CTE

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Final Report

An Assessment of Highway Impacts on Ecological Function in Palustrine Forested Wetlands in the Upper Coastal Plain of North Carolina

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16. Abstract

Population growth and transportation needs have resulted in highway construction in wetlands. The impact of highway construction on wetland ecological functions is not clearly understood, although the alteration of wetland ecological functions such as hydrologic flux and storage, biological productivity, and nutrient flux by highways has been observed. An assessment of the effects of fill and culvert-type highway crossings on ecological functions in palustrine forested wetlands in the Upper Coastal Plain of North Carolina was performed. A combination of functional indicators, which were used as surrogate measures of wetland function, were tested in the field, and the most predictive ones were utilized in a response surface model. Using the functional indicator field data, a general functional assessment strategy/methodology was refined and functional response surface models were created to synthesize assessment results. Differences were detected between study areas upstream and downstream of the crossings and a reference area for five wetland functions—hydrologic flux and storage (180-245%), plant productivity (35-80%), biogeochemical cycling and storage (85-115%), decomposition (90-125%), and community/wildlife habitat (120-205%). Results from this exploratory study suggest that fill and culvert-type crossings disrupt or alter wetland ecological functions, mainly upstream. This study can be used in formulating a needed regional functional assessment protocol for this wetland type in the Southeast United States. Continued research is required to refine wetland response models, and to determine the effects of highway construction at increasing time intervals after highway completion.

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ABSTRACT

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CHAPTER ONE

INTRODUCTION AND OVERVIEW

WETLAND STATUS

It is estimated that 87 million hectares of wetlands existed in the conterminous United States before European settlers arrived (Roe and Ayers, 1954). Dahl and Johnson (1991) determined that by the mid-1980's approximately 42 million hectares remained, or 48 percent of the original estimate.

Clearing and drainage for agriculture, development, forestry, and flood control are major reasons for wetland conversion. As long ago as 1850, the federal government helped speed conversion by encouraging wetland drainage with the Swamp Land Act. There have also been substantial conversions from one wetland type to another. Dahl and Johnson (1991) estimated in the decade from the mid-70's to the mid-80's, roughly a half million hectares were converted from one wetland class (National Wetland Inventory classification scheme) to another. Most of the change was from swamps to shrub-scrub and marsh wetlands.

The rate of conversion of wetlands for agriculture and forestry has substantially decreased in the past decade due to restrictions mandated in Section 404 of the Clean Water Act (1972, 1982), and the Swampbuster provisions of the Food Security Act of 1985. But pressure to fill in or drain wetlands has not decreased. As the population continues to grow and transportation needs increase, highway construction poses an increasing threat to the quality and functionality of our remaining wetland resources.

WETLAND VALUES AND FUNCTIONS

It was well into the nineteenth century before scientists began to study wetlands exclusively and document, measure and interpret their ecological function on the landscape (Mitsch and Gosselink, 1994). As more information became available, the view of wetlands as wastelands began to change.

Wetlands began to be recognized for their benefits to man - their values. Today many wetland values are acknowledged by society. A list of wetland values is included in Table 1.1. These values are the result of natural physical and chemical processes that occur in wetlands, hereafter referred to as wetland functions (Richardson, 1994). Wetland functions result from the unique position of wetlands on the landscape, which is the convergence of uplands with deepwater habitats.

Table 1.1. A list of general wetland values. (from Richardson, 1994)

1. Flood control (conveyance), flood storage
2. Sediment control (filter, filter for waste)
3. Water quality
4. Waste water treatment system
5. Nutrient removal from agricultural runoff and wastewater systems
6. Recreation
7. Open space
8. Visual-cultural
9. Hunting
10. Preservation of flora and fauna (endemic, refuge)
11. Timber production
12. Shrub crops (cranberry and blueberry)
13. Medical (streptomycin)
14. Education and research
15. Erosion control
16. Food production (shrimp, fish, ducks)
17. Historical, cultural, and archaeological resources
18. Threatened, rare, and endangered species habitat

As such, wetlands are ecotones, and assume some characteristics of both upland and deepwater habitats. This unique combination of topographical and hydrologic characteristics gives rise to the wetland ecological functions listed in Table 1.2.

WETLAND LEGISLATION

There is no single wetlands regulatory law. Wetlands are protected by an array of laws and regulations, which are enforced by more than one governmental agency.

President Carter's Executive Order 11990 directed all federal agencies to minimize the destruction, loss or degradation of wetlands, and to preserve and enhance their natural beneficial values.

Table 1.2. A list of general wetland functions. (from Richardson, 1994)

1. Hydrologic flux and storage
   a. Aquifer (groundwater) recharge to wetland and/or discharge from
   b. the ecosystem.
   c. Water storage reservoir and regulator
   d. Regional stream hydrology (discharge and recharge)
   e. Regional climate control (evapotranspiration)
2. Biological productivity
   a. Net primary productivity
   b. Carbon storage
   c. Carbon fixation
   d. Secondary productivity
3. Biogeochemical cycling and storage
   a. Nutrient source or sink on the landscape
   b. C, N, S, P, etc. transformations (oxidation/reduction reactions)
   c. Denitrification
   d. Sediment and organic matter reservoir
4. Decomposition
   a. Carbon release (global climate impacts)
   b. Detritus output for aquatic organisms (downstream energy source)
   c. Mineralization
5. Community/wildlife habitat
   a. Habitat for species (unique and endangered)
   b. Habitat for algae, bacteria, fungi, fish, shellfish, wildlife and wetland plants
   c. Biodiversity

Section 404 of the Federal Water Pollution Control Act of 1972 (PL 92-500) and subsequent amendments are known as the Clean Water Act, which give the U.S. Army Corps. of Engineers along with the U.S. Fish and Wildlife Service (USFWS) and the Environmental Protection Agency (EPA) jurisdiction over all dredge and fill activities in "waters of the U. S." which includes virtually all wetlands. In addition to the federal law, coastal wetlands are regulated in North Carolina by state legislation, including the Dredge and Fill Act (NCGS 113-229) and the Coastal Area Management Act (NCGS 113A-100 *et seq*). Under the Clean Water Act, the unavoidable destruction of wetlands for development usually requires that the lost wetland acreage, and the accompanying functions and values, be replaced or mitigated by the developer. Dredge and fill activities associated with highway construction are regulated by these laws. And while the Federal Highway Administration (FHWA) is responsible for providing safe and efficient highways under Order 5660.1A of the Department of Transportation (DOT), the FHWA is committed to the protection , preservation, and enhancement of the nation's wetlands to the fullest extent practicable during the planning, construction and operation of highway facilities (Rossiter and Crawford, 1983).

HIGHWAY CONSTRUCTION AND WETLANDS

Where highways cross wetlands, engineers can employ two different structures. A pile-supported structure (bridge) can be used to elevate the highway over the wetland, or the crossing can be filled in with soil, and relatively narrow culverts can be used for water transference under the fill material roadbed. Occasionally a wetland basin is crossed utilizing fill on the outer, less deep fringes, and a piling supported bridge is used to cross the main channel.

It is recognized that pile-supported highway structures for crossing wetlands are ecologically less disruptive than fill and culvert-type crossings (Shuldiner et al, 1979). However, pile-supported bridges are more expensive, and fill and culvert crossings are still commonly built over streams of small to moderate flow in North Carolina (Cindy Bell, NCDOT, personal correspondence).

Earth fill can produce detrimental ecological effects in wetlands (Shuldiner et al, 1979). A partial list is included in Table 1.3.

Table 1.3. Partial list of effects of earth fill on wetland ecological functions. (from Shuldiner et al, 1979)

1. Modification of the hydrologic regime
2. Increased water turbidity
3. Alteration of water circulation patterns
4. Removal of natural filtration systems
5. Alteration of biological productivity
6. Alteration of nutrient flux

Modification of hydrology is likely the most ecologically disruptive consequence of highway crossings on wetlands. Fill and culvert crossings can impound water upstream, and they may also concentrate flows, deepening some channels and altering the natural distribution of water in the wetland (Parizek, 1970).

Any alteration of hydrological regime or flow pattern can have effects that extend beyond the highway structure both up and downstream, depending on the hydrology of the stream and the size of the wetland. Detrimental effects such as timber die-off (Stoeckeler, 1965) have been attributed to ponding upstream of fill and culvert crossings. How the altered hydrological regime impacts the other wetland functions such as biological productivity, biogeochemical cycling and storage, decomposition, and community wildlife and habitat has not been quantified. What has become apparent is that wetland values such as water quality, timber production, food production, and recreation may also have been diminished by highways.

There is a paucity of information on what effects highways may have on wetland ecological functions and how severe and geographically far-reaching those effects are. There are no explicit methodologies or procedures for the accurate quantification and assessment of highway impacts on wetland ecological function. At the present there is no accepted, reliable procedure for quantifying and assessing highway impacts on wetlands, nor is there an accurate, data-based framework for determining how much mitigation is required for highway impacts on wetlands.

OBJECTIVES OF RESEARCH

The objective of this research is to assess the impacts of highway construction on wetland functions. The study focuses on existing highway crossings of palustrine forested wetlands in the Coastal Plain of North Carolina. Specifically, we investigated and measured highway impacts on five key wetland functions. These functions are hydrological flux and storage, plant productivity, biogeochemical cycling and storage, decomposition and community/wildlife habitat (Richardson, 1994). An array of functional indicators for each wetland function was selected and used to quantify wetland functional levels. A framework is developed, based on the indicators deemed most valuable and easy to determine, which can be used to assess past highway construction impacts on palustrine forested wetland functions. This method was developed to predict future highway impacts on wetland functions *a priori*. In this phase of our research we attempted to formulate a functional assessment framework for testing future highway crossing construction effects. This study is also intended to provide a baseline from which to formulate strategies for framework development for other types of wetlands in the future.

In Chapter Two a brief description and analysis of mitigation is presented along with the rationale for using functional assessment as a tool for achieving quantifiable mitigation results.

Chapter Three is a description of a functional assessment procedure, which explores different functional indices alternatives and methodologies. It was executed in an attempt to quantify the impacts of fill and culvert-type road crossings on Riverine (Lower perennial) wetland systems in the Upper Coastal Plain of North Carolina.

CHAPTER TWO

FUNCTIONAL ASSESSMENT OF A DISTURBANCE ON WETLAND ECOLOGY-A GENERAL APPROACH

INTRODUCTION

Wetland Regulatory Legislation, Mitigation and Functional Assessment Relationships

Since settlement began, it has been estimated that the contiguous United States has lost 53 percent of its wetlands due to drainage and other human activities (Mitsch and Gosselink, 1993). The magnitude of this loss becomes clearer when the many wetland values to society are considered. Until the last 25 years, wetland conversion was viewed as progress - but subsequent scientific inquiry into the nature of wetlands has shown that wildlife habitat, primary production, water quality, nutrient storage and recycling capabilities, and many more wetland values and benefits are lost or compromised when wetlands are converted to terrestrial ecosystems.

There is no single wetland protection law, rather, a collection of legislation combines to monitor and regulate wetland development. The major law regulating wetland development and conversion is Section 404 of the Federal Water Pollution Control Act of 1972 (PL 92-500), which empowers the U. S. Army Corps of Engineers (USACOE) to oversee all dredge and fill activities in the "waters of the United States", which by definition includes virtually all wetland areas. When unavoidable impacts to wetlands are deemed to be significant enough, the ACOE may require the developer to "mitigate" or replace/restore wetlands elsewhere to compensate for lost wetland values.

The National Environmental Policy Act (NEPA) of 1969 describes mitigation alternatives for projects receiving federal money, and is used as a guideline for other mitigation. Under NEPA there are five mitigation options:

1. Avoidance of the impact by not taking a certain action or part of an action.
2. Minimization of the impacts by limiting the degree of magnitude of an action and its implementation.
3. Repairing or restoring the impact on the affected area.
4. Reducing or eliminating the impact over time by preservation and maintenance operations during the life of the action.
5. Compensating for the impact by replacing or providing substitute resources.

The goal of *mitigation* is the retrieval/replacement of lost wetland *functions* due to development or disturbance, but up until recently mitigation success was judged primarily on restoration of a single function, wildlife habitat. There was no accepted method for quantifying and characterizing hydrologic flux and storage, biological productivity and the other wetland ecosystem functions. Now more specific and comprehensive recovery of wetland functions is viewed as the appropriate measure of mitigation success (Ken Jolly, USACOE, personal correspondence). But for many years mitigation was generally conducted by attempting to restore the hydrology, and by replacing selected native plant species in hope that key or important fauna would be able to use the area for habitat. In coastal states such as North Carolina, wetland development and subsequent mitigation is regulated through a combination of Section 404 and state regulations. The following list and brief analysis of mitigation types describes the four older, more widely employed, mitigation types;

1. Avoidance or minimization of damage, which is the most logical type from an ecological standpoint, because dependence on technology to replace natural wetland values is decreased.
2. Habitat restoration, which is usually not ecologically comprehensive but restores attributes conducive to the survival of certain wildlife species.
3. Compensation or replacement, which is probably the most common type of mitigation being practiced today, consists of compensating for the loss of habitat (function) by intensive management on available lands either on or off-site. The quality of the habitat to be used as replacement is frequently analyzed using the U. S. Fish and Wildlife's Habitat Evaluation Procedure (HEP). This procedure quantifies habitat for chosen species in terms of "habitat units" which are used as a basis for computing replacement areas.
4. Artificial habitat construction is the recreation of damaged habitat, with total reliance on technological and scientific skill to replace lost habitat (function).

There are different forms of compensation or replacement mitigation. *In-kind replacement* requires that the same type of habitat, which was lost, be replaced either on or off-site. *Out-of-kind replacement* allows a different type of habitat to be used as a replacement for lost habitat. *Equal replacement* is the replacement of lost habitat with an equal amount of habitat units on or off-site. The habitat units can be for the same or for different species as those on the impacted site. *Relative replacement* is the replacement of habitat based on the relative value of one or more species.

In-kind, on-site mitigation is preferred, but when site limitations preclude it (which in practice is usually the case), off-site equal or relative replacements are allowed.

This type of flexibility can be viewed as a strength, but with latitude in choice comes risk. The likelihood that the lost habitat will not be equitably replaced is increased as off-site and out-of-kind replacement is undertaken. Trade-offs between what is lost and what is replaced force value judgments that are usually reached and defended based on HEP analyses. They are often further altered by judgments by the regulating agencies, which are not backed by scientific data or predictive models. Therefore one of the perceived strengths of the mitigation process, flexibility in replacement type, can also become a weakness as the risk of losing habitat increases with the out-of-kind, off-site type of habitat replacement.

The most extreme and risky form of wetland mitigation is creation, where technology and science are utilized in an attempt to recreate a destroyed ecosystem. The failure rate of created wetlands to date has been high (Roberts, 1993). Problems with wetland creation include the failure to recreate the desired hydrology, low fertility of the created wetland soil, invasion by exotic plant and tree species, lack of reestablishment of lost plant and animal species due to isolation, and lack of specific and clear ecological goals for success.

Society perceives wetland mitigation as having failed when the values associated with the destroyed wetland are not adequately recovered by the mitigation. In all cases the values associated with a wetland such as flood conveyance, wildlife habitat, food and timber production, and water quality are the result of restoring wetland functions (Richardson, 1994). Wetland functions are the natural physical and chemical attributes of a wetland that in combination eventuate wetland values. They include hydrologic flux and storage, biological productivity, biogeochemical cycling and storage, decomposition, and community/wildlife habitat.

When, as in the past, community/wildlife habitat or HEP units are the primary metric for quantifying wetland values, then the preponderance of wetland functional attributes are being ignored. It can be argued that wildlife habitat is an integrator of the other functions, and as such is a good indicator of overall wetland value. While this may be true, it is difficult to justify a procedure where so much potentially useful information for mitigation execution is not being utilized. Recently there has been an acknowledgment by regulating government agencies that a broader scope of wetland functions must be considered when wetland mitigation plans are being formulated, or the risk of failure is increased (Roberts, 1993). For the practice of mitigation to become more advanced and scientifically sound, functional characteristics must be measured and included in assessment procedures.

Currently there are very few methods in the literature for the quantification and assessment of wetland function. Brinson (1993) proposed a hydrogeomorphic classification system for wetlands which uses measures of three wetland functional components, geomorphic setting, water source and its transport, and hydrodynamics. Using this generalized classification system, a specific "profile" of wetland functions is developed for each wetland classification type. Appropriate functional indicator parameters are then chosen and measured to indirectly assess a wetland's functional level based on data collected at reference areas of the same wetland classification type. Brinson stops short of recommending appropriate functional indicators for measurement. Rather, he states that each wetland type is intrinsically different from the others and functional indicators should be chosen based on the unique natural ecological characteristics of the particular wetland classification type, which can be somewhat localized and specific. It follows that the next logical step in the development of functional assessment methodology is to generate a list of possible wetland functional indicators, and to begin testing them in the field in different wetland types, so that the methodology for assessment moves toward more specific ecological criteria and guidelines that can be applied under varying wetland conditions.

As development increases around the country, it is inevitable that wetland areas will experience greater pressure from society for conversion. Methods to accurately quantify existing wetland functions are needed so that lost wetlands can be meaningfully compared to the proposed mitigation replacements. In order to facilitate the replacement of wetland values lost by wetland development, and to ensure greater success in the future for mitigation projects, a more comprehensive approach to assessing wetland ecological processes, which includes all the recognized functional attributes, is needed.

OBJECTIVES

The objective of this chapter is to develop a general functional assessment procedure that will quantify all the major recognized wetland functions (hydrologic flux and storage, biological productivity, biogeochemical cycling and storage, decomposition and community/wildlife habitat). First, for each wetland function, a list of functional indicators will be suggested. From the indicators deemed most practical and best suited to a particular wetland type and mitigation situation, a quantitative framework will be constructed. This framework will then be used to develop a functional response surface model which can be used to model how a particular wetland's functions compare to an undisturbed reference wetland and other similar wetland types in the region.

METHODS

The projected goals of the functional assessment must be considered before any measurements are made. This will dictate how the assessment methodology should be implemented in the field. For instance, the assessment can be utilized to characterize a wetland ecosystem scheduled to be impacted. The levels of wetland functions can be determined and then used to quantify the degree of perturbation imposed on the site after alteration of the wetland has been completed. Specific mitigation goals can be created based on pre-disturbance functional levels. Alternatively, if a site has been previously impacted, then undisturbed wetlands of the same type should be located and used as reference areas. These reference areas can then be assessed and the measurements taken in them used as a basis for mitigation objectives.

As mentioned previously, the five ecosystem level wetland functions are hydrologic flux and storage, biological productivity, biogeochemical cycling and storage, decomposition, and community/wildlife habitat. Each function must be quantified in the field by measuring various parameters or functional indicators. Possible functional indicators that could be utilized singularly or in combination to describe the various wetland functions are listed in Table 2.1.

Table 2.1. Various field indicators of wetland function.

Hydrologic Flux and Storage:
1. water surface elevation (stage)
2. flow rate (Q)
3. depth
4. area of inundation
5. tidal amplitude

Biological Productivity:

Woody vegetation
1. basal area
2. density
3. species composition

Herbaceous vegetation
1. species composition
2. percent cover
3. biomass/net primary productivity

Biogeochemical Cycling and Storage:
1. soil nutrient analysis
2. water quality analyses
3. nitrogen and phosphorus concentrations
4. metals/toxics
5. dissolved oxygen
6. temperature
7. pH
8. salinity
9. tidal flushing
10. sediment traps
11. ^{137}Cs soil tracer method (Ritchie and McHenry, 1990)
12. dendrogeomorphic features-adventitious roots (Hupp and Morris, 1990)
13. oxidation/reduction potential

Decomposition:
1. cotton strip assay (Harrison et al, 1988)
2. litter bags
3. coarse woody debris or deadwood

Community/wildlife habitat:
1. USFWS Habitat Evaluation Procedures for various species of wildlife
2. macroinvertebrate surveys
3. woody species richness and diversity
4. herbaceous species richness and diversity
5. multivariate analyses

This list is not intended to exclude other functional indicator measurements, but to suggest some "best indicators" of wetland functions based on the literature and other recent studies (Brinson 1995, Richardson 1994).

RESULTS AND DISCUSSION

Data Synthesis and Presentation-The Response Surface

After functional indicator data have been gathered in the field, they must be synthesized for evaluation. Depending on the situation, the data from the pre-disturbance or reference wetlands can be used as the standard to which the post-disturbance data are compared. Richardson (1994) advocates a useful way to visualize a large amount of data at one time, with an ecosystem response surface (Figure 2.1). Figure 2.1 is a hypothetical example of how different wetland areas, or data that compares the same wetland before and after disturbance, can be presented. The circle represents the reference or undisturbed wetland indicator function, scaled to 100%. The axes for the wetland functions, e.g., hydrologic flux, and productivity, are also scaled to 100% and they are used to graphically portray how the disturbed wetland functional indicators compare to reference levels.

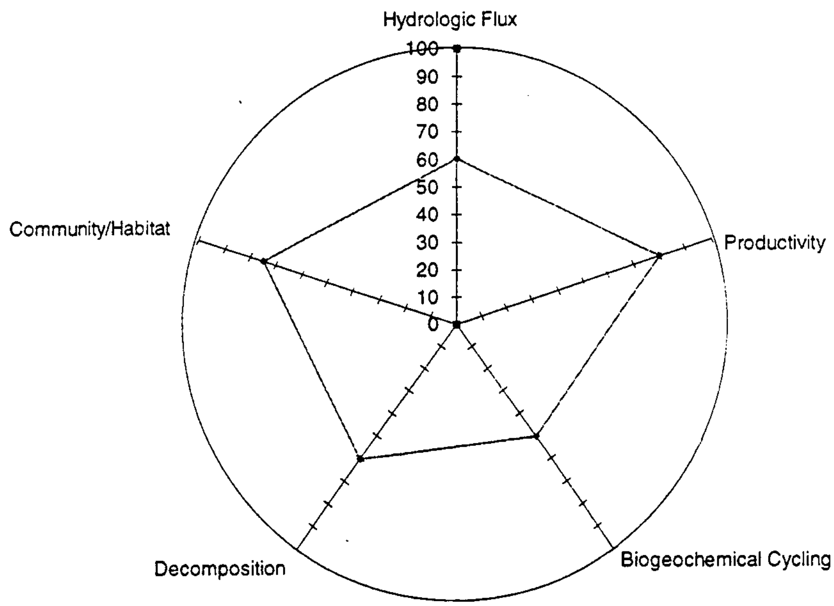


Figure 2.1. Hypothetical ecosystem response surface for comparing a disturbed wetland to an undisturbed or reference wetland (after Richardson, 1994).

For example, the hydrologic flux indicator, when measured at the disturbed wetland, is functioning at only 60% of the reference or undisturbed area, the productivity indicator at 80%, and so on.

This response surface is also useful for another comparison. If several areas need to be compared to a reference area or to pre-disturbance conditions for one functional indicator, then the axes can be used to display how the different areas compare to the reference area for that particular indicator (Figure 2.2). Graphs like Figure 2.2 can be constructed for all functional indicators measured so that each indicator level at each study site can be compared. For instance, if five disturbed sites are to be compared to a reference area functional indicator or if the areas need to be compared to their pre-disturbance condition, then it can be illustrated how these areas have changed based on a chosen functional indicator.

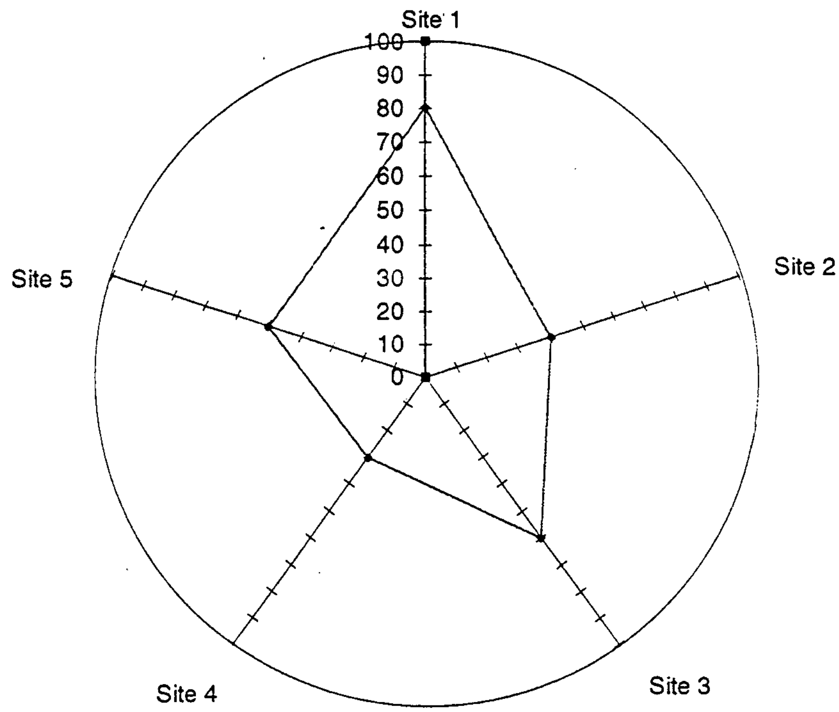


Figure 2.2. Hypothetical unifunctional response surface comparing the same indicator across five different sites. The reference area value is placed at 100% and is represented by the outer circle.

In Figure 2.2, if the indicator chosen is biogeochemical cycling, however measured, then the functional response surface indicates that Site 1 is functioning at 80% and Site 2 is functioning at 40% of the reference or pre-disturbance levels etc.,.

Regulatory Applications

This type of functional assessment procedure can be applied to many regulatory situations. Presently there are no formal finalized mitigation regulations (Ken Jolly, USACOE, personal correspondence) that can be applied to the process of determining mitigation replacement ratios, although there is an interagency working group whose goal is to present recommendations for the entire mitigation process (51 FR 41220, Nov. 13, 1986). The EPA has issued guidelines which the NCDOT follows, along with the regulatory agencies (Dennis Pipkin, NCDOT, personal correspondence). It specifies that for each acre of land impacted by development, two acres are required for restoration mitigation, three acres for creation mitigation, four acres for enhancement mitigation, and 10 acres for preservation mitigation.

The selection of the mitigation ratio required to replace lost wetland function is not a precise or quantitative process. Rather the ratios are arbitrarily determined by the participating federal and state agencies based on the rarity of an impacted wetland, the risk of failure in attempting to replace a particular type of wetand/wildlife habitat, and whether or not the replacement/mitigation areas are in-kind, and on site. The ratio is, in general practice, lower if they are in-kind and on-site, and higher if not (Ken Jolly, USCOE, personal correspondence). If the mitigation areas are not in-kind and on-site, then the replacement ratios are higher.

The functional assessment methodology presented here is intended to facilitate a rational, quantitative choice of compensatory mitigation ratios. By measuring the levels of all wetland ecological functions present in a disturbed wetland using functional indicators, and comparing them to pre-disturbance or reference levels, a more accurate estimation of wetland functional loss can be accomplished. This type of field-based, scientific assessment leads to a much more pragmatic, quantitative basis to rest decisions on, when justifying mitigation ratios. This method does not intend to make judgments on which function (or value) is more "important" or "desirable". The goal of this method is to establish a starting point for determining mitigation ratios which is backed by a thorough, scientific investigation of the ecological properties of a given wetland.

Another practical application of the ecological functional assessment methodology is monitoring the functional recovery of compensatory restoration/creation sites. Baseline data can be taken after restoration or creation projects are installed. From these data the progress of ecological functional recovery can be monitored by taking future measurements at the site. The progress at the site can be compared to reference area data to determine how effective the project is in restoring overall wetland ecological function in the long run, which should be any mitigation project's main goal.

The Need For Continued Methodology Development

Table 2.1 is not intended to be an exhaustive or comprehensive functional indicator list. The types of measurements listed are suggestions which may be desirable in some instances, but other measurements can and should be employed as the situation dictates. The type of wetland being assessed should suggest the particular type of measurements to be made. For instance, if a freshwater wetland is situated on a peat soil substrate, then water table measurements may be the most important hydrologic functional indicator. Peat depth may be an important biogeochemical storage functional indicator for that wetland. If the wetland is a brackish marsh, then peat accumulation may be a defining biogeochemical characteristic, etc. It should be noted here that when describing the community/wildlife habitat function, care should be exercised in the choice of species to evaluate. If less important species are selected for HEP or if more important species are excluded, mitigation goals can be biased and the success of the habitat mitigation effort may be jeopardized.

It is clear that when an array of functional indicator choices are available, varying results will be obtained depending on which indicators are utilized. The possibility of collecting data that are not as pertinent or valuable to the mitigation process because the "wrong" indicator was employed is real. This problem can only be solved through careful attention to how a functional indicator relates to a given wetland type. When the "correct" or best functional indicators for a particular wetland type are determined through field trials, the functional assessment process can be standardized, so that an acceptable level of consistency is reached and reproducible, dependable results can be used for mitigation.

Also, measurement and analysis of all five of the functional indicator categories mentioned in Table 2.1 can be expensive and requires a high level of skill, but it is important to include all areas for a comprehensive analysis of wetland function. The breadth of wetland ecological function is such that proficiency in multiple disciplines is required to assess a full array of functional indicators. From a regulatory standpoint this presents a problem, in that the personnel resources required to execute an in-depth functional assessment of a wetland may not be available. Addressing this problem, without ignoring the need for quality assessments, is challenging. The expense involved in performing a full-blown functional assessment also presents a real challenge. Time in the field, field equipment, laboratory facilities, and instrumentation all contribute to make the investment in a quality functional assessment of a wetland a substantial one. Clearly a simplified, less expensive functional assessment methodology, which is also a quantitatively sound one, is needed.

Wetland functional assessment procedures are new and virtually untested in the field. Determination of the best, most quantitatively sound set of functional indicators for a given wetland classification will have to be done on a case-by-case basis. This will of necessity have to be performed by qualified scientists at considerable expense. But once appropriate functional indicators have been chosen for a particular wetland classification, through rigorous testing, it is likely that indicators that require less formal training, are less expensive to collect and analyze, and which synthesize the information that the more difficult and expensive indicators convey, will be found. Complex and expensive indicators can then be replaced by simpler, more practical indicators, which are acceptable synthesizers of wetland function, and the original expense of gathering initial functional assessment indicators can be offset. The goal of the methodology presented here is the development of a list of relatively inexpensive and dependable wetland functional indicators that can be used in quantifying wetland functional loss for compensatory purposes. The field work and the evaluation and determination of which wetland functional indicators to use for a particular wetland is the next step in meeting the challenge.

When appropriate functional indicators are determined, response surface models can be used to illustrate how disturbed wetland ecosystems compare to pre-disturbance or reference areas. In addition, different areas can be compared for a particular wetland ecological function using a response surface. Finally, response surfaces can be used to track functional gains or losses in a created wetland.

CONCLUSIONS

Wetlands benefit society in many ways and have been acknowledged as possessing what are now commonly called wetland values. Wetland ecosystem functions- hydrologic flux and storage, biological productivity, biogeochemical cycling and storage, decomposition, and community/wildlife habitat, are the physical and chemical generators of wetland values. There are many laws and regulations concerning wetland disturbance and compensatory requirements, but there is no accepted methodology for assessing wetland ecological functions. Therefore there is no quantitative method for evaluating development impacts on wetland function, so that mitigation can be performed in accordance with the extent of functional disturbance. A functional assessment procedure, based on Brinson's general guidelines for placing wetlands in a particular hydrogeomorphic classification, can be used to place wetlands in comparative wetland types. Specific recommendations for possible functional indicators, which are not presently included in hydrogeomorphic classification, are needed with the basis for functional indicator selection being contingent on the type of wetland classification and the expertise of the workers. A response surface model is a tool that synthesizes and illustrates collected data in a way which facilitates meaningful comparisons between disturbed and undisturbed wetlands of the same type is presented. Meaningful comparisons between a disturbed wetland and the pre-disturbance condition or a reference area are possible with this methodology. It can assist in formulating reasonable mitigation ratios. It can also be used to monitor the functional development of created wetlands. For meaningful functional assessment to become a reality, much more work is needed to determine accurate and affordable functional indicators for the many wetland classification types. An application of this functional assessment approach is presented in Chapter Three.

CHAPTER THREE

AN ASSESSMENT OF HIGHWAY IMPACTS ON ECOLOGICAL FUNCTION IN PALUSTRINE FORESTED WETLANDS IN THE UPPER COASTAL PLAIN OF NORTH CAROLINA

INTRODUCTION

Freshwater wetlands perform many ecological functions on the landscape. Those functions include the retention of floodwater (Novitsky 1979 and Verry and Boelter 1979), sediment (Brinson et al., 1981a), and nutrients (Knight et al., 1984), biogeochemical transformations (Faulkner and Richardson, 1989), organic decomposition (Mitsch and Gosselink, 1993), and primary productivity (Mitsch and Ewel, 1979 and Brinson et al., 1981b). Palustrine forested wetlands, as classified by Cowardin et al. (1979) perform all of these functions to some degree (Mitsch and Gosselink, 1993).

The wetland ecological functions previously mentioned (also see Table 1.2) can singly or in combination convey to wetlands attributes that are valued by man. These attributes are often referred to in the literature as wetland values (Table 1.1). Wetland values include but are not limited to aesthetic and heritage value, outdoor recreation, wildlife habitat, flood moderation, water quality improvement, and aquifer recharge (Richardson, 1994 and Mitsch and Gosselink, 1993). Wetlands' benefits to man and wildlife have not been widely realized or acknowledged until the second half of this century, and only relatively recently have steps been taken to legally protect wetlands, so that the values associated with them are not diminished.

The Federal Highway Administration (FHWA) is responsible for providing safe and efficient highways, and under Order 5660.1A of the Department of Transportation

(DOT), the FHWA is committed to the protection , preservation, and enhancement of the nation's wetlands to the fullest extent practicable during the planning, construction and operation of highway facilities (Rossiter and Crawford, 1983).

In the coastal plain of North Carolina, population and commercial growth have increased, so that highway construction across wetlands is inevitable. The North Carolina Department of Transportation currently builds fill and culvert-type road crossings across smaller stream channels, rather than pile supported bridges, because they are less expensive. Fill and culvert-type crossings have been documented for altering wetland ecological functions (Table 1.3). In accordance with the law, construction of highway crossings over wetlands should minimize impacts that alter or reduce wetland functions.

Very few studies have been done on the effects of highway construction on wetland ecological functions. While studies have been done elsewhere on highway impacts on wetlands (Parizek, 1970, Scheidt, 1967, and Weber and Reed, 1976) none have been done in this area of the county, and scant recent work in this area is in the literature. In order to mitigate highway impacts on wetlands properly, in proportion to actual ecosystem disruption, an assessment procedure which indicates what type of ecological damage has been done, and to what degree, is needed. This study presents a method for accurately assessing the impact of highway construction on wetland ecological functions.

OBJECTIVES

The objective of this research is to create a methodology to investigate and assess the impacts of highway construction on wetland ecological functions. This study focuses on existing highway crossings of palustrine forested wetlands in the upper coastal plain of North Carolina. The goals are to investigate and measure highway impact on five key wetland ecological functions. These functions are hydrological flux and storage, biological productivity, biogeochemical flux and storage, decomposition, and community/wildlife habitat (Richardson, 1994). In this portion of the study, plant productivity is used to assess biological productivity (a macroinvertebrate study was also conducted by King et al., 1997). It should be noted that while decomposition is a biogeochemical process, for the purposes of this study it is treated as a separate function because of its importance to wetland ecosystem integrity.

Specific functional indicators are used to quantify ecological function levels. From these data, indicators are chosen that are the most effective gauges of wetland functional change. A framework or methodology for assessment is then developed, based on the indicators deemed most efficient, practical and cost effective. The goal is to use this methodology to assess past highway construction impacts on palustrine forested wetland ecological functions. In addition, it is hoped that this methodology can be used to predict any disturbance such as future highway impacts on wetland functions *a priori*, or to monitor the short-term/long-term effects of highway construction on ecosystem functions.

METHODS

Research Sites

Two research sites were located on Interstate 40 in Sampson County near Newton Grove, North Carolina (Figure 3.1).

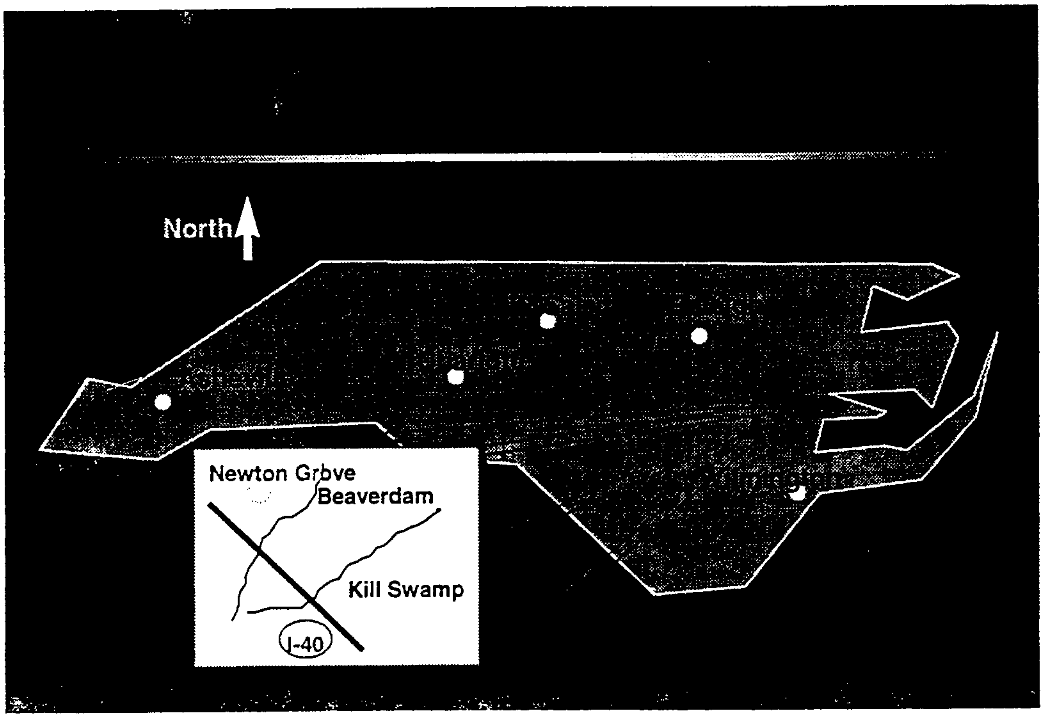


Figure 3.1 General location of study sites in the Upper Coastal Plain of North Carolina.

There the highway crosses two third order streams in the greater Cape Fear River basin. The northerly crossing is over Beaverdam Swamp and the southerly crossing is over Kill Swamp. The crossings are located approximately 1.8 kilometers apart (Figure3.2).

These sites were chosen for study for five reasons-

1- both crossings are fill-culvert type construction, which means that fill dirt is used to support the road over the floodplain and only one central box culvert and two smaller peripheral overflow pipes are used to convey stream flow from the upstream side under the road to the downstream side.

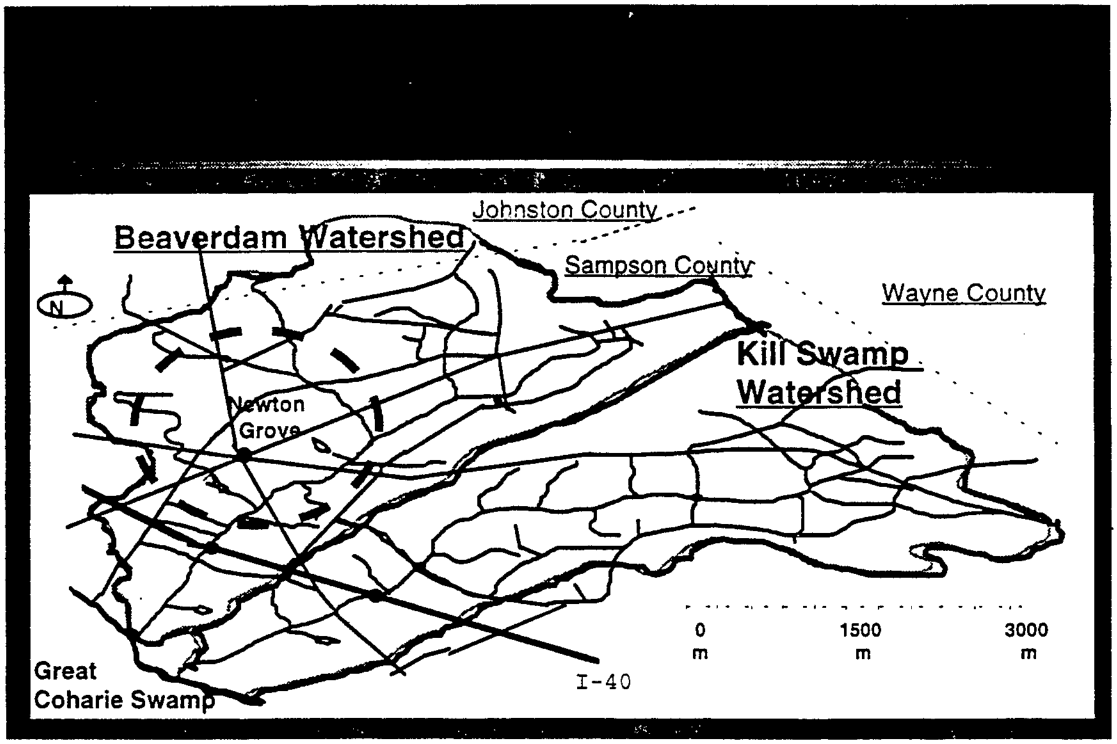


Figure 3.2 **Detailed map of study sites.**

2- they both have Bibb-Johnston association soils, which are Typic fluvaquents, and are classified as hydric soils (USDA, SCS).

3- watershed areas upstream are similar; Beaverdam Swamp equals 24.1 square kilometers and Kill Swamp equals 18.4 square kilometers.

4- the flow rates are similar for 50 year precipitation events; Beaverdam Q/50yr $=30.9$ meter^3 sec^{-1} and Kill Q/50yr$=28.3$ meter^3 sec^{-1} as calculated by NCDOT engineers.

5- considering the above information and the similar land use patterns upstream, they were deemed the best sites available in eastern North Carolina that had the potential for quantification of wetland ecological functions impacted by a recently constructed highway crossing ($\cong$7 years).

An inquiry was made to the NCDOT to see if any projects were scheduled for the research grant period in eastern North Carolina which would include fill and culvert-type crossings. There were no such crossings scheduled for the research grant period. Therefore it was necessary to use pre-existing fill and culvert crossings for this study. This situation precluded a "before and after construction" type research design, and necessitated the use of a reference area to ascertain what the undisturbed condition of the crossings may have been like before the road was constructed.

A reference area was located approximately 350 meters upstream from the crossing at the Beaverdam Swamp study area. It was chosen due to its relatively undisturbed condition, meaning the forest was free of indications of recent (detectable) logging activity. Also, indications of consequential, recent beaver activity were absent. The reference area was chosen after a thorough reconnaissance of the sites, and was deemed the next best reference alternative available since a pre-construction analysis was precluded.

Each crossing consisted of upstream and downstream impact study areas which started at the shore of the stream at the toe of the slope of the crossing and extended upstream or downstream approximately 220 meters. Where the crossing was constructed at Beaverdam Swamp, the width of the stream, where constant inundation occurs, is approximately 200 meters. The width of Kill Swamp where I-40 crosses is approximately 180 meters. In total there were five study sites located on the two crossings - one upstream and downstream from each bridge and the reference area upstream of the Beaverdam crossing.

Reference Area Analysis

To help verify that the impacted crossing areas and the reference area selected were similar before road construction began, aerial photographs obtained from the NCDOT were analyzed to determine percent crown closure for each area. The degree of stocking can be measured on aerial photographs by estimating the percent crown closure (Paine, 1981), which is defined as the percent ground area covered by a vertical projection of the tree crowns. If the percent crown closure (degree of stocking) is high for all the areas, and tree crowns in them are large enough to be considered approaching maturity, it is logical that the study areas had been in a relatively undisturbed state for a considerable time before road construction began. If this was the case, then at the least it can be confirmed that there was no visible sign of forest disturbance at the sites within 7-8 years of road construction.

The photograph scale was 1 cm =98 feet (1 inch = 250 ft) and the photograph size was 45 x 45 cm (18 x 18 inches). Using post-construction photographs and a Bausch and Lomb stereo zoom transfer scope, the exact position of the crossings was transferred to pre-construction photos taken in 1978, which was approximately eight years before the crossings were built. The study areas - upstream, downstream, and reference - were then delineated on the pre-construction photographs and percent crown closure was estimated following Paine (1981). The actual photographic percent crown closure measurements employed the tree cramming technique developed by Pope *et al.* (1961).

Functional Analysis

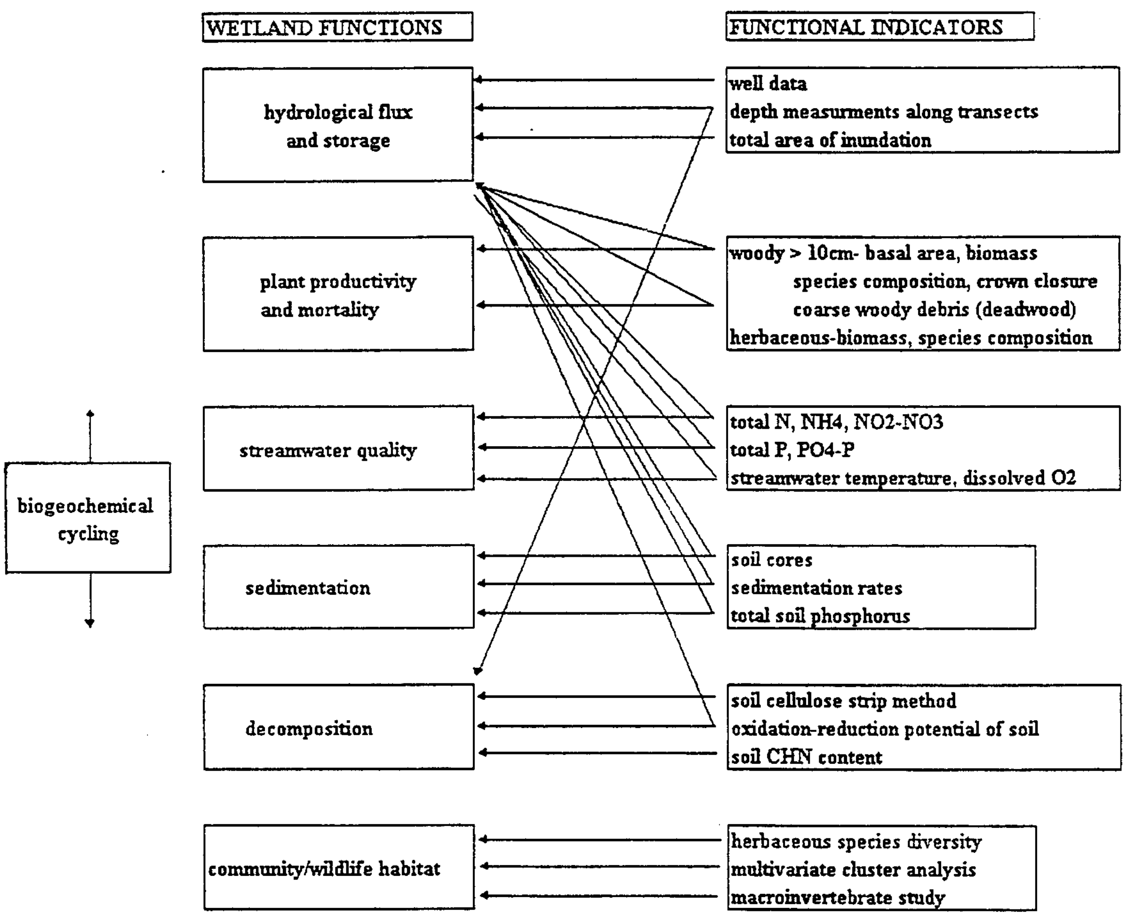
Five wetland functions were assessed. They were hydrological flux and storage, plant productivity, biogeochemical cycling and storage, decomposition, and community/wildlife habitat.

Attempting to measure wetland ecological functions for the purpose of comparing different wetlands is a recent development in wetland science. A practical solution to the problem of measuring complex, seasonal ecological functions is to use surrogate functional indicators, which reflect in some way the character and scope of the ecological function of interest. For example, a list of functional indicators for hydrologic flux and storage might include flow, velocity, depth, or stage etc.,.

For each wetland ecological function of interest in this study, there were several functional indicators chosen for measurement. The relationships between the wetland ecological functions of interest and the functional indicators chosen for measurement in this study, which are induced or are affected by them, are presented in Figure 3.3. The rationale for the selection of the particular functional indicators chosen to characterize the five wetland functions is presented below (following the order in Figure 3.3). For hydrological flux and storage stream stage, depth, and area of inundation or reach data from each area was needed, so the indicators are self-explanatory.

Basal area was used to describe stocking and species composition of the woody species onsite.

Figure 3.3. List of wetland ecological functions and the indicators them in this study.



Total above-ground biomass is a general indicator of total community production. Canopy closure is another measure of stocking and is an indirect indicator of how much sunlight reaches the forest floor, an important consideration for herbaceous productivity. Coarse woody debris is used as an indicator of the level of tree mortality, and indirectly the amount of stress. Biomass and species composition are very common descriptors of the herbaceous plant community.

Many parameters are available to an investigator to describe streamwater quality. We chose to measure nitrogen and phosphorus to determine if there was a nutrient gradient between reference, upstream and downstream study areas, and if the crossing was having an effect on nutrient loads. Stream water temperature and dissolved oxygen levels, which are important to aquatic life and which are important regulators of decomposition, were measured for the same reason, to determine if there was a gradient between study areas.

Sedimentation and total soil phosphorus were measured to see if areas which might have been experiencing higher sedimentation rates (i.e., areas upstream of the crossings or just downstream of the crossing embankment) were also phosphorus enriched, which has implications for plant and microbial growth.

Cellulose strips (sometimes called the cotton strip assay) were used to quantify the rate of organic matter decomposition in the soils, which indicates the rate of nutrient cycling onsite. Soil oxidation-reduction potential was measured to correlate soil oxygen conditions with the rate of decomposition. Soil carbon and nitrogen content also describe how the decomposition process has enriched the soil.

In this study, we concentrated on describing the structure of the herbaceous and woody plant communities and used benthic macroinvertebrate data to assess wildlife habitat. Herbaceous plant diversity provides information on the number of species present and the health of the plant populations. Multivariate cluster analysis is a tool which combines vegetative and environmental data such as water depth and soil chemistry and differentiates areas based on their similarities and differences. Macroinvertebrates have been used for many years as indicators of overall stream ecosystem health and habitat quality.

Two types of comparisons of functional indicators were made. First, functional indicator levels from upstream and downstream areas were contrasted. Significant differences in functional indicator levels between the upstream and downstream areas were considered to suggest that the fill and culvert-type crossing had impacted ecological function. The reference area was used to estimate undisturbed, baseline functional data for the two sites, and was used to compare the impact sites to what their pre-disturbance condition may have resembled.

The presentation of data concerning the five wetland functions and the functional indicators used to measure them in the rest of this chapter will follow the order contained in Figure 3.3.

1. Hydrological Flux and Storage

Water level recorders, model WL 40 manufactured by Remote Data Systems, Wilmington, N. C., were used to monitor stream stage levels. Two water level recorders were installed on each side of the crossings. One was located near the embankment. The other was located as far away as allowable to still be sighted from an elevation surveying instrument from the top of the highway embankment. The position of the farthest water level recorders upstream and downstream at each crossing was such that the effect of the crossing on the actual stream stage was minimized.

The elevation above sea level of the calibration mark on the water level recorders was determined by a NCDOT survey team to within 0.02 cm, so that the surface water elevation upstream and downstream from the crossings could be compared. The recorders were programmed to sample water elevations twice a day, at 8AM and 8PM.

Transects ran parallel to the road, across the topography of the stream and floodplain areas. At each site a transect was located approximately 20 meters (1 chain) from the road embankment. Additional transects were then located approximately 40 meters (2 chains) upstream or downstream from the first transect location. Random sampling points along the transects were located on a 40 meter (2 chain) interval. The size of the reference area was constrained to nine sampling points along three transects which were spaced approximately 40 meters (2 chains) apart. This was to ensure an undisturbed and homogenous buffer area around the reference sampling points, so that edge effects due to disturbance could be minimized.

Sampling points used for vegetation measurements were also used to determine water depth measurements along each transect at each site. Depths were measured during November and December of 1995. Total areas of inundation were estimated by measuring to the water's edge from the outermost points along each transect. Upstream/downstream inundated areas were then compared by connecting the transect endpoints and calculating the inundated areas.

2. Plant Productivity and Mortality

Woody vegetation basal area and species composition data were collected using a variable plot size timber inventory for trees ≥ 10 cm diameter breast height (dbh) upstream and downstream at both crossings as well as for the reference area. A ten factor prism was used. Sampling points were located as described in the hydrologic flux and storage section. From the timber inventory data the number of trees per acre by species and diameter class were calculated using standard forest inventory equations (Wagner, 1984). These data equations were utilized to estimate above-ground biomass of the woody plant component of the vegetation. Separate equations for *Taxodium distichum* (after Schlesinger, 1976) and the hardwood species (after Dabel and Day, 1977) were used to calculate biomass values for leaves, branches, and stems. These values were combined to compute total above-ground biomass. Species composition was determined using both biomass and basal area as indices. Woody vegetation percent canopy closure was determined at each herbaceous vegetation sampling point, as described below, using a spherical densiometer.

The length of all standing and downed coarse woody debris ≥ 10 cm was measured within a 15m radius of each plot center. Each snag or downed stem was placed into one of two diameter size categories, 10-30 cm, and >30 cm and an estimate of volume was made using these two size categories.

Herbaceous vegetation data was collected from ten sampling points upstream and downstream at each crossing, and from six sampling points in the reference area. Only six points were sampled in the reference area because its size limited the sampling area to nine total sampling points. Two sampling points per transect were sampled. Both points were located on the same side of the stream channel, so that one point was closer to the channel and one was closer to the edge of the swamp. Herbaceous species frequency was determined using the line-intercept method. Two random bearings were chosen and species were recorded every meter for twenty meters from plot center on each line.

Herbaceous biomass was determined using two 0.25 square meter clip plot sub-units for each plot. The location of the clip plots was determined by selecting a random bearing and distance, no greater than 8 meters from plot center. The vegetation from each clip plot was sorted by species, washed to remove any sediment, and then oven-dried at 70° Celsius and weighed. The mean of the weight of each species for each plot was determined.

3. Biogeochemical Cycling and Storage

Stream water quality

Nitrogen and phosphorus concentrations in the stream water were measured twice, on June 6, 1995 and on July 3, 1995. On 6/6/95 duplicate grab samples were taken at two sampling points on the transects adjacent to the crossings and the transects farthest away from the crossings, so that the perimeter of the study area could be compared to the interior, more impacted area. Also, grab samples were taken at one sampling point along two transects in the reference area. On 7/3/95 duplicated grab samples were taken at two sampling points from each of the four transects closest to the crossings, so that if nutrient gradients existed along the stream they would be detected. Also, grab samples were collected at two points along two transects in the reference area.

Stream water samples were analyzed following Environmental Protection Agency protocol for the analysis of water quality samples (USEPA, 1979). The samples were filtered through a 0.45 micron filter (except for the total nitrogen and phosphorus samples) and refrigerated until analysis (ammonium was analyzed within 24 hours). Nitrogen analysis was done on a TRAACS 800 spectrophotometer. Ammonium (NH_4^+) levels were determined by the Berthelot reaction. Nitrite-nitrate (NO_2 - NO_3) levels were determined by copper-cadmium reduction. After a persulfate digestion, total nitrogen levels were determined by hydrazine reduction.

Phosphorus levels were measured using a Beckman DU-64 spectrophotometer. Soluble reactive phosphorus (PO_4-P) and total P levels are determined by the Murphy-Riley phospho-molybdate blue complex reaction (total P levels were determined after persulfate digestion).

Stream water temperature and dissolved oxygen levels were measured at the sampling points along each transect using an Orion pH multi-meter during the spring and summer of 1996.

Sedimentation

Sediment accumulation upstream and downstream of the crossings was determined from the previously mentioned soil cores using ^{137}Cs as a soil marker.

This technique utilizes the historical fact that an enormous amount of above-ground nuclear testing took place in1964, before the practice was banned in the same year. ^{137}Cs, a product of the nuclear fission explosions, was blown into the atmosphere and then redeposited shortly after in rainfall. If the soil where the ^{137}Cs fell back to Earth contained clay, the cesium was adsorbed to the clay particles and essentially immobilized. Since 1964, there have been no more emissions of the 1964 magnitude, so that the ^{137}Cs peak detected in soil columns in this part of the world date to the 1964 era.

Any soil above the strata where the ^{137}Cs peak occurs in the soil profile is considered to have been deposited since 1964, therefore accurate estimates of sediment accumulation since 1964 can be made.

^{137}Cs decays by emitting a beta particle and then moments later emitting a gamma ray of a specific energy. A gamma ray spectrometer (HP-Ge Ortec with a preamplifier and multi-channel analyzer) was used to detect these gamma rays and determine where the highest activity of ^{137}Cs occurred in the soil core profiles, thus dating that particular strata to the 1964 era. From the depth of the soil overlying the peak strata, sedimentation rates were calculated.

Soil from each core was analyzed for total soil phosphorus following the procedures of Sommers and Nelson (1972) and the USEPA. Soil from each 2-cm increment was digested by wet ashing using nitric and perchloric acids on a Westco digestion block. The level of phosphorus in the resulting solution was measured spectrophotometrically by the Murphy-Riley phospho-molybdate blue complex reaction on the TRAACS 800.

4. Decomposition/Soil Chemistry

Decomposition rates were measured with the cotton strip assay method (Harrison *et al*, 1988) during August of 1995 and April/May of 1996. Specially prepared cotton cloth, which is 99.99% pure cellulose, was inserted vertically into the soil, left for 7-18 days to partially decompose, then retrieved from the soil. After cleaning and drying, the cloth was cut into 5 cm crossections. Hence, soil decomposition rates could be determined by 5 cm increments from the soil surface down to a depth of 25 cm. The tensile strength of the various cotton strip sections was then determined on a Syntech tensiometer. Total decomposition rates and decomposition rates at the various soil depths were then determined mathematically using a linear transformation of the data (Hill et al., 1988).

Soil redox potential was measured using platinum electrodes installed in the soil permanently following the technique developed by Faulkner and Richardson (1989).

Soil cores were taken in the same locations as described in the vegetation section. The percentage of carbon and nitrogen in the soil was measured by dividing the soil cores into 2 centimeter increments to a depth of 30 cm, drying and grinding the soil, passing it through a 2 mm sieve, and then analyzing it on a Perkin Elmer Series II CHNS/O Analyzer mode l 2400.

5. Community/Wildlife Habitat

Herbaceous species diversity was determined using the Shannon-Weaver diversity index (Shannon and Weaver, 1949). Multivariate cluster analyses were performed on the herbaceous and woody vegetation data using PC-ORD version 2.0 (McCune and Mefford, 1995). Also a cluster analysis was performed on the herbaceous vegetation data which incorporated some of the other environmental variables measured in the study.

Cluster analysis classifies species, variables, or sites. It is an explicit way to identify groups and find structure in data taken in the field. It can establish differences or similarities between sites, and can detect relationships between communities and the environment by analyzing the groups formed by the cluster analysis with respect to external variables (van Tongeren, 1995). Euclidean distance, a measure of dissimilarity, was used in the cluster analyses. It uses the abundance of species to group sites with similar species composition. Euclidean distance is strongly sensitive to species richness and dominant species in the sample total (van Tongeren, 1995). Plots that are more similar are clustered together, and plots that are not as similar are separated by cluster boundaries.

The analysis of macroinvertebrate abundance and diversity as functional indicators can be found in King et al., 1997. The study found that both herbivore abundance and overall diversity were highest within 10 meters of the highway embankment, and they decreased at essentially a linear rate with distance from the road. The reference area had the lowest herbivore abundance and diversity, suggesting that road construction created a disturbance which increased macrophyte density and the number of habitat types.

RESULTS

Statistical Design and Testing

The goal of this research was to determine if wetland ecological functions are affected by fill and culvert highway crossings, and if so, to what extent. In order to quantify wetland functions, a primary goal of the project was to identify functional indicators which best described the five main wetland functions previously described.

Since the NCDOT did not have any projects scheduled which would have allowed a "before and after construction" analysis of crossing impacts during our funding period, we were forced to use sites that were already completed. It was hypothesized that if there were significant differences in the levels of any functional indicators among upstream, downstream, and reference areas, then those differences would suggest that the road was affecting wetland ecological function and future "before and after construction" studies could be designed accordingly.

For statistical testing, each upstream and downstream area at each crossing was treated as a separate block, as was the reference area, for a total of five blocks. Plots within blocks were the experimental units. Analysis of variance was performed in Statistical Analysis Systems (SAS) using the Ryan-Einot-Gabriel-Welsch test for multiple comparisons among three or more means. In some instances the upstream areas were combined into a single block, as were the downstream areas, and compared to the reference area.

Reference Area Analysis-Percent Crown Closure For Study Areas

A statistical comparison of the pre-construction percent crown closure for all the study areas is presented in Table 3.1.

Table 3.1. Statistical comparison of percent crown closure for the study areas (circa 1978).

		mean
Kill Swamp upstream	A	81.76
Kill Swamp downstream	AB	79.05
reference area	AB	78.57
Beaverdam Swamp upstream	B	77.02
Beaverdam Swamp downstream	B	77.02

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate.

There is no statistical difference in percent crown closure between the reference area and the other study areas. Since the photographs were taken in the winter when the trees were without leaves, the tendency is to underestimate percent crown closure somewhat. No study area had complete crown closure at that point, but all areas were within 10-15 percent of complete closure and full stocking. The pre-construction photographs indicate that there were a substantially higher number of trees with large crowns (15-20 meters in diameter) at the Beaverdam Swamp impact areas than at the Kill Swamp impact areas or the reference area. Timber cruise data from 1995 verifies that there are larger diameter trees at Beaverdam Swamp now (Table 3.2) based on the mean diameter of trees in each area. These analyses support the view that the areas have not been impacted by activities other than road building, to the extent that it is possible to determine by field reconnaissance and aerial photograph analysis.

Table 3.2. Mean tree diameter by study area (circa 1995).

	mean
Beaverdam Swamp upstream	14.35
Beaverdam Swamp downstream	12.78
Kill Swamp downstream	11.17
reference area	10.46
Kill Swamp upstream	8.03

Where tree removal was detected, at Beaverdam Swamp upstream and Kill Swamp downstream, it was clear that the disturbance was confined to narrow strips which were on the periphery of the study areas, and no data was taken within 40-50 meters of those small areas (NCDOT staff suggested that private landowners may have taken some trees out as highway construction gave them better access to the areas, but again, these areas were small and were buffered from the actual study areas). Therefore the reference area, which was in an apparent undisturbed condition before road construction began, has remained undisturbed according to the best information available, and the other study areas have received only very minor human disturbance other than the road construction itself for many years.

The following results follow the order of functional indicators given in Figure 3.3, which are used as metrics for the corresponding wetland functions.

1. Hydrologic Flux and Storage

All the hydrological data (well data, depth measurements, total areas of inundation etc.) were used to determine if there were detectable differences in stream stage and areal coverage upstream and downstream of the highway. Changes in stream hydroperiod and reach caused by highway construction would be expected to induce or drive changes in the other functions being investigated. Detection and documentation of hydrological alteration due to highway construction, if it exists, was then crucial to the success of the assessment framework.

The mean upstream/downstream difference in water surface elevation for the study, from March 1995 until October 1996 (18 months), for Beaverdam and Kill Swamp was 19.8 and 2.25 cm respectively. These numbers represent the surface elevation (stage) difference between the wells situated just upstream and downstream of the crossings, meaning the water surface elevation just upstream of the crossings was more than that just downstream. The width of the crossings (or length of the culverts) are approximately 70 meters at both sites, so it is logical that the stream surface upstream of the crossing should be somewhat higher than the surface downstream to maintain flow, but these are very low gradient streams and the difference at Beaverdam Swamp is in excess of what the natural gradient of the stream would cause. At Kill Swamp the upstream/downstream stage difference is negligible, indicating that after most precipitation events there is unimpeded flow through the culvert, but at Beaverdam Swamp the constantly elevated stream surface upstream suggests that the road may be affecting stream flow and causing ponding upstream. Figure 3.4 is a hydrograph for a typical one-week period at Beaverdam Swamp. Figure 3.5 is a hydrograph of the same week for the Kill Swamp crossing.

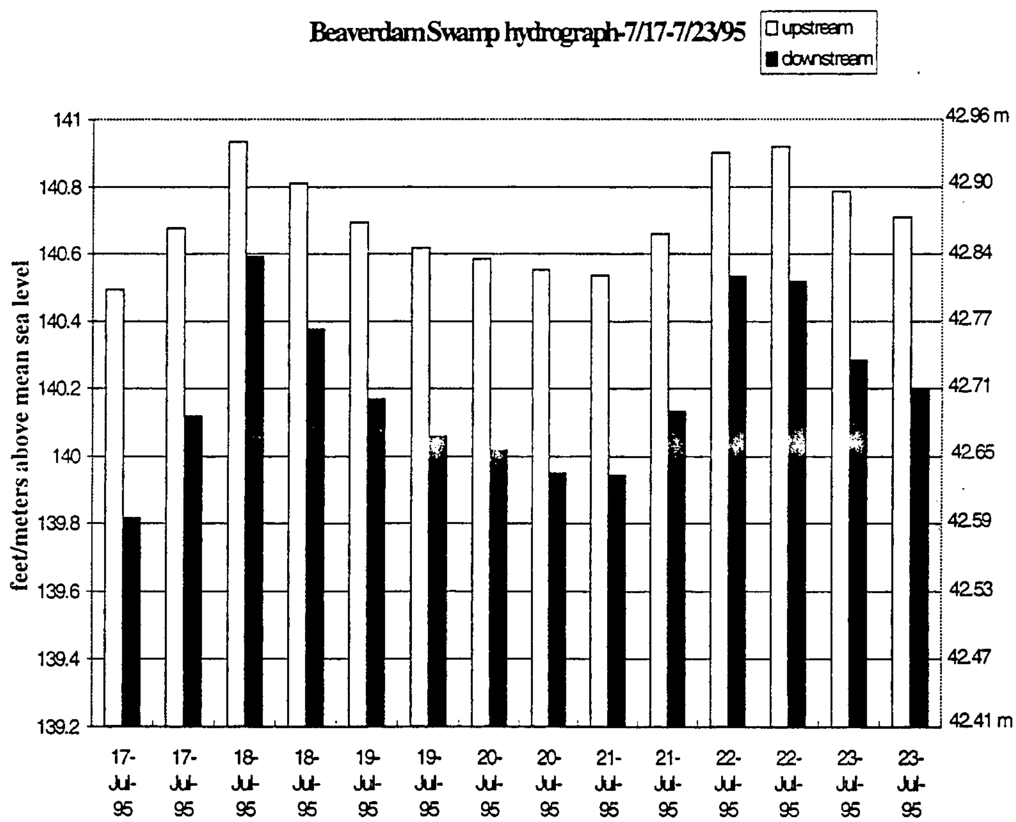


Figure 3.4. Hydrograph of a typical week for Beaverdam Swamp crossing.

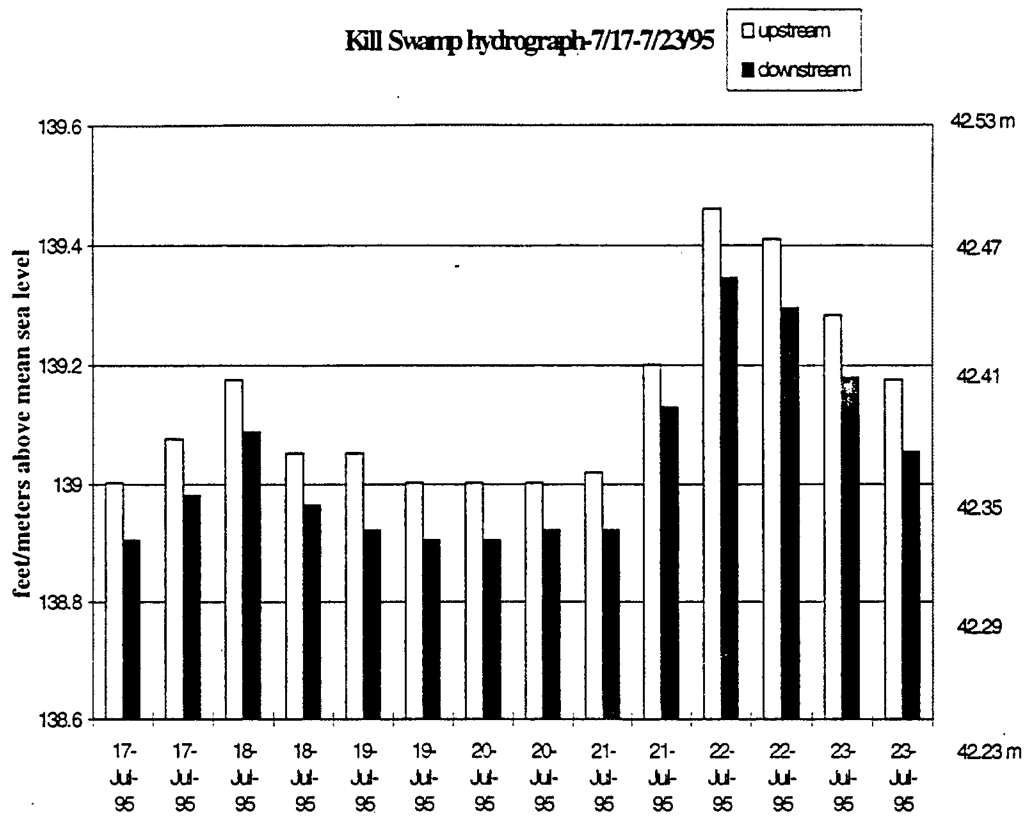


Figure 3.5. Hydrograph of a typical week for Kill Swamp crossing.

The upstream/downstream differences averaged over each month of the study for Beaverdam Swamp are shown in Figure 3.6. The maximum difference was 23.8 cm during May and the minimum difference was 17.1 cm during July. The average monthly upstream/downstream difference at Kill Swamp ranged from 0.5-2.9 cm, which is an insufficient amount of ponding to warrant further interpretation here.

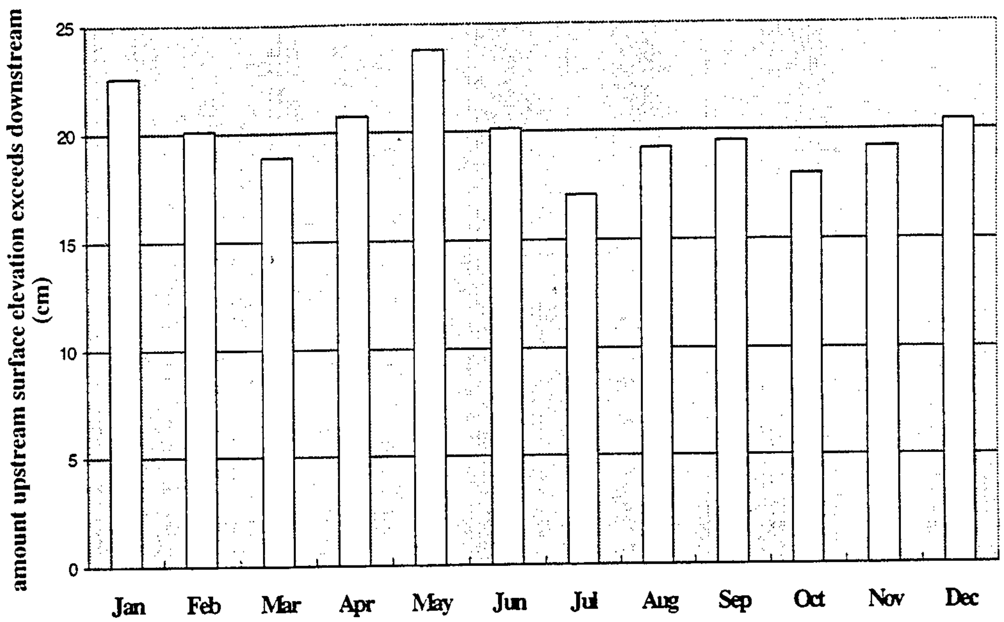


Figure 3.6. Average monthly upstream/downstream stream stage differences at Beaverdam Swamp for the project period.

However, the percentage time that water levels upstream of the Beaverdam Swamp crossing are at or exceeding the study period average is of importance, because prolonged inundation of 19 cm or more is enough ponding to stress less flood tolerant woody and herbaceous species (Whitlow et al., 1979, van der Valk, 1991). Figure 3.7 illustrates the percentage of time that the stream stage in the upstream area of Beaverdam Swamp exceeds the downstream areas, and by how much (again, the wells just upstream and downstream of the crossings are where the data are taken from).

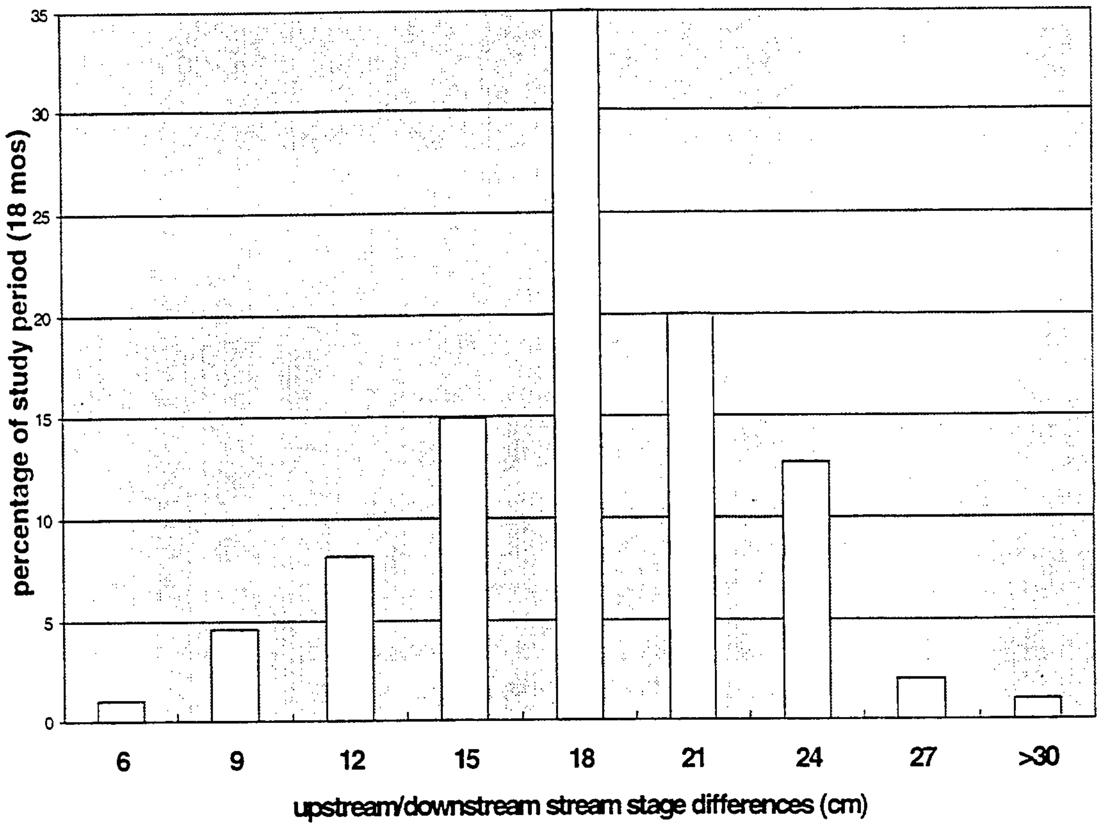


Figure 3.7. Amounts by which the stream stage upstream at Beaverdam Swamp exceeded the downstream stage, and the percentage of study period (18 months) that the differences occurred.

The upstream area stream stage was 18 cm or more over the downstream stage approximately 70% of the study period, and from Figure 3.6 it can be seen that often this amount of difference was present during the growing season (mid March to November, Sampson County Soil Survey, 1981). The amount and length of inundation upstream suggests that the woody and herbaceous vegetation in the upstream area of Beaverdam Swamp which are not flood-tolerant are being stressed and possibly being converted to more flood-tolerant species.

Water depth data taken at plot centers (two sampling dates in 1996, during low stream stage conditions) conforms to the stream stage pattern, with the mean of all the upstream plots for Beaverdam and Kill Swamp being 10.7 and 2 cm deeper respectively than the downstream plots. The difference at Beaverdam Swamp is statistically significant. The reference area plots were significantly more shallow than all the sites beside the road crossings. The difference in mean depth between upstream Beaverdam Swamp and the reference area averaged 22.6 cm, and the difference for Kill Swamp upstream and the reference area averaged 15.2 cm.

Measurements for total area of inundation were made twice, during August and November of 1995. On both occasions, there had been no measurable precipitation for at least one week, so that if ponding was detected during these times, it would be logical to assume that ponding occurs most or all of the time. The upstream area of Beaverdam Swamp was larger within 60 meters of the crossing by an average area of 0.12 hectares (ha), with a range of 0.08-0.16 ha. The downstream area of Kill Swamp within 60 meters of the crossing was larger than the upstream by and average of 0.003 hectares. The greater area of inundation at Beaverdam Swamp compared to Kill Swamp coincides with the upstream/downstream stream surface elevation and stream depth data, suggesting that the road crossing may have altered the hydrology of the Beaverdam Swamp site significantly and created ponding there.

2. Plant Productivity and Mortality

The forest covering the study areas is typical of the forest covering wetter, frequently or constantly inundated ecosystems in the southeastern US. The primary components are bald cypress (*Taxodium distichum* (L.)) and swamp tupelo (*Nyssa sylvatica* var. *biflora* Marshall). For a complete listing of tree species found in the study areas, see Appendix II. The woody vegetation survey indicates that there are substantial differences in basal area between upstream and downstream areas of the crossings, and between these areas and the reference area (Figure 3.8).

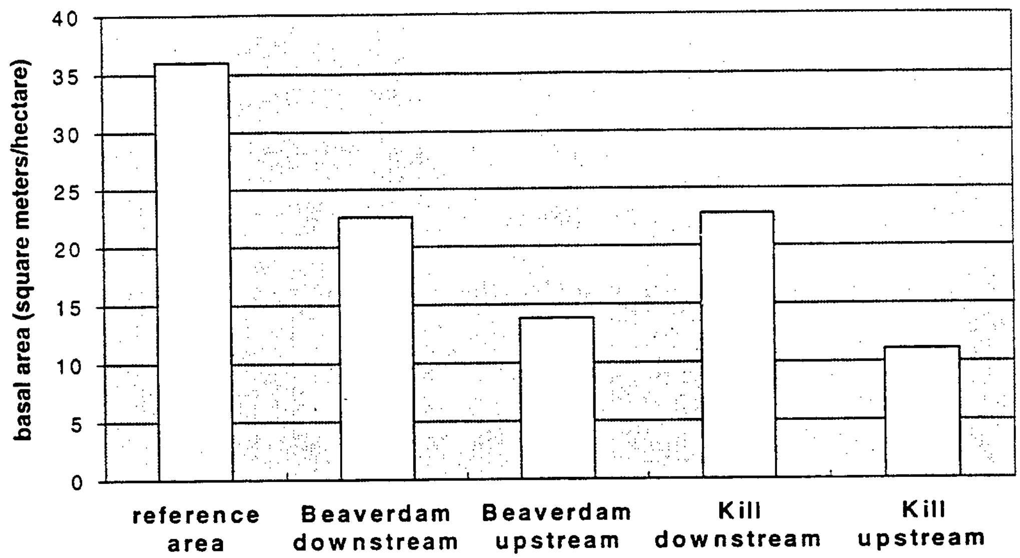


Figure 3.8. Basal area of woody vegetation by study area (down=downstream, up=upstream).

The decrease in basal area as one goes from the downstream to the upstream areas at both crossings is striking. At Beaverdam Swamp there is 39% less basal area upstream of the crossing (22.5 vs. 13.8 m^2/ha). At Kill Swamp, there is 52% less basal area upstream than downstream (22.8 vs. 11.1 m^2/ha). And the downstream areas at each crossing, where more basal area exists, both have substantially less basal area than the reference area, which is fully occupied by trees. Beaverdam Swamp downstream has 38% less basal area than the reference area (22.5 vs. 36.0 m^2/ha) and Kill Swamp downstream has approximately 37% less basal area than the reference area (22.8 vs. 36.0 m^2/ha).

From Figure 3.9, where species composition by basal area and study area is graphed, it can be seen that cypress is the dominant species on Beaverdam Swamp, cypress, tupelo, and the other species are much more balanced at the reference area, and tupelo is the dominant species at Kill Swamp, although Kill Swamp upstream, with its low basal area, is more balanced species-wise (a list of tree species present at the research sites is in Appendix II). All study areas have cypress, tupelo, and maple present. The "others" category includes sweetgum, yellow-poplar, loblolly pine, and ironwood for the reference area and green ash, willow oak and black willow for Kill Swamp upstream. These are less flood-tolerant species, except for black willow, which contributes the most basal area to the "others" category at Kill Swamp upstream. For each of the other species, less than 5 individuals were found at either site. It should be noted that "others" comprise a substantial component in the reference area, where the water depth is not as great as at the other study areas, suggesting the ponding induced by the highway may have eliminated these species near the crossing.

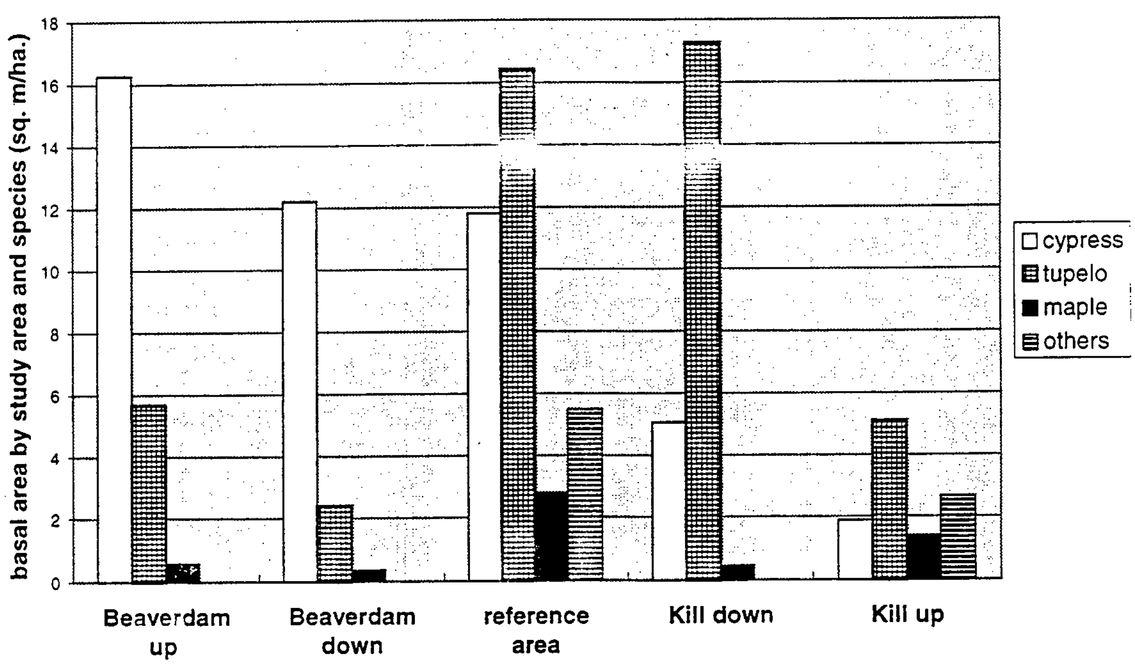


Figure 3.9. Basal area by study area and species.

Total above-ground biomass by species and study area is presented in Table 3.3. The reference area ranked first in basal area (Figure 3.8), but was second in total biomass, even though it had more trees, because the trees there were not as mature and had a lower average dbh (Table 3.2).

Table 3.3. Total above-ground biomass (oven-dried) by study area and species (kg/ha).

location	species	leaves	branches	stem	total
Beaverdam downstream	cypress	570	2543	99750	102863
	tupelo	806	4635	21875	27316
	maple	88	511	2466	3065
				grand total	**133244**
Beaverdam upstream	cypress	466	2194	88242	90902
	tupelo	396	2284	10939	13619
	maple	45	262	1300	1607
				grand total	**106128**
reference area	cypress	234	1097	44156	45487
	tupelo	1398	8056	38280	47734
	maple	237	1913	9202	11352
	sweet gum	331	1364	6383	8078
	others	85	486	2228	2799
				grand total	**115450**
Kill downstream	cypress	116	566	23010	23692
	tupelo	1950	11334	56720	70004
	maple	45	262	1300	1607
				grand total	**95303**
Kill upstream	cypress	102	389	14227	14718
	tupelo	712	4122	20066	24900
	maple	425	2432	11093	13950
	others	251	1462	7422	9135
				grand total	**62703**

Older, larger trees are proportionally much heavier than small trees because their stems are much more massive. Therefore, even though a forest stand is fully stocked with trees, the biomass of a stand that is not fully stocked can be greater if the trees there are older, because the stems of older trees gain weight exponentially with increasing diameter (see the equations in Schlesinger, 1976 and Dabel and Day, 1977).

The Beaverdam downstream study area contained approximately 15% more biomass than the reference area, and approximately 25% more than the Beaverdam upstream area. The reference area contained approximately 8% more biomass than the Beaverdam upstream area and approximately 17% and 45% more than the Kill Swamp downstream and upstream areas respectively.

Species composition by study area and biomass can be seen in Figure 3.10. Species composition looks similar using either basal area or biomass as an index. The Beaverdam study areas have a much higher proportion of cypress than the other study areas, while tupelo dominates at Kill Swamp. Cypress and tupelo biomass are balanced in the reference area.

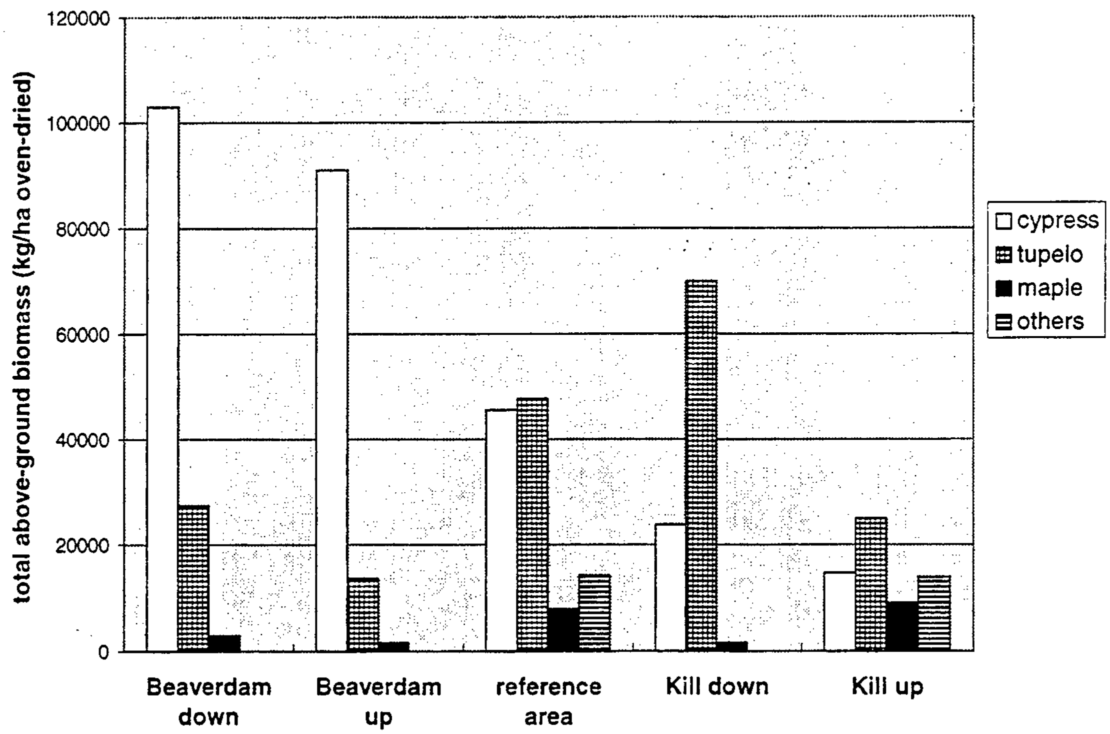


Figure 3.10. Total above-ground biomass by study area for the major tree species (in kg/ha oven-dried weight).

The relative absence of bald cypress at Kill Swamp suggests that economic incentives for selective logging could have triggered its removal where it was possible to get the trees out of the swamp, although logging most certainly must have taken place decades ago, prior to the road construction. The importance of black willow there may also indicate some sort of disturbance took place many years ago, because it often is a pioneer species. A past disturbance would make vegetation comparisons between study areas more difficult to interpret, although the disturbance, if indeed one occured, happened 15-20 years before road construction began.

Densiometer readings from the field were converted to percent crown closure and percent crown closure for each area was computed. Percent crown closures for each area were compared statistically and the results are in Figure 3.11.

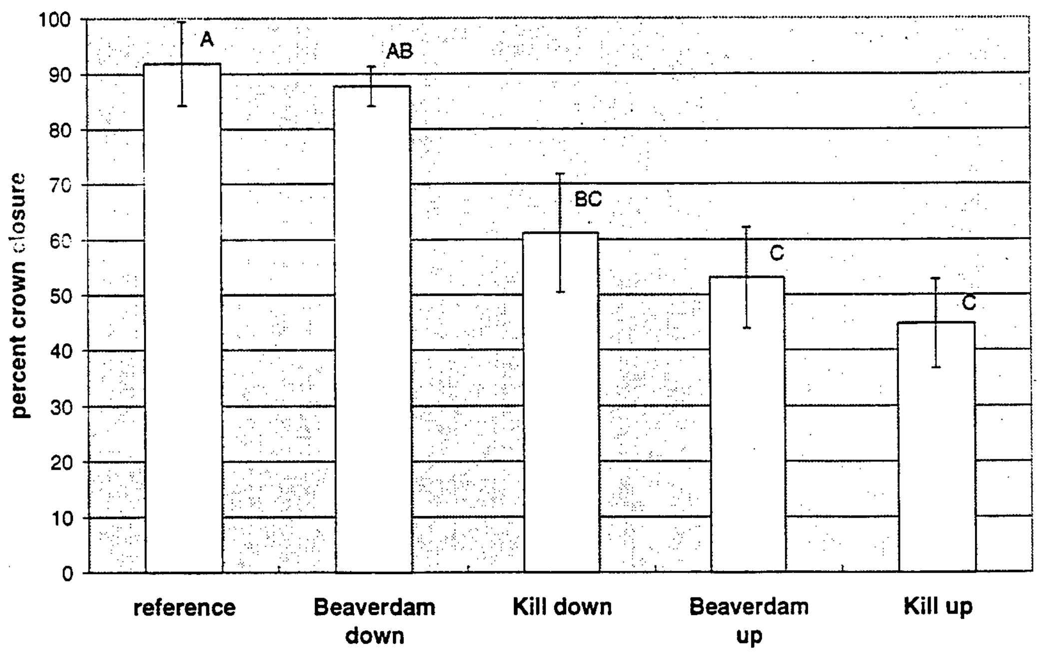


Figure 3.11. Statistical comparison of percent crown closure by study area (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

The reference area had significantly more crown closure than all areas except Beaverdam Swamp downstream. The upstream areas, which are not statistically different, had statistically less percent crown closure than all areas except Kill Swamp downstream. When the areas are grouped by upstream, downstream or reference location, the reference area is not statistically different from the downstream areas, but those areas had a statistically higher percent crown closure than the upstream areas (Table 3.4).

Percent crown closure in the reference and Beaverdam downstream area has increased from the estimates made from the 1978 aerial photographs by approximately 10 percent, while it has decreased in all other areas approximately 15 to 35 percent.

Table 3.4. Statistical comparison of percent crown closure between upstream/downstream/reference areas.

		mean
reference area	A	91.83
downstream areas	A	74.44
upstream areas	B	48.88

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different.

Coarse woody debris total volume results are presented in Table 3.5. Statistical analysis (SAS, 1989) determined that the volume of coarse woody debris is significantly higher at Beaverdam Swamp upstream than the other study areas. Figure 3.12 is an illustration of the average coarse woody debris volume per plot by study area.

Table 3.5. Coarse woody debris, standing and downed combined (meter^3/ha).

Location	total vol.	>30cm
Beaverdam Swamp upstream	36.13	23.52
Kill Swamp downstream	13.45	6.73
Beaverdam Swamp downstream	14.97	6.37
Kill Swamp upstream	12.22	4.02
reference area	5.50	1.58

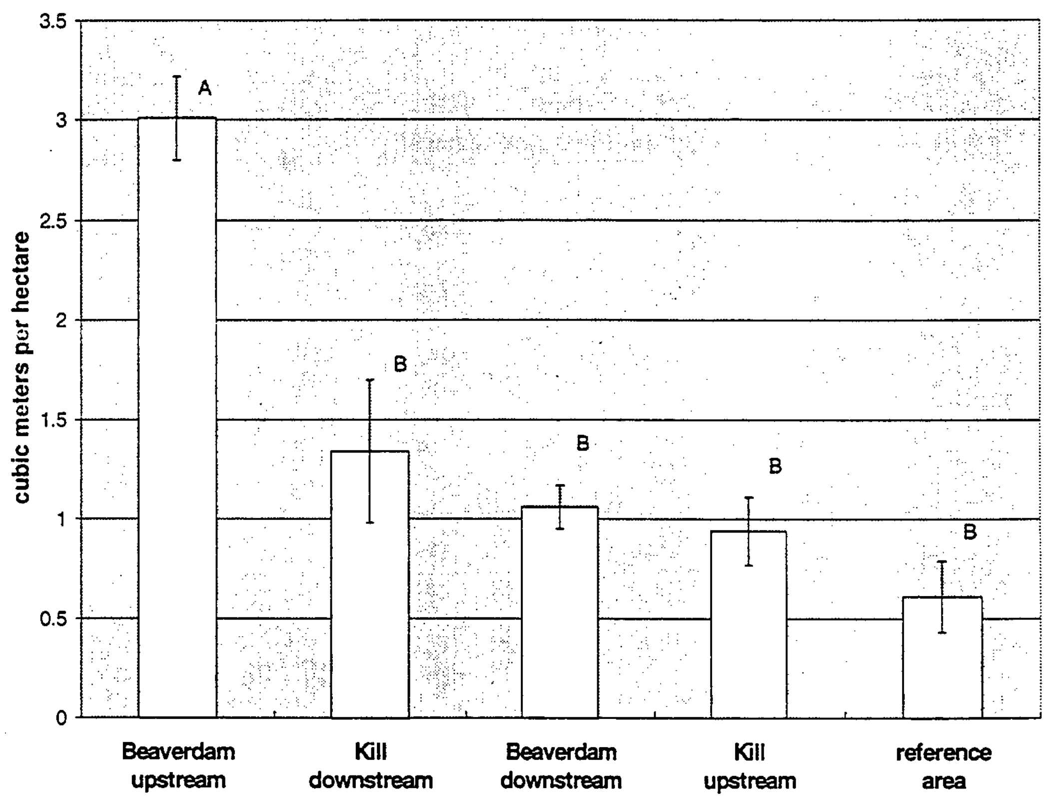


Figure 3.12. Mean volume of coarse woody debris per plot for each study area (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different). This test controls for Type I experimentwise error rate.

When the upstream and downstream areas are grouped, using the same statistical test, the upstream areas contained significantly more coarse woody debris than the downstream areas, and the reference area contained significantly less coarse woody debris than either upstream of downstream areas (see Table 3.6).

Table 3.6. Coarse woody debris mean volume totals for plots grouped by upstream, downstream and reference areas (units are m^3/plot).

upstream areas	A	1.93
downstream areas	B	1.18
reference area	C	0.61

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate.

Herbaceous biomass data was compared statistically between the study areas. The results are in Figure 3.13.

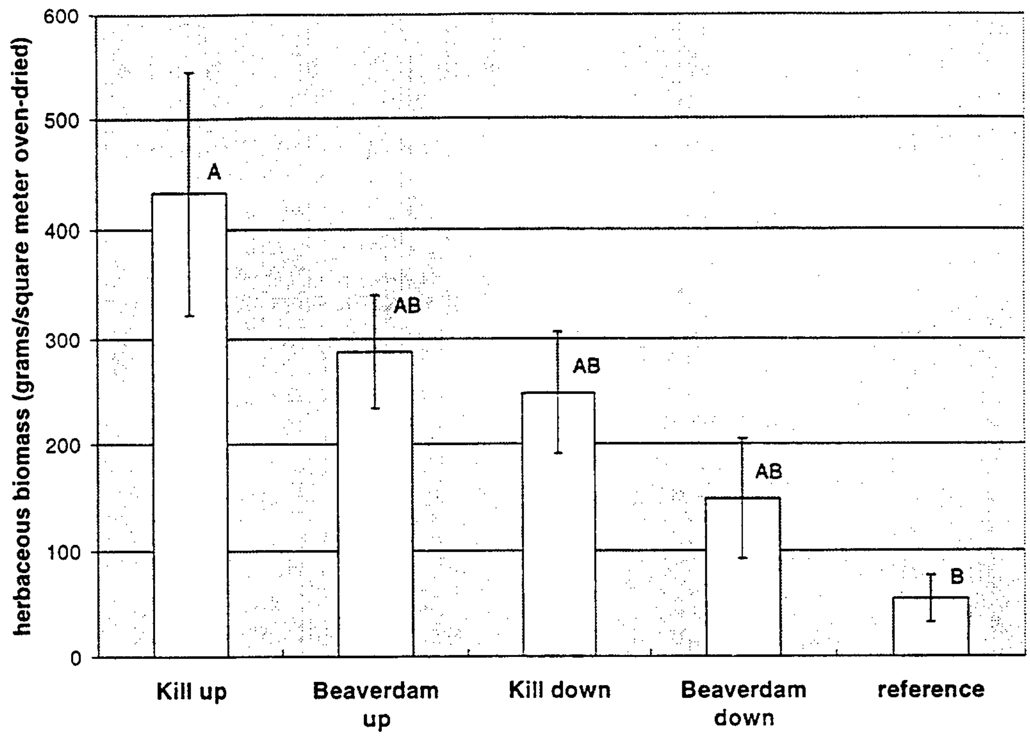


Figure 3.13. Statistical comparison of herbaceous biomass by study area (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

The Kill Swamp upstream area has significantly more herbaceous biomass than the reference area. It is not significantly different from any of the other areas. A statistical comparison of the areas when grouped by upstream, downstream, and reference areas is shown in Table 3.7. The downstream areas are not significantly different from the other areas. The upstream areas have significantly more herbaceous biomass (~ 6x) than the reference area.

There were 18 herbaceous species identified in the entire study area (a list is in Appendix III). *Murdannia kiesak*, which is the most common herbaceous species on the sites, is an invasive exotic. A list of the most important herbaceous species by study area is in Table 3.8. The other most important species are *Leersia oryzoides*, or cutgrass, and *Boehmeria cylindrica*, or false nettle.

Table 3.7. Statistical comparison of herbaceous biomass between upstream/downstream/reference areas.

		mean
upstream areas	A	90.02
downstream areas	AB	49.57
reference area	B	13.60

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate.

Table 3.8. List of the most important herbaceous species by study area.

spp.	mk	b	lo	ph	bc	sp	ec	lp	sc	ps
Bd	x	x		x	x					
Bu	x		x	x		x	x	x		
ref	x								x	
Kd	x		x		x	x				
Ku	x		x	x	x					x

Bd=Beaverdam downstream, Bu=Beaverdam upstream, ref=reference, Kd=Kill downstream, Ku=Kill upstream

mk=*Murdannia kiesak*, b=*Bidens* spp., lo=*Leersia oryzoides*, ph=*Polygonum hydropiperoides*, bc=*Boehmeria cylindrica*, sp=*Spirodela polyrrhiza*, ec=*Echinocloa crusgalli*, lp=*Ludwigia palustris*, sc=*Saururus cernuus*, ps=*Penthorum sedoides*

3. Biogeochemical Cycling and Storage

Stream water quality

The results of the first stream water nutrient concentration data showed no significant differences between upstream and downstream areas of the study for either the June or July 1995 sampling dates. For each nutrient concentration analyzed, NH_4-N, NO_2 and NO_3-N, total N, PO_4-P and total P, there was no significant difference detected between the reference area and Beaverdam Swamp up and downstream areas. Also, there was no significant difference detected between the Kill Swamp upstream and downstream areas. However, there were significantly higher concentrations of each nutrient analyzed for (2-10X) in Kill Swamp than at Beaverdam Swamp or the reference area. Table 3.9 contains a list of the ranges for the respective stream water nutrients analyzed for on the two sampling dates in 1995 by study area.

Table 3.9. Concentration ranges for stream water nutrients in the study areas (μg/l).

	Beaverdam Swamp	Kill Swamp	reference area
NH_4	68-123	568-1387	78-90
NO_2-NO_3	43-144	1222-2220	42-182
Total N	909-1027	2892-4308	1002-1493
PO_4-P	61-177	188-292	79-151
Total P	159-227	405-858	151-210

Although there appeared to be upstream/downstream decreasing gradients for some nutrients, for others the concentration increased downstream, indicating sources of nutrients along the study area perimeter. Both sampling days were during or just after sizable rain events, so that runoff from surrounding areas was coming into the study areas as we sampled. It was decided that since the study areas were not “closed systems” regarding water-borne nutrients, that determining the nutrient sink capability of the streams within the study areas, within the scope of the project, would be impossible, and no further sampling for water-borne nutrient concentrations was done. Streamwater nutrients were not useful in this study as functional indicators because of the many sources adjacent to the study areas, but they may be useful in areas where agriculture is not the major component of the local economy.

The streamwater temperature and dissolved oxygen parameters were gathered later in the study, on four occasions in April, May, June, and July of 1996. These parameters were chosen as indicators because of their importance to aquatic life and decomposition. Readings were taken simultaneously for both indicators in the morning at each plot center, in order to measure dissolved oxygen levels at the lower range of their daily cycle. Two people were used to sample the upstream/downstream areas simultaneously, one up and one down. This was done to minimize differences in upstream/downstream data due to diel fluctuations in the parameters as the sun rose. The results of the statistical analyses of the stream temperature data were quite variable. A summary of the data is in Table 3.10. One statistical analysis model (model 1) contained all the study areas, and the other (model 2) analyzed the data after study areas were combined on an upstream/downstream basis.

Generally, the upstream areas were warmer than the downstream areas. Also, as spring became summer, the reference area temperature levels fell to the lower end of the temperature range for the study areas.

Table 3.10. Statistical summary of stream water temperature data (°C) (B=Beaverdam Swamp, K=Kill Swamp, u=upstream, d=downstream, ref=reference area.

model 1, all research areas included

4/18/96		mean	5/26/96		mean	6/26/96		mean
Ku	A	15.79	Bu	A	29.78	Ku	A	25.36
ref	B	15.19	ref	B	25.98	Kd	A	25.34
Bu	AB	14.19	Ku	B	25.08	Bd	A	24.46
Bd	AB	13.82	Kd	B	24.94	Bu	B	22.58
Kd	AB	13.51	Bd	B	24.55	ref	C	19.72

7/2/96		mean	7/23/96		mean
Bd	A	29.29	Bu	A	25.69
Kd	B	27.05	Ku	AB	25.50
Ku	BC	26.15	Bd	ABC	24.87
Bu	C	25.71	ref	BC	24.64
ref	C	25.00	Kd	C	24.45

model 2, upstream and downstream areas from both crossings combined

4/18/96		mean	5/26/96		mean	6/26/96		mean
ref	A	15.19	u	A	27.64	d	A	24.86
u	A	14.99	ref	AB	25.98	u	B	24.03
d	B	13.70	d	B	24.71	ref	C	19.72

7/2/96		mean	7/23/96		mean
d	A	28.39	u	A	25.59
u	B	25.94	d	B	24.70
ref	C	25.00	ref	B	24.64

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different.

The statistical results of the dissolved oxygen samplings, done in June and July of 1996 are in Table 3.11.

Table 3.11. Statistical analysis of the dissolved oxygen concentrations for the study areas.

		dissolved O_2 (mg/l)
6/26/96		
reference area	A	2.60
Kill Swamp downstream	AB	2.18
Beaverdam Swamp upstream	AB	1.98
Beaverdam Swamp downstream	BC	1.82
Kill Swamp upstream	C	1.22
7/2/96		
Kill Swamp upstream	A	0.73
reference area	A	0.54
Beaverdam Swamp upstream	A	0.43
Kill Swamp downstream	A	0.37
Beaverdam Swamp downstream	A	0.31
7/23/96		
reference area	A	0.88
Beaverdam Swamp upstream	AB	0.72
Kill Swamp upstream	B	0.52
Kill Swamp downstream	C	0.08
Beaverdam Swamp downstream	C	0.02

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate.

Stream water dissolved oxygen measurements were taken with the temperature measurements, but due to technical difficulties, data from June and July were the only measurements which could be used. In June the reference area had the highest dissolved oxygen levels, and the downstream areas showed higher dissolved oxygen levels than the upstream areas. This was significant at Kill Swamp, even though the upstream and downstream temperatures were almost identical there. There was no significant difference in dissolved oxygen at Beaverdam Swamp. When the statistical model combined the upstream and downstream areas from both crossings there was no significant difference between the upstream and downstream areas but the reference area was significantly different from the upstream/downstream areas.

On July 2, the dissolved oxygen levels were much lower than in June, and there were no statistical differences detected in dissolved oxygen concentrations between the study areas, or between the combined upstream/downstream areas. On July 23, the dissolved oxygen concentrations were similar to those of July 2 except that the downstream areas at both crossings had very low concentrations, which made the differences between them and the upstream areas significant. The reference area again had the highest dissolved oxygen levels. There were significant differences between the upstream/downstream/reference areas.

Sedimentation

Sedimentation rates for the study areas were not found to be statistically different, due to the high variability between sampling points. The results of the statistical analysis are in Table 3.12. At Kill Swamp, the sedimentation rate was higher downstream than upstream, suggesting that road construction may have caused some sediment runoff downstream. The reference area rate was the lowest, suggesting that disturbance may have contributed to the higher rates at the sites beside the highway crossings.

Table 3.12. Statistical comparison of the sedimentation rates for the different study areas (mm/year).

		mean
Kill Swamp downstream	A	4.71
Beaverdam Swamp upstream	A	4.14
Beaverdam Swamp downstream	A	3.55
Kill Swamp upstream	A	3.49
reference area	A	3.06

The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate.

Total soil phosphorus concentrations were similar between the sites, with one study area being statistically higher in mean concentration than the others. The results are plotted in Figure 3.14. Kill Swamp downstream had significantly higher total soil phosphorus than Kill Swamp upstream. Since the sedimentation data indicates no differences between the amount of sediment deposition between the sites, it suggests that the difference in total soil phosphorus concentration at Kill Swamp is not a result of phosphorus deposition associated with eroded mineral soil, but rather the amount of litter/organic matter present in the area.

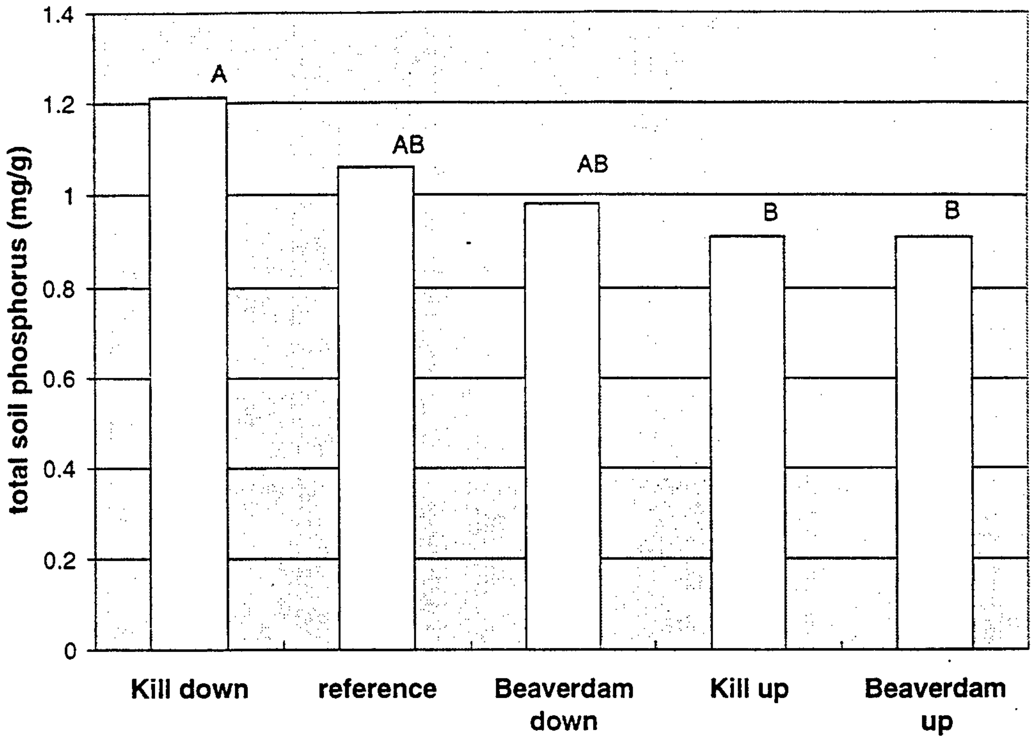


Figure 3.14. Statistical comparison of total soil phosphorus content (mg/g) by study area. Depth = 0-25 cm. (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

Decomposition/Soil Chemistry

Soil decomposition rates were measured twice during the study, in August of 1995 and in April/May of 1996. Statistically analyzed results from 1995 are in Figure 3.15. There was a statistically higher decomposition rate at Beaverdam Swamp downstream than at Beaverdam upstream, and these two areas had statistically higher decomposition rates than the Kill Swamp or reference areas in August of 1995.

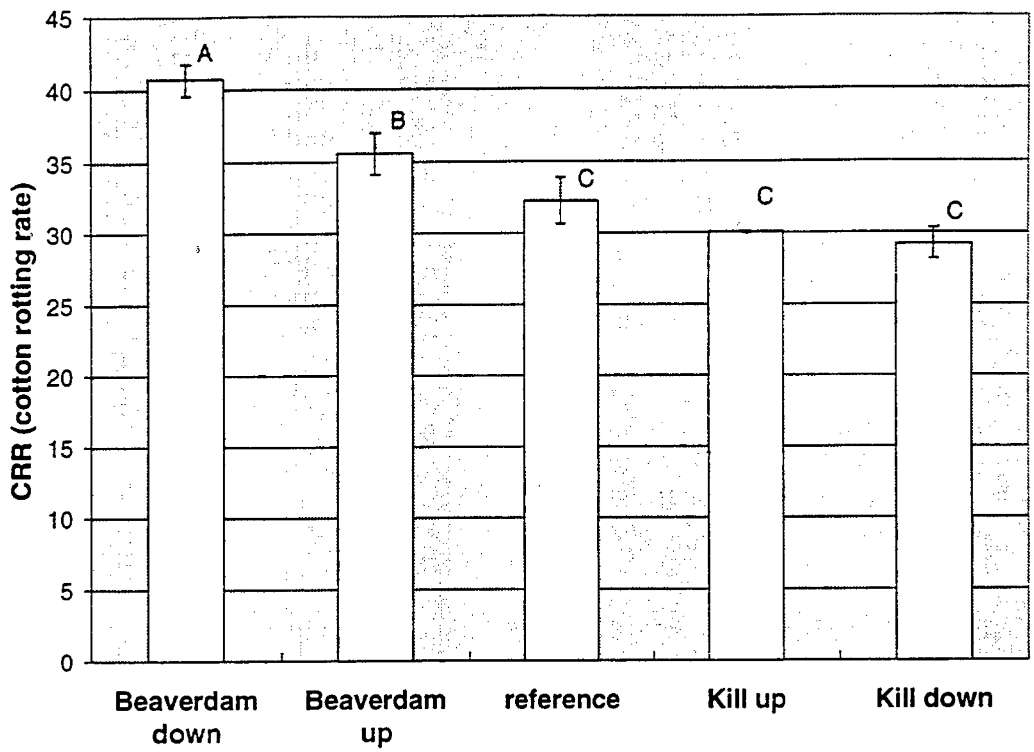


Figure 3.15. Statistical comparison of 1995 soil decomposition rates by study area. CRR is the number of cotton strips that will decompose to approx. 50% tensile strength in one year (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

The statistical results from the April/May 1996 cloth installation are presented in Figure 3.16. In 1996 there was again a faster decomposition rate at Beaverdam Swamp downstream than at Beaverdam Swamp upstream, but there was no significant difference between Beaverdam Swamp downstream and the Kill Swamp and reference areas. The standard errors in 1996 were approximately 2-3 times greater than in 1995, which makes sorting out any differences more difficult statistically. Decomposition rates were slower in 1996, probably due to lower water temperatures. But in both years the decomposition rate was slower at Beaverdam Swamp upstream than at Beaverdam Swamp downstream.

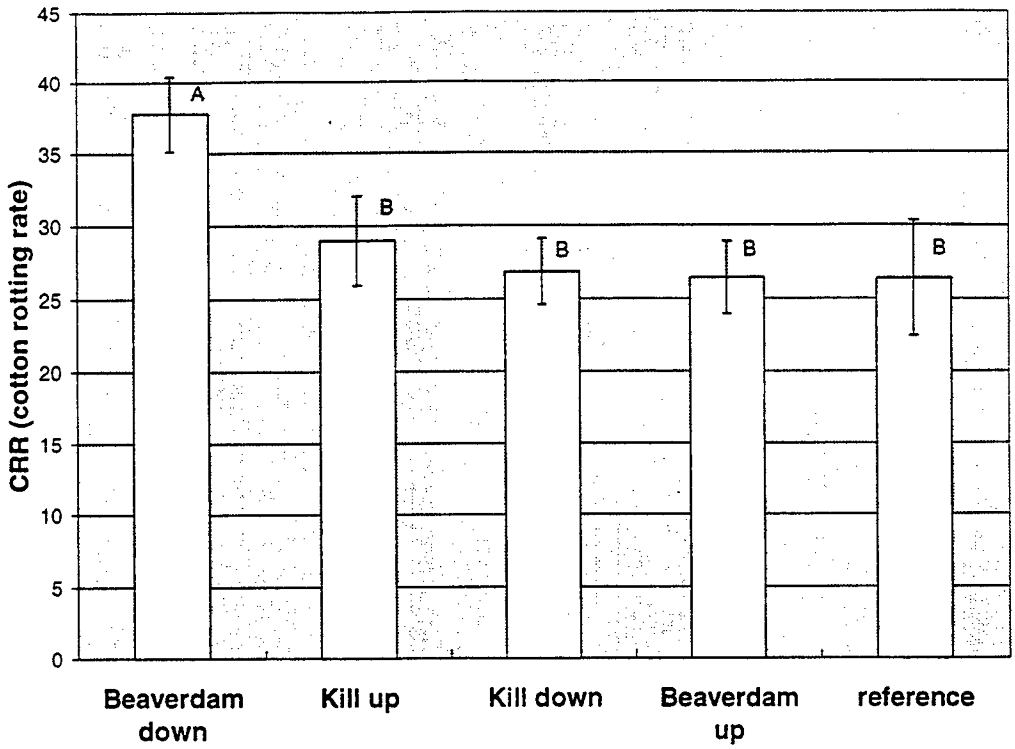


Figure 3.16. Statistical comparison of 1996 soil decomposition rates by study area. CRR is the number of cotton strips that will decompose to approx. 50% tensile strength in one year (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

A pilot study was done to check the methods used for soil oxidation-reduction (redox) potential. Five platinum probes were placed in a one meter square inundated plot upstream at Beaverdam Swamp. One probe was placed at each corner to a depth of 15 cm. After repeated attempts it was apparent that the readings were so variable that the utility of the data being collected was suspect. When two probes were placed within 2 cm of each other at the same 15 cm depth and readings taken, the results were highly variable between the probes. While acknowledging that microsite differences were to be expected, the high degree of variability lead to the abandonment of soil redox as a useful functional indicator in these wetlands.

Soil chemistry analysis relating to decomposition consisted of soil samples being taken and analyzed for carbon and nitrogen content. The statistical comparison of soil carbon content by site is in Figure 3.17. The reference area has a significantly higher soil carbon content than Kill Swamp (~2X) and Kill Swamp has a significantly higher soil carbon content than Beaverdam Swamp (~6X). There are no significant differences between upstream and downstream areas. The statistical comparison of soil nitrogen content by site is in Figure 3.18. The reference area has the highest soil nitrogen content, being significantly higher than all study areas except Kill Swamp upstream. The nitrogen soil levels by area have a similar relationship to each other as the carbon soil levels, and would be identical if it were not for the higher degree of variation in the soil nitrogen data.

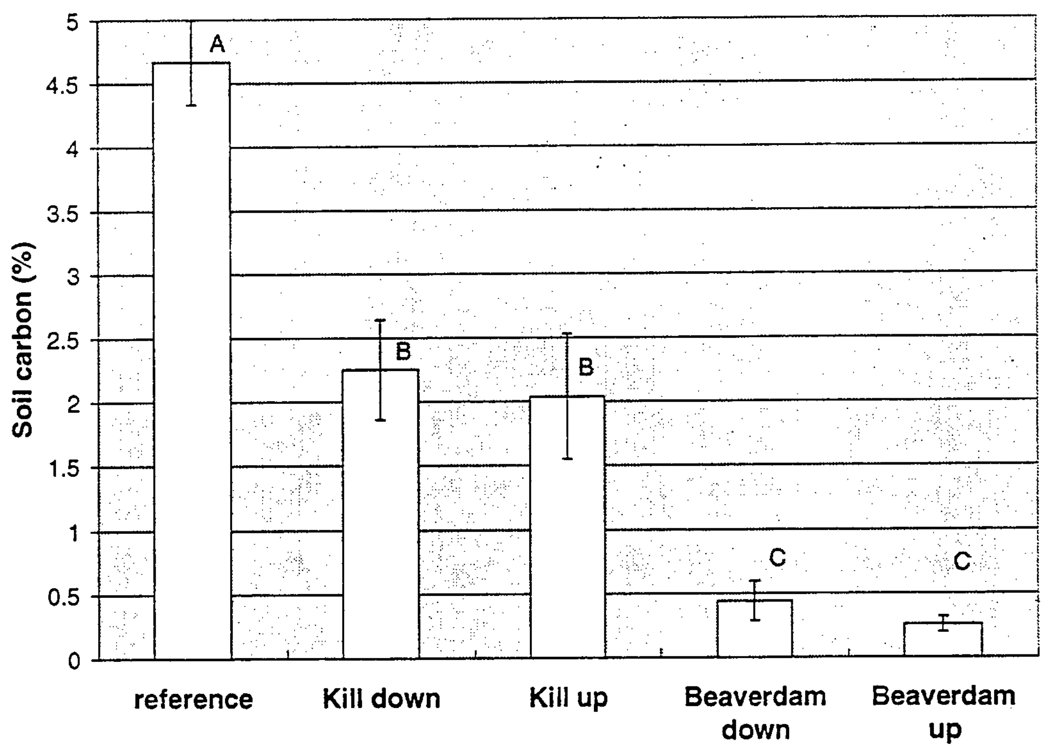


Figure 3.17. Statistical comparison of soil carbon content (%) by study area (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha =0.05.

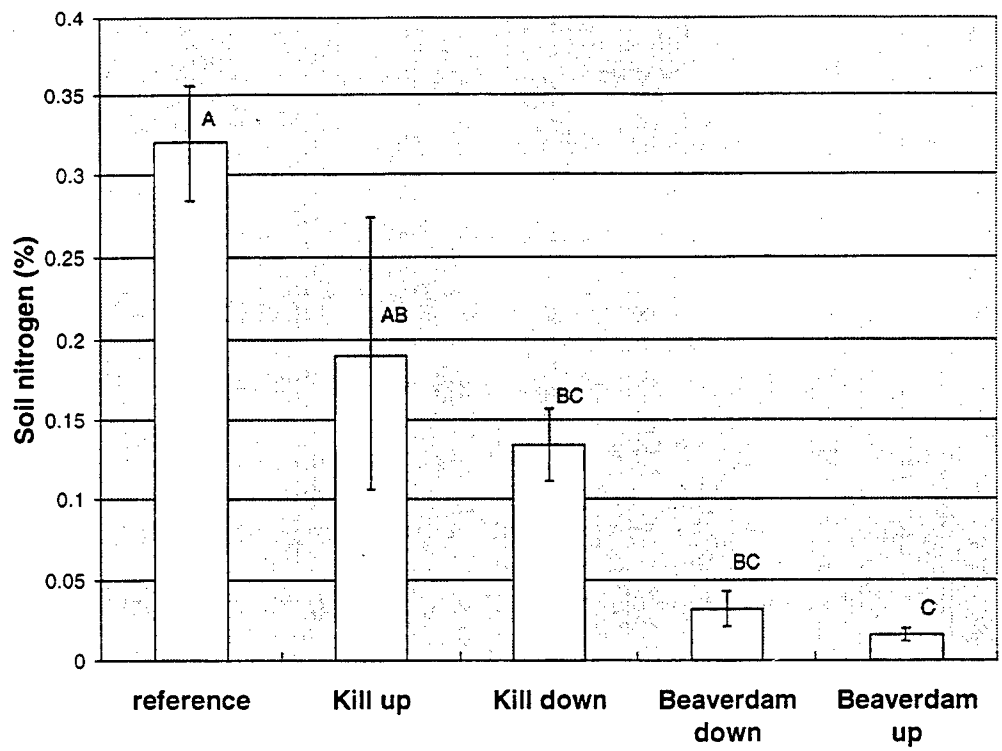


Figure 3.18. Statistical comparison of soil nitrogen content (%) by study area (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

Community/Wildlife Habitat

Herbaceous plant diversity was computed by the Shannon-Weaver index (Shannon and Weaver, 1949). The formula is $H' = -\Sigma\, (Xi/Xo) \ln (Xi/Xo)$, where H' is the Shannon-Weaver diversity index, Xo is the total number of species in the sample, and Xi is the number of individuals in species i. The Shannon-Weaver index has a range from 0 to over 7, where 0 indicates a community of one species, and 7 indicates a community where many species occur and the variability between samples is very high. The overall herbaceous diversity on all sites was low. We tested for statistical differences between the upstream/downstream areas and the reference area. As seen in

Figure 3.19, the upstream areas at Beaverdam and Kill Swamp have similar diversity, but they are statistically more diverse than the downstream areas and the reference area. Diversity in the downstream areas is identical using this index. The reference area had the lowest diversity of all the study areas.

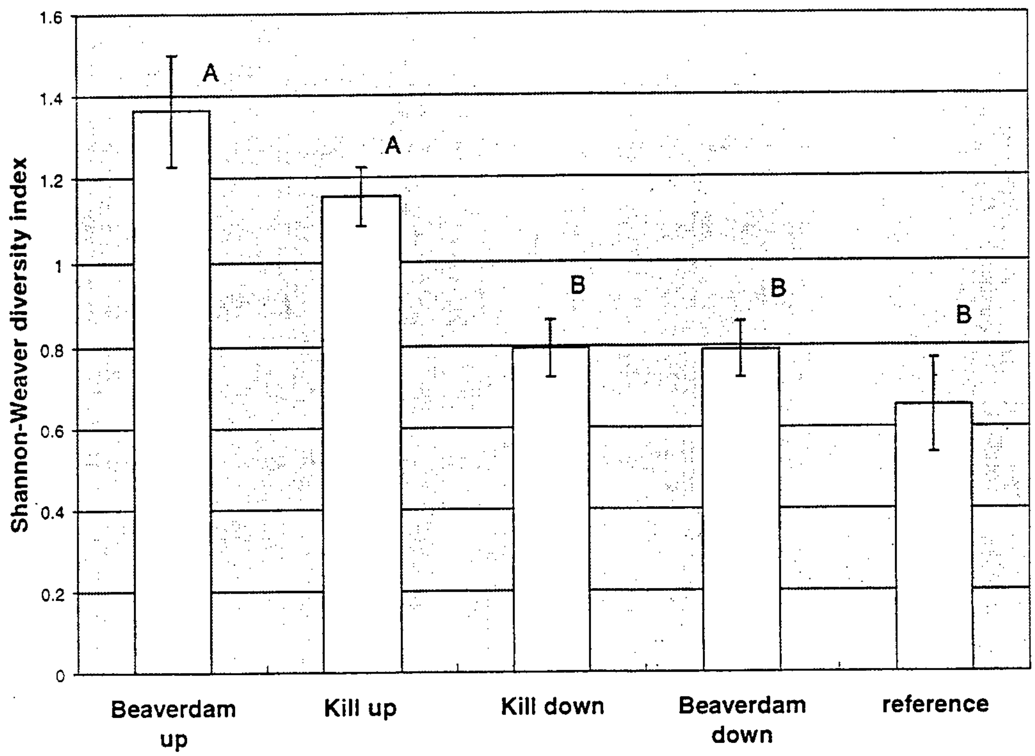


Figure 3.19. Statistical comparison of herbaceous plant diversity by study area (The Ryan-Einot-Gabriel-Welsch multiple F test, alpha = 0.05. Means with the same letter are not significantly different. This test controls for Type I experimentwise error rate).

A multivariate cluster analysis of the herbaceous species data was performed using PC-ORD version 2.0 (McCune and Mefford, 1995. Figure 3.20 is a cluster analysis of the herbaceous species data constructed using Euclidean distance and farthest neighbor linkage. Cluster analysis is a tool used to point out similarities between plots graphically, in this case species composition. Plots which are most similar in species compostion (or with respect to chosen environmental variables) are grouped together first, and then these groups are combined based on similarity values calculated by Euclidean distance. The greatest resolution of groups is seen at the left of the graph. When plots from the same study area or the same upstream/downstream area are found grouped, similarities in the environment there are suggested.

Divisions between the primary groups in Figure 3.20 occur between kd3 and bd10, bu3 and bd9, kd8 and ku4, and ku11 and r25. The primary groups in Figure 3.20 are small, but most consist of plots from the same upstream/downstream areas. The reference area plots, scattered throughout the cluster analysis, are not very similar to each other from this perspective. This cluster analysis, which contains only small (5-6 plots) groups from the different study areas, does not convey an overriding, clear pattern from which to conclude there is an upstream/downstream difference in the herbaceous species composition. However, the groups that do show up suggest that the process of herbaceous species compositional shifts may have begun and are still in progressing.

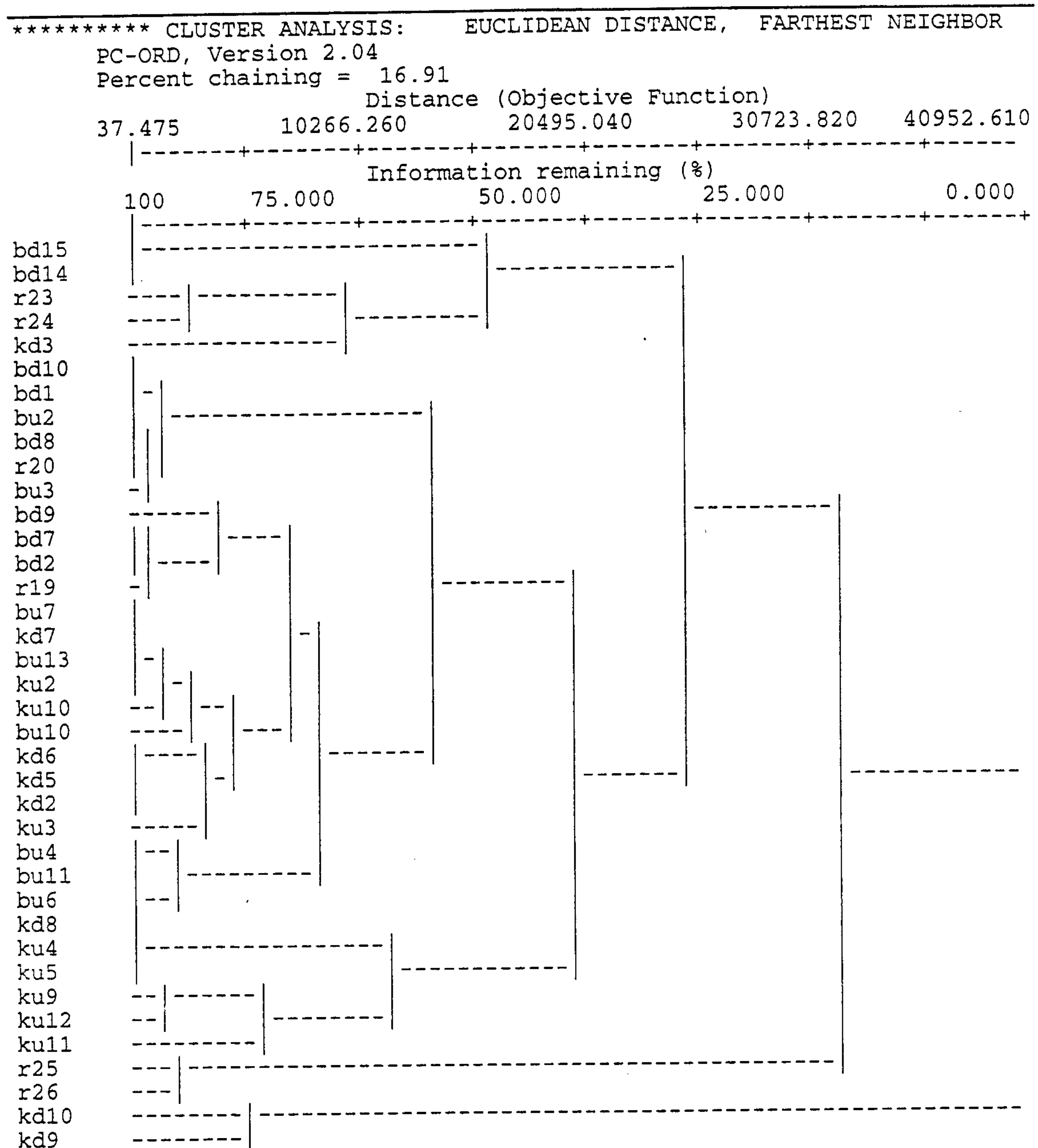


Figure 3.20. Cluster analysis of herbaceous vegetation by plots (bd=Beaverdam downstream, bu=Beaverdam upstream, r=reference area, kd=Kill downstream, ku=Kill upstream).

Figure 3.21 is a cluster analysis of the plots by tree species composition and basal area. Similarities between upstream/downstream areas are clearer in this cluster analysis, which divides the plots into three main groups. The upper group, from bd15 to bu11 contains 6 downstream plots, all but one from Beaverdam Swamp. The second and largest group, from bd8 to ku9, contains mostly upstream plots, with the Beaverdam Swamp upstream plots near their downstream counterparts at the top of the group and the Kill Swamp plots at the bottom. Four reference area plots are in this group, and three of them are in the upper part of the group, where more Beaverdam Swamp and downstream plots are located. The lower group, from r24 down is comprised mainly of Kill Swamp downstream plots. Two reference area plots are grouped in the Kill Swamp downstream group. From a tree species composition standpoint, the cluster analysis indicates that the upstream areas are different from the downstream areas, and the two downstream areas are different from each other.

```
********** CLUSTER ANALYSIS:    EUCLIDEAN DISTANCE,  FARTHEST NEIGHBOR
PC-ORD, Version 2.04
Percent chaining =   4.83
                         Distance (Objective Function)
        .000           4.632           9.263          13.895      18.526
        |-------+-------+-------+-------+-------+-------+-------+------
                         Information remaining (%)
     100.000          75.000          50.000          25.000        000
        |-------+-------+-------+-------+-------+-------+-------+------
bd15    --|----------|
bd9     |-|          |------------------|
bd7     |            |                  |
bu12    ------|------|                  |-------------------------------
kd5     ------|                         |
bd14    -|---|                          |
bd10    -|   |--------------------------|
bu11    -----|
bd8     |
bd2     |
r19     |--|
bu6     |  |
bu7     |  |
ku3     |  |------|
kd8     |  |      |
bd1     |  |      |-----|
bu3     |--|      |     |
r20     |         |     |
r26     ----------|     |-----|
bu4     ||              |     |
bu13    ||--|           |     |
ku12    ||  |           |     |
r23     -|  |-----------|     |
bu10    |   |                 |
ku4     |   |                 |
ku10    |---|                 |-----------------------------------------
ku11    |                     |
ku5     |                     |
ku9     |                     |
r24     |--|                  |
kd10    |  |----|             |
r25     || |    |             |
kd6     ||-|    |             |
kd3     ||      |-------------|
kd9     ||      |
ku2     |       |
kd7     --|-----|
kd2     --|
```

Figure 3.21. Cluster analysis of tree species composition by basal area by plots (bd=Beaverdam downstream, bu=Beaverdam upstream, r=reference area, kd=Kill downstream, ku=Kill upstream).

Numerous cluster analyses were performed on the data, to try and detect which environmental variables, when combined with vegetation data, resulted in separation of the upstream/downstream/reference areas into discernible groups. This would suggest which environmental variables were more valuable in the quantification of possible road construction effects on wetland ecological functions. The final cluster analysis is presented in Figure 3.22. The vegetation variables included in Figure 3.22 are percent canopy closure, basal area, herbaceous biomass and herbaceous diversity. The environmental variables included are water depth, percent soil carbon, percent soil nitrogen, total soil phosphorus, and soil decomposition rate. These variables proved valuable as functional indicators in this analysis.

```
********** CLUSTER ANALYSIS:    EUCLIDEAN DISTANCE,   FARTHEST NEIGHBOR
Percent chaining =    5.76
                          Distance (Objective Function)
       6.640      13808.300      27609.960      41411.620   55213.280
         |-------+-------+-------+-------+-------+-------+-------+------
                          Information remaining (%)
     100.000          75.000         50.000         25.000        .000
         |-------+-------+-------+-------+-------+-------+-------+------
bd15     |
bd14     ||
r25      |||
r26      |||
bd7      |||-----|
kd10     | |     |
kd9      | |     |
bd9      |-|     |--------|
r19      |       |        |
r23      |       |        |
r24      |       |        |
ku12     --------|        |---------------|
bd8      -|--|            |               |
r20      -|  |--|         |               |
bd2      |---|  |         |               |
bu11     |      |---------|               |
ku10     |      |                         |
bu3      |-|    |                         |
kd7      | |----|                         |-------------------------------
ku11     | |                              |
kd5      --|                              |
bd10     |                                |
bu12     |--|                             |
kd8      |  |------------------|          |
kd6      |  |                  |          |
bd1      ---|                  |----------|
bu4      ||                    |
bu10     ||--------|           |
bu13     -|        |-----------|
bu6      |--|      |
kd2      |  |------|
ku3      |  |
kd3      ---|
bu7      |----|
ku4      |    |-------|
ku2      |----|       |---------------------------------------------------
ku5      |            |
ku9      -------------|
```

Figure 3.22. Cluster analysis of vegetation and environmental variables by plot The vegetation variables included are percent canopy closure, basal area, herbaceous biomass and herbaceous diversity. The environmental variables included are water depth, percent soil carbon, percent soil nitrogen, total soil phosphorus, and soil decomposition rate (bd=Beaverdam downstream, bu=Beaverdam upstream, r=reference area, kd=Kill downstream, ku=Kill upstream).

The plots are split into two main groups between kd5 and bd10. These two main groups are split into two smaller subgroups between ku12 and bd8 and kd3 and bu7. The main upper group contains most of the downstream and all of the reference plots, while the main lower group contains most of the upstream plots. In this cluster analysis, a more perceptible upstream/downstream difference emerges. It demonstrates that water depth and soil conditions can indeed be helpful in detecting differences between upstream and downstream areas, differences that may be induced by disturbance. When the soil decomposition rate is included in the environmental variables, the separation between downstream/reference and upstream areas is not quite as distinct, but the separation is clear enough for soil decomposition to be considered a valuable diagnostic variable also.

DISCUSSION

This section follows the presentation order of wetland functions/indicators found in Figure 3.3.

1. Hydrologic Flux and Storage

Of all the defining ecological functions of a wetland, hydrologic flux and storage are the most important, because the sources and volume of water present in a wetland delimit the biological productivity, biogeochemistry, decomposition and community/wildlife habitat functions. An assessment of the hydrological characteristics of a wetland can also reveal very helpful information about rate changes for the other ecological functions. If fill and culvert-type crossings do impact wetland ecological function, the most probable and direct path would be through an alteration in hydrology, which would then be expected to influence the other wetland ecological functions, with possible effects similar to those listed in Table 1.3.

The results suggest that the hydrology of the sites has been impacted by the fill and culvert crossing structures, although to different degrees. The upstream/downstream difference in stream surface elevation at the Beaverdam Swamp site indicates that the crossing structure may be causing ponding upstream of the highway (a 19.8 cm difference), while at Kill Swamp the ponding effect is much less pronounced (a 2.25 cm difference). Again, these data are from the wells situated within 15 m of the crossings up and downstream.

The average monthly differences between the upstream and downstream areas at Kill Swamp for the study period ranged from 0.5-2.9 cm, which is not enough difference to expect a dramatic ecological functional change there. However, at Beaverdam Swamp, the average upstream stage ranged from 17.1-23.8 cm higher than the downstream stage during the study period (Figure 3.6), and the difference was 18 cm or more for 70% of the study period, often during the growing season (Figure 3.7). The depth and persistence of such inundation upstream of the crossing indicates that altered hydrology may have a significant effect on the other wetland functions at Beaverdam Swamp upstream. Also, differences in levels of the other wetland ecological functions measured during the study indicate that stream surface elevation and stream depth were meaningful and useful functional indicators.

Since the water surface elevation comparisons are made from the wells which are closest to the crossing embankment upstream and downstream (all are within 10-15 meters), the ponding effect's extent upstream at either site is not clear from the hydrograph data alone.

To investigate how water moved through the sites, the data from the wells situated approximately 150 meters up and downstream from the crossings was combined with the data from wells close to the crossings. Stream depth was calculated at each well and then used to compare how relative stream depth changed at each well during and after rainfall events.

At Kill Swamp, stream depth fluctuations during a given precipitation event at the upstream well locations were uniform (as water levels changed, depths changed at the respective wells at similar rates and by the same amounts). The water surface elevation upstream was slightly greater than downstream, indicating that some ponding was occurring upstream as stream levels rose, as noted before. When water levels receded after a precipitation event, the depth changes followed the same pattern (similar rates and amounts), only in reverse. Ponding exceeding the study period average of 2.25 cm did not ever persist more than 36-48 hours during the study period, indicating that the culvert was allowing ponded water to drain through to downstream without excessive delay.

At Beaverdam Swamp, stream depths at each well, during and after precipitation events, exhibited a different general pattern (see Figure 3.23). For the precipitation event depicted in Figure 3.23, as water rose and stream depth increased, depth 150 meters upstream and at both locations downstream increased simultaneously by approximately 60 cm. Stream depth just upstream of the crossing increased by 30 cm, only half the amount of increase at the other three points. During falling water surface levels, the pattern reversed itself and the water depth just upstream fell approximately 20 cm while at the other points it fell 45-50 cm. Stream depth just upstream of the crossing increased and decreased, uniformly over the course of the study, by approximately one-half the amount of the other locations during stream stage fluctuations, which indicates a hydrological change at this location due to highway flow-through design.

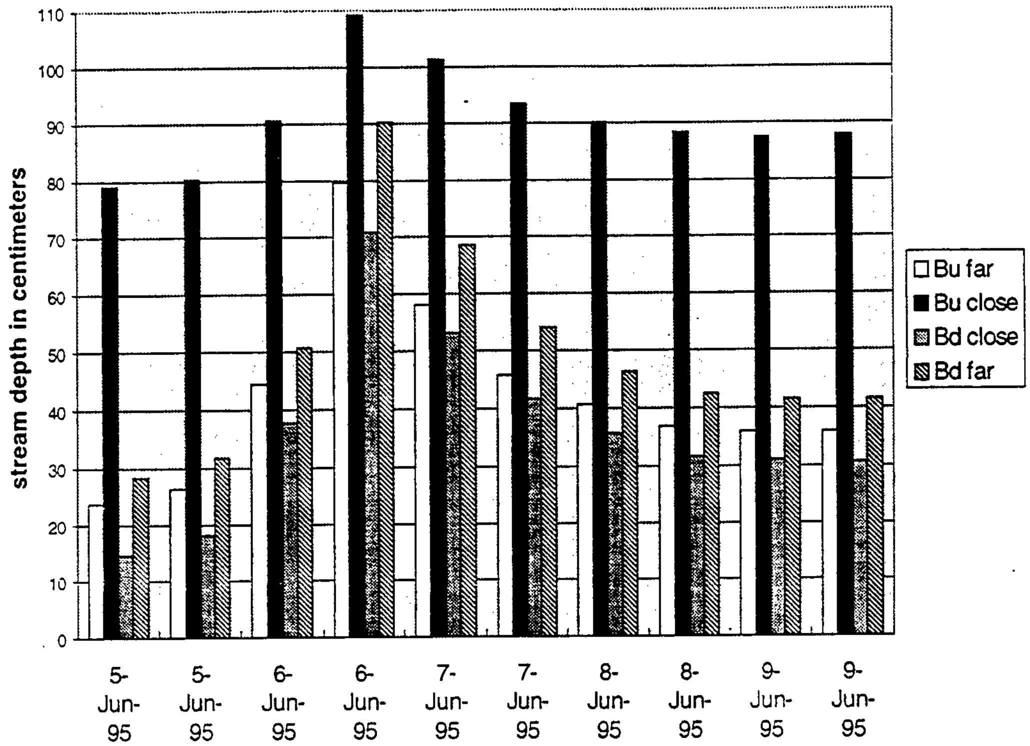


Figure 3.23. Comparisons of depths at the different well locations at Beaverdam Swamp during the course of a precipitation event (depth measurements are given for 8 AM and 8 PM each day). Bu far = the well farthest upstream, or approx. 150m from the crossing, Bu close = the upstream well close to the crossing (approx. 10m), Bd close = the downstream well close to the crossing (approx. 10m), and Bd far = the downstream well farthest from the crossing (approx. 150m).

The most logical explanation for this is that during rising stream levels, the stream channel just upstream of the crossing was expanding laterally and spreading out faster than at the other locations, inundating peripheral low-lying areas close to the crossing. This allowed the water that would have increased stream depth at the well just upstream to flow to and inundate those low lying areas rather than piling up at the culvert and increasing stream depth there by the same amount as at the other well locations. This hydrologic pattern indicates that the culvert had the capacity to transmit some of the water downstream during rising water, but not all of it. At Beaverdam Swamp, the stream was ponded upstream on average approximately 20 cm higher than downstream, and the depth changes during precipitation events suggest that flooding of low-lying areas just upstream of the crossing was common during rising stream stage. As at Kill Swamp upstream, ponding at Beaverdam Swamp upstream exceeding the average amount measured over the course of the study (19.8 cm) did not usually persist longer than 36-48 hours except after the biggest rain events, such as Tropical Storm Allison and Hurricanes Bertha and Fran.

After it was determined that the upstream study area at Beaverdam Swamp had an elevated stream surface compared to the downstream area, and that stream depth changes at the four wells during fluctuating stream levels were not synchronized, the next step in documenting possible crossing impacts to stream hydrology was to determine the size of the area of ponding upstream. Unfortunately, the resolution of our well location scheme, which was inhibited by the cost of the wells, was not extensive enough to precisely determine the extent of ponding upstream at Beaverdam Swamp. However, stream depth measurements made at plot centers were helpful in estimating how far upstream the ponding effect persists. The mean stream depth at Beaverdam Swamp, measured on two occasions, was 10-11.6 cm deeper upstream than downstream (the measurements were taken during low water conditions so that differences due to the road crossing's influence would not be confounded by high water levels).The mean for all the upstream plots was used as a benchmark to which individual plot depths were compared. Plot depths were consistently deeper than the mean on the third transect upstream from the crossing, but not the fourth. The third transect was approximately 100 meters upstream from the crossing embankment. It is assumed that since stream depths to this point upstream were greater than the overall mean of all the upstream plots, then the extent of ponding can be estimated to extend at least to the third transect upstream, or approximately 100 meters. The area associated with this ponding is approximately 1.4 ha. at low stream levels and would be expected to increase to two hectares or more as stream stage rises. At Kill Swamp there was not a significant upstream/downstream difference in stream depth, so ponding there, while present, is considered minimal.

The area of inundation upstream at Beaverdam Swamp, during low water levels, was slightly larger than the corresponding downstream area. At Kill Swamp there was a very minimal difference, with the downstream area actually being larger. Again the pattern of a more pronounced hydrological perturbation at Beaverdam Swamp is borne out by the data, although the method for area determination was not as precise as with surveying or global positioning technology. The comparison of areas of inundation at low stream stages is complicated by the fact that stream width varies naturally with slight variations in topography in low gradient streams in that part of the state. If ponding is occurring, it is possible to still observe no substantial differences in areas of inundation, if the stream banks are steeper upstream than downstream. In that case, the water depth would increase, but the area of inundation may increase by a smaller amount proportionally, and be of less value as an indicator of ecological function (or a more complicated indicator) than stream surface elevation or stream depth.

The hydrological variation detected upstream of the crossing at Beaverdam Swamp suggests that the other wetland functions, which interact with hydrological flux and storage, will be affected in that study area. The results of the other ecological functional indicators, to be discussed in the next sections, also show differences in ecological function between the study areas, and as a group validate the usefulness of the functional indicators used to measure hydrological flux and storage in these low-gradient palustrine wetland systems. Conversely, it is difficult to predict what changes in ecological function should be expected at Kill Swamp based on the much less pronounced hydrological differences measured there. The reason for the less pronounced depth differences at Kill Swamp may be due to culvert elevation differences between Beaverdam and Kill Swamp crossings.

2. Plant Productivity and Mortality

Spence (1982) reported that the distribution of plant species and associations in a wetland is determined primarily by water depth, and alterations of hydroperiod and water depth are known to cause changes in plant associations and communities. Studies have indicated that an increase in water depth causes changes in the herbaceous plant community, with increased mortality of the the less flood-tolerant species and a decline in species diversity (van der Valk et al., 1994). Varying responses to inundation by woody species is also well documented, and tolerance levels have been described for many bottomland and swamp species (Whitlow and Harris, 1977).

The woody vegetation survey data contains large upstream/downstream differences in basal area at both sites and a notable positive difference in basal area in the reference area. However, there does not appear to be a large difference in stand age among the study areas now, and the % crown closure analysis of the study areas before highway construction (Table 3.1) shows the areas to be uniformly stocked from a tree density standpoint. Since the areas were forested relatively uniformly before highway construction, the differences in basal area among study areas now suggests some type of disturbance, which would suggest that the road crossing is contributing to the basal area differences. The lack of evidence of logging and the unlikelihood and lack of evidence of fire lend confidence to the probability of highway effects, although, as noted before, it cannot be stated conclusively that the highway caused the die-off of timber due to the lack of pre-highway construction data on tree density and hydrology.

Another possible explanation for the upstream/downstream basal area differences is beaver damage. Evidence of beaver activity is present at both sites, both up and downstream, but it is not recent and, after very close inspection of all four crossing areas, seems uniform in its dispersal over both upstream and downstream areas. The number of trees which had been girdled by beavers was less at Kill Swamp than at Beaverdam, but the number at Beaverdam Swamp was not high (estimated at 10-12 trees per hectare). The reference area had minimal evidence of beaver activity. While beaver activity is important to the interpretation of the basal area data, it does not appear to be concentrated in any particular areas, namely upstream of the crossings, where basal area was consistently lower.

We hypothesized that fill and culvert-type crossings may attract beavers, since the culvert effectively narrows a 150-200 meter wide stream to less than 10 meters where the culvert is installed. Six other fill and culvert crossings were visited along four lane highways in the area, approximately 40-80 kilometers away, and all six crossings had evidence of beaver activity. So it may be that the crossings affect vegetation indirectly by attracting beavers to the immediate area. The beaver hypothesis was not a major component of the research proposal, so further investigation was beyond the scope of this project, but it is probable, based on field observations that beavers are attracted to the narrow stream reaches created by culvert construction, and influence wetland ecological function indirectly.

Woody species biomass results follow the same general pattern as the basal area results, although the reference area had less biomass than one area, Beaverdam Swamp downstream. Basal area and biomass give varying results when stands of different ages are being compared. Basal area is much less sensitive to tree age and is a better measure of occupancy or stocking. It is a good measure of how well stocked an area is, regardless of age, and for that reason is probably a more useful functional indicator than woody biomass if the differences in stand ages (or average dbh for each stand) are described and noted. Basal area is "normalized" with respect to stand age, allowing stands of different ages to be compared on the basis of how fully stocked they are, rather than on the weight of the trees present.

While beaver activity is certainly responsible for some tree mortality in the crossing areas and contributes to the differences in basal area and biomass between the crossings and the reference area, beaver activity is not solely responsible for those upstream/downstream differences. The ponding that occurs at both crossings has raised the water level upstream, and the permanent rise in water level, especially at Beaverdam Swamp, where the average stream depth increase is greatest, has stressed the trees there, especially the less flood tolerant hardwood species. The percentage of total basal area that is contributed by swamp tupelo, which is less flood tolerant than cypress, is 12%

lower at Beaverdam upstream than at Beaverdam downstream. The percentage of basal area that is contributed by cypress is 13% higher at Beaverdam upstream than at Beaverdam downstream, which indicates that ponding may be having an adverse effect on less flood tolerant species.

Twenty-two percent of the basal area in the reference area is composed of tree species such as red maple and sweetgum, which are not as tolerant of inundation for long periods of time as cypress and swamp tupelo (Whitlow and Harris, 1977). The percentage of basal area made up by such less tolerant species at both Beaverdam crossing areas and the Kill Swamp downstream area is only about 2%, which suggests that the hydrological disturbance caused by stream-crossing construction has created stream depths too deep for their survival. The average difference in water depth between the reference area and Beaverdam upstream was 22.6 cm. *Taxodium distichum*, *Nyssa sylvatica* var. *biflora*, *Fraxinus pennsylvanica*, and *Salix nigra* are the only species that are present on the study sites which can tolerate more than one year of constant inundation without dying. Several species found in the reference area which would not be able to survive the inundation that occurs upstream at Beaverdam Swamp include *Acer rubrum*, *Carpinus caroliniana*, *Liquidambar styraciflua*, *Liriodendron tulipifera*, and *Pinus taeda*. The lack of any substantive amount of basal area of these less flood-tolerant species upstream of the crossing suggests that the alteration of the hydrology, which resulted in ponding, has not only decreased basal area but also woody species diversity.

An important consequence of the ponding at Beaverdam upstream is that the water levels there may not recede enough during drought periods to allow the regeneration of the dominant cypress species. Cypress seeds will not germinate and survive under inundated conditions (Brown, 1984), so it is probable that when the trees persisting there die, the likelihood of woody regeneration is low.

Coarse woody debris (standing and on the ground) provide shelter for birds and aquatic wildlife, and represent a slowly released nutrient sink that is used by aquatic organisms on-site and downstream. In forest ecosystems under high water stress, coarse woody debris volume levels are often higher (Stoeckeler, 1965). The amount of coarse woody debris present at the sites was quite variable (Table 3.7). At Kill Swamp and at Beaverdam Swamp downstream, the amount of coarse woody debris was 2-3 times the amount present in the reference area. Beaverdam Swamp upstream had approximately 6 times the amount of the reference area, which was statistically significant. That much coarse woody debris indicates a stress level much higher than is present at the reference area, and when considered with the water depth results, suggests that living conditions for less flood-tolerant tree species at Beaverdam Swamp upstream are harsher than at the other crossing areas. When the study areas were grouped by upstream/downstream location and statistically compared, the upstream areas had statistically higher amounts of coarse woody debris than the downstream and reference areas. This result is no doubt due to the unusually high amount of debris at Beaverdam Swamp upstream, but what is interesting is that the downstream areas still had significantly more coarse woody debris than the reference area. That suggests more stress near the crossings (upstream and downstream) for trees. Beaver activity no doubt magnifies the differences, but it would be presumptuous to assume that it accounts for the entire difference.

The results of the herbaceous biomass survey (Fig. 3.13) show a trend towards more biomass upstream than downstream, and a significantly lower amount in the reference area. Although the upstream areas support from 42-48% more biomass, the differences are not significant due to a high degree of variability. The percent canopy closure (Fig. 3.11) shows that in the areas where there is greater canopy closure there is less herbaceous biomass. The downstream areas at Kill and Beaverdam Swamps have 27-40% greater canopy closure than the upstream areas, and the reference area has the most canopy closure of any area. The loss of canopy closure in the upstream areas coupled with an increased amount of herbaceous biomass indicate a long-term shift in primary production from woody to herbaceous species. The trends shown by the percent canopy closure and herbaceous biomass indicate that they are useful functional indicators.

3. Biogeochemical Cycling and Storage

Stream water quality

Stream water nutrient concentration levels do not indicate any significant nutrient absorption at either site, and nutrient removal is often a characteristic of lower flow velocity wetland ecosystems. But as stated before, that is probably due to the fact that there are numerous peripheral sources of nutrient input along the streams in and near the study areas apparent in the data, which overwhelm any study area absorption gradient that may be present, and make comparisons between study areas impossible.

Stream water temperature (Table 3.10) was warmer in the upstream areas in the spring and warmer in the downstream areas in the summer. A possible explanation for this is that the upstream areas, which have a lower percent crown closure than the downstream areas, allow more solar radiation through to the water in the spring. By late June foliage on the trees and the emergent macrophytes is full, so that the upstream areas, which have more macrophytic biomass, would have less direct sunlight reaching the water surface, and less heat being absorbed there. The downstream areas, while having more canopy cover, do not have as much emergent macrophyte biomass, and are also shallower, so that as the daytime temperatures increase, the shallower areas downstream see a rise in temperature over the deeper, more stable upstream areas (especially Beaverdam Swamp). The lower temperatures in the reference area probably result from the much higher canopy closure there and less direct solar radiation.

On July 26 and August 2 the downstream areas had warmer water temperatures-Beaverdam Swamp was significantly warmer, even after a 3.5 cm rainfall event the day before sampling on July 26 (precipitation records for the area were obtained from the North Carolina Climatologist's Office in Raleigh, N. C. for the study period). This may be explained by the fact that the upstream areas are deeper, especially at Beaverdam Swamp, so that high summer temperatures are moderated by the greater water mass. On July 23, the pattern of the 6/26 and 7/2 samplings is reversed, and the upstream areas are warmer. This may have been brought about by the exceptionally hot, dry weather during the preceeding week, which drove water temperatures up during the day. The upstream areas, especially Beaverdam Swamp, are deeper would be expected to cool less overnight than the shallower downstream areas.

In its entirety, the dissolved oxygen data (Table 3.11) shows a great deal of variability. On June 26 the only significant upstream/downstream difference was at Kill Swamp, where the downstream area had a higher concentration than the upstream area. The range of concentrations was much higher on this date than later dates, probably due to a 3 cm precipitation event on July 1, which also seems to have depressed streamwater temperatures somewhat below what they were one month before. On July 2, there was no significant difference in dissolved oxygen levels, which were much lower than on the previous sampling date. On July 23, the upstream areas at both crossings had significantly higher dissolved oxygen concentrations than the downstream areas. Again, the range of concentrations was quite low, but especially low downstream. The significant differences between the reference, upstream, and downstream areas indicates that dissolved oxygen concentration gradients can be important at both sites.

Differences between the upstream and downstream areas were detected in stream water temperature and dissolved oxygen concentration levels. The differences between areas are not consistent between sampling dates for temperature or dissolved oxygen, which indicates that more intense sampling is needed for these parameters to be useful when attempting to assess differences between study areas. Both these parameters seem to be closely associated with seasonal climatic and vegetational changes, and should be tracked on a long-term basis. These two components of wetland ecosystem function, particularly dissolved oxygen concentration, have often been dismissed as functional indicators because of the variability just mentioned, but it is clear that they may be valuable functional indicators if the complexity of changes inherent in their character can be better understood, and they can be analyzed in a uniform manner.

Sedimentation

There were no upstream/downstream/reference area differences in sedimentation rates (Table 3.12), rendering sedimentation rate a poor functional indicator in low gradient coastal plain ecosystems. Its usefulness would be more likely on streams that carry higher sediment loads, such as those in the Piedmont or mountain areas.

Total soil phosphorus concentrations (Fig. 3.14) were relatively uniform between all sites. The concentrations downstream at both sites were higher than upstream areas, and the difference at Kill Swamp was significant. The correlation coefficient for the percent soil carbon data (Figure 3.5) and the total soil phosphorus data (Figure 3.17) is 0.63, which indicates that soil P is moderately well associated with soil carbon content. The difference in soil phosphorus between the study areas was not nearly as great as the difference in soil carbon and nitrogen, suggesting that soil phosphorus flux dynamics, while somewhat tied to soil organic matter, do not appear to be as closely tied to soil organic matter as soil nitrogen. Total soil phosphorus was the only soil chemical functional indicator which exhibited an upstream/downstream difference at the study sites. Since the difference was detected at the Kill Swamp site, it does not appear that the difference can be attributed to hydrologic disturbance or ponding.

4. Decomposition\Soil Chemistry

The two rounds of soil burial cloth tests had consistent results (Figs. 3.15 and 3.16). In both installations the upstream area of Beaverdam Swamp had a significantly slower rate of decomposition that all the other areas, and as has been pointed out, the hydrology in that area shows more upstream/downstream variation than Kill Swamp or the reference area. The results suggest that a long-term increase in stream depth has decreased the decomposition rate at the Beaverdam Swamp upstream area. A great deal of research has been done on the effect of hydroperiod on decomposition rates in wetlands (Bell and Sipp, 1975; Brinson, 1977; Yarbro, 1979; Cuffney and Wallace, 1987). Generally, flooding has been shown to increase the rate of decomposition. However, Happell and Chanton (1993) and Lockabie et al. (1996) have shown that decomposition rates are slowed by prolonged inundation. The findings of this research corroborate these more recent studies.

With decreased decomposition rates, it can be predicted that the upstream area at Beaverdam Swamp will experience a slow build-up of organic matter over time which will eventually decrease the depth of the stream just upstream of the crossing. Without this extra storage capacity, when stream levels rise after precipitation events, it is logical to assume, from the hydrological pattern documented there, that the area upstream which is affected by ponding will increase, and the time that it takes for the ponded water to flow through the culvert will also increase from the present level.

There was not a statistical difference between up and downstream areas in soil carbon and soil nitrogen levels, but the differences between the crossings and the reference area was sizable. The differences between the two crossing study areas could possibly be due to the difference in stream water nutrient loading, but that would not explain the large difference between the reference area and the Beaverdam crossing study areas, which are just a few hundred meters apart on the same stream. The relative uniformity of differences between the carbon and nitrogen levels at the crossing areas and the reference area may indicate that the driving mechanism behind the differences is the same, and that the soil storage capacity at the study sites has been altered by disturbance. Soil chemistry functional indicators are helpful in contrasting the different study areas, but in this case are difficult to interpret. It can be said however, that the reference area, which has had the least amount of recent disturbance, has a much more nutrient-rich soil, which has important ecological ramifications for the crossing study areas.

5. Community/Wildlife Habitat

The herbaceous diversity across the study areas was statistically higher at the upstream areas (Fig.3.19). Openings in the tree canopy there have presumably made competition for sunlight less intense and increased the area where herbaceous growth is possible. Diversity is highest at Beaverdam Swamp upstream, where the water is, on average, the deepest, which indicates that there may be more flood tolerant species there or there are more microsite elevational differences. The important species that are commonly shared between the upstream study areas are *Murdania keisak*, *Leersia oryzoides*, and *Polygonum hydropiperoides*. *Murdania* is equally important in both areas, but *Leersia* and *Polygonum* are much more important at Kill upstream, where the water is shallower than at Beaverdam Swamp. *Spirodela polyrrhiza*, a floating plant adapted to deeper open water, is important at Beaverdam Swamp upstream but not at Kill Swamp upstream. *Echinochloa crusgalli* and *Ludwigia palustris*, which are wetland plants but not aquatic plants, are important at Beaverdam Swamp upstream but not at Kill Swamp upstream. Drier microsites or coarse woody debris are likely microsites for these type plants in areas of deeper water. So it appears that a complex combination of factors, increased sunlight, relatively deep water and numerous, less deep microsites all may be contributing to the increased diversity at the upstream areas, and in particular, Beaverdam Swamp upstream. The relative lack of diversity at the reference area is probably due to the high basal area/canopy closure characteristic, which decreases the amount of sunlight reaching the herbaceous layer and the relatively uniform, shallow stream depth there. Herbaceous plant diversity appears to correlate positively with microsite diversity and disturbance levels.

The multivariate cluster analyses (Figs. 3.20-23) subtly reinforce the hypothesis that vegetative data reflect the influence of disturbance - similar species exist under similar conditions. The fact that the upstream areas were clearly separated from the downstream areas, and the downstream areas were often more similar to the reference areas agrees with the overall pattern that has developed from the hydrologic flux data, which indicates that the upstream areas have been noticeably altered from the downstream areas, and that the crossing areas are different from the reference area.

Figure 3.22 is a cluster analysis of the plots that combined vegetation data and some of the other functional indicators measured in the study. The fact that the upstream and downstream areas were separated clearly in the dendrogram confirms that employing a cluster analysis, which combines vegetation with other environmental variables such as water depth, soil chemistry characteristics and soil decomposition, can help identify definitive functional indicators, and can facilitate the differentiation between disturbed and undisturbed areas within a common wetland type.

A summary of the statistical results by study area is presented in Table 3.13. The differences in the functional indicators between the upstream and downstream areas at Beaverdam and Kill Swamp and the reference area are shown by columns with different letters. Table 3.14 contains a summary of the statistical results from the model that combined the upstream and downstream areas from both crossings and compared them to the reference area for a few selected indicators.

Table 3.13. Statistical summary of results by study area.

	Beaverdam downstream	Beaverdam upstream	Reference area	Kill downstream	Kill upstream
stream depth	B	A	C	B	B
% crown closure	AB	C	A	BC	C
herbaceous biomass	AB	AB	B	AB	A
coarse woody debris	B	A	B	B	B
soil phosphorus	AB	B	AB	A	B
soil decomposition '95	A	B	C	C	C
Soil carbon	C	C	A	B	B
Soil nitrogen	BC	C	A	BC	AB
Herbaceous diversity	B	A	B	B	A

Table 3.14. Statistical summary of results with study areas grouped by upstream/downstream and reference location.

	upstream	downstream	reference area
% crown closure	B	A	A
Herbaceous biomass	A	AB	B
Coarse woody debris	A	B	C

Key differences between highway study sites and the reference area were detected in stream depth, percent crown closure, soil decomposition rates, soil carbon and nitrogen and herbaceous diversity. At Beaverdam Swamp, where a more pronounced upstream/downstream difference was detected, differences were found in stream depth, percent crown closure, coarse woody debris, soil decomposition rate, and herbaceous diversity. At Kill Swamp significant differences were only found in the soil phosphorus content, and herbaceous diversity indicators (there was also a substantial difference in basal area). The apparent lack of highway effects at Kill Swamp suggest that the upstream/downstream stream depth differences are important and are the major factor which has induced changes at Beaverdam Swamp. Differences in culvert elevation may be a factor in these differences. The fact that there are differences in basal area, herbaceous diversity and soil phosphorus concentration between upstream and downstream areas at Kill swamp, with only a small difference in the hydrological indicators between upstream and downstream areas, suggests that the forest that was present before highway construction consisted of a lower percentage of flood-tolerant species, or that repeated inundation along with a slightly raised water level stressed trees that were established based on the pre-construction hydrology.

At the crossings described in this study, some functional indicators described differences between upstream and downstream areas better than others. A list of ecosystem functions and the functional indicators which were most helpful in describing upstream/downstream differences is presented in Table 3.15.

Response Surface Models and Impact Assessment

The data summaries in Tables 3.13-14 are informative but do not convey quantitatively much about the relative differences between study areas. A much better way to synthesize functional indicator data is with response surface models (Richardson, 1994), that make comparisons of sites based on an individual indicator or which can be used to compare sites utilizing multiple indicators. The benefits of this type of approach are (1) an integrated and scaled analysis of ecosystem function is presented, (2) study area comparisons can be made to a reference area and (3) a graphical model provides a clear basis for site comparisons to agencies and the public.

Table 3.15. List of wetland ecosystem functions and the functional indicators which were determined to be useful in describing them in this study.

Ecosystem function	Functional indicator
1. Hydrologic flux and storage	stream surface elevation stream depth relative stream depth
2. Plant productivity	woody species basal area percent crown closure coarse woody debris herbaceous biomass
3. Biogeochemical cycling and storage	
Stream water quality	stream water temperature* dissolved oxygen concentration*
Sedimentation	sedimentation rate* total soil phosphorus concentration
4. Decomposition	soil decomposition rate soil C and N concentration*
5. Community/wildlife habitat	herbaceous diversity multivariate cluster analysis macroinvertebrate assessment

*these functional indicators were either too difficult to interpret or they did not indicate upstream downstream differences in this study, but may be useful in detecting disturbance at other sites.

Figure 3.24 is a response surface model which compares the crossing sites by basal area.

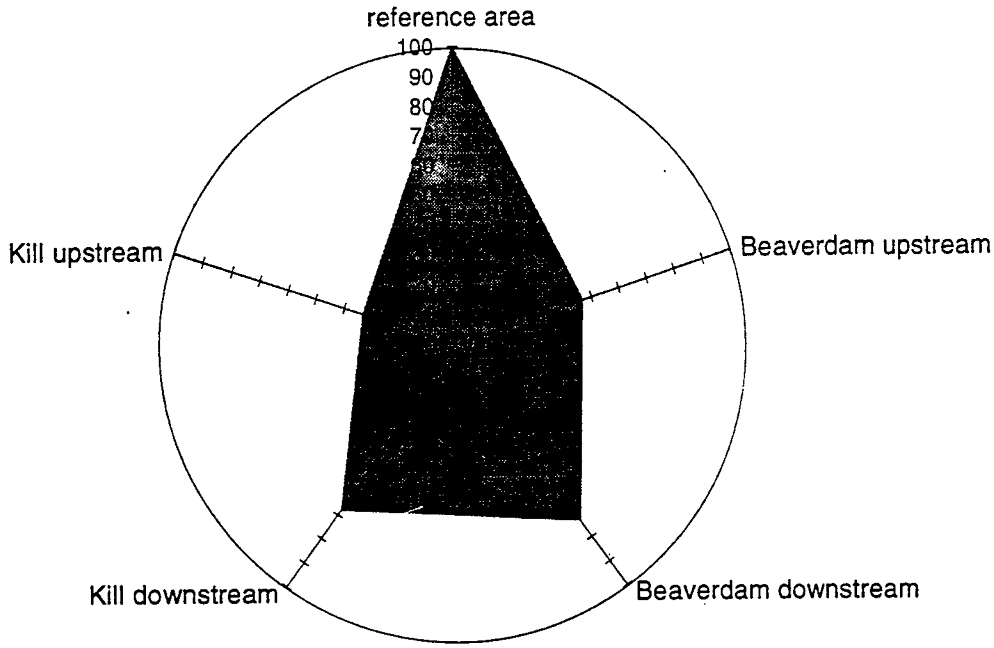


Figure 3.24. Response surface model comparison of crossing areas by basal area.

In this response surface model, as in the ones to follow, the reference area is used as a benchmark to which the other areas are compared. Functional indicator measurements from the reference area are scaled to 100%, and the functional indicator values from the other areas are then compared to the reference area using the percentile scale. For instance, the basal area at Beaverdam Swamp downstream is approximately 73% of that measured in the reference area, and the basal area at Beaverdam Swamp upstream is approximately 48% of that found in the reference area.

Figure 3.24 illustrates that the basal area in the upstream areas is approximately 25-35% less than in the downstream areas, and that the downstream areas both have about 30% less basal area than the reference area.

In Figure 3.25 several functional indicators for Beaverdam Swamp downstream are combined and presented. A functional indicator that detected differences between study areas from each of the five major ecological functions, hydrologic flux and storage, plant productivity, biogeochemical cycling and storage, decomposition, and community/wildlife habitat, is used. Any indicator can be employed in the model-in the following figures the indicators chosen were selected based on how well they depicted differences in ecological functions between study areas. The scale for Figure 3.25 is greater than for Figure 3.24 because there are functional indicator values in this study area which are in excess of the values measured in the reference area. The inner circle represents the reference area functional indicator levels (100%) and the outer circle represents 200% of those reference area levels. Response surface models for the Beaverdam Swamp upstream, Kill Swamp downstream, and Kill Swamp upstream areas are presented in Figures 3.26-3.28.

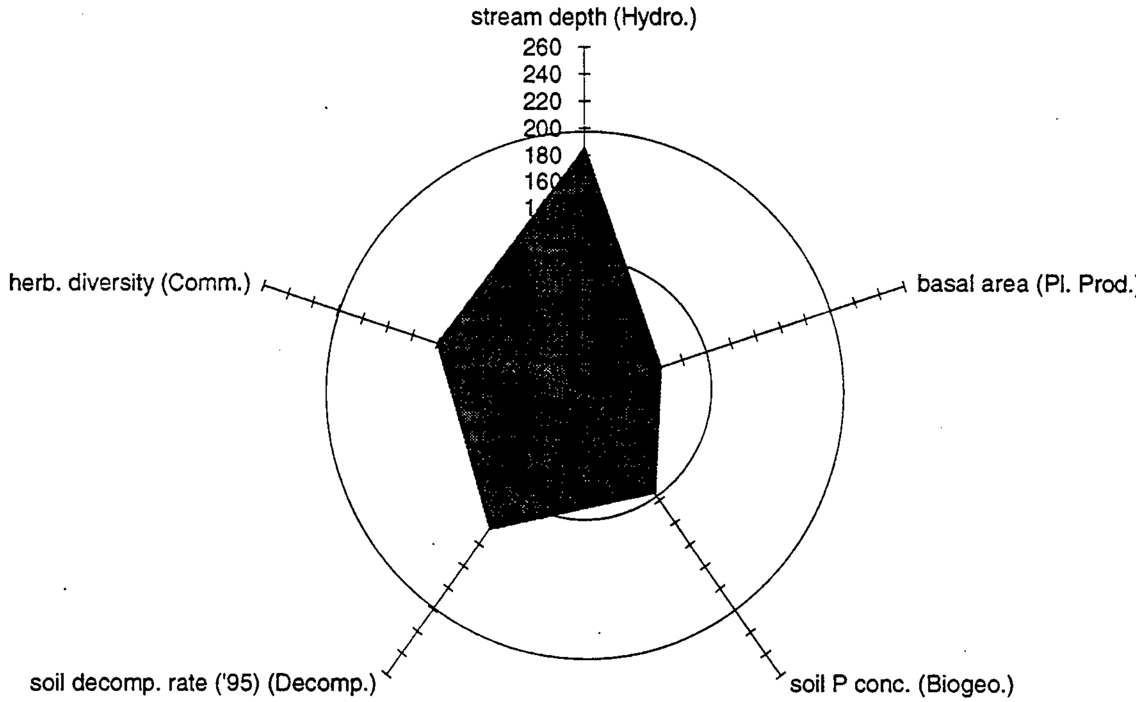


Figure 3.25. Response surface model for the Beaverdam Swamp downstream study area, with the wetland ecological function that the functional indicator relates to in parentheses.

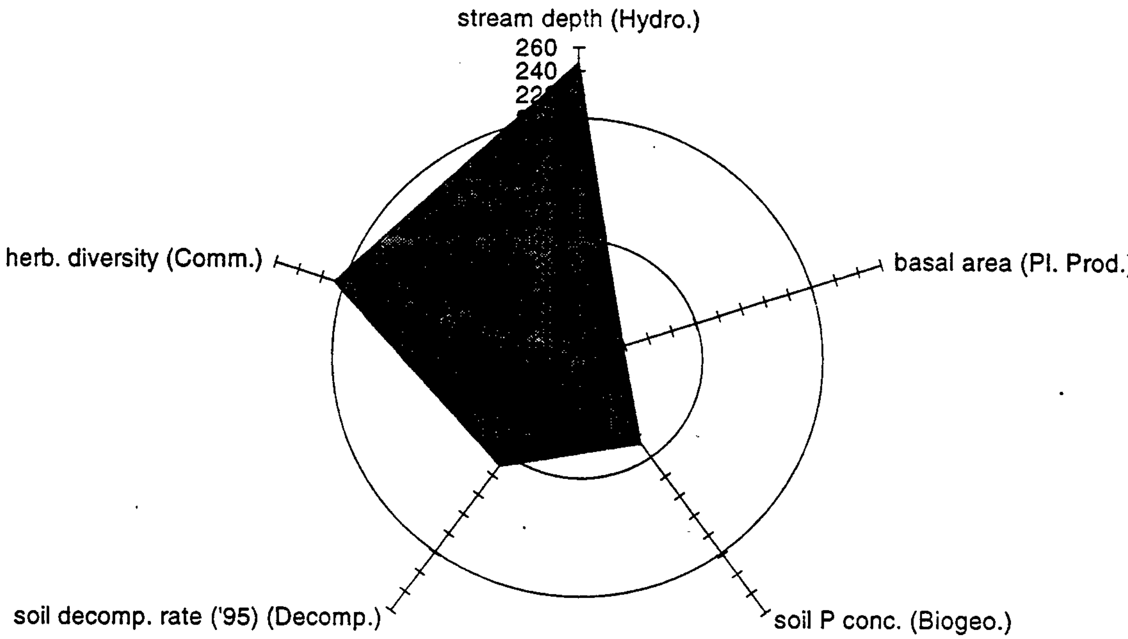


Figure 3.26. Response surface model for the Beaverdam Swamp upstream study area, with the wetland ecological function that the functional indicator relates to in parentheses.

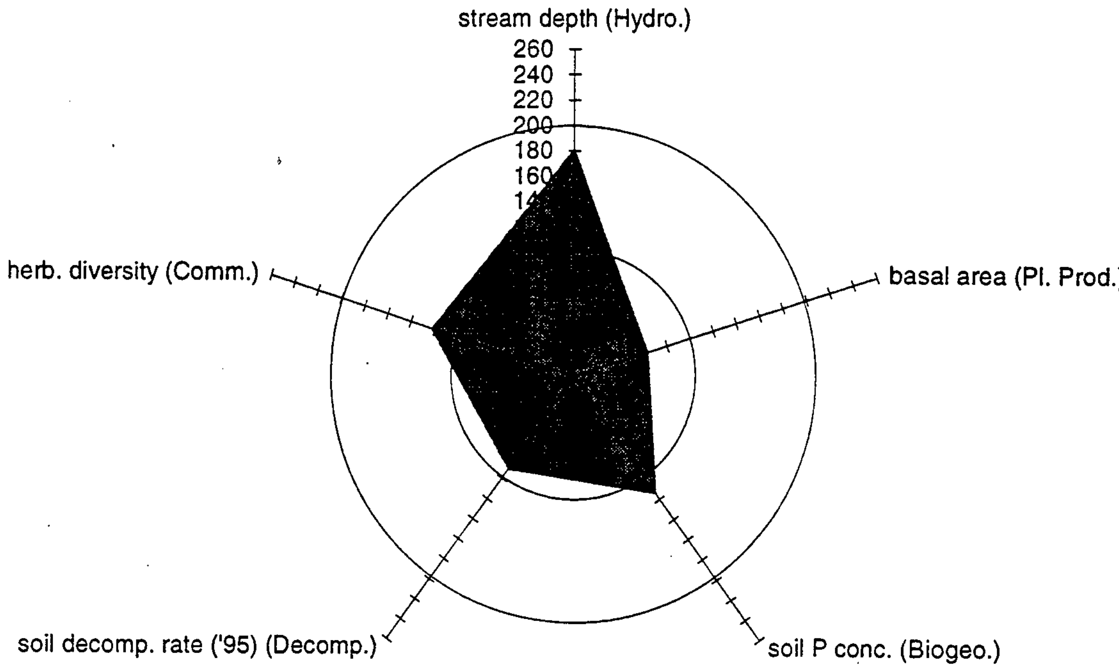


Figure 3.27. Response surface model for the Kill Swamp downstream study area, with the wetland ecological function that the functional indicator relates to in parentheses.

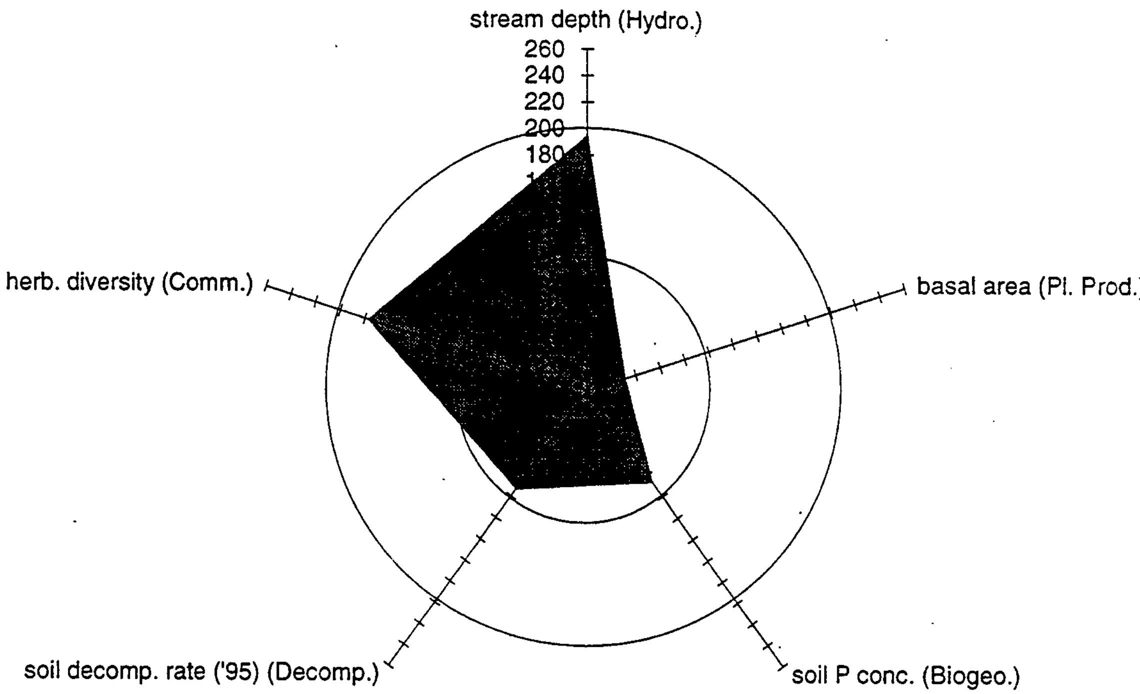


Figure 3.28. Response surface model for the Kill Swamp upstream study area, with the wetland ecological function that the functional indicator relates to in parentheses.

Using the response surface models the functional indicator data can be interpreted and useful comparisons can be made between the study areas and between the study areas and the reference area. Figures 3.25-26 illustrate that stream depth is greater in the Beaverdam study areas than in the reference area, and Beaverdam upstream is approximately 60% deeper than downstream, relative to the reference area. Basal area is lower in the Beaverdam study areas than in the reference area, approximately 40% downstream and 60% upstream. Soil phosphorus levels downstream are comparable to the reference area, but upstream they are approximately 15% less. Soil decomposition rates are higher at both Beaverdam study areas than at the reference area, but the rate upstream is approximately 20% less than downstream. Herbaceous diversity is greater at the study sites than at the reference site, but is nearly 100% greater at the upstream area relative to the reference area.

At Kill Swamp (Figs.27-28), stream depth is greater at the study areas than at the reference area, with no meaningful difference between upstream and downstream areas. Basal area comparisons are similar to those at Beaverdam swamp. Soil phosphorus concentrations are higher downstream than reference or upstream areas, but do not vary from the reference area considerably. Soil decomposition rates at the study areas are similar to each other and to the reference area. Herbaceous diversity is approximately 20% higher downstream and 80% higher upstream than in the reference area.

Taking the study area models as a group, basal area is lowest where stream depth is highest and vice versa. Soil phosphorus concentrations are greater in the downstream areas and the soil decomposition rate is higher there also. Herbaceous diversity is not as great when basal area is high and stream depth is at lower levels. Also, herbaceous diversity and stream depth are much greater in the study areas than in the reference area.

Comparing the shaded areas to the reference area circles, the downstream areas are most similar to the reference area, Kill swamp being more similar. The upstream areas are most dissimilar from the reference area than the downstream areas, the upstream area at Beaverdam swamp being more dissimilar. From the models, the upstream indicators which are the most different from the reference area indicators are stream depth, basal area, and herbaceous diversity, suggesting the greatest highway impacts are to the hydrological, productivity, and community functions at those locations.

Impact Assessment

Quantitative assessment of the impact of the highway crossing's presence on wetland ecological function can be done in many ways (assuming that the hydrological perturbation, which is likely caused by the fill and culvert crossings, drives the differences in the other functional indicators measured, which is also a logical assumption). Arguably the most logical method would be to use the reference area as a present-day surrogate for the undisturbed condition at each study area, since it has been confirmed that the woody vegetation component of these wetland ecosystems was relatively similar at each study area before highway construction began. Under these assumptions, the response surface models can be used to assess the highway's impact. The percentage differences between the reference area and the crossing study areas can be measured on the response surface models for each indicator, and can be used to assess the impact to each ecological function individually at each study area and/or can be combined to illustrate the impact to the ecosystem as a whole. Using Beaverdam Swamp upstream as an example, the value from the response surface model for each ecological function is:

$$F = H + P + B + D + C$$

where F = functional difference (%), H = hydrological flux and storage, P = plant productivity and mortality, B = biogeochemical cycling and storage, D = decomposition, and C = community/wildlife habitat

1. H = hydrologic flux and storage-stream depth..............................245%
2. P = plant productivity and mortality-basal area................................38%
3. B = biogeochemical cycling and storage-total soil phosphorus 85%

4. D = decomposition-soil decomposition rate 110%

5. C = community/wildlife habitat-herbaceous diversity 209%

The percentage difference from the reference area for each ecological function is:

Functional difference (%) = Study area value – reference area value

1. H =hydrologic flux and storage-stream depth..............................+145%

2. P =plant productivity and mortality-basal area...................................-62%

3. B =biogeochemical cycling and storage-total soil phosphorus...............-15%

4. D =decompositioin-soil decomposition rate ....+10%

5. C =community/wildlife habitat-herbaceous diversity ....+109%

Deviations from the reference state, whether positive or negative, should be considered to be impact related, and converted to a positive value, for a combined total difference of 341%. Dividing this total by five, which gives equal weight to each ecological function, gives an overall ecological functional difference of 68.2% between the Beaverdam Swamp upstream area and the reference area. If the same calculations are made for the other study areas, the differences in ecological function between them and the reference area are:

Beaverdam Swamp downstream ....176%÷5 = 35.6%

Kill Swamp downstream ....163%÷5 = 32.6%

Kill Swamp upstream ....261%÷5 = 52.2%

When the data are presented this way, it is clear that the upstream areas, particularly Beaverdam Swamp, have experienced greater alteration from the reference area than the downstream areas, where the impact level is almost identical between the two sites.

The next step is determining the level of impact which would require compensation. This should be an ecologically based process which draws on as much functional assessment data as possible. The process of determining the level of impact over which compensation would be required is complicated by the many levels of complexity (size, water source, etc.) in these type ecosystems, by the expense involved in compensation, and by the difficulty inherent in successfully restoring riparian palustrine systems in the coastal plain eco-region. The many viewpoints range from calling for mitigation for any impacted ecological function or for total avoidance due to the difficulty of restoration/creation, to the abandonment of the view that all wetlands are important and need to be protected by a high level of compensation. A position somewhere in the middle, which combines ecological science with the reality of the unavoidability of continued wetland disturbance is needed. Our integrated ecosystem model provides a method for determining the level of impact. Our model suggests that the upstream sites are functioning at 32-48% of the level of the reference area (100% - the difference values), several years after highway construction. The downstream sites are functioning at 64-67% of the level of the reference area.

From an impact assessment standpoint the hydrological flux and storage indicator data is clearly the most valuable, because differences in the hydrologic functional indicators can be expected to generate differences in most, if not all of the other ecological functions. While the differences in the hydrologic functional indicators between the crossing areas and the reference area measured in this study ranged from 80-150%, a 20% increase in stream depth can create a hydrologic pattern of permanent rather than intermittent inundation in low-lying areas in these low gradient streams, which would cause a shift in plant productivity and mortality, and the effect would cascade down through the remaining wetland ecological functions. A practical, measurable and ecologically sound recommendation for the maximum amount of alteration allowed in the hydrologic flux and storage ecological function before compensation is required is therefore 20%. A caveat that the alteration of *any* ecological function by more than 20% should require compensation is also recommended, based on the fact that all the wetland ecological functions are interrelated and connected, and alteration of one ecological function can be expected to alter some or all of the others. This level of impact allows for some disturbance or development without the compensation "penalty". When human activities breach the functional disturbance threshold, the wetland functional assessment methodology presented here gives regulatory agencies a valuable tool. It is a precise, repeatable metric which detects and measures the functional impacts of disturbance, give a numerical comparison of impacted and reference areas which facilitates the calculation of wetland mitigation ratios, and provides a monitoring tool for assessing mitigation progress where it is deemed necessary.

This study suggests that the fill and culvert-type crossings have influenced and altered wetland ecological functions at these sites. Although this research was exploratory in nature with the main goal of methodological development, enough evidence is present to safely state that:

1. using pre-construction data (aerial photographs) it can be determined that the study areas and the reference area were similar in regards to plant productivity and disturbance levels.

2. based on the logical assumption that the other wetland ecological functions in the crossing study areas were functioning in a similar way as the reference area before highway construction, the construction of fill and culvert-type crossings across these streams has altered the hydrology, at one crossing significantly, and in so doing altered the remaining wetland ecological functions also.

3. further research is needed to explicitly explore the cause and effect relationship between the construction of fill and culvert-type highway crossings and wetland ecological function impacts.

4. the useful wetland ecological functional indicators identified in this study should be utilized as a starting point for future research aimed at assessing the functional impacts that fill and culvert-type crossings have on these type of wetland ecosystems.

Useful functional indicators for this particular classification or type of wetland have been identified (Table 3.13), and therein lies a key to the value of this research. The most widely accepted wetland functional assessment methodology from a regulatory standpoint is the Hydrogeomorphic Classification system (HGM) (Brinson, 1993). Smith *et al.* (1995), in a USCOE technical report, use HGM as the basis for determining wetland classification, and using the classification of the wetland as a starting point, determine the functional indices most likely to be useful in the functional assessment of that particular wetland. The variation inherent in similarly classified wetlands that are geographically far removed is recognized in this methodology and it is suggested that regional subclasses be identified, so that more appropriate functional indictors can be chosen for measuring the ecological functions of a particular wetland classification in a particular region. The study sites in this research would be classified as Riverine (Lower perennial). Currently there is not a suggested list of indicators for this classification that are recommended by the USACOE, and no results of field trials which have identified and verified useful functional indicators for Riverine (Lower perennial) wetlands can be found in the literature. So, this research has succeeded in at least partially completing the list of useful functional indicators for Riverine (Lower perennial) wetlands in this region of the United States, by actual field testing and measurement on-site.

The fact that some functional indicators employed in this research, such as sedimentation rate or stream water nutrient concentration, did not reveal differences in the upstream/downstream/reference areas does not preclude their usefulness elsewhere. As set forth by Smith (1995), the appropriateness of functional indicators will be in large part determined by the classification type of the wetland of interest. Useful functional indicators in this study may not be as applicable to organic soil flats or lacustrine fringe wetlands and vice versa, so each particular wetland classification will have to have its own functional indicator list developed.

A peripheral point that is raised by this research, which has not been adequately addressed by any functional assessment methodology currently in use, is one of assessment timing. A crucial question for any functional assessment methodology should be, "What is the appropriate length of time that should be allowed to elapse before an accurate functional assessment of disturbance should be made?" The sites studied in this research had been disturbed approximately 7-8 years before any functional assessment measurements were made, and a strong argument can be made that the sites had still not reached "equilibrium" in ecologically functional terms. There are indications that tree survival and regeneration at Beaverdam Swamp upstream may be limited in the long-term by ponding, so that when the trees that occupy the site now die, regeneration may not be forthcoming due to increased water depth, and a total shift in primary productivity from woody to herbaceous plants may occur. The impacts of disturbance on ecological function may not be fully realized for many more years. So it is important to remember, as functional assessment methodology is advanced from its present, formative stage, that ecological functional change can take years develop after disturbance, and accurate ecological functional assessments must consider this fact. Certainly many impacts can be observed and measured in the short-term after disturbance, but some impacts may take years to be fully realized.

CONCLUSIONS

This research suggests that fill and culvert-type highway crossings are influencing wetland ecological function by establishing that there are differences in wetland ecological function levels between the upstream, downstream and reference study areas. Further study is warranted to more clearly define the extent of that influence.

- There were differences detected in hydrological flux and storage at Kill Swamp and Beaverdam Swamp.
- Mean stream surface elevation was 19.8 and 2.25 cm higher upstream than downstream at Beaverdam Swamp and Kill Swamp respectively.
- Stream depth averaged 10-12 cm more upstream at Beaverdam Swamp than downstream, which was statistically significant, and ponding there was estimated to extend at least 100 m upstream.

- There were differences in plant productivity detected between study areas. The reference area had approximately 30% more basal area per hectare than the downstream areas, which had 30-50% more basal area than the upstream areas.
- Tree biomass in the upstream study areas was approximately 10-40% lower than in the other areas.
- The reference area contained several less flood-tolerant species than the other areas, suggesting that stream hydrology had been disturbed near the highway.
- The reference area had a significantly higher percent crown closure than the upstream study areas.
- Coarse woody debris volume was highest in the upstream areas and lowest in the reference area.
- The upstream study areas had a significantly higher amount of herbaceous biomass than the reference area, suggesting a trend in increased herbaceous biomass production where basal area and canopy closure had decreased.
- No statistical differences in biogeochemical cycling and storage were found except for total soil phosphorus, which was higher in the downstream study areas.
- The soil decomposition rate was significantly lower upstream at Beaverdam Swamp than downstream, suggesting that ponding was inhibiting decomposition there.
- Herbaceous diversity was significantly higher at the upstream study areas than at the reference area.
- Cluster analyses showed separation between the upstream and downstream study areas when herbaceous data was analyzed using other functional indicators as

environmental variables, indicating that there statistical differences in ecological function between the upstream and downstream study areas.

- Useful functional indicators were identified for each wetland ecological function (see Table 3.15). Other indicators, which did not provide statistical differences between the study areas may be useful in other wetland types.
- The summary and synthesis of wetland functional information is facilitated by using response surface models, which allow comparisons between various functional indicators and the sites where they were measured.
- A method for assessing the impact of disturbance on wetland ecological functions was presented, using the results and response surface models from this study. The assessment procedure showed that the upstream study areas had experienced more functional alteration than the downstream study areas, when compared to the reference area.
- A wetland functional disturbance threshold of 20% is recommended, after which compensation for lost ecological function is warranted.
- Currently there are no specific lists of useful functional indicators available from regulatory agencies, such as the USACOE, which recommend functional assessment techniques, and the useful indicators identified through field testing in this study are an important and much needed starting point for the compilation of such lists.
- This research points out the fact that the HGM Classification system and its functional assessment methodology need to better address the problem presented by short term vs. long-term ecological function change brought about by disturbance, and how to accurately assess functional change in that context.

Further research, which utilizes pre and post-construction data is needed to better elucidate the impacts of highway construction in wetlands. This "pilot" research suggests many useful functional indicators for continued study in palustrine forested coastal plain wetlands.

RECOMMENDATIONS

The results of our research indicate that there are ecological functional differences between the study areas and the reference area in addition to differences between upstream and downstream study areas at each site. As noted in the Results section, the defining ecological function for any wetland is hydrological flux and storage. Results suggest that at each upstream study area this function was altered by the fill and culvert-type crossing, most notably at Beaverdam Swamp, where mean stream depth and stream surface elevation were greater than in the downstream area. This change in hydrological flux and storage induces change in the other wetland ecological functions such as vegetation productivity, biogeochemical cycling and storage, decomposition and community/wildlife habitat.

The crossings were engineered based on estimated stream flow quantities during a fifty-year storm event. Culvert size and placements were designed so that the road bed would not be eroded by high stream flows during a large precipitation event. At each crossing, one reinforced concrete box culvert (18'x 6' at Beaverdam Swamp and 12'x 8' at Kill Swamp) was used to accommodate normal stream flow and 2 peripheral culverts at each site (72" at Beaverdam Swamp and 60" at Kill Swamp) were installed to convey the extra stream flow when it exceeded normal flow rates. These peripheral pipes were elevated so that they only conveyed flow when stream stages were high. It appears that the design is assumed to convey normal stream flows without ponding if it is built to convey a much greater flow during fifty year precipitation events.

However, it appears that at Beaverdam Swamp the overwhelming interest in preventing roadbed erosion during high stream flows has resulted in reduced conveyance of normal stream flow causing ponding upstream. It is recommended that estimates of both storm and normal stream flows be made and the specifications of the culverts (both size and elevation criteria) be designed to assure normal stream flow conveyance beneath the road without any ponding upstream, as well as sizing to ensure adequate storm flow. At Beaverdam and Kill Swamps, the culverts (all three combined) reduced the effective width of the streams by approximately 85%. If effective stream width reduction is calculated using only the main culvert, which is only stream water conveyance during normal flow, the effective stream width is reduced by 90-92% by the crossing designs. When stream flow is constricted to a reach this narrow, the elevation of the bottom of the culvert is critical and must be low enough to assure unimpeded flow. If a small error is made and the culvert is installed above the proper elevation, the ponding effect will be magnified by the constriction of the stream reach inherent in the design.

It is recommended that the NCDOT revisit fill and culvert-type crossings built recently (along US 117, US 17, or I-40 etc.) to determine if the elevation of the culverts allows normal flows without ponding. If ponding is occurring, then the upstream stream elevation will be more than one or two centimeters in excess of the downstream elevation.

Rectification of upstream ponding at fill and culvert-type crossings can be acheived by precise installation of the proper size culverts. If it is difficult to achieve elevational accuracy (± 5 cm) when installing culverts then the number and size of culverts should be increased to ensure unimpeded stream flow. The wetland areas impacted by fill and culvert-type crossings will be reduced by the recommended design changes that accommodate both low and high flow conditions. The added cost of culvert construction should be offset by reduced mitigation costs as functional assessment techniques become more precise in determining areas of impact.

APPENDIX I

List of tree species found in the study areas

Acer rubrum L.	red maple
Carpinus caroliniana Walt.	ironwood
Fraxinus pennsylvatica Marsh.	green ash
Liquidambar styraciflua L.	sweet gum
Liriodendron tulipifera L.	yellow-poplar
Nyssa sylvatica var. *biflora* (Walt.) Sarg.	swamp tupelo
Pinus taeda L.	loblolly pine
Quercus phellos L.	willow oak
Salix nigra Marsh.	black willow

APPENDIX II

List of herbaceous species found in the study areas

Bidens spp.	beggar ticks
Boehmeria cylindrica L.	false nettle
Cyperus stigosus L.	umbrella sedge
Echinocloa crusgalli (L.) Beauv.	barnyard grass
Hydorlea quadrivalvus Walter.	water-pod
Leersia oryzoides (L.) Swartz	cut grass
Ludwigia palustris (L.) Ell.	water-primrose
Lycopus virginicus L.	water horehound
Murdannia kiesak (Hasskarl) Hand.-Mazz.	Asian spiderwort
Onoclea sensibilis L.	sensitive fern
Penthorum sedoides L.	ditch stonecrop
Peltandra virginica L.	arrow-arum
Polygonum hydropiperoides Michx.	smartweed, knotweed
Rhyncospora corniculata (Lam.) Gray	horned-rush
Saururus cernuus L.	lizard's-tail
Scirpus cyperinus (L.) Kunth	bulrush
Spirodela polyrrhiza (L.) Schlied.	duckweed
Typha latifolia L.	cattail

LITERATURE CITED

Bell, D.T., and S.K. Sipp, 1975. The litter stratum in the streamside forest ecosystem. Oikos 26:391-397.

Brinson, M.M., B.L. Swift, R.C. Plantico, and J.S. Barclay, 1981a. Riparian Ecosystems: Their Ecology and Status, U.S. Fish and Wildlife Service, Biol. Serv. Prog., FWS/OBS-81/17, Washington, D.C., 151p.

Brinson, M.M., A.E. Lugo, and S. Brown, 1981b. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annu. Rev. Ecol. Systematics 12:123-161.

Brinson, M.M. 1977. Decomposition and nutrient exchange of litter in an alluvial swamp forest. Ecology 58:601-609.

Brinson, Mark M. 1993. A Hydrogeomorphic Classification for Wetlands, Wetlands Research Program Technical Report WRP-DE-4, U. S. Army Corps of Engineers Waterway Experiment Station, Vicksburg, Mississippi.

Brown, Claire A., 1984. Morphology and biology of cypress trees. In *Cypress Swamps*, K.C. Ewel and H.T. Odum eds. Univ. of Florida Press, Gainesville Fl.

Cowardin, L. M., V. Carter, F.C. Golet, and E.T. LaRoe, 1979. Classification of Wetlands and Deepwater Habitats of the United States, U.S. Fish and Wildlife Service Pub. FWS/OBS-79/31. Washington, D.C., 103 p.

Cuffney, T.F., and J.B. Wallace, 1987. Leaf litter processing in Coastal Plain streams and floodplains of southeastern Georgia, USA. Arch. Hydrobiol. (Suppl.) 76:124.

Dahl, T.E., and C. E. Johnson, 1991. *Wetlands Status and Trends in the Conterminous United States Mid-70's to Mid-80's*, U. S. Dept. of Interior, Fish and Wildlife Service, Washington, D. C., 21 p.

Dabel, C. V., and F.P. Day. 1977. Structural comparisons of four plant communities in the Great Dismal Swamp, Virginia. Bulletin of the Torrey Botanical Club 104:352-360.

Faulkner, S. P. and C. J. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils, in Constructed Wetlands for Wastewater Treatment, D.A. Hammer, ed., Lewis Publishers, Chelsea, Michigan, pp. 41-72.

Happell, J.D. and J.P. Chanton. 1993. Carbon remineralization in a north Florida Swamp forest: effects of water level on the pathways and rates of soil organic matter decompostion. Global Biogeo. Cycles 7:3 pp. 475-490.

Harrison, A. F., P. M. Latter, and D. W. H. Walton, 1988. *Cotton Strip Assay: An Index of Decomposition in Soils*, Institute of Terrestrial Ecology, Natural Environment Research Council, U.K., Titus Wilson and Son, Ltd.

Hill, M.O., P.M. Latter, and G. Bancroft. 1988. Standardization of rotting rates by a linearizing transformation,in Cotton Strip Assay: An Index of Decomposition in Soils, A.F. Harrison, P.M. Latter, and D.W.H. Walton, eds., Institute of Terrestrial Ecology, Natural Environment Council, Titus Wilson and Son, Ltd.

Hupp, C. R., and E. E. Morris. 1990. A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. Wetlands 10:1 p.107-124.

Knight, A.W., and R.L. Bottorf, 1984. The importance of riparian vegetation to stream ecosystems, in California Riparian Systems: Ecology, Conservation, and Productive Management, R.F. Warner and K.M. Hendrix, eds., Univ. Calif. Press, Berkley, pp. 160-167.

Lockabie, B.G., A.L. Murphey, and G.L. Sommers, 1996. Hydroperiod influences on nutrient dynamics in decomposing litter of a floodplain forest. Soil Sci. Soc. Am. J. 60:1267-1272.

McCune, B., and M. J. Mefford. 1995. PC-ORD. Multivariate Analysis of Ecological Data, Version 2.0. MjM Software Design, Gleneden Beach, Oregon.

Mitsch, W.J., and K.C.Ewel, 1979, Comparitive biomass and growth of cypress in Florida wetlands, Am. Midl. Nat. 101:417-426.

Mitsch , W. J. and J.G. Gosselink, 1993. *Wetlands,* Van Nostrand Reinhold Company. New York, N.Y.

North Carolina Climatology Office, Dept. of Environment, Health and Natural Resources, Division of Natural Resources, North Carolina State University, Raleigh, N.C.

Novitzki, R.P., 1979. Hydrologic characteristics of Wisconsin's wetlands and their influence on floods, stream flow, and sediment, in Wetland Functions and Values: The State of Our Understanding, P.E. Greeson, J.R. Clark, J.E. Clark, eds., American Water Resource Association, Minneapolis, Minn., pp. 377-388.

Paine, D. P., 1981. *Aerial Photography and Image Interpretation for Resource Management.* John Wiley and Sons, Inc. New York.

Parizek, R. B., 1970. Impact of highways on the hydrologic environment. Proceedings of the First Annual Geomorphology Symposia Series.
In Environmental Geomorphology, Binghampton, N. Y. (Oct. 16-17, 1970).

Pope, R.B., C. D. Maclean, and D. A. Bernstein, 1961. Forestry uses of aerial photos, Pacific Northwest Forest and Range Experiment Station, USDA Forest Service, Portland OR.

Richardson, C. J., 1994. Ecological functions and human values in wetlands: a framework for assessing forestry impacts. Wetlands 14:1, p.1-9.

Ritchie, J. C., and J. Roger McHenry. 1990. Application of radioactive fallout Cesium-137 for measuring soil erosion and sediment accumulation rates and patterns: a review. Journal of Environmental Quality 19:215-233.

Roberts, L., 1993. Wetlands trading is a loser's game, say ecologists. Science 260:1890-1892.

Roe, H. B., and Q. C. Ayers, 1954, *Engineering for Agricultural Drainage*, McGraw-Hill, New York, 501 pp.

Rossiter,J.A., and R.D. Crawford, 1983, Evaluation of artificial wetlands in North Dakota: recommendations for future design and construction. Transportation Research Record, 948:21-25.

SAS Institute, Inc. 1989. SAS/STAT User's Guide. Version 6. Cary, North Carolina., USA.

Scheidt, M. E. 1967. Environmental effects of highways. ASCE Journal of Sanitary Engineering, Proc., 93:5 p. 17.

Schlesinger, W.H. 1976. Biogeochemical limits on two levels of plant community organization in the cypress forest of Okefenokee Swamp. Doctoral thesis. Cornell University, Ithaca, New York.

Shannon, C. E., and W. Weaver. 1949. The mathematical theory of communication. Urbana, IL: University of Illinois Press.

Shuldiner,P.W., D.F. Cope and R.B. Newton. 1979. *Ecological effects of highway fills on wetlands-research report.* NCHRP Report 218A p. 34.

Smith, R. Daniel, Alan Ammann, Candy Bartoldus, and Mark M. Brinson. (1995). An approach to assessing wetland functions using Hydrogeomorphic Classification, reference wetlands, and functional indices. Wetlands Research Program Technical Report WRP-DE-9, US Army Corps of Engineers Waterways Experiment Station.

Spence, D.H.N. 1982. The zonation of plants in freshwater lakes. Advances in Ecological Research 12:37-125.

Stoeckeler, J. H. 1965. Drainage along forest swamp roads: lessons from Northern Europe. Journal of Forestry 63(10):772.

United States Department of Agriculture, Soil Conservation Service. 1985. Soil Survey of Sampson County, North Carolina.

United States Environmental Protection Agency. 1979. Methods for chemical analysis of water and wastes. USEPA. Cincinnati, Ohio.

van der Valk, A.G. 1991. Response of wetland vegetation to a change in water level. *in* C. M. Finlayson and T. Larson, editors. Wetland management and restoration- Proceedings of a workshop, Sept. 12-15, 1990. Swedish Environmental Protection Agency, Solna, Sweden.

van der Valk, A.G. 1994. Assessing the impacts of an increase in water level on wetland vegetation. Ecological Applications, 4(3) pp. 525-534.

van Tongeren, O.F.R., 1995. Cluster analysis, in Data Analysis in Community and Landscape Ecology, R.H.G. Jongman, C.J.F. ter Braak, and O.F.R. Tongeren, eds.,Cambridge University Press.

Verry , E.S., and D.H. Boelter, 1979. Peatland hydrology, in Wetland Functions and Values: The State of Our Understanding, P.E. Greeson, J.R. Clark, J.E. Clark, eds., American Water Resource Association, Minneapolis, Minn., pp. 389-402.

Weber, W. G., and L. A. Reed, 1976. Sediment runoff during highway construction. ASCE Civil Engineering 46:3 p.76.

Whitlow, T.H., and Richard W. Harris. 1979. Flood tolerance in plants: a state-of-the-art review, U.S. Dept. of Commerce Technical Report E-79-2, National Technical Information Service.

Yarbro, L.A., 1979. Phosphorus cycling in the Creeping Swamp floodplain ecosystem and exports from the Creeping Swamp watershed. Ph.D. diss. Univ. of North Carolina, Chapel Hill (Diss. Abstr. 80-13989).