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Groffman, Gold, Husband, Simmons, & Eddleman (URI)

Narragansett Bay Estuary Program

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FINAL REPORT

NARRAGANSETT BAY PROJECT

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AN INVESTIGATION INTO MULTIPLE USES OF VEGETATED BUFFER STRIPS

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SUMMARY

While the use of buffer zones to protect wetlands and surface water bodies is mandated by Rhode Island law, there is very little information available on the factors that control the effectiveness of these zones under Rhode Island conditions. Research was needed to determine the criteria that should be considered in the design and maintenance of buffer zones in the landscape. The goal of our study was to provide information on the suitability of any particular piece of ground as a "vegetated buffer strip (VBS)" for water quality protection and wildlife habitat given information on soils, vegetation, geomorphology and surrounding land uses. For the water quality studies, our approach was to make intensive, site specific measurements of pollutant removal from surface and subsurface flow at a small number of well instrumented buffer strips. We also developed a "microbial index" of buffer zone pollutant removal capacity that was then measured at a larger number of sites differing in soils, vegetation, geomorphology and surrounding land uses. The intensive site specific measurements of actual buffering activity were used to calibrate this index. For the wildlife studies we determined species richness of birds and herpetofauna and described wildlife habitat parameters in buffer strips. We then developed a model to prescribe buffer requirements to protect wetland-dependent wildlife species.

<u>Surface removal studies</u>

For the surface removal studies, field plots were established as part of a USDA SCS funded study to compare 10 different grasses for their ability to take up added nitrogen (N). All plots received 177 kg N/ha over three applications, and N loss to groundwater was measured using tension plate lysimeters. Loading rates of N as nitrate (NO_3^-) to groundwater ranged from 2 to 106 kg N/ha. N removal efficiencies for the grasses (calculated as N not leached/total N input) ranged from 40 to 99% of the N applied in summer but was less than 30% in winter. Denitrification (conversion of NO_3^- to N gas) was measured in soils from the surface runoff plots subjected to simulated runoff, in contained incubations. Under these conditions, denitrification N removal efficiencies (calculated as denitrification rate/total N addition) ranged from 1 to 50% per day. Denitrification rates were higher in upland grass plots than in either well drained or poorly drained forest plots, likely due to acid soil in the forests. Similar to uptake of N by grass, denitrification is likely to be ineffective at removing N from surface runoff during winter, even if flow through the buffer is controlled.

Major findings and management implications from the surface removal studies include:

1. Different grass species have dramatically different abilities to act as sinks for NO_3^- under Rhode Island conditions.

2. Winter removal of NO_3^- by plants or denitrification is unlikely.

3. Upland, limed, grass plots had a higher capacity to remove added NO_3^- by denitrification than either upland or wetland forest plots.

4. Low pH in some forest soils in Rhode Island limits their capacity for denitrification.

5. If flow through the VBS is controlled, denitrification can remove large amounts off NO_3^- (up to 50% of addition per day). New non-point source pollutant control mechanisms that combine aspects of VBS and artificial wetlands may be highly effective NO_3^- sinks in the landscape.

Groundwater studies

To determine the ability of natural riparian buffers to attenuate groundwater contaminants we related changes in groundwater quality to physical and chemical features of upland and wetland buffers. A network of groundwater monitoring wells were placed in different soil drainage classes (moderately well drained, somewhat poorly drained, poorly drained, and very poorly drained soils) adjacent to streams at three sites. A "spike" of NO_3^-, copper, and a conservative tracer (chloride) was added upgradient of all wetland and upland locations. Attenuation of NO_3^- and copper was assessed by observing changes in the relative chloride ratios compared to the application ratios. Complete attenuation of copper was found at all sites, due to the high affinity of soil organic matter for copper. Wetland locations were found to remove most of the NO_3^- from groundwater, while upland sites showed considerable variability with regard to NO_3^- attenuation. Depth to the water table was the only site factor significantly related to NO_3^- attenuation. When depth to groundwater was less than 60 cm. (which corresponds to the root zone depth), considerable attenuation was observed.

Major findings and management implications from the groundwater studies include:

1. As expected, movement of a contaminant pulse through groundwater in a riparian site was quite slow. Mean velocities ranged from 0.07 to 0.2 m/day.

2. Complete attenuation of copper was observed at all locations, suggesting that riparian buffers are strong sinks for this metal.

3. NO_3^- attenuation in upland VBS ranged from 21 to 99% and increased significantly as the plume moved downslope.

4. Almost complete NO_3^- removal was observed in both poorly and very poorly drained soils.

5. Depth to water table during the study was significantly correlated with NO_3^- attenuation in the upland locations. Where the mean water table was within the expected rootzone (within 60 cm of the surface) considerable attenuation occurred. When the mean water table was below the root zone (deeper than 60 cm) little removal was noted.

6. The NO_3^- removal mechanisms were not identified in this study. Denitrification, plant uptake, and microbial immobilization may all have accounted for the removal observed. Repeating this experiment in early spring, before considerable plant uptake begins, could shed light on removal mechanisms.

The results of the water quality aspect of our study demonstrated that selected portions of riparian buffers have the capacity to remove NO_3^- and copper from groundwater in the summer. The implications of these results for management purposes are that buffer widths required for pollutant removal will vary based on the soil types present. The ability of selected somewhat poorly drained soils and all poorly drained soils to completely remove the groundwater contaminants studied was particularly noteworthy. Our study suggests that certain riparian lands upslope and on drier soils than recognized wetlands have the capacity to improve groundwater quality part of the year, and can serve to protect both wetland and surface water ecosystems during summer months. The summer removal rates observed may not result in permanent removal of pollutants, since plant uptake (a major removal mechanism in the summer) may be a temporary sink for nutrients. To determine the long term role of VBS in the transition zone between uplands and wetland will require study during early spring, when plant uptake is limited, and will also require site specific assessments of the proportion of contaminated groundwater which is likely to interact with the riparian buffer.

Microbial index studies

To develop an easily measured "index" of VBS pollutant attenuation capacity, we evaluated two microbial parameters as predictors of the pollutant attenuation observed in our groundwater studies. Denitrification enzyme activity (DEA) and microbial biomass carbon (C) were measured within the sites used in our groundwater studies and were compared with observed patterns of pollutant attenuation. Both DEA and microbial biomass C were higher in wetland areas than in upland areas, and in surface soils versus subsurface (groundwater) zones. The microbial parameters were significantly correlated with pollutant attenuation observed in the well monitoring networks, but high plant uptake of NO_3^- reduced their predictive power. At some sites, especially in the uplands, high pollutant attenuation was observed when the microbial parameters predicted low attenuation (a conservative error of prediction). In no cases was high attenuation predicted and not observed. DEA was measured in surface and subsurface soils in 20 locations across the Hunt-Potowomut (RI) watershed and in two artificial wetland types in Massachusetts. This sampling confirmed that wetland areas have higher buffering potential than upland zones and that soil pH controls variability in forest denitrification potential across the landscape. The artificial wetlands supported as high or higher denitrification capacity than the natural wetlands.

Major findings from the index work include:

1. Surface soil microbial biomass C and/or DEA may be useful as conservative indicies of VBS pollutant attenuation potential.

2. Plant uptake appears to be the major NO_3^- sink in upland VBS while plant uptake, denitrification and microbial immobilization operate as sinks in wetland VBS. There is a strong need for further research on pollutant attenuation in VBS during periods when plants are not active.

3. Soil pH appears to be more important than geomorphic position or upland land use in controlling VBS pollutant attenuation capacity across the landscape. There is a strong need for further research to determine the landscape scale factors that control soil pH.

4. The artificial wetland systems studied had as high or higher potential for biological attenuation of pollutants than native wetlands.

Our site specific studies of surface and groundwater removal suggested that there are several key factors that control the water quality maintenance value of VBS. Understanding how these factors are expressed across the landscape will be essential for determining areas to be preserved as VBS, for assessing the performance of existing VBS, and for designing and evaluating new non-point source pollutant control techniques. The microbial indicies developed may be useful tools for these landscape scale assessments. Further research is needed to determine how these indicies perform during periods when plant uptake is not active, and to determine standard protocol for their implementation as assessment tools.

Wildlife studies

Most often buffers are associated with pollution abatement and water quality, however, buffers may be very important to wildlife and represent unique habitats with inherent values. Values of wetland buffers may include: (1) increased species richness; (2) sites for foraging; (3) corridors for dispersal; (4) escape from flooding; (5) sites for hibernation; (6) areas for breeding and nesting; (7) areas of low predator and nest parasite density; and (8) buffering of disturbances from outside the wetland. Research was needed to examine the use by wildlife of buffers on red maple swamps, our most common freshwater wetland.

The species richness and diversity of herpetofaunal and mammal communities in four red maple swamp buffers were measured from March to November 1989. In addition, wildlife habitat parameters were described for those same buffers. Trapping resulted in the capture of 2,064 individuals, comprised of 12 species of amphibians (n = 1,668), 2 species of reptiles (n = 3), and 12 species of mammals (n = 395). The most remote, undisturbed site, Great Swamp, had the highest number and diversity of herps, while the most disturbed site, Frenchtown, had the highest number and diversity of mammals. A remarkably high number of herps and mammals were recorded on all four of these wide buffers. Three rare (for Rhode Island), area-sensitive mammal species were captured on buffers: water shrew, smoky shrew, and southern bog lemming.

Bird communities along the cline from upland oak to red maple swamp were censused in southern Rhode Island in June and July 1989 to document the composition of these communities, determine territory requirements of component species for inclusion in a wetland buffer model, determine whether any species were significantly associated with upland, transition, or wetland forest. Vegetation was measured in 0.04 ha plots along these transects to determine if vegetation characteristics were explaining bird species distributions. Four sites were selected for study, three in the Hunt-Potowomut basin and one in the Great Swamp Management Area. Forty-four species were detected at the combined four sites, of which 19 were Neotropical migrant species potentially of concern for wildlife management. The bird community composition was very similar to that in two other studies done in forested wetlands in southern New England. Three measures of occurrence; numbers of song detections, numbers of total detections, and density; were analyzed to determine if birds were significantly associated with upland, transition, or wetland habitat. Two species, eastern wood pewee and rufous-sided towhee, were significantly more associated with upland forest that with wetland forest. Three species, common yellowthroat, Canada warbler, and northern waterthrush, were either significantly associated with wetland forest or were always recorded in this habitat type. Vegetation variables most often associated with the occurrence of common species included those related to forest overstory trees and a well-developed shrub layer. Buffer strips around forested wetlands should emphasize greater width to preserve habitat for forest interior wetland birds that are Neotropical migrant species and to maintain populations of upland dependent species, if these are management goals.

A buffer width model is offered which is based on four factors: (1) habitat suitability; (2) wildlife spatial requirements; (3) access to upland and/or transitional habitats; and (4) noise impacts on feeding, breeding, and other life functions. Suggested buffer widths range from 32 m to 100 m, or more in the case of threatened or endangered species. The final buffer should be the minimum width necessary to satisfy all requirements of the wildlife species that we wish to include in a management plan.

I. INTRODUCTION

A. The nature of non-point source pollution

The movement of pollutants from terrestrial environments into water bodies is a critical threat to water quality in many areas, including Narragansett Bay. Terrestrial land uses can deliver substantial loads of sediments, nutrients and toxic compounds into water bodies, leading to eutrophication, sedimentation, and biological decline. Understanding the terrestrial sources of pollutants, and the biological, chemical and physical factors affecting their fate and transport in terrestrial environments is essential to the maintenance of water quality for drinking supplies, navigation, fisheries and recreational uses.

Extensive basic and applied research has addressed specific "point source" pollutants over the last 20 years, and many advancements have been made towards their control. Non-point source (NPS) pollutants have proven more difficult to study and control however. These pollutants have multiple, diffuse sources and are affected by a wide range of chemical, physical and biological factors as they travel across the landscape (Cain et al. 1989). Small, diffuse non-point sources can multiply and interact to cause significant degradation of ground and surface water bodies however, and these pollutants have become proportionately more significant as point source controls have increased (Duda 1989). The diverse origins of NPS pollutants greatly complicate their control because a wide range of control strategies are required, and these controls must be applied in a total landscape or watershed context (Bedford and Preston 1988, Whigham et al. 1988). Many of these controls are expensive and difficult to justify at a local scale, given the diffuse and often poorly defined nature of NPS pollutants. Furthermore, landscape and watershed scale approaches to control are seldom attempted in scientific studies and are difficult to implement in a legislative context. Control of NPS pollution thus provides significant challenges to scientists and policy makers.

NPS pollutants can be classified based on their source, their chemical nature or their mode of transport across the landscape. In a heterogeneous landscape, NPS pollutants from agriculture, industrial areas, highways and suburban development often become highly mixed, making it difficult to pinpoint original sources. Chemically, NPS pollutants can be conveniently classified as nutrients, metals or organics. These pollutants can either be transported primarily by water flowing across the surface of the land (surface runoff), or by subsurface flow (groundwater). Different chemical types of pollutants with different modes of transport are affected by different physical, chemical and biological processes as they move across the landscape and thus pose distinct challenges for control (Hemond and Benoit 1988).

Control of NPS pollution requires a mixture of engineering approaches along with preservation, maintenance and augmentation of pollutant mitigation processes inherent in the environment. Surface runoff is frequently controlled by engineered drainage systems and detention basins that reduce the erosive power of runoff and allow for settling and stabilization of waterborne pollutants. Runoff control systems are often dependent on and/or are designed to augment natural infiltration and biological degradation processes present in soil. Groundwater borne pollutants are more difficult to control since subsurface flow is difficult to isolate and treat. Biological and chemical pollutant degradation mechanisms in the subsurface are poorly understood and are not readily amenable to engineering solutions.

B. Vegetated buffer strips

Vegetated buffer strips (VBS) are an example of a NPS pollutant control mechanism that relies heavily on pollutant degradation mechanisms inherent in the environment. VBS are defined as "small strips of grass or other vegetation that are used to trap pollutants moving from land areas before they enter water bodies (SCS 1989)". VBS potentially can serve to intercept pollutants moving in both surface runoff and subsurface flow, and can facilitate a variety of biological and chemical pollutant attenuation mechanisms. Two types of VBS have potential to control NPS pollutants:

a) "Engineered" VBS are designed and created to reduce the transport of surface water contaminants from low density of development areas. These strips require land grading, the use and maintenance of selected plant materials, and the construction of some type of strucutre immediately upgradient of the strip to minimize channelization and erosion of the VBS. For regulatory purposes, research on "engineered" VBS needs to be directed towards improving pollutant removal through improvements in the design and maintenance of these strips.

b) "Natural" VBS represent the undisturbed ecosystems which occur between upland land uses and surface water bodies. For regulatory purposes, research on "natural" VBS needs to focus on identifying those landscape features which should be protected from development due to their inherent values for water quality maintenance and/or wildlife habitat.

Despite the emerging widespread use of VBS, there are several unresolved scientific issues relating to their effectiveness for controlling both surface runoff and groundwater pollutants (Hayes et al. 1988). Understanding of the physical processes that intercept pollutants, and of the chemical and biological processes that degrade pollutants in VBS are incomplete. Generating this understanding is essential for evaluation of the effectiveness of VBS in different situations and for developing management strategies to enhance their performance. The value of VBS for aesthetic purposes and for the maintenance and stimulation of wildlife in the landscape are also poorly characterized.

The concept of active maintenance of buffer zones for water quality protection originated from research that found that strips of riparian forest vegetation were important in maintaining stream water quality in areas of intensive agriculture (Karr and Schlosser 1978, Lowrance et al. 1984, Jacobs and Gilliam 1985, Peterjohn and Correll 1985). Riparian strips were found to effectively impede surface runoff moving out of agricultural fields, reducing sediment delivery to water bodies and increasing infiltration of surface flow. Soluble pollutants in both surface and subsurface flow moving through riparian zones were found to be subject to plant uptake, microbial degradation and chemical immobilization by soil particles. Major unresolved questions relating to the effectiveness of riparian filters center around the long term fate of pollutants trapped in sediments, soil and vegetation in the zones and to the effects of pollutants on biological resources in the buffer. The latter question has increased in importance as interest (and legislation) in wetland preservation has increased in recent years since riparian zones are often dominated by wetland ecosystems.

C. Factors controlling the effectiveness of VBS

For VBS to be effective NPS pollutant control mechanisms they must physically intercept pollutants and then either chemically or biologically remove or degrade them. "Engineered" VBS are designed to reduce water velocity, causing sediments to be deposited (Dillaha et al. 1989, Brinson 1988). Physical interception of surface runoff is complicated by the strong tendency of water to move in discrete channels. Such channelization can substantially reduce pollutant attenuation by VBS as water and pollutants can rapidly flow through the channels into either receiving water or wetlands. Once begun, channelization tends to increase in severity. Some type of engineered control system is necessary to insure that the erosive, channelizing force of runoff is dissipated before runoff enters the VBS. If uniform, "sheet flow" is achieved, sediments will be deposited and soluble pollutants will be subject to biological and chemical attenuation. For sheet flow to occur, land shaping activities coupled with revegetation and flow spreaders are required. An unmanaged natural riparian landscape is generally not capable of inducing sheet flow from storm runoff.

The ability of upland VBS to physically impede groundwater flow is low. Indeed, the major control mechanism for surface runoff in VBS is to increase infiltration of surface flow into soil, stimulating movement of soluble pollutants into groundwater (Schwer and Clausen 1989). The ability of both engineered and natural VBS to affect groundwater pollutants is dependent on the ability of plant roots, microorganisms and chemical binding processes to be active in the saturated zone through which groundwater flows (Hemond and Benoit 1988). In upland areas, groundwater is usually well below the area of highest tree root and microorganism density (Cooke and Cooper 1988, Pinay and Decamps 1988). In wetlands, groundwater approaches the surface, creating a high potential for biological and chemical attenuation of pollutants (Nixon and Lee 1986).

Chemical mechanisms of pollutant attenuation that operate in VBS arise from the ability of soil mineral and organic components to absorb certain chemical species. Clay and organic matter surfaces contain negative charges that can absorb cations and many polar organic compounds. Most toxic metals are cations as is ammonium (NH_4^+), a major inorganic form of soil nitrogen (N). Cation absorption by soil is a dynamic process however, and any cations absorbed on a soil particle can be displaced and released to the soil solution. Furthermore, the cation absorbing capacity (cation exchange capacity, CEC) of soil is finite, and is controlled by the amount of clay and organic matter present. Clay and organic matter are in short supply in Rhode Island upland soils in general, and in subsurface soils in particular.

Biological pollutant attenuation mechanisms in VBS include plant uptake of nutrients and microbial processing of nutrients, metals and organics. Plant uptake of nutrients is dependent on the ability of plants to intercept and remove nutrients from either surface or subsurface flow. Plants differ greatly in their selectivity for particular nutrient forms and in the rate at which they take up nutrients. More importantly, nutrients trapped in plant tissues can later be released back into the soil solution as these tissues decompose (Nixon and Lee 1986). Storage of nutrients in structural tissues of trees (Ehrenfeld 1987), or in grass tissues that are later harvested (Brown and Thomas 1978) represent effective pollutant removal mechanisms. Clearly, some form of plant community management is necessary to maintain an effective plant uptake sink for nutrients in VBS.

Microorganisms have the ability to degrade organic compounds as food resources and to absorb (immobilize) nutrients and metals into their tissues to support growth. Microbial immobilization is reversible; nutrients that are absorbed can later be released, or mineralized, depending on the amount of nutrient available in soil (Hemond and Benoit 1988). Nitrate (NO_3^-), the most mobile form of N, can be converted to N_2 gas by certain microorganisms that respire NO_3^- in the absence of oxygen. Wet (oxygen poor), organic rich wetland soils are thought to be excellent sites for this process (denitrification), but its significance in wetland soils may be greatly overestimated (Bowden 1987). Microbial processes in subsurface soils are poorly understood but are likely strongly inhibited by a lack of organic carbon to support growth (Smith and Duff 1988, Parkin and Meisinger 1989). Research is needed to establish if microbial processes can significantly contribute to pollutant attenuation in VBS and if strategies can be devised to favorably manage these processes in the environment.

While most research on NPS pollutants has focused on field scale dynamics of specific pollutants and control mechanisms, a larger scale approach is needed when considering the cumulative impacts of land use changes on large water bodies such as Narragansett Bay (Preston and Bedford 1989). The use of VBS must be considered in relation to upstream and downstream sources of pollutants and biological resources. VBS and other NPS control mechanisms must be considered in the context of total watershed management since receiving water quality is the product of integrated, watershed scale factors.

A major landscape scale issue concerns the role of upland VBS as buffers for wetlands versus the role of wetlands as buffers for streams, lakes and coastal water bodies. Protection of wetlands is explicitly mandated by both federal and state legislation, yet the role of wetlands as landscape scale pollutant attenuation mechanisms has been extensively studied (Lowrance et al. 1984, Jacobs and Gilliam 1985, Pinay and Decamps 1988, Cooke and Cooper 1988, Hemond and Benoit 1988, Neely and Baker 1989) and is frequently cited. Wetlands are effective as buffers because pollutants in both surface and subsurface flow come into contact with surface soil and vegetation, maximizing the potential for biological and chemical attenuation of pollutants (Nixon and Lee 1986). Given that it is difficult to obtain this type of contact in upland areas (especially for subsurface flow), the development of VBS for wetland protection is problematic. Roman and Good (1985) presented a comprehensive approach for establishing upland buffers for wetlands in the New Jersey pinelands but they stressed that the mechanisms operating in upland buffers were not well characterized, and that much further research was required. The ability of upland areas to act as buffers is a critical question. We must determine if we will need to implement stronger controls on upland pollutant sources to protect wetlands, or if we will rely on wetlands to buffer receiving water bodies from upland land uses (with the potential for wetland degradation). Further, the upland/wetland transition zone may be critical for water quality maintenance in the landscape, and these zones may need to be re-evaluated if water quality is to be considered in wetland delineation and regulation efforts.

Landscape analysis of VBS must also consider aesthetic and wildlife habitat values of buffer areas. The maintenance of open space and wildlife (including rare and/or endangered species) can be advanced by the use of VBS for water quality maintenance, and can provide additional justification for their implementation as part of watershed management plans (Brown et al. 1987). Quantitative evaluation of aesthetic values is problematic however since it is based on subjective factors, and there are no generally accepted criteria or models currently available. The wildlife value of buffers is difficult to assess since wildlife habitat suitability is affected by many factors that vary widely in different types of VBS. Food, nesting and roosting resources, and buffer width and edge, will control wildlife abundances and densities in VBS. Research in existing VBS is required to assess their value as wildlife habitat.

D. Goals of this research

The goal of our study was to provide information on the suitability of any particular piece of ground as a "buffer strip" for water quality protection and wildlife habitat given information on soils, vegetation, geomorphology and surrounding land uses. For the water quality studies, our approach was to combine intensive, site-specific measurements of pollutant removal from surface and subsurface flow in buffer strips with measurements of a "microbial index" of buffering capacity at a large number of sites differing in soils, vegetation, geomorphology and surrounding land uses. The intensive site-specific measurements of actual buffering activity will serve to calibrate this index. The surface removal studies were part of ongoing USDA-SCS and Rhode Island Water Resources Center funded research projects and some of the data and text in chapters I and II can also be found in reports to those agencies. For the wildlife studies we determined species richness of birds and herpetofauna and described wildlife habitat parameters in buffer strips. We then developed a model to prescribe buffer requirements to protect wetland-dependent wildlife species.

II. SURFACE REMOVAL STUDIES

A. Introduction

Engineered VBS were first developed as formal NPS pollutant control mechanisms for control of sediment and nutrient outputs in surface runoff from agricultural operations (Dillaha et al. 1989). VBS were designed to reduce the velocity of runoff to encourage deposition of sediment and infiltration of water. Design and site criteria for given sediment loads have been established and are being formalized as a "practice standard" by the SCS (1989).

Although VBS have proven abilities to trap sediments moving from agricultural areas, their ability to remove soluble pollutants, and their long-term ability to remove sediments is not well established (Magette et al. 1987). For sediment control, it is essential that the tendency of flowing water to "channelize" be contained. If water flows in discrete channels, the VBS will be eroded and sediment deposition and water infiltration will not occur. In a survey of 33 VBS in Virginia, Dillaha et al. (1989) found that the majority of VBS were ineffective for water quality improvement of surface runoff because most flow was channelized. In addition to channelization, long term sediment accumulation can reduce the ability of VBS to slow flow velocities and stimulate deposition (Lee et al. 1989). Accumulated sediments and sediment-borne pollutants in VBS become potential sources of pollutants that can be released if the VBS is disturbed by large storms, agricultural activities, construction, or other events (Brinson 1988).

Soluble pollutants moving in surface runoff are subject to biological and chemical attenuation mechanisms in VBS. Cations (including metals) can be adsorbed on cation exchange sites on clay and organic matter, and nutrients are subject to plant uptake, microbial immobilization and denitrification. These mechanisms depend on establishing contact between pollutants and soil and/or plant roots. This contact time is reduced if infiltration and percolation is rapid. There is thus concern that designing and managing VBS to maximize infiltration actually transfers a surface runoff pollutant problem into a groundwater problem (Schwer and Clausen 1989). Concern about transfer of soluble pollutants to groundwater in VBS is especially important in areas of highly permeable soils and low slope such as Rhode Island (Valiela and Costa 1988, Winter 1988). It is important to recognize that VBS are not likely to be used as sole NPS control devices for surface runoff in Rhode Island. Diffuse, low energy surface runoff (typical of agricultural fields) is rare in Rhode Island. High energy/high volume runoff (typical of urban and suburban areas), which is a common problem in Rhode Island, cannot be controlled solely by VBS (Magette et al. 1987, Scott 1988). In this area, VBS are more likely to be found downstream of detention basins or other runoff and sediment control devices. The dynamics of soluble pollutants in these VBS is thus of critical importance given Rhode Island conditions.

In this study, our goal was to focus on aspects of surface runoff dynamics in VBS that have not been well studied and that have relevance to local conditions. Our objective was to quantify processes that remove NO_3^- from infiltrating and percolating water, and to understand the factors controlling these processes across the landscape. We compared the ability of 10 different grass VBS to retain added N. We analyzed how much N moves from VBS into groundwater using lysimeters and measured denitrification and N immobilization in selected grass and forest VBS. Our results have application to assessing the ability of VBS to attenuate pollutants moving in surface runoff.

B. Methods

Field plots

This experiment was conducted at the University of Rhode Island's Peckham Farm, in Kingston, Rhode Island. Experimental VBS were established as part of an ongoing USDA SCS study on soil classified as a Typic Dystrochrept. The surface horizon of this soil contained 0.92% organic carbon, 2.53% organic matter, and a pH of 5.89. The full scope of the buffer strip study consisted of 10 different species of grasses, each replicated three times, making a total of 30 plots (Table 2-1). Each plot measured 3 m x 5 m in size, with a 0.70 m alleyway between alternate plots. The switchgrass and big bluestem were both propagated in a greenhouse and when they had grown to a height of approximately four inches, were then transplanted into their respective plots. The remainder of the grasses were grown from seed in the field. Situated above the grass plots was a 25 m x 100 m oat field which was graded to a 2% slope to provide runoff onto the grass plots.

On April 20, 1988, the grass plots, along with the oat field, were treated with 33 kg N/ha in the form of urea. Additional treatments with urea were confined to the grass plots. On July 22, 1988, 96 kg N/ha was added as a top dressing, and on September 30, 1988 another 48 kg N/ha was added to boost root and rhizome production for the winter. The plots were sprayed on June 8, 1988 with 0.28 kg/ha of Buctril and .07 kg/ha of Banvel, both of which are for broadleaved weed control. Additional spot treatment for *Agropyron repens*, was accomplished using a 2% glyphosate solution. The plots were mowed four times to a height of three inches to suppress weeds, allow the grasses to fill the plots, and to prevent the grasses from going to seed.

Ceramic lysimeter plates (27 cm diameter, Morton et al. 1988) were installed below each grass plot to collect water percolating below the root zone as part of the ongoing SCS study. The plates were installed in triplicate below each of the ten species at an average depth of 70 cm. Leachate samples were collected for selected events in late 1988 and have been collected for every percolation event since March 10, 1989 as part of the SCS study. Nitrate in leachate samples was quantified using an ion chromatograph. Loading estimates were derived from measured flux per event and concentrations observed during the event.

For microbial process studies, only two of the grass treatments, common reeds canarygrass (*Phalaris arundinacea*) and tribute tall fescue (*Festuca arundinacea*) were used. The reeds canarygrass was chosen because it is well adapted to a moist environment (such as riparian areas) and has the ability to uptake large quantities of nutrients. The tall fescue was chosen because it is a common plant used in many other studies. We also located microbial process study plots in well and poorly drained riparian forest sites adjacent to the grass VBS study area. The poorly drained site was situated in a soil that was classified as an Aquic Dystrochrept, with a depth to mottles of 25 cm. The surface horizon of this soil contained 8.5% organic carbon, 0.32% total N, and a pH of 3.52. In the well drained site, the soil was classified as a Typic Dystrochrept, with a 66 cm. depth to mottles. This soil contained 5.1% organic carbon, 0.25% total N, and a pH of 4.23. Sub-plots of the forest sites were treated with lime (20 lbs/ft^2 in the poorly drained, 10 lbs/ft^2 in the well drained) to test for pH limitation of microbial N processes.

TABLE 2-1

Grass species and plot numbers in vegetative buffer strip experiment. Plots established as part of USDA SCS funded research.

Grass species	Plot numbers
Big bluestem	10, 15, 27
Bromegrass	4, 11, 24
Garrison creeping foxtail	2, 19, 30
Kentucky bluegrass	8, 16, 25
Orchardgrass	7, 20, 22
Perennial ryegrass	1, 13, 23
Reeds canarygrass	5, 17, 29
Sweet vernalgrass	9, 14, 26
Switchgrass	3, 12, 21
Tall fescue	6, 18, 28
Forest plots	
Well drained - oak, maple	WDF
Poorly drained - maple	PDF

Microbial process experiments

These experiments were done as part of a research project funded by the Rhode Island Water Resources Center. Denitrification was measured using soil core techniques described by Groffman and Tiedje (1989a). Cores of 15 cm depth and 2 cm diameter were removed from soil, placed in plexiglass tubes and sealed with rubber serum stoppers. Cores were subjected to a variety of experiments.

Experiment 1. The first experiment began on July 6, 1988 and consisted of three parts carried out over a three day period.

Part 1:

The first part of the experiment measured *in situ* rates of denitrification and N_2O production in the different plots. Ten soil cores were sampled from each of the forest plots and each of the grass treatments. These cores were immediately capped and 5 ml of acetylene was added to every other core. Acetylene was added to inhibit the final step in the denitrification process, allowing us to quantify denitrification rates by measuring the accumulation of nitrous oxide (N_2O) in the sealed cores (Yoshinari and Knowles 1976). The remainder of the cores received 5 ml air. To ensure a proper diffusion of the acetylene throughout the core sample, the atmosphere in each core was mixed by pumping five times with a 60 cc syringe. These cores were then placed in the ground and left to incubate for a total of 6 hours. A 3 ml gas sample was taken from each of the cores after two and six hours of incubation. The samples were placed into 10.25 x 65 mm evacuated glass tubes and were returned to the laboratory. After taking the six hour sample, the caps were removed and the cores brought to the laboratory where they were stored at 4^{o} C overnight.

Part 2:

In part 2 of the experiment, the cores were incubated under anaerobic conditions to test for oxygen limitation of denitrification. The following day, the cores were recapped using the rubber septum stoppers and were alternately evacuated, by means of a vacuum pump, and refilled using 99.999% pure nitrogen gas. All cores received 5 ml acetylene, were mixed and were incubated at 22^{o} C for 6 hours, with gas samples taken at two and six hours. After the second sampling, the caps were once again removed from the cores, and the cores were stored at 4 C overnight.

Part 3:

In part 3, cores were amended with either NO_3^- or NO_3^- and glucose to test for NO_3^- and carbon limitation of denitrification. The following day, five cores from each of the plots were subjected to treatment with a 100 ppm NO_3^--N solution, while the other five cores from each of the plots were treated with a 100 ppm NO_3^--N and 1000 ppm glucose solution. Cores were sealed, made anaerobic and incubated and sampled as described above.

Experiment 2. The second experiment was conducted on the three days between July 25-27, 1988. The objective was to assess how much denitrification would occur following an addition of NO_3^-, thus simulating a runoff event. Ten cores were taken from each of the reeds canary grass, tall fescue, well drained forest, and poorly drained forest sites. Five cores from each of these sites were then treated with 10 ml of a 100 ppm NO_3^--N solution, and the other five cores were treated with 10 ml of a 100 ppm NO_3^--N and 1000 ppm glucose solution. The cores were capped, amended with acetylene and incubated and sampled as described above. After the six hour sample was taken, the cores were uncapped and stored at 4 C overnight. The cores were resealed, and incubated on the following two days to follow the response of the cores to the amendments over a three day period.

Experiment 3. This experiment was designed to determine if low pH limited denitrification in the forest plots. On August 9, 1988, two new 30 m^2 forest plots were created, one located in the poorly drained section and the other located in the well drained section. These plots received 20 lbs/ft^2 and 10 lbs/ft^2 of lime, respectively.

Denitrification rates were measured in the well drained forest on September 6, 1988, and in the poorly drained forest on September 23, 1988. The experimental procedure consisted of taking 20 cores from each of the four forest plots, and returning these cores to the lab. Each set of 20 cores were then broken into 4 sets of 5, with each set receiving one of the following treatments; 10 ml distilled water, 10 ml of a 100 ppm NO_3^--N solution, 10 ml of a 100 ppm NO_3^--N and 1000 ppm glucose solution, and the forth set receiving no amendment. All cores were then capped, amended with acetylene and incubated and sampled as described above.

Before any of the cores from experiments 1, 2 and 3 were discarded, their weight and length were recorded. The head space of each core was then measured by calculating the difference between the pressure within the core, as measured with a pressure transducer, at atmospheric pressure and with 5 ml air added. Gas samples were analyzed for N_2O on a Tracor model 540 gas chromatograph equipped with two electron capture detectors and four 2 m columns packed with Porapak Q. The data were expressed on an areal basis using bulk density values determined on each core. Results were analyzed using the ANOVA procedure in the SAS statistical package. A duncans multiple range test and an LSD test of significance with a 0.05 confidence interval, were used to determine differences between treatments both within and between plots.

Experiment 4. In this experiment, denitrification was measured over an 8 day period in NO_3^- amended (4 µg/g soil) soils held in 946 ml mason jars in the laboratory. Levels of mineral N in soil were also measured to assay immobilization and re-mineralizataion of the added NO_3^-. Each jar contained 100 g soil, and denitrification and soil mineral N levels were measured 1 day before addition of NO_3^- and 1, 3 and 8 days following the addition. Denitrification was measured by sealing the jar with a lid containing a black rubber septum, adding 10 ml of acetylene, and taking gas samples at 2 and 6 hours following sealing. Acetylene was removed from the jars by evacuating and refilling the jar with air three times. The jars were left unsealed between denitrification measurements. Mineral N was extracted from 5 g subsamples with 2 N KCl and analyzed on an Alpkem RFA 300 continuous flow analyzer.

Experiment 5. To further assess the factors controlling denitrification in natural forest VBS, we measured denitrification enzyme activity (DEA) in 14 forest VBS in the Hunt-Potowomut watershed (Figure 2-1). At 7 locations (4 along Sandhill Brook, 1 along Frenchtown Brook, 1 along Mauney Brook, 1 along Fry Brook) we collected soil samples (0-15 cm) in upland (moderately well drained soils) and wetland areas (poorly or very poorly drained soils) during December 1988 and January 1989. DEA was measured using the anaerobic assay described by Smith and Tiedje (1979). Sieved soils (3 replicates per site) were amended with nitrate (200 µg/g), dextrose (40 µg/g) chloramphenicol (10 µg/g) and C_2H_2 (10 kPa) and incubated under shaken, anaerobic conditions for 90 minutes. Gas samples were taken at 30 and 90 minutes, stored in evacuated glass tubes, and analyzed for N_2O as described above. Soils were made anaerobic by repeated evacuation and flushing with oxygen-free gas.

Figure 2-1. Sampling locations for experiment 5. At each of the seven locations, samples of surface soil were taken in upland (moderately well drained or somewhat poorly drained) and wetland (poorly drained or very poorly drained) soils.

This assay measures the maximum potential denitrification enzyme activity present in soil at the time of sampling (Groffman 1987).

C. Results

The average concentrations of NO_3^- in leachate from the 10 grass species ranged from 0.9 to 31.6 mg/L. Loading rates of NO_3^- to groundwater ranged from 2.2 to 106.1 kg/ha. The 10 grasses can be broken into four groups based on their loading rates (Table 2-2). All grasses received 177 kg N/ha as fertilizer. N removal efficiencies (calculated as N not leached/total fertilizer N input) for the grasses thus ranged from 40 to 99% of the N applied.

In experiment 1, aerobic, in situ rates of denitrification and N_2O flux were insignificant in all plots (Table 2-3). Anaerobic, unamended rates were also very low, but were somewhat higher in the poorly drained forest plot than in the other plots (Table 3). Nitrate amended denitrification rates were higher ($p < 0.05$) than either aerobic or anaerobic unamended rates in all plots (Table 2-3). Nitrate and carbon amended rates were higher ($p < 0.05$) in all plots other than the reeds canary grass (Table 2-3).

In experiment 2, denitrification rates in NO_3^- and NO_3^- and carbon amended cores were in the order of: tall fescue > reeds canary grass > poorly drained forest > well drained forest (all differences $p < 0.05$, Table 2-4). Nitrate and carbon amended rates were higher than NO_3^- only amended rates in all plots, but the differences were significant in the poorly drained forest and reeds canary grass plots only (Table 2-4). The NO_3^- amendment in experiment 2 simulated a 31.8 kg N/ha addition. Denitrification N removal efficiencies for the different plots, (calculated as denitrification rate/total N addition), ranged from 1 to 50% per day (Table 2-4). Results from experiment 3 indicated that the lime amendment significantly increased soil pH in the forest plots, but denitrification did not increase accordingly (Table 5).

When NO_3^- was added to soils held in mason jars in the laboratory (experiment 4), soil from the reeds canarygrass plot showed the strongest denitrification response followed by the tall fescue and well drained forest plots (Figure 2-2). The poorly drained forest soil showed no denitrification response to added NO_3^- in this experiment. All soils other than the poorly drained forest were able to immediately absorb the added NO_3^- (Figure 2-3). However, levels of mineral N increased in all soils over the eight day incubation.

In experiment #5, we observed strong relationships between soil moisture and DEA ($r^2 = 0.56$, Figure 2-4) and between soil pH and DEA ($r^2 = 0.43$, Figure 4). In a multiple regression analysis, soil moisture and pH accounted for 78% of the variability in DEA among the 14 sites sampled.

TABLE 2-2

Nitrate leaching losses (kg N/ha) and approximate N removal efficiencies for grasses in vegetative buffer strip experiment, Spring 1988 - Spring 1989. Data collected as part of USDA SCS funded research.

Grasses	Nitrate leaching losses	N removal efficiency
orchardgrass, sweet vernalgrass	2.0 to 10.0 kg/ha	94-99%
tall fescue, creeping foxtail	10.0 to 25.0 kg/ha	86-94%
perennial ryegrass, big bluestem	25.0 to 50.0 kg/ha	72-86%
kentucky bluegrass, reed canarygrass	> 50.0 kg/ha	<70%
bromegrass, switchgrass		

TABLE 2-3

Denitrification rate (g N ha^{-1} d^{-1}) in soil cores in response to amendments, 880706[1]

	Well drained forest	Poorly drained forest	Tall fescue	Reeds canary
Aerobic				
No amendment	-4.6^{c}	-8.7^{c}	-4.0^{b}	2.0^{b}
C_2H_2	-21.3^{c}	-16.3^{c}	-9.0^{b}	-2.0^{b}
Anaerobic with C_2H_2				
No amendment	1.1^{c}	13.1^{c}	1.0^{b}	1.0^{b}
NO_3^-	1306^{b}	1402^{b}	17208^{a}	15208^{a}
NO_3^- and glucose	2155^{a}	2951^{a}	21702^{a}	15819^{a}

[1]Values followed by different supercripts within columns are significantly different at $p < 0.05$ in a one-way analysis of variance with a Duncan's multiple comparisons test. The tall fescue and reeds canarygrass plots had significantly higher ($p < 0.05$) denitrification than the forest plots when all treatments were combined, using the same statistical tests.

TABLE 2-4

Denitrification (g N ha^{-1} d^{-1}) response and N removal efficiencies for soil cores treated with simulated runoff containing either NO_3^- or NO_3^- and glucose, 880725[1]. Total NO_3^--N addition = 31,800 g N/ha.

Plot	NO_3^- amended		NO_3^- and glucose amended	N removal efficiency (% N denitrified/day) NO_3^- amended	N removal efficiency (% N denitrified/day) NO_3^- and glucose amended
Well drained[c] forest	311		408	1	1.3
Poorly drained[bc] forest	365	*	1439	1.2	4.5
Tall fescue[a]	7889		16186	25	51
Reeds canary[b] grass	4537	*	9139	14	29

[1]Plots followed by different supercripts showed significantly different denitrification activity over both amendments at $p < 0.05$ using a one-way analysis of variance with a Duncan's test of multiple comparisons.

*Indicates significant difference between NO_3^- and NO_3^- and glucose treatments at $p < 0.05$ with a t-test.

TABLE 2-5

Denitrification (g N ha^{-1} d^{-1}) in soil cores from limed and unlimed forest plots treated with simulated runoff, September 1988

Plot	pH	NO_3^- amended	NO_3^- and glucose amended
Poorly drained forest			
Limed	4.79	365	1102
Unlimed	3.52	336	1606
Well drained forest			
Limed	4.94	31	72
Unlimed	4.23	101	685

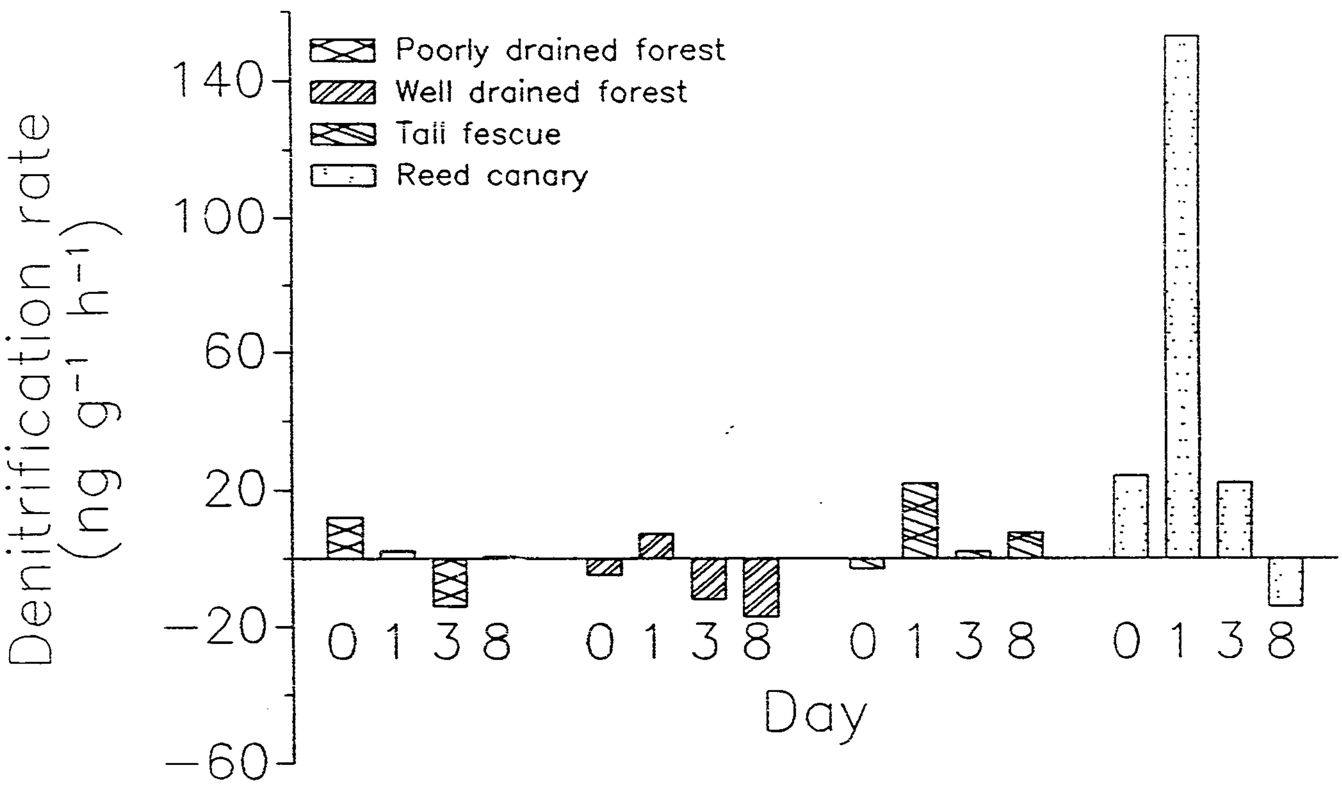


Figure 2-2. - **Denitrification response to added NO_3^- (4 ug/g soil) over 8 days in soil held in mason jars.**

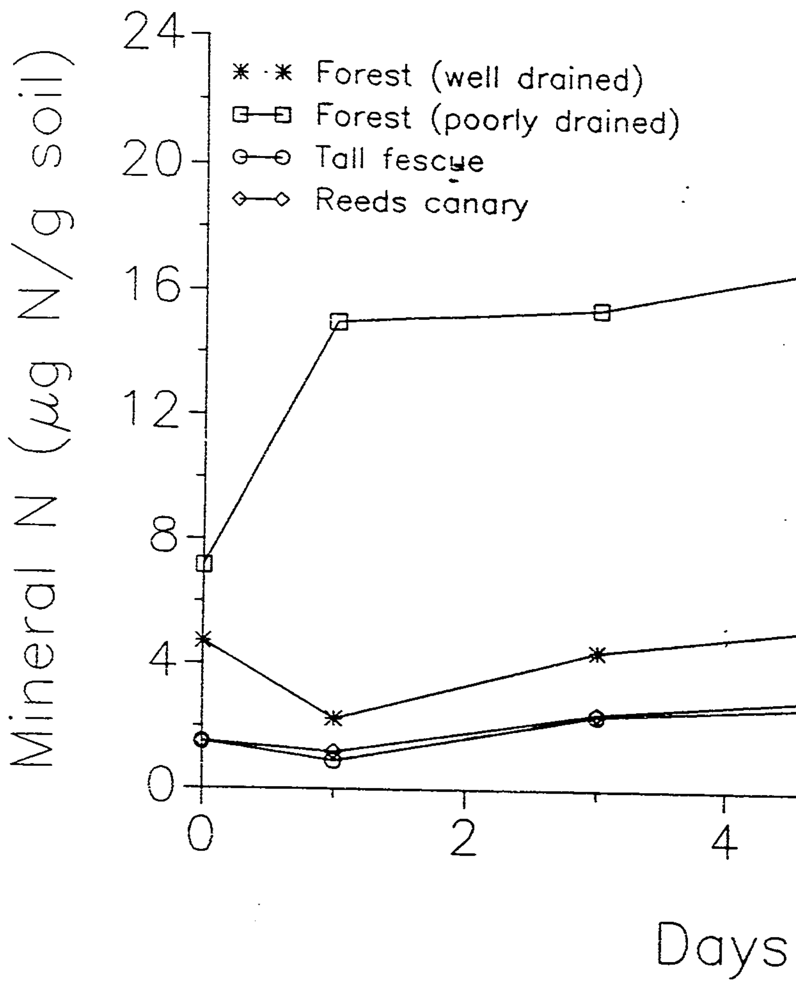


Figure 2-3. - **Mineral N in soils held in mason jars that received 4 ug/g soil NO_3^--N immediately before day 1 sampling.**

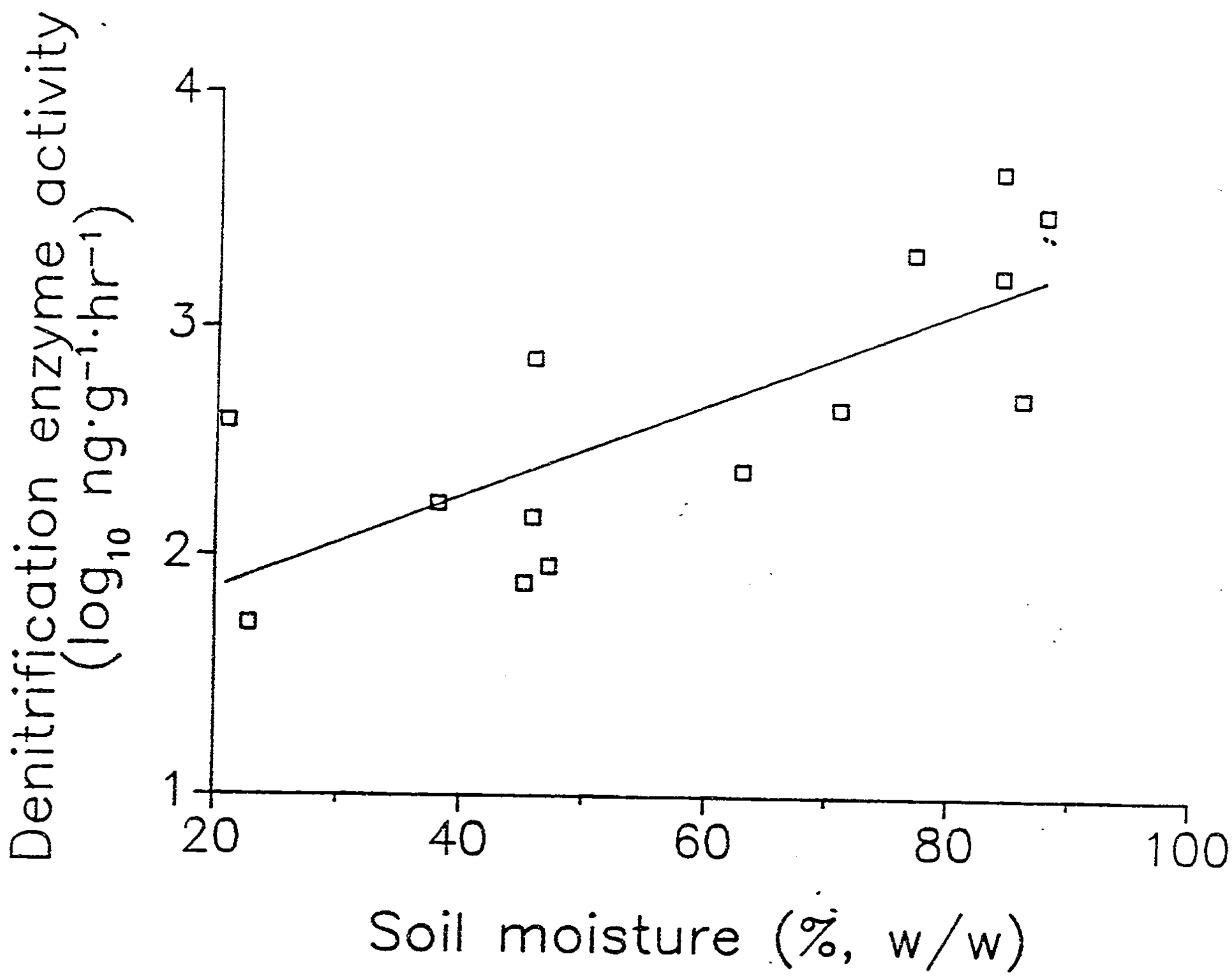


Figure 2-4. Log_{10} denitrification enzyme activity versus soil moisture in surface soils from 14 sites in the Hunt-Potowomut watershed.

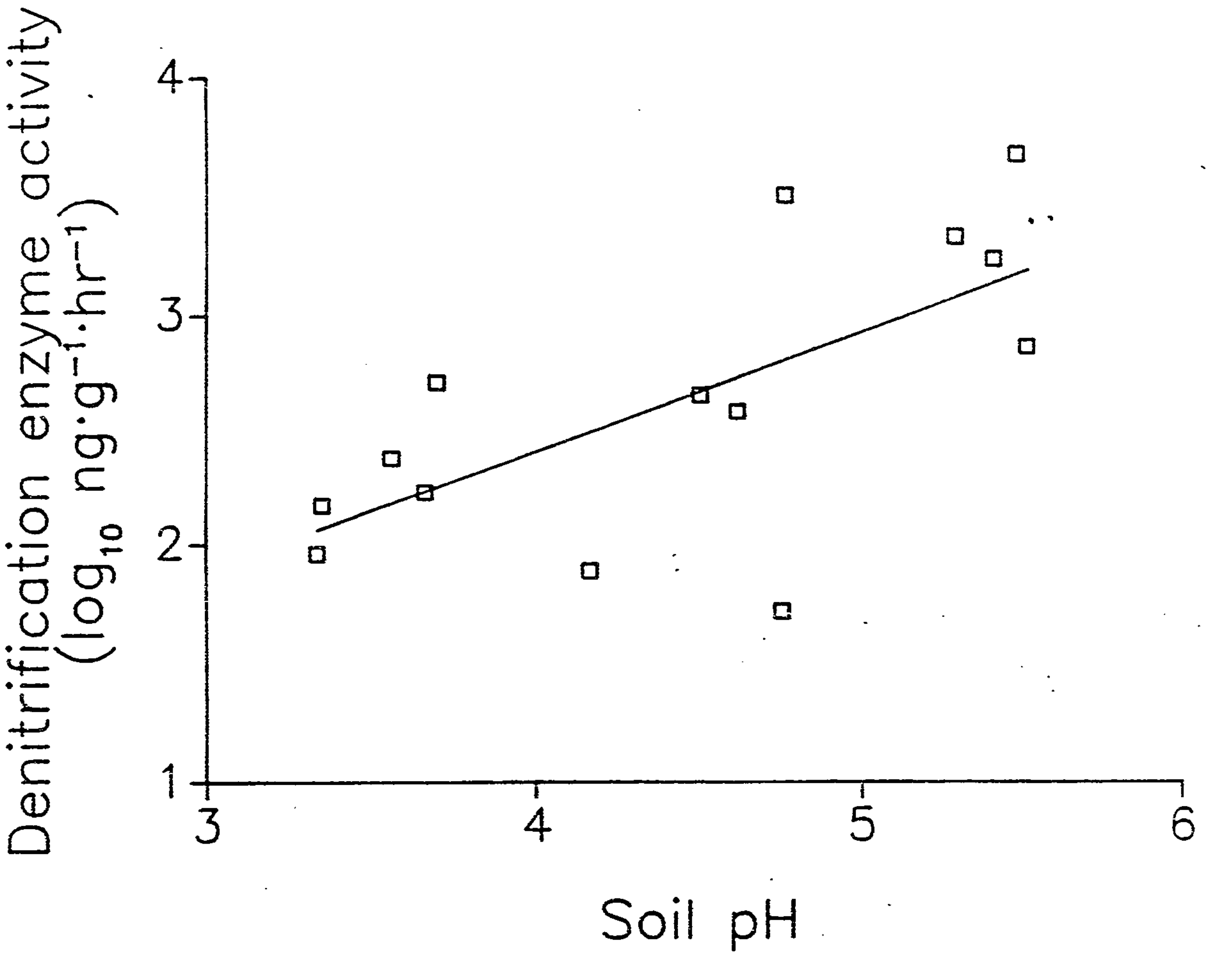


Figure 2-5. Log_{10} denitrification enzyme activity versus soil pH in surface soils from 14 sites in the Hunt-Potowomut watershed.

D. Discussion

The results from the lysimeters suggest that the different grasses have dramatically different abilities to take up N. It is important to note that all grasses received 177 kg N/ha as fertilizer and therefore all grasses demonstrated a significant ability to take up N. However, N removal efficiencies (calculated as N not leached/total N fertilizer input), were rather low in many cases. The amount of N added was greater than what normallly occurs in surface runoff, and efficiencies would likely be higher at lower rates of N additions. These results were obtained during spring and summer, when N removal efficiencies should been highest due to high plant uptake of N and low leaching during this period. Results from a related SCS study conducted at the URI agronomy farm with VBS established and fertilized during winter found much lower N removal efficiencies by grass. In the winter study, tall fescue-perennial ryegrass VBS were established and fertilized with 33 kg N/ha in September. An average of 21 kg N/ha (64%) was leached during the winter months. These results suggest that winter N removal efficiencies by VBS will be much lower than summer removal efficiencies. It is also important to note that the ability of grasses to take up N fertilizer does not necessarily predict their ability to remove N from runoff. N moving in runoff will travel either across the soil surface or can percolate rapidly through macropores as "gravitational" water, which is distinct from more slowly moving "plant available" water (Brady 1974). Once the runoff event is over, percolation rates will decline (shift to micropore flow), and the potential for plant uptake of N will increase. Although our results demonstrate important differences in the inherent ability of different grasses to absorb N under Rhode Island VBS conditions, further research is needed to fully quantify their ability to remove N from infiltrating water originating from surface runoff.

The ultimate fate of N taken up by vegetation in VBS is uncertain. N in plant tissues can be released during senescence and decomposition and can be re-mineralized and released into the soil solution. Some type of plant harvest is necessary to minimize this problem (Brown and Thomas 1978). Harvest of grasses is more straightforward than for trees since it is physically easier to accomplish and it stimulates regrowth of the grasses. Tree removal, or woodlot management, is more complex to accomplish and can involve considerable disturbance to soil in the VBS. Tree regrowth is often not immediate following harvest and thus tree removal can temporarily reduce VBS performance (Hemond and Benoit 1988). Storage of N in woody tissues of trees could be considered to be a long-term sink for N. Ehrenfeld (1987) found that most N taken up by trees affected by septic tank leach field outputs went to leaves (60-100%). Wetland trees did not exhibit any ability for long term storage of N from septic leach fields in that study.

Data from experiment 1 found that in situ rates of denitrification in the plots were very low and were limited by the presence of oxygen and an absence of NO_3^- and/or glucose. Oxygen and NO_3^- control of denitrification have been reported in many studies (Tiedje 1988). However, the strong stimulation caused by glucose that was observed in the forest plots and not in the grass plots was surprising since the forest plots have high levels of organic matter relative to the grass plots. It is possible that tillage, fertilization and liming over time have increased substrate availability (both physical and biochemical) to microbes in the grass plots. Since the carbon amended incubations in experiment 1 were carried out under anaerobic conditions, the stimulation that we observed was due only to increasing the supply of substrates to denitrifiers and not to a reduction in soil oxygen levels caused by general stimulation of heterotrophic microbes as has been observed in other studies (Rice et al. 1988, Groffman and Tiedje 1988). The data suggest that carbon availability to denitrifiers may limit denitrification activity in forest soils more than has previously been thought. Struwe and Kjøller (1989) also found C stimulation of denitrification in temperate forest soils.

We expected the forest plots to have a much higher potential for denitrification than the grass plots, since forest soils (especially poorly drained forests) generally have higher moisture and organic matter levels than upland agricultural soils. In contrast to our expectations, soils from the grass plots exhibited consistently higher denitrification activity than soils from the forest plots. In experiment 1, the grass plots had higher activity than the forest plots in anaerobic incubations of soil amended with either NO_3^- or NO_3^- and glucose. In experiments 2 and 4, the grass plots had higher activity than the forest plots in response to NO_3^- additions made to simulate runoff. These results suggest that carbon availability to microbes is higher in the grass plots than in the forest plots (discussed above), and that the population of denitrifiers is bigger and/or more active in the grass plots than in the forest plots. Warwick and Hill (1989) also found that removal of NO_3^- from surface flow through forests was low.

Experiment 3 was done to test the hypothesis that low pH limited denitrification activity in the forest plots relative to the grass plots. Although we successfully raised the pH in the forest plots from 3.5 to 4.8 in the poorly drained plot and from 4.2 to 4.9 in the well drained plot (pH in the grass plots was 5.3), denitrification rates did not increase accordingly. Denitrification was actually lower in the limed plots than in the unlimed plots in most cases. This is likely due to the fact that microbes in soil are adapted to in situ physical and chemical conditions, and changing the pH thus reduced their activity (Koskinen and Keeney 1982). In the long term, raising soil pH should lead to the development of a different microbial community, with higher denitrification activity (Parkin et al. 1985). These results suggest that forested buffers in Rhode Island may not always be effective denitrification sinks for NO_3^-, and that managing these zones to increase denitrification is not simple and requires long-term study.

Results from experiment 5 suggested that wetland VBS are likely to more effective sinks for NO_3^- than upland VBS, and confirmed the importance of pH in controlling denitrification capacity in forested VBS. Regression analysis showed that the wetland sites, and the sites with highest pH had highest denitrification capacity (as indicated by DEA). Wetlands have more anaerobic conditions than uplands, and thus were expected to have higher denitrification potential. However, measurement of actual denitrification activity in wetlands has rarely been attempted and is likely much lower than potential activity (Bowden 1987). Determining the landscape scale factors that control forest soil pH (geologic setting, parent material, upland land use, etc) will be useful for assessing VBS performance across the landscape. Bartlett et al. (1979) also found that pH had an influence on the ability of red maple swamp soils to denitrify.

Denitrification N removal efficiencies calculated from experiment 2 (Table 4) were quite high in the grass plots, suggesting that up to 50% of a very large N addition (> 30 kg N/ha) could be denitrified per day. These results must be interpreted with great caution however, since they were obtained with soil cores and are not field measured fluxes. Adding amendments to confined soil cores does not allow for free drainage and thus stimulates the development of anaerobiosis in the cores and maximizes the accessibility of NO_3^- to denitrifiers. In experiment 4 (soil held in mason jars), simulated runoff was less confined and the denitrification response was less intense and was relatively brief. Measurements of denitrification in soil cores taken from field VBS that have received runoff are necessary to validate these results.

The simulated infiltration experiments were useful in several regards however. First, the role of carbon in increasing N removal efficiency in all plots is clear. The carbon effect is likely due both to increasing substrate availability to denitrifiers (discussed above) as well as to depletion of soil oxygen levels resulting from a general stimulation of heterotrophic activity. These results suggest that runoff containing high levels of available carbon (feedlot or manured field runoff for example) may be more amenable to treatment in VBS than runoff that is low in available carbon. Second, the results suggest that if free drainage can be prevented, significant denitrification can occur in VBS. Flow of runoff through the VBS could be controlled either by surface contour engineering or subsurface drainage manipulation, creating a hybrid NPS pollutant control mechanism that utilizes aspects of VBS, detention basins and artificial wetlands.

Results from the experiment 4 (mason jars) suggest that immobilization of N by soil microbes is not a reliable N removal mechanism in VBS. All soils (other than the poorly drained forest) were able to absorb a spike of 4 ppm NO_3^-, but all soils showed net N mineralization over the next 8 days. These results suggest that N that is immobilized during runoff events may later be released to the soil solution and may either be leached to groundwater or carried out of the VBS in the next runoff event. On the other hand, immobilization may be useful as a temporary sink for NO_3^-, allowing plant access to the N that is re-mineralized. This is especially important in light of the fact that N moving rapidly in runoff may not be particularly accessible to plant roots as discussed above.

E. Summary of major findings and management implications

1. Different grass species have dramatically different abilities to act as sinks for NO_3^- under Rhode Island conditions.

2. Winter removal of NO_3^- by plants or denitrification is unlikely.

3. Upland, limed, grass plots had a higher capacity to remove added NO_3^- by denitrification than either upland or wetland forest plots.

4. Low pH in some forest soils in Rhode Island limits their capacity for denitrification.

5. If flow through the VBS is controlled, denitrification can remove large amounts off NO_3^- (up to 50% of addition per day). New non-point source pollutant control mechanisms that combine aspects of VBS and artificial wetlands may be highly effective NO_3^- sinks in the landscape.

III. GROUNDWATER STUDIES

A. Statement of Problem and Objectives

Riparian buffers have been recently promoted as a land management technique for surface water quality protection (Lowrance et. al., 1984, Jacobs and Gilliam, 1985). Numerous studies have demonstrated the effectiveness of riparian buffers to reduce concentrations of selected contaminants; however, there appears to be marked differences in pollutant attenuation between buffers due to differences in soil, hydrology and vegetation. Studies are required that identify those features associated with riparian buffers that have the greatest promise for pollutant removal in order to focus future regulatory measures.

The effectiveness of a riparian buffer will vary based on a number of factors including:

(a) The mode of hydrologic transport (groundwater vs. surface water) that carries pollutants from the upland to surface water.
(b) Characteristics of the specific water quality contaminant of interest including: Solubility, affinity to sediment and possible attenuation mechanisms.
(c) The attenuation potential of a buffer including: Potential plant uptake, microbial activity, soil adsorption capacity and chemical fixation potential.
(d) The proportion of upland contaminants which travel through the active portion of the buffer before reaching surface water.

The concept of using a continuous strip of "natural" vegetation located along the shoreline to achieve water quality improvement is best suited to removing contaminants transported from upland areas by groundwater. Groundwater is often discharged along buffers abutting streams and ponds and has been shown to have the potential to interact with the rootzone of riparian vegetation before reaching surface waters (Hill, 1989). In contrast, surface runoff from residential lands usually occurs as channelized flow from storm drains. Without engineered structures at the outlet, storm water runoff will degrade a riparian buffer, creating eroded channels which hold little potential to attenuate water quality contaminants (Dillaha et. al., 1988). Reducing water quality contamination from storm water runoff has been recently investigated in Rhode Island by the Storm Water Management and Erosion Control Committee of RI DEM (Scott, 1988). The committee recommended site modifications and a variety of engineered basins as possible solutions to improve stormwater quality. Vegetated buffer strips (VBS) were recommended for selected circumstances; however, the committee recommended that land forming activities (specifically uniform grading and the installation of a level spreading device) precede the creation of the strip, suggesting that a natural buffer alone could result in channelized flow.

The focus of this study was to investigate the ability of riparian buffers to attenuate the groundwater contaminants, NO_3^-N and copper. The primary focus of the groundwater and microbial studies was N attenuation; however, at the suggestion of NBP personnel copper was added to the groundwater to gather information on heavy metal attenuation through riparian buffers. The study was conducted on both upland and wetland riparian zones. Because previous studies have demonstrated that organic matter and soil moisture can influence contaminant removal, the study focused on the relationship of soil drainage classes to attenuation. Detailed soils data are available through the RIGIS data base which could permit rapid identification of critical riparian zones if soil type can be related to pollutant removal.

Sources and Fate of Nitrate-N and Copper

Anthropogenic sources have been shown to measurably increase N concentrations in ground and surface waters. Nitrogen is a common pollutant from unsewered residential areas (Cain et al., 1989; Koppleman, 1978), agricultural regions (Keeney, 1986; Baker et al., 1975) and has been found in increasing concentrations in precipitation. NO_3^-N is a drinking water contaminant (USEPA, 1976) and has been found to be the limiting nutrient to accelerated eutrophication in coastal bays and estuaries (Ryther and Dunstan, 1971).

In determining nonpoint N loading to estuaries and coastal bays, increasing attention is focusing on both the sources and potential sinks that occur within a watershed. Most anthropogenic N in groundwater is in the NO_3^-N form, which is highly mobile in groundwater. Considerable effort has been directed towards documenting sources of NO_3^-N loading to groundwater (Keeney, 1986); however, few evaluations have been performed on the quality of groundwater entering a particular stream. Groundwater recharge represents a major portion of river flow in the northeastern U.S. where annual precipitation greatly exceeds evapotranspiration.

Nitrate can be removed from groundwater by plant uptake, microbial immobilization or by denitrification. For N removal to occur through plant uptake, groundwater must enter the rootzone during the growing season. Nutrient retention by plants may be significant during the growing season, however this potential sink is often considered temporary due to the eventual decay and release of these substances back into the environment (Bowden, 1987). Denitrification requires anaerobic conditions with an available carbon source. Generally the soil denitrification potential is expected to increase with increasing organic matter and increasing periods of saturation within the rootzone (Tiedje, 1988)

Anthropogenic additions of heavy metals from domestic and industrial activities have come to exceed the natural sources of heavy metals. Evidence of this is demonstrated by the abundance of heavy metals in surface waters and sediments near populated and industrial areas (Bender, 1989).

Copper in the cupric (Cu^{+2}) ion form has been shown to be toxic to aquatic organisms (USEPA, 1985). However, the cupric ion is readily sorbed onto surfaces of solids, particularly clays and organic matter. In addition, the cupric ion forms complexes and precipitates with such inorganic and organic compounds as humates, hydrous iron and manganese oxides (Baker and Chesnin, 1975). Copper may also be incorporated into biological systems or residues (Keeney and Wildung, 1977). Copper is not considered to be highly mobile in groundwater due to the array of constituents to which it can sorb or complex.

B. Experimental Approach

Determining the fate of a groundwater contaminant moving through a riparian buffer can be confounded by the influences of dilution as well as spatial and temporal variations of the contaminant. In this study we applied a solution of NO_3^-N and copper along with a conservative tracer (chloride) to trenches upgradient of both upland and wetland zones of a riparian buffer. The change in concentration of either NO_3^-N or copper relative to the chloride concentration in the groundwater was then observed at various depths as the contaminants moved through the buffer region. The dilution of the introduced contaminant plume was determined by the dilution of the chloride concentrations observed. A decline in the NO_3^-N or copper concentrations in excess of that caused by dilution was attributed to attenuation (Trudell, et. al., 1985).

In developed areas groundwater concentrations of both chloride and NO_3^-N can be elevated due to septic system leachate. N attenuation could be studied using changes in NO_3^-N: chloride ratios without adding a spiked solution. However, septic system leachate can flow considerable distances in a narrow concentrated plume (Cantor and Knox, 1985). If the direction or the relative concentrations of the contaminants within the plume vary with time, separating attenuation from dilution becomes difficult. Adding a contamination plume to the shallow groundwater minimizes the reliance on existing sources which can vary with time and space.

Groundwater monitoring wells were placed at two depths within soils differing with respect to water table depths (soil drainage classes). Shallow wells were installed in each soil drainage class to observe attenuation within the portion of the biologically active soil zone that is likely to be saturated during the period of seasonal high water table each year. Because groundwater elevations during the experiment were not certain to be at the seasonal high water level and since an applied plume may not travel on the surface of the groundwater due to infiltration from the surface, deeper wells were installed in each drainage class.

C. Methods

Site Selection

Three study areas were selected, each containing a soil catena ranging from moderately well drained to very poorly drained soils. The length of the soil catenas ranged from 25 to 60 meters. Two study areas were chosen within the Hunt-Potowomut watershed, one abutting Sandhill Brook in North Kingstown, and one abutting Frenchtown Brook in East Greenwich (figures 3.1, 3.2a, b, c,). The third study area (which will be referred to as the Laurel Lane study area) was located in South Kingstown abutting a tributary to the Usquepaug River (figures M.1 a,b,c). The Sandhill and the Laurel Lane study areas were located on soils derived from stratified drift deposits, while the Frenchtown study area was located on glacial till soils. The sites were chosen after an extensive survey of the Hunt-Potowomut watershed and were chosen because they represented a range of geomorphic and upland landuse conditions were accessible for study, and appeared to be free of major hydrologic or biological anomolies. All sites were located by April, 1989. All sites were on private land. Landowner permission for conduct of the study was not obtained until June 1989.

The Sandhill Brook study area (site) was located immediately adjacent to an unsewered dense residential development (lot sizes were approximately 1/4 acre). The Frenchtown Brook study area was located near an unsewered low-density residential development (lot sizes were approximately 2 acres). The Laurel Lane study area was located in a wooded area approximately 1/3 of a mile from a golf course and approximately 1/2 mile from any residences. All of the study areas consisted of an upland hardwood forest grading to a forested red maple swamp.

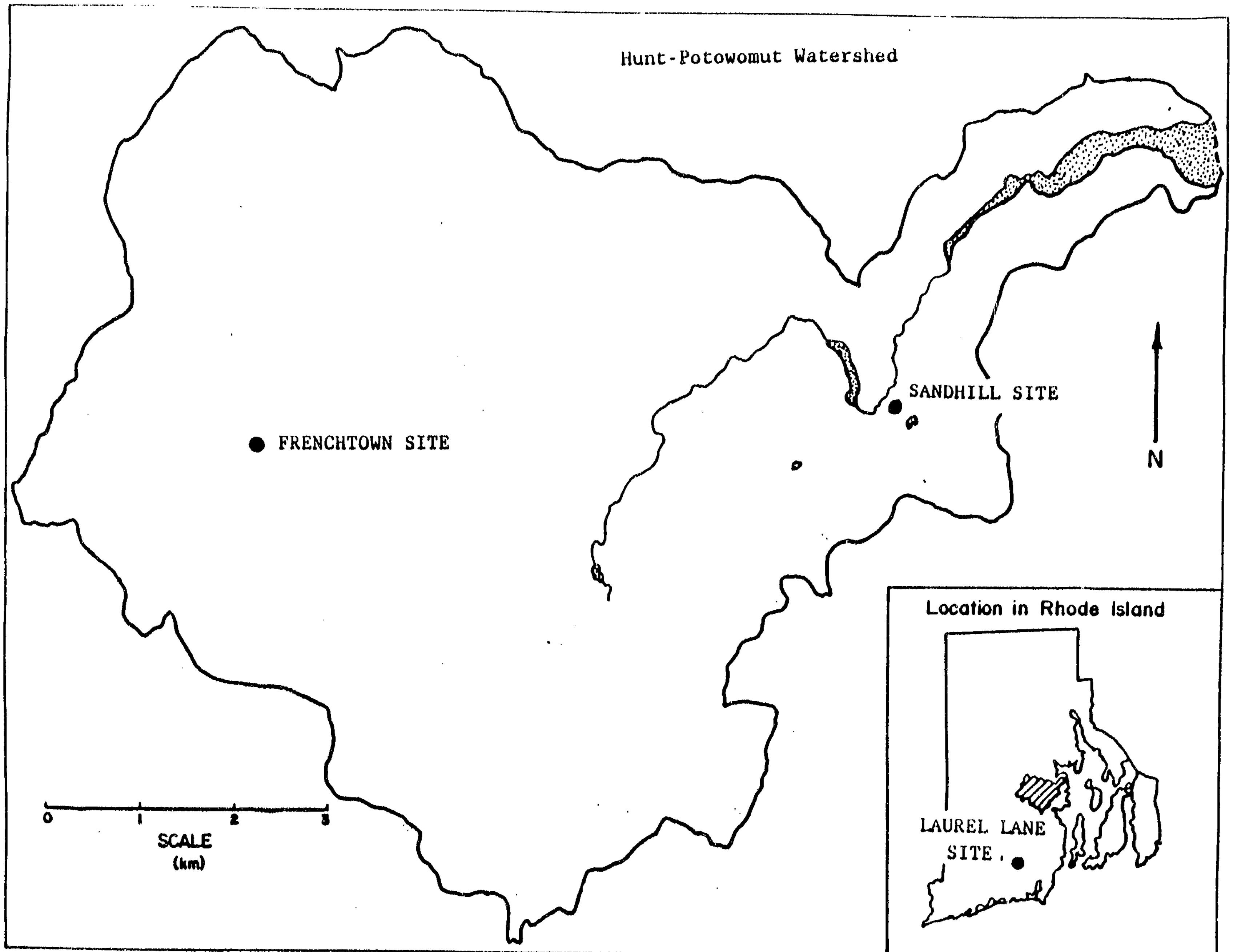


Figure 3.1: Location of the Frenchtown, Sandhill, and Laurel Lane study areas relative to the Hunt-Potowomut watershed.

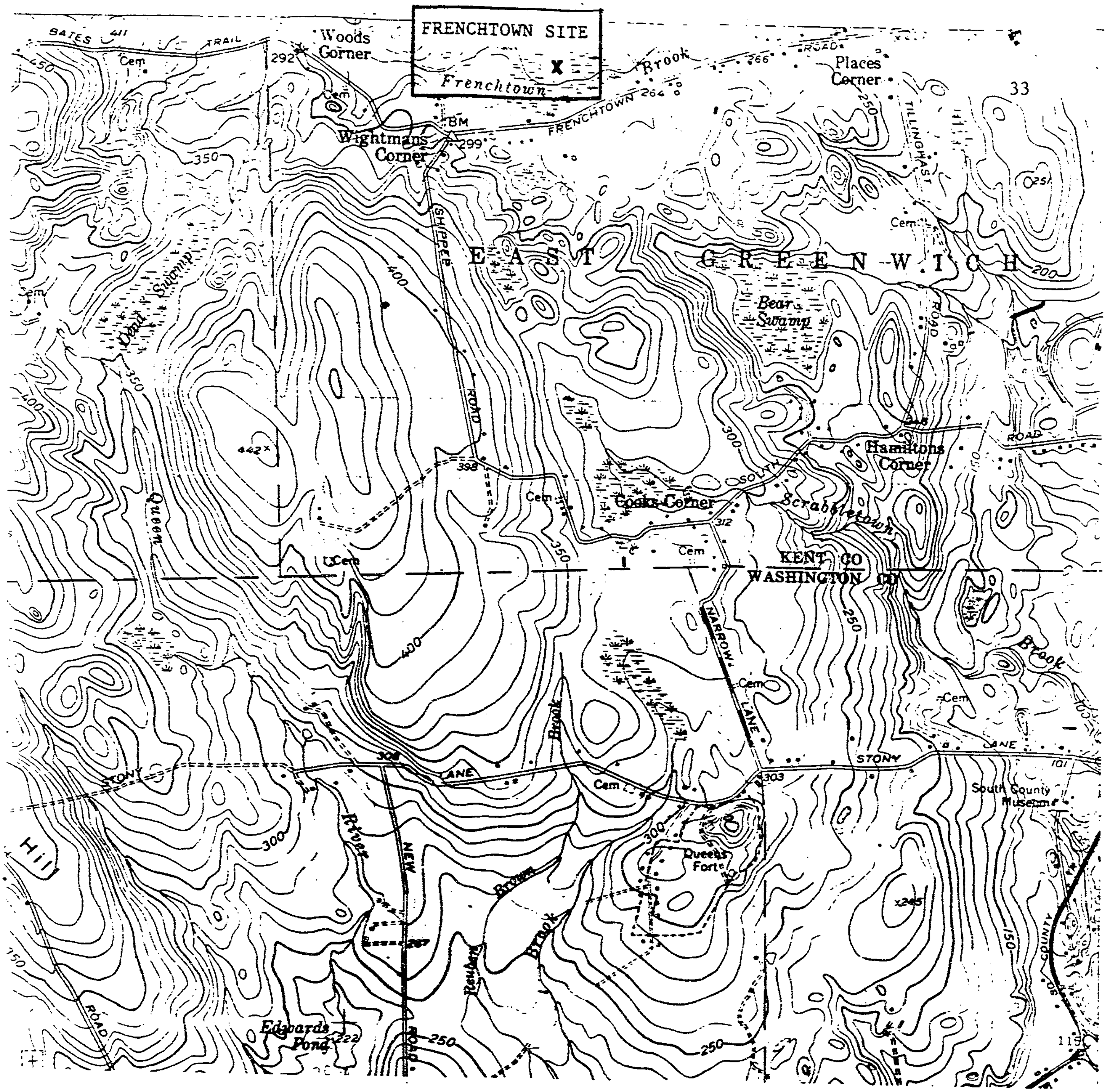


Figure 3.2a: Location of the Frenchtown study area (USGS, 1970).

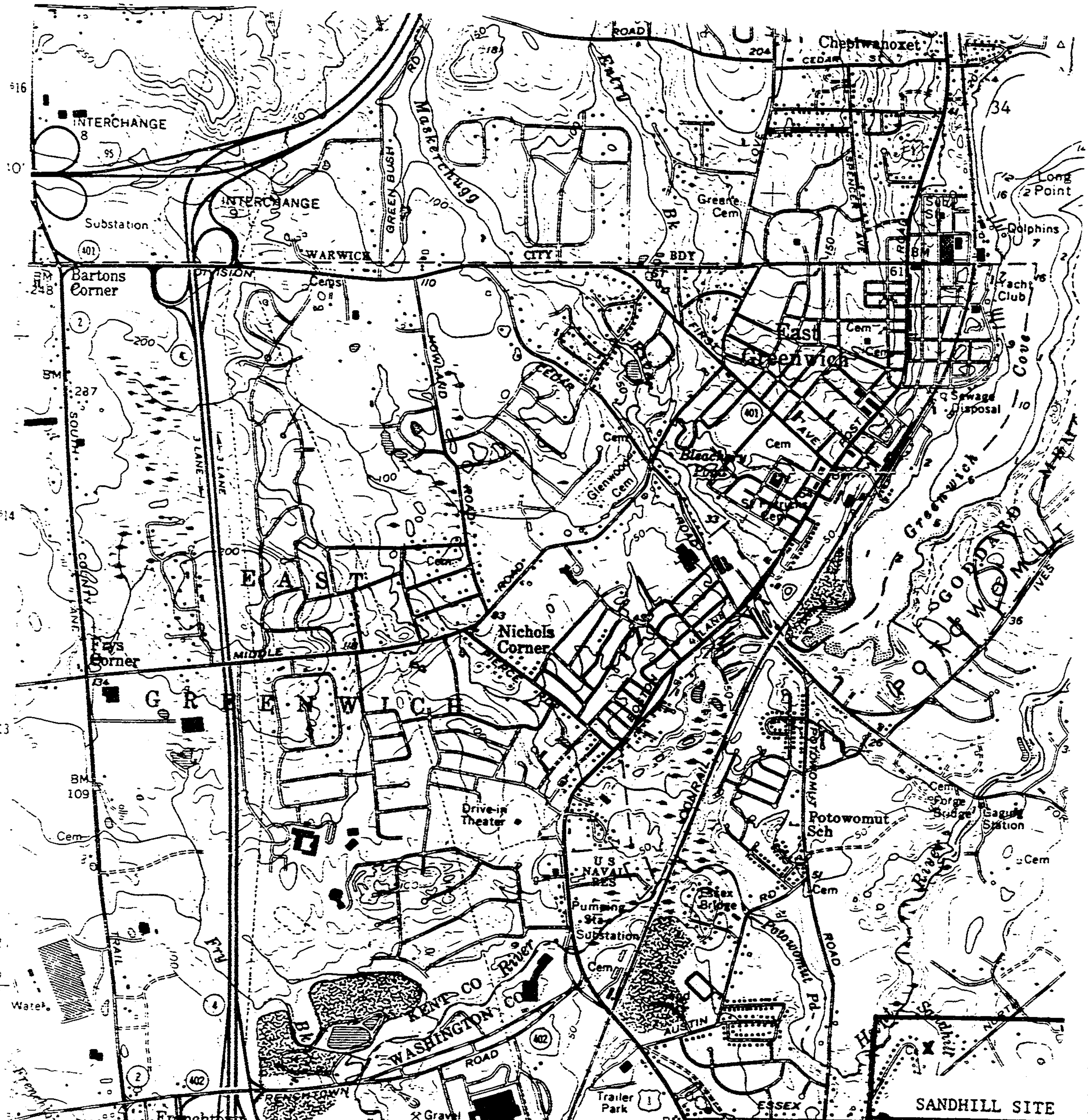


Figure 3.2b: Location of the Sandhill study area (USGS, 1975).

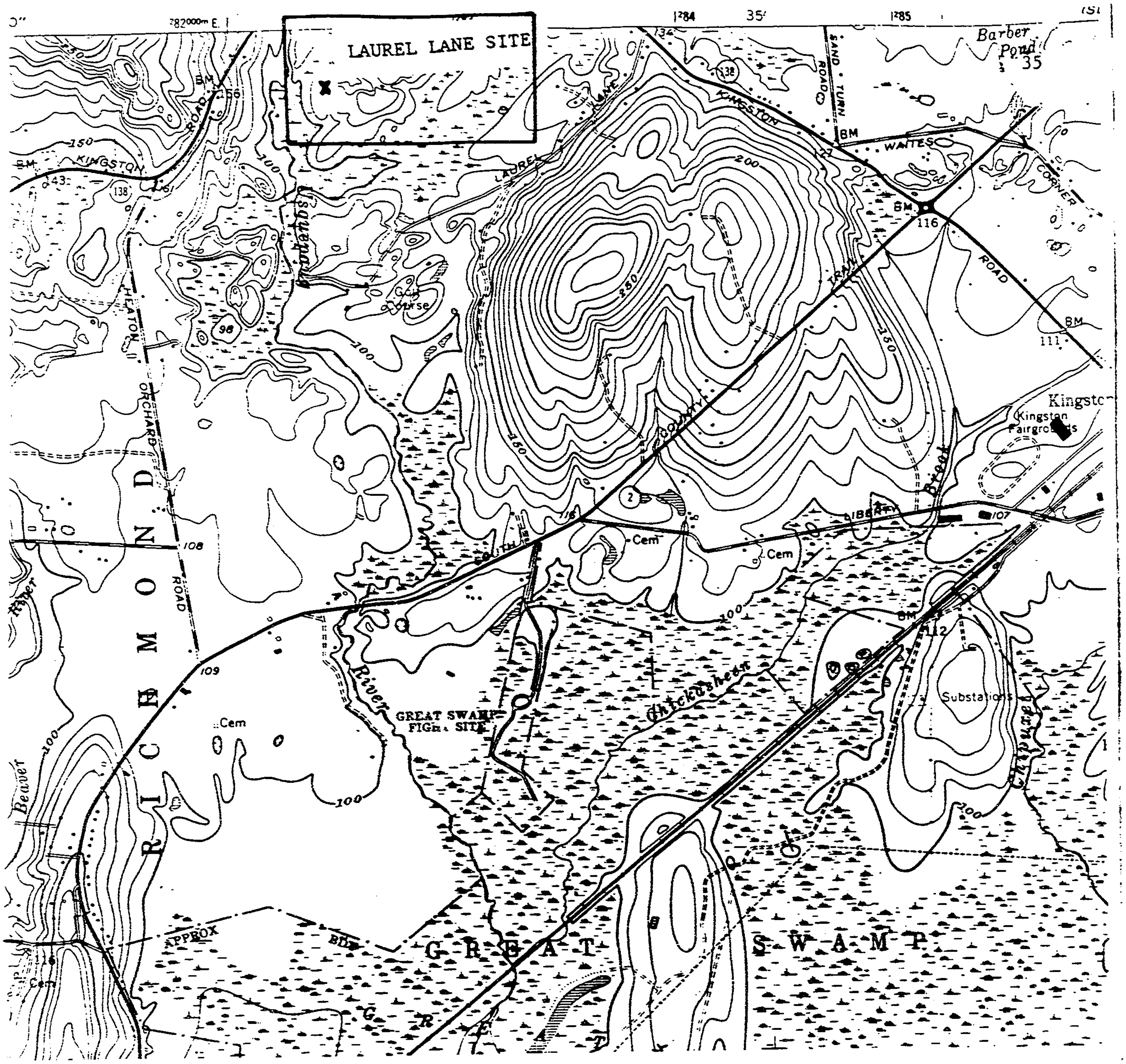


Figure 3.2c: Location of the Laurel Lane study area (USGS, 1975).

Site Layout

Water sampling well transects were located perpendicular to the direction of groundwater flow within each soil drainage class represented (figures 3.3a, b, c). Groundwater was monitored in moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD), and very poorly drained (VPD) soil drainage classes. Drainage classes represent degrees of soil wetness, and are based primarily on morphological features, such as mottles and gleying, which indicate depth to the high water table during the growing season (table 3.1). Mottles and gleying form during saturated periods when soil temperatures are above 5^{o} C as a result of chemically reducing conditions which occur when the soil oxygen supply is depleted by microbial activity (Sokoloski, 1989). Within a given mineral substrate or soil parent material, the soil types which develop in response to different degrees of soil wetness are classified into a drainage catena (Brady, 1984).

The soils of each study area were classified by University of Rhode Island and Soil Conservation Service soil scientists. The depth to the expected high water table during the growing season and the depth to the C horizon was determined at each well transect, using soil morphology. The soils in each study site, for each of the drainage classes represented, are listed in table 3.2. Appendix 3.Ia, b contains a complete description of each soil series. In contrast to many regions in the United States, the Rhode Island Soil Survey (Rector, 1981) did not identify SPD soils as distinct soil series. Generally SPD soils are classified within MWD soil series in Rhode Island. In this study, SPD soils were identified using the criteria from table 3.1.

Groundwater flow direction was determined at each site by analysis of water table elevations. This was accomplished by augering a minimum of 12 holes to the water table at each study site, then allowing the groundwater to equilibriate overnight. The relative groundwater elevations were measured with a surveying level, and the direction of groundwater flow was determined by triangulation (Driscoll, 1986). At the Laurel Lane study area the groundwater flow direction had already been established in a previous study, which used groundwater elevations from several different dates in 1987 (Simone and Bachand, 1988).

Each well transect was approximately six meters in width. Sampling wells within each transect were separated by a minimum distance of one meter to eliminate the potential for affecting groundwater flow or quality at a well when sampling occurs at a neighboring well.

Within each study area two sampling locations were established. The upland location consisted of wells in the moderately well drained and somewhat poorly drained soils, while the wetland location consisted of wells located in the poorly drained and very poorly drained soils. These sites will be referred to as the "upland locations" and the "wetland locations", respectively.

Well Construction and Placement

The wells were constructed of 4.04 cm. PVC schedule 40 pipe. The screened portion was of the same material with 0.0254 cm. slots (Atlantic Screen Inc, Milton, DE) to allow water entry into the wells. The screened portion of the well was attached to the solid portion with a PVC collar. The end of the screened section was fitted with a pointed PVC well point. The top of each well was covered with a fitting which facilitated a screw on cap. All fittings were attached with PVC cement (Master Plumber, Multi-Purpose Cement). Each

Figure 3.3a: Plan view of the Frenchtown site. Shows the relative locations of groundwater sampling wells, soil drainage classes, and contaminant apllication trenches.

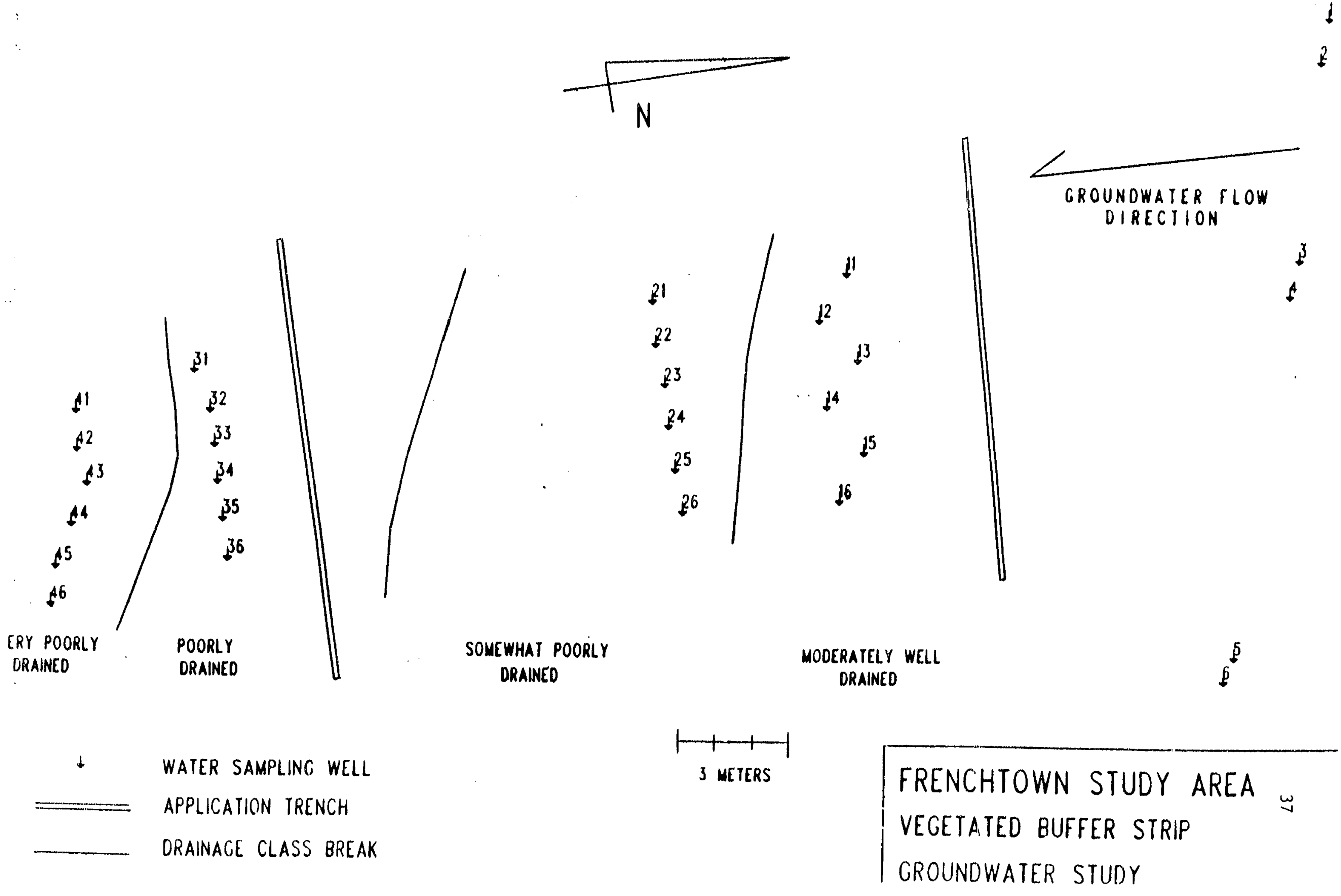


Figure 3.3b: Plan view of the Sandhill site. Shows the relative locations of groundwater sampling wells, soil drainage classes, and contaminant apllication trenches.

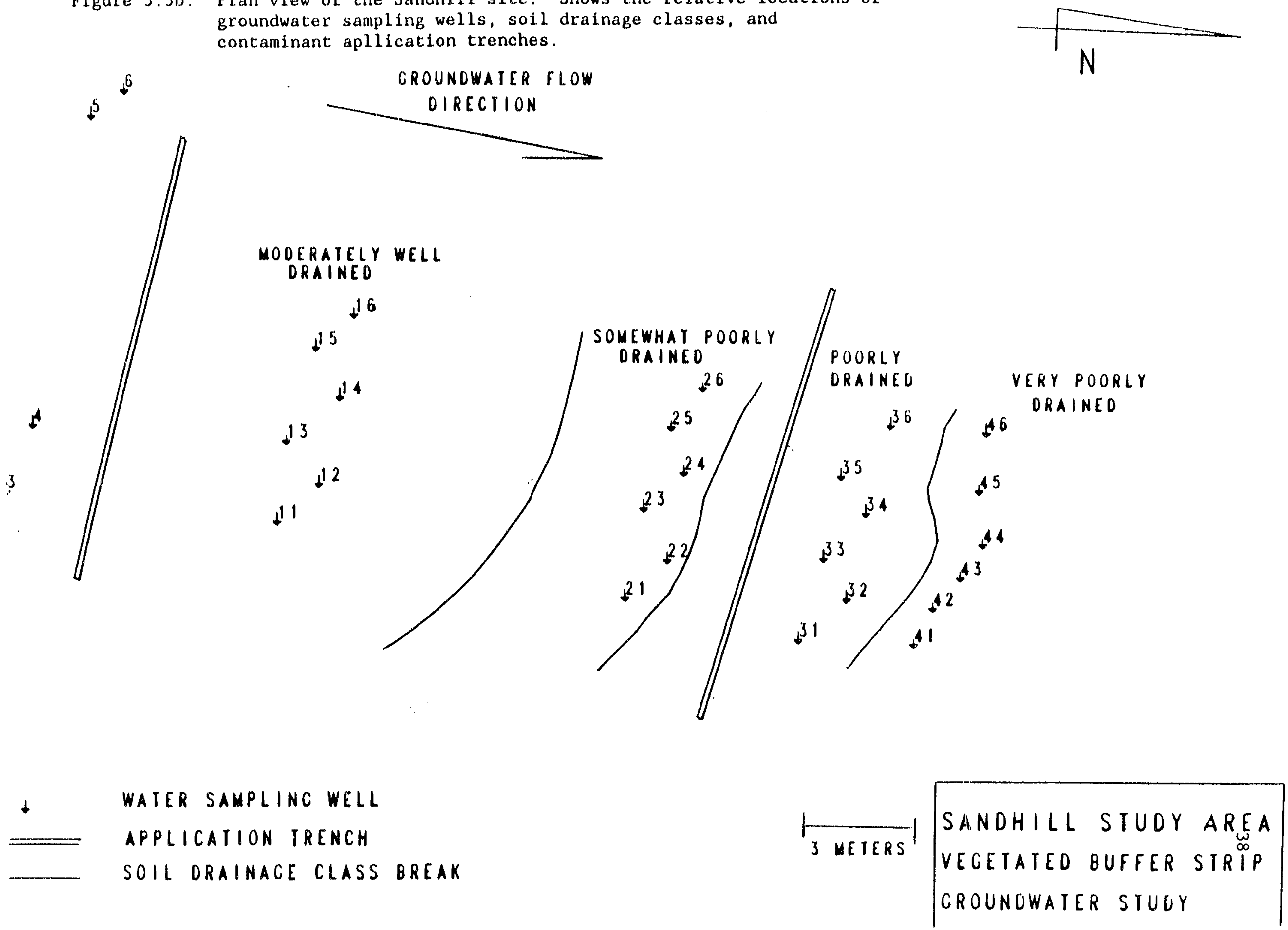


Figure 3.3c: Plan view of the Laurel Lane site. Shows the relative locations of groundwater sampling wells, soil drainage classes, and contaminant apllication trenches.

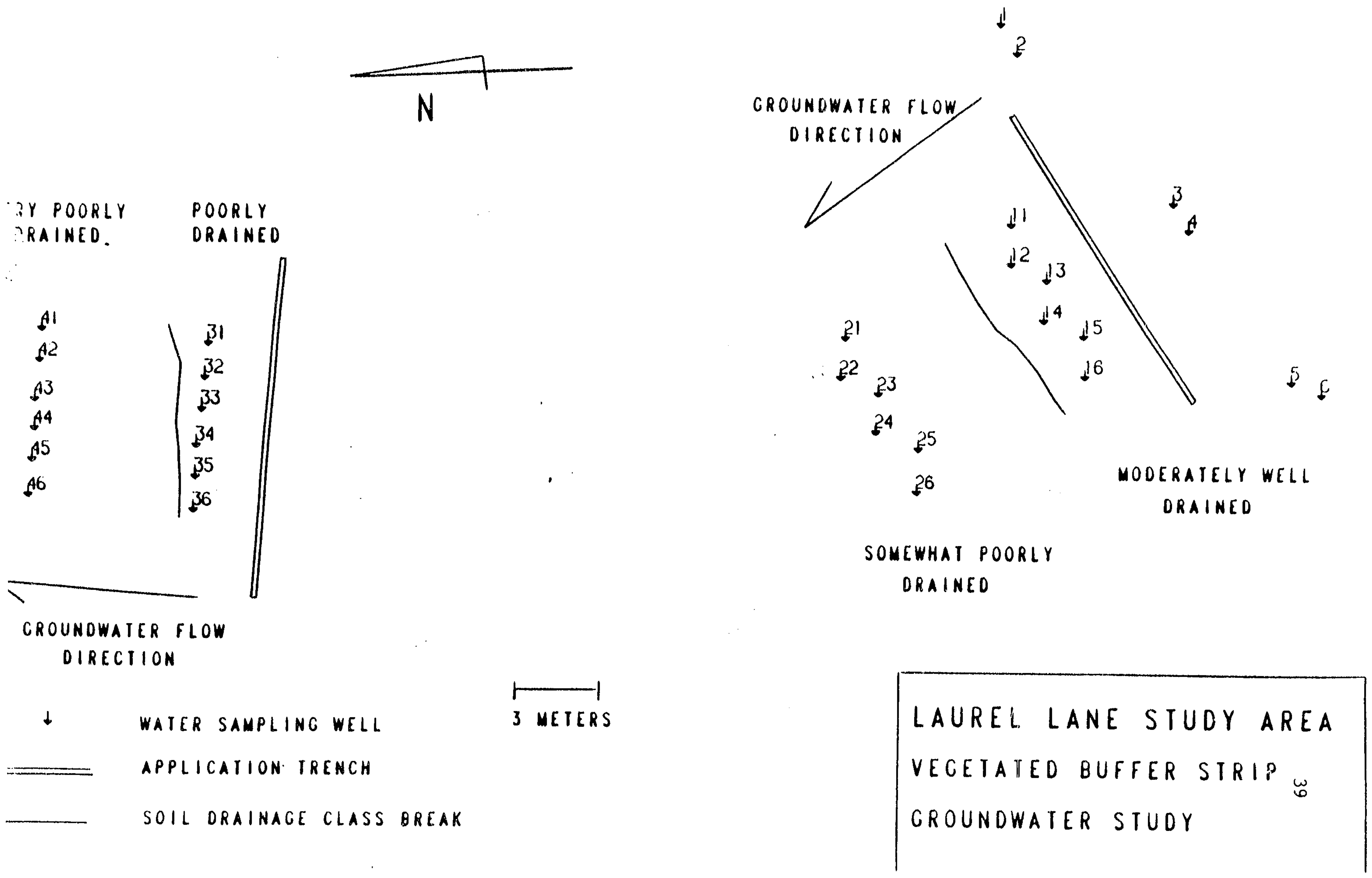


Table 3.1: Description of soil drainage classes, after Wright and Sautter (1979).

Excessively drained soils exhibit bright colors and are usually coarse-textured. The soils have rapid permeabilities, very low water-holding capacities, and the subsoils are free of mottling.[a]

Somewhat excessively drained soils have bright colors and are rather sandy. These soils also have rapid permeabilities, low water-holding capacities, and the subsoils are free of mottles.

Well drained soils drain excess water readily, but contain sufficient fine material to provide adequate moisture for plant growth. The subsoils are free of mottles to a depth of at least 3 ft (91 cm) and the colors are generally bright yellow, red, or brown.

Moderately well drained soils may have any texture but their internal drainage is restricted to some degree. Mottles are common in the lower part of the subsoil, generally at a depth of 18 to 36 in. (46 to 91 cm). These soils may remain wet and cold later in the spring, but are generally suited for agricultural use.

Somewhat poorly drained soils remain wet for long periods of time due to slow removal of water. These soils generally have a slowly permeable layer within the profile or a high water table. Mottles are common in the subsoil at a depth of 8 to 18 in. (20 to 46 cm).

Poorly drained soils have water tables at or near the surface during a considerable part of the year. Dark, thick, surface horizons are commonly associated with these soils and gray colors usually dominate the subsoils. Mottles are frequently found within 8 in. (20 cm) of the soil surface.

Very poorly drained soils are saturated by a high water table most of the year. These soils generally have thick black surface horizons and gray subsoils. Topographically, soils of this drainage class usually occupy level or depressed sites and are frequently ponded with water.

Table 3.2: Soil series classification of each soil drainage class at each site.

	Site		
Drainage Class	Sandhill	Frenchtown	Laurel Lane
Moderately Well Drained	Sudbury*	Sutton*	Deerfield
Somewhat Poorly Drained	Walpole*	Sutton*	Deerfield
Poorly Drained	Walpole*	Leicester*	Wareham
Very Poorly Drained	Scarboro*	Palms*	Scarboro

* Indicates that the soils are considered taxajuncts of the listed classification

well was vented to allow the water within the well to equilibriate with the ambient ground water elevation. Wells were installed by augering a hole, either with a 4 inch hand auger or a 4 inch power auger, to the proper depth, and placing the well into the hole. A surveying level was used to ensure that the well screens were installed at the proper depth. The screened portion of the well was backfilled with a medium sand to prevent clogging of the well screen with fine soil particles. The solid portion of the well was backfilled with native soil to approximately 10 cm below the surface, then backfilled with approximately 8 cm. of 1 cm. bentonite pellets to prevent channeling of surface water down the side of the well. Native soil was used for the remainder of the backfilling. Soil was mounded around the base of each well to prevent surface ponding of water around the well casing (to avoid well contamination from channeling of water along the side of the well). The relative elevation of the ground surface and the top of each well was determined with a surveying level.

Both upland and wetland locations were instrumented with three "shallow" wells and three "deep" wells in each drainage class, along with an additional well transect in the area immediately upgradient of the study site. The "upgradient" wells were used for background water quality analysis. A total of 18 wells were established in each upland location. The shallow wells were installed with the upper limit of the well screen beginning at the depth of the seasonal high water table (based on soil morphology) and extending down 30 cm. The deep wells were installed with the screened portion of the well beginning at 30 cm. below the depth of seasonal high water table and extended down to 100 cm below the seasonal high water table.

A total of 12 sampling wells were installed in each wetland location, with six in the PD soils and six in the VPD soils. The shallow wells at each transect were screened from 7 cm below the ground surface (to allow the bentonite seal to block out surface water) to 10 cm below the depth of the solum. The purpose of the shallow wells was to sample the water which was in the biologically active zone of the soil. The deep wells were installed with the screened portion of the well beginning at 10 cm below the depth of the solum and extending down 70 cm.

Immediately upgradient of each upland and wetland sampling location, a 12.0 by 0.2 meter chemical application trench was established perpendicular to the direction of groundwater flow. The bottom of the trench was 15 cm above the water table. The purpose of the trench was to ensure that the chemical application reached the groundwater quickly and evenly. The application trench in the Frenchtown upland site was only established to within 30 cm of the water table, due to the inherent difficulty of digging in till soils. The application trench spanned the width of the well transects with an excess of 3 meters on either side, to diminish the possibility of the natural variation in the groundwater flow direction moving the chemical plume outside the area of the sampling wells (see figures 3.4a, b, c).

Contaminant Application

At each site, chemicals were applied in solution form in three doses of 489 liters each. Each dose was applied over a period of 3 hours, and the three doses were applied over a week long period (table 3.3). The concentrations and application rates used were chosen to ensure that the contaminant plume would be detected after dilution from recharge and mixing with the groundwater. NO_3^-N concentrations selected are somewhat higher than concentrations normally expected in septic system effluent (30 - 80 mg/l)(USEPA, 1980).

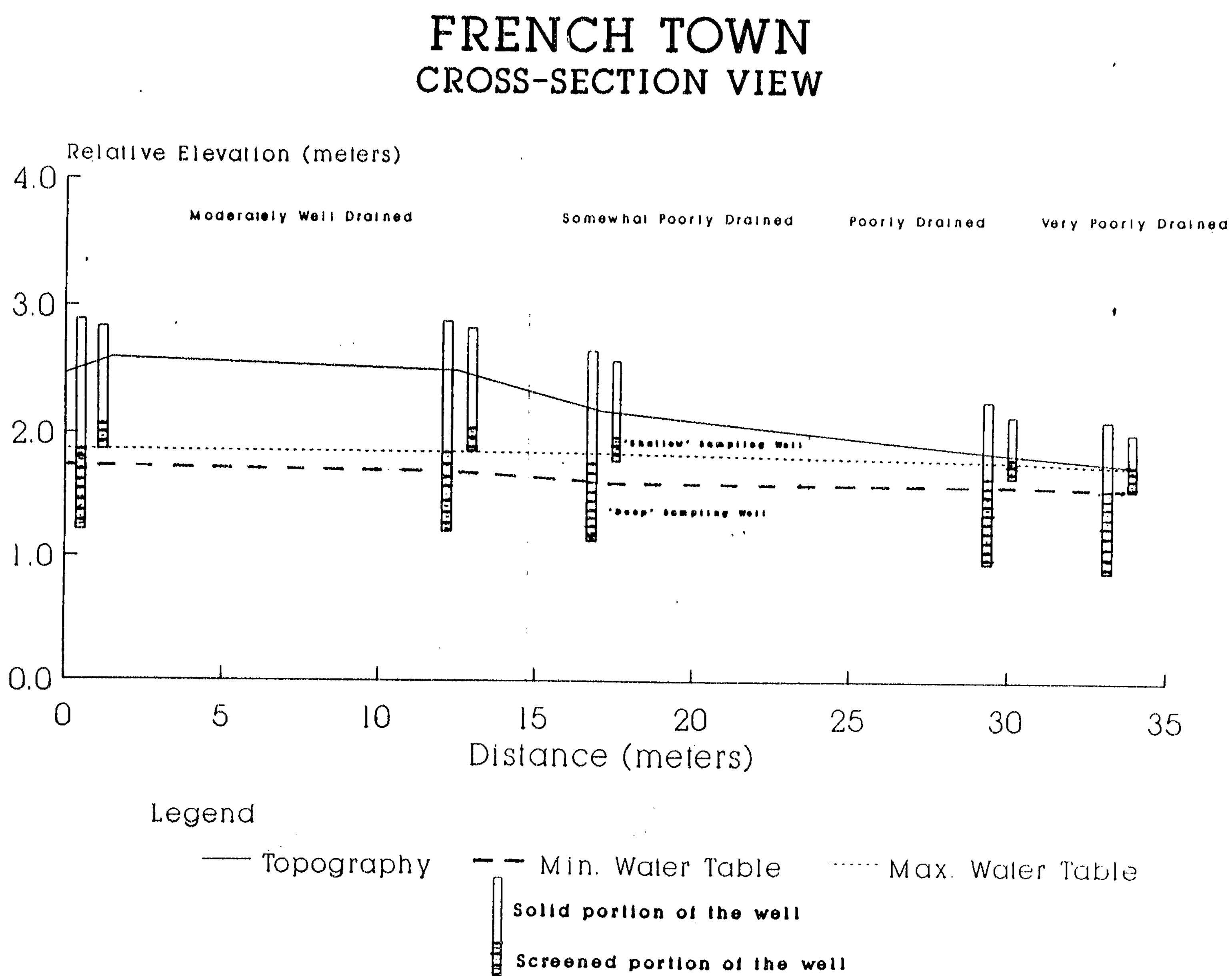


Figure 3.4a: Cross-section view of the Frenchtown site. Shows the relative locations of groundwater sampling wells, soil drainage classes and the maximum and minimum water table elevations observed during the study period.

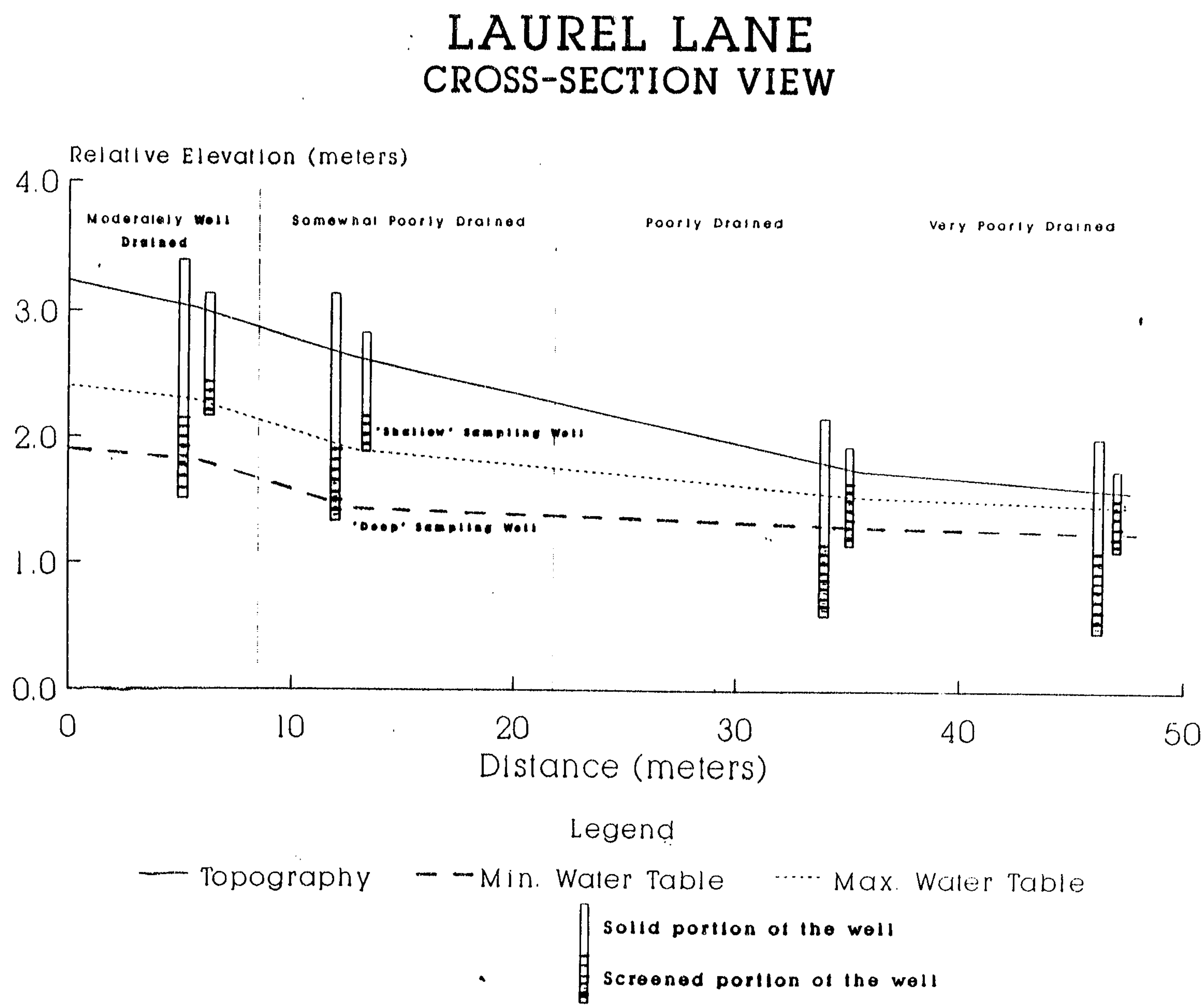


Figure 3.4c: Cross-section view of the Laurel Lane site. Shows the relative locations of groundwater sampling wells, soil drainage classes and the relative maximum and minimum water table elevations observed during the study period.

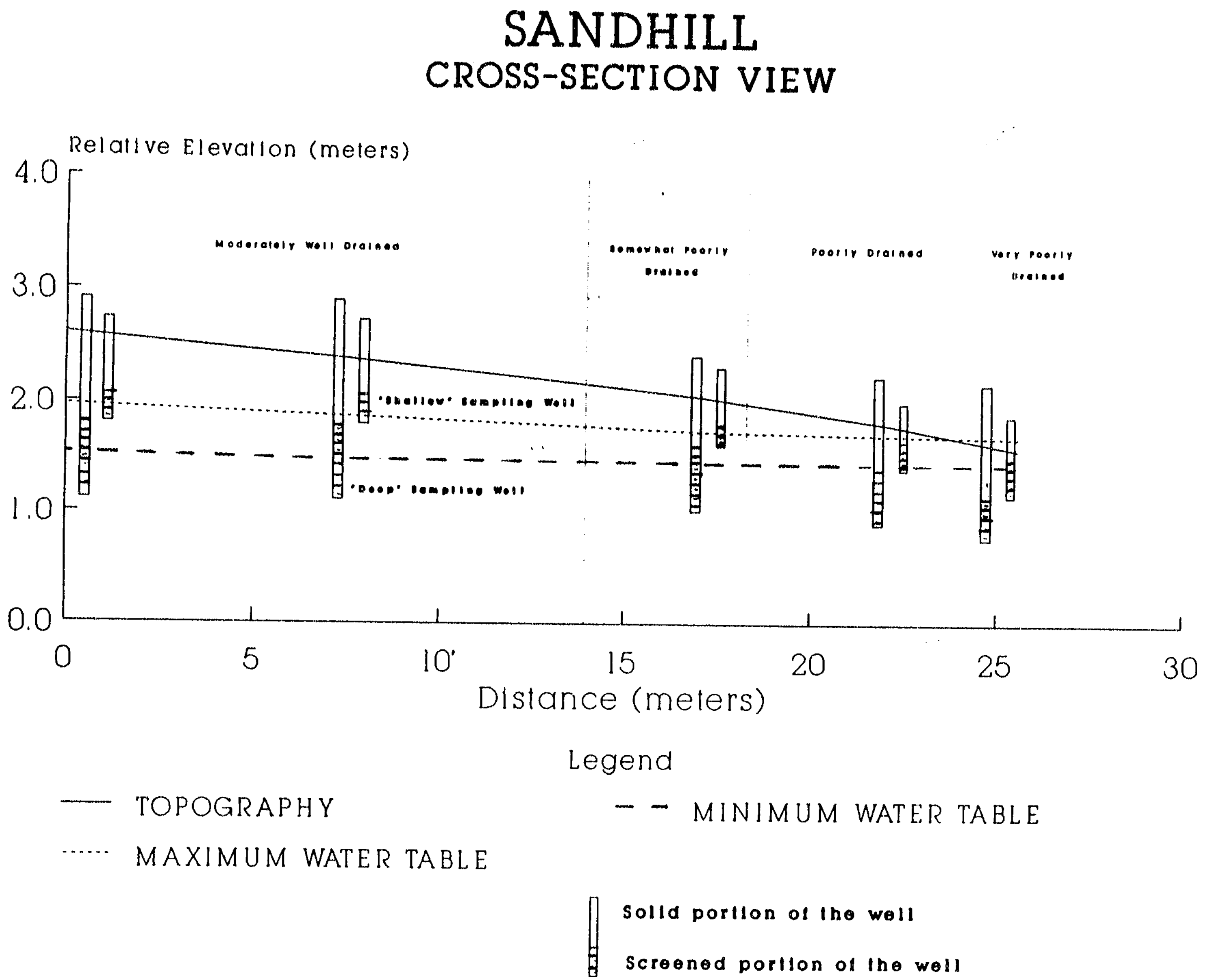


Figure 3.4b: Cross-section view of the Sandhill site. Shows the relative locations of groundwater sampling wells, soil drainage classes and the relative maximum and minimum water table elevations observed during the study period.

Table 3.3: Application dates of contaminants at each site.

Application Dates

Location	Application	Study Site		
		Laurel Lane	Frenchtown	Sandhill
Upland				
	1	8/23/89	8/08/89	8/16/89
	2	8/24/89	8/09/89	8/17/89
	3	8/25/89	8/15/89	8/18/89
Wetland		Laurel Lane	Frenchtown	Sandhill
	1	7/14/89	6/26/89	6/29/89
	2	7/18/89	6/27/89	7/05/89
	3	7/19/89	6/30/89	7/06/89

However, the total mass of NO_3^-N applied over a week, simulates the expected weekly load from a septic system serving a 3 to 4 person residence (USEPA, 1980). Dry, reagent grade chemicals were mixed into solution on site using water obtained from the associated nearby stream at each site. The proper concentrations were ensured by preweighing the chemicals and mixing them into a pre-established volume of water in a polyethylene carboy placed at the site. Because the stream concentrations of copper and NO_3^-N were less than 2% of the application concentrations (table 3.8) no adjustment was made for these constituents at application. Chloride, however, was found at concentrations ranging from 10 - 25 mg/l in the streams. The mass of reagent chloride added to the application trenches was adjusted to account for the chloride in each of the stream waters. The chemical concentrations, for each site, are listed in table 3.4. Permission to apply chemicals to the application trenches was requested from the RIDEM UIC program and was obtained in May 1989.

Well Sampling

Each well was sampled weekly beginning immediately prior to chemical application. Sampling was continued in each drainage class until the chloride concentrations returned to background levels.

Prior to each sampling, three well volumes of water were removed from each well to ensure that a sample representative of the ambient groundwater was obtained. Water was evacuated from the wells with a hand vacuum pump (Soil Moisture Corp., Santa Barbara, Ca.) through 3/8" polyethylene hose, and samples were drawn through 1/8" polyethylene tubing (Tygon Brand, Norton Performance Plastics, Akron, Ohio). Samples were collected into sample rinsed acid washed polyethylene bottles (Nalge Co., Rochester, NY). Within 48 to 72 hours of sample collection the samples were filtered (2.5um) and preserved with 0.8 ml/l concentrated sulfuric acid (APHA, 1985). Samples were stored at 4 degrees celcius until analysis.

Soil and Water Analysis

Water analysis for NO_3^-N was performed using a colorimetric, automated cadmium reduction method (Alpkem Corp., 1986) on a Alpkem RFA 300 Rapid Flow Analyzer. Water analysis for chloride was performed using a colorimetric, automated ferricyanide method (Alpkem Corp., 1986) on a Alpkem 300 Rapid Flow Analyzer. The copper analysis was performed using a atomic absorption, direct aspiration method (US EPA, 1979) on a Perkin-Elmer Atomic Adsorption Spectrophotometer.

Water table elevations were measured monthly at each site, in sampling wells which were undisturbed for a minimum of 24 hours prior to elevation measurement, using a conductivity based water elevation meter (Soiltest Inc, Evanston, Il) and a tape measure (table 3.5). Groundwater temperatures were measured on a monthly basis in sampling wells immediately after evacuation and groundwater recharge of the wells. (table 3.6)

Soil samples were obtained from each transect at all sites, at the screened sampling depths and at the surface. The pH of each of the soils was measured in triplicate using a electrometric method (U.S. EPA, 1979) with a glass electrode (Corning Glass Works, Corning, NY).

The percent organic matter of each of the soil samples was determined in duplicate by a carbon loss on ignition method. The ignition method consisted

Table 3.4: Concentration of applied contaminants in the upland and wetland sites.

Contaminant Application Concentrations

Applied Chemical (Form)	Application Concentration (mg/l)	
	Upland Site	Wetland Site
Nitrogen (Potassium Nitrate)	180	120
Chloride (Potassium Chloride)	360	120
Copper (Cupric Sulfate)	30	15

Table 3.5: Averaged relative water table elevations throughout study period at each site.

Drainage Class	High Water Table Elevation	Low Water Table Elevation	Average Water Table Elevation
	Water Table Elevation (m) [a]		
Sandhill Study Area			
Upgradient Wells	29.970	29.537	29.745
Moderately Well Dr.	29.883	29.487	29.688
Somewhat Poorly Dr.	29.755	29.471	29.626
Poorly Dr.	29.728	29.458	29.603
Very Poorly Dr.	29.709	29.453	29.583
Frenchtown Study Area			
Upgradient Wells	28.871	28.722	28.779
Moderately Well Dr.	28.861	28.688	28.728
Somewhat Poorly Dr.	28.850	28.614	28.711
Poorly Dr.	28.793	28.596	28.681
Very Poorly Dr.	28.752	28.566	28.649
Laurel Lane Study Area			
Upgradient Wells	29.408	28.909	29.159
Moderately Well Dr.	29.304	28.825	29.065
Somewhat Poorly Dr.	28.908	28.438	28.673
Poorly Dr.	28.547	28.307	28.427
Very Poorly Dr.	28.480	28.266	28.373

a) A datum was established at each site and arbitrarily assigned an elevation of 30 meters.

Table 3.6: Average groundwater temperatures in each soil drainage class at each site.

Groundwater Temperatures (°C)

Frenchtown Study Area	June 27	July 11	Aug 15	Oct 6	Nov 10
Upgradient	15.3	16.0	16.7	14.0	11.5
Moderately Well Drnd	13.7	14.0	15.5	14.0	12.0
Somewhat Poorly Drnd	14.2	14.0	15.5	14.0	11.5
Poorly Drnd	15.1	16.0	17.0	13.8	11.0
Very Poorly Drnd	15.6	16.0	16.8	13.8	10.2

Sandhill Study area	July 16	Aug 16	Sept 28	Nov 10
Upgradient	13.5	16.5	15.7	13.0
Moderately Well Drnd	15.0	-	15.2	12.5
Somewhat Poorly Drnd	15.0	-	14.8	12.5
Poorly Drnd	15.5	18.4	14.8	12.0
Very Poorly Drnd	16.5	19.7	14.5	12.0

Laurel Lane Study Area	Aug 17	Sept 7	Oct 4	Nov 27
Upgradient	16.0	15.2	14.2	11.0
Moderately Well Drnd	16.0	15.5	14.2	9.3
Somewhat Poorly Drnd	15.8	15.4	14.2	9.7
Poorly Drnd	17.0	15.5	14.2	9.0
Very Poorly Drnd	16.5	15.5	14.2	9.0

of preweighing oven dry (24 hours at 104 degrees Celcius) soil samples and placing them in an oven at 1000 degrees Celcius for 20 minutes. The samples were cooled in the oven and the weights were obtained when the samples were cool enough to touch. The percent organic mater was calculated using the following relationship:

$$\text{\% Organic Matter} = \frac{\text{(oven dry weight - final weight)}}{\text{oven dry weight}} \times 100\% \qquad (1)$$

D. Results and Discussion

To quantify the amount of attenuation observed from the applied NO_3^-N and copper, the application ratios of NO_3^-N and copper to chloride (table 3.7) was compared with the observed ratios. The result of the comparison is considered the percent loss due to attenuation (percent attenuation). This expression was determined in the following manner:

$$\text{Percent Attenuation} = \frac{\text{(application ratio - observed ratio)}}{\text{application ratio}} \times 100\% \qquad (2)$$

The background concentrations of chloride, copper, and NO_3^-N, were established prior to attenuation analysis (tables 3.8a, b). In the upland sites the background concentrations were determined using the data obtained from the upgradient wells, throughout the study period. Due to the spatial variation in the ambient groundwater concentrations of chloride and NO_3^-N through the drainage catena, the background concentrations in the wetland sites could not be established with confidence using the data obtained from the upgradient wells. In the wetland sites the background concentrations were determined using the concentrations obtained in the sampling wells prior to chemical application.

To distinguish sample contamination due to the introduced chemical plume from variations in background concentrations in the upland sites, Chebyshev's rule was used to limit attenuation analysis to those samples with a concentration greater than at least 96% of the background samples (i.e. concentrations greater than the mean plus five times the standard deviation of the background samples) (Bhattacharyya and Johnson, 1977).

In the wetland locations the coefficient of variation of background samples was considerably larger than in the upland locations, partially as a result of smaller sample size. The highest observed background concentrations were much lower than five standard deviations above the background sample mean. Attenuation analysis was therefore extended to data from samples which had chloride concentrations greater than three standard deviations above the mean concentrations (greater than atleast 89% of the background data based on Chebyshev's rule). In all wetland and upland locations, all samples included in the attenuation analyses had chloride concentrations greater than the highest background levels observed at a given location. A total of 956 samples were analyzed for chloride and NO_3^-N. Only 12% were found to be affected by the contamination plume, using the criteria given above.

Hydrologic Characteristics

The travel time for the contamination plumes to reach the first set of downgradient wells averaged between 25 and 30 days for both the Laurel Lane and Sandhill sites and was almost 60 days at the Frenchtown site. Mean daily velocities of the forward edge of the plume ranged from 0.07 m/day at the upland Frenchtown site,- to 0.2 m/day at the upland Sandhill site. The velocities observed result from a combination of hydrologic characteristics, specifically hydraulic gradient; hydraulic conductivity; and porosity of the media as well as from the method that the contamination source was applied. During application, a 20.8 cm depth of "spiked" solution was applied to the application trenches during 3 three hour periods within one week. This application rate (6.9 cm/hour for 3 hours) is much higher than that of a typical onsite sewage disposal system (5 cm/day) and may have increased the natural groundwater flow rate and modified the typical flow path of infiltrating waters. However, the coarse grained media found at all sites is not conducive to creating groundwater mounding, so we expected hydraulic gradients to be relatively unaffected by the application. Preliminary estimates of the average linear velocity of each site (Freeze and Cherry, 1979) were determined from hydraulic conductivity, estimated by grain size analysis (Masch and Denny 1966), and the average hydraulic gradient measured during the study. Average linear velocities were estimated at approximately 0.3 m/day for both Frenchtown and Sandhill sites, while the estimated linear velocity at the Laurel Lane site is estimated to be in the range of 0.05 m/day. While the estimated velocities vary from predicted values, due to the inherent difficulties in estimating hydraulic conductivity and flowpaths, we believe the observed rates of movement were not dramatically higher than rates expected from natural groundwater flow. Thus residence times were likely to be representative of natural conditions. The nearest monitoring wells were placed more than 2 m downgradient from the application trench to reduce the chance that the contamination plume would move immediately to the sampling wells.

A contamination plume was observed at all locations except the Frenchtown wetland well transects. Although background chloride concentrations at that location were consistently less than 10 mg/l, no samples were obtained with elevated chloride concentrations. Our experimental design was not adequate to determine whether the plume moved away from the wells or was diluted by recharge and dispersion. No data on copper or NO_3^-N attenuation from the Frenchtown wetland location are included in the results and discussion section.

In estimating the percent attenuation of NO_3^-N, the choice of a background concentration of NO_3^-N and chloride can affect the attenuation values obtained. To compute attenuation (equation 2), the background concentrations of both NO_3^-N and chloride must be subtracted from the observed concentrations. By the nature of the equation, the value selected for the background concentration of chloride becomes less important as the observed chloride concentrations in the plume increase. When the observed chloride concentrations were close to background levels, the choice of the background chloride concentration markedly affected the estimate of percent attenuation. Appendix 3.II lists all observations which were considered to be affected by the contamination plume. Attenuation estimates are given for the average background chloride concentrations and the highest background chloride concentrations in appendix 3.III.

An example of the effect of the choice of background concentration for chloride in the estimate of percent attenuation is shown in the comparison of

Table 3.7: Ratios of copper and NO_3^-N to chloride in the upland and wetland sites.

Application Concentration Ratios

Chemical: Chloride	Upland Sites	Wetland Sites
Nitrogen: Chloride	1:2	1:1
Copper: Chloride	1:12	1:8

the estimates obtained at the moderately well drained soil at the Laurel Lane upland site. Chloride concentrations in the plume averaged 13.5 mg/l, less than 10 mg/l higher than the average background concentration (4.96 mg/l). The percent attenuation using the average background concentration of chloride (4.96 mg/l) was 22.8% while the percent attenuation determined using the highest background chloride (6.47 mg/l) (table 3.8a) concentration observed was 5.2% (table 3.9).

The percent attenuation of NO_3^-N determined using the high background concentration of chloride is the more conservative estimate of attenuation and is used in the discussion of the results.

Nitrogen Attenuation

The results of the NO_3^-N attenuation study are summarized in table 3.9. High NO_3^-N attenuation was consistantly observed in the wetland locations, while the upland locations showed considerable variation in attenuation of NO_3^-N. Within each of the three upland sites, groundwater sampled in the SPD soils had greater attenuation rates than the groundwater sampled in the MWD soils. At the Frenchtown and Sandhill sites almost complete attenuation occurred in the SPD soils, while the Laurel Lane SPD soils had less than 30% attenuation. In the MWD soils only the Sandhill site showed high removal rates (>60% attenuation). The other sites had no significant attenuation at the MWD transects. The negative attenuation value for the MWD Frenchtown location was the result of one well which had NO_3^-N concentrations higher than chloride concentrations for three consecutive sampling dates. The negative attenuation values generally occurred when the highest concentrations of both NO_3^-N and chloride were observed. This well may have been contaminated by an outside source; however, we had no direct evidence of contamination and since the timing of the elevated samples coincided with the expected appearance of the plume at that well we did not adjust our dataset. Excluding this well yielded an average attenutaion value of 75% for the MWD Frenchtown location.

Biological pollutant attenuation mechanisms in riparian buffers include plant uptake of nutrients and microbial processing of nutrients, metals and organics. Plant uptake of nutrients is dependent on the ability of plants to intercept and remove nutrients from either surface or subsurface flow. Plants differ greatly in their selectivity for particular nutrient forms and in the rate at which they take up nutrients. More importantly, nutrients trapped in plant tissues can later be released back into the soil solution as these tissues decompose (Peverly, 1985, Nixon and Lee, 1986). Over time, plant demand for nutrients can be exceeded by inputs into the VBS, further reducing the plant sink (Omernik et al. 1981, Bowden, 1987). Jacobs and Gilliam (1985) suggested that most plant nutrient uptake in riparian buffers will be by annuals, that have no potential for long term nutrient storage. Storage of nutrients in structural tissues of trees (Ehrenfeld, 1987), or in grass tissues that are later harvested (Brown and Thomas, 1978) represent effective pollutant removal mechanisms. Clearly, some form of plant community management is necessary to maintain a long term plant uptake sink for nutrients in riparian buffers.

Microorganisms have the ability to degrade organic compounds as food resources and to absorb (immobilize) nutrients and metals into their tissues to support growth. Microbial immobilization is reversible; nutrients that are absorbed can later be released, or mineralized, depending on the amount of

Table 3.8a: Background concentrations of NO_3^-N and chloride at each site.

Nitrate-N and Chloride Background Concentrations

Site	Location	# obs	Nitrate-N Low conc mg/l	Nitrate-N High conc mg/l	Nitrate-N Ave conc mg/l	Nitrate-N Standard Error mg/l	Chloride Low conc mg/l	Chloride High conc mg/l	Chloride Ave. conc mg/l	Chloride Standard Error mg/l
Frenchtown										
	Stream	13	0.06	0.30	0.15	0.02	--	--	--	
	Upland	32	ND[a]	ND	ND	ND	1.03	4.30	2.53	0.12
	Wetland	13	ND	ND	ND	ND	3.33	9.74	6.18	0.68
Sandhill										
	Stream	8	0.68	1.47	0.94	0.15	--	--	--	--
	Upland	45	0.09	10.86	4.99	0.48	7.88	27.96	17.86	0.40
	Wetland	11	ND	0.93	0.15	0.08	1.25	17.28	7.69	1.34
Laurel Lane										
	Stream	8	0.25	1.37	0.55	0.17	--	--	--	--
	Upland	47	ND	0.10	ND	ND	2.91	6.47	4.96	0.15
	Wetland	10	ND	0.56	0.22	0.07	2.64	10.19	6.81	0.91

a) ND: below detection limit of 0.01 mg/l of NO_3^-N.

Table 3.8b: Background concentrations of copper at each site.

Copper Background Concentrations

Site	Location	# Obs.	Copper Low conc mg/l	High conc mg/l	Ave conc mg/l	Standard Error mg/l
Frenchtown						
	Upland	12	ND[a]	ND	ND	--
	Wetland	13	ND	ND	ND	--
Sandhill						
	Upland	15	ND	0.661	0.097	0.050
	Wetland	11	ND	0.097	ND	--
Laurel Lane						
	Upland	15	ND	ND	ND	--
	Wetland	11	ND	ND	ND	--

a) ND: Below detection limit of 0.09mg/l of copper.

Table 3.9: Average attenuation values for NO3-N and copper along with other selected site characteristics.

Drainage Class	Depth	# Obs	% Nitrate-N Attenuation (high bkrd Cl)	Standard Error	% Copper Attenuation (high bkrd cl)	Distance from Application (m)	pH	% Organic Matter	Average Depth to Groundwater (cm)
	Average Attenuation Values								
Frenchtown Study Area									
Moderately Well Drained	Shallow	0				4.05	4.04	1.85	78.5
	Deep	10	-35.98	62.42	100.00	4.05	5.12	0.86	78.5
Somewhat Poorly Drained	Shallow	0				8.64	4.43	1.68	48.2
	Deep	7	97.53	2.46	100.00	8.64	4.92	0.73	48.2
Poorly Drained	Shallow	0				2.70	4.28	3.88	17.7
	Deep	0				2.70	5.18	0.70	17.7
Very Poorly Drained	Shallow	0				6.58	4.47	27.95	11.8
	Deep	0				6.58	4.82	1.22	11.8
Sandhill Study Area									
Moderately Well Drained	Shallow	0				5.23	5.26	1.83	71.3
	Deep	8	61.84	11.65	100.00	5.23	5.60	2.14	71.3
Somewhat Poorly Drained	Shallow	0				14.86	5.49	1.78	44.5
	Deep	12	98.91	1.09	100.00	14.86	5.72	0.99	44.5
Poorly Drained	Shallow	19	71.46	9.19	100.00	2.36	5.91	7.17	21.6
	Deep	8	100.00	0.00	100.00	2.36	5.89	2.56	21.6
Very Poorly Drained	Shallow	2	90.68	2.56	100.00	5.78	5.42	30.68	1.2
	Deep	13	96.91	1.01	100.00	5.78	5.44	5.88	1.2
Laurel Lane Study Area									
Moderately Well Drained	Shallow	0				2.36	5.09	1.27	94.6
	Deep	4	5.16	5.64	100.00	2.36	5.01	1.01	94.6
Somewhat Poorly Drained	Shallow	0				9.60	4.86	2.22	92.1
	Deep	12	21.45	6.57	99.98	9.60	5.07	0.71	92.1
Poorly Drained	Shallow	0				2.50	4.73	2.13	29.1
	Deep	6	92.08	6.42	100.00	2.50	5.54	0.62	29.1
Very Poorly Drained	Shallow	1	70.80	0.00	100.00	14.61	4.70	5.10	20.1
	Deep	6	97.23	0.29	100.00	14.61	5.10	0.33	20.1

Table 3.10: Spearman's rank correlation coefficient for percent nitrate- N attenuation versus various site characteristics.

Spearman Rank Correlation Coefficients

Comparison Variable	% Nitrate-N Attenuation (high bkrd cl)	95% Confidence Level
Distance	0.42	0.44
pH	0.40	0.44
% Organic Matter	-0.05	0.44
Upland Depth to Groundwater	-0.83*	0.83
Wetland Depth to Groundwater	-0.14	0.90

* Significant at the 0.05 level.

nutrient available in soil (Hemond and Benoit, 1988). Denitrification represents a more reliable attenuation process, resulting in export of NO_3^- as N gas. Wet (oxygen poor), organic rich wetland soils are thought to be excellent sites for denitrification but its significance in wetland soils may be greatly overestimated (Bowden, 1987). Microbial processes in subsurface soils are poorly understood but are likely strongly inhibited by a lack of organic carbon to support growth (Smith and Duff, 1988, Parkin and Meisinger 1989).

Spearman's rank test (Bhattacharyya and Johnson, 1977) was performed to compare attenuation observed in the upland transects to a number of site characteristics including:

a) Mean water table elevation during study period
b) pH of the soil
c) organic matter content
d) travel distance from the application trench
e) microbial parameters (see chapter IV)

Only mean water table elevation proved to be statistically significant ($p<0.05$) to NO_3^-N attenuation (table 3.10). In the upland sites the highest attenuation corresponded to the shallowest groundwater levels.

Although the well transects at each site were established on identical drainage classes, we observed considerable differences between sites in summer water table elevations in the upland transects (table 3.11). During the study period (July-October) average water table elevation of the upland transects ranged from 107 to 95 cm below the ground surface at the MWD and SPD Laurel Lane transects, respectively, to less than 50 cm for the SPD transects at both Frenchtown and Sandhill where complete attenuation occurred. The water table elevation differences between sites may have ramifications for nutrient removal in the upland locations since the root depth in the transition zone of Rhode Island red maple swamps was found to be 0.6 m (Allen, 1988). Groundwater below the active rootzone will not be subjected to nutrient removal via plant uptake. Since all data used in this study were obtained during the growing season between July and October, plant uptake of nutrients is a reasonable mechanism for attenuation.

As indicated on table 3.9, virtually complete NO_3^-N removal occurred at those well transects where water table elevations were less than 60 cm below the surface. The moderately well drained transect at Sandhill was the only additional upland location where the water table rose to within 60 cm of the surface during the study. This transect showed considerable attenuation (62%) of NO_3^-N.

Although the screened portion of the deep wells at the Laurel Lane wetland locations were installed below 0.6 m, due to the deep solum of those soils, nearly complete attenuation was observed. Water table elevations at those locations was within 0.25 m of the surface and the contaminant plume may have traveled through the root zone before reaching the deep wells. On the other hand, denitrification in the groundwater below the rootzone may have eliminated the NO_3^-N. Repeating the experiment in the early spring, before initiation of plant uptake, should elucidate this question.

NO_3^-N attenuation in the upland locations at our study sites ranged from nearly 100% to 0% at sites with virtually the same organic matter content. Our results suggest that organic matter content in saturated zones of upland soils is not predictive of NO_3^-N attenuation.

Soil organic matter was quite low at all upland locations (<2.2%)(table 3.9); however, Trudell et al. (1986) have suggested that denitrification of NO_3^-N added to mineral subsoil may not be initially carbon limited.

Table 3.11: Averaged high, low and average depth to water table throughout study period in all well transects at each site.

Water Table Elevations
During Study
July - November 1989

Soil Drainage Class	Depth to High Water Table Elevation (meters)	Depth to Low Water Table Elevation (meters)	Depth to Average Water Table Elevation (meters)
Sandhill Study Area			
Upgradient Wells	0.640	1.073	0.865
Moderately Well Drained	0.518	0.914	0.713
Somewhat Poorly Drained	0.316	0.600	0.445
Poorly Drained	0.091	0.361	0.216
Very Poorly Drained	-0.114	0.142	0.012
Frenchtown Study Area			
Upgradient Wells	0.601	0.750	0.693
Moderately Well Drained	0.652	0.825	0.785
Somewhat Poorly Drained	0.343	0.579	0.482
Poorly Drained	0.065	0.262	0.177
Very Poorly Drained	0.015	0.201	0.118
Laurel Lane Study Area			
Upgradient Wells	0.828	1.327	1.067
Moderately Well Drained	0.729	1.208	0.946
Somewhat Poorly Drained	0.734	1.204	0.921
Poorly Drained	0.210	0.450	0.291
Very Poorly Drained	0.130	0.344	0.202

Table 3.12: Monthly precipitation and groundwater elevation levels throughout study period compared to mean long term levels.

Monthly Precipitation Levels

Month	1989 (cm)	20 Year Mean (cm)	Monthly Departure (cm)	Annual Cumulative Departure (cm)
June	12.85	7.42	5.44	9.52
July	16.61	7.59	9.02	18.54
August	14.50	11.32	3.18	21.72
September	15.85	10.44	5.41	27.13
October	17.88	10.06	7.82	34.95

Groundwater Levels

Month	Average Departure from Longterm Median (cm)	Percent of Wells Which Set or Equalled Monthly High Water Level Records (%)
July	45.93	4.76
August	56.69	19.05
September	58.52	23.81
October	91.44	66.67

* based on data from USGS water resource data from 21 wells located throughout Rhode Island

Trudell et al. (1986) observed complete denitrification of an introduced "slug" of NO_3^-N in sand media with soil organic matter less than 0.2%. The authors questioned the viability of this limited carbon source if continuous NO_3^-N was introduced for prolonged periods and suggest that attenuation of NO_3^-N from continuous sources (e.g. septic systems) may differ from attenuation of one-time additions of NO_3^-N

The spring, summer and fall of 1989 were extremely wet. Between April 1 and October 31, 1989 precipitation totaled 109.4 cm, a 60% increase over the 20-year mean for those months. Tables 3.12a and b compare long term mean monthly precipitation and groundwater levels to 1989 values for Rhode Island. Groundwater levels measured by USGS (USGS, 1989) in selected wells throughout Rhode Island were also unusually high during the study period. Virtually all of the 21 wells included in the USGS long term monitoring program had groundwater levels above the long term median from July through October, 1989, the period of time when monitoring occurred. In October, 1989 two-thirds of the wells monitored exceeded or equalled the highest recorded water levels observed during that month. Based on long term records, it appears that groundwater levels at the study sites were likely to be unusually high.

Throughout the study period the shallow upland wells were never in the saturated zone. The shallow wells were installed to monitor chemical transformations in groundwater near the surface. The exact location of the upland shallow well screens was based on soil morphologic features formed during the seasonal high water period. Allen et al. (1988) found that summer water table elevations in Rhode Island riparian forests are considerably lower than the seasonal high water table elevations. Repeating the contamination monitoring study in the spring, when groundwater is at its annual peak might generate results from the shallow upland wells where biological activity is expected to be higher than in the groundwater surrounding the deep wells. Plant uptake of nutrients is expected to be minimal during early spring, therefore providing an assessment of other N removal mechanisms.

Copper Attenuation

At all sites virtually complete attenuation of copper was observed (table 3.9). Appendix 3.II lists the copper concentrations of those samples considered. A comparison of the stability of different metals retained in organic-metal complexes ranks them in the general relative order: Cu> Pb> Fe> Ni> Mn> Co> Zn >Mg (Baker and Chesnin, 1975). Thus, in soils with organic residues, the relative mobility of copper is less than the relative mobility of the other metals listed. The ability of a soil to adsorb cations such as copper is characterized by the term "cation exchange capacity." Cation exchange capacity arises from negatively charged sites on clay and organic matter.

Based on a crude, but conservative, mass balance analysis, the cation exchange capacity was sufficient, at all study areas, to readily sorb the copper added. In the upland sites, only the deep wells were within the saturated zone of the soil. The organic matter content of the saturated zone of the upland deep wells was generally quite low, between 0.71% and 1.01% for all well transects, except the moderately well drained well transect at the Sandhill site (table 3.9). Clay is typically less than 5% in most Rhode Island soils, and is not a major source of cation exchange sites. Based on cation exchange capacity estimates of 250 and 400 meq/100 grams of organic matter (Leeder, 1978), the expected cation exchange capacity of the soil was a minimum of 2.0 meq/100 grams of soil, neglecting any contribution from clay minerals.

Conservatively assuming the 12 meter wide plume mixed only within the upper 0.15 meters of the groundwater and a soil bulk density of 1.2 g/cm^3, complete attenuation of the 45 grams of copper in solution (represents the maximum amount of copper applied) would be achieved within 0.52 meters of "horizontal" flow. The wetland sites have considerably higher organic matter contents and would be expected to have an even higher attenuation potential. However, according to Keeney and Wildung (1977) it has been commonly observed that in waterlogged soils containing sufficient organic matter to effect a decrease in Eh, the mobility of many metals is increased. Continued application of cations, such as copper and other heavy metals, to soil could conceivably saturate the cation exchange capacity of the soil. Copper may also be removed from solution by precipitation and complexing with other elements, enhancing soil attenuation capacity above that estimated by cation exchange capacity.

Adsorption of cations, such as copper, requires adequate contact with cation exchange sites. Inadequate contact could result from the solution moving rapidly through the soil macropores. It is difficult to predict the transport behavior of copper within groundwater flow systems. In many subsurface environments adsorption and precipitation reactions cause fronts of heavy metal contaminants, such as copper, to move very slowly relative to the velocity of the groundwater (Freeze and Cherry, 1979). Our results support this view, but do not rule out the possibility that copper from upland land uses can move through riparian zones in certain circumstances.

E. Summary

To determine the ability of natural riparian buffers to attenuate groundwater contaminants we related changes in groundwater quality to physical and chemical features of upland and wetland buffers. A network of groundwater monitoring wells were placed in different soil drainage classes (moderately well drained, somewhat poorly drained, poorly drained and very poorly drained soils) adjacent to streams at three sites. A "spike" of nitrate-N (NO_3^-N), copper, and a conservative tracer (chloride) was added upgradient of all wetland and upland locations. Attenuation of NO_3^-N and copper were assessed by observing changes in their relative chloride ratios compared to the application ratios. Complete attenuation of copper was found at all sites, due to the high affinity of soil organic matter for copper. Wetland locations were found to remove most of the NO_3^-N from groundwater, while upland sites showed considerable variability with regard to NO_3^-N attenuation. Depth to the water table was the only site factor significantly related to NO_3^-N attenuation. When depth to groundwater was less than 60 cm. (which corresponds to the root zone depth), considerable attenuation was observed.

F. Major Findings

1. As expected, movement of a contaminant pulse through groundwater in a riparian site was quite slow. Mean velocities ranged from 0.07 to 0.2 m/day.

2. Complete attenuation of copper was observed at all locations, suggesting that riparian buffers are a strong sinks for this metal.

3. NO_3^-N attenuation in upland VBS ranged from 21 to 99% and increased significantly as the plume moved downslope.

4. Almost complete NO_3^-N removal was observed in both poorly and very poorly drained soils.

5. Depth to water table was significantly correlated with NO_3^-N attenuation in the upland locations. Where the mean water table was within the expected rootzone (within 60 cm of the surface) considerable attenuation occurred. When the mean water table was below the root zone (deeper than 60 cm) little removal was noted.

6. The NO_3^-N removal mechanisms were not identified in this study. Denitrification, plant uptake, and microbial immobilization may all have accounted for the removal observed. Repeating this experiment in early spring, before considerable plant uptake occurs, could shed light on removal mechanisms.

Management implications

The results of the water quality aspect of our study demonstrated that selected portions of riparian buffers have the capacity to remove NO_3^- and copper from groundwater in the summer. The implications of these results for management purposes are that buffer widths required for pollutant removal will vary based on the soil types present. The ability of selected somewhat poorly drained soils and all poorly drained soils to completely remove the groundwater contaminants studied was particularly noteworthy. Our study suggests that certain riparian lands upslope and on drier soils than recognized wetlands have the capacity to improve groundwater quality part of the year, and can serve to protect both wetland and surface water ecosystems during summer months. The summer removal rates observed may not result in permanent removal of pollutants, since plant uptake (a major removal mechanism in the summer) may be a temporary sink for nutrients. To determine the long term role of VBS in the transition zone between uplands and wetland will require study during early spring, when plant uptake is limited, and will also require site specific assessments of the proportion of contaminated groundwater which is likely to interact with the riparian buffer.

Laurel Lane Study Site

Moderately well drained soil: Deerfield loamy fine sand

O horizon	4 cm. to 0 cm.
Solum	0 cm. to 61 cm.
Depth to maximal seasonal wetness	65 cm.

Somewhat poorly drained soil: Deerfield loamy fine sand

O horizon	7 cm. to 0 cm.
Solum	0 cm. to 45 cm.
Depth to maximal seasonal wetness	32 cm.

Poorly drained soil: Wareham loamy fine sand

O horizon	25 cm. to 0 cm.
Solum	0 cm. to 35 cm.

Very poorly drained soil: Scarboro mucky sandy loam

O horizon	30 cm. to 0 cm.
Solum	0 cm. to 39 cm.

Appendix 3.Ib: Description of soil series at the Frenchtown and Sandhill sites.

1. Frenchtown site- Moderately well drained soil

Oe	8 to 0 cm	partially decomposed organic matter
A	0 to 12 cm	black (10YR 2/1) sandy loam
E	12 to 16 cm	very dark grayish brown (10YR 3/2) sandy loam
B	16 to 56 cm	dark reddish brown (5YR 3/3) sandy loam
2C	56 cm +	dark reddish brown (5YR 3/2) gravelly loamy coarse sand to gravelly coarse sandy loam. Lenses of fine sandy loam.

Water Table = 71 cm from surface. (after heavy rains)

The site is positioned in a transitional area between glacial till and outwash materials. The upper material has glacial outwash textures and appearance but is poorly sorted indicating that it has been reworked at some time. Beneath the outwash material are layers or lenses of finer materials.

The B and 2C horizons have a high oxidized iron content resulting in their atypical colors.

The series that best fits this profile is a Sutton Taxadjunct. The profile does not fit the criteria for Sutton because the colors of the Bw and 2C are too red and the texture of the 2C horizon is too coarse. The series allows for fine sandy loam, sandy loam or very fine sandy loam in the C horizon. These differences are minor and the soil is similar enough in morphology and behavior to Sutton that little or nothing would be gained in recognizing a new series.

2. Frenchtown site- Somewhat poorly drained soil

Oe	8 to 0 cm	partially decomposed organic matter
A	0 to 10 cm	black (10YR 2/1) mucky fine sandy loam
Bw	10 to 66 cm	dark brown (10YR 3/3 to 4/3) sandy loam with lenses of dark brown (10YR 3/3) fine sandy loam and dark reddish brown (5YR 3/3) gravelly coarse sandy loam.
C	66 cm+	dark brown (10YR 3/3) gravelly loamy coarse sand.

Water Table = 30 cm (after heavy rains)

Appendix 3.Ib: Description of soil series at the Frenchtown and Sandhill sites.

2. Frenchtown Site- Somewhat poorly drained soil (cont.)

There is no series that would fit this profile because we do not map a somewhat poorly drained, friable till soil. The soil would be classified taxonomically as an Aeric Haplaquept. The soil would behave similarly to the Sutton and Leicester Taxadjuncts. The depth to the water table of the profile is lower than the depth to the water table of the Sutton Taxadjunct and higher than the depth to the water table of the Leicester Taxadjunct.

3. Frenchtown Site- Poorly drained soil

Oe	11 to 0 cm	partially decomposed organic matter
A	0 to 21 cm	black (10YR 2/1) mucky sandy loam
C1	21 to 71 cm	dark brown (10YR 3/3) sandy loam
C2	71 to 76 cm	grayish brown (10YR 5/2) loamy fine sand
C3	76 cm +	grayish brown (10YR 5/2) loamy coarse sand to coarse sandy loam (borderline textures)

Water Table = 21 cm (after heavy rains)

The series that best fits this profile is a Leicester Taxadjunct. The profile does not fit the criteria for the Leicester series because the texture of the A and lower C horizons is too coarse. The series allows fine sandy loam, very fine sandy loam and loam in the A horizon and fine sandy loam and loam in the C horizon. The series also allows thin pockets or lenses of silt loam, loamy sand, or sand in the C horizon. These differences are minor and soil is similar enough in morphology and behavior to Leicester that little or nothing would be gained in recognizing a new series.

4. Frenchtown site- Very poorly drained soil

Oe	0 to 8 cm	partially decomposed organic matter
Oa	8 to 13 cm	black (10YR 2/1) muck
A	13 to 20 cm	black (10YR 2/1) mucky silt loam
C1	20 to 56 cm	dark brown (10YR 3/3) silt loam to very fine sandy loam
C2	56 cm +	dark brown (10YR 3/3) gravelly sandy loam to gravelly fine sandy loam

Water Table = 6 cm (after heavy rains)

Appendix 3.Ib: Description of soil series at the Frenchtown and Sandhill sites.

4. Frenchtown site- Very poorly drained soil (cont.)

The series that best fits this profile is a Palms Taxadjunct. The profile does not fit the criteria for Palms because the organic layer is too thin. The soil is similar enough in morphology and behavior to the Palms series that little or nothing would be gained in recognizing a new series.

5. Sand Hill Site- Moderately well drained soil

Oe	6 to 0 cm	partially decomposed organic matter
Ap	0 to 22 cm	very dark brown (10YR 2/2) sandy loam
Bw	22 to 60 cm	very dark grayish brown (2.5Y 3/2) sandy loam
2C	60 to 100 cm	dark yellowish brown (10YR 4/4) loamy sand with common fine distinct olive brown (2.5Y 4/4) mottles and common medium distinct (10YR 5/8) mottles.

Water Table = 62 cm (after heavy rains)

The gravel content of the solum ranges from 0 to 5 percent and the gravel content of the 2C ranges from 5 to 10 percent. The dark colors in the solum are due in part to the dark mineralogy of the parent material. Below 100 cm, many coarse prominent dark red (2.5YR 3/6) mottles are found

The series that best fits this profile is a Sudbury Taxadjunct. The value of the Ap and the chroma of the Bw are too low to meet the criteria for Sudbury. In addition, the texture of the 2C horizon is too fine. The series allows for textures of gravelly sand to very gravelly coarse sand in the 2C. These differences are minor and the soil is similar enough in morphology and behavior that little or nothing would be gained in recognizing a new series.

6. Sand Hill Site- Somewhat Poorly drained soil

Oe	7 to 0 cm	partially decomposed organic material
A	0 to 9 cm	black (10YR 2/1) fine sandy loam
Bw	9 to 51 cm	very dark grayish brown (2.5Y 4/2) sandy loam with few faint fine mottles (low chroma)
2C	51 cm +	dark grayish brown (10YR 4/2) very gravelly loamy sand

6. Sand Hill Site- Somewhat poorly drained site (cont.)

Water Table = 32 cm (after heavy rains)

The gravel content of the solum is 0 to 5 percent and the gravel content of the 2C is 35 to 40 percent.

The drainage class of this pit, as indicated by the low chroma mottles at 51 cm, is poorly drained. However, the break between the somewhat poorly drained and poorly drained soils occurs within a narrow band and the pit was dug at the lower end of that band. An auger boring of the upper end of the band showed that the mottles occurred at a lower depth.

The series classification for this profile is Walpole.

7. Sand Hill Site- Poorly drained soil

Oe	8 to 0 cm	partially decomposed organic matter
A1	0 to 14 cm	black (10YR 2/1) mucky fine sandy loam
A2	14 to 45 cm	black (10YR 2/1) fine sandy loam
C1	45 to 69 cm	very dark grayish brown (10YR3/2) very fine sandy loam
C2	69 cm +	very dark gray (10YR 3/1) loamy coarse sand and beddings of very dark gray (10YR 3/1) loamy fine sand and very dark grayish brown (10YR 3/2) very fine sandy loam

Water Table = 22 cm (after heavy rains).

The series that best fits this profile is a Scarboro Taxadjunct. The profile does not fit the criteria for a Scarboro because the texture of the C horizon is too fine. The series does not allow for very fine sandy loam in the C horizon. These differences are minor and little or nothing would be gained by recognizing a new series

Appendix 3.Ib: Description of soil series at the Frenchtown and Sandhill sites.

8. Sand Hill Site- Very poorly drained soil

A1 0 to 10 cm black (10YR 2/1) mucky sandy loam

A2 10 to 26 cm black (10YR 2/1) mucky silt loam

2C 26 cm + very dark gray (10YR 3/1) loamy sand

Water Table = 10 cm

The coarser texture of the A1 horizon seems to be due to a deposition of coarse materials.

The series that best fits this profile is a Saco Taxadjunct. The profile does not meet the criteria for Saco because the texture of the 2C horizon above 100 cm is too coarse. The series allows for textures of silt loam or very fine sandy loam above 100 cm. The differences are minor and little or nothing would be gained in recognizing a new series.

Appendix 3.II: All samples considered affected by contamination plume at each site.

FRENCHTOWN UPLAND SITE

Sample Date (yr,mo,day)	# Days since Application	Site	Well	Observed Nitrate-N (mg/l)	Observed Chloride (mg/l)	Observed Copper (mg/l) [a]	Nitrate-N % Attenuation	Copper % Attenuation
890906	59	FT	13	0.00	6.13	ND	100.00	100.00
890914	67	FT	13	0.45	9.04	ND	80.57	100.00
890921	74	FT	13	18.21	10.53	ND	-492.42	100.00
890928	81	FT	13	21.47	17.14	ND	-236.51	100.00
891006	90	FT	13	17.65	18.17	ND	-155.93	100.00
891012	96	FT	13	1.53	18.97	ND	78.98	100.00
891020	104	FT	13	0.71	7.22	ND	50.10	100.00
891027	111	FT	13	0.29	6.51	ND	73.10	100.00
891102	116	FT	13	0.19	6.43	ND	81.67	100.00
891012	96	FT	15	0.62	7.55	ND	60.62	100.00
891020	104	FT	22	0.43	9.37	ND	82.71	100.00
891020	96	FT	24	0.00	7.43	ND	100.00	100.00
891027	111	FT	24	0.00	6.67	ND	100.00	100.00
891102	116	FT	24	0.00	6.10	ND	100.00	100.00
891012	96	FT	26	0.00	6.47	ND	100.00	100.00
891027	111	FT	26	0.00	7.04	ND	100.00	100.00
891102	116	FT	26	0.00	6.04	ND	100.00	100.00

a) ND: Below detection limit of 0.09 mg/l of copper.

Appendix 3.II: All samples considered affected by contamination plume at each site.

SANDHILL UPLAND SITE

Sample Date (yr,mo,day)	# Days since Application	Site	Well	Observed Nitrate-N (mg/l)	Observed Chloride (mg/l)	Observed Copper (mg/l)	Nitrate-N % Attenuation	Copper % Attenuation
891020	65	SH	12	10.29	45.03	ND	37.86	100.00
890921	36	SH	13	6.28	44.11	ND	84.02	100.00
890928	43	SH	13	7.28	43.38	ND	70.30	100.00
891005	50	SH	13	10.18	46.06	ND	42.66	100.00
891012	57	SH	13	6.69	31.57	ND	6.06	100.00
891020	65	SH	13	8.78	44.40	ND	53.83	100.00
890901	16	SH	15	1.52	33.74	ND	100.00	100.00
891012	57	SH	15	4.15	38.62	ND	100.00	100.00
890824	8	SH	21	3.43	34.74	ND	100.00	100.00
890901	16	SH	21	4.96	38.86	ND	100.00	100.00
890906	21	SH	21	4.98	39.01	ND	100.00	100.00
890914	29	SH	21	5.72	39.15	ND	86.96	100.00
890921	36	SH	21	2.95	33.55	ND	100.00	100.00
890928	43	SH	21	1.45	31.77	ND	100.00	100.00
891005	50	SH	21	4.64	36.99	ND	100.00	100.00
891026	71	SH	22	4.62	31.63	ND	100.00	100.00
890914	29	SH	26	0.21	32.72	ND	100.00	100.00
890921	36	SH	26	0.00	31.94	ND	100.00	100.00
891026	71	SH	26	2.34	33.35	ND	100.00	100.00
891102	78	SH	26	1.43	34.36	ND	100.00	100.00

Appendix 3.II: All samples considered affected by contamination plume at each site.

LAUREL LANE UPLAND SITE

Sample Date (yr,mo,day)	# Days since Application	Site	Well	Observed Nitrate-N (mg/l)	Observed Chloride (mg/l)	Observed Copper (mg/l)	Nitrate-N % Attenuation
890831	8	LL	13	2.42	11.15	ND	1.73
890918	26	LL	15	5.81	20.26	ND	17.26
890925	33	LL	15	4.38	16.01	0.167	10.46
891004	42	LL	15	2.01	9.92	ND	-8.80
890918	26	LL	21	3.60	11.50	ND	-36.30
890925	33	LL	21	15.63	53.37	0.095	33.69
891004	42	LL	21	2.34	13.24	ND	33.24
891011	49	LL	21	3.18	17.35	ND	42.77
891004	42	LL	24	16.12	56.14	ND	35.42
891011	49	LL	24	14.37	40.42	ND	15.96
891016	54	LL	24	5.36	18.06	ND	9.39
891023	61	LL	24	6.71	21.31	ND	11.09
891101	70	LL	24	3.74	14.11	ND	5.20
891016	54	LL	26	2.80	16.78	ND	47.00
891023	61	LL	26	2.51	14.11	ND	36.42
891101	70	LL	26	4.47	17.84	ND	23.10

Appendix 3.II: All Samples considered affected by contamination plume at each site.

SANDHILL WETLAND SITE

Sample Date (yr,mo,day)	# Days since application	Site	Well	Observed Nitrate-N (mg/l)	Observed Chloride (mg/l)	Observed Copper (mg/l)	Nitrate-N Attenuation %	Copper Attenuation %
890707	8	SH	31	16.36	29.27	ND	15.03	100.00
890711	12	SH	31	19.25	33.24	ND	17.15	100.00
890714	15	SH	31	17.43	24.79	ND	-18.35	100.00
890718	19	SH	31	11.90	22.06	0.102	1.01	100.00
890721	22	SH	31	5.90	31.01	ND	72.38	100.00
890803	35	SH	31	5.57	24.44	ND	61.97	100.00
890921	84	SH	31	0.38	25.67	ND	98.50	100.00
890928	91	SH	31	0.61	29.25	ND	97.60	100.00
891005	98	SH	31	0.75	29.11	ND	96.81	100.00
890721	22	SH	34	7.82	22.22	ND	36.25	100.00
890726	27	SH	34	0.33	22.07		98.51	-
890816	48	SH	35	0.00	35.38	ND	100.00	100.00
890823	55	SH	35	0.00	39.23	ND	100.00	100.00
890901	64	SH	35	0.00	43.74	ND	100.00	100.00
890906	69	SH	35	0.00	52.39	ND	100.00	100.00
890914	77	SH	35	0.00	32.03	ND	100.00	100.00
890921	84	SH	35	0.00	46.58	ND	100.00	100.00
890928	91	SH	35	0.00	31.01	ND	100.00	100.00
891005	98	SH	35	0.00	52.61	ND	100.00	100.00
890803	35	SH	36	1.61	23.02	0.487	88.62	100.00
890809	41	SH	36	1.35	27.31	ND	93.00	100.00
890816	48	SH	36	0.45	48.83	ND	99.21	100.00
890823	55	SH	36	0.03	32.69	ND	100.00	100.00
890901	64	SH	36	0.00	44.61	ND	100.00	100.00
890921	84	SH	36	0.00	28.76	0.109	100.00	100.00
890928	91	SH	36	0.00	52.68	ND	100.00	100.00
891005	98	SH	36	0.00	46.65	ND	100.00	100.00
890809	41	SH	44	0.89	23.89	ND	94.61	100.00
890823	55	SH	44	0.24	65.68	ND	99.84	100.00
890901	64	SH	44	0.00	47.47	ND	100.00	100.00
890906	69	SH	44	0.00	43.83	ND	100.00	100.00
890914	77	SH	44	0.00	54.00	ND	100.00	100.00
890921	84	SH	44	0.00	41.43	0.097	100.00	100.00
890928	91	SH	44	0.00	44.46	ND	100.00	100.00
891005	98	SH	44	0.00	54.95	ND	100.00	100.00
890901	64	SH	45	1.24	25.30	ND	92.80	100.00
890914	77	SH	45	1.20	25.04	ND	92.96	100.00
890921	84	SH	45	1.24	22.78	ND	91.37	100.00
890928	91	SH	45	1.08	33.19	ND	95.94	100.00
891005	98	SH	45	1.19	23.74	ND	92.31	100.00
890928	91	SH	46	1.46	21.92	ND	88.87	100.00
891005	98	SH	46	1.26	24.95	ND	92.48	100.00

Appendix 3.II: All Samples considered affected by contamination plume at each site.

LAUREL LANE WETLAND SITE

Sample Date (yr,mo,day)	# Days since application	Site	Well	Observed Nitrate-N (mg/l)	Observed Chloride (mg/l)	Observed Copper (mg/l)	Nitrate-N Attenuation %	Copper Attenuation %
890807	24	LL	32	0.34	20.63	ND	98.82	100.00
890907	55	LL	32	0.38	15.62	ND	97.03	100.00
890913	61	LL	32	0.36	15.90	ND	97.54	100.00
890925	73	LL	32	0.24	16.93	ND	99.66	100.00
890918	66	LL	34	0.26	16.28	ND	99.42	100.00
891004	82	LL	36	3.52	18.44	ND	60.03	100.00
890824	41	LL	41	0.40	17.17	ND	97.42	100.00
890831	48	LL	41	0.38	16.98	ND	97.65	100.00
890907	55	LL	41	0.44	17.12		96.79	-
890913	61	LL	41	0.34	15.95	ND	97.89	100.00
890918	66	LL	41	0.45	15.90	ND	96.00	100.00
890925	73	LL	41	0.38	17.12		97.64	-
890913	61	LL	42	2.00	16.28	ND	70.80	100.00

Appendix 3.III: Comparison of attenuation values using different background chloride concentrations.

Drainage Class	Depth	# Obs	% Nitrate-N Attenuation (high bkrd Cl)	Standard Error	% Nitrate-N Attenuation (ave bkrd Cl)	Standard Error
			Average Attenuation Values Computed Using Different Background Chloride Values			
Frenchtown Study Area						
Moderately Well Drained	Shallow	0				
	Deep	10	-35.98	62.42	-9.22	50.69
Somewhat Poorly Drained	Shallow	0				
	Deep	7	97.53	2.46	98.19	1.83
Poorly Drained	Shallow	0				
	Deep	0				
Very Poorly Drained	Shallow	0				
	Deep	0				
Sandhill Study Area						
Moderately Well Drained	Shallow	0				
	Deep	8	61.84	11.65	80.38	5.42
Somewhat Poorly Drained	Shallow	0				
	Deep	12	98.91	1.09	99.42	0.57
Poorly Drained	Shallow	19	71.46	9.19	75.46	7.89
	Deep	8	100.00	0.00	100.00	0.00
Very Poorly Drained	Shallow	2	90.68	2.56	92.20	1.94
	Deep	13	96.91	1.01	97.37	0.87
Laurel Lane Study Area						
Moderately Well Drained	Shallow	0				
	Deep	4	5.16	5.64	22.75	0.84
Somewhat Poorly Drained	Shallow	0				
	Deep	12	21.45	6.57	30.46	5.04
Poorly Drained	Shallow	0				
	Deep	6	92.08	6.42	94.47	4.57
Very Poorly Drained	Shallow	1	70.80	0.00	81.23	0.00
	Deep	6	97.23	0.29	98.19	0.17

IV. DEVELOPMENT OF AN INDEX OF BUFFERING CAPACITY

A. Introduction

Research presented in the previous chapters strongly suggests that the factors controlling the effectiveness of any particular piece of ground as a buffer strip for water quality protection are complex, and that there is great variability in VBS performance across the landscape. Similar results have been found for other studies of biogeochemical processes where hydrology, geology, biology and chemistry interact to affect the fate and transport of pollutants in the environment (Risser et al. 1988, Bedford and Preston 1988). To conduct detailed, landscape scale assessments of VBS, or other NPS control mechanisms, would require establishing a large number of intensive sampling networks and would be prohibitively expensive and time consuming. These time and cost constraints make it extremely desirable to develop indicies of buffering capacity that can be measured at a large number of sites across the landscape that differ in vegetation type, geomorphology, soil type and surrounding land uses. Indicies would potentially be useful for elucidating landscape and regional scale controls over biogeochemical processes, and for conducting rapid site-specific assessments for planning or regulatory purposes.

The key to the development of indices of biogeochemical parameters is to establish quantitative relationships between parameters that are easy to measure and those that are hard to measure (Schimel et al 1988). Parameters that are the integrative product of multiple interacting physical, chemical and biological factors are most useful for reflecting the complex control of biogeochemical processes. In this regard, biological variables such as plant productivity, species composition, and soil organic matter levels are likely more useful than static variables such as soil profile characteristics or geologic strata. On the other hand, biological variables that respond dramatically to short term fluctuations in temperature and water will not be useful as indicators of longer term biogeochemical parameters.

In this study we evaluated two microbial parameters as indices of pollutant attenuation in VBS. We hypothesized that microbial biomass C content and denitrification enzyme activity (DEA) should be strong indicators of the buffering capacity of different locations. Studies in Michigan found DEA to be a strong predictor of actual annual denitrification N gas production (Groffman and Tiedje 1989b), and current studies in Rhode Island have found that microbial biomass C is a strong indicator of N leaching losses from different land uses (Groffman et al. 1988). These parameters are strong candidates for indicies of VBS performance because they are directly related to pollutant attenuation mechanisms important in VBS (denitrification, immobilization, cation exchange) and they are the long term product of interacting physical, chemical and biological parameters (van Veen et al. 1984, Carter 1986, Azam et al.1986, Groffman 1987). We measured DEA and microbial biomass C in the soils at our well monitoring sites and at a range of sites across the Hunt-Potowomut watershed to 1) establish relationships between these parameters and the attenuation observed in our well monitoring networks, 2) to elucidate the factors responsible for the attenuation observed in our well monitoring networks and 3) to see if patterns observed in our well monitoring networks were consistent across an entire watershed.

B. Methods

Sampling

Soil samples were collected at the surface (0-15 cm) and from depths corresponding to the depths of well screening at each of the 3 intensive well monitoring sites during July and August of 1989. To collect the deep samples, overlying soil was carefully removed so that the sample consisted of the deep material only. This sample collection did not guarantee that our deep samples were free of surface material (Parkin and Meisinger 1989), but was sufficient for our purposes of general comparison. The total number of soil samples for analysis at the well monitoring sites was: 3 sites x 4 locations x 3 depths = 36.

Samples were also collected at 10 locations across the Hunt-Potowomut watershed and at an artificial wetland in Massachusetts during August - November of 1989 (Table 4-1, Figure 4-1). These sites were selected to replicate the geomorphic and upland land use stratification of our well monitoring sites. At each site, samples were collected at two locations; at the boundary between the moderately well drained and somewhat poorly drained soil (upland VBS), and at the boundary between the poorly drained and very poorly drained soil (wetland VBS). Samples were collected at the surface (0-15 cm) at 15 cm below the mean seasonal high water table (inferred from soil profile characteristics) and at 50 cm below the mean seasonal high water table. The deep samples corresponded to the shallow well screening depths at our well monitoring sites. The artificial wetland receives surface runoff from a large shopping mall (Emerald Square). Effluent passes through a wet meadow area and then through an open water (approximately 0.5 m depth) treatment area. There are three replicate wet meadow and open water "cells". In the wet meadow cells, samples were collected approximately 10 meters from the effluent inlet, in the open water cells, sediments were collected approximately 4 meters from the effluent inlet. The total number of soil samples for analysis of watershed patterns was: 10 sites x 2 locations x 3 depths = 60 (plus two artificial wetland sites = 62).

All samples were stored at 4^{o} C between the time of sampling and all analyses. Percent moisture was determined gravimetrically and soil pH was measured with a glass electrode in a 1:1 slurry of soil and water.

Microbial assays

Denitrification enzyme activity (DEA) was measured using the procedure described in chapter II of this report. Microbial biomass C content was measured using the chloroform fumigation-incubation method (Jenkinson and Powlson 1976). In this method, soils (3 replicates per soil) are fumigated to kill and lyse microbial cells in a soil sample. The fumigated sample is inoculated with fresh soil, and microorganisms from the fresh soil grow vigorously using the killed cells as substrate. The flush of carbon dioxide (CO_2) released by the actively growing cells during a 10 day incubation is directly proportional to the amount of C and N in the microbial biomass of the original sample. A proportionality constants (0.45) is used to calculate biomass C from the CO_2 flush. Carbon dioxide was measured on a Tracor 540 gas chromatograph equipped with an electron capture detector.

Table 4-1

Sites for watershed scale microbial index sampling

Sub-basin	Geomorphic setting	Upland land use
Frenchtown - Mauney (FM1-B)	Stratified drift	Low density residential
Frenchtown - Mauney (FM1-A)	Stratified drift	Low density residential
Scrabbletown (SC-6)	Stratified drift	Undeveloped
Hunt River (HP-9)	Stratified drift	Undeveloped
Frenchtown - Mauney (FM-7)	Glacial till	Non-intensive agriculture
Scrabbletown (SC-5)	Glacial till	Low density residential
Hunt River (HP-5)	Glacial till	Medium density residential
Hunt River (HP-10)	Till/alluvium	Undeveloped
Fry (F-10)	Till/alluvium	Undeveloped
Hunt River (HP-6)	Till/alluvium	High density residential
Emerald Square (MA) (ES)	Artificial wetland Open water cells Wet meadow cells	

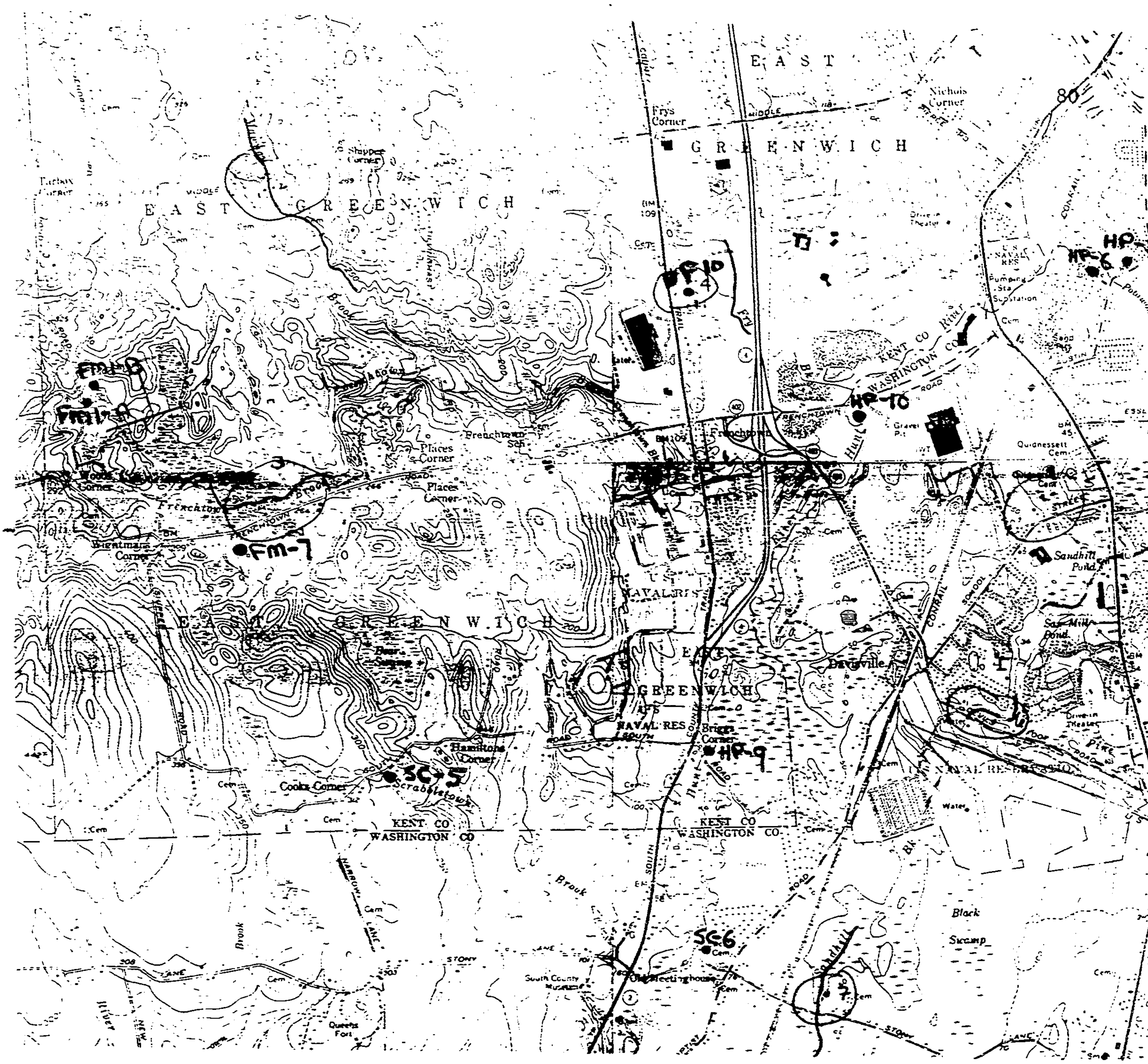


Figure 4-1. Site locations for watershed scale sampling. Map also shows locations #1-7 from figure 2-1.

Analysis

Correlation and regression analysis were used to analyze relationships between attenuation and microbial parameters in the well monitoring sites, and between DEA and pH in the watershed scale sampling sites. For these analyses, the attenuation values based on "high background chloride levels" in "deep" wells from Table X, Chapter III, and the surface soil DEA and pH values from Table 3, (this chapter) were used. We did not analyze relationships between Cu attenuation and microbial parameters because there was no variation in Cu attenuation among our sites. Microbial biomass C values for the watershed scale sampling sites are not presented due to problems with our instrument for measuring CO_2 production. These data, along with data on microbial biomass N content, will be available at a later date.

C. Results

Well monitoring sites

Spatial patterns of DEA were similar at all three sites, but differed by an order of magnitude between sites (Figures 4-2 - 4-4). At all sites, DEA was low in the upland areas (moderately well and somewhat poorly drained soils) relative to the wetland (poorly drained and very poorly drained soils). DEA levels were lowest in the outwash site with an undeveloped upland (Laurel Lane) and were highest in the outwash site with a developed upland (Sandhill) and at the till site. DEA levels were higher in surface soils than in the subsurface at all locations. Over all sites, the Spearman rank correlation coefficient between surface soil DEA and observed NO_3^- attenuation was 0.49. If the till site was eliminated from the analysis, this correlation increased to 0.83 ($p < 0.05$).

Microbial biomass C showed similar patterns to DEA (Table 4-2). Biomass C levels were lowest at the Laurel Lane site and equally high at the Frenchtown and Sandhill sites. Biomass C increased when moving from moderately well drained to somewhat poorly drained to poorly drained soils, and then declined in the very poorly drained soil at all sites. Only surface soil data on biomass C is reported due to problems with our instrument for CO_2 analysis. Data on biomass C in subsurface soils will be reported at a later date. Over all sites, the Spearman rank correlation coefficient between surface soil microbial biomass C and observed NO_3^- attenuation was 0.62 ($p < 0.05$).

Watershed scale sampling

At sites across the Hunt-Potowomut watershed, DEA was higher in wetland soils than in upland soils at most locations. Over all sites (including the well monitoring sites), DEA in upland soils averaged 115.2 ng N g^{-1} h^{-1} versus 264.6 ng N g^{-1} h^{-1} in wetland soils ($p < 0.06$). The artificial wetland sites had DEA equal or greater than the native wetland sites. There were no consistent differences between DEA at sites in different geomorphic positions and with different upland land uses. Only surface soil DEA levels are presented since these were most predictive of attenuation in the well monitoring sites.

At both the well monitoring and watershed scale sampling sites, pH tended to be higher in wetland than in upland sites. There were strong relationships between DEA and pH in both upland (r^2=0.41, $p < 0.02$) and wetland (r^2=0.32, $p < 0.03$) sites.

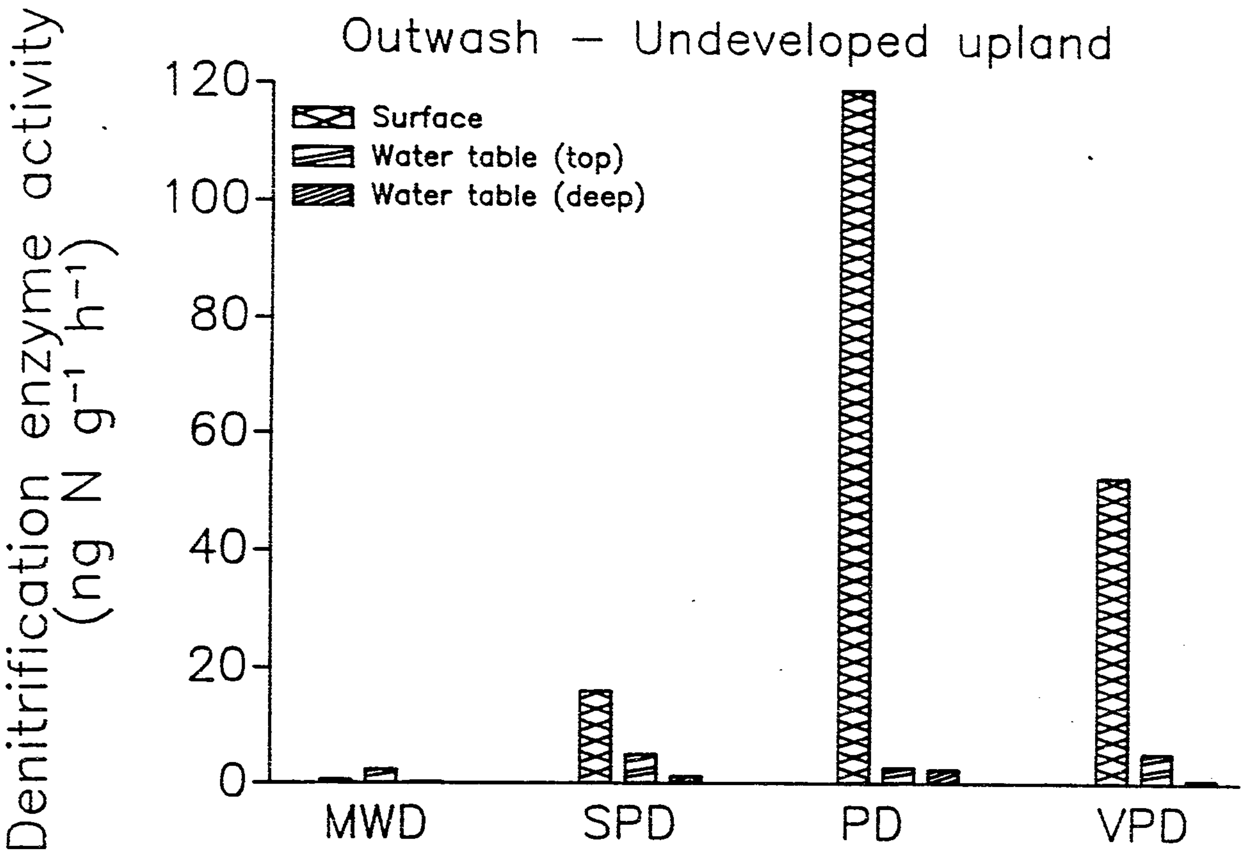


Figure 4-2. Denitrification enzyme activity in moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD) soils at the Laurel Lane well monitoring site.

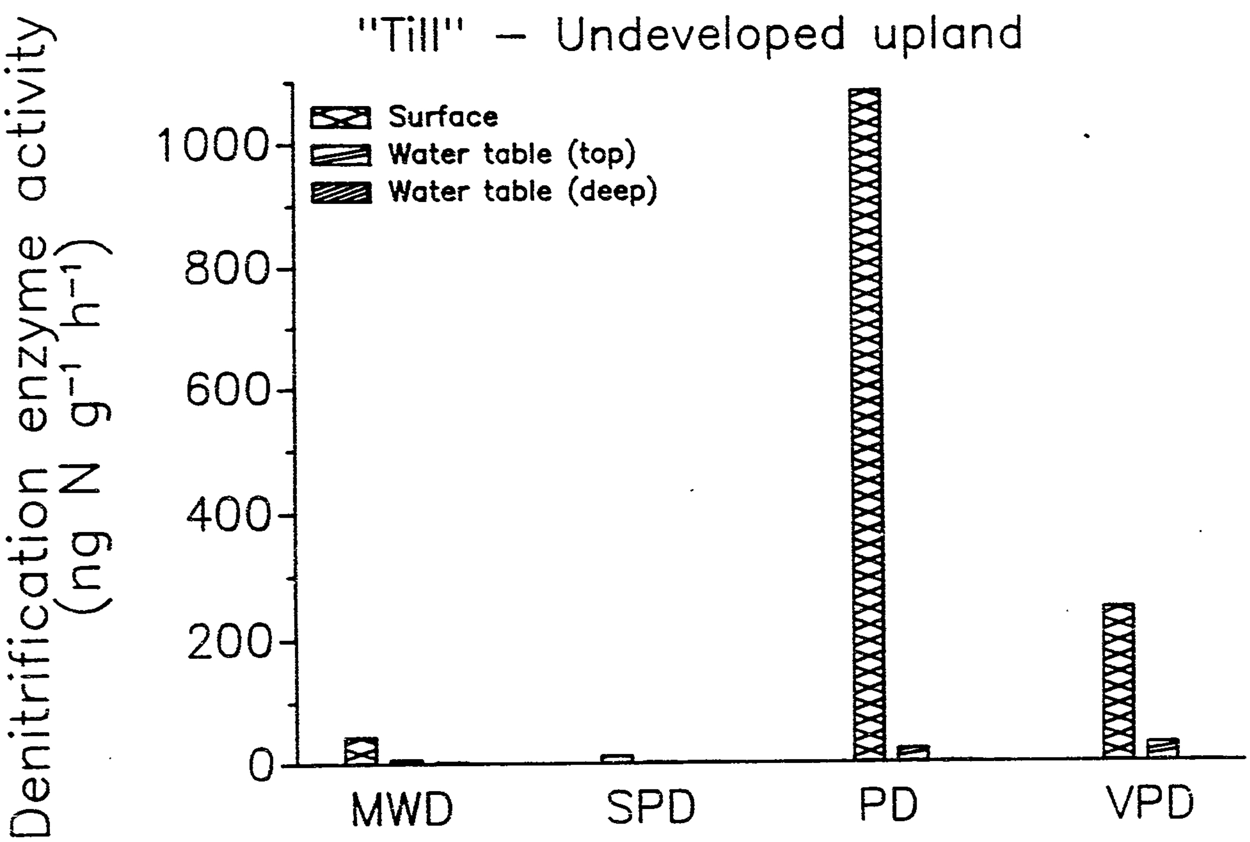


Figure 4-3. Denitrification enzyme activity in moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD) soils at the Frenchtown well monitoring site.

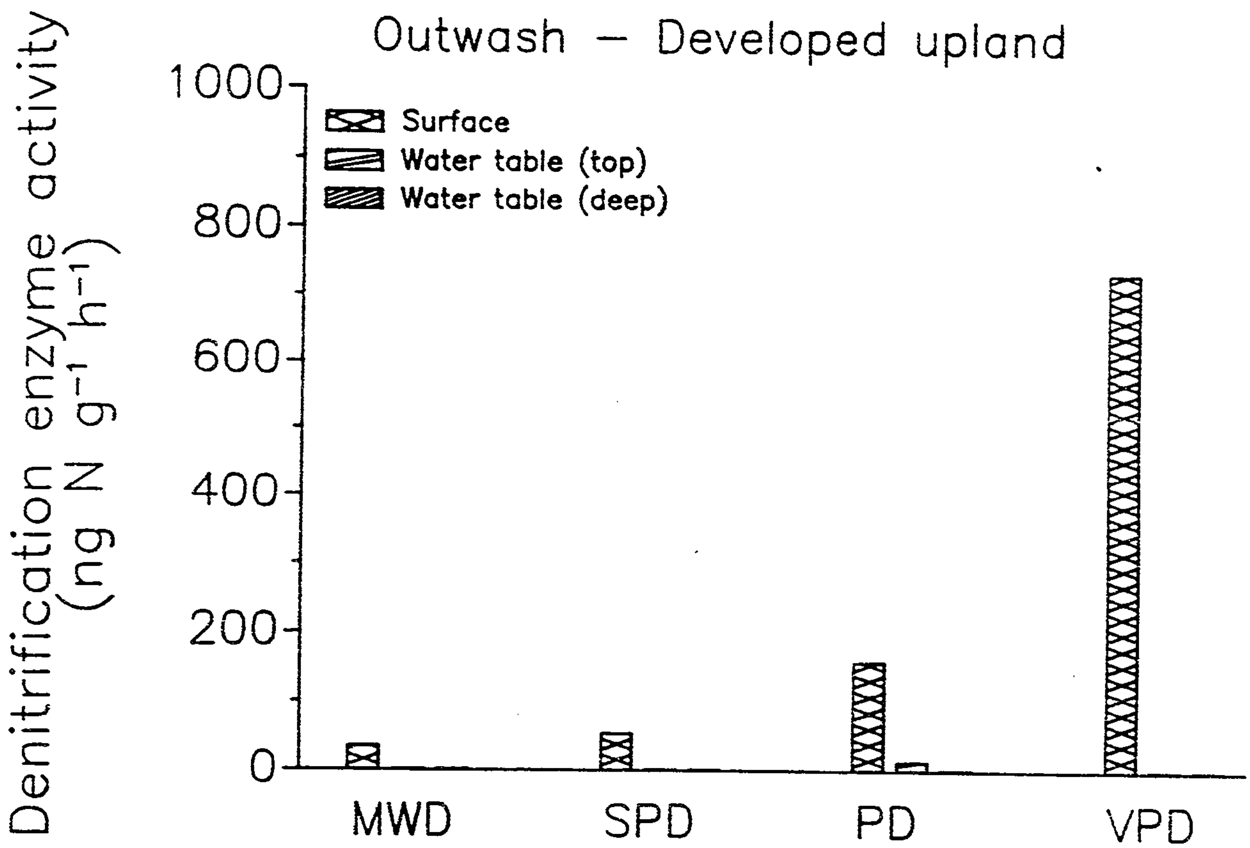


Figure 4-4. Denitrification enzyme activity in moderately well drained (MWD), somewhat poorly drained (SPD), poorly drained (PD) and very poorly drained (VPD) soils at the Sandhill well monitoring site.

Table 4-2

Soil microbial biomass C in surface soil (0-15 cm) and observed nitrate attenuation (%, from Table 3-3) at well monitoring sites.

Site	Microbial biomass C (μg/g soil) mean (standard error)	Observed % Attenuation
Frenchtown (till, undeveloped upland)		
Moderately well drained soil	336 (177)	0
Somewhat poorly drained soil	396 (81)	98
Poorly drained soil	1400 (138)	ND
Very poorly drained soil	499 (95)	ND
Laurel Lane (outwash, undeveloped upland)		
Moderately well drained soil	107 (68)	5
Somewhat poorly drained soil	279 (128)	27
Poorly drained soil	650 (31)	94
Very poorly drained soil	ND	97
Sandhill (outwash, developed upland)		
Moderately well drained soil	393 (24)	62
Somewhat poorly drained soil	430 (101)	99
Poorly drained soil	435 (104)	100
Very poorly drained soil	169 (169)	97

ND - not detectable

Table 4-3

Denitrification enzyme activity (DEA ng N g^{-1} h^{-1}) and pH in surface soils at watershed scale sampling sites

Site ID	Geomorphic setting	Upland land use		DEA	pH
LL*	Outwash	Undeveloped	Upland	15.9	3.88
			Wetland	52.6	3.92
FT*	Till	Low density residential	Upland	11.1	3.86
			Wetland	247.4	4.16
SH*	Outwash	High density residential	Upland	53.7	4.40
			Wetland	732.5	5.86
FM1-B	Outwash	Low density residential	Upland	162.5	4.71
			Wetland	153.8	5.05
FM1-A	Outwash	Low density residential	Upland	91.9	4.69
			Wetland	1004.8	5.33
SC-6	Outwash	Undeveloped	Upland	40.5	4.28
			Wetland	29.7	4.60
HP-9	Outwash	Undeveloped	Upland	42.0	5.04
			Wetland	156.9	4.52
FM-7	Till	Non-intensive agriculture	Upland	558.7	5.79
			Wetland	396.6	6.00
SC-5	Till	Low density residential	Upland	5.3	5.02
			Wetland	17.9	4.81
HP-5	Till	Medium density residential	Upland	29.9	4.92
			Wetland	660.0	4.64
HP-10	Till/alluvium	Undeveloped	Upland	84.4	3.38
			Wetland	130.2	3.58
F-10	Till/alluvium	Undeveloped	Upland	M	4.11
			Wetland	18.4	4.17
HP-6	Till/alluvium	High density residential	Upland	29.4	4.59
			Wetland	40.9	5.24
ES	Artificial wetland		Wet meadow cells	300.3	5.43
			Open water cells	1987.5	5.75

*Well monitoring sites, upland = somewhat poorly drained soil, wetland = very poorly drained soil
FM - Frenchtown-Mauney
SC - Scrabbletown
HP - Hunt River
ES - Emerald Square artificial wetland

D. Discussion

We were able to establish some useful relationships between microbial parameters and pollutant attenuation, but several factors reduced the strength of these relationships. Microbial biomass C was the strongest predictor of attenuation, but low biomass C values in the very poorly drained soils reduced the strength of the relationship. DEA was a strong predictor of attenuation only if data from the Frenchtown (till, undeveloped upland) site was eliminated. Surface soil levels of DEA and microbial biomass C were the strongest predictors, which is a very useful result for the development of a widely applicable index technique (it is very labor intensive to delineate and collect samples from specific subsurface zones). Both parameters were conservative predictors of attenuation; in no case did they predict high attenuation when low attenuation was observed. The microbial parameters failed as predictors only when high attenuation was observed in locations where low attenuation was expected.

We hypothesized that DEA would be a strong predictor of NO_3^- attenuation in VBS because denitrification is potentially a major sink for NO_3^- in buffer areas. Although "potential" measurements of denitrification are only tenuously related actual N gas fluxes at any point in time, we have found that DEA is strongly related to annual N gas flux at a site (Groffman and Tiedje 1989b). DEA has been found to be a poor predictor of hourly or daily denitrification rates, due to the presence of a large pool of inactive (but detectable by the DEA procedure) enzymes in soil (Smith and Parsons 1985, Groffman 1987, Martin et al. 1988). Due to this pool, DEA varies very little over the course of the year relative to factors such as hourly denitrification rates, soil moisture or soil NO_3^- levels. DEA should be viewed as a long-term, integrative product of multiple physical and biological factors. Long term factors are more likely to be useful predictors of complex and longer term (annual) biogeochemical processes such as annual N gas flux than of short term phenomena such as hourly or daily N gas flux.

Microbial biomass C was expected to be a useful predictor of pollutant attenuation in VBS because it represents the source of activity for microbial immobilization of nutrients and metals. Similar to DEA, microbial biomass C is a long term product of multiple physical, chemical and biological parameters (van Veen et al. 1984, Carter 1986) and should be more strongly related to long term, complex biogeochemical processes than to short term phenomena. Microbial biomass C should be a more general indicator of microbial processes at a site than DEA, which may explain why it was a stronger predictor of attenuation.

The predictive power of the microbial parameters was weakened by the importance of plant uptake as a sink for NO_3^- in the VBS during the groundwater study (chapter III). This was especially important in upland sites where high NO_3^- attenuation was observed in areas with very low DEA and microbial biomass C. This was particularly a problem at the Frenchtown site, and eliminating this site from correlation analysis greatly improved the predictive power of DEA. Since plant uptake may not be a reliable sink for NO_3^- on a seasonal and/or long term basis (see discussions in previous chapters), DEA, in combination with estimates of water table depth, may serve as a useful, conservative index of long term NO_3^- attenuation capacity at a site. In addition, the predictive power of the microbial parameters is likely to be much greater during periods when plants are inactive. We hope to test this idea by repeating our well monitoring experiment in early spring of 1990.

We expected to observe low levels of DEA in upland areas since the anaerobic conditions necessary for denitrification are rare in upland soils. We observed a very strong relationship between "depth to water table" and attenuation capacity among our sites (see chapter III), which suggests that when groundwater pollutants are closer to the surface, they are more likely subject to biological and chemical attenuation processes. Our measurements of DEA and microbial biomass C suggest that in upland VBS, plant uptake is the major potential source of attenuation, while denitrification, microbial immobilization and plant uptake are likely to be active in wetlands. Determining the reliability of plant uptake as a nutrient sink will be critical for evaluating the potential of upland areas to act as effective buffers for wetlands. Determining the ability of microbial parameters to function under low temperatures (when plants are not active) will be critical for evaluating the potential of wetland areas to act as effective buffers for surface water bodies. Poorly drained soils, which usually fall in the upland/wetland transition zone, had the highest levels of DEA at two of the three well monitoring sites. Our results suggest that wetland delineation and regulation efforts directed toward water quality should consider the potentially high water quality maintenance value of these soils.

The high biomass C and DEA observed at the Sandhill site (outwash with high density residential development in uplands) relative to the Laurel Lane site (outwash with undeveloped upland) suggest that pollutant inputs may induce buffering capacity. Nutrient inputs can stimulate both plant (Ehrenfeld 1987) and microbial (Smith and Duff 1987) populations, increasing their potential to act as sinks. Alternatively, the higher attenuation observed at the Sandhill site may only be a function of the shallower depth to groundwater at this site relative to the Laurel Lane site. Further comparisons of VBS with developed and non-developed uplands is necessary to resolve this question. Understanding the factors controlling the limits of "induced buffering potential" will be important for buffer planning and assessment in undeveloped areas.

The watershed scale sampling confirmed observations from the well monitoring sites (and from watershed scale sampling reported in chapter II) that wetland sites have higher denitrification potential than upland sites. However, we did not observe consistent patterns of DEA with geomorphic setting and upland land use in the watershed scale sampling. Data from the well monitoring sites suggested that till sites have a higher inherent potential to support denitrification than outwash sites, and that intensive upland land use can induce high denitrification potential in outwash sites that would normally have low potential. However, the watershed scale results suggest that other factors, such as depth to water table and/or soil pH either influence or override effects of geomorphic position and land use on denitrification potential. On the other hand, differences in soil pH across the landscape may be related to geomorphic position and/or present and/or past land use. Future work is needed to establish the landscape scale factors that control soil pH. There is also a need for comparative measurements of DEA in sites of differing geomorphic setting and upland land use that control for soil pH and depth to water table.

Evaluating the landscape scale controls over denitrification potential are important for planning and assessment of VBS. The microbial parameters, DEA and microbial biomass C, will be useful conservative tools for this evaluation. Our data from the artificial wetland site provide an example of how these tools can be applied. DEA data from this site suggests that the wet meadow cells denitrify at a higher rate than many (but not all) native wetland sites. The open water cells supported the highest DEA levels measured in our sampling. This high biological attenuation capacity, coupled with strong hydrologic control of influent and effluents at the site, suggest that these artificial wetlands are effective NPS pollutant control mechanisms.

E. Summary of major findings and management implications

1. Surface soil levels of microbial biomass C and/or DEA may be useful as conservative indicies of VBS pollutant attenuation potential.

2. Plant uptake appears to be the major NO_3^- sink in upland VBS while plant uptake, denitrification and microbial immobilization operate as sinks in wetland VBS. There is a strong need for further research on pollutant attenuation in VBS during periods when plants are not active.

3. Poorly drained soils, which usually fall in the upland/wetland transition zone, appear to have a high potential water quality maintenance value that should be considered in wetland delineation and regulation efforts oriented towards water quality.

4. Soil pH appears to be more important than geomorphic position or upland land use in controlling VBS pollutant attenuation capacity across the landscape. There is a strong need for further research to determine the landscape scale factors that control soil pH.

5. The artificial wetland systems studied had as high or higher potential for biological attenuation of pollutants than native wetlands.

Management implications

Our site specific studies (chapters II and III) suggested that there are several key factors that control the water quality maintenance value of VBS. Understanding how these factors are expressed across the landscape will be essential for determining areas to be preserved as VBS, for assessing the performance of existing VBS, and for designing and evaluating new NPS control techniques. The microbial indicies developed may be useful tools for these landscape scale assessments. Further research is needed to determine how these indicies perform during periods when plant uptake is not active, and to determine standard protocol for their implementation as assessment tools.

V. WILDLIFE STUDIES

A. Introduction

Significance of Wetland Buffers to Wildlife Species. Wetland buffers usually are established to protect adjacent wetlands. We will consider a wetland buffer to be comprised of (1) the transition zone that lies between what is unconditionally identified as upland and wetland, and (2) a portion of the adjacent upland. The width of the buffer, or how far the buffer should extend into the upland, is one of the subjects of this report.

Most often, buffers are associated with pollution attenuation and water quality. However, buffers may be important to wetland wildlife, and represent unique habitats with important inherent values. Why do these transition zones support such high diversities and abundances of wildlife? Several values of wetland buffers to wildlife have been suggested:

1. Species richness
2. Sites for foraging
3. Corridors of dispersal
4. Escape from flooding
5. Sites for hibernation
6. Areas for breeding and nesting
7. Areas of low predator and nest parasite density
8. Buffering of disturbances from outside of the wetland

1. Species richness. The ecological conditions in buffer transition zones are highly dynamic; they resemble neither true wetland nor upland, but fluctuate greatly and may resemble each during different times of the year or from one year to the next (Brown et al. 1987). Plant and animal diversity in buffer zones is comprised of both wetland and upland species, and may contain species specific to the zone. Thus, the habitat overlap in the transition zones of buffers often yields a relatively high species diversity (MacArthur and Pianka 1966, Allen 1962, Ranney 1977). Studies of fauna in forest edges have supported this trend; Vickers et al. (1985) in a Florida study found that herpetofauna diversity and abundance were greater along the edges of six wetlands compared to adjacent wetland and upland areas. Harris and Vickers (1984) reported that virtually all mammals, because they are cursorial and most eat plants, reside in "peripheral" areas such as transition zones. In a Florida study, Harris and McElveen (1982) found a great abundance and diversity of breeding birds in the ecotone between cypress wetlands and clearcut areas.

A wetland buffer also represents an "edge," which is defined as "the place where plant communities meet or where successional stages or vegetative conditions within plant communities come together (Thomas et al. 1979)." The concept of edge has been well established in the field of wildlife biology ever since Aldo Leopold (1933) stated that wildlife is a "phenomenon of the edge." Leopold's premise was that most species of wildlife needed more than one habitat type to survive; this was supported by his observations that wildlife "occurred where the types of food and cover which it needs come together." Subsequent studies reported increases in species richness of birds (Lay 1938, Beecher 1942, Galli et al. 1976, Laudenslayer and Balda 1976, McElveen 1977, Gates and Gysel 1978), mammals (Bider 1968, Forsyth and Smith 1973, Matthiae and Stearns 1981)), and other groups of fauna.

In recent years, studies have revealed that edge creation is not a panacea of wildlife management; in fact, certain types of edges have negative effects on wildlife (reviewed in Noss 1987). The effect of the edge is often a function of whether it is inherent or induced (Thomas et al. 1979). Inherent edges are those which are the product of natural phenomena; they represent a mixing of communities with a continuum of successional stages. Inherent edges result when disturbances such as fire, storms, flooding or other natural events create heterogeneity in the natural landscape (White 1979, Sousa 1984, Pickett and White 1985). Generally, inherent edges result in high diversity and abundance of wildlife species.

Induced edges, those which are the result of human manipulation, often have the opposite effect. Induced edges have more abrupt changes between habitats; the resulting fragmentation of habitats may produce an "island effect" (MacArthur and Wilson 1967). Small, isolated habitats are less capable of supporting as many species of wildlife, particularly area-sensitive, interior species. These interior species may be replaced by more opportunistic "weed" species that are associated with early successional stages. As induced edge increases, the faunal composition of the landscape shifts from species associated with large, stable ecosystems to those usually restricted to more temporal habitats (Faaborg 1980, Samson 1980, Noss 1981, 1983, Samson and Knopf 1982, Harris 1984, Noss and Harris 1986). For example, human development of land may create a boundary of early succession habitat around a forested wetland. The disturbed habitat may prove to be a source of "weed" species of plants and animals which will invade the wetland and alter its natural diversity (Janzen 1983). Janzen (1983,1986) suggests that opportunistic animals which reach high densities in surrounding disturbed habitats often invade pristine, interior habitats. The results can be excessive trampling, browsing and grazing, and seed predation.

Harris (1984) gives us a general principle, "The greater the contrast between habitat types, the greater the edge effect." Brown et al. (1987) state that, "The negative effects of induced edge are easier to understand when one considers that edge is more than a one-dimensional boundary between habitat types. Effects of an open, disturbed habitat penetrate some distance into an adjoining wooded natural area; hence, edge has a width of influence." Wetland buffers are zones of transition, which function as inherent edges between wetland and upland communities. The wider the buffer, the less influence the adjacent disturbed area will likely have on the wetland community.

2. Sites for Foraging. Wetland buffers provide foraging sites for many species of wildlife. Harris et al. (1979) concluded that peak mast production for uplands and wetlands were different; winter and spring is the peak fruiting season for most wetland species, while summer and fall were peak for most upland species. Thus, animals that live near or in wetland buffers are likely to have more mast available to them.

3. Corridors of dispersal. Most animals before reaching sexual maturity disperse from the home range in which they were born and reared to a new location. The distances that animals disperse may range from less than 0.3 km among some small mammals (Alfred and Beck 1963) to about 160 km by red foxes (*Vulpes vulpes*) (Phillips et al. 1972). Dispersing animals are subject to higher mortality rates than resident animals, because they are young, inexperienced, and travel through unfamiliar or hostile terrain. Dispersal is important to species survival because (1) it helps maintain genetic heterozygosity, (2) areas of depleted populations are repopulated (Fahrig and Merriam 1985), and (3) a species can spread into suitable habitat as it becomes available (Robinson and Bolen 1984).

Wetland buffers may serve as corridors for animal dispersal between blocks of habitat. As humans create more isolated "habitat islands," corridors of dispersal may become increasingly important to the maintenance of localized wildlife populations. Small populations cut off from other populations may experience a "genetic bottleneck," inbreeding depression, a loss of heterozygosity, and a variety of other genetic problems, all of which may result in decreased fitness. The loss of genetic diversity may increase the immediate, as well as long-term probability of extinction of the species (Soule 1980, 1986; Wilcox 1980). Wildlife corridors can be defined as strips or parcels of land which allow safe passage of wildlife between larger blocks of habitat (Brown et al. 1987). Such corridors can increase the effective breeding size of a population by allowing the intermixing of genetic material from various small populations.

Brown et al. (1987) review the use of corridors by a variety of wildlife species. They note that carnivores such as cougars, bobcats, bears, and otters use corridors extensively in order to link habitat blocks that would otherwise be inaccessible to them. These top-level carnivores could not support themselves energetically on smaller habitat blocks. Wild turkeys routinely use corridors for travel, foraging, and roosting areas (Gehrken 1975, Kilpatrick et al. 1987). Many species of birds and small mammals use fencerows and hedgerows because they provide the safety of dense plant cover (Bull et al. 1976, Wegner and Merriam 1977). In one study (MacClintock et al. 1977), interior forest birds were found to use isolated forest fragments that were connected by corridors to extensive forest blocks.

The "edge effect" also affects the value of buffers as corridors for wildlife. Narrow "line" corridors show many of the negative effects of induced edges and generally function only as corridors of travel; wider "strip" corridors can retain some of the values of interior conditions (Brown et al. 1987). Generally, the wider the corridor, the more actual habitat there is for wildlife species (Forman 1983). Numerous studies of bird species richness in riparian zones of varying width have supported this (Stauffer 1978, Tassone 1981). Tassone (1981) suggests that buffers should be at least 100 m wide on larger streams to provide adequate corridors between larger habitat blocks.

It is intuitively logical that wildlife use buffers as travel lanes which interconnect larger habitat blocks. In a review of the subject, Adams and Dove (1989) found little empirical evidence which documents the use and value of such corridors. However, they concluded that the advantages of corridors outweigh the disadvantages, particularly in urban-suburban settings; they also noted that much more research on corridors is needed.

4. Escape from flooding. Wetland buffers often provide a refuge for some terrestrial wetland species during times of high water. Harris and Vickers (1984) found that many species of mammals that reside in wetlands moved to peripheral areas during times of high water levels. This suggests that their survival depends upon the availability of dryer habitats in surrounding transition zones (Larson 1981).

5. Sites for hibernation. Many heterothermic mammals avoid freezing temperatures by hibernating in a burrow or cave. Hibernation in mammals is an inactive state in which all metabolic activities are lowered, and body temperature significantly drops. For example, a woodchuck's (*Marmota monax*) body temperature may drop from an active level of 37° C to less than 4° during hibernation (Cockrum 1962). Buffer zones represent important sites for the hibernation of many mammals; wetland water tables are too high to allow the creation of burrows.

Many herp species also hibernate (DeGraaf and Rudis 1986). For example, the marbled (Ambystoma opacum) and spotted (Ambystoma maculatum) salamanders hibernate in burrows created by rodents. The redback salamander (Plethodon cinereus) may hibernate down to 38 cm in the soil or in rock crevices; although they have been found hibernating in aquatic habitats (DeGraaf and Rudis 1986).

6. Areas for breeding and nesting. Buffers on wetlands sometimes have high avian species diversity. Forman et al. (1976) found that forest islands in New Jersey supported a higher species diversity of birds than did equally-sized sample plots in forest interiors; both forest edge and interior birds bred and nested in the buffers. The impact of forest fragmentation on birds was later reviewed in detail by Whitcomb et al. (1981).

7. Areas of low predator and nest parasite density. A number of both wild and domestic predators and parasites are associated with disturbed habitats, induced edges, and human development. These predators and parasites are often attracted to induced edges where they can seriously reduce interior species populations. Whitcomb et al. (1976) showed that nest parasites (brown-headed cowbirds [Molothrus ater]), nest predators (small mammals, grackles [Quiscalus sp.], jays [Cyanocitta sp.], and crows [Corvus sp.]), and non-native nest-hole competitors (e.g., European starlings [Sturnus vulgaris]) commonly invade narrow riparian strips and small forest tracts. In a Michigan study, Gates and Gysel (1978) showed that forest-field edges acted as "ecological traps;" birds nesting near the edge had smaller clutches and were more subject to predation and cowbird parasitism than those in fields or interior forest habitats.

Nest predation studies by Wilcove (1985) and Wilcove et al. (1986) documented significantly higher rates of predation for birds in small forest tracts compared to large tracts, in forests surrounded by houses compared to forests surrounded be agricultural land, and a decreasing distances to forest edge. These studies indicated the deleterious effects of predation may extend 300-600 m inside a forest border.

8. Buffering of disturbances from outside of the wetland. Some wildlife species are very sensitive to disturbances from outside of their habitats. Buffers probably tend to ameliorate disturbances, and may improve species richness by simply adding size to the wetland habitat island. Lynch and Whitcomb (1978) reviewed research on long-term shifts in the composition of avian communities relative to the (1) size of the habitat unit, (2) its level of isolation, and (3) level of human-related disturbances to such areas. They concluded that there was no consensus as to the relative importance of these three factors, and that the disturbance factor is confounded with size and level of isolation.

In a Wisconsin study, Temple (1986) found that over a third (16/43) of the bird species ordinarily found on forest tracts were not found in smaller-sized fragments. These birds also were absent in larger-sized tracts that lacked a core area more than 100 m from sharp, induced edges; this phenomenon was even more apparent in narrow corridors. Tilghman (1987) examined bird richness in 32 Massachusetts woodlots which were isolated by urban landuses and ranged in size from 1 to 69 ha. She found that size was the most important factor (explained 79% of variability) in explaining changes in woodland bird richness. The number of bird species increased rapidly along with patch size up to about 25 ha (75% of "pool" species); beyond that size the slope of the species-area curve became more gradual.

In a Virginia study, Tassone (1981) characterized a number of species that require large tracts of undisturbed, closed-canopy forest habitat. These fragment-sensitive or interior species were most common in Virginia riparian buffers that were greater that 62 m in mean width; he suggested that until more research is done, buffers should be at least 60 m in width to provide breeding areas for fragment-sensitive species. On larger streams, buffers should be at least 100 m in width to optimize use of the riparian zone by avifauna. In a Maine study (Small and Johnson 1985), the minimum suggested riparian buffer strip was 75 m to protect woodland bird communities. This value was further supported by Johnson (1986) in a study of avian use of lakeshore buffer strips by breeding birds.

Robbins (1980) reviewed the literature on area-sensitive birds and conducted new research on the subject. He pointed out that area-sensitive species are mainly long-distance insectivorous migrants that winter in the New World tropics (e.g., flycatchers [Family Tyrannidae], vireos [Vireo sp.], and wood warblers), whereas edge habitats were filled with short-distance migrants (eg., blackbirds, catbirds [Dumatella carolinensis], house wrens [Troglodytes aedon], jays, robins [Turdus migratorious], starlings, etc.) and non-migratory, resident species. Whitcomb et al. (1981) concluded that (1) the number of edge species is negatively correlated with habitat island size, (2) the number of forest-interior species is positively correlated with size, and (3) no correlation exists between ubiquitous species and size.

Noise from humans and their associated activities may have a significant influence on certain species, however, research into the impacts of noise on wildlife is scant (Brown et al. 1987). Robertson and Flood (1980) found that bird species diversity was negatively correlated with amount of disturbance between sites along the Ontario shoreline. Brown et al. (1987) state that, "Several species which nest in Florida (yellow-billed cuckoo [Coccyzus americanus], yellow-throated vireo [Vireo flavifrons], American redstart [Setophaga ruticilla], and pileated woodpecker [Dryocopus pileatus]) were more abundant in, or restricted to, the less disturbed transects."

Although the exact impacts of disturbances on wildlife are yet to be quantified, they may include at least: auditory interference of activities such as courtship and mating, prey location, predator detection, and homing; stress-related effects; sleep disruption; annoyance; and decreased reproduction (Brown et al. 1987). In addition to disturbance factors, nearby human development has been documented to cause increased predation and harassment by cats and dogs (Errington 1936, McMurry and Sperry 1941, Bradt 1949, Hubbs 1951, Parmalee 1953, Eberhard 1954, Korschgen 1957, Smith 1966, Corbett et al. 1971, Jackson 1971, Gavitt 1973, George 1974, Gill 1975, van Aarde 1980). Cats associated with households have home ranges up to 253 ha (Warner 1985); the home range of free-ranging dogs in Baltimore was about 1.7 ha (Beck 1973).

In a review of the management of amphibians, reptiles, and small mammals in North America, Gibbons (1988) outlines the premise that, "Amphibians, reptiles, and small mammals need special consideration in environmental management because: (1) they are significant biotic components in terrestrial and freshwater habitats; (2) research and management efforts have lagged behind those on other vertebrates; (3) a stronger understanding of their ecology and life history is needed to guide management decisions; and (4) their importance has not been promoted satisfactorily to develop the proper public attitude."

According to Bury et al. (1980), reptiles and amphibians (a.k.a. herps or herpetofauna) make up 30% of the native terrestrial vertebrate species of North America north of Mexico. The herps on many sites have a greater biomass than that of birds or mammals; the potential for herps to influence the overall community is great. Amphibians, reptiles, and small mammals are often the top-level predators in an ecosystem. Most of the time we tend to exclude herps and small mammals from consideration in habitat evaluation and management:

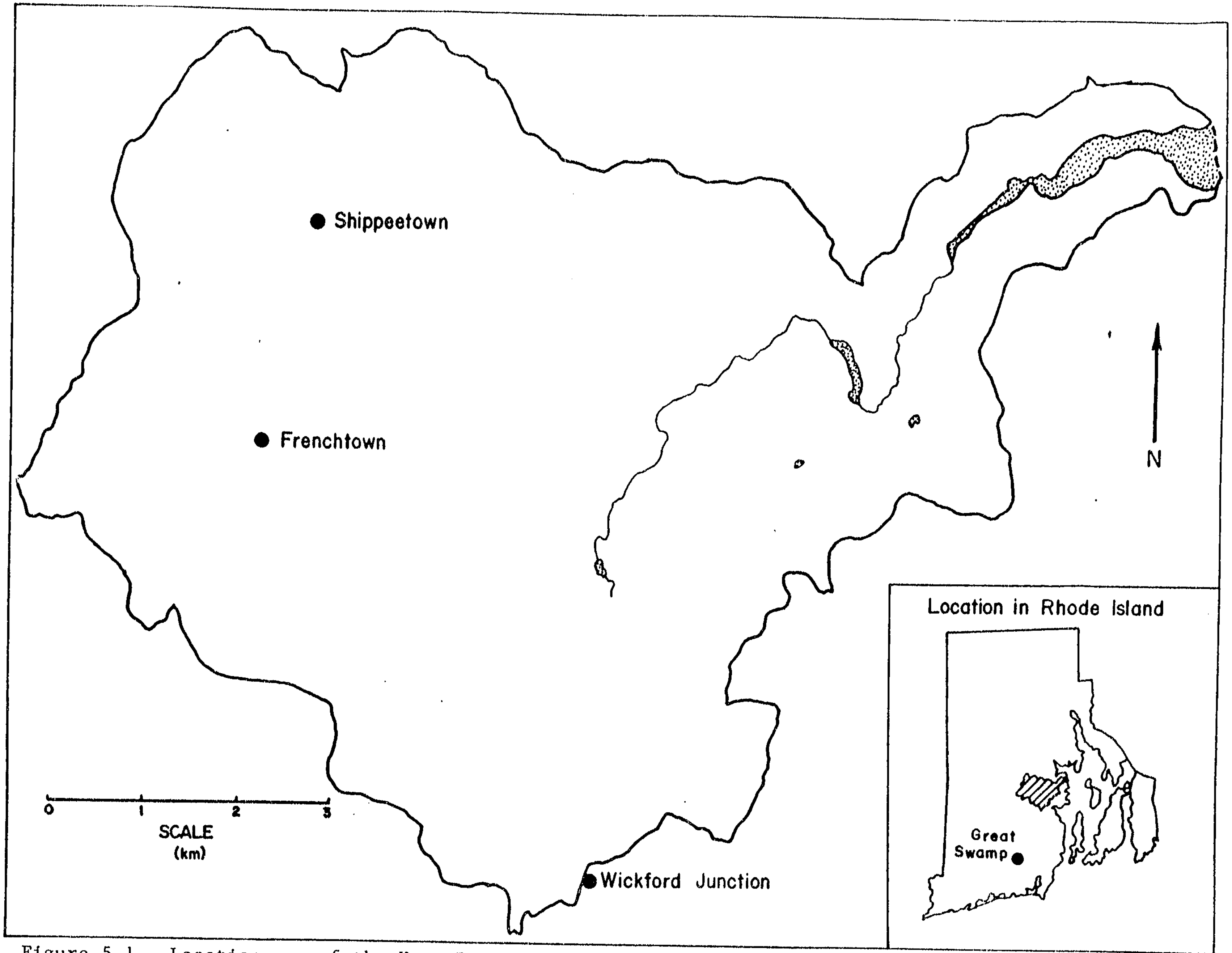


Figure 5-1. Location map of the Hunt-Potowomut Watershed and four red maple swamp study sites.

they have a secretive nature and are seldom observed. Unfortunately, we know little about the specific habitat requirements of small mammals; even less is known about most herps. We do know that the ecological reactions of herps and small mammals are often highly site specific and depend upon details of the local environment (Bury et al. 1980).

Our study was designed as a pilot. Because of limited funding and resources, the scope of the work included a general survey of herps found in relatively wide buffer strips. The results of this study yields baseline information which may be useful in future studies of buffers of less width or those located in areas of greater disturbance. Field methods of this study were designed to: (1) determine species richness of herpetofauna in buffer strips, and (2) describe wildlife habitat parameters in those same buffer strips. In addition to meeting the planned objectives, we incidentally gathered information on species richness of small mammals in buffer strips.

The objectives of the avian portion of the study were to: (1) document the occurrence of breeding bird species across the transition from upland oak forest to red maple swamp wetland, (2) determine which species were more associated with upland forest and with red maple swamp, (3) determine the configuration of bird territories at the upland/wetland boundary, and (4) determine characteristics of vegetation present in the upland/wetland transition that might result in changes in bird distribution.

B. Study Areas

The study areas are mostly in the Hunt-Potowomut River watershed, which is in the towns of East Greenwich, North Kingstown, and West Warwick, Rhode Island. This watershed was chosen as the primary site because the proposed research is part of a larger water quality study of this area sponsored by the Narragansett Bay Project (NBP). Land uses nearby include new and existing residential developments, industry, commerce, farms, sand and gravel mines, highways, wetlands and a variety of other types of open space. The area is relatively undeveloped when compared to the major urban centers surrounding Providence, but will ultimately experience substantial growth.

Preliminary selection of buffer strip study sites was made from aerial photographs, and final selection of four sites (Fig. 5-1) was made from the ground in February, 1989. Criteria for the selection of sites included location in the Hunt-Potowomut drainage because of the geographic restrictions on the study, presence of a relatively wide upland forest around the adjacent wetland, large wetland size to avoid swamp area effects on species richness (Merrow, in prep.), and isolation from large-scale disturbances. We had a great deal of difficulty finding four sites within the Hunt-Potowomut watershed that met our criteria, and finally chose the fourth and last site off the watershed in the Great Swamp Management Area. The lack of available study sites on the watershed reflects the level of development which has already occurred. The Great Swamp transects were also selected to provide an undisturbed site for comparison to the Hunt-Potowomut sites. Because the two avian sampling transect lines at Great Swamp were adjacent to each other, data from avian censuses and vegetational measurements were combined for the analyses. The following sites were chosen for the study:

<u>Great Swamp Site</u>. This site (outside of the Hunt-Potowomut watershed) is located in the Great Swamp State Management Area, South Kingstown, RI. The management area is 1,200 ha of undeveloped land. A herpetofaunal capture array

was located near a red maple wetland, part of a streamside wetland complex adjacent to the Chipuxet River, which flows into Worden's Pond. The wetland edge is sharply defined at this site with a well-drained oak upland. The wetland has a dense understory of sweet pepperbush (Clethra alnifolia) and greenbriar (Smilax sp.). Two transects were placed in this area for sampling avifauna, one oriented in a north-south direction and one on an east-west axis toward the river.

Wickford Junction Site. The red maple wetland here is part of a large forested wetland which is drained by an intermittent stream. The site is fairly isolated area with the exception of a railroad track adjacent to the upland. The track isolates the site from a marsh and swamp complex on the opposite (west) side of the tracks. The wetland edge is fairly well defined at this site with vegetation similar to the Great Swamp site. One transect for sampling avifauna was placed on the south end of this wetland, oriented on a north-south axis.

Shippeetown Road Site. This site lies next to a red maple wetland along an intermittent stream that flows into a perennial stream about 0.2 km away. The area along the stream is relatively undisturbed, but the area is surrounded by houses of various ages (3-30 years). The wetland edge here is fairly well defined, with oaks (Quercus sp.) and maples (Acer sp.) Dominating the upland and red maple dominating the wetland overstory. One transect for sampling birds was oriented on a north-south axis and extending all the way across this wetland.

Frenchtown Road Site. This is a streamside wetland located along an intermittent stream adjacent to Frenchtown Creek, a perennial stream. Within the past several years, a housing development (approx. 1 ha lots) has been placed next to the wetland and creek. The wetland edge at this site is not well defined and slopes gradually from the creek towards the upland. There is a gradual change in vegetation along this gradient. The wetland understory is fairly dense with sweet pepperbush the most abundant plant.

C. Methods

Herpetofaunal and Small Mammal Trapping

The removal method modified from the trapping array system of Campbell and Christman (1982) was used to assess amphibian and reptile abundance. One capture array was placed 15.2 m from the wetland edge on each of the buffer strip sites (Fig. 5-2). Each array was comprised of three 15.2-m drift fences arranged in a T-shaped pattern (modified from Clawson et al. 1984). A can trap was placed at the ends of each drift fence, and two can traps were placed on either side of the mid-points of each drift fence. Can traps were set flush with the ground, and holes were punched in the cans to help prevent drowning of specimens. Arrays were placed in the field in early March by the Wildlife Populations class at URI. Arrays were checked by students each day from 19 March to 19 April, 1989 (31 days) as part of a class project, and from 2 May to 22 November, 1989 (138 days) by undergraduate students paid from this NBP grant. The total number of trap nights in these two periods was 11,232. In addition to collecting herp data, a number of mammals were captured in the traps. Mammal data represent only a portion of the mammal community, because can traps select for insectivores and other mammals that cannot jump out of the cans. Data for herps and mammals reflect relative abundances of those species captured.

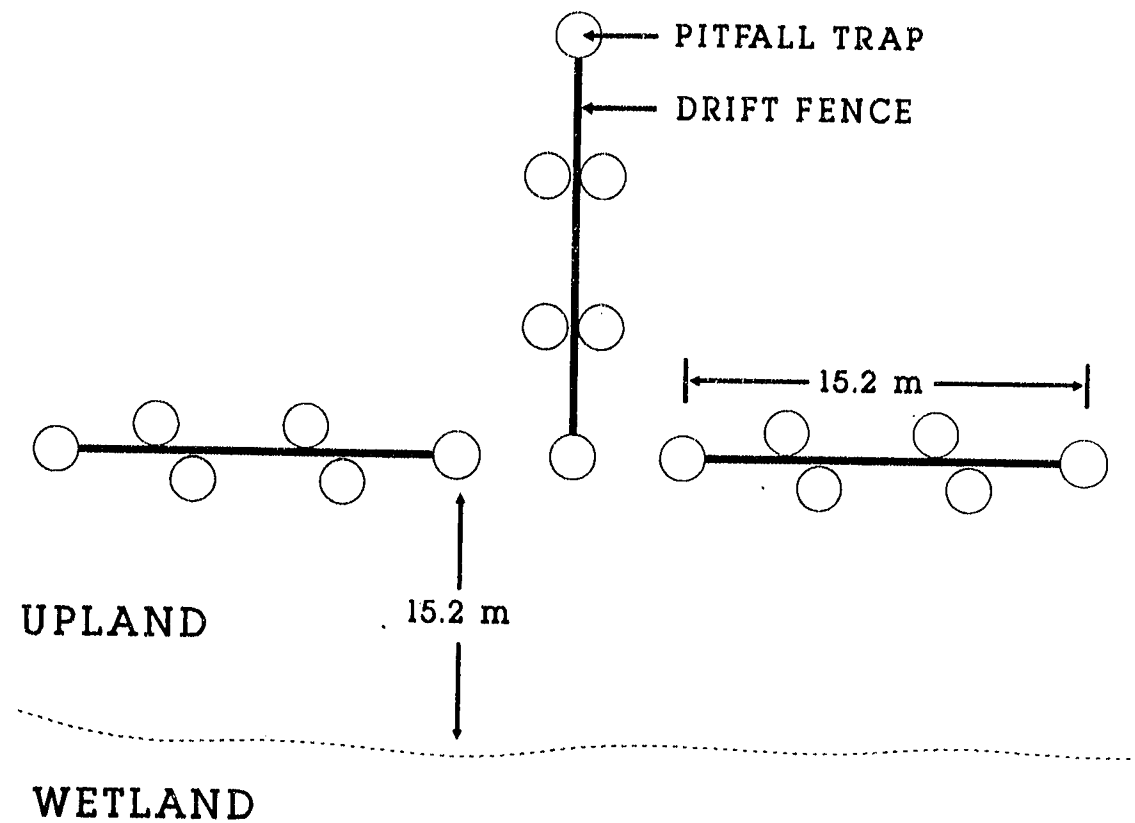


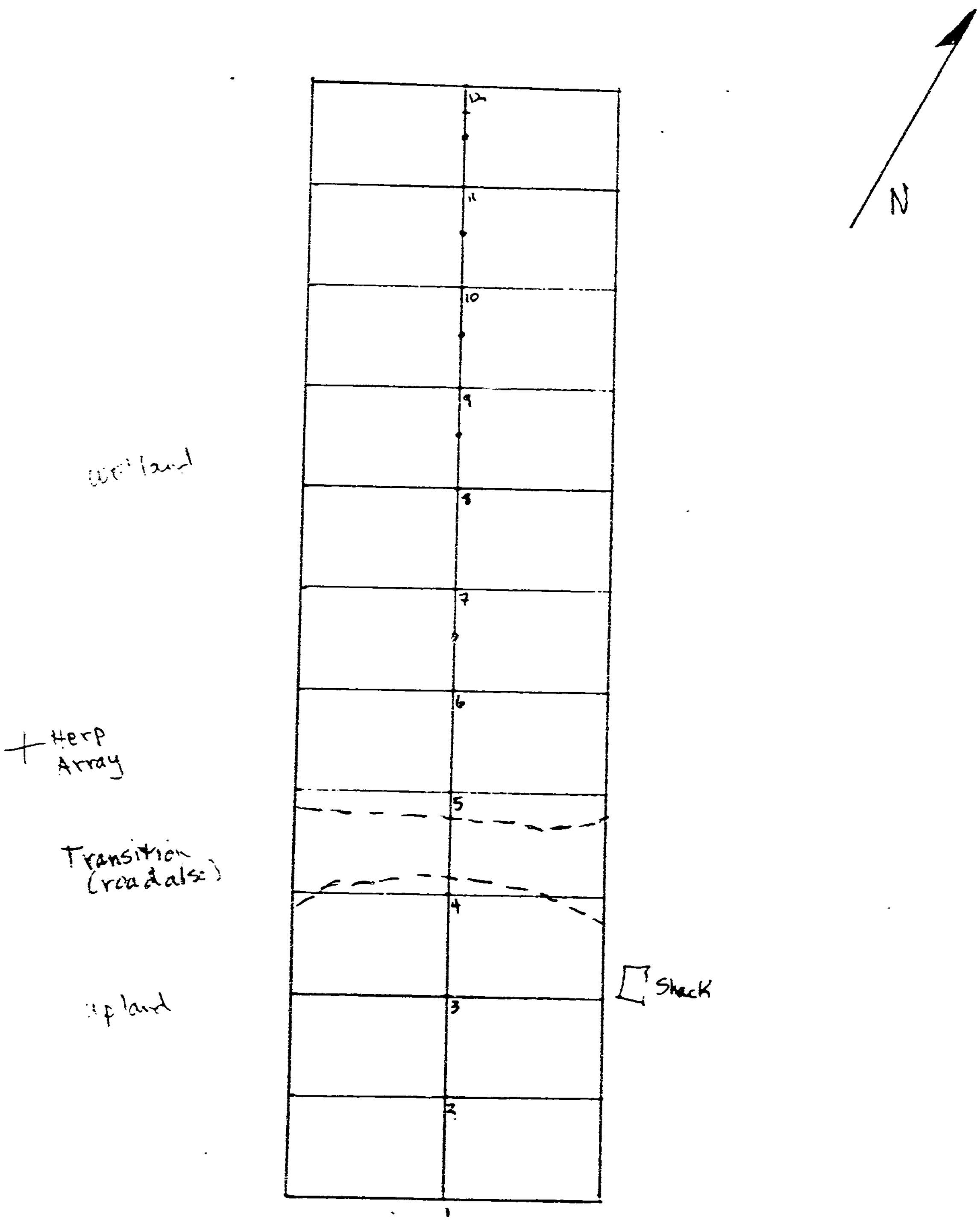
Fig. 5-2. Schematic of a typical herpetological trapping array.

Habitat data for each buffer strip site were gathered using a plotless point-quarter method (Cottam and Curtis 1956) for understory and dominant and codominant trees, in conjunction with a line intercept method (Canfield 1941) for the shrub layer. The following variables were measured for each site: (1) canopy coverage; (2) average diameter at breast height (dbh); (3) basal area of stand; (4) average height of dominant and codominant trees; (5) species composition; (6) shrub density; (7) stem density; (8) slope; (9) distance to nearest disturbance; (10) juxtaposition of other land uses; and (11) number, diameter, and total length of fallen dead logs per ha.

Avian Sampling

At each site, transects were oriented at approximately a 90° angle to the upland/wetland boundary (Fig. 5-3). A staff compass was used to place transect lines on consistent bearings. Flagging was placed at the beginning of each transect line and at every 20 m along the line. In some cases, flagging also was placed at 10-m intervals, because dense understory vegetation precluded readily sighting flagging placed every 20 m. Transects were started at a maximum of 100 m into the upland, but at least 50 m from forest edge to avoid including birds characteristic of forest edges. Consequently, only the transects at Great Swamp extended 100 m into upland because of disturbances adjacent to the other sites. Transect lines extended through the transition zone of each wetland, then at least 100 m into the wetland. In some cases (Shippeetown and Great Swamp I), bearings of transect lines were changed along the line to avoid veering too closely to clearings or other disturbances. Boundaries of the upland, transition zone, and wetland were mapped using the occurrence of indicator vegetation species (Fig. 5-3). Avian census transects always passed within 80 m of the herpetofaunal and small mammal trapping arrays, but were oriented to include as much upland forest as possible. Consequently, the distance from the transects to the trapping arrays varied from 0 (Great Swamp II) to 80 m (Frenchtown). At all sites in the Hunt-Potowomut basin, the only areas in which such transects could be sited from upland to wetland forest were covered by transects. This resulted in only one transect line at Shippeetown and Wickford Junction and two at Frenchtown. Transect width was set at 60 m (30 on each side of the transect line) because visibility was too poor in the dense shrubs of the wetlands to allow wider transects. The area of the sites was 1.00 ha (Great Swamp II), 1.30 ha (Wickford), 1.64 ha (Frenchtown), 1.92 ha (Great Swamp I), and 2.04 ha (Shippeetown).

Breeding bird populations were censused by using territory mapping (Williams 1936, International Bird Census Committee 1970). Limitations of this method have been discussed (Eagles 1981, and others), but it was selected as the census technique because a measure of territory size for dominant species was needed to develop the buffer width model. Briefly, an observer would walk along each transect line, noting all visual and auditory detections of all birds. To avoid problems associated with declines in singing of most birds in late morning, counts were conducted starting at 1/2 hour before sunrise and ending at 0930 or earlier. These observations were marked on scale maps of the transect lines. At least eight replicates were run on each study site on different days between 1 June and 31 July 1989. Starting points were alternated at each end of the transect on different days to avoid problems with decline in detections after sunrise. All simultaneous registrations of birds of the same species were noted to facilitate mapping of separate territories. When all censuses were completed, all observations of separate species were



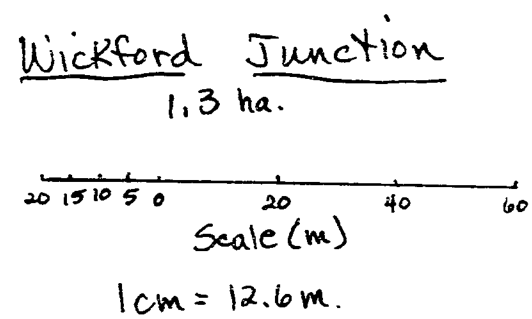


Figure 5-3. Field map of Wickford Junction site showing typical set-up of transect line and habitat types.

combined on one clean map. Clusters of observations were interpreted as separate individuals, and the outer perimeter points of these clusters were connected to delineate separate territories. For development of the buffer model, territory sizes for selected abundant species were determined using a compensating polar planimeter calibrated to the scale of the maps. These data are included under the Buffer Model section. In addition to the determination of territory size, numbers of detections of each bird species and the location of territory centers was also used for some analyses. Numbers of detections, both auditory (songs and calls) and visual, were recorded in each habitat type (upland, wetland, and transition). Territory centers in each habitat type were located by determining the arithmetic center of the mapped territory and recording the habitat type in which it occurred.

Density of birds was determined by calculating the proportion of each territory that overlapped the plot for each habitat type, summing these values for each habitat type, then dividing this total by the area of that habitat type on the plot. This resulted in different density measurements for each habitat type on each study area. Density was not calculated for species for which the number of detections was too small (<3 detections on only one of the eight censuses) or for non-territorial species such as brown-headed cowbirds.

Vegetation Sampling

Vegetation was measured in 0.04-ha circular plots centered at the beginning of the transects and at 20-m intervals from the starting point (James and Shugart 1970, James 1978). Variables recorded within each vegetation plot included percent canopy coverage using a densiometer, canopy height using an optical rangefinder, coverage and mean height of shrubs from 20 points within the circle, percent coverage by water, shrub density within two arm's-width transects from the center of the circle to its perimeter, and number and dbh for each tree within the circle. Trees were defined as any woody plant with a dbh of >7.5 cm, while shrubs were defined as any woody plant >1 m tall and <7.5-cm dbh. In order to calculate the total number of shrubs/ha, the total stems in the two transects was multiplied by 50 (James 1978). The total basal area of trees (in m^2/ha) was calculated by summing the individual basal areas of all trees in the plot and multiplying by 25. The habitat type at the center of each circle was recorded to determine if the three types differed in vegetative characteristics.

Statistical Analyses

Data on bird species occurrences are presented as descriptive phenomena. Avian habitat type preferences were analyzed using the resource preference index of Neu et al. (1974). This index uses a $\underline{X}^2$ statistic to determine if usage is significantly different than availability for all habitat types combined, then determines whether each habitat type is used more (=selected) or less (=avoided) than expected using a Bonferroni $\underline{z}$ statistic. Habitat types considered were upland, wetland, and transition. The area of each habitat type on each study area (the availability of each habitat type) was determined using the planimeter (see above). The analysis was conducted on all species with $\geq$10 registrations for both song detections only and for all detections. Kruskall-Wallis tests (Wilkinson 1987) were used to determine if densities of selected species of birds differed across the upland-wetland transition zone. Densities of each species were determined in the three habitat types on each study site as described above. Habitat type was the grouping variable used for this analysis.

Characteristics of vegetation were analyzed using the Statistical Analysis System (SAS Institute 1985). ANOVA with a Duncan's multiple range test was used to test for differences in each variable among upland, transition, and wetland sites at each study area. The variables were then regressed on density of the most common birds using Spearman's rank correlation (Conover 1980:252). Unless noted differently, significance was assigned at $\underline{P}=0.05$.

D. Results

Herpetofauna and Mammals

Trapping resulted in the capture of 2,064 individuals, comprised of 12 species of amphibians (n = 1,668), 2 species of reptiles (n = 3), and 12 species of mammals (n = 395) (Tables 5-1, 5-2). The number of herps and mammals caught on each site ranged from 250 to 704 (mean = 417) and from 62 to 143 (mean = 99), respectively (Tables 5-3, 5-4).

The most remote and least disturbed site, the Great Swamp, yielded the highest number of herps (n = 703), followed by Wickford (n = 393), Shippeetown (n = 322), and Frenchtown (n = 250). The distribution of capture data for traps did not differ from normal (Komogorov-Smirnov test, $\underline{P}<0.05$, Wilkinson 1987). Nonparametric comparisons (Kruskal-Wallis one-way analysis of variance, Wilkinson 1987) of captures of herps between the four sites indicated that the Great Swamp site had significantly more captures that the other sites (Table 5-5). A similar test for mammals showed that Frenchtown had significantly more captures than did Shippeetown or Wickford (Table 5-6).

Species richness and diversity indices for herps and mammals were calculated for each site (Tables 5-7 and 5-8, respectively). Two diversity indices were chosen which incorporate both species richness and evenness into a single value: Shannon's Index (Shannon and Weaver 1949) and Hill's Index (Hill 1973). Shannon's Index is equal to the average degree of uncertainty in predicting to what species an individual chosen at random from a sample will belong. Hill's Index is an estimation of the number of "abundant species."

The Great Swamp site had the greatest species richness (n = 13) and diversity of herps, followed by Wickford, Frenchtown, and Shippeetown (Table 5-7). The Great Swamp (Hill's Index = 7.74) had more than twice as many abundant herp species than did the site with the next highest value (Wickford, Hill's Index = 3.79). The species richness of all sites was about equal (Table 5-8). Indices of diversity for herps were greatest at Frenchtown (Table 5-8).

A measure of evenness, Evenness 1 (E1 of Ludwig and Reynolds 1988; equal to the J' value of Pielou 1975, 1977), was calculated for herps and mammals found at each site (Tables 5-7 and 5-8). If all species captured were equally abundant, then the evenness would be maximum and would decrease toward zero as the relative abundances of the species diverge away from evenness. Evenness in herps was considerably higher (E1 = 0.80) at the Great Swamp, whereas the other sites were about the same (range = 0.61-0.66). Evenness in mammals was higher at Frenchtown and Wickford (E1 = 0.61 and 0.60, respectively) than at Shippeetown and the Great Swamp (E1 = 0.47 and 0.46, respectively).

Recruitment pulses of green frogs and American toads were responsible for the high total number of herps at the Great Swamp in July and August (Fig. 5-4). In July, 94 and 79 (out of 202) *Rana* and *Bufo* were captured, respectively, whereas in August, 13 and 130 (out of 150) were captured at the

Table 5-1. Species of herpetofauna captured in four Rhode Island buffer zones.

Species	No. Captured
Class Amphibia:	
Eastern newt (*Notophthalmus viridescens*)	26
Spotted salamander (*Ambystoma maculatum*)	49
Marbled salamander (*Ambystoma opacum*)	7
Four-toed salamander (*Hemidactylium scutatum*)	45
Red-backed salamander (*Plethodon cinereus*)	607
Green frog (*Rana clamitans*)	115
Pickeral frog (*Rana palustris*)	11
Northern leopard frog (*Rana pipiens*)	17
Wood frog (*Rana sylvatica*)	359
American toad (*Bufo americanus*)	426
Woodhouse's toad (*Bufo woodhousei*)	2
Spring peeper (*Hyla crucifer*)	1
Class Reptilia:	
Painted turtle (*Chrysemys picta*)	1
Common garter snake (*Thamnophis sirtalis*)	2
Grand Total	1668

Table 5-2. Species of mammals captured in four Rhode Island buffer zones.

Order and species	Number Captured
Order Insectivora:	
Short-tailed shrew (Blarina brevicauda)	84
Masked shrew (Sorex cinereus)	237
Smoky shrew (Sorex fumeus)	1
Water shrew (Sorex palustris)	4
Order Rodentia:	
Southern bog lemming (Synaptomys cooperi)	1
Meadow vole (Microtus pennsylvanicus)	8
Woodland jumping mouse (Napaeozapus insignis)	9
Red-backed vole (Clethrionomys gapperi)	22
Eastern meadow jumping mouse (Zapus hudsonicus)	2
White-footed mouse (Peromyscus leucopus)	24
Order Lagomorpha:	
Eastern cottontail (Sylvilagus floridanus)	2
Order Marsupialia:	
Opossum (Didelphis marsupialis)	1
Grand Total	395

Table 5-3. Capture numbers of 14 species of herps on four wetland buffers in Rhode Island.

Species	Frenchtown	Site Great Swamp	Shippeetown	Wickford	Totals
Eastern newt	0	26	0	0	26
Spotted salamander	0	20	0	29	49
Marbled salamander	0	7	0	0	7
Four-toed salamander	2	34	0	9	45
Red-backed salamander	80	208	92	227	607
Green frog	2	105	2	6	115
Pickeral frog	4	0	0	7	11
Northern leopard frog	3	7	4	3	17
Wood frog	89	44	181	45	359
American toad	70	247	43	66	426
Woodhouse's toad	0	2	0	0	2
Spring peeper	0	0	0	1	1
Painted turtle	0	1	0	0	1
Common garter snake	0	2	0	0	2
Totals	250	704	322	393	1,669

Table 5-4. Capture numbers of 12 species of mammals captured in four Rhode Island buffer zones.

Species	Site Frenchtown	Great Swamp	Shippeetown	Wickford
Short-tailed shrew	36	12	17	19
Masked shrew	77	69	60	31
Smokey shrew	0	0	1	0
Water shrew	0	0	1	3
Southern bog lemming	0	1	0	0
Meadow vole	3	3	1	1
Woodland jumping mouse	8	1	0	0
Red-backed vole	10	9	1	2
Eastern meadow jumping mouse	1	1	0	0
White-footed mouse	7	5	7	5
Eastern cottontail	1	0	0	1
Opossum	0	0	1	0
Totals	143	101	89	62

Table 5-5. Statistical comparisons (Kruskal-Wallis) of the total number of herps captured (n) in 4 wetland buffers in Rhode Island.

Site	Great Swamp(703)	Wickford(393)	Shippeetown(322)	Frenchtown(250)
Great Swamp	---	0.005**	0.002**	0.000***
Wickford		---	0.788	0.060
Shippeetown			---	0.154

** $\underline{P}<0.01$, *** $\underline{P}<0.001$

Table 5-6. Statistical comparisons (Kruskal-Wallis) of the total number of mammals captured (n) in four wetland buffers in Rhode Island.

Site	Great Swamp (101)	Wickford (62)	Shippeetown (89)	Frenchtown (143)
Great Swamp	---	0.053	0.631	0.079
Wickford		---	0.114	0.050*
Shippeetown			---	0.000***

* $P<0.05$, *** $P<0.001$

Table 5-7. Species richness and diversity of herpetofauna in four wetland buffers in Rhode Island.

Site	No. Species	Shannon Index	Hill's Index	Evenness 1
Frenchtown	7	1.29	3.62	0.66
Great Swamp	13	2.05	7.74	0.80
Shippeetown	5	0.98	2.67	0.61
Wickford	9	1.33	3.79	0.61

Table 5-8. Species richness and diversity of mammals in four wetland buffers in Rhode Island.

Site	No. Species	Shannon Index	Hill's Index	Evenness 1
Frenchtown	8	1.27	3.58	0.61
Great Swamp	8	0.96	2.60	0.46
Shippeetown	8	0.98	2.67	0.47
Wickford	7	1.17	3.22	0.60

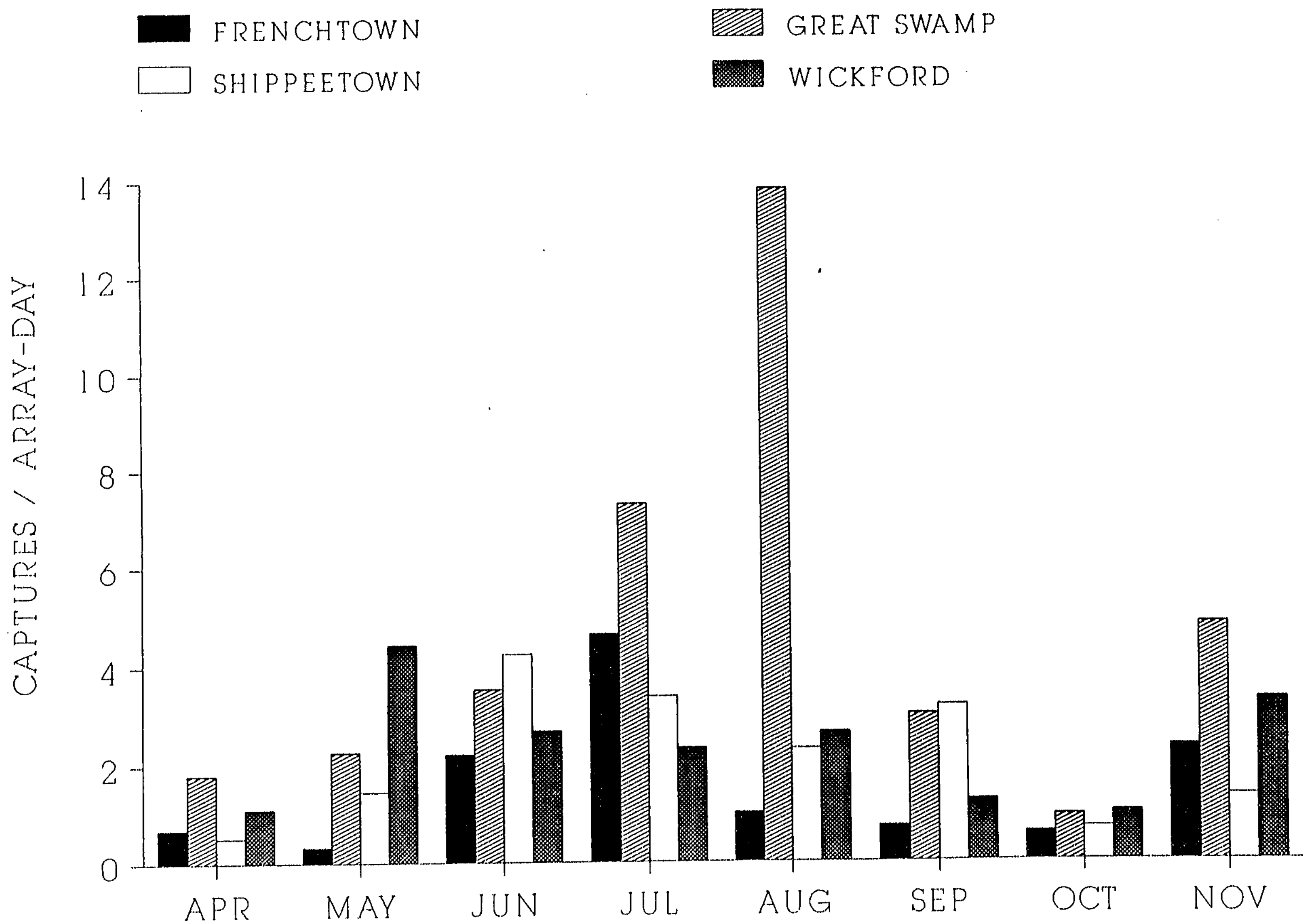


Figure 5-4. Total herp captures/array-day by month for four red maple swamp buffers in Rhode Island.

Great Swamp. Mammal captures increased on all sites in June and July, possibly the result of recruitment pulses (Fig. 5-5).

Vegetation Analysis and Site Measurements

The vegetative characteristics of all sites were very similar. The Great Swamp had the greatest species richness, yet the lowest densities of overstory stems, understory stems, shrubs, and fallen logs (Table 5-9). The Shippeetown site had the greatest density of overstory and understory stems, while Wickford had the highest shrub density and number and length of fallen logs (Table 5-9). Litter depths were 5.0 cm or greater, except at Shippeetown (3.3 cm).

Average buffer widths did not meet the desired 100-m mark, but three were at least 76 m; only Shippeetown had a narrower width (mean = 32 m). Although the Shippeetown site had a narrow buffer width, it was the second furthest from a disturbance. Distance to a permanent water source varied greatly (range = 24-366 m); the range of distances to temporary water was considerably narrower (range = 9.8-22.9 m).

The distance to disturbance varied greatly (range = 46-207 m), as did the nature of the disturbances. The Frenchtown buffer was bordered by a sharp, induced edge that was created when a large-lot housing complex was established circa 1986. The habitat outside the buffer is comprised of manicured lawns with landscaping. The Great Swamp site is an example of an inherent edge or a more natural transition zone. The only nearby disturbance is a State-owned shooting range, which is infrequently used, and a power line right-of-way; otherwise, the site is in 1,200 ha of undisturbed land. The Shippeetown site was separated from long-established houses on a two lane road by 171 m of transitional habitat. The buffer's edge, although induced, was not as sharply defined as the edge at Frenchtown. The only disturbance at the Wickford site was an often-used railroad track 84 m away.

Avian Community Composition

Forty-four species occurred in the combined study sites during the 1989 nesting season (Table 5-10). Species richness was highest at Great Swamp (30 species, followed by Shippeetown (30 species), Frenchtown (26 species), and Wickford (23 species). Eight of these species are resident, non-migratory species, 19 are short-distance migrants that winter mainly in the southern U.S., and 17 are Neotropical migrants that nest in North America and spend the winter in the Caribbean or Central and South America (Robbins et al. 1989). Fifteen species occurred at all four sites, including northern flicker, great crested flycatcher, black-capped chickadee, tufted titmouse, white-breasted nuthatch, veery, wood thrush, gray catbird, red-eyed vireo, black-and-white warbler, ovenbird, common yellowthroat, scarlet tanager, northern cardinal, and rufous-sided towhee (Table 5-10). Nineteen of the 44 total species were detected ten or more times when all data were combined (Table 5-11).

Individuals detected on at least two of the eight censuses totalled 213 for all the plots, although not all of the birds in this total had territories actually centered on the plots. The most common birds with territories on or near the plots included common yellowthroat (22 territories), gray catbird (19), ovenbird (19), veery (17), black-and-white warbler (16), black-capped chickadee (14), rufous-sided towhee (14), tufted titmouse (11), red-eyed vireo (10), and wood thrush (9) (Table 5-12).

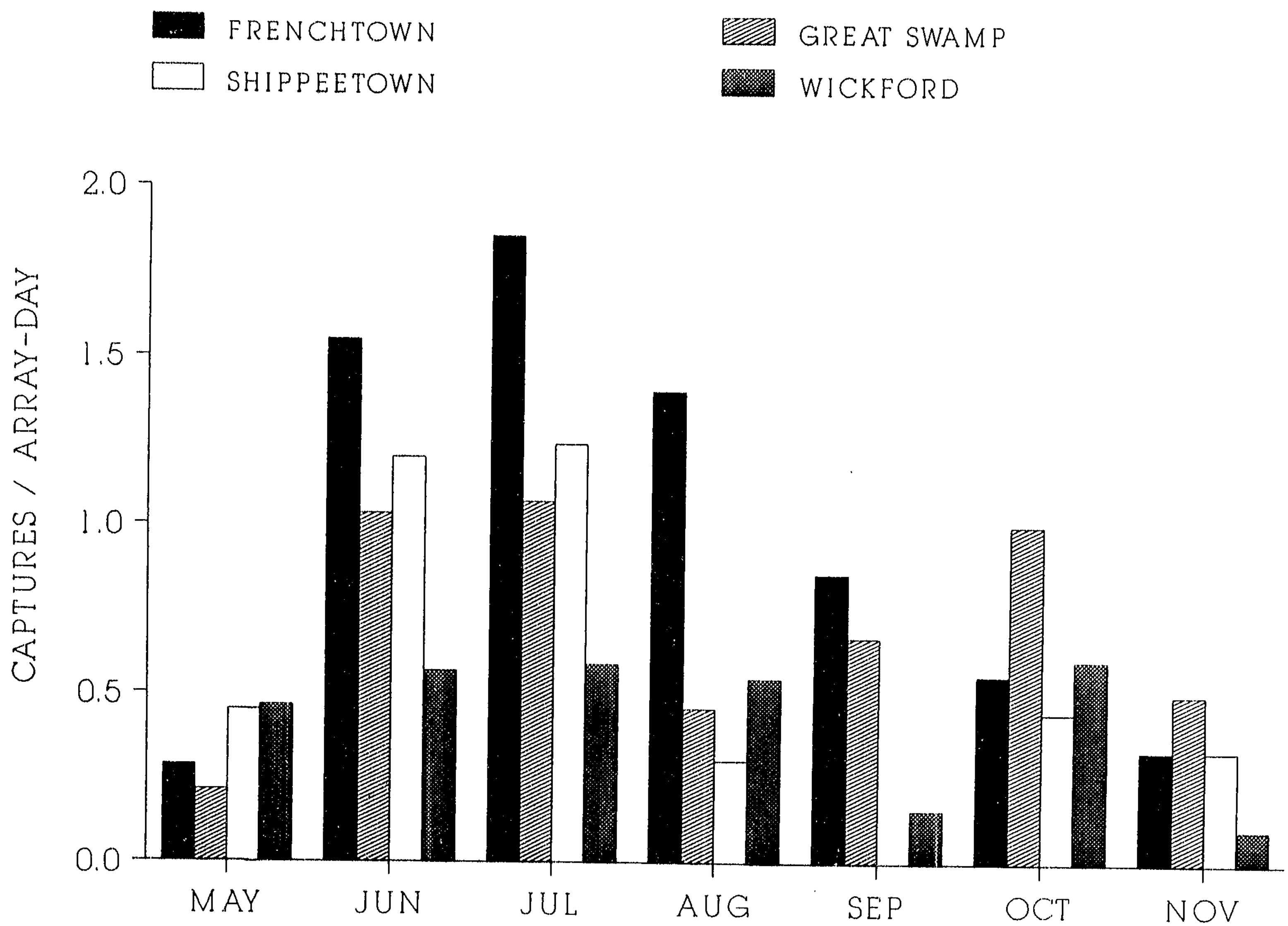


Figure 5-5. Total mammal captures/array-day by month for four red maple swamp buffers in Rhode Island.

Table 5-9. Major vegetative and site parameters characterizing the four herp array sites (mean values from vegetation analysis).

Veg. or site parameter	Site Frenchtown	Great Swamp	Shippeetown	Wickford
Species richness	7	10	6	6
Basal area (m^2/ha)	26.6	19.4	22.2	18.2
Avg. dbh (cm)	20.3	18.0	16.8	19.6
Canopy cover (%)	74	95	99	83
Density:				
Overstory (stems/ha)	457	190	805	264
Understory (stems/ha)	1,076	316	1,541	494
Shrubs (clumps/ha)	333	580	676	879
Fallen Logs:				
Number (no/ha)	34	34	40	47
Total length (m/ha)	232.2	180.2	237.8	347.8
Average log width (cm)	17.3	12.5	14.5	13.5
Litter depth (cm):	6.9	5.6	3.3	5.1
Slope (%):	2	7	6	8
Distances (m):				
Avg. buffer width	85	85	32	76
Dist. to perm. water	24	106	26	366
Dist. to temp. water	13	10	18	23
Dist. to disturbance	46	207	171	84

Table 5-10. Species and number of territories/ha of birds recorded on or adjacent to forested upland-wetland plots at each study site, May-July 1989. An "X" indicates insufficient numbers of detections to calculate density.

Species	Frenchtown	Shippeetown	Wickford	Great Swamp
Ruffed grouse (*Bonassa umbellus*)		X	X	
Belted kingfisher (*Ceryle alcyon*)				X
Downy woodpecker (*Picoides pubescens*)	0.10		X	
Hairy woodpecker (*P. villosus*)				X
Northern flicker (*Colaptes auratus*)[b]	X	X	0.33	0.41
Eastern wood-pewee (*Contopus virens*)[a]	0.22	0.51		0.43
Great crested flycatcher (*Myiarchus crinitus*)[a]	0.38	0.50	X	X
Eastern kingbird (*Tyrannus tyrannus*)[a]			X	
Blue jay (*Cyanocitta cristata*)[b]		0.32	0.68	.13
American crow (*Corvus brachyrhynchos*)[b]	X	X		
Black-capped chickadee (*Parus atricapillus*	1.33	1.30	1.32	1.25
Tufted titmouse (*P. bicolor*)	0.72	1.06	0.57	1.03
White-breasted nuthatch (*Sitta carolinensis*)[b]	0.75	0.51	X	0.64
Brown creeper (*Certhia americana*)[b]		X		
Carolina wren (*Thyrothorus ludovicianus*)		X	0.95	X

Table 5-10. Continued.

Species	Frenchtown	Shippeetown	Wickford	Great Swamp
House wren (*Troglodytes aedon*)[b]	X	X		
Ruby-crowned kinglet (*Regulus calendula*)[b]				X
Veery (*Catharus fuscescens*)[a]	1.83	0.70	1.12	1.36
Wood thrush (*Hylocichla mustelina*)[a]	0.63	0.76	0.82	0.49
American robin (*Turdus migratorius*)[b]	X	0.46		X
Gray catbird (*Dumatella carolinensis*)[b]	1.90	0.74	2.20	1.67
Northern mockingbird (*Mimus polyglottos*)[b]	X	X		
European starling (*Sturnus vulgaris*)		X		
Red-eyed vireo (*Vireo olivaceus*)[a]	1.94	0.44	1.09	0.11
Blue-winged warbler (*Vermivora pinus*)[a]		X	0.20	X
Yellow warbler (*Dendroica petechia*)[a]	X			0.20
Chestnut-sided warbler (*D. pensylvanica*)[a]			0.69	0.78
Black-throated green warbler (*D. virens*)[a]		X		
Black-and-white warbler (*Mniotilta varia*)[a]	1.12	0.84	1.60	1.72
American redstart (*Setophaga ruticilla*)[a]	0.23			

Table 5-10. Continued.

Species	Frenchtown	Shippeetown	Wickford	Great Swamp
Ovenbird (*Seiurus aurocapillus*)[a]	2.02	0.95	1.31	1.56
Northern waterthrush (*S. noveboracensis*)[a]	0.23			0.28
Common yellowthroat (*Geothlypis trichas*)[b]	1.24	0.39	1.21	2.37
Canada warbler (*Wilsonia canadensis*)[a]	0.43			0.30
Scarlet tanager (*Piranga olivacea*)[a]	0.66	0.68	1.26	0.57
Northern cardinal (*Cardinalis cardinalis*)[b]	X	0.60	0.30	0.27
Rose-breasted grosbeak (*Pheucticus ludovicianus*)[a]		0.64	0.36	
Rufous-sided towhee (*Pipilio erythrophthalmus*)[b]	1.05	0.22	1.21	0.93
Chipping sparrow (*Spizella passerina*)[b]		X		X
Song sparrow (*Melospiza melodia*)[b]	X	X		X
Swamp sparrow (*M. georgiana*)[b]				X
Red-winged blackbird (*Agelaius phoeniceus*)[b]				X
Brown-headed cowbird (*Molothrus ater*)[b]	X			X
American goldfinch (*Carduelis tristis*)[b]		X		X

[a]Neotropical migrant species
[b]Short distance migrant species

Table 5-11. Habitat type preferences of bird species using numbers of auditory and visual detections in each type. Only species detected at least 10 times were included in the analysis. All study areas were combined, preference determined using technique of Neu et al. 1974.

	Habitat type			
Species	Wetland	Transition	Upland	X^2
Northern flicker	7 (0)[a]	4 (0)	4 (0)	0.38
Eastern wood pewee	1 (-)	13 (0)	25 (+)	34.56[b]
Great crested flycatcher	8 (0)	6 (0)	3 (0)	0.72
Blue jay	6 (0)	5 (0)	8 (0)	2.39
Black-capped chickadee	52 (0)	38 (0)	40 (0)	1.64
Tufted titmouse	19 (0)	23 (0)	26 (0)	6.04[b]
White-breasted nuthatch	4 (-)	12 (0)	12 (0)	8.20[b]
Veery	55 (0)	46 (0)	25 (0)	2.83
Wood thrush	19 (0)	12 (0)	22 (0)	6.65[b]
American robin	2 (-)	6 (0)	13 (+)	15.00[b]
Gray catbird	45 (0)	21 (0)	21 (0)	5.46
Red-eyed vireo	24 (0)	18 (0)	10 (0)	1.54
Chestnut-sided warbler	4 (0)	9 (0)	3 (0)	3.72
Black-and-white warbler	29 (0)	32 (0)	20 (0)	1.31
Ovenbird	23 (0)	22 (0)	30 (+)	7.23[b]
Common yellowthroat	48 (+)	32 (0)	13 (-)	8.63[b]
Scarlet tanager	10 (0)	6 (0)	7 (0)	0.59
Northern cardinal	8 (0)	1 (-)	5 (0)	4.39
Rufous-sided towhee	14 (-)	21 (0)	39 (+)	28.08[b]

[a] Number of detections (selectivity, where + is selected, - is avoided, 0 is selected in a proportion not significantly different from random).
[b] Significant at $P \leq 0.05$.

Table 5-12. Habitat type preferences of bird species locations of centers of mapped territories. Only species with at least 10 territories included in the analysis. All study areas were combined, preference-determined using technique developed by Neu et al. 1974.

	Habitat type			
Species	Wetland	Transition	Upland	$\underline{X}^2$
Black-capped chickadee	7 (0)[a]	4 (0)	3 (0)	0.59
Tufted titmouse	5 (0)	1 (-)	5 (0)	3.54
Veery 6 (0)	3 (0)	3 (0)	4.37	
Wood thrush 4 (0)	3 (0)	2 (0)	0.10	
Gray catbird 11 (0)	3 (0)	5 (0)	3.31	
Red-eyed vireo	5 (0)	3 (0)	2 (0)	0.44
Black-and-white warbler	8 (0)	3 (0)	5 (0)	1.58
Ovenbird 8 (0)	5 (0)	6 (0)	0.50	
Common yellowthroat	16 (+)	4 (0)	2 (-)	9.97[b]
Rufous-sided towhee	5 (0)	5 (0)	4 (0)	0.11
All Neotropical migrants	50 (+)	24 (0)	23 (0)	5.75
All short distance migrants and permanent residents	57 (0)	22 (-)	37 (0)	11.07[b]

[a] Number of detections (selectivity, where + is selected, - is avoided, 0 is selected in a proportion not significantly different from random).
[b] Significant at $\underline{P} \leq 0.05$.

Habitat Type Preferences of Birds

Eight species were distributed among habitats in a pattern significantly different from random (Table 5-11). Of these, six either preferred or avoided at least one of the habitat types. Eastern wood pewee, white-breasted nuthatch, American robin, and rufous-sided towhee all significantly avoided forested wetland habitat. This avoidance of wetland habitat was especially pronounced in eastern wood pewee, because only one detection of a non-territorial bird was recorded within a wetland (Table 5-11). Only one species, northern cardinal, showed either preference or avoidance of transition habitat between upland forest and wetland forest. This species' distribution was not significantly different from random, however (Table 5-11). Four species, including eastern wood pewee, American robin, ovenbird, and rufous-sided towhee, occurred more often than expected in upland forest, while only one species, common yellowthroat, avoided upland forest (Table 5-11).

If the location of centers of territories were used to indicate habitat type preferences, only one species used habitat in a pattern departing from random selection (Table 5-12). Common yellowthroats avoided upland forest and selected wetland forest. When the location of territory centers for all species grouped into migratory status was analyzed, Neotropical migrants had a slight, though non-significant, selection of wetlands (Table 5-12). Short-distance and permanent resident species avoided transition zones for territory center placement, however.

Density of territories differed across the upland-wetland transition for three species--tufted titmouse, wood thrush, and common yellowthroat (Table 5-13). Eastern wood pewee and veery densities differed across the cline, but only neared significant differences. Of these species, veery and common yellowthroat territories occurred in greater density in wetland forest, eastern wood pewee and tufted titmouse in upland forest, and wood thrush in the transition zone (Table 5-13).

Canada warbler and northern waterthrush were not recorded often enough to include in any of the three analyses (Table 5-10); but occurred only in forested wetland habitat. These species have been recorded more frequently in other studies of red maple swamps than in this study (Swift et al. 1984, Merrow, in prep.)

Relationships between Vegetation and Bird Density

Only six bird species showed significant correlations between density and any of the vegetation variables (Table 5-14). Eastern wood pewee density was negatively correlated with water coverage and positively correlated with tree basal area. Density of black-capped chickadees was negatively correlated with basal area of trees, while density of white-breasted nuthatch was positively correlated with basal area of trees. Veery density was strongly associated with both height and coverage of shrubs and with coverage by water. Common yellowthroat density showed a similar relationship, with the added high correlation with shrub stem density. Finally, ovenbirds were more abundant at sites with tall canopy, larger tree dbh, and greater basal area; and less abundant where shrub coverage was great. Two variables, canopy coverage and tree stem density, were not related to density of any bird species.

Table 5-13. Differences in density of selected bird species across the upland-wetland transition, all study areas combined.

Species	Density in wetland (no./ha)	Density in transition zone (no./ha)	Density in in upland (no./ha)	Kruskal-Wallis test statistic	P
Eastern wood pewee	0	0.24	0.57	5.01	0.08
Great crested flycatcher	0.26	0.29	0.10	0.62	0.73
Blue jay	0.21	0.24	0.40	0.04	0.98
Black-capped chickadee	1.48	1.15	1.28	1.89	0.39
Tufted titmouse	0.81	0.35	1.38	6.12	0.05
White-breasted nuthatch	0.10	0.72	0.59	3.85	0.15
Veery	1.60	1.40	0.77	4.65	0.10
Wood thrush	0.34	1.12	0.57	6.84	0.03
Gray catbird	2.14	1.12	1.88	3.04	0.22
Red-eyed vireo	0.70	1.52	0.48	1.55	0.46
Black-and-white warbler	1.26	1.38	1.32	0.24	0.89
Ovenbird	0.83	1.61	1.94	2.35	0.31
Common yellowthroat	2.33	1.20	0.38	6.12	0.05
Scarlet tanager	0.66	1.00	0.74	1.69	0.43
Northern cardinal	0.25	0.05	0.58	1.00	0.60
Rose-breasted grosbeak	0.42	0.07	0.26	1.08	0.58
Rufous-sided towhee	0.42	1.51	2.52	3.96	0.14

Table 5-14. Species with significant correlations between density and vegetation variables. Additional species with densities not correlated with vegetation variables included great crested flycatcher, blue jay, tufted titmouse, wood thrush, gray catbird, black-and-white warbler, scarlet tanager, northern cardinal, rose-breasted grosbeak, and rufous-sided towhee.

	Vegetation variable[a]						
Species	Canopy height (m)	Shrub height (m)	Shrub coverage (%)	Water coverage (%)	Shrub density (No./ha)	Tree dbh (cm)	Basal area (m^2/ha)
Eastern wood pewee	NC[b]	NC	NC	-0.60[c]	NC	NC	0.73
Black-capped chickadee	NC	NC	NC	NC	NC	NC	-0.60
White-breasted nuthatch	NC	NC	NC	NC	NC	NC	0.72
Veery	NC	0.64	NC	0.78	0.58	NC	NC
Ovenbird	0.58	NC	-0.54	NC	NC	0.52	0.65
Common yellowthroat	NC	0.64	0.52	0.74	0.72	NC	NC

[a]Percent canopy coverage and tree density (no./ha) were not significantly correlated with density of any bird species.

[b]NC = not correlated ($P>0.05$).

[c]Spearman correlation coefficient ($P<0.05$).

E. Discussion

Herpetofauna

It is impossible to accurately compare densities or even relative abundances between our buffer sites and those in other studies; techniques and methods of study vary. In a Florida study (Campbell and Christman 1981), 30 arrays (4-legged arrays) were operated for a total of 7,432 array-days in 11 habitat types. Habitats ranged from river swamp to sand pine scrub. Their traps collected 1,644 specimens of 43 species of amphibians and reptiles for an average of 0.22 specimens per array-day, or 1.55 specimens per array each time that they were checked. Their arrays were equipped with both cans and wire mesh traps. On average, arrays with cans only collected about 35 specimens per array over a period from mid-March to mid-November. Campbell and Christman (1981) reported Shannon Indexes which ranged from 1.00 for river swamps to 2.46 for rosemary scrub habitat; the mean Shannon Index for the 11 habitat types was 1.85.

In a Missouri study (Clawson and Baskett 1982), arrays (4-legged arrays with wire funnels) in upland forests that were checked from March to November collected 2,545 individuals comprising 13 species of amphibians (n = 1,406) and 22 species of reptiles (n = 1,139). We collected an equal number of amphibian species in our study; the greater number of reptiles captured in their study may be the result of latitudinal differences in abundance, and the greater effectiveness of funnel traps in capturing reptiles (Campbell and Christman 1981). Clawson and Baskett (1982) had a success rate of about 0.55 amphibians per array-day (12 arrays set for 214 trap days).

We collected 1,668 herps (14 species) in only 500 array-days (3-legged arrays). On average we trapped 3.34 herps per array-day. This relatively high trap success may be an indication of equally high herp densities in land adjacent to the red maple swamps that we studied. Buffer zones may provide critical habitat for many amphibians, which they spend most of their lives in terrestrial zones, yet lay their eggs in water.

Our Shannon indices for herps averaged 1.41 (Table 5-7), compared to 1.85 in the Florida study (Campbell and Christman 1981). The greater Florida values were the result of more herp species (43 species vs. 14 in our study). Considering the depauperate herpetofauna taxa in New England, our diversity indices are relatively high. This is the result of a high degree of evenness of species, that is, most species are relatively abundant.

Because of the limited scope of our study (4 sites examined for 1 year), we are unable to comment with any confidence on between site differences. It appears that the Great Swamp site supported a greater number of herps than did the other sites. This may have been the result of ecological factors or simply an aberration. It is interesting to note, however, that this site was (1) the most remote, (2) closest to temporary water, (3) furthest from disturbances, (4) moderate in shrub density, and (5) by far the lowest in understory and overstory stem densities. Additional research is needed to determine how important these and other variables are to herp abundance.

Mammals. The inadvertent capture of small mammals in the herp arrays revealed the presence of three species thought to be rare in Rhode Island, as well as relatively high densities of many other species. Four water shrews were captured: one at Shippeetown and three at Wickford. These captures represent two of four known locations for this species in RI (Layne and Shoop 1971, Husband et al. 1989). It is notable that this semiaquatic species was captured in buffer zones adjacent to red maple swamps; water shrews usually are characterized as living "among boulders along mountain streams or in sphagnum moss along mountain lakes (Whitaker 1980)." However, in northern New Hampshire, water shrews were found more than 100 m from streams in mature northern hardwood stands (DeGraaf and Rudis 1983).

The capture of a smoky shrew at Shippeetown is another rare find. Cronan and Brooks (1961) indicated that this species might "possibly" be extant in the western highlands of the State. Layne and Shoop (1971) last reported a smoky shrew in Rhode Island in 1970. In addition, the southern bog lemming captured at the Great Swamp was the third recorded in RI (R. Enser, RI Natural Heritage Prog., pers. commun.). Red maple swamps and their surrounding buffers may be critical habitats for all three of these rare, area-sensitive species.

The capture rate for small mammals at the four sites was about 0.79 per array-day. The greatest number (143) and diversity (Shannon's index = 1.27, Hill's index = 3.58) of small mammals occurred at Frenchtown (Table 5-8). The numbers (Table 5-4) and diversity (Table 5-8) of small mammals appeared to increase with degree of disturbance and amount of induced edge. Further research is needed to determine if this trend is only apparent or real.

In Wisconsin, Matthiae and Stearns (1981) studied mammalian species-area relationships and the influence of adjacent landscape on species richness in forested habitats. They observed that species richness generally increased with patch size; rural sites were most diverse in mammals; small mammals and medium-sized nocturnal omnivores used urban islands as refuges; and lower species diversities and abundances were found in islands in the urban-rural transition zone.

The distribution and abundance of small mammals also has been related to water levels and the associated vegetative communities. In a study of prairie potholes (Pendleton 1984), small mammals were found to choose habitats based on soil moisture levels. Deer mice were associated with dry sites, meadow voles with moist sites, while masked shrews and short-tailed shrews were most common in transitional habitats intermediate in moisture. This, along with trap type, may explain the relatively high proportion of masked and short-tailed shrews in our capture.

Avian Community Composition

The list of species observed in the four forested wetlands closely resembled bird communities in other studies of red maple swamps in southern New England (Swift et al. 1984, Merrow, in prep.). The species richness of 44 compares to 39 in southern Rhode Island (Merrow, in prep.) and 46 in Massachusetts (Swift et al. 1984). The slight differences in species composition might be explained by the relatively small size of some plots in the Rhode Island study (which was designed to examine habitat-area effects on species richness) and the inclusion of plots of earlier successional stage in the Massachusetts study (Swift et al. 1984). Differences in the community composition of birds from other studies related mostly to the order of abundance of species, with wood thrush, gray catbird, red-eyed vireo, ovenbird, and rufous-sided towhee being relatively more abundant in our study. Two species among the ten most abundant species in the other studies were Canada warbler and northern waterthrush. These species were probably present in greater numbers, but were not detected as often because of the later dates of censuses compared to the other studies. Both of these species undergo exceptional declines in singing behavior after the young are hatched, but also may be more characteristic of moister forests than our sites. Several species were present on our plots because of the effects of nearby forest edges in the case of Shippeetown and Frenchtown and the presence of open habitat near the Chipuxet River at Great Swamp. These include belted kingfisher, eastern kingbird, house wren, American robin, northern mockingbird, European starling, blue-winged warbler, yellow warbler, chipping sparrow, song sparrow, swamp sparrow, red-winged blackbird, and American goldfinch.

Neotropical migrant species comprised 45.5% of the overall bird community. This proportion is lower than the 60-80% that should be expected for undisturbed eastern deciduous forest (Terborgh 1989:71). The reason for this difference is unclear because of the lack of existing similar data sets for large tracts of mature forest in southern New England, but may be related to declines of Neotropical migrant birds that nest in forest interiors, as noted elsewhere in the East (Robbins et al. 1989, Terborgh 1989:46). The presumed reasons for such declines are increases in short distance migrant and permanent resident competitors, habitat loss on wintering areas, predators, and brown-headed cowbirds (which are nest parasites). The latter possibility is supported by circumstantial evidence, because adult black-and-white warblers were observed feeding young cowbirds at Frenchtown.

The bird community did not differ substantially among sites, indicating that all four were either relatively undisturbed or were equally subject to disturbance and the effects of nearby edge habitat. The Great Swamp site was selected to provide a relatively undisturbed site, but does have a powerline, old road, and shooting range within 150 m of both transects.

Eight species are sensitive to forest area size (ie. they do not occur regularly in small forest tracts of <100 ha) or to both forest area size and degree of isolation from other forest tracts (Robbins et al. 1989). The former include hairy woodpecker, veery, wood thrush, red-eyed vireo, ovenbird, scarlet tanager, and rose-breasted grosbeak; the latter includes black-and-white warbler. Most of these species do not occur with regularity in forests of <400 ha in the middle Atlantic states. Because veery, wood thrush, red-eyed vireo, black-and-white warbler, and ovenbird are among the 10 most common species on the plots, forest area is probably not yet limiting these birds in the Hunt-Potowomut basin, although the populations could be elevated by immigration from more suitable areas (Terborgh 1989). The main reason for their relative abundance is probably the moderate degree of fragmentation and isolation of forests that has occurred to date (cf. Temple and Cary 1988). As development increases in the basin, these species may be expected to decline (Robbins et al. 1989).

Habitat Type Preferences of Birds

Two species were significantly classed as upland birds by at least two of the three types of analyses, eastern wood pewee and rufous-sided towhee. Tufted titmouse, white-breasted nuthatch, and American robin were indicated as more associated with upland by at least one analysis. Of these species, eastern wood pewee is perhaps the most associated with dry forests, avoiding wetland forest almost entirely (Robbins et al. 1989). Pewee is also the only Neotropical migrant species in the upland group, tufted titmouse is a permanent resident, and white-breasted nuthatch is a short distance migrant species.

Only one species was indicated as more common in wetland forest, common yellowthroat. This species is often associated with a variety of wetland types, but tends to be a habitat generalist (Kahl et al. 1985). Two species that are common in forested wetlands in southern New England that were not common on our plots are also nearly obligate to wetlands, Canada warbler and northern water thrush. (Merrow, in prep.). All of our detections of these species were in wetland habitat.

In designing buffers around forested wetlands (see below), the goals of maintaining a wetland would have to be considered before designating a buffer width for birds. If upland species are to be maintained, or Neotropical migrant wetland species are to be preserved, a wider buffer would be necessary. If the goal is only to maintain wetland birds such as common yellowthroat, a narrower buffer would suffice.

Several limitations of our data probably resulted in our failure to detect higher densities of some birds in wetlands or vice versa. The width of upland around all sites may still have been insufficient to eliminate edge effects, both from nearby open habitats (resulting in detection of several species more characteristic of early successional habitats) and from effects of wetland species being at higher density in upland immediately adjacent to the wetlands. If younger wetland birds tend to occur in less suitable habitat, a spillover effect from the wetland might have swamped our attempts to detect density differences in uplands. The distance at which edge effects operate is unknown, but edge species often nest >65 m into other habitat types (Yahner 1988). The effects of cowbird nest parasitism, on the other hand, may extend >300 m into forested habitat from adjacent edges (Robbins et al. 1989). Future studies to fully document avian occurrence and habitat preferences across upland-wetland transitions in forested habitat should extend plots at least 300 m into each habitat rather than the 100 m used in this study.

An additional limitation of this study was in the small sample size for many species and the small size of the study areas. Plot sizes for sampling birds in eastern deciduous forest are optimally ≥40 ha (Terborgh 1989), but the limitations of the Hunt-Potowomut study sites imposed 1-2 ha plots.

Relationship between Vegetation and Bird Densities

The vegetation variables most related to abundance of birds in forested wetlands were density and coverage by shrubs, size of trees, and coverage of the substrate by water. In other studies of the relationship of vegetation to bird occurrence in red maple swamps, shrub variables and hydrology were also important determinates of avian species occurrence (Merrow, in prep.) and density (Swift et al. 1984). Across the upland-wetland transition, eastern wood pewee was associated with the dry substrate and larger trees of upland habitat, while veery and common yellowthroat occurred more often at sites with water coverage and more shrubs. Ovenbirds occurred across the range of forest habitats at sites with large, tall trees and few shrubs. Finally, white-breasted nuthatch occurred at sites with high tree basal area, possibly because of their need for tree bole surfaces for foraging.

Our ability to detect additional relationships between vegetation variables and bird species other than the six species with significant correlations was limited by small sample size of some species, and by the uniformity of vegetation among the sites (low variance in most variables despite variation in bird density). The vegetation variables with significant correlations with density of >1 species were those that varied along the cline from upland to wetland (shrub variables, tree basal area, and water coverage). Upland tended to have lower values for shrub variables, higher tree basal area, and no water coverage in comparision to wetlands.

Future research should consider bird communities and vegetational responses outside the breeding season, document the change in bird communities across the upland-wetland transition at sites with a broader surrounding upland to avoid possible edge effects that we noted, focus on larger areas outside the Hunt-Potowomut basin, be conducted for at least two years to incorporate possible changes in weather and other variables, and attempt to explain why species are distributed as they are across the transition from upland to wetland (eg. examine habitat, food resources and usage, nesting biology and associated nest parasitism and predation). These tasks are all involved with attempting to account for differences in habitat use by birds in different seasons, to account for differences in sex or age of birds by enlarging sample sizes over this limited study, and to expand the data base to allow use of more sophisticated statistical techniques by limiting variation in data (Weller 1988).

F. Buffer Model

The Approach

Ideally, land managers would have at their disposal a simple, accurate model that would prescribe buffer requirements to protect wetland-dependent wildlife species. As an initial step toward the construction of such a model, we will modify the general model offered by Brown et al. (1987). This model is based on four factors: (1) habitat suitability; (2) wildlife spatial requirements; (3) access to upland and/or transitional habitat; and (4) noise impacts on feeding, breeding, and other life functions of wildlife. Our approach was to gather existing information on the more sensitive wetland-dependent species which would be adversely affected by landscape alterations. This information was used in conjunction with site-specific information gathered from our field research. A set of standards for habitat suitability was then developed for area-sensitive, interior species likely to occur in Rhode Island wetlands and their buffers. Spatial requirements were evaluated and a minimum diameter for a low-stress suitable habitat was calculated. This value is considered the minimum width necessary to prevent the harmful effects of human development on wetland animals. Access requirements to uplands or transitional habitats for semiaquatic animals such as turtles was also considered. Finally, the effects of noise on wildlife was factored into the buffer zone requirements. The end product is a first-stage model which will serve as a conservative guide for establishing buffer zone requirements. Additional research will refine and focus the model.

Habitat Suitability Guidelines

The following set of guidelines are meant to be minimum standards for forested wildlife habitat in and adjacent to red maple swamps:

1. Tree canopy height greater than 9 m;
2. Tree canopy closure greater than 70%;
3. At least 3 trees greater than 62 cm diameter at breast height (dbh) per ha;
4. More than 0.1 snags greater than 62 cm dbh per ha;
5. Average shrub height greater than 0.6 m but less than 2.5 m;
6. Shrub canopy closure greater than 25%.

At present these standards are based on the professional opinions of the authors, and supported by Rhode Island forest site evaluations (Wilcox 1984). Such habitat suitability standards will be refined as more knowledge of wildlife requirements is gained through additional research.

Spatial Requirements

Spatial requirements of wildlife vary greatly. Mammals require a home range to secure food and water, breed, rest, play, etc. By definition: "Home range is the area over which an individual travels in its normal daily activities (Blair 1953)." Average home range sizes (in ha, for males:females) for some small mammals captured in our study are (Cockrum 1962, DeGraaf and Rudis 1983): short-tailed shrew, 0.21:0.49; water shrew, 0.20-0.32 ha, sex unknown; meadow jumping mouse, 0.36:0.37; woodland jumping mouse, 1.23:0.96; red-backed vole, 0.77:0.23, southern bog lemming, 0.32:0.40, and meadow vole, 0.13:0.08. These sizes also vary by season and by habitat type and condition.

Little is known about the home range size of many salamander species (Dawley 1989). The green frog had a home range from 20 m^2 to 200^2 in southern Michigan near a stream and lake (Martof 1953). For redback salamanders, home ranges of 13 m^2 for females and about 24 m^2 for males were measured in a northern hardwood forest in Michigan (Kleeberger and Werner 1982). For many other common species, such as the pickeral frog and the northern leopard frog, no home range data have been collected (DeGraaf and Rudis 1986). Other species, such as the spotted salamander, may not exhibit a "home range," but may travel in similar tracks each year to and from ponds (Shoop 1968).

We do know what a typical "amphibious" life cycle is: young begin life in the water, move to terrestrial habitats to feed and grow for sometimes 2-4 months, usually remain terrestrial as adults, and then return to the water to breed and lay eggs. However, some amphibians are totally aquatic, while others spend their lives on land. The buffer widths in our study apparently were sufficient to support large numbers of herps and mammals. It should be noted, however, that the narrowest buffer, Shippeetown (avg. width = 32 m), had the lowest and second lowest species diversity of herps (Table 7) and mammals (Table 8), respectively. The other three buffers had average widths of at least 76 m.

We measured the territory sizes (in ha) of the following area-sensitive (forest interior species), Neotropical birds found on our sites: veery (mean = 0.39, SD = 0.17, range = 0.17-0.70, N = 14, ovenbird (mean = 0.23, SD = 0.10, range = 0.08-0.34, N = 10), black-and-white warbler (mean = 0.30, SD = 0.26, range = 0.06-0.79, N = 10), great-crested flycatcher (mean = 0.46, range = 0.36-0.56, N = 2), and eastern wood peewee (mean = 0.69, SD = 0.11, range = 0.60-0.84, N = 3). These values agree with the generalization (Brown et al. 1987) that many interior songbirds do not live in plots smaller than 0.4 ha (diameter = 72 m).

Based on the territory sizes of the forest-interior bird species, our knowledge of small home range requirements, and our herp densities, we suggest for red maple swamps that the minimum buffer diameter be 100 m (the diameter of a 0.8-ha circle) of stress-free suitable habitat.

Access to Uplands or Transitional Habitat

Wetland buffers must be preserved which provide sites for semiaquatic turtles to lay their eggs. Typically, a female turtle moves to a transition of upland site, digs a hole (nest), deposits her hard-shelled eggs in it, and then covers the nest with soil. In some species (eg., snapping turtles [*Chelydra serpentina*]) hatchlings may overwinter in the nest. Turtles are very vulnerable to predation and other interferences during the nesting portion of their life cycle. Brown et al. (1987) state that a minimum 15.24-ft upland buffer along the entire length of a river is recommended for turtles; we suggest that this figure be adopted as a minimum buffer width for most species of turtles in RI.

Buffers of animals or plants of special concern (Appendices A and B) should be established with the advice of the RI Heritage Program.

Buffer Requirements for Noise Attenuation

Certain wildlife species are very sensitive to noise. The effects of loud noises on wildlife are not well understood; they may include at least: auditory interference of activities such as courtship and mating; prey location and feeding; predator detection; homing; stress-related effects; sleep disruption; annoyance; and decreased reproduction (Brown et al. 1987). Brown (1981) gives a method for attenuating highway noise to background levels by using forested buffers. Buffer zone width can be calculated from a formula that includes noise level of the source and a noise attenuation coefficient for the buffer. The calculated buffer width is that which is necessary to reduce noise levels to background (40dBA).

Brown's formula is:

$$B_w = \ln(40/N_i)/-A_c$$

Where:

B_w = buffer width in meters

A_c = attenuation coefficient:

- use 0.0066 for paved buffer;
- use 0.0121 for nonforested buffer;
- use 0.0174 for heavily forested buffer.

N_i = noise level of the source:

- use 50 dBA for residential areas;
- use 60 dBA for nonarterial general traffic;
- use 70 dBA for commercial areas.

To reduce the noise generated by a commercial area to background levels, a paved buffer of 85 m would be required, whereas a heavily forested buffer of 32 m would yield the same effect. Some interpolation may be necessary for noise levels which fall between those cited.

Final buffer width determination

All four of the above factors should be considered before establishing a buffer's width (Fig. 5-6). The buffer should be the minimum width necessary to satisfy all requirements. In the case of threatened or endangered species within the wetland area, a buffer greater than the maximum 100 m suggested in this model may need to be established (Good and Roman 1985). In addition, certain habitats which are important to wildlife or are unique to an area should have a minimum buffer size of 100 m (Good and Roman 1985).

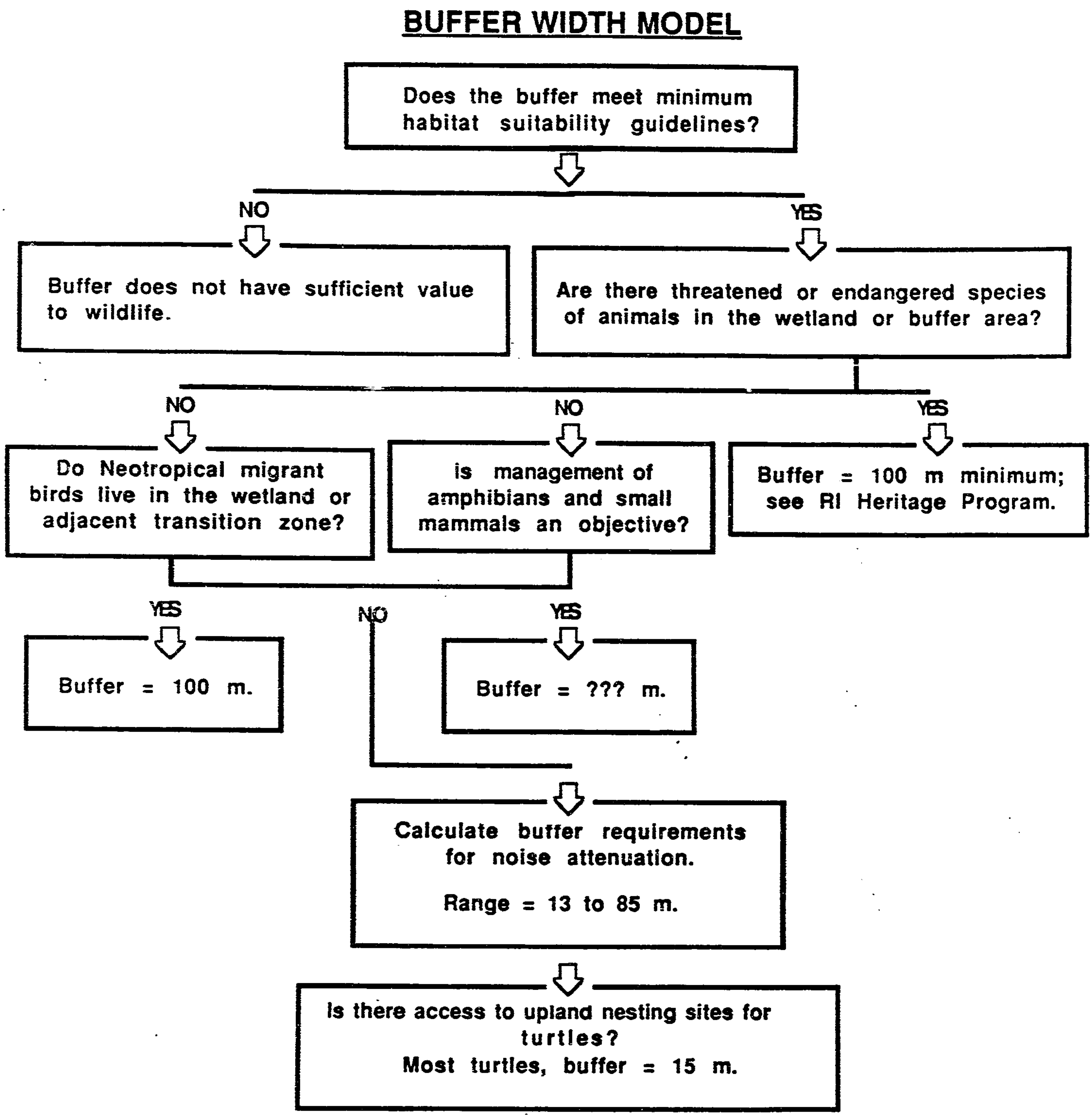


Figure 5-6. Flowchart of wetland buffer width model for red maple swamps in Rhode Island.

APPENDIX A

Rhode Island Natural Heritage Program

Rhode Island Natural Heritage Program
Animal Species of Special Concern - November 1988

Definitions of State Status

(SE) State Endangered — Native species in imminent danger of extirpation from Rhode Island. These species meet one or more of the following criteria:

1. A species currently under review for listing by the U.S. Fish and Wildlife Service as Federally endangered or threatened.

2. A species with 1 or 2 known or estimated total occurrences in the state.

3. A species apparently globally rare or threatened, and estimated to occur at approximately 100 or fewer occurrences range-wide.

(ST) State Threatened — Native species which are likely to become state endangered in the future if current trends in habitat loss or other detrimental factors remain unchanged. These species meet one or more of the following criteria:

1. A species with 3 to 5 known or estimated occurrences in the state.

2. A species with more than 5 known or estimated occurrences in the state, but especially vulnerable to habitat loss.

(SSI) State Interest — Native species not considered to be State Endangered or State Threatened at the present time, but occur in 6 to 10 sites in the state.

(C) Species of Concern — Native species which do not apply under the above categories but are additionally listed by the Natural Heritage Program due to various factors of rarity and/or vulnerability, or for which status information is presently not well known.

(SX) State Extirpated — Native species which have been documented as occurring in the state but for which current occurrences are unknown. When known, the last documentation of occurrence is included. If an occurrence is located for a SX species, that species would automatically be listed in the State Endangered category.

(FE) Federally Endangered — Listed as Federally Endangered.

(FT) Federally Threatened — Listed as Federally Threatened.

Species	Common Name	Status
Invertebrates:		
Cicindela dorsalis	Northern Beach Tiger Beetle	SX
Nicrophorus americanus	American Burying Beetle	SE
Speyeria idalia	Regal Fritillary	SE
Hemileuca maia maia	Buck Moth	ST
Enalagma recurvatum	Barrens Bluet Damselfly	SE
Williamsonia lintneri	Banded Bog Skimmer Dragonfly	SE
Fish:		
Lampetra appendix	American Brook Lamprey	SSI
Acipenser oxyrhynchus	Atlantic Sturgeon	ST
Amphibians:		
Ambystoma opacum	Marbled Salamander	C
Gyrinophilus porphyriticus	Northern Spring Salamander	SSI
Hemidactylium scutatum	Four-toed Salamander	C
Scaphiopus holbrookii	Eastern Spadefoot	ST
Rana pipiens	Northern Leopard Frog	SSI
Reptiles:		
Caretta c. caretta	Atlantic Loggerhead (+)	FT
Chelonia m. mydas	Atlantic Green Turtle (+)	FT
Dermochelys c. coriacea	Atlantic Leatherback (+)	FE
Lepidochelys kempi	Atlantic Ridley (+)	FE

(+) occurring in offshore waters only

Clemmys insculpta	Wood Turtle	SSI
Malaclemys terrapin	Northern Diamondback Terrapin	ST
Carphophis amoenus	Eastern Worm Snake	SSI
Elaphe obsoleta	Black Rat Snake	SSI
Heterodon platyrhinos	Eastern Hognose Snake	C
Storeria occipitomaculata	Northern Redbelly Snake	C
Thamnophis sauritus	Eastern Ribbon Snake	C
Crotalus horridus	Timber Rattlesnake	SX (1966)

Birds: (Species listed in reference to nesting sites only, except in the case of the Federally listed Bald Eagle and Perigrine Falcon which occur as transients or wintering birds.)

Species	Common Name	Status
Podilymbus podiceps	Pied-billed Grebe	SX
Phalacrocorax auritus	Double-crested Cormorant	SSI
Botaurus lentiginosus	American Bittern	SE
Ixobrychus exilis	Least Bittern	SSI
Ardea herodias	Great Blue Heron	SSI
Casmerodius albus	Great Egret	SSI
Egretta caerulea	Little Blue Heron	SSI
Egretta thula	Snowy Egret	SSI
Bubulcus ibis	Cattle Egret	SSI
Nycticorax nycticorax	Black-crowned Night Heron	SSI
Nycticorax violaceus	Yellow-crowned Night Heron	SSI
Plegadis falcinellus	Glossy Ibis	SSI
Anas crecca	Green-winged Teal	SSI
Anas discors	Blue-winged Teal	SSI
Anas strepera	Gadwall	C
Lophodytes cucullatus	Hooded Merganser	C
Cathartes aura	Turkey Vulture	C
Haliaeetus leucocephalus	Bald Eagle	FE
Pandion haliaetus	Osprey	SSI
Circus cyaneus	Northern Harrier	SE
Accipiter striatus	Sharp-shinned Hawk	SX
Accipiter cooperii	Cooper's Hawk	ST
Accipiter gentilis	Northern Goshawk	SSI
Falco peregrinus	Peregrine Falcon	FE (S/A)
Rallus elegans	King Rail	SSI
Rallus longirostris	Clapper Rail	SSI
Porzana carolina	Sora	SSI
Gallinula chloropus	Common Moorhen	SX
Charadrius melodus	Piping Plover	FT
Bartramia longicauda	Upland Sandpiper	ST
Haematopus palliatus	American Oystercatcher	SSI
Sterna dougallii	Roseate Tern	FT
Sterna hirundo	Common Tern	C
Sterna antillarum	Least Tern	SSI
Tyto alba	Common Barn Owl	ST
Asio otus	Long-eared Owl	C
Aegolius acadicus	Northern Saw-whet Owl	C
Chordeiles minor	Common Nighthawk	C
Melanerpes carolinus	Red-bellied Woodpecker	SSI
Melanerpes erythrocephalus	Red-headed Woodpecker	C
Dryocopus pileatus	Pileated Woodpecker	SSI
Empidonax virescens	Acadian Flycatcher	SSI
Eremophila alpestris	Horned Lark	SSI
Hirundo pyrrhonota	Cliff Swallow	ST

S/A = Endangered due to similarity of appearance.

Catoptrophorus semipalmatus Willet SSI

Birds (continued)

Corvus ossifragus	Fish Crow	C
Troglodytes troglodytes	Winter Wren	C
Cistothorus palustris	Marsh Wren	SSI
Regulus satrapa	Golden-crowned Kinglet	C
Vermivora chrysoptera	Golden-winged Warbler	SX (1960)
Parula americana	Northern Parula	SX
Dendroica caerulescens	Black-throated Blue Warbler	SX
Dendroica cerulea	Cerulean Warbler	SSI
Dendroica fusca	Blackburnian Warbler	C
Helmitheros vermivorus	Worm-eating Warbler	C
Icteria virens	Yellow-breasted Chat	SE
Pooecetes gramineus	Vesper Sparrow	SE
Ammodramus savannarum	Grasshopper Sparrow	ST
Ammodramus maritimus	Seaside Sparrow	C
Zonotrichia albicollis	White-throated Sparrow	C
Junco hyemalis	Dark-eyed Junco	C
Icterus spurius	Orchard Oriole	C

Mammals:

Sorex fumeus	Smoky Shrew	C
Sorex palustris	Water Shrew	SSI
Sylvilagus transitionalis	New England Cottontail	C
Synaptomys cooperi	Southern Bog Lemming	C
Martes pennanti	Fisher	SSI
Lynx rufus	Bobcat	ST

Endangered Plant Species in Rhode Island (in taxonomic order):

Scientific Name	Common Name	Extant Sites
Lycopodium annotinum	Stiff Clubmoss	1
Lycopodium inundatum var. robustum	Robust Bog Clubmoss	1
Asplenium montanum	Mountain Spleenwort	1
Asplenium rhizophyllum	Walking Fern	1
Pellaea atropurpurea	Purple Cliff-brake	1
Sagittaria teres	Slender Arrowhead	3
Carex collinsii	Collins' Sedge	1
Eleocharis melanocarpa	Black-fruited Spike-rush	1
Eleocharis tricostata	Three-angled Spike-rush	1
Fuirena pumila	Umbrella Grass	2
Psilocarya scirpoides	Long-beaked Bald Rush	2
Rhynchospora inundata	Horned Rush	4
Rhynchospora torreyana	Torrey's Beaked Rush	2
Scleria triglomerata	Tall Nut-rush	1
Orontium aquaticum	Golden Club	1
Galearis spectabilis	Showy Orchis	1
Malaxis unifolia	Green Adder's Mouth	2
Platanthera flava var. herbiola	Pale Green Orchis	2
Platanthera hookeri	Hooker's Orchid	1
Saururus cernuus	Lizard's Tail	1
Arceuthobium pusillum	Dwarf Mistletoe	1
Polygonum puritanorum	Pondshore Knotweed	1
Minuartia stricta	Rock Sandwort	2
Minuartia glabra	Smooth Sandwort	2
Clematis occidentalis	Purple Clematis	1
Caulophyllum thalictroides	Papoose-root	1
Adlumia fungosa	Climbing Fumitory	2
Parnassia glauca	Grass-of-Parnassus	1
Dalibarda repens	Dewdrop	1
Sanguisorba canadensis	Canadian Burnet	1
Linum sulcatum	Grooved Flax	1

Endangered Plant Species in Rhode Island (continued):

Scientific Name	Common Name	Extant Sit[e]
Helianthemum dumosum	Bushy Rockrose	4
Rotala ramosior	Toothcup	1
Ludwigia sphaerocarpa	Round-fruited False Loosestrife	1
Aralia racemosa	Spikenard	1
Hydrocotyle verticillata	Saltpond Pennywort	1
Andromeda polifolia	Bog Rosemary	1
Kalmia polifolia	Pale Laurel	1
Sabatia kennedyana	Plymouth Marsh Pink	2
Stachys hyssopifolia	Hyssop-leaved Hedge Nettle	2
Houstonia longifolia	Long-leaved Bluets	1
Linnaea borealis var. *americana*	Twinflower	1
Eupatorium leucolepis var. *novae-angliae*	New England Boneset	5
Liatris borealis	Northern Blazing Star	4
Sclerolepis uniflora	Sclerolepis	1

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