A GEOGRAPHIC INFORMATION SYSTEM ANALYSIS OF SUBMARINE GROUNDWATER DISCHARGE ON THE EASTERN SHORE OF VIRGINIA

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ABSTRACT

Recent research has indicated that submarine groundwater discharge (SGWD) occurs in the Chesapeake Bay, and that discharged groundwater can be contaminated with nutrients and toxic substances. Although discharge has been measured directly on a local level, a convenient method for determining the spatial distribution of SGWD on a large scale has not yet been devised. Although current watershed mass balance approaches have been used to estimate gross SGWD, this method cannot identify areas at high risk for contaminant input. A family of innovative computer mapping programs called Geographic Information Systems (GIS) have emerged from the software industry that serve not only as mapping tools but also as databases designed to manage spatial data. The GIS approach allows straightforward manipulation and presentation of data that is spatially related. In the research presented here, a GIS was applied to the problem of large-scale determination of SGWD and nutrient loads for the Eastern Shore of Virginia. Computerized data layers including land use, hydraulic gradient and soil permeability were used to create a GIS model of SGWD on Virginia's Eastern Shore. The model was used to predict spatial distribution of SGWD as well as the nutrient loading from nearshore agriculture and on-site waste treatment technologies. While sufficient field data for reliable calibration and verification of the model did not exist, the available data and the output of the GIS model were consistent. The methodology developed predicts that 90% of the impact from nearshore land use occurs in the southwestern quadrant of the Eastern Shore. In addition, the model determined locations where urban land use (septic tanks) may present a nutrient and coliform contamination danger. The model identified areas susceptible to contamination through discharge to a resolution of about 33 meters, showing that GIS can function as a local as well as regional management tool. The GIS framework is also expandable. The model can be modified to accommodate new data as research continues into the phenomenon of submarine groundwater discharge. This synthesis represents the first large-scale groundwater discharge analysis of the Eastern Shore and exists as part of an ongoing investigation to characterize groundwater-surface water interactions of the Eastern Shore.

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I. INTRODUCTION

The Chesapeake Bay is a 166,000 km^2 embayment on the Eastern Seaboard of the United States (Cronin, 1971; cited by Reay and Simmons, 1992). Its salt waters are fed by watersheds from Virginia, Pennsylvania, Maryland, West Virginia, New York and Delaware. The volume of the Chesapeake Bay is 60.5 billion cubic meters and the Bay has 7400 km of shoreline (Gillelan and Mackins, 1983; as cited by Reay and Simmons, 1992). Major ports include Baltimore, Philadelphia, and Hampton Roads. Recreational and commercial fishermen rely on the waters of the Chesapeake, and the Bay is an important home to a variety of marine and avian life. Because of a wide variety of recreation, economic and natural resource issues, great emphasis has been given to restoring and maintaining the water quality in the Chesapeake Bay.

Since the late 1970's, an integrated regional effort has been underway to halt the degradation incurred to the Chesapeake Bay by anthropogenic activities. Researchers concentrated their efforts on studying the Chesapeake Bay system, and limiting nutrient and toxic input to the bay from point and non-point sources. In 1983, the United States Environmental Protection Agency published recommendations for remediation of the Bay (USEPA, 1983). Two of the major problems identified in this report were nitrogen and phosphorus input, which result in eutrophication. The report recommended nitrogen and phosphorus input be controlled by waste water treatment technology and non-point pollution control. Although a few sources (including agricultural field runoff) were considered in the study, the effect of the non-point source of submarine groundwater discharge (SGWD) was not acknowledged. Once an unrecognized source of contaminant loading into the Chesapeake Bay, the significance and processes of SGWD have only begun to be understood and appreciated. Although major steps have been taken toward controlling point and some non-point sources, the lack of understanding of the phenomenon of groundwater seepage and aquifer contamination has prevented action being taken towards controlling this non-point source. It has even been suggested (Dillaha, 1990) that some methods currently used to control non-point pollution, such as sedimentation ponds and porous pavement, may actually increase pollution through SGWD.

Submarine groundwater discharge occurs from any aquifer whose waters are in intimate contact with a body of open water with lower hydraulic head. The groundwater in the aquifer flows downgradient and through the sediments under the influence of the higher upland head *h* as shown in Figure 1. Johannes (1980) suggested this condition was prevalent around the perimeter of most bodies of water. The discharge may support many nearshore ecosystems, and is not a problem in and of itself. Submarine groundwater discharge only becomes a problem if the discharged groundwater is contaminated by agricultural or urban land use. Contaminant transport by SGWD is complicated by chemical and biological transformations, such as ammonification or denitrification, that take place as groundwater passes through the benthic (bottom) zone to the water. Thus the quality of the discharged water is not necessarily that of the groundwater. Recent studies have detailed the role of SGWD as a medium for nutrient input to proximal surface waters, and it has been suggested that toxic input may also be a concern (Reay and Simmons, 1992). Studies are now under way to determine the impact of pesticide application on SGWD (Hubbard-personal communication, 1993). Because SGWD has been shown to be impacting the water of the Chesapeake Bay (Reay and Simmons, 1992) a strategy should be developed that addresses both diffuse source and subsurface nitrogen inputs to the Chesapeake Bay (Staver and Brinsfield, 1990).

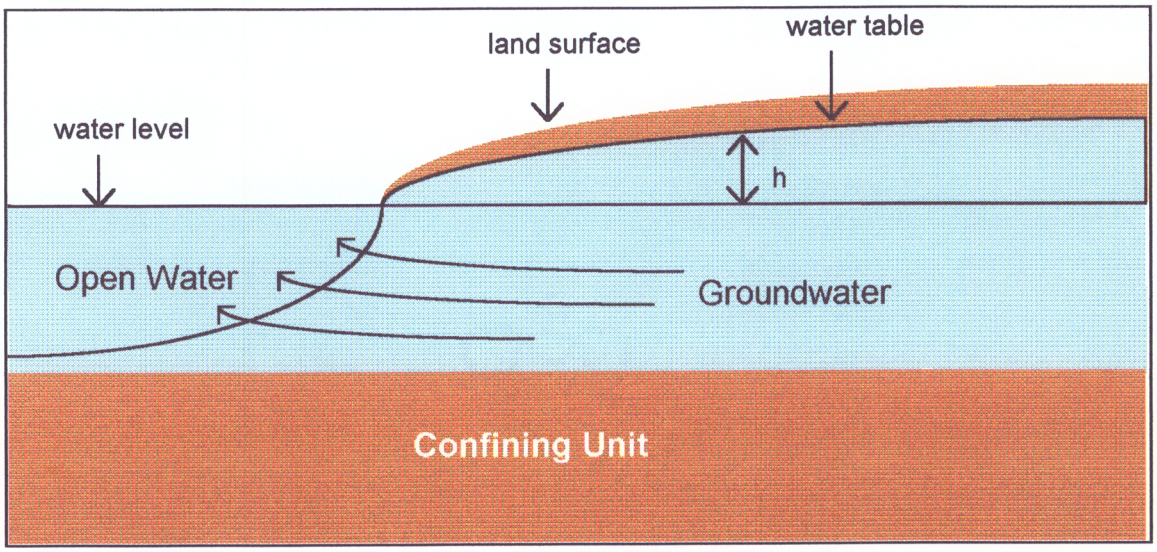


Figure 1. Schematic of SGWD

A correlation between upgradient land use and discharged groundwater quality has been found in the Chesapeake Bay, indicating that fertilizers and on-site waste treatment methods adversely affect down-gradient surface water. Land use near the Bay is typically agricultural and residents frequently use on-site waste treatment technology (*i.e.*, septic systems) with little or no riparian zone buffering the Bay. This land use configuration has lead to contamination of the surficial aquifer with oxidized nitrogen from fertilizer, as well as ammonium and phosphate from septic systems. This contaminated groundwater will move downgradient to be discharged in the nearby surface water body. The cumulative impact of this discharge over the Chesapeake Bay shoreline has been shown to be significant (Simmons, 1989), and because discharge rates and nutrient fluxes are highly variable, a method is needed to determine the effects of SGWD on a spatial level. From the management perspective, a valid spatial analysis of the effects of SGWD is necessary to find out which areas are most impacted by this phenomenon as well as to determine where to allocate resources to best decrease the threat of contamination from discharge.

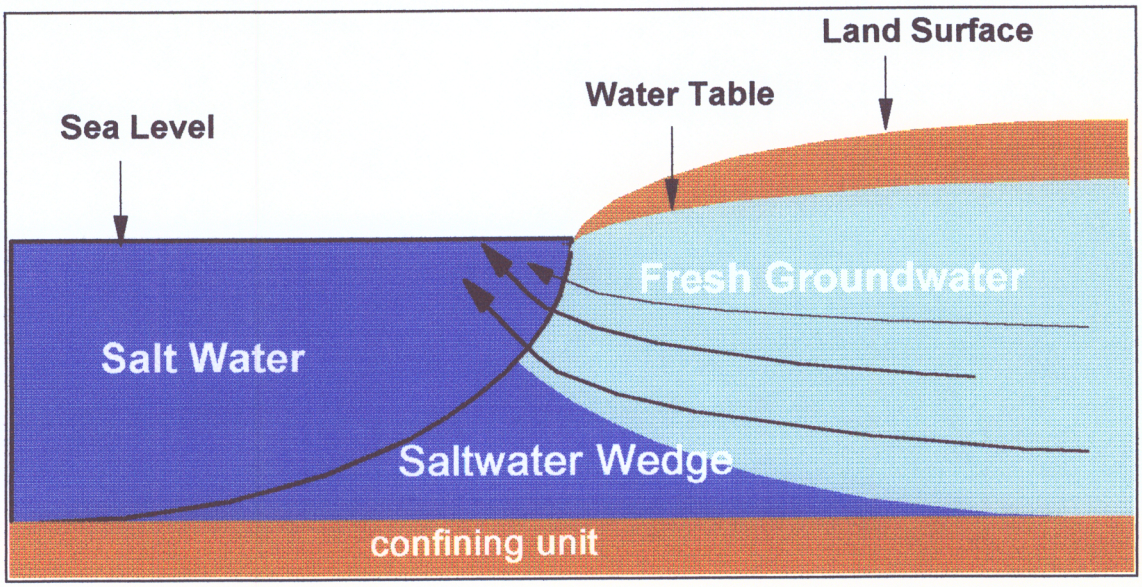


Figure 2. SGWD in marine environments

When SGWD occurs along salt waters such as the Chesapeake Bay, the groundwater discharge area is forced closer to the shore by the underlying dense sea water as shown in Figure 2. Thus, along the Eastern Shore the bulk of the discharge is thought to occur within 100 meters of shoreline (Reay, 1993). The rate of discharge is dependent

locally upon the ability of the soil to transmit water as well as the driving force of the local hydraulic gradient. The analysis of discharge in marine environments is hampered, however, by tidal and wave action that disrupt the steady-state conditions of the hydraulic gradient required for many groundwater-surface water interaction models. It is thought that in marine environments the majority of discharge takes place near low tide, when the local hydraulic gradient is steepest (Reay, 1993).

Two approaches typically have been taken to quantify SGWD. The first is a mass balance of water around the surficial aquifer and the second is the use of calculated hydraulic head in potential flow theory with a finite differencing scheme to determine outflow. Neither of these methods allow simple analysis of nearshore SGWD. The water mass balance approach, while easy to apply, has no spatial component, and elements of the hydrologic system can be highly variable or unknown. Numerical methods do incorporate a spatial component, but the required data are frequently unavailable and these schemes can be too complex for large-scale analysis. A different method incorporating both the simplicity of the mass balance approach and the spatial characteristics of the finite element method would be helpful in the analysis of SGWD.

Geographic Information Systems (GIS) facilitate analysis and presentation of spatially related data. These computer programs can be thought of as a database for geographic modeling where each feature is assigned attribute codes in database format. A GIS combines all the features for manipulating the attributes in database fashion with the ability to manipulate attributes in relation to spatial features. One key feature of modeling with a GIS approach is that the framework set up for this problem can be enhanced, taking different contaminants into account, and modifications can easily be made to the model as greater understanding of the geology and physical processes become available.

Spatial data such as soil permeability and large-scale hydraulic gradients that will allow analyses of the problem of SGWD are available for much of Virginia (Shanholtz *et al.*, 1989). A methodology was developed and applied to the Eastern Shore of Virginia with GIS data available from the state and digitized layers from various other resources with the objective of analyzing nearshore SGWD. This model avoids the complicated finite element approach, retains the valuable spatial analysis component, and remains

simple enough that the methodology can easily be transplanted to a different region of the bay or country.

A model of this type is not without limitations, however. Because of the high local variability of SGWD it is impossible to predict relative discharge and contamination with certainty. In addition, large-scale quantitative analysis is difficult because of the limited availability of seepage data. While the model cannot predict quality nor quantity of SGWD without sufficient calibration, it can predict likelihood of discharge and contamination on a scale from regional (county) to local (a particular inlet). Other limitations (such as scale and distance problems) are necessarily introduced by the methodology of the GIS model. These sources of error are detailed in Chapter V. In addition, other influences such as biological processes and riparian zones are not considered because of lack of data for these factors. The GIS approach can include these factors as data become available.

There is a method of point readings of seepage, provided by a seepage meter inserted in nearshore sediments. These meters are typically placed directly onto the sediments from the shoreline to 10m-50m offshore. Once in place, the meters trap groundwater passing through the sediments, giving a measurement of groundwater flux and collecting a sample for laboratory analysis. Seepage data is available for several sites on the Eastern Shore, and while the data were not exhaustive enough for a full calibration, they did characterize the functioning of the model as well as provide a rough estimate of quality and quantity of nearshore groundwater discharge.

Although groundwater discharge is becoming recognized as a significant means of nutrient enrichment, no method has been developed to evaluate spatial distribution of SGWD on a large scale. Such a method can serve as a guide to future sampling, or identify trouble spots for management. The method presented in this research does not require computationally intensive programming nor estimates of the hydrologic cycle components, yet retains an important spatial element. The GIS method complements rather than replaces existing techniques for measuring SGWD.

The objectives of this research are:

1. Examine the feasibility of a GIS-based approach to the problem of SGWD.

2. Locate shore areas on the Eastern Shore subject to potentially high SGWD.

3. Locate shore areas on the Eastern Shore subject to potentially high nutrient loading through SGWD.

4. Locate shore areas on the Eastern Shore subject to potentially high coliform loading through SGWD.

5. Use available field data to estimate performance of the model.

6. Use the GIS model to gain an understanding of the spatial distribution of SGWD on the Eastern Shore, and examine its potential as a management tool.

II. LITERATURE REVIEW

History of the Problem

Although the Chesapeake had long been in decline, there was no integrated effort to study and respond to the water quality problems associated with the Bay until 1975. At that time, Congress directed the United States Environmental Protection Agency (USEPA) to study the problems of the Chesapeake Bay and recommend remedial action (USEPA, 1988). In 1977 the Bay Study identified three major areas of concern: nutrient enrichment, toxic substance introduction, and a decline in submerged aquatic vegetation. Since then, steps have been taken in the Chesapeake Bay watershed to reduce nutrient enrichment including non-point pollution control and enhanced municipal waste water treatment facilities. The Chesapeake Bay agreements of 1983 and of 1987 between the states and localities of the Chesapeake Bay drainage basin are expected to positively impact the Chesapeake Bay (USEPA, 1988). Although the steps taken toward remediation of the Chesapeake Bay have had beneficial effects, the hydrologic and ecologic systems are not completely understood, in particular the role of submarine groundwater discharge (Simmons, 1989).

Interaction of Saline & Fresh Water

Saltwater - freshwater interactions are complicated by several factors. First, tidal action and wave action tend to disrupt the steady-state groundwater flow system (Riedl *et al.*, 1972). Furthermore, the dynamics of mixing between two fluids with differing densities creates a complicated flow regime, especially in unsteady tidal conditions. Analytical descriptions of the saltwater-freshwater interface, while only approximations, are useful in understanding the nature of SGWD and groundwater-surface water interactions.

While there is debate about the time and origin of the first mathematical description of the freshwater-saltwater interface, credit is generally given to the nineteenth century scientists Ghyben and Herzberg (Reilly and Goodman, 1985). This simple formulation

given below can provide a reasonable estimate of the location of the freshwater saltwater interface for a surficial aquifer.

$$z = \frac{\rho_f}{\rho_s - \rho_f} h \qquad (1)\ \text{(Reilly and Goodman, 1985)}$$

Where:

z = depth from sea level to interface
ρ_S = density of salt water
ρ_f = density of fresh water
h = hydraulic head of aquifer

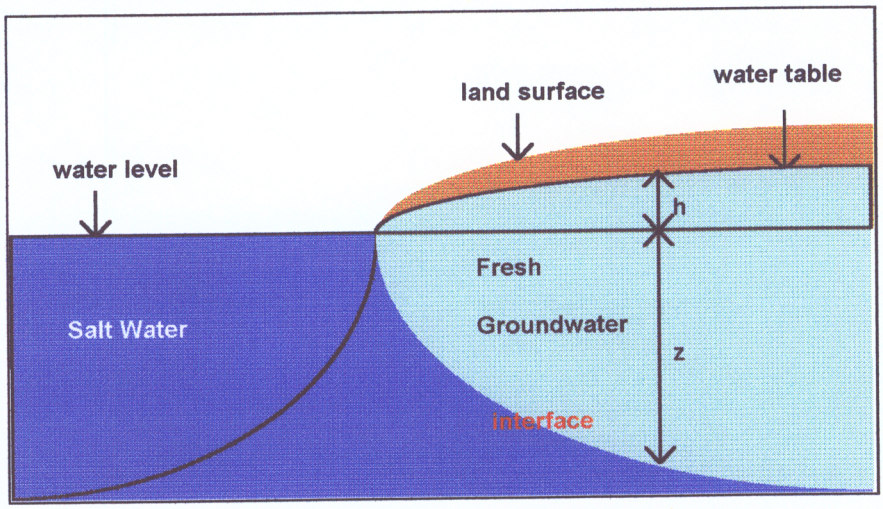


Figure 3. The Ghyben-Herzberg principle

Although this description is adequate for an approximation of the location of the freshwater-saltwater interface, it is limited by the no mixing and static case assumptions (Dupuit assumption). Under these assumptions, there can be no SGWD because the thickness of the freshwater is zero at the shoreline, as shown in Figure 3 and expressed in Formula 1.

The next advancement in the theoretical description of the nature of the freshwater-saltwater interface came with the work of Hubbert (1940) and Muskat, (1937) who described the flow field in terms of potential theory (from Reilly and Goodman, 1985).

This description is valid when the fluids are in motion, but again this formulation assumes a discrete boundary between the salt and fresh water.

$$Z = \frac{\rho_f}{\rho_s - \rho_s} h_f - \frac{\rho_s}{\rho_f - \rho_s} h_s \quad (2) \quad \text{(Hubbert, 1940)}$$

Where:

Z = depth to interface
ρ_S = density of salt water
ρ_f = density of fresh water
h_f = head of fresh water
h_S = head of saltwater

Thus the interface is a function of the head of the salt water and fresh water, expanding the accuracy of the Badon Ghyben-Herzberg formulation. The above theory was used by Vacher (1988) to predict the region of fresh groundwater on strip-island lenses. Vacher also modified Van der Veer's solution for a phreatic aquifer discharging into salt water (Van der Veer, 1977). This formulation for the situation shown in Figure 4 predicts freshwater discharge for a surficial aquifer with the inclusion of recharge.

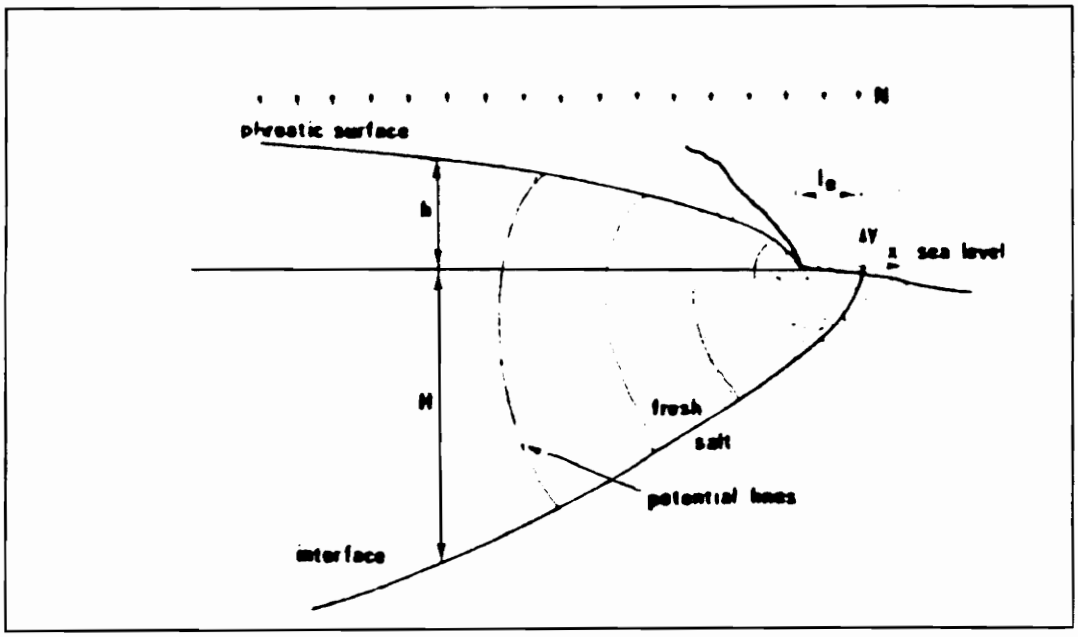


Figure 4. Van der Veers strip-island solution

(Van der Veer, 1977)

The discharge solution can be modified to read:

$$Q = \frac{\alpha}{\alpha + 1} R(M - x) \qquad (3)\ (\text{Vacher, 1988})$$

Where:

$\alpha = \rho_f / (\rho_s - \rho_f)$
M = distance to lateral midpoint of aquifer
R = recharge
Q = discharge across boundary
x = distance inland

This solution allows calculation of discharge and size of the discharge face with the assumptions of an infinite strip island, uniform recharge and hydraulic conductivity, steady-state and no mixing. At x = 0, the boundary is the shoreline, and thus the volumetric flux into open water. Except for the density expression, this formula is basically a mass balance of water into and out of the aquifer.

The use of potential theory was expanded by Glover (1959), who described the streamlines of steady flow of groundwater to the ocean. Although this formulation assumes a sharp interface between the interstitial saline and fresh water, it does allow for the offshore discharge of groundwater. This is shown as:

$$\gamma^2 - \frac{2Q}{\gamma K} X - \frac{Q^2}{\gamma^2 K^2} = 0 \qquad (4)\ (\text{Glover, 1959})$$

Where:

$\gamma = 1 / \alpha$
Q = discharge
K = hydraulic conductivity
X = distance inshore

Glover's equation is convenient because it relates discharge to hydraulic conductivity, densities, and position. Although this formulation assumes a discrete boundary, it still can be valid if the system is considered advection dominated (Glover,

1959). The formula yields the streamlines (the path of fluid flow) given in Figure 5. Glover also commented on tidal action that causes sea water to diffuse in the discharge face, a phenomenon noticed in most studies of SGWD. Fetter (1972) expanded on this technique to find accurate analytic solutions to the freshwater-saltwater interface underneath strip-islands. The Dupuit assumptions of Glover and Fetter precluded discharge. Although the many assumptions of the previous formulas make analysis by potential theory methods questionable for real world application, the theory can demonstrate which variables influence submarine groundwater discharge, and enhance an understanding of how the flow system affects groundwater-surface water interactions.

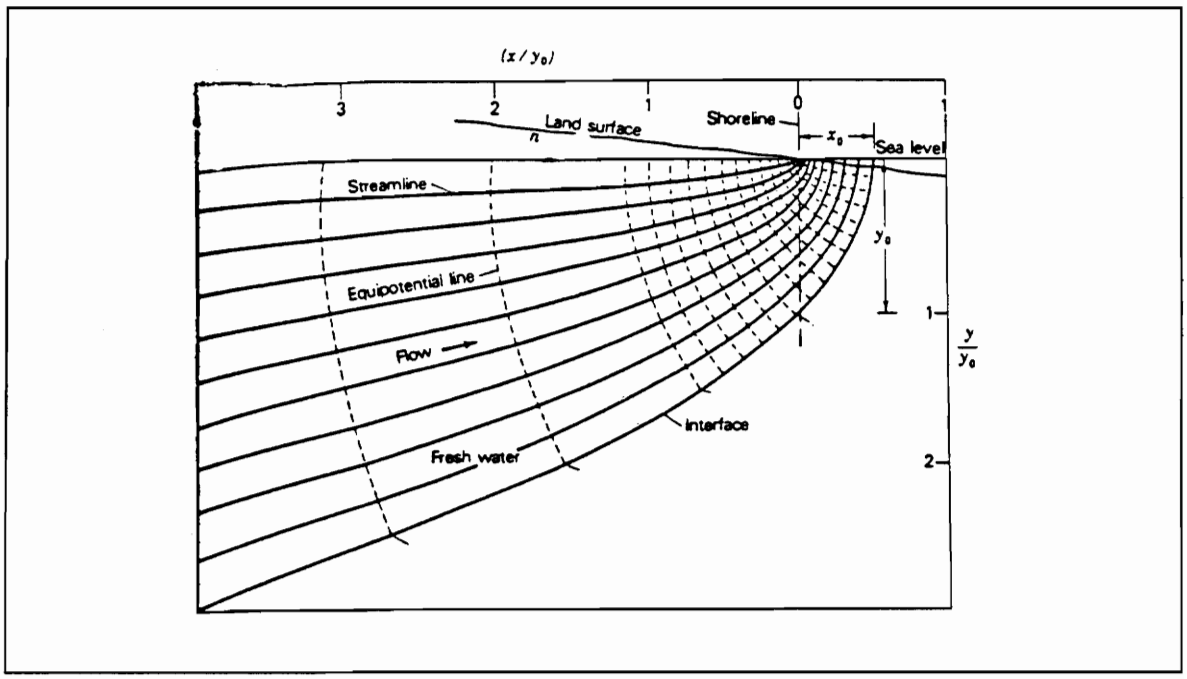


Figure 5. Potential theory flow net

(Glover, 1959)

Although these methods are quite sophisticated, none take into account that the interface is not discrete, but constantly mixing, moving and dispersing in response to tidal action and varying groundwater head. This creates a zone of mixing between the fresh groundwater and interstitial sea water, as shown in Figure 6. The black arrows in this figure represent the movement of freshwater, and red the cycling of seawater. Cooper (1959) developed a theory to explain such mixing, as well as the unexpected seaward limit of the freshwater-saltwater interface. Cooper theorized that freshwater was continuously

mixing with saltwater near the interface, and because the resulting mixture overlies denser saline water, the mixture consequently seeps out to the ocean. Because the mixture is constantly being transported from the interface, a landward flow of salt groundwater must occur to replace the mixed salt water. Henry (1959, 1964) and Kohout (1960) expanded and confirmed Cooper's hypothesis and Glover's formulation. Henry (1964) first used the advection-diffusion equation to describe the salt-freshwater interface. Although much work has been done using miscible fluids to describe these systems, the methods of Glover and others remain the best for advection dominated systems (Fetter, 1972; Reilly and Goodman, 1985). Serfes (1991) presented analytically the impact of tides on the hydraulic gradient, and found that the gradient could change direction during a tidal variation.

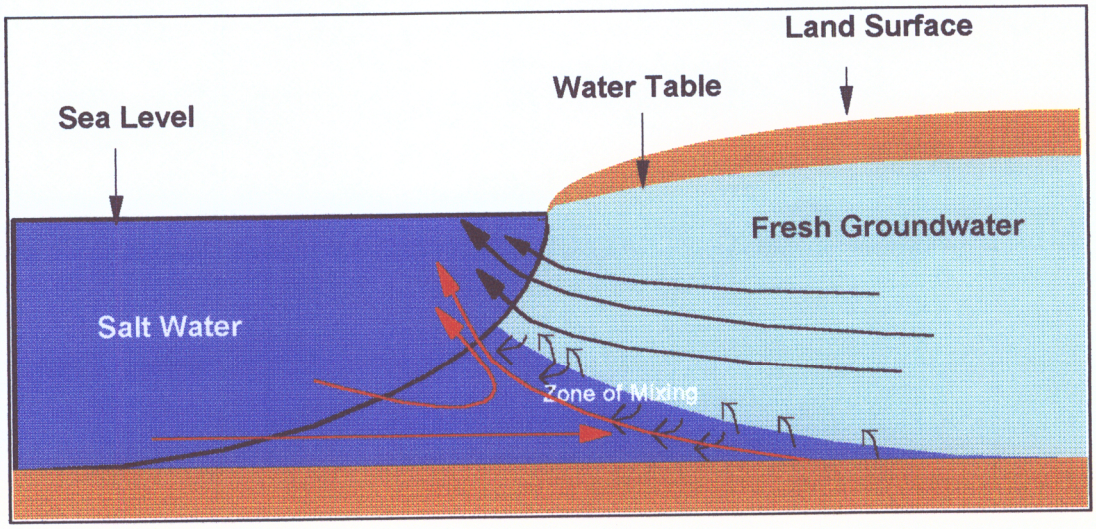


Figure 6. Mixing of interstitial saltwater and fresh groundwater

The water mass balance approach (as shown in Figure 7) has been used with moderate success in a few cases. Because estimates of all components of the hydrologic cycle are available except SGWD, it is a simple matter to solve for this unknown as in Equation 5.

$$SGWD=Rainfall-Evapotranspiration-Runoff-Leakance-Pumping \quad (5)$$

This method does not require detailed knowledge of the physical system, but rather the hydrologic cycle. The problem is that uncertainty and seasonal variability of the estimates associated with many of the components of the balance prevent accurate calculation of SGWD. Additional limitations of this approach are the inability to predict location of

discharge and nutrient loads without additional modeling methods. These problems are overcome to a large degree by discrete modeling, which will be discussed later.

Several reviews of SGWD-associated literature are available (Cooper *et al.* 1964; Reilly and Goodman, 1985; Reay and Simmons; 1992) for further reference.

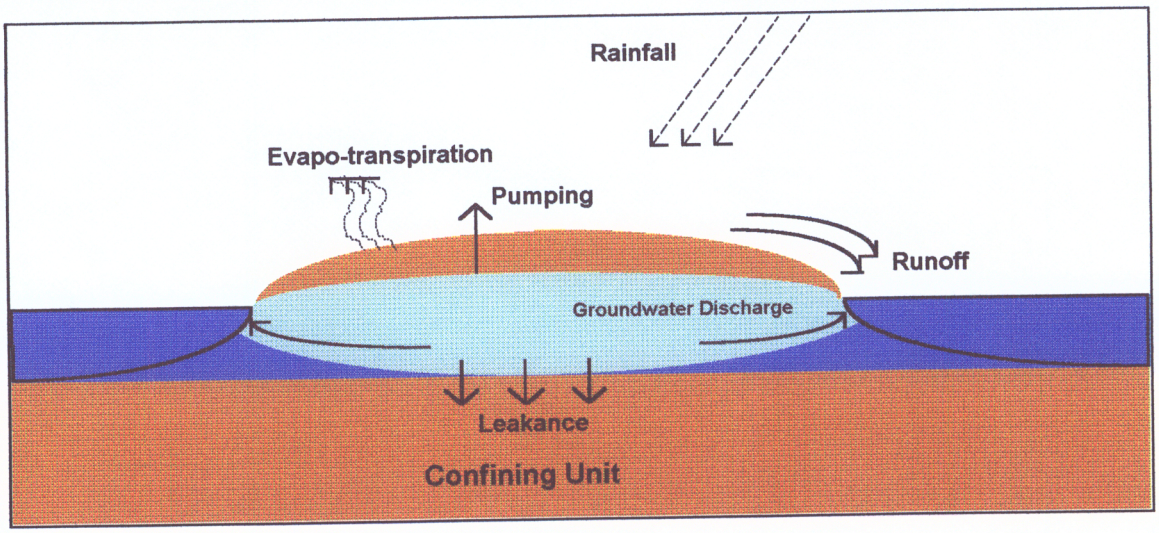


Figure 7. Hydrologic system of an oceanic strip-island

Submarine Groundwater Discharge

Submarine Groundwater Discharge (SGWD) typically occurs when groundwater has a higher head than the proximal surface water, as shown schematically in Figure 2. This seepage contributes to all types of surface water, carrying contaminants present in the surficial aquifer such as nutrients, pesticides and minerals. Although SGWD can occur from confined aquifers many miles out at sea (Zektzer *et al.*, 1973; Simmons, 1986), only SGWD from a vulnerable unconfined aquifer with typical nearshore discharge will be considered in this discussion. This discharge that occurs from the surficial aquifer is analogous to baseflow in streams. Groundwater discharge to surface water is the most poorly understood component in many hydrologic systems, not only because it is difficult to quantitatively measure, but because SGWD is dependent on many localized physical features. Reay and Simmons (1992) and Johannes (1980) have written thorough overviews on the subject of groundwater discharge that give further information.

Methods of Prediction and Detection of SGWD

Finite element models can incorporate a nutrient loading model, as well as give a spatial analysis of SGWD. Unfortunately, finite element models are very complex; detailed estimates of parameters are frequently unavailable and powerful computing resources are necessary for even a simple scheme. Any finite element scheme applied to a marine environment must necessarily be complicated by the temporal variations of tides as well as the miscibility of fluids with different densities. If contaminants are to be included, a methodology for contaminant transport must be developed. Unfortunately, the large amount of spatial data required by this more accurate approach does not exist. Numerical schemes also require extensive physical data, which may not be available for this type of computation, especially for a large scale analysis, for which data processing may be cumbersome.

Zektzer *et al.* (1973) provided a review of available techniques for detecting SGWD. Most of the techniques reviewed (electric conductivity, temperature, saltwater anomalies, radon) were intended to detect gross, point sources of SGWD for exploitation as water supplies. The radon method (Kir'yakov *et al.*, 1982) is also suited only to point groundwater discharge. The presence of radon in freshwater can act as a tracer in the marine environment to pinpoint locations of high discharge. Resistivity logs have shown that theory overpredicts depth of the fresh water-salt water interface in Long Island, NY (Bokuniewicz and Pavlik, 1990). A biological indicator of SGWD has even been suggested - the grazing mud snail (Williams *et al.*, 1991). Grazing mud snails seek out areas with high algal levels, which may occur due to higher levels of nutrient transport from SGWD. Recently, a direct measurement device has been used for the analysis of SGWD and the contaminants it carries.

Direct Observation Of SGWD: The seepage meter.

The seepage meter, illustrated in Figure 8, has been a valuable tool for the investigation of SGWD. Lee (1977) first used a seepage meter on lakes to investigate the most uncertain component in hydrology and limnology -- SGWD. This seepage meter was a successful attempt to provide data not only on the rate of groundwater discharge, but also to collect samples of the groundwater being discharged. Lee's seepage meter

consisted of a capped cylinder implanted in the bottom sediments. A small, balloon-like device collected water samples, and its filling was timed to estimate groundwater flux. A device was contemporaneously developed to measure seepage in New Zealand (Lock and John, 1978). Lock and John's investigations were the first of many to show an exponential die-off of SGWD with distance offshore. These studies demonstrated the importance that SGWD plays in the hydrologic cycle and provided impetus for further research into the phenomenon.

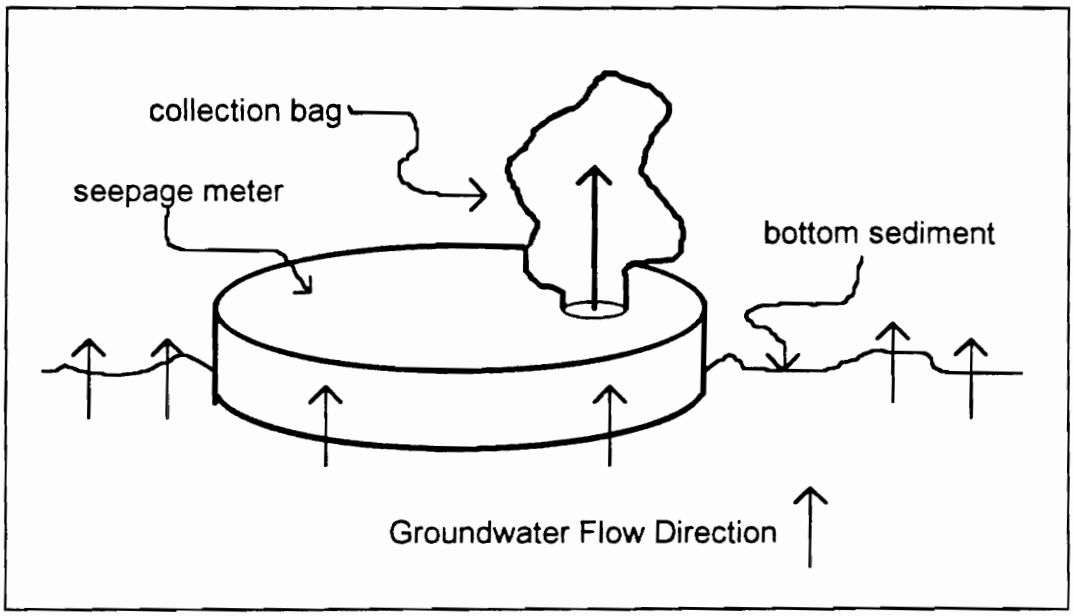


Figure 8. A seepage meter

Several researchers (Lee and Cherry, 1978, Lee and Hynes, 1978) began using mini-piezometers along with seepage meters to investigate SGWD along stream beds. This piezometer use allowed the measurement of groundwater head and nutrient content 0.6 m below the stream. This advancement not only allowed for more accurate understanding of the flow, but also permitted the examination of chemical transformation of contaminants by transport through the benthic zone. Good correlation was found between the interstitial water sampled by the piezometer and discharged groundwater sampled by seepage meter for conservative contaminants, but no significant correlation was found with non-conservative contaminants such as nutrients. This indicates that chemical

transformations may be taking place as groundwater seeps through the bottom sediments (Lee and Hynes, 1978).

Investigations also began with seepage meters on other water bodies. Lee's exponential decay function was supported by seepage meter measurements in the Great South Bay in New York (Bokuniewicz and Zeitlin, 1980) and in three Canadian lakes (Shaw and Prepas, 1990a and b). In Florida, data were taken that suggested that seepage meters overestimated nutrient input to lakes because of the anaerobic conditions that resulted by placement of the seepage meter (Belanger and Mikutel, 1985). Despite this, many studies have been able to obtain nitrate fluxes (*e.g.*, Reay, 1992; Simmons *et al.*, 1991; Simmons *et al.*, 1992). Zimmerman *et al.* (1985), while investigating the role of dissolved phosphate in groundwater seepage from marshes, noted large changes in groundwater seepage with tidal heights, an important result for the analysis of SGWD in marine environments. Tides sometimes cause sea water to infiltrate into the aquifer (Serfes, 1991).

Optimal placement of seepage meters was studied by Shaw and Prepas by simulation using the Monte Carlo method. This work assists in selection of sampling locations and frequencies (Shaw and Prepas, 1990a). It was found that most variation in meter readings was the result of "spatial variability of lake sediments." An average transect consisting of ten meters would properly predict entire lake seepage 70% of the time, and repeating the test or using more meters only slightly increased accuracy. These concepts were put into practice (Shaw and Prepas, 1990b) to measure seepage in several lakes in Alberta, Canada. A more rugged and versatile seepage meter was designed to study discharge into the Great Lakes (Cherkauer and McBride, 1988). This seepage meter was designed to be operated remotely (no divers necessary for installation) as well as suited to a harsher environment (strong current and wave action). The remote logging seepage meter should be a step forward in the ability to analyze seepage data (Reay and Walthall, 1992). By this time, the seepage meter was used in marine and estuarine environments and became a tool in the investigation of nutrient transport by SGWD (Valiea *et al.*, 1978; Bokuniewicz, 1980; Simmons, 1986 and 1988; Simmons and Love, 1987; Simmons and Netherton, 1987; Lewis 1987; Libelo *et al.* 1991).

Significance of SGWD for Contaminant Transport

Groundwater discharge has been recognized to be an important but overlooked factor in water budgets (Johannes, 1980). Concerning the Chesapeake Bay, a study (Libelo *et al.*, 1991) on the estuarine James River concluded that anthropogenic activity adversely affected discharged groundwater quality and a rough extrapolation showed that "groundwater nutrient inputs to the Chesapeake Bay system may represent 30% of the total flux into the system." Another study (Simmons, 1988) of SGWD off the Eastern Shore indicated that groundwater flux could account for nutrient loading of the Chesapeake Bay on the order of a major tributary (such as the James). This was very significant because SGWD was not even considered to be contributing to nutrient enrichment in the Chesapeake Bay (MacIntyre *et al.*, 1989). It was noted in 1987 (Libelo *et al.*, 1991) that the current Chesapeake Bay model (Hydroqual, 1987) did not include groundwater seepage in the water and nutrient budgets. Johannes (1980) said quite appropriately "overlooking the fact (that SGWD occurs) could lead to serious misinterpretation of ecological data in studies of coastal pollution, of benthic zonation and productivity, and of the flux of dissolved substances and overlying water." The land use patterns associated with contamination of surface water from the non-point source of SGWD are shown schematically in Figure 9. Notice that the sources frequently associated with surficial aquifer contamination may cause increased levels of damaging SGWD. Nitrate and pesticides are of concern in agricultural areas, and septic systems serving homes are a source of ammonium ions and coliform contamination risk.

Some studies of SGWD have occurred on marine areas without seepage meters. Weiskel and Howes (1991) investigated nitrogen flux in a small Massachusetts embayment, comparing three models and field methods. While the mathematical models predicted similar quantities, these quantities were typically 60% higher than field results. In addition, this study indicated that it is possible to correlate nitrogen flux accurately with up-gradient land use. A study was conducted on Cape Cod, Massachusetts comparing three methods of quantifying nitrogen inputs to a small cove (Giblin and Gaines, 1990). High levels of nitrate in the groundwater were determined to be derived from on-site septic systems (ammonium seems to be the primary form of nitrogen associated with urban land use on the Eastern Shore, however [Reay, 1992]). This nitrate contamination was highly variable within the aquifer. A five-fold difference was found in prediction of

nutrient loading between a water budget method, the seepage meter estimates, and a salt and water budget at the mouth of the cove.

More recent studies in the Chesapeake Bay found that near-shore land use practices affect quality of SGWD as shown in Figure 9. A high degree of correlation has been found between oxidized nitrogen and agricultural areas, as well as between reduced nitrogen and phosphate and suburban-septic areas on the Eastern Shore (Simmons 1989; MacIntyre *et al.* 1989; Libelo *et al.*, 1991; Simmons *et al.*, 1991; Reay *et al.*, 1992; Miles and Reay, 1993). Results of some of these studies even indicate that some of the practices involved with controlling non-point pollution may be adversely impacting groundwater quality, and thus the water quality of the Chesapeake Bay (Diliha, 1990; Reay *et al.*, 1992). Effects of forest buffers and riparian zones as a management strategy for nutrient removal have been studied (James *et al.*, 1991; Reay *et al.*, 1991; Smedley, 1993). Sedimentation ponds, tillage practices, and other non-point pollution control techniques may need to be modified to prevent the degradation of surficial aquifers (Reay, 1993 personal communication). Other studies have provided an understanding of the role the sediment plays in nutrient transport by way of groundwater discharge (Lee *et al.*, 1980; Simmons, 1989; Reay and Gallagher, 1991; Reay *et al.*, 1992; Reay 1992).

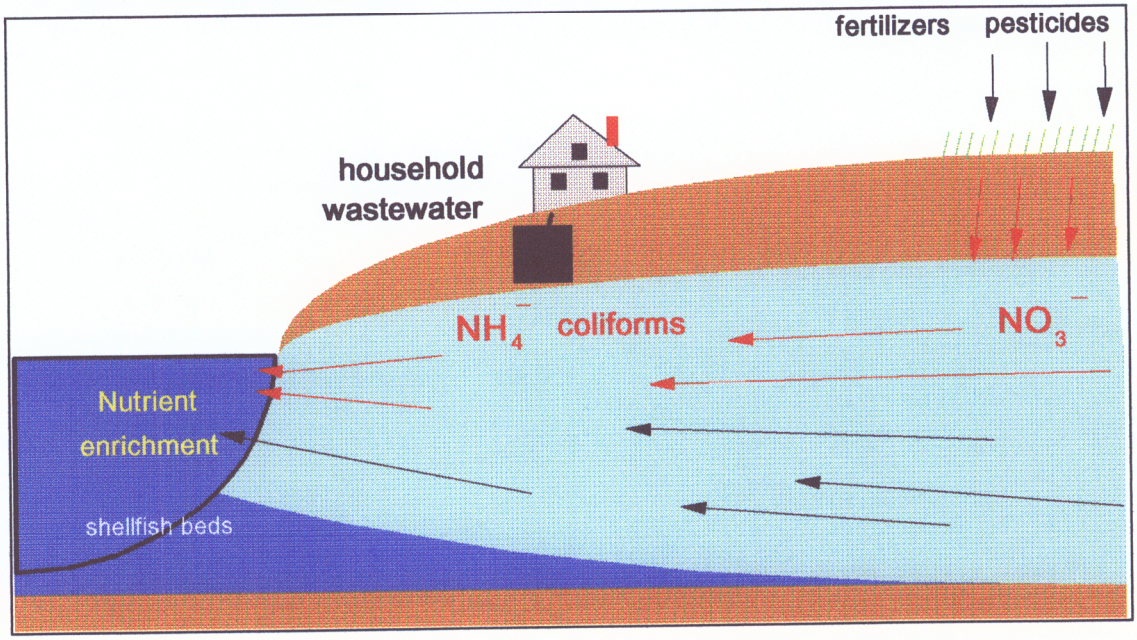


Figure 9. Impacts of land uses on discharged water quality

Other recent research has concentrated on determining groundwater flow characteristics of the Chesapeake Bay System. Staver and Brinsfield (1991) showed that discharge was proportional to recharge, and that discharge followed a seasonal pattern, with heavy rates of recharge during winter and spring. They came to two other important conclusions: "tidal discharge conditions will not influence the long-term net discharge of groundwater" and the "strict dependence of flow rates on water table elevation." Tidal action has been shown to drive short-term discharge rate in areas of high conductivity, while low-permeability sediment type, such as clay, has been shown to nearly stop discharge (Zimmerman, 1991).

Geographic Information Systems

A Geographic Information System (GIS) is a marriage of computer-aided design (CAD) and database management (DBM). This combination of computing concepts allows for the analysis of any type of spatial data as well as the format for visual representation and easy recognition of output of these analyses. A variety of GIS systems are available, ranging from inexpensive educational programs to complete multi-user programs with huge databases. There are several GIS texts available, (Tomlin, 1990; Taylor, 1991; Burrough, 1991) as well as numerous articles (Caswell, 1992; International Journal of Geographic Information Systems, published by Taylor and French, London; Geo Info Systems, published by Astor Publishing Corperation, Eugene, Oregon).

GIS systems fall into two basic categories: vector-based and raster-based. A vector-based GIS system represents objects as points, lines and areas, whereas a raster-based GIS represents objects in a matrix of small cells, each cell containing an associated attribute. An important aspect of this work is to test the viability of a technique that does not require extensive computing resources. The PC-based Atlas*GIS (Strategic Mapping, 1992) was chosen for several reasons. It is a moderately priced vector based system that can operate on almost any type of PC, the import/export routine converts to and from many other popular file formats, and Atlas's menu-driven interface is easy to use.

GIS Uses in Resource Management

Geographic information systems have found many uses in resource management. Some examples of these uses are groundwater resource planning and management (Bishop *et al.*, 1990), forest management for non-point pollutant risk (Lull and Potts, 1990), and the assessment of pollution potential from fertilizers (Halliday and Wolfe, 1991). Animal waste pollution potential has also been assessed by a GIS system (Heatwole and Shanholtz, 1991). An interesting adaptation has been made by joining GIS with a groundwater flow model and surface runoff model for phosphate mining reclamation (Ross et al., 1990). Many other similar analyses are available in a wide variety of investigations, the above being only a representative sample of the uses of GIS. Of particular interest is the risk-based analysis such as that used by Tim *et al.* (1992), which identified areas at high risk of erosion in a Westmoreland County, Virginia. This type of

analysis does not attempt to quantify erosion, but rather locate areas at risk for erosion. Layers are built up that are representative of the parameters that affect erosion, and a measure of erodability is developed. A GIS approache typically uses this risk based analysis when data for model calibration is unavailable. Beeler (1988) classified groundwater areas by just such a method with a GIS.

The Eastern Shore in Virginia

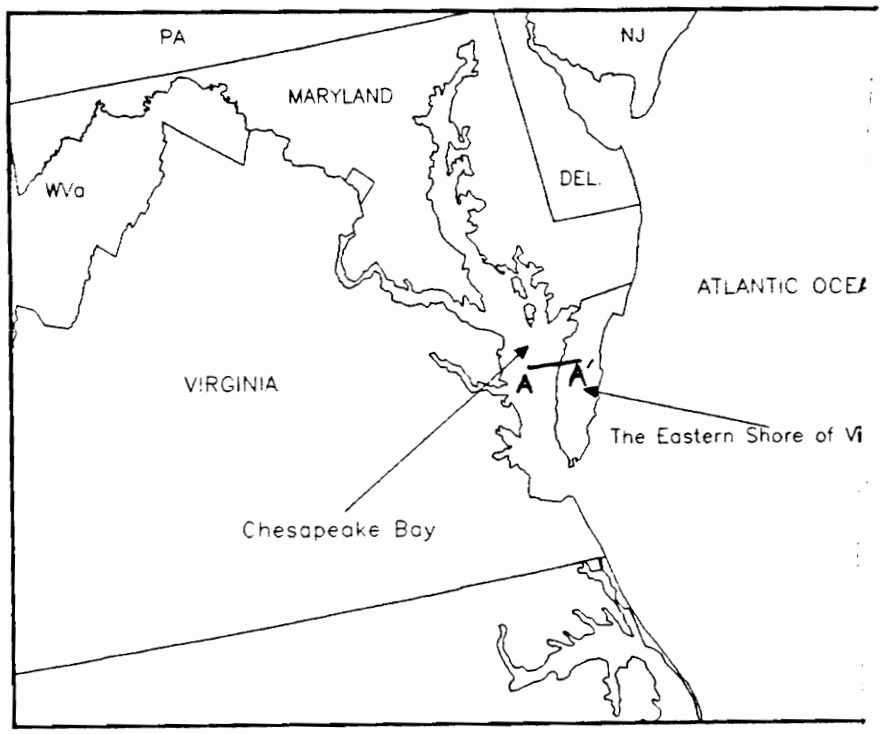


Figure 10. The Eastern Shore

The Eastern Shore in Virginia consists of Accomack and Northampton Counties, and is located across the Chesapeake Bay from Virginia proper, as shown in Figure 10. Located on the southern tip of the Delmarva peninsula in the Coastal Plain physiographic province, the Eastern Shore is a narrow, 112.6 km long peninsula, oriented in a general north-south direction, bounded on the east by the Atlantic Ocean and on the west by the Chesapeake Bay. The Atlantic edge consists of barrier islands, tidal flats and marshes. Moving westward across the shore are well-drained lowlands, followed by poorly drained

areas towards the center, and back again to well-drained lowlands, marshes and tidal flats on the Chesapeake side (Richardson, 1992). The Eastern Shore of Virginia is approximately 2500 square kilometers. The average width of the peninsula is 13 km in Accomack County and 10 km in Northampton County (Simmons, 1989).

Fennema and Newton (1982) presented a comprehensive survey of the groundwater resources on the Eastern Shore. There are no major streams on this flat, moderately well drained land; 30 of the 43 inches per year of rainfall are lost to runoff and evapotranspiration. The area is largely rural, with agriculture and food-processing industries (Fennema and Newton, 1982; Richardson, 1992). There are many small wells located in the surficial aquifer serving homes and small water distribution systems, as well as some larger industrial and seasonal agricultural withdrawals. The surficial aquifer is considered, however, to be in relative equilibrium with a withdrawal capacity twice current consumption despite saltwater encroachment problems in lower aquifers (Richardson, 1992). Groundwater allocation on Virginia's Eastern shore breaks down into three basic categories, 43% commercial, industrial, and municipal water treatment, 38% agricultural, and about 19% for domestic wells (Laczniak, 1988 as cited by Hamilton *et al.* 1989)

The surficial Pleistocene aquifer is referred to as the Columbia aquifer. This aquifer consists of unconsolidated shell, gravel, sand and some clay lenses. The bottom of this aquifer is between about 0-100 ft below mean sea level, and is typically less than 100 ft thick. The aquifer has been estimated to have an average hydraulic conductivity of 15 ft/day (Fennema and Newton, 1982) and a model calibration run yielded an estimate of 18.1 ft/day (Harsh and Laczniak, 1986). Hamilton *et al.* (1989) have also surveyed the groundwater quality of the Delmarva Peninsula and present an overview of the hydrology. A typical cross-section (line A-A' of Figure 10) is shown in Figure 11. Leakance through the Yorktown-Eastover confining unit between the Columbia aquifer and the Upper Yorktown-Eastover aquifer is typically small except in the few areas where the confining unit is missing. The surficial aquifer is stable, and the water table is close to the ground surface, making the aquifer vulnerable to contamination.

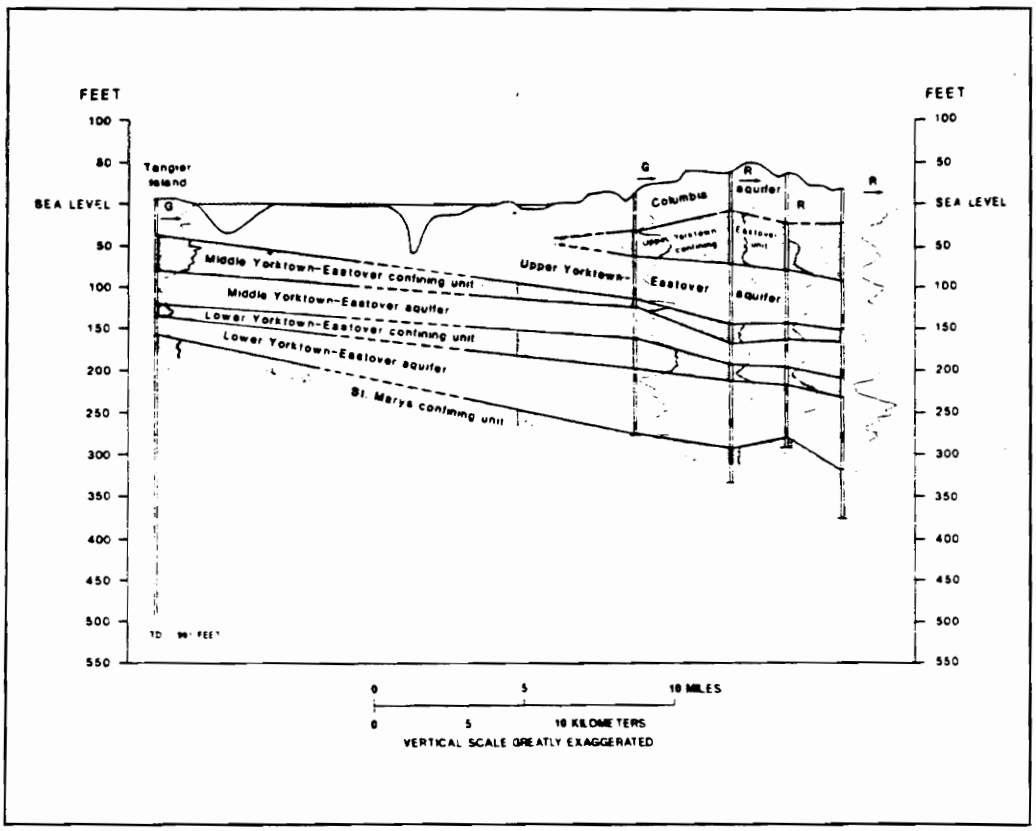


Figure 11. Cross-section of the Eastern Shore
(VA Water Control Board, 1974)

III. METHODS AND MATERIALS

Introduction

The procedure can be divided into four basic parts: 1) identify and obtain appropriate spatial data 2), import data from VirGIS (Virginia GIS mapping project) and digitize from other sources, 3), manipulate the data layers to create a model of the pertinent processes involved with SGWD and its associated contaminants, and 4), compare the model with seepage meter investigations. The result of the procedure was a model that was used to estimate the spatial distribution of nearshore groundwater discharge from the Eastern Shore, as well as the water quality. The Eastern Shore's shoreline was discretized by soil type, and segments with high likelihood of contamination were identified. Data on soil permeabilities and hydraulic gradient were used to estimate volumetric discharge, and land use data determined which discretized segments were likely to transmit contaminated water. The volumetric discharge and contamination attributes were combined to yield a value for each segment that represents risk for nutrient flux. These data were used to construct a framework to analyze groundwater discharge patterns on the Eastern Shore of Virginia, both on the large scale as well as the small.

Spatial Data

Several types of spatial data were necessary to construct this model. First, it was necessary to gauge the volumetric rate of SGWD. This required a combination of driving force (hydraulic head) and ability to transmit water (permeability). Spatial data to estimate of regional hydraulic head was obtained from maps developed by the USGS (Richardson, 1992). While no spatial data was found for nearshore aquifer permeability, relative nearshore permeabilities were estimated from the nearshore soil type. Spatial data for soil types was obtained from VirGIS, a statewide GIS project (Shanholtz and Flagg, 1990). VirGIS is currently organizing a spatial database for land use, soil type, water feature, elevation, and other critical information used for estimating erosion rates, as well as other agriculturally related phenomena.

The second type of data required by this model was land use. Agricultural land use is associated with nitrate and potential pesticide contamination, and urban with ammonium and coliform contamination. Agricultural and pastural land use was available through

VirGIS, and population centers on the Eastern Shore were digitized from USGS 7.5' quad maps. The soils and gradient data were used to estimate volume of discharge; land use characterization was used to estimate contamination of the discharged groundwater. A cross between these two maps ranked the areas as to the likelihood of having both high discharge rates as well as high contamination probability. This attribute was the one of highest concern from a planning and management standpoint.

The basic unit of the map was a segment of the perimeter of the Eastern Shore proper. The segments represented the smallest element of the shoreline for which there is comprehensive data (the soils data). There were 1993 of these segments in the perimeter, and they were manipulated on a spatial level with the previously discussed data to obtain a piecewise description of the discharge along the Eastern Shore.

Data Assembly

Table 1. VirGIS data layers

Data Layer	Components
Water Features	Marshes Areal Water Features Streams
Land Uses	Agricultural Pasture Non-agricultural
Soils	Soil features Areal Water Features

The first step in the construction of the GIS model began with obtaining data from the VirGIS project of the Virginia Division of Soil and Water Conservation, and converting the data into Atlas*GIS. Soil, water features, and land use data were obtained in matrix form, each on 1/9 (about 33m x 33m) hectare resolution. The layers purchased are detailed in Table 1. Additional information on the data can be found in VirGIS report ISSL 90-1 (Shanholtz and Flagg, 1990) and ISSL 88-13 (Shanholtz *et al.*, 1989) for Accomack and Northampton counties, respectively. These documents commented on the accuracy and methods of digitizing, as well as on the methods of checking the final maps. Spatial ANalysis System (SPANS) version 5.2 (INTERA TYDAC, 1991) was used as a translational tool because the available data was less expensive in raster rather than vector

format. In addition, the vector files were substantial and awkward to manipulate easily on a PC. Unfortunately, Atlas*GIS cannot translate this matrix of data at this point in time. Ultimately, the time and effort required in the translation would have justified purchasing the vector versions of these maps.

After receiving the matrices, the data were converted to a vector format (USGS dlg3) using SPANS. One unique feature of SPANS is that it can convert to and from many GIS formats. The conversion proceeded from the original raster database to a SPANS internal raster format, and then to the SPANS quadtree (a space-saving form of a raster). Data was enquaded for each of the maps at a resolution of 15m (less than that of the original rasters), to maintain the spatial integrity of the database. Next, the features were converted to SPANS internal vector format at a resolution of 30m and finally to the USGS dlg3 format that could be converted into Atlas*GIS format for analysis.

Because each layer from each map was imported separately into Atlas*GIS, some tedious registration of layers was necessary. It also seems that the Atlas GIS dlg3 import routine does not handle topology well; upon comparing the Northampton County soils survey it became evident that the 1/9 hectare VirGIS grid did not have adequate resolution to portray all small features, for example narrow beaches. These excluded features were digitized manually with the aid of county soil surveys (Cobb and Smith, 1989) and checked with the Accomack County aerial surveys (Accomack Co. soils office). Inspection of the Accomack soils showed that the soils layer had better resolution than that of Northampton County. After completing the registration process, each feature of the soils map was given a value for maximum and minimum permeability as found in the soil survey.

Features such as Chincoteague Island, Fisherman's Island, Tangier Island and any feature surrounded completely by water were excluded from consideration in this study. Thus small islands and offshore marsh areas were ignored. Although groundwater does flow out of these areas, the water transmitted is typically tidally generated, and significant nutrient or toxic input will not occur because inspection of the USGS 7.5' quads and the VirGIS layers show no impacting land-uses associated with these areas. One notable exception to this rule was Chincoteague Island. Here there was significant contamination of groundwater occurring due to the extensive use of septic systems fed from on-shore

water sources. Although it is believed this land area contributes significant nutrient loads to the surrounding waters, no hydraulic gradient datum was available, so Chincoteague Island was not incorporated into this study.

With the soil and water features maps imported into Atlas, the next layer added was the surficial aquifer elevation. The 10-ft (3.048 m) contour line was manually digitized from a map of average Columbia Aquifer water table heights (Richardson, 1992, pg. 54). The water table height was assumed to vary linearly by distance to the nearest water feature, yielding an estimate of large-scale hydraulic gradient for any point along the SGWD zone. Some necks, such as the one shown in the inset in Figure 14, necessitated a waypoint in the middle of the narrowest part of the neck to ensure that the hydraulic gradient is measured over land. The distance was first calculated from the segment to the waypoint, and then the distance from the waypoint to the water table elevation was added for each feature on the neck. It should be noted that the water table map is not digitized at the 30 m resolution of the others, and may be a source of errors as discussed in Chapter V.

Table 2. Digitized Maps

Layer Name	Source Map
10' groundwater elevation iso-line	Richardson, 1992
Suburban Septic	USGS 7.5' quads
Modifications to Soils Maps	County Soil Surveys

Concurrently, a layer containing population centers for the Eastern Shore was digitized from USGS 7.5' topographical maps. This layer, along with the other digitized layers, is given in Table 2. Population centers were digitized with special attention to those residential areas near the shoreline. It was thought that because there are limited municipal waste water treatment facilities on the Eastern Shore, these areas are significant sources of reduced nitrogen and coliforms to the surficial aquifer, as shown by Simmons (1988). In conjunction with the agricultural and pasture lands, the result was a groundwater pollution potential layer. The final segmented discharge layer (the perimeter of the shore) was revised to include distance to, area of, and type of the nearest pollution potential figure. The result of the area and distance was a quasi-normalized layer of the

propensity to pollute (meaning the figures obtained have meaning only relative to each other).

Next, a layer was created that defines the SGWD zone, which in this study was defined to be the point where the soils map meets the areal water features map. This layer can be considered a map of the shoreline, or alternatively, the boundary line of the Eastern Shore. Because a distinction had to be made how far up inlets and streams SGWD takes place, water features classified as areal by VirGIS, as well as a few areal features from blue-line streams of area over about 25 acres were classified as locations of potential SGWD. Features in blue-line streams that were included had significant open-water stretches where discharge could occur and not be included as a baseflow to a stream. Undoubtedly SGWD occurs into streams, but this should be included as a point load from a stream and should not be included in an inventory of SGWD.

Spatial Manipulation

The spatial manipulation took place on the layers listed in Table 3. Figures 12 and 13 are flowcharts that aid in the visualization of the methodology used. The SGWD zone was then split by the soils layer and given a geometric average permeability (Table 4) from the permeabilities listed in the soil survey (Cobb and Smith, 1989), and these smaller discharge zones were each given a value of hydraulic gradient (distance away from the 10 ft iso-line divided by 10 ft). Table 4 also lists the length that each soil type was manifested along the perimeter of the shore. The distance measured for each boundary segment was the distance from the midpoint of the boundary segment to the closest portion of the 10 ft contour line. These features were then considered a quasi-normalized layer of propensity to discharge, containing both gradient and nearshore permeability for each segment. Some soil features were upwards of 10 km long measured along the shoreline. These large features were broken down into smaller segments (about 1 km each), such that the smaller segments could portray other variables like the gradient more accurately (on a smaller scale). Instead of a large segment assigned one gradient (or contamination, etc.), there were several smaller segments, each assigned a different attribute.

Table 3. List of Data layers

Layer	Purpose
boundary line	discharge perimeter
soils	permeability, segmentation
water table elevation	hydraulic gradient
agriculture & pasture	contamination of surficial aquifer
towns	contamination of surficial aquifer
marshes	areas with offshore discharge

Table 4. List of permeabilities

Soil name	Mapping Unit	Permeability (cm/hr)	Total perimeter length (m)
AhA	Arapahoe loam	7.62	0
AmA	Arapahoe-Magothy complex	7.62	19448
AsE	Assateague sand	50.8	3538
AtD	Assateague fine sand	50.8	465
BeB	Beaches	50.8	45623
BhB	Bojac loamy sand	28	20047
BkA	Bojac sandy loam	16.51	4345
BoA	Bojac fine sandy loam	16.51	86084
CaA	Camocca fine sandy	16.51	703
ChA	Chicoteaugue silt loam	1.52	29716
DrA	Dragstone fine sandy	16.51	3733
FhB	Fisherman fine sand	50.8	2353
FmD	Fisherman-Assateague complex	50.8	0
FrB	Fisherman-Camocca complex	50.8	0
MaA	Magotha fine sandy	7.62	12368
McA	Magotha-Chincoteague complex	6.6	180720
MoB	Molena sandy loam	28	2930
MoD	Molena loamy sand	28	119603
MuA	Munden sandy loam	7.62	10853
NmA	Nimmo sandy loam	16.51	3773
PoA	Polawana loamy sand	28	72
SeA	Seabrook loamy sand	28	67
UpD	Udorthents and Udipsamments	16.51	6530

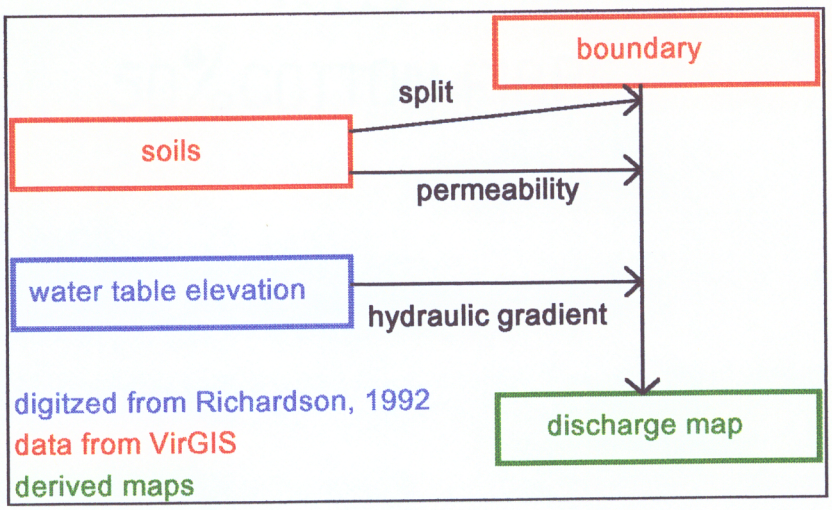


Figure 12. Derivation of SGWD map

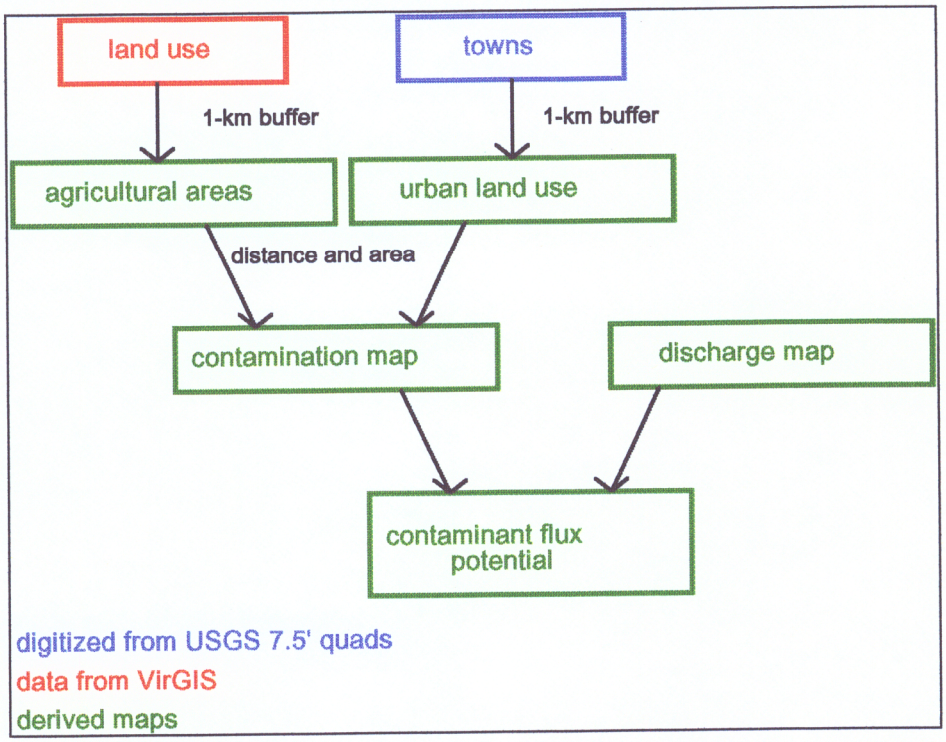


Figure 13. Derivation of the contamination map

With the boundary in place, the next objective was to determine which of the segments of the discharge boundary line were likely to be under the influence of which contaminants. A schematic of the process is shown in Figure 13. The methodology

chosen was to split all town, pasture and agricultural features by a one kilometer buffer of shore, and insert into the attribute file both the distance from the shore to that feature and the area of that feature. The one kilometer distance was chosen as an arbitrary buffer, because it is thought that the contaminants have moved only a limited distance in the forty or so years since fertilizers have been applied to agricultural fields on the Eastern Shore (Simmons, 1993-personal communication). The one kilometer buffer can be justified assuming movement of water in the surficial aquifer of just under 5 cm per year. Thus, each incremented boundary had both the distance to the potentially contaminating feature, as well as its impact (presumed proportional to area). In the same manner, a marsh layer was created to show the weighted proximity of marsh features to the discharge boundary. Although other sources of contamination may affect groundwater quality of the Eastern Shore, such as animal waste lagoons (e.g., Ritter and Churnside, 1986), only agricultural, pasture and town features were included in this study. As more comprehensive spatial data becomes available, however, it could be incorporated into the framework of this analysis.

A similar process of buffering was used to determine which segments were directly proximal to marsh areas. The buffer distance used was 50 meters. This marsh attribute may assist in the understanding of the groundwater flow regime as discussed in Chapter IV. An additional value was added as an attribute to each boundary increment- the distance to the aquifer divide. This distance represents the linear distance over which recharge takes place. With limiting assumptions this will allow a calculation of the quantity of discharge similar to Equation 4 for each segmented boundary. A mass balance is performed around each segment, and the sum of each of these fragments will represent total discharge if the Eastern Shore is assumed a strip island with uniform recharge and evapotranspiration, constant hydraulic conductivity, and no pumping. The segments were uniformly scaled to sum to 112 km on either side of the strip-island Eastern Shore. Thus, both the Chesapeake and Atlantic shorelines have 112 km of idealized shoreline, equal in length to that of the hydraulic divide. In an attempt to maintain a conservative estimate, the recharge distance used was the linear distance to the center of the aquifer, without regard to the overland condition used in determining the gradient.

The output from the spatial manipulation consists of four layers; marsh areas, the discharge potential, potential for contaminant flux, and recharge distance.

Spatial Analysis

With the spatial manipulation complete, what remained was to analyze and present the data, which are the output layers. The study area is presented in Figure 14. Stars denote sampling areas used in analyzing the effectiveness of the spatial analysis. Summary data for the output layers are presented in tabular format as well as in the form of maps. The maps were broken down by category or quantiles of equal data size. Output layers were examined for patterns of discharge, as well as gain an overall conception of groundwater discharge. Discussion of the output layers follows in the next section.

The method of determining the quality of the modeling technique was with seepage meter readings. An average discharge figure was found by taking meter readings from several sites where sampling has occurred at various times of the year. These meter readings were correlated to the data derived as above, and were used to evaluate the effectiveness of this modeling technique. Vacher's M distance was used to compute the strip island lens-type solution, as discussed in the next section. The total discharge figures found by these methods are subject to the limitations discussed in Chapter IV.

Materials:

The VirGIS data was created as described in Shanholtz et al. (1989) and Shanholtz and Flagg (1990). File format manipulation was accomplished with the PC-based SPANS v 5.1 running on a 486-33 PC under OS/2 (INTERA TYDAC, 1991). Atlas*GIS versions 1.1 and 2.0 were used as the GIS analysis tool on a 486-50 and 386-20 PCs under DOS (Strategic Mapping, Inc., 1992) Digitizing was done on a Summagraphics Summasketch Plus Digitizer. Atlas Import/Export version 2.0 (Strategic Mapping, Inc., 1992) converted files between dlg3 files and Atlas format. USGS 7.5' quad maps and USGS Open-File Reports were used as source maps for digitizing. Final soils maps were checked against the soils maps of Northampton County (Cobb and Smith, 1989) and the aerial photographs in the Accomack County soils office.

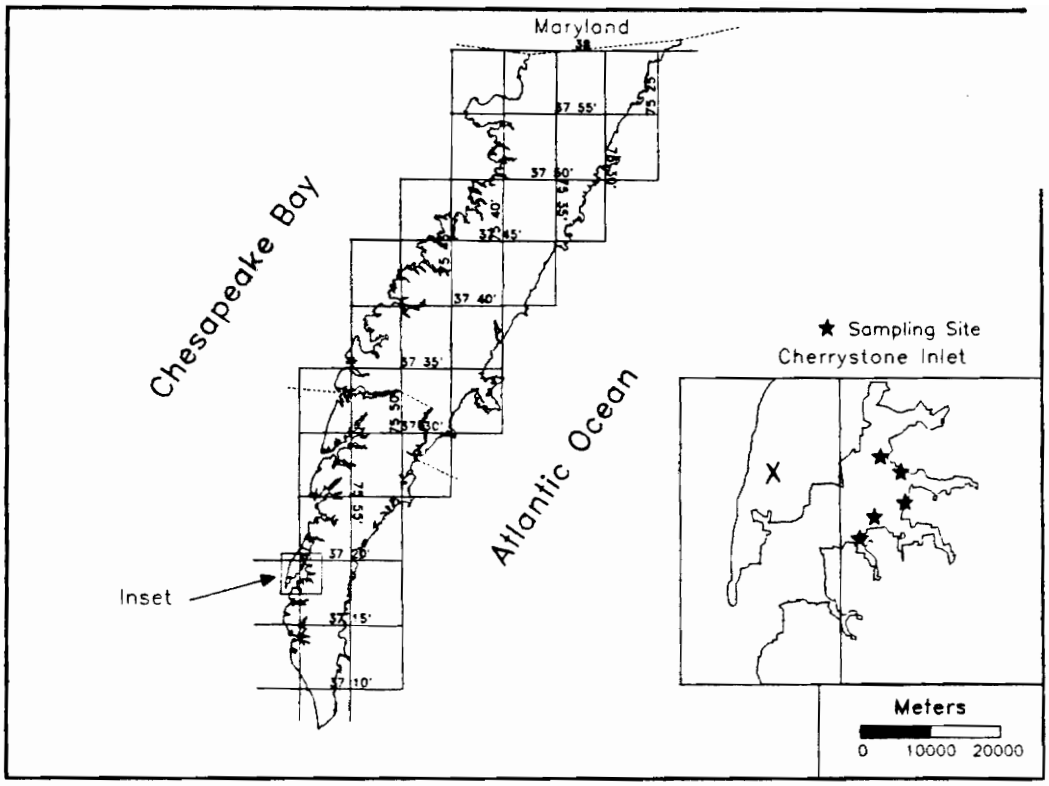


Figure 14. Study area

IV. RESULTS AND DISCUSSION

The model developed in this research is characterized by the discretization of the Eastern Shore perimeter into 1993 shoreline segments. Each of these segments has attributes for length, gradient, nearshore permeability, proximal land use area, distance to the nearest impacting land use, type of polluting feature and recharge distance. To aid in data visualization, some maps were organized into 4 or 5 quantiles, representing 4 or 5 equal ranges of data for that map. Colors represent the ranges of the quantiles by red, yellow, green and blue from high to low. Thus for Figure 15, red represents the highest 25% of permeabilities, blue the lowest 25%. Color has no influence on the values manipulated by the database, however. Color just indicates into which quantile the feature's datum lies. In addition, it should be noted that these figures represent data available in far finer resolution than the scale of these maps allow. Any area of shoreline on these maps can be examined much closer, as in Figures 30-33. This chapter is organized such that volumetric discharge results are presented and discussed first, followed by contamination results. Limitations are then presented, along with correlation to sampled areas and an estimate of discharge. Concluding this chapter are some comments on the GIS approach to the problem of SGWD, as well as some recommendations for future work.

Table 5 is an overview of the length of the shoreline in the various quadrants of the Eastern Shore. Inspection of Table 5 indicates that there is almost twice as much shoreline exposed on the Chesapeake side of the Eastern Shore. Tables such as this show different important attribute sums by quadrant of the Eastern Shore, as defined by county (Accomack or Northampton) and recipient water (Chesapeake or Atlantic). This method allows quantitative comparison of the data, rather than the qualitative nature of the previously discussed maps.

Table 5. Length of shoreline (km) by county and water feature

	Chesapeake	Atlantic	Sum
Accomack	321	204	525
Northampton	289	137	426
Sum	610	341	951

Discharge-Associated Results

This subsection will examine the aspects of this research associated with the derivation of the volumetric discharge information for the Eastern Shore. Permeability, gradient, and discharge potential data are presented and discussed.

Figure 15 is a view of the permeabilities along the Eastern Shore on a large scale. This permeability map illustrates the effect of erosion on near-shore soil types. The southwestern portion of the Eastern Shore (a higher energy nearshore environment not protected by barrier islands or wetlands) typically has nearshore soils of greater permeability than the rest of the Eastern Shore. Figure 16 shows the segments in marsh areas. Note the conspicuous absence of marsh areas in the southwestern quadrant of the map. These values for permeability are quantitatively shown in Table 6. Also notice the order of magnitude difference in the nearshore permeabilities for east and west Northampton County. Nearshore SGWD is more likely to occur in the higher permeability soils, and therefore in this quadrant of the map. SGWD may also be occurring in the areas where the less permeable silt loam is present, but the groundwater may be discharging farther offshore, possibly due to the presence of the marshes (see Nippert, 1993).

Table 6. Mean permeability along Eastern Shore (cm/hr)

	Chesapeake	Atlantic	Both
Accomack	12.27	3.00	8.69
Northampton	23.0	2.51	10.82
Both	17.3	2.82	12.14

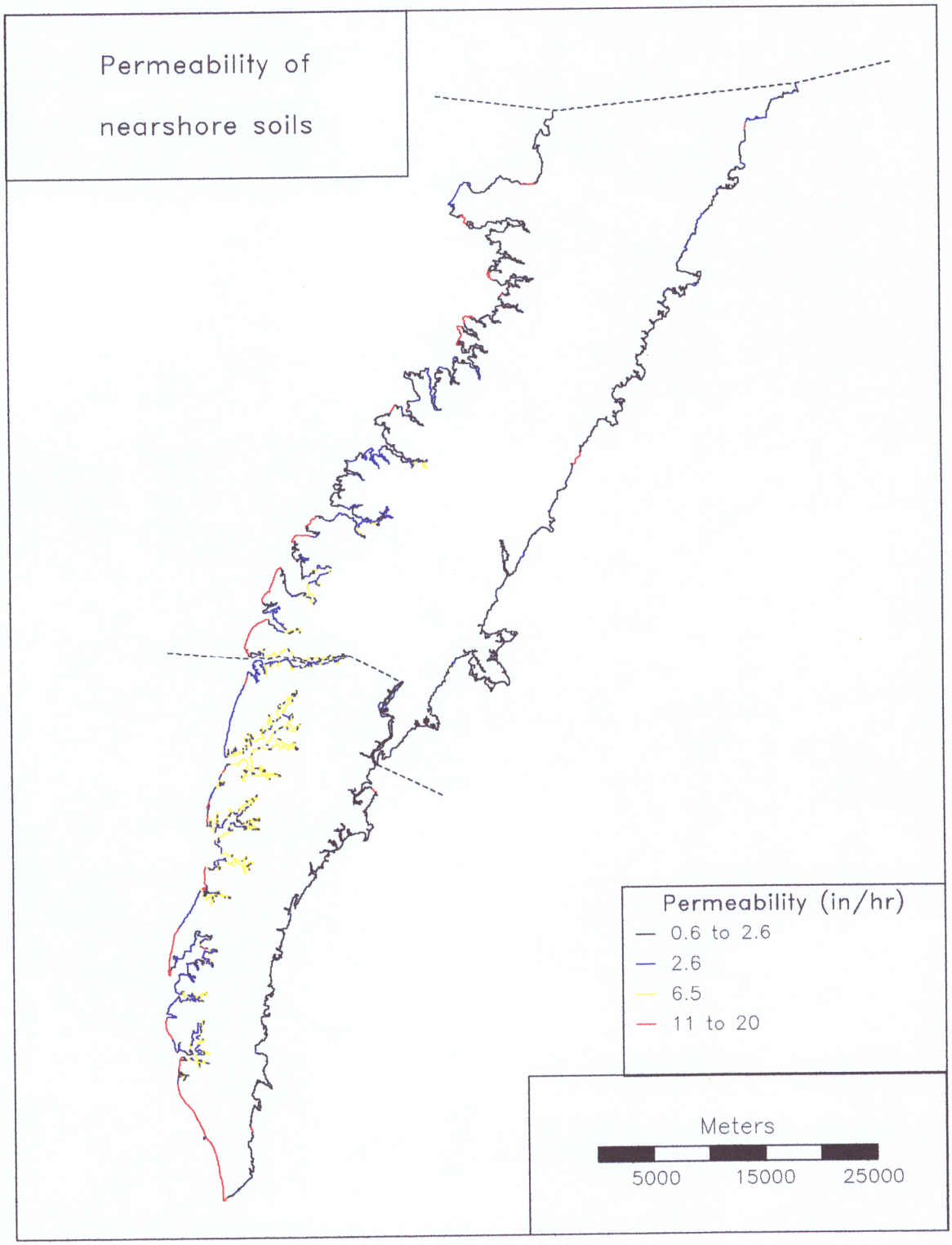


Figure 15. Permeability along the Eastern Shore

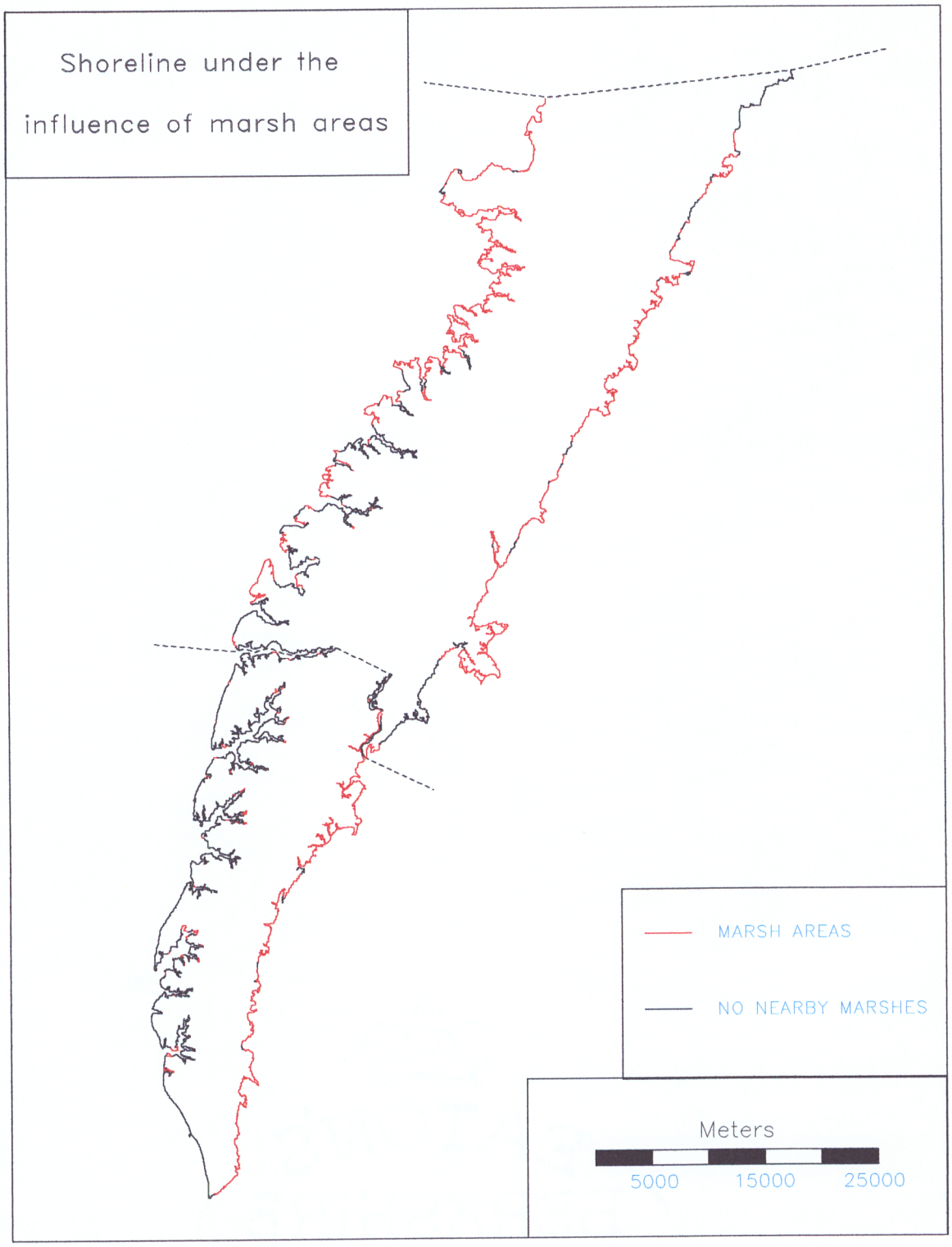


Figure 16. Marsh areas

The higher permeability type of nearshore soil configuration is also likely to see greater saline groundwater discharge from periodic tidal action causing greater mixing between the fresh groundwater and the proximal Chesapeake Bay water (see Riedl *et al.*, 1972). Thus, saltwater may be cycled more quickly through these environments. Measurements of vertical hydraulic conductivity were made in several locations and the relative correlation between field measurements and model output is displayed in Table 7. This table shows that while nearshore soil types are related to nearshore vertical conductivity, they are not an interchangeable quantity and the permeability of a specific soil type should be taken as displaying only the trend in nearshore vertical conductivities. It has also been suggested that in low energy coves the nearshore soil type may not be reflective of the nearshore conductivity because of silt settling from the water column in low-energy locations (Reay, 1993, personal communication).

Table 7. Vertical hydraulic conductivity versus nearshore soil permeability

Sample location	Permeability (GIS model) (cm/hr)	Measured Hydraulic Conductivity K_V (cm/hr) (Reay and Gallagher, 1991)
Eyreville	50.8	15.8
Old Castle	16.5	8.1
Eyrehall	16.5	3.96
Scotts	1.524	0.09

Figure 17 is a gradient map, showing the 10 foot (3.048 m) water table contour line that yielded the gradient. The gradient is steepest in the finger-like inlets on the Chesapeake side of the Eastern Shore. Table 8 lists mean gradient for each of the quadrants on the Eastern Shore.

Table 8. Mean hydraulic gradient along Eastern Shore (m/m)

	Chesapeake	Atlantic	Both
Accomack	0.0018	0.0024	0.0020
Northampton	0.0030	0.0028	0.0029
Both	0.0023	0.0026	0.0024

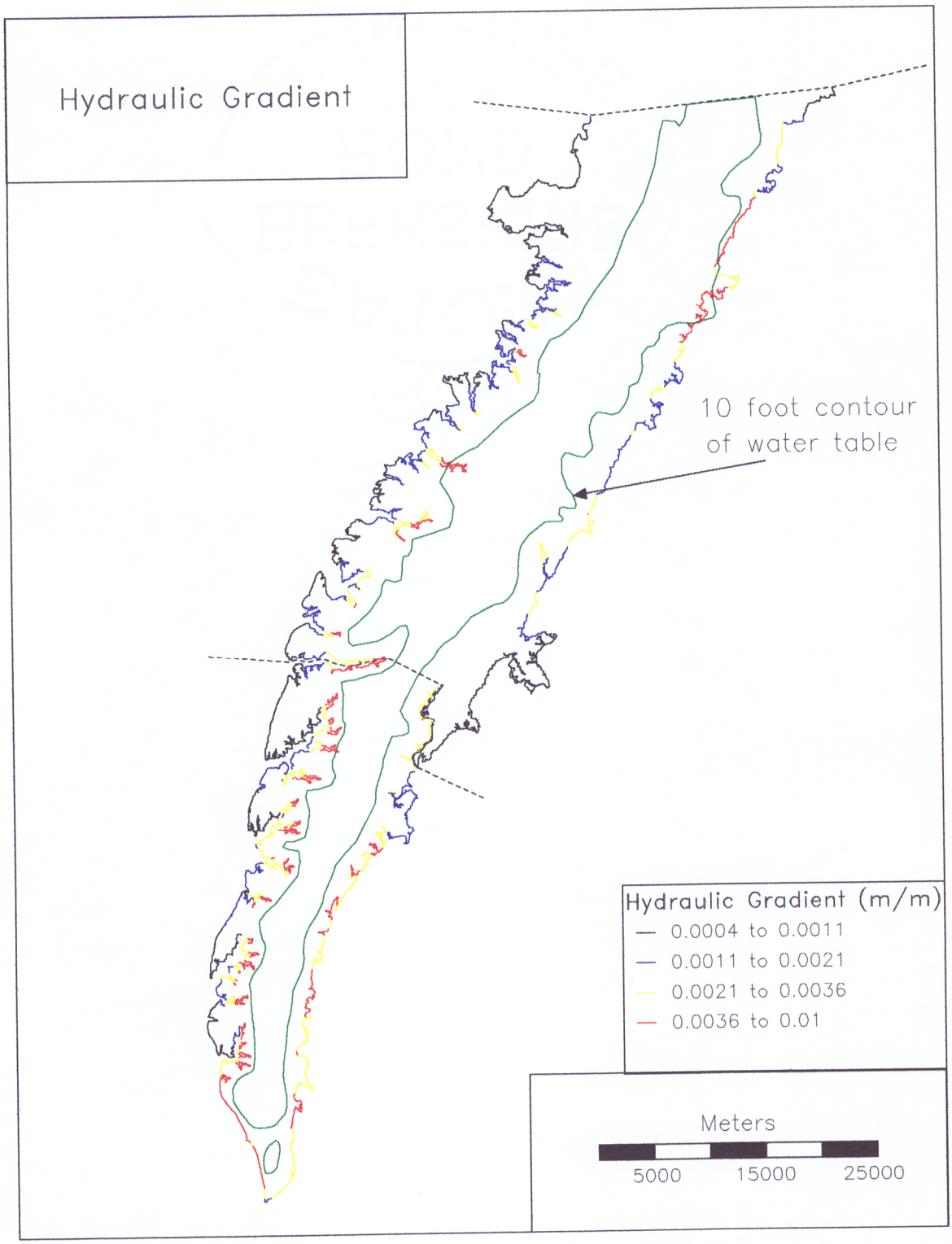


Figure 17. Hydraulic gradient along Eastern Shore

The gradient map shown in Figure 17 presents the proximity of a particular discharge boundary segment to the ten foot (3.048 m) contour of the Columbia Aquifer. Figure 18 represents a plot of this gradient beginning at the northwest corner of the Eastern Shore, continuing along the perimeter and ending at the northeast corner. The peaks are errors associated with the water table map. In Figure 18 the spurious gradient value where the 10 ft contour line crosses into open water can be seen. If these errors are removed, Figure 19 may more accurately represent the gradient along the Eastern Shore. A corrected gradient value of 0.01 was assigned a for every segment with a gradient over 0.01. If the Eastern Shore can be thought of as Van der Veer's homogeneous strip island, the values associated with each segment are proportional to discharge. Although Van der Veer's solution calls for many assumptions that are questionable, such as homogeneous aquifer and uniform recharge, we can assume that on a large scale this map can also represent the volume of fresh water being discharged. Following this reasoning, the large scale discharge of fresh water is relative to the gradient map for a homogeneous aquifer assuming no recharge or leakance. Under these assumptions the aquifer is uniform and steady-state, so discharge will be proportional to the hydraulic gradient only.

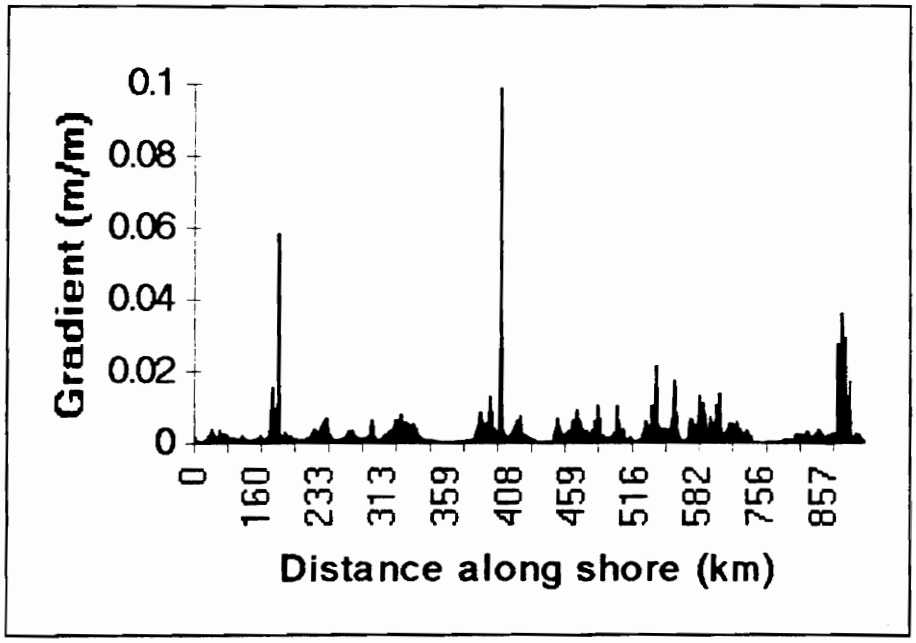


Figure 18. Gradient v. length along shoreline

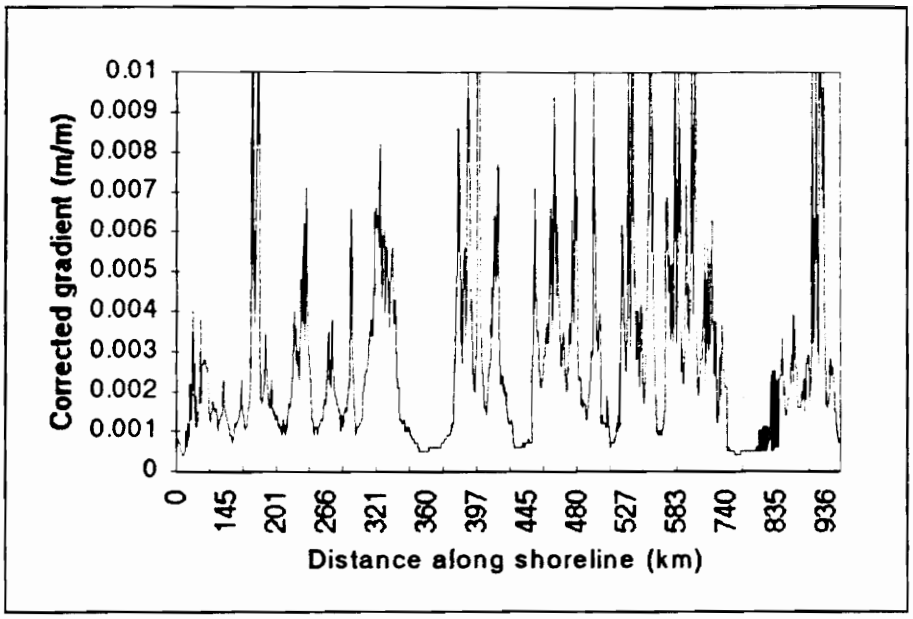


Figure 19. Corrected gradient along Eastern Shore

The two previously discussed map layers, permeability and gradient, illustrate the most important aspects of the groundwater flow system. First, the large scale gradient represents the driving force of the freshwater discharge. Second, the permeability map reflects both a parameter for the smaller scale behavior pattern of groundwater discharge with respect to volumetric groundwater discharge, as well as the ability of a segment to transmit water. High discharge areas were predicted by calculating which areas show not only the ability to transmit water, but also the driving force to discharge. The discharge potential, seen in Figure 20, is a straight multiplication of the i segments' attributes for gradient and permeability as follows:

$$\text{discharge potential}_i = \text{gradient}_i * \text{permeability}_i \quad (6)$$

Thus the multiplication of these attributes results in the combination seen in Figure 20. A plot of discharge versus length is given in Figure 21. By visual inspection we see that the most severe discharge was predicted to occur along the Chesapeake shore of Northampton County. This is corroborated quantitatively by the data in Table 9. These values, like the nutrient flux potentials are dimensionless, and are measures of potential to discharge groundwater. The eastern side of the Eastern Shore typically shows low

likelihood of nearshore discharge with its vast stretches of thick, dense Chincoteague silt loam.

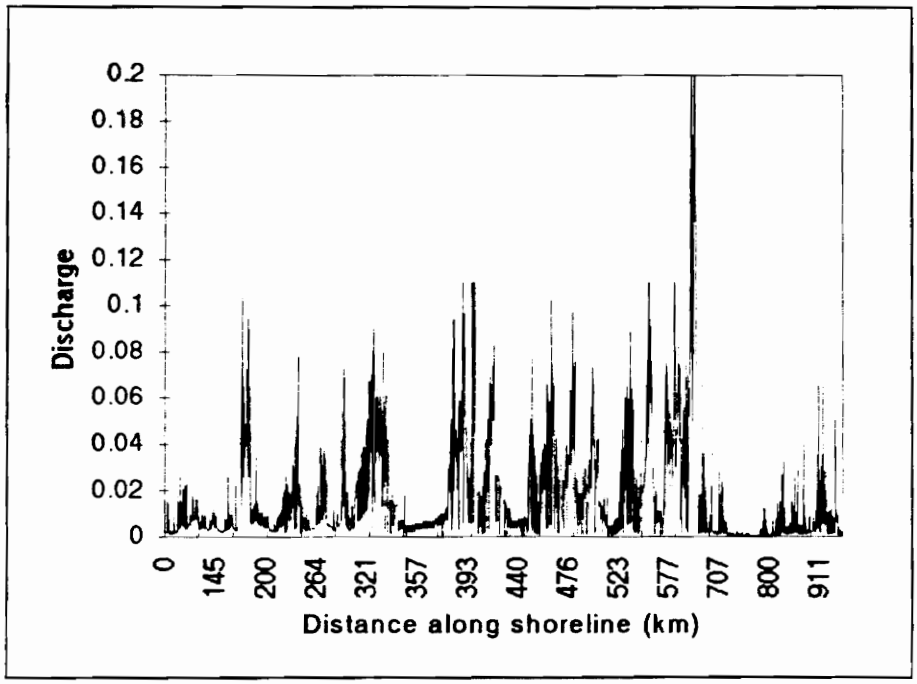


Figure 20. Discharge potential along Eastern Shore

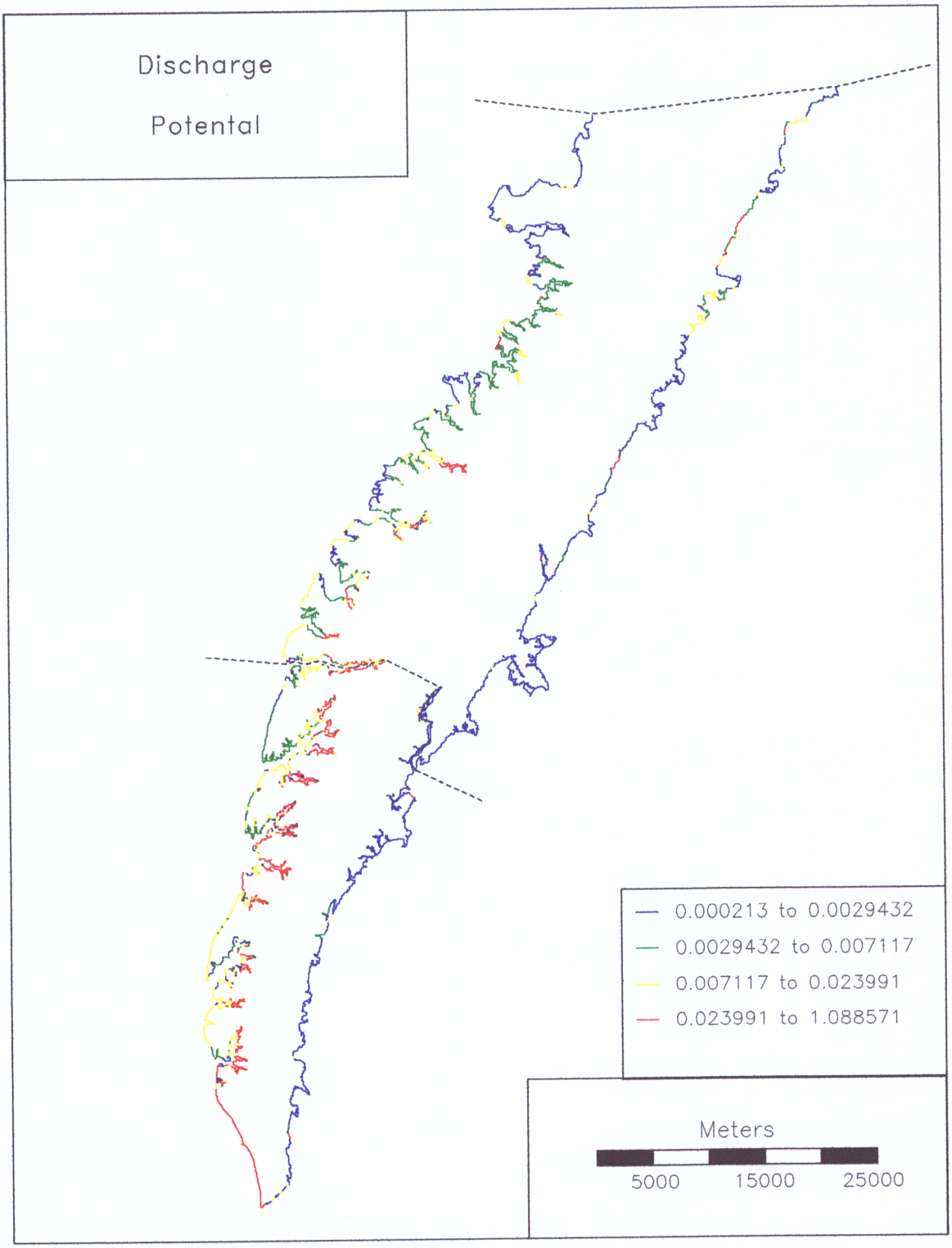


Figure 21. Discharge potential for the Eastern Shore

Notice in Figure 17 that the gradient is steeper on the southern Atlantic side of the peninsula. The same region in Figure 20 predicts little discharge because of the low permeability of the silt loam in this region (Figure 15). That the gradient is steep may be a consequence of the low permeability soils near the discharge zone acting as an aquaclude, actively preventing the movement of water towards the Atlantic thus elevating the water table beyond that of a homogeneous aquifer. The marsh areas where this soil type is abundant can be seen in Figure 16. It has been suggested that groundwater may flow under clay lenses in marsh areas, to be discharged farther from shore (Nippert, 1993; Reay, 1992). The driving down would also increase resistance to flow causing the aforementioned elevation of the water table.

Table 9. Mean volumetric flux potential

	Chesapeake	Atlantic	Both
Accomack	0.0084	0.0030	0.0063
Northampton	0.0279	0.0027	0.0198
Both	0.0176	0.0029	0.0123

Contamination-Associated Results

Figures 22, 23 and 24 show urban, pasture, and agricultural land use on the Eastern Shore. These were the features considered in this study as potential sources of pollutants to the surficial aquifer. All features past the one km buffer were eliminated from consideration, and a proximity function was developed for each of the segments. This function is the contamination potential with a measure of severity of:

$$\text{contamination potential}_i = \text{area}_i/\text{distance}_i \qquad (7)$$

where the area_i and distance_i are the area of and distance to the nearest potential contaminating feature. Segments with more than one type of contaminating feature were summed, and equal weighting was used for each type of contamination source. Sensitivity of this equal-weighting assumption is discussed later. Figure 28 is the map resulting from the above measure of severity. Some segments did not have any contaminating feature within the 1-km buffer, and were represented by black lines. For example, in Figure 28, black lines indicate that there is no contaminating feature associated with that boundary element.

Maps of the segments under the influence of urban, pasture, and agriculture land use are seen in Figures 25, 26 and 27, respectively. Notice the high densities of agriculture are on the southwestern edge of the Eastern Shore. Fields are typically very close to the discharge boundary in this region, and coupled with the high discharge potential previously discussed, would cause this region to experience the most damaging SGWD. Agricultural areas are not only vulnerable to contamination by nutrient input, but toxics from pesticide use as well. This vulnerability, however, has not been reflected in discharge measurements of pesticides in Eastern Shore groundwater (Hubbard, 1993, personal communication). Unlike agricultural land use, urban land use only accounts for a small portion of discharge, as shown in Figure 25 (and Table 10). The areas that are under the influence of urban land use are the areas at risk of contamination from coliforms. Figure 28 includes many areas that have no potentially contaminating feature, so discharge (no matter how extensive) can be eliminated from consideration.

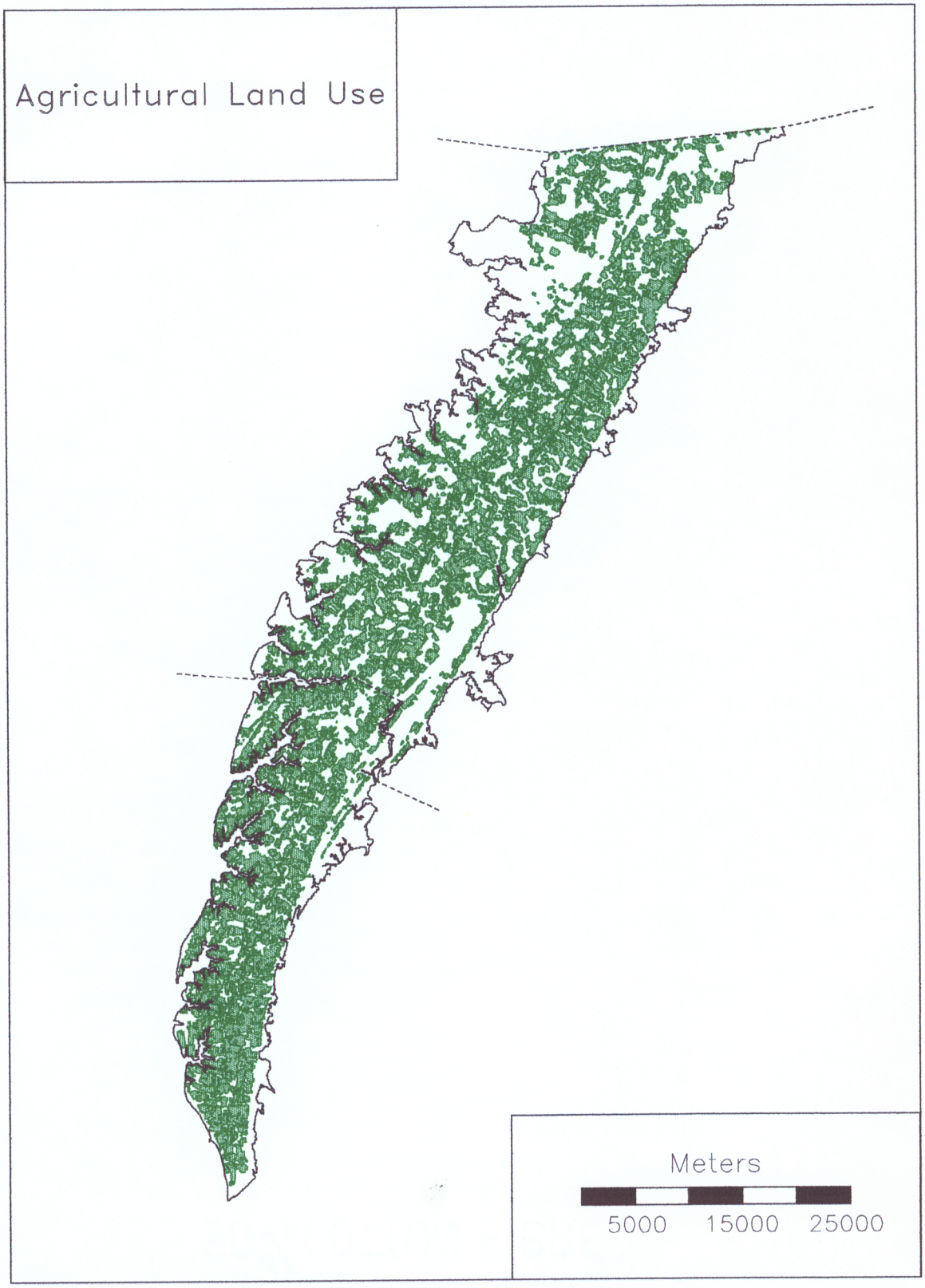


Figure 22. Agricultural land use

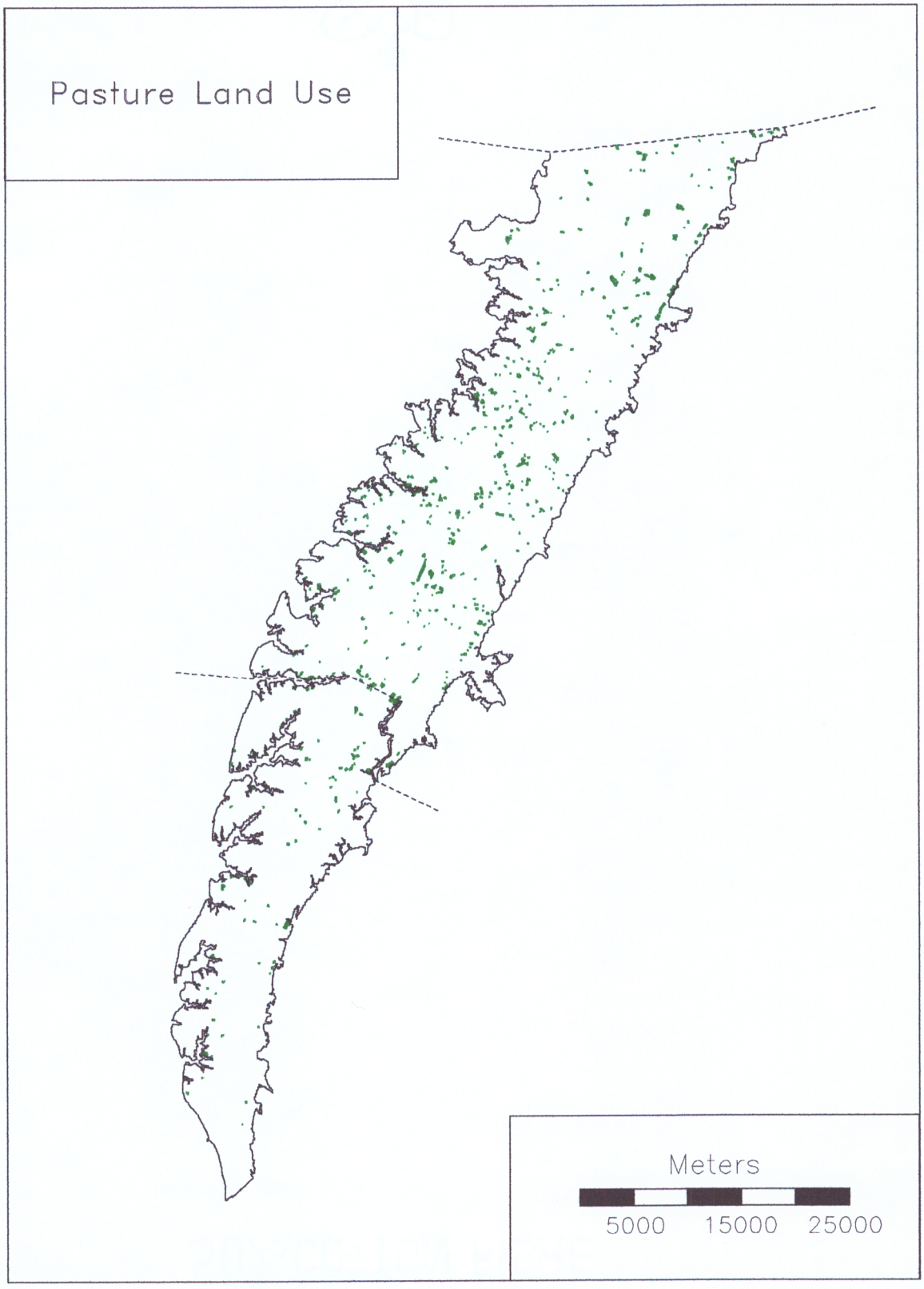


Figure 23. Pasture land use

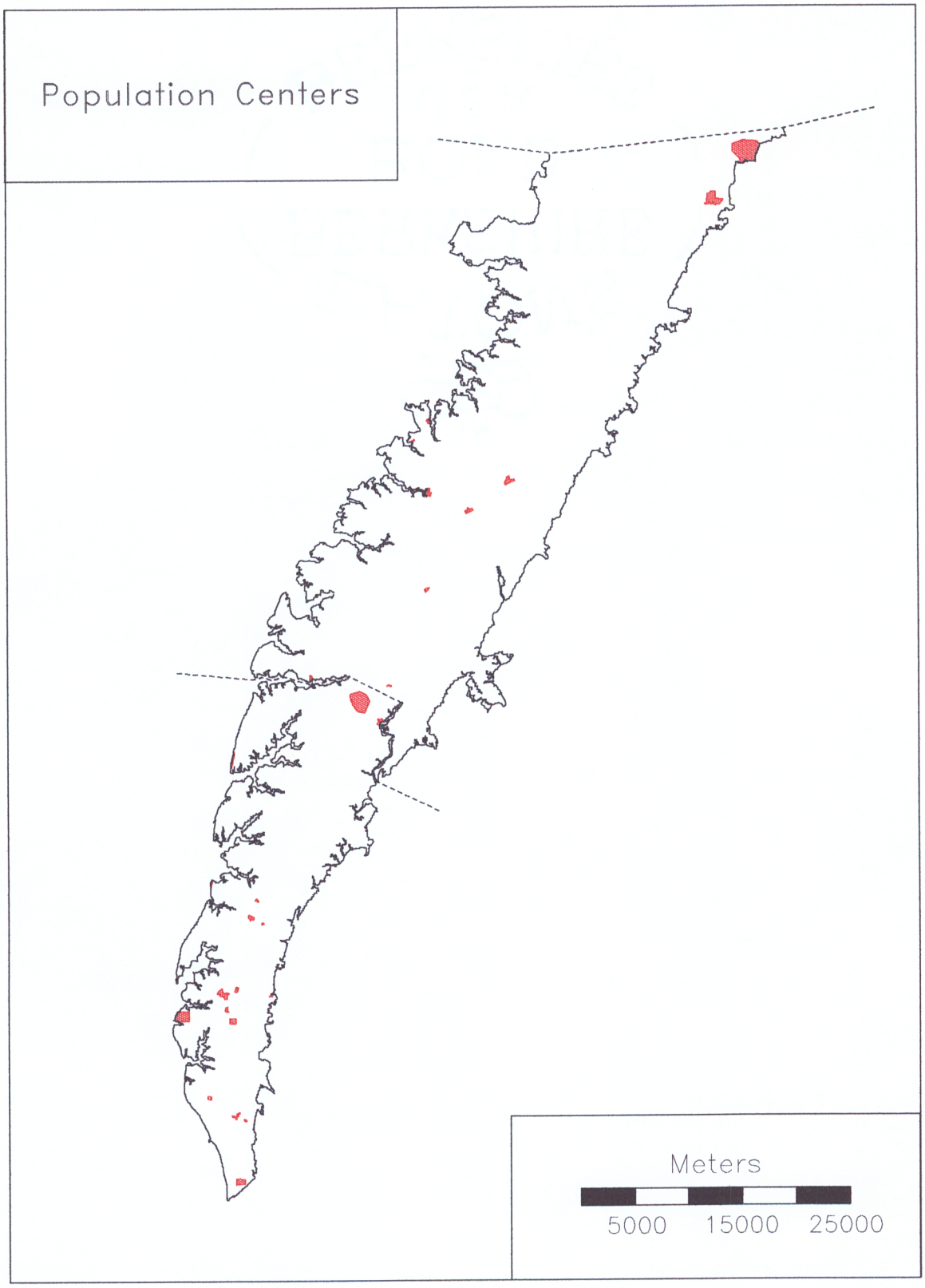


Figure 24. Nearshore population centers

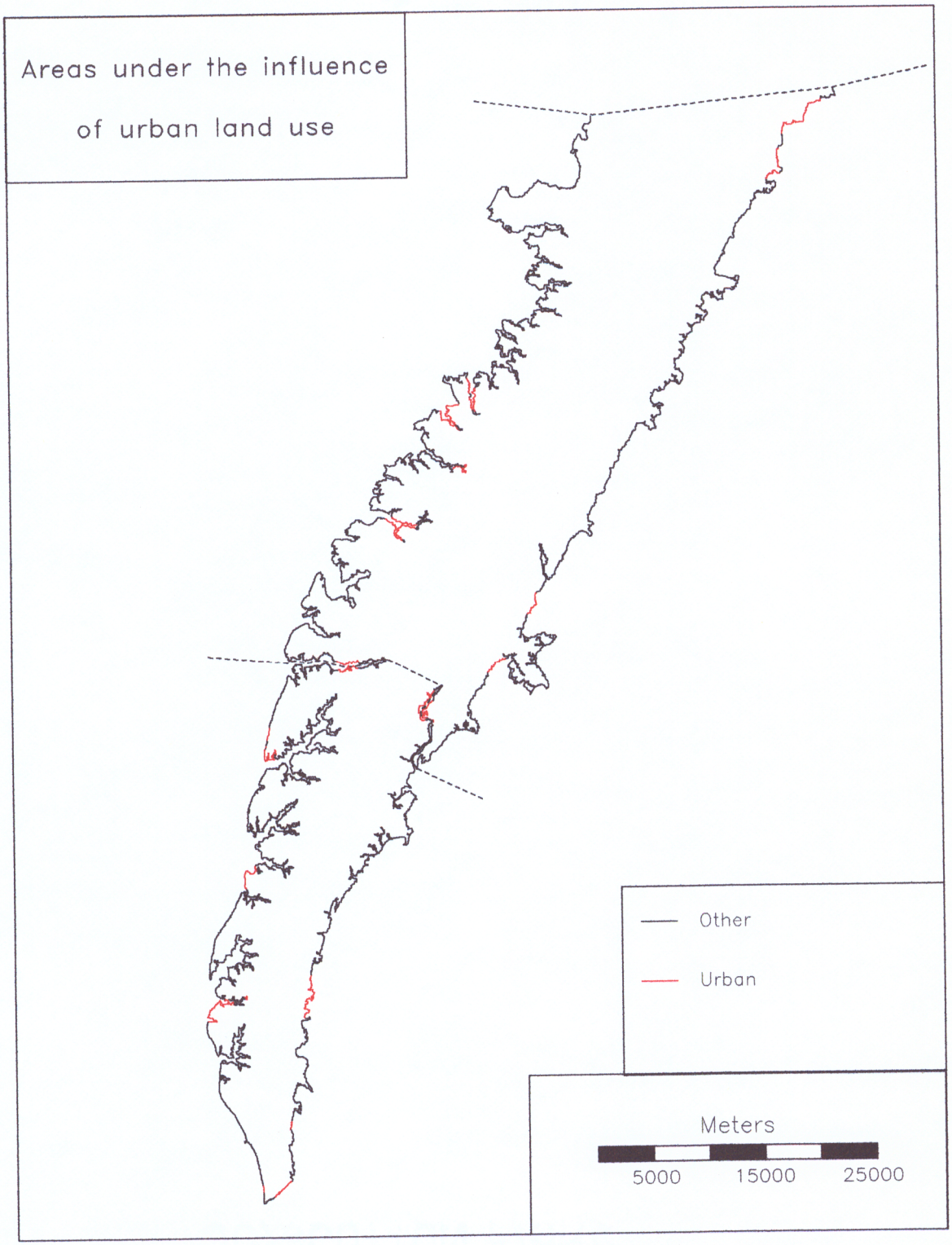


Figure 25. Segments under the influence of urban land use

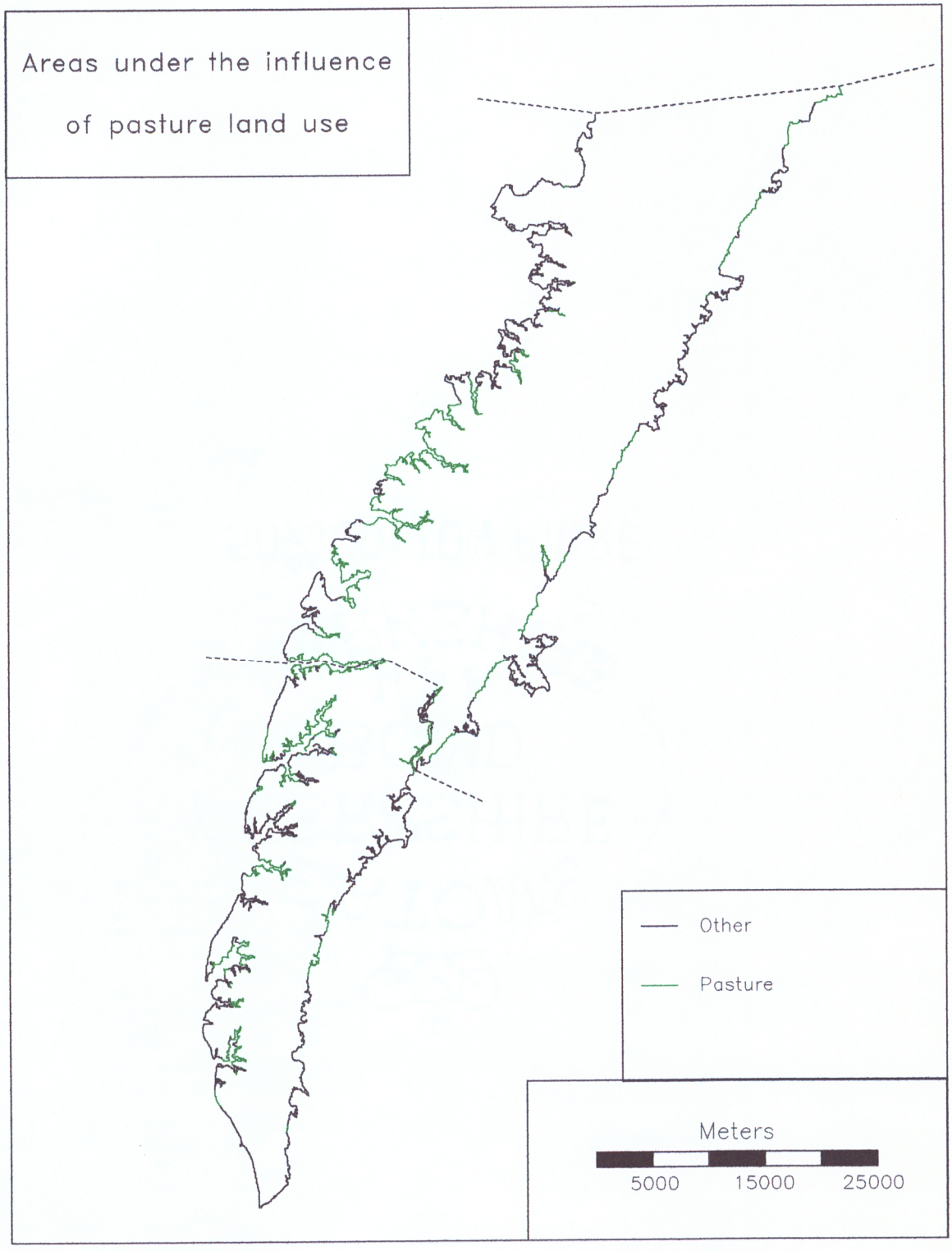


Figure 26. Segments under the influence of pasture land use

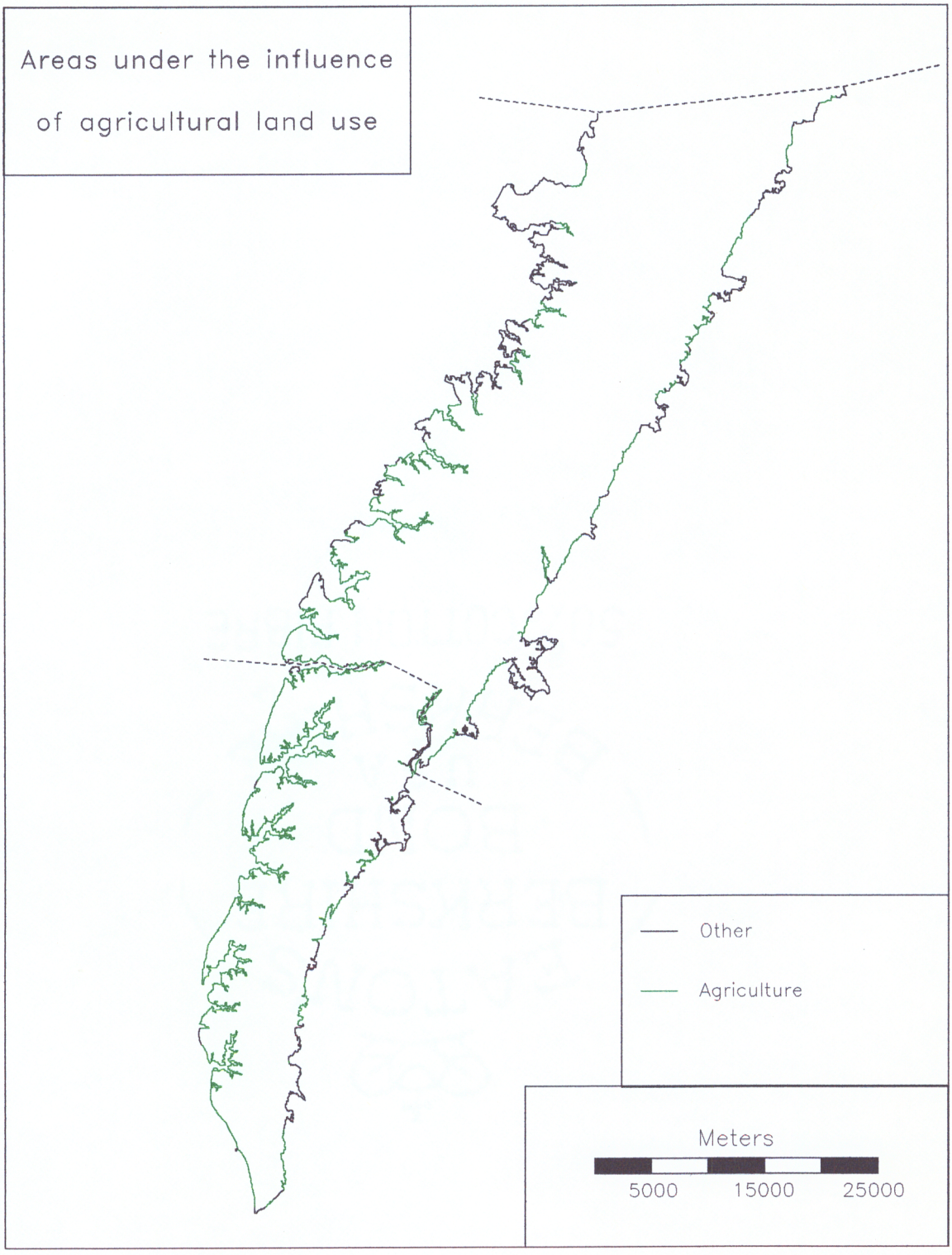


Figure 27. Segments under the influence of agricultural land use

Table 10. Nearshore land use by recipient water

	Chesapeake Bay (km)	Atlantic Ocean (km)	Overall (km)
Agriculture	455 (75%)	131(38%)	587 (62%)
Pasture	282 (46%)	78 (23%)	357 (38%)
Town	45.3 (7%)	27.3 (8%)	72.6 (7.7%)
No feature	137 (22%)	173 (51%)	311 (33%)

Table 10 shows the total length of shoreline devoted to various land uses on the Atlantic, Chesapeake, and the combined total. These numbers denote the presence of a contaminating feature, without regards to the level of contamination from agricultural, pasture or town. There is about a quarter of the shoreline that has no contamination potential. An even smaller amount is the length of shoreline devoted to towns. Notice that the Chesapeake Bay side has much more agriculture and pasture land use than the Atlantic side. Coupled with the higher likelihood of high discharge in this area, this area should see elevated nutrient flux.

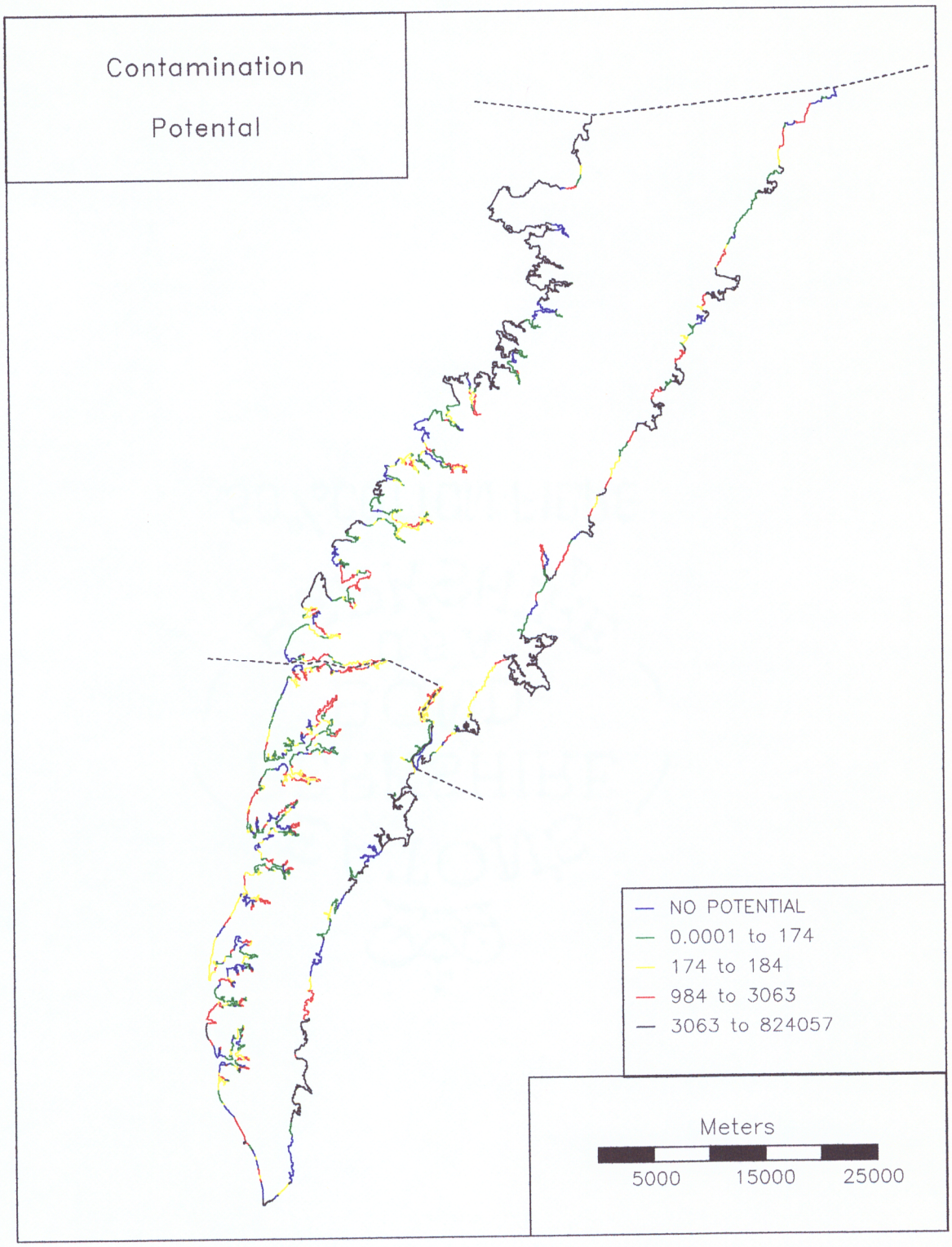


Figure 28. Contamination potential

When the contamination map was combined with the discharge map, the result was a measure of the relative ability of discharge through a particular boundary segment to contaminate surface water. Figure 29 is the result of the measure of contaminant flux potential:

$$\text{flux potential}_i = \text{contamination potential}_i * \text{discharge potential}_i \qquad (8)$$

for each of the i segments. Figure 29 is a combination of a segment's ability to transmit water, driving force, and likelihood to contaminate. The information plotted in Figure 30 was the data from Equation 8 versus length around shoreline, starting in the northwest corner of the study area. The red areas are areas of highest likelihood to contaminate. Again, western Northampton County was predicted by this model to be the most adversely affected by contamination. A segment with no contaminating feature was represented on the map by black. Table 11 represents the sum of the discharge severities from Equation 8, that is, the sum of the product of discharge and nutrient flux potential for each segment, without regard for the variability of pollution potential between agricultural, pasture, and urban land use. It shows that 93% of the contaminating discharge will occur into the Chesapeake Bay rather than the Atlantic Ocean.

Western Northampton County, as well as the southern portion of western Accomack County exhibited a combination of permeable soils, moderate hydraulic gradient, and dense agriculture that led to a high likelihood of contamination. Qualitatively, the majority of contamination from nearshore SGWD was predicted to occur in this area. The overall picture of nutrient flux via SGWD can be shown by Table 11, a tabular form of Figure 22. Note first the factor of nearly two between total shoreline on the Chesapeake Bay side and the Atlantic Ocean side, next the average permeability. Note that most of the higher permeability soils are on the western side, showing that nearly eight times as much discharge may be happening on the Bay side. Then note the differences in contamination. Not only did the model predict higher discharge on the Chesapeake Bay side, but an order of magnitude greater contamination.

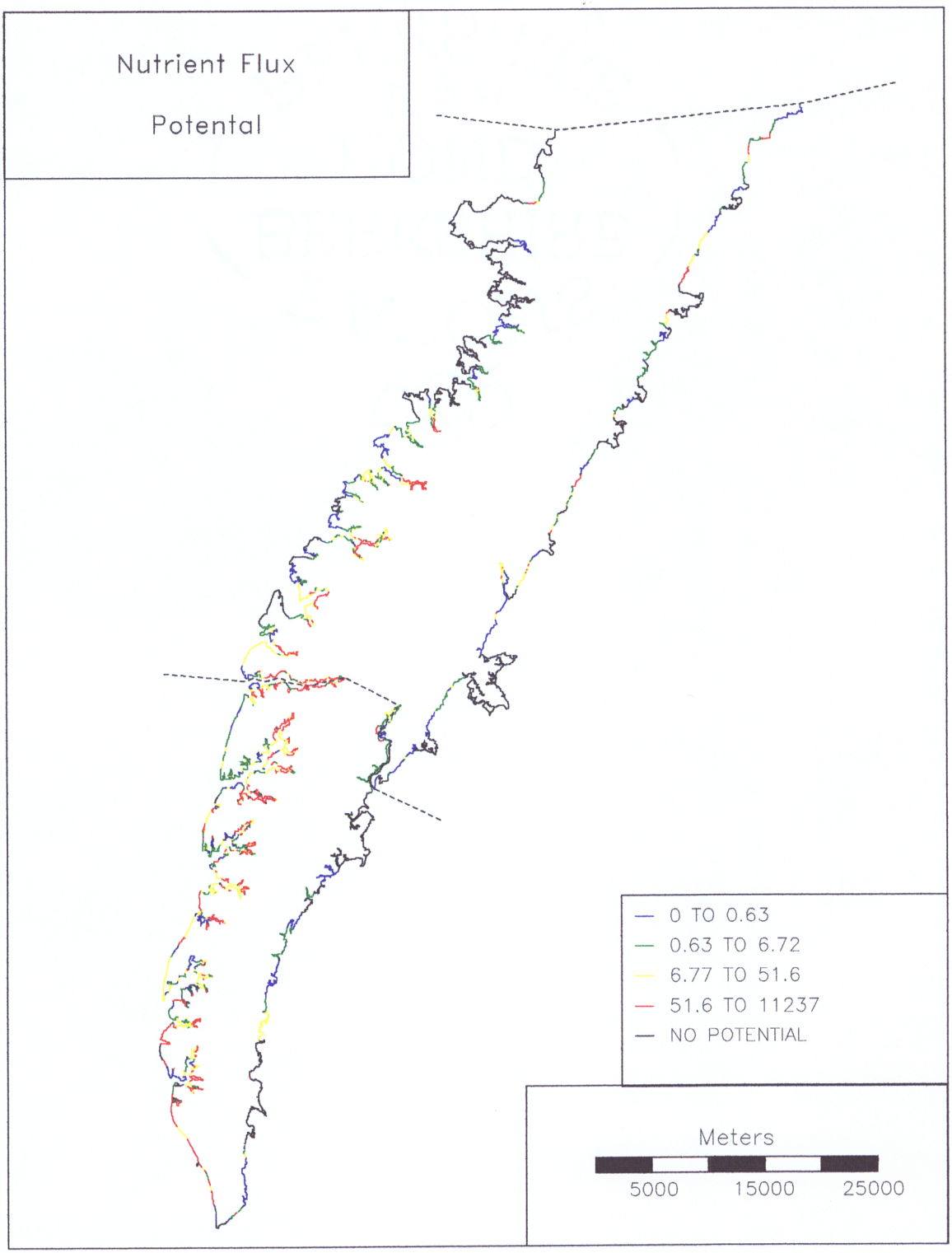


Figure 29. Nutrient flux potential

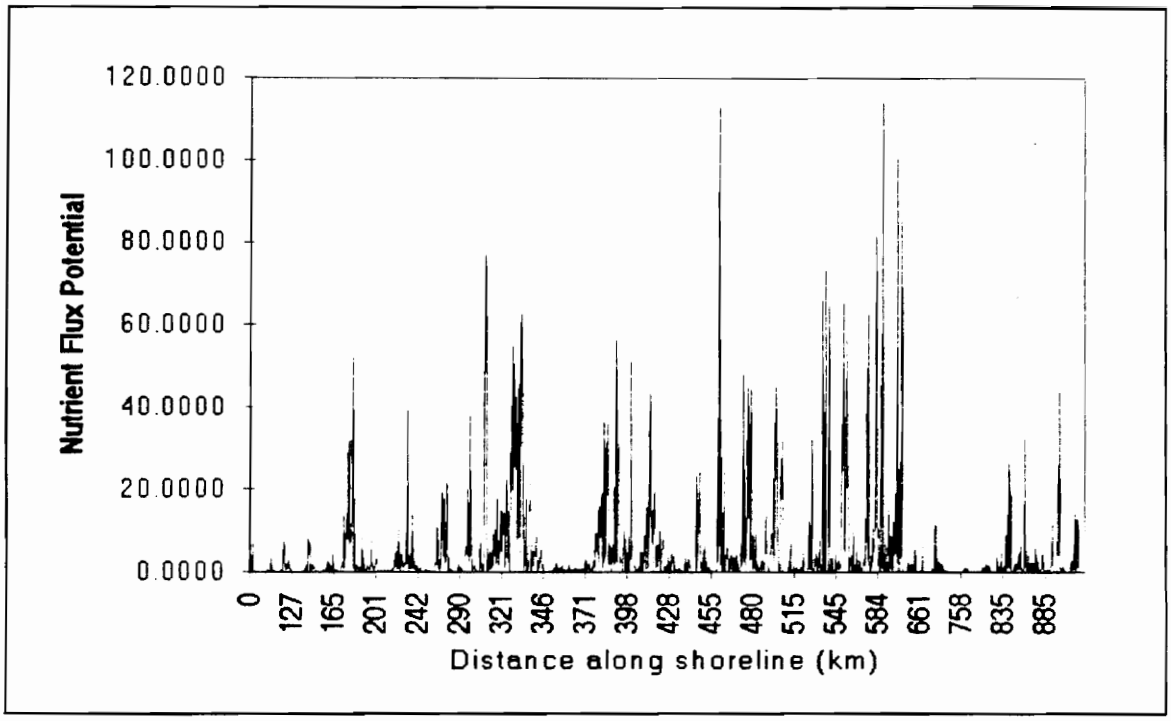


Figure 30. Nutrient flux potential v. length for the Eastern Shore

Table 11. Unscaled nutrient flux potential by county and recipient water

	Chesapeake	Atlantic	Sum
Accomack	490 (19%)	151 (6%)	667 (25%)
Northampton	1900 (74%)	28.7(1%)	2019 (75%)
Sum	2390 (93%)	179 (7%)	2685 (100%)

Table 12. Max, min, and mean for the attributes

statistic	Mean	Min	Max	Std Dev
length of segment (m)	477	0.111	13000	976
soil permeability (in/hr)	4.79	0.6	20	4.74
hydraulic gradient (m/m)	0.002	0.000355	0.0989	.00402
discharge severity	0.0128	0.00213	1.089	.0376
contamination potential	219.4	0	3500	312.74
nutrient flux potential	5.08	0	136	12.24

Table 12 presents some summary statistics for the attributes for each of the i segments for the entirety of the Eastern Shore.

Sensitivity Analysis

To understand the effects of different weighting schemes, two other weighting schemes were chosen besides the equal-weight scheme already discussed. One scheme counted urban land use as being half again as much as agricultural and pasture land use, and the other scheme counted pasture land use as half of agricultural and urban land use. Both schemes differed from the equal-weight scheme by less than two percent overall for nutrient flux potential. This indicated that the equal weight scheme chosen was not sensitive to variable contamination from the different land uses. The lack of distinction between different land uses is appropriate for the large scale, but locally, the different schemes may change the nutrient flux potential for a given small area.

Correlation to Sampled Areas

The quality of this method of modeling was evaluated by examining several sites where studies have been conducted with seepage meters, wells, and mini-piezometers. The best example is Cherrystone Inlet, on the western shore of Northampton County. Figures 31, 32, 33, and 34 show permeabilities, gradient, discharge and nutrient flux quantiles, respectively. Figure 31 includes the soils layer, and it is possible to see how the perimeter was segmented by the soils map. Also, at this scale it is easy to see the variation in gradient with distance to the 10' water elevationin Figure 32. Table 13 shows the sampling areas in Cherrystone Inlet (Reay, 1993) and the values of model output for both discharge and flux potential, as marked on Figures 33 and 34. The model output corresponds well to these areas, both in discharge as well as nutrient flux, as shown in Figures 36 and 37.

Figure 35 is a contour plot of nitrate concentrations in Cherrystone Inlet. Notice that the location of high concentration in Figure 35 is near Eyreville (Figure 34), a location of high discharge. The model did not, however, predict the other area of high contamination in the upper left of Cherrystone Inlet in Figure 35. This area may be the result of a stream discharging into the inlet at this location. No conclusion can be reached, however, because no discharge measurements were made in this area, and the nutrient flux from SGWD in this area is unknown.

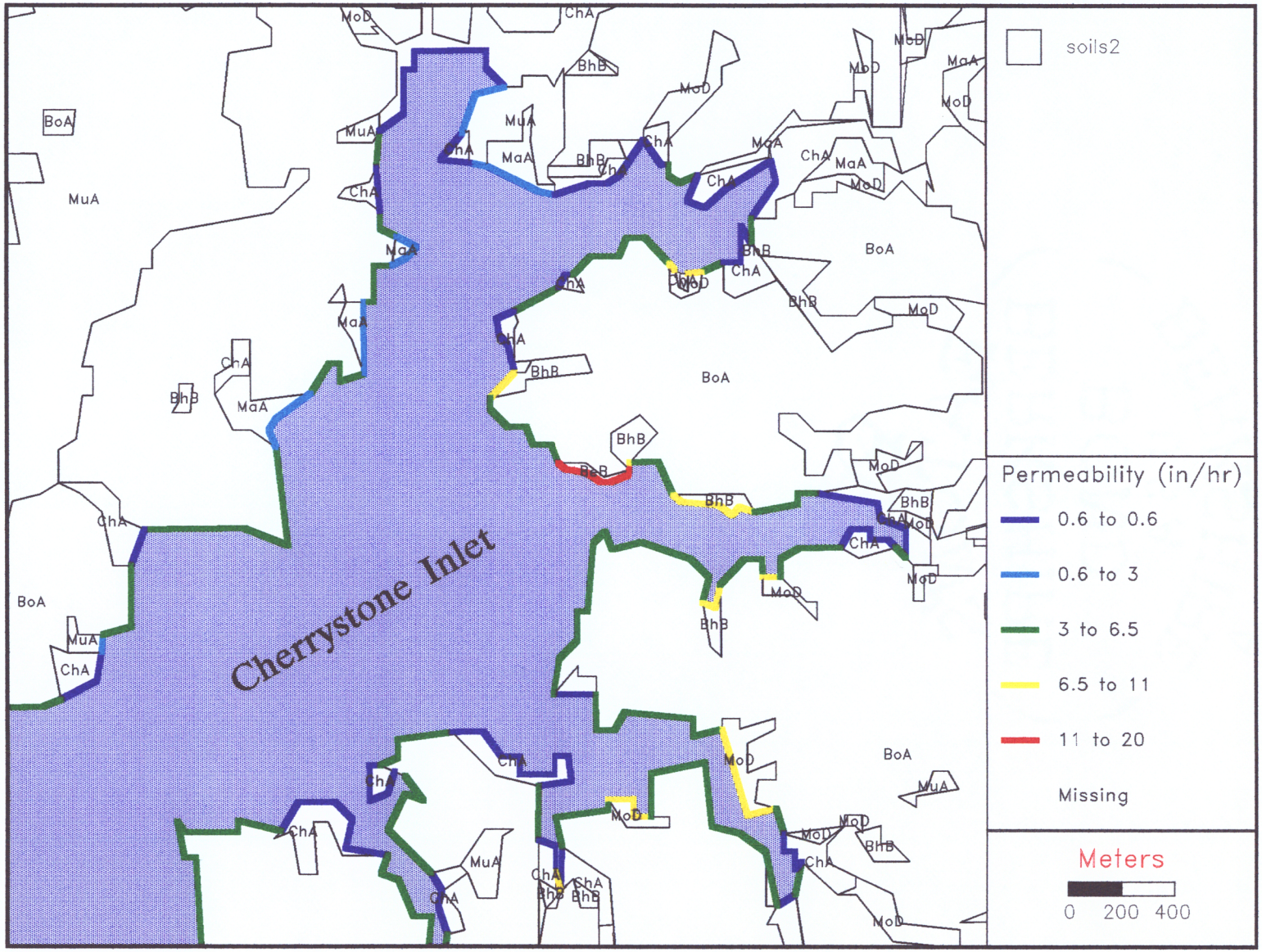


Figure 31. Cherrystone Inlet permeabilities and soils

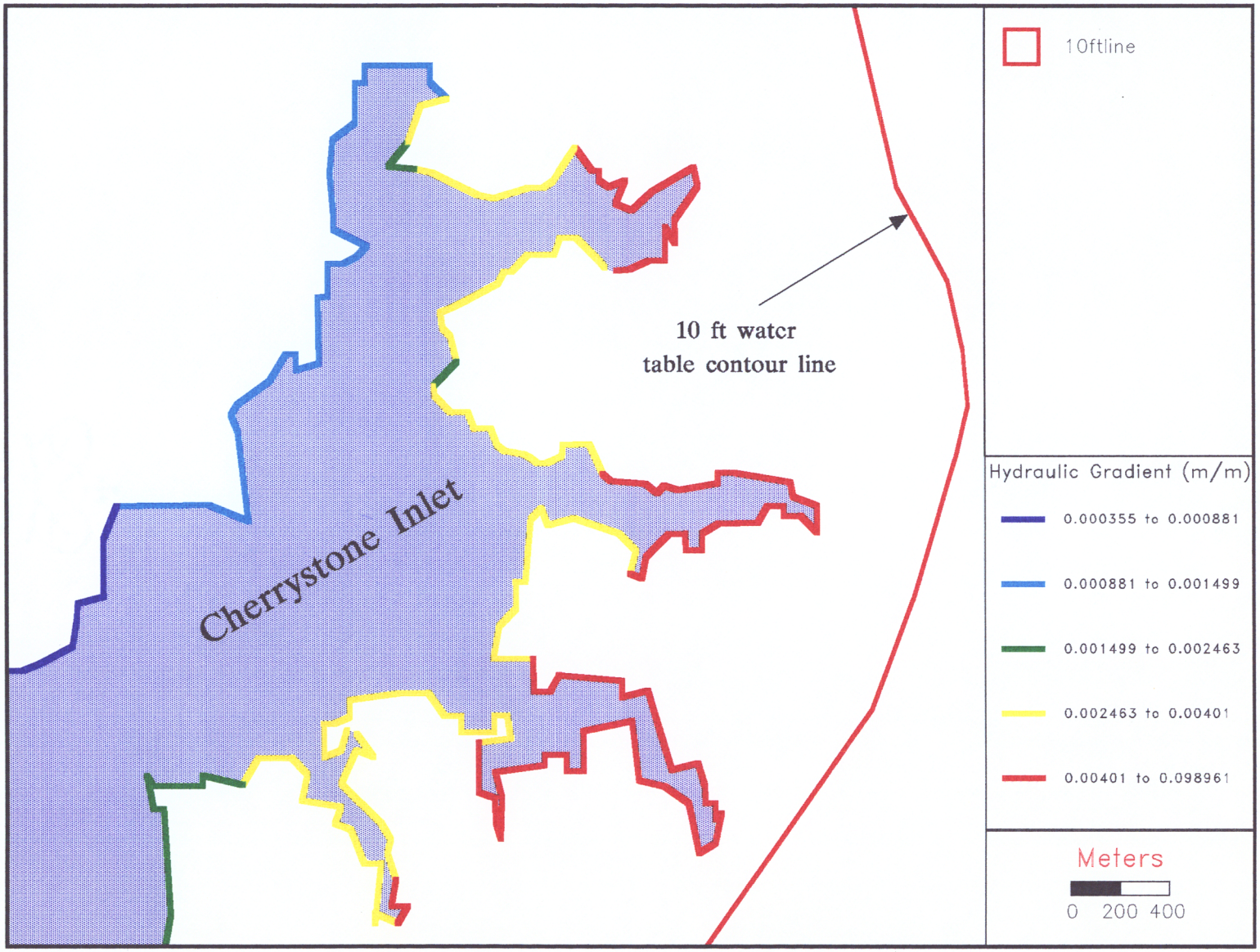


Figure 32. Cherrystone Inlet hydraulic gradient and water table elevation

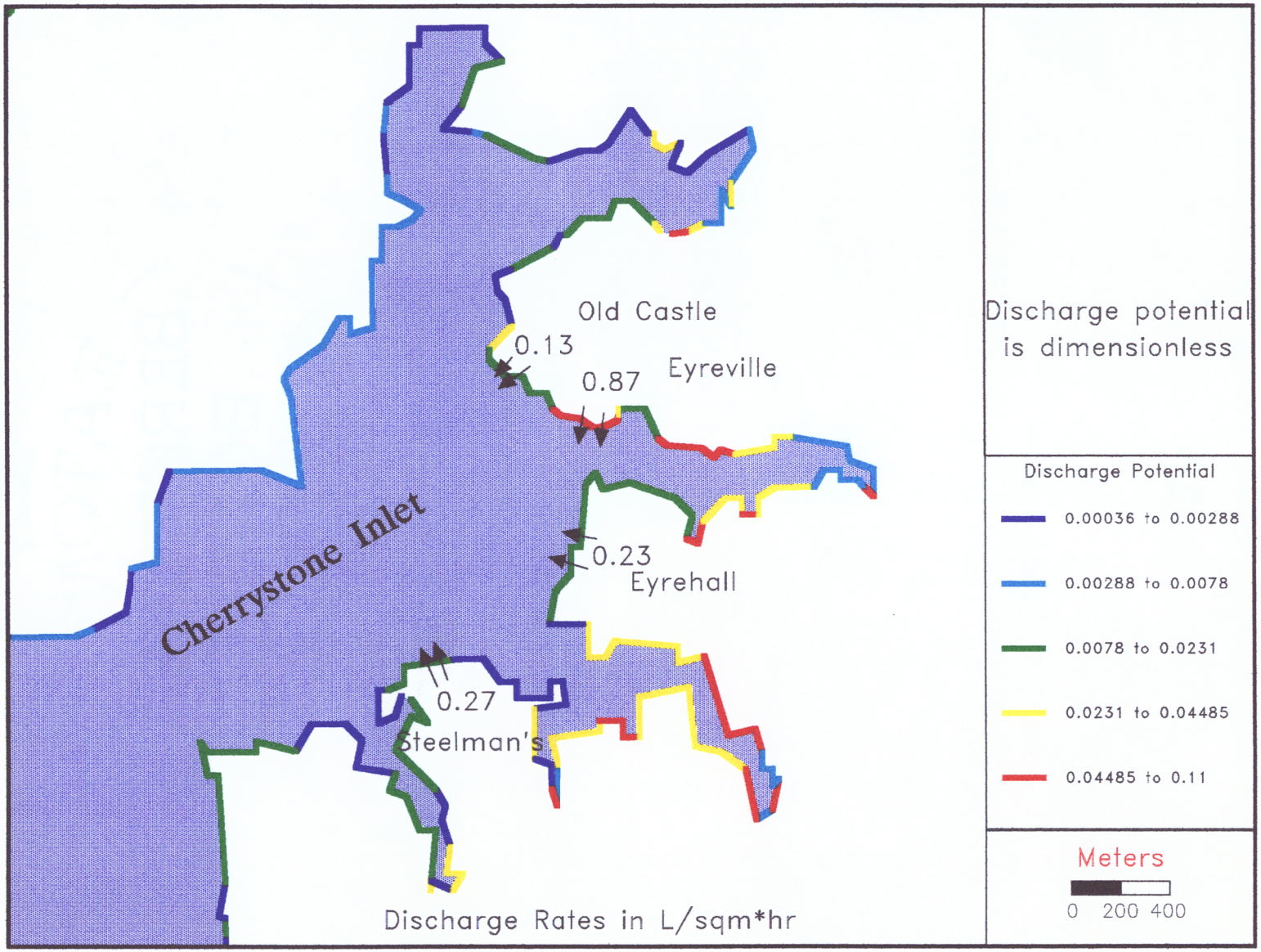


Figure 33. Potential discharge, Cherrystone Inlet

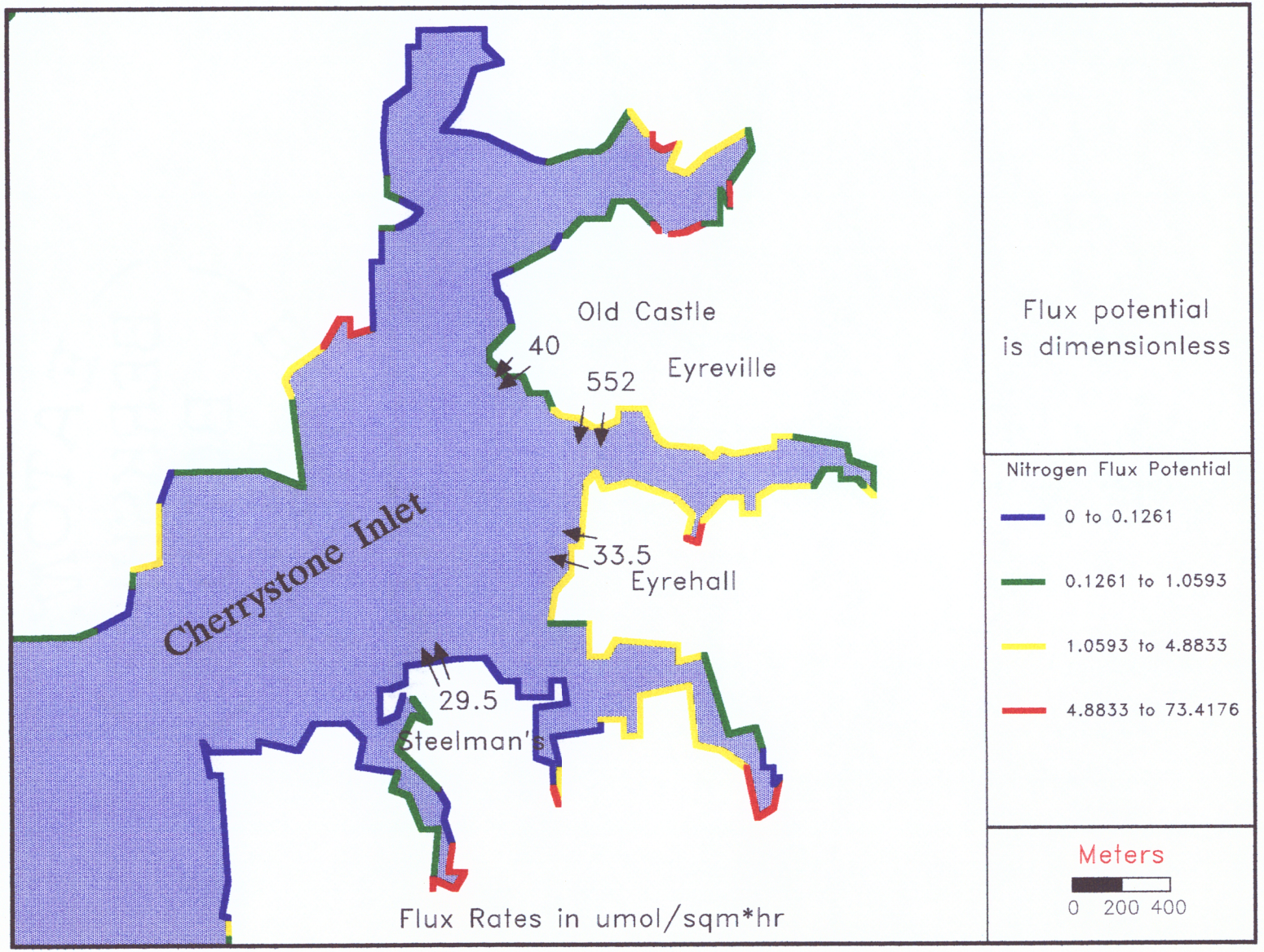


Figure 34. Potential for nutrient flux, Cherrystone Inlet

Table 13. Field measurements and model results

	measured discharge L /sqm*hr	predicted discharge severity	average nitrogen flux (umol/sqm*hr)	flux potential
Old Castle	0.13	0.0164	40	0.690
Eyrehall	0.23	0.0203	33.5	2.5
Steelman's	0.27	0.0183	29.5	0.101
Eyreville	0.87	0.0596	552	3.24

Another area of comparison was Steelman's Landing (see Figure 14), on the eastern edge of Northampton County. (Note Steelman's Landing is not the same as the Steelman's referred to in Cherrystone Inlet). The model predicted very low discharge because of a small gradient as well as low-permeability Chincoteague silt loam. In addition, predictions of nitrate concentrations were low because of the distance of agricultural fields from the discharge zone. As predicted by the model, both low discharge rates as well as low contamination were found by Simmons (1988).

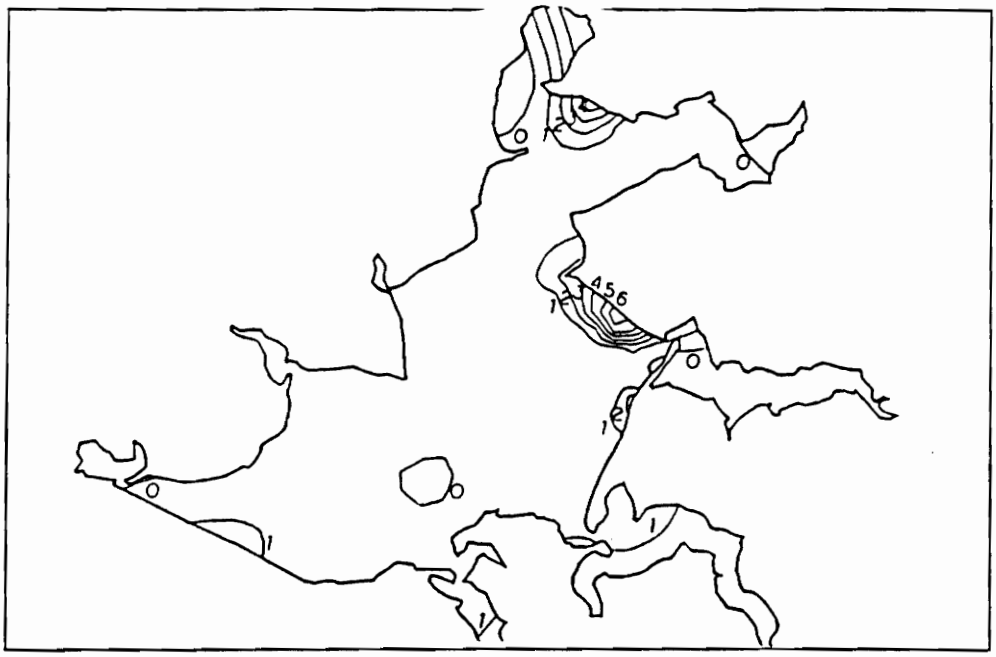


Figure 35. Nitrate concentrations in Cherrystone Inlet

(After Reay and Simmons, 1992)

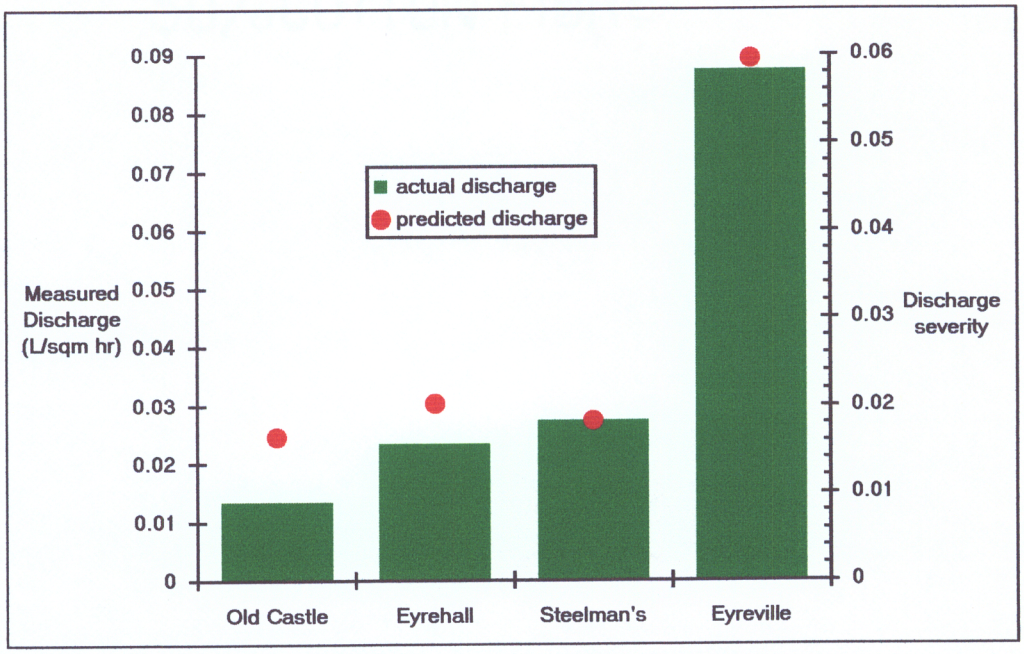


Figure 36. Volumetric discharge correlation

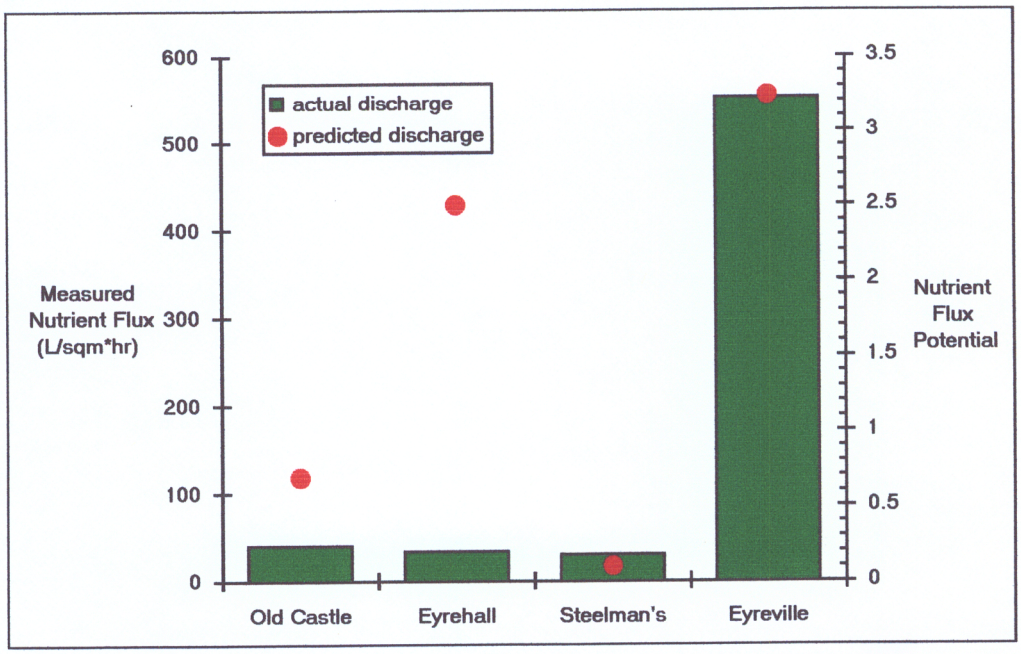


Figure 37. Nutrient Flux correlation to sampled sites

Figures 36 and 37 and the previous discussion indicate that the model seems to be correlated well on a local scale. Because these were the only data for the Eastern Shore, a comprehensive calibration of the model is unfortunately impossible. The volumetric flux in Figure 36 was very well correlated, showing that the model can predict gross SGWD under these conditions. The nutrient flux, which was not as well correlated, shows that factors other than proximity to agricultural features may be influencing nutrient flux from the sediments. These influences will be discussed in the Limitations subsection.

Limitations

Like all models, this model has limitations and sources of error. The most significant introductions of large scale errors stemmed from the representation and manipulations of the 10 ft line as digitized from Richardson (1992). This map was not meant to be an accurate map of the ten-foot-contour line, and digitizing added more (but hopefully small) errors. There is even an area where this line crosses into open water, which can be seen in Figure 17. All digitizing errors aside, it is impossible for the piezometeric to be 3.048 meters above sea level, 100 meters out into a channel unless the aquifer becomes semi-confined by the marsh feature. If this is true, it would support Nippert's (1993) work about semi-confined flow beneath marsh areas. Another variability in the contour is fluctuations of about one meter between rainy winter and spring and the drier summer and fall. The error introduced here, however, is constant with respect to all segments, and thus irrelevant for a year-long average.

Another problem associated with the 10 ft contour appears as a result of the limitations of the GIS. The gradient is determined by the absolute distance of the boundary feature from this line. In addition to the inability to determine local variations in gradient, the shortest distance along land from the boundary feature is not always a straight line. Notice the spit of land marked with an x in Figure 14. The straight line distance from the water table to the segment is likely shorter than the path follwed by the water. Although a more accurate distance can be found manually, time constraints outweigh the improved accuracy that would be achieved by this method. Instead, the waypoints were used as described in Chapter III and the error generated by this was small because this model was driven more by the order-of-magnitude differences in permeability. In addition, the total error introduced by misplacement of the gradient line itself is likely to

be on the order of magnitude of the error previously discussed. The groundwater flow regime may beinfluenced by other effects due such as rainfall ponding on the necks on the necks.

Localized, small-scale sources of error are likely to be very common due to the presence of hidden clay lenses in more permeable soils, or alternatively, particularly sandy or gravely areas within lower permeability soils. This source of error is believed to be highly local, and will be subsumed into the background on the larger scale of this model. Seepage meter data for calibration also has difficulties. The high variability of seepage is demonstrated while sampling; two meter readings just meters apart can frequently be off by a factor of two or more (Reay, 1992). Again, however, order of magnitude differences in actual seepage between low seepage sites and high was what this model focused on.

Another source of error was the variability of soil permeability with depth and within a family of soils. Sometimes an order of magnitude difference existed within the reported permeability of a particular soil (Cobb and Smith, 1989). It is hoped that the geometric mean taken from the available data will accurately reflect a typical soil type's permeability. A second problem is that the permeability values for FmD, BeB, AsE, AtD, CaA, FrB and FhB soils were all given "greater than" a certain value in the soil surveys. Fortunately, there were not too many examples of these types of soils on the shoreline. The minimum listed value was taken for these types of soil.

Another area of question was the length of the shoreline. As shown frequently in the arena of popular science, shorelines can be represented as fractals, meaning there is no absolute basis of determining the distance of the shoreline. On a larger scale, the perimeter will be smaller without great detail. As scale becomes smaller and smaller, more and more convolutions of the shoreline are represented by an increasing perimeter. Thus, the figure of about 950 km of shoreline used with resolution down to objects of 33m would decrease if resolution was to become larger, and increase if resolution was to decrease. This effect of scale should be noted, but its complete analysis is beyond the scope of this research. When comparing the GIS maps with corresponding topographic maps, the 33m resolution seemed capable of capturing significant features for this type of study.

It has been suggested, as previously mentioned, that low energy areas in coves may have lower permeabilities than the nearshore soil type would suggest. Preliminary evidence (Reay, 1993) also indicated that this may be true. The Scotts area was characteristic of this type of nearshore environment. While this area was predicted to have high discharge, limited data suggest that the sediments were actually taking up nutrients from the water column, indicating that the nutrient flux process may be shut down in this location. If this were true for the entirety of the shore, the high prediction for nutrient flux potential may be moderated because of the abundance of coves in this area. As more data become available about this aspect of SGWD it can easily be incorporated into the existing structure of this GIS model.

This work was also limited by the available data. Other factors, such as the effects of riparian zones, biological activity in the benthic zone, local gradient, and marsh areas have not been studied closely enough to allow their consideration in the flux potential developed in this research. As these topics are studied, they can be included in the GIS framework with the simple addition of new attributes for each of the segments. Despite all the limitations discussed in this section, there was still good correlation with field data. As more seepage data becomes available, a definitive calibration can take place.

A prediction of discharge

Bearing the aforementioned limitations in mind, two methods were used to determine raw discharge, the recharge formulation, and seepage meter estimates. Both can only be considered rough estimates, and significant errors should be anticipated. Every effort was made to produce a conservative result while including all the considerations concerning SGWD. The results should be regarded with the weight accorded them as discussed here and in Chapter III.

The Van der Veer formulation becomes similar to the mass balance, which can be reformulated to the scaled distance such that for each of the segments:

$$\text{discharge}_i = \text{recharge} * \text{length}_i / \text{distance}_i \qquad (9)$$

where recharge is 13 inches per year as discussed in the literature review, and the distance is the Van der Veer distance (the distance to the midpoint of the aquifer). This

formulation assumed a scaled recharge area from the midpoint of the aquifer. Other assumptions were that the aquifer is homogeneous, recharge is uniform, and the model corresponded to the strip island of Van der Veer (1977). In addition, we assume insignificant leakance, as well as no pumping. Table 14 summarizes the discharge, and Figure 38 shows the recharge distance, which is directly proportional to discharge. Table 15 and Figure 39 is the multiplication of the recharge distance by the contamination potential. Thus Table 15 contains dimensionless values, analogous to the nutrient flux potential.

Table 14. Van der Veer's Discharge by recipient water
(10^6 m^3/yr)

	Chesapeake	Atlantic	Total
Accomack	98.2 (35%)	96.1 (35%)	194 (70%)
Northampton	48.6 (17.5%)	35.5 (12.5%)	84 (30%)
Total	147 (52.5%)	132 (47.5%)	278 (100%)

Table 15. Contamination potential with Van der Veer's discharge
(10^6)

	Chesapeake Bay	Atlantic Ocean	Total
Accomack	8900 (30%)	9000 (30%)	17860 (60%)
Northampton	10000 (34%)	1640 (6%)	11640 (40%)
Total	18900 (64%)	10640 (36%)	29540 (100%)

It is noteworthy that with the previous assumptions, the wider areas of the Eastern Shore had the highest discharge (Figure 38) - just opposite of the permeability and gradient results. This significantly mediated the problem for nutrient flux in the southwestern portion of the study area when the recharge map was combined with the contamination map. Figure 39 shows a shift of pollution potential northward to these wider areas of the Eastern Shore, but most of the high risk areas remain the same. Northampton County also still had the highest nutrient input for this scenario as well, showing the dominating role the contamination potential plays in determining nutrient flux potential. The study area is 1386 square kilometers, which is slightly larger than the total area of 843 assumed by this method. This helped yield a more conservative estimate of volumetric discharge. The overall volumetric estimate is on the order of the $460*10^6$ m^3/yr found by the mass balance method (the aquifer receives yearly 13 in/yr over 1386

sq. m). For comparison, the flow in the James River is approximately $18*10^6$ m^3/day. While this mass balance approach may be valid on the large scale, it did not correlate well to the sampled sites. It is, however, another method of looking at the causes behind SGWD, especially recharge.

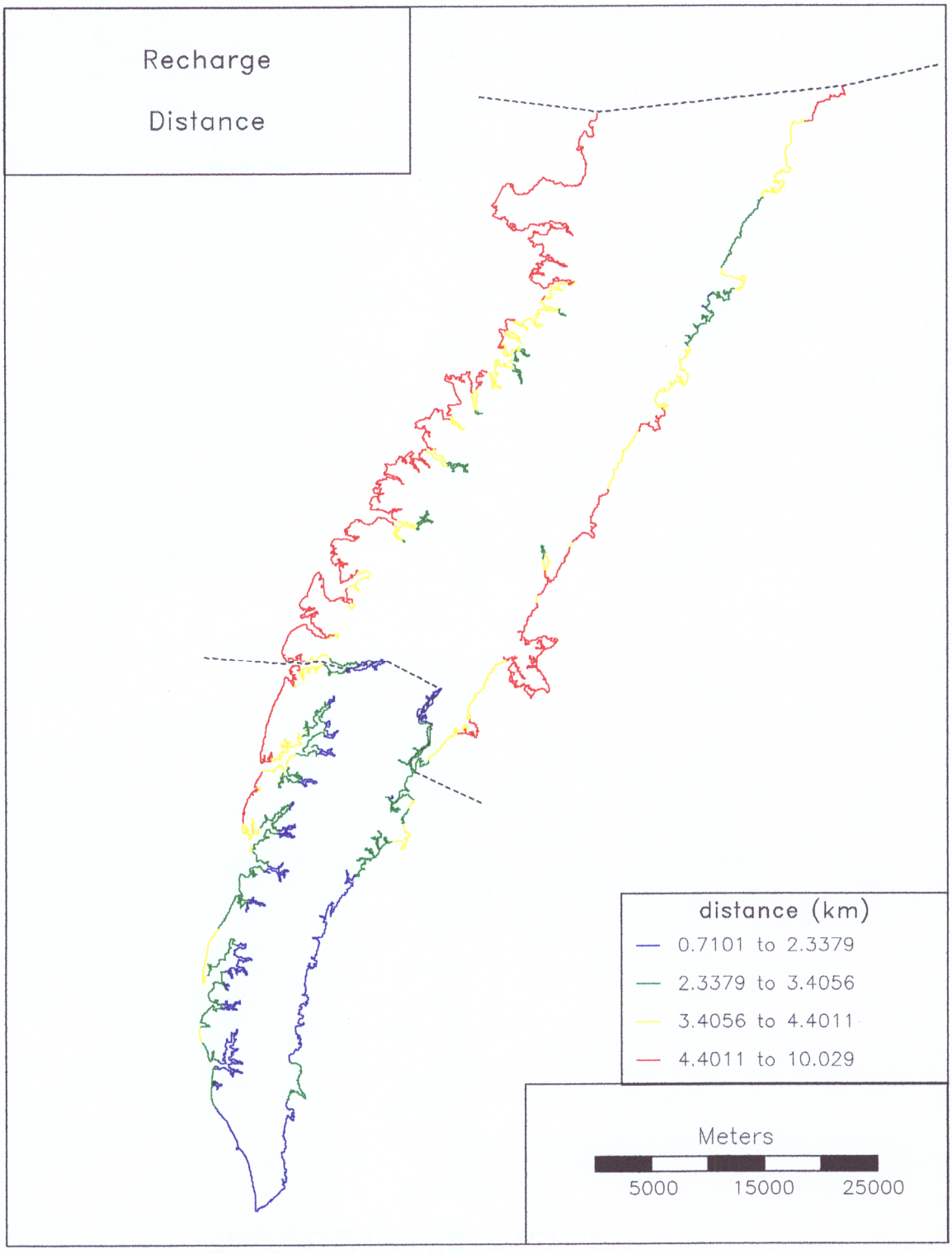


Figure 38. Recharge distance

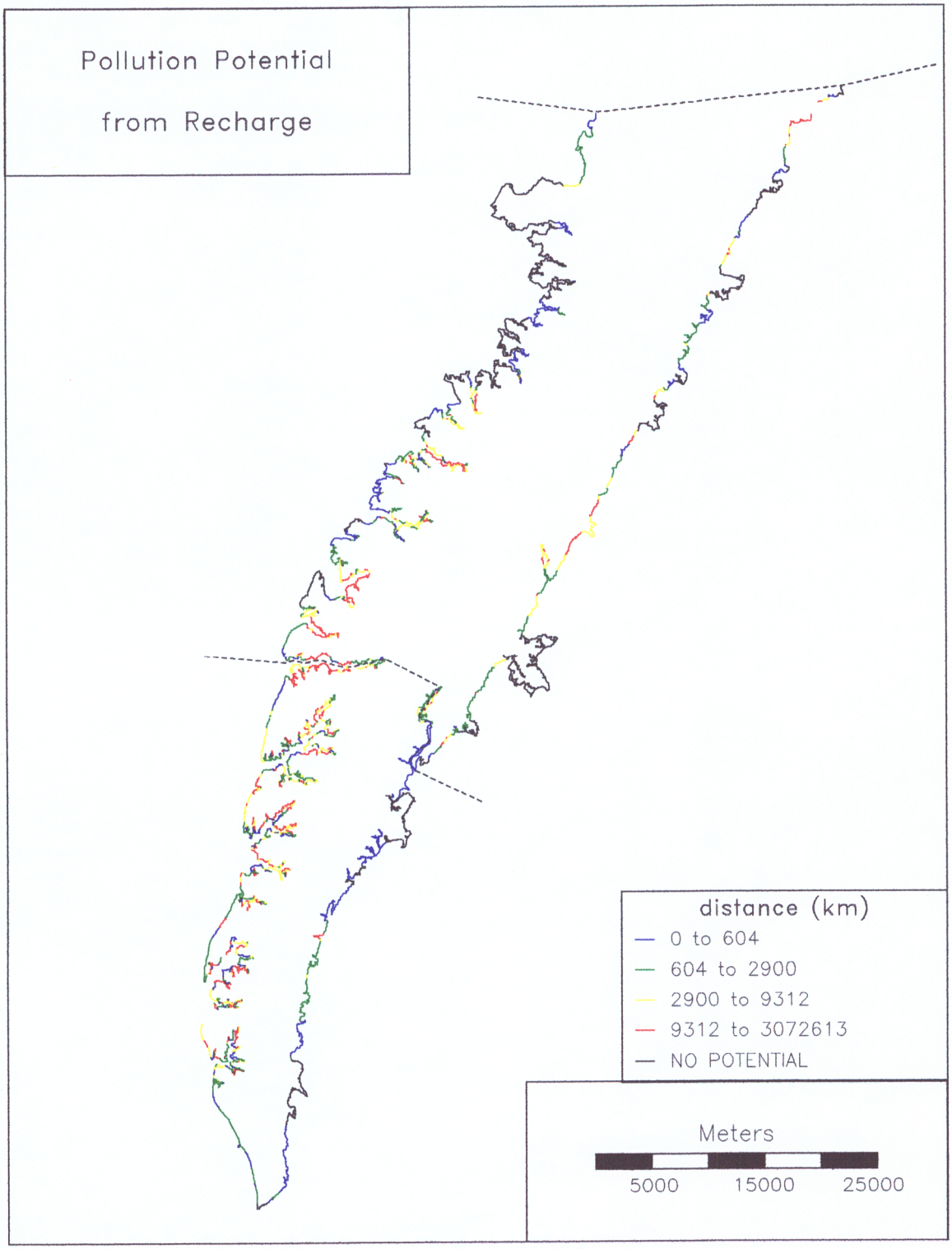


Figure 39. Recharge combined with contamination potential

The method of seepage meters used the same data found in the Correlation subsection to obtain a result for discharge and nutrient flux. Each segment's potentials were calibrated to the seepage meter readings from Table 15. Discharge was considered to take place uniformly over the entire length of the segment, and 10 m out into open water. When the spatial analysis was complete, the sum of the discharge around the Eastern Shore was significantly less than the $460*10^6$ m^3/yr of recharge to the aquifer. This may be the result of incorrect seepage meter readings, or meter readings not characteristic of the entirety of the study area. In addition, the figure of $12*10^6$ m^3/yr may be valid when considering nearshore discharge only (for this analysis discharge was only considered to occur 10 m from shore). Using a buffer of 100 m from shoreline instead of 10 m would bring the total volumetric discharge figure in line with the previous estimates while elevating the nutrient input by a factor of ten. In addition, seepage meters are thought to be 50-60% efficient, thus some of the shortfall could be accounted for by this source of error (Reay, 1993, personal communication). This may be offset, however, by the fact that seepage meters collect the mixture of salt and freshwater that is being discharged. Table 17 shows the estimates for nutrient flux in kg/yr for the quadrants of the Eastern Shore.

Table 16. Discharge by recipient water - seepage meter estimates
(10^6 m^3/yr)

	Chesapeake Bay	Atlantic Ocean	Total
Accomack	2.87	.65	3.53
Northampton	8.52	.39	8.91
Total	11.4	1.04	12.4

Table 17. Nutrient flux - seepage meter estimates
(kg/yr)

	Chesapeake Bay	Atlantic Ocean	Total
Accomack	8090	2200	10300
Northampton	31400	470	31900
Total	39500	2660	42140

The GIS approach for analysis of SGWD

Assembly of the required data sets comprised the bulk of the effort dedicated to this research. As methods of data transfer become more developed, the time spent on this procedure will decrease. The GIS facilitated the spatial analysis of many data in a method that was straightforward and sensible. The attribute file was easily accessible in the common dBase III+ format. This allowed easy access and analysis of attributes using a simple data base manager. It is felt that a GIS system is an ideal tool for the management of this type of data. The problem of data availability was discussed in the limitations section of this model. The GIS system allows analysis of SGWD to occur on a regional scale for the entire Eastern Shore, as well as examinatin on a local scale such as Cherrystone Inlet. The previously mentioned benefits of the GIS approach indicate that it may be a valuable management tool in an attempt to manage regional SGWD by concentrating efforts on the locally high areas of contamination potential through SGWD.

Recommendations for Future Work

As seepage meter readings become more available and standardized, it may be possible to correlate SGWD more thoroughly to the gradient, permeability, and land use data presented in this study. An extensive evaluation of the local groundwater flow regime would be helpful as well. The GIS is an ideal tool to inventory and analyze the type of spatial data required to predict SGWD. As more data become available, the GIS prediction of discharge will increase in accuracy. Additionally, as advances are made in determining effects of other factors, such as those discussed in the limitations section, other data may be added to the existing framework of the model to correct for these factors.

V. SUMMARY AND CONCLUSIONS

The GIS approach to large scale analysis of SGWD was useful for the prediction and analysis of SGWD. Significant limitations exist for this method, however. While pointing out areas at risk for high levels of contamination, an overall exact figure for contamination is fraught with uncertainty, and the figures generated should be recognized as the order-of-magnitude estimates that they are. While research will continue into the phenomenon of SGWD, without further advances in the data collection and theory of the physiography and hydrology of the nearshore environment, the analysis is limited to the scope presented in this study. While much GIS work is done on the level of finding high-risk, optimal or worst case type scenarios, concrete numbers from this type of analysis always have some degree of uncertainty related to the quantity and quality of calibration data available. Additional quantitative data in the form of seepage meter readings, or possibly a derivation of near-shore gradient data would enhance the reliability of the total estimate of seepage. The available data, however, correlated well to model output. Assuming this correlation is not happenstance, the GIS approach will prove an effective tool in management and analysis of SGWD.

Conclusions can be broken down into six categories:

1. Data were not available on the scale necessary to predict local discharge accurately by any method, including numerical methods. The extreme local variability of seepage as measured by seepage meters indicated a vast amount of data for aquifer permeability, hydraulic gradient, nearshore sediment environment, infiltration measurements, nutrient content of the aquifer, and nearshore flow are required. Data collection of this intensity will likely not occur in the foreseeable future, so any modeling technique that gives a concrete value for total or spatial variability is questionable. The local variability of seepage should average out in the larger scale of this model.

2. The southwestern portion of the study area exhibited characteristics of high permeability soils and moderate to steep hydraulic gradient indicating that greater nearshore discharge and seawater cycling are likely to occur in these

areas. In addition, this area did not have the marshy buffer characteristic of much of the Eastern Shore.

3. Nutrient enrichment by SGWD was predicted to be most severe in the southwestern quadrant of the Eastern Shore. The presence of intensive agriculture near the shore and likelihood of high volumetric discharge combine to make this area the most susceptible to contamination. This model predicted more than 90% of nutrient loading from the Eastern Shore to occur into the Chesapeake Bay, instead of the Atlantic Ocean. The bulk of this figure consists of discharge from Northampton County. According to the model, more than 7/10 of the Eastern Shore's nearshore environment is under the influence of agricultural areas, including pastures. At the most, 1/10 the nearshore environment is under the influence of urban land use. About 1/4 of the Eastern Shore's perimeter has no proximal impacting land use, indicating that these areas may not be at risk for damaging SGWD, regardless of the volume of discharge.

4. Discharge proximate to population centers were found by this modeling method. Areas such as these with high likelihood of coliform contamination should not be used as locations for shellfish beds. Individual septic systems were not considered in this study but could also present a threat to the commercial seafood industry.

5. While field data are not extensive enough to allow reliable calibration and verification, there is good correlation between field data and model results on a local scale. Remembering the admonitions of the limitations subsection, the strip island method yields $270*10^6$ m^3/yr of discharge, and the seepage meter estimates $12*10^6$ m^3/yr of nearshore discharge. Calibration with available nutrient flux data shows about 42,000 kg of nitrogen may be delivered to the nearshore environment via SGWD.

6. A GIS approach is a valuable analysis tool for the investigation of SGWD. Manipulation of spatial data was a crucial part of this research, and GIS is an efficient method of handling the type of data used in this research, as well as data for any highly spatial research. Because data can be analyzed on both a

local and regional scale, the GIS methodology developed here could prove an important management tool. Any management strategy developed to limit nutrient input by medium of SGWD should pay special attention to the southwestern portion of the study area, where the greatest impact from SGWD is predicted.

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Vita

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I was born in Binghamton, NY July 19, 1968 to Carol A. and Walter P. Wynn. Eventually moving to Virginia in 1972, I attended West Springfield High School and graduated in 1986. I earned my Eagle Scout, and was involved in playing rock and roll. In 1986 I began attending Virginia Tech. I received my B.S. in Engineering Science and Mechanics as well as a minor in mathematics in 1990. I then began study in environmental engineering, and received my Master's in May, 1993.

My career goal is a consulting position with a small organization where I can work with computers. I also enjoy playing the jazz bass in a local group, and hope to continue after I leave here. In the future, I would like to return to school for a Ph.D. I am an EIT, and I hope to move toward my PE.