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Subsurface Hydrodynamics and Nutrient Exchange within an Extensive Tidal Freshwater Wetland

William G. Reay
College of William and Mary - Virginia Institute of Marine Science

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SUBSURFACE HYDRODYNAMICS AND NUTRIENT EXCHANGE WITHIN AN EXTENSIVE TIDAL FRESHWATER WETLAND

A Thesis

Presented to

The Faculty of the School of Marine Science

The College of William and Mary in Virginia

In Partial Fulfillment

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Master of Arts

by

William Glendon Reay

1989

This thesis is dedicated
to my parents and grandmother,
whose love and support is never ending.

APPROVAL SHEET

This thesis is submitted in partial fulfillment of

the requirements for the degree of

Master of Arts

William Glendon Reay

Approved August, 1989

Carlton H. Hershner, Ph.D.
Chairman

Michael E. Bender, Ph.D.

Gerald H. Johnson, Ph.D.

William G. MacIntyre, Ph.D.

Bruce J. Neilson, Ph.D.

Norman Bartlette Theberge, J.D., LL.M

Richard L. Wetzel, Ph.D.

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ABSTRACT

Located between upland and riverine systems, extensive tidal freshwater wetlands are influenced by a variety of recharging water sources and their respective nutrient contents. Conversely, tidal wetlands discharge interstitial waters and solutes to surface waters during periods of aerial exposure. Geohydrologic and model simulation methodology were utilized in order to aid in the understanding of wetland subsurface flow dynamics, its influence upon pore water nutrient chemistry, and its role in nutrient exchange with adjacent surface waters. Interstitial water nutrient chemistry was monitored along three transects extending from the uplands to the creekbank edge. Surface waters were also monitored throughout the 13 month study period.

Measurements of soil dry bulk density, percent organic matter, fiber content, and horizontal hydraulic conductivity were conducted along a 118 meter transect from the creekbank edge to the high marsh/upland interface. Results indicate vertical and lateral heterogeneity of these physical and hydraulic soil properties within the upper one meter soil profile. Multivariate statistical techniques best described the transect as four separate soil types. General regions of soil types followed wetland elevational regions, these include: the creekbank, levee, low marsh flat, and high marsh regions. Fiber content was identified as the measured parameter which best explained variations in wetland soil permeability.

Vertical and horizontal hydraulic head fluctuations were monitored utilizing piezometer/well arrays along the 118 meter transect. Direct measurement of interstitial water seepage flow from the subaqueous portion of the creekbank to adjacent surface water was determined. Model simulation of subsurface hydrodynamics were made in order to provide water table fluctuations, estimates of horizontal seepage, and pore water budgets along the transect. Field measurements of marsh surface elevations and hydraulic soil properties were incorporated into the model to allow for comparison between simulated and observed results.

Spatial variations in soil properties, and subsurface hydrodynamics indicate that an extensive tidal freshwater wetland cannot be considered as a homogeneous unit. It may be described more accurately as three distinct, yet interactive regions (creekbank, low marsh flat,and high marsh), with varying potentials for surface and interstitial water exchange. The creekbank, experiencing large water table oscillations and hydraulic gradients, was the most dynamic and tidally influenced region. These hydrodynamic characteristics resulted in substantial subsurface water transport and dilution of interstitial waters by recharging surface waters within the creekbank region. Due to extremely low hydraulic gradients and ponding of water, horizontal seepage was minimal within the low marsh flat. Moderate hydraulic gradients in conjunction with highly permeable soils were conducive for significant horizontal seepage within the high marsh. Hydrologic evidence indicates a potential for nutrient rich shallow groundwater recharge within the high marsh region. Sensitivity analysis within the creekbank region indicates that aquifer depth exhibits the largest influence on interstitial water discharge followed by soil permeability and specific yield properties of the aquifer respectively. Inverted results, as those found within the creekbank region, were obtained for the high marsh region.

Interstitial water nitrogen and total phosphorus levels were variable and a function of depth, location, and time. However, several generalities and patterns appeared relatively consistant. Creekbank pore waters were relatively enriched with oxidized inorganic forms of nitrogen relative to low and high marsh regions. Creekbank ammonium, total nitrogen and phosphorus interstitial pools were intermediate, whereas, dissolved organic nitrogen levels was the lowest of the three regions sampled. The low marsh flat was inorganic nitrogen poor, and intermediate with respect to dissolved organic nitrogen, relative to creekbank and high marsh regions. Pore waters within the low marsh were significantly enriched with dissolved total phosphorus as compared to the creekbank and high marsh regions. High marsh interstitial waters displayed reduced levels of nitrate and nitrite, while levels of ammonium, dissolved organic and total nitrogen were elevated in relation to the creekbank and low marsh flat. Interstitial total phosphorus levels within the high marsh were significantly lower than the low marsh and approximately equal to the creekbank region. The role and influence of subsurface hydrodynamics upon pore water nutrient concentrations and spatial variations are discussed.

Spatial and temporal potential patterns of nutrient exchange between surface water and pore waters of various wetland regions are identified. Dissolved oxidized inorganic forms of nitrogen were imported throughout the sampling period by the creekbank, low marsh flat, and high marsh regions. Ammonium flux, due to seepage , was predominantly from the wetland to surface waters; the high marsh exhibited a greatest potential for ammonium export. The high marsh was a source of dissolved organic nitrogen throughout the study, while the low marsh flat and creekbank regions may best be characterized as sources during winter, spring, and summer months, and potential sinks during the fall. The high marsh exhibited the potential to export dissolved total nitrogen throughout the year, whereas, the low marsh flat and creekbank exhibit export potential during spring and summer months. Patterns of total phosphorus exchange were from high marsh, and low marsh regions throughout the year, while exchange between creekbank and surface waters was minimal and temporally variable. Hydrodynamics within each wetland region must be considered in conjunction with pore water chemistry, in order to fully understand nutrient and solute transport potentials.

SUBSURFACE HYDRODYNAMICS AND NUTRIENT EXCHANGE WITHIN AN EXTENSIVE TIDAL FRESHWATER WETLAND

I. GENERAL INTRODUCTION AND STUDY OBJECTIVES

During the last decade, the degradation of coastal water quality due to increased nutrient input has received a great deal of attention. Considerable research has concentrated on understanding the physical and chemical interactions between tidal wetlands and adjacent surface waters. However, this effort has primarily focused upon surface water interactions. Water movement through a wetland soil is of significant ecological importance. It influences a wide variety of processes including evapotranspiration, soil aeration, geochemistry, and solute transport.

Situated between upland and riverine systems, extensive tidal freshwater wetlands are influenced by a variety of *in situ* and allochthonous processes. These processes effect the geohydrologic and pore water chemistry characteristics within the system. Upland, riverine and eaolian sediment sources and depositional processes, in concert with *in situ* peat formation and humification, produce a soil substrate that regulate subsurface water storage and transmission. Evapotranspiration and subsurface drainage provide mechanisms for pore water removal, while upland groundwater, surface water infiltration, and precipitation are sources of water recharge. Interstitial water nutrient dynamics and chemistry are influenced by a soils properties, transport of water through the soil, and chemical characteristics of recharging water.

This study utilizes geohydrologic and model simulation methodology to increase our understanding of subsurface hydrodynamics, its effect upon interstitial water nutrient chemistry, and to identify general patterns of interstitial water nutrient exchange between a tidal freshwater wetland and adjacent surface waters. The investigation is divided into three sections, study topics are: 1) physical and hydrophysical properties of a tidal freshwater wetland soil, 2) field monitored and model simulated wetland subsurface hydrology, and 3) interstitial water nutrient chemistry. Results from the individual topic investigations are integrated in order to further our understanding of advective pore water nutrient exchange with surface waters.

Objectives of this study are:

1) to characterize a tidal freshwater wetland soil according to its physical and hydrophysical properties;

2) to identify relationships between soil properties;

3) to determine the magnitude and spatial variability of subsurface hydrodynamics and primary controlling mechanisms;

4) to identify spatial and temporal variations of pore water nutrient levels and processes responsible for such variations;

5) to identify potential nutrient exchange patterns between various wetland soil regions and adjacent surface waters.

II. STUDY SITE DESCRIPTION

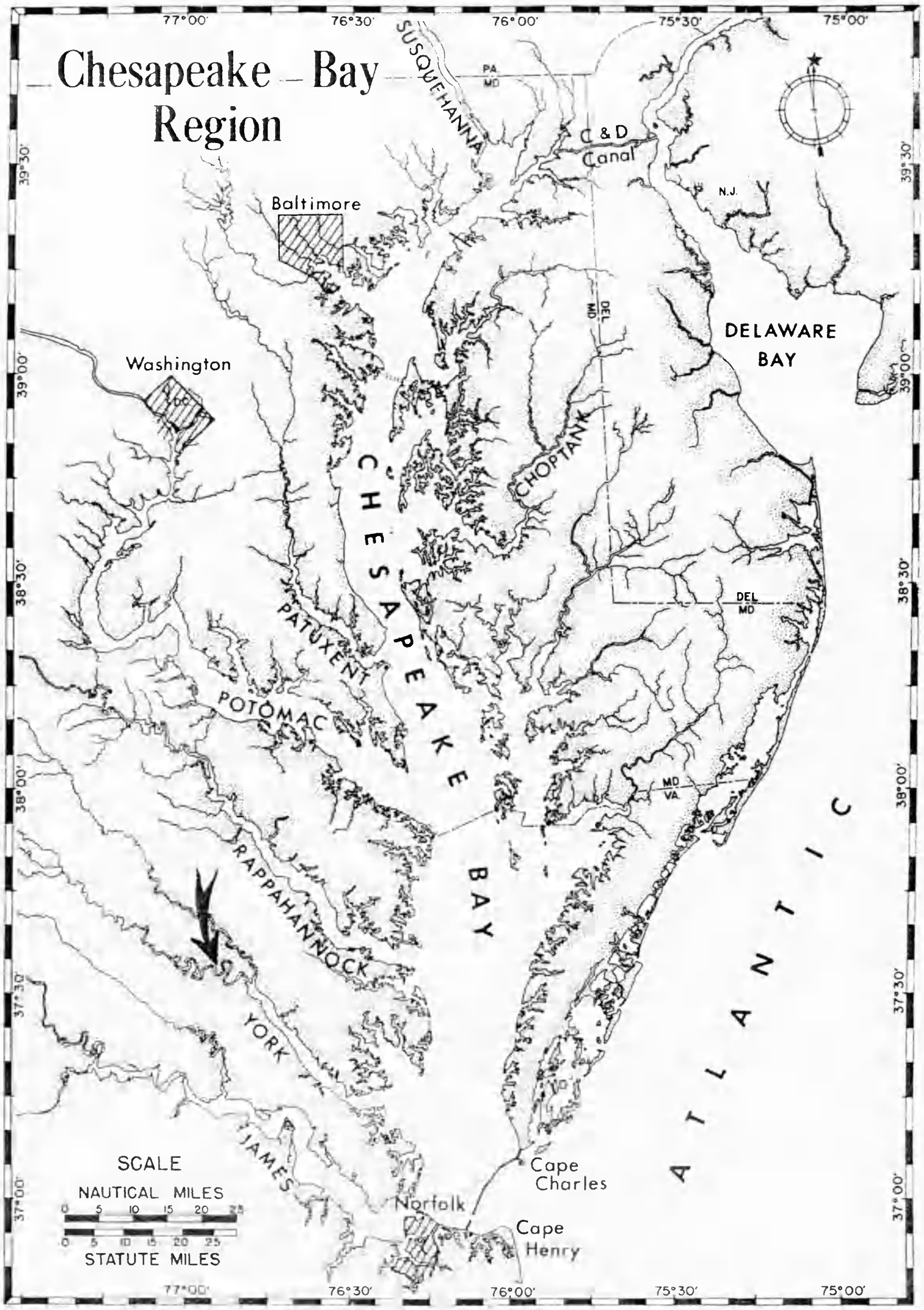
Field work was conducted at Sweethall Marsh (37^{o} 34'N, 76^{o} 54'W) located on the Pamunkey River, an upper tributary to the York River Estuary (Fig. 2.1). The 401 hectare extensive marsh (Silberhorn and Zacherle, 1987) is in a transition zone between oligohaline and freshwater, and is dominated by freshwater vegetation. A twenty year salinity record, 1968-1987 (VIMS, unpublished data), shows a range from 0 to 5ppt with a mean salinity of 0.5ppt. Mean tidal range was 82 centimeters , with mean spring and neap tidal ranges of 95 and 65 centimeters respectively (NOAA, 1987).

The vegetation pattern of a 118 meter transect normal to the creekbank can be subdivided into four general elevational regions: the creekbank, levee, low marsh flat and high marsh. The creekbank is vegetated exclusively by *Peltandra virginica* while the levee region is predominantly *Spartina cynosuroides*. The low marsh flat is best characterized as a mixed community. Dominant species include: *Peltandra virginica, Zizania aquatica, Hibiscus moscheutos, Leersia oryzoides, Scirpus robustus* and *Polygonum* species. *Typha latifolia* dominates the high marsh region.

The creekbank was relatively steep and located along an eroding bank of the thoroughfare. It grades sharply (7cm/meter) to a levee crest 6.7 meters away. The backside of the levee sloped (0.5cm/meter) into an expansive low marsh flat, which was 13 centimeters below the levee crest elevation. The low marsh flat graded (1.0cm/meter) into a narrow high marsh region. The upland interface is a steep (14cm/meter) eroding sandy bank surmounted by a row crop agricultural field. The creekbank and levee regions comprise 5 and 15 percent of the transect respectively, the extensive low marsh flat 70 percent, and the high marsh the remaining 10 percent According to Frey and Basan's (1978) saltmarsh developmental classification system and Odum's (1984) geomorphological and ecological description of tidal freshwater marsh developmental stages, this transect is representative of a youthful marsh.

Large scale bioturbation of the marsh soil was common to the study site. Burrowing activity of *Uca minax* (brackish water fiddler crab) was a prominant feature throughout the marsh. Burrows vary in size with a diameter of approximately 1 to 2 centimeters. *Ondatra zibethica* (common muskrat) excavations, ditching and tunneling activities were also common features but appeared to be concentrated along the creekbank and levee regions. Tunnels caused by *Ondatra zibethica* were on the order of 15 to 20 centimenters in diameter.

Figure 2.1: Chesapeake Bay region, showing the location of Sweethall Marsh, Virginia.



Chesapeake Bay Region
77°00'
76°30'
76°00'
75°30'
75°00'
39°30'
39°00'
38°30'
38°00'
37°30'
37°00'
SUSQUEHANNA
PA
MD
C & D Canal
N.J.
Baltimore
Washington
D.C.
DEL
MD
DELAWARE BAY
CHOPTANK
CHESAPEAKE BAY
PATUXENT
POTOMAC
RAPPAHANNOCK
YORK
JAMES
MD
VA
ATLANTIC
Cape Charles
Cape Henry
Norfolk
SCALE
NAUTICAL MILES
0 5 10 15 20 25
0 5 10 15 20 25
STATUTE MILES

III. PHYSICAL AND HYDROPHYSICAL PROPERTIES OF WETLAND SOIL

Introduction

Located between upland and riverine systems, tidal freshwater wetlands are subject to a variety of sediment sources and depositional processes. Spatial and temporal variations of allochthonous sediment sources, *in situ* peat formation and decomposition processes provide for a myriad of wetland soil characteristics. Allochthonous sediment sources include upland, riverine and aeolian components. Knowledge of wetland soil properties and their spatial heterogeniety is essential to the prediction of water movement and chemical interactions within the soil matrix. Commonly measured soil properties of wetlands include grain size distribution, dry bulk density, percent organic matter, and fiber content. These soil parameters provide insight into soil structure, and may be used to differentiate and classify soil types existing within a wetland.

In order to understand water table behavior and discharge/recharge functions of a wetland, it is essential to know water storage and transmission characteristics of the soil saturated zone. Hydraulic conductivity (K) and specific yield (Sy) are primary controlling factors in the hydrologic function of a soil. Hydraulic conductivity is a measure of the permeability of a porous media to a fluid; K is a function of both the porous media and the interstitial fluid. Typical hydraulic conductivity values range from 10^{-3} to 1cm/sec for clean sands and 10^{-6} to 10^{-3} cm/sec for clay-sand mixtures (Davis and Deweist, 1966). Specific yield, which is equivalent to the drainable porosity of an unconfined aquifer, is an estimate of the water yielding capacity of a soil. It can be defined as the percent of water released per unit area of aquifer material given a unit drop in water table elevation. Specific yield values range from 2 percent for aquifer material composed of clay sized particles to 30 percent for sandy aquifer material (Davis and Dewiest, 1966). Hydrophysical soil properties are influenced by the structural and compositional characteristics of the soil matrix, these include: botanical composition, fiber content, degree of humification and compaction, volumetric water content, grain size and configuration, and bulk density.

3.1 Experimental Methods

Soil properties were measured along a transect running perpendicular to the thoroughfare and extending to the upland edge. An elevated walkway was constructed parallel to the transect to minimize marsh surface disturbance and provide easy access. Sampling stations, 1through 8, were located 0.7, 2.7, 6.7, 15.0, 30.0, 60.0, 100.0, and 118.0 meters from the creekbank respectively. Replicate cores were taken with a modified Davis peat corer (1.4cm i.d.) at depth intervals of 0-20, 20-40, 50-70, and 80-100 centimeters. Random subsamples were taken from each 20cm core and analyzed for dry bulk density, percent organic matter, and fiber content. A total of nine subsamples were taken from each core; three subsamples were used in the determination of each physical soil property. The arithmetic mean of the subsamples was used for the final parameter value for the particular core. Subsamples used in dry bulk density determinations were oven dried at 105^{o}C for 24 hours and weighed. Dry bulk density is expressed as dried mass per field condition volume of the subsample. An estimate of soil organic matter was determined by combusting the dried subsamples at 500^{o}C for 5 hours followed by reweighing. Soil organic matter is expressed as a percentage weight loss from combustion of the dried subsample. Fiber content was determined by weighing the oven dried material retained on a 1mm mesh screen sieve after removal of large mineral particles (i.e., sand grains). Fiber content is expressed as dried mass of fiber per field condition volume of the subsample.

In order to describe grain size distribution, a series of cores were taken with the Davis peat corer along the transect at the depth intervals stated above. Cores were taken at the creekbank/levee region, low marsh flat, and the high marsh near the upland interface. Each 20 centimeter core segment was homogenized, subsampled and analyzed for grain size by wet sieving and pipet analysis (Folk, 1980). The Wentworth grain size scale was used to separate the soil into sand, silt, and clay size fractions.

Saturated volumetric water content within the upper 35 centimeters of soil was determined for the four general marsh regions; the creekbank, levee, low marsh flat, and high marsh. Replicate cores (40cm length, 7.6cm. i.d.) were taken prior to marsh surface exposure by the ebbing tide. These soil samples were assumed to be saturated with respect to water. Cores were sealed and immediatly returned to the laboratory. Saturated samples were taken at 5 centimeter intervals, weighed, oven dried at 105^{o}C for 24 hours, and reweighed. Percent water content is expressed as a percentage weight loss of the saturated sample per field condition volume of the sample (Boelter and Blake, 1964). An estimate of soil water yielding capacity was determined by taking another set of replicate cores when the water table had reached a minimum level. Water table elevations were monitored by nearby piezometers and wells. Assuming no change in soil volume, percent water loss over a tidal cycle was determined by the difference in volumetric water content between saturated and gravity drained samples. Drained soil samples were only taken at the creekbank and levee crest where observations indicated greatest tidal influence on the water table.

Soil horizontal hydraulic conductivity (K) were determined *in situ* on undisturbed saturated marsh soil. Measurements were made along the transect at the following depth intervals: 0-20, 20-40, 50-70, and 80-100 centimeters. Sampling stations incorporated the four general elevational regions: the creekbank, levee, low marsh flat, and high marsh. Hydraulic conductivity of the 0-20 centimeter depth interval was measured by the auger hole technique with utilizing Hooghoudt's equation (eq. 3.1) (Luthin, 1957; Rycroft et. al., 1975). Rate measurements continued until the cavity was 25% recharged.

$$K = [(2.3)(r^2)(Ah)][\log(h_1/h_2)][(2c+r)(t_2-t_1)]^{-1} \qquad \text{(eq. 3.1)}$$

Where:

K = Horizontal hydraulic conductivity (cm/sec)

Ah = Cavity shape factor (cm)

t_2-t_1 = Time interval for water level change from h_1 to h_2 (cm)

r = Radius of auger hole or cavity (cm)

h_1 = Distance from water table to water level at t_1 (cm)

h_2 = Distance from water table to water level at t_2 (cm)

c = Distance from water table to base of augered cavity (cm)

Hydraulic conductivity of the remaining depth intervals were determined following the piezometer tube method and utilization of equation 3.2 (Luthin, 1957; Rycroft et. al., 1975).

$$K = (3.14)(r^2)[\log_2(h_1/h_2)][(Ah)(t_2-t_1)]^{-1} \qquad \text{(eq. 3.2)}$$

In principle, both methods measure the rate of water recharge into the auger hole/piezometer following evacuation of water. A modification to the methods was necessary to maintain the integrity of the augered cavity. Once the cavity was augered, a 20 centimeter long screen mesh cylinder (0.5cm mesh size) was inserted into the augered hole or PVC casing (3.5cm i.d., 4.5cm o.d.) and installed at the proper depth location. Soil hydraulic conductivity was measured at three locations within each general marsh region and each auger hole/piezometer test was replicated to assure consistency. The arithmetic mean was taken as the hydraulic conductivity estimate for a particular depth interval within a region. Temperature of the recharging water was recorded at several sites. In most instances, there was a slight initial decline of hydraulic conductivity with recovery time.

Therefore, reported values were determined from stabilized terminal readings (Rycroft et. al., 1975; Hemond and Goldman, 1985).

3.2 Results

Grain size distribution descriptive statistics are given in Table 3.1. The high marsh/upland edge station had a large sand component, approximately 44% on a dry weight basis. This was primarily due to input from the eroding sandy bank of the adjacent uplands. The creekbank/levee and low marsh flat regions were primarily composed of a silt-clay sediment mixture of riverine origin. The creekbank/levee site had a slightly higher percent silt and lower percent clay composition than the low marsh flat site. This gradation of grain size may be explained by the settling of silt-sized particles within tidal water as it inundates the creekbank/levee region (Pestrong, 1977).

Lateral and vertical variations in bulk density, percent organic matter, and fiber content along the transect are given in Figures 3.1 to 3.3. Descriptive statistics for the three soil parameters are given in Tables 3.2 to 3.4. Surface to 20cm depth interval bulk density values decreased from the creekbank (station #1) (0.576g/cm^3) to the high marsh site (station #8) (0.247g/cm^3). The levee region (station #3) (0.440g/cm^3) displayed bulk density values intermediate between the creekbank and high marsh sites. Variation with depth is apparent at the back-levee site (station #4), low marsh/high marsh transitional zone (station #7), and the high marsh/upland edge interface (station #8). The decrease in bulk density values with depth at the back-levee site, from 0.462 to 0.256g/cm^3, was apparently due to an increase in organic matter. Mean percent organic matter and fiber content values increased from 16.1 percent and 0.008g/cm^3 at the 0 to 20cm depth interval to 27.4 percent and 0.011g/cm^3 at the 80-100cm depth interval respectively. Bulk density values increased with depth at the low marsh/high marsh and high marsh/upland edge transition zones; values increased from 0.202 to 0.610g/cm^3 and 0.247 to 0.578g/cm^3 respectively.

Visual observations of cores from the low marsh/high marsh transition zone and high marsh indicate that an increase in sand content with depth was responsible for the increase in soil bulk density with depth. Variability in terrestrial sand input to the high marsh regions appears to have caused discrepencies in data. The bulk density, percent organic matter, and fiber content data indicate a very fibrous, organic soil of low bulk density within the upper 40 centimeters og the high marsh/upland edge interface. However, the single core used in grain size analysis indicates a fairly dense, sandy sediment.

A multivariate statistical technique, Q-mode factor analysis (Di Zhou et. al., 1983) was used to establish, simplify, and classify relationships between sampling sites based on physical soil parameters. The reader is referred to Davis (1986) for a general overview of principal components and factor analysis. The raw data matrix [X] of soil parameters versus sample location was transformed by dividing each value in a given variable column by the highest value observed in that column. This scaling served to give each variable an equal weighting. The scaled matrix was then row-normalized. The major product moment, or matrix of cosine similarity [W], was calculated by multiplying the standardized data matrix [X] by its transpose $[X]^T$. Eigenvalues and eigen vectors (principal components) were determined for the similarity matrix [W]. The sum of the eigenvalues of the cosine similarity matrix [W] represents the total variance of the data set. Eigenvalues, percent sums of squares, and cumulative percent sums of squares are given in Table 3.5. Principal component I, representative of soil bulk density, accounted for the majority (87%) of total variation in the data set. Principal component II, representative of percent organic matter and uncorrelated with principal component I, accounted for 12 percent of the total variation. The third principal component, representative of fiber content, showed an insignificant contribution (1%<) in explaining total variance and was eliminated from further analysis. This simplified the analysis by reducing the dimensions of the data. Fiber content, being a size class measure of soil organic matter, had a high correlation (r^2=0.72) with percent organic matter. The strong positive relationship between the two soil parameters indicate the redundant nature of information contained within the data set. Principal components I and II may be considered as initial factors.

Factor loadings indicate the correlation of a sample with a particular factor. In order to provide more interpretable results, factors were orthogonally rotated (varimax rotatation) to position sample vectors closer to extreme regions or the origin of the factor axes (i.e., adjusting factor loadings so they are closer in value to 1 or 0). Communalities, determined by the sum of the squared factor loadings of a particular sample, indicate how well a sample vector is explained by individual factors. Orthogonally rotated factor loadings and communalities are given in Table 3.6 and loadings are plotted in Figure 3.4. Examination of communalities indicate that all sample locations are well represented by the two factors. Rotation of the factors distributed the total variance explained by the factors more evenly; rotated factor I and II explained 54 and 45 percent of the total variance respectively. Figure 3.4 indicates two major groupings of the sample vectors; one group loading high with respect to factor I and another with respect to factor II. End member samples which are exemplified by extreme soil properties, were identified as station code #4 and #25. These sampling stations correspond to the creekbank 80-100cm and the low marsh flat/high marsh transitional region 0-20cm depth interval respectively. Mean soil parameter properties for the end members stations are:

Factor #	Sta./depth	Code#	Bulk density	% Organic	Fiber content
1	1/80-100cm	4	0.662g/cm^3	10.2	0.002g/cm^3
2	7/0-20cm	25	0.202g/cm^3	29.4	0.021g/cm^3

Having factor loadings between end member stations, intermediate stations may be described as compositional mixtures of the end members. Determination of end member association was accomplished by an oblique rotation of the factor loadings. Examination of the denormalized, oblique rotated loadings matrix, shows that the marsh transect may be differentiated into four general soil types. Characterized by varying degrees of end member association, the four soil types may be described as follows: 1) greater than 75% to 100% association with factor I end member, 2) greater than 50% to 75% association with factor I end member, 3) greater than 75% to 100% association with factor II end member, and 4) greater than 50% to 75% association with factor II end member. By subdividing the marsh transect into these four general soil classifications, the transect was differentiated into four general regions (Fig. 3.5). These four regions correspond to the four general elevational regions: the creekbank, levee, low marsh flat, and high marsh. Soil classification identifies similar soil types along the creekbank, levee, and deeper low marsh flat/high marsh transition and high marsh regions. Whereas these soils exhibit similar bulk densities, percent organic matter and fiber content, grain size analysis (Table 3.1) shows the creekbank/levee region to be primarily composed of silt-clay sized particles while the deeper low marsh flat/high marsh transition and high marsh regions are characterized by a relatively high sand content.

Horizontal hydraulic conductivity measurements were made in the four general soil regions identified by Q-mode factor analysis. Vertical and lateral variation in hydraulic conductivity was observed along the transect from the creekbank to the high marsh site. Mean hydraulic conductivity values are plotted in Figure 3.6 and descriptive statistics are given in Table 3.7. Hydraulic conductivity measurements followed a log-normal distribution. Cochrans C and Bartlett-Box procedures were used to test for homogeniety of variances. One-way analysis of variance of log transformed data ($F_{.05,14,30}$=1.06, Prob.=0.000) indicates that soil permeability was not homogeneous along the transect. The creekbank 80-100cm depth interval was excluded from analysis due to lack of replication; soil permeability was so low within the region that the sample sites were inundated by tidal waters before proper measurements could be made.

A Tukey HSD multiple comparison test, at the 0.05 alpha level, indicates significant differences with depth at the levee and high marsh sampling locations. The 0-20cm (2.3 x 10^{-4}cm/sec) and 20-40cm (2.8 x 10^{-4}cm/sec) depth intervals were significantly different from the 50-70cm (1.1 x 10^{-5}cm/sec) depth interval at the levee. The high marsh showed significant differences between the 0-20cm (1.9 x 10^{-3}cm/sec) depth interval and the 50-70cm (6.0 x 10^{-5}cm/sec) and 80-100cm (7.6 x 10^{-5}cm/sec) depth intervals. Significant lateral variations occured in the 0-20cm depth interval between the creekbank and high marsh site; mean hydraulic conductivity values were 1.1 x 10^{-4}cm/sec and 1.9 x.10^{-3}cm/sec respectively. At the 20-40cm depth interval, the creekbank site (1.6 x 10^{-5}cm/sec) varied significantly from the levee (2.8 x 10^{-4}cm/sec), low marsh flat (5.4 x 10^{-4}cm/sec), and the high marsh (1.1 x 10^{-3}cm/sec) sites. Significant differences were also found between the levee (1.1 x 10^{-5}cm/sec) and low marsh flat (2.2 x 10^{-4}cm/sec) site at the 50-70cm depth interval. No significant differences in hydraulic conductivity were found at the 80-100cm depth interval between the levee, low marsh flat, and high marsh.

The relationship between log transformed hydraulic conductivity and various soil parameters were determined by simple linear regression. The correlation matrix of relationships between mean hydraulic conductivity, bulk density, percent organic matter, and fiber content is given in Table 3.8. All regression line slopes were significantly different from 0 at the 0.05 alpha level. Positive linear relationships were observed between hydraulic conductivity and percent organic matter, and fiber content. Dry bulk density displayed an inverse relationship with hydraulic conductivity and the other soil parameters (Fig. 3.7).

A step-wise inclusion multiple regression procedure was utilized in order to provide linear prediction models and assess which of the independent soil parameters explained most of the variation observed in soil hydraulic conductivity values. An overall F test indicates that the multiple soil parameter linear model explained 76 percent of the variability observed in hydraulic conductivity values ($F_{.05,3,12}$=12.53) (Table 3.9). The multiple linear prediction equation (eq. 3.3) for horizontal hydraualic conductivity is:

$$K = 10^{(62.76(X1) - 4.22(X2) - 0.05(X3) - 1.74)} \qquad \text{(eq. 3.3)}$$

Where,

K = Horizontal hydraulic conductivity (cm/sec)

X1 = Fiber content (g/cm^3)

X2 = Dry bulk density (g/cm^3)

X3 = Percent organic matter

Partial F tests indicate that the addition of bulk density and percent organic matter to the model, did not significantly improve the prediction ability of hydraulic conductivity over fiber content alone. Fiber content accounted for 69 percent of the variation observed in permeability estimates.The prediction equation (eq. 3.4) excluding bulk density and percent organic matter is:

$$K = 10^{(85.33(X1) - 4.70)} \qquad \text{(eq. 3.4)}$$

The high marsh region, where surface soils were characterized by low bulk densities, and high fiber and organic contents, exhibited mean volumetric water content values of over 90 percent (Fig. 3.8). Deeper sandy sediments (below 50cm from the surface) within the high marsh region were characterized by a volumetric water content of approximately 50 percent. Soil volumetric water content values approximated 80 percent within the creekbank and levee regions. Soil water content of the low marsh flat, approximately 85 percent, was intermediate between the high marsh and creekbank/levee regions. Estimates of specific yield of the creekbank and levee soils were 2 and 3 percent respectively.

3.3 Summary and Discussion

Spatial variations of soil properties within an extensive tidal freshwater wetland indicate that the soil was not a homogeneous unit. Multivariate analysis best described a 118 meter transect running from a creekbank to the upland edge as four separate general soil classes (Fig. 3.5). These four general soil classes correspond to the general elevational regions found within a typical extensive tidal wetland: the creekbank, levee, low marsh flat, and high marsh. The creekbank region, a narrow region approximately three meters in width and exclusively vegetated by*Peltandra virginica* , was characterized by high soil bulk densities (0.566g/cm^3), low percent organic matter (11.7%), and low fiber content (0.004g/cm^3). The upper one meter soil profile of the creekbank appeared to be homogeneous with respect to soil organic and fiber content, and bulk density. The levee region extended 20 meters inward from the creekbank and was predominately vegetated by *Spartina cynosouroides*. Soil characteristics within the levee region displayed vertical and lateral differences and appeared transitional between the creekbank and the extensive low marsh flat. Soil bulk densities varied from 0.506 to 0.230g/cm^3, decreasing with distance from the creekbank. Percent organic matter and fiber content varied from 11.4 to 32 percent and 0.001 to 0.019g/cm^3 respectively, these properties increased in value with distance from the creekbank. Tunneling activity of *Ondatra zibethica* was concentrated along the creekbank and levee region. It appears that the inorganic, clay-like soil properties found within these regions allow for such macroporosity, whereas interior marsh soils would immediately subside or collapse in on such structures.

Vegetated by a high diversity of plants and comprising the majority of the transect, approximately 80 percent, the low marsh flat comprised the third general soil type. The third soil type extended from the back-levee to the low marsh/high marsh interface, a distance of 100 meters. Mean bulk density, percent organic matter, and fiber content values were $0.274 g/cm^3$, 27.0 percent, and $0.017 g/cm^3$ respectively. As with the creekbank, the low marsh flat displayed minimal variation in soil properties with respect to depth. The high marsh, about 18 meters wide and predominantly vegetated by *Typha latifolia*, comprised the fourth soil type. This region displayed the greatest variation of soil properties with respect to depth. Bulk density increased from 0.247 at the 0-20cm depth interval to $0.578 g/cm^3$ at the 80-100cm depth interval, while percent organic matter and fiber content decreased from 29.6 to 13.7 percent and 0.027 to $0.005 g/cm^3$ respectively. The high marsh region may be described as a thin layer of peat substrate overlying a sandy, inorganic sediment of terrogenous origin. Incorporating all sampling stations, overall mean values for a one meter depth along the transect were $0.427 g/cm^3$ for bulk density, 18.2 for percent organic matter, and $0.010 g/cm^3$ for fiber content.

Soils of similar physical properties were found along the creekbank, below 50 centimeters at the levee, and at depth within the low marsh flat/high marsh transitional zone and high marsh regions. However, grain size analysis and visual observation indicate these soils differ significantly. The creekbank/levee soils display low sand content (5.1% on a dry weight basis), whereas sand sized sediment dominated the 80-100cm depth interval in the high marsh (greater than 60% on a dry weight basis). Although limited, grain size data indicates that the creekbank/levee and low marsh flat regions are dominated by riverine input of silt and clay sized particles. The high marsh received a large contribution of sand sized particles from the eroding uplands in addition to silt and clay sized particles of riverine origin.

Soil parameters measurements concur with other investigations of tidal freshwater wetlands. Percent organic matter estimates were highest in the high marsh and low marsh regions, approximately 30 percent; regions were dominated by *Typha latifoilia* and mixed vegetation communities respectively. Minimum percent organic matter estimates (11.8%) occured along the *Peltandra virginica* dominated creekbank. Whigham and Simpson (1975) reported percent organic matter values ranging from 14 to 40 percent for a tidal freshwater wetland along the Delaware River. Investigators also noted a similar decreasing trend of soil organic matter as one moved from the high marsh towards the creekbank. Bowden (1982) reported organic matter values ranging from 40 to 75 percent along the North River in Massachussetts. Percent organic matter estimates along the Chickahominy River in Virginia varied between 20 and 59 percent (Hoover, 1984; Odum et. al., 1984). Dry bulk density and fiber content values were not given by the above investigators.

Peat soil, as defined by Vollmer (1967), is "organic soil formed by the accumulation in wet areas of the partially decomposed remains of vegetation, and with less than 20 percent of mineral matter". High riverine and upland contributions of inorganic material, in conjunction with lower biomass and density of roots and rhizomes of tidal freshwater vegetation (Silberhorn et. al., 1974; Garofalo, 1980), deterred the formation of peat soil within the study site. This observation concurs with Odum (1984) who reported that peat formation within northern and southern tidal freshwater wetlands is not as common as in brackish or salt marshes. The *Typha latifolia* dominated high marsh soil was the most closely related soil type to a typical peat within the extensive tidal freshwater wetland.

Spatial heterogeneity of hydraulic conductivity was closely associated to variations in soil types found within the wetlands. Significant relationships existed between hydraulic conductivity and various physical soil properties (Fig. 3.7). The coefficient of determination (r^2) of log transformed hydraulic conductivity and dry bulk density was 0.66, 0.55 with percent organic matter, and 0.72 with fiber content. Verry and Boelter (1978) found a similar positive relationship between hydraulic conductivity and fiber content (r^2=0.54) and inverse relationship with bulk density (r^2=0.54).

Utilizing a multiple linear regression procedure, fiber content was identified as the measured parameter which best explained variations of soil permeability observed throughout the wetlands. Being a size fraction measurement of organic matter within the soil, fiber content provides information on the degree of humification within a soil. Soils that have undergone a high degree of humification or decomposition will tend to be less permeable as compared to soils experiencing little or none (Rycroft et. al., 1975). Fiber sized organic matter apparently increases channelization within the soil matrix, thereby increasing a soils ability to transmit water. Prediction equations were developed in order to provide hydraulic conductivity estimates when given specific soil properties (i.e., fiber content, bulk density, percent organic matter). The linear regression model (eq. 3.3), incorporating fiber content, bulk density, and fiber content, accounted for 76 percent of the variability of field hydraulic conductivity measurements. A simplified model (eq. 3.4), incorporating only fiber content, accounted for 69 percent of the total variation observed in soil permeability.

Since hydraulic conductivity of the wetland soil was highly dependent on its fiber content, temporal variations in soil permeability may occur in response to root formation during the course of the growing season. Since *Typha latifolia* and *Spartina cynosouroides* have dense shallow root systems, regions dominated by such vegetation have an increased potential for temporal soil permeability variations. Booth (1989) reported increases in soil organic matter within tidal freshwater wetland *Spartina cynosouroides* region throughout the growing season.

The creekbank region did not display significant variation in soil permeability with respect to depth. Soil structure within the upper one meter of this region was fairly uniform nature, consisting of a dense, low organic soil of clay/silt sized particles. The creekbank region lacked a dense fibrous root system, since it was dominated by *Peltandra virginica* which has a large fleshy tuber. Having mean hydraulic conductivity values that ranged between 1.1 x 10^{-4} and 6.3 x 10^{-6}cm/sec, the creekbank region was the least conducive to water transmission.

Hydraulic conductivity within the levee varied significantly with depth. This region, as shown in Figure 3.5, displayed large differences in soil structure within the upper one meter profile. Mean soil permeability estimates within the 0-20cm and 20-40cm depth intervals were 2.3 x 10^{-4} and 2.8 x 10^{-4}cm/sec respectively, while the 50-70cm and 80-100cm depth intervals displayed mean hydraulic conductivities estimates an order of magnitude less. *Spartina cynosouroides*, which dominates the levee region, has a shallow dense root system. Data indicates a mean fiber content of 0.007g/cm^3 in the upper forty centimeters, whereas the 50-70cm and 80-100cm depth intervals had fiber content values of 0.002 and 0.001g/cm^3 respectively. Bulk density and percent organic matter did not show a substantial change with depth.

The upper one meter of soil within the low marsh flat was a uniform low density, fibrous organic soil (Fig. 3.5), thus hydraulic conductivity did not vary significantly with depth. Comprising the majority of the transect, the low marsh flat had the potential for relatively rapid horizontal water movement. Horizontal mean hydraulic conductivity values ranged from 1.4 x 10^{-3} to 2.2 x 10^{-4}cm/sec.

The most permeable soil, 1.9 x 10^{-3}cm/sec, occured within the *Typha latifolia* dominated high marsh surface-20cm depth interval. *Typha latifolia* has a dense shallow root system similar to *Spartina cynosouroides*; mean fiber content and percent organic matter at this depth interval was 0.027g/cm^3 and 29.6 respectively. Hydraulic conductivity measurements varied significantly with depth; the 0-20cm depth interval differed significantly from the underlying 50-70cm (6.0 x 10^{-5}cm/sec) and 80-100cm (7.6 x 10^{-5}cm/sec) depth intervals. Differences in permeability were due to variations in soil structure of the individual depth intervals (Fig. 3.5). A organic fibrous soil was underlain by a dense sand/silt/clay mineral sediment. Harvey (1986) observed similar results of a highly permeable shallow marsh soil underlain by a less permeable sand/clay substrate.

Knott et. al. (1987) suggests that humified organic matter may act to seal interstitial pore spaces between mineral particles and cause a decrease in substrate permeability.

In summary, there was a increasing trend in hydraulic conductivity within the upper 40 centimeter profile as one moved from the creekbank towards the high marsh (Fig. 3.6). Significant lateral differences were observed, most noticably within the upper 40 centimeters between the high marsh and creekbank edge where there was a two order of magnitude difference.

Rycroft et. al. (1975) provides a comprehensive summary of non-tidal wetland hydraulic conductivity studies. Permeability values range from 6 x 10^{-8}cm/sec for highly humified peat to 5 x 10^{-3}cm/sec for undecomposed peat. Several soil permeability investigations have been conducted in brackish and salt marshes, however, no published studies were found for tidal freshwater wetlands where botanical composition exhibits such high diversity. A summary table of these studies is given in Table 3.10. Comparison of these studies indicate that permeability values reported in this study are similar to brackish and salt marshes found along the Atlantic coast.

Saturated tidal freshwater marsh soil, on a percent volume basis, was primarily composed of water; mean volumetric water content values ranged between 77 and 90 percent. Results concur with other investigations on various wetland peat types. Boelter (1964) reports volume water contents for several types of saturated peats, these include: 1) undecomposed moss peats (96.6%), 2) decomposed moss peats (86.5%), and 3) herbaceous peats (86.9%). Dasberg and Neuman (1976), investigating inland wetland soil properties, reported values exceeding 90 percent and Agosta (1985) reported values ranging from 80 to 85 percent for a east coast *Spartina* dominated saltmarsh. Less dense organic soils, like those found in the low marsh and high marsh regions, had a higher volumetric water content than the more dense inorganic soils found along the creekbank. Incorporation of water into the root and rhizome matter is partially responsible for this trend.

Estimates of specific yield along the creekbank and levee region were 2 and 3 percent respectively. This indicates that after several hours of drainage (i.e., over a tidal cycle), approximately 95% of the water was retained within the drained soil matrix. Retention of the water is due to strong capillary action and the small drainable pores found within clay/silt substrates. Specific yield estimates would be expected to increase as one moved from the creekbank towards the upland, since the low marsh flat and high marsh soils are composed of higher organic and fibrous material. Investigations on specific yield estimates within wetlands systems is relatively sparse. Harvey (1986), working in a *Spartina alterniflora* saltmarsh, reported a mean value of 3.2 percent. Dasberg and Neuman (1977) report values of 20 to 30 percent, and Boelter (1965) reported a wide range of estimates (10 to 79%) for several inland peat types.

Figure 3.1: Spatial distribution of soil dry bulk density along wetland transect.

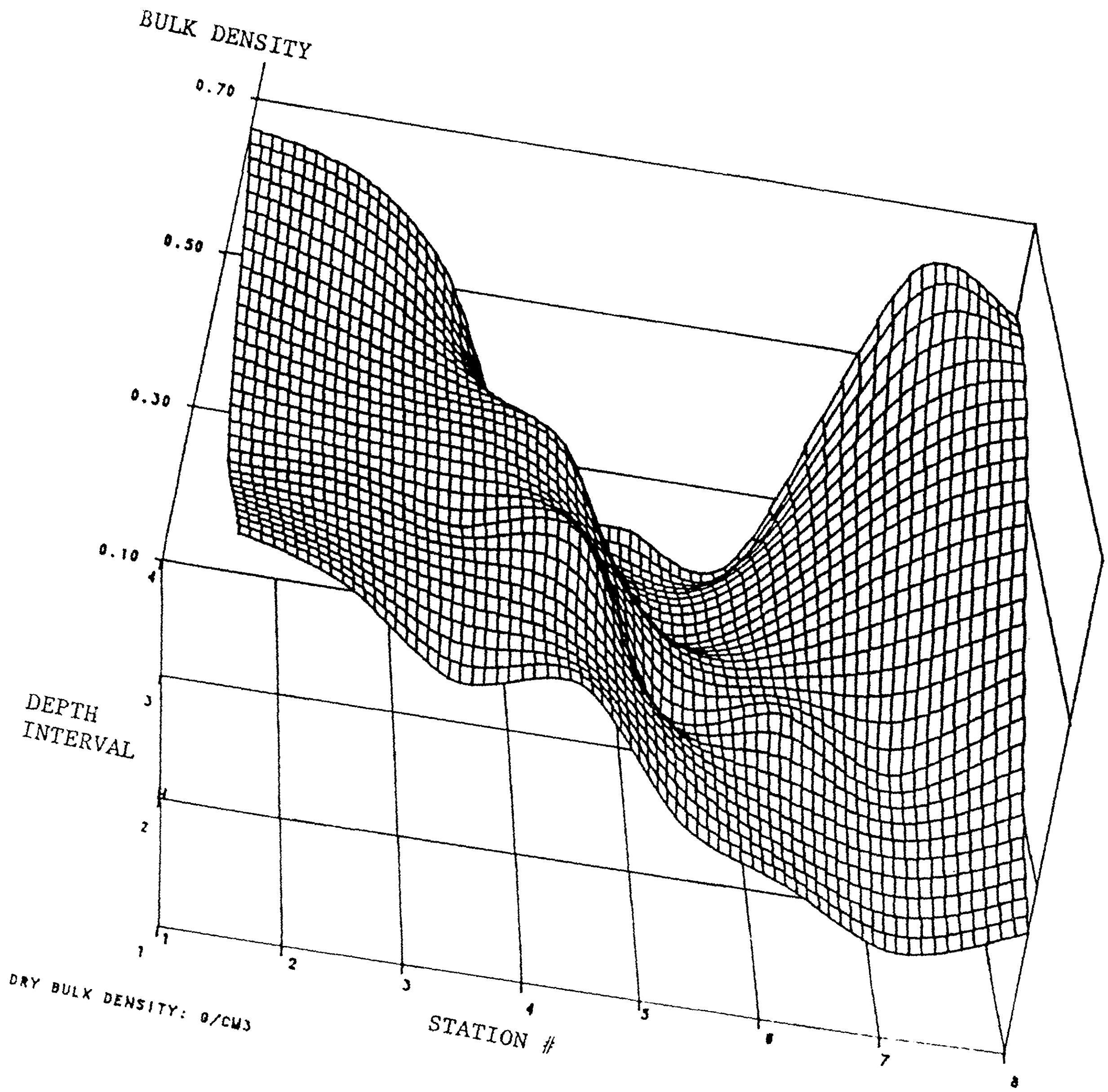
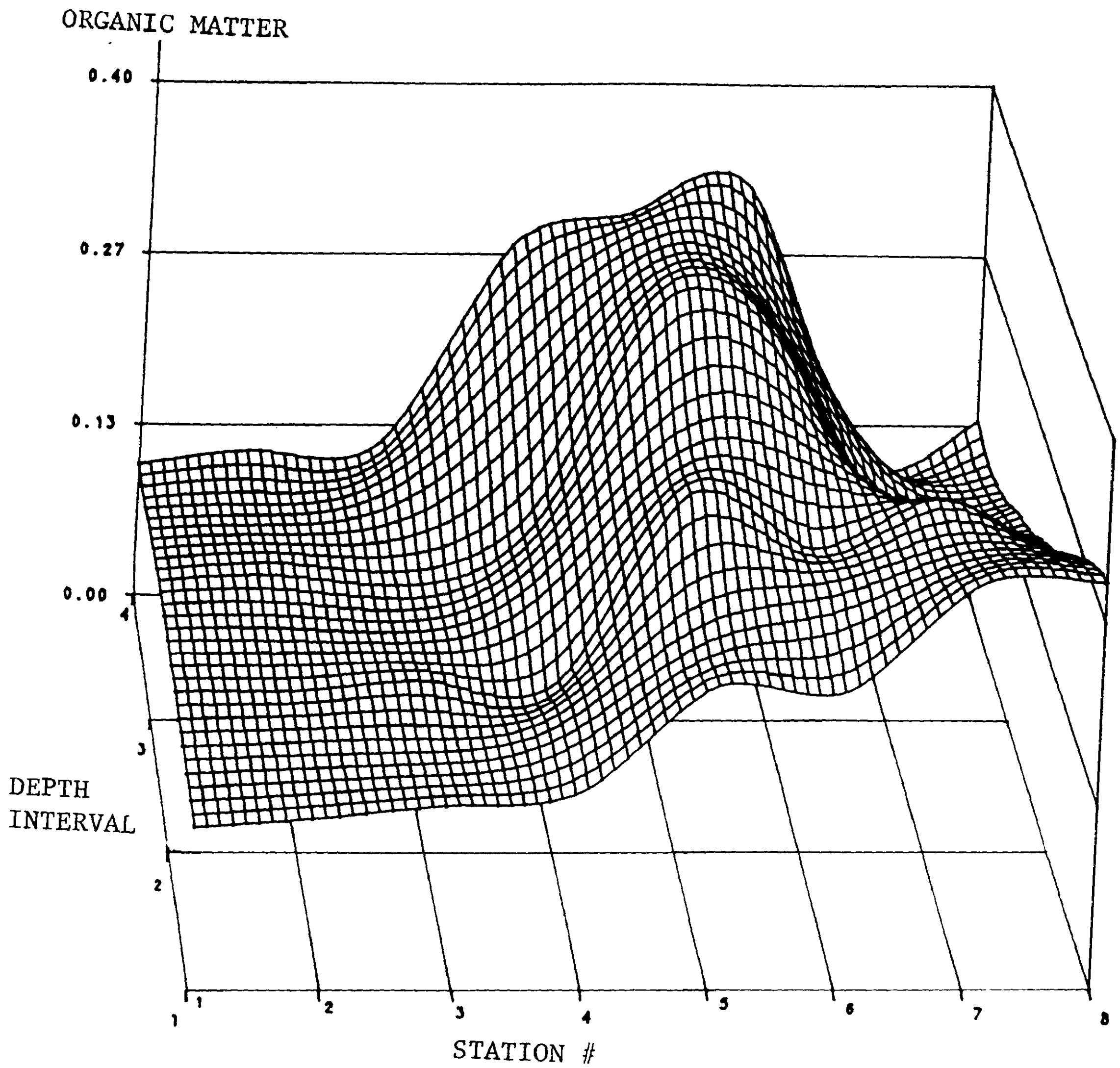
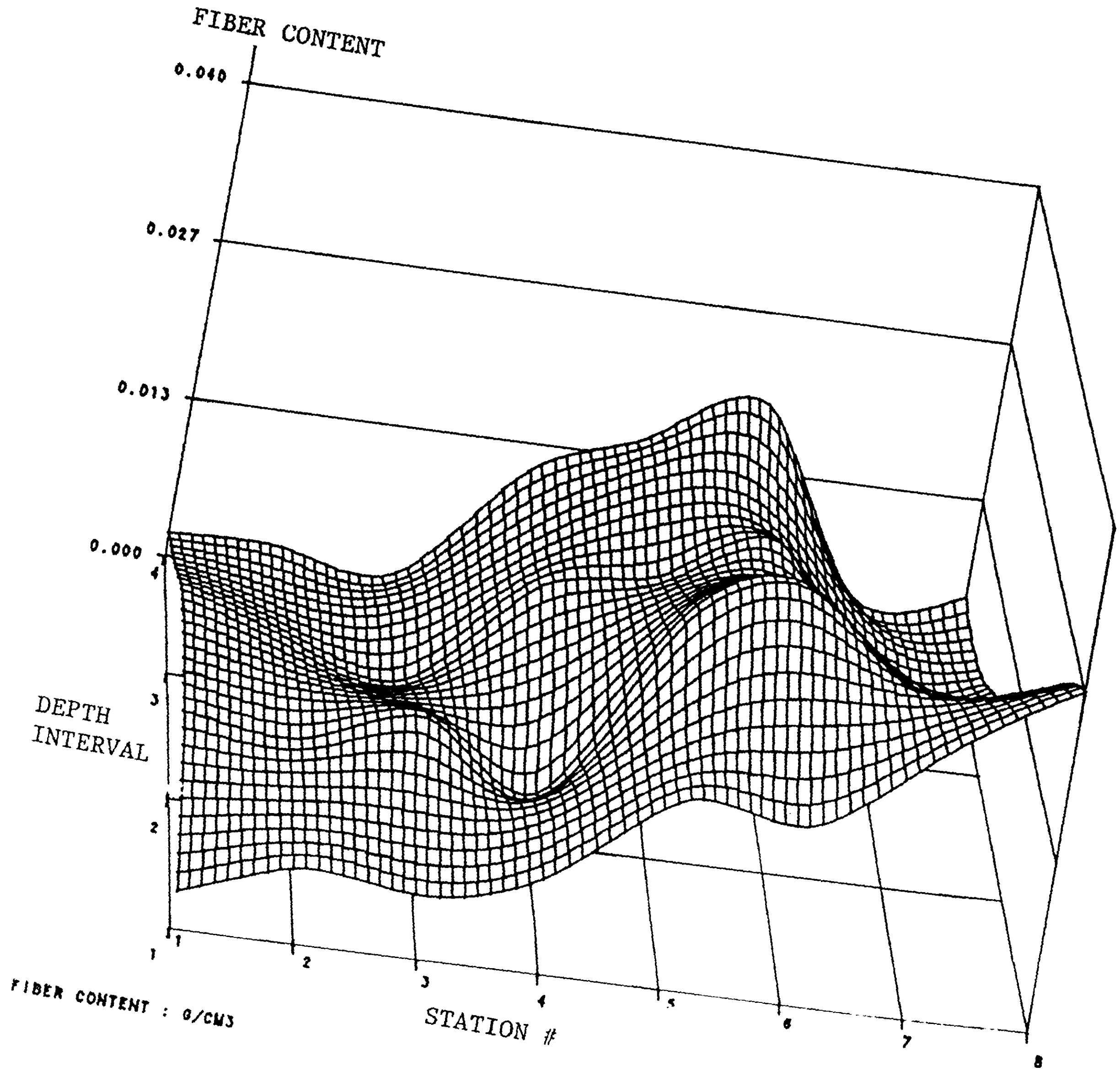

BULK DENSITY
0.70
0.50
0.30
0.10
DEPTH
INTERVAL
4
3
2
1
1
2
3
4
5
6
7
8
STATION #
DRY BULK DENSITY: G/CM3

Figure 3.2: Spatial distribution of soil percent organic matter along wetland transect.



ORGANIC MATTER
0.40
0.27
0.13
0.00
DEPTH
INTERVAL
4
3
2
1
1
2
3
4
5
6
7
8
STATION #

Figure 3.3: Spatial distribution of soil fiber content along wetland transect.



FIBER CONTENT
0.040
0.027
0.013
0.000
DEPTH
INTERVAL
1
2
3
4
1
2
3
4
5
6
7
8
STATION #
FIBER CONTENT : G/CM3

Figure 3.4: Plot of factor loadings on the two varimax factor axes.

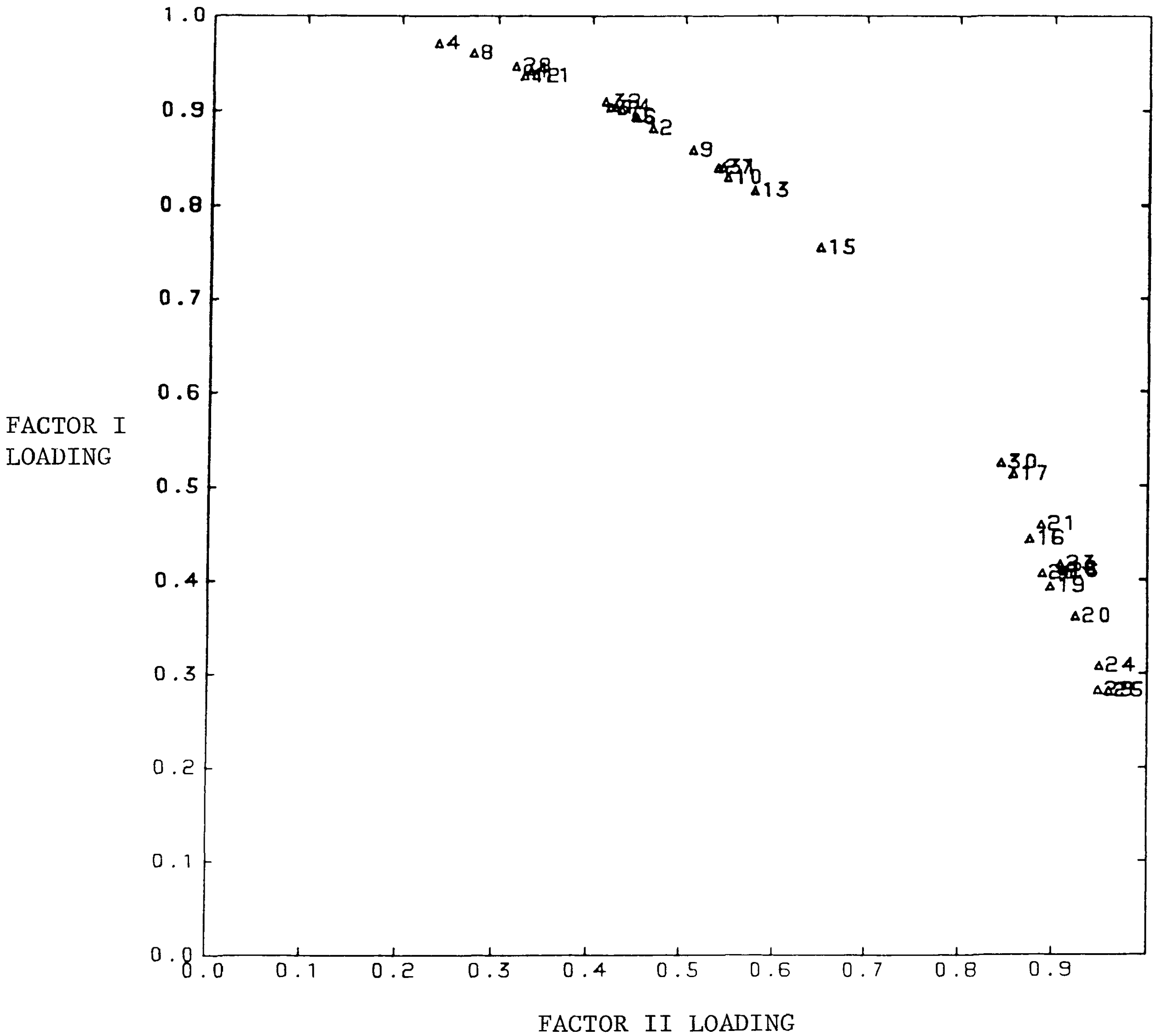
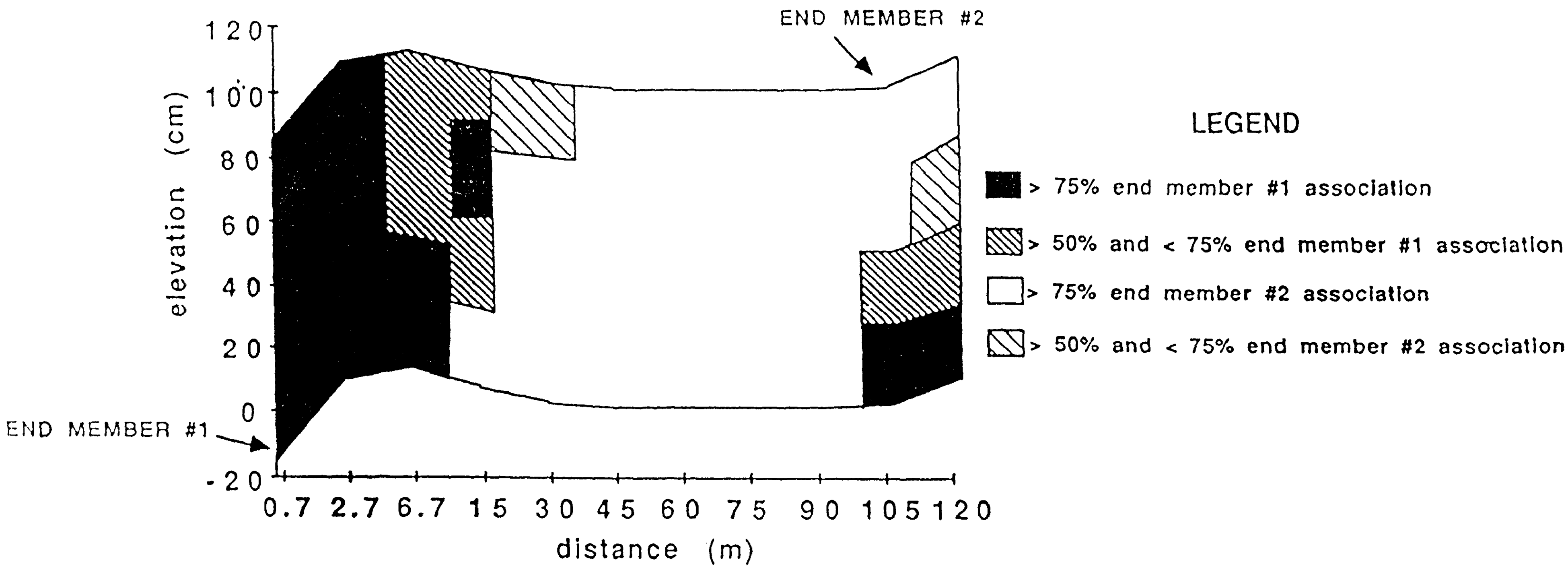
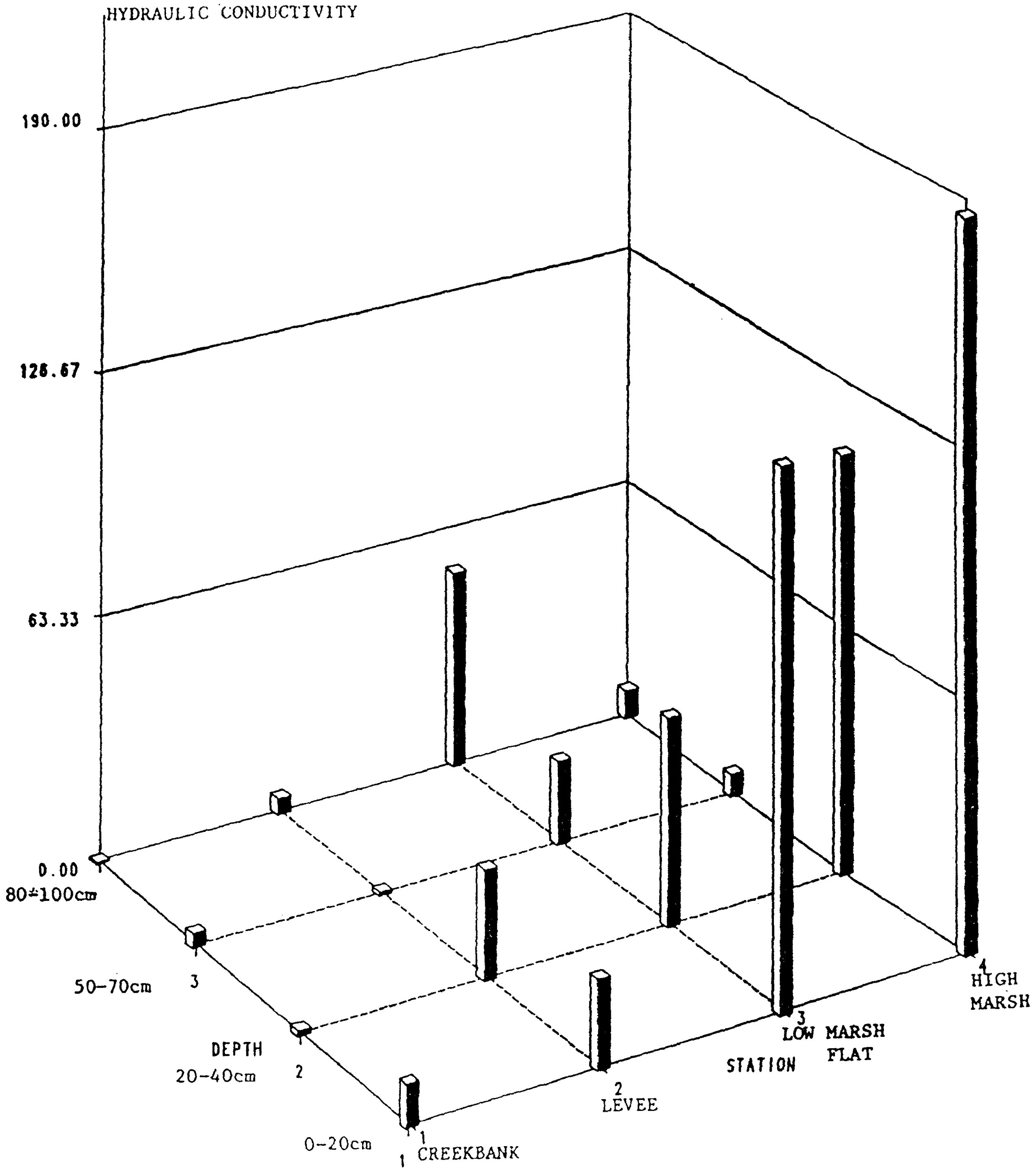

FACTOR I LOADING
FACTOR II LOADING
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0

Figure 3.5: Wetland soil characterization based on factor end member association.



END MEMBER #2
END MEMBER #1
elevation (cm)
120
100
80
60
40
20
0
-20
0.7 2.7 6.7 15 30 45 60 75 90 105 120
distance (m)
LEGEND
> 75% end member #1 association
> 50% and < 75% end member #1 association
> 75% end member #2 association
> 50% and < 75% end member #2 association

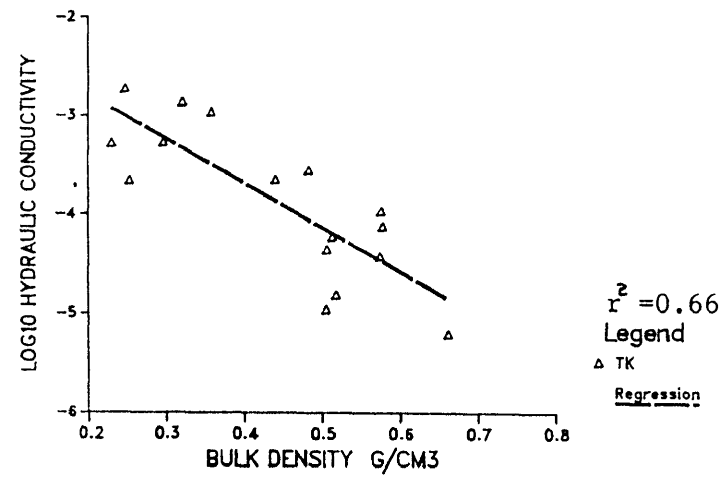
Figure 3.6: Spatial variations in soil horizontal hydraulic conductivity along wetland transect.


HYDRAULIC CONDUCTIVITY
190.00
126.67
63.33
0.00
80-100cm
50-70cm
3
DEPTH
20-40cm
2
0-20cm
1
1
CREEKBANK
2
LEVEE
STATION
3
LOW MARSH
FLAT
4
HIGH
MARSH

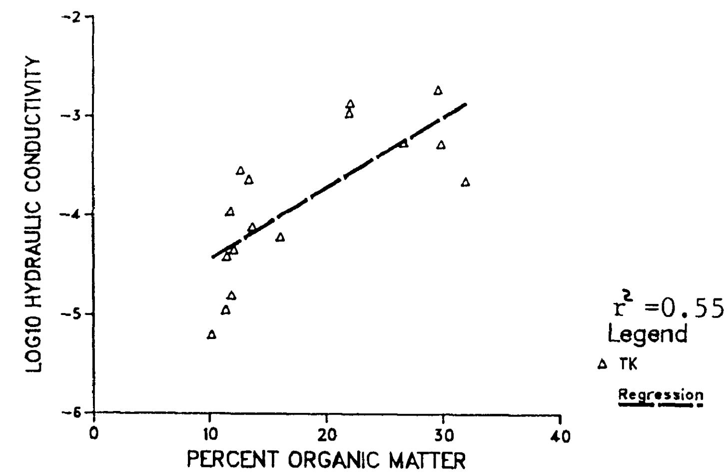
K*10-5 CM/SEC

Figure 3.7: Horizontal hydraulic conductivity
and soil parameter relationships.

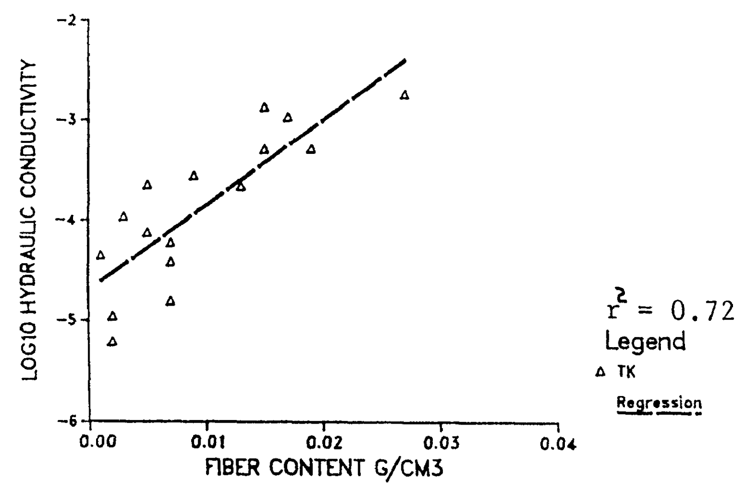
DRY BULK DENSITY



PERCENT ORGANIC MATTER

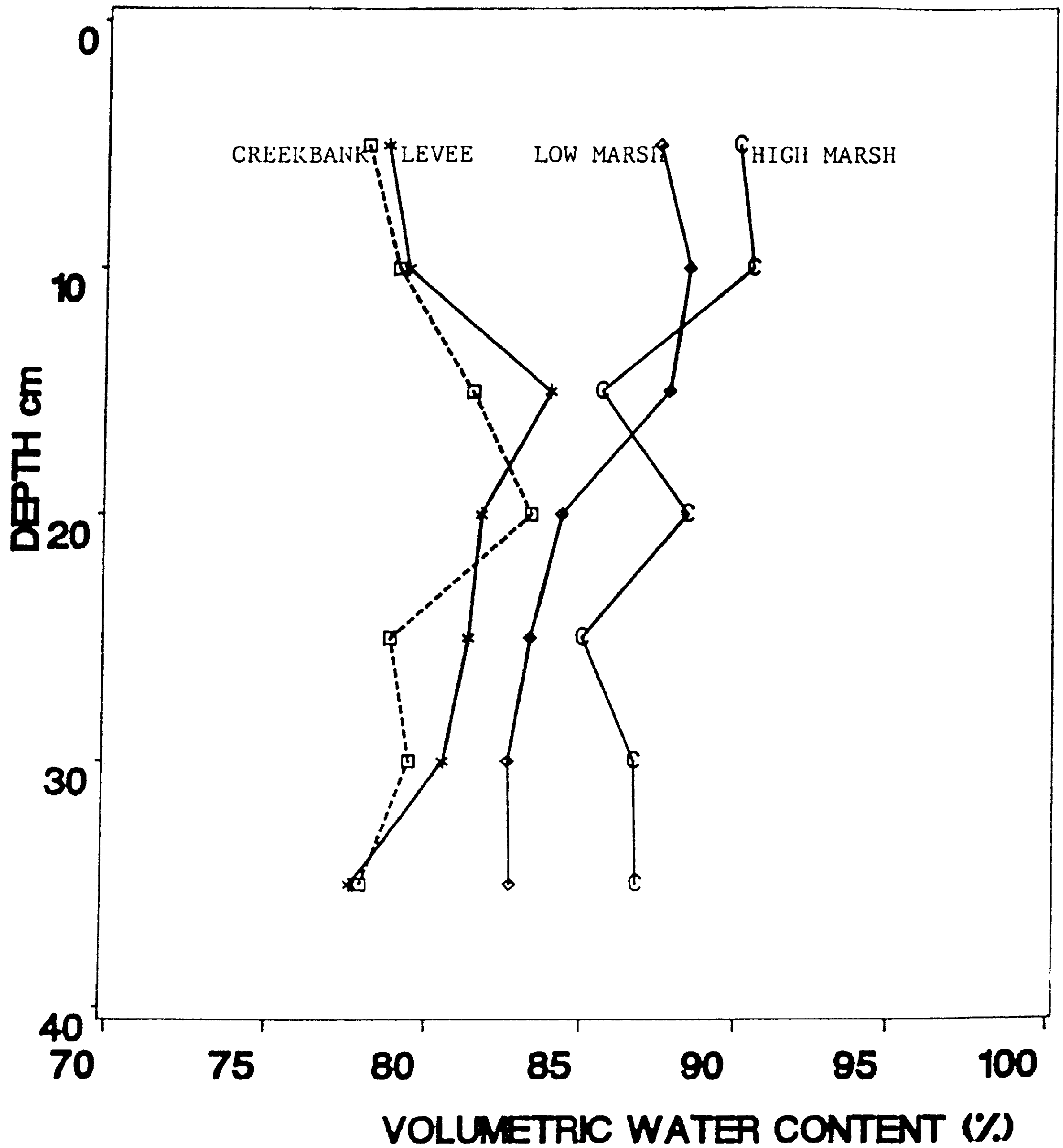


FIBER CONTENT



note: Hydraulic conductivity is
expressed in cm per second

Figure 3.8: Wetland soil volumetric water content.



CREEKBANK
LEVEE
LOW MARSH
HIGH MARSH
DEPTH cm
0
10
20
30
40
70
75
80
85
90
95
100
VOLUMETRIC WATER CONTENT (%)

Table 3.1: Grain size distribution of wetland soil along transect.

Station/Depth	% Sand	% silt	% clay
Creekbank/Levee			
surface-20cm	5.8	29.6	64.6
20-40cm	3.4	34.9	61.7
50-70cm	5.1	30.2	64.7
80-100cm	6.1	25.1	68.8
Low Marsh Flat			
surface-20cm	5.9	21.8	72.3
20-40cm	2.1	22.2	75.7
50-70cm	2.1	18.3	79.6
80-100cm	3.6	23.6	72.8
High Marsh/Upland Edge			
surface-20cm	56.4	13.8	29.8
20-40cm	30.5	18.1	51.4
50-70cm	27.1	18.4	54.5
80-100cm	65.3	12.1	22.6

Table 3.2: Descriptive statistics of soil dry bulk density along wetland transect.

DRY BULK DENSITY (G/CM3)

Station/Depth	Mean	Std.Err	N	Station/Depth	Mean	Std.Err	N
Station 1				Station 5			
surface-20cm	0.576	0.049	4	surface-20cm	0.321	0.028	3
20-40cm	0.518	0.030	4	20-40cm	0.296	0.031	3
50-70cm	0.574	0.028	3	50-70cm	0.253	0.011	3
80-100cm	0.662	0.025	3	80-100cm	0.230	0.023	3
Station 2				Station 6			
surface-20cm	0.530	0.016	3	surface-20cm	0.262	0.031	3
20-40cm	0.514	0.025	3	20-40cm	0.319	0.017	3
50-70cm	0.538	0.028	3	50-70cm	0.291	0.019	3
80-100cm	0.627	0.017	3	80-100cm	0.218	0.005	3
Station 3				Station 7			
surface-20cm	0.440	0.021	3	surface-20cm	0.202	0.014	3
20-40cm	0.483	0.006	3	20-40cm	0.264	0.016	3
50-70cm	0.505	0.048	4	50-70cm	0.433	0.029	3
80-100cm	0.506	0.016	3	80-100cm	0.610	0.067	3
Station 4				Station 8			
surface-20cm	0.462	0.026	4	surface-20cm	0.247	0.014	3
20-40cm	0.499	0.034	4	20-40cm	0.358	0.042	4
50-70cm	0.463	0.034	3	50-70cm	0.513	0.028	3
80-100cm	0.256	0.004	3	80-100cm	0.578	0.028	3
				For all samples	0.427	0.014	102

Table 3.3: Descriptive statistics of soil percent organic matter along wetland transect.

PERCENT ORGANIC MATTER

Station/Depth	Mean	Std.Err	N	Station/Depth	Mean	Std.Err	N
Station 1				Station 5			
surface-20cm	11.8	0.4	4	surface-20cm	22.1	1.3	3
20-40cm	11.9	0.1	4	20-40cm	26.7	0.7	3
50-70cm	11.5	0.7	3	50-70cm	32.0	1.3	3
80-100cm	10.2	0.3	3	80-100cm	29.9	1.2	3
Station 2				Station 6			
surface-20cm	12.5	0.5	3	surface-20cm	21.6	2.4	3
20-40cm	12.0	0.5	3	20-40cm	22.6	2.8	3
50-70cm	12.0	0.2	3	50-70cm	28.8	0.9	3
80-100cm	11.3	0.8	3	80-100cm	31.9	0.9	3
Station 3				Station 7			
surface-20cm	13.4	0.4	3	surface-20cm	29.4	1.0	3
20-40cm	12.7	1.1	3	20-40cm	26.4	2.5	3
50-70cm	11.4	0.5	4	50-70cm	14.6	0.4	3
80-100cm	12.1	0.2	3	80-100cm	10.8	0.7	3
Station 4				Station 8			
surface-20cm	14.6	0.6	4	surface-20cm	29.6	0.8	3
20-40cm	12.7	0.7	4	20-40cm	22.0	1.9	4
50-70cm	16.1	1.4	3	50-70cm	16.1	1.3	3
80-100cm	27.4	0.8	3	80-100cm	13.7	0.7	3
				For all samples	18.2	0.8	102

Table 3.4: Descriptive statistics of soil fiber content along wetland transect.

FIBER CONTENT (G/CM3)

Station/Depth	Mean	Std.Err	N
Station 1			
surface-20cm	0.003	0.001	4
20-40cm	0.007	0.002	4
50-70cm	0.007	0.004	3
80-100cm	0.002	0.001	3
Station 2			
surface-20cm	0.006	0.001	3
20-40cm	0.006	0.004	3
50-70cm	0.004	0.001	3
80-100cm	0.002	0.001	3
Station 3			
surface-20cm	0.005	0.001	3
20-40cm	0.009	0.003	3
50-70cm	0.002	0.001	4
80-100cm	0.001	0.001	3
Station 4			
surface-20cm	0.008	0.003	4
20-40cm	0.004	0.001	4
50-70cm	0.011	0.003	3
80-100cm	0.011	0.002	3

Station/Depth	Mean	Std.Err	N
Station 5			
surface-20cm	0.015	0.001	3
20-40cm	0.019	0.001	3
50-70cm	0.013	0.004	3
80-100cm	0.015	0.006	3
Station 6			
surface-20cm	0.014	0.005	3
20-40cm	0.023	0.007	3
50-70cm	0.017	0.006	3
80-100cm	0.019	0.002	3
Station 7			
surface-20cm	0.021	0.003	3
20-40cm	0.016	0.002	3
50-70cm	0.005	0.001	3
80-100cm	0.004	0.002	3
Station 8			
surface-20cm	0.027	0.002	3
20-40cm	0.017	0.005	4
50-70cm	0.007	0.003	3
80-100cm	0.005	0.002	3
All samples	0.010	0.001	102

Table 3.5: Eigenvalues, percent sums of squares, and cumulative sums of squares for the principal components of soil parameters along wetland transect.

VARIABLE	EIGENVALUE	% SUMS OF SQUARES	CUMULATIVE % SUMS OF SQUARES
Bulk Density	27.8826	87.13	87.13
% Organic Matter	3.8723	12.10	99.23
Fiber Content	0.2449	0.77	100.00

Table 3.6: Varimax rotated factor loadings and end member identification.

Station/Depth	Station Code #	Communality	Factor I Loading	Factor II Loading
1/0-20cm	1	0.999	0.9422	0.3336
1/20-40cm	2	0.995	0.8821	0.4653
1/50-70cm	3	0.992	0.9037	0.4187
1/80-100cm*	**4**	**1.000**	**0.9712**	**0.2381**
2/0-20cm	5	0.999	0.8944	0.4460
2/20-40cm	6	0.998	0.8931	0.4479
2/50-70cm	7	0.998	0.9014	0.4307
2/80-100cm	8	0.999	0.9610	0.2744
3/0-20cm	9	0.998	0.8592	0.5096
3/20-40cm	10	0.988	0.8296	0.5475
3/50-70cm	11	0.994	0.9374	0.3399
3/80-100cm	12	0.985	0.9367	0.3274
4/0-20cm	13	0.999	0.8156	0.5778
4/20-40cm	14	0.998	0.9044	0.4246
4/50-70cm	15	0.993	0.7548	0.6506
4/80-100cm	16	0.964	0.4440	0.8756
5/0-20cm	17	0.999	0.5126	0.8578
5/20-40cm	18	0.998	0.4119	0.9103
5/50-70cm	19	0.960	0.3931	0.8977
5/80-100cm	20	0.985	0.3611	0.9244
6/0-20cm	21	1.000	0.4593	0.8882
6/20-40cm	22	0.958	0.4074	0.8897
6/50-70cm	23	0.998	0.4169	0.9080
6/80-100cm	24	0.997	0.3081	0.9496
7/0-20cm*	**25**	**0.999**	**0.2815**	**0.9593**
7/20-40cm	26	0.999	0.4092	0.9119
7/50-70cm	27	0.994	0.8401	0.5372
7/80-100cm	28	0.999	0.9474	0.3191
8/0-20cm	29	0.980	0.2823	0.9487
8/20-40cm	30	0.991	0.5249	0.8458
8/50-70cm	31	1.000	0.8407	0.5411
8/80-100cm	32	1.000	0.9099	0.4146
		Variance	54.047	45.188
		Cum. Variance	54.047	99.234

* denotes factor end members

Table 3.7: Descriptive statistics of soil horizontal hydraulic conductivity along wetland transect.

Station/Depth interval	Mean	Std.Error	N
Creekbank			
0-20cm	$1.1x10^{-4}$	$3.7x10^{-5}$	3
20-40cm	$1.6x10^{-5}$	$4.8x10^{-6}$	3
50-70cm	$3.8x10^{-5}$	$3.1x10^{-5}$	3
80-100cm	$6.3x10^{-6}$	NA	1
Levee			
0-20cm	$2.3x10^{-4}$	$1.3x10^{-5}$	3
20-40cm	$2.8x10^{-4}$	$8.1x10^{-5}$	3
50-70cm	$1.1x10^{-5}$	$4.2x10^{-6}$	3
80-100cm	$4.5x10^{-5}$	$1.3x10^{-5}$	3
Low Marsh Flat			
0-20cm	$1.4x10^{-3}$	$3.6x10^{-4}$	3
20-40cm	$5.4x10^{-4}$	$2.0x10^{-4}$	3
50-70cm	$2.2x10^{-4}$	$6.4x10^{-5}$	3
80-100cm	$5.2x10^{-4}$	$2.5x10^{-4}$	3
High marsh/Upland edge			
0-20cm	$1.9x10^{-3}$	$7.5x10^{-4}$	3
20-40cm	$1.1x10^{-3}$	$6.9x10^{-4}$	3
50-70cm	$6.0x10^{-5}$	$2.3x10^{-5}$	3
80-100cm	$7.6x10^{-5}$	$3.6x10^{-5}$	3
For entire sample	$4.2x10^{-4}$	$4.0x10^{-4}$	46
For entire sample (without mineral soil at 50-70cm and 80-100cm depth interval at high marsh)	$4.9x10^{-4}$	$4.3x10^{-4}$	39

Table 3.8: Soil parameter correlation (r) matrix.

VARIABLE	LOG K	BULK DENSITY	% ORG. MATTER	FIBER
LOG K	1.00			
BULK DENSITY	-0.81	1.00		
% ORG. MATTER	0.74	-0.94	1.00	
FIBER CONTENT	0.83	-0.84	0.85	1.00

Table 3.9: Step-wise inclusion multiple regression of soil parameters with horizontal hydraulic conductivity.

STEP #1 VARIABLE ADDED: FIBER CONTENT

MULTIPLE R: 0.83
R SQUARE: 0.69

ANALYSIS OF VARIANCE	DF	MEAN SQUARE	F
REGRESSION	1	5.91	31.01
RESIDUAL	14	0.19	

VARIABLE VALUE
FIBER CONTENT: 85.33
(CONSTANT): -4.70

STEP #2 VARIABLE ADDED: BULK DENSITY

MULTIPLE R: 0.86
R SQUARE: 0.73

ANALYSIS OF VARIANCE	DF	MEAN SQUARE	F
REGRESSION	2	3.14	17.73
RESIDUAL	13	0.18	

VARIABLE VALUE
FIBER CONTENT: 52.31
BULK DENSITY: -2.10
(CONSTANT): -3.45

STEP #3 VARIABLE ADDED: % ORGANIC MATTER

MULTIPLE R: 0.87
R SQUARE: 0.76

ANALYSIS OF VARIANCE	DF	MEAN SQUARE	F
REGRESSION	3	2.17	12.52
RESIDUAL	12	0.17	

VARIABLE VALUE
FIBER CONTENT: 62.76
BULK DENSITY: -4.22
% ORG. MATTER: -0.05
(CONSTANT): -1.74

$$K(\text{cm/sec}) = 10^{(85.3*\text{fiber})-4.7}$$

$$K(\text{cm/sec}) = 10^{(62.7*\text{fiber})-(4.2*\text{bulk density})-(0.05*\%\text{organic matter})-1.7}$$

Table 3.10: Literature cited field measurements of soil hydraulic conductivity within tidal wetlands.

Reference	Location	Method	Hydraulic Conductivity (cm/sec)
Harvey(1986)	Brackish Salt Marsh, Virginia	seepage tube	7.4×10^{-4} (mean value)
Yelverton & Hackney (1985)	Salt Marsh, Maryland	auger hole	3.5×10^{-4} (mean value)
Jordan & Correll (1985)	Brackish Salt, Marsh, Maryland	auger hole	7.2×10^{-3} ($4.2 * 10^{-3}$ S.D.)
Knott et al(1987)	Salt Marsh, Massachusetts	permeameter/ seepage tube	7.8×10^{-4} (mean value)
Gardner(1976)	Salt Marsh, South Carolina	permeameter/ seepage tube/ auger hole	3.3×10^{-5} (mean value)
Reay (this study)	Tidal Fresh-Water Marsh, Virginia	auger hole/ seepage tube	4.9×10^{-4} (4.3×10^{-4} S.E)

IV. WETLANDS SUBSURFACE HYDROLOGY

Introduction

Wetland systems may be considered as phreatic aquifers where the water table, delineating the zone of saturation, is directly influenced by the atmosphere. The water table is therefore a dynamic surface where the hydrostatic pressure varies with and equals atmospheric pressure. Transient water influx and efflux within a wetland system is responsible for water table fluctuations, variations in water storage characteristics, and changes in the relative importance of water transport mechanisms (Ingram, 1983).

Soil water removal processes operating within a tidally influenced wetland include evapotranspiration and subsurface drainage. Evapotranspiration is a combination of direct evaporation of water within a soil matrix and its associated litter layer, and evaporation of water from wetland vegetation. Several studies of east coast tidal wetlands have quantified the effect of evapotranspiration on soil water budgets (Hemond and Fifield, 1982; Dacey and Howes, 1984). Seepage is the transport of interstitial water due primarily to differences in hydraulic pressure and elevation between regions of interest. The relative importance and magnitude of effluent seepage from tidal wetlands to adjacent soil regions or surface waters is quite variable (Gardner, 1976; Jordan and Correll, 1985; Agosta, 1985; Yelverton and Hackney, 1986; Harvey et. al., 1987).

Interstitial water lost by evapotranspiration or effluent seepage can be replaced through a variety of mechanisms. Influent seepage, from adjacent surface waters (Jordan and Correll, 1985; Yelverton and Hackney, 1986; Harvey et. al., 1987) and/or upland regions via groundwater recharge (Valiela et. al., 1978; Carr and Blackley (b), 1986) has been shown to influence water table elevations and water storage within tidal wetlands. Meteorologic sources of water are variable, but when coincident with marsh surface exposure, affect pore water pressures and water table behavior (Carr and Blackley (b), 1986). Soil water may also be replenished by diffuse surface flow followed by vertical infiltration (Hemond and Burke, 1981; Hemond et. al., 1984; Jordan and Correll, 1985; Yelverton and Hackney, 1986; Carr and Blackley (a) and (b), 1986; Harvey et. al., 1987). Sources of diffuse surface flow include inundating tidal waters and runoff from adjacent upland regions during periods of high rainfall.

Determination of subsurface hydraulics in conjunction with soil hydrophysical properties is required to calculate the exchange of interstitial water within a wetland system. With a knowledge of wetland geohydrologic properties, the application of theoretical equations of flow in porous media is possible. Fluid dynamic models may be prepared to simulate subsurface hydraulics within a defined system, this would allow for predictive ability and provides a tool to further our understanding of such systems.

4.1 Elementary Theory of Subsurface Flow

Water moves through a soil matrix in response to variations in energy. The total energy of a moving unit mass of water, at a particular point, is described by Bernoulli's equation (4.1).

$$h = (v^2/2g) + (p/w) + z \qquad \text{(eq. 4.1)}$$

Where:

p = pressure (mass/length^2)

w = specific weight of water (length^3/mass)

v = velocity of flow (length/time)

g = gravitational acceleration (length/time^2)

z = elevation above a reference level (length)

h = hydraulic head (length)

Since water moves through aquifers at relatively low flow velocities, the kinetic energy component ($v^2/2g$) equation may be neglected. This simplifies the definition of hydraulic head so that it may be described by pressure (p/w) and elevation (z) components. Water flow through a porous medium occurs in the direction of decreasing head (h). The change in hydraulic head with distance is known as the hydraulic gradient (dh/dl). Reactions between the water and soil medium, and frictional drag of water particles moving past each other and through the porous material, result in a resistance to water movement through a soil (Ingram, 1983). This resistance is responsible for the decrease in hydraulic head along a flow path.

Darcy's law (eq. 4.2) provides the basic empirical foundation for laminar flow through a cross-sectional area of porous media.

$$Q = -KA(dh/dl) \qquad \text{(eq. 4)}$$

Where:

Q = Discharge (length^3/time)

A = Cross-sectional area perpendicular to direction of flow (length^2)

K = Hydraulic conductivity (length/time)

dh/dl = Hydraulic gradient (dimensionless)

Darcy's law indicates that discharge (Q) is proportional to the cross-sectional area (A), permeability of the porous media (K), and energy loss (dh) along the flow path, and inversely proportional to the length of the flow path. The negative sign indicates that discharge is in the direction of decreasing hydraulic head. Darcy's law is valid and applicable where viscous forces dominate inertial forces, such flows may be described as laminar. As flow rate velocities increase, the relationship between specific discharge (Q/A) and hydraulic gradient (dh/dl) may no longer remain linear, and Darcy's law is inapplicable. Because of the low velocities involved, most subsurface flow of natural water may be described by Darcy's law.

The fundamental equation describing steady state two dimensional water flow within a homogeneous (aquifer properties constant from point to point) and isotropic (aquifer properties constant in all flow directions at a single point) aquifer, can be derived by combining Darcy's law with the principle of mass conservation (eq. 4.3). K and h are as previously defined, and x and z are horizontal and vertical flow components respectively.

$$K\,(d^2h/dx^2 + d^2h/dz^2) = 0 \qquad \text{(eq. 4.3)}$$

In order to describe time-variant flow, it is necessary to incorporate an aquifer storage coefficient which is equal to specific yield (Sy) within an unconfined aquifer (eq. 4.4).

$$d^2h/dx^2 + d^2h/dz^2 = (Sy/K)(dh/dt) \qquad \text{(eq. 4.4)}$$

Analytical solutions to equation (4.4) are difficult due to the time variation of the water table boundary condition. By use of Dupuit's approximation, simplification to a one dimensional equation describing horizontal flow is possible (eq. 4.5).

$$K\,(d^2h^2/dx^2) = 2(Sy)(dh/dt) \qquad \text{(eq. 4.5)}$$

Application of Dupuit's approximation requires that simplifying assumptions be made regarding the flow field. Assumptions include: 1) flow is horizontal, 2) velocity is uniform with depth, and 3) hydraulic gradients are small and equal to the slope of the water table. The Dupuit approximation should be used with caution in regions with a vertical flow component . Such regions may be characterized by large hydraulic gradients and/or vertical variations in soil permeability that would produce depth variations in flow velocity.

4.2 Experimental Methods

Hydraulic head and watertable elevations were monitored over various tidal cycles by utilization of multi-level piezometer and well arrays. Piezometer sampling depth intervals were 20-40 and 80-100 centimeters. The well cavity extended from 20 to 100 centimeters below the soil surface. Instruments were located at the same locations along the transect used in the soil characterization study. Piezometers were constructed of metal pipe (3cm i.d.) with a 20 centimenter long mesh opening forming the cavity. Wells were of similar construction but with a 80 centimeter cavity opening. Piezometers and wells were installed by augering out a thin walled casing (3.5cm i.d.) to the appropriate depth and inserting the instrument into the casing followed by removal of the casing. Rubber seals were fixed above the cavity and immediatly below the marsh surface to eliminate water migration down the side of the instrument.

Following installation, piezometers and wells were purged several times to remove debris and ensure adequate recharge. Piezometers, wells, tidal gauge, and marsh transect profile elevations were surveyed and referenced to a common point. Measurements of hydraulic head were accomplished manually by lowering a millimeter scale into the piezometer/well and recording the depth at which an electrical circuit was completed by contact with the water surface. Tidal changes within the thoroughfare were monitored by a semi-continous recording tidal gauge located at the study site.

The direct measurement of interstitial water seepage flow from the subaqueous portion of the creekbank to the adjacent surface water was determined using seepage meters designed according to Lee (1977) with modifications by Bokuniewicz and Zeitin (1980).

Seepage meters were positioned just below mean low water in close proximity to the creekbank nutrient monitoring wells. Seepage meters were deployed over complete tidal cycles and macroscopic seepage rates were calculated according to Lee (1977).

4.3 Subsurface Hydrologic Model Description

A hydrologic model was developed to simulate hydraulic head fluctuations and horizontal flow within a wetland system over specified tidal conditions. The simulated system incorporated surface elevations of the 118 meter transect used in the soil characterization and subsurface hydraulics study, and field values of specific yield (Sy) and soil hydraulic conductivity (K). Soil hydraulic conductivity values were spatially assigned according to the soil classification system developed by factor analysis (Fig. 3.5). Permeability estimates from the 0-20cm depth intervals were used since this represented the region where the most mobile interstital water occured.

The bottom boundary of the flow system was assumed to be one meter in depth, this corresponded to low permeabilty soils identified during field investigations. Approximately a two order of magnitude difference in mean hydraulic conductivity measurements between the 0-20cm and 80-100cm depth intervals occured at the creekbank (1.1×10^{-4}cm/sec vs. 6.3×10^{-6}cm/sec) and high marsh regions (1.9×10^{-3}cm/sec vs. 7.6×10^{-5}cm/sec). Vertical variations in K within the levee region approximated a one order of magnitude difference between the 0-20cm and 80-100cm depth intervals (2.3×10^{-4}cm/sec vs. 4.5×10^{-5}cm/sec). The low marsh flat did not demonstrate significant vertical variations in horizontal hydraulic conductivity. However, given the extremely low hydraulic gradients displayed within this region, the effect of aquifer depth on discharge estimates was minimized. The right boundary was set at the high marsh/upland interface where the sand-clay sediment exibited low mean hydraulic conductivity values (7.6 x 10-

5cm/sec). Both the bottom and right boundaries were expressed as no flow boundaries. Wang and Anderson (1982) provided a mathematical solution for these boundary conditions. The left boundary, located at the creekbank, was time variable and followed tidal cycle fluctuations. The water table, representing the final boundary, was time variable and determined by the model.

Several assumptions were made in order to simplify modeling efforts. Hydraulic head, at a particular nodal point along the transect, was assumed to equal tidal elevation if that nodal point was inundated by tidal waters. Hydraulic head was equal to the wetland surface elevation immediately following surface exposure or when surface water ponding existed at a particular nodal point; in both cases the soil was assumed to be saturated. Simulated horizontal flow did not occur when the entire wetland was inundated by tidal waters, since hydraulic head was assumed equal throughout the entire wetland. The model assumed instantaneous soil water recharge and discharge when a particular nodal point was reinundated or exposed by surface waters respectively. Field observations show that the above simplifying assumptions are realistic. Simulation model time steps were set at six minute intervals.

Simulated hydraulic head estimates were determined by a finite difference solution of the governing equation (eq. 4.5). An implicit form of the approximation, utilizing Gauss-Seidel iteration, was chosen due to its ability to remain stable given a wide range of model time step intervals (Remson et. al., 1971; Wang and Anderson, 1982). Horizontal discharge estimates to and from adjacent hydraulic cells were determined by integration of the governing equation with respect to x and substitution of appropriate boundary conditions (eq. 4.6) (Davis and DeWeist, 1966; Bear, 1972). Q and K are as previously defined, h_0 and h_1 are hydraulic head measurements at two adjacent nodal points, and L is the distance between the two points.

$$Q = (K/2L)(h_0^2-h_1^2) \qquad \text{(eq 4.6)}$$

Discharge through vertically stratified hydraulic cells, which were used to separate regions of various soil characteristics as identified by factor analysis (Fig. 3.5), were determined by equation 4.7 (Bear, 1972). K' and K" are the hydraulic conductivities of the respective subunits, and L' and L" are the horizontal lengths of each subunit within the vertically stratified hydraulic cell.

$$Q = (h_0^2 - h_1^2)/[(L'/K') + (L''/K'')] \qquad \text{(eq. 4.7)}$$

Evaluation of model performance was achieved by comparison of field collected and model simulated spatial and temporal variations of hydraulic head. Difference indices, mean absolute error (MAE) and root mean square error (RMSE) provided estimates of the average magnitude of error between simulated and field measured results (Willmott, 1982; Willmott et. al., 1985). Both MAE and RMSE are in length units similar to hydraulic head measurements. RMSE should be regarded as a high estimate of actual error since it is relatively more effected by outlying observations than MAE.

Sensitivity analysis was employed to assess the relative response of discharge over a tidal cycle to variations in the following aquifer characteristics: 1) hydraulic conductivity, 2) specific yield, and 3) aquifer depth. Soil and aquifer depth parameters were normalized in order to allow for comparison. Normalization was accomplished by dividing each parameter value by the mean field value. Parameters were varied over a range reflecting realistic values determined from field observations and review of literature. Sensitivity analysis was utilized for creekbank and high marsh, where large differences in soil properties, surface elevations and hydraulic gradients existed between the two regions.

4.4 Field Results

Variations in hydraulic head and watertable elevations were monitored over tidal cycles on December 1, 1986, and March 3, 6, and 7 of 1987. March 3, 1987 watertable fluctuations for specified locations within the upper one meter of soil are given in Figures 4.1 through 4.4. The creekbank site, located 70 centimeters from the creekbank edge, displayed the greatest variation in hydraulic head and watertable elevation over a tidal cycle and with depth (Fig. 4.1). The decline in watertable elevation, as measured by the well, and hydraulic head within the 20-40cm depth interval of the soil, demonstrated similar patterns and magnitudes. Water level drawdown within the well and the 20-40cm piezometer were 22.9 and 23.7 centimeters respectively. Similar water level response by the well and 20-40cm piezometer indicates that the majority of horizontal flow is occuring within the upper 40 centimeters of the soil profile. Water level elevations in the 80-100cm piezometer were relatively unresponsive and always lower than hydraulic head measurements at the 20-40cm depth interval. Lower hydraulic head with depth indicates vertical flow from upper to lower soil levels. However, given the unresponsive nature of the deep piezometer and low soil permeability at this depth interval (6.3×10^{-6}cm/sec), such vertical flow is believed to be minimal. The rapid increase in water table elevation, following inundation by tidal waters, signifies that vertical infiltration was a primary mechanism of pore water recharge within the creekbank region. Saturation of the soil was usually complete within 20 minutes of tidal water inundation.

Piezometers and the well responded in a similar fashion at the levee crest station (Fig. 4.2), despite significant variations in soil permeability with depth. Within two hours of exposure by surface waters, water table and hydraulic head elevations declined sharply to approximately 20 centimeters below the surface. It should be noticed that little time lag existed between the decline of surface water and interstitial water within this region. Following the period of rapid decline, the water table and hydraulic head elevations stabilized to a slow steady decline throughout the majority of the tidal cycle. Horizontal recharge of interstitial water was evident by the rapid rise in water table and hydraulic heads in the latter part of the tidal cycle. The levee region was not inundated during this particular tidal cycle, however, field measurements during early and latter parts of the cycle indicate that hydraulic heads and water table closely followed adjacent surface water elevations. This pattern is an indication of increased soil permeability due to channelization of flow (i.e. pipe flow) within the upper 20 centimeters. Tunneling and burrowing activities of *Ondatra zibethica* were found at station 2, which was approximately 4 meters creekward of the levee crest station. Water table and hydraulic heads at station 2 followed surface water elevations throughout the monitored tidal cycles. Excavation of this site following the study, revealed a subsurface tunnel (approximately 20cm in diameter) in close vicinity to station 2. Such tunneling activity allows for a free connection of surface waters to marsh creekbank, levee, and interior regions, thus influencing hydraulic gradients and the exchange of material within these regions.

The low marsh region displayed minimum variation in hydraulic head with depth, and was characterized by a saturated soil condition, ponding of surface waters, and extremely low hydraulic gradients (Fig. 4.3). Following 48 hours of exposure, the water table remained within 1.5 centimeters of the soil surface. Surface ponding gradually decreased throughout the tidal cycle due to evapotranspiration, vertical infiltration, and/or surface runoff. Recharge of low marsh flat interstitial water to the levee and high marsh regions was minimal or non-existant. Hydraulic gradients throughout the high marsh and a large majority of the back-levee region indicated water movement towards the marsh interior.

The high marsh station, located near the upland interface, displayed significant temporal and spatial variations in hydraulic heads and watertable (Fig. 4.4). Following surface exposure, the shallow piezometer and well were characterized by a slow steady decline, which approximated 5.5cm over a tidal cycle. There was only an additional decline of 2 centimeters following an exposure period of 48 hours. Horizontal recharge of interstitial water was evident just prior to tidal inundation, this was expected given the relatively high soil permeabilities of the region. Hydraulic head at the 80-100cm depth interval was elevated in relationship to the well and shallow piezometer. This hydraulic head pattern suggests groundwater recharge from the adjacent uplands to the wetland. Soil permeability of the sand/silt lens at the 80-100cm depth interval (7.6 x 10^{-5}cm/sec) was significantly less than the overlying 0-20cm and 20-40cm depth intervals (1.9 x 10^{-3} and 1.1 x 10^{-3} cm/sec respectively). This lower permeability could be partially responsible for the unresponsive nature of the deep piezometer.

Direct estimates of seepage from the creekbank to adjacent surface waters were made over two tidal cycles. Seepage was monitored over a stormtide and spring tide event in order to compare extreme conditions. The stormtide event, December 1, 1986, had a high water elevation of 144.4cm above the reference point. The marsh levee was exposed approximately 4.5 hours while the creekbank remained inundated throughout the tidal cycle. The spring tide sampling period, May 3, 1987, had a reference high water elevation of 113.8cm with the levee and creekbank regions being exposed for approximately 11.0 and 6.0 hours respectively. Due to low hydraulic conductivity estimates along the creekbank, seepage meters were sampled only once over a complete tidal period. Mean cumulative discharges for the storm event and spring tide condition were 0.75 (S.E.=0.11, N=3) and 1.42 (S.E.=0.07, N=3) liters/meter^2/tidal cycle respectively. The stormtide discharge should be considered a minimum since marsh surface exposure by tidal water and hydraulic gradients within the creekbank region were kept minimal. This preliminary data indicates an inverse relationship between seepage discharge and water surface elevation as reported by Lee (1977).

4.5 Model Results

Simulated creekbank water table draw-down and associated time lag from tidal surface waters were in good agreement with field observations (MAE =1.1cm; RMSE =1.5cm) (Fig. 4.5a). Simulated water table response late in the tidal cycle conforms to the idea that vertical infiltration is the dominant recharge mechanism within the creekbank region. This was demonstrated by the rapid increase in water table elevation as the site was inundated by surface waters. Disparity existed between the predicted and field measured water table elevation at the levee crest (MAE =16.0cm; RMSE =17.6cm) (Fig. 4.5b). As previously discussed, this region was influenced by pipe flow conditions which were not accounted for by the model.

Model performance at the back-levee site was in general agreement (MAE =2.6cm; RMSE =2.8cm) (Fig. 4.6a) with field observations. Since this region occured within a transitional zone from levee (K= 2.3 x 10^{-4}cm/sec) to low marsh flat (1.4 x 10^{-3}cm/sec), under-estimation of water table draw-down (7.7cm versus 5cm) may have occured due to model usage of the lower soil permeability estimates within this region. Field data indicates horizontal recharge towards the end of the tidal cycle, whereas model simulation shows a slow steady drop in water table. Increased horizontal interstitial water recharge is an indication of a more permeable soil.

There was good agreement between model predicted water table elevation and observed elevations within the low marsh flat region (MAE =1.0cm; RMSE =1.4cm) (Fig. 4.6b). Most deviation occured early in the tidal cycle when model code provided for immediate response of the water table with surface waters if the site is inundated. Water table elevations within this region indicated ponding of surface and drained interstitial water from upland and back-levee sites. Discharge from the high marsh and back levee regions, to the low marsh flat, maintained ponding and 100 percent water saturation of the soil throughout the majority of the tidal cycle. Model simulation provided excellent agreement within the high marsh (MAE =0.4cm; RMSE =0.5cm) (Fig. 4.7); both the pattern and extent of water table draw-down were accurately predicted. It should be noted that recharge from upland groundwater sources were not incorporated into the model.

Discharge curves for a simulated tidal cycle (March 3, 1987) show great spatial and temporal variation (Fig. 4.8a-d). Creekbank seepage rates (Fig. 4.8a) increased early in the tidal cycle and reached a maximum of approximately 700ml/hr. The increase in discharge was a result of increased hydraulic gradients due to ebbing surface waters. As creekbank interstitial water was transported to surface water due to horizontal effluent seepage, and the ebbing of surface water ceased, hydraulic gradients and discharge rates began to decay. Horizontal porewater movement was terminated when the site was inundated by surface waters. Creekbank discharge lasted approximately six hours over the simulated tidal cycle. It is important to recognize that discharge patterns, rates, and time periods will vary depending on tide amplitude and symetry. Discharge at the levee crest (Fig. 4.8b) was characterized by an initial low rate that increased throughout the tidal cycle to a maximum of 500ml/hr. Discharge was terminated abruptly due to inundation of surface waters. Horizontal discharge, from the levee crest towards the creekbank, continued for approximately 10 hours. The low marsh flat region (Fig. 4.8c) displayed similar patterns as the levee crest, however, maximum horizontal disharge rates (20ml/hr) were greatly reduced due to low hydraulic gradients. The simulated tidal discharge curve for the high marsh region (Fig. 4.8d) showed an initial maximum seepage rate of 800ml/hr followed by a gradual decline due to decay in hydraulic gradient. High marsh discharge continued for approximately ten hours over the simulated tidal cycle.

Interstitial water budgets were constructed utilizing model results for the creekbank, levee, low marsh flat, and high marsh regions of the transect. Effluent (discharge) and influent (recharge) seepage were calculated by the model, whereas, vertical infiltration was determined by difference to complete mass balance of water. Horizontal flows and vertical infiltration are expressed as mean values determined from three consequtive tidal cycles that inundated the entire marsh transect during the study period. *Ondatra zibethica* influence was incorporated in the model by having a nodal point 2.7 meters from the creekbank follow surface water elevations. A model comparison was made as to the effect of this large scale bioturbation to a creekbank and levee region that does not experience such tunneling activity.

Interstitial water movements and budgets for the creekbank region, with and without the tunneling activity of *Ondatra zibethica*, are given in Figures 4.9 and 4.10 respectively. Tunneling activity allows for bi-directional drainage within the creekbank region (0.7-2.7 meters), whereas, a uni-directional drainage pattern occurs in the absence of such large scale bioturbation. Horizontal discharge within 0.7 meters of the creekbank edge towards surface water was greater without interior tunneling activity (4.18 versus 2.47 liters/meter/tidal cycle) due to the sustainance of a steeper hydraulic gradient within the region. Water table elevations at the 1.7 meter nodal point did not decrease as rapidly or with the same magnitude as if under the influence of bi-directional flow. Without the influence of tunneling activity, discharge decreases rapidly as one move towards the marsh interior. There was over a four fold decrease in cumulative discharge across the vertical faces from the 0.7 (4.18 liter/meter/tidal cycle) to the 2.7 meter nodal point (0.90 liter/meter/tidal cycle). This decrease in subsurface seepage is due to the decrease in water table draw-down and hydraulic gradient as one moves away from the creekbank edge. With the simulated tunneling activity, horizontal discharge from the adjacent interior cell (1.7 meter node point) towards the creekbank was minimized and recharge of the creekbank was dominated by vertical infiltration. In the absence of tunneling activity, horizontal influent from the interior cell towards the creekbank was the dominant mechanism of pore water recharge to the region. However, vertical recharge becomes the primary mechanism of recharge within several meters of the creekbank when flow from adjacent interior regions diminished. Horizontal recharge of creekwater through the creekbank towards more interior marsh regions was minor. This was due to the combination of low soil permeabilities and sustenance of hydraulic gradient from the more interior regions toward the creekbank during the majority of the simulated tidal cycles.

Interstitial water movements and budgets for the levee region (Figs. 4.9 and 4.10) indicate that horizontal discharge occurs towards the creekbank and low marsh interior. Flow direction generally followed surface elevational profiles. The effect of muskrat tunneling activity within the creekbank region acted to increase the magnitude of flow and the spatial range of the levee region that drains toward the creekbank. Vertical infiltration was the primary mechanism of pore water replacement within the majority of the levee region. Simulated horizontal recharge to the levee from creekbank regions was minimal.

The low marsh region was characterized by low horizontal water movement on the order of 0.07 to 0.10 liters/meter/tidal cycle (Fig. 4.11a). The low marsh flat region had a potential for relatively large horizontal flows due to high soil permeabilites ($K= 1.4 \times 10^{-3}$cm/sec), but low topographic relief and hydraulic gradients created the low flow conditions. An upward vertical component (0.01-0.02 liters/tidal cycle) was calculated in order to assure mass continuity over the inundating tidal cycles. This upward component indicates a surplus of water within this region, beyond the point of soil saturation. Low marsh regions in close proximity to the high marsh, received surplus water from high marsh regions via subsurface drainage. Model results correspond to field observations of surface water ponding within the majority of the low marsh flat region. Model simulation indicated no horizontal or vertical recharge within this region due to its soil remaining saturated with negligable hydraulic gradients throughout the tidal cycles.

Model results indicate the high marsh to be a region of substantial horizontal discharge (Fig. 4.11b). Mean horizontal discharge over a tidal cycle increases as one moves towards the marsh interior, while horizontal and vertical recharge decrease. Although hydraulic gradients in the landward high marsh cells decay rapidly, flow from these cells sustain the hydraulic gradients in the adjacent more interior cells. This process, in conjunction with highly permeable soils, was responsible for the increased discharge of the interior high marsh cells. Although total horizontal discharge increases as one moves from the upland/high marsh interface towards the marsh interior, net discharge decreases. Therefore, more water is being removed and not replaced by horizontal flow as one moves landward in the high marsh region. Horizontal discharge from adjacent more landward regions and vertical infiltration of inundating surface water are the predominant mechanisms for soil water recharge within the high marsh region. The importance of vertical infiltration increases as one moves landward in the high marsh region. Although this region exhibited relatively high soil permeability, horizontal recharge prior to tidal inundation was minimal due to the low topographic relief. Low topographic relief results in adjacent nodal points being inundated within a short time period of each other, thus effectively reducing the amount of time available for horizontal recharge. It should be noted that on tidal cycles where the maximum tidal height is in close proximity to surface elevations of the high marsh region, more time will be allowed for horizontal recharge to occur.

Sensitivity analysis within the creekbank region indicates that aquifer depth exhibited the largest influence on interstitial water discharge, followed by hydraulic conductivity and specific yield properties of the aquifer respectively (Fig. 4.12). Inverted results, as those found within the creekbank region, were obtained for the high marsh region (Fig. 4.13).

4.6 Summary and Discussion

Examination of vertical and horizontal heads and watertable fluctuations along a transect from the creekbank edge to the high marsh/upland interface, suggest that an extensive tidal freshwater marsh does not respond as a single hydraulic unit during periods of aerial exposure. It is better described as three distinct yet interactive hydraulic cell types.

The most dynamic and tidally influenced cell is located along the narrow creekbank margin and is characterized by relatively large, rapid oscillations of the water table and hydraulic gradients. The lower portion of the back-levee and interior low marsh cell is distinguished by minimal water table movement and low hydraulic gradients; this region appears to be relatively stagnant with respect to horizontal water movement. The third general hydraulic cell, located within the high marsh region, displayed water table fluctuations and hydraulic gradients that were intermediate between the creekbank and back-levee/low marsh flat regions. Hydraulic evidence suggests shallow groundwater recharge to the high marsh region from the adjacent uplands. These three general hydraulic cells or regions are similar to those reported by Gardner (1973).

Vertical variations in hydraulic head were small as compared to lateral differences throughout the majority of the marsh transect, indicating that the majority of subsurface flow is horizontal. Movement of interstitial water was concentrated within the upper 40 centimeters of the wetland soil profile, this concurs with investigation by Jordan and Correll (1985), and Harvey (1986). The direction of horizontal subsurface flow after aerial exposure, generally followed that of the surface elevation profiles.

When modeled as an unconfined aquifer and incorporating marsh surface elevations and hydrophysical soil properties, simulated hydraulic heads along the transect were in good agreement with field observations over space and time (Figs. 4.5 to 4.7). Exception occured within the levee region where *Ondatra zibethica* tunneling activity was responsible for pipe flow conditions which were not incorporated into the model.

The export of interstitial water from immediate creekbank regions towards adjacent surface waters via horizontal seepage has been investigated in a variety of salt and brackish tidal marshes, but review of literature found no such studies within tidal freshwater wetlands. Given the wide variety of surface elevational profiles, aquifer depths, soil hydrophysical properties, and tidal ranges, there is general agreement of cumulative flux estimates per meter of marsh creekbank over a tidal cycle. Gardner (1976) reported 6.91

liters, Jordan and Correll (1985) estimated 43 liters across the creekbank face and 16 liters across a vertical face 0.5 meters interior to the creekbank, Yelverton and Hackney (1986) and Harvey et. al. (1987) estimated 39.6 and 11 liters respectively. Mean cumulative discharge for this study, through a one meter wide vertical face 0.7 meters interior to the creekbank, was 4.18 liters per tidal cycle. This value appears low compared to previous studies, however, hydraulic conductivity estimates used in the previously mentioned studies, except Gardner (1976), were several times greater. Direct estimates of horizontal seepage per tidal cycle (1.42 liters/m^2) are in approximate agreement to model predicted estimates (2.47 liters and 4.18 liters with and without *Ondatra zibethica* tunneling activity respectively) under similar tidal conditions. Discrepency between measured and predicted values may be due to lower soil permeability (6.3 x 10^{-6}cm/sec) at the depth the seepage meters were deployed; model simulations within this region used a hydraulic conductivity value of 1.1 x 10^{-4}cm/sec.

Recharge of displaced interstitial water within the creekbank occurs from a combination of vertical infiltration and horizontal discharge from adjacent interior regions. Horizontal recharge through the creekbank by surface waters was minimal. Although characterized by the lowest soil permeabilities, the creekbank region had the greatest potential for water and solute exchange with surface waters. This was primarily due to steep surface elevational profiles and tidal action facilitating the formation of relatively steep hydraulic gradients during periods of aerial exposure.

Comprising the majority of the transect, the lower portion of the back-levee and interior low marsh flat displayed very little horizontal transport of interstitial water (Fig. 4.11a), although there existed a potential for relatively rapid horizontal water movement due to relatively high soil permeabilities. Isolation from tidal influence, low hydraulic gradients, and ponding of surface water were responsible for the low horizontal flux estimates. Discharge from the back-levee and high marsh regions helped maintain interior region soils in a saturated condition. Hemond and Fifield (1982) reported that horizontal seepage played a minor role in pore water movement within interior portions of a New England salt marsh. Vertical movement in response to evaporation demands dominated flow within the region. This region appears to have a low potential for exchange of water and solutes with surface waters due to seepage and recharge mechanisms. The ability to receive solute input from inundating waters would increase during the growing season when evapotranspiration potential increases (Dacey and Howes, 1984), and following surface exposure for time periods greater than a tidal cycle.

Moderate hydraulic gradients in conjunction with high permeable soils were responsible for substantial horizontal effluent seepage within the high marsh region (Fig. 4.11b). Observations of hydraulic head (Fig. 4.4) indicate a potential recharge of upland groundwater to this region. Groundwater input would aid in the stabilization of the water table during periods of exposure. The magnitude of groundwater input within the upper meter of the wetland soil is believed to be limited by the low soil permeability (7.6×10^{-5}cm/sec) within the recharge zone. Vertical infiltration by inundating surface waters, and horizontal influent seepage from adjacent more landward regions were the primary mechanisms of soil water recharge within the high marsh. Importance of vertical infiltration decreased as one approached the low marsh flat.

Model sensitivity analysis indicates that creekbank and high marsh discharges increase as the hydraulic conductivity, specific yield, and depth of aquifer increase. However, the relative importance of each parameter differs between the two regions. Aquifer depth exerted maximum observed influence on creekbank discharge, whereas its relative effect in the high marsh region appeared minimal. Due to the form of the discharge equation (eq. 4.6), regions displaying large hydraulic gradients, such as the creekbank region, will be influenced to a greater extent by aquifer depth. Hydraulic conductivity displayed an intermediate influence within both regions. As with aquifer depth, an increase in soil permeability leads to a rapid decrease in the hydraulic gradient, providing a negative feedback mechanism. Variations in specific yield displayed maximal effects in the high marsh and minimal in the creekbank region. As specific yield properties of a soil increase, more water must be released from the soil in order to produce a unit decline in the water table. Therefore, an increase in a soils specific yield will keep hydraulic heads and gradients relatively high in regions displaying large surface elevational differences. The combination of highly permeable soil and sustenance of a relatively high hydraulic gradient lead to the maximal influence in the high marsh region. Although large hydraulic gradients were maintained within the creekbank region, the influence of specific yield was over shadowed by the soils low permeability. Results from sensitivity analysis within the creekbank region concur with Harvey et. al. (1987), who reported that geomorphological factors (i.e., aquifer depth, marsh surface elevations) influence cumulative discharge to a greater degree than soil hydrophysical properties (i.e., hydraulic conductivity, specific yield).

Figure 4.1: Tidal variations in creekbank hydraulic heads and watertable, March 3, 1987. Horizontal reference line represents soil surface elevation.

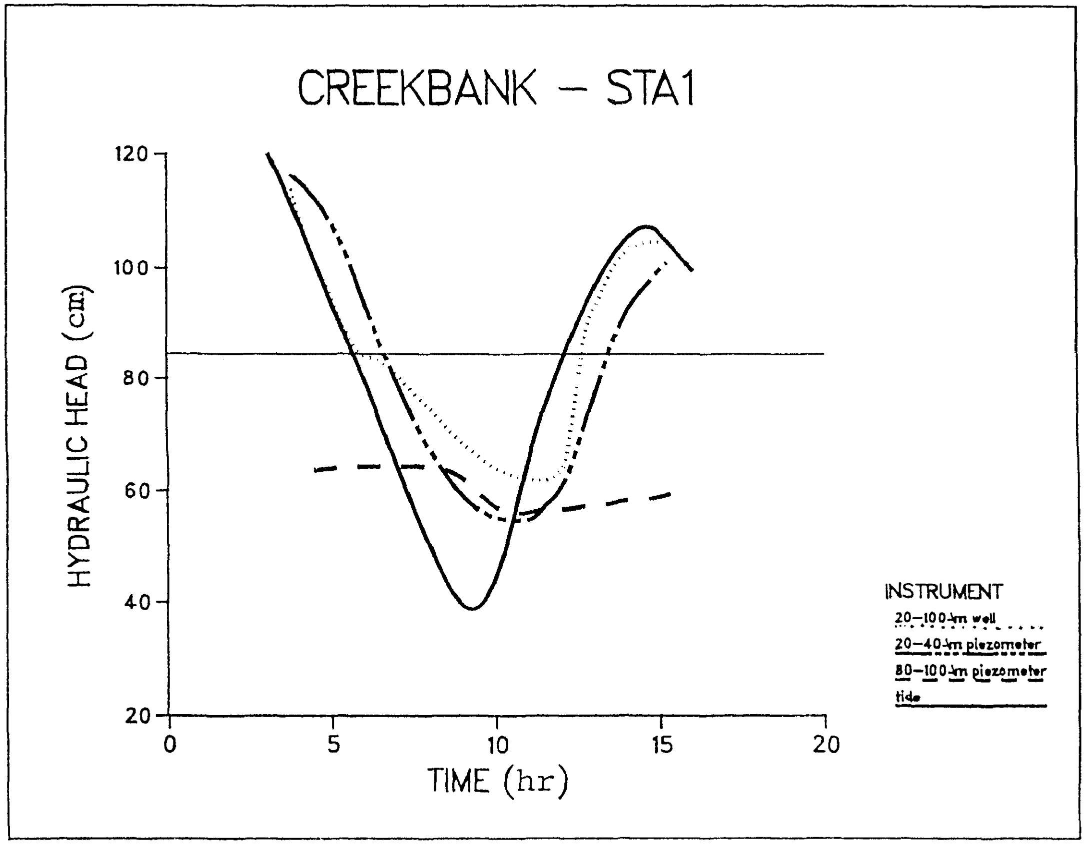

CREEKBANK – STA1
HYDRAULIC HEAD (cm)
120
100
80
60
40
20
0
5
10
15
20
TIME (hr)
INSTRUMENT
20–100-cm well
20–40-cm piezometer
80–100-cm piezometer
tide

Figure 4.2: Tidal variations in levee crest hydraulic heads and watertable, March 3, 1987. Horizontal reference line represents soil surface elevation.

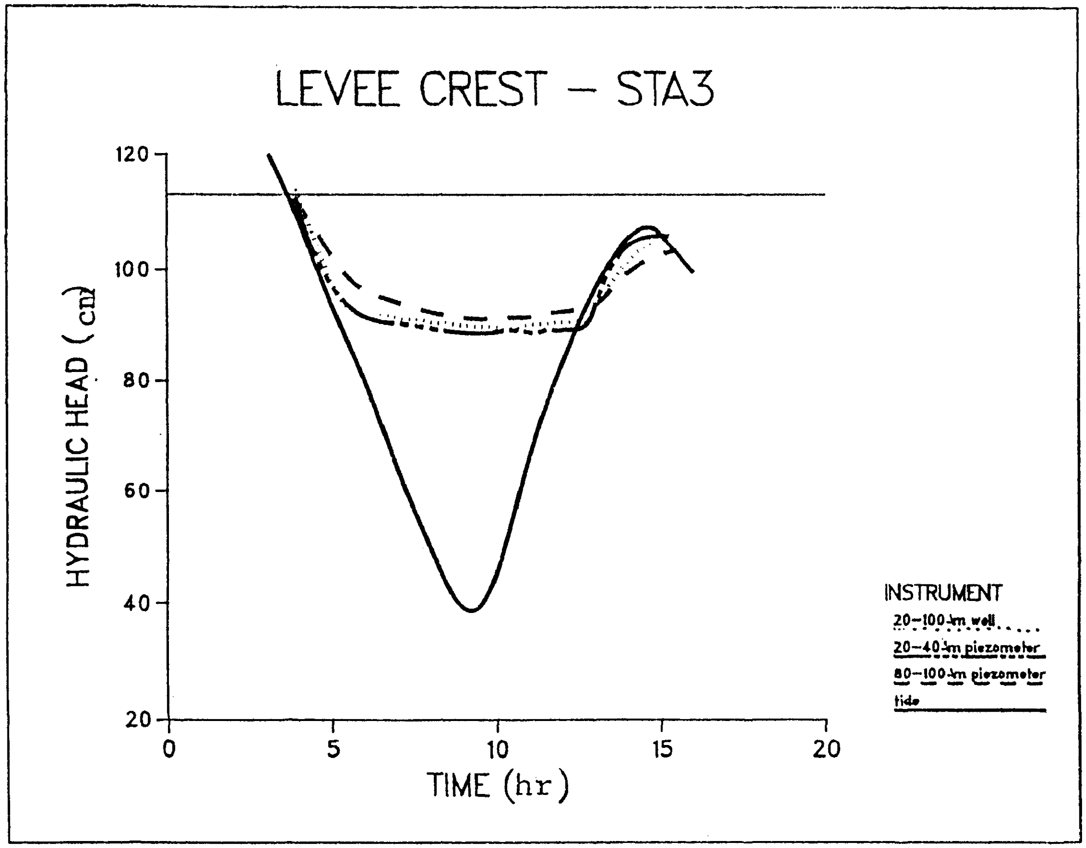
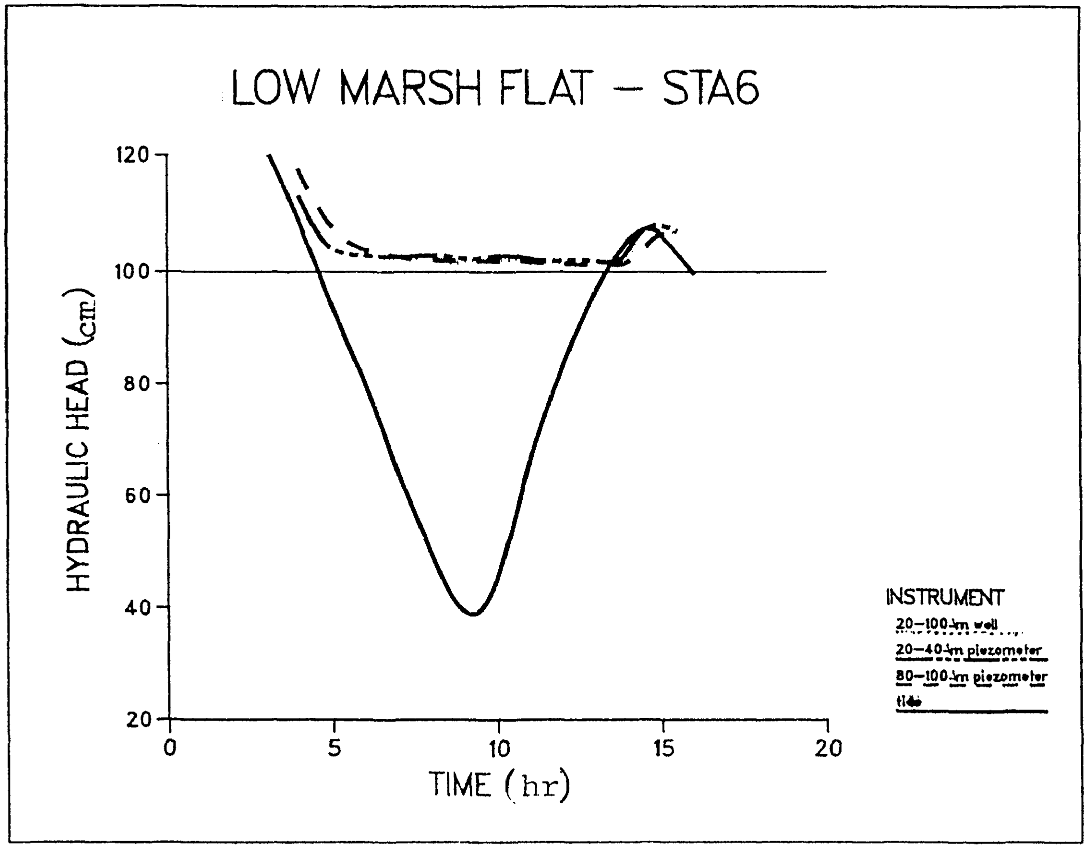

LEVEE CREST – STA3
HYDRAULIC HEAD (cm)
120
100
80
60
40
20
0
5
10
15
20
TIME (hr)
INSTRUMENT
20–100-m well
20–40-m piezometer
80–100-m piezometer
tide

Figure 4.3: Tidal variations in low marsh hydraulic heads and watertable, March 3, 1987. Horizontal reference line represents soil surface elevation.



LOW MARSH FLAT — STA6
HYDRAULIC HEAD (cm)
120
100
80
60
40
20
0
5
10
15
20
TIME (hr)
INSTRUMENT
20–100-cm well
20–40-cm piezometer
80–100-cm piezometer
tide

Figure 4.4: Tidal variations in high marsh hydraulic heads and watertable, March 3, 1987. Horizontal reference line represents soil surface elevation.

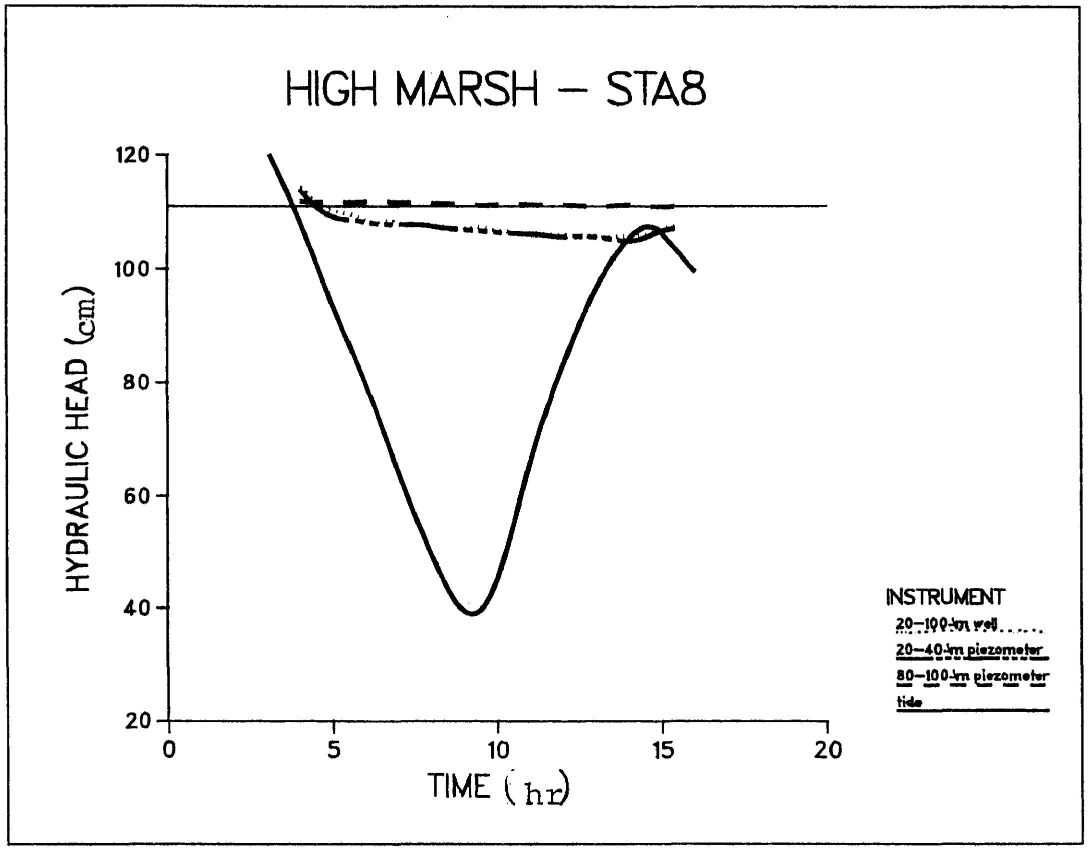
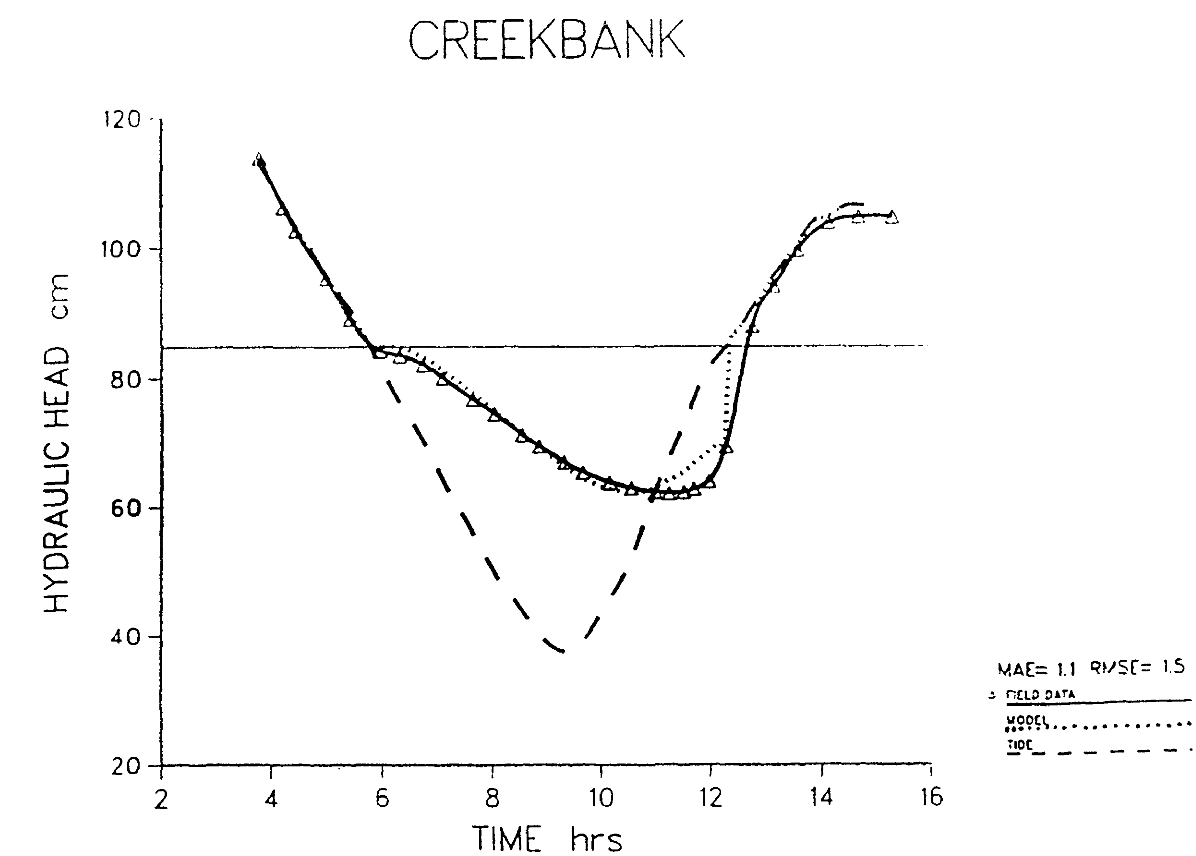

HIGH MARSH — STA8
HYDRAULIC HEAD (cm)
120
100
80
60
40
20
0
5
10
15
20
TIME (hr)
INSTRUMENT
20–100-cm well
20–40-cm piezometer
80–100-cm piezometer
tide

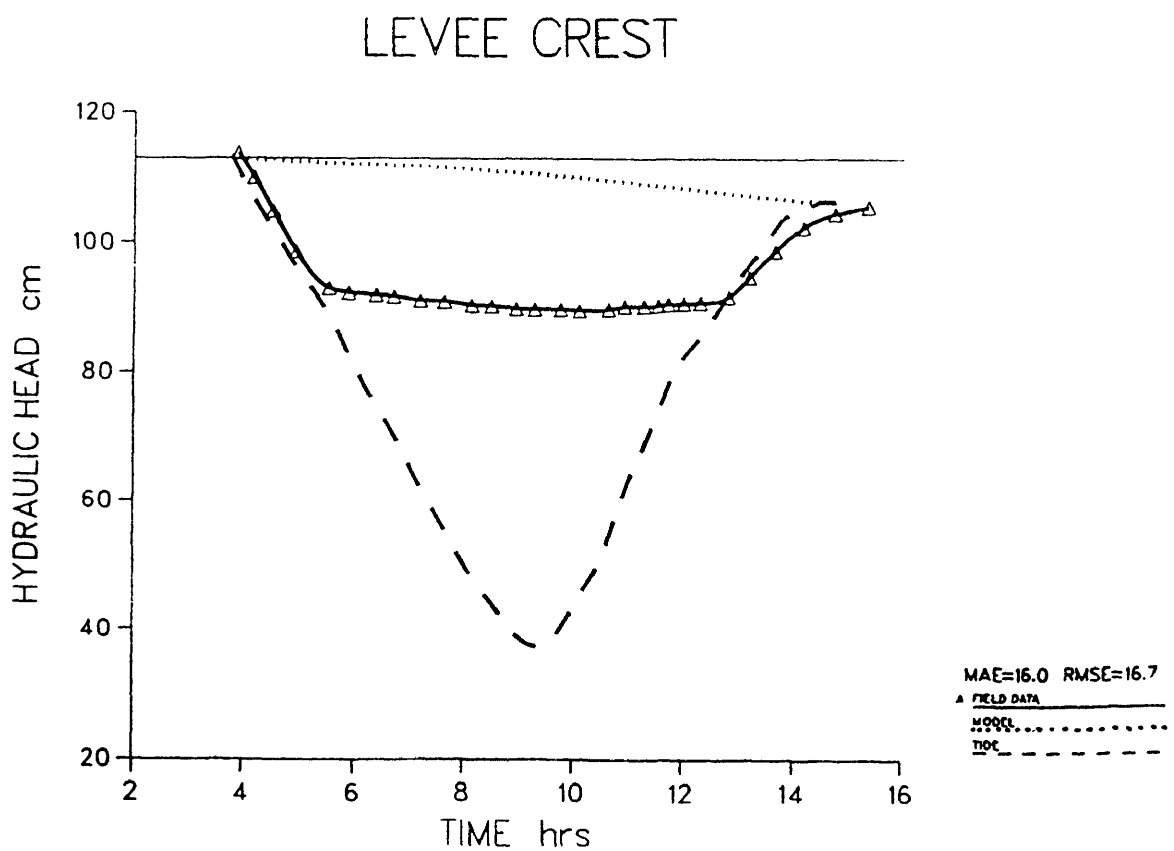
Figure 4.5: (a) Simulated watertable response within creekbank region, March 3, 1987.

(b) Simulated watertable response within levee region, March 3, 1987.

Horizontal reference line represents soil surface elevation.


CREEKBANK
HYDRAULIC HEAD cm
TIME hrs
MAE= 1.1 RMSE= 1.5
FIELD DATA
MODEL
TIDE

(a)

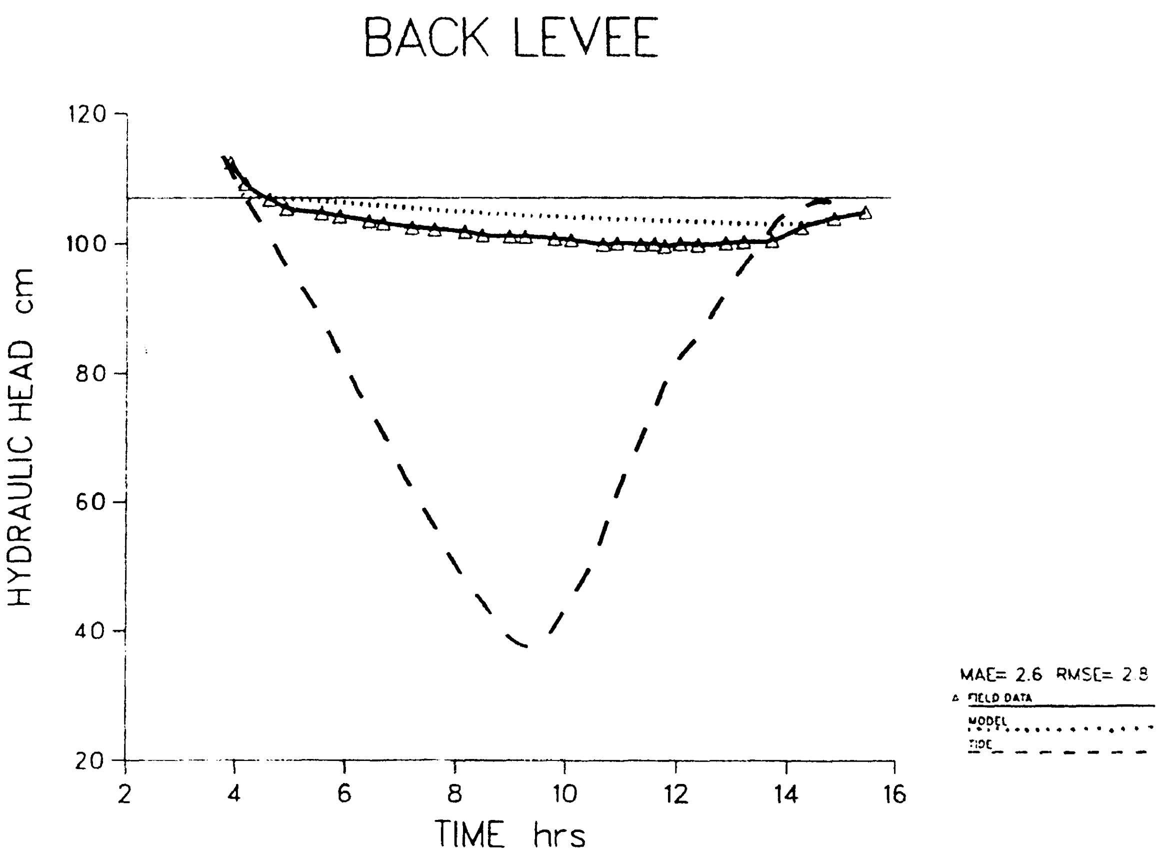

LEVEE CREST
HYDRAULIC HEAD cm
TIME hrs
MAE=16.0 RMSE=16.7
FIELD DATA
MODEL
TIDE

(b)

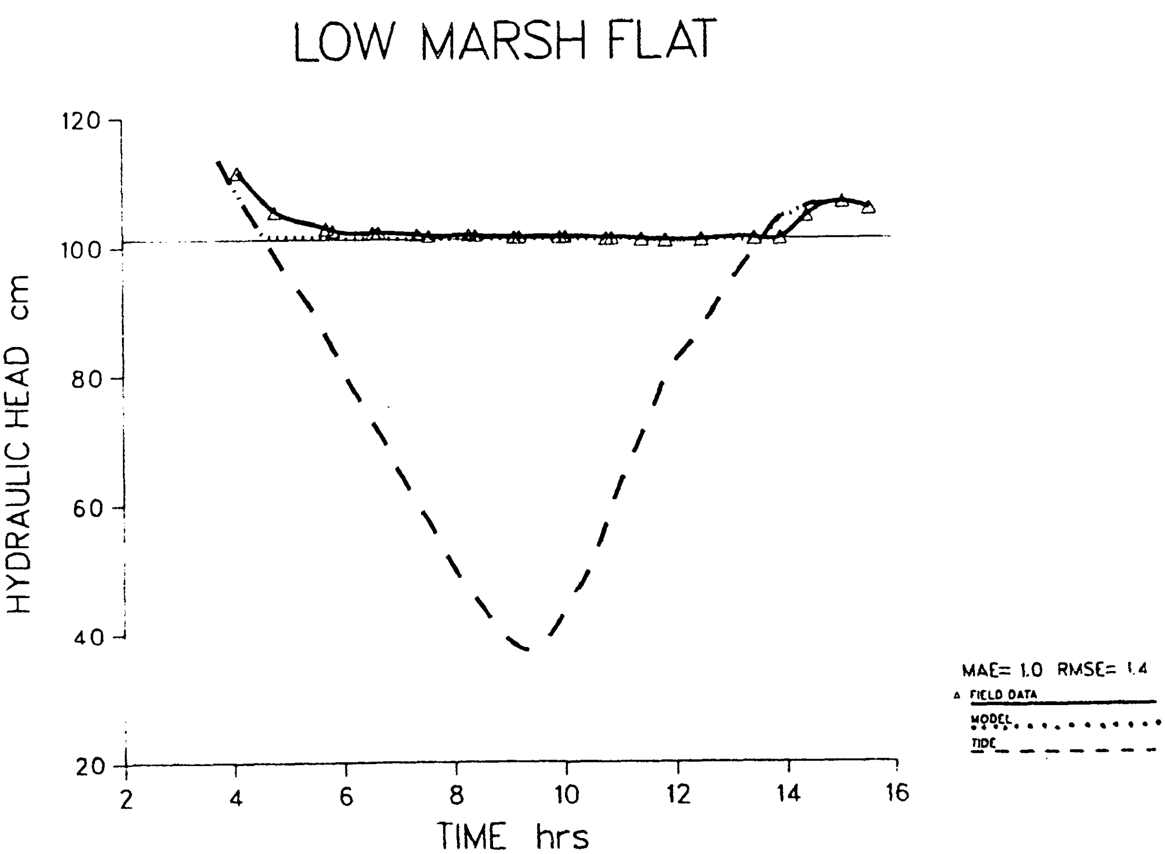
Figure 4.6: (a) Simulated watertable response within back-levee region, March 3, 1987.

(b) Simulated watertable response within low marsh region, March 3, 1987.

Horizontal reference line represents soil surface elevation.


BACK LEVEE
HYDRAULIC HEAD cm
120
100
80
60
40
20
2
4
6
8
10
12
14
16
TIME hrs
MAE= 2.6 RMSE= 2.8
FIELD DATA
MODEL
TIDE

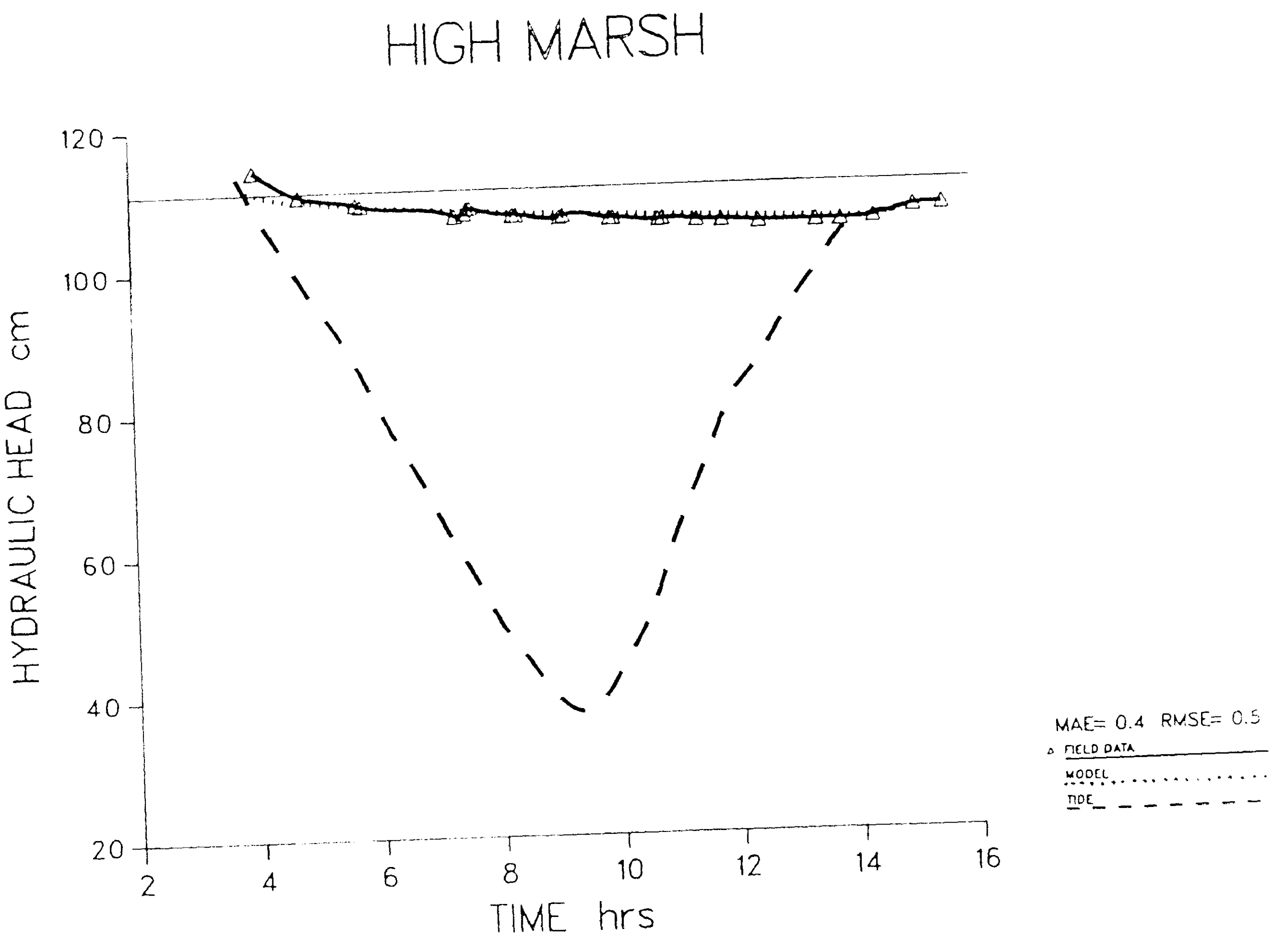
(a)


LOW MARSH FLAT
HYDRAULIC HEAD cm
120
100
80
60
40
20
2
4
6
8
10
12
14
16
TIME hrs
MAE= 1.0 RMSE= 1.4
FIELD DATA
MODEL
TIDE

(b)

Figure 4.7: Simulated watertable response within high marsh region, March 3, 1987.

Horizontal reference line represents soil surface elevation.



HIGH MARSH
120
100
80
60
40
20
HYDRAULIC HEAD cm
2
4
6
8
10
12
14
16
TIME hrs
MAE= 0.4 RMSE= 0.5
FIELD DATA
MODEL
TIDE

Figure 4.8: Simulated discharge curves for
March 3, 1987.
(a) creekbank, (b) levee crest,
(c) low marsh flat, (d) high marsh.

Negative values indicate effluent
discharge towards the creek.

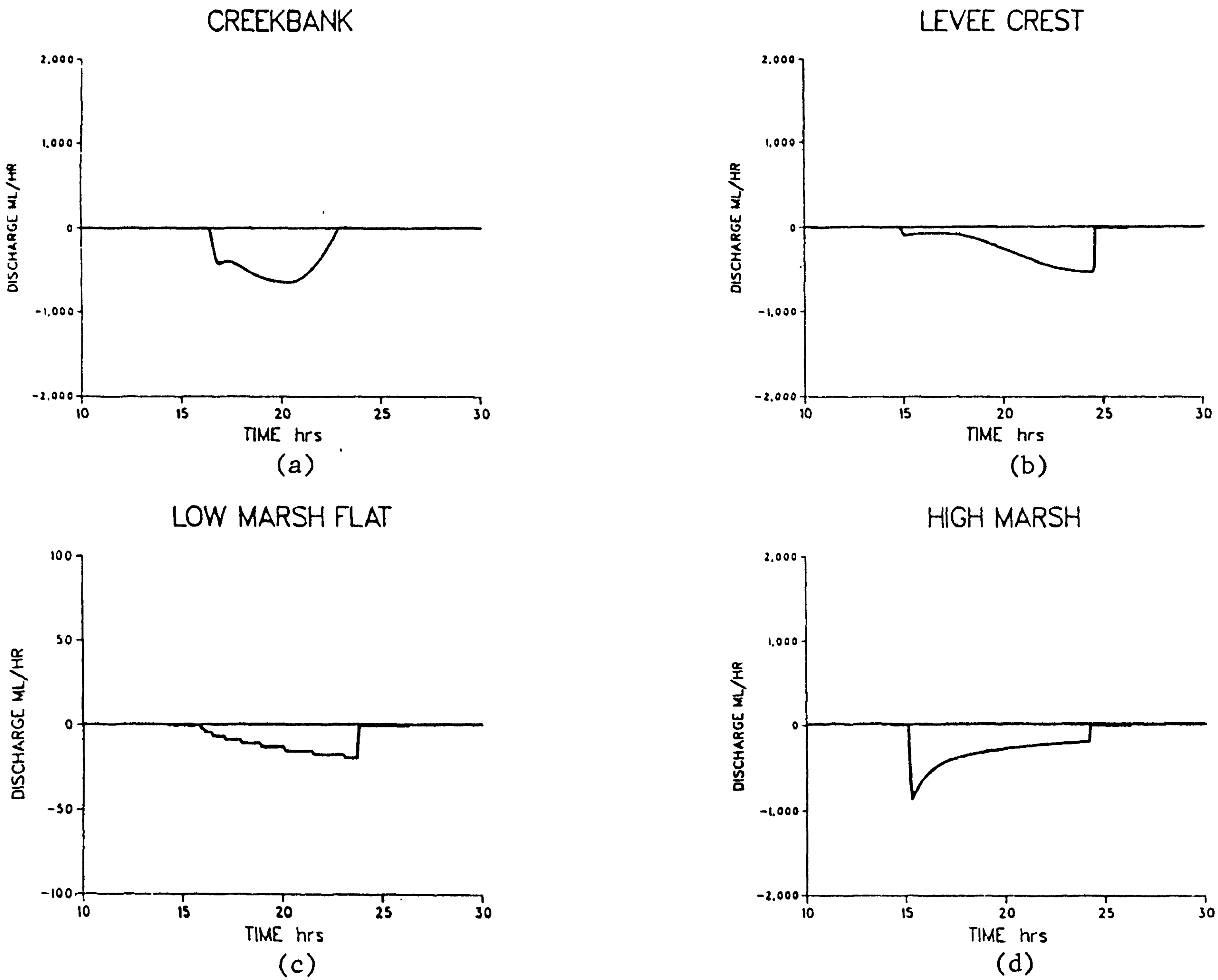
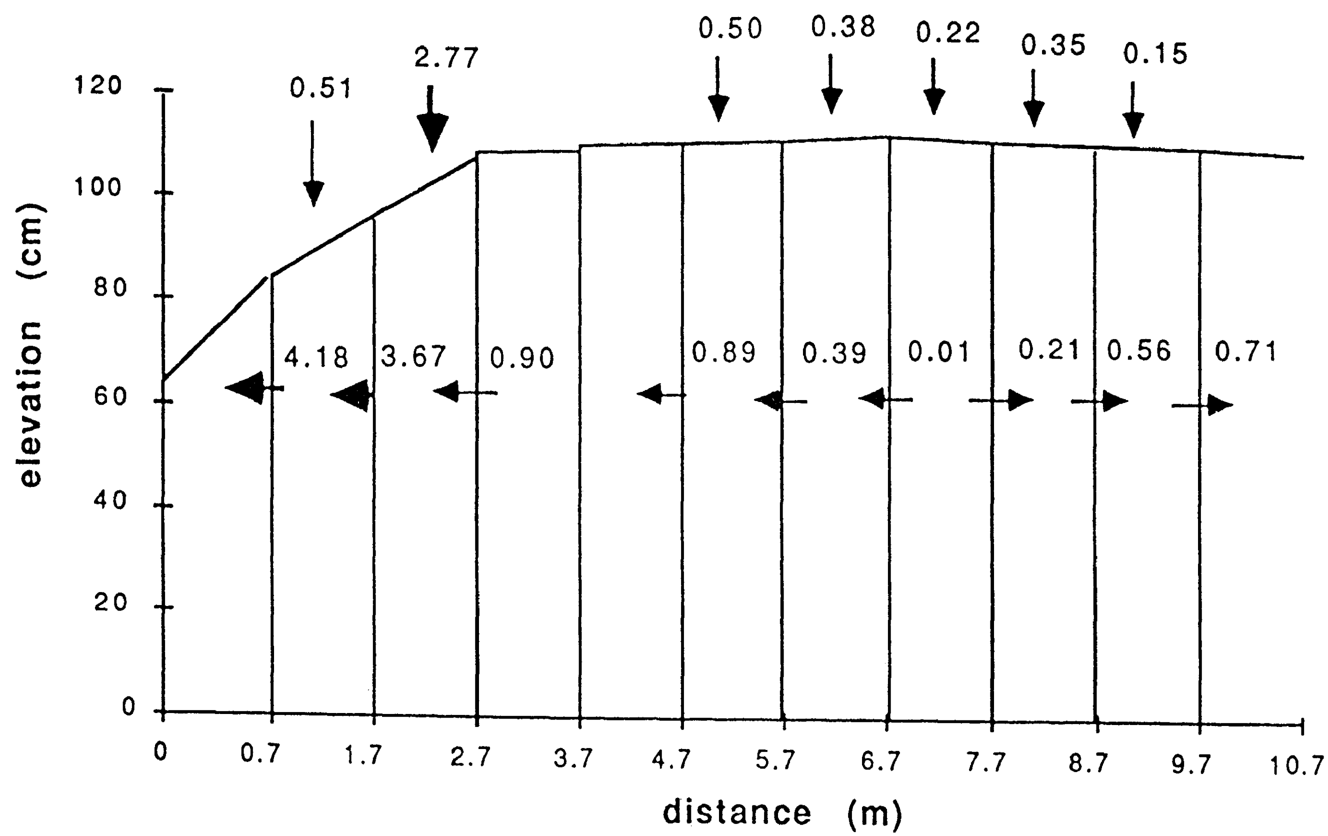

CREEKBANK
LEVEE CREST
LOW MARSH FLAT
HIGH MARSH
DISCHARGE ML/HR
TIME hrs
2,000
1,000
0
-1,000
-2,000
100
50
-50
-100
10
15
20
25
30
(a)
(b)
(c)
(d)

Figure 4.9: Creekbank and levee pore water budgets without the influence of *Ondatra zibethica.*

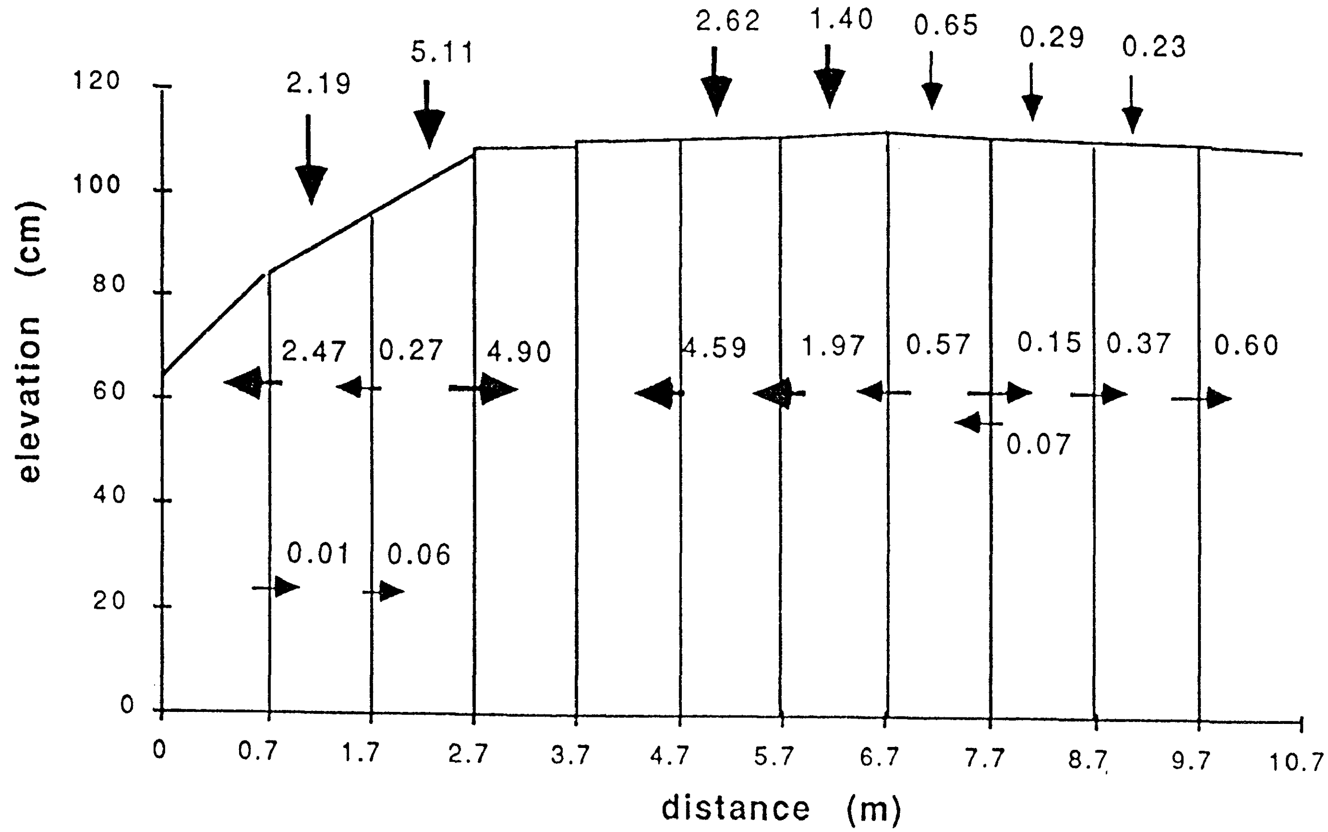
Pore water budgets are in liters per tidal cycle.



0.51
2.77
0.50
0.38
0.22
0.35
0.15
4.18
3.67
0.90
0.89
0.39
0.01
0.21
0.56
0.71
elevation (cm)
120
100
80
60
40
20
0
0
0.7
1.7
2.7
3.7
4.7
5.7
6.7
7.7
8.7
9.7
10.7
distance (m)

Figure 4.10: Creekbank and levee pore water budgets under the influence of *Ondatra zibethica.*

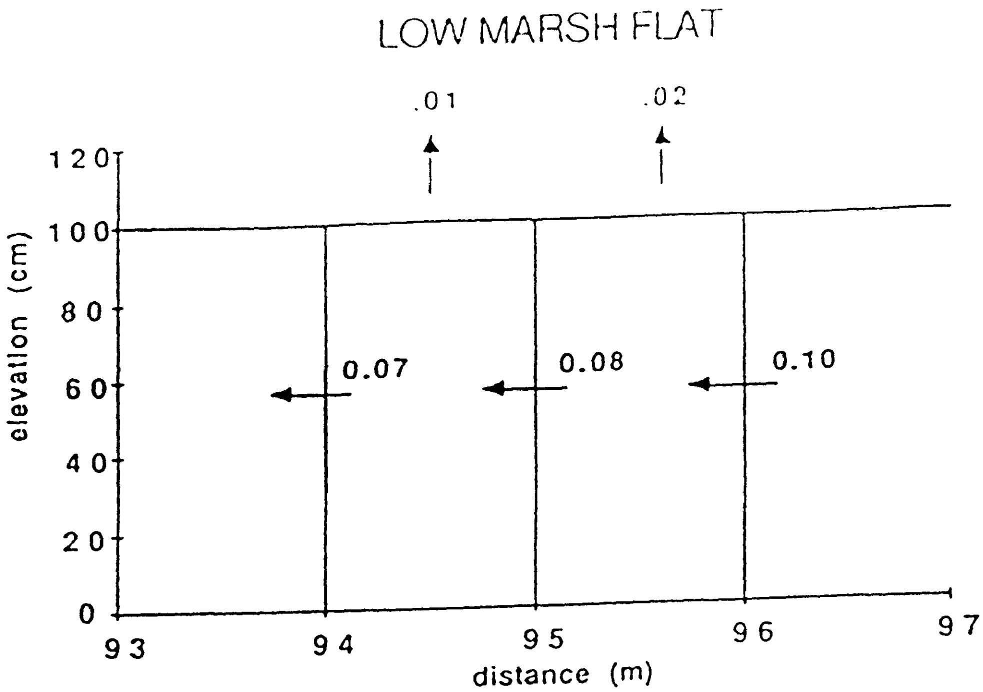
Pore water budgets are in liters per tidal cycle.



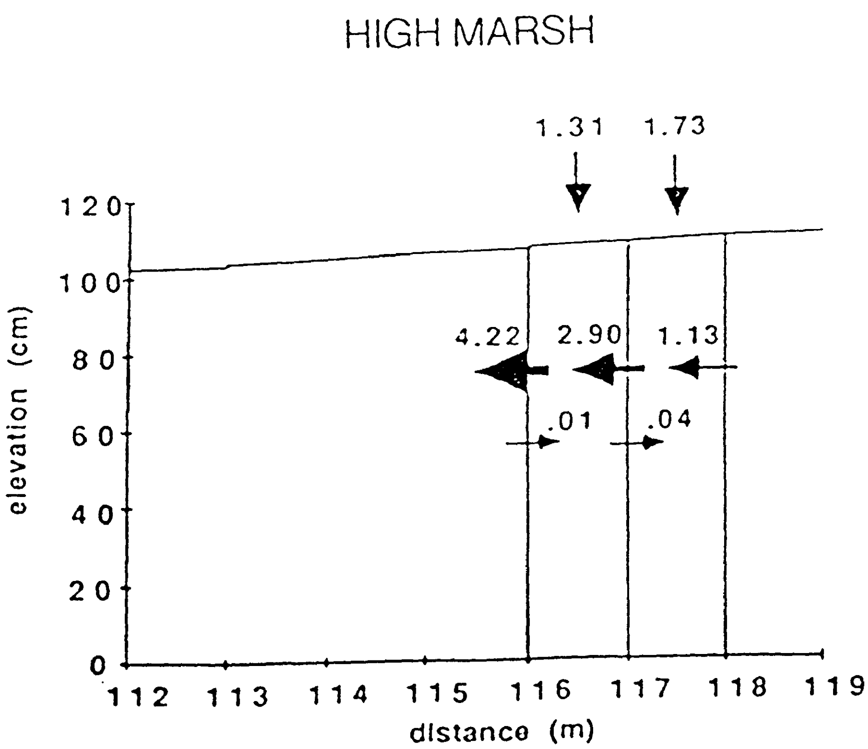
elevation (cm)
120
100
80
60
40
20
0
distance (m)
0
0.7
1.7
2.7
3.7
4.7
5.7
6.7
7.7
8.7
9.7
10.7
2.19
5.11
2.62
1.40
0.65
0.29
0.23
2.47
0.27
4.90
4.59
1.97
0.57
0.15
0.37
0.60
0.07
0.01
0.06

Figure 4.11: (a) Low marsh pore water budget.
(b) High marsh pore water budget.

Pore water budgets are in liters per tidal cycle.


LOW MARSH FLAT
.01
.02
120
100
80
60
40
20
0
0.07
0.08
0.10
93
94
95
96
97
elevation (cm)
distance (m)

(a)


HIGH MARSH
1.31
1.73
120
100
80
60
40
20
0
4.22
2.90
1.13
.01
.04
112
113
114
115
116
117
118
119
elevation (cm)
distance (m)

(b)

Figure 4.12: Creekbank model sensitivity analysis

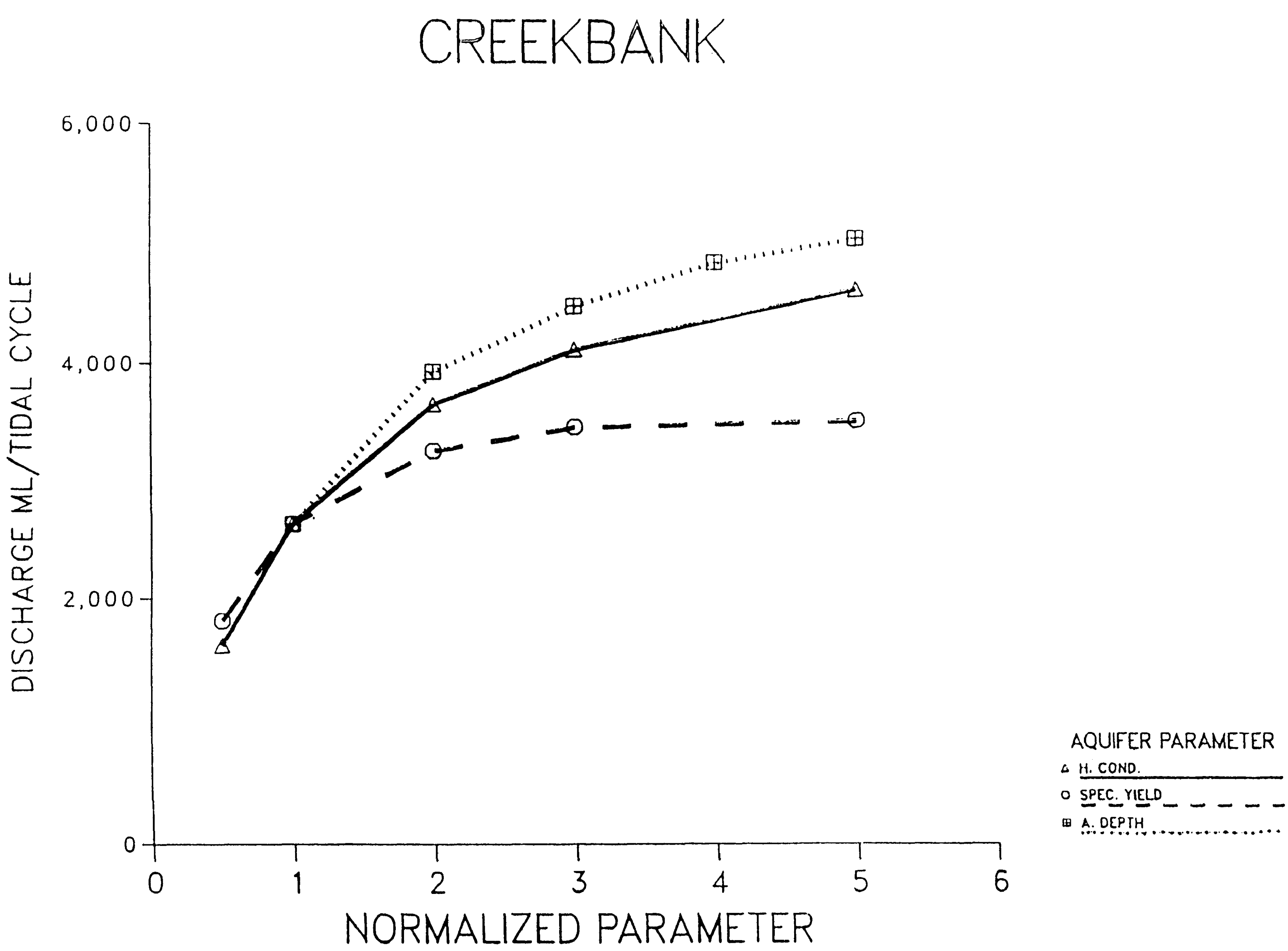
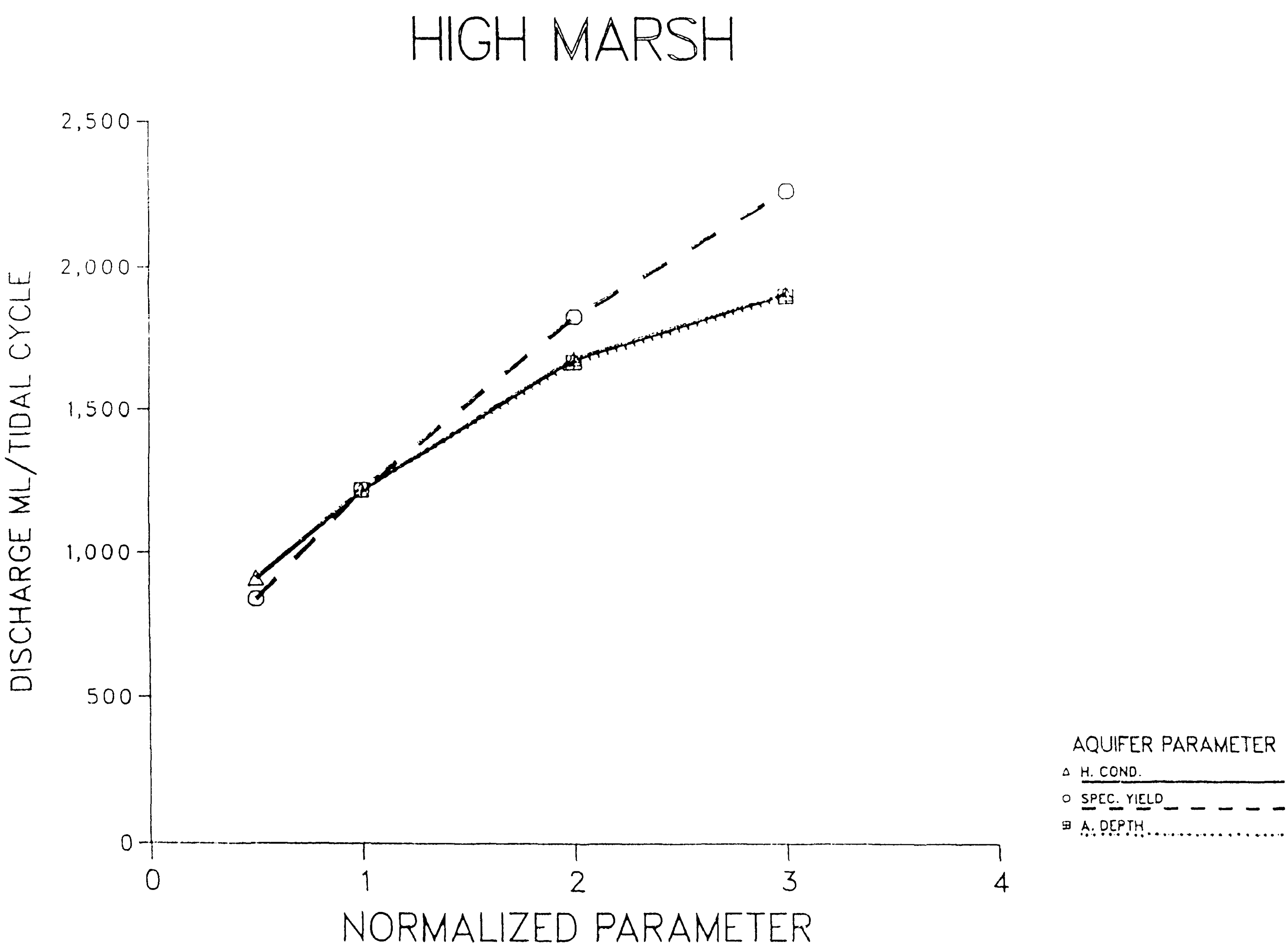

CREEKBANK
DISCHARGE ML/TIDAL CYCLE
6,000
4,000
2,000
0
0
1
2
3
4
5
6
NORMALIZED PARAMETER
AQUIFER PARAMETER
H. COND.
SPEC. YIELD
A. DEPTH

Figure 4.13: High marsh model sensitivity analysis.



HIGH MARSH
DISCHARGE ML/TIDAL CYCLE
2,500
2,000
1,500
1,000
500
0
0
1
2
3
4
NORMALIZED PARAMETER
AQUIFER PARAMETER
H. COND.
SPEC. YIELD
A. DEPTH

V. INTERSTITIAL WATER NUTRIENT CHEMISTRY AND EXCHANGE POTENTIAL

Introduction

Interstitial water nitrogen and phosphorus chemistry, and availability of these nutrients within a wetland soil is influenced by a wide variety of interacting physio-chemical and biological processes, degree of soil saturation, environmental conditions, and characteristics of recharging water (Clymo, 1983). Sources of recharging water include inundating surface waters, precipitation, seepage from adjacent wetland soil regions, and upland groundwater sources. During periods of exposure, subsurface seepage and evapotranspiration may export pore water from the soil allowing entry of atmospheric oxygen and recharging water into the soil matrix. A saturated soil condition effectively reduces the exchange of oxygen and solutes with the atmosphere and/or recharging water. Typically a two layered soil system will develop, a shallow oxidized region overlying a thicker anaerobic soil region. The aerobic/anaerobic interface is transient in nature, being influenced by vegetation and hydrodynamic processes within the region.

Due to the anaerobic nature of wetland soils, interstitial waters are generally enriched with reduced forms of nutrient species. Organic nitrogen and ammonium are the primary species of dissolved nitrogen occuring within wetland soils. If not utilized by the vegetation, ammonium accumilates within the soil due the the discontinuation of organic nitrogen mineralization. Nitrate and nitrite, the remaining inorganic nitrogen species, generally do not accumilate due to rapid microbial mediated denitrification processes. Denitrification is a process where microbial communities utilize nitrogenous oxides as terminal electron acceptors during the oxidation of organic matter, whereby, they are reduced and removed from the system either as nitrogen gas (N_2) or nitrous oxide (N_2O).

Phosphorus is more closely associated with sediment than nitrogen species, however, its availibilty increases under saturated anaerobic soil conditions. Factors effecting this increased availability include reduction of metal phosphate minerals to more soluble forms and the reduction of ferric oxides on clay/silt particles which subsequently release phosphate. Therfore, interstitial waters are generally enriched in phosphate and other soluble forms of phosphorus relative to surface waters. Vegetative uptake and reprecipitation of insoluble ferric phosphates within oxidized regions contribute to the decrease of available phosphate within the system. For a detailed review of nutrient transformations within wetland soils, the reader is referred to Delaune et. al. (1976) and Reddy and Patrick (1984).

5.1 Experimental Methods

Interstitial water nutrient samples were collected bi-monthly during the spring (March-May) and summer (June-August) seasons, and monthly during the fall (September-November) and winter (December-February) seasons. Samples were collected from three transects which extended from the uplands to the wetlands edge at the thoroughfare. Each transect consisted of four sampling stations, these include: the uplands, high marsh, low marsh flat and creekbank/levee regions. Each sampling station consisted of a three well array, except for the upland station which consisted of a single well. Sampling depths at each station within the wetland were 20-40, 50-70, and 80-100 centimeters. The upland stations consisted of a single well that sampled the upper limit of the phreatic aquifer. Replicate water samples were also taken from the thoroughfare during all sampling periods.

Monitoring wells were constructed of PVC pipe (4cm inside diameter) which were capped at the bottom and had a removable threaded top. Wells were perforated for a 20 centimeter distance and wrapped with plastic screening to prevent the entry of large particles. Wells were installed as outlined in the Subsurface Hydrodynamics methodolgy section. A bentonite sealant was installed at the surface to help prevent vertical migration down the side of the well by overlying waters. Collection of interstitial water was accomplished by first evacuating the well with a vacuum pump and collecting recharging water into a 250ml polyethylene sample bottle. All sampling instruments were flushed with deionized water between sample collections. Samples were stored on ice until return to the lab.

Within two hours of collection, samples were filtered with combusted and deionized water rinsed Whatman A/E glass filters. Water samples were analyzed spectrophotometrically for dissolved inorganic nitrogen species, total nitrogen, and total phosphorus. Ammonium was analyzed following the hypochlorite/phenol procedure (U.S. EPA, 1983). Nitrite was determined by diazotizing with sulfanilamide and coupling with N-(1-naphyl)-ethylenediamine to form an azo dye (U.S. EPA, 1983). Nitrate was determined after reduction to nitrite by use of Cu-Cd column. Nitrate was calculated as the difference between the reduced sample (nitrate + nitrite) and the non-reduced sample (nitrite). Total nitrogen (inorganic + organic fixed nitrogen) was determined by first subjecting the sample to a potassium persulfate digestion. Following digestion, HCl solution and borate buffer were added and the sample was analyzed as in the nitrate procedure (D'Elia et. al., 1977). Urea was used for total nitrogen standards. Total dissolved phosphorus samples were subjected to an ammonium persulfate digestion followed by adjustment of pH. The sample was then analyzed as in the Ascorbic acid procedure for orthophosphate (U.S. EPA, 1983).

5.2 Results

Nonparametric analysis of variance by ranks (Kruskal-Wallis Test) was utilized to determine if significant variations in mean nutrient concentrations existed between depth intervals within a specified marsh location, and between marsh locations at corresponding depth intervals. Nutrient data incorporating all sampling periods were used for statistical analysis. If significant variations did occur between depths or sample locations, a nonparametric multiple comparison test for unequal sample sizes was used to determine which samples significantly differed from one another (Zar, 1984). All statistical test used an alpha level of 0.05.

Mean creekbank interstitial water nutrient concentrations, with respect to time and depth, are given in Figures 5.1 and 5.2. Descriptive statistics incorporating all samples of the study is given in Table 5.1. Mean sampling date nitrate concentrations (Fig. 5.1a) varied from non-detectable to 23.8uM/liter. Nitrate concentrations exhibited a general decreasing pattern with increasing depth. The 20-40cm sampling depth varied significantly from the 80-100cm depth interval. Mean nitrate levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 5.4, 2.4, and 1.7uM/liter respectively. Temporal variations, similar to that of creek surface water, were exhibited by the 20-40cm depth interval. Elevated levels (>10uM/liter) occured in the winter and early spring months, followed by a sharp decline and stabilized lower levels (<2uM/liter) throughout the summer and fall months. The 50-70cm and 80-100cm pore water samples remained relatively low and uniform throughout the entire sampling period. Nitrite contribution to the creekbank interstitial water nitrogen pool was negligable; mean sampling date concentrations varied from non-detectable to less than 0.5uM/liter (Fig. 5.1b). Analysis of variance by ranks indicated no significant variations of nitrite with respect to depth. A distinct temporal trend similar to nitrate levels at the 20-40cm sampling depth was not observed for nitrite levels at the corresponding depth interval.

Mean creekbank sampling date ammonium concentrations (Fig. 5.1c) varied from 1.0-13.5uM/liter with no obvious seasonal trends. Mean ammonium levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 6.4, 4.9 and 4.7uM/liter respectively. The 20-40cm depth interval varied significantly from the 50-70cm and 80-100cm depth intervals. Dissolved organic nitrogen was the primary constituent of dissolved nitrogen within creekbank interstitial waters. Dissolved organic nitrogen approximated 80 percent of the total nitrogen pool. Mean sampling date concentrations varied from 15.7 to 57.3uM/liter (Fig. 5.1d). Data indicates a possible decreasing trend of dissolved organic nitrogen levels from the spring through the fall with a subsequent increase during the late winter season. Significant variations with depth were not observed. Mean dissolved organic nitrogen levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 33.4, 31.8, and 30.0uM/liter respectively. Total dissolved nitrogen concentrations follow a similar temporal pattern as organic nitrogen (Fig. 5.2a); mean sampling date variations ranged from 21.4 to 74.9uM/liter. Total nitrogen concentrations within the 20-40cm depth interval were significantly higher than concentrations at the 80-100cm depth interval. Mean dissolved total nitrogen levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 44.0, 38.2, and 36.5uM/liter respectively.

Creekbank total phosphorus levels displayed a peak during the spring followed by a rapid decline and stabilized low levels for the remainder of the study (Fig. 5.2b). Mean sampling date concentrations varied from 0.7 to 15.8uM/liter. Total phosphorus concentrations did not differ significantly with respect to depth. Mean concentration levels during the study period were 3.9, 3.8, and 3.8uM/liter for the 20-40cm, 50-70cm, and 80-100cm depth intervals respectively.

Mean low marsh flat interstitial water nutrient concentrations, with respect to time and depth, are given in Figures 5.3 and 5.4. Descriptive statistics incorporating all samples and sampling dates are given in Table 5.2. Mean sampling date pore water concentrations of nitrate and nitrite ranged from non-detectable to 2.2uM/liter and non-detectable to 0.3uM/liter respectively (Figs. 5.3a and 5.3b). Nitrate and nitrite concentrations did not vary significantly as a function of depth. Mean nitrate levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 1.1, 1.0, and 0.9uM/liter respectively. Overall mean nitrite levels throughout the study period were undetectable to 0.1uM/liter for all three depth intervals. Nitrate and nitrite levels did not exhibit observable seasonal trends.

Ammonium was the primary inorganic nitrogen species within the low marsh flat, however, concentrations appeared extremely low for a wetland soil. Mean sampling date concentration levels ranged from 0.4 to 7.8uM/liter (Fig. 5.3c). Data does not indicate a discernable seasonal pattern. Mean ammonium levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 4.3, 3.5, and 3.2uM/liter respectively. Ammonium levels differed significantly between the 20-40cm and 80-100cm depth intervals. Dissolved organic nitrogen comprised approximately 85 percent of the total nitrogen pool. Mean sampling date concentration levels varied from 16.2 to 62.8uM/liter. Data indicates a decreasing trend of dissolved organic nitrogen from the spring through the fall with a subsequent increase during the winter season (Fig. 5.3d); this trend was observed at all sampling depth intervals. Significant differences of dissolved organic nitrogen levels between depth intervals were not observed. Mean concentrations during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 34.8, 33.3, and 33.6uM/liter respectively. Total dissolved nitrogen concentrations follow a similar temporal pattern as organic nitrogen (Fig. 5.4a); mean sampling date levels range from 20.4 to 65.9uM/liter. No significant variations of total dissolved nitrogen with respect to depth were observed. Mean dissolved total nitrogen concentrations during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 40.4, 38.0, and 38.1uM/liter respectively. Total phosphorus levels were variable and did not follow a consistant pattern (Fig. 5.4b); mean sampling date concentrations range from 1.3 to

15.4uM/liter. Low marsh flat mean total phosphorus concentrations for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 7.5, 9.0, and 6.3uM/liter respectively. The 50-70cm depth interval generally exhibited elevated mean monthly total phosphorus concentrations during the summer and fall months; overall concentrations at the 50-70cm depth differed significantly from the 80-100cm depth interval.

Mean high marsh interstitial water nutrient concentrations, with respect to time and depth, are given in Figures 5.5 and 5.6. Descriptive statistics incorporating all samples and sampling dates is given in Table 5.3. Interstitial water nitrate and nitrite concentrations within the high marsh region were a minor component of the total dissolved nitogen pool. Mean sampling date concentrations of nitrate and nitrite ranged from non-detectable to 7uM/liter and nondetectable to <0.3uM/liter respectively (Figs. 5.5a and 5.5b). The final sampling date in February 1987 displayed an anomily of increased nitrate concentration level (7uM/liter) within the 20-40cm depth interval. Mean nitrate concentrations in the 50-70cm depth interval were influenced by a single well that typically had elevated nitrate levels. Upon removal of this well, a sand lens originating in the adjacent uplands was identified. This would increase the potential for relatively nutrient rich upland groundwater or tidal water to recharge the soil region around the well. Overall nitrate and nitrite levels did not significantly differ between the three sampled depth intervals. Mean sampling date ammonium concentrations (Fig. 5.5c) varied from 1.0 to 25.2uM/liter. The 20-40cm depth interval ammonium concentrations indicate a temporal pattern of high levels occuring in the winter and early spring and decreasing through the summer and fall months. Mean ammonium levels during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 9.4, 9.0 and 2.9uM/liter respectively. The 20-40cm and 50-70cm depth intervals differed significantly from the 80-100cm depth interval.

Dissolved organic nitrogen comprised approximately 85 percent of the total nitrogen pool within the high marsh. Mean sampling date concentration levels varied from 26.9 to 75.0uM/liter (Fig. 5.5d). Data indicates a decreasing trend of dissolved organic nitrogen from the spring through the fall with a subsequent increase during the winter season, this trend was observed at all sampling depth intervals. Mean dissolved organic nitrogen concentrations during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 55.8, 45.3, and 40.2uM/liter respectively. Overall mean dissolved organic nitrogen levels displayed significant variations with depth, the 20-40cm depth interval varied significantly from the 50-70cm and 80-100cm depth intervals. Total dissolved nitrogen concentrations followed a similar temporal pattern as organic nitrogen (Fig. 5.6a); mean monthly variations ranged from 28.8 to 89.4uM/liter. Significant variations of total dissolved nitrogen with depth were observed; the 20-40cm sampling depth differed significantly from the 50-70cm and 80-100cm depth intervals, and the 50-70cm depth varied significantly from the 80-100cm depth interval. Mean total dissolved nitrogen concentrations during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 65.5, 55.7, and 43.7uM/liter respectively.

High marsh total phosphorus levels showed a steady decrease from spring months through the fall and subsequent winter (Fig. 5.6b); mean sampling date concentrations ranged between 0.7 and 7.7uM/liter. The 80-100cm depth interval exhibiteded elevated levels in relationship to the other sampling depths throughout the entire study. Mean total phosphorus concentrations during the study period for the 20-40cm, 50-70cm, and 80-100cm depth intervals were 3.4, 3.2, and 4.6uM/liter respectively. The 80-100cm depth interval varied significantly from the 50-70cm depth interval.

Descriptive statistics for nutrient concentrations within the creek water as a function of time are given in Figures 5.7 and 5.8. Mean sampling date nitrate levels varied from 1.4 to 56.1uM/liter (Fig. 5.7a). Nitrate levels exhibited a distinctive seasonal pattern. High monthly mean concentrations occured in early spring (18uM/liter) and winter (35.6uM/liter); this coincided with regional periods of high rainfall, surface runoff, and decreased water column productivity. Summer and fall months were characterized by relatively low nitrate concentrations, mean seasonal values were 7.4 and 7.7uM/liter respectively. Nitrite was a minor constituent of the creekwater nitrogen pool and did not display a discernable seasonal pattern (Fig. 5.7b). Mean nitrite sampling date concentrations varied between non-detectable and 0.7uM/liter. Ammonium levels within creek surface waters remained relatively low throughout the sampling period with mean sampling date concentrations varing from 1.7 to 7uM/liter (Fig. 5.7c). Winter ammonium concentrations (4.4uM/liter) were elevated relative to spring (3.3uM/liter), summer (3.5uM/liter), and fall levels (2.8uM/liter).

Dissolved organic nitrogen comprised the majority of the total nitrogen pool within creek waters during the study period; mean sampling date levels varied from 6.2 to 46.4uM/liter (Fig. 5.7d). Mean spring (37.2uM/liter) and fall (30.8uM/liter) levels of dissolved organic nitrogen within creekwaters were elevated with respect to summer (28.8uM/liter) and winter (17.0uM/liter) levels. This pattern could be a reflection of increased water column productivity during the spring and fall seasons. Total nitrogen levels displayed a similar seasonal pattern as expressed by nitrates with elevated spring (51.4uM/liter) and winter (54.6uM/liter) levels (Fig. 5.8a); mean summer and fall concentrations were 40.1 and 41.5uM/liter respectively. Total nitrogen was dominated by inorganic species, specifically nitrate, during the winter and early spring and organic nitrogen throughout the remainder of the year. Mean sampling date total phosphorus levels displayed a seasonal pattern characterized by a spring peak (2.1uM/liter). Creekwater total phosphorus levels decreased during the summer (1.1uM/liter) and fall (0.9uM/liter) with a slight increase occured during in the winter months (1.3uM/liter) (Fig. 5.8b).

Mean dissolved total nitrogen and phosphorus concentrations within the shallow phreatic upland aquifer are given in Figure 5.9. Total nitrogen levels displayed an observable seasonal pattern characterized by high spring mean concentrations (621.2uM/liter) followed by a steady decline during the summer (388.6uM/liter) and fall (28.1uM/liter), with a subsequent increase during the winter (398.7uM/liter) (Fig. 5.9a).

Nitrate and dissolved organic nitrogen were the dominant nitrogen species within the shallow upland aquifer. Drought conditions existed during the summer and fall months of the study, and on several occasions monitoring wells remained dry during fall sampling periods. Dissolved total phosphorus levels were minimal throughout the entire study (Fig. 5.9b). Seasonal mean total phosphorus concentrations were 1.3, 1.0, 0.7, and 1.0uM/liter for spring, summer, fall, and winter respectively. Low levels were expected due to a high affinity of phosphorus forms with mineral sediment in oxygenated aquifers.

Significant variations in pore water nutrient chemistry occured laterally between the high marsh, low marsh flat, and creekbank regions. Lateral variations in well nutrient concentrations, as a function of depth and time, are given in Figures 5.10 to 5.12. Overall mean creekbank nitrate concentrations, at all sampling depth intervals, were significantly higher than the corresponding depth intervals in low marsh and high marsh regions. The most pronounced variation between regions occured at the 20-40cm sampling depth interval; mean nitrate levels of pore water from the creekbank, low marsh flat, and high marsh were 5.4, 1.1, and 1.1uM/liter respectively (Fig. 5.10a). Creekbank nitrite levels at the 20-40cm depth interval differed significantly from the corresponding depth interval in the low marsh and high marsh (Fig. 5.10b). Significant differences between the 50-70cm and 80-100cm depth intervals were not detected (Figs. 5.11b and 5.12b). Overall mean nitrite levels for all sampling locations were 0.1uM/liter or less.

Significant differences in ammonium concentrations between the creekbank, low marsh flat, and high marsh regions varied with sampling depth. Mean high marsh (9.4uM/liter) and creekbank (6.4uM/liter) levels, at the 20-40cm depth interval, were significantly higher than levels observed in the low marsh flat (4.3uM/liter) (Fig. 5.10c). The 50-70cm depth interval showed significant variations in ammonium concentrations between the high marsh (9.0uM/liter) and low marsh flat (3.5uM/liter) (Fig. 5.11c). Mean ammonium levels at the creekbank (4.7uM/liter) 80-100cm sampling depth interval varied significantly from corresponding depths in the low marsh flat (3.2uM/liter) and high marsh (2.9uM/liter) (Fig. 5.12c).

Mean dissolved organic nitrogen concentrations for all sampling depths within the high marsh region were significantly higher than levels observed in the low marsh and creekbank regions. The largest variation occured at the 20-40cm depth interval, mean dissolved organic nitrogen levels for the high marsh, low marsh and creekbank were 55.8, 34.8, and 33.4uM/liter respectively (Fig. 5.10d). Similar significant differences, as those for dissolved organic nitrogen levels, were observed for total nitrogen concentrations. As with dissolved organic nitrogen, the largest variation between regions occured at the 20-40cm depth interval; mean dissolved total nitrogen levels during the study for the high marsh, low marsh and creekbank were 65.5, 40.4, and 44.0uM/liter respectively.

Significant differences in total phosphorus levels varied with sampling depth. Low marsh flat mean dissolved total phosphorus levels were significantly higher than the high marsh and creekbank regions at the 20-40cm and 50-70cm sampling depth intervals (Figs. 5.10f and 5.11f). Mean total phosphorus concentrations, at the 20-40cm depth interval, for the low marsh, high marsh, and creekbank regions were 7.5, 3.4, and 3.9uM/liter respectively, while the 50-70cm sampling depth interval concentrations were 9.0, 3.2, and 3.8uM/liter respectively. Significant differences at the 80-100cm depth interval occured between the high marsh and creekbank, mean total phosphorus levels were 4.6 and 3.8uM/liter respectively (Fig. 5.12f).

5.3 Summary and Discussion

Dissolved pore water nitrogen species and total phosphorus concentrations, as a function of depth, location, and time, were highly variable, however, several generalities and patterns appeared relatively consistant. Organic nitrogen was the dominant form of dissolved nitrogen within the wetland; this form was responsible for approximately 85 percent of the total dissolved nitrogen pool. Mean sampling date concentrations of dissolved organic nitrogen varied from 15.7 to 75uM/liter. Jordan and Correll (1985), investigating nutrient dynamics within a Chesapeake Bay brackish low marsh, reported similar levels of dissolved interstitial organic nitrogen; levels ranged between 43 and 64uM/liter. The inorganic nitrogen pool was dominated by ammonium which accounted for approximately 10 percent of the dissolved total nitrogen present within interstitial waters. Mean sampling date ammonium concentrations throughout the marsh ranged from 0.4 to 25.2uM/liter. Bowden (1984), working within a Delaware tidal freshwater marsh, reported mean annual free ammonium levels of 83uM/liter at a 20-22cm depth interval. Chambers and Harvey (1988) found mean ammonium levels of 16.8uM/liter within the high marsh and 38.4uM/liter within the low marsh of a tidal freshwater wetland in Virginia. Contribution of nitrate and nitrite to the pore water total nitrogen pool was minor, except within the creekbank region during winter and early spring months. Pore waters were generally enriched in dissolved phosphorus as compared to creek surface waters; mean sampling date concentrations throughout the wetland ranged from 0.7 to 15.8uM/liter. Elevated concentrations of dissolved total phosphorus were observed in the low marsh flat. Chambers and Harvey (1988) reported mean soluble reactive phosphorus (SRP) concentrations of 25.9 and 21.0uM/liter for the high marsh and low marsh respectively.

Gardner (1973), investigating pore water chemistry in a South Carolina tidal salt marsh, identified three hydrogeochemical settings that influenced pore water chemical parameters. He recognized creekbank regions as hydraulically active that drain reduced interstitial water to adjacent surface waters during periods of aerial exposure. The creekbank soil matrix is recharged, with respect to water content, by inundating tidal waters. This serves to alter creekbank interstitial water chemistry by actively mixing reduced pore waters with oxygenated surface waters. Pore water chemistry within the marsh interior, which he classified as hydraulically stagnant and isolated, appeared to follow theoretical closed system chemistry. The final hydrogeochemical setting occured near the upland margin where pore water chemistry was influenced by fresh groundwater contribution into the region.

Investigating interstitial water chemistry and subsurface hydrology within upper Chesapeake Bay tidal brackish marshes, Jordan and Correll (1985) reported decreased nutrient levels within pore waters in close proximity to tidal creeks. This pattern of reduced nutrient levels supports the hypothesis of increased pore water drainage and surface water recharge within creekbank regions of tidal wetlands. Jordan and Correll (1985) refer to this water exchange mechanism as "tidal-pumping". Yelverton and Hackney (1986) reported decreased dissolved organic carbon pore water concentrations within the creekbank region of a North Carolina salt marsh relative to interior sampling sites; all marsh sampling sites were enriched relative to inundating creek waters. The narrow creekbank region was responsible for over ninty percent of the total seepage export along the investigated transect (Yelverton and Hackney, 1986).

Lateral pore water nutrient variations observed in this study may be explained, in part, by hydrodynamic processes. Results of field investigations and the subsurface hydrologic model indicate that an extensive tidal freshwater wetland cannot be considered a homogeneous unit. It may be described more accurately as three distinct, yet interactive regions (creekbank, low marsh flat, high marsh), with varying potentials for nutrient exchange with surface and subsurface waters.

The creekbank, experiencing large water table oscillations and hydraulic gradients, was the most dynamic and tidally influenced region. These hydrodynamic characteristics result in significant subsurface water transport and dilution of interstitial waters with inundating surface waters. Surface waters were generally nitrogen rich with respect to nitrate and nitrite, and relatively poor with respect to ammonium, dissolved organic nitrogen, and total phosphorus as compared to wetland interstitial waters. Pore water nutrient levels within the creekbank region of the tidal freshwater wetland were relatively enriched with oxidized inorganic forms of nitrogen relative to low marsh and high marsh regions. The 20-40cm sampling depth region of the creekbank displayed a similar nitrate and nitrite seasonal pattern as creek surface waters, however, nitrate concentrations were reduced in magnitude several fold. Nitrate levels of greater than 10uM/liter characterized the creekbank 20-40cm depth interval during the winter and early spring months followed by a sharp decline and stabilized levels, typically less than 2uM/liter, throughout the summer and fall. Chalmers (1977) reported a similar pattern of high pore water nitrate levels occuring in winter months. The presence of elevated nitrate levels within creekbank interstitial waters is due to nitrate enriched infiltrating surface waters and may reflect decreased microbial activity during cooler periods. Creekbank ammonium levels, mean sampling date concentrations ranging from 1.0 to 13.5uM/liter, were generally intermediate between the high marsh and low marsh flat, and enriched relative to creek surface waters. Mean dissolved organic nitrogen concentrations within the creekbank ranged from 15.7 to 57.3uM/liter; these levels were the lowest of the three regions sampled, but enriched relative to creek surface waters. Total nitrogen and phosphorus interstitial pools within the creekbank region were intermediate between the high marsh and low marsh flat. Visual observation of sediment cores within the creekbank region indicate that insoluble ferric oxides may be found at depths greater than one meter. Such iron precipitants provide a removal mechanism of dissolved interstitial phosphorus compounds.

The back-levee and low marsh flat comprised the second and most extensive general hydrologic region within the wetland. Due to extremely low hydraulic gradients and ponding of water, the low marsh flat soils were typically fully saturated with respect to water content and did not display significant horizontal water transport. Saturated soil conditions in conjunction with negligible subsurface discharge, creates a condition that is unfavorable for water and solute exchange with surface waters. Molecular diffusion, evaporation, and biological processes are therefore the primary controlling factors for water and solute exchange capability within the region. The low marsh flat was inorganic nitrogen poor relative to high marsh and creekbank regions. Inorganic nitrogen generally contributed less than 15 percent to the mean total nitrogen pool. Dissolved organic nitrogen levels within the low marsh were intermediate between the high marsh and creekbank regions. Compared to creek surface waters, low marsh pore waters were enriched with dissolved nitrogen during the winter, spring and early summer and became relatively nitrogen poor in the late summer and fall. Reduced levels of total nitrogen may reflect the regions limited exchange and recharge potential. Pore waters within the low marsh were significantly enriched with dissolved total phosphorus as compared to the creekbank and high marsh. Mean total phosphorus levels at the 20-40cm, 50-70cm, and 80-100cm depth intervals were 7.5, 9.0, and 6.3 respectively. Jordan and Correll (1985) reported that phosphate concentrations within brackish low marsh regions were consistantly higher than levels observed in high marsh regions. Mean 20-40cm depth interval total nitrogen to total phosphorus ratios for the study were 11.3:1, 5.4:1, and 19.3:1 for the creekbank, low marsh flat and high marsh respectively. A low TN:TP ratio of 5.4:1 indicates the potential for nitrogen limitation within the low marsh.

Defining the third general hydraulic region, the high marsh was characterized by moderate hydraulic gradients and highly permeable organic soils. These properties were conducive for significant horizontal discharge of interstitial waters. Model and field hydrologic evidence suggests that water discharged during aereal exposure is replenished by inundating surface waters and upland groundwater sources. Infiltration of surface waters would dilute high marsh interstitial waters with respect to total dissolved nitrogen, however, these waters would serve as a source of nitrate and dissolved oxygen. Since upland groundwaters exhibit high nitrogen levels, predominantly in the forms of nitrate and dissolved organic nitrogen, it may serve as an important contributer of nitrogen to the wetlands high marsh region. The importance of groundwater will be site specific, however, its role in wetland nutrient budgets should not be overlooked. Using field measured soil permeability and hydraulic gradient estimates of $7.6*10^{-5}$cm/sec and 0.028

respectively, groundwater discharge per meter width to the upper meter of the high marsh was 0.08 liters per hour. Although groundwater discharge appears low, dissolved total nitrogen levels were elevated (621.2uM/liter for spring mean) within the groundwater. When discharge and total nitrogen levels are coupled, total nitrogen flux (50uM/hour) to the immediate high marsh region from upland groundwater sources is significant. Groundwater flux estimates are dependent upon seasonal variation of groundwater nutrient content and water table elevation. The reported flux of 50uM of total nitrogen per hour should be considered a maximum since it incorporated spring nutrient concentrations and water table elevations. Valiela et. al.(1978), investigating nutrient fluxes within a New England salt marsh, reported that groundwater is a significant source of nitrate and dissolved organic nitrogen to the wetland system.

Nitrate and nitrite concentrations were relatively insignificant contributors to the total nitrogen pool within the high marsh, typically on the order of 1 to 2 percent. Interstitial waters within the organic soil region of the high marsh (i.e., upper 70 centimeters) displayed elevated ammonium levels relative to the low marsh and creekbank; mean sampling date concentrations varied between 2.5 and 25.2uM/liter. The 20-40cm sampling depth interval showed a seasonal trend of high winter and early spring levels followed by a decrease and stabilization of reduced levels in the summer and fall months. Elevated ammonium levels during the winter and early spring coincide with elevated nitrate levels found within inundating surface waters and upland groundwaters. This pattern reflects biologically reducing processes and highlights the importance and potential use of tidal freshwater wetlands for non-point source water interception and purification. Chalmers (1977) reported a similar ammonium pattern of minimal early summer levels when vegetative production was typically at a maximum. Dissolved organic and total nitrogen levels, within the high marsh, were elevated and differed significantly from corresponding depth intervals at the low marsh and creekbank sampling sites; mean sampling date concentration ranged from 26.9 to 75.0uM/liter. Interstitial total phosphorus levels in the high marsh were significantly lower than low marsh levels and approximately equal to the creekbank region.

Nixon (1980) provides a qualitative summary of net annual nutrient tidal fluxes between tidal marshes and coastal waters; reviewed studies incorporated large geographic and salinity ranges. Methodoligies used to estimate water and nutrient fluxes varied, however, results indicate distinct patterns of nutrient transport. In summary, the investigations indicate that tidal marshes export dissolved organic nitrogen and total phosphorus, while importing oxidized inorganic nitrogen species (nitrate and nitrite). Flux patterns concerning ammonia were not as succinct, results indicate that the wetland may serve as a source or a sink.

Given the site specific varaibility of subsurface transport within a region and non-conservative nature of nutrient species, subsurface flux estimates (mass/time x area) were not calculated. However, the potential for import and export of dissolved nutrients between surface and interstitial waters of various wetland regions will be discussed. General patterns and relative contribution of nutrients to the marsh from surface waters, and vice-versa, were calculated as the difference between surface and interstitial water concentrations at a particular point in time (i.e. sampling dates). Interstitial water concentrations from the 20-40cm depth interval were used in the nutrient exchange potential determinations since it represented the most mobile interstitial water sampled. If these general exchange patterns are considered along with subsurface hydrodynamics within the various wetland regions (summarized in section 4), one is able to achieve a better understanding of the nutrient exchange potential between the wetland and adjacent surface water masses.

Dissolved oxidized inorganic forms of nitrogen (nitrate and nitrite) were imported throughout the sampling period by the three sampled regions (Figs. 5.13a, 5.14a, and 5.15a). Exception to this import only occured twice within the creekbank region. A strong seasonal pattern of nitrate import to the marsh was observed; high import values were associated with winter and early spring months when surface water nitrate levels reach a maximum. Nitrite is not included in the general discussion due to its negligible contribution to surface and interstitial water nitrogen pools. Ammonium flux, due to seepage, was predominantly from the wetland to surface waters. The high marsh region (Fig. 5.15b), appeard to have a greater ammonuim export potential than the low marsh flat (Fig. 5.14b) or creekbank (Fig. 5.13b) during the winter and spring seasons. The high marsh was a source of dissolved organic nitrogen throughout the entire study (Fig. 5.15c), while the low marsh flat (Fig. 5.14c) and creekbank (Fig. 5.13c) regions may best be characterized as a source during the winter, spring and summer months, and a potential sink during the fall season.

Examination of total dissolved nitrogen and phosphorus potential for exchange best exemplifies differences between the three general tidal wetland regions. The high marsh exhibited the potential to export dissolved total nitrogen (Fig. 5.15d) and phosphorus (Fig. 5.15e) throughout the study period. When water nutrient chemistry is coupled with subsurface hydrologic model results, it is evident that the high marsh exports total nitrogen and phosphorus to surface waters. As interstitial water drains from the high marsh to the saturated low marsh flat, surface ponding of the excess water is exposed to inundating waters of subsequent tides or lost through surface runoff to tidal creeks.

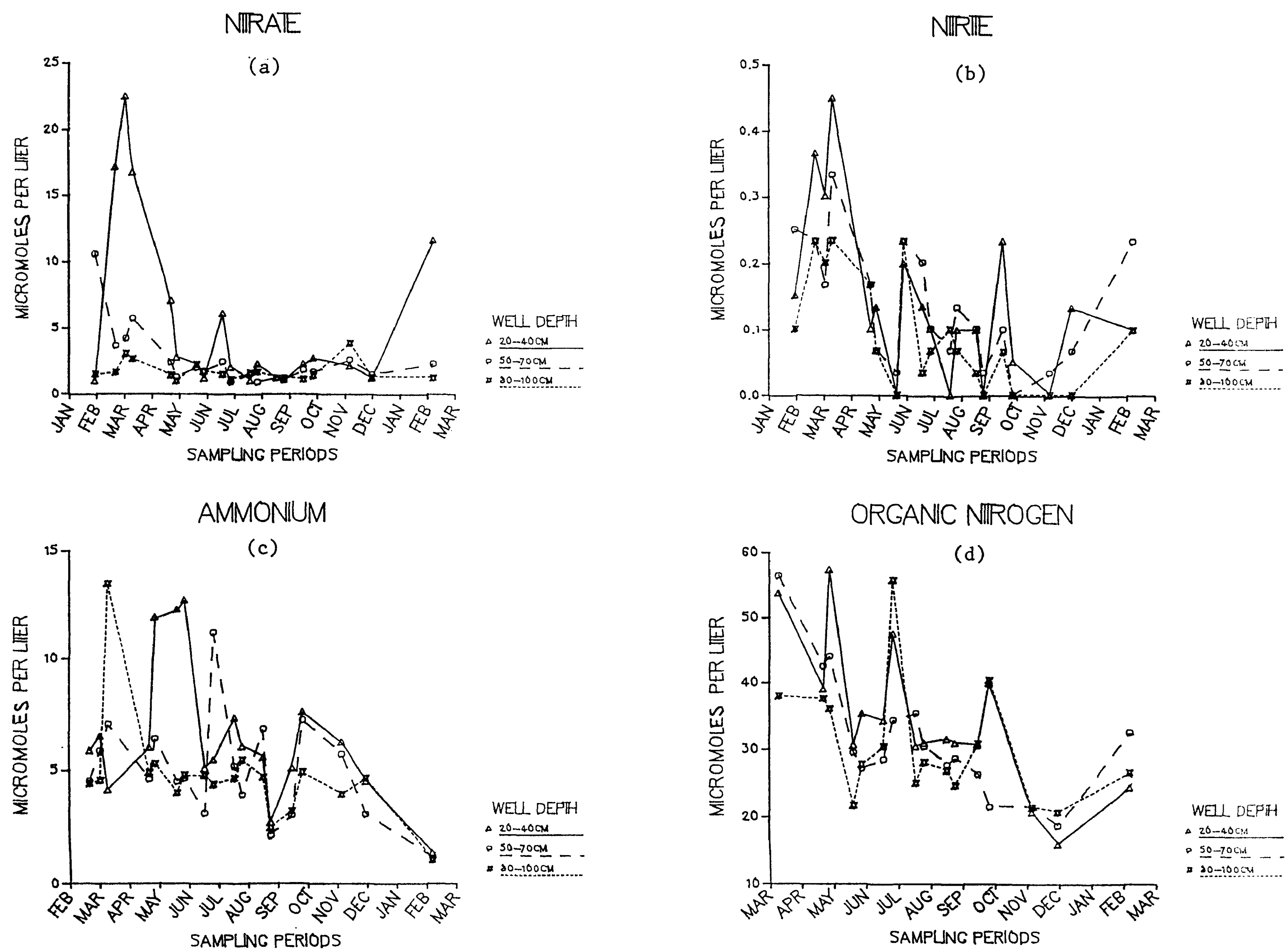
Nutrient exchange potential within the low marsh flat depicts a great variation with respect to dissolved total nitrogen. Potential total nitrogen exchange alternates between export during the late spring and summer months and import during the fall, winter, and early spring when inundating surface waters exhibit elevated levels of nitrate. The low marsh flat exibited the greastest magnitude of dissolved total phosphorus export potential compared to the high marsh and creekbank regions. Since lateral subsurface transport was minimal to non-existant within this region, the low marsh flat region may be considered as a resevoir of dissolved total phosphorus. This resevoir is unavailable for export or exchange with surface waters aside from molecular diffusion at the soil/water interface.

The potential for dissolved total nitrogen exchange between surface waters and the creekbank region followed a similar pattern as did the low marsh flat; export occuring in the spring and summer followed by import in the fall and winter months. However, the creekbank region differs from the low marsh flat by actively exchanging significant amounts of interstitial water with surface waters over a tidel cycle. Total phosphorus exchange between surface waters and the creekbank appear variable and minimal throughout the majority of the study. Elevated levels of total phosphorus within the creekbank interstitial water were responsible for the a typical exchange potential during March and April.

Jordan and Correll (1985) concluded that seepage was a primary export mechanism of nutrients from a brackish tidal wetland to adjacent surface waters. Authors reported that potential export of phosphate, ammonium, and dissolved organic nitrogen, from the wetland due to seepage, is 19-40, 17-23, and 46-68 percent respectively. Gardner (1976), investigating nutrient exchange within a salt marsh whose soil permeabilty was orders of magnitude less than reported by Jordan and Correll (1985), concluded that seepage was a minor nutrient transport mechanism as compared to molecular diffusion.

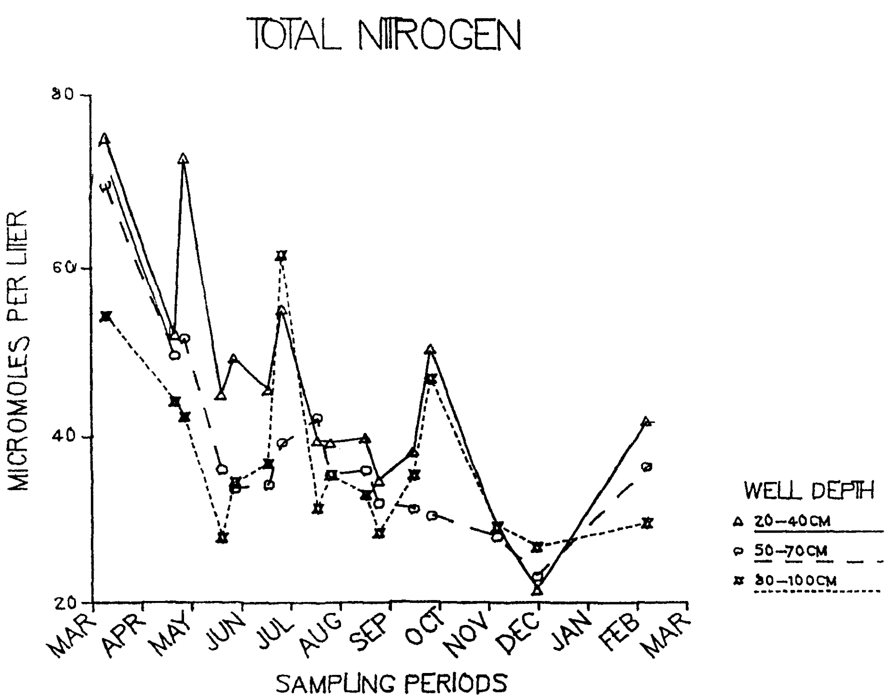
Results of this investigation show that seepage can explain general dissolved nutrient flux patterns between wetlands and coastal surface waters. Review of literature indicates that tidal wetlands import oxidized inorganic nitrogen, and export of dissolved organic nitrogen and total phosphorus, whereas the direction of ammonium transport was variable (Nixon, 1980). The importance of seepage as a nutrient transport mechanism between wetlands and surface is site specific between wetlands (Gardner, 1975; Jordan and Correll, 1985) and within a particular wetland. This study emphasizes the importance of advective flow as a solute transport mechanism within tidal freshwater wetlands and determination of a wetland nutrient budgets and flux determinations must incorporate this mechanism in addition to diffusion controlled processes.

Figure 5.1: Mean creekbank nutrient concentrations
as a function of time and well depth.
(a) nitrate, (b) nitrite,
(c) ammonium, (d) organic nitrogen

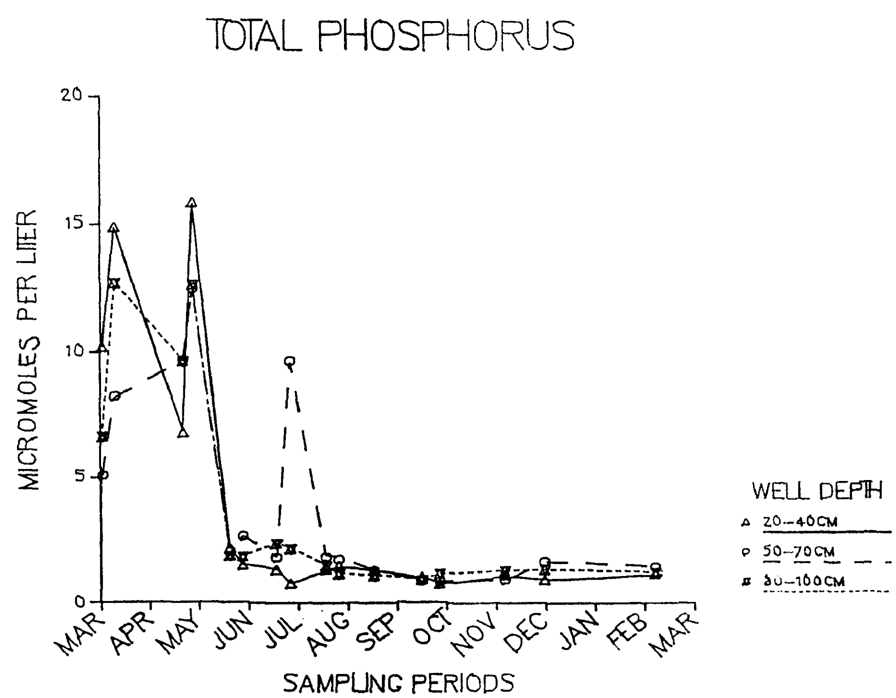


NITRATE
(a)
NITRITE
(b)
AMMONIUM
(c)
ORGANIC NITROGEN
(d)
MICROMOLES PER LITER
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

Figure 5.2: Mean creekbank nutrient concentrations
as a function of time and well depth.
(a) total dissolved nitrogen,
(b) total dissolved phosphorus.


TOTAL NITROGEN
MICROMOLES PER LITER
80
60
40
20
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

(a)


TOTAL PHOSPHORUS
MICROMOLES PER LITER
20
15
10
5
0
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

(b)

Figure 5.3: Mean low marsh nutrient concentrations
as a function of time and well depth.
(a) nitrate, (b) nitrite,
(c) ammonium, (d) organic nitrogen

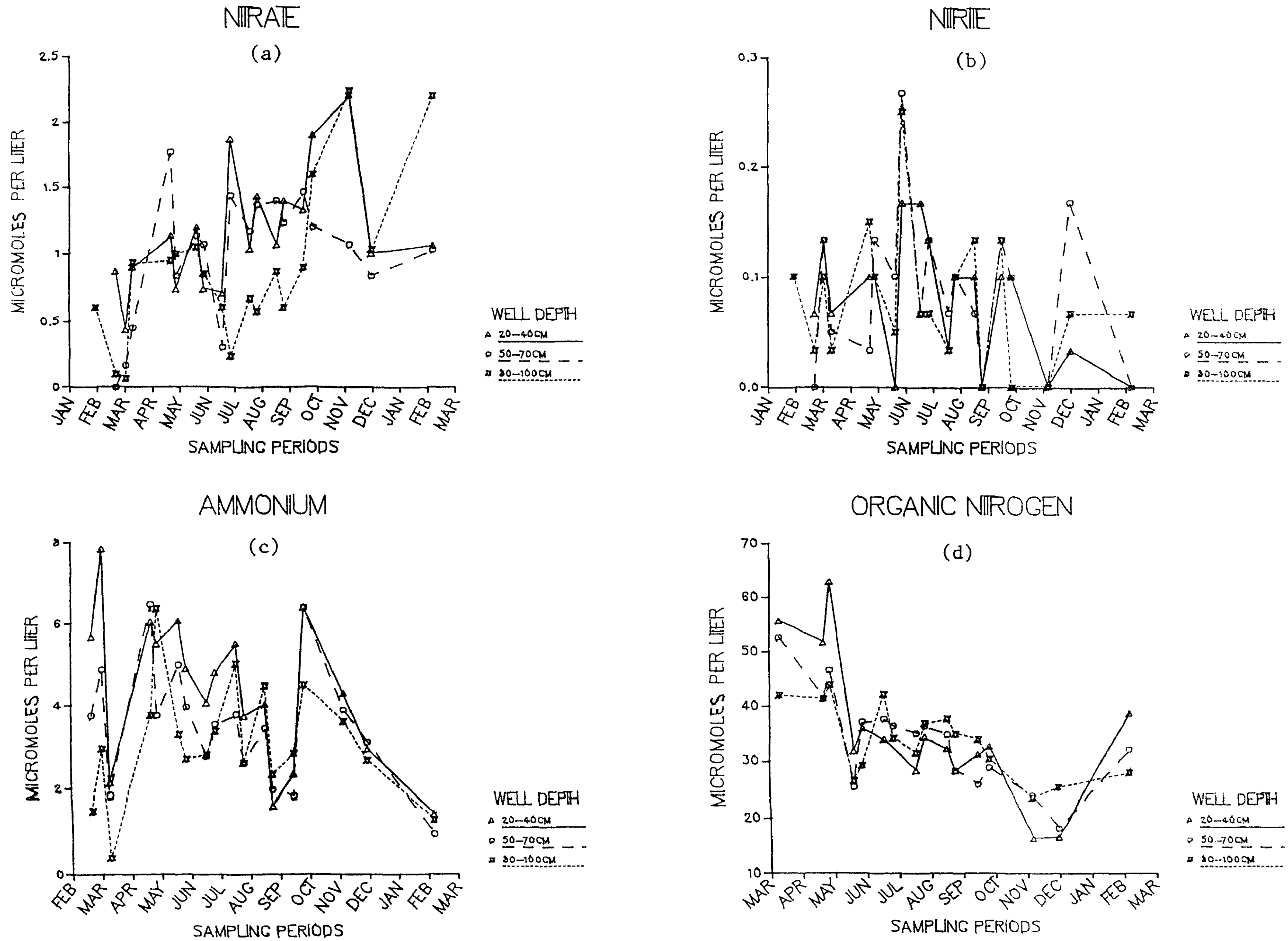
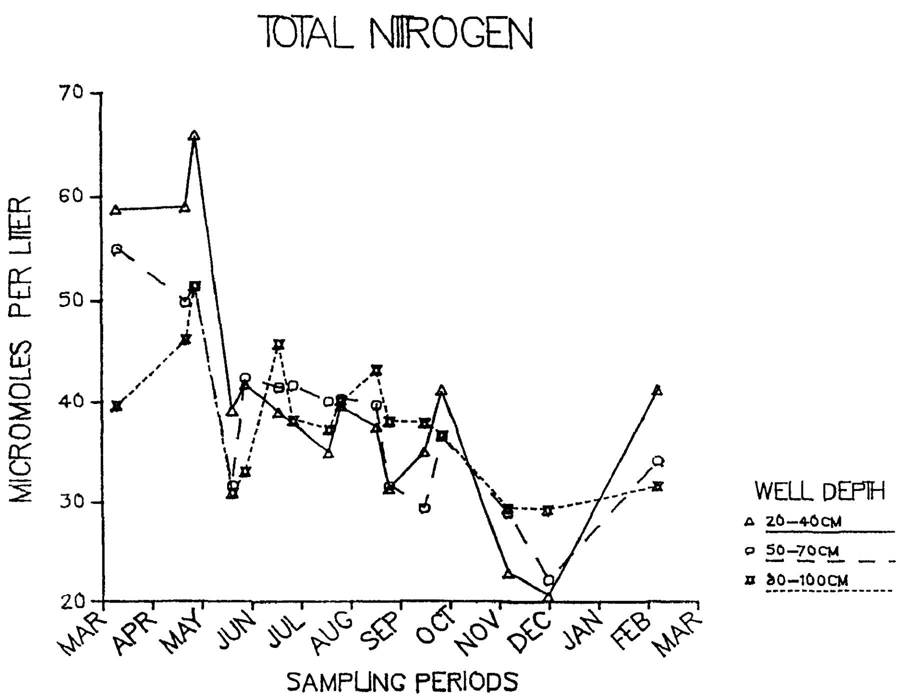
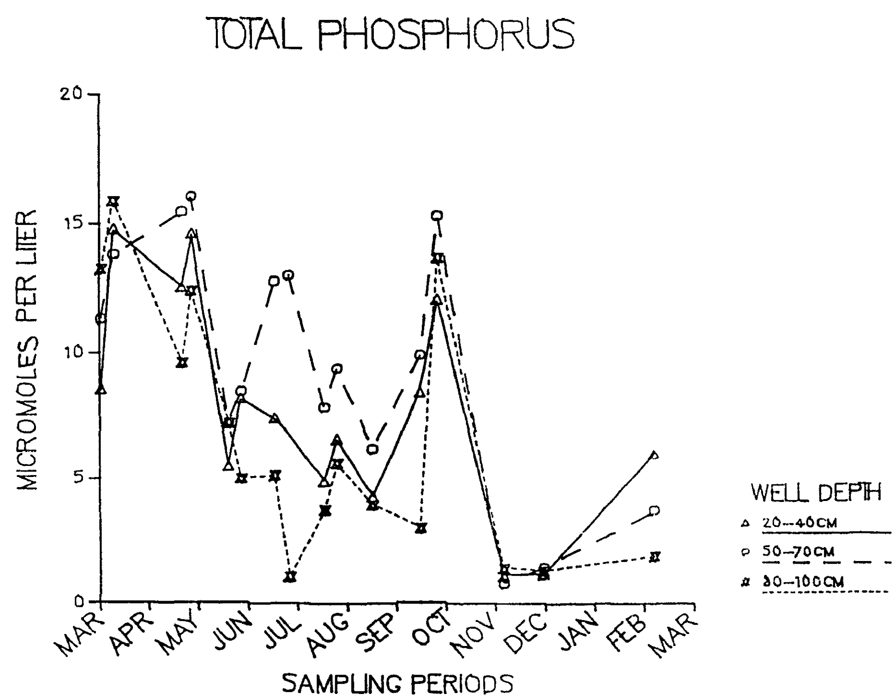

NITRATE
(a)
MICROMOLES PER LITER
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM
NITRITE
(b)
AMMONIUM
(c)
ORGANIC NITROGEN
(d)

Figure 5.4: Mean low marsh nutrient concentrations
as a function of time and well depth.
(a) total dissolved nitrogen,
(b) total dissolved phosphorus.

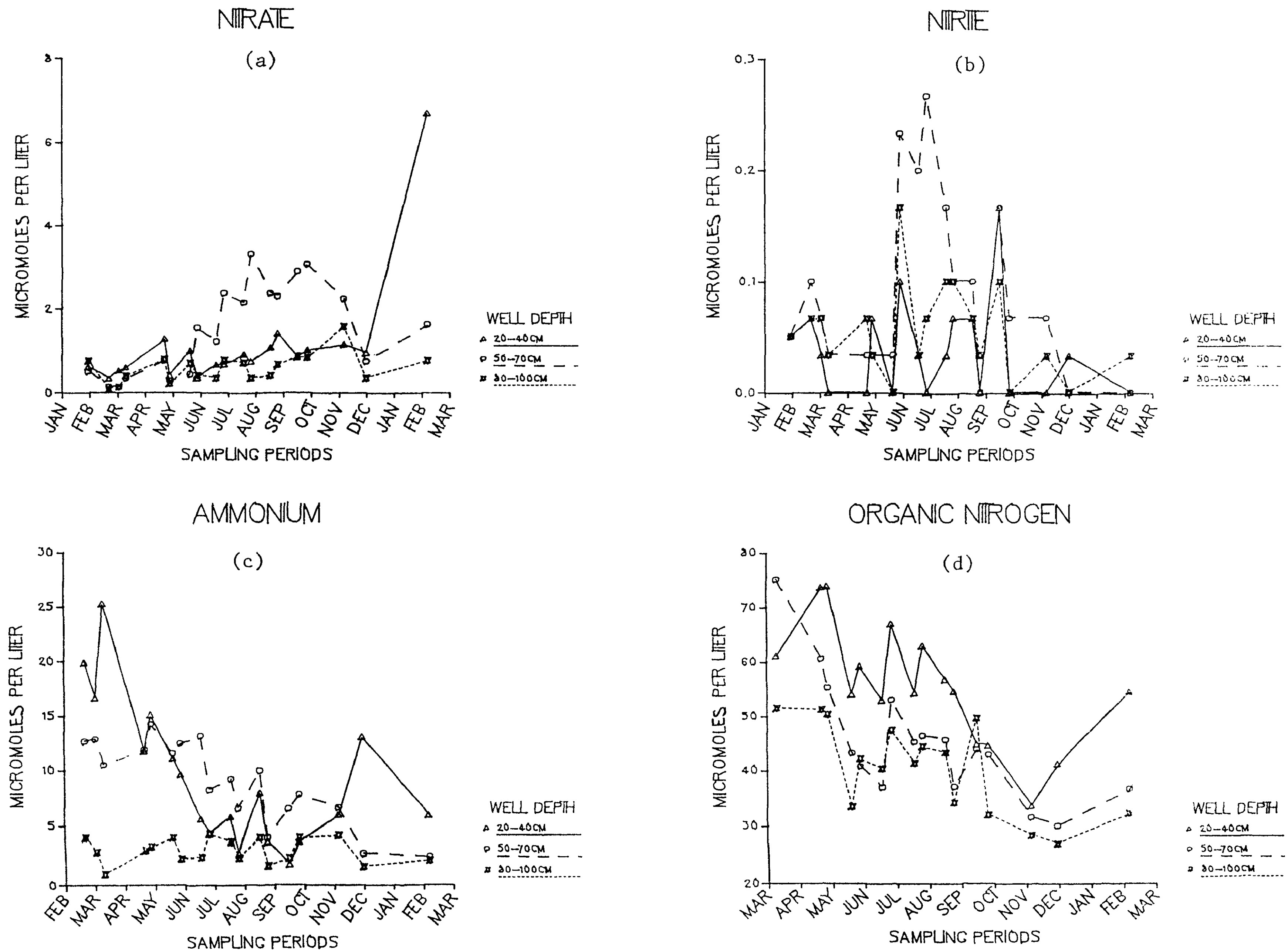

TOTAL NITROGEN
MICROMOLES PER LITER
70
60
50
40
30
20
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

(a)


TOTAL PHOSPHORUS
MICROMOLES PER LITER
20
15
10
5
0
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

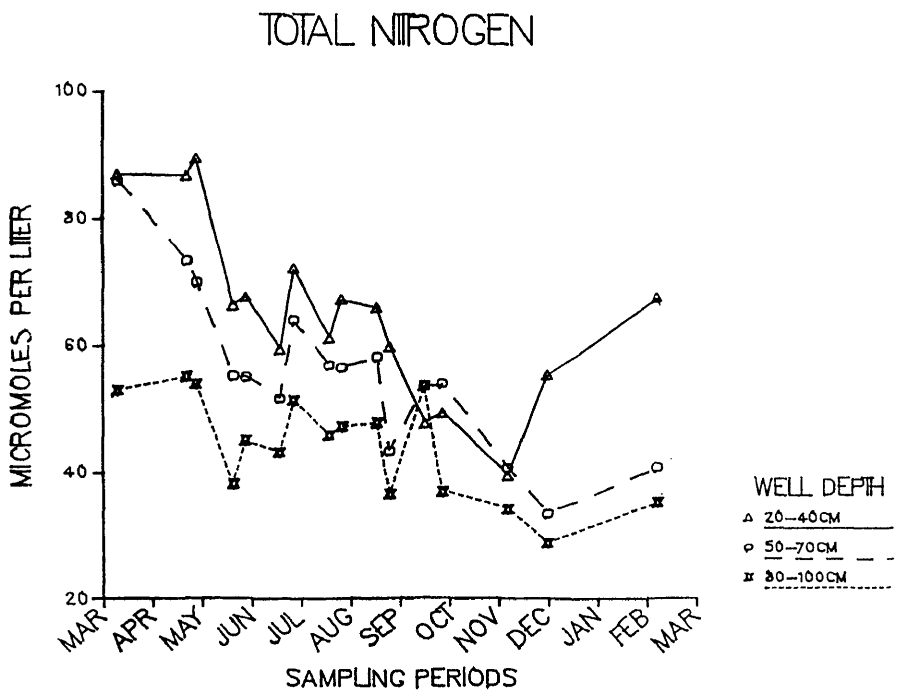
(b)

Figure 5.5: Mean high marsh nutrient concentrations
as a function of time and well depth.
(a) nitrate, (b) nitrite,
(c) ammonium, (d) organic nitrogen

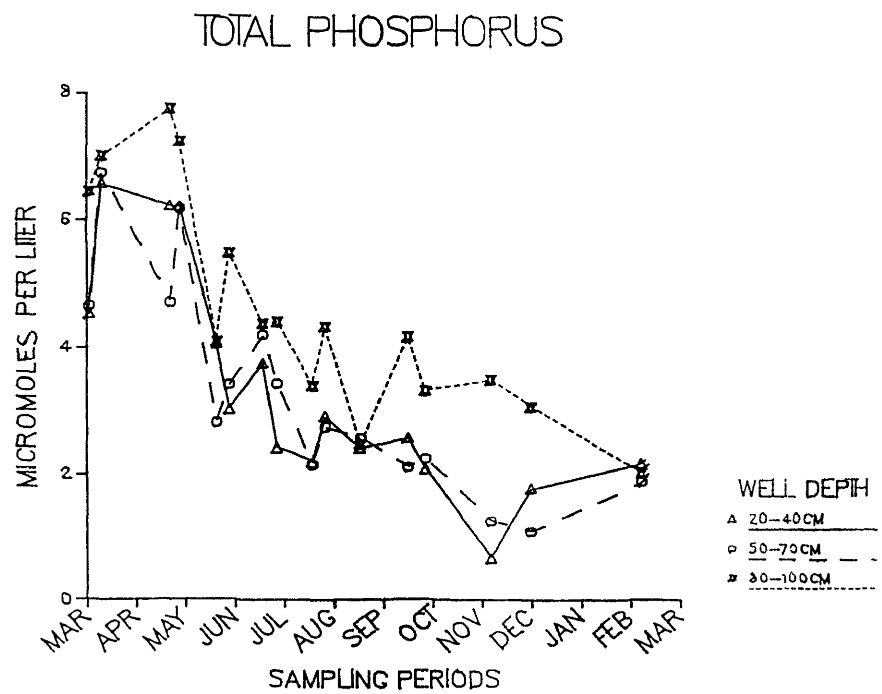


NITRATE
(a)
MICROMOLES PER LITER
SAMPLING PERIODS
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC JAN FEB MAR
WELL DEPTH
20–40CM
50–70CM
80–100CM
NITRITE
(b)
MICROMOLES PER LITER
SAMPLING PERIODS
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC JAN FEB MAR
WELL DEPTH
20–40CM
50–70CM
80–100CM
AMMONIUM
(c)
MICROMOLES PER LITER
SAMPLING PERIODS
FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC JAN FEB MAR
WELL DEPTH
20–40CM
50–70CM
80–100CM
ORGANIC NITROGEN
(d)
MICROMOLES PER LITER
SAMPLING PERIODS
MAR APR MAY JUN JUL AUG SEP OCT NOV DEC JAN FEB MAR
WELL DEPTH
20–40CM
50–70CM
80–100CM

Figure 5.6: Mean high marsh nutrient concentrations
as a function of time and well depth.
(a) total dissolved nitrogen,
(b) total dissolved phosphorus.


TOTAL NITROGEN
MICROMOLES PER LITER
100
80
60
40
20
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

(a)


TOTAL PHOSPHORUS
MICROMOLES PER LITER
8
6
4
2
0
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS
WELL DEPTH
20–40CM
50–70CM
80–100CM

(b)

Figure 5.7: Mean creek water nutrient concentrations
as a function of time.
(a) nitrate, (b) nitrite,
(c) ammonium, (d) organic nitrogen

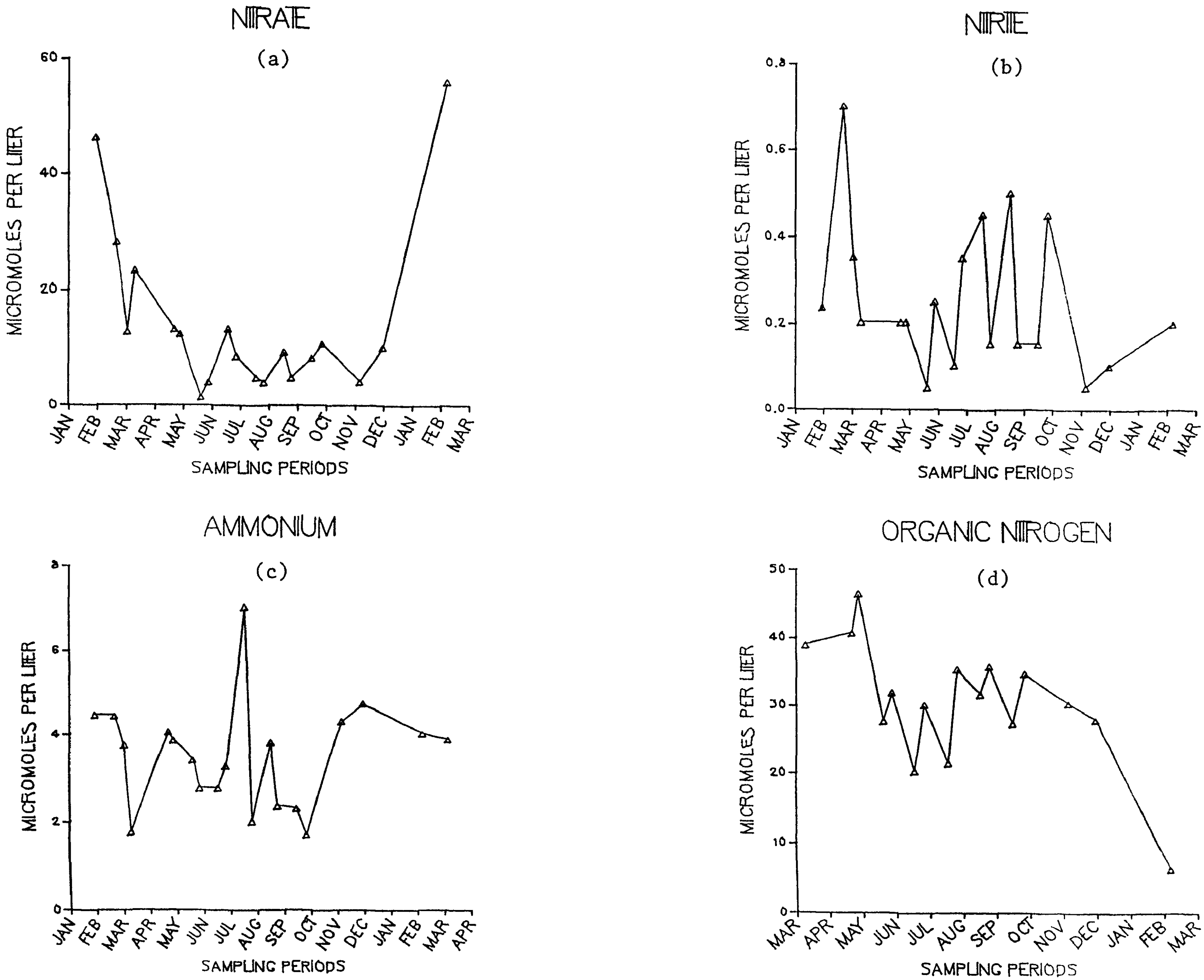
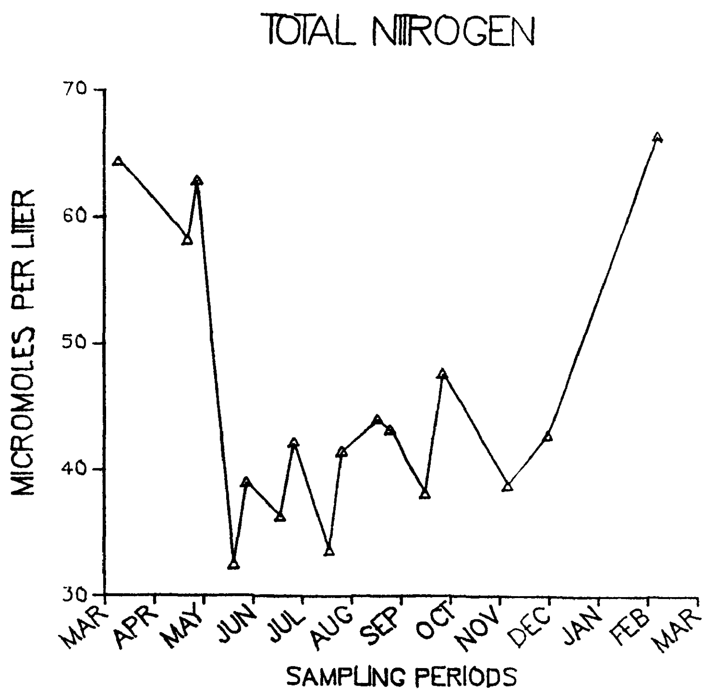
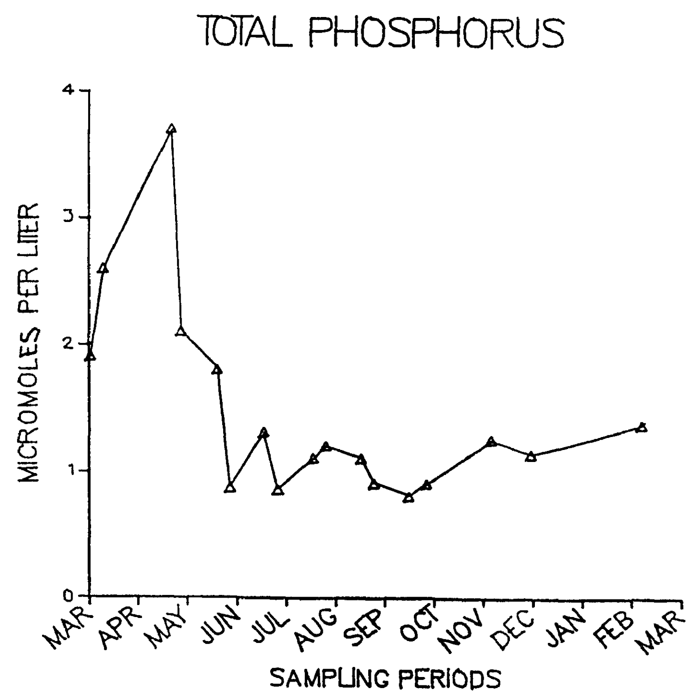

NITRATE
(a)
MICROMOLES PER LITER
SAMPLING PERIODS
NITRITE
(b)
MICROMOLES PER LITER
SAMPLING PERIODS
AMMONIUM
(c)
MICROMOLES PER LITER
SAMPLING PERIODS
ORGANIC NITROGEN
(d)
MICROMOLES PER LITER
SAMPLING PERIODS

Figure 5.8: Mean creek water nutrient concentrations
as a function of time.
(a) total dissolved nitrogen,
(b) total dissolved phosphorus.

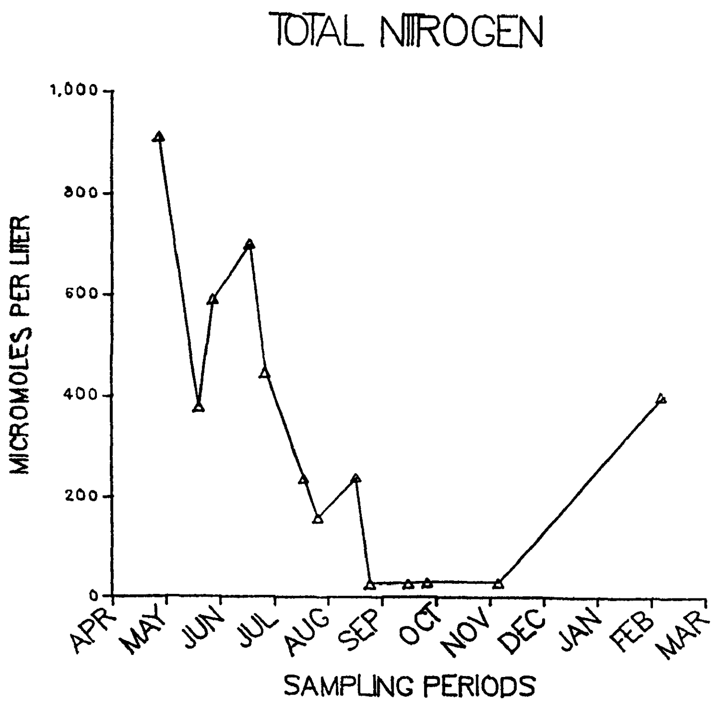

TOTAL NITROGEN
MICROMOLES PER LITER
70
60
50
40
30
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS

(a)

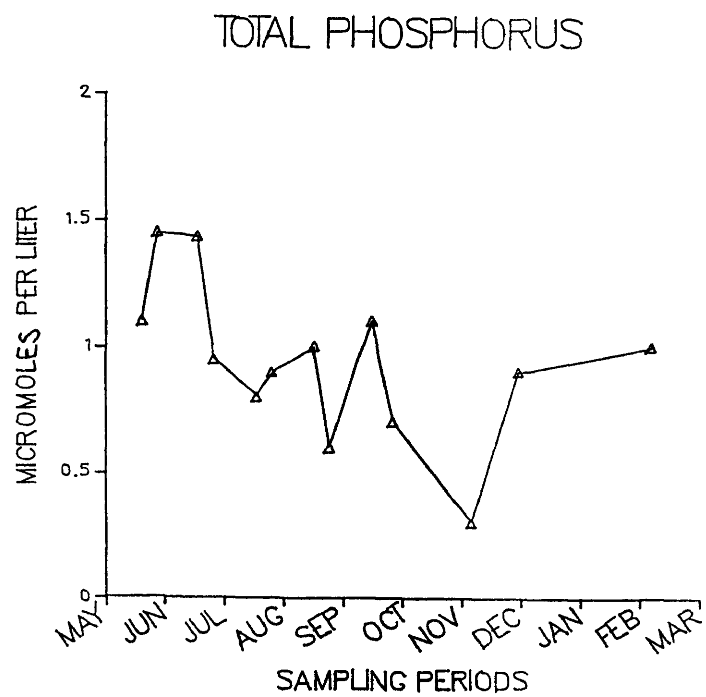

TOTAL PHOSPHORUS
MICROMOLES PER LITER
4
3
2
1
0
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS

(b)

Figure 5.9: Mean upland groundwater nutrient
concentrations as a function of time.
(a) total dissolved nitrogen,
(b) total dissolved phosphorus.

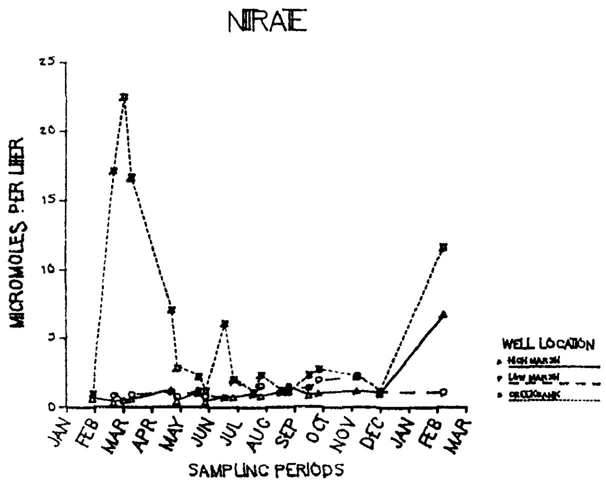

TOTAL NITROGEN
1,000
800
600
400
200
0
MICROMOLES PER LITER
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS

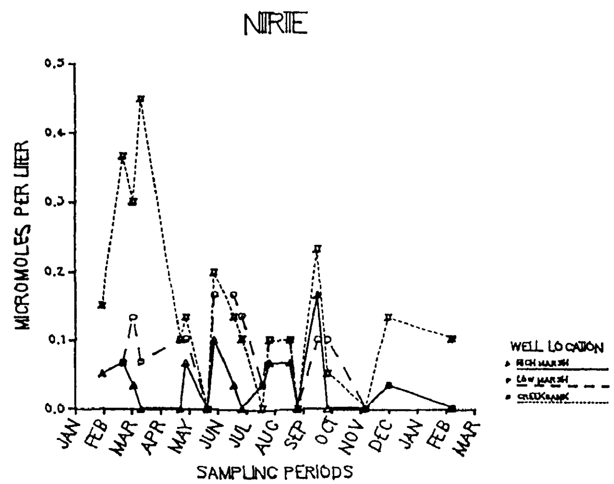
(a)

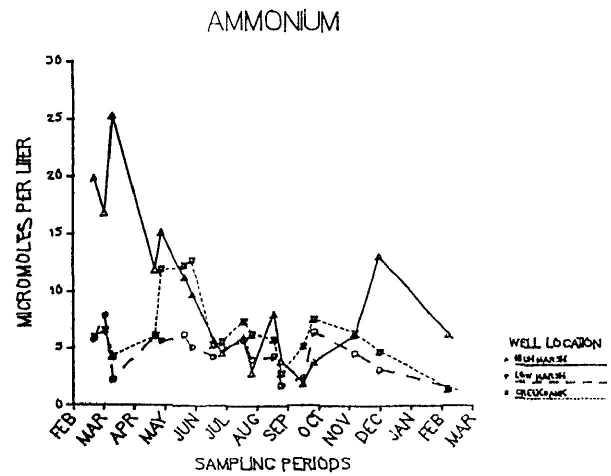

TOTAL PHOSPHORUS
2
1.5
1
0.5
0
MICROMOLES PER LITER
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
JAN
FEB
MAR
SAMPLING PERIODS

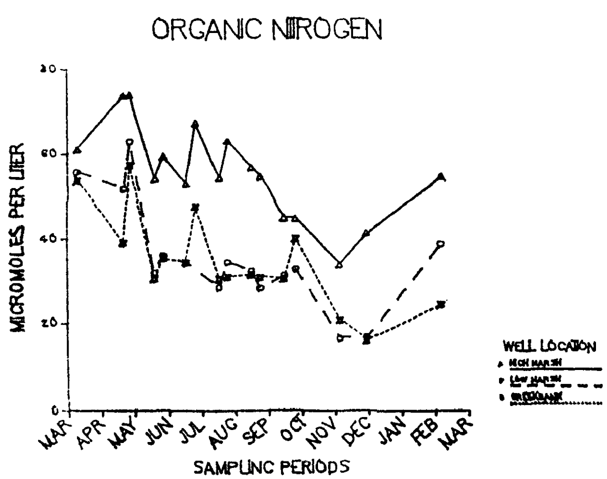
(b)

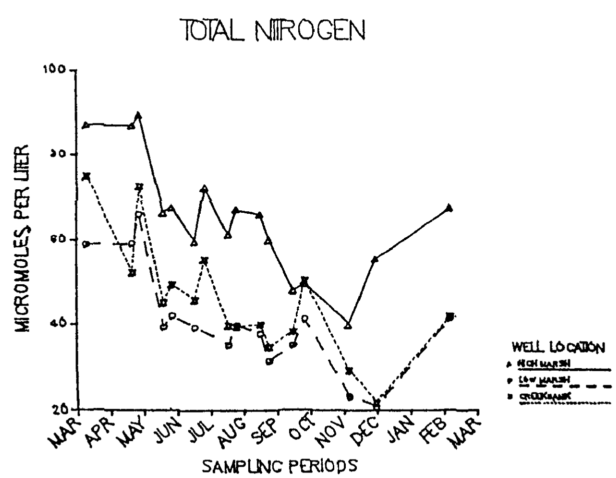
Figure 5.10: Mean 20-40cm depth interval pore water nutrient concentrations as a function of transect location and time.
(a) nitrate, (b) nitrite, (c) ammonium,
(d) organic nitrogen, (e) total nitrogen,
(f) total phosphorus.


NITRATE
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
(a)


NITRITE
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
(b)


AMMONIUM
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
(c)


ORGANIC NITROGEN
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
(d)


TOTAL NITROGEN
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
(e)

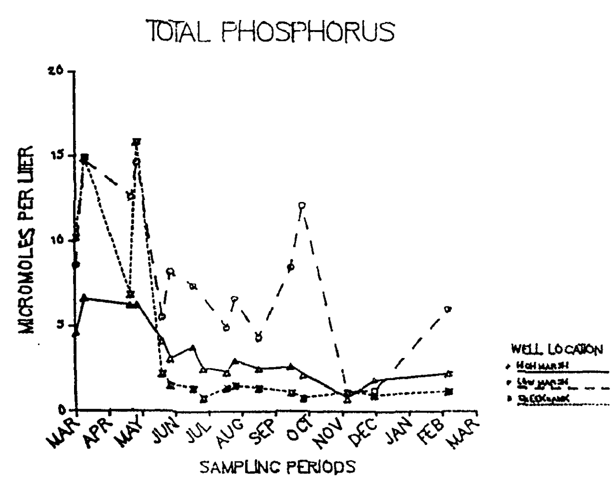

TOTAL PHOSPHORUS
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
(f)

Figure 5.11: Mean 50-70cm depth interval pore water nutrient concentrations as a function of transect location and time.
(a) nitrate, (b) nitrite, (c) ammonium,
(d) organic nitrogen, (e) total nitrogen,
(f) total phosphorus.

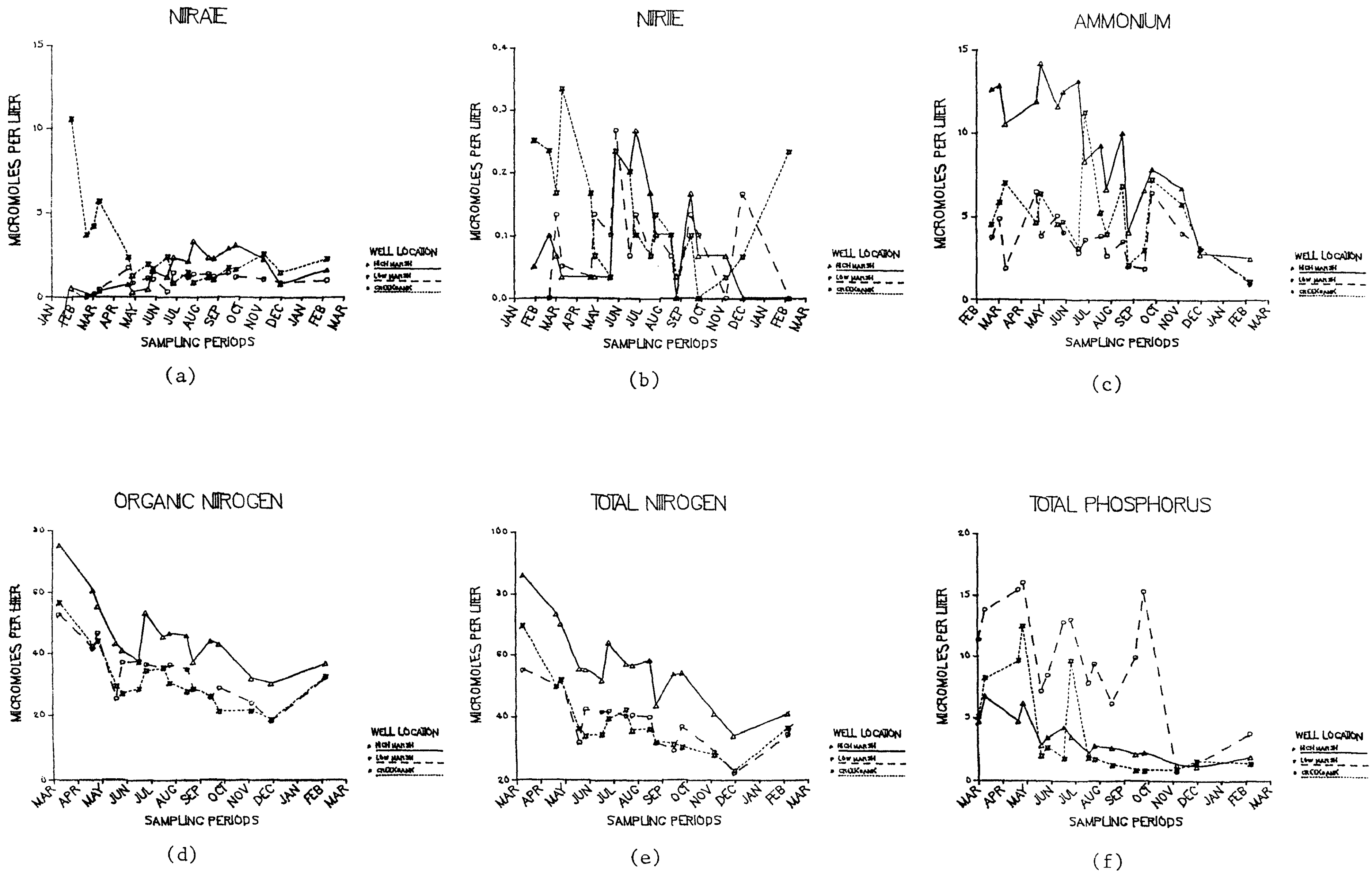

NITRATE
NITRITE
AMMONIUM
ORGANIC NITROGEN
TOTAL NITROGEN
TOTAL PHOSPHORUS
MICROMOLES PER LITER
SAMPLING PERIODS
WELL LOCATION
HIGH MARSH
LOW MARSH
CREEKBANK
(a)
(b)
(c)
(d)
(e)
(f)

Figure 5.12: Mean 80-100cm depth interval pore water nutrient concentrations as a function of transect location and time.
(a) nitrate, (b) nitrite, (c) ammonium,
(d) organic nitrogen, (e) total nitrogen,
(f) total phosphorus.

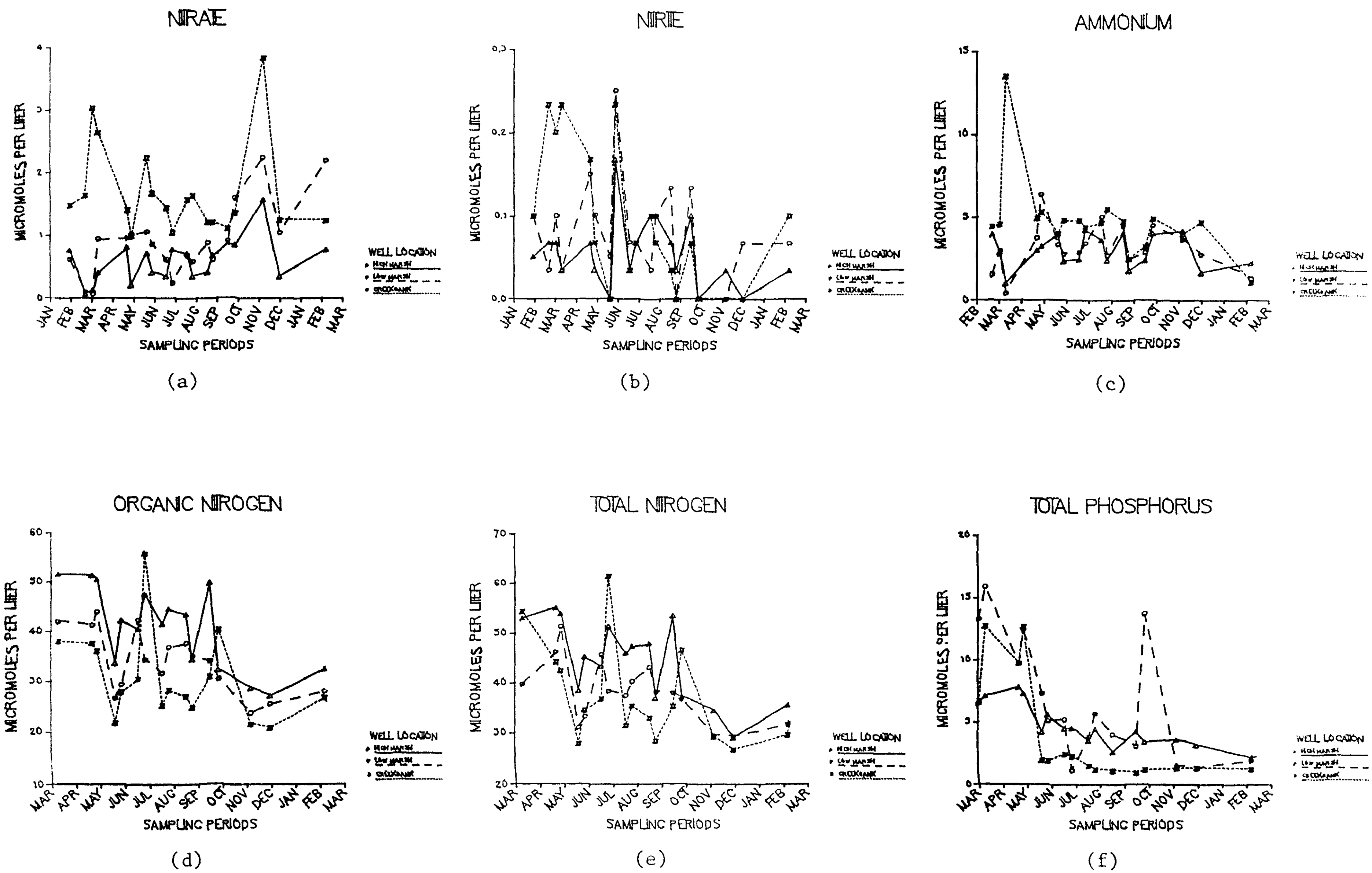
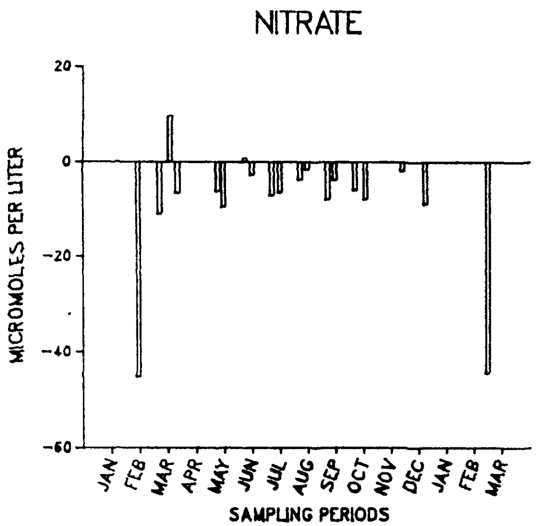
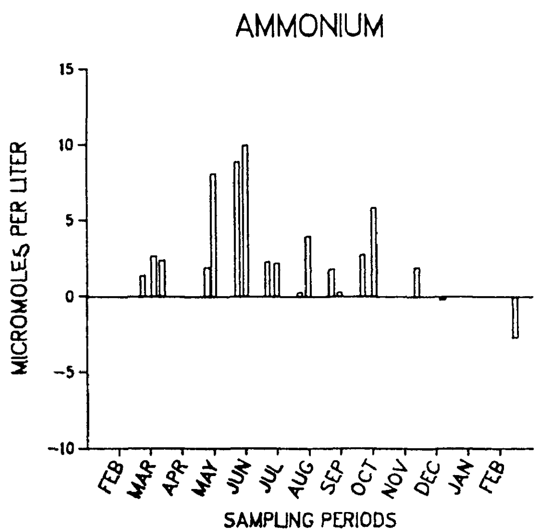
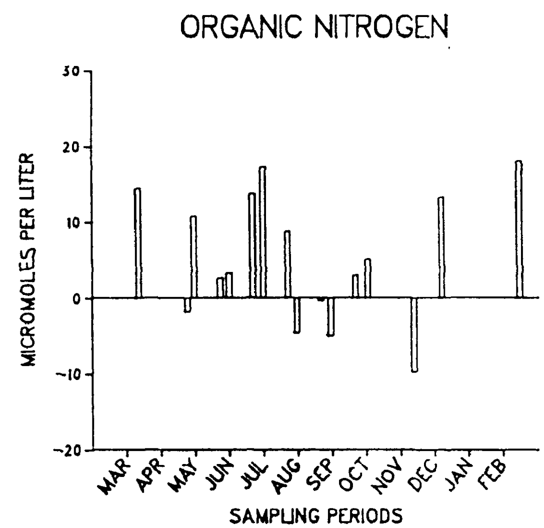

NITRATE
MICROMOLES PER LITER
SAMPLING PERIODS
(a)
NITRITE
MICROMOLES PER LITER
SAMPLING PERIODS
(b)
AMMONIUM
MICROMOLES PER LITER
SAMPLING PERIODS
(c)
ORGANIC NITROGEN
MICROMOLES PER LITER
SAMPLING PERIODS
(d)
TOTAL NITROGEN
MICROMOLES PER LITER
SAMPLING PERIODS
(e)
TOTAL PHOSPHORUS
MICROMOLES PER LITER
SAMPLING PERIODS
(f)
WELL LOCATION

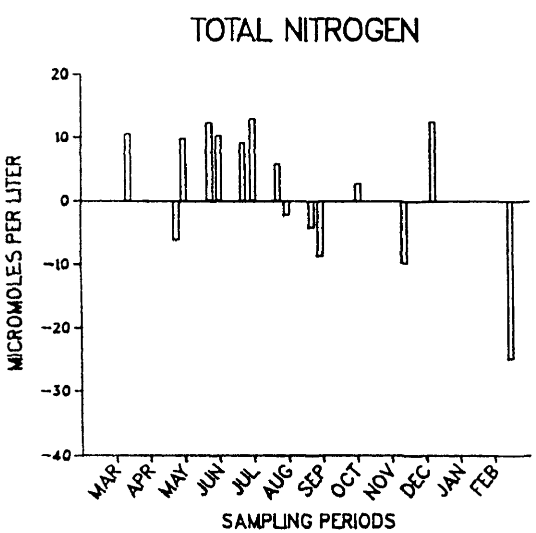
Figure 5.13: Potential nutrient exchanges between the creekbank and adjacent surface waters. (a) nitrate, (b) nitrite, (c) ammonium, (d) organic nitrogen, (e) total nitrogen, (f) total phosphorus.

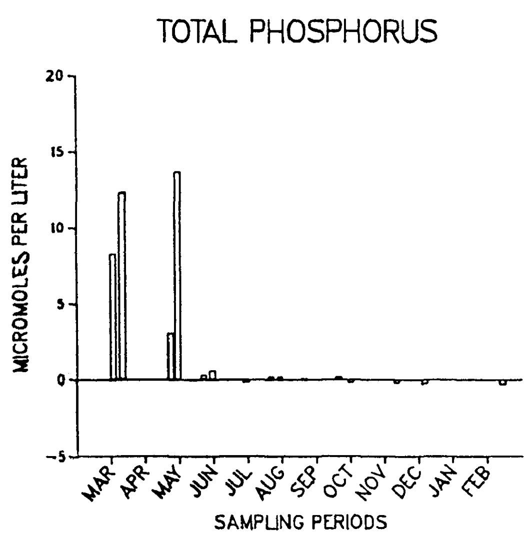
Positive values indicate an export from the marsh to surface waters.







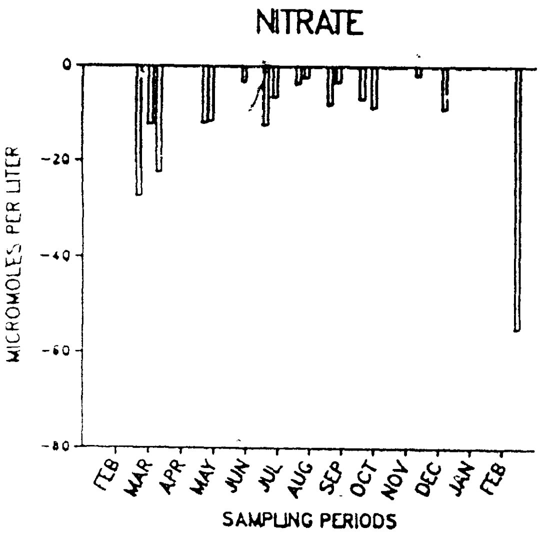


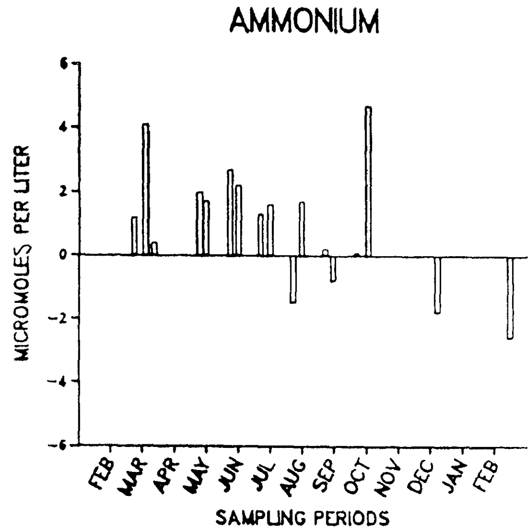


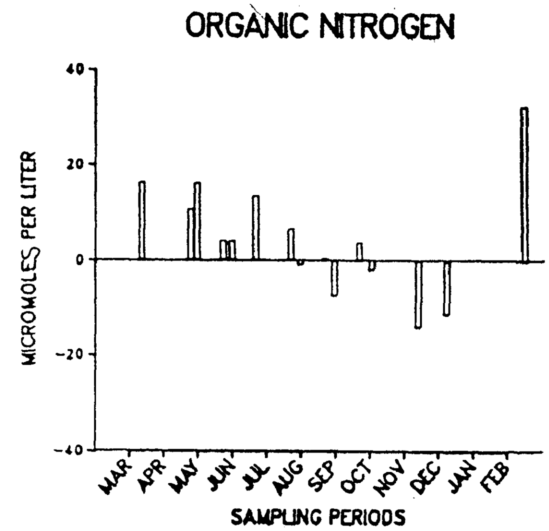
POSITIVE VALUES INDICATE
AN EXPORT FROM THE MARSH

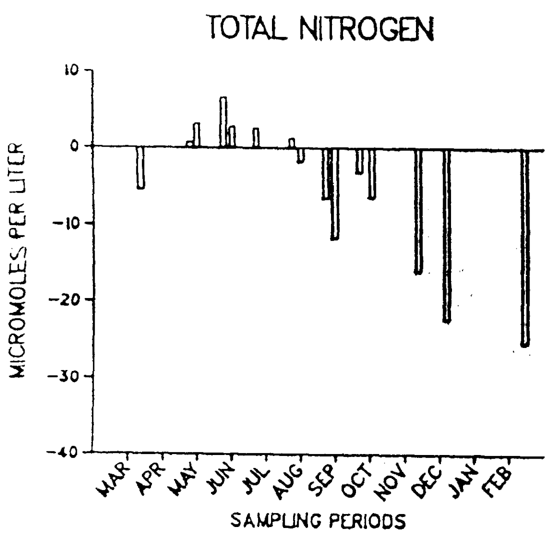
Figure 5.14: Potential nutrient exchanges between the low marsh and adjacent surface waters. (a) nitrate, (b) nitrite, (c) ammonium, (d) organic nitrogen, (e) total nitrogen, (f) total phosphorus.

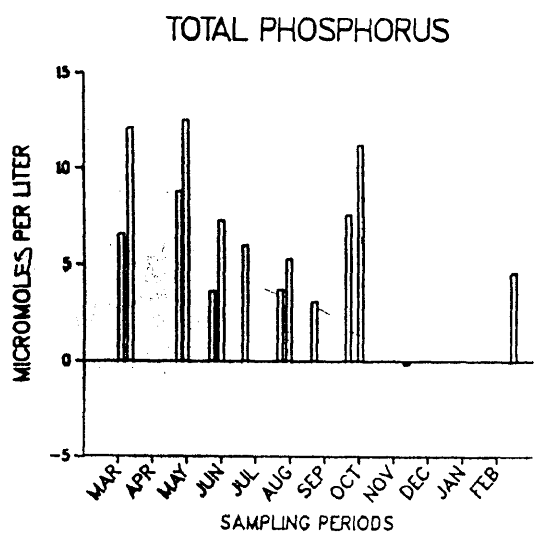
Positive values indicate an export from the marsh to surface waters.







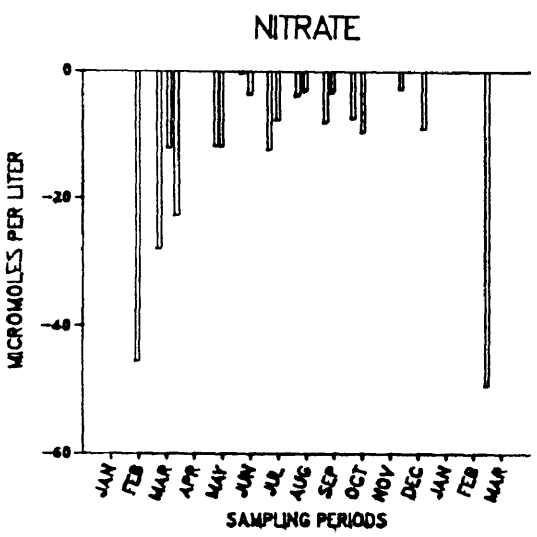


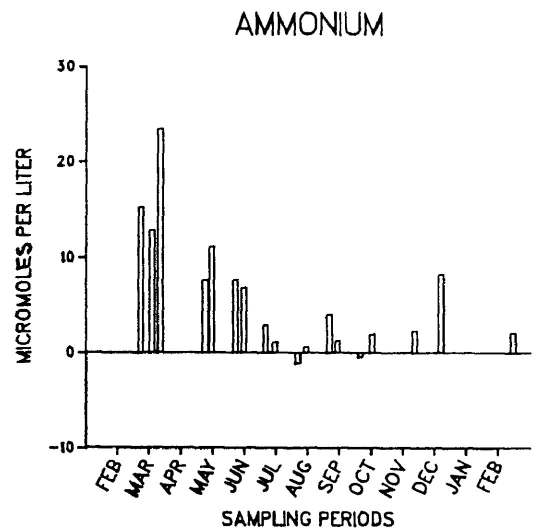


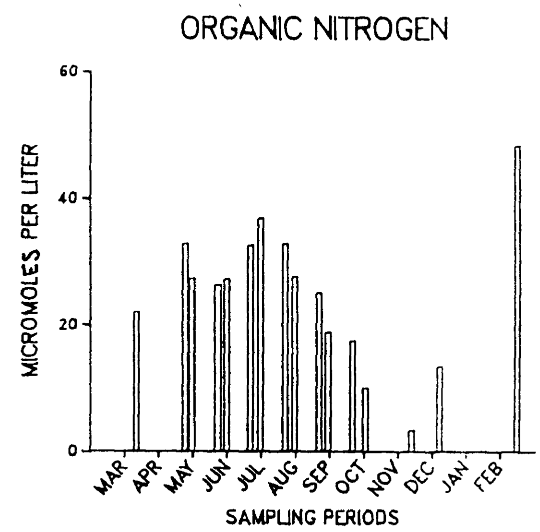
POSITIVE VALUES INDICATE
AN EXPORT FROM THE MARSH

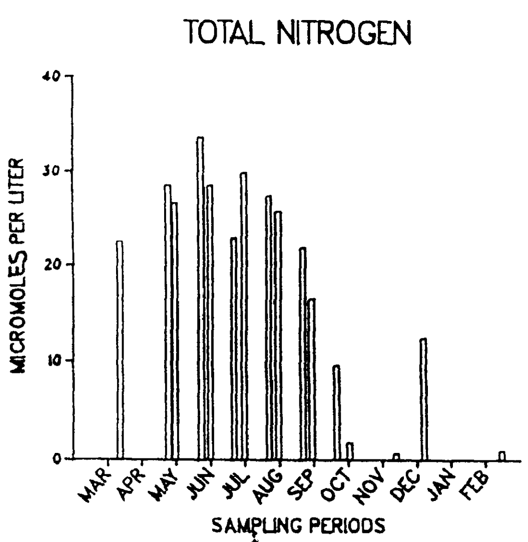
Figure 5.15: Potential nutrient exchanges between the high marsh and adjacent surface waters. (a) nitrate, (b) nitrite, (c) ammonium, (d) organic nitrogen, (e) total nitrogen, (f) total phosphorus.

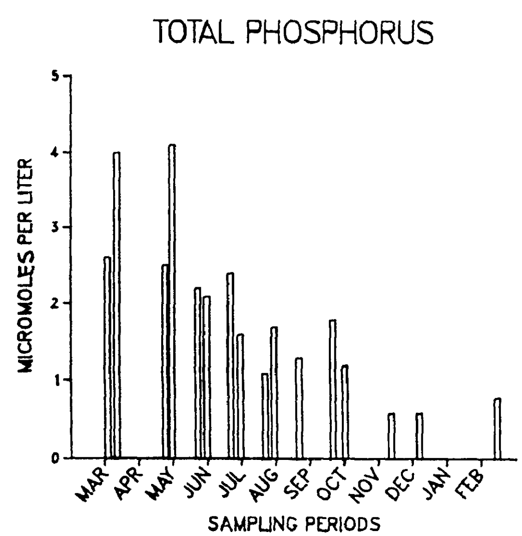
Positive values indicate an export from the marsh to surface waters.











POSITIVE VALUES INDICATE
AN EXPORT FROM THE MARSH

Table 5.1: Descriptive statistics of creekbank pore water nutrient concentrations.

Concentrations are expressed as uM/liter.

Species	Depth	Mean	Std.Error	N
NO3	20-40cm	5.4	1.0	54
	50-70cm	2.4	0.4	55
	80-100cm	1.7	0.2	56
NO2	20-40cm	0.1	<0.1	54
	50-70cm	0.1	<0.1	55
	80-100cm	0.1	<0.1	56
NH4	20-40cm	6.4	0.6	50
	50-70cm	4.9	0.5	53
	80-100cm	4.7	0.6	53
DON	20-40cm	33.4	1.8	41
	50-70cm	31.8	1.7	45
	80-100cm	30.0	1.7	45
TN	20-40cm	44.0	2.2	42
	50-70cm	38.2	1.9	45
	80-100cm	36.5	2.0	45
TP	20-40cm	3.9	0.9	42
	50-70cm	3.8	0.7	45
	80-100cm	3.8	0.7	45

Table 5.2: Descriptive statistics of low marsh pore water nutrient concentrations.

Concentrations are expressed as uM/liter.

Species	Depth	Mean	Std.Error	N
NO3	20-40cm	1.1	0.1	52
	50-70cm	1.0	0.1	51
	80-100cm	0.9	0.1	50
NO2	20-40cm	0.1	<0.1	52
	50-70cm	0.1	<0.1	51
	80-100cm	0.1	<0.1	50
NH4	20-40cm	4.3	0.4	51
	50-70cm	3.5	0.3	51
	80-100cm	3.2	0.3	47
DON	20-40cm	34.8	2.0	42
	50-70cm	33.3	1.4	43
	80-100cm	33.6	1.4	39
TN	20-40cm	40.4	2.0	43
	50-70cm	38.0	1.5	43
	80-100cm	38.1	1.5	41
TP	20-40cm	7.5	0.7	43
	50-70cm	9.0	0.9	43
	80-100cm	6.3	1.1	41

Table 5.3: Descriptive statistics of high marsh pore water nutrient concentrations.

Concentrations are expressed as uM/liter.

Species	Depth	Mean	Std.Error	N
NO3	20-40cm	1.1	0.2	56
	50-70cm	1.5	0.3	56
	80-100cm	0.6	0.1	56
NO2	20-40cm	0.1	<0.1	56
	50-70cm	0.1	<0.1	56
	80-100cm	0.1	<0.1	56
NH4	20-40cm	9.4	1.2	54
	50-70cm	9.0	1.2	54
	80-100cm	2.9	0.2	54
DON	20-40cm	55.8	2.3	45
	50-70cm	45.3	2.3	48
	80-100cm	40.2	1.5	46
TN	20-40cm	65.5	3.0	45
	50-70cm	55.7	3.1	48
	80-100cm	43.7	1.5	46
TP	20-40cm	3.4	0.3	44
	50-70cm	3.2	0.3	48
	80-100cm	4.6	0.4	47

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Vita

William Glendon Reay

Address

Department of Biology
Virginia Polytechnic Institute and
State University
Blacksburg, Virginia 24061
Telephone (703) 231-5602

Personal Data

Birth Date: February 6, 1959
Marital Status: Married

Education

Ph.D.,	Microbiology, expected 1991, VPI&SU, Blacksburg, Virginia Research Topic: Effects of Submarine Groundwater Discharge and Land-Use Patterns on Sediment Nutrient Flux and Microbial Populations. Major Professor: Dr. George M. Simmons, Jr.
M.A.,	Marine Science, 1989, The College of William and Mary, Virginia Institute of Marine Science, Williamsburg, Virginia Research Topic: Subsurface Hydrodynamics and Nutrient Exchange within a Extensive Tidal Freshwater Wetland. Major Professor: Dr. Carl H. Hershner
B.S.,	Biology, 1983, George Mason University, Fairfax, Virginia Research Topic: Effects of Urbanization on Stream Biota and Water Quality. Major Professor: Dr. Ted Bradley

Professional Employment

Present:	Graduate Research Assistant, Dept. of Biology, VPI&SU, Blacksburg, Virginia
Summer 1989:	Teaching Assistant, Bermuda Biological Station for Research, St. Georges, Bermuda

1988:	Research Assistant, Chesapeake Research Consortium, Gloucester Point, Virginia.
1984-88:	Graduate Research Assistant, Virginia Institute of Marine Science, College of William and Mary, Gloucester Point, Virginia
1983:	Executive Intern, Coastal States Organization, Washington, D.C.