Donald R. Belk. EVALUATION OF HYDROLOGIC RESTORATION IN THE ALLIGATOR SWAMP: A WATER BALANCE APPROACH. (Under the direction of Dr. Jonathan D. Phillips) Department of Geography and Planning, November, 1990.

Measurement of the water balance is essential to evaluating the hydrologic component of a wetland ecosystem. Furthermore, water budgets can provide an estimation of inputs and outputs as a basis for predicting impacts to the wetland.

This thesis attempts to evaluate the impact of hydrologic changes that occurred after the installation of drainage control structures in canals within the Alligator Swamp. Three categories of study sites were evaluated. The first category consisted of former Atlantic white cedar (<u>Chamaecyparis thyoides</u>) wetlands that were logged during the mid-1980s. These logging sites were denoted as "post-altered" because they were located within an area where drainage control structures and canal backfilling were being used to restore the hydrology to pre-disturbance conditions. The second category, "altered", consisted of similar logging sites where drainage canals remained unimpeded. The third category, an existing stand of white cedar, represented a "natural" area. Comparisons were made of the water balances from each category. Changes in soil moisture and water table elevation fluctuations were the components of the water balance measured in the field.

The research goal was to determine whether the hydrologic regime in the post-altered area was more similar to the natural or to the altered study areas. A hydrologic model that forecasts changes in the water balance following artificial drainage was used in conjunction with the field measurements to assess the impact of the hydrologic restoration.

Results of the model revealed that the hydrology of the post-altered area closely approximated that of the natural area. This was attributed to the model's incorporation of surface subsidence and the loss of surface moisture from drainage. However, results of the water balance measurements showed that no significant differences existed between the altered and post-altered sites. Abnormally high precipitation during the study period and the loss of drainage canal conveyance capacity were two factors responsible for the lack of differences between the two hydrologically-distinct study area. Furthermore, the loss of conveyance capacity resulting from the natural deterioration of drainage canals within the study area has led to the recovery of the natural hydrologic regime.

EVALUATION OF HYDROLOGIC RESTORATION
IN THE ALLIGATOR SWAMP:
A WATER BALANCE APPROACH

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CHAPTER I

INTRODUCTION

Scientific research into wetland functions and values has changed the public's perception of wetlands. Once viewed as miasmatic wastelands, these areas are now highly regarded for their economic, aesthetic, and biological properties (Greeson et al, 1979; Bardecki, 1984; Farber and Costanza, 1987). As the study of wetlands has progressed, the need persists for further research into wetland hydrology. Hydrologic processes are the primary influence on wetland ecology and maintenance, yet the interactions between hydrology and wetland functions are not fully understood (Carter, 1986; Mitsch and Gosselink, 1986).

Measurement of the water balance is essential to evaluating the hydrologic component of any ecosystem. Furthermore, water budgets provide a first approximation of inputs and outputs as a basis for hydrologic models and predictions of impact (Carter, 1986). Wetland water budgets are particularly important since investigations of sedimentary inputs and outputs to wetlands are dependent upon water movement. Despite this need, many previous evaluations of wetland water budgets have either failed to measure all components, or have not used site-specific data (LaBaugh, 1986). The need exists, therefore, for a field-oriented hydrologic study in which care is taken in measuring all components of the water balance. This is concurrent with the need for expanded research and basic data collection which focusses upon wetland hydrology and its relation to other functions (Carter, 1986).

This thesis involves the compilation of water budgets for the purpose of examining the impact of hydrologic restoration in the Alligator Swamp of North Carolina.

CHAPTER II

PURPOSE STATEMENT

The purpose of this thesis is twofold. First, it will compare the variations in water table fluctuations and soil moisture within a regenerated, second growth stand of Atlantic white cedar and both canalized (altered) and hydrologically restored (post-altered) clear-cut white cedar bogs. Secondly, changes in water table levels will provide data for the compilation of water balances to determine the impact of hydrologic restoration on the water availability in post-altered wetlands. The goal of this research is to determine whether the hydrologic regime in the post-altered wetlands is more similar to the undrained or to the altered (canalized) study areas.

Additionally, this thesis will be the foundation of long-term research on the effects of hydrologic restoration and the success or failure of the regeneration of Atlantic white cedar (*Chamaecyparis thyoides*) in the Alligator Swamp (Noffsinger and Belk, 1989).

CHAPTER III

GEOGRAPHICAL CONTEXT

The analysis of changes in water regimes associated with natural, altered, and post-altered wetlands can be placed in the context of at least two major traditions or subdisciplines of geography. First, the project is a study in hydrology, which is an important subdiscipline of physical geography (Gregory, 1985). Second, the setting of the study - wetlands altered by human activity - places this work in the tradition of man-land or man-environment studies in geography (Goudie, 1986).

Hydrology

Hydrology is the study of the occurrence, distribution and movement of water over, on, and under the earth's surface. Knowledge of hydrology is essential in the fields of agriculture, botany, forestry, soil science, geology, geomorphology, and ecology. The realm of geography encompasses or overlaps all of these; hence, specialists from many fields have contributed to the development of hydrology as a distinct discipline. Consequently, according to Ward (1967), hydrology has grown "from the outside towards a center".

The hydrologic cycle has long fascinated humankind. With its inherent elegance, this cyclical movement was seen as a great principle of order, inexorably linked to the success of agriculture and, ultimately, to human survival. Philosophers and theologians from medieval to modern times have viewed the hydrologic cycle as irrefutable evidence of the "wisdom of God" (Tuan, 1968). Geographers remain intrigued by its conceptual elegance. R. J. Chorley, one of the world's most influential geomorphologists, defined the hydrologic cycle as a "natural manifestation of great pervasiveness, power, and beauty that transcends man's territorial and intellectual boundaries" (Chorley, 1969).

Biswas (1970) traced the history and development of hydrology from antiquity (the first recorded evidence of water resources work occurred in 3200 B.C.) through the nineteenth century, when many of the theoretical foundations of modern hydrology were began. Some of the most important achievements included work in groundwater hydrology by the British geologist William Smith (1769-1839), who is credited with combining the disciplines of hydrology and geology, and by the Frenchman Henri Darcy (1803-1858), who developed the theoretical basis of soil and groundwater flow (Darcy's law). The work of John Dalton (1766-1844), who first recognized the relationship between evaporation and vapor pressure, led to a generalized theory of vapor pressure (Dalton's law of partial pressures) as well as to a method for the estimation of evaporation (Dalton's equation). Other significant achievements included the 1862 publication of the first "modern" hydrology manual by Nathaniel Beardmore, and the beginning of systematic gaging of streams in representative watersheds of the United States (Biswas, 1970). Chow (1964) traced the development of hydrology through periods of observation (1400-1600 A.D.), measurement (1600-1700), experimentation (1700-1800), modernization (1800-1900), empiricism (1900-1930), rationalization (1930-1950), and theorization (1950-present).

One of the most important contributions made to the understanding of hydrological-climatic relationships evolved from the work of the distinguished geographer C. W. Thornthwaite, who was concerned with practical problems of crop irrigation. He developed a system of climate classification based on water surplus and deficiency, where climatic boundaries were determined rationally by comparing precipitation and evapotransporation, rather than by the vegetative regimes adopted by the Koppen classification. Thornthwaite calculated water need based on air temperature, latitude, and seasonality. (Thornthwaite, 1948). His climate classification system has been discussed in many physical geography textbooks (for example, Strahler and Strahler, 1983). The practicality of Thornthwaite's system was confirmed when the United States Department of Agriculture adopted the soil-water balance as the fundamental criterion in the U.S. system of soil classification (Soil Survey Staff, 1975).

The water balance concept was further refined by Thornthwaite and his associate, J. R. Mather (Thornthwaite and Mather, 1955; 1957; Mather, 1978) and is now accepted as a basic tool in environmental analysis, including hydrological studies in wetlands (Dooge, 1972; Carter et al, 1979; Carter, 1986; LaBaugh, 1986).

Human-Environment Interaction

Human-environment interaction is a fundamental theme in geography. The environmental crises besetting modern society have lead some geographers to reexamine this theme as one of overriding importance in the discipline. Ronald F. Abler, in a presidential address to the Association of American Geographers, declared that geographers "should speak of physical geography and of human-environment interactions in the same way we should speak of the world's human geography...(with a) discourse rooted in places, regions, and interconnections.." (Abler, 1987).

Guelke (1989) advocates adopting human-environment interaction as geography's fundamental concern. This would provide a basis for a stronger and more coherent discipline. Although physical geography and the study of physical processes are often viewed as entities that are quite separate from human geography, Guelke states that "scientific knowledge of physical processes is human knowledge, and is relevant in human geography to the degree that people incorporate this knowledge in the way they relate to their environment...in dealing with problems relating to human activities on the earth, geographers would not ignore spatial and locational factors, but these factors would be incorporated within the human-environment theme." Guelke adopts a strict environmentalist viewpoint by stating "few people have integrated the possible consequences of their present actions with future prospects of their own survival...how people see themselves in a global context, and how these understandings are reflected in their lifestyles becomes a question for geographical analysis." For Guelke, this idea means that the coherence of the environmental-determinist approach to geography could be regained without "readopting its discredited causal thesis" (Guelke, 1989).

The theme of human-environment interaction has a rich tradition in geography. The influence of human impact on the environment was first explored in detail by George Perkins Marsh, who in 1864 published the monumental work <u>Man and nature: or physical geography as modified by human action</u>. This book was probably the most important landmark in the history of the study of human-environment interaction

(Goudie, 1986). Marsh, who drew his inspiration from the great generalist Alexander von Humboldt, defined geography as "the science of the absolute and relative conditions of the earth's surface and of the ambient atmosphere and the investigation of the relations of action and reaction between man and the medium he inhabits" (Lowenthal, 1958).

Elisee Reclus, the prominent French geographer of the late nineteenth century, drew from the ideas of Marsh in his two-volume work The Earth (1871). Carl O. Sauer (1889-1975), the great historical-cultural geographer, was strongly influenced by Marsh, especially while developing his concept of destructive exploitation. Sauer is credited with reintroducing the American public to the work of Marsh (Speth, 1977). Today, George Perkins Marsh is considered the nation's first conservationist and the "father of the environmental movement" (Curtis et al, 1982).

The discipline of geography, with its focus on the analysis of human-environment interaction, is well-positioned to address the myriad of concerns about the present state of the environment. Alteration of wetlands has been a significant human activity since the beginning of colonial settlement. For three hundred years, the wetlands of the Albemarle-Pamlico peninsula have undergone change or destruction as the nature of land-use has shifted from plantation agriculture to widespread logging to large-scale corporate farming activities (McMullan, 1984). Yet, only in the last 30 years have the inherent values of wetlands and the human impact of wetland destruction been recognized.

Prudent management of North Carolina's last remaining wetlands is thus a question for geographical analysis, as the issues of growth, development and the preservation of wetland values must be resolved. As such, this thesis may appropriately be viewed in the broader context of the man-environment tradition of geography.

CHAPTER IV

BACKGROUND

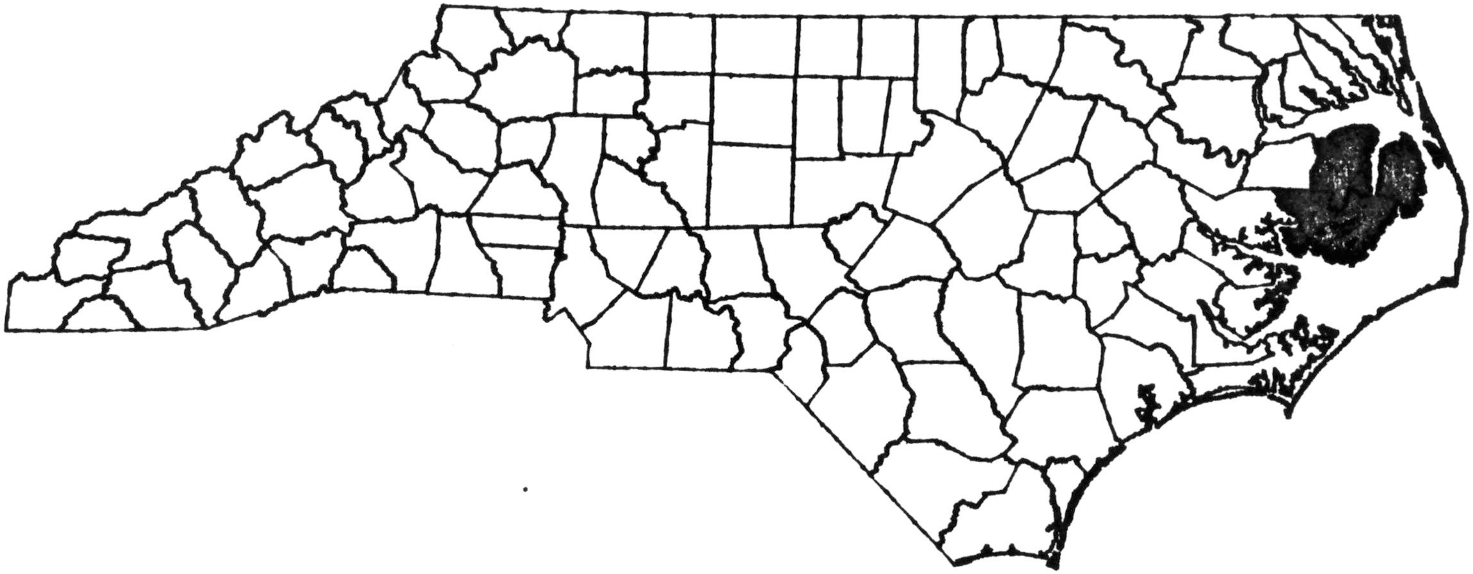
Location and Context of the Study

The Alligator Swamp, sometimes known as the Little Dismal Swamp, lies on the mainland of Dare County in northeastern North Carolina (Figure 1). The Dare County peninsula is bordered by the Alligator River to the west, the Albemarle Sound to the north, and to the east by the Croatan and Pamlico Sounds. Mainland Dare County is the easternmost part of the low-lying Albemarle-Pamlico peninsula (Figure 2). It contains a mosaic of wetland ecosystems, including large areas of pocosins. Pocosins are found on the lower coastal plain of the southeastern United States and are concentrated in North Carolina. They comprise more than 50% of North Carolina's freshwater wetlands (Richardson et al, 1981). According to the United States Fish and Wildlife Service's wetland classification system (Cowardin et al, 1979) pocosins are of the palustrine system, with two vegetational classes: (1) Scrub-shrub, with a shrub understory and scattered pond pine (Pinus serotina), and (2) Broadleaf evergreen shrubs such as titi-bush (Cyrilla racemiflora), sweet bay (Magnolia virginiana), and red bay (Persea borbonia). Pocosins are associated with several other wetland vegetation communities (Kologiski, 1977). Pocosins are frequently saturated and are also

Figure 1

NORTH CAROLINA

showing location of Albemarle–Pamlico Peninsula*



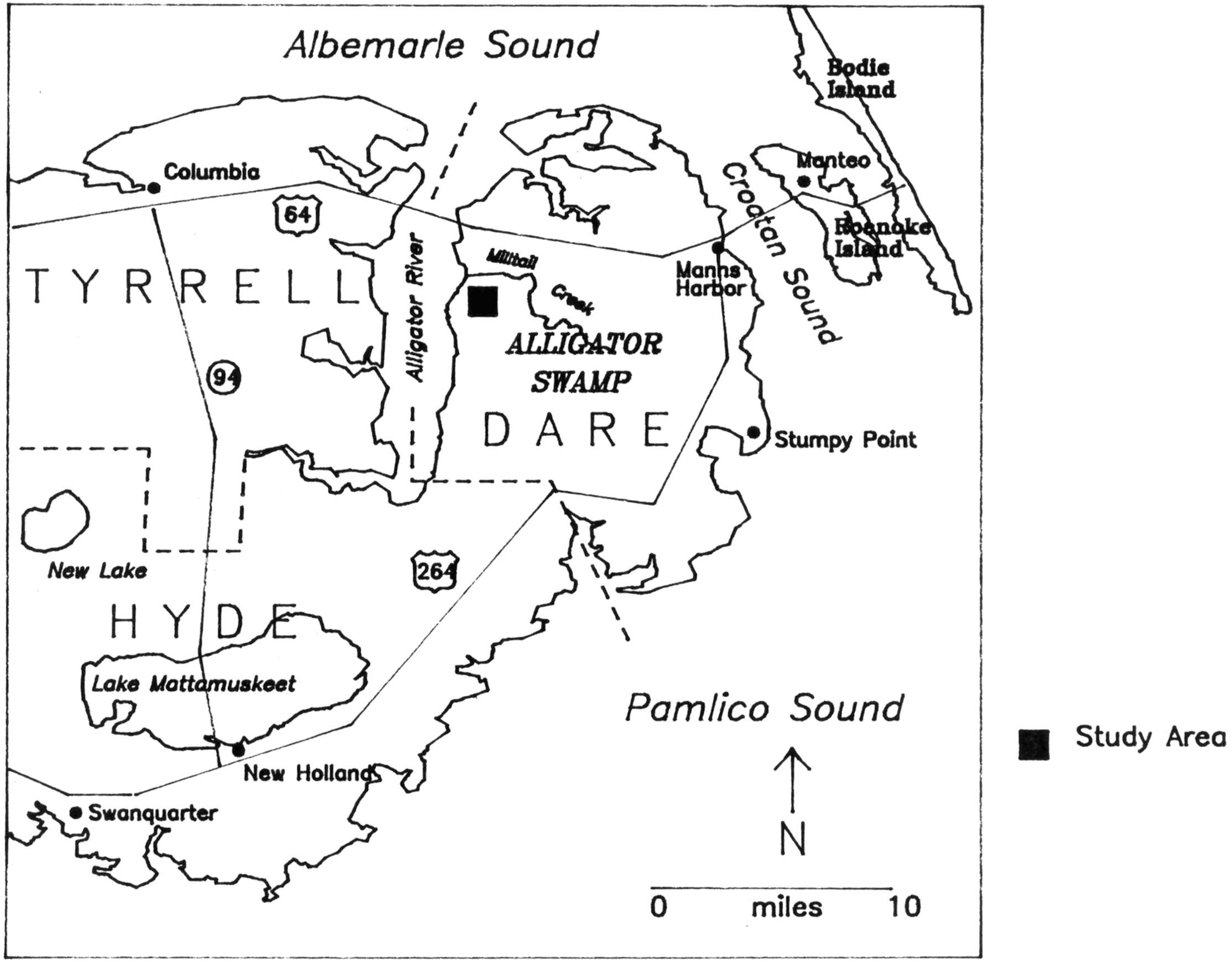
*Dare, Hyde, and Tyrrell County

■ Albemarle–Pamlico Peninsula

Figure 2

ALBEMARLE–PAMLICO PENINSULA

with selected features



subject to periodic burning. These fires are a crucial factor in the structure and composition of pocosin vegetation (Christensen et al, 1981). Pocosins are evergreen shrub bogs characterized by an extended hydroperiod, with ombrotrophic (nutrient-poor) and predominately organic soils of high acidity. The term <u>bog</u> is defined as a peat-accumulating wetland that has no significant inflows from the surrounding area and supports acidophilic mosses such as sphagnum (Dooge, 1972; Mitsch and Gosselink, 1986).

Pocosins generally occur in four topographic settings: (1) blocked drainage systems; (2) Carolina bays; (3) ridge and swale systems associated with ancient dune or beach ridges on the lower Coastal Plain terraces; and (4) seeps, springs and slow-flowing streams in the Sandhills region (Christensen et al, 1981; Otte, 1981). Blocked drainage systems, in which most pocosins occur, developed 10,000 to 15,000 years ago during the Wisconsin glaciation (the last major ice advance during the Pleistocene) as sediment and organic debris settled and clogged streams. The stream valleys became filled with the accumulation of peat, which spread laterally over the entire interstream divide and formed a peat dome (hence the derivation of the term 'pocosin' from the Algonquian term for "swamp on a hill") (Otte, 1981). These organic deposits lie atop the Pamlico morphostratigraphic unit (msu), a Pleistocene formation of arenaceous clays and argillaceous sands that lies east of the Suffolk Scarp (Daniels et al, 1978). Carolina bays, which are characterized by elliptical, peat-filled depressions, occur abundantly in a broad band across the Coastal Plain of the southeastern United States. They vary in size from less than 50 meters in length to more than 8 kilometers long (Lake Waccamaw in Columbus County, North Carolina). Other freshwater lakes associated with Carolina bays are White Lake and Black Lake in Bladen County. The geologic origin of Carolina bays is uncertain, and many controversial theories have been proposed. Ridge and swale and sandhills pocosins are relatively small in total area, and have not received a great deal of attention in the literature (Sharitz and Gibbons, 1982).

The pocosin wetlands of North Carolina, which in 1962 comprised nearly 70% of the nation's pocosins, once covered more than 890,000 hectares (2.2 million acres) in the lower Coastal Plain (Wilson, 1962). Today, however, only 31% (281,000 ha, or 695,000 acres) of North Carolina pocosins remain in a natural state. The remaining 607,000 ha (1.5 million acres) have been partially or totally altered by human activities. As recently as 1984, the pocosins of North Carolina were being lost at the rate of 17,600 ha (43,500 acres) per year (Tiner, 1984).

Pocosins remain the least known of wetland ecosystems. The relative isolation and inaccessibility of these wetlands have caused them to be largely ignored by the scientific community until recently (Richardson, 1983). Commercial interest in peat mining, logging, and agriculture, countered by a national interest in wetland preservation brought pocosin wetlands to the attention of the general public (DeBlieu, 1987). Pocosins are now recognized as unique, vital ecosystems of great importance to the economy of North Carolina, especially for their roles in maintaining estuarine water quality, and for providing habitat for wildlife species.

The Alligator Swamp, located on the Dare County peninsula between the Alligator River and the Pamlico Sound, contains large areas of pocosins and associated wetland ecosystems, which comprise more than 50% of the county's total area. Natural pocosins in Dare County were estimated at 55,850 ha (138,000 acres) by Wilson (1962). Richardson, et al. (1981) estimated 5% of this area had been altered by human activities, including logging. Logging of baldcypress (*Taxodium distichum*) and Atlantic white cedar (*Chamaecyparis thyoides*) (commonly known as 'juniper') was a major enterprise in the late nineteenth and early twentieth century (Earley, 1987). Logging continues to be the major impact on Dare County wetlands, most of which were owned by Prulean Farms, a subsidiary of the Prudential Life Insurance

Company. Another large portion of natural pocosins lies in the southern part of the Dare peninsula within the Dare County Bombing Range, which is owned by the federal government for the purposes of training military pilots in bombing techniques.

Prudential subsequently sold its holdings in Dare County (and some adjacent land in Tyrrell County) to the Nature Conservancy, a conservation organization based in Virginia. Negotiations between the Nature Conservancy and the federal government resulted in the establishment of the Alligator River National Wildlife Refuge in March, 1984. The refuge encompasses more than 55,600 ha (137,411 acres) of coastal plain ecosystems, primarily pocosin and mixed pine and hardwood swamp forest (Dean, 1984). The refuge is perhaps best known for research into the reintroduction of the red wolf (Canis rufus) into a natural habitat. The species is presently extinct in the wild (Venters, 1989). Most of the refuge area exists in an undeveloped condition, but many sections throughout the refuge have been recently logged for Atlantic white cedar. Clearcutting of white cedar, mostly second-growth stands regenerated from the logging activities from the 1890's through the 1920's, began in 1979. Some cutting is slated to continue until 1996, because timber rights were sold by First Colony Farms prior to ownership by Prudential and the subsequent creation of the Alligator River National Wildlife Refuge. The United

States Fish and Wildlife Service, the federal service charged with management of the refuge, will attempt to mitigate the effects of artificial drainage through a management plan to restore the natural hydrology of a section of the refuge. Goals of this restoration include the establishment of favorable waterfowl habitat and the regeneration of Atlantic white cedar (Noffsinger, 1989). The plan calls for the installation of water control structures (flashboard risers) in a canal south of Milltail Creek (see Figure 3). The Grassy Patch canal extends from Dry Ridge North Road and drains into the Alligator River at Cypress Point. The proposal also includes the permanent filling of 14 smaller, adjacent canals. Flashboard risers will be installed each end of the Grassy Patch canal so that water levels can be manipulated to a slight extent in order to maintain the canal access road. This management plan will be one of the first wetland restoration efforts undertaken on the Albemarle-Pamlico peninsula. Furthermore, the plan provides an opportunity to study what shall be defined as a post-altered wetland: a recovering ecosystem that is undergoing no further alteration from silvicultural practices, and with drainage canals permanently closed. The plan put forth by the Alligator River National Wildlife Refuge staff offers a premise for the study of this recovery.

Pocosin Communities: Successional Relationships

Otte (1987) defines two major pocosin types found in the prevalent blocked drainage system setting: (1) primary pocosins that appear to be the climax ecosystem on coastal peat domes, with some areas that may be several thousand years old; and (2) secondary pocosins, which exist solely because of recent human alterations. Secondary pocosins are at most several hundred years old. Primary pocosins are climax communities that began as marsh systems dominated by aquatic macrophytes and grasses, succeeded to cypress and Atlantic white cedar swamp forests, and finally to shrub bog pocosins (Otte, 1981).

The literature is ambiguous with regards to the climax stage of a pocosin and whether it succeeds an Atlantic white cedar forest or vice versa. This ambiguity exists largely because of the complexities and uncertainties involving the role of fire in pocosin succession. Otte (1981), when determining the origin of peat in the Croatan National Forest, found white cedar peat overlain by pocosin peat (primarily consisting of shrub vegetation). Within the pocosin peat, he found patches of marsh peat, which indicate local, wetter conditions possibly marking areas of relatively deep peat burns that left shallow, open pools for some period of time. Otte based his pattern of pocosin community development on such peat profiles. He also categorized the influence of fire based on four states of recovery which depend on the type of burn. He stated that a fire in an area of shallow peat that is severe enough to destroy the root and rhizome mat will be repopulated by the same species as was present before the fire; i.e., Atlantic white cedar.

Buell and Cain (1943) stated that if protected from fire, the Atlantic white cedar forest will naturally succeed to a shrub bog. Penfound (1952) noted that evergreen shrub bogs are formed when Atlantic white cedar is removed by cutting or fire, but that a white cedar swamp naturally gives way to a red bay-sweet bay community when protected from fire.

Descriptions of North Carolina forests by Ashe and Pinchot (1897) and Korstian and Brush (1931) indicate that Atlantic white cedar existed in a vast range throughout the coastal plain prior to human settlement. The presence of white cedar logs beneath shallow peats in the Holly Shelter and Green Swamps (Kologiski, 1977; Otte, 1981) and the presence of white cedar stumps in peat of the Hoffman Forest (Daniels, et al., 1977) support the contention that vast white cedar forests once existed prior to human disturbance, only to be replaced by pocosins that were created primarily by chronic human disturbance of the forests.

N. L. Christensen (1981) contends, however, that pocosins are stable, and except at their margins, show little evidence of succeeding to white cedar or cypress forests. Radiocarbon dates of white cedar stumps at the base of 1 to 2 meters of peat in a Hoffman Forest shrub bog show that 5,000 to 7,000 years before present, a white cedar swamp existed. This indicates that pocosins are actually very stable ecosystems. Christensen states that instead of creating pocosins, human disturbance actually extends pocosin boundaries (Christensen, et al., 1981). Activities such as logging leave extensive areas of slash and brush that provide fuel for fires that encourage the establishment of fire-tolerant pocosin species. Logging, therefore, may have increased the extent of pocosins and decreased swamp forests (Ash, et al., 1983).

It is unrealistic to theorize on the natural succession of pocosin communities in the absence of fire (Christensen, et al., 1981). The accumulation of decaying organic material on the surface, the accumulation of the peat layer, seasonal fluctuations in the water table, and the flammability of shrub vegetation indicate that fire is inevitable, and eventually these areas will burn. Once pocosins are established, the character of the vegetation favors a fire regime that tends to perpetuate that vegetation.

Kologiski (1977) made an extensive survey of plant communities of the Green Swamp in Brunswick County, North Carolina, and devised a hierarchical scheme consisting of five major vegetation systems. Only one of his systems fits the definition of a true pocosin as described in a community profile of pocosins and Carolina bays by Sharitz and Gibbons (1982). Kologiski's white cedar bog system included two community types: one with a dense canopy over deep organic soils, and the other with a mixed Atlantic white cedar/loblolly pine over shallow organic soils underlain by sandy clay loam. Kologiski speculated the white cedar forest would succeed to an evergreen bay forest following logging. Additionally, fire intensity would determine if the evergreen bay forest reverts to a pocosin or white cedar forest (shallow burn), a sedge bog (deep burn/high water table), or deciduous bay forest (deep burn/low water table). However, the suitability of Kologiski's classification scheme outside of the Green Swamp has yet to be determined (Christensen et al, 1981); thus, the successional relationships among common pocosin community types remain a matter of speculation (Sharitz and Gibbons, 1982).

It would appear that post-fire regeneration of an Atlantic white cedar versus a pocosin community is indeterminate (Frost, 1987). Time of year, height of water table, and depth of burn are interrelated factors which will determine the potential of a plant community to occupy a site disrupted by fire. For example, white cedar will regenerate if a fire kills the canopy but does not burn deep enough to destroy the seed bank within the soil. A deep peat burn, however, destroys the seeds and prevents the reestablishment of white cedar, thus favoring other pocosin species. Korstian (1924) stated that slash fires were beneficial to regeneration after logging. However, a second burn some years after would eliminate white cedar. Korstian noted that seed bank viability was the most important factor in Atlantic white cedar regeneration. He stated that despite repeated logging from the period following the American Revolution to 1924, good restocking generally resulted. Korstian observed that if the original stand of white cedar is destroyed, together with the seed bank, succession would be toward a pine-dominated community.

Secondary pocosins are defined by Otte (1987) as those areas where scrub-shrub vegetation has occurred after documented human intervention in Atlantic white cedar forests. Such secondary pocosins do not exhibit the well-defined vegetation community patterns found in primary pocosins. When Atlantic white cedar forests were destroyed during the intense logging activity of the late 1800's through the 1920's, the construction of drainage ditches and barge canals for timber removal disrupted the natural flow of water and nutrients. This set the stage for an artificial ombrotrophic environment (Otte, 1987). Fires that burned in the logged areas destroyed the seed bank, allowing the pocosin species that were present in the cedar forest to quickly regenerate. Some areas that remained wet or did not burn completely recovered as white cedar forests. These areas can be found today in the Dismal Swamp of Virginia and in the Alligator River National Wildlife Refuge. The Alligator River stands that have regenerated constitute the previously mentioned present-day logging sites with the refuge. Thus, the post-altered pocosin area that will be studied in this thesis originates from a secondary pocosin.

Atlantic White Cedar

Atlantic white cedar (Chamaecyparis thyoides) is an evergreen aromatic tree with narrow, pointed spirelike crowns and slender, horizontal branches. The height of mature specimens ranges from 15 to 27 meters (50 to 90 feet). White cedar is generally found in wet, peaty, acidic soils where it forms pure stands (Little, 1980). Atlantic white cedar is geographically distributed along a narrow band of the eastern coastal United States, restricted to freshwater wetlands from Maine to Mississippi (Laderman, 1982). There are no quantifiable estimates of the presettlement range of Atlantic white cedar, but geographic place names may provide a rough estimate of the former existence of Atlantic white cedar forests during the last

300 years (Dill et al, 1987; Frost, 1987; Belk, 1989). The extent of Atlantic white cedar forests has been steadily reduced since colonial times, as these forests were used for their valuable timber. Pioneers used the durable wood for log cabin floors and shingles, and during the American Revolution, the wood was burned to make charcoal for gunpowder. White cedar wood remains valuable today. It is highly regarded as an exterior building material because of its resistance to rot, and as an interior wood because of its aromatic properties (Little, 1980; Earley, 1987).

The most extensive Atlantic white cedar forests extant in North Carolina, and probably in the world, are located in the Alligator River National Wildlife Refuge (Moore and Laderman, 1989). White cedar in this region occurs in pure, even-aged stands, and in lowland swamp forests where it is mixed with lowland conifers and hardwoods. The shrub layer in pure stands most frequently contains sweet pepperbush, highbush blueberry, dangle huckleberry, large gallberry, inkberry, possumhaw, fetterbush, maleberry, and evergreen bayberry (Moore and Carter, 1987). The mixed stands contain such species as red maple, blackgum, water tupelo, and baldcypress, along with scattered pond pine, loblolly pine, and occasional sweet bay (Levy, 1987).

Atlantic white cedar is very intolerant to shade. It does not reproduce beneath its own living canopy or that of a deciduous swamp forest. Fire provides the opportunity for white cedar to repopulate a site or colonize an adjacent site occupied by other wetland communities. An intense fire, however, destroys the seed bank and prevents the reestablishment of cedar. Hydrologically, the moisture regime of white cedar appears to lie on a gradient between swamp forest (wetter) and pocosin (drier) (Frost, 1987). Golet and Lowry (1987) monitored water table activity in six Atlantic white cedar wetlands in Rhode Island over a six year period to analyze the relationships between water regimes, tree growth, and other site characteristics. They did not find a general relationship between water regime and annual radial growth, but indicated that more extensive data on soil moisture may provide more evidence for such a relationship.

The Atlantic white cedar has been extirpated from over 90% of its presettlement range in North Carolina (Frost, 1987, 1989). Traditionally, Atlantic white cedar has regenerated sufficiently to permit extensive logging (Korstian, 1924). However, present-day encroachment into environments otherwise suitable for the growth of white cedar has diminished its natural regenerating capacity.

Several factors may account for the failure of white cedar to regenerate. Artificial drainage in white cedar wetlands alters the soil regime by causing dehydration, compaction, and subsidence. These factors lead to the shrinkage of the organic layer and to its eventual replacement by mineral soil. Artificial drainage has also altered natural fire recurrence intervals, allowing fires in drier peats to burn intensely enough to destroy the seed bank. Suppression of fire allows logging slash to compress the soil, and shade out young seedlings.

Undoubtedly, many other factors influence the failure of regeneration in logged Atlantic white cedar stands. Soil and plant chemistry, algal and microbiotic relationships, shade and seed bank density, the relationships of competing vegetation, and the health and seed-bearing capability of adjacent uncut specimens, to name a few, would be essential elements of a detailed environmental study which is beyond the scope of this thesis. However, an examination of soil properties and moisture regimes could provide insight into the soil and hydrologic conditions present in the white cedar wetlands of the Alligator Swamp.

Organic Soils

Organic soils are characteristic of waterlogged or poorly drained environments. They are created only in the presence of anaerobic microorganisms which transform the accumulated organic material into peat or muck by processes of decomposition. Anaerobic conditions are essential for the accumulation of organic soils, thus, such deposits are formed in bogs, marshes, swamp forests, and other wetlands (Gorham, 1957; Davis and Lucas, 1959; Moore and Bellamy, 1974).

The physical characteristics of organic soils are determined, to a large extent, by the degree of decomposition of the organic material. Degree of decomposition can be determined by measuring one of the chemical or physical characteristics that change as decomposition advances. Farnham and Finney (1965) used fiber content to determine this property. Later, Boelter (1969; 1972b; 1974) used bulk density as a measure of the degree of decomposition.

Boelter (1964) stated that water retention and hydraulic conductivity are important physical properties that largely determine the hydrologic characteristics of organic soils. These properties are related to porosity and pore size distribution which in turn are related to the degree of decomposition. As decomposition proceeds, the size of organic particles decreases, resulting in decreasing total porosity and higher bulk density. Thus, undecomposed peat has a relatively low bulk density and contains large pores that drain readily, while highly decomposed sapric peats have a much higher bulk density and retain more water. Boelter (1965) found hydraulic conductivities of various

Minnesota peats to cover a wide range of values which were related to pore size distribution. While undecomposed peats allowed rapid water movement, highly decomposed peats had rates of hydraulic conductivity lower than clays. The hydraulic conductivity of highly decomposed (sapric) peat has a range of several orders of magnitude and may be as high as some fibric peats. This large variation is caused by such phenomena as piping, buried wood, and variations in peat stratigraphy (Siegel, 1988).

Organic soil deposits in North Carolina have been estimated to cover nearly 500,000 ha (Lucas, 1982). They occupy three geographic settings. The largest deposits occur east of the Suffolk Scarp, a relic shoreline of 75,000 years BP. These peatlands, also identified by the term Blacklands, lie atop the Pamlico morphostratigraphic unit (msu), which is the youngest exposed seabed that constitutes the Lower Coastal Plain. These deposits include the Great Dismal Swamp, The Albemarle-Pamlico peninsula, the Pamlico-Neuse peninsula, and the Alligator Swamp. The next largest deposits of organic soils include the Croatan and Hoffman Forests and the Green Swamp, which lie atop the Talbot and Wicomico morphostratigraphic units east of the Surry Scarp, a relic shoreline of 300,000-400,000 years BP (Daniels et al, 1978). Carolina Bays are the third setting for extensive organic soil deposits in North Carolina.

Organic soil deposits in the Lower Coastal Plain contain primarily colloidal muck soils of two main types: (1) an upper region of brownish-black, fine grained, highly decomposed sapric peat that overlies (2) a dark reddish brown, decomposed though slightly fibrous sapric peat. The differences between these two types can be attributed to their origin. The upper peat represents accumulation in a swamp forest environment, while the lower, more fibrous peat is indicative of a shallow marsh environment (Ingram and Otte, 1981; 1982; Otte, 1981; Ingram, 1987). When saturated, both mucks have the general appearance and consistency of black axle grease or chocolate pudding (Ingram, 1987). These colloidal mucks are formed in poorly aerated, low energy environments (Dolman and Buol, 1967; 1968). Colloidal muck soils are characteristically sticky, plastic, and very acid. They have extremely low values of hydraulic conductivity (Boelter, 1965). Colloidal mucks do not compress, but flow when put under pressure. Such characteristics have made the utilization and development of the deeper North Carolina Blacklands extremely difficult; however, areas underlain by thinner organic deposits have more agronomically desirable properties and have been intensively developed (Lilly, 1981a; 1981b).

Organic soils are grouped under the order Histosol, one of ten soil orders that cover the world's soils, according to the U.S. system of classification. Histosols are further categorized by three suborders: Saprist, Hemist, and Fibrist, which differentiate Histosols on the basis of degree of decomposition of the organic material (Boelter, 1969; McKinzie, 1974). Most Histosols in North Carolina are classified within the Great Group Medisaprist. The prefix medi- indicates that the soil (1) has no significant humilluvic horizon (material moved downward by leaching) and (2) has a mean annual temperature of 47^{o}F with more than a 9^{o}F variation between winter and summer means (Soil Survey Staff, 1975).

Peatlands have long been altered by human activities. In the United States, and particularly in North Carolina, organic soil deposits have been important in the development of agriculture and timber resources. Organic soils were especially important in the years before the widespread availability of commercial fertilizers, because these soils naturally release nutrients such as nitrogen and phosphorus that are important for crop growth. The human effort at management of these areas, which usually begins with artificial drainage, has been the catalyst for research into organic soil deposits. Artificial drainage in organic soils has been historically recognized as the precursor to one of the major problems encountered in peatland utilization: subsidence.

Subsidence is the loss in elevation of the soil profile after an organic soil is artificially drained. It is an inherent problem in the management of Histosols for agricultural and, to a lesser extent, silvicultural practices (Maki, 1974; Skaggs et al, 1980). Subsidence can be attributed to several factors, including (1) shrinkage and dehydration, (2) burning, (3) consolidation due to loss of the bouyant force of groundwater, (4) compaction from tillage or logging operations, and (5) biochemical oxidation.

Shrinkage is characteristic of both drained and undrained Histosols. The withdrawal of moisture from the surface layers by evapotranspiration (or drawdown from a drainage canal) may cause high moisture tensions in the root zone, which results in a decrease in volume above the groundwater level, or phreatic surface. In undrained Histosols, however, shrinkage is a seasonal phenomenon, and constitutes only a change in volume, not a physical change in soil mass. Shrinkage can be the major contributor to subsidence in an artifically drained Histosol if the surface layer dries (Skaggs et al, 1980). Skaggs and Barnes (1976) reported that colloidal mucks shrink to one-third to one-half of their initial volume when air dried. The drying process is irreversible, and results in stone-hard aggregates of granular structure, similar to ground coffee. Colloidal materials that have been irreversibly dried will not regain their original volume and consistence even after months of soaking.

Fire is a perennial problem in Histosols. Nearly all Histosols in North Carolina have lost some surface elevation due to burning. Many of the small lakes within the deep peat regions of the Lower Coastal Plain may have originated from peat fires. Some former areas of extensive organic deposits have been subjected to such intense, repeated burnings that today only the mineral soil surface remains (Frost, 1989). Peat fires are particularly hazardous during the dry season. They are extremely difficult to extinguish and may smolder for months.

Consolidation and compaction are often combined in the term "densification". When drainage occurs, the bouyant force of groundwater is lost in the upper layers. The deeper layers are then forced to bear an increased weight of 1 gram per square centimeter for each centimeter of drawdown. This causes consolidation and compaction of the soil layers below the water table. When heavy equipment is used in farming or logging operations, the compaction of these layers is intensified.

The primary cause of subsidence is decay of organic matter. When an area is artificially drained, the water table drops to allow oxygen to enter the soil, creating the aerobic conditions necessary for decomposition. Over time, this decay results in increasingly smaller soil organic matter content, and a concentration of the mineral content of the soil (Lilly, 1981a).

Effects of Logging on Soil Properties

A key to understanding the hydrologic characteristics of altered organic soils is to examine the effect that logging operations may have on soil properties. While logging may not represent the final alteration that is indicative of agriculture or peat-mining operations, it nonetheless constitutes a profound change in the nature of the underlying material and must be considered when attempting to evaluate the hydrologic response of an altered soil.

Harvest of Atlantic white cedar in North Carolina is usually done with an amphibian feller-buncher, a caterpillar-like machine that is specifically designed for harvesting wetland timber. Large, hydraulically-operated arms on the feller-buncher seize the erect tree and shear it at the base. A worker on foot then removes the tops and branches, which are used for traction for the skidder that pulls the trunks to the road (Laderman, 1989). Cedar harvesting leaves the immature, unharvested cedars vulnerable to windthrow. Many shallow-rooted, windthrown cedars were observed lying on the fringes of the logging sites within the study area. Uprooting by windthrow can disturb large amounts of material, often resulting in complete inversion of the soil profile (Schaetzl, 1986).

The soil disturbance from the logging procedure results in compaction and displacement of the mineral layer. The resultant effects on the clay loams and sandy loams that underlie the organic layer are increased bulk density, reduced infiltration capacity, and decreased porosity.

These changes are not permanent. Research in the Mississippi coastal plain on clay loam soils affected by tree-length skidding showed a return to pre-disturbance conditions in eight years. Soils disturbed by wheel ruts, however, recovered after twelve years (Dickerson, 1976).

Wetland Hydrology

Hydrology is the primary factor in the establishment and maintenance of wetlands and wetland processes (Gosselink and Turner, 1978; Mitsch and Gosselink, 1986). These processes are often disrupted by human activities, and the Albemarle-Pamlico peninsula, with extensive areas of pocosins, is one region that has been affected.

The hydrologic characteristics of hydraulic conductivity, water table fluctuations, surface runoff and nutrient relationships in a natural and an altered pocosin have been examined by Daniel (1981), based upon his instrumentation and analysis of three watersheds on the Albemarle-Pamlico peninsula. He examined the hydrology of three watersheds that are characteristic of the Albemarle-Pamlico peninsula. The Van Swamp basin is a forested wetland which exists primarily in a natural state, although some drainage for road access has taken place. The Albemarle Canal watershed typifies the heavily-drained agricultural areas that are prevalent on the peninsula (Daniel, 1978). The third watershed he analyzed was located between Pungo Lake and Phelps Lake in the northeast corner of the Pungo National Wildlife Refuge. This basin exemplifies the pocosin system, with deep organic soils and vegetation consisting of evergreen bay, shrub, and pond pine.

Daniel (1981) states that the hydrologic regime of pocosins is influenced by four input-output events. Precipitation is, in most cases, the only input source. The major outputs are evapotranspiration and runoff, with groundwater recharge being the least significant of the four components (Heath, 1975). Evapotranspiration is the most important output during the late summer and early fall, with runoff being the predominant output during months of increased precipitation. Richardson (1982) reported that evapotranspiration accounts for 60% to 70% of water output during the summer and fall in an undisturbed pocosin. Daniel (1981) reported rainfall to exceed potential evapotranspiration in all but the driest years. These conditions explain the elevated water tables within pocosins.

Several examples show how the hydrology of pocosin wetlands has been altered by the widespread installation of artificial drainage systems for timber management and agriculture. Initially, the clearing of ditch right-of-ways and the removal of vegetative cover reduces transpiration and increases evaporation. However, ditching will lower water table levels in the vicinity of the drainage canals. This reduces the length of time the ground is saturated, thus decreasing evaporation. Transpiration will increase somewhat as the vegetation recovers. Surface runoff, which exists primarily as overland flow in a natural bog or pocosin (Bay, 1969; Heath, 1975; Ash et al., 1983) is increased when drainage canals are cut. Overland flow in a bog ecosystem is diffuse. Discharge is spread over a wide area because of the relatively low relief of a wetland. When drainage canals are constructed, the discharge is directed toward outflow points, thus concentrating runoff. Drainage canals cause an increase in the rate of runoff (increases in maximum flow events), though not a significant increase in total annual runoff volume (Daniel, 1981).

Additionally, drainage canal construction alters the properties of organic wetland soils. Ditching increases decomposition, changing the physical properties of the peat, so that bulk density increases and macro-pore space and permeability decrease. The subsidence that results from water table drawdown may change the surface topography by forming slightly convex surface between ditches. This is due to the convex water table between ditches. (Boelter, 1972a; Verry and Boelter, 1979; Skaggs et al, 1980; Daniel, 1981; Gregory et al, 1984).

Wetland Evaporation Studies

Sharitz and Gibbons (1982), in a community profile of southeastern shrub bog ecosystems, stated that evapotranspiration was the least-understood component in determining the hydrologic characteristics of pocosins.

Two approaches are commonly employed in determining evaporation from a water surface. The mass-balance or vapor flux approach has its theoretical foundations in Dalton's Equation, which states that the rate of evaporation is proportional to the difference between the vapor pressure of the water (or leaf) surface and the vapor pressure of the air directly above the surface:

$$E = C(e_s - e_a) \tag{1}$$

where E is the rate of evaporation, C is a coefficient that is a function of wind speed, and e_s and e_a are the saturation vapor pressures at the surface and the overlying air, respectively.

The energy budget method involves the partitioning of available net solar radiation into its different categories of use at the earth's surface. The energy required for evaporation, Q_E, is determined from the energy budget equation

$$Q_E = Q_s - Q_{rs} - Q_l - Q_c \pm Q_g \pm Q_v \tag{2}$$

where Q_s is the shortwave solar radiation, Q_{rs} is reflected shortwave radiation, Q_l is the longwave radiation from the water body, Q_c is the sensible heat transfer to the air, Q_g is the change in stored energy, and Q_v is the energy transfer between the water body and bed. Evaporation from the water surface (E_o), is then determined by

$$E_o = Q_E/pL \text{ in mm/sec} \tag{3}$$

where p is air density (g cm^{-3}) and L is the latent heat of vaporization of water (590 cal g^{-1} or 2.47×10^{3} J g^{-1}). While the energy budget approach is the most accurate (Winter, 1981; Carter, 1986), obtaining exact measurements for the separate energy components is time-consuming and requires expensive equipment. The following discussion centers on two formulae which are more practical for use by the researcher, and employ readily available meteorological data.

Thornthwaite (1948) developed his formula from the standpoint of the energy balance approach. His method uses air temperature as an index of the energy available for evapotranspiration. Thornthwaite's formula assumes air temperature is correlated with the combined effects of net radiation, and that a fixed relationship exists between those portions of net radiation used for heating and those used for evaporation (Thornthwaite, 1954; Ward, 1967; Dunne and Leopold, 1978). The formula takes the form

$$PE = 1.6(10T_a/I)^a \qquad (4)$$

where PE is potential evapotranspiration in cm $month^{-1}$, T_a is mean monthly air temperature in degrees Celsius, I is the annual heat index, and $a = 0.49 + (1.79 \times 10^{-2})I - (7.71 \times 10^{-5})I^2 + (6.75 \times 10^{-7})I^3$.

Penman (1948) developed a formula which effectively combines the mass-balance and the energy budget approaches, as well as allowing the use of easily-obtained data. Derivation of the Penman formula is described in detail by Ward (1967), Dunne and Leopold (1978), Lee (1980) and Shaw (1988).

Both the Penman and Thornthwaite formulae have found worldwide acceptance and applicability, and can be considered the main tools of hydrologists and climatologists in the estimation of evapotranspiration (Ward, 1967; Shaw, 1988).

Measurement of evapotranspiration (ET) is a special concern in the computation of water balances for wetlands. A multitude of empirical formulae, including the previously discussed Penman and Thornthwaite equations, have been developed to determine evapotranspiration. However, because of the many physical and biological factors that affect ET, Lee (1980) stated that no reliable method exists for estimating ET rates based on simple weather data or computation of potential evapotranspiration. While these formulae are able to predict evapotranspiration of open water, grasslands, and arable crops with sufficient accuracy, their values are subject to doubt when applied to wetlands. The reason for this is that the mosaic of swamp vegetation is difficult to describe in terms of the necessary parameters (Van der Molen, 1988). Stalks of dead and decomposing vegetation left over from earlier years, and present above or between the actively growing vegetation, will influence the energy exchange with the atmosphere, usually reducing ET. This situation exists in the logged white cedar wetlands of the Alligator Swamp, although it is compounded by the presence of logging slash and buried timber. Van der Molen further states that such "untidy" natural vegetation patterns are in strong contrast with the well-defined open water and trimmed grass surfaces used to calibrate the popular formulae; therefore, they are not directly useful for wetland hydrology problems (Van der Molen, 1988).

In the past, hydrological studies in wetlands have estimated ET as a residual in the water balance equation. Carter (1986) and LaBaugh (1986) clearly demonstrated the inherent problems in using residuals. More recently, studies have been undertaken which attempt to relate wetland ET to evaporation from a free water surface, but there has been much disagreement as to whether the presence of wetland vegetation increases or decreases the loss of water over that which would occur from an open body of water (Carter, 1986; Mitsch and Gosselink, 1986; Koerselman and Beltman, 1988). Bay (1967), for example, determined that losses via evapotranspiration from some northern Minnesota bogs were 88% to 121% of open water evaporation. Eisenlohr (1966) determined ET losses from vegetated prairie potholes in North Dakota were 10% lower than from those devoid of vegetation.

Ingram (1983), in an extensive literature review of ET studies in bogs, fens, and mires, stated that actual evapotranspiration from bogs is approximately equal to potential evapotranspiration. Ingram concluded that rates of ET depend upon the vegetation type. Considering evapotranspiration from wetlands is rarely limited by inadequate water supply, such factors as species type, stomatal openings, vegetative cover and density, and stage of development could cause rates to vary from one wetland type to another. Definitive studies in comparative rates of ET from different plant types are lacking, however (Carter et al, 1979). Linacre's 1976 review of evapotranspiration from swamps concluded that the presence and nature of vegetation have relatively minor influences on rates of ET, at least during the active growing season (Linacre, 1976). These reviews show, according to Koerselman and Beltman (1988), that models for determining evapotranspiration have not been calibrated with field data from peatlands.

Koerselman and Beltman's hydrological study of a fen in the Netherlands, which measured all water balance components, was undertaken to address this need. The objective of the study was to relate actual ET to pan evaporation and to Penman's potential evaporation from a free water surface (Penman's E_o). Evapotranspiration from lysimeters filled with peat and vegetation was compared with evaporation from lysimeters filled with water, and with Penman's formula. They found that ET from the vegetated lysimeters exceeded evaporation from water-filled lysimeters by a factor of 1.7-1.9 and was 0.7-0.8 times less than evaporation calculated by the Penman formula. They developed a regression equation that described ET from fens (in mm day^{-1}) as a function of Penman's E_o on a monthly basis: $ET = 0.73E_o + 0.16$ ($r=0.97$). They concluded that their equation was accurate, given the reliability of the lysimeter technique, and furthermore, that it could provide a model for the calculation of actual ET from potential ET that could be determined from routine weather data. They stated their equation may also apply to other peatlands in the temperate zone under conditions of high water tables, but that further calibration of such models is needed (Koerselman and Beltman, 1988; Koerselman, 1989).

An earlier study by Dolan et al (1984) used a diurnal water-table fluctuation technique as a method for measuring evapotranspiration in a central Florida freshwater marsh. They noted some particular problems with lysimeter studies in wetlands, notably, that lysimeter studies involve the physical isolation of individual plants or plant parts which could possibly subject them to unrealistic micro-climatic conditions. Dolan et al found their technique to be more suitable since relatively simple instrumentation could provide long-term measurements of ET for an entire plant community, not just an isolated patch. They also devised a regression formula for computing monthly evapotranspiration: $ET = 6.83 \times 10^{-5} BS + 0.10$, where B is the average aboveground live biomass, and S is the average saturation deficit. They found that total ET for the study period (one year) proved to be nearly identical to that predicted by the Thornthwaite technique. They concluded that the close agreement between the yearly totals for the regression model and the Thornthwaite estimates may indicate that an empirical relationship based principally on monthly average air temperature is adequate for yearly estimates but not for monthly estimates (Dolan et al, 1984).

The above studies by Koerselman and Beltman (1988) and Dolan et al (1984) illustrate the utility of both the Penman and Thornthwaite formulae in obtaining reasonably accurate estimates of ET. However, historical evidence suggests that the Penman formula, which combines the mass-transfer and energy budget approaches and employs a wider range of data inputs, is more reliable for shorter (monthly) estimates (Ward, 1967; Mather, 1978).

Recovery of Artifically Drained Wetlands

Hydrologic changes induced by artificial drainage are a function of the time that has elapsed since draining occurred. Previous research has shown that peatlands drained for silviculture will eventually regain the hydrologic characteristics that were altered by drainage. Much of this research has been conducted in Finland. In addition to being a leading exporter of timber, Finland contains ten percent of the world's peatlands. Investigation of the long-term influence of forestry drainage on peatland hydrology is of particular importance in a country where nearly one-fifth of the total land area has been drained for silvicultural practices (Mustonen and Seuna, 1972; Seuna, 1980).

Seuna (1980) examined the drainage trends for two 5 km^2 basins in southeastern Finland. He analyzed 21 years of hydrologic data for both basins in a natural condition, then for an additional 18 years after one basin was drained for silviculture. During the 18-year post-drainage period, there was a decreasing trend in the influence of drainage. At the end of the period, Seuna found that mean annual runoff had decreased to pre-drainage levels. He attributed this to two factors: (1) an increase in evapotranspiration caused by a new stand of timber, and (2) the impairment of drainage ditches, which reduces water carrying capacity.

Impairment of drainage canals is an important consideration in the hydrologic recovery of artificially drained areas in eastern North Carolina. The general trend of eastern North Carolina drainage canals is toward loss of conveyance capacity. Because of uncontrolled vegetative growth and loss in canal cross-sectional area due to erosion, Swicegood and Kriz (1973) found water-carrying capacity losses in eastern North Carolina canals ranging from 17 to 61 percent over a six-year period. Such rates would require complete channel modification within 8 to 10 years.

Physical processes involved in canal degradation have been described by Stephenson and Steila (1982). They found drainage canals in the Great Dismal Swamp to be susceptible to such factors as slope failure or slumping, the formation of alcoves at channel toeslopes, and the hydrostatic popout of soil from channel slope banks, known as a "moyocker". Such phenomena led to a 70 to 80 percent loss of water-carrying capacity within only 2 years.

Phillips (1988) presented a preliminary model of channel deterioration. This exponential model provides a method for estimating the change in channel roughness (Manning's n) over time, which corresponds to changes in the canal bankfull discharge. Increases in the roughness coefficient are most pronounced in highly vegetated channels, which are prevalent in canals constructed for silvicultural practices.

Phillips stresses the importance of channel degradation in wetland hydrologic modelling. Considering that environmental management policies are often based on such models, the recovery of wetland hydrologic regimes based upon channel deterioration may become a factor in policy decisions. This is particularly relevant when a determination of wetland status is required.

Wetland functions that are characteristic of natural systems, such as flood retention and nutrient filtering, are known to be disrupted when artificial drainage occurs (Ash et al, 1983). Recovery of artificially drained wetlands implies a return to these and other functions over time; i.e., a return to stability. Phillips (1985a), in a theoretical assessment of the stability of artificially drained wetlands, determined that the rate of channel degradation following canal construction is the key variable determining wetland hydrologic stability. When channel degradation is rapid, the system responds by quickly reaching a new equilibrium. The hydrologic regime of the wetland will remain in a metastable condition, unless the drainage canals are periodically renovated. Rehabilitation of canal carrying capacity, including sediment removal and the clearing of debris, results in a recurrence of disequilibrium, or instability (Phillips and Steila, 1984; Phillips, 1985a; 1985b).

The concept of artificial drainage channel degradation inducing stability in the wetland hydrologic regime is important in evaluating the impact of hydrologic restoration in the Alligator Swamp. The influence, if any, of the installation of drainage control structures will be directly related to the viability of drainage canals within the altered and post-altered study areas.

CHAPTER V

METHODOLOGY

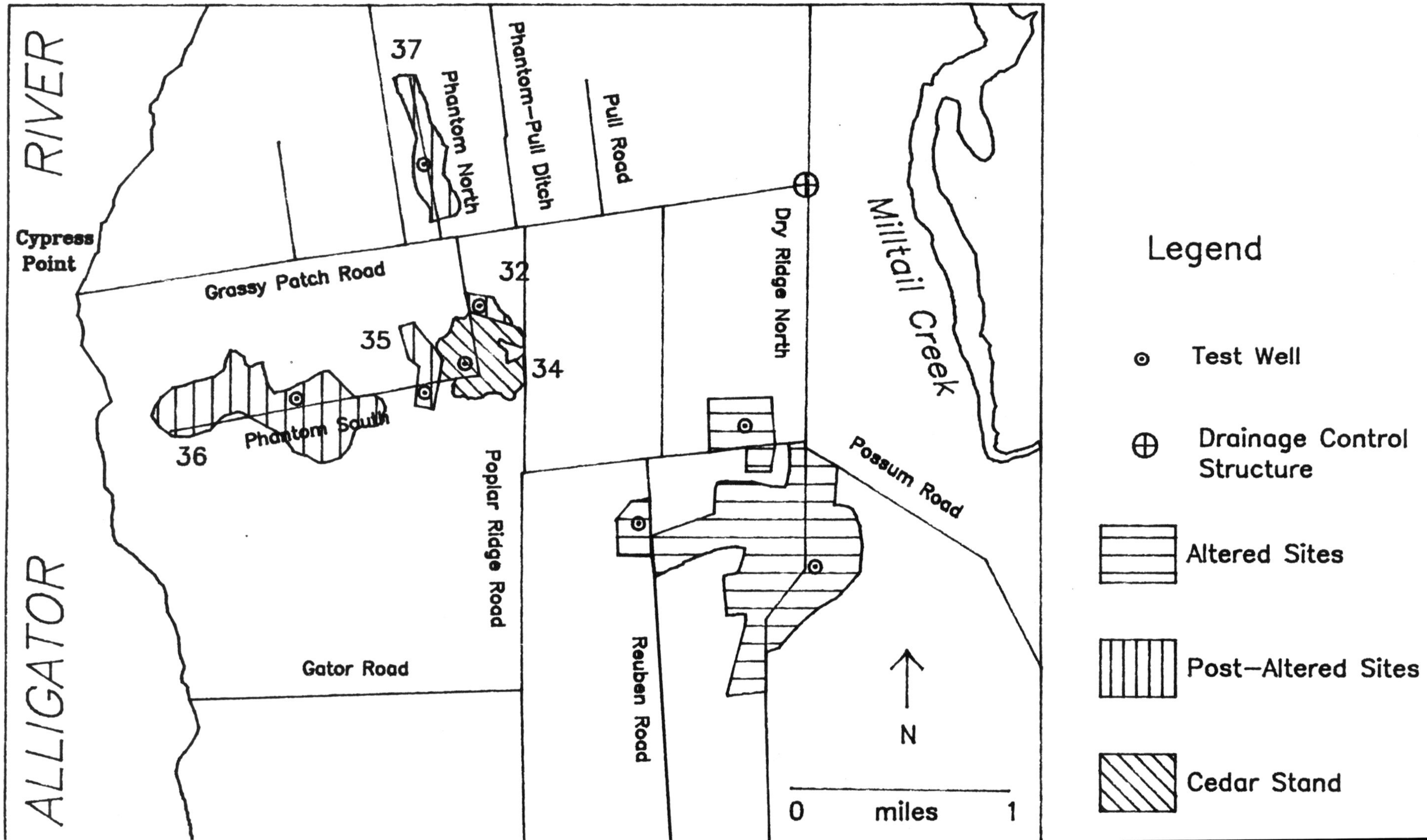
The study area lies within the Alligator River National Wildlife Refuge on the mainland peninsula of Dare County, North Carolina (Figure 2). The specific study sites within this area consist of timber tracts used by Atlantic Forest Products for the logging of Atlantic white cedar from 1980 to 1984 (Figure 3). These tracts were mapped by a registered land surveyor in 1979. These individual tract maps contain the coordinates and distances obtained by the surveyor and were verified in the field by use of a compass traverse (Compton, 1962). A reference traverse was plotted from the intersection of Pull Road and Grassy Patch Road to the intersection of Grassy Patch Road and the Phantom-Pull Drainage Ditch.

Eight sites were selected for the computation of water balances. These sites were chosen because logging tracts provide a homogeneous surface in which to examine water table fluctuations and soil characteristics. These areas are homogeneous in that they: (1) are clear-cut, therefore unshaded, areas; (2) were logged during 1983 and 1984; (3) contain logging slash throughout the area; (4) exhibit a high degree of surface disturbance; (5) lie within the same soil series (Pungo) as determined from the Dare County soil survey (Soil Conservation Service, 1989); and (6) lie within

Figure 3

STUDY AREA

showing location of sites and test wells



the same wetland classification as determined from National Wetlands Inventory maps. NWI maps delineate wetlands according to the classification scheme adopted by the U.S. Fish and Wildlife Service (Cowardin et al, 1979). Atlantic white cedar wetlands are classifed as System: Palustrine; Class: Forested; Subclass: Needle-leaved evergreen.

Sites 18, 20, and 21 are currently influenced by artificial drainage, and thus are representative of altered wetlands. Sites 32, 35, 36, and 37 lie within the post-altered Grassy Patch-Phantom area. Site 34 represents a natural area. It is an uncut white cedar stand containing specimens ranging from approximately 70 to 90 years of age. This range of ages was determined by taking core samples from the largest specimens with an increment borer. This site was located by examination of 1987 aerial photography and has been verified in the field.

Site 34 represents one of the last remaining stands of Atlantic white cedar that regenerated from the large-scale logging operations of the early 20th century (Peacock and Lynch, 1982). Site 34 lies within the Grassy Patch area.

The drainage area of each site was estimated using an assumptive method described by Phillips (1984). These drainage basins are delineated by existing drainage canals, roadbeds (which act as natural levees), and the natural water bodies of Alligator River and Milltail Creek.

Artificial drainage practices result in changes to the water balance components of a region, as well as changes in the properties of organic soils which predominate in the study area (Skaggs et al, 1980). The water availability of an altered area must be compared to that of a morphologically similar, undrained area for these water balance changes to be measured.

A model was chosen that could best assess the impact of hydrologic restoration in a canalized swamp system. Novikov and Gonchorova (1981) developed a model (henceforth referred to as NAG) for forecasting changes in the water balance following drainage. Because drainage decreases groundwater levels in a swamp, water availability (or water loss) can be evaluated from the evaporation changes that take place when water tables are lowered. Also, changes in runoff due to evaporation changes can be determined as the difference between potential evaporation from drained areas and evaporation from a natural swamp.

The methodology applied in this thesis, particularly the comparisons of water table fluctuations between natural, altered and post-altered wetlands is well-suited for the use of NAG, as evaporation is the primary mechanism for water table drawdown in these areas. Therefore, NAG will allow a assessment of the water availability in the hydrologically restored (post-altered) site based on evaporation differences between the canalized and natural areas.

The water availability of the altered area, R_{alt}, is given by

$$R_{alt} = R_{nat} - R_{ts} \tag{5}$$

where R_{nat} is the water availability in a natural area, and R_{ts} represents the total decrease in water availability from the onset of artificial drainage practices. It is generally known that drainage in peat soils causes considerable subsidence of the surface layer, though this impact decreases over time as well as with the depth of the peat layer (Jongedyk et al, 1954; Slusher et al, 1974; Yevdokimova et al, 1976; Skaggs et al, 1980; Lilly, 1981a, 1981b; Parent et al, 1982). Because of this subsidence Novikov and Goncharova (1981) maintain that estimates of changes in groundwater storage (a key water balance component) cannot be made by simply comparing water table levels before and immediately after drainage. Thus,

$$R_{ts} = R_t + R_{th} \tag{6}$$

where R_t represents the water content in the surface layer of peat lowered by subsidence. It is expressed as

$$R_t = \Delta d_t \times C_{nat} \tag{7}$$

where Δd_t represents the change in peat thickness after drainage (subsidence). The C_{nat} value is the mean soil moisture content within the cedar stand (Site 34), which represents the Atlantic white cedar wetland in its predisturbed condition. The term d_t, peat thickness before drainage (in meters), was calculated for values ranging from 51 to 90 inches (130 to 229 cm). This range corresponds to depths of organic soil (Pungo muck) indicated in the Dare County soil survey. It also coincides with observations made in the Hoffman Forest by Maki (1974), who reported peat thicknesses of 100-200 centimeters. The subsidence of the altered sites (Δd_t) was then determined by the empirical equation

$$\Delta d_t = 0.18 \times K \times d_t^{0.35} \times d^{0.64} \tag{8}$$

where d is drainage canal depth (m) and K is a coefficient of peat density (1.0 = dense peat, 1.4 = average peat and 2.2-2.7 = loose peat; Novikov and Goncharova, 1981). To determine if the NAG values agreed with other subsidence data from Lower Coastal Plain organic soils, subsidence values from Equation (8) were compared to results reported by Skaggs and Barnes (1976) and Tant (1979). Tant measured subsidence in Belhaven and Pungo soils in Washington County, North Carolina over a six-year period. Field experiments by Skaggs and Barnes (1976), followed by Skaggs (1980), were also conducted in Washington County at a site near Lake Phelps. Comparisons between their findings and the NAG subsidence equation are given in the Results chapter.

R_{th} represents moisture losses under artificial drainage due to moisture reduction in the upper peat layer after water table drawdown and is expressed by

$$R_{th} = (h_0 - h_t)(C_{nat} - C_{dr}) \qquad (9)$$

where h_0 represents height of the water table before drainage (mean water table elevation in the cedar stand, Site 34) and h_t is water table height after drainage (mean water table elevation of the altered or post-altered sites). C_{dr} represents the mean soil moisture content in the altered and post-altered sites (Novikov and Goncharova, 1981).

After drainage canals are plugged, the water level in the canal will eventually equal the level of the water table. This assumption is based on the observations of Gilliam and Skaggs (1981) and Daniel (1981), who observed that the water tables in organic soils drained for agricultural rarely fall to the depth of the canal. With water table elevations constantly higher than canal water levels, the difference in hydraulic head forces water from the adjacent soil to the canal. Furthermore, because of the low permeability of organic soils, drainage canals primarily serve as surface drainage. It is the outflow function of canals that keeps water table levels artificially low (Gilliam and Skaggs, 1981). Thus, the plugging of a drainage canal will serve to decrease the hydraulic gradient between the canal and the water table. Over time, gravity seepage into the canal will diminish, and the canal water level will fluctuate in response to precipitation and evaporation in the same manner as the water table (Daniel, 1981). Once the equilibrium between canal and groundwater surface levels is reached, water availability of the post-altered sites, R_{pa}, is determined by

$$R_{pa} = R_{nat} - (E_{canals} - E_{nat}) \times A_{canals} + R_t \qquad (10)$$

where E_{canals} is the evaporation loss from the drainage canals, E_{nat} is evapotranspiration from the natural area (Site 34), and A_{canals} is the surface area of canals, expressed as a percentage of total basin area. R_t was described in Equation (7).

R_{nat}, the water availability in a natural area, is given via the water balance approach (Mather, 1978). Water balances can be computed for areas of every scale, from a soil profile to a large watershed (Dunne and Leopold, 1978). For a small watershed, the generalized water budget can be expressed as follows:

$$P - ET - R \pm CSM \pm CWT = 0 \qquad (11)$$

where P is precipitation, ET is evapotranspiration, R is runoff, CSM is change in soil moisture, and CWT is change is groundwater storage. This expression of the water balance is similar to that of Pegg and Ward (1971). They conducted a hydrologic study for a small catchment in which CSM and CWT were monitored to estimate ET.

Each component of the water balance was evaluated separately, with runoff treated as a residual. Precipitation, air temperature, temperature of the dew point, and wind speed data were obtained from the Dare County Bombing Range, which has recorded daily weather data since 1986. The Range weather station is located approximately 3.7 km (6 mi) from the study area.

Evaporation was calculated by the Penman formula, which combines the mass transfer and energy budget methods for calculating evaporation from open water. The mass transfer method calculates the upward flux of water vapor from the water surface, while the energy budget method balances incoming and reflected solar radiation with the energy used for evapotranspiration and energy transfers and storage (Dunne and Leopold, 1978; Shaw, 1988). The basic expression of the Penman formula can be written:

$$E_o = \frac{\Delta H}{\gamma} + E_a \qquad (12)$$

where E_o is the energy for evaporation, Δ represents the slope of the curve of saturation vapor pressure plotted against temperature, H is net radiation, E_a describes the contribution of mass transfer to evaporation, and γ is the psychrometric constant (0.66 mb/degree Celsius). Data necessary for the calculation of evaporation include mean air temperature (T_a), mean temperature of the dew point (T_d), hours of sunshine (n), and wind speed (u). T_a, T_d, and u were collected from the bombing range weather station. Values of n are normally obtained via the use of a Campbell-Stokes recorder. For this study, measurements of n were obtained from monthly summaries of climatological data from the National Weather Service office at Cape Hatteras, which is located approximately 74 km (46 mi) southeast of the study area. Computation of evaporation (in mm day^{-1}) using the four variables mentioned above was accomplished via the use of tables published by McCulloch (1965).

Potential evapotranspiration (PE) was calculated by the model developed by Koerselman and Beltman (1988). Developed specifically for calculating evapotranspiration in temperate zone peatlands, this equation describes evapotranspiration from fens (ET, mm day^{-1}) as a function of Penman's E_o (mm day^{-1}) on a monthly basis:

$$ET = 0.73E_o + 0.16 \tag{13}$$

Change in soil moisture (CSM) was measured from samples collected from the sites. These samples were stored in plastic zip-lock bags (to prevent moisture loss) for transport to the laboratory, where they were weighed, dried in an oven for 12 hours at 105, and reweighed to determine soil moisture content (Briggs, 1977; Shaw, 1988). Water content was calculated on an oven-dry weight basis then converted to a volume measure using the oven-dry weight per unit wet bulk volume (Boelter and Blake, 1964). Samples were collected weekly.

Changes in groundwater storage (CWT) were determined from fluctuations in the water table. Water table levels were determined from readings taken from test wells installed at the sites. Maki (1974) used a similar methodology for measuring water tables in pocosins. Each test well was installed as near as possible to the center of the logging site. Readings were taken twice-weekly from June 2 through September 1.

The test wells were made of 3.175 cm diameter PVC pipe cut into approximately 1.8 m (six-foot) lengths. Each end of the pipe was capped. The pipe was perforated by slits cut in a spiral pattern, approximately 15 centimeters apart, in a 1.2 m section of the pipe. The pipe was sheathed in nylon filter cloth to prevent soil from entering the pipe. The sheathed section of pipe was placed into a hole bored into the organic soil to a depth of approximately 2 meters using a large-bore auger designed for use in peat soils (Hodges, 1989). Water table levels were measured from the top of the test well, which protruded 25 to 30 cm above the ground. Depth from the surface was measured using a battery-operated sensing device, which consisted of a weighted electrode attached to a coated copper cable. The cable was incremented into 15.24 cm (six-inch) sections. Upon contact with water, the electrode would activate a meter in the control box of the device.

CHAPTER VI

EXPECTATIONS OF THE STUDY

The effect of the hydrologic restoration in altered white cedar bogs in the Alligator River National Wildlife Refuge will be analyzed by examining water balances for the individual clear-cut sites. As previously discussed, these water balances will be compared to those in sites where the drainage network remains unimpaired by control structures.

The obvious effect of installing drainage control structures is to essentially stop runoff from the canals, which would normally flow into the Alligator River and Milltail Creek. Although runoff will be measured as a residual in the water balance computations, a significantly lower value from this residual is expected in the post-altered sites. However, the degree to which drainage canal conveyance capacity has deteriorated may affect this value, and the expected difference could be considerably less.

Changes in water table levels for both altered and post-altered sites are related primarily to drawdown from evapotranspiration. Water table levels in the post-altered sites are expected to be higher because of the drainage control structure. Because the structures will block canal outflow, there will be an increased amount of water available for evaporation. However, both groups are assumed to have equal rates of evaporation and transpiration, as the altered and post-altered sites are comparable in vegetation, soils, and history of disturbance. Therefore, the altered group is expected to show more pronounced water table fluctuations and higher values of runoff.

Another important consideration when evaluating the water balances of the post-altered sites will be the effects created by roadbeds. Roadbeds in the Alligator River National Wildlife Refuge, as in most altered wetlands in eastern North Carolina, are constructed from the spoil created when drainage canals are cut.

While the impact of drainage ditches are well-documented, the effects of roadbeds on wetland hydrology are not fully understood (Laderman, 1989; Winter, 1988). Maki (1974) described the roadbeds in the Hoffman Forest of North Carolina as having noticeable barrier effects. He stated that the low hydraulic conductivity of the spoil material impedes infiltration from the canal. The roadbed, despite being composed primarily of organic material with an abundance of roots and buried logs, compacts to such a degree that it effectively serves as a dam, blocking lateral water movement.

Roadbeds certainly act as barriers to surface water movement. On several occasions, ponding of surface water at roadbeds was observed in the study area. With a sufficient fetch of wind, surface water washes against the banks of the roadbed, creating an environment for sediment deposition when the roadbed is oriented perpendicular to the prevailing winds.

The effectiveness of roadbed barriers is related to both the composition of the spoil material and groundwater hydraulics. If the water level in the canal is below the groundwater head, the pressure of the upwelling groundwater may move it into the canal, if the roadbed material is permeable enough to permit transmission. The upwelling water is also capable of lifting the colloidal mucks in the ditch and flowing around them. There is also some gravity drainage from the area adjacent to the canal. If the water level in the canal is above the groundwater head, gravity allows water to move in and out of the more permeable material, and presses the mucks against the canal bottom, restricting seepage (U.S. Fish and Wildlife Service, 1986).

Differences in water level fluctuations within the post-altered sites may be attributable to the aforementioned factors. This is particularly applicable to Sites 32, 35, and 36. Sites 32 and 35 are adjacent to the Phantom Canal, while site 36 is bordered by Phantom Road South.

Observations of water levels in the altered sites (18, 20, and 21) may also reflect the influence of roadbeds (Site 21 lies adjacent to the Dry Ridge North Canal). Because the drainage network of the altered sites remains unimpeded by control structures, these observations may provide a clue of the degree to which drainage canal water levels are stabilized with the water level of the Alligator River. Frost (1989) speculated that drainage canals may actually prevent extreme seasonal drawdown of water tables by stabilizing their levels with the larger water bodies of the Alligator River and Pamlico Sound. Uniformity in water table fluctuations in the study area could be attributable to this phenomenon, but drainage canal conveyance capacity could be a factor in preventing such stabilization.

Trends in water table fluctuations for the altered and post-altered study sites should be similar, although the residual runoff value should be lower in the post-altered sites. If the groundwater storage component of the water budget equation differs significantly, such factors as the effect of roadbeds and the physical condition of drainage canals must be considered.

In summary, this study expects to reveal the impact of hydrologic restoration in the Alligator Swamp through the analysis of groundwater fluctuations and water balance comparisons. These analyses will demonstrate that the hydrologic regime of the post-altered study area is similar to that of a natural, undrained area.

CHAPTER VII

RESULTS

NAG Calculations

The Novikov and Gonchorava (NAG) model (1981) forecasts water balance changes after drainage (see Methodology). It considers both the evaporation changes and surface subsidence that occur when water tables are lowered by artificial drainage practices. The resulting decrease in water availability from the onset of drainage, R_{ts}, is given by Equation (6). Water content in the peat surface lowered by subsidence, R_t, is a product of the NAG subsidence equation (8) and the mean soil moisture content of the natural area, Site 34 (C_{nat}). Equation (8) includes drainage canal depth, peat thickness, and a coefficient of peat density (K).

The dense sapric peat that is characteristic of Alligator Swamp histosols resulted in a peat density coefficient (K) value of K = 1.0. Pre-drainage peat thickness (d_t) ranged from 130 to 230 cm (51 to 91 in). This range was determined from observations by previous field researchers in eastern North Carolina swamps (including Daniel, 1981; Ingram and Otte, 1981, 1982; Otte, 1981; Lilly, 1981; and Ingram, 1987), as well as information from the Dare County Soil Survey (Soil Conservation Service, 1989).

Drainage canal depth (d) was determined by measurements taken at various intervals along the length of the canals. Depths averaged 2.29 m (7.5 ft); thus, $d^{0.64} = 2.29^{0.64} = 1.70$. When solving for the NAG subsidence values in Equation (8), where d_t ranged from 130 to 230 cm, subsidence of the peat surface after drainage ranged from 0.34 to 0.41 m (See Table 1). Given the 20-year period since the canals were cut, this range of subsidence values (1.70 - 2.05 cm yr^{-1}) agrees with earlier results by Skaggs and Barnes (1976) and Tant (1979). They estimated peat subsidence in North Carolina blacklands to average 1.02 and 2.70 cm yr^{-1}, respectively.

R_t, the product of subsidence and pre-disturbance soil moisture, ranged from 0.14 to 0.17 for the given range of thicknesses (Table 1). The other factor in Equation (6) is R_{th}, which represents the moisture loss in the upper peat surface from water table drawdown.

The range of values for R_{ts}, total moisture decrease as a result of drainage, are listed in Table 1. This range (0.107 m - 0.192 m) represents a forecast of the decrease in moisture (as a depth per unit area) from artificial drainage.

TABLE 1

SUMMARY OF NAG CALCULATIONS

Subsidence (Δd_t) for pre-drainage peat thicknesses (d_t) of 130-230 cm (51-90 in):

$\Delta d_t = 0.18 \times K \times d_t^{0.35} \times d^{0.64}$ where $d = 2.29$ m, and

Water content in subsided peat layer, R_t, where

$$R_t = \Delta d_t \times C_{nat} \quad (C_{nat} = 0.41)$$

d_t (cm)	Δd_t (m)	R_t*
130-141	0.34	0.14
142-152	0.35	0.14
153-165	0.36	0.15
166-179	0.37	0.15
180-191	0.38	0.16
192-206	0.39	0.16
207-221	0.40	0.16
222-230	0.41	0.17

Total moisture decrease as a result of drainage, R_{ts}, per site for range of R_t (0.14-0.17)

Water availability in the altered sites, R_{alt}, given as

$$R_{alt} = R_{nat} - R_{ts},$$

using the mean R_{ts} value, and where $R_{nat} = 0.551$

Site	R_{ts}*	R_{alt}*
37	0.143 - 0.173	-
36	0.107 - 0.137	-
35	0.147 - 0.177	-
32	0.143 - 0.173	-
21	0.162 - 0.192	0.374
20	0.146 - 0.176	0.390
18	0.142 - 0.172	0.394
	Mean water availability for the Altered Group:	0.386

---

*Value denotes depth per unit area.

TABLE 2

SUMMARY OF NAG CALCULATIONS (CONT'D)

Water availability in the post-altered sites, R_{pa}, given as

$$R_{pa} = R_{nat} - (E_{canals} - E_{nat}) + R_t$$

where $R_{nat} = 0.551$, $E_{canals} = 0.006$, $E_{nat} = 0.004$, and $R_t = 0.155$

$$Rpa = 0.704 \text{ m (depth per unit area)}$$

Final Results:

$R_{alt} = 0.386$

$R_{pa} = 0.704$

$R_{nat} = 0.551$

The predicted values of R_{ts} were applied to Equation (5) to determine R_{alt}. The average water availability of a natural area, R_{nat}, was calculated from the water balance of Site 34. Results from Equation (5) are summarized in Table 1. The mean water availability for the altered group, as the average of midpoint values for each site, was 0.386 m.

Water availability in the post-altered group was determined from Equation (10), which reflects the water availability in the natural area minus the mean canal evaporation and forest evapotranspiration values for the study period. Post-altered water availability also includes R_t, which represents water content in the layer of peat that was lost to subsidence. The final results for the NAG model (Table 2) showed that the water availability in the post-altered area (0.704 m) surpassed that of the natural area (0.551 m), with the lower mean value of the altered group (0.386 m) reflecting the decreased water availability expected in a drained system.

Use of the NAG model to determine water availability in the altered and post-altered areas required supporting data from the calculation of water balances, represented by Equation (11). Change in Soil Moisture (CSM) and Change in Water Table (CWT) were the field-measured components of the water balance. The results from the soil sample analyses and the test well observations and their application to the water balance are discussed below.

Soils

Values of soil moisture at 1 m were found to be very consistent (see Table 3). The significance of differences in the mean soil moisture content among the post-altered and altered sites during the sampling period was tested with the t-test (Davis, 1973). At the 95% level of certainty, no significant differences in soil moisture were found among the individual altered sites or the individual post-altered sites. Additionally, no significant difference was found between the two groups (See Table 4). Soil samples were analyzed for organic matter content to determine if a difference existed for this parameter. Again, no significant differences were found (See Table 4). As expected, t-tests reflected significant differences between the post-altered and natural groups and the altered and natural groups (See Table 4). The mean soil moisture percentage of samples collected from the cedar stand (Site 34) was 41.14% and mean percentage of organic matter was 6.46%. Throughout the sampling period, soil water values of the cedar stand samples did not differ significantly.

TABLE 3

SUMMARY OF SOIL DATA

	Percent Soil H_2O		Percent Organic Matter[1]	
	Mean	Standard Deviation	Mean	Standard Deviation
Altered Group				
Site 18	86.22%	2.81	11.10	0.15
Site 20	83.42	7.53	12.46	3.06
Site 21	89.89	1.73	9.00	1.10
Group	86.51	5.81	10.85	2.26
Post-Altered Group				
Site 37	88.44	1.29	10.91	1.19
Site 36	89.13	0.62	10.27	0.49
Site 35	79.95	7.22	11.05	0.67
Site 32	88.88	0.97	10.44	1.01
Group	86.60	5.18	10.68	0.85
Cedar Stand				
Site 34	41.14	8.47	6.46	1.73

---

[1] Percentage of organic matter at field H_2O.

TABLE 4

RESULTS OF T-TESTS
SOIL DATA

Percent Soil H_2O

	N	Mean	Std Dev.	DF	Sig. Level	Crit. Value	t Value
Post-Altered	16	86.60	5.18	26	0.025	2.056	-0.05
Altered	12	86.51	5.11				
Post-Altered	16	86.60	5.18	4	0.025	2.776	-10.26
Natural	4	41.14	8.47				
Altered	12	86.51	5.11	4	0.025	2.776	-10.12
Natural	4	41.14	8.47				

Percent Organic Matter

	N	Mean	Std Dev.	DF	Sig. Level	Crit. Value	t Value
Post-Altered	16	10.68	0.85	13	0.025	2.160	0.27
Altered	12	10.85	2.26				
Post-Altered	16	10.68	0.85	3	0.025	3.182	-4.73
Natural	4	6.46	1.73				
Altered	12	10.85	2.26	7	0.025	2.365	-4.06
Natural	4	6.46	1.73				

Based upon this analysis, and the fact that no changes in soil moisture content were found during the sampling period, the CSM component of the water balance was zero.

Water Tables

Data from field observations of water table levels were analyzed using one-way analysis of variance, or ANOVA. ANOVA measures both the variance among sample means and the variance within samples. These two measures of variance constitute a statistic called the F-ratio (Davis, 1973).

No significant differences were found between sites within the altered group (Table 6). Within the post-altered group, there were no statistically significant differences among the individual sites, with the exception of Site 36. The extreme value of mean water table elevation for this site is attributable to the test well's close proximity to the roadbed. The deeper water table readings at Site 36 are indicative of the drawdown that occurs near drainage canals. The water table elevations at Site 36 resulted in a negative mean water table elevation for the post-altered group. This result was not plausible considering the blockage of the drainage canals in the post-altered area; therefore, Site 36 data were deleted for the water table elevation analysis.

Water table elevation data (Table 5) reveal that the range of mean elevations is 2.01 cm. The standard deviations for all sites in all groups are high; in most cases the standard deviation value surpasses the mean.

Overall results from the analyses of water table elevations did not concur with expectations. As can be seen from Table 7, no statistically significant differences exist between groups.

Line graphs illustrating the changes in water table elevations during the study period are shown in Figures 4, 5, and 6.

Calculation of Water Balances

Meteorological data from the Dare County Bombing Range were compiled for the precipitation and evapotranspiration components of the water balance equation. Rainfall for June totaled 18.99 cm (7.48 in), more than 8 cm (3 in) above normal. July rainfall of 15.06 cm (5.93 in) was slightly above the average value of 13.51 cm (5.32 in), while August precipitation measured 21.82 cm (8.59 in). The August figure was over 6.68 cm (2.5 in) above the monthly norm. Mean precipitation for each observation period was 4.65 cm.

TABLE 5

SUMMARY OF WATER TABLE DATA[1]

Water Table Elevation, in centimeters[2]

Altered Group	Mean	Standard Deviation
Site 18	1.31cm	6.89
Site 20	2.23	5.06
Site 21	5.37	7.29
Group	2.91	6.52
Post-Altered Group		
Site 37	1.50	3.73
Site 36	-6.01	5.29
Site 35	2.27	4.82
Site 32	1.61	5.29
Group	-0.23	5.86
Excluding Site 36	1.81	4.59
Cedar Stand		
Site 34	0.90	5.04

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[1] Figures reflect mean height of water table throughout the study period.

[2] Negative values indicate height of water table below surface.

TABLE 6

RESULTS OF ANALYSIS OF VARIANCE
WATER TABLE DATA

Altered Site Comparisons

Water Table Elevation, in centimeters

	N	Mean	Std Dev.	DF	Sig. Level	Crit. Value	F Ratio
Site 21	12	5.37	7.29	24	0.05	4.28	1.60
Site 20	13	2.22	5.06				
Site 21	12	5.37	7.29	37	0.05	3.28	1.35
Site 18	13	1.31	6.89				
Site 20	13	2.22	5.06	25	0.05	4.26	0.15
Site 18	13	1.31	6.89				

Post-Altered Site Comparisons

	N	Mean	Std Dev.	DF	Sig. Level	Crit. Value	F Ratio
Site 37	11	1.50	3.73	23	0.05	4.30	0.19
Site 35	13	2.27	4.82				
Site 37	11	1.50	3.73	48	0.05	2.82	0.17
Site 32	13	1.60	5.29				
Site 35	11	2.27	4.82	37	0.05	3.28	0.23
Site 32	13	1.60	5.29				

TABLE 7

RESULTS OF ANALYSIS OF VARIANCE
WATER TABLE DATA

Group Comparisons

Water Table Elevation, in centimeters

	N	Mean	Std Dev.	DF	Sig. Level	Crit. Value	F Ratio
Post-Altered	37	1.80	4.59	74	0.05	3.95	0.71
Altered	38	2.90	6.52				
Altered	38	2.90	6.52	49	0.05	4.02	0.95
Natural	12	0.90	5.04				
Post-Altered	37	1.80	4.59	86	0.05	3.10	0.72
Natural	12	0.90	5.04				

Evaporation (Penman's E_o, mm day^{-1}) was calculated from temperature and wind speed data collected at the Bombing Range weather station. Evaporation was highest in June (6.07 mm day^{-1}). Penman's E_o measured 5.79 mm day^{-1} and 5.86 mm day^{-1} for July and August, respectively. Mean evaporation for each observation period was 5.89 mm. The evapotranspiration component of the water balance was determined by Koerselman and Beltman's (1988) equation where $ET = 0.73E_o + 0.16$. Mean ET for each observation period was 0.45 cm.

Change in soil moisture (CSM) was zero, and change in groundwater storage (CWT) was computed from the fluctuation in water table elevation for each period.

The results of the water balance computations for the study sites correlated well ($r = 0.85$) with the excess moisture conditions reflected in the analysis of the water table elevations. Results from the water balance formula (Equation 11) are summarized in Table 8. Each site had an average of 4.54 centimeters of water remaining after accounting for the components of ET and CWT. This value represents the runoff residual. Runoff was not evident in the form of outflow from the deteriorated drainage system; therefore it was represented by canal storage. The range of the mean water balance values was small (1.01 cm).

TABLE 8

WATER BALANCE SUMMARY

Mean Precipitation: P = 4.65cm
Mean Evapotranspiration: ET = 0.45
Mean Water Table Flux: CWT = 0.33

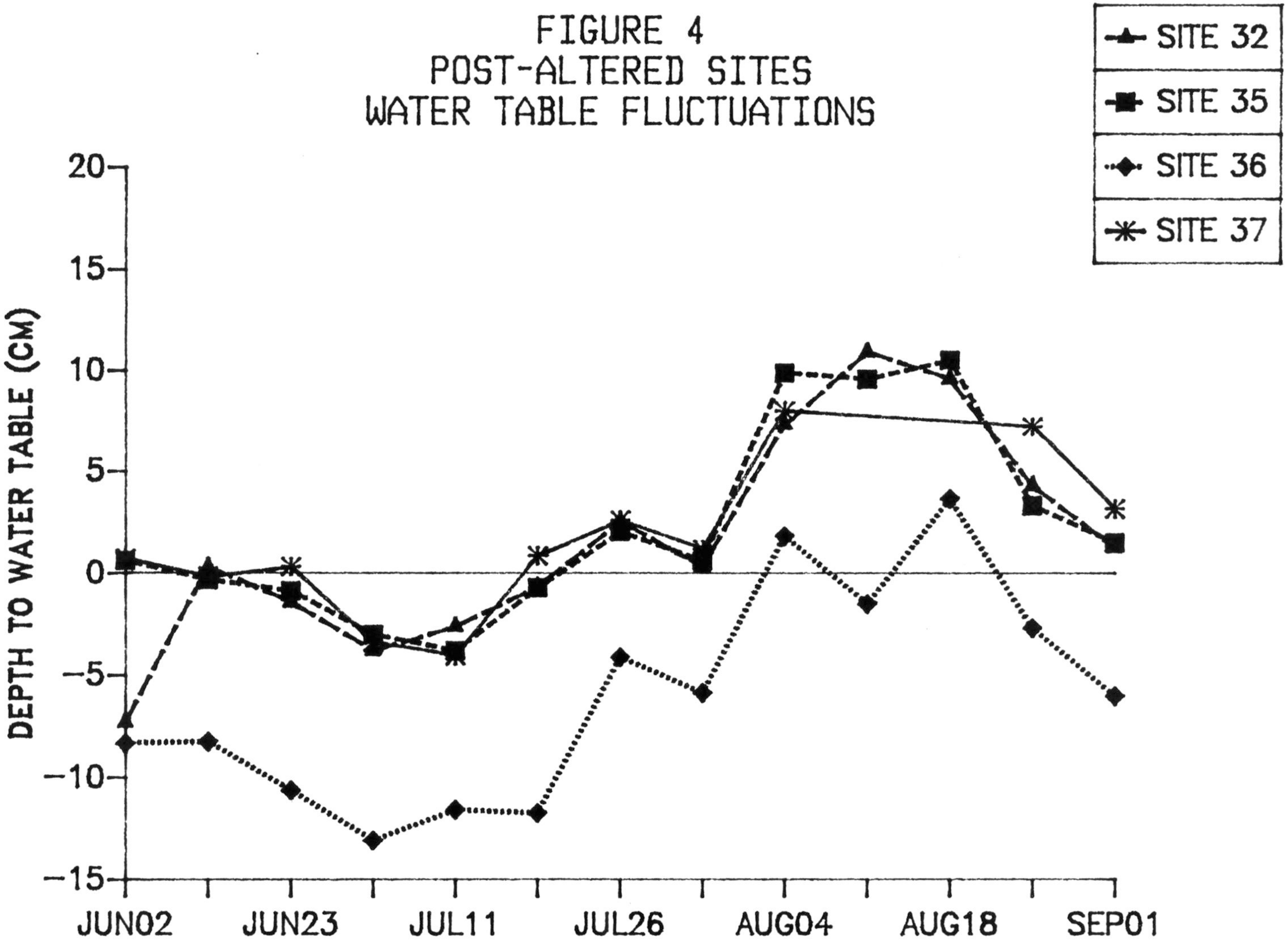
$(P - ET - R \pm CSM \pm CWT = 0)$

	Mean Water Balance[1]	Sum[2]
Altered Group		
Site 18	4.81cm	57.74cm
Site 20	4.17	50.04
Site 21	4.67	55.99
Group	4.55	
Post-Altered Group		
Site 37	4.00	39.97
Site 36	4.41	52.90
Site 35	4.29	51.48
Site 32	4.93	59.17
Group	4.42	
Cedar Stand		
Site 34	5.01	55.08
All Sites	4.66	

[1] Reflects water balance computation for each observation during the study period.

[2] Total water balance for the study period.

FIGURE 4
POST-ALTERED SITES
WATER TABLE FLUCTUATIONS



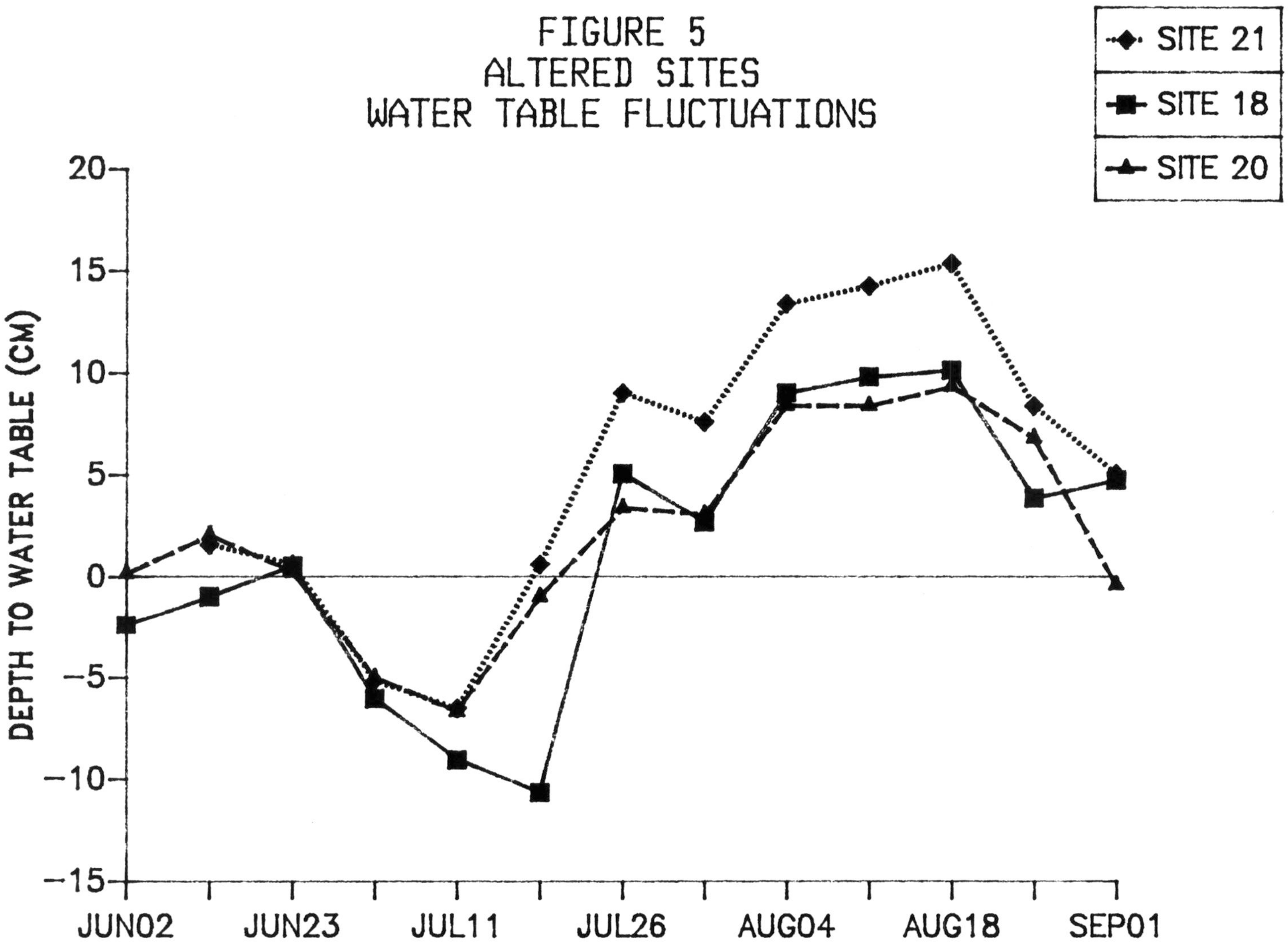
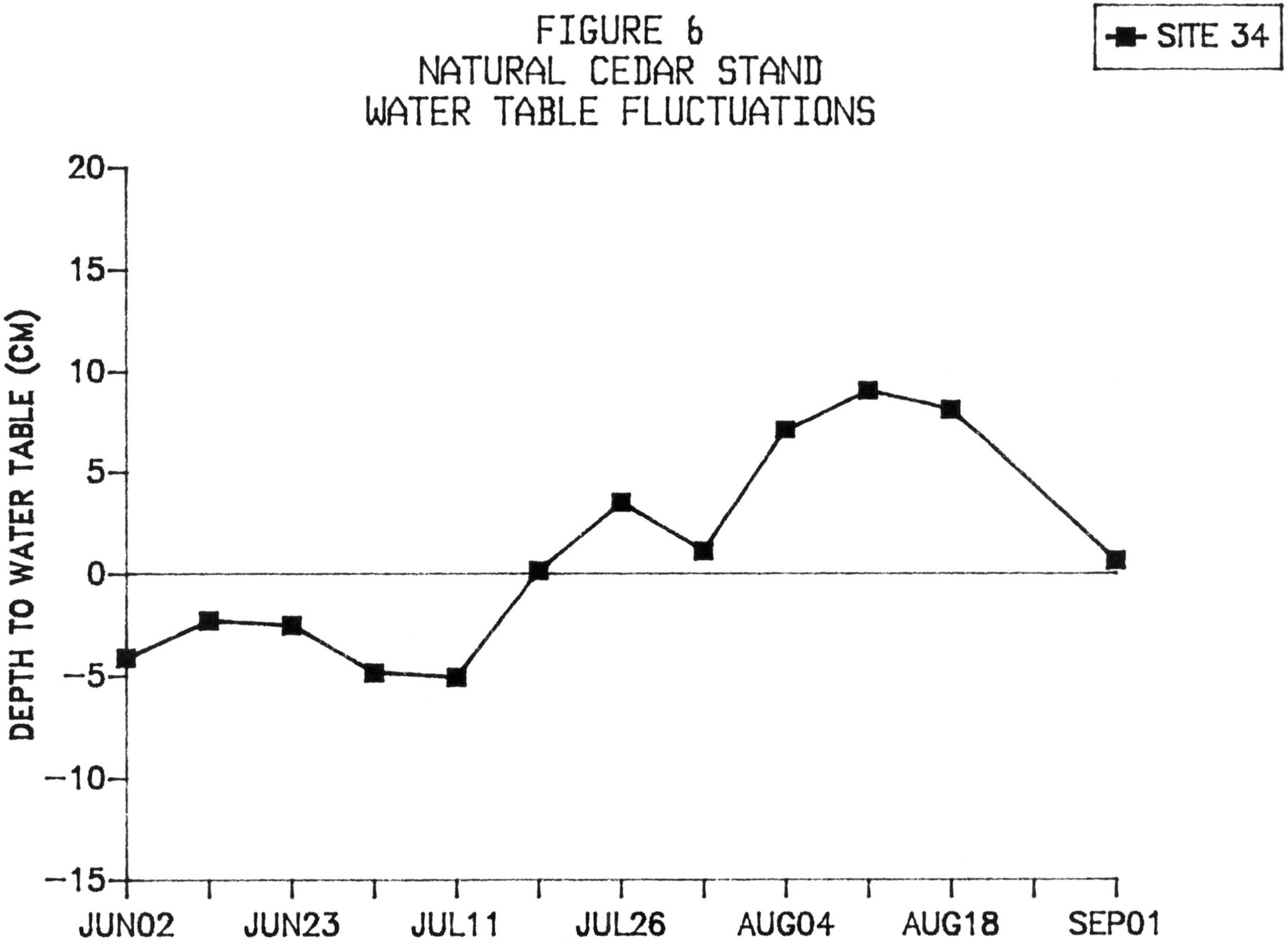


FIGURE 5
ALTERED SITES
WATER TABLE FLUCTUATIONS

FIGURE 6
NATURAL CEDAR STAND
WATER TABLE FLUCTUATIONS



CHAPTER VIII

DISCUSSION

The results of the NAG model showed that the post-altered white cedar wetland surpassed the natural white cedar bog in terms of water availability. The NAG model incorporated surface subsidence and the consequent surface layer moisture loss in forecasting water balance changes from artificial drainage practices. The water balance computations from field data did not consider these factors; thus, they did not reveal any significant differences between the altered, post-altered and natural areas. In addition, the results from water table elevation data (Table 5) showed that the mean elevation of the post-altered group was lower than that of the altered group. Such results clearly contradict the expectations of a higher mean water table level in the area where drainage canals were blocked with control structures.

The miniscule range of mean water balance values (1.01 cm) between three hydrologically distinct areas is a direct result of the wetter than normal conditions. The lack of differences between water balances, specifically the component of water table elevation, appears to indicate that a simple water budget approach to evaluating hydrologic restoration in this environment may not be viable, at least during periods of above-normal precipitation. However, this finding is inconclusive, and a more detailed examination is needed.

Following is a discussion of several factors that may account for the lack of significant differences in water balances between the three study areas, as well as the anomaly in the results of the test well readings.

Microtopography of the white cedar forest. The cedar stands that once existed at the post-altered sites probably possessed the hummocky microtopography that is characteristic of white cedar forests. Although care was taken to avoid the remnant hummocks during siting and placement of the test wells, it is possible that some of the post-altered site wells were placed in slightly elevated locations, which would have reflected lower water table elevations relative to the surface.

Topography of the pre-peat surface. Research by Oaks and Coch (1973), followed by Daniel (1981) and Otte (1981), clearly demonstrated that the pre-peat (Pleistocene) surface of the Albemarle-Pamlico region exhibits a well-defined dendritic drainage pattern. Characteristics of this pattern are broad, shallow stream valleys with corresponding interstream divides, or "highs" (Daniel, 1981). It is conceivable that the altered group may be located near such a high. A sloping mineral layer underlying the peat surface could cause locally higher water table levels, especially considering the barrier effects of the roadbed network.

Loss of canal conveyance capacity. As previously discussed, the canals within the Alligator River NWR have not been regularly maintained. Fallen trees and other vegetation were frequently observed in all canals. The nearest outlet constructed to drain the altered area to the Alligator River is a canal adjacent to Gator Road. This road lies within one of the most inaccessible areas in the Refuge. Gator Road has overgrown with vegetation to such an extent that it is no longer passable by vehicle. On one visit, the Gator Road canal was not seen initially because of the dense growth. When it was finally located, saplings and dense vegetation were thriving within the canal itself. Clearly, this outlet no longer functions as such. Given these conditions, the canals within the altered area are no longer viable along their entire length. Such a situation could result in elevated water table levels within this area.

Conditions of excess moisture. Ordinarily, late summer is a period of high evaporative demand and pronounced drawdown of the water table, with relatively lower amounts of rainfall. However, during the 3-month study period June through August, precipitation at the Dare County Bombing Range totaled 55.94 cm (22.02 in). Normal precipitation for this period, based on long-term data from the National Weather Service stations at Cape Hatteras and Elizabeth City, is 39.12 cm (15.4 in). The large amounts of rainfall resulted in standing water throughout the study area. Because vegetation moisture requirements were fulfilled and moisture was not limiting, actual evapotranspiration equaled the potential rate. Thus, the similar trend in water table elevation changes among all sites excluding Site 36 (Figure 7) is accountable primarily to losses by evaporation, particularly if runoff is negated by loss of canal conveyance capacity.

The loss of canal conveyance capacity may best explain the failure to meet the expectations of this study. As stated earlier in a discussion of the recovery of artificially drained wetlands, the rate at which drainage canals deteriorate may be the key variable that determines the resumption of natural hydrologic functions. Phillips and Steila (1984), Lilly (1981), and Phillips (1985) have speculated that canal degradation serves as an adjustment mechanism for artificially drained wetlands to reach a new state of equilibrium in the years following drainage. Results of this thesis suggest that drainage canals within the altered and post-altered study areas have deteriorated to such an extent that the hydrologic regime has reached a state of stability. This condition will persist as long as the canals are not cleared or reexcavated.

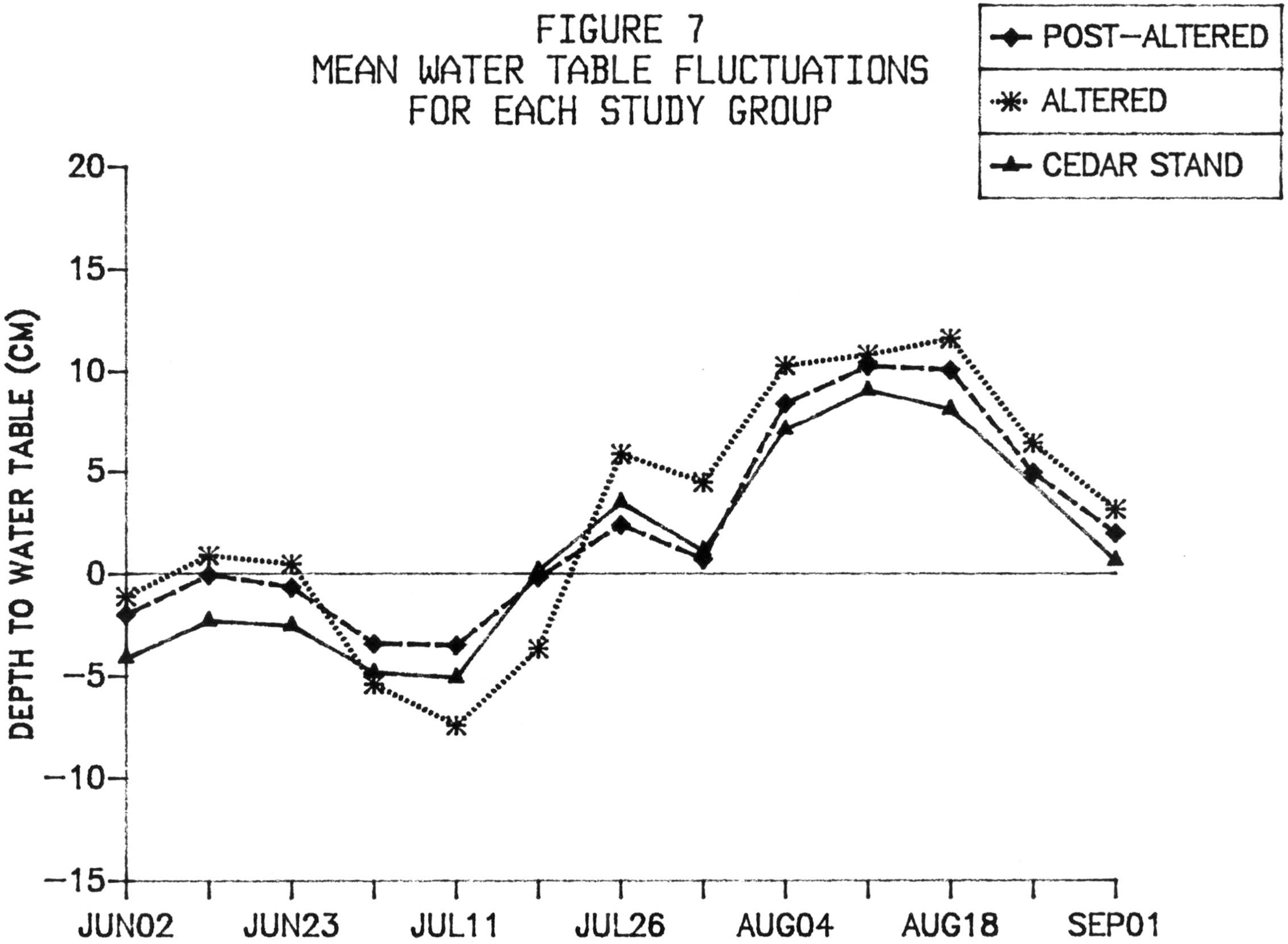


FIGURE 7
MEAN WATER TABLE FLUCTUATIONS
FOR EACH STUDY GROUP

This finding is consistent with previous research in eastern North Carolina drainage systems. Swicegood and Kriz (1973) reported losses of conveyance capacity that were significant enough to require complete channel modification within 8 to 10 years following drainage. Stephenson and Steila (1982) found conveyance capacity losses of 70 to 80 percent only 2 years after drainage.

The deterioration of drainage canals within the study area has, therefore, led to the adjustment of a new hydrologic equilibrium. This resumption of stability implies a restoration of 'natural' hydrologic functions. Furthermore, the results of this thesis suggest that natural occurrences such as vegetative growth, sedimentation, slumping, and debris clogs can restore hydrologic equilibrium in an artificially drained ecosystem. Consequently, the installation of drainage control structures has little, if any, effect on a system where the most significant loss of water is vertical, via evapotranspiration, rather than the horizontal flow through drainage canals.

The effects of artificial drainage on Atlantic white cedar regeneration in the Dare County mainland have not yet been documented (Moore and Laderman, 1989). However, the drier soil conditions caused by water table drawdown from artificial drainage will increase the likelihood of fire events that favor the establishment of pocosin vegetation. (See Chapter IV - Pocosin Communities: Successional Relationships for a discussion of pocosin and white cedar wetland vegetative regimes.) Historically, environmental conditions have favored the continued presence of Atlantic white cedar, as macrofossil evidence has indicated (Otte, 1981). However, the role that spontaneous fires, lightning, saltwater intrusion, and hurricane windthrow played in originally opening habitat for white cedar colonization is completely obscured by the area's history of drainage and logging (Moore and Laderman, 1989).

The results of this thesis suggest a restoration of natural hydrologic functions. It is premature to speculate as to whether this restoration is a benefit or a hindrance to white cedar regeneration, particularly when preliminary data show a wide variance in seedling density among the clear-cut stands (Robert E. Noffsinger and Joy A. Schmertmann, unpublished data, 1989). This variation implies that different conditions existed within the substrate of the forests prior to logging. Nonetheless, analysis of soil samples showed that the logged stands were nearly uniform in soil moisture and organic matter content. Because all of these stands were logged during a one-year period (summer 1983-1984), this uniformity can be attributed to post-disturbance deposition of the mineral elements in the soil. If the uniformity in soil properties exists in other sites logged during the same time period, it could serve as a starting point of investigation into the variance in white cedar regeneration among the cut stands.

The results of this thesis will serve as a preliminary basis for further observation of hydrologic conditions in the Alligator River National Wildlife Refuge. The water balance approach to evaluating hydrologic restoration can be a low-cost, field-reliable, and successful application, and if applied over a longer and more 'average' period, the results should yield an accurate assessment of the effects of drainage control in a non-agricultural setting.

Research Needs

Sampling of water table elevations should be conducted over a longer time span to reflect periods of average precipitation. Normally, summer and fall are the driest months. Because the groundwater head would be below the canal water surface, an analysis of water table fluctuations during periods of extended drawdown may yield a more pronounced effect from drainage canals. Canal water levels should also be monitored during the sampling period. A comparison of the fluctuations between canal surface levels and groundwater levels could demonstrate how canal conveyance capacity has been reduced. Although the viability of drainage canals has been determined intuitively from the analysis of water table fluctuations, a definitive conclusion could be obtained by measuring flow velocity along various reaches of the canal. Such data would accurately determine canal outflow response to individual rainfall events.

In addition, sampling from an expanded test well grid would produce a more accurate assessment of water table elevation changes. For example, test wells placed along a traverse from the Phantom South canal to the Grassy Patch canal to Milltail Creek would allow for an evaluation of the water table on a larger scale, and would be especially useful in evaluating the influence of these features.

The subsidence equation in the NAG model produced results that were in agreement with prior eastern North Carolina studies; however, surface subsidence should be determined empirically over an extended period of time, using a methodology similar to that described by Tant (1979). In one hypothetical scenario, Site 32 would be cleared of logging slash prior to permanent marker installation while Sites 37 and 35 would remain uncleared. This would allow a determination of the effect of logging debris on subsidence.

Soil sampling should be expanded to include the remaining logged areas within the Refuge, as well as the remaining white cedar forests. Such data could enhance current knowledge of white cedar moisture regimes, particularly soil moisture status. More extensive soil moisture data are needed to improve the regression models formulated by Golet and Lowry (1987).

Evapotranspiration from extant white cedar forests and logged areas should be determined empirically through the use of lysimeters. The results of such a study would be a valuable addition to literature on evapotranspiration from bogs, and particularly North Carolina white cedar wetlands. Additionally, such results are needed to calibrate the Koerselman and Beltman model (Koerselman and Beltman, 1988).

CHAPTER IX

SUMMARY AND CONCLUSIONS

This thesis attempted to evaluate the hydrologic changes that occurred after the installation of drainage control structures in canals in the Alligator River National Wildlife Refuge. A water balance approach was used with changes in soil moisture and water table elevation fluctuations being the field-measured components of the water balance.

Three categories of study sites were evaluated. The first category consisted of former Atlantic white cedar (Chamaecyparis thyoides) stands that were harvested during the summers of 1983 and 1984. These logged sites were located within an area where drainage control structures and canal backfilling were used in an effort by the U.S. Fish and Wildlife Service to restore the natural hydrologic regime. This category was denoted as post-altered, since drainage canals would no longer serve as outlets for runoff. The second category consisted of similarly logged white cedar stands within an area where the drainage canals remained unimpeded. Such an area was representative of forested wetlands where artifical drainage has altered the local hydrology. Water balances from the post-altered and altered study areas were compared to determine the effectiveness of hydrologic restoration. This comparison was contrasted with the water balance of an extant white cedar stand which contained specimens ranging from 70 to 90 years of age. This natural area represented the third category.

The goal of this research was to determine whether the hydrologic regime in the post-altered wetlands was more similar to the undrained or to the altered (canalized) study areas. A hydrologic model that forecasts changes in the water balance following artificial drainage was used in conjunction with the water balance measurements to assess the impact of hydrologic restoration in a canalized swamp system.

It was expected that the installation of drainage control structures would result in higher elevations and less pronounced fluctuations in the water table within the post-altered area. Final water balance computations showed that no significant differences in the field-measured components existed between the post-altered and altered study areas. However, results from the hydrologic model revealed that the water availability in the post-altered area closely approximated that of the natural area. This was attributed to the fact that the model incorporated surface subsidence and the consequent surface layer moisture loss from artificial drainage. The water balance computations did not consider these factors.

Two factors were primarily responsible for the lack of significant differences in the water balances of the altered and post-altered groups. Abnormally high precipitation resulted in water table levels above the surface during most of the study period; thus, soil moisture percentages remained constant, and the similarity in water table elevation changes could be attributed to evaporation losses. However, the loss of canal conveyance capacity best explained the failure to meet research expectations. Based upon earlier research and the fact that canals within the study area are not maintained, it was concluded that the natural deterioration of drainage canals has led to the recovery of the natural hydrologic regime. Consequently, the installation of drainage control structures had little or no effect on a system where the most significant loss of water is vertical, via evapotranspiration, rather than the horizontal flow through drainage canals.

The Fish and Wildlife Service staff of the Alligator River National Wildlife Refuge will continue the work begun in this thesis. While management policies cannot be implemented on the basis of the preliminary findings presented here, some implications to refuge management have become apparent. Specifically, when drainage canals are not maintained, it has been shown that they naturally lose their conveyance capacity over time. This degradation could eliminate the need and expense of backfilling and control structure installation in some areas. Also, the monitoring of water table elevations will be important as the Refuge staff continues its analysis of white cedar regeneration.

The preservation and encouragement of the Atlantic white cedar forests of the Alligator Swamp is one of the main objectives of the U.S. Fish and Wildlife Service. However, this goal is overshadowed by the specter of sea level rise. Indeed, a slight increase in the base level could virtually eliminate the gravity drainage function of the canals, regardless of their condition. The gradual flooding that would result could make soil conditions intolerable for white cedar seedling survival. Furthermore, such flooding would lengthen the recurrence interval between fire events which are critical to the regeneration of white cedar.

Clearly, the future of Chamaecyparis thyoides in the Alligator Swamp is, at best, tenuous. It is hoped that the information presented in this thesis can make a contribution toward assessing this threat, and be a useful tool for sound management of the endangered species of the Alligator River National Wildlife Refuge.

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