, to 2010, the year of the most recent LCD.

Wetland Change Analysis using Regional LCDs. The magnitude and primary locations of wetland change (including loss and gain) were identified by calculating the quantity (total area in hectares) and percentage (of total area of wetlands) of wetland change at the MCD scale. They were calculated for the following time periods at the Anderson Level-I category of all general wetlands: 1984 to 1992 using the CBLCD and 1992 to 1996, 1996 to 2001, 2001 to 2006, and 2006 to 2010 using C-CAP. The same analysis was repeated for developed land cover classes (*e.g.*, urban and built-up lands), natural/undeveloped classes (*e.g.*, forest and rangeland), and agricultural/cultivated land cover. A time series of maps illustrating the "hot spots" of wetland change at the MCD scale was created for each category (*e.g*., developed, undeveloped, and agricultural/cultivated) by visually defining MCDs with the highest percentages of wetland change (*e.g.*, loss and gain). Hot spots were also defined as clusters of MCD's with similar high (or low) densities of wetland permitting.

Descriptive Analysis of Wetland Permits. For the descriptive analysis of the permit records, permits were first assigned one of the following wetland system classifications by overlaying each dataset onto an NWI layer: estuarine, lacustrine, marine, palustrine, and riverine wetland (Cowardin *et al*. 1979). Due to variations in the positional accuracy and projection and coordinate systems of NWI polygons and permit records, permit location points did not always fall inside a NWI polygon. In this case, the nearest NWI polygon attributes were transferred to permit locations within a 400 meter radius. This radius was calculated using the ArcGIS Incremental Spatial Autocorrelation tool, which used z-scores to identify the 400 meter distance as the statistically significant peak z-score that reflected the radius at which clustering of permits was most pronounced. Permits with distances to the nearest NWI polygon greater than 400 meters were excluded from this part of the analysis. Of the 26,968 permits received from the

USACE and respective state agencies, about13% (n = 3,451) were excluded because of insufficient geographic information or duplication.

ArcGIS was used to measure the quantity and percentage of permits in each state, county, MCD, and wetland system. The quantities and percentages of permits as well as the quantities and percentages of change in the number of permits from one time period to another were calculated at approximately 5 year intervals from 1980 to 2010. These time periods were intentionally staggered to complement the time periods used to analyze wetland change using the C-CAP and CBLCD regional data sets: 1984, 1992, 1996, 2001, 2006, and 2010. Because the results from analyzing of the quantity of permits alone may be potentially spatially biased due to highly developed areas also being located on coastal landscapes, two variables analyzing the density of permits were normalized by dividing the quantity of permits in each MCD by 1) the area of the respective MCD, and 2) the area of wetland cover in the respective MCD. The density of the change in the quantity of permits was also calculated for each of the previously mentioned time periods.

Hot Spot Analysis of Wetland Change and Permitting. First, I identified the locations (*e.g.,* MCDs) and wetland systems with the highest densities of wetland permitting. Second, spatial and temporal trends of permitting were analyzed from 1980 to 2010 using the selected time periods. In order to identify the locations of statistically significant hot and cold spots of wetland permits by year. Third, the ArcGIS Optimized Hot Spot Analysis tool was used to examine the z-scores (*e.g*., the intensity of spatial clustering) and p-values (*e.g*., the statistical significance) of the spatial distribution of the permit densities, normalized by the area of each MCD, for each year (Chainey, 2011;

ESRI, 2016). This method was chosen because it identifies statistically significant clusters of high (hot) and low (cold) values of permit density. All permit types were given the same weight, because there were no attributes (*e.g.*, area of impact) that could be confidently used to differentiate the magnitude of impacts of each permit. For example, an MCD with a low density of permits could contain one large institution (*e.g.,* industrial or academic institution) that acquired a permit that lasted for 5 years and negatively impacted the nearest wetland as well as wetlands downstream from the institution.

Geo-statistical Analysis of the Correlation between Wetland Permitting and Change. The end goal was to determine if the locations and time periods of high permit density were randomly distributed and to relate it to different land covers using correlation variables. It could also be used to verify that activities that may have been exempt from permitting (*e.g.*, agriculture) still had a strong, positive correlation with wetland loss. Inversely, it would support the concept that lower levels of wetland permitting would occur in locations that maintained undisturbed, natural land cover (*e.g.*, forests). First, I calculated the following variables at the MCD scale for each time period: 1) the acreage and density of wetland coverage converted from wetland to another land cover type and vice-versa, 2) the acreage and density of developed, natural, and agricultural coverages, and 3) the density of wetland permits. An exploratory analysis was performed using normal probability plots and the Shapiro-Wilk normality test to test if the distributions of wetland density and permitting were normal. The non-parametric, Spearman’s Rank Correlation test was used to test if the distribution of the density of wetland permits against acreages of developed, natural, agricultural land covers were random.

With respect to the independent variables, the number and percentage of permits were calculated in metropolitan statistical areas as defined by the U.S. Census Bureau to measure the distance between urbanized areas and permits used for development. Third, the number and density of total housing units (Den_THU) were calculated according U.S. Census Bureau from 1980, 1990, 2000, and 2010. Suburban sprawl has led to series of residential developments along with the creation and expansion of roadways throughout the Delmarva Peninsula resulting in wetland alteration and removal. The density of impervious roads (Den_IMPRD) was calculated using a 30 meter resolution LCD provided by the Chesapeake Bay Program Office (CBPO, 2016). Next, the number and percentage permits within a floodplain defined by the FEMA 100-year flood hazard layer was calculated. For counties without a FEMA-defined layer, attributes from the SSURGO database were queried to create a floodplain layer that was appended to the FEMA layer (Sangwan, 2013). Existing research has shown that development and wetland disturbances within floodplains magnify the impacts of flooding events (Brody *et al*., 2008). Lastly, the correlations between the density of wetland loss (Den_WLOSS), the density of total permits (Den_TPERM), the percentage of total permits issued in the floodplain (Prct_PERM_FLD), and the percentage of wetland loss that occurred within the floodplain (Prct_WLOSS_FLD) at the MCD scale were calculated in R using the Kendall Tau rank correlation test. The function was used to test the strength of association between the variables that were not normally distributed.

3.3 **Results**

The results section consists of a summary of: 1) the area, percentage and density of wetland cover and change, 2) the area, percentage and density of the distribution and change of wetland permitting at various spatial scales and wetland types, 3) hot spot locations and time periods of wetland change and permitting, 4) the percentages of permits in urban areas and floodplains, and 5) the correlations between wetland density, the density of wetland change, and the acreages and densities of potential natural and anthropogenic drivers of wetland change.

3.3.1 Wetland Change on the Delmarva Peninsula

The spatial and temporal analyses of wetland change were conducted on the MCD, county and state scales. Wetland change on the Delmarva Peninsula at the state and county scales from 1984 to 2010, according to CBLCD and C-CAP, is illustrated in Appendix A. During this period, wetland cover decreased by 2% (9.353 ha) (Figure 3.2 and Appendix A). The highest statistically significant ($p < .05$) percentage of loss occurred between 1996 and 2001. During this period, Delaware experienced a 2.5% loss, Maryland a 2.2% loss and Virginia a 1.4% loss. On the county scale, Wicomico (-3.3%, 952 ha), and Sussex (-2.4%, 1,454 ha), counties experienced the highest percentages of loss. With respect to wetland gain, the highest quantity and percentages of net wetland gain occurred from 2006 to 2010. The entire study area experienced a 0.18% (811 ha) of net wetland gain.

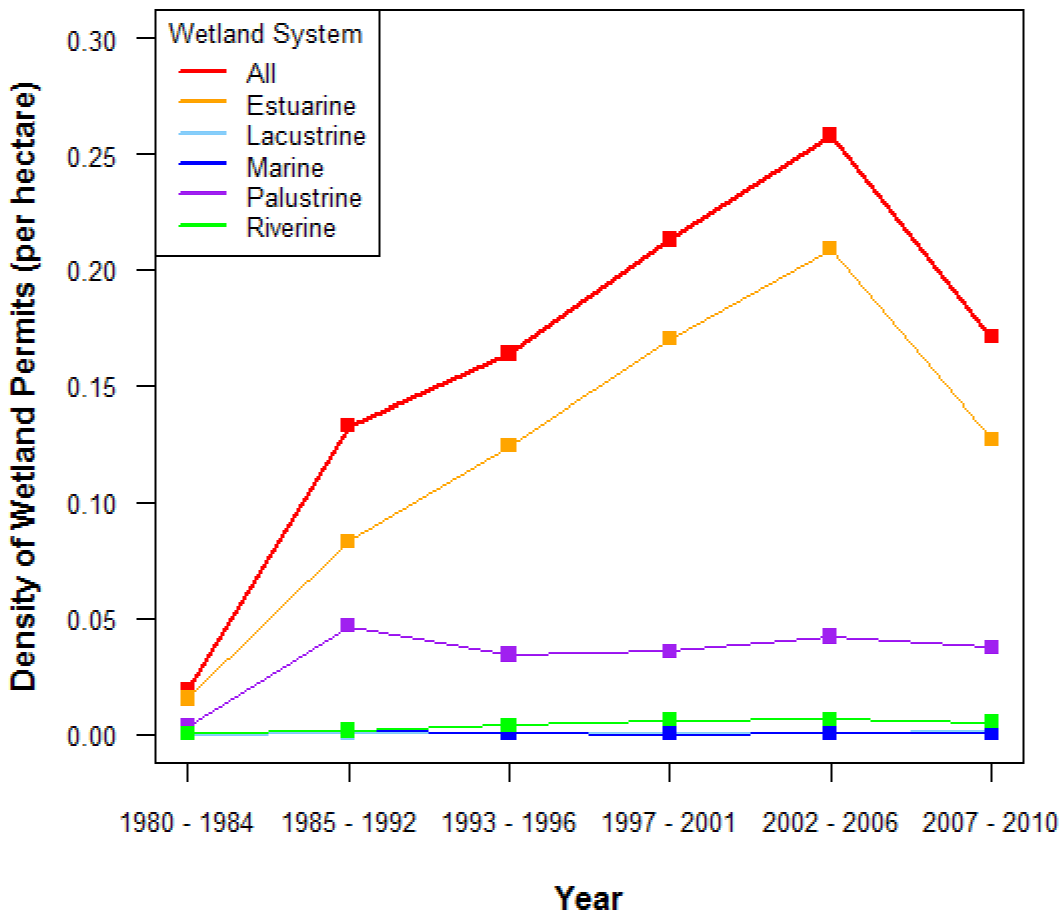

Figure 3.2 Time Series of line graphs of the density of wetland permits by wetland system from 1980 to 2010. The densities of wetland permits for lacustrine wetlands were not high enough to appear on the graph.

3.3.2 Hot Spots of Wetland Change

The primary hotspots of wetland change occurred in the following regions (labeled in Figure 3.3): between Salisbury and Pocomoke City, MD (Section A), urban corridors like Wilmington, DE on the northern portion of the peninsula (Section B), on wildlife refuges south of Cambridge, MD (Section C) and Dover, DE (north of Section D), on riverine systems, on barrier islands like the Assateague Islands (east of Sections A and E), in bay areas like Rehobeth Bay (east of Section D), and near major rivers on the northwestern portion of the peninsula like the Choptank River and Chester River (Section F). Hot spots of wetland change (e.g., locations with highest intensity of spatial clustering of wetland change within the 400 meter distance threshold) could be categorized by identifying geographic locations with clusters of pixels with common land cover conversions and anthropogenic and/or natural factors: urbanization, suburban sprawl, and sea level rise (See Figure 3.3). With respect to urbanization, the top 5 MCDs with the highest percentages of wetland loss, ranging from 7% to 12% were located in heavily urbanized areas in the northeastern portion of the peninsula like Elkton, MD (west of Section B) and in tourist hubs like Ocean City, MD and Ocean View, DE (east of Section A). Suburban sprawl, the migration of people out of urban areas to less developed areas adjacent to urban areas, was a prominent driver of wetland loss in southwestern DE (Central Section D) and along U.S. Routes 13 and 113 in DE (surrounding Section D) due to subsequent infrastructure expansion (*e.g.,* roadways), and sea level rise.

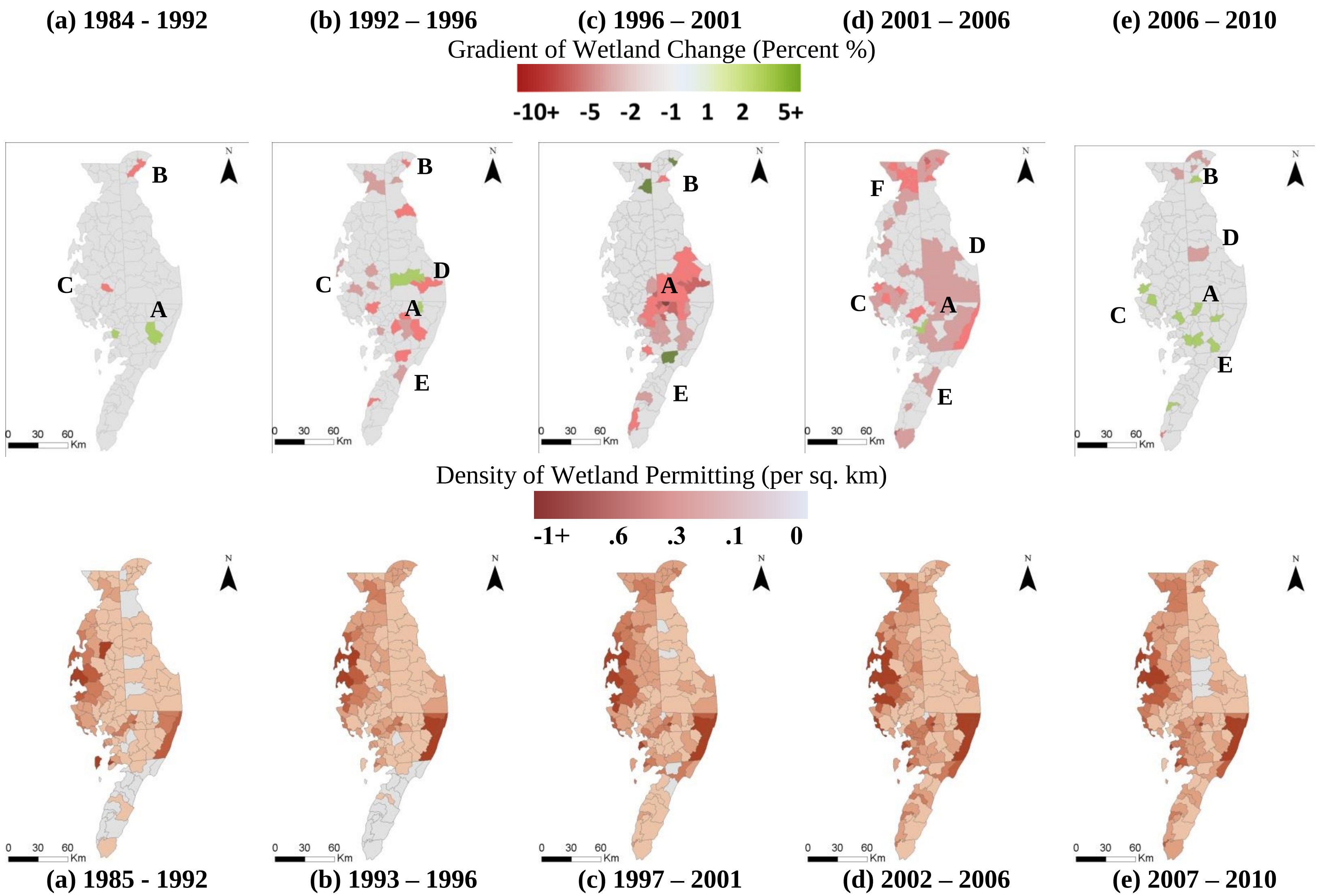

Figure 3.3 Maps illustrating: 1) wetland loss by percentage at the minor civil division scale from 1984 to 2010 (A –E): (A) CBLCD from 1984 to 1992 and (B-E) C-CAP from 1992 to 2010, and 2) the density of wetland permitting (per ha) at the minor civil division scale from 1984 to 2010.

3.3.3 Wetland Permitting Trends and Patterns

By Permit Type. Of the 23,293 federal wetland permits analyzed on the Delmarva Peninsula in Delaware, Maryland, and Virginia, 55.6% were categorized as General, 17.8% State Programmatic, 12.1% Nationwide, 11.0% Individual, and 3.5% Letters of Permission (Table 3.1). The majority of the permits were located in Maryland (91%) mainly due to Maryland representing 51% of the Peninsula, where rapid coastal development and agricultural activity has occurred over the 30 year period. When comparing the states' ratios of the count of each permit type to the total count of permits in the state, the majority of General permits (61%) were distributed in Maryland. On the other hand, Delaware issued almost 50% more Nationwide permits (almost 60%) and twice as many Individual permits (21%) than Virginia. The vast majority of State Programmatic permits (78%) were issued in Virginia.

Hot Spots of Wetland Permitting. Wetland alteration permits in Maryland were mostly concentrated on the central coast of the Chesapeake Bay (Queen Anne and Kent counties) (north of Section C), around and south of urbanized areas like Salisbury, and on the Atlantic Coast near Ocean City (Section A), (Figure 3.3). The Blackwater National Wildlife Refuge (Section C) is unique because it is dominated by emergent and forested wetlands that are at high risk to sea level rise, and is surrounded by heavy agriculture (Phillips, 2007). It is also adjacent to the suburbs of Cambridge, MD that have experienced increases (*e.g.*, migrating) due to communities from the Washington, DC and Annapolis, MD metropolitan moving to this area because of its convenient location, and affordable housing market, and natural façade.

The highest quantity and density of permits issued in Delaware occurred in northern DE in highly urbanized areas like Wilmington and Dover (Between Sections B and D) as well as on coastal developments along the Atlantic coast in Sussex County stretching from Rehobeth Bay to Assawoman Bay (east of Section A and D). The lowest quantity of permits was issued in the central portion of the peninsula.

On the Eastern Shore counties of Virginia (Section E), the highest concentration of permits were issued along on the western coast on the Chesapeake Bay and in the northern portion near Chincoteague, while the least permits were issued in the central uplands along the highway (13) that passes through both counties.

3.3.4 Temporal Trends and Patterns of Wetland Permitting

The temporal trend of permit issuance reflects the scale and type of wetland alteration for a given year. The periods of increase and decrease in the number of permits issued parallels the wetland change trends of loss and increase from 1984 to 2006 and a decrease until 2010 (See Figure 3.2) The surge in permitting along the DE coast and the DE, MD border occurred between 1993 and 2001 (*e.g.,* the period with the highest quantity of wetland loss). The exceptions to the trend occurred in Accomack and Northampton counties in VA which both experienced a continuous increase permitting (A-NPDC, 2011). This reinforced the fact that both counties also having abnormally high quantities of wetland loss compared to the other counties, potentially due to continuous shoreline protection to combat sea level rise and coastal flooding.

Delaware experienced a gradual increase in the density of nationwide permits that lasted from 1991 to 1997, followed by a gradual decrease only to experience shorter and smaller scale surges from 2002 to 2005 and 2007 to 2010. In Maryland, the density of granted permits followed a trend that suggested a shifting of dominant permit types from Letters of Permission in the 1980's to Nationwide and General permits in the early 1990's, to General permits from the late 1990's, to the mid 2000's to State Programmatic permits in the late 2000's. Virginia, State Programmatic General permits showed a gradual increase from1997 to 2003 and a gradual decrease until 2010, while General permits were abruptly surpassed by the State Programmatic permits in 2010.

By Wetland Type. By spatially overlaying permits on NWI polygons, the study assessed the intensity of wetland disturbance over time and the types of wetland systems impacted by anthropogenic and natural drivers of wetland change from 1980 to 2010. In DE, the majority of the permits were associated with palustrine systems (n = 669; 2.9% of all permits in all states) followed by estuarine wetlands (n = 402; 1.7%) (See Figure 3.4 and Table 3.1). The number of permits issued in palustrine wetlands appeared low, because of the omission of numerous DE permitting records due to insufficient geographic information (e.g. latitude and longitude) needed to map the permits. Nationwide permits accounted for 70% of the permits in DE, while standard (individual) permits accounted for 25% of the permits in the state. In Maryland, almost 78% of permits were directed towards projects that fell near estuarine systems, while almost 20% of the permits had the potential to impact palustrine wetlands (See Figure 3.4, Table 3.1, and Table 3.2). In VA, general and nationwide permits accounted for over two-thirds of permits issued in the state.

	Wetland Permit Type									
	Letter of Permission		Nationwide		General		Individual		State Programmatic	
State	n	% of Total	n	% of Total	n	% of Total	n	% of Total	n	% of Total
Delaware	23	0.1%	814	3.5%	42	0.2%	290	1.2%	200	0.9%
Maryland	810	3.4%	1,967	8.4%	12,908	54.9%	2,281	9.7%	3,234	13.8%
Virginia	3	0.0%	65	0.3%	122	0.5%	6	0.0%	729	3.1%
Total	836	3.6%	2,846	12.1%	13,072	55.6%	2,377	10.1%	4,163	17.7%

Table 3.1 Breakdown of state and federal Section 404 permits by type from 1980 to 2010 by state.

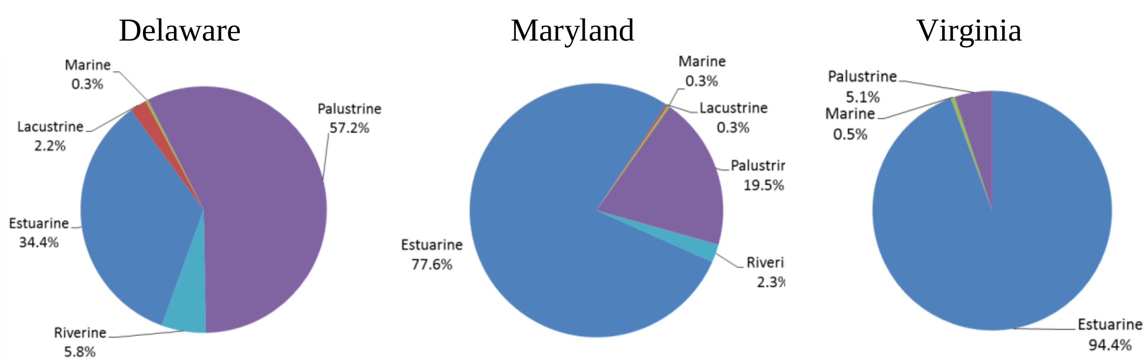

Figure 3.4 The Study Area's Section 404 permits by nearest wetland system type and state: 1980 – 2010. Note –The data excludes 200 state programmatic permits issued in DE.

	Wetland Permit Type						
Wetland Type	LOP	Nationwide	General	Individual	State	Total	% of Total
Estuarine	600	1,595	10,539	1,879	3,109	17,722	
	3.4%	9.0%	59.5%	10.6%	17.5%		76.1%
Lacustrine	3	25	31	8	19	86	
	3.5%	29.1%	36.0%	9.3%	22.1%		0.4%
Marine	21	12	6	29	2	70	
	30.0%	17.1%	8.6%	41.4%	2.9%		0.3%
Palustrine	4	1,131	2,204	603	731	4,858	
	0.1%	23.3%	45.4%	12.4%	15.0%		20.9%
Riverine	22	83	292	58	102	557	
	3.9%	14.9%	52.4%	10.4%	18.3%		2.4%
Grand Total	835	2,846	13,072	2,577	3,963	23,293	
	3.6%	12.2%	56.1%	11.1%	17.0%		

Table 3.2 Study Area's Section 404 permits by nearest wetland system type: 1980 – 2010. Within the study area, 3,675 permits lacked digital NWI data and/or fell outside of the 400m buffer and subsequently were not included.

The percentage of wetland permits granted outside of urban areas in 1980 (*e.g.*, the beginning of the study period) was 80% greater than the percentage granted inside urban areas (See Figure 3.5). From 1980 to 1994 the majority of wetland permits were issued outside of urban areas, with the peak occurring during the late 1980s (almost 100%). This trend reinforced patterns of suburban sprawl and coastal development. The continuous gap was not diminished until 1994. The disparity between urban and non-urban permits leveled off until 2004, when the gap began to increase again between 2004 and 2010 (*e.g.*, 60% outside versus 40% inside. The urban areas with the highest concentrations of (overall) permits were all located in Delaware.

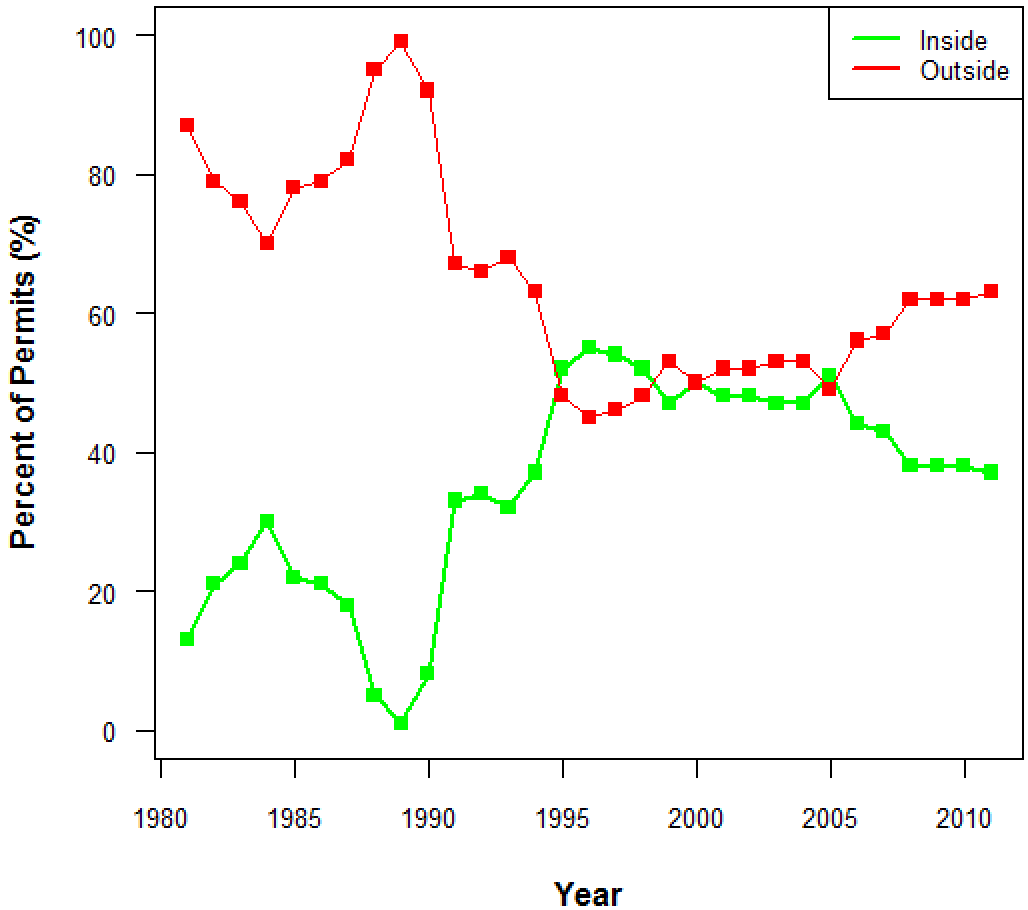

Figure 3.5 Line graphs showing the percentage of Section 404 permits issued within U.S. Census urban areas on the Delmarva Peninsula: 1980 – 2010.

According to the ArcGIS Hot Spot Optimization tool, highest statistically significant clusters of permitting in the early 1990's was located on the western side of the peninsula stretching from Elkton, MD (Section B) to south of Cambridge, MD (Section D) (See Figure 3.3). Towards the mid-1990s the hot spots of permitting were concentrated in the mid to southeastern coastal urban areas like Ocean City, MD (Section A). In the late 1990's the hot spots shifted back to the western coastlines of the peninsula. In the early 2000's, hot spots stretched from Ocean City, MD inland into Wicomico County (surrounding Section A), and in the late 2000's, shifted back to western coast near metropolitan areas like Cambridge, and Easton, MD (Section F).

3.3.4 Correlation of wetland permit density and land use change

According to the Q-Q plot, the change of wetland permitting density and the change of density of wetland coverage variables followed a non-linear pattern, which suggests that the variables are not distributed normally. According to the Spearman rank correlation test, the only time periods with statistically significant correlations between the change of wetland permitting density and the change of wetland coverage (*p*-values ($p < .05$)) were from 1992 to 1996 and 1996 to 2001. Both time periods had weak correlation coefficients (.267 for 1992 to 1996 and .384 for 1996 to 2001).

As previously explained, the Spearman rank correlation test was used to examine the statistical significance and the strength of the relationships between the change in the density of wetland permits and the acreage of developed, natural, and agricultural land covers. With respect to the developed land covers, only one temporal combination had a statistically significant ($p < 0.05$), but weak correlation coefficient (-.181): the 1985 to

1992 wetland permit period versus the 1992 acreage. For natural land cover, seven out of the eleven tests had a statistical significance of $p < 0.01$ with correlation coefficients ranging from -.345 to -.235. With respect to conversion to agricultural land cover, ten of the eleven tests had a statistical significance of $p < 0.01$ with correlation coefficients ranging from -.356 to -.235 (See Appendix C).

When evaluating the impact of wetland loss on growing floodplain communities, results show that on average 49% of permits on the peninsula were issued in the 100-year floodplain. When comparing the results on a state by state scale, Virginia had the highest percentage of permits issued in the 100-year floodplain, which is rooted to the counties' narrow width and location on the coastal edge of the peninsula. Only 47% of MD's permits fell inside the floodplain. Delaware had 51% of its permits issued in the floodplain. The majority of the counties (14) had over 50% of their permits issued in the floodplain. When examining temporal trends, both MD and DE experienced an increase in permits issued inside the floodplain from the early 1990's (approximately 33%) to 2001 (almost 60%). From 2002 to 2006, the percentage fell to around 30% in MD and below 25% in DE. The percentages gradually returned to above 50% by 2010. On the other hand, the percentage of floodplain permits hovered between 80% and 100% in VA, mainly because of geographic location and shape of the eastern shore of VA (See Figure 3.6).

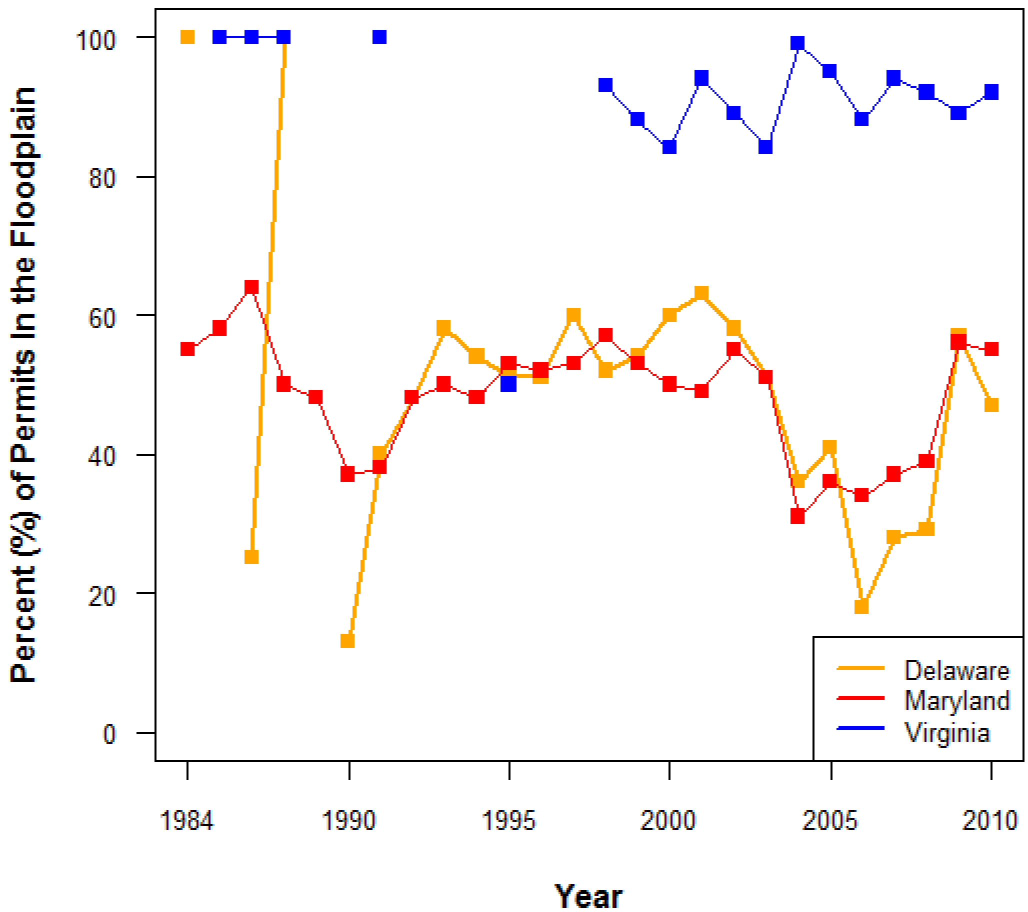

Figure 3.6 Line graphs showing the percent of permits issued within the floodplain on the Delmarva Peninsula: 1980 – 2010. The floodplain layer was produced by merging the FEMA Flood Hazard layer with a queried SSURGO soil layer.

With respect to the naturally induced wetland change, the percentage of permitted activities in the floodplains from 1980 to 2000 had a significant positive correlation ($p < 0.05$) with the percentage of wetland loss that fell inside the floodplain (See Table 3.3). All of the permitting variables, except the percentage of permits issued in the floodplain had a statistically significant correlation with the remaining wetland change variables. The impervious roadway variable had a significant positive correlation with the density of total housing units, and significant negative correlations with wetland loss and permitting variables. The relationships between density of total permits and percentage of wetland loss in the floodplain were not significant. The residential development variable had statistically significant correlations with both permitting variables, but the density of total permits had a negative *z*-score. The density of wetland loss variable also had a significant negative correlation.

Variable 1	Variable 2	Significance	tau value	Z-Score
Density of Impervious Roads	Den_THU	0.000	0.740	12.241
	Den_TPERM	0.739	0.020	0.333
	Den_WLOSS	0.047	-0.120	-1.985
	PRCT_PERM_FLD	0.001	-0.202	-3.338
	PRCT_WLOSS_FLD	0.896	-0.008	-0.130
Density of Total Housing Units	Den_WLOSS	0.028	-0.133	-2.194
	Den_TPERM	0.048	0.119	1.976
	PRCT_PERM_FLD	0.005	-0.169	-2.787
	PRCT_WLOSS_FLD	0.520	0.040	0.643
Density of Wetland Loss	Den_TPERM	0.012	-0.152	-2.518
	PRCT_PERM_FLD	0.831	-0.013	-0.213
	PRCT_WLOSS_FLD	0.046	-0.126	-2.000
Density of Permits	PRCT_PERM_FLD	0.050	0.119	1.963
	PRCT_WLOSS_FLD	0.024	0.142	2.265

Table 3.3 The results of the Kruskall Tau rank correlation test on the relationships between the following permitting, development, and wetland loss indicators: total housing units (Den_THU), the density of impervious roads (Den_IMPRD)the density of permits issued inside the floodplain (DEN_TPERM), the density of the area of wetland loss (DEN_WLOSS), the percentage of permits issued inside the floodplain (Prct_PERM_FLD), and the percentage of wetland loss that occurred in the floodplain (Prct_WLOSS_FLD).

3.4 Discussion

This geographic assessment of wetland loss, permitting, and drivers of wetland change provided a better understanding of how permitting activities have impacted wetlands on a regional scale that crosses multiple political boundaries. These results provide ecologists, wetland managers, and policymakers with a guide on how to spatially and temporally assess the impacts of permitted activities. More importantly, it illustrates where compensatory mitigation and restoration efforts should be directed based on development patterns around urban centers in flood prone areas. First, the results identified intense wetland permitting and loss between 1996 and 2001. Wicomico County, MD and Sussex County, DE (Sections A and D), that surround Salisbury, MD, each lost over 2.5% in wetland area and experienced continuous urban development (See Figure 3.3). The analysis could not directly attribute a specific area of wetland loss to each permitted activity. However the time period accounted for approximately 70% of the total permits distributed in Sussex County and 40% of permits issued in Wicomico County.

The next key period of loss occurred between 2001 and 2006, and occurred in three major regions: in the northern corner (Sections B and F), on the southwestern edge (Section C), and along the Atlantic Coast (Sections A, D, and E) (See Figure 3.3). The results highlight areas already heavily urbanized like Wilmington, DE and Elkton, MD that have higher percentages of wetland loss due to a smaller area of wetland cover in comparison to other Delmarva Counties. However, the communities continue to experience sprawl and conversion of agriculture and wetlands to developed land covers. During this time period, both VA counties had 90% of their permits issued to address sea level rise and storm surge events like Hurricane Isabel in 2003 (A-NPDC, 2011). Overall, the results point towards urban development and sprawl that are occurring on MD coasts, surges of inland development in DE near major rivers, and infrastructure revitalization in coastal communities in VA that are vulnerable to sea level rise and flooding events. See Appendix B for the tables and pie graphs illustrating Section 404 permits by nearest wetland system type from 1980 – 2010 .Planners and developers should take into consideration the location and frequency of permits to address re-occurring problems like sea level rise and coastal erosion.

3.4.1 Wetland Types influenced by Permitting

Though the majority of permits were issued for estuarine wetlands, the majority of wetland loss occurred in palustrine forested and emergent wetlands. Two potential reasons for these results are: 1) more Section 404 regulations are applied to estuarine wetlands, and 2) a greater abundance of small projects on homes (e.g., the building of docks) that result in the loss of small patches of estuarine wetlands. With hot spots of wetland permitting located in suburban areas along transportation corridors between coastal and upland communities, these results also imply that residential development from suburban sprawl and subsequent roadway expansion will continue to disturb, disconnect, and replace palustrine wetlands. In order to maintain "No Net Loss" and to prevent increased flooding from increasing the impervious surfaces, entities like the MD State Highway Administration will need to focus compensatory mitigation efforts on transportation corridors that connect urban centers. For example, the proposed railway system for commuters and shipping along Highway 113 in the portion of the Peninsula would require nationwide and general permits along with extensive compensatory mitigation due to the large scope and impacted area of wetlands. Cutting edge wetland mapping programs combined with cooperative wetland banking and BMPs will help states and localities monitor and identify wetland ecosystems that are appropriate for restoration and preservation.

The results also show that wetlands are affected by other anthropogenic drivers like agriculture, which are exempt from the Section 404 program. This finding has significant policy implications because it reinforces the need for the expansion and funding of wetland preservation and restoration programs like the U.S. Department of Agriculture Wetland Easement Program, which has concluded. Programs like these that support BMPs have become vital towards not infringing on the agricultural industry and simultaneously maintaining the hydrological functions and landscape conditions of wetlands.

3.4.2 Challenges with Existing Permitting Processes

Ecologists and wetland scientists have voiced concerns of potential problems related to the permitting process that negatively impact wetlands, habitats, and water quality (Connolly, 2006; Copeland, 2010). First, officials have no systematic method of reporting and tracking wetland change on an annual scale. The closest mechanisms that are used to track the spatial distribution of wetlands and temporal trends and patterns of change are the National Wetland Inventory (NWI) geospatial dataset – when the dataset is updated, NWI Wetlands the Status and Trends project which is conducted every 5-10 years, and the Natural Resource Inventory (NRI), all of which have spatial and temporal limitations. For example, the frequency of each state's source imagery for the NWI geospatial dataset and NWI Wetlands Status and Trends project varies (*e.g.*, leaf on/off, season, tidal period), which limits the feasibility and accuracy of time series analysis of wetland change on regional scales (Taylor, 2014). Similar to the NWI Wetlands Status and Trends project, the NRI is also based on sampling, wetland classes, but it the excludes wetlands on federal lands (Brooks *et al*., 2004; Stein *et al.*, 2012; Taylor, 2014). Researchers should continue to enhance mapping techniques that can be used to systematically analyze or identify critical areas of wetland change.

Second, legal and political issues complicate the effective regulation, enforcement, and evaluation of Section 404 permits. Issues that might lead to less effective wetland regulation include the ambiguity of regulatory definitions/terminologies like "fill material" and "waters of United States", the exemption of select activities, and the omission of wetland functionality and water quality variables from the formal review process. For example, activities related to wetland farming, ranching, forestry, and wetland draining activities are often exempt from review, despite their direct impacts on wetland coverage, functionality, and sustainability.

Third, policymakers and permit applicant have aimed to decrease the complexity and amount of time required to review and approve permits. However, it wasn't until the late 1980s that officials and scientists began to redirect their basis of decision making to include spatial and temporal trends of wetland loss (Connolly *et al*., 2005, The White House, 1993; Tiner & Finn, 1986; Tiner *et al*., 2011; Tiner *et al*., 2012). Despite the implementation of "No Net Loss" policy, the additional research is still needed to evaluate the efficiency and effectiveness of the permitting system in reducing wetland loss on a regional scale. In order to evaluate the status of “no net loss” benchmarks, federal and state agencies must track the acreage of net wetland loss on state and nationwide scales (ELI, 2010; MDE, 2016). Maryland and Virginia’s “no net loss” initiatives are tied to their non-tidal regulation programs. Maryland’s strategy requires mitigation measures for any wetland loss and a monitoring of wetland change on the watershed scale. Virginia’s strategy also has compensatory measures and incentive programs to reduce wetland loss and promote gain.

Finally, the indicators used to evaluate the impacts and efficiency of the permit distribution are limited in scale, breadth, and frequency (Peyre *et al*., 2001). None of the states in this study utilize a tool that goes beyond sampling techniques to track permitting patterns and their potential impacts on the acreage of wetland cover. For example, when evaluating the impacts of permit applications, the evaluator may not be required or equipped with the resources to consider the cumulative impacts that a high concentration of permits or the historical impacts of permitted activities around the project site may have had on the subwatershed(s) wetland coverage, water quality, or habitat. Currently, federal and state agencies have pioneered the development of geospatial tools (*e.g.*, the Wetlands Resource Registry) that integrate socioeconomic and environmental indicators to improve the design and implementation of policies dealing with wetland restoration, water quality, and smart growth (Moglen *et al*., 2011). However, inclusion of wetland change and permitting indicators could strengthen the efficiency of these tools.

3.4.3 **Limitations and Challenges**

The primary limitations of the permitting data sets used are as follows: limited attributes in the permitting databases with respect to permitted activities, the lack of, or coarse resolution of, attributes to the georeferenced permits, a lack of variables that quantified the area of wetland cover impacted by the wetland permit, and the challenge of accounting for wetland loss caused by activities exempt from the Clean Water Act and state and local regulations. First, the permitting GIS data set from the U.S. Army Corps of Engineers did not include a complete attribute field with information on the area of wetland cover impacted by each project. It is important to note that the area of wetland change may not have always directly reflected the area of impact from each project Due to the moderate resolution of the 30 meter land cover datasets used in this study, the calculated area and percentage of wetland change could have been the result of one or more of the permitted projects. Second, a challenge came with standardizing and georeferencing the permitting data set from the state of DE. The state's permitting data set did not have specific location information (i.e., latitude or longitude) to georeference or plot the data. It may have impacted the spatial and temporal analysis of hot spots of permitting that may or may not have occurred in the minor civil divisions throughout the Delmarva Peninsula. However, the data set did contain information in select fields (*e.g.*, project name, applicant, and waterway), that allowed the data to be incorporated into the temporal analysis of the annual total number and density of permits on the following spatial scales: the entire study area, state, and county. Third, the data set also lacked attributes regarding the spatial area impacted by the permitted activity. This, reinforces the need for the updating of the attributes of the digital records of the permits, as well as further research on how to assign weights (*e.g.*, magnitude or intensity) to the impacts of specific activities according to variables like the area of the project site, the time span of the project, or the distance of the project to a wetland. Finally, annual time series maps of the hot spots at the MCD were created from 1980 to 2010. Finally, this analysis did not drill down to examine the spatial and temporal relationship between specific permit types and the calculated area of impact or conversion from wetland to non-wetland. Challenges included: 1) the complexity and continuous evolution of the legal definition of wetlands and "Waters of the U.S.," and 2) the existence of activities exempt from the permit review procedure between 1980 and 2010.

3.5 **Conclusion**

This study was conducted to identify spatial and temporal trends and patterns of wetland change and permitting on the Delmarva Peninsula between 1980 and 2010. Spatially, palustrine wetland loss was concentrated around urban centers and expanded into suburbs. Estuarine wetland loss was concentrated on the Atlantic Coast, on the south western corner, and northeastern corner of the peninsula. Temporally, wetland loss and permitting increased until 2006, except in VA where permitting continues to increase, likely to counter drivers of wetland loss that are not directly human mediated such as sea level rise. The majority of permits were distributed in MD (91%) for urban development in the tourist center of Worcester County. Permits were mainly distributed outside of urban areas in suburbia until the early 1990's, and did not begin targeting the suburbs again until 2004. The Corps mostly issued nationwide permits (56%), followed by state program permits (19%). Even though the majority of permits were issued for estuarine wetland projects (76%), the majority of wetland loss occurred in palustrine emergent and forested wetlands (78%). This finding highlighted the issue that permits are being issued in part to respond to hazards, such as shoreline erosion. The results supported the hypothesis that wetland permitting and loss had statistically significant correlations with residential development, impervious roads, and floodplain characteristics.

General permits dominated the permit types issued on the Peninsula. As indicated in the results, general permits likely signaled urban and suburban sprawl. Unlike individual permits, general permits are designed to expedite the application process for projects that have minimal impacts on wetlands. However, each permit is reviewed independently, meaning that the continual issuance of permits for residential neighborhoods in a suburban corridor could have major long term, cumulative impacts on hydrological conditions (*e.g.*, connectivity, stormwater retention, and streamflow) surrounding the developed areas. As shown in past research, individual (standard) permits are often issued for development projects with larger footprints into wetlands (Brody *et al* 2007). Examples of these projects include the conversion of forests and wetlands into impervious roads and non-roads (*e.g.*, buildings and rooftops), both of which reduce the ability of wetlands to absorb stormwater and nutrient loads. Developers and emergency managers need more geospatial information regarding the cumulative impacts of urban development and climate change. For example, sea level rise could be driving permits activities like seawall installation, which could alter estuarine wetland habitats. On the other hand, suburban sprawl could be driving residential development, resulting in the conversion of non-tidal palustrine wetlands. Future research is needed to contextualize the area of impact according to policies and regulations active during the respective time period that the development activities were permitted.

Wetlands on the Delmarva Peninsula are in danger of continued loss if permitting systems and restoration efforts do not account for direct and indirect drivers of wetland change on multi-dimensional spatial scales. Wetland loss will lead to a loss in functions like flood protection, water quality, and biodiversity. The statistical results supported the hypotheses that wetland change and permitting have positive associations with urbanization, suburban residential development, and agriculture. Variability in the associations of distance to urban centers, roadways, and floodplain combined with land cover ratios provided insight on the increased vulnerability of upland and isolated palustrine, forested wetlands to residential development. Wetland change's statistically significant correlation with landscape conditions (*e.g.,* slope) reinforced existing research that relates impervious surfaces to increases in surface runoff and flood risks. The results also supports concept that the conversion of tidal wetlands is linked to coastal urban development as well as sea level rise.

4 CHAPTER 4: A CASE STUDY OF THE LOCAL AND LANDSCAPE CONDITIONS THAT CAN BE USED TO PREDICT WETLAND CHANGE IN SUSSEX COUNTY, DE

ABSTRACT

Wetlands provide ecosystem services that are critical to hydrological conditions, biodiversity, and water quality. To ensure that wetland habitats are protected and restored, an effective means of identifying spatial and temporal trends and patterns of wetland change and the drivers of wetland change is needed, especially in states in the Chesapeake Bay Watershed and on the Delmarva Peninsula. Sussex County, DE was selected as the pilot site because of its history of wetland loss and its dependence on tourism, agriculture, shipping, and urban development. Multivariate logistic regression was used to develop two models to predict the risk of wetland loss as a function of physical and anthropogenic indicators. To reduce the multicollinearity and autocorrelation of the selected predictor variables, a variance inflation factor (VIF) analysis was performed on the original suite of independent variables related to wetland characteristics, topography, land cover, hydrogeomorphology, climate, and Section 404 policies. The application of the VIF reduced the original count of independent variables from 213 (Model 1) to 66 (Model 2). Trends of wetland change were calculated using the National Oceanic and Atmospheric Administration – Coastal Change Analysis Program and the National Wetland Inventory. All 8,003 of the observations were used to build the model . The local and landscape variables and their influence on the variance of the model from highest to lowest (according to the four principal component groupings) was residential development and permitting (0.45), landscape conditions and hydrology (0.26), urbanization (0.18), and precipitation and erosion (0.10). This provided a

relatively moderate fit to the data (AUC =0.62). Scientists, planners, and policymakers can use the model and methodology to create new models and maps that incorporate finer resolution source data. Tools like this will help prioritize areas vulnerable to wetland loss, evaluate the impacts of permitted activities and mitigation measures, and predict future wetland change based on forecasted land use and land cover change trends.

4.1 **Introduction**

Wetlands play a critical role in maintaining balances in a variety of ecosystem conditions like hydrological conditions, biodiversity, and water quality (Chase *et al.*, 2003; Tablante *et al*,. 2002). Wetlands reduce flooding, mitigate nutrient loads, restore groundwater, and provide habitat for migratory species (Moglen *et al.*, 2007; Neilson *et al.*, 2007; Rogers & McCarty, 2000). Numerous plant and animal species depend on wetland ecosystems for vegetation and refuge (Bachman *et al.*, 1998; Rogers & McCarty, 2000). The environmental and economic value of these ecological services has been magnified by continuous wetland degradation and loss. Scientists and policymakers need geostatistical models of wetland change and its drivers in order to construct policies that will reduce the negative impacts of activities permitted by Section 404 of the Clean Water Act (CWA). The purposes of this paper were to test the hypothesis that wetland change is predictable from landscape conditions and context, and to develop a statistical model to predict wetland loss as a function of the landscape, wetland permitting, climate, and socioeconomic variables using the case study from Sussex County, DE.

4.1.1 Drivers of Wetland Change

Wetland functions and resilience are often influenced by their landscape position, regional climate patterns, and impacts from human development (Brooks *et al.*, 2004; Hartmann & Goldstein, 1994; Weller *et al.*, 2007). With the Delmarva Peninsula possessing over one third of the wetlands in the Chesapeake Bay watershed, wetland loss continues to increase the risks of flooding in coastal communities as well as water quality degradation from unregulated agriculture and the poultry industry. Historically, agriculture was the primary land use that degraded and prompted the removal of freshwater and tidal wetlands. Agricultural activities like drainage and stream diversion have permanently altered hydrologic conditions, fragmented habitats, and interrupted surface and groundwater flow/tables. Hydrologic alterations of wetland landscape conditions have also weakened the ability of wetlands to absorb and drain the nutrients and sediments from agriculture and silviculture. Since the late 1700's, the states that comprise the Delmarva Peninsula have seen an average of 50% of wetlands converted into non-wetlands like agriculture and impervious surfaces (Dahl, 1990; Fretwell, Williams, & Redman, 1996). Between the 1950 and 1970, the primary drivers of wetland loss on the peninsula consisted of agricultural cultivation, dredging/channelization, ponding, and urbanization (Tiner & Finn, 1986). The Mid-Atlantic Region continues to experience surges in population growth and residential development, which has increases the demand for policies and comprehensive plans that address the impacts of increasing impervious surfaces as well as sea level rise (Jantz *et al.*, 2010; Jantz *et al.*, 2011; Klemas, 2007). For example, the expansion of infrastructure like sea walls to protect residential and commercial structures from stormwater runoff and flash flooding has been linked to wetland loss and fragmentation (Dahl & Stedman, 2013).

4.1.2 Wetland Management Policies

In order to address negative impacts of wetland loss on water quality and stormwater management, scientists and policymakers have had to implement legislation like the Food Security Act of 1985 ("Swampbuster") and a series of "U.S. Farm Bills". These legislations promoted wetland restoration and conservation by discouraging wetland conversion through financial penalties, introducing wetland mitigation banking options, and creating long term easements to limit wetland alteration (Connolly *et al.*, 2005; ELI 2015). Federal and state agencies continue to seek improvements in enforcing compensatory mitigation to counter wetland loss and implementing a formal wetland mapping procedures and standards to improve the spatial and temporal monitoring of "No Net Loss" goals. However, the nation still lacks a common wetland management, monitoring, and protection law that defines universal regulations, jurisdictional definitions, delineation and mapping methods, and data management standards (Connolly *et al.*, 2005; ASWM, 2016).

Policies like the Section 404 - Wetland Permitting Program of the Clean Water Act (CWA) were created to reduce wetland loss from the discharge of fill materials and to increase wetland restoration (Connolly *et al.*, 2005). However, these policies and regulations have not been able to meet "No Net Loss" goals due to unintentional complications like evolving definitions of the "Waters of the U.S." as well as intentionally exempted activities like agriculture and silviculture. Also, the regulatory powers in wetland permitting programs often falls in the hands of state and local government agencies. These entities are often limited in capacity to effectively monitor the processes and impacts of the permitted activities and to enforce compensatory mitigation. The impact of a permitted activity often depends on the project scope, the area of influence, the surrounding landscape conditions, the permit type, and the nature of the permitted activity. For example, Section 404 general permits are designed to have minimal adverse impacts to wetlands, while individual permits tend to be more complex with negative impacts on wetlands requiring compensatory mitigation measures (Brody *et al.*, 2005; Taylor, 2014).

The impacts of permitted activities also reflect the relationship between socioeconomic priorities as well as physical conditions (*e.g.,* landscape) at spatial scales larger than the project site and temporal scales longer than the length of the project. For example, estuarine wetlands have often fallen outside of the scope of the agricultural conservation initiatives.. Plus, these initiatives focused on the management and efficiency of human efforts, not natural processes like climate change and habitat. Communities located in areas that are challenged by human and non-human drivers as well as ongoing impacts (*e.g.,* urbanization, water quality degradation, silviculture sea level rise, salinization, and recession) continue to struggle to design and maintain sustainable wetland management plans.

Despite federal, state, and local legislation, wetland loss has increased in some locations due to exemptions in the permitting system and limited resources to monitor and enforce wetland regulations. Urbanization has replaced agriculture as the primary threat to wetland conversion on the peninsula due to commuters and retirees sprawling from urban centers like Washington, DC, Philadelphia, PA, and Norfolk, VA into suburban and rural areas filled with palustrine, forested wetlands. The introduction, expansion, and revitalization of impervious surface roadways and building structures has been linked to increases in surface runoff, increases in pollutants in waste and stormwater runoff, and decreases in ground water recharge (Brody *et al.*, 2015). Subsequently, numerous species of vegetation have suffered from eutrophication, which comes from nutrient loading and from deoxidized runoff from heated impervious surfaces (Kaller *et al.*, 2013; Kirwan & Megonigal, 2013). In order for "No Net Loss" goals to be achieved across multiple state and hydrological boundaries more research is needed forecasting wetland change as a function of landscape conditions, wetland permitting, climate, and socioeconomic variables..

4.1.3 Differences in Wetland Permitting

When addressing wetland change and permitting on a long term, regional scale, it is challenging to account for numerous differences in federal, state, and local wetland policies, regulations, definition, and mapping methods (ELI, 2008; ASWM, 2016). For example, federal wetland delineation methods can be applied to federal and state wetlands. However, states have created delineation methods that can be applied to non-federal wetlands. Contrasts in wetland definitions and delineations methods complicate the ability to assess trends, patterns, drivers, and impacts wetland change. These contrasts have the potential to lead to inconsistencies in regulations and permitting, like underestimating riparian buffers or exempting activities that could pose a threat to downstream wetland habitats. Currently, each state on the Delmarva Peninsula possesses a different role and responsibility in evaluating permit applications, monitoring wetland change, and enforcing compensatory mitigation regulations. Often states must base their conservation plans on different economic circumstances, transportation systems, and development plans (ELI, 2008). There are also contrasts in "No Net Loss" policies and incentives to encourage wetland preservation. One commonality between all states is the need for accurate time series analyses and geospatial models that accurately identify and predict spatial and temporal trends and patterns of wetland change.

4.1.4 Existing Research

The applicability of geographic assessments of wetland change and vulnerability can be improved by integrating landscape context variables with land use and land cover change databases and socioeconomic indicators (Hollister *et al.*, 2004; Phillips, 2004). However, most existing studies on the status of wetlands on Delmarva Peninsula either focus solely on local (micro scale) non-tidal systems, generic landscape indicators (e.g., elevation and slope), geological properties, hydrogeomorphic functions; or agricultural development (Hussein & Rabenhorst, 2001; Nagler *et al.*, 2009; Rabenhorst *et al.*, 2001; Tiner, 2005; Whigham *et al.*, 2007). On the other hand, policymakers have primarily relied on the U.S. Fish and Wildlife Service, State Agencies, and think-tanks for research on the status and trends of wetlands on regional to national scales (Dahl, 1990, 2000; Mayer, 2011; Tiner *et al.*, 2011; USACE, 2010; USFWS, 2011). Studies are either over-simplified with no local, contextual value, too site specific for regional scale application and repetition, exclusive of socioeconomic or landscape characteristics, or too coarse in spatial resolution to identify and track local patterns and trends. There has been a recent increase in research on how wetland change and policies relate are correlated with one another on a variety of spatial (local and regional) and temporal (annual versus decadal) scales (Thomas & Lamb, 2005).

The backbone of this study was based on previous studies that discuss the use of integration methods to analyze the drivers of wetland change, the risk of wetland loss, and regional vulnerability in the Mid-Atlantic region of the U.S. (Daniels & Cumming, 2008; Gutzwiller & Flather, 2011; Ji, 2007; Locantore *et al.*, 2004; Smith & Tran, 2003). Various studies have analyzed the relationships between wetland change and drivers like urbanization, sea level rise, flooding and topography to create models that forecast wetland habitats most vulnerable to conversion (Brody *et al.*, 2015; Daniels & Cumming, 2008; Gutzwiller & Flather, 2011). For example, Daniels and Cumming (2008) identified topography, distance to roadways and urbanized areas, and topography as a few of the primary drivers and predictors of wetland loss in a basin in Costa Rica (Daniels & Cumming, 2008). However, most studies do not analyze the drivers of wetland conversion on regional, long term scales. Gutzwiller and Flather (2011) utilized a regional approach to identify the key anthropogenic and landscape predictor variables as well as a target spatial scale to incorporate into a model that predicted the risk for wetland habitat loss in the southern U.S. Their results identified the surrounding land cover as the dominant predictor variable followed by urbanization, patch size, propriety variables at the micro scale (within a 570m buffer). Landscape and local conditions are two components that must be geospatially monitored and assessed in order to track wetland change and design legislation to protect and restore wetlands.

When selecting locations for restoration projects and choosing BMPs to achieve water quality standard goals, policymakers have also had to consider the trends, patterns, drivers and impacts of wetland permit parameters and activities. For example, Brody *et al.* (2015) found that the magnitudes of flood risks in Florida and Texas were directly related to wetland permit type (Brody *et al.,* 2015). Larger scale projects with a larger footprint in wetland habitats posed the greatest risk to increasing impervious surfaces, which could increase the frequency and intensity of flood events. However, permitted activities for smaller projects increased the potential for alterations to hydrologic conditions (e.g., wetland connectivity, surface runoff patterns, and streamflow). Clusters of small projects (*e.g.,* cookie-cutter neighborhoods) were prone to have larger cumulative impacts than isolated, sporadic large projects (Connolly *et al*., 2005; Connolly, 2006). Planners and scientists now have the ability to selectively incorporate predictor variables like population growth, residential development, and land cover change into geospatial models, which can improve the predictability and accuracy of the wetland change models.

4.1.5 Objective

This study contributed to the knowledge gap that exists in understanding the hierarchical relationships between wetland change, and the physical and anthropogenic factors that influence the change. The specific objectives were as follows: 1) to develop a predictive statistical model for the risk of wetland loss for a case study, Sussex County, DE that can be applied to wetland management and policies, 2) to test the model with different combinations of independent variables, and 3) to explain how the model's indicators can be used to understand the drivers and prioritize policies to reduce wetland loss. The hypotheses are as follows: 1) the likelihood of wetland loss can be understood through a quantitative analysis of landscape context on a county scale, 2) the likelihood of wetland loss will have a strong relationship with a series of variables related to the distance of a wetland to development, agriculture, and water, 3) the likelihood of wetland loss will have a strong relationship with slope and elevation, and 4) agriculture and development will be the independent variables with the highest influence on wetland loss in the principal component analysis.

This study's contribution to the body of knowledge about the indicators of wetland loss included examining the risk of wetland change in relation to the characteristics of wetlands, areas adjacent to wetlands, and the landscape surrounding the wetlands, and the density of Section 404 permits within a specified buffer around a wetland(s). "Risk" was defined as the probability for wetland change (from a wetland to a non-wetland land cover) given that a particular geographic location (*e.g.*, a pixel) was classified as a wetland. This distinction was made to reiterate that the model was not created to predict the spatial area or location of wetland loss, but rather the predictability that a wetland or a portion of a wetland will change or be converted into a non-wetland.

4.2 **Data and Methods**

First, the binary dependent variable, wetland loss (WLOSS), was derived from 30 meter land cover data from the National Ocean and Atmospheric Administration – Coastal Change Analysis Program (NOAA – C-CAP) and the U.S. Fish and Wildlife Service National Wetland Inventory (NWI). The land cover history of each pixel was traced over three time periods (1992, 2001, and 2010). This dichotomous trajectory represented whether each pixel had been “unchanged” (0) or “changed” (1) by the final date (2010) in the land cover database series. Second, the WLOSS layer was overlaid onto the NWI layer to identify the NWI polygons that experienced wetland loss. Third, groupings of statically significant orthogonal (“rotated”) predictor variables were identified through principal component analysis in order to evaluate the significance of different variables that potentially increased wetland loss. Fourth, a logistic regression model with a binary variable was created using the coefficients of rotated groupings obtained via principal component analysis. Finally, the model’s strength of predictability and accuracy were analyzed.

4.2.1 Study Area

Sussex County, DE sits on the central-eastern portion of the Delmarva Peninsula, is 3,098 km^2 in area (See Figure 4.1). The county falls in the Coastal Plain Physiographic Province, and is adjacent to the Delaware Bay to the north and to the Atlantic Ocean to the east. The Sussex County economy has historically been the center of Delaware’s agriculture, poultry, and coastal tourism/recreation (DNREC, 2013). The county was chosen for this study because of its variation in land cover types, a complex spatial distribution of freshwater and non-tidal wetlands, and constant pressures from seasonal tourism, coastal development, sea level rise, and storm surges. The wetlands on the Peninsula are also managed with a variation of policies from a matrix of governmental entities, which includes the Delaware Department of Natural Resources and

Environmental Control (DNREC) and multiple jurisdictions of U.S. Army Corps of Engineers (USACE).

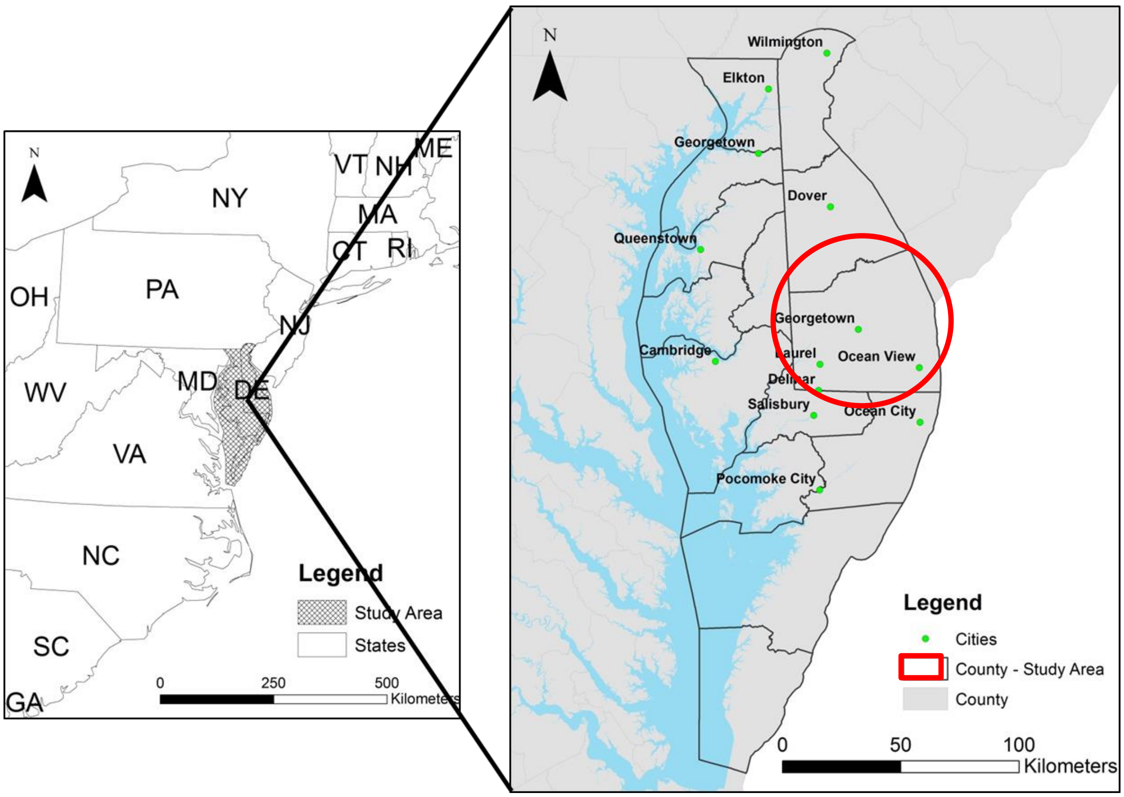

Figure 4.1 The Study Area – Sussex County, DE (highlighted in **Red**) on the Delmarva Peninsula (Source: US. Census Bureau)

4.2.2 Source Data

In a regional descriptive analysis of wetland trends and patterns, Tiner *et al.* (2011) found that major drivers of wetland change in the state of Delaware were related to the cultivation of land for agriculture, urbanization, and alterations in hydrogeomorphic conditions (*e.g.*, hydro periods, and limitations in wetland management policies). In order to define the binary, dependent variable of wetland loss wetland loss, the land use and land cover data from the NOAA – C-CAP were used to identify pixels (Table 4.1) that were classified as wetland cover in 1992, 2001, and/or 2010, to determine if they either maintained wetland cover classification, or were converted into a non-wetland cover (*e.g.*, agriculture, developed space, forest, open space, or water). These time periods were selected because C-CAP land cover data was available in 4 to 5 year increments from 1992 to 2010, and also maintained a constant methodology and classification scheme. This made it possible to explore the statistical relationships between wetland loss and anthropogenic and landscape drivers/stressors. Wetland loss (WLOSS), the binary response variable, served as the dependent variable, and was an indicator of wetland loss between 1992, 2001, and 2010. If wetland loss occurred, the pixel was reclassified 1; if the pixel classification did not change from wetland, the pixel was reclassified 0. The WLOSS pixels were then converted from a rasterized pixel to a vector point layer, and were also spatially joined with NWI polygons to identify NWI polygons that experienced wetland loss between 1992 and 2010.

In addition to direct extraction or alteration, wetlands respond to changes in landscape conditions and processes. In order to assess the drivers of wetland change, a number of landscape condition and local (proximity) variables were combined into a multi-layer, raster geodatabase. Similar to studies by Daniels and Cumming (2008) and Gutzwiller and Flather (2011), the independent variables were divided into two categories (local and landscape) of wetland drivers in order to account for the wetland conversion conditions and process that cross multiple spatial scales. Local variables (polygon-based variables) were derived with respect to the U.S. Fish and Wildlife Service NWI geospatial dataset(s) (Table 4.1). The suite of local variables included elevation (USGS) (Wetlands at lower elevations were expected to be more prone to conversion to water, and wetlands at higher elevations to be more prone to conversion to developed land covers) and slope (Wetlands at higher slopes were expected to be more prone to conversion). Elevation and slope (10 meter) for the WLOSS points was from the U.S. Geological Survey National Elevation Dataset. Three decadal annual precipitation data was obtained from the National Climatic Data Center 1981-2010 US Climate Normals data series.

Landscape variables were derived primarily from a hybrid 30-m land cover data set that comprised of NOAA's C-CAP (circa 1992 - 2010) (Table 4.1). As discussed in Daniels and Cumming (2008) distance surfaces add a dimension to the analysis that considers temporally and spatially dynamic variables. For example, wetland habitats closer to roads and urban centers may be more vulnerable wetland conversion than large patches of forested wetlands in rural floodplains. The predictor variables regarding distance were as follows: developed areas (C-CAP), agriculture (CBP), metropolitan statistical areas (US Census), shoreline (CBP), streets (CBP), floodplains (CBP), and open water (C-CAP; CBP). Impervious road and non-road (*e.g.*, buildings) data (10 meter) were obtained from the Chesapeake Bay Program. A binary indicator variable was also created to identify if wetland polygons and points of wetland loss were in tidal or non-tidal areas (Tidal wetlands were also expected to be at higher risk of conversion). The final suite of calculated variables (in all capital letters in Table 4.1) were calculated with respect to the designated buffers of 3,000m and 3,500m around NWI polygons. The buffers were determined by analyzing the z-score results from the Incremental Spatial Autocorrelation tool in ArcGIS, which identified the radius distance with the most intense clustering of NWI polygons .

Layer	Data Source	Description	Justification
Dependent Variables (abbreviation)			
Wetland change trajectory (WLOSS)	NOAA - Coastal Change Analysis Program (C-CAP); 30m Resolution; 1992, 2001, 2010; the National Wetland Inventory via the US Fish and Wildlife Service	Pixels were coded for no change (0), and change (1), and noise depending on whether they remained wetlands, were converted to another land cover. Any NWI polygon that contained 1 or more “change” pixels was identified as a WLOSS polygon	Binary Dependent variables; an NWI polygon that contained 1 or more “change” pixels was identified as a WLOSS polygon
Local Independent Variables			
Elevation Average Elevation (AVG_ELEV) Range of Elevation (RANGE_ELEV)	Digital elevation model (DEM) via the USGS National Elevation Data set; 10 m resolution	the average elevation at the specified buffer	The higher the elevation, the higher the probability of wetland conversion to developed land cover. The lower the elevation, the higher the probability of wetland conversion to agriculture or water
Average Slope (AVG_SLOPE) Range of Slope (RANGE_SLOPE)	Digital elevation model (DEM) via the USGS National Elevation Data set; 10 m resolution	Calculated from DEM and expressed as the median slope inside an 8-cell moving window; A pixel value is assigned to each point of wetland loss	The higher the slope, the higher the likelihood of wetland conversion due to runoff and erosion
Landscape Condition - Independent Variables			
Distance to Streets (DIST_STR)	USGS – Chesapeake Bay Program; vector	For each wetland patch polygon, the average distance of the wetland polygons to the closest street polygons	The smaller the distance from impervious land cover, the higher the potential for wetland conversion
Distance to Floodplain (DIST_FLD)	Chesapeake Bay Program Combination of floodplain boundaries from the FEMA – Flood Hazard Layer and select attributes from USDA - NRCS Soil Survey Geographic (SSURGO) Database; vector	For each wetland polygon, the distance to the floodplain boundary	The lower the distance to the floodplain, the higher the likelihood for wetland conversion to open water, agriculture near or adjacent to a stream system, or developed land cover on the coast (*e.g.* Ocean City, MD)
Distance to Metropolitan Statistical Areas (DIST_UAC)	US Census Bureau; 2010; vector	For each wetland polygon and each point (*e.g.*, pixel centroid) of wetland loss, the distance to the closest MSA polygon	The smaller the distance from impervious land cover, the higher the potential for wetland conversion
Distance to Shoreline (DIST_SHR)	USGS – Chesapeake Bay Program; vector	For each wetland polygon and each point (*e.g.*, pixel centroid) of wetland loss, the distance to the shoreline boundary	The smaller the distance from shoreline , the higher the potential for wetland conversion
Distance to Agriculture (DIST_AGR)	2012 DE Land Use Database; vector	For each wetland polygon and each point (*e.g.*, pixel centroid) of wetland loss, the distance to the nearest agricultural polygon	The smaller the distance from agriculture, the higher the potential for wetland conversion

Local Independent Variables (cont'd)			
Layer	Data Source	Description	Justification
Distance to Open Water (DIST_WAT)	Chesapeake Bay Program; National Hydrography Dataset (NHD); the National Wetland Inventory (NWI) ponds/lakes; vector	For each wetland polygon and each point (*e.g.,* pixel centroid) of wetland loss, the distance to the nearest water polygon	The smaller the distance from water, the higher the potential for wetland conversion
Distance to Developed areas (DIST_DEV)	Chesapeake Bay Program; Combination of NAVTEQ, impervious surface data sets, and state land cover datasets to identify developed land cover; vector	For each wetland polygon and each point (*e.g.,* pixel centroid) of wetland loss, the distance to the closest developed area polygon.	The smaller the distance from developed land cover, the higher the potential for wetland conversion
Change in Developed Area (CT_URBC)	NOAA – C-CAP; 30 m resolution	For each wetland buffer polygon, the change in the spatial area of developed land cover	The smaller the distance from developed land cover, the higher the potential for wetland conversion
Tidal Classification (TIDAL)	the National Wetland Inventory (NWI); vector	If a wetland polygon or a point (*e.g.,* pixel centroid) of wetland loss fell inside or intersected a tidal wetland polygon, it was classified as tidal (1). If it fell outside a tidal wetland polygon, then it was classified as non-tidal (0)	Tidal wetlands would have a higher the potential for wetland conversion due to inundation and erosion resulting from sea level, sinking, and storm surges
Land Cover Spatial Area (AREA_LC_YR) Proportion (PRCT_LC_YR) Percent Change (PRCT_LC_YR1_YR2)	NOAA – C-CAP (1992, 2001, 2010); 30 m resolution	The spatial area and percentage of each Anderson Level-I land cover that fell inside the specified buffer around select NWI polygons: developed, agriculture, rangeland, forest, water, wetland, and barren (hectare)	Larger quantities and higher percentages of developed, agricultural, and water land cover increase the probability of wetland conversion due development, agriculture, or inundation; Larger quantities and higher percentages of forest, rangeland, and barren land cover increase the probability of wetland conversion due to suburban development, agriculture, or ongoing wetland/soil excavation
Housing Units Total (SUM_THU_YR) Density(DEN_THU_YR) Average Density (AVG_DEN_YR)	US Census Bureau (1980, 1990, 2000, 2010); vector	For each wetland polygon, the total number of housing units in the block groups that intersect the specified buffer around the polygon; the density of housing units was calculated by dividing the total number of housing units by the spatial area of the buffer; the average density was calculated by dividing the number of housing units in the buffer by the number of wetland polygons in the buffer	Wetlands surrounded by higher quantities of housing units would have a higher probability of being converted into residential land uses like impervious surfaces or turf grass

Local Independent Variables (cont'd)			
Layer	Data Source	Description	Justification
Average Annual Precipitation (AVG_PRCP)	National Oceanic and Atmospheric Administration's (NOAA) National Climatic Data Center (NCDC) - 1981–2010 U.S. Climate Normals (m)	The average tri-decadal annual precipitation between 1981 and 2010	Higher quantities of precipitation and intensity storms would increase the probability for wetland loss due to runoff and subsequent inundation and/or erosion; Higher quantities of precipitation would increase the probability for wetland loss due to runoff and subsequent inundation and/or erosion
Impervious surface density (DEN_IMP)	Chesapeake Bay Program Impervious Road and Non-road layers, 10 m resolution	The density of impervious cover per the area of each buffer polygon; aggregated to 30m resolution (per hectare)	The higher the density of impervious land cover, the higher the potential for wetland conversion
Wetland permit density Quantity of General Permits (CT_GEN) Density of General Permits (DEN_GEN) Quantity of Individual Permits (CT_IND) Density of Individual Permits (DEN_IND)	United States Army Corps of Engineers (USACE) Maryland Department of the Environment (MDE) Delaware Department of Natural Resources and Environmental Control (DNREC)	The count and point density of each permit type from 1980 to 2010 (per hectare)	The higher the density of permits, the higher potential for wetland conversion; The higher the density of individual permits; the higher potential for wetland change over a larger area than general permits, because general permits tend to be smaller in scope and have minimal adverse impacts on wetlands on or near the project site

Table 4.1 The dependent variable (WLOSS) and proposed independent (predictor) variables, along with their theoretical justification for consideration as a driver of wetland change.

4.2.3 Statistical model building

The statistical analysis performed in R using functions related to variance inflation factors (VIFs) and principal component analysis (PCA). First, for each raster cell in the WLOSS geodatabase, the dependent variable (WLOSS) and its corresponding values for each of the proposed independent variables were extracted from the database (n = 8,003 total units). Second, a correlation matrix was created in R to graphically analyze potential relationships in the full suite of independent variables and to complement my literature based understanding of Sussex County's landscape dynamics and wetland management, trends, and patterns.

Third, VIFs were calculated in order to distinguish the multicollinearities between the independent variables and to assess the independent influence of each variable on the risk of wetland loss. A common rule of thumb that values of VIF that are greater than 10 could represent serious multicollinearity (Neter, 1996). However, variables with higher values do not always nullify the regression analysis results and are not required to be removed from the model (O'Brien, 2007; Gutzwiller & Flather, 2010). For this study, independent variables with an "infinite" VIF goodness of fit, were deemed to have high multicollinearity (*e.g.,* with one or more of the other variables) and were removed from the pool of independent variables. The VIF was recalculated until all of the independent variables had a goodness of fit less than was not infinite, or was deemed necessary. VIF was chosen over methods like stepwise regression, because it allowed the user to assess multicollinearity and to control the order of selection and removal of the independent variables.

Fourth, a PCA was performed to: 1) identify linear groupings of the independent variables that are uncorrelated with one another and 2) determine which groupings variables had the strongest relationships and influence of the variance of the probability of wetland loss. The set of landscape (driver) variables were rotated along orthogonal axes via the principal components analysis (PCA) and varimax rotation functions in R, which decreased multicollinearity and increased the ability to differentiate the independent variables that had the highest loadings in each grouping (Gutzwiller & Flather, 2010). VIF and PCA methods have been used together in numerous ecological modelling studies on predicting wetland vulnerability (McCauley *et al.*, 2013), the drivers of wetland conversion (Daniels & Cumming, 2008; Gutzwiller & Flather, 2010; Sanneke *et al.*, 2013). In order to address the limited ability to interpret PCA results, the VIF analysis was performed prior to the PCA. The end goals were to identify the groupings of the predictor variables and to rank each grouping by its proportion of loadings. After running the PCA with the varimax function, the similarities between variables high loadings (*e.g.*, infinite) in each of the four to five resulting PCA groupings were examined. Finally, a prediction model was constructed using the logistic regression model in R, and consisted of the remaining variables (n = 49).

4.2.4 Model validation and performance assessment

First, an ANOVA Chi-squared test was executed to evaluate the statistical significance and variances of the predictor variables of the model. Second, the model was validated with the testing data set that was set aside prior to generating the logistic regression model. Third, the model's suite of independent variables was evaluated by calculating the Akaike Information Criterion (AIC). When evaluating the overall quality of model, the AIC estimates the loss of information associated with the model's parameters (Anderson, 2010). AIC is statistical metric that has been used in numerous analyses related to ecological, climate change, land cover change modelling (Overmars *et al.*, 2003; Rutherford *et al.*, 2008; Ghazoul, 2010; Warren & Seifert, 2011).

Fourth, the Area under the curve (AUC) and the receiver operating characteristic (ROC) plots (See Figures 4.2) were generated to evaluate the accuracy and strength of the model's predictions. The ROC curve illustrates a model's probability of successfully predicting wetland loss (sensitivity; the x-axis) versus the probability of successfully predicting no change (specificity; the y-axis). With AUC ranging from 0 to 1, a model with a strong discriminative ability would have an out-bowed curve with a peak closer to the upper left corner of the plot (1,1), and would have an AUC closer to 1. On the other hand, a model with a random relationship, virtually no discriminative ability, between the predictor and the outcome would display a curve closer to a 45-degree angle, and would have an AUC closer to 0.5 (Daniels & Cumming, 2008; Steyerbery *et al.*, 2010) Finally, I analyzed the model coefficients used in the logistic regression equation, based on the orthogonally rotated variables, in order to decipher the strength and importance of the independent variables in predicting wetland loss.

4.3 **Results**

The results section consisted of summaries of the rotated principal component (PCA) analyses, the variance inflation factor (VIF) analyses, the logistic regression models (GLM), and the validation of each model's performance and accuracy.

Overall, the results provided support for the hypothesis that wetland loss in Sussex County, DE is predictable from landscape setting and conditions. The importance of landscape conditions in this instance provided beneficial insights into the trends, patterns, and processes of wetland loss. The landscape variables chosen for the model were moderately strong and statistically significant ($p < 0.05$) correlates of changes in wetland coverage. When interpreting principle components based on their coefficients, the relative order of influence of the drivers of wetland loss from highest to lowest was: residential development and wetland permits (RC1), landscape conditions with respect to water (RC2), impervious surfaces and urban areas (RC3), and precipitation and soil erosion (RC3). This order suggests a hierarchical structure of drivers in which the physical landscape (hydrogeomorphic conditions and processes) may provide the greatest explanation for landscape process that are more regional in nature, subsequently driving wetland loss at a larger (regional and non-political) and temporal (decadal) scales. The results support the hypotheses that models founded on landscape conditions and characteristics can be used to predict wetland loss. Residential development, urbanization, and permitting have driven wetland loss in urban, suburban, coastal, and upland areas throughout the county and Delmarva Peninsula. It also supports the hypotheses that wetlands at lower elevations and tidal conditions are at risk to loss to inundation due to factors and processes like sea level rise and erosion. The results also reiterate the importance of decreasing multicollinearity (*e.g.*, minimizing the number of independent variables prior to a PCA) in order to decrease prediction errors and to increase accuracy.

4.3.1 Predictor Variable Selection - Variance Inflation Factor

The independent variables for the tested models included RC1, RC2, RC3, and RC4 (See Table 4.2). These variables represented the principal components after magnifying the loadings by maximizing the sum of the variances of the squared loadings. The model's composite predictor variables included 49 independent variables that were queried using the variation inflation factor function. The model's AIC was 6,711. Also, the model only had three statistically significant composite predictor variables (RC1, RC2, and RC3) at the 95% confidence interval level ($p < 0.05$).

Model	*N*	AIC	AUC	RC1 Residential Development and permits (e^B)	RC2 Landscape -Hydrology (e^B)	RC3 Impervious Surface/ Urbanization (e^B)	RC4 Precip./ Soil Erosion (e^B)	Intercept (e^B)	Accuracy
1	213	6,711	0.64	0.07 (1.07)	-0.38 (0.66)	0.15 (1.16)	0.04 (1.04)	-1.77 (0.17)	0.42

Table 4.2 Model performance statistics computed with independent samples and model coefficients. The model included 49 independent variables determined by a variation inflation factor analysis. Key abbreviations include: AIC, Akaike information criterion; AUC, area under the curve for receiver operation characteristic (ROC) plots; and e^B, exponentiation of the coefficient. The only coefficients for the model that were significant at the P < .05 level were RC1, RC2, and RC3.

4.3.2 Principal Component Analysis

As previously discussed, the original set (n = 213) of predictor variables was condensed to 49 variables by performing the variance inflation factor analysis, a regression analysis to test for statistical significance, and PCA varimax rotation analyses to identify predictor variables with low communality values (See Table 4.3). The final PCA varimax rotation was run with the factors identified by the original distribution of PCA eigenvalues. This section summarizes the results of final PCA varimax rotation by illustrating the rotated structure matrices.

The final composite predictor variables were comprised of 49 variables (See Table 4.4). With respect to the binary dependent variable of wetland change, communalities were relatively high (greater than or equal to 0.50) except for variables related to distance to agriculture, tidal classification, average slope, the ranges of elevation and slope, and the acreages of Census block groups, which supports the concept that VIF and PCA reduce dimensionality while explaining the majority of the variance in the original variables (Table 4.2) (Daniels & Cumming, 2008). The rotated matrix (Table 4.3) had a factor of four components.

Model 1 (n = 49)			
Component	Eigenvalue	Variance (%)	Cumulative (%)
1*	16.24	45	45
2*	9.46	26	72
3*	6.46	18	90
4	3.60	10	100

Table 4.3 Loadings and percentage of variance accounted for by each of the four principal components extracted through principal component analysis Components with an asterisks (*) were considered statistically significant ($p < 0.05$).

One PCA with varimax rotation decreased the dimensionality of the 49 variables into four components with statistically significant eigenvalues greater than one, which explained 100% of the original variance (Table 4.2). Communalities were relatively high (above 0.50), except for the tidal classification (0.51), the distance to agriculture (3000m: 0.50; 3500m: 0.53), the majority of topographic elements (e.g., elevation and slope), and the acreage of Census block groups (3000m: 0.49; 3500m: 0.52). This confirmed that PCA reduces the dimensionality of multicollinear variables. It also suggested that the VIF analyses should be repeated to further reduces the dimensionality.

The rotated structure matrix (Table 4.3) shows that the first component (RC1) had high loadings from the quantity of general and individual permits (average of 0.82), the quantity of total housing units at 3000m and 3500m (an average of 0.83 between 1980 and 2010), and the average density of housing units at 3000m and 3500m (an average of 0.88 between 1980 and 2010) . The densities of individual and general permits increased as the density of the total housing units increased, reflecting the trend of residential development and wetland permitting. This second component was called the residential gradient.

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On the second component (RC2), the following variables loaded highly with loadings greater than or equal to 0.75: the distance to the floodplain at both scales (an average of -0.81), the average elevation (-0.82), the distance to the shoreline at both scales (an average of -0.82), and the distance to water at both scales (an average of -0.77). This component represented the hydrological and agricultural gradients of the county (RC2). These results reinforce the relationship with lower elevations and shorter distances to floodplains and shorelines, suggesting that wetlands became more frequently inundated or exposed to saline conditions, which could be correlated with wetland loss.

On the third component (RC3), the following variables loaded had moderate to high loadings that were greater than or equal to 0.66: the area of impervious surfaces at both scales (an average of 0.85) and the change in the developed urban areas at both scales with an average of 0.67. This component represented the urbanization gradient of the county (RC3). This result reinforces the relationship of the excavation and filling of wetlands to expand roadways and construct buildings.

The fourth component (RC4) was called the precipitation and erosion gradient, because the average soil erosion facts and average quantities of precipitation loaded highly at both scales with precipitation averaging 0.87 and the soil erosion factor averaging 0.84.

4.3.3 Model performance and coefficients

With respect to the model, all four predictor variables were statistically significant at $p < 0.05$. The order of influence of the drivers of wetland loss revealed that the residential development and permitting gradient (RC1) was the most important correlate of wetland loss, followed by the landscape conditions and hydrology gradient (RC2), urbanization gradient (RC3), and the precipitation and soil erosion gradient (RC4). The exponentiation of the logarithmic model coefficients (e^B) was calcuated. The magnitude of the model coefficients for RC1 was 1.07 (Table 4.2). As RC1 loading scores increased, the quantities of total housing units, the densities of total housing units, and quantities of wetland permits increased. The positive sign on the RC1 coefficient suggested that the likelihood of wetland loss increased as RC1 scores increased. For every unit increase of the residential development and permitting gradient (*e.g.*, increased construction of homes or alteration of the property), the likelihood of wetland loss was approximately equal ($e^B = 1.07$).

Results also revealed that landscape condition and hydrology affected the probability of wetland loss (B = -0.38). The likelihood of loss increased as the elevation and distances to the floodplain and shoreline decreased. For every unit increase in RC2, the likelihood of wetland loss was one-third less ($e^B = 0.66$). The third influential predictor in the model was RC3 (B = 0.15), the urbanization gradient. As the RC3 scores increased, the spatial area of impervious surfaces and area of land converted into developed land cover increased at the same magnitude ($e^B = 1.16$). The final predictor in the model was RC4 (B = -0.04) was the precipitation and soil erosion gradient. As the RC4 scores increased, the percentage of quantity of precipitation and the soil erosion factor increased at the same magnitude ($e^B = 1.04$), suggesting wetland loss may have been the result of precipitation events (e.g., flash flooding and seasonal flooding) and subsequent soil erosion from factors like unstable soil and increased streamflow.

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	Component				
Predictor	RC1 (Residential/ Permitting)	RC2 (Landscape/ Hydrology)	RC3 (Urban)	RC4 Precip/ Erosion	Commonalitie s
Tidal classification (1= tidal, 0 = non-tidal)	0.27	0.65	-0.04	0.15	0.51
Average distance to agriculture (3000m)	0.39	0.54	-0.24	0.06	0.50
Average distance to agriculture (3500m)	0.40	0.55	-0.23	0.07	0.53
Average distance to developed (3000m)	-0.13	0.49	-0.63	0.33	0.76
Average distance to developed (3500m)	-0.13	0.50	-0.62	0.34	0.77
Average distance to floodplain (3000m)	-0.16	-0.8	-0.19	-0.07	0.71
Average distance to floodplain (3500m)	-0.17	-0.82	-0.19	-0.07	0.75
Count of General Permits (#) (1980 -2010) (3000m)	0.82	-0.07	-0.06	-0.08	0.69
Count of General Permits (#) (1980 -2010) (3500m)	0.84	-0.07	-0.07	-0.09	0.72
Count of Individual Permits (#) (1980 -2010) (3000m)	0.82	0.01	-0.10	-0.04	0.69
Count of Individual Permits (#) (1980 -2010) (3500m)	0.83	0.01	-0.10	-0.04	0.70
Area Impervious Surface (ha)(3000m)	0.35	0.12	0.84	0.07	0.84
Area Impervious Surface (ha)(3500m)	0.35	0.14	0.85	0.08	0.86
Average Soil Kfactor (3000m)	0.02	0.10	0.00	0.83	0.70
Average Soil Kfactor (3500m)	0.03	0.08	0.02	0.84	0.71
Average precipitation (1981-2010) (3000m)	0.00	-0.10	0.10	0.87	0.78
Average precipitation (1981-2010) (3500m)	0.00	-0.10	0.09	0.87	0.78
Average elevation (m) (3000m)	-0.27	-0.82	-0.13	-0.03	0.76
Range of elevation (m) (3500m)	-0.12	0.05	0.38	0.09	0.17
Average slope (m) (3000m)	-0.17	-0.05	0.40	0.11	0.20
Range of slope (m) (3500m)	-0.05	0.10	0.37	0.09	0.16
Average distance to shoreline (m) (3000m)	-0.18	-0.82	-0.26	0.03	0.78
Average distance to shoreline (m) (3500m)	-0.18	-0.82	-0.26	0.03	0.78
Average distance to streets (m) (3000m)	-0.10	0.51	-0.59	0.37	0.76
Average distance to streets (m) (3500m)	-0.11	0.52	-0.59	0.39	0.79
Total housing units (#) (1980)(3000m)	0.84	0.22	0.32	0.02	0.86
Total housing units (#) (1990)(3000m)	0.86	0.24	0.32	0.00	0.90
Total housing units (#) (2000)(3000m)	0.83	0.25	0.36	0.00	0.87
Total housing units (#) (2010)(3000m)	0.79	0.30	0.38	0.02	0.86
Acreage of Block groups (3000m)	-0.25	-0.64	0.05	0.13	0.49
Avg. den. of housing units (per ha) (1980)	0.82	0.34	0.02	0.04	0.79
Avg. den. of housing units (per ha) (1990)	0.90	0.32	0.03	0.02	0.91
Avg. den. of housing units (per ha) (2000)	0.90	0.34	0.06	0.01	0.94
Avg. den. of housing units (per ha) (2010)	0.89	0.38	0.10	0.01	0.94
Total housing units (#) (1980) (3500m)	0.85	0.23	0.33	0.02	0.88
Total housing units (#) (1990) (3500m)	0.87	0.25	0.32	0.00	0.92

Predictor	Component				Commonalities
	RC1 (Residential/ Permitting)	RC2 (Landscape/ Hydrology)	RC3 (Urban)	RC4 Precip/ Erosion	
Total housing units (#) (2000) (3500m)	0.83	0.27	0.36	0.00	0.89
Total housing units (#) (2010) (3500m)	0.79	0.31	0.39	0.02	0.87
Acreage of Block groups (3500m)	-0.26	-0.66	0.08	0.12	0.52
Avg. den. of housing units (per ha) (1980)(3000m)	0.84	0.32	0.01	0.04	0.81
Avg. den. of housing units (per ha) (1990)(3000m)	0.90	0.30	0.01	0.01	0.91
Avg. den. of housing units (per ha) (2000)(3000m)	0.91	0.33	0.05	0.00	0.94
Avg. den. of housing units (per ha) (2010)(3000m)	0.89	0.37	0.08	0.01	0.94
Average distance to urban areas (m)(3000m)	-0.29	-0.41	-0.59	0.12	0.62
Average distance to urban areas (m)(3500m)	-0.30	-0.42	-0.59	0.13	0.63
Area of urban change (ha) (1980 to 2010)(3000m)	0.40	0.24	0.67	0.07	0.67
Area of urban change (ha) (1980 to 2010)(3500m)	0.41	0.26	0.68	0.07	0.71
Average distance to water (m)(3000m)	-0.27	-0.76	-0.24	0.02	0.71
Average distance to water (m)(3500m)	-0.28	-0.78	-0.24	0.02	0.75

Table 4.4 Rotated loadings and percentage of variance accounted for by each of the four statistical significant principal components extracted through principal component analysis.

After running the model, the performance and accuracy of predictor variable were calculated. As illustrated in Figures 4.2, the model had an AUC of 0.62.

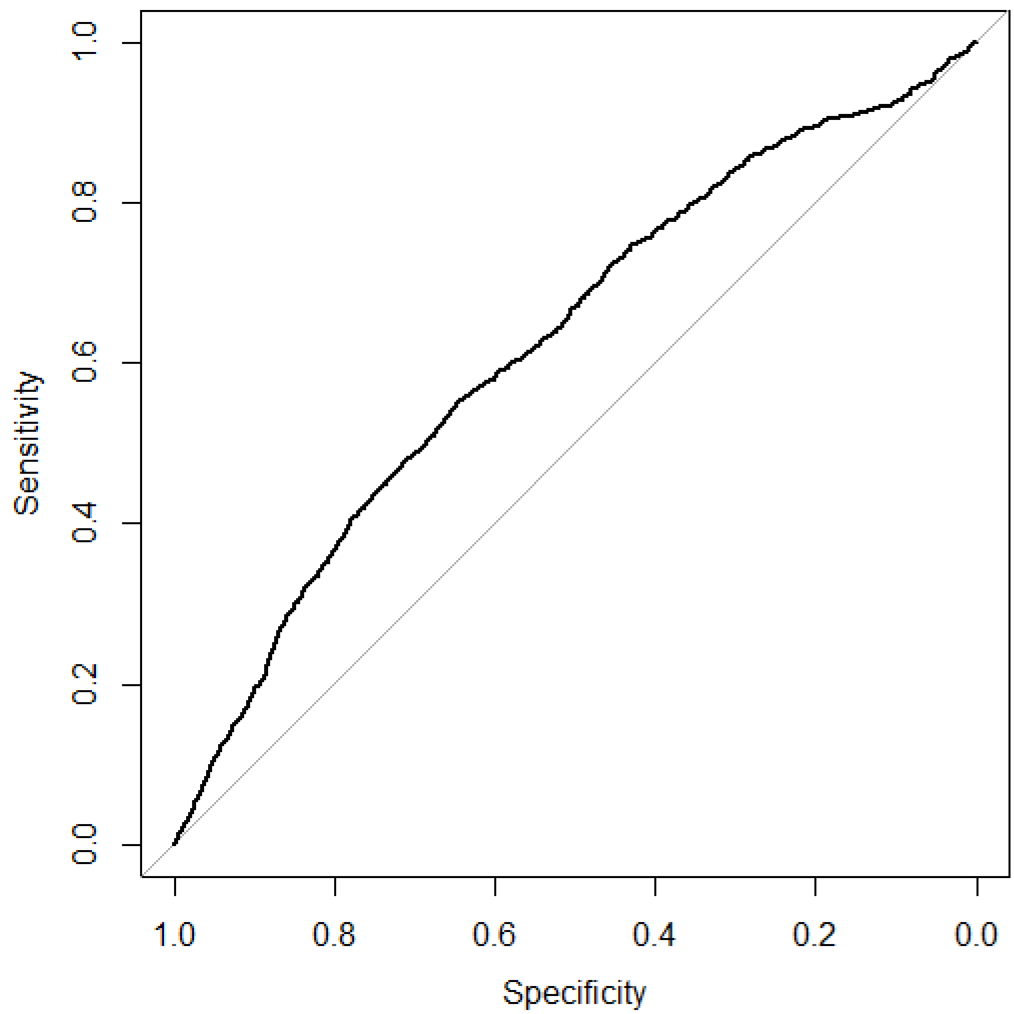


Figure 4.2 This plot reflected the Area under the Curve (AUC) (0.62) with specificity represented on the x-axis and sensitivity on the y-axis. The ROC curve illustrates a model's probability of successfully predicting wetland loss (sensitivity; the x-axis) versus the probability of successfully predicting no change (specificity; the y-axis).

4.4 **Discussion**

4.4.1 Key Drivers of Wetland Change

These results support the hypothesis that wetland change is predictable in Sussex County when accounting for residential development, permitted activities, and urbanization as the major drivers of wetland loss. According to the PCA results, the quantities of total housing units and wetland permits had a major influence on the probability of wetland loss. The strong relationships with landscape conditions and hydrology also reinforced the trend of the loss of estuarine wetlands to development, sea level rise, and erosion. Land cover change matrices and maps illustrated that development in and around Sussex County cities like Georgetown has been a product of suburban sprawl from urban areas like Salisbury, MD. Transportation corridors between Ocean City, Salisbury, and Dover have been the hotspots of development and roadway expansion, which has increased the need for infrastructure revitalization regarding stormwater, floodplain, and water quality management. The county has experienced increases in impervious cover due to development and roadway expansion as well as conversion of agriculture and wetlands into developed land cover. Previous studies on wetland change and modeling have also concluded that statistically significant relationships exist between wetland loss distances to urban areas, roadways, and agriculture (Daniels & Cumming, 2008; Gutzwiller & Flather, 2011).

The results also support the hypothesis that permitted activities like development and excavation for infrastructure also represent significant drivers of wetland loss in uplands as well as lowlands. It also reinforces how roadways like the US-13 corridor from northern DE to southern DE and US-9 that connects tourists to Rehobeth Beach continued to experience increases in impervious cover, and continue to stress wetland conservation. During the 30 year time period, planners and developers had to adapt to drivers of growth as well as changes in landscape conditions due to development as well as natural processes and climate change (*e.g.* increases in sea levels and surface runoff). Adaptations often resulted in the alteration or removal of wetlands in order to expand infrastructure (*e.g.*, storm walls, dredging, drainage) to reduce the negative impacts of the physical processes on residential and commercial structures. Overall, the results suggest a need for future monitoring and assessments of wetland change due to agriculture, silviculture, and urban growth. Time series and regional land cover change analyses as well as incentivized reporting from farmers and county officials. Despite improvements in tracking trends and patterns of wetland loss, policymakers and planners need a clearer understanding of the importance, variance, and magnitudes of the drivers of wetland loss at different scales.

4.4.2 Assessment of the model

The high quantity of multicollinear variables made it challenging to: 1) decipher the commonalities of the variables in each principal 2) determine the appropriate variables to remove after analyzing the VIFs, and 3) accurately predict wetlands that would experience loss as well as wetlands that would not experience loss. Daniels and Cumming (2008) used two similar methods of predicting wetland conversion and resulted in AUC values of 0.79 and 0.81, respectively. Gutzwiller & Flather's (2011) modelling of wetland loss in the U.S. Forest Service's Southern Region consisted of 10 to 13 variables and resulted in an average AUC of 0.72. Different objectives, landscape conditions, methodologies, the resolution or scale of source data, modeling parameters, and performance indicators all complicated the ability to compare and contrast studies. Despite these limitations, the results fall within the moderate (good) range of performance for modeling the risk of wetland loss.

4.4.3 Implications to Wetland Management and Smart Growth

The geostatistical analysis of the drivers of wetland change and permitting suggested that numerous associations have relevance to urban and community planning, infrastructure, wetland conservation, agricultural easements, and floodplain management. First, wetlands near urban centers, neighborhoods, roadways, and waterbodies are vulnerable to degradation and loss. Federal, state, and local officials must create policies that counteract negative impacts from permitted activities that threaten palustrine, forested wetlands. Despite the implementation Section 404 permitting programs and wetland conservation policies like Swampbuster, wetland change has continued to occur, because of exempted activities and limited or inconsistent monitoring and enforcement initiatives. The risk of wetland loss has manifested from traditional, high impact, large scale projects like ditching, dredging and excavation. Conservation planning must also address the long-term, cumulative impacts resulting from an increase in the frequency of small-scale projects. For example, residential and infrastructure expansion magnify the influence of landscape conditions like slope and the erosion factor. States and localities must continue to implement conservation efforts like requiring buffers around wetlands and critical areas.

4.5 Conclusion

Results supported the hypothesis that wetland change can be driven by regional, moderate scale processes related to trends of urban development and changes in hydrological conditions. Addressing challenges like autocorrelation and multicollinearity improved my model's performance and accuracy. This study also illustrated the complexity of identifying predictors of wetland change. Local and landscape conditions are interwoven with natural processes like hydrologic connectivity and climate change that occur at the macro-scale, but require data sets that are maintained at finer resolutions or at local scales. In order to improve the performance, accuracy, goodness of fit, and percentage of explanation, future research should explore implementing statistically and spatially sound methods of random sampling and scale selection. By geostatistically and quantitatively examining and mapping wetland change, scientists can provide policymakers with a clearer scientifically substantiated picture of critical areas that vulnerable to wetland change as well as areas ripe for sustainable restoration. Wetland management plans must be founded on with regional geospatial data and analyses rather than solely localized, site specific projects that may indirectly miss regional temporal and spatial trends and patterns. While this study only explored wetland change statistically, the results can be used as a template for future statistical models and mapping tools that evaluate the impacts of physical and anthropogenic drivers on wetland vulnerability, resilience and functionality.

5 CHAPTER 5: CONCLUSION

This research evolved from a desire to improve the understanding of how physical, socioeconomic, and political factors drive wetland change on a regional scale over a 30 year time period. What seemed like straight forward tasks, to determine the quantities, primary locations, and predictors of wetland loss, proved to very complex as spatial and temporal trends, patterns, and impacts of land cover change and wetland permitting records on the Delmarva Peninsula have rarely been examined together. The Delmarva Peninsula was used for this study because of its variation in land cover types, heavy concentrations of freshwater and tidal wetland cover, trends of population growth, active participation in the Clean Water Act - Section 404 wetland permitting program, and its unique position between the Chesapeake Bay, the Delaware Bay, and the Atlantic Ocean.

Wetland inventories and permitting databases on the Delmarva Peninsula were not historically structured to be used as tools to measure impacts from physical and anthropogenic processes, plus estimates of regional wetland change were often assessed by episodic, localized field studies, which limited systematic and regional wetland change analyses. The acquisition of Section 404 permit records from federal and local agencies provided an opportunity to provide descriptive analyses and maps on wetland change and permitting, and a geostatistical model on the predictability of wetland change. These deliverables could improve the understanding of wetland change and the efficiency of federal, state, and local wetland permitting policies on a regional scale. As literature on historic and future wetland change was explore, the following questions were developed:

1. What were the spatial areas and locations of wetland change from 1984 to 2010 on the Delmarva Peninsula measured by existing geospatial data sets?
2. What physical and anthropogenic drivers of land use and land cover change were correlated with wetland loss on the Delmarva Peninsula?
3. What information did the spatial and temporal distribution of wetland permits and wetland loss patterns provide regarding the influence of wetland change drivers and the impacts of the wetland permitting system?
4. What wetlands, watersheds, and counties were most vulnerable to wetland loss due to physical, socioeconomic and policies that drive wetland change?

This research took a tiered approach to quantifying the spatial and temporal patterns of wetland change and the impacts of permitted activities and natural processes on the Delmarva Peninsula. Chapter 2, assessed wetland change on the Delmarva Peninsula from 1984 to 2010 using regional land cover data sets. Chapter 3 analyzed spatial and temporal trends and patterns of Section 404 permitting on the Delmarva Peninsula. Chapters 2 and 3 examined wetland change and permitting at a broad, regional scale, while Chapter 4 concentrated on one county, Sussex County, DE, by examining and prioritizing predictors of wetland change. Finally, Chapter 4 using a geostatistical model assessed the predictability of wetland change in in Sussex County, DE with respect to local conditions and landscape context. Below, the major findings from each chapter are summarized, and are followed by a discussion on priorities for future research.

Chapter 2, focused on analyzing moderate resolution (30 meter) regional land cover data sets analyzed to: 1) examine the spatial agreement of national and state land cover data sets along with the National Wetland Inventory (NWI), 2) to quantify historical wetland changes on the Delmarva Peninsula at multiple spatial scales between 1984 and 2010, 3) to identify differences in the spatial area of wetland change and discuss the source of and implications for these differences, and 4) to investigate the extent to which drivers of wetland change can be identified using existing land cover data sets (LCDs). In the past, the monitoring of wetland change on long term temporal and larger spatial scales has been limited due to the high cost of systematically acquiring high resolution imagery and wetland feature data from site studies. Studies with respect to wetland change have mainly occurred over small (site-specific) scales or at large (multi-state) scales that could not be used to link trends of wetland change to county or regional wetland management and urban development strategies due to the coarse resolution of source imagery or outdated field data. This study quantified the spatial area and percentage of wetland change from 1984 to 2010 at 4 to 8 year intervals of the LCDs and identified primary locations of wetland loss over the peninsula which stretches through 14 counties and 3 Mid-Atlantic States.

Results showed data from the National Oceanic and Atmospheric Administration – Coastal Change Analysis Program and the Chesapeake Bay Land Cover Data Series had the highest percentage of spatial agreement; the regional LCDs had an average 75% agreement with state LCDs along with a 76% agreement with the NWI. Findings also showed that most of the statistically significant changes in the wetland cover in the study area were observed in the time period between 1992 and 2001, while a loss decreased from 2001 to 2010. From 1984 to 2010, the peninsula experienced lost approximately 9,000 hectares of wetlands. Sussex County, DE experienced the most wetland loss. The second tier of counties (Dorchester, Wicomico, and Worcester, MD) that experienced wetland loss were all located on the southern portion of the peninsula adjacent to the Chesapeake Bay, and had a combined area of loss greater than the total acreage of wetland loss in the remaining 10 counties, excluding Sussex County, DE. And, the hot spots of wetland loss were located inside or on the fringe or urbanized areas, suburban residential areas, and on coastal metropolitan and natural areas. However, looking at specific parcels impacted by activities (*e.g.*, agriculture) exempt from wetland management regulations like Section 404 of the Clean Water Act may provide insight into the commonalities and drivers of the hot spots of wetland loss. City planning strategies like buffering and agricultural easements must be based on a clear understanding wetland change trends and patterns in order to implement effective policies to control flooding and to prevent the fragmentation of wetlands that absorb surface runoff and nutrients from agriculture.

Chapter 3, focused on a time series analyses of wetland change, permitting, and their drivers. Results showed wetland permitting and loss increased from 1984 to 2006 and decreased from 2006 to 2010, a trend that was constant in 12 out of the 14 counties, and also temporally complemented the Housing Market Boom. The majority of wetland loss occurred with palustrine wetlands, and the majority of permits were distributed for activities that had the potential to negatively impact estuarine wetlands. State policies implemented along with the housing recession appeared to have contributed to the decline in permitting as well as in wetland loss, except in the coastal areas in VA vulnerable experiencing sea level rise. The ability to quantify and compare the spatial and temporal distribution of wetland permits and change at a natural (*e.g.*, hydrological units) and an anthropogenic (*e.g.*, US Census Bureau minor civil divisions) scale can help policymakers, emergency managers, and developers design and implement policies and projects that incorporate a clear understanding of the cumulative impacts of urban development, agriculture, and climate change.

In the past, the high cost of obtaining and contextualizing multi-jurisdictional wetland permitting databases over large scales has been a restricting factor in expanding the study of the relationship between wetland change and permitting in regions that cross major watersheds and socio-political boundaries. Studies in this area have mainly occurred on a national level, in the Gulf Coast region, or in select subwatersheds of the Chesapeake Bay Watershed, and have focused primarily on compensatory mitigation measures rather than linking permitting trends and patterns with impacts on the spatial distribution of wetlands. This study quantified the number, percentage, and density of wetland permits on an annual and 4 to 8 year intervals in accordance to LCD, which was compared to similar calculations regarding wetland change, and tested for correlations with indicators representing potential drivers of wetland change permitting at cumulative, state, county, minor civil division, and hydrological unit scales. Findings showed how permitting and loss followed trends and patterns of development were directly related to development trends with respect to with statistically significant correlations between the densities of housing units, the area of impervious roadways, the area of wetland loss. However, looking at the patterns from drivers outside of urbanization may provide insight into the driving forces behind permitting and subsequent loss of wetlands. For example, wetland permitting in the 2 counties in VA continued to experience wetland loss from 2006 to 2010, while the other counties in MD and DE saw a decrease in wetland loss. These 2 counties were dominated by coastal communities and experienced percentages of wetland loss 3 to 4 times higher than other counties on the peninsula. These characteristics may be linked to the landscape conditions (*e.g.*, floodplain) of the counties, which introduced the notion that permitted activities were not only the driving forces behind loss, but that climate change (*e.g.*, sea level rise) and landscape conditions were also the driving forces behind wetland permitting. The spatial area of the impact of permitted activities and subsequent compensatory mitigation measures were not considered in relating wetland permitting to loss in this study, further, previous studies have shown that states have been unable to meet "No Net Loss" goals despite implementing measures like increasing restrictions on permitted activities and amending building and zoning codes to reduce wetland disturbance and extraction. To quantify the impact of agricultural and silvicultural activities and sea level rise, future research could combine GIS data like agricultural easement records with other remote sensing based data sets like the US Department of Agriculture - Crop Data Layer, the NOAA Sea level Rise Viewer, and the Chesapeake Bay Program Phase 6 Land Use Dataset.

Chapter 4 provided insights into the predictability of wetland change vulnerability in conjunction with anthropogenic and physical drivers with respect to local wetland conditions and surrounding landscape context. Using principal component analysis (PCA) and spatial autocorrelation analysis tools to identify the most influential groupings of predictor variables related to socioeconomic, wetland permit, land cover, topographic, hydrogeomorphic, and climate conditions, a series of logistic regression models were developed to compare the predictability of wetland change. The different suites of predictor variables and subsequent component groupings were created using regional and state LCDs, wetland permitting records, climate data. Results revealed the following gradients of the predictor variables by order of influence: residential development and permitting, the landscape and hydrological conditions, urbanization, and precipitation and erosion This reinforced previous research that identified development, agriculture, sea level rise and topography as the primary drivers of wetland loss. These results also coincide with previous studies that have shown how the accuracy and area under the curve calculations have been improved by reducing the quantity of multicollinear predictor variables (Daniels & Cummings, 2008). Future research could use the model developed in this study as a template to expand future analysis to include more predictor variables related to the socioeconomics (*e.g.*, income and property value), agriculture (*e.g.*, crop type), and climate (*e.g.*, drought and temperature) context surrounding the wetlands.

Future research priorities include the continued monitoring of wetland change and permitting over space and time as well as advances in modelling to assess the impact of physical processes and anthropogenic activities. The NWI, regional and state land use and land cover data sets, and wetland permitting records have provided critical information on wetland management and change since the early 1980s and the continuation and enhancement of all of this data is important to assessing future changes in the spatial distribution and functionality of wetlands. While the research presented in this dissertation focused mainly on wetland loss, wetland gain due to conservation and restoration initiatives are also very important to understanding the effectiveness of the implementation and enforcement of wetland management policies. Creating spatially and temporally accurate geostatistical analyses and maps from high resolution LCDs and wetland permitting records with attributes that reflect the spatial area of wetlands impacted by permitted activities can improve the understand of the drivers and impacts of wetland change (Brody *et al*., 2008; Daniels & Cummings, 2008). Future NWI-plus coupled with higher resolution LCD series will be key in expanding analyses of wetland change to larger regions like the Chesapeake Bay watersheds and understanding how sub-regions with different wetland management, smart growth, and climate change policies across a variation of physiographic and climate regions can be cooperatively reshaped so that more states and counties can set, meet, and sustain “No Net Loss” goals, while creating comprehensive plans that consider critical wetland areas that are vital to flood control, nutrient reduction, and biodiversity.

APPENDIX A

Breakdown of area and percentages of wetland change in the study area portioned by county and state from 1984 to 2010. (Note: The statistics are according to CBLCD from 1984 to 1992 and C-CAP from 1992 to 2010.) State percentages of wetland change were calculating by dividing the total area of change for the state by the total area of change for the entire study.

	1984 - 1992		1992 -1996		1996 - 2001		2001 - 2006		2006 - 2010	
County	ha	% diff	ha	% diff	ha	% diff	ha	% diff	ha	% diff
Kent, DE	-3.6	-0.03%	-198.1	-0.42%	-59.6	-0.13%	-299.1	-0.64%	-36.8	-0.08%
New Castle	-37.3	-0.53%	-70.3	-0.39%	14.1	0.08%	-213.5	-1.19%	-6.4	-0.04%
Sussex	-38.3	-0.17%	-40.6	-0.07%	-1,454.5	-2.43%	-778.9	-1.33%	-8.4	-0.01%
Total for Delaware	-79.1	-0.19%	-309.0	-0.25%	-1,499.9	-1.20%	-1,291.5	-1.05%	-51.6	-0.04%
Caroline	-27.0	-0.09%	-34.5	-0.24%	17.0	0.12%	-108.7	-0.75%	19.2	0.13%
Cecil	-8.8	-0.06%	-15.8	-0.29%	27.7	0.51%	-137.6	-2.52%	-16.2	-0.30%
Dorchester	-36.8	-0.07%	-567.8	-0.81%	-75.2	-0.11%	-653.7	-0.94%	191.6	0.28%
Kent, MD	-24.1	-0.15%	-1.8	-0.02%	26.9	0.27%	-78.7	-0.78%	-2.8	-0.03%
Queen Anne	-85.3	-0.32%	3.4	0.02%	24.3	0.15%	-107.8	-0.68%	-44.5	-0.28%
Somerset	122.4	0.17%	-218.2	-0.50%	-411.6	-0.94%	-213.5	-0.49%	286.5	0.66%
Talbot	-69.4	-0.36%	-0.3	0.00%	41.8	0.35%	-96.2	-0.81%	21.6	0.18%
Wicomico	3.0	0.01%	-81.3	-0.28%	-952.8	-3.34%	-321.3	-1.17%	171.8	0.63%
Worcester	15.9	0.04%	-686.1	-1.23%	-314.6	-0.57%	-760.0	-1.39%	298.9	0.55%
Total for Maryland	-110.2	-0.03%	-1,602.3	-0.62%	-1,616.6	-0.63%	-2,477.4	-0.98%	926.1	0.37%
Accomack	0.3	0.00%	-530.8	-0.88%	300.3	0.50%	-507.8	-0.85%	-12.1	-0.02%
Northampton	-0.5	0.00%	-44.6	-0.20%	-163.0	-0.72%	-232.5	-1.03%	-51.0	-0.23%
Total for Virginia	-0.3	0.00%	-575.5	-0.69%	137.3	0.17%	-740.2	-0.90%	-63.1	-0.08%
Total	-189.5	-0.04%	-2486.7	-0.54%	-2,979.2	-0.64%	-4,509.2	-0.98%	811.4	0.18%

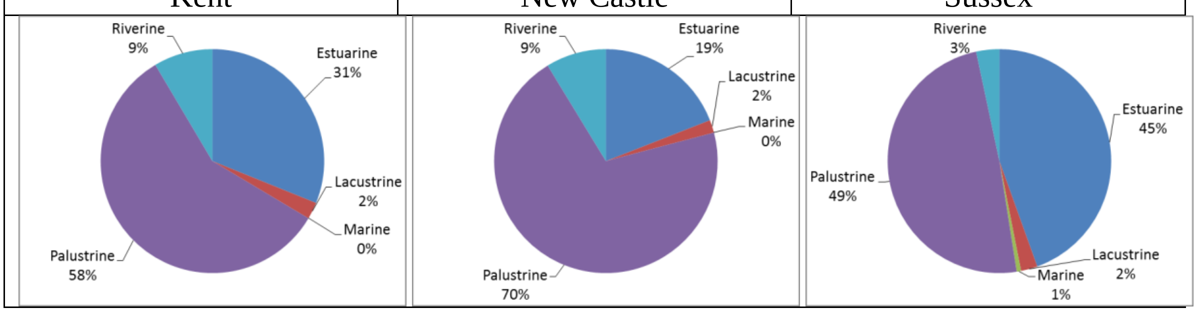
APPENDIX B

State by state tables and graphs Section 404 permits by nearest wetland system type from 1980 – 2010

	LOP	Nationwide	General	Individual	State	Total	% of Total
Estuarine	11	240	151	0	402	402	
	2.7%	59.7%	37.6%	0.0%	100.0%		1.7%
Lacustrine	1	21	4	0	26	26	
	3.8%	80.8%	15.4%	0.0%	100.0%		0.1%
Marine	0	1	3	0	4	4	
	0.0%	25.0%	75.0%	0.0%	100.0%		0.0%
Palustrine	10	499	160	0	669	669	
	1.5%	74.6%	23.9%	0.0%	100.0%		2.9%
Riverine	1	53	14	0	68	68	
Total	23	814	42	290	0	1169	0.3%
	2.0%	69.6%	3.6%	24.8%	0.0%		5.0%

Table 3.4. Delaware Section 404 permits by nearest wetland system type: 1980 – 2010. Within the area, 321 permits lacked digital NWI data, were duplicated and/or fell outside of the 400m buffer and subsequently were not included. The average distance from permit to NWI wetland = 70.0 m and median distance from permit to NWI wetland = 30.9 m. 200 permits were included in this portion of the analysis but excluded from the rest of the study due to the lacking of latitude and longitude attribute values.

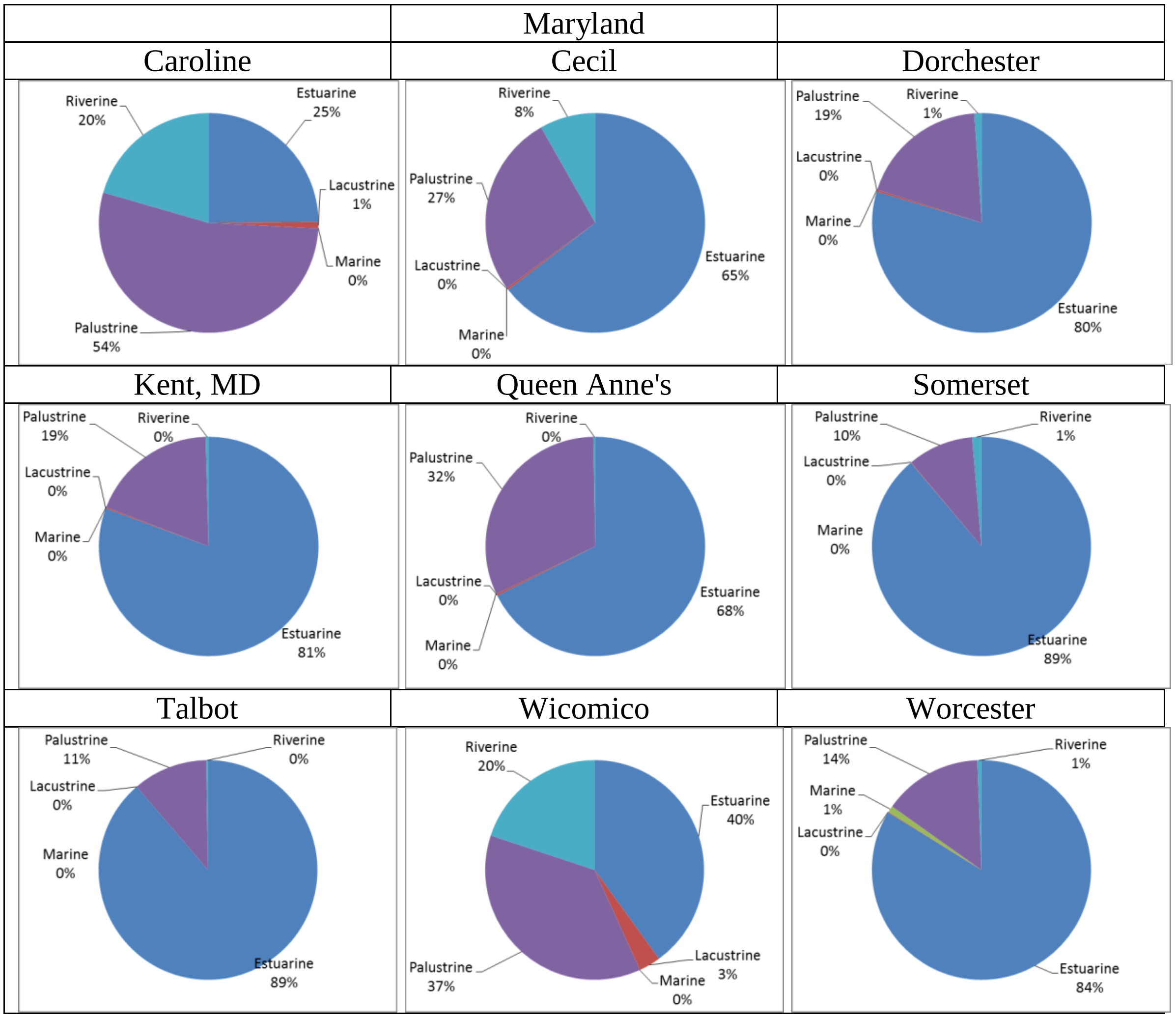
	Delaware	
Kent	New Castle	Sussex



Study Area's Section 404 permits issued in DE by county and nearest wetland system type: 1980 – 2010. Note –The data excludes 200 state programmatic permits issued in DE.

Type	LOP	Nationwide	General	Individual	State	Total	% of Total
Estuarine	587	1,296	12,158	2,407	14,041	16,448	
	3.6%	7.9%	73.9%	14.6%	85.4%		70.6%
Lacustrine	2	4	25	29	31	60	
	3.3%	6.7%	41.7%	48.3%	51.7%		0.3%
Marine	21	9	31	0	61	61	
	34.4%	14.8%	50.8%	0.0%	100.0%		0.3%
Palustrine	179	628	2,629	706	3,436	4,142	
	4.3%	15.2%	63.5%	17.0%	83.0%		17.8%
Riverine	21	30	336	102	387	489	
	0.5%	0.7%	8.1%	2.5%	9.3%		2.1%
Total	810	1,967	12,908	2,281	3,234	17,966	
	25.0%	60.8%	399.1%	70.5%	100.0%		77.1%

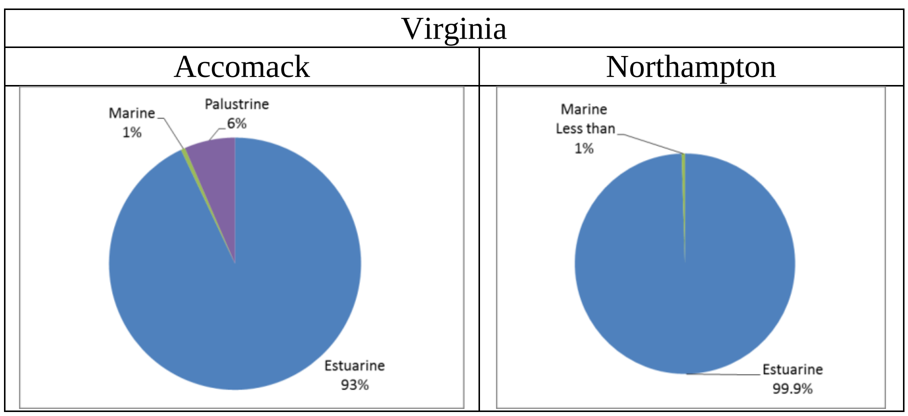
Maryland Section 404 permits by nearest wetland system type: 1980 – 2010. Within the area, 2,104 permits lacked digital NWI data and/or fell outside of the 400m buffer and subsequently were not included. The average distance from permit to NWI wetland = 42.3 m and median distance from permit to NWI wetland = 10.3 m.



Study Area's Section 404 permits issued in MD
by county and nearest wetland system type: 1980 – 2010.

Type	LOP	Nationwide	General	Individual	State	Total	% of Total
Estuarine	2	59	109	702	170	872	
	0.2%	6.8%	12.5%	80.5%	19.5%		3.7%
Marine	0	2	1	2	3	5	
	0.0%	40.0%	20.0%	40.0%	60.0%		0.0%
Palustrine	0	4	18	25	22	47	
	0.0%	8.5%	38.3%	53.2%	46.8%		0.2%
Total	2	65	128	729	195	924	
	0.2%	7.0%	13.9%	78.9%	21.1%		4.0%

Virginia Section 404 permits by nearest wetland system type: 1980 – 2010. Within the area, 1,455 permits lacked digital NWI data, were duplicated, and/or fell outside of the 400m buffer and subsequently were not included. The average distance from permit to NWI wetland = 45.5 m and median distance from permit to NWI wetland = 7.6 m.



Study Area's Section 404 permits issued in VA
by county and nearest wetland system type: 1980 – 2010.

APPENDIX C

Correlation analysis of changes in wetland permitting density and changes in acreage of agricultural, developed, and natural land cover

Spearman correlation coefficient test for statistical significant relationships between changes in the density of wetland permitting and the acreage of developed, natural, and agricultural land cover classes between 1980 and 2010. The statistical significance is represented by the number of asterisks (*): $p < .05$ (*), $p < .01$ (**), $p < .001$ (***).

			Developed Land Cover			
Permitting Period	1984	1992	1996	2001	2006	2010
1980 – 1984 Coefficient	-0.132	-	-	-	-	-
Significance	0.127	-	-	-	-	-
1985 – 1992 Coefficient	-0.142	-0.181	-	-	-	-
Significance	0.100	0.036*	-	-	-	-
1993 – 1996 Coefficient	-	0.148	0.134	-	-	-
Significance	-	0.088	0.121	-	-	-
1997 – 2001 Coefficient	-	-	0.129	0.124	-	-
Significance	-	-	-0.135	-0.150	-	-
2002 – 2006 Coefficient	-	-	-	-0.074	-0.085	-
Significance	-	-	-	0.392	-0.074	-
2007 – 2010 Coefficient	-	-	-	-	-0.089	-0.089
Significance	-	-	-	-	0.305	0.304
			Natural Land Cover			
1980 – 1984 Coefficient	-0.134	-	-	-	-	-
Significance	0.121	-	-	-	-	-
1985 – 1992 Coefficient	-0.345	-0.303	-	-	-	-
Significance	0.000**	0.000**	-	-	-	-
1993 – 1996 Coefficient	-	-0.114	-0.116	-	-	-
Significance	-	0.187	0.179	-	-	-
1997 – 2001 Coefficient	-	-	-0.235	-0.235	-	-
Significance	-	-	0.006**	0.006**	-	-
2002 – 2006 Coefficient	-	-	-	-0.277	-0.280	-
Significance	-	-	-	0.001**	0.001**	-
2007 – 2010 Coefficient	-	-	-	-	-0.295	-0.297
Significance	-	-	-	-	0.001**	-0.295
			Agricultural/Cultivated Land Cover			
1980 – 1984 Coefficient	-0.111	-	-	-	-	-
Significance	0.200	-	-	-	-	-
1985 – 1992 Coefficient	-0.233	-0.232	-	-	-	-
Significance	0.007**	0.007**	-	-	-	-
1993 – 1996 Coefficient	-	-0.203	-0.205	-	-	-
Significance	-	0.018*	0.017*	-	-	-
1997 – 2001 Coefficient	-	-	-0.311	-0.315	-	-
Significance	-	-	0.000**	0.000**	-	-
2002 – 2006 Coefficient	-	-	-	-0.334	-0.336	-
Significance	-	-	-	0.000**	0.000**	-
2007 – 2010 Coefficient	-	-	-	-	-0.358	-0.356
Significance	-	-	-	-	0.000**	0.000**

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