12-1996

An Ecological Assessment of Wetland Mitigation Projects in East Tennessee

Kim Pilarski
University of Tennessee, Knoxville

To the Graduate Council:

I am submitting herewith a thesis written by Kim Pilarski entitled "An Ecological Assessment of Wetland Mitigation Projects in East Tennessee." I have examined the final electronic copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Geography.

Sally P. Horn, Major Professor

We have read this thesis and recommend its acceptance:

Carol Harden, James A. Drake

Accepted for the Council:
Carolyn R. Hodges

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

To the Graduate Council:

I am submitting herewith a thesis written by Kim Pilarski entitled "An Ecological Assessment of Wetland Mitigation Projects in East Tennessee." I have examined the final copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Geography.

Sally P. Horn

Sally P. Horn, Major Professor

We have read this thesis
and recommend its acceptance:

Accepted for the Council:

Associate Vice Chancellor and
Dean of The Graduate School

AN ECOLOGICAL ASSESSMENT OF WETLAND MITIGATION PROJECTS IN EAST TENNESSEE

A Thesis
Presented for the
Master of Science
Degree
The University of Tennessee, Knoxville

Kim Pilarski
December 1996

DEDICATION

This thesis is dedicated to my children
Ian Edward Pilarski Turner
and Alexandra Jane Pilarski Turner.
I did this for you and because of you.
You are both my shining stars, geographers in the making.

ACKNOWLEDGMENTS

I would like to express my gratitude and appreciation to everyone who contributed to this project. Sincerest thanks and appreciation are extended to Dr. Sally P. Horn, my committee chairperson and advisor throughout my graduate career at the University of Tennessee, Knoxville. Her advice, suggestions, and editing of this manuscript were invaluable, and her continual support, friendship, patience, and encouragement has led me to open many doors I might otherwise have passed. I would also like to thank Dr. Carol P. Harden for her friendship, as well as her input on this research. Her courses and enthusiasm served to develop my interest in water resources. Dr. James A. Drake provided valuable ecological insight and suggestions, as well as field assistants, lab space, and friendship. I would also like to thank Dr. M. Ted Nelson for his continual support and encouragement during my employment with the Tennessee Valley Authority. His interest and appreciation of the role of geographers outside of academia allowed me to gain valuable experience and to begin a career I truly love.

Several government agencies and individuals helped make this study possible. With input of staff from the Tennessee Department of Transportation, the City of Maryville, the Environmental Protection Agency, and Tennessee Valley Authority, I was able to select research sites. Mr. Bruce Eslinger graciously allowed me access to his property. The University of Tennessee Department of Geography provide field equipment, and the University of Tennessee Department of Ecology also provided field equipment and lab space.

Special thanks are extended to the faculty, staff, and fellow graduate students in the Department of Geography. Dr. Sidney Jumper, former department head, encouraged the development of a top-notch group of faculty that truly cares about students. His leadership and insight are much appreciated. I would also like to thank fellow graduate students Kendra McSweeney, Lisa Northrop, Glenn Hyman, John and Jacque Benhart, Mike Daly, Patrick Solomon, as well as my best friends Sarah Gault and Brian Huskey. Together and separately you provided a network of support, humor, and occasional childcare, and also a level of intellectual and social stimulation that I will always treasure.

Support and friendship was also provided by my numerous co-workers at the Tennessee Valley Authority. Particular thanks to my River Action Team, as well as other folks from Water Management, especially Dr. Larry Bray, Judy Bair, and Kirk Kelley. Anne Ferrell, my running partner and confidante, also deserves thanks.

I would also like to thank Jeffrey R. Duncan for his input on my sampling design, dedicated field assistance, and also for his love, encouragement, and friendship. He helped me through many a dark hour when all the work ahead seemed truly overwhelming. Similarly I would like to extend a heartfelt thanks to my parents, Harry and Carol Pilarski. Their love of the natural world during my childhood and beyond led to

my interest in environmental protection and thus this research. Lastly, I would like to thank my children, Ian and Alexandra Pilarski Turner. Their unwavering love, support, patience and understanding made this all possible.

ABSTRACT

Federal wetland protection regulations stipulate that developers who destroy natural wetlands are required to construct mitigation or replacement wetlands. Despite the frequency of wetland mitigation, few studies have evaluated the ability of mitigated wetlands to mimic the ecological function and community composition of natural wetlands. Fewer still compare mitigation sites with existing natural wetlands in the same ecological region.

Studies from other freshwater habitats suggest that comparisons of the ratios of functional feeding groups (FFGs) of benthic macroinvertebrates (i.e., shredders, scrapers, collectors, predators) may provide useful information about the ability of mitigated wetlands to mimic the ecological functions of natural wetlands.

In this study I examined the ratios of FFGs of benthic macroinvertebrates in three mitigated wetlands and in a natural reference wetland in the Ridge and Valley physiographic province of the southeastern United States. Using a multihabitat sampling protocol, I sampled open water areas, *Typha latifolia* zones, and areas of mixed emergent vegetation < 1 m in height. Chi-square tests indicate that the ratios of FFGs in the mitigated wetlands are significantly different than those of the natural wetlands. Based on these results, I conclude that the mitigated wetlands have a much different trophic structure than the natural wetland, and thus do not replace the ecological function of natural wetlands lost to development. The results from this study raise questions about the long-term viability of these systems and the use of wetland mitigation as a natural resource management strategy

LIST OF ABBREVIATIONS

Clean Water Act	CWA
Course Particulate Organic Matter	CPOM
Fine Particulate Organic Matter	FPOM
Functional Feeding Group	FFG
Geographic Information System	GIS
National Wetlands Inventory	NWI
Tennessee Department of Environment and Conservation	TDEC
Tennessee Department of Transportation	TDOT
Tennessee Valley Authority	TVA
Tennessee Wildlife Resources Agency	TWRA
United States Army Corps of Engineers	COE
United States Department of Agriculture	USDA
United States Environmental Protection Agency	EPA
United States Fish and Wildlife Service	USFWS

1. INTRODUCTION

Wetland conservation has come under intense scrutiny, as natural resource managers seek to balance private property rights with public concerns about water quality issues, biodiversity, and wildlife conservation. Wetlands regulation has fallen into intense legal debate in the last decade, because protecting wetlands spans both public and private interests. Wetlands contain water, a public resource, therefore they fall under the purview of public domain. But they are also land, sometimes dry. Land is seen as a private resource, and government intervention into private property rights is often problematic.

As a means to balance private property rights with concern about water quality and wetlands as a unique natural resource, regulations allow developers to "mitigate" impacts to wetlands. Webster's New Collegiate Dictionary (Soukanov 1984) defines "mitigate" as "to make or become less severe or intense: to moderate." In the simplest sense, wetlands mitigation can mean simply avoiding a wetland area. Wetland mitigation may also mean restoring a degraded wetland by improving the hydrology of the site or planting wetland vegetation. In many cases, however, wetland mitigation refers to the process of creating a wetland in an adjacent upland area, to compensate for natural wetlands lost to development.

The stated goal of mitigation in Section 404 of the Clean Water Act (CWA) is the "functional replacement" of the natural wetlands. Wetland functions have been broadly defined as the hydrologic, biogeochemical, and ecological processes that occur in wetlands (National Research Council 1995). Hydrologic functions such as short-term surface water storage and the resulting reduction of flood damage are easily understood. In fact, wetland hydrology is easily recreated. The more complex ecological and biogeochemical processes are less well understood, and thus more difficult to recreate (Reimhold 1994).

Wetlands have vital roles in the maintenance of habitat, water quality, and biodiversity. Ecosystem-level wetland functions are often based on vegetation, faunal components, sanctuary refuge value for fish, wildlife, and waterfowl, as well as food chain production and export to adjacent ecosystems (Reimhold 1994). The long-term success of mitigated wetlands in replacing these functions is largely unknown. Monitoring of wetland mitigation projects has focused primarily on hydrology and vegetation (Newling and Landin 1985, Landin et al. 1989, Pacific Estuarine Research Laboratory 1990); further, most monitoring projects have focused on coastal wetlands (Kusler and Kentula 1989, U.S. Army Corps of Engineers 1990).

Do mitigated wetlands mimic the ecological function of natural wetlands?

The use of mitigation as a resource management tool in the absence of a means to evaluate its ecological success begs several questions. How should ecological function be assessed? If mitigation does not, in fact, create ecologically functional wetlands, should it still be used as a mechanism for balancing natural resource concerns with private property rights?

This thesis examines the ability of mitigated wetlands to replace or to mimic the ecological functions of natural wetlands in the Ridge and Valley ecoregion of East Tennessee. The main objective of this research was to compare functional feeding groups of benthic macroinvertebrates in three mitigated wetlands with that of a natural/reference wetland. As a link between wetland vegetation and higher order consumers like amphibians, fish and waterfowl, wetland macroinvertebrates are critical components of wetland ecosystems. Their numbers and kinds can provide indication of habitat quality and ecosystem function. In this study I quantitatively compared differences in community composition of functional feeding groups (FFGs) between the natural system and the mitigated or created systems to results from studies of other aquatic systems, and drew conclusions from those results about the functional viability of the mitigated wetlands. Prior to this study, FFGs had not been used to evaluate the community structure of mitigated wetlands; thus as a second objective of this study I evaluated the use of this methodology as a means to monitor wetland mitigation projects.

It is hoped that this study will contribute to a better understanding of the ecological processes underlying wetland mitigation as well as the use of mitigation as a wetland conservation tool. Wetlands are uncommon in the Ridge and Valley ecoregion. In areas where wetlands are often small and isolated, what impact will mitigation have on landscape-level biodiversity? Connections between terrestrial systems, aquatic systems, and individual wetlands are critical to the support of many organisms; reduction in average size, total number, linkages, and density of wetlands will result in loss of landscape-level functions such as maintenance of biodiversity, water quality and natural hydrologic flow regimes (National Research Council 1995).

The role of wetland mitigation in the larger arena of restoration ecology is also important. The science of restoration ecology is still in its development stages, and much scientific debate centers on the ability of humans to restore or recreate degraded or lost ecosystems to self-perpetuating functional natural ecosystems. Cairns (1993) stated: "Abundant evidence indicates that preserving ecosystems is far less expensive than restoring them." Should mitigation even be an option, given the paucity of evidence that supports it?

Organization of the Thesis

This thesis begins with background information on wetlands, wetland loss, and a discussion of wetland mitigation and ways in which the success of mitigation efforts have been measured (Chapter 2). Specific attention is given here to the use of macroinvertebrates in aquatic environmental studies. I justify the use of FFGs in this study and describe studies from other systems. Chapter 3 is a description of study sites and research methods. In Chapter 4, I present research results from each wetland community assessment. I discuss those results in Chapter 5 in the context of similar research. In Chapter 6, I conclude with a discussion of the implications of my results with respect to the use of wetland mitigation as a conservation tool. I consider the implications of the study for future research needs, monitoring and assessment requirements, and the larger issues of landscape-level biodiversity.

2. WETLAND MITIGATION AND THE USE OF FUNCTIONAL FEEDING GROUPS TO COMPARE NATURAL AND MITIGATED WETLANDS

Definition of Wetlands

A wetland is fundamentally what the term implies – wet land. Wetlands are transitional zones between dry terrestrial habitats and aquatic systems, and as such exist in a state of flux. These environments have several features in common: shallow water or saturated soil, and biota that has adapted to the fluctuating hydro-regime. Regional and local differences in vegetation, hydrology, water chemistry, soils, topography, climate, and other factors result in the development of a wide variety of wetlands.

The interaction of hydrology, vegetation, and soil results in the emergence of characteristics unique to wetlands (Environmental Laboratory 1987), and has formed the framework for defining wetlands for regulatory purposes. Official federal guidelines for defining wetlands are based on three main characteristics: (1) the presence of water; (2) soils that differ from adjacent uplands; and (3) the presence of vegetation adapted to the wet conditions (Mitsch and Gosselink 1986). Current regulatory procedures for defining and delineating wetlands are described in the *1987 Corps of Engineers Wetlands Delineation Manual* (Environmental Laboratory 1987). This manual describes wetlands as "those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions."

(Environmental Laboratory 1987, p.13) The diagnostic environmental characteristics include the presence of hydrophytic species of vegetation, hydric soils, and permanent or periodic inundation during the growing season sufficient to permit the growth of hydrophytic vegetation.

Ecological and Economic Significance of Wetlands

The environmental characteristics that make wetland environments unique also serve to create a valuable resource. As a transitional zone between terrestrial and aquatic environments, wetland areas perform ecological functions that can be translated into both socioeconomic values and environmental quality benefits (Kusler 1983, Adamus 1983, Mitsch and Gosselink 1986, Salvesen 1990). Organic productivity within wetlands provides vital nutrients and habitat for fish, waterfowl, shellfish, and other animals; coastal wetlands are especially important habitats for estuarine and marine fish and shellfish, various waterfowl, shorebirds and wading birds. It is estimated that 90% of the marine species of commercial value use estuarine environments at some point in their life-cycle (Kusler1983).

Because wetlands are transition zones from uplands to aquatic systems, they have a high species diversity, with species adapted to both the aquatic and terrestrial systems (Mitsch and Gosselink 1986). Changes in the hydroperiod affect vegetation; cycles of plant growth are often based on the changes in water level that naturally occur in wetland systems. With shifts in the vegetational community, the faunal community will vary as well. Ecological niches exist that can support not only fully aquatic plants and animals, but also partially aquatic and terrestrial organisms (Kusler et al. 1994).

Wetland environments also provide important hydrologic functions (Carter et al. 1979, Greeson et al. 1978, Daniel 1981, Adamus and Stockwell 1983, Sather and Smith 1984, Salvesen 1990). Wetlands serve as natural buffer zones within watershed, slowing down and absorbing excess water during storm events. As water moves through wetlands, spreading out over a wide flat area, velocity of runoff decreases, flood peaks and storm flows are attenuated, and tributary and main-channel peak flows are desynchronized (Carter 1986). Carter (1986) cites studies that indicate flood peaks are significantly lower (by as much as 80%) in watersheds that preserve large areas of wetlands. By slowing the flow of water, wetlands promote the deposition of sediments that would otherwise be carried downstream. In riverine systems, significant amounts of sediment are deposited in adjacent wetland areas when flooding occurs. This provides additional benefits for reservoirs, dams, and flood control structures within the watershed; sedimentation significantly shortens the life span and effectiveness of dams and flood control structures, and impacts the ecological health of reservoir systems.

Located between both aquatic and terrestrial systems, wetlands intercept runoff from both land and water. Through the mechanisms of element cycling, sediment deposition, ion and molecule adsorption, and temperature modification, the quality of water leaving a wetland may be substantially improved over that entering the wetland (Carter et al. 1979, Sather and Smith 1984, Mitsch and Gosselink 1986, Burke et al. 1988). By retaining sediment, wetlands serve as sinks for chemicals sorbed to the sediments. Anaerobic and aerobic processes such as denitrification and chemical precipitation remove organic and inorganic nutrients and toxic materials, leading to improved water quality (Kadlec and Kadlec 1978, Mitsch and Gosselink 1986, Phillips 1989).

Despite the economic and ecological significance of wetlands, the loss and alteration of wetland habitats has been drastic. The Wetland Loss Report, carried out by the United States Fish and Wildlife Service (USFWS), estimated that 200 years ago there were 87,668,000 ha of wetlands in what is now the continental United States. By the 1980s that total had decreased to 42,420,000 ha.

Wetlands in Tennessee

No complete wetlands inventory has ever been conducted in Tennessee, and no national wetlands inventory has ever accurately quantified the wetlands of Tennessee (Governors Interagency Wetlands Committee 1994). At best, wetland inventory data have been described as limited for Tennessee (Trettin 1995). A 1994 study conducted for the State of Tennessee used six separate datasets to determine the acreage of wetlands in Tennessee. There is wide variation in the different inventories, however, due to differences in defining, identifying, and delineating the resource base being inventoried, and differences in the accuracy of the methods employed in the inventory. An indirect estimate of the state's "wetlands capability base," defined as areas that are <u>capable</u> of supporting wetlands based on the occurrence of hydric soils, projects that Tennessee has approximately 808,000 ha of wetland capability base. A breakdown of this estimate shows that only 4% of the hydric soils occur in the eastern Ridge and Valley section of the state. Statewide, most of this area has been converted to non-wetland uses (e.g. agriculture) and is no longer classified as wetland.

An estimate of wetland area in Tennessee by Dahl (1990) gave a current figure of 317,948 ha present in the mid-1980s. The study then added this figure to United States Department of Agriculture (USDA) agricultural drainage statistics to conclude that approximately 782,548 ha probably existed in the 1780s. A study conducted in 1988 by the Tennessee Department of Environment and Conservation (TDEC) used both aerial photography and satellite imagery to determine that Tennessee had 258,227 ha of vegetated wetlands; of that figure 230,684 ha (89%) were in western Tennessee and 27,543 ha (11%) were in the rest of the state.

Other estimates of current wetland area are based on the National Wetlands Inventory (NWI) maps produced by the USFWS. These are produced from United States Geologic Survey (USGS) 7.5" quadrangle maps digitized into a Geographic Information System (GIS). The Tennessee Wildlife Resources Agency (TWRA) digitization of NWI maps concludes that a minimum of 316,532 ha of wetlands exists in the state of Tennessee.

Within the Tennessee Valley region, wetland losses in bottomland hardwood forests, mesic riverine forest, and riverine marshes and swamps have been extensive. S. Davis (personal communication) estimated that by the mid-1980s Tennessee had lost 59% of its original wetlands. The vast majority of the losses both in the southeast and nationally occurred during a period from the mid 1950s to the mid 1970s. Draining and clearing of inland wetlands for farming accounted for most of the losses. Wetland loss due to agricultural conversions appear to be decreasing; during the last twenty years, industrial, residential, and urban development account for the most wetland loss both in Tennessee and nationally (Salvesen 1990, Governor's Interagency Wetlands Committee 1994). Transportation impacts (highway and airport construction) in particular are having an increasing impact on wetland loss in Tennessee (Governor's Interagency Wetlands Committee 1994).

Wetland Regulation

Wetlands are the only ecosystem to be completely regulated across all public and private lands within the United States (Blumm and Zahela 1989). The most important, and probably most controversial, of the several Federal laws, executive orders and administrative rules that are designed to protect wetlands from being converted or modified is Section 404 of the CWA. Section 404 derives from the United States Army Corps of Engineers (COE) broad powers to control the navigable waters of the United States under the Rivers and Harbors Act of 1899. Originally passed as part of the 1972 Federal Water Pollution Control Act Amendments, and reauthorized in 1977 as the CWA, Section 404 gives the COE enforcement power over the discharge of any dredged or fill materials into the waters of the United States, including wetlands. The United States Environmental Protection Agency (EPA) also has oversight over the 404 permitting process.

Under Section 404, the COE is responsible for issuance or denial of permits for development activities taking place within wetlands. When the COE receives an application for a 404 permit, the application is processed and a public notice is issued. A 30 day comment period follows during which time the COE, as well as local, state, and federal agencies (including the USFWS and the National Marine Fisheries Service) review the application. The Section 404 permit program requires that the impact of a proposed dredge and fill project be evaluated in terms of the public interest, including such factors as flood control, navigation, recreation, water supply, and environmental and socioeconomic concerns.

Mitigation

Section 404 of the CWA stipulates that unavoidable adverse environmental impacts of the proposed project must be mitigated. The Council on Environmental Quality, which oversees the 404 program, states that mitigation includes:

> a) avoiding the impact altogether by not taking a certain action or parts of an action;

b) minimizing impacts by limiting the degree or magnitude of action and its implementation;

c) rectifying the impact by repairing , rehabilitating, or restoring the affected environment;

d) reducing or eliminating the impact over time by preservation and maintenance operations during the life of the action;

e) compensating for the impact by replacing or providing substitute resources or environments.

In general, once the COE grants a permit with a mitigation requirement, the mitigation must follow one of four strategies, based upon the above criteria: avoidance/minimization, restoration, enhancement, or creation (Salvesen 1990). The process of wetland mitigation may involve (1) enlarging a natural wetland at one location to compensate for a loss in the same wetland at another location; (2) creation of an entirely new replicate wetland within the same hydrologic reach as the impacted wetland; or (3) construction of a replacement wetland at an off-site location (Larson 1986).

The goal of wetland mitigation, as incorporated into the Section 404 regulation, is "to restore and maintain the chemical, physical, and biological integrity of our Nation's waters (Kruczynski 1989, p. 144). Ideally, mitigated wetlands should replace all the ecological functions provided by the destroyed wetlands. The biotic structure and hydrologic function of the created wetland should mimic that of the destroyed wetland, to ensure that the ecological functions lost are replaced.

Assessing the Success of Wetland Mitigation

There is a range of opinions about the success of wetland mitigation projects; indeed, there is a range of opinions as to what even constitutes the term "success" in evaluating created wetlands. As stated above, success from an ecological standpoint indicates that the mitigated wetland replaces the ecological functions of the destroyed, previously existing wetland. From a regulatory standpoint, success of projects is often rated on the basis of compliance with permit requirements, or even whether or not the projects were implemented (Quammen 1986). To insure <u>no net loss</u> of wetland habitat values, it is necessary to have high expectations for wetlands constructed or modified for the purpose of mitigating lost wetland habitat; functions of constructed and natural wetlands should be similar, but how similar is debatable (Zedler 1990). A project's success is often very site-specific. It may mean replacing <u>all</u> aspects of a natural system; for other sites, replacement of some functions to some level may be deemed sufficient (Kentula et al. 1993).

Several studies have indicated successful establishment of wetland environments as a result of mitigation (Seneca et al. 1976, Newling and Landin 1985, Zentner 1988, Erwin 1989). Most studies, however, have focused on coastal marshes on the Atlantic Coast. In addition, most studies of mitigation projects have been qualitative, site specific case studies (Baker 1984, Reimold and Cobler 1986, Mason and Slocum 1987). Kentula et al.(1993) point out, however, that case studies of single sites or comparisons of pairs of sites do not provide information that can be used with any certainty to make ecosystem management decisions .

Few studies have used reference sites to assess to what degree mitigated wetlands replicate natural wetlands in the same ecoregion (Quammen 1986, Kentula et al. 1993). Some wetland scientists are concerned that wetland restoration and habitat replacement attempts are not replacing lost wetland values (Race 1985, Quammen 1986, Larson 1988, Zedler 1990). Regulatory and development interests recognize the uncertainty of wetland mitigation; Salvesen (1990, p. 105) describes guidelines for successful mitigation, and states "wetland mitigation is still a risky business that offers few if any rules to follow and no guarantees that a created or restored wetland will perform as planned." Even EPA has a "conservative policy" on mitigation because of the scientific uncertainty associated with constructed wetlands (Cuipek 1986).

In many cases, wetland mitigation projects have failed totally, or have provided marginal wetland functions (D'Avanzo 1986). An EPA study conducted in Portland, Oregon between 1980-1986 examined eleven palustrine emergent marshes created to compensate for wetlands lost to development (Gwin and Kentula 1990). The study compared planned and existing hydrology, wetland area, wetland shape, slopes of banks, and vegetation with permit specifications and construction plans. Results indicated that none of the wetlands were designed or constructed as permitted. Gwin and Kentula estimated a cumulative loss of 1.48 ha, or 29% from the 5.10 ha that were to be created. In addition, they found that vegetation to be planted as part of the permit requirements did not occur on the sites; the number of wetland species ranged from 0% to 7% of the total number of plant species found on the created wetlands.

Despite more that a decade of experience and thousands of mitigation projects, the technology of wetland creation in the United States is largely seen as experimental (Race 1985). The uncertainty of wetland mitigation effectiveness underscores the need for scientific monitoring and evaluation of wetland restoration and creation projects. Regardless of conflicting evidence as to the success of replacement wetlands, mitigation is still seen as a viable option for developers whose construction activities impact wetlands. Given the tenuous status of our wetland resources, it is imperative that an adequate scientific research base is developed to gauge whether or not mitigated wetlands actually replace natural wetland habitats.

Functional Groups of Macroinvertebrates: Indicators of Aquatic Function

Benthic macroinvertebrates are key components of wetland ecosystems. They are an integral part of the food web; they serve as the link between algae and microorganisms (their primary food resources) and fish, wading birds, waterfowl, and other vertebrates for which they are prey (Sacco et al. 1987, Erwin 1989b). Finally, they exhibit sensitivity to watershed conditions and demonstrate stability in assemblage structure over time (Rosenberg and Resh 1993). Macroinvertebrates have been used in aquatic research because they (1) are large enough to be observed with the unaided eye (greater than 0.5mm), (2) are abundant enough to be readily collected, and (3) have lifecycles of suitable length (several weeks to 1-2 years) for short-term seasonal or annual field investigations (Cummins 1992). This study is the first to use functional feeding groups (FFGs) to evaluate the degree to which mitigated wetlands mimic the ecological function of natural wetlands in the same ecoregion; however, there is a rich literature on the use of macroinvertebrates in other aquatic ecological studies.

Benthic macroinvertebrate communities, found in association with the aquatic substrate (Wetzel 1983), have been utilized to assess environmental conditions in temperate zone lacustrine systems in studies dating from the early 1900's (Streever and Crisman 1994). Benthic macroinvertebrates have been used as indicators of habitat and water quality in stream and lake ecosystems for both short-term and long-term studies (Reid 1985, Schaeffer et al. 1985, Corkum 1989, Zedler 1990, Rosenberg and Resh 1993). Benthic organisms (amphipods, crayfish, oligochaetes) also act as bioindicators, and are responsive to major stressors within aquatic environments - excess sediment, altered hydrology, contaminants, and changes in nutrient cycling (Liebowitz et al. 1991). In addition, their presence and relative abundance can give information as to the condition of the aquatic substrate (Packard and Stiverson 1975).

Research on macroinvertebrates to a large degree has focused on descriptions of species and their distributions in lotic and lentic systems (Wetzel 1983). Comparatively few studies have focused on macroinvertebrates in inland wetland systems (Brooks and Hughes 1986, Adamus and Brandt 1990, Liebowitz et al.1991, Kentula et al.1993, Streever and Crisman 1994). Macroinvertebrate abundance and their relationship to waterfowl populations in the upper Midwest have been examined by several researchers (Krull 1970, Voights 1976, Huener 1984, Broschart and Linder 1986, Frederickson and Reid 1988). An additional focus of study has been the differences in invertebrate communities between natural wetlands and those utilized for wastewater treatment (Jones and Clark 1987). Helgen (1995) recently completed a study of 35 isolated depressional wetlands located in the North central Hardwood Forest ecoregion; she examined invertebrates, water and sediment chemistry, and land cover types and vegetation. No studies have examined the response of wetland invertebrate communities to fragmentation of regional wetland resources (Adamus and Brandt 1990).

Benthic macroinvertebrate populations have been used to evaluate stream restoration projects because of the functional role they play in stream ecosystems and their effectiveness as monitors of environmental conditions (Whiles and Wallace 1992, Richards et al. 1993). Cummins (1973) suggested that the success of any aquatic resource rehabilitation or management strategy should be based on a basic knowledge of freshwater ecosystem structure and function. The basic aspect of freshwater function is energy cycling and flow, the heart of which lies in the processing of organic matter by aquatic invertebrates such as insects and other arthropods. Food intake, tissue assimilation, and waste release are significant trophic mechanisms in aquatic systems, and form the "cement" holding biological communities together (Cummins 1973).

Functional performance, the viability of plant and animal communities within an ecosystem, has been recommended as the appropriate scientific standard for assessing the success of artificial and restored wetlands (Larson 1988, Maltby 1988). It has been suggested that ecosystem function provides greater understanding of the interaction of populations, energy cycling, nutrient exchange in a community, and hence response to perturbation (Cairns 1974). A common problem in experimental design, however, is determining the appropriate taxonomic classification for specific faunal groups (Fredrickson and Reid 1988). This issue of "taxonomic sufficiency" is a research problem in which the level of identification is balanced against the need for information. Organisms are identified to the taxonomic level necessary and sufficient to meet the study's objectives (Ellis 1985, Ferraro and Cole 1990). The effort needed to identify aquatic macroinvertebrates to genus or species is often unnecessary when the goal is to identify the functional roles of the organisms within an ecosystem (Fredrickson and Reid 1988).

Research by Cummins (1973) shows that variations in the relative dominance of functional feeding groups (FFGs) of macroinvertebrates between similar ecosystems indicate changes in community structure and function, and furnishes practical insights about that structure --even more so than typical species lists or diversity indices. Rather than classifying macroinvertebrates to taxonomic levels, the FFG classification system is based on research by Cummins (1973), which groups macroinvertebrates according to a limited set of feeding adaptations and their basic nutritional resource categories. Such grouping allows an association between organisms and food categories: (1) detritus - coarse particulate organic matter (CPOM) or fine particulate organic matter (FPOM) and the associated microbiota; (2) periphyton - attached algae and associated material; (3) live macrophytes; and (4) prey (Cummins and Wilzbach 1985).

Benthic macroinvertebrates are classified into four groups, based on mouthpart morphology and feeding behavior. Table 1 summarizes the four groups.

Few studies have examined differences in benthic macroinvertebrate populations between natural and mitigated wetlands, despite their potential to document wetland function. Streever and Crisman (1993) compared populations of meiobenthic cladocerans between natural and created wetlands in Florida. Their research found that cladoceran assemblages of some constructed wetlands mimicked those of natural wetlands, but the range of assemblages found in constructed wetlands was narrower than that found in natural wetlands. Erwin (1989b, 1991) found that invertebrate communities were partitioned differently among macrophyte communities in both a constructed wetland and a natural reference wetland in Florida. A study comparing abundance and production of macroinvertebrates from natural and artifically established salt marshes in North Carolina found no significant differences between the two systems (Sacco et al. 1987). Broschart

TABLE 1

Functional Feeding Group Classifications[1]

GENERAL CATEGORY	GENERAL PARTICLE SIZE RANGE OF FOOD (μ m)	FEEDING MECHANISM	DOMINANT FOOD
Shredder	$>10^3$	chewers & miners	vascular plant tissue
Collector	$>10^3$	Filter or suspension feeders; sediment or deposit feeders	detrital particles
Scraper	$>10^3$	mineral & organic scrapers	herbivores: algae and associated material
Predator	$<10^3$	swallowers & piercers	carnivores

[1]Adapted from Cummins (1973).

and Linder (1986) examined differences in aquatic invertebrate abundance, biomass, and diversity between natural marshes and adjacent created wetlands in South Dakota; their results indicated that a significantly greater mean number of taxa and a larger mean number of all macroinvertebrates were present in the created wetlands.

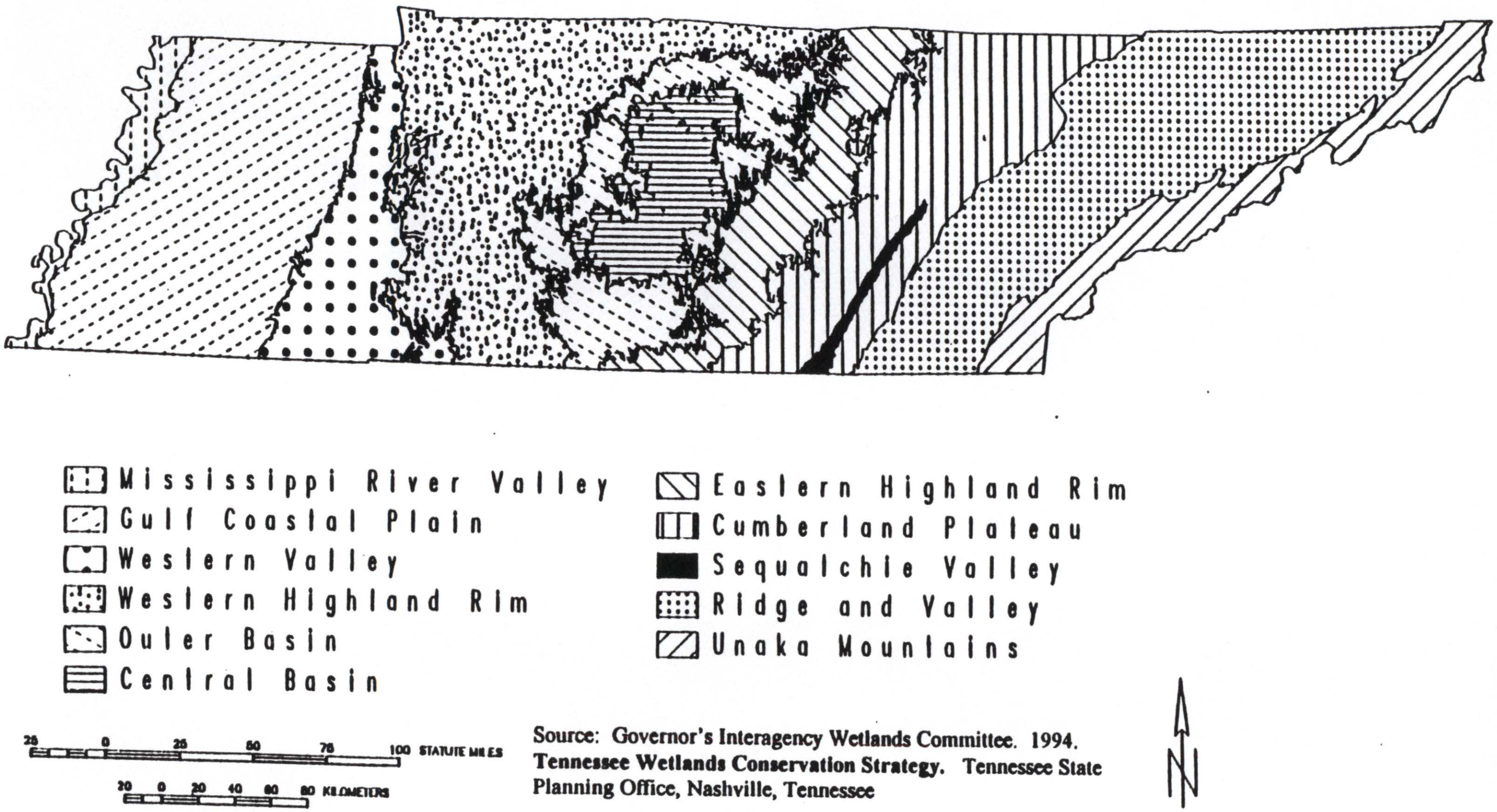
This limited number of studies points to a real need for additional comparative studies to assess the degree to which mitigated wetlands mimic natural wetlands within similar ecoregions (Adamus and Brandt 1990). Wetland mitigation is still considered a viable tool to compensate for wetlands lost to development, yet monitoring and assessment on any level is cursory at best once these wetlands are constructed. In the East Tennessee study region, customary monitoring by state and federal personnel consists of assessing the success, evaluated as 75% survival rate, of vegetation reestablishment -- no monitoring of fauna is conducted (L. Mazanti, Personal Communication 1993). The goal of this study is to provide a comparison of FFGs of benthic macroinvertebrates between three mitigated wetlands and a natural wetland in the East Tennessee Ridge and Valley ecoregion. It is hoped that this comparison can give insight about the "success" of created wetlands, in terms of replacing the ecological function of the naturally occurring wetlands that they are designed to replace.

3. STUDY SITES AND METHODS

All four sites sampled in this study are herbaceous emergent wetlands occurring in the Central Appalachian Ridge and Valley ecoregion as defined by Omernik (Figure 1) (1987). This region is bounded by the Blue Ridge province to the east and the Cumberland plateau to the west. Geologically the area is highly folded and faulted, composed of narrow lowland valleys having an altitude of 244 m to 275 m. The valleys are surrounded by long, narrow, even-topped parallel ridges extending in an northeast to southwest direction. The ridges range in elevation from 455 m to 944 m, capped by sandstone and dolomite; valleys are developed in limestone, dolomite, chert, and shale. Ridges in most cases rise to only 30 m to 92 m above the valley floors. Drained primarily by the Tennessee River, drainage patterns in the Ridge and Valley generally follow a trellised drainage pattern . Annual precipitation in this are averages 1371 mm, with approximately 45% of this total occurring in April through September. The average annual temperature is 14° C; summer temperatures average 24° C degrees, and winter temperatures 4° C.

Dominant soils in the broader valleys of the Ridge and Valley range from moderately deep to shallow over soft shale and clayey limestone. Subsoils are generally clayey or shaly with moderate to slow permeability. Limestone sinks are common.

Wetlands in the Ridge and Valley area have received little study (Koryak 1982, Smith and Michael 1982, Governor's Interagency Wetlands Committee 1994), but in general differ



Source: Governor's Interagency Wetlands Committee. 1994. **Tennessee Wetlands Conservation Strategy.** Tennessee State Planning Office, Nashville, Tennessee

Figure 1
Physiographic Regions of Tennessee

markedly from wetlands in the Mississippi Alluvial Valley and the Gulf Coastal Plain. In the Ridge and Valley ecoregion, wetlands are generally smaller, less abundant, located farther from one another, and have narrower riparian zones (Koryak 1982, Smith and Michael 1982, Pearson 1994). This is in large part due to the geologic history of the area; wetlands are generally found where drainage patterns are poorly defined and geologically immature. In the Ridge and Valley, drainage patterns are well-developed and low-lying poorly drained areas that favor wetland development are rare (Hack 1969).

Site Descriptions

Research sites (Figure 2) were selected through interviews with personnel from the COE, Tennessee Department of Transportation (TDOT), the Tennessee Valley Authority (TVA), and the EPA, based on permits issued by their respective agencies. The natural wetland site was selected based on examination of USGS quadrangle maps, NWI maps, interviews with TDEC and TWRA personnel, and site visits. Directions to sites are provided in Appendix C. Four natural wetlands were visited. Three sites were deemed unsuitable, based on criteria set forth by Brooks and Hughes (1986). See Appendix A for a list of sites considered and their locations. Selection criteria included large areas of permanent vegetation or accumulated organic matter, heterogeneity in vegetative structure and water width and depth, abundance of large woody plants or aquatic vascular macrophytes, apparent abundance of vertebrates, and little or no anthropogenic disturbance. The three sites that were rejected were located adjacent to large industrial

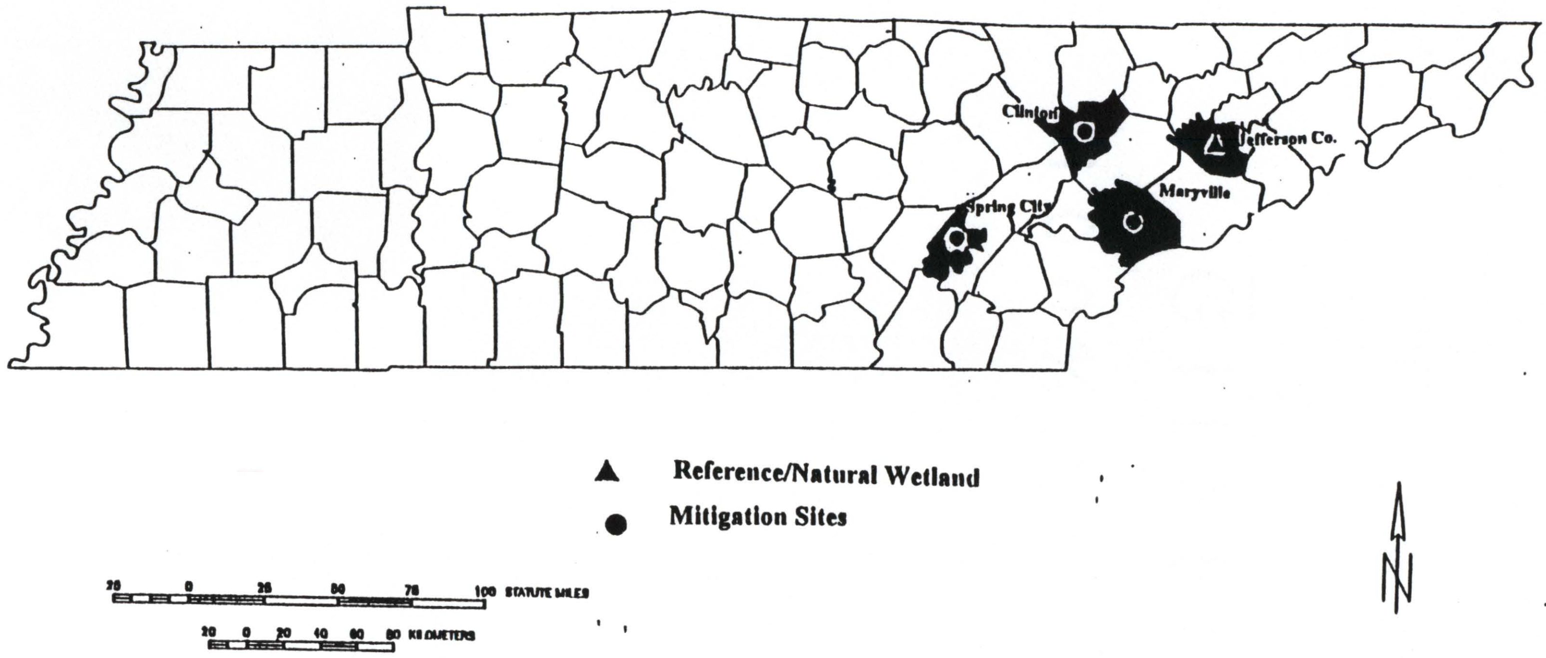


Figure 2
Location of Study Sites

areas. Two of those sites had a documented history of toxic waste contamination, specifically polychlorinated biphenyls.

Mitigated wetland size ranged from approximately 0.8 to 1.4 ha, while the natural wetland was approximately 7.2 ha. Based on field reconnaissance, vegetation in mitigated wetlands consisted of mixed herbaceous wetlands species, including *Typha latifolia, Eleocharis spp., Scirpus* spp., *Polygonum spp.*, and *Juncus effusus* (plant names follow Reed 1988). Vegetation in the natural wetland consisted primarily of *Polygonum spp., Impatiens capensis, Juncus effusus*, and *Typha latifolia,* with the latter clearly dominating areal coverage. Appendix B gives directions to the study sites.

Natural Wetland - Jefferson County, Tennessee

The natural reference wetland (36° 05'25" N, 83° 37" W; 247 m elevation; Figure 3) is located in Jefferson County, Tennessee. It is primarily a meandering drainway associated with a first order stream. The soils in this area are described as undifferentiated alluvial soils (Moon et al. 1935). The wetland is located in a floodplain area of Blue Spring Branch, bordered on the North and west sides by a small-scale cattle operation. The south side of the wetland area is a wooded area with some agricultural usage. The east perimeter of the wetland is bordered by a two-lane road, Eslinger Road. The surrounding land uses have been historically agricultural.

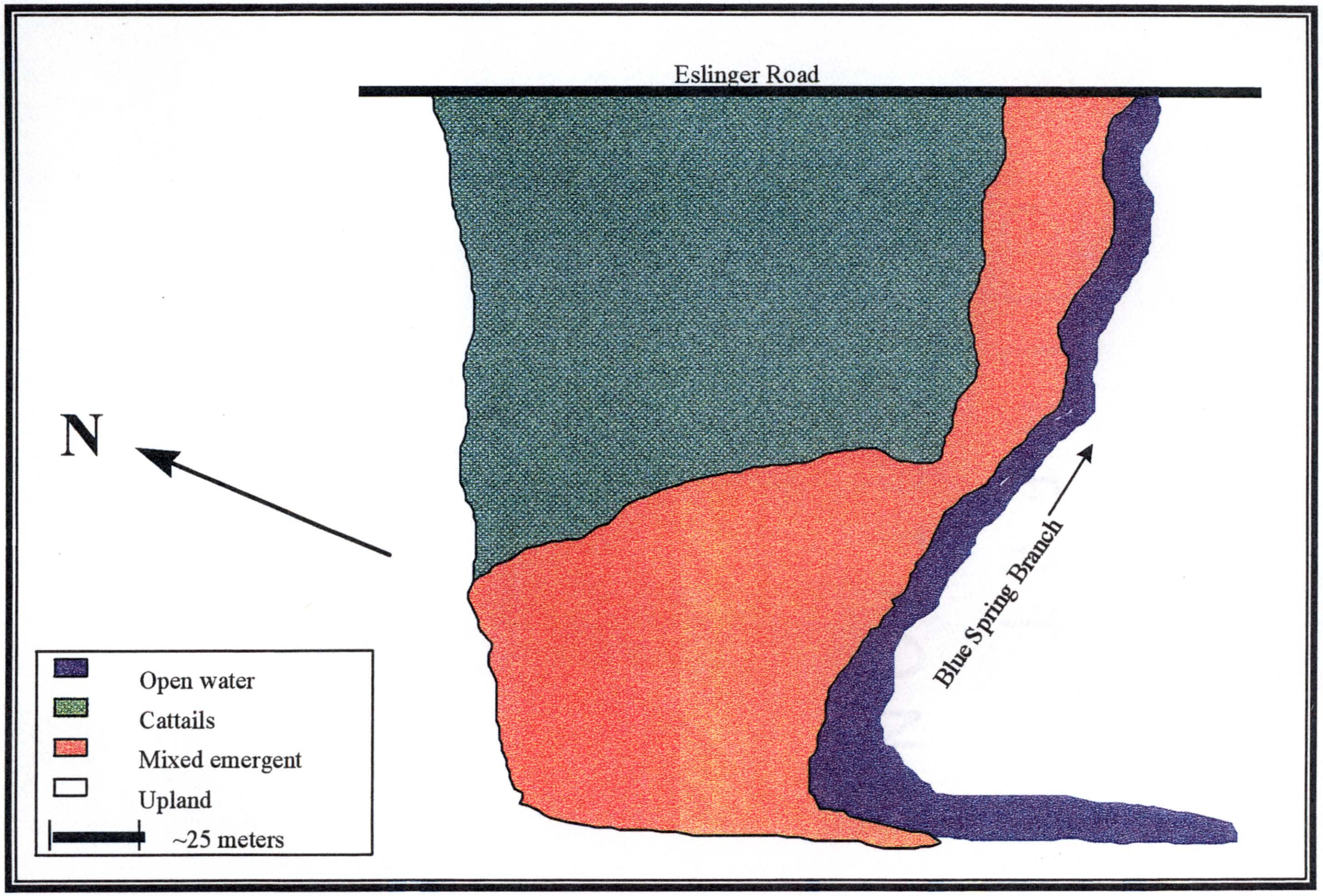


Figure 3
Natural Wetland Site Diagram, Jefferson County, Tennessee

Discussion with the owner of the marsh, Mr. B. Eslinger, indicated that the wetland developed after his grandfather stopped clearing the area in the early 1900s. He has been told by Soil Conservation Service personnel that the wetland was the largest in Jefferson County, and that it is also considered a jurisdictional wetland subject to Section 404 regulation (B. Eslinger, personal communication).

The actual wetland area is approximately 7.2 hectares. Blue Spring Branch spreads out and runs through the marsh area. Figure 3 depicts the major plant communities. The east edge of the marsh is a red maple swamp. Dominants are red maple (*Acer rubrum*) and black willow (*Salix nigra*) with an understory of grasses, smartweed (*Polygonum spp.)*, rush (*Juncus effusus*), jewelweed (*Impatiens capensis*). Moving east, the vegetation changes into an open carr of small woody plants (30 percent cover) of black willow (*Salix nigra*) interspersed with cattail (*Typha latifolia*). The remainder of the wetland area, approximately 1.6 ha, is open marshland including a few patches of open water. The dominant vegetation is cattail (*Typha latifolia*). The west edge of the marsh is composed of a mix of smartweed (*Polygonum spp*) and cattail (*Typha latifolia*), with a few patches of open water. In the patches of open water are aquatic water milfoil (*Myriophyllum sp.*) and watercress (*Nasturtium officinale*).

Wildlife viewed during site visits include various species of turtles and snakes, black crowned night herons, red wing black birds, redtail hawk, and several unidentified species of frogs and toads. In addition, whitetail deer tracks were seen along the edges of the marsh.

Mitigated Wetland - Clinton, Tennessee

The Clinton wetland (36° 05' 25" N, 84° 06" W; 246 m elevation; Figure 4) is located in Clinton, Tennessee adjacent to the Clinch River. Prior to development, there was one large (approx. 1.4 ha) wetland area and two smaller wetlands along a confined drainage ditch that connected to the larger wetland on both ends. Historical land use in the area was primarily livestock grazing; the wetland area itself lies on TVA property.

Soils in the wetland areas are described as Staser loam (Moneymaker 1981). Staser loam is found in bottomland areas, located mainly along the Clinch River. Tennessee Valley Authority biologists classified a 1.2 ha portion of the wetland as a palustrine emergent wetland (Cowardin et al. 1979). The remaining 0.2 ha was described as a palustrine scrub/shrub wetland (Cowardin et al. 1979). The smaller wetlands were classified as a palustrine forested wetland and a palustrine aquatic bed wetland (Cowardin et al. 1979); together these two wetlands comprised an area of approximately 0.2 ha. A site description completed in 1978 by TVA personnel indicated that very few wetlands of this type exist along the Clinch River and the wetland should be considered unique for the area. The site was considered important to wetlands wildlife in the area and of special importance to migrant-wintering waterfowl. Large numbers of migrant waterfowl were

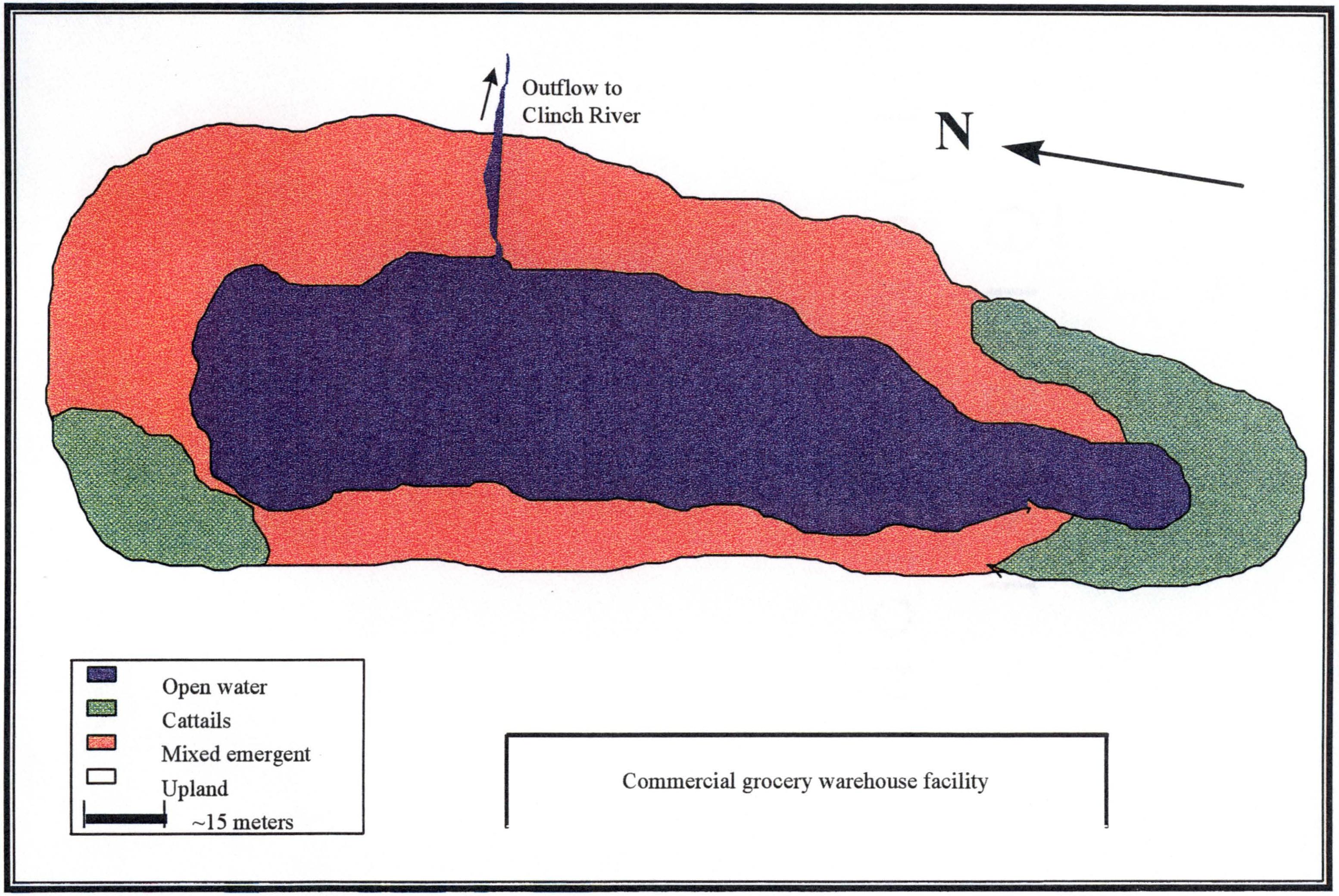


Figure 4
Mitigated Wetland Site Diagram, Clinton, Tennessee

known to frequent the wetland during the fall and winter. The site was also deemed important to resident Canada geese, wading birds, and migrating shorebirds.

According to TVA documents (internal memos and letters) the city of Clinton acquired a large portion of the agricultural land in the Eagle Bend area of the Clinch River for industrial park development. Land immediately adjacent to the wetlands was purchased by a large grocery chain for development of a distribution center in 1988. TVA recognized that the existing wetlands could be directly or indirectly impacted by construction activities on the adjacent tract, and thus sought to quantify any adverse wetland impacts that might occur to the area.

Construction plans were evaluated, and it was determined that to offset the loss of the smaller 0.2 ha sites, a 1 m berm would be constructed at the downstream end of the larger wetlands area. It was felt that this would increase the depth of water over the existing wetland while inundating additional areas, thus minimizing construction costs while creating a higher quality wetland. An upstream berm was also constructed just downstream from the originally proposed detention pond. This berm would negate the need for a stormwater detention facility as the main site storm drain (121 cm diameter) would no longer outfall into the wetlands area. Rather, the proposed upstream berm would divert the flow directly to the Melton Hill Reservoir through a proposed swale. This revision also negated the need for the 152.4 m of 61 cm storm sewer. This swale would have a minimum elevation of 247 m in order to maintain the existing wetlands area east of the truck shop. The source of water for the mitigated wetlands area would be the stormwater runoff from the roof drains. In addition to construction activities to enhance mitigation, TVA documents recommended that various types and quantities of wetlands plants and/or trees be planted in and along the proposed wetlands area. Construction activities were completed in 1988.

Site visits in 1992 and 1993 indicated that the wetland site had developed in large part into a pond, with smaller areas of wetland on the fringes of the deeper areas (Figure 4). At the shallow south end of the wetland, dominant vegetation included cattail (*Typha latifolia*), smartweed (*Polygonum spp.*), and rush (*Juncus effusus*). Open water areas are interspersed with American lotus (*Nelumbo lutea).* The north end of the wetland is more heavily vegetated, with an area of red maples (*Acer rubrum*) approx. 10 m in height, that appear to be dead or dying. There is a lesser understory of black willow (*Salix nigra*), and an herbaceous area dominated by cattail (*Typha latifolia*) and rush (*Juncus effusus*).

Wildlife viewed included waterfowl and wading birds such as nesting Canada geese, great blue herons, and black crowned night herons; other birds seen in the wetland area included red winged blackbird, and red-tailed hawk. Deer and raccoon tracks were frequently observed.

Mitigated Wetland - Maryville, Tennessee

Prior to mitigation, the Maryville wetland (36° 05' 25" N, 84° 06" W; 292 m elevation; Figure 5) was an upland field area approximately 0.9 ha in size adjacent to Laurel Bank Branch. The area is surrounded on two sides by agricultural usage (primarily grazing) and suburban housing developments. The wetland is within the Maryville city limits. Soil survey maps describe the existing soils in the area as Prader silt loam, a poorly drained soil of the bottomlands (Elder et al., 1953).

The site was chosen to mitigate the loss of approximately 0.8 ha of bottomland hardwood wetlands adjacent to Laurel Bank Branch in order to develop a site for Nippondenso Company, LTD. Approximately 2405 cubic meters of on-site material were placed in the wetland area to increase the existing elevation approximately 8.5 m. The toe of the fill was placed approximately 3 m from Laurel Bank Branch. Site development was for a power switchyard and a truck access road for the company's manufacturing plant.

Mitigation plans called for excavation of 0.9 ha to a bottom elevation suitable for creation of an emergent marsh/forested wetland. This elevation was described as approximately .3 m above the existing stream bed elevation of Laurel Bank Branch where it enters the property on the southern boundary. Small canals were excavated between Laurel Bank Branch and the created wetland in order to provide a water source for the wetland. A 7.6 m buffer between the Laurel Bank Branch and the created wetland was constructed,

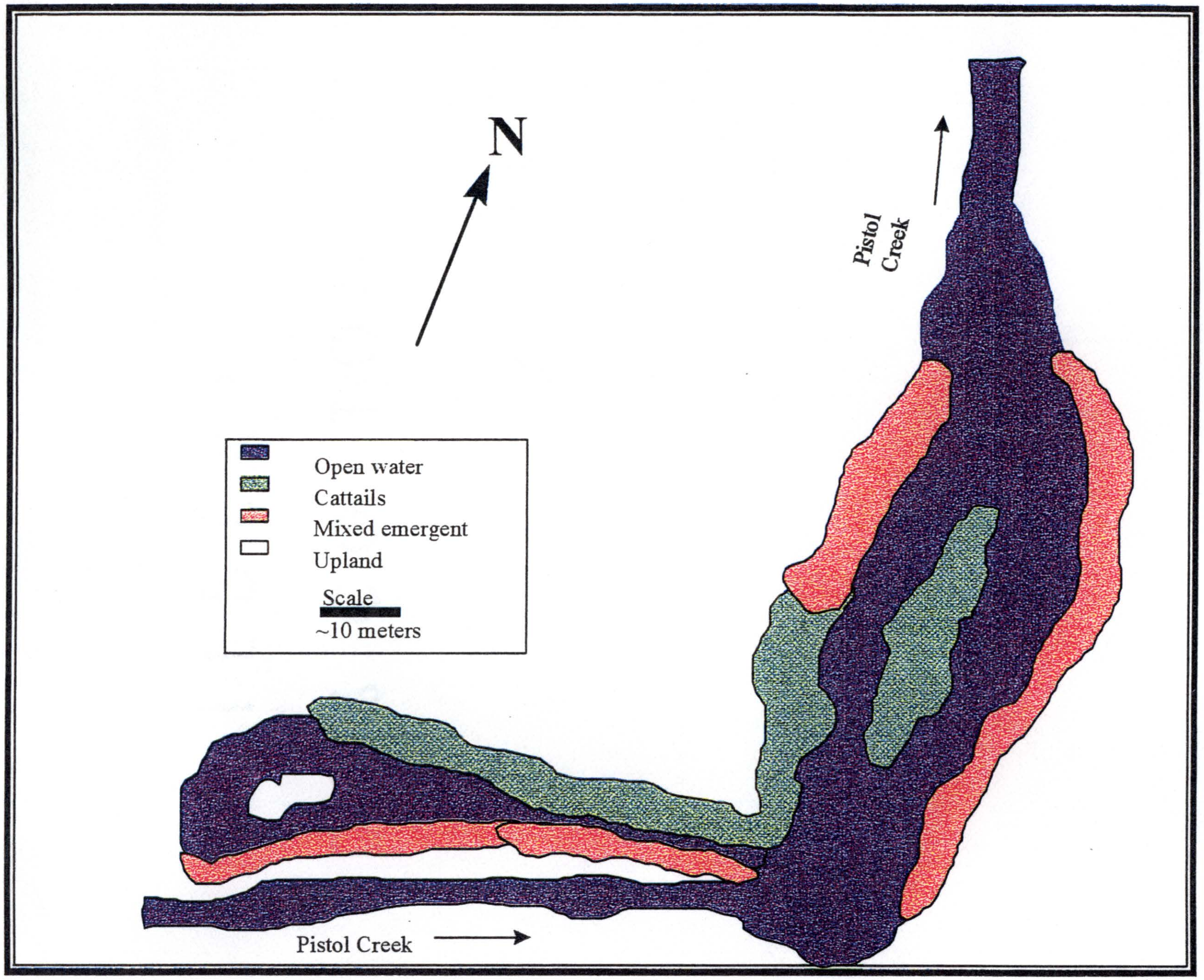


Figure 5
Mitigated Wetland Site Diagram, Maryville, Tennessee

except where tie-ins (channels) were constructed. The created wetland was to be seeded and/or sprigged by utilizing cuttings from vegetation in the existing wetlands adjacent to the site. Selected trees, designated by the COE, were to remain undisturbed in the created wetland, and grading activities were limited to within 3 m of the trees to insure root systems were not damaged. Mitigation activities were completed in 1988.

Site visits in 1992 and 1993 revealed the establishment of some herbaceous wetland vegetation, along with woody species such as buttonbush (*Cephalanthus occidentalis*), black willow (*Salix nigra*) and red maple (*Acer rubrum*) mixed in with upland species (Figure 5). Dominant herbaceous species include cattail (*Typha latifolia*) rice cutgrass (*Leersia oryzoides)*, spikerush (*Eleocharis spp.)* and rush (*Juncus effusus*). Large areas of open water exist. Wildlife species observed included wood ducks, black crowned night heron, muskrat, and various species of snakes and turtles. Upstream from the wetland area, Laurel Bank Branch has been channelized into a straight open channel, with the existing riparian shrub vegetation removed from the banks. This was done by the owner of the cattle field, to allow the cattle easy access to the creek. On every site visit 5-10 cattle were observed in the stream.

Mitigated Wetland - Spring City, Tennessee

The Spring City wetland (35° 45’ 45” N, 84° 00” W; 225 m elevation; Figure 6) is a small (0.8 ha) mitigated wetland located in Spring City, Tennessee,. Prior to mitigation, the site was an upland open field area and small riparian zone adjacent to Town Creek. The area

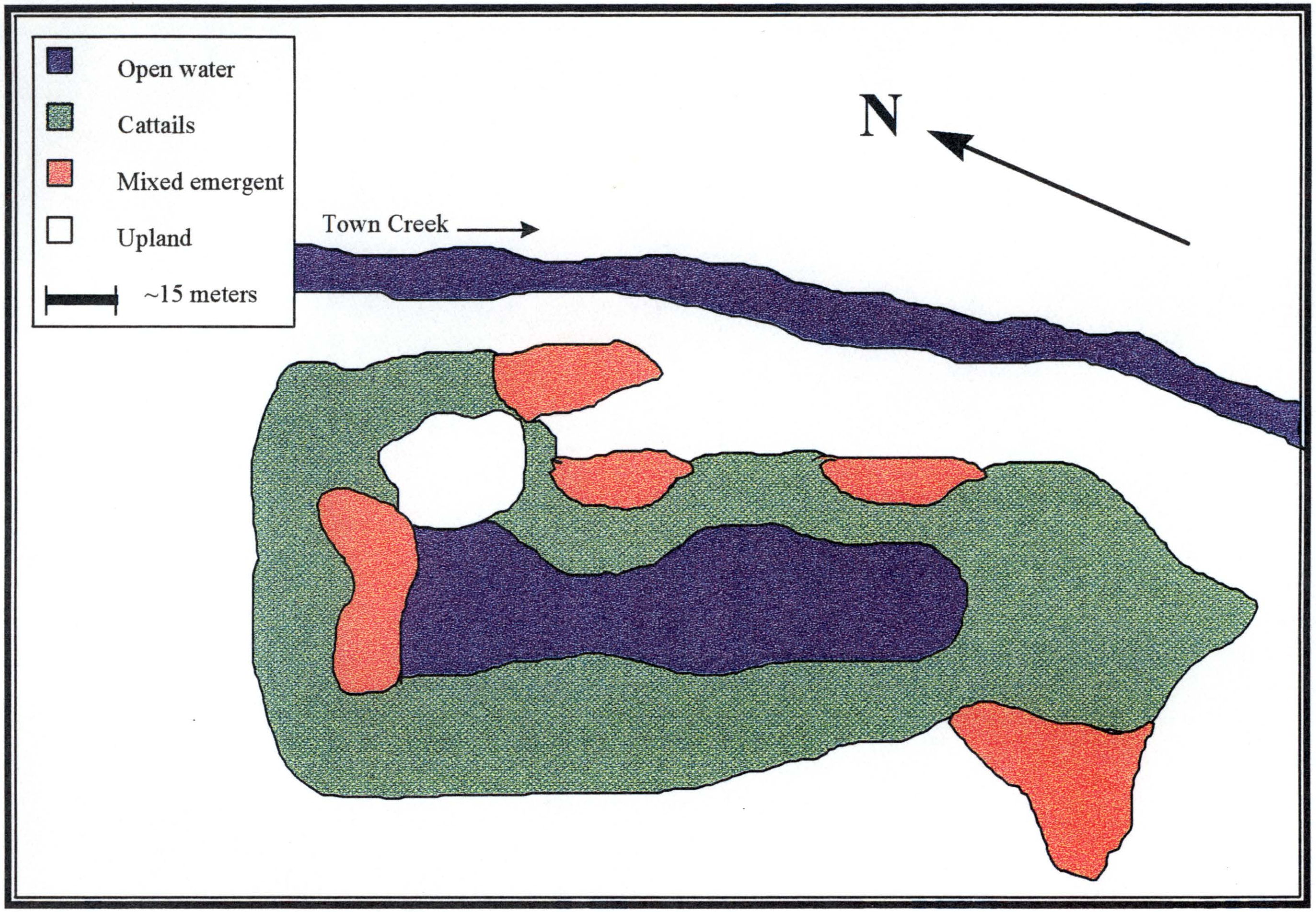


Figure 6
Mitigated Wetland Site Diagram, Spring City, Tennessee

is bordered on three sides by roads; a gas station/convenience store and auto parts store are located within 30 m of the wetland. The remaining side is a fenced open field area. The wetland is within the Spring City limits, and is essentially surrounded by urban/suburban land uses. Soil survey maps describe the existing soils in the area as Taft silt loam, an imperfectly drained light-colored soil found on terraces (Hasty et al. 1940).

The wetland was constructed in 1990 to mitigate the loss of approximately 0.8 ha of forested wetlands along Town Creek. According to documents provided to me by the Environmental Protection Agency, the previously existing wetlands were filled by the TDOT, in violation of Section 404 of the CWA. Construction of a highway ramp for State Road 68 at the intersection of State Road 69 in the floodplain of Town Creek exacerbated flooding downstream in Spring City, Tennessee. Channel improvements, including deepening and widening of the creek and the construction of concrete liners along the stream banks, and filling of the adjacent wetlands were construction activities designed by TDOT to alleviate flood damage, however these activities were cited by the EPA as violating CWA regulations. Tennessee Department of Transportation was ordered by the EPA to restore the normal flow and habitat function of Town Creek and to offset the loss of the wetlands by restoration of an adjacent 0.8 ha of wetlands.

Based on site plans provided by TDOT, stream restoration consisted of the removal of the concrete liners and the placement of gabion-basket control structures planted with wetland tree species to control in-stream flow and enhance habitat. The south bank was graded to an elevation suitable for the development of wetland conditions and to serve as a flood-flow channel. Wetland restoration consisted of the grading of the north floodplain to an elevation consistent with that of identified adjacent wetlands, and the reforestation of the entire area with 450 wetland trees per acre.

Documentation of site activities indicate that fill material previously placed in the wetland north of the ramp was removed to original ground level. Prior to grading, however, the area was found to contain a considerable amount of rock scattered through it. Mitigation plans were changed to include additional excavation activities, and it was recognized that additional excavation to remove rock material would leave some shallow depressions (less than 0.3 m). Tree species (seedlings approximately 30 cm high) planted in the mitigation site included black willow (*Salix nigra*), box elder (*Acer negundo* L.), sweetgum (*Nyssa sylvatica*), green ash (*Fraxinus pennsylvanica* var. *subintegerrima)*, and red maple (*Acer rubrum*). Trees were planted on 3 meter centers (based on the use of seedlings or saplings) at a ratio of 450 trees per 0.4 ha. Mitigation was completed in November 1990. In late 1991, the area had been mowed by local residents, who complained that the site was harboring rats and snakes. According to TDOT records, trees were to be recounted and replanted as necessary.

Though restoration plans included a monitoring program, my conversations with EPA personnel indicate that this monitoring has been very inconsistent (L. Mazanti, personal communication). Environmental Protection Agency documents indicate that a monitoring program including vegetative monitoring, hydrologic monitoring, and in-stream habitat recovery assessment should occur to ensure that mitigation requirements were being met on a long-term basis. The restoration plan described vegetative monitoring as the measurement of the growth and density of the planted trees and any volunteer species on a quarterly basis until success (75% survival) is demonstrated for five consecutive years. Vegetative monitoring did not, however, include herbaceous species. Hydrologic monitoring included the installation of three piezometers and one crest gage throughout the mitigation site and the monitoring of basic water quality parameters, such as temperature, dissolved oxygen, pH, conductivity and suspended solids, on a quarterly basis. In-stream habitat recovery was to be assessed by sampling the benthic macroinvertebrates in the stream, using a quantitative method, and comparing the community diversity, density and abundance to an unimpaired upstream community. According to EPA personnel, however, this monitoring has not yet occurred.

Site visits during 1993 indicated good establishment of herbaceous wetland vegetation (Figure 6). Dominant species include cattail (*Typha latifiolia)*, and rush (*Juncus effusus*). The actual wetland area, that with standing water and saturated soil, does not cover the entire area designated as that to be mitigated. The actual wetland area is approximately half of the area, the remainder is primarily upland field.

Wildlife usage of the site is evident. During site visits, whitetail deer tracks were visible, and several types of wetland bird species, including redwing black birds and black crowned night herons were present. I also saw killdeer, woodcock, as well as an abundance of tadpoles and flying aquatic insects such as dragon flies and damselflies.

Methods

Samples were collected during a two week period during April and May of 1993. I chose this time period based studies of aquatic macroinvertebrates by Plafkin et al. (1989), whose work indicates that density and species richness tend to be the richest when sampling is conducted in late Spring. Each wetland was sampled once. This sampling method is based on the assumption that differences in community composition and functional feeding groups of benthic macroinvertebrates reflect real differences among sites, and not seasonal trends (Corkum 1989).

Following Plafkin et al (1989) I chose a multihabitat sampling approach. The four sites were stratified by cover (vegetation type) based on the dominant emergent plants (Ross and Murkin 1989); representative habitats common to all sites were then chosen. Three common habitats were determined: 1) open water area (standing water with <10% vegetated area); 2) areas of *Typha latifolia*; and 3) mixed emergent vegetation < 1m in height. Samples were collected along transects oriented across/perpendicular to the hydrologic gradient of the wetland (Liebowitz et al. 1991). Five sampling stations were randomly selected in each representative habitat, using a random number table in conjunction with compass headings. An Ekman dredge was used to collect benthic macroinvertebrates at each sampling site (Krull 1970, Driver 1977). The body of the

Ekman dredge consists of a square or rectangular box of stainless steel 25 cm on each side. The lower opening is closed by a pair of strong metal jaws that oppose each other and are closed tightly by springs. When fully pulled apart, they leave the whole bottom of the box open. Two strong external springs, when released by a "messenger" mechanism, snap the jaws shut. A spring mechanism at the top of the sampler provides a means of releasing the jaws. To gain an additional measure of control, and to effectively push the dredge through aquatic vegetation, the dredge was mounted on a pole using swivel bolts fitted with wing nuts; the sampler was then positioned and thrust down into the substrate to a depth of 15 cm. Samples were transferred immediately to a standard white enamel sorting pan. Samples were sieved in the field with ambient water to remove coarse materials and placed in 1 liter containers. Samples were returned to the laboratory and sieved through a standard no. 70 sieve (210-μm mesh) and preserved in 80% ethanol.

Organic material, soil clumps, and small rocks frequently contaminated samples. To separate the benthic macroinvertebrates from contaminants, samples were picked under 20X magnification in the laboratory. Picking times ranged between 1 and 6 hours per sample, depending on the amount of extraneous material. All benthic macroinvertebrates were classified into functional feeding group following the scheme summarized by Cummins and Wilzbach (1985), Table 1, Chapter 2 of this thesis. Samples from the individual habitats were preserved and counted separately (Plafkin et al. 1989); after the data were analyzed from individual habitats, it was aggregated for large scale comparisons between the wetland systems.

Inter-wetland and intra-wetland differences between the invertebrate samples were analyzed using chi square analysis (Sokal and Rohlf 1981), comparing expected values from the reference site to the observed values from the mitigation sites. A level of significance of 5% was selected. To determine where differences within categories (vegetation zones) existed, a standard residual was calculated (Sokal and Rohlf 1981). If the standard residual fell between -2 and 2, it was assumed that there was no significant difference between the two categories.

4. RESULTS

Density of Benthic Macroinvertebrates

As shown in Table 2, a total of 3,582 benthic macroinvertebrates were collected from the four sites. The natural wetland had the highest number of organisms collected, 1328. In the mitigated sites the total number of organisms ranged from 1295 (Maryville) to 337 (Clinton).

Differences in FFGs Between Natural and Mitigated Sites

Comparative proportions of shredders, scrapers, collectors and predators at the four sites are summarized in Table 3 and depicted graphically in Figure 7. In the natural wetland, shredders were the dominant FFG, comprising 61% of the organisms collected. The dominant shredder organisms were isopods and amphipods. Collectors constituted 18% of the sample, and scrapers, primarily gastropods, made up 17% of the total. Predators composed 4% of the total number of benthic macroinvertebrates collected. In the Maryville site, collectors were the dominant FFG, composing 71% of the total number of benthic macroinvertebrates collected.

Table 2

Total numbers of organisms within functional feeding groups collected from mitigated and natural wetlands

	Reference wetland	Clinton	Maryville	Spring City	Total
Functional Group					
shredder	811	15	19	4	849
scraper	223	8	247	364	842
collector	238	254	917	174	1583
predator	56	60	112	80	308
TOTAL	1328	337	1295	622	3582

Table 3

Comparative ratios of FFGs collected from reference wetland and four mitigated sites.

SITE	SHREDDERS (% OF TOTAL)	SCRAPERS (% OF TOTAL)	COLLECTORS (% OF TOTAL)	PREDATORS (% OF TOTAL)
reference wetland	61	17	18	4
Clinton	5	2	75	18
Maryville	<1	19	71	9
Spring City	<1	59	28	13

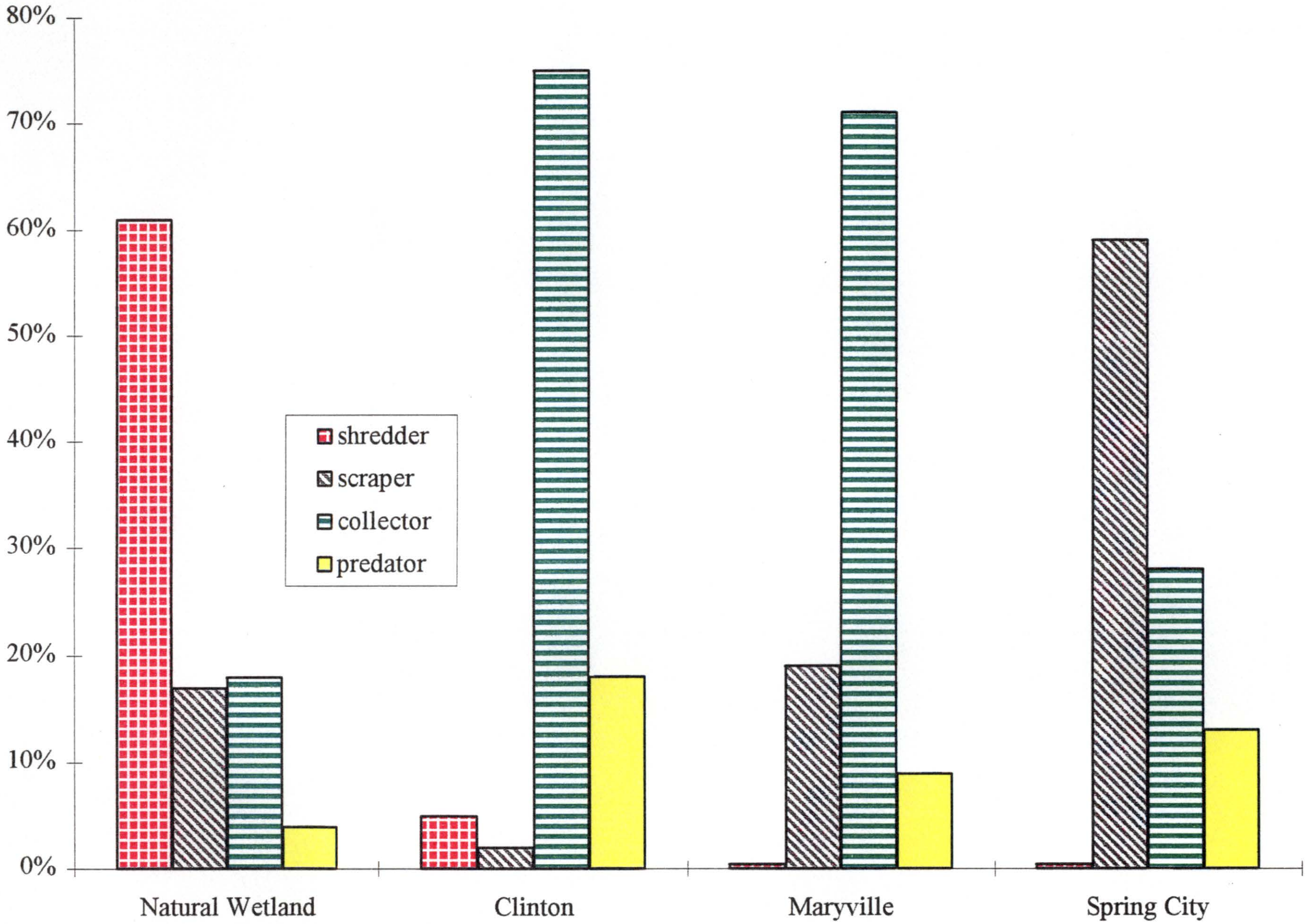


Figure 7
Comparative Ratios of Functional Feeding Groups

Collectors were primarily midge flies (chironomids). Scrapers were the next most important group (19%). Predators composed 9% of the sample, and shredders made up less than 1% at the Maryville site. Collectors (primarily chironomids) were also the most dominant FFG at the Clinton site, comprising 75% of the sample. Shredders comprised 5% of the sample, and predators comprised 18% of the total number of individuals collected. Scrapers comprised 2% of the total. Results from the Spring City site indicate that scrapers (principally gastropods) were the most dominant FFG, composing 59% of the sample. Collectors were the second most common group at 28%. Predators composed 13%, and shredders made up less than 1% of the sample.

Oligochaetes (aquatic worms) and chironomids were the most abundant organisms collected from the mitigated wetlands. Due to the large numbers of chironomids represented in the samples, 10% of the count of collectors was subtracted and added to the count for predators, to account for different food preferences among chironomids. (Cummins and Wilzbach 1985). Data included in the tables reflects this adjustment.

Inter-habitat Variation in FFG

Distribution of FFGs varied by habitat type within wetlands. In the natural reference wetland, (Table 4) over one-half (752) of the total organisms were found in the mixed emergent zone (strata 3). Separating the FFGs out, 65% of the shredders were found in the mixed emergent zone strata. Scrapers preferred the open water zone (52%). Collector organisms were found in high proportion (53% of total collectors) in the mixed

Table 4

Number of benthic macroinvertebrates collected in three habitat zones in natural wetland, Jefferson County, Tennessee

	Typha latifolia. zone	Open water zone	Mixed emergent zone	TOTAL
Functional Group				
shredder	152	130	529	811
scraper	46	116	61	223
collector	51	60	127	238
predator	12	9	35	56
TOTAL	261	315	752	1328

emergent zone. Sixty-two percent of predators were also found in the mixed emergent zone in the natural wetland.

Table 5 summarizes data from the Clinton site. The majority of total organisms (146/337) were collected within the *Typha latifolia* (cattail) zone. Shredders were found primarily in the mixed emergent zone (93%). Scrapers preferred the cattail zone (75%). Collectors were more evenly distributed; 41% were found in the cattail zone, 37% in the open -water zone, and 20% in the mixed emergent zone. Fifty-five percent of the predators were found in the cattail zone.

In the Maryville site (Table 6), most of the organisms were collected from the open water zone. Unlike the previous two sites, however, there is not a wide variation between the

Table 5

Number of benthic macroinvertebrates collected in three habitat zones in mitigated wetland, Clinton, Tennessee

	Typha latifolia zone	Open water zone	Mixed emergent zone	TOTAL
Functional Group				
shredder	1	0	14	15
scraper	6	1	1	8
collector	106	95	53	254
predator	33	19	8	60
TOTAL	146	115	76	337

Table 6

Number of benthic macroinvertebrates collected in three habitat zones in mitigated wetland, Maryville, Tennessee

	Typha latifolia zone	Open water zone	Mixed emergent zone	TOTAL
Functional Group				
shredder	11	2	6	19
scraper	85	81	81	247
collector	252	446	219	917
predator	26	60	26	112
TOTAL	374	589	332	1295

three strata. Forty-five percent were collected from the open- water zone, 28% from the cattail zone, and 25% from the mixed emergent zone. Fifty-seven percent of the shredders were found in the cattail zone; scrapers were almost evenly distributed across the three habitat types. Forty-seven percent of the collectors were found in the open-water zone. Predators were found most often in the open-water zone (53%).

Like the Maryville site, most of the organisms collected in the Spring City site (Table 7) were from the open-water zone. The two sites are also similar in that there is not a wide variation between the three strata; 45% of the organisms were found in the open-water zone, 21% in the cattail zone, and 32% in the mixed emergent zone. Only four shredders were found; three of the four were collected in the cattail zone. Scrapers were almost evenly distributed between the three habitat types. Sixty-five percent of the collectors were found in the open-water zone. This zone also contained the most predators, with 58% of the total.

Table 7

Number of benthic macroinvertebrates collected in three habitat zones in mitigated wetland, Spring City, Tennessee

	Typha latifolia zone	Open water zone	Mixed emergent zone	TOTAL
Functional Group				
shredder	3	1	0	4
scraper	104	124	136	364
collector	20	114	40	174
predator	9	47	24	80
TOTAL	136	286	200	622

Chi-square Analysis

Chi-square values were calculated based on proportions of the FFGs. Chi-square tests indicate that the ratios of FFGs in the mitigated wetlands differ significantly from that of the natural wetland (Table 8). To determine where similarities within groups exist, a standard residual (SR) was calculated for each category (Sokal and Rohlf 1981); if the SR was between 2 and -2, no significant difference existed between that particular FFG ratio and that found in the natural wetland. As shown in Table 8, there are some similarities between the mitigated wetlands and the natural wetland with regards to ratios of particular FFGs. In the Clinton site, there were no similarities. In the Spring City site, the percentage of collectors was similar to that of the reference wetland. In the Maryville site, the percentages of scrapers and predators were similar.

Further analysis was done to compare similarities between the mitigated and reference sites with regards to FFGs found in specific vegetation zones in the wetlands. Comparing proportions of FFGs found in the *Typha latifolia* zone (Table 9), chi square tests indicate significant differences in the proportions of FFGs collected from the mitigated and reference wetland. Some similarities did exist, however, between the proportion of shredders found in the Clinton site and the reference wetland. The Spring City site was similar in proportions of collectors and predators collected from the *Typha* zone. The proportions of scrapers and predators from the Maryville site also were similar to those found in the reference wetland.

Chi-square analysis of results from the open water zone also show significant difference with regards to proportions of FFGs (Table 10). Some similarities exist, however. In the open water zone, the proportion of scrapers found in the Spring City site and the reference wetland is similar. The proportion of predators collected from the Maryville site in the open water zone is also similar to the proportion collected from the open water zone in the natural wetland. Marked differences were found in the mixed emergent zone (Table 11). Though overall proportions of FFGs differed in this habitat zone between the mitigated and reference wetland, proportions of predators from both the Clinton site and Maryville site corresponded to those in the reference wetland. Proportions of scrapers were also similar in the Maryville wetland.

Table 8. Chi- square analysis of functional feeding groups of benthic macroinvertebrates (presented as percent of total) collected from a natural reference wetland and mitigated wetland sites in east Tennessee.

OVERALL CHI-SQUARE ANALYSIS

FFG	natural wetland	Clinton	chi-square	standard residual	sig.dif?	natural wetland	Maryville	chi-square	standard residual	sig.dif?
shredder	61	5	51.4	7.1	yes	61	<1	3600	600	yes
scraper	17	2	13.2	3.6	yes	17	19	0.23	0.47	no
collector	18	75	43.3	6.5	yes	18	71	39.	6.2	yes
predator	4	18	49	7	yes	4	9	2.7	1.6	no
			156.9					3642.4		
$\chi^2=156.9$ df=3 $p<.001$						$\chi^2=3642.4$ df=3 $p<.001$				

	natural wetland	Spring City	chi-square	standard residual	sig.dif?
shredder	61	<1	3600	60	yes
scraper	17	59	103.7	10.1	yes
collector	18	28	0.76	0.81	no
predator	4	13	6.2	2.4	yes
			3710.6		
$\chi^2=3710.6$ df=3 $p<.001$					

Table 9. Chi -square analysis of functional feeding groups of benthic macroinvertebrates (presented as percent of total) collected from *Typha latifolia* vegetation zone in a natural reference wetland and mitigated wetland sites in east Tennessee.

CHI- SQUARE ANALYSIS											
FFG	**natural wetland**	**Clinton**	**chi-square**	**standard residual**	**sig.dif?**		**natural wetland**	**Maryville**	**chi-square**	**standard residual**	**sig.dif?**
shredder	58	<1	0.84	0.9	no	shredder	58	3	1008.3	31.7	yes
scraper	18	4	49	7	yes	scraper	18	23	1.0	1.03	no
collector	20	73	38.4	6.1	yes	collector	20	67	32.9	5.7	yes
predator	4	22	302	17.3	yes	predator	4	7	1.2	1.1	no
			390.2						1043.5		
$\chi^2=390.2$ df=3 p<.001						$\chi^2=1043.5$ df=3 p<.001					
	natural wetland	**Spring City**	**chi - square**	**standard residual**	**sig.dif?**						
shredder	58	2	1568	39.5	yes						
scraper	18	76	44.2	6.6	yes						
collector	20	15	1.6	1.2	no						
predator	4	7	1.2	1.1	no						
			1615								
$\chi^2=1615$ df=3 p<.001											

Table 10. Chi square analysis of functional feeding groups of benthic macroinvertebrates (presented as percent of total) collected from open water zone in a natural reference wetland and mitigated wetland sites in east Tennessee.

CHI SQUARE ANALYSIS											
FFG	**natural wetland**	**Clinton**	**chi square**	**standard residual**	**sig.dif?**		**natural wetland**	**Maryville**	**chi square**	**standard residual**	**sig.dif?**
shredder	41	<1	3280.5	57.2	yes	shredder	41	<1	3280.5	57.27564	yes
scraper	37	<1	2664.5	51.6	yes	scraper	37	14	37.78	6.146543	yes
collector	19	83	49.3	7.0	yes	collector	19	76	42.75	6.538348	yes
predator	3	17	11.5	3.3	yes	predator	3	10	4.9	2.213594	no
			6005.								
χ^2=6005 df=3 p<.001						χ^2=6005 df=3 p<.001					
	natural wetland	**Spring City**	**chi square**	**standard residual**	**sig.dif?**						
shredder	41	<1	3280.5	57.2	yes						
scraper	37	43	0.8	0.97	no						
collector	19	40	11.0	3.3	yes						
predator	3	17	11.5	3.3	yes						
			3303.8								

Table 11. Chi square analysis of functional feeding groups of benthic macroinvertebrates (presented as percent of total) collected from mixed emergent vegetation zone in a natural reference wetland and mitigated wetland sites in east Tennessee.

CHI SQUARE ANALYSIS											
FFG	**natural wetland**	**Clinton**	**chi square**	**standard residual**	**sig.dif?**		**natural wetland**	**Maryville**	**chi square**	**standard residual**	**sig.dif?**
shredder	70	18	150.2	12.2	yes	shredder	70	2	2312	48.1	yes
scraper	8	<1	112.5	10.6	yes	scraper	8	24	4	2	no
collector	17	70	40.1	6.3	yes	collector	17	66	36.3	6.1	yes
predator	5	11	3.2	1.7	no	predator	5	8	1.1	1.1	no
			306						2353.4		
	expected	**Spring City**	**chi square**	**standard residual**	**sig.dif?**						
shredder	70	<1	9660.5	98.2	yes						
scraper	8	59	44.1	6.6	yes						
collector	17	28	4.3	2.1	no						
predator	5	13	4.9	2.2	no						
			9713.8								

5. DISCUSSION

Results of this study show that the mitigated wetlands have a much different trophic structure, as inferred from the macroinvertebrate community, than that of a natural wetland in the same ecoregion. In the natural wetland, shredders were by far the dominant FFG. In two of the mitigated wetlands studied, collectors were the dominant FFG, whereas scrapers were the dominant FFG in the third site. This indicates that the ecological function of these mitigated wetlands is much different than that of natural wetlands. Specifically, in the natural wetland the most common macroinvertebrates are those that feed on vascular plant tissue (i.e., shredders), while the mitigated wetlands were dominated by macroinvertebrates that feed on algae (i.e., collectors and scrapers). Observed differences in FFGs in the mitigated wetlands are likely related to surrounding land uses, and age of the sites. Nutrient inputs from surrounding land uses may impact the community structure of the wetlands studied. In terms of age, the mitigated sites are dominated by a patchy, early successional stage vascular plant community; light penetration in the mitigated sites is greater, favoring increased algal growth.

Functional Feeding Groups in Aquatic Systems

Some comparisons can be drawn between the results of this study and other studies of wetland macroinvertebrates. Most studies, however, have focused primarily on assessment of density and biomass of specific taxa or overall species richness. Few studies have investigated proportions of FFGs present in wetlands, either mitigated or natural.

However, in some instances, results from other wetland macroinvertebrate studies can be further analyzed and FFG classifications derived from lists of taxa collected.

One general similarity between this study and other studies is the relative abundance of collector organisms found in wetlands. Studies of both natural and impacted wetlands indicate that chironomids (collectors) and oligochaetes (collectors) tend to dominate the macroinvertebrate fauna of shallowly flooded emergent wetlands. In some wetlands, chironomids and oligochaetes were virtually the only organisms collected (Bergman et al. 1977). Chironomids, oligochaetes, and gastropods (scrapers) were found to be the predominate groups in emergent wetlands studied in the northern United States and Canada (Tebo 1955, Anderson & Hooper 1956, Whitman 1976). Benke et al. (1979) collected macroinvertebrates from a forested wetland in Georgia, and listed oligochaetes, *Tanypodidinae* midge fly larvae, a caddis fly (*Phylocentropus*), and chironomids as the most abundant taxa. The majority of these are classified as collectors (Merrit & Cummins 1984).

The high numbers of collectors found in the mitigated sites is also consistent with observations made by Magee et al. (1993). In a study of macroinvertebrate taxa of black willow (*Salix nigra)* wetlands in northeastern Missouri, taxa classified as collectors (Sphaeriidae, Chironomidae, Oligochaeta) were the most abundant; shredders (isopods and amphipods) were also important. Taxa classified as predators were present in 9% of the samples. Chironomids were present in over 50% of the samples; oligochaetes occurred in over 40%. These results are similar to the ratios of FFGs found in the Maryville and Clinton sites. It is difficult to draw conclusions from the similar results of this study and that of Magee et al., since their study sampled a much different type of wetland than the mitigated wetlands I studied. Magee et al. sampled seasonally flooded areas lacking understory vegetation, using artificial substrates baited with two organic litter layers. Estimates of total frequency of macroinvertebrates collected in the Missouri study were based only on the 7 taxa comprising 62-79% of the total frequency of occurrence, and did not include results of the entire 55 taxa of macroinvertebrates collected.

Proportions of FFGs found in the mitigated wetlands in east Tennessee support a model being developed by Helgen (1995) to assess wetlands. Wetlands with known anthropogenic impacts were found to have an abundance of Chironomids (collectors); least impacted sites had greater species richness. In a study of mitigated wetlands in Florida, Erwin (1991) using species richness of macroinvertebrates as a criteria for determining the "health" of mitigated wetlands. In a qualitative assessment of preserved wetlands versus created wetlands, he found species richness to be greater in preserved wetlands. Because Erwin's study only examined the presence of species, and not the numbers of organisms, it is impossible to determine proportions of FFG and compare his findings to this study.

Using a similar sampling design to this study, Bataille and Baldassarre (1993) sampled three vegetational zones (shallow marsh, composed primarily of mixed emergent vegetation; deep marsh composed of cattails and bulrushes; and open water) of a seasonal, semipermanent, and permanent wetland. Though the study examined macroinvertebrate abundance and distribution in relation to waterfowl feeding habits and focused primarily on emergent insects and gastropods, Bataille and Baldassarre found that chironomids were the most abundant taxa in all three wetlands. Their FFG ratio for collectors from all sites is similar to the results from the Clinton and Maryville sites (both mitigated sites).

The macroinvertebrate faunal community of the natural wetland in this study is similar to those described in some studies of forested wetlands, in which shredders play a larger role. Batema (1987) found that shredders (primarily isopods and amphipods) accounted for 78% of the macroinvertebrate fauna of a forested wetland in Missouri. Sklar and Conner (1979) found an equal distribution of amphipods (shredders), oligochaetes (collectors), and gastropods (scrapers) in a study of macroinvertebrates of a alluvial river swamp in Louisiana. Wharton et al. (1981) describe an unpublished study by Sniffen which determined that isopods (shredders) dominated the invertebrate biomass in a North Carolina swamp, while oligochaete worms (collectors) were numerically the most abundant organisms.

Wetlands dominated by herbaceous vegetation have been found to have high numbers of gastropods (scrapers) (Voights 1973, Reid 1983). This is consistent with the results of this study for the Spring City site, which had a high proportion of scrapers. Gastropods were the dominant taxa collected from that site.

Many studies have examined FFG populations in aquatic systems other than wetlands. Studies done on running water (lotic) systems indicate that streams approximately 0.5-10 m wide, with a well-developed overhead canopy, typically have shredder/ scraper/ collector/predator ratios of >25: <10:>50:~10 (Cummins and Wilzbach 1985). The proportion of shredders is similar between the lotic system and the natural wetland I studied. Both have a large input of organic material, primarily leaf litter in the case of the stream system and standing emergent vegetation in the case of the natural wetland. Shredders in the natural wetland, as in lotic systems, function to convert large pieces of organic matter (leaves, needles, wood, and other plant parts) into smaller particles. Similar results were found in the proportions of scrapers and predators. There is a greater difference in the proportions of collectors compared to a lotic system. Collectors feed on small particles of organic matter, either by filtering from the passing water or gathering from deposits in the sediments. The lotic system Cummins and Wilzbach studied had a collector ratio greater than 50% of total organisms sampled; the natural wetland's ratio of 18% is markedly lower.

Cummins and Wilzbach also examined the FFG ratios in a stream with an open canopy, with low shrubs and/or herbs and/or grasses. In this case the ratio of shredder/scraper/collector/predator was found to be >10:~25:>40:~10. Collectors dominate this system. Stream orders 4-6, characteristically 10-30 m wide, with a riparian habitat characterized as open variable, trees and/or shrubs had a ratio of shredders/scrapers/collectors/predators of <5:>40:>50:~10. Large rivers, with a stream order > 6 (approximately > 30 m wide) had a ratio of <10:<10:>75:~10. The changes in ratios seen in the lotic system has been described by Vannote et al. (1980) as the river continuum concept. Changes in the relative abundances of FFGs along a river tributary system from headwaters to mouth correspond directly with the changes in input of organic matter, as the river system widens and the riparian zone decreases. Proportions of shredders decrease, while proportions of scrapers increase as light and nutrient availability increase. The proportion of collectors also increases dramatically in the larger streams, as fine particulate organic matter (FPOM) is concentrated from upstream sources.

Applying this concept to the results from the mitigated wetlands, some parallels can be seen. The ratios seen in the mitigated wetlands correspond fairly well to that observed in the larger order stream described by Cummins and Wilzbach. A stream with riparian habitat described as open variable, trees and/or shrub characteristically has proportions of collectors typically greater than 50%. In both the Maryville and Clinton wetlands, collectors comprise over 70% of the organisms collected. This indicates an abundance of FPOM. Large amounts of FPOM are either the result of leaf/plant litter processing, transport from other sources, or high densities of algae. Given that shredders, which are primarily responsible for producing FPOM, comprise less than 1% of the sample, it appears that the FPOM used by collectors in these sites is largely composed of living algal cells.

Algae in the mitigated wetlands may be due to both the depth of the mitigated sites and vegetative structure. In the natural wetland, vegetation was virtually continuous over the entire wetland; open water areas were uncommon. In the mitigated sites, however, open water in some instances comprised over 50% of the areal extent of the wetland, while vegetation was irregularly distributed. Based on the river continuum concept (Vannote et al. 1980), as vegetative/organic input declines in an aquatic system, shredder populations decrease. In open water areas where light and nutrient input are higher, algal production is higher, thus favoring higher populations of collectors and scrapers. Urban and agricultural land use surrounding the mitigated wetlands also provide nutrient-rich runoff that may cause increased algal growth. While an examination of algae may give some information about primary production within wetlands, an examination of higher trophic levels gives a better understanding of the trend in community dynamics over a longer time scale.

High proportions of scrapers also indicate increased microphyte/algae production (Cummins 1992). This was evident in the Spring City site. Though I did not collect data on algae, I did observe more algae in this site than the Maryville or Clinton sites. The Spring City wetland had a higher proportion of scrapers than the other mitigated sites or the natural wetland. Shallower water than the other sites, favoring increased light penetration, may explain the increased abundance of scrapers.

Human Impacts and Land Use Influences

Patterns of increased proportions of collectors in the mitigated wetlands follow the pattern of a study of the relationship between human impact and invertebrate community structure carried out by Kerans et al. (1992). In assessing the biological integrity of lotic systems, they examined 18 characteristics of invertebrate assemblages. Of the 18 characteristics, four were related to FFGs: relative abundance of shredders, gatherers, detritivores, and chironomids. They hypothesized that as human impacts increase, proportions of collectors increase, while proportions of shredders and scrapers decrease. Results seen in the mitigated wetlands support their latter hypothesis, as proportions of shredders are markedly lower in the mitigated sites, and the Clinton site exhibits significantly lower proportions of scrapers than the mitigated wetland. Kerans et al. also hypothesize that proportions of chironomids increase with human impacts. This patterns is also reflected in my dataset, in which all mitigated sites exhibited relatively high numbers of these organisms.

While FFG analysis can give information about the normal biological functioning of the natural wetland, it can also give information about the degree to which human impact (mitigation) has influenced the function of the mitigated systems. In lower order lotic systems described by Cummins (1992), the overhead canopy of riparian vegetation restricts aquatic photosynthesis, limiting in-stream primary production. Algae-eating invertebrates, such as periphyton-grazing scrapers and collectors are less plentiful in these systems. Larger streams and rivers, in which overhead canopy is reduced, have larger proportions of scrapers and shredders. In the mitigated wetland sites, large trees and shrubs were not present to shade the wetland. Algae production is greater in these systems than in the natural wetland, which has unbroken vegetative cover and some overhead canopy. This could explain the high proportions of collectors and scrapers observed in the mitigated sites.

Richards and Host (1994) correlated land use influences with macroinvertebrate populations in a stream system, and found that non-forest land uses and housing density were related to increased nutrient supply in streams. The resulting algal growth was correlated with increased proportions of collectors. All the mitigated sites had adjacent land uses that were urban/suburban or agricultural, thus making increased nutrient loading likely. The abundance of collectors was therefore not unexpected. Agricultural land uses in the form of livestock grazing also surround the natural wetland, thus nutrient loading is likely. However in the natural wetland the abundant emergent vegetation may remove most nutrients, leaving little for algal growth.

Consideration of the land use setting of the reference wetland and the mitigated sites is important in evaluating the relationships between land use and FFGs. Selection of natural reference wetlands for this study was difficult, based on Brooks and Hughes (1988) criteria for little or no anthropogenic disturbance. Natural wetlands located in pristine, undisturbed settings are virtually nonexistent in the Ridge and Valley region, based on my search for study sites. The reference site selected was surrounded by rural, agricultural land uses, but still fulfilled the other criteria Brooks and Hughes (1988) deemed important (see below). Kentula et al. (1992) question whether pristine or undisturbed wetlands are valid comparisons for mitigated wetlands surrounded by human impacts. They suggest using natural wetlands located in similar land use settings, so as to set reasonable criteria for successful replacement of wetland function. Brooks and Hughes' (1988) selection criteria included large areas of permanent vegetation or accumulated organic matter, heterogeneity in vegetative structure and water width and depth, abundance of large woody plants or aquatic vascular macrophytes, apparent abundance of vertebrates, and little or no anthropogenic disturbance. Combining their criteria regarding vegetative structure, vertebrate abundance, and physical heterogeneity with the recommendations of Kentula et al.'s (1992) regarding similar land use settings provides for a reference wetland that is exposed to similar ecological conditions and thus reflects the potential structure and function of the mitigated sites. In this study, both the natural wetland and the Maryville site were surrounded by similar agricultural land use, yet with the exception of the proportions of scrapers, the results still differ markedly.

To effectively evaluate the Spring City and Clinton sites, a natural wetland in a more urban setting might provide a more valid comparison than the reference wetland used in this study. In my search for reference wetlands at the onset of this study, however, I was unable to find more than one wetland that fulfilled the criteria of Brooks and Hughes (1988). While the evaluation of only a single reference wetland led to uncertainty in comparing mitigated sites to the reference site, the paucity of undisturbed wetlands in the Ridge and Valley region made this necessary. Additional inventories of wetlands in the Ridge and Valley region may identify other natural wetlands that could be used in future studies.

Ecological Considerations

The community structure of macroinvertebrate FFGs may impact the populations of other macroinvertebrates and vertebrate species that depend upon them as a food source (Kent 1994). As a food source for fish, insectivorous birds, and certain mammals and herpetofauna, macroinvertebrate communities are the basis of wetland's ability to support a biologically diverse wildlife community (Horner and Raedecke 1989). Joyner (1980) found that one of the variables by which breeding ducks selected ponds was the number of invertebrate taxa present. Kaminski and Prince (1981) also studied waterfowl feeding behavior and found that feeding was concentrated in areas with the highest number of invertebrate families.

Differences in the FFG populations between my study sites support studies that correlate vegetation composition and structure with invertebrate populations. Reid (1983) indicates that invertebrate community composition in wetlands is dependent on the density and type of aquatic plants present. Leaf structure, shape, and surface area are related to invertebrate abundance. Murkin and Murkin (1989) describe the vegetative communities of wetlands as the essential structural components of the habitat of invertebrates;

variations in vegetative community and structure influences the type and density of macroinvertebrate communities. There is also a relationship between the patchiness of habitat; sparsely distributed vegetation results in diminished populations of invertebrates, and such wetlands have are less attractive to waterfowl and other wildlife (Gwin and Kentula 1986). The patchy nature of the vegetation in the mitigated sites may explain the distinct differences of FFGs seen between those sites and the natural wetland. With a relatively uniform distribution of hydrophytic vegetation, the natural wetland exhibits a much different macroinvertebrate community structure. It can be assumed that it would have a much different, more diverse community of vertebrates that use the macroinvertebrates as a food source.

Though some planting of woody trees and shrubs was required in the mitigated sites, the herbaceous hydrophyte communities sampled were the result of natural revegetation. In a study of wetland mitigation effectiveness conducted for EPA, Gwin and Kentula (1990) suggest the length of time it takes for a mitigated wetland to revegetate by natural means is often so long as to prevent functional replacement of the natural wetland. In the time it takes for a fully-vegetated wetland to develop, the functions and values of the wetland are essentially lost.

Development of a functional wetland, complete with a ecologically viable faunal community is in many ways related to studies of ecosystem disturbance and recovery. Recovery of aquatic systems is dependent upon the type of disturbance; Whiles and

Wallace (1992) characterize disturbances as press or pulse. Press disturbances involve changes in or recreation of the physical structure of a system, whereas pulse disturbances are temporal and do not involve changes to the physical structure of the system. Utilizing this characterization, mitigation would be characterized as a press disturbance, a type of disturbance which is associated with longer recovery periods than pulse disturbances.

Gore (1982) examined trends in colonization and establishment of FFGs in reclaimed strip-mined river channels, and found that collectors were the initial colonizers. As detritivores, collectors are able to find high quality food even on bare substrate. This corresponds to the results found in the Clinton and Maryville sites, which had high proportions of collectors. Gore also concluded that as habitat complexity increased, trophic complexity increased. The natural wetland, with its rich mosaic of vegetative communities, has a complex array of habitats. The proportions of FFGs found in the natural wetland is correspondingly more evenly distributed than the proportions found in the mitigated sites.

Sampling variability may affect the results of this study. I did not control for water depth within the habitat areas sampled, thus there was some variability in the depth of areas sampled. Though no areas were sampled that were over 0.6 m in depth, invertebrate populations do vary based on physical variations in depth and substrate (Reid 1983, Murkin and Murkin 1989).

The age of the mitigated sites may also have an impact on the results of this study. At the time of the study, the Clinton and Maryville sites were five years old, and the Spring City site was three years old. This study was in essence a snapshot in time. Ecological succession and community development in the mitigated sites was at a relatively early stage, when compared with the natural wetland. Current knowledge of wetland ecosystem development indicates that wetlands change in time from young, incipient stages to older, more mature stages characterized by plant communities typical of more mature wetlands and increased organic matter in the soil (Odum 1987). Studies describing long-term ecosystem development of mitigated wetlands are lacking however. The oldest mitigated wetlands are still less than 20 years old, making it difficult to predict changes in the ecosystem over time (Kadlec 1987). Though FFGs may differ markedly at the time of sampling, if habitat diversity increases and human impacts remain limited, the mitigated sites may approach the community structure of the natural wetland over time. Additional studies of these same sites in the future would provide insight about their continued ecological development.

Distance from "source" areas, and even the hydrology of the sites may influence the community composition of the benthic macroinvertebrates. The Maryville site appeared to have some hydrologic connectivity with Pistol Creek; it is likely that some colonization may occur during high flow events, when the stream floods into the wetland. The Spring City and Clinton sites were relatively isolated from adjacent source areas. Distance from source areas for colonizing organisms is one of the most important factors influencing community development following disturbance in aquatic systems (Niemi et al. 1990).

The community structure of these wetland systems depends on the assembly history of the sites (Drake 1990, 1991, Drake et al. 1993). The FFG communities in existence at the mitigated sites are a function of the sequencing of colonization at each site; as different types of FFGs colonized the new wetland sites, a specific, though dynamic FFG community structure developed. The ecological patterns, as evidenced in the community structure of FFGs found in the mitigated wetlands, depend on a complex set of biotic and abiotic factors, making it difficult to make strict comparisons between sites based on the results of this study.

The age of the sites, size, hydrology, vegetation type, and surrounding land use are all factors that affect the community composition of the sites. The interaction of biotic and abiotic factors work to create functioning ecosystems. Even within ecoregions, wetland types vary. The findings of this study indicate that the complexity and dynamics of wetland systems make it difficult to make predictions about the ecological success of wetland mitigation.

Functional Feeding Groups: The Methodology as an Assessment Tool for Wetland Mitigation

This study was the first attempt to assess the ability of mitigated wetlands to mimic the ecological function of natural wetlands through an examination of FFGs. Classification of benthic macroinvertebrates into FFGs is less time-consuming than identification to species level, and provided the appropriate resolution to give needed information about the wetland system.

An examination of the literature showed there are no standard sampling techniques for benthic macroinvertebrates in wetlands (Ross and Murkin 1989, Liebowitz et al. 1991); based on this research, a standardized sampling procedures for analysis of FFGs in wetlands is needed. Though I used an Ekman dredge for sampling, based on recommendations by Krull (1970) and Driver (1977), I believe a richer assortment of benthic macroinvertebrates could be obtained if a standard "D" shaped 304 mm (12 inch) dip net was used. My experience in this study indicated the Ekman dredge performed poorly when large amounts of vegetation were present. Using a net would also decrease the time needed to process samples, as the Ekman dredge extracted a large amount of sediment and organic material that needed to be sieved from the samples prior to processing. I would recommend using the same multihabitat sampling approach, taking a minimum of five samples per habitat zone.

Liebowitz et al. (1991) recommend that wetland invertebrate sampling techniques be standardized with respect to wetland type, season, equipment, and sorting procedures. I concur with this recommendation, especially with regard to wetland type. A sampling design similar to that of this study can provide the basis for a rapid assessment of the ecology of the mitigated wetlands. It is hoped this study will provide additional information that will guide future decision-making with regards to wetland monitoring and assessment.

6. CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER STUDY

In assessing the FFGs of the mitigated wetlands compared to the natural wetland, my objective was to draw conclusions about the "biotic integrity" of the mitigated sites. Karr (1987) describes biotic integrity as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region." In comparing the general results of the FFG analysis, it appears that there are some very clearcut differences between the natural and mitigated wetlands. This raises questions about the biotic integrity of the mitigated sites, based on Karr's 1987 definition.

The marked differences in community composition between the mitigated and natural wetlands indicate that mitigation does not replace the ecological function of wetlands lost to development. In fact, this study has raised larger questions than it has answered. Specific issues include a need for additional research on wetland ecology, both in natural and mitigated wetlands, and policy-oriented issues dealing with wetland monitoring and assessment. The results of this study also raise questions about the role of wetland mitigation in resource management and protection.

Research Needs

This study is one of few that have characterized the FFG distributions found in wetlands. Additional ecological research is needed, focusing on wetland ecology within ecoregions, including the functional roles of faunal communities within those ecoregions. This is especially vital if mitigated wetlands are to truly function as replacements for natural wetlands lost to development. Without an understanding of the types and functional roles of faunal communities found in wetlands, especially macroinvertebrate communities which are so important in food webs, an understanding of the essential element of biological community development is lacking. Much research has focused on forested wetlands; additional research should be conducted to characterize FFG proportions in natural wetlands with emergent herbaceous plant communities. Additional research is needed to determine the types and proportions of FFGs found in various types of wetlands, both on an ecoregion-level and in general, especially related to vegetation type.

Little is known about the colonization mechanisms in mitigated wetlands. Though some research has focused on other ecosystems, more information is needed regarding colonization rates in wetlands, the types of organisms that initially colonize these new wetlands, and how communities develop and evolve over time. Some research has recommended stockpiling and utilizing substrate from the existing wetlands to accelerate natural colonization mechanisms, since it contains seed banks and in some instances macroinvertebrates (Erwin 1991, Zentner 1994). Comparative studies examining the use of wetland substrate vs. upland soils would give important information about colonization and ecosystem development. Long-term monitoring of mitigated wetlands is needed to detect changes in the faunal and floral communities. Other studies have indicated mitigated wetlands are vulnerable to invasion by exotic species (Erwin 1991). What is the resistance to biological invasion and resilience from ecological disturbance of mitigated wetlands?

Wetland functions and values are intimately linked to upland ecosystems (Naiman and. Decamps 1990, National Research Council 1995). This study examined mitigated wetlands that were essentially isolated from native habitats by roads, agricultural land uses, adjacent structures or parking lots. When mitigated wetlands are created in what is (or will become) an urbanized landscape with no habitat connectivity, what is the long-term ecological viability of those systems, and what impact will be felt in terms of regional biodiversity? Can the ecological functions of mitigated wetlands be maintained, when they are created in what is (or will become) an urbanized landscape with no habitat connectivity?

This raises additional questions beyond the scope of this study about habitat fragmentation and landscape ecology. Wetland ecology and mitigation projects should be considered not as site specific/single habitat creation projects, but as part of a larger landscape composed of different types of habitats. A recognition of the importance of the surrounding landscape and its functions will lead to the creation of viable, functional wetland systems instead of small, isolated wetland patches that are vulnerable to ecological degradation.

Policy Implications

Beyond ecological questions, this study raises issues about wetland regulations and permitting requirements. In all of the wetland mitigation projects examined in this study, only cursory descriptions of the existing wetlands were included in the permits. If the true goal of wetland mitigation is to recreate the functional value of natural wetlands, permits should require a detailed assessment of the existing natural wetland before the wetland can be impacted or destroyed by development. Mitigation then should focus on recreating, as much as possible, the system that will be impacted. Wetland hydrology, vegetative communities, and faunal communities should be assessed prior to any construction or development. Macroinvertebrate surveys, using the methodology described in this study, can provide information on the trophic structure and ecology.

My investigations of wetland mitigation in East Tennessee revealed that no formal monitoring of mitigated wetlands for permit compliance and successful ecosystem development exists. Monitoring by state and federal regulatory personnel (EPA, TDEC, USGS) of the wetlands in this study involved assessing the survival (75%) of vegetation planted as required by the mitigation permit and in the case of the Spring City wetland, some hydrologic monitoring. Long-term monitoring of ecosystem development is lacking, as are any types of faunal surveys.

I concur with the recommendations of Kentula et al. (1993) who recommend routine assessments be conducted the first few years following wetland construction/mitigation, when vegetative communities and hydric soils are developing. Routine monitoring should involve visual assessment of the sites to determine if problems exist, and to record wetland development. Data collected during routine monitoring should include:

- wetland type
- surrounding land use
- water depth
- water flow patterns
- indirect evidence of hydrology (water marks, drift lines)
- vegetation coverage
- faunal observations
- photographic records
- descriptive narrative

Kentula et al. (1993) also suggest additional, comprehensive assessments be conducted three to five years after construction of emergent wetlands. Such an assessment would include collecting detailed information regarding hydrology, soil development, vegetation, water quality, and faunal assessments including observations, habitat evaluations, and species or community specific sampling. Depending on the (intended) functional values of the mitigated wetland, not all of the above characteristics would need to be assessed, but in cases in which ecological function is a concern, information about vegetation and faunal composition and structure is vital.

I believe that ecological function, measured by FFG, should be included as a stated success criteria for wetland mitigation projects, and that monitoring programs should include an analysis of wetland macroinvertebrates. Requiring routine assessments to be conducted annually, and additional comprehensive monitoring on a five-year cycle would provide sound scientific data for evaluating the success of individual mitigation projects, as well as assist research efforts regarding wetland ecology and ecosystem development.

Considerations arise regarding cost of monitoring and assessment. Property owners and developers are resistant to additional regulatory burdens placed on development, and they communicate that resistance to government and elected officials. Wetland regulations are currently under review, and federal laws are being revised at the House and Senate level to in most cases loosen current regulatory constraints. While it is unlikely that any additional regulatory requirements will be included in wetland protection laws, there remains a need to expand the science base so as to effectively regulate and monitor the resource the laws are designed to protect.

As stated in the Clean Water Act, the goal of wetland mitigation is no net loss of wetland functions or values. In regions like the Ridge and Valley of Tennessee, where natural wetlands are rare, mitigation does not appear to function as a viable natural resource protection mechanism, based on the results of this study. Though the original intent of wetland regulation considered mitigation as a viable alternative to balance economic development with environmental concerns, this study indicates wetland creation is an inexact science. Without policy based in sound scientific research, wetland loss will continue. As these losses accelerate, so too will the loss of wetland functions. The resultant impacts will be felt in not only economic terms, but also in overall loss of biodiversity.

It is hoped that this study can serve to increase our knowledge of wetland ecosystems. Analysis of FFGs in wetlands can be an effective methodology to monitor and or assess wetland ecology, but we need additional research that will characterize this and other aspects of wetland ecology. This study is but one piece of the larger puzzle. Ewel (1991, p. 92) states "...the construction of a wetland *de novo*, once it is determined that it is functionally equivalent to one that has been lost, will be an important milestone in marking our progress in achieving that understanding and in protecting the diversity of organisms that evolution has provided." Until additional research allows us a greater understanding of wetland creation and ecosystem development, there should be a greater emphasis on preserving and protecting existing wetlands. Cairns (1991) sums up this view with his statement that "...since precise replication of predisturbance condition seems highly improbably at present, preservation is definitely more practical than restoration."

LITERATURE CITED

LITERATURE CITED

Adamus, P.R. 1983. A method for wetlands functions assessment. Vol. II. **FHWA Assessment Model.** Environmental Division. Office of Research. U.S. Department of Transportation. Federal Highway Administration, Washington, D.C. FHWA-IP-82-24.

Adamus, P.R. and L.T. Stockwell. 1983. **A Method for Wetland Functional Assessment**. Report No. FHWAIP-82-23. U.S. Department of Transportation, Federal Highway Administration, Washington, D.C.

Adamus, P.R. and K. Brandt. 1990. **Impacts on Quality of Inland Wetlands of the U.S.: A Survey of Indicators, Techniques, and Applications of Community-Level Biomonitoring Data**. U.S. EPA Environmental Research Laboratory, Corvallis, Oregon.

Anderson, R.O., and F.F. Hooper. 1956. Seasonal abundance and production of littoral bottom fauna in a southern Michigan lake. **Annual Review of Entomology** 75: 259-270.

Anderson, T. 1988. **Inland Wetlands Protection at the Local Government Level: Ecological and Institutional Considerations**. Master's Thesis, Cornell University.

Baker, G.F. 1984. **An Analysis of Wetland Losses and Compensation Under the Clean Water Act Section 404 Program: Managing Natural Resources through Mitigation**. M.S. Thesis, University of San Francisco, San Francisco, California.

Bataille, K.J., and G.A. Baldassare. 1993. Distribution and abundance of aquatic macroinvertebrates following drought in three prairie pothole wetlands. **Wetlands 13(4): 260-269.**

Batema, D.L. 1987. **Relations among wetland invertebrate abundance, litter decomposition, and nutrient dynamics in a bottomland hardwood ecosystem**. Ph.D. Dissertation, University of Missouri, Columbia, Missouri.

Bergman, R.D. et al. 1977. Water birds and their wetland resources in relation to oil development at Storkerson Point, Alaska. Resource Publication 129, U.S. Fish and Wildlife Service, Washington, D.C.

Blumm, M.C. and D.B. Zahela. 1989. Federal wetlands protection under the Clean Water Act: Regulating ambivalence, intergovernmental tension, and a call for reform. **University of Colorado Law Review** 60: 695.

Brinson, M.M. 1992. **A Hydrogeomorphic Classification of Wetlands.** U.S. Army Corps of Engineers, Washington, D.C. Technical report WRP-DE-4.

Brooks, R.P. and Hughes, R.M. 1986. Guidelines for assessing the biotic communities of freshwater wetlands. In **Proceedings of the National Wetlands Symposium: Mitigation of Impacts and Losses**, Kusler, J.A. et al. (eds.), pp. 276-282. Association of State Wetland Managers, Berne, New York.

Broschart, M.R. and R. L. Linder. 1986. Aquatic invertebrates in level ditches and adjacent emergent marsh in a South Dakota wetland. **Prairie Naturalist** 18: 167-178.

Burke, D.G., E.J. Meyers, R.W. Tiner, Jr., and H. Groman. 1988. **Protecting Non-Tidal Wetlands**. American Planning Association, Chicago, Illinois.

Cairns, J., Jr. 1974. Indicator species vs. the concept of community structure as an index of pollution. **Water Resources Bulletin** 10(2)338-347).

Cairns, J., Jr. 1993. Is restoration ecology practical? **Restoration Ecology** 1(1): 3-7.

Carter, V. 1986. An overview of hydrologic concerns related to wetlands in the United States. **Canadian Journal of Botany** 64(1-3): 364-374.

Carter, V., M.S. Bedinger, R.P. Novitzki, and W.O. Wilen. 1979. Water resources and wetlands. In **Wetland Functions and Values: The State of our Understanding.** P.E. Greeson, et al. (eds.), pp. 344-376. American Water Resources Association, Minneapolis, Minnesota.

Corkum, L.D. 1989. Patterns of benthic invertebrate assemblages in rivers of northwestern North America. **Freshwater Biology** 21: 191-205.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. **Classification of Wetland and Deepwater Habitats of the United States**. Washington, D.C.: U.S. Fish and Wildlife Publication FWS/OBS-79/31.

Cuipek, R.B. 1986. Protecting wetlands under the Clean Water Act 404: E.P.A.'s conservative policy on mitigation. **National Wetlands Newsletter** 8: 12-14.

Cummins, K.W. 1973. Trophic relations of aquatic insects. **Annual Review of Entomology** 18: 183-206.

Cummins, K.W. 1992. Invertebrates. In: **The Rivers Handbook**, pp. 234-250, P. Calow and G.E. Petts (eds.), Blackwell Scientific Publications, Oxford, Great Britain.

Cummins, K.W., and M.A. Wilzbach. 1985. **Field Procedures for Analysis of Functional Feeding Groups of Stream Macroinvertebrates**. Contribution 1611, Appalachian Environmental Laboratory. University of Maryland Press, Frostburg, Maryland.

Dahl, T.E. 1990. **Wetlands Losses in the United States 1780's to 1980's**. U.S. Department of Interior, Fish and Wildlife Service, Washington, D.C..

Daniel, C.C. III. 1981. Hydrology, geology, and soils of pocosins: a comparison of natural and altered systems. In **Pocosin Wetlands**, Richardson, C.J., et al. (eds.) pp. 69-108. Hutchison Ross Publishing Company, Stroudsbury, Pennsylvania.

D'Avanzo, C. 1986. Vegetation in freshwater replacement wetlands in the northeast. In **Mitigating Freshwater Replacement Wetlands in the Northeast: An Assessment of the Science Base**, Larson, J.S., and C. Neill (eds.) pp. 53-81. The Environmental Institute, University of Massachusetts, Amherst, Massachusetts:

Davis, S. 1991. Personal Communication. Wetland Ecologist, Tennessee Valley Authority, Norris, Tennessee.

Drake, J.A. 1990. The mechanisms of community assembly and succession. **Journal of Theoretical Biology** 147:213-233.

Drake, J.A. 1991. Community assembly mechanics and the structure of an experimental species ensemble. **American Naturalist** 137(1): 1-26.

Drake, J.A., T.E. Flum, G.J. Witteman, T.Voskuil, A.M. Hoylman, C. Creson, D.A. Kenny, G.R. Huxel, C.S. LaRue, and J.R. Duncan. 1993. The construction of an ecological landscape. **Journal of Animal Ecology** 62: 117-130.

Driver, E.A. 1977. Chironomid communities in small prairie ponds: some characteristics and controls. **Freshwater Biology** 7: 121-133.

Elder, J.A. et al. 1953. Soil Survey, Blount County, Tennessee. U.S. Department of Agriculture, Washington, D.C.

Ellis, D. 1985. Taxonomic sufficiency in pollution assessment. **Marine Pollution Bulletin** 16: 459-468.

Eslinger, B. 1993. Personal Communication. Land owner, Eslinger Marsh, Jefferson County, Tennessee.

Environmental Laboratory. 1987. **Corps of Engineers Wetlands Delineation Manual**, Technical Report Y-87-1. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Erwin, K.L. 1989a. Freshwater marsh creation and restoration in the southeast. In **Wetland Creation and Restoration: The Status of the Science, Volume I**, J.A. Kusler and M.E. Kentula (eds.), pp. 233-248. Island Press, Washington, D.C.

Erwin, K.L. 1989b. Wetland evaluation for restoration and creation. In **Wetland Creation and Restoration: The Status of the Science, Volume I**, J.A. Kusler and M.E. Kentula (eds.), pp. 429-458. Island Press, Washington, D.C.

Erwin, K.L. 1991. **An Evaluation of Wetland Mitigation in the South Florida Water Management District**. Contract #C89-=0082-Al. West Palm Beach Florida: South Florida Water Management District.

Ewel, K.C. 1991. Diversity in wetlands. **Evolutionary Trends in Plants** 5(2): 90-92.

Ferraro, S.P., and F.A. Cole. 1990. Taxonomic level and sample size sufficient for assessing pollution impacts on the California Bight macrobenthos. **Marine Ecology Program Series** 67: 251-262.

Fredrickson, L.H. and F.A. Reid. 1988. Invertebrate response to wetland management. In **Waterfowl Management Handbook, Fish and Wildlife Leaflet 13.** U.S. Fish and Wildlife Service, Washington, D.C.

Gore, J.A. 1982. Benthic invertebrate colonization: source distance effects on community composition. **Hydrobiologia** 94: 183-193.

Governor's Interagency Wetlands Committee. 1994. **Tennessee Wetlands Conservation Strategy.** Tennessee State Planning Office, Nashville, Tennessee.

Greeson, P.E., J.R. Clark, and J.E. Clark (eds). 1978. **Wetland Functions and Values: The State of Our Understanding**. American Water Resources Association, Minneapolis, Minnesota.

Gwin, S.E. and M.E. Kentula. 1990. **Evaluating Design and Verifying Compliance of Wetlands Created Under Section 404 of the Clean Water Act in Oregon**. EPA/600/3-90/061. Corvallis, Oregon: Environmental Research Laboratory.

Hack, J.T. 1969. The Area, Its Geology: Cenozoic Development of the Southern Appalachians, In **Distributional History of Biota of the Southern Appalachians, Part I.** Holt, P.C. (ed.), pp. 1-17. Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

Hasty, A.H. et al. 1940. **Soil Survey, Rhea County, Tennessee**. U.S. Department of Agriculture, Washington, D.C.

Helgen, J. 1995. Biological assessment of reference wetlands. **Bulletin of the North American Benthological Society** 12(1): 150.

Horner, R.R. and K.J. Raedeke. 1989. **Guide for Wetland Mitigation Project Monitoring.** Washington State Transportation Commission, Olympia, Washington.

Huener, J.D. 1984. **Macroinvertebrate Response to Marsh Management**. M.S. Thesis, Utah State University, Logan, Utah.

Jones, R.C. and C.C. Clark. 1987. Impact of watershed urbanization on stream insect communities. **Water Resources Bulletin** 23(6): 1047-1055.

Joyner, D.E. 1980. Influence of invertebrates on pond selection by ducks in Ontario. **Journal of Wildlife Management** 44: 700-705.

Kadlec, R.H. 1987. Monitoring wetland responses. In **Increasing Our Wetland Resources**, J. Zelazny and J.S. Feierabend (eds.)., pp. 114-120. National Wildlife Federation, Washington, D.

Kadlec, R.H. and J.A. Kadlec. 1978. Wetlands and water quality. In **Wetland Functions and Values: The State of Our Understanding,** pp. 436-546. P.E. Greeson, et al. (eds.) American Water Resources Association, Minneapolis, Minnesota.

Kaminski, R.M. and H.H. Price. 1981. Dabbling duck activity and foraging responses to aquatic macroinvertebrates. **Auk 98: 115-126.**

Karr, J.R. 1987. Biological monitoring and environmental assessment: a conceptual framework. **Environmental Management** 11: 249-256.

Kent, DM 1994. Managing wetlands for wildlife. In **Applied Wetland Science and Technology,** pp.306-329. DM Kent (ed.). CRC Press, Boca Raton, Florida.

Kentula, M.E., J.C. Sifneos, J.W. Good, M. Rylko, and K. Kunz. 1992. Trends and patterns in Section 404 permitting requiring compensatory mitigation in Oregon and Washington. **Environmental Management** 16: 109-119.

Kentula, M.E., R.P. Brooks, S.E. Gwin, C.C. Holland, A.D. Sherman, J.C. Sifneos. 1993. **An Approach to Improving Decision Making in Wetland Restoration and Creation.** C.K. Smoley, Chelsea, Michigan.

Kerans, B.L., J.R. Karr, and S.A. Ahlstedt. 1992. Aquatic invertebrate assemblages: spatial and temporal differences among sampling protocols. **Journal of the North American Benthological Society** 11(4): 377-390.

Koryak, M. 1982. Wetland regulation in Appalachia. In: **Proceedings of the Symposium on Wetlands of the Unglaciated Appalachian Region**, B.R. McDonald (ed.). West Virginia University, Morgantown, West Virginia.

Kruczynski, W.L. 1989. Options to be considered in preparation and evaluation of mitigation plans. In **Wetland Creation: The Status of the Science, Volume I**. Kusler, J.A. and M.E. Kentula (eds.) pp. 549-554. Island Press, Washington, D.C.

Krull, J.N. 1970. Aquatic plant-macroinvertebrate associations and waterfowl. **Journal of Wildlife Management** 34(4): 707-718.

Kusler, J.A. 1983. **Our National Wetland Heritage: A Protection Guidebook**. Environmental Law Institute, Washington, D.C.

Kusler, J.A. and M.E. Kentula (eds.). 1989. **Wetland Creation and Restoration: The Status of the Science.** EPA/600/3-98/0. Environmental Protection Agency, Corvallis, Oregon.

Landin, M.C., E.J. Clairain, Jr., C.J. Newling. 1989. Wetland habitat development and long-term monitoring at Windmill Point, Virginia. **Wetlands** 9(1): 13-26.

Larson, J.S. 1988. Wetland creation and restoration: an outline of the scientific perspective. In **Increasing Our Wetland Resources**, Zelazny , J. and J.S. Feierabend (eds.) pp. 73-79. National Wildlife Federation, Corporate Conservation Council, Washington, D.C.

Liebowitz, N.C., L. Squires, and J.P. Baker. 1991. **Research Plan for Monitoring Wetland Ecosystems**. U.S. E.P.A. 600/3-91/010. U.S. Environmental Protection Agency, Washington, D.C.

Magee, P.A., L.H. Fredrickson, and D.D. Humburg. 1993. Aquatic macroinvertebrate association with willow wetlands in northeastern Missouri. **Wetlands** 13(4): 304-310.

Maltby, E. 1988. Global wetlands - history, current status, and future. In **The Ecology and Management of Wetlands, Volume I: Ecology of Wetlands**, Hook, D.D. et al. (eds.). Timber Press, Portland, Oregon.

Mason, C.O., and D.A. Slocum. 1987. Wetland replication - does it work? In **Proceedings of the 5th Symposium on Coastal and Ocean Management, Volume I**, pp. 1183-1197. American Society of Civil Engineering, New York.

Mazanti, L. 1993. Personal Communication. Wetland Regulator, Environmental Protection Agency, Region IV, Atlanta, Georgia.

Merritt, R.W. and K.W. Cummins (eds.). 1984. **An Introduction to the Aquatic Insects of North America (2nd Edition).** Kendall/Hunt Publishing, Dubuque, Iowa.

Mitsch, W.J. and J.C. Gosselink. 1986. **Wetlands**. Van Nostrand Reinhold Company Inc., New York.

Moon, J.W. et al. 1935. **Soil Survey, Jefferson County, Tennessee**. U.S. Department of Agriculture, Washington, D.C.

Moneymaker, R.H. 1981. Soil Survey of Anderson County, Tennessee. U.S. Department of Agriculture, Washington, D.C.

Murkin, E.J. and H.R. Murkin (eds). 1989. **Marsh Ecology Research Program: Long-term Monitoring Procedures Manual.** Delta Waterfowl and Wetlands Research Station, Portage la Prairie, Manitoba, Canada.

Naiman, R.J. and H. Decamps (eds.). 1990. **The Ecology and Management of Aquatic-Terrestrial Ecotones**. United Nations Educational, Scientific, and Cultural Organization, Paris.

National Research Council. 1995. **Wetlands: Characteristics and Boundaries.** Commission on Geosciences, Environment, and Resources, Washington, D.C.

Newling, C.J., and M.C. Landing. 1985. **Long-Term Monitoring of Habitat Development at Upland and Wetland Dredged Material Disposal Sites, 1974-1982**. Technical Report D-85-5. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Niemi, G.J., P. DeVore, N. Detenbeck, D. Taylor, J.D. Yount, A. Lima, J. Pastor, and R.J. Naiman. 1990. Overview of case studies on recovery of aquatic systems from disturbance. **Environmental Management** 14: 571-588.

Odum, W.E. 1987. Predicting ecosystem development following creation and restoration of wetlands. In **Increasing Our Wetland Resources**, J. Zelazny and J.S. Feierabend (eds.)., pp. 67-70. National Wildlife Federation, Washington, D.C.

Omernik, J.M. 1987. Ecoregions of the coterminous United States. **Annals of the Association of American Geographers** 77: 118-125.

Packard, G.C. and R.K. Stiverson. 1975. Abundance and production of macroinvertebrates from natural and artificially established salt marshes in North Carolina. **American Midland Naturalist** 96(2): 487-493.

Pacific Estuarine Research Laboratory. 1990. **A Manual for Assessing Restored and Natural Coastal Wetlands with Examples from Southern California.** California Sea Grant Report No. T-CSGCP-021. La Jolla, California.

Pearson, S.M. 1994. Landscape-level processes and wetland conservation in the southern Appalachian Mountains. In Trettin, C.C. et al. (eds.) **Wetlands of the Interior Southeastern United States, Journal of the Water, Air, and Soil Pollution Society** 77: (3-4): 321-332..

Phillips, J.D. 1989. Fluvial sediment storage in wetlands. **Water Resources Bulletin** 25: 867-873.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M Hughes. 1989. **Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish.** U.S. Environmental Protection Agency, EPA/444/4-89-001.

Quammen, M.L. 1986. Measuring the success of wetlands mitigation. **National Wetlands Newsletter** 8: 6-8.

Race, MA. 1985. Critique of present wetlands mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. **Environmental Management** 9(1): 71-82.

Reed, P.B., Jr. 1988. **National List of Plant Species that Occur in Wetlands: National Summary**. U.S. Fish and Wildlife Service Biological Report 88(24).

Reid, F.A. 1983. Aquatic macroinvertebrate response to management of seasonally-flooded wetlands. M.S. Thesis. University of Missouri, Columbia, Missouri.

Reid. F.A.. 1985. Wetland invertebrates in relation to hydrology and water chemistry. In: **Water Impoundments for Wildlife**: a Habitat Management Workshop, Knighton, M.D. et al. (eds.), pp.72-79. North Central Forest Experiment Station General Technical Report NC-100, St. Paul, Minnesota.

Reimhold, R.J. 1994. Wetland functions and values. In D.M. Kent (ed.) **Applied Wetland Science and Technology**, pp. 55-78. CRC Press, Boca Raton, Florida.

Reinold, R.J. and S.A. Cobler. 1986. **Wetlands Mitigation Effectiveness**. E.P.A. Publication 68-04-0015. Environmental Protection Agency, Washington, D.C.

Richards, C., and G. Host. 1994. Examining land use influences on stream habitats and macroinvertebrates : a GIS approach. **Water Resources Bulletin** 30(4): 729-738.

Richards, C., G.E. Host, and J.W. Arthur. 1993. Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. **Freshwater Biology 29: 285-294.**

Rosenberg, D.M. and V.R. Resh. 1993. **Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall,** New York, New York.

Ross, L.C.M. and H.R. Murkin. 1989. Invertebrates. In **Marsh Ecology Research Program: Long-Term Monitoring Procedures Manual**, E.J. Murking and H.R. Murkin (eds.), pp. 35-39. Delta Waterfowl and Wetlands Research Station, Portage La Prairie, Manitoba, Canada.

Sacco., J.N., F.L. Booker, and E.D. Smith. 1987. Comparison of the macrofaunal communities of a human-created salt marsh at two and fifteen years of age. In **Increasing Our Wetland Resources, Corporate Conservation Council Proceedings**, Zelazny. J., and J.S. Feierabend (eds.), pp. 282-285. National Wildlife Federation Conference. Washington, D.C

Salvesen, D. 1990. **Wetlands: Mitigating and Regulating Development Impacts**. The Urban Land Institute, Washington, D.C.

Sather, J.M. and R.D. Smith. 1984. **An Overview of Major Wetland Functions and Values**. U.S. Fish and Wildlife Report FWS/OBS-84/18. Department of Interior, Washington, D.C.

Schaeffer, O.J. et al. 1985. Evaluation of a community-based index using benthic indicator organisms for classifying stream quality. **Water Pollution Control Federation Journal** 57(2): 167-171.

Seneca, E.D., S.W., Broome, W.W. Woodhouse, L.M. Cammen, and J.T. Lyon. 1976. Establishing Spartina alterniflora marsh in North Carolina. **Environmental Conservation** 3: 185-188.

Sklar, F.H. and W.H. Conner. 1979. Effects of altered hydrology on primary production and aquatic animal populations in a Louisiana swamp forest. In **Proceedings 3rd Coastal Marsh and Estuary Management Symposium,** Day, J.W., Jr., R.E. Turner and A.J. Mumphrey, Jr. (eds.), Pp. 191-208. Louisiana State University, Division of Continuing Education, Baton Rouge, Louisiana.

Smith, L.S. and E.P. Michael. 1982. Values of wetlands in the unglaciated Appalachian region. In: B.R. McDonald (ed.) **Proceedings of the Symposium on Wetlands of the Unglaciated Appalachian Region.** West Virginia University, Morgantown, West Virginia.

Sokal, R.R. and F.J. Rohlf. 1981. Biometry: Principles and Practice of Statistics in Biological Research, 2nd edition. W.H. Freeman. New York.

Soukanov, A.H. (ed.) 1984. **Webster's II New Riverside University Dictionary**. The Riverside Publishing Company, Houghton Mifflin Company, Boston, Massachusetts.

Streever, W.J. and T.L. Crisman. 1993. A preliminary comparison of meiobenthic cladoceran assemblages in natural and constructed wetlands in central Florida. **Wetlands** 13(4): 229-236.

Tebo, L.B. 1955. Bottom fauna of a shallow eutrophic lake, lizard lake, Pocahontas County, Iowa. **American Midland Naturalist** 54: 89-103.

Trettin, C.C. 1995. Wetlands of the Interior Southeastern United States. **Journal of the Water, Air, and Soil Pollution Society** 77 (3-4).

United States Army Corps of Engineers. 1990. **Water Quality Standards for Wetlands: National Guidance.** Office of Water Regulations and Standards. U.S. Environmental Protection Agency, Washington D.C. EPA 440/S-90-011.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. **Canadian Journal of Fisheries and Aquatic Sciences** 37: 130-137.

Voights, D.K. 1973. Aquatic invertebrate abundance in relation to changing marsh vegetation. **American Midland Naturalist** 95: 313-322.

Wetzel, R.G. 1983. **Limnology.** Sanders College Publishing, New York, New York.

Wharton, C.H., V.W. Lambour, J. Newson, P.V. Winger, L.L. Gadd, and R. Mancke. 1981. The fauna of bottomland hardwoods in southeastern United States. In **Workshop on Bottomland Hardwood Forest Wetlands of the Southeastern United States**, Clark, J.R. and J. Benforado (eds.), pp. 87-160. Elsevier Scientific Publishing Company, New York, New York.

Whiles, M.R. and J.B. Wallace. 1992. First-year benthic recovery of a headwater stream following a 3-year insecticide-induced disturbance. **Freshwater Biology** 28: 81-91.

Zedler, J.B. 1990. **A Manual for Assessing Restored and Natural Coastal Wetlands**. Pacific Estuarine Research Laboratory, San Diego State University, San Diego, California.

Zentner, J. 1988. Wetland restoration in coastal California. In **Increasing Our Wetland Resources, Corporate Conservation Council Proceedings**, Zelazny. J., and J.S. Feierabend (eds.), pp. 216-219. National Wildlife Federation Conference. Washington, D.C.

Zentner, J. 1994. Enhancement, restoration and creation of freshwater wetlands. In D.M. Kent, ed., **Applied Wetland Science and Technology**, pp. 126-166. CRC Press, Inc., Boca Raton, Florida.

APPENDICES

Appendix A

Natural Wetlands Removed from Study Consideration (based on personal communication with Tennessee Wildlife Resources personnel)

Name	Location	Reason
Greater Alcoa Marsh	Alcoa, TN	Potential PCB contamination
Lesser Alcoa Marsh	Alcoa, TN	Potential PCB contamination
Amnicola Marsh	Chattanooga, TN	Located immediately adjacent to industrial facilities

Appendix B

Directions to Study Sites

- Natural Wetland, Jefferson County
  Highway 11E North from Knoxville; Right on Eslinger Road. Wetland is located approximately 2 miles past junction with Hwy. 11E, on right.
  Note: Site access must be approved by property owner, Mr. Bruce Eslinger.

- Mitigated Wetland, Clinton, Tennessee
  Interstate 75 North from Knoxville; Exit at Highway 61 (Clinton Exit). Take Highway 61 west into Clinton. Left on Eagle Bend Road (Eagle Bend Industrial Park). Food Lion Distribution Facility is located approximately 1.2 miles on right; Turn into Parking Lot. Wetland is located between warehouse and Clinch River.
  Note: Site access must be approved by both TVA and on-site Food Lion personnel.

- Mitigated Wetland, Maryville, Tennessee
  Highway 129 South from Knoxville. In Maryville, take Highway 321 West. At Maryville Industrial Park, turn right. Wetland is located behind the Nippondenso facility.
  Note: Site access must be approved by City of Maryville.

- Mitigated Wetland, Spring City, Tennessee
  Interstate 75 South from Knoxville. Exit at Highway 68. Take State Road 68 to State Road 29. Turn left (North) into Spring City. Wetland is located behind an auto parts store immediately before the intersection with State Road 68.
  Note: Site access must be approved by Tennessee Department of Transportation.

Appendix C

Functional Feeding Groups As An Assessment Methodology: Recommendations

- Natural reference wetland and mitigation sites should have similar surrounding land uses.
- Standardized sampling procedures, based on wetland type.
- Sample macroinvertebrates in different habitats within the wetland. Sampling should be undertake in representative habitats common to all sites.
- Sampling should occur in early to mid-Spring.
- Sample representative habitats with standard 304 mm (12") "D" net; collect a minimum of five samples per habitat type.
- Assessment of functional feeding groups should occur in year one (post mitigation), year five, and in five year intervals.
- To gain a fuller understand of wetland function, the analysis of functional feeding groups should be part of an overall assessment of vegetation reestablishment, hydrology, and wildlife utilization.

VITA

Kim Pilarski was born in South Bend, Indiana on November 13, 1960 the daughter of Harry E. Pilarski and Carol Broxholm Pilarski. She attended elementary and high school in Buchanan, Michigan and Naples, Florida, graduating from Buchanan High School in June, 1979. She attended Michigan State University and the University of Tennessee, graduating with a Bachelor's of Arts in August, 1983. In 1989 she reentered the University of Tennessee, and in December of 1996 received a Master of Science degree in Geography.

Kim is currently employed as an Environmental Scientist by the Tennessee Valley Authority's Clean Water Initiative. She has two children, Ian Edward Pilarski Turner and Alexandra Jane Pilarski Turner.