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ESTIMATE OF NITRATE FLUX TO REHOBOTH AND INDIAN RIVER BAYS, DELAWARE, THROUGH DIRECT DISCHARGE OF GROUND WATER

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ESTIMATE OF NITRATE FLUX TO REHOBOTH AND INDIAN RIVER BAYS, DELAWARE, THROUGH DIRECT DISCHARGE OF GROUND WATER

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ABSTRACT

Agricultural fertilizer application, animal (poultry) waste, and wastewater disposal practices of the past 40 years have resulted in widespread nitrate contamination of ground water in coastal Sussex County, Delaware. Discharge of contaminated ground water to Rehoboth and Indian River bays is suspected of being a significant contributor to elevated nutrient concentrations in these surface water bodies, resulting in excessive phytoplankton growth and other related problems.

The estimated nitrate loading to Rehoboth and Indian River bays through the direct discharge of nitrate contaminated ground water is derived from a model that incorporates two ground-water discharge estimation methods with geostatistical analysis of nitrate concentrations in 16 contributing drainage sub-basins. Estimated ground-water discharge rates to the bays range from 20.7 to 39.2 million gallons per day (7.84×10^4 to 1.48×10^5 cubic meters per day). The potential nitrate-nitrogen load to the bays is estimated to be in the range of 1303 to 2500 pounds per day (591 to 1134 kilograms per day) on an average annual basis. Nitrate-nitrogen loading rates for the individual contributing sub-basins range from 6.5 to 120 pounds per acre-year (7.3 to 135 kilograms per hectare-year). These estimates do not consider the potential for denitrification that likely occurs within the organic-rich bay-bottom sediments.

There is substantial variation in ground-water flux, potential nitrate-nitrogen flux, and areal nitrate-nitrogen loading rates between the sub-basins that contribute ground water to Rehoboth and Indian River bays. These variations are attributed largely to differences in nitrate concentrations in ground water, although differences in ground-water fluxes account for some of the variation. Differences in nitrate concentrations in ground water are primarily due to land-use characteristics. Differences in estimated ground-water fluxes are due to spatial variation in the natural characteristics of the aquifer and to modeling assumptions.

Nitrate-nitrogen fluxes and loading rates to the bays generally are highest for sub-basins that have historically had intensive poultry production. Within this group, the Indian River north, Indian River south, and Piney Neck sub-basins have the largest nitrate-nitrogen fluxes and loading rates. These sub-basins contribute almost 50 percent of the total nitrogen load through direct ground-water discharge to the Indian River Bay drainage basin, but only comprise about 30 percent of the land area.

Areal nitrogen loading rates from ground water to surface water as calculated in this study are similar to those estimated by previous investigators for the Rehoboth Bay drainage basin. Areal loading rates calculated in this study for the Indian River Bay basin and for sub-basins in which poultry growing is a predominate land use are greater than the maximum rate previously estimated. This indicates that the loading rates developed previously are reasonable for all land-use types and areas except those that have had intensive poultry production in the past.

INTRODUCTION

Purpose and Scope

In the Inland Bays region (fig. 1), eutrophication, caused in part by excess input of nutrients, has been identified as one of the most important water-quality problems in the region (Delaware Inland Bays Estuary Program, 1991). Given the relatively high nitrate concentrations in ground water near the bays and the importance of ground water in the area's hydrologic budget, nitrate input to the bays by direct ground-water discharge is likely to be an important contributor to the eutrophication problem. This report provides estimates of the potential flux of nitrate by direct ground-water discharge into Rehoboth and Indian River bays.

As it is expected that the readership has varying degrees of technical background, this report provides only general discussions of the computational methods used in the study. More detail is provided, however, on model construction and model limitations so that readers familiar with the technical details can make informed decisions regarding use of the results.

The estimation procedure used in this study relies upon existing data and information. The water-budget and Darcian models previously developed by Andres (1987b) are modified and used as the basis for the ground-water discharge estimates. The results are given primarily in tabular form. Maps and other illustrations developed in the course of the research may be examined at the offices of the Delaware Geological Survey (DGS).

Acknowledgments

E. R. Grace (University of Delaware, Department of Geography) provided invaluable help to the project by writing programs to automate many of the data handling and calculation procedures. M. A. Levan, J. P. Zickler, and C. S. Howard assisted in this project. Ronald Graeber of the Delaware Department of Natural Resources and Environmental Control (DNREC) provided ground-water quality data from facilities holding permits for spray irrigation of wastewater. Technical review of the report was provided by E. J. Englund (U. S. Environmental Protection Agency), J. M. Shafer (Illinois State Water Survey), and R. W. Greene (DNREC).

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Previous Work

Inland Bays Region

A large body of literature pertaining to ground-water quality and nutrient loading in the Inland Bays watershed is available. Some of the published ground-water quality studies include Robertson (1977, 1979), Ritter and Chirnside (1982, 1984), Denver (1986), and Andres (1991a, 1991b). In addition, a number of nutrient loading studies of the Inland Bays have been done by Ritter and his colleagues at the University of Delaware, leading up to the landmark report by Ritter (1986). Some recent geologic and hydrologic studies pertinent to the study area include Johnston (1973, 1976, 1977), Chrzastowski (1986), James et al. (1989, 1990), Andres (1986, 1987a, 1987b, 1989, 1991c), Talley (1987, 1988a, 1988b), Talley

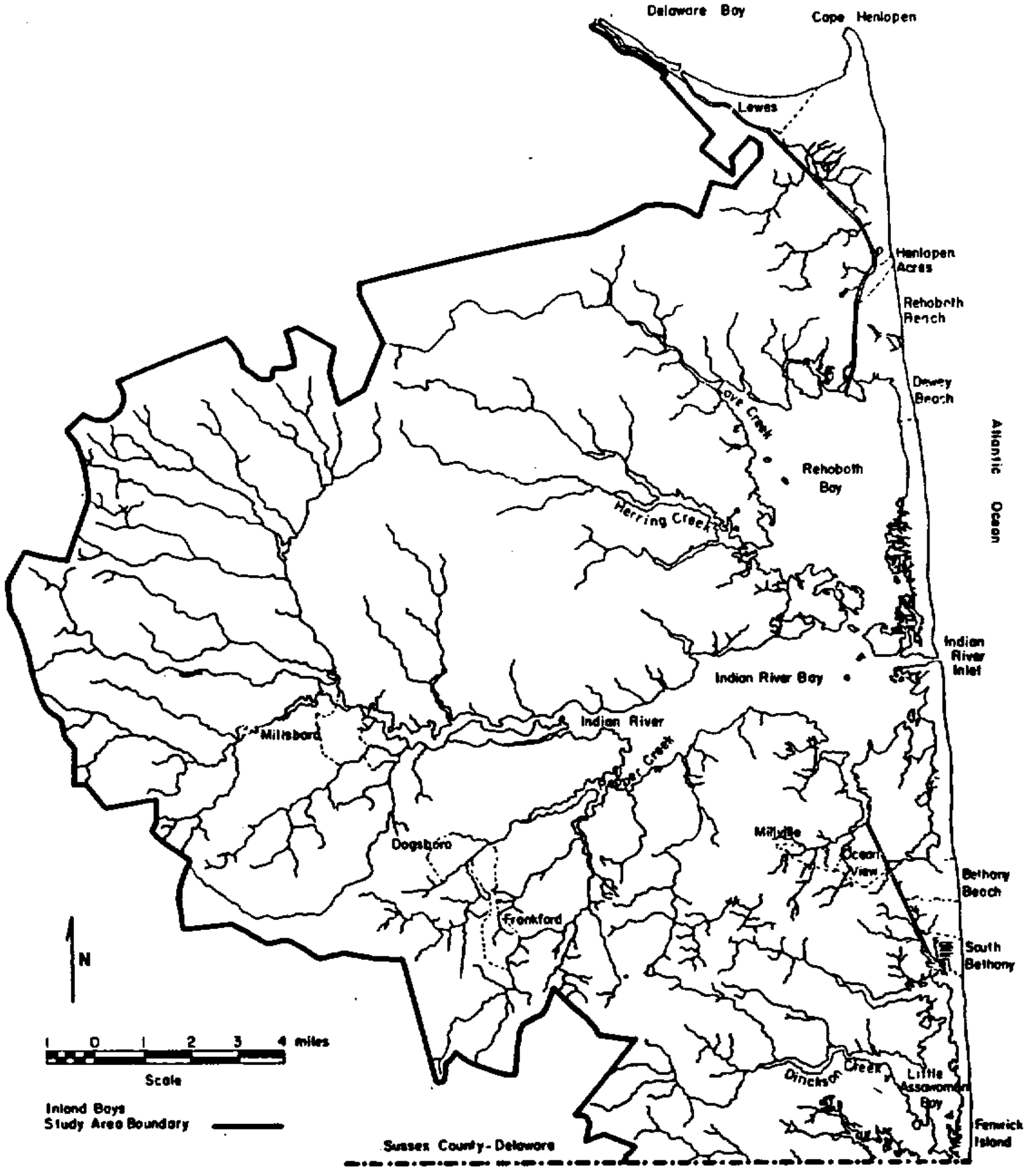


Figure 1. Map of Inland Bays region.

and Andres (1987), and Talley and Simmons (1988).

Almost 1000 wells have been sampled in the Inland Bays watershed (Robertson, 1977, 1979; Ritter and Chirnside, 1982, 1984; Denver, 1989; Andres, 1991b). In addition, Bachman (1984) and Denver (1986) reported the results of several hundred analyses from samples collected in adjacent Maryland and southwestern Delaware where land-use, geologic, and hydrologic conditions are similar. These investigations have documented several important facts.

- Nitrate is the dominant form of nitrogen in shallow (less than 100 ft) ground water throughout most of the Inland Bays watershed, although nitrate-nitrogen concentrations greater than 0.5 mg/L occur in less than five percent of the wells sampled in an area south of Indian River Bay or in deeper confined aquifers. This pattern of occurrence is due to natural geologic and geochemical factors.

- Nitrate in ground water is a problem over much of the Inland Bays watershed because soils are well drained, ground water contains appreciable dissolved oxygen, and the aquifer is unconfined. Nitrate is generally not chemically stable where soils are poorly drained, the ground water is anoxic, and the sediments contain appreciable organic carbon. In addition, the aquifer is generally protected by a confining layer in these areas.

- As a direct result of agricultural and wastewater disposal practices over the past 30 to 40 years, nitrate occurs at significant concentrations (nitrate-nitrogen greater than 5 mg/L) in over 50 percent of the wells sampled. Nitrate concentrations exceeding the U. S. EPA maximum contaminant level (10 mg/L nitrate-nitrogen) occur in over 20 percent of all wells sampled in the area. Nitrate contamination occurs at all depths in the aquifer, in some locations at depths exceeding 90 ft below land surface.

- Nitrate concentrations are usually higher in agricultural and mixed agricultural-residential areas than in forested and mixed forested-residential areas. The highest nitrate concentrations occur in areas with intensive poultry production.

- Ammonia nitrogen concentrations greater than 0.05 mg/L are usually only found in shallow wells located close to poultry farms and agricultural and domestic sewage waste disposal sites in areas of poorly drained soils.

Other Studies

Valiela et al. (1990) reviewed studies of the flux of ground-water borne nutrients into coastal ecosystems. They found that nitrate loading causes increased growth of macroalgae and phytoplankton, reduction of seagrass beds, and reductions of the associated fauna. Weiskel and Howes (1991) quantified the flux of ground water and nutrients into coastal waters using both Darcian and water-budget models coupled with extensive ground-water level and chemistry data. The field data were then used to calibrate nutrient loading models.

Oberdorfer et al. (1990), through the use of water-budget and Darcian ground-water models, found that direct ground-water discharge contributed 20 to 50 percent of the nutrient load to Tomales Bay, California, depending on season. Giblin and Gaines (1990) also used a water-budget model to estimate ground-water flux into a marine cove. Shafer and Varljen (1990) presented the results of a geostatistical study of nitrate in ground water. Their work focused on methods of determining the spatial correlation of nitrate.

Hydrogeologic Framework

The Columbia aquifer is the hydrologic unit of this study. The name has been used in a number of reports to describe the near-surface water-yielding rocks of the Delmarva Peninsula (Bachman, 1984; Bachman and Wilson, 1984; Andres, 1987a; Talley, 1988a; Talley and Andres, 1987). The name was derived from the Columbia Formation and Columbia Group as described in Delaware by Jordan (1962, 1964). In general, the Columbia aquifer is the same as the unconfined aquifer, water-table aquifer, or Pleistocene aquifer as denoted in many earlier publications. In this report, the use of "the aquifer" will refer to the Columbia aquifer unless noted otherwise. Several lithostratigraphic units form the Columbia aquifer in the study area. Their hydrogeologic characteristics are summarized in Table 1.

Over much of Delaware, the Columbia aquifer is a complex hydrologic unit that is generally unconfined, although it may be locally confined or vertically stratified into unconfined and confined sections, especially in coastal Delaware (Andres, 1991a). The known thickness of the aquifer is highly variable in the study area, ranging from a minimum of about 75 ft to a maximum of over 200 ft (Andres, 1987a; Talley, 1988a). An important characteristic of the aquifer is the spatial heterogeneity of thickness, permeability, and lithology. These properties greatly influence patterns of ground-water flow and chemistry.

The Columbia aquifer discharges both uncontaminated and nitrate-contaminated fresh ground water to Rehoboth and Indian River bays and to the streams draining into the bays. Johnston (1973, 1976, 1977) found that about 80 percent of streamflow is derived from ground-water discharge. Similarly, Ritter (1986) estimated that 80 percent of the nitrogen entering the bays during a normal year is derived from ground-water discharge.

The Pocomoke aquifer, which underlies the Columbia, may also contribute fresh water that flows upward through the Columbia aquifer to the bays, but convincing evidence for this has not yet been found. Further, the Pocomoke aquifer would probably not contribute much nitrate as the results of numerous chemical analyses have shown that nitrate concentrations in the parts of the Pocomoke containing water with less than 250 mg/L chloride are usually less than 1 mg/L nitrate-nitrogen (Talley and Andres, 1987).

METHODS

Geostatistics

Geostatistics, a branch of applied statistics, is commonly used in natural resources evaluations to describe and estimate the spatial distribution of phenomena of interest. Examples are ground- or water-table surface elevation, ore grade, or pollutant distribution. Geostatistics is also employed to assess probabilities and risks for environmental hazards.

Semi-variograms and kriging are the primary geostatistical methods used in this

Table 1. Geologic units comprising the Columbia aquifer and their hydrologic functions. From Andres (1991a).

LITHO-STRATIGRAPHIC UNIT	LITHOLOGY	HYDROLOGIC FUNCTION
Unnamed Holocene deposits	Variable and complex assortment of sand, silt, clay, organic material, and gravel.	Minor component of Columbia aquifer. Controls locations of recharge and discharge. If saturated, capable of yielding minor quantities of water to wells.
Delaware Bay deposits	Sand, medium to coarse, with scattered gravel, compact silty clay, and organic-rich silty clay.	Probably a minor component of Columbia aquifer. Hydrologic function dependent on lithology. May yield small quantities of water to wells.
Omar Formation	Silt, clay, and fine sand, with varying amounts of shell and organic material, and lesser amounts of medium to coarse sand and gravel. Locally, may be fine to coarse sand. Fine-grained beds more common where unit is greater than 30 ft. thick.	Leaky confining layer or confining layer. Has strong influence on chemical composition of ground water and on rates and directions of ground-water flow. At best can yield small quantities of water to wells.
Beaverdam Formation	Sand, medium to coarse, with varying amounts of gravel, fine sand, silt, and clay found in relatively discontinuous lenses and layers. Generally becomes coarser with depth. Fine-grained beds more common in upper one-third of unit.	Major component of Columbia aquifer. Source of stream baseflow and recharge to deeper aquifers. Yields moderate to large quantities of water to wells. Lower half of aquifer usually more permeable than upper half.
Bethany Formation	Silt, clay, and sand in varying proportions with minor amounts of gravel. Silt and clay beds tend to form a relatively continuous layer.	Fine-grained beds form base of the Columbia aquifer and function as a leaky confining layer. Thicker sand layers form the Pocomoke aquifer. Functions as part of the Columbia aquifer where sands are in hydraulic connection with sands of overlying units.

report. A semi-variogram (hereafter, variogram) is a method used to quantify the commonly observed relationship that samples close together will tend to have more similar values than samples far apart (Englund and Sparks, 1988). The variogram thus describes the spatial correlation structure of the parameter of interest. Kriging is a generic name for a group of minimum-error-variance estimation techniques that cover a variety of methods for weighted-moving-average interpolation (Journel, 1987; Englund and Sparks, 1988). The spatial correlation structure described by the variogram is used in kriging as the weighting function. Named after D. G. Krige, kriging was developed in the 1950s and 1960s in the field of mineral resource evaluation (David, 1977). It has been increasingly used for environmental studies over the past 10 years (Journel, 1987). Both variograms and kriging are described in more detail in following sections.

Variogram Modeling

A variogram is a plot of one-half the mean squared difference of paired sample measurements (commonly referred to as gamma squared) as a function of the distance between samples (Clark, 1979). In practice, theoretical variograms, which are models of the spatial dependence between samples, are fit to sample variograms (plots of sample pair differences). The theoretical variograms are then used to estimate nitrate concentrations (see Kriging).

The computer program VARIO from GEOEAS (Englund and Sparks, 1988) was used to construct two-dimensional sample and theoretical variograms. Lag distances were adjusted to obtain the smoothest experimental variogram with the restriction that no fewer than 30 sample pairs be used for each lag spacing. Variogram models were determined by visually examining the fit between the model and data. Each data set was also tested for anisotropy. The program GEOPACK (Yates and Yates, 1990) was used to check the accuracy of the GEOEAS calculations.

Kriging

Kriging is "... a weighted-moving-average interpolation method where the set of weights assigned to samples minimizes the estimation variance, which is computed as a function of the variogram model and locations of the samples relative to each other, and to the point or block being estimated" (Englund and Sparks, 1988, p. xiv). In this study, two-dimensional ordinary block kriging is used to calculate estimates of the concentration of nitrate-nitrogen and the associated estimation variance. Discussion of the mathematical basis of kriging is beyond the scope of this report. Interested readers are referred to David (1977), Journel and Huigbregts (1978), Davis (1986), and Clark (1979) for detailed developments of the subject.

The kriging procedure is used to interpolate the value of nitrate on a grid basis. Because block-centered kriging was used, the block size was set to one-quarter the mean minimum distance between sampling points as suggested by David (1979). The minimum and maximum coordinates of each sub-basin and shoreline were determined to define the kriging grid area.

The variogram models used in kriging were tested with the XVALID routine of GEOEAS. This program sequentially removes each data point from the data set and estimates a value at the location of the deleted data point from the remaining data points. The characteristics of the estimated data and the differences between the estimated and actual values are then statistically analyzed. Better fitting models more closely reproduce the original data set and have smaller differences between estimated and actual values than poorer fitting models.

Data Sources and Preparation

Ground-water quality and some hydrogeologic data used in this study are taken from Andres (1991b) and other unpublished sources (DNREC permit compliance monitoring reports and well completion reports; DGS ground-water information system). All data are available at the DGS offices.

Water quality data were obtained from 479 wells (fig. 2). Well coordinates were determined by first plotting locations on U. S. Geological Survey 7.5-minute maps and manually measuring latitude and longitude. The geodetic positions were then converted into Delaware State Plane Coordinates using the computer program GPPCGP (National Geodetic Survey, 1987). Well screen depths are accurately known for over 90 percent of the wells used as sampling points. All samples were taken from wells that obtain water from the Columbia aquifer.

Data preparation is a major part of all geostatistical studies. Careful screening of the data set is necessary to ensure that the data conform to the assumptions inherent in the modeling techniques. Examples of problems addressed by screening are identification of sample groupings or populations, removal of duplicate samples, and treatment of extreme data values and spatial trends.

Temporal Averaging. Temporal averaging is necessary because the samples were collected between winter 1988 and spring 1990. During this time period most wells were sampled only once, but many monitoring and public supply wells were sampled multiple times. The average of multiple samples from a single well were used in the analysis. On initial assessment, the grouping of data collected over a three year time period seems less than ideal because there is seasonal and yearly variation in nitrate concentration at any point in the aquifer due to variations in nitrogen input and ground-water flow. However, additional consideration tends to support using all of the data.

Temporal variation in nitrate concentration tends to be largest in the shallowest parts of the aquifer because nitrate input varies with seasonal or yearly fluctuations in agricultural land use or wastewater disposal practices. Almost all wells screened less than 35 ft below land surface used in the data set are monitoring wells that were sampled multiple times over several seasons and years. These sample results were averaged for each well to better represent long-term average nitrate input. Further, sample results from shallow monitoring wells are averaged with sample results from deeper monitoring wells at the same location (see Spatial Averaging, p. 10). Analysis of nitrate concentrations in deeper wells, screened more than 35 ft below land surface, shows that short-term (seasonal and less than two years) variations in nitrate concentration are usually less than the spatial variation between sampling points. This is most likely a result of the interaction between mixing in the aquifer and the long-term (greater than five years) consistency in nitrogen input. Therefore, the data set provides a reasonable representation of nitrate distribution in the aquifer

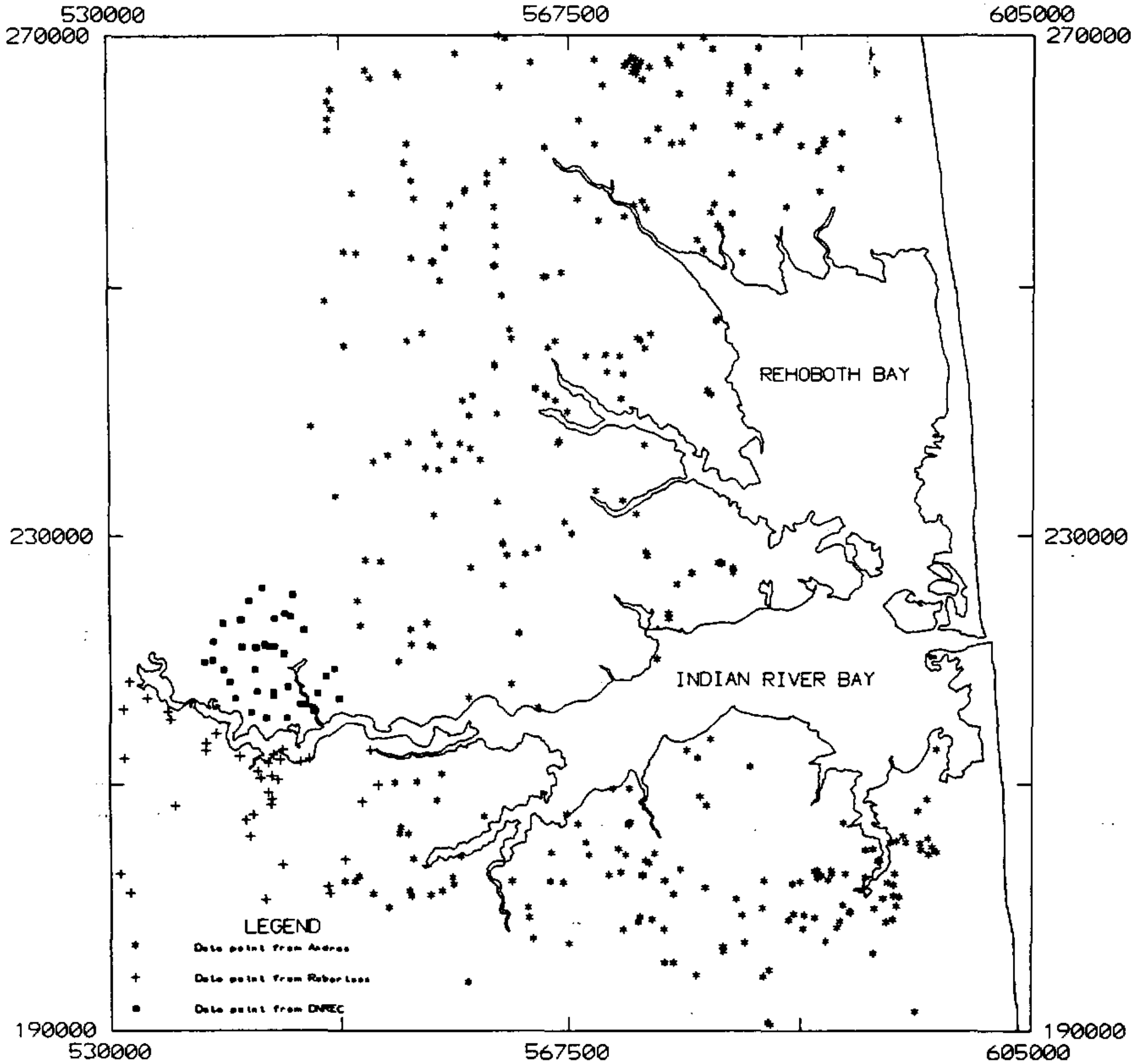


Figure 2. Map showing locations of sample points. Coordinates are Delaware State Plane Coordinates (1927) in feet.

because the analysis is centered on 1989, results from shallow wells are time averaged, most samples were collected from deeper wells where temporal variations are small, and the bulk of the aquifer is more than 35 ft below land surface.

Spatial Averaging. At a number of locations, two or three monitor wells were installed at different depths at the same location (well clusters). Because the variogram and kriging procedures are two-dimensional, results from the wells comprising the well clusters are averaged. Usually, only the results from the deeper two wells are included in the average because they sample the bulk of the aquifer, while the shallowest well usually samples the overlying leaky confining layer. In cases where the shallowest well is screened in the aquifer, results from all wells in a well cluster are included in the average.

Results from sampling well clusters have shown that nitrate concentration can vary with depth below a point on land surface (Denver, 1989; Andres, 1991a). This phenomenon presents some difficulty for two-dimensional variogram modeling because many of the samples used in this study were collected from domestic wells that withdraw water from a 5- to 10-foot-long well screen, whereas the aquifer saturated thickness is usually 60 to 90 ft.

On first inspection it may appear that the only way to treat this problem is to obtain a representative sample of bulk aquifer nitrate concentration by using only data from well clusters, or by doing three-dimensional modeling. These data are not available; therefore, some means of assessing the impact of this lack of data is needed. First, calculations may be biased by using samples that were collected from a narrow portion of the aquifer where nitrate concentrations are substantially lower or higher than the true bulk aquifer concentration. This does not appear to be the case as significant numbers of samples were collected from all depth ranges (Andres, 1991a). Second, there is only a weak statistically determined trend between decreasing nitrate concentrations and increasing well depths (Andres, 1991a; Bachman, 1984). It appears that the problem of not using true bulk aquifer samples in the analysis should not invalidate two dimensional modeling but will cause some difficulty in constructing models.

Population Groupings. Because nitrate occurrence and distribution are controlled by land use and hydrogeologic factors, subdivision of the data set is based on major differences in these factors. Land use information for this study was derived principally from Ritter and Chirnside (1982). This work was supplemented by the author's field observations, interviews with residents, and analysis of maps and aerial photographs in order to estimate the levels of poultry production mentioned herein.

As areas having intensive poultry production have higher nitrate concentrations than other areas (Robertson, 1977, 1979; Ritter and Chirnside, 1982, 1984; Andres, 1991a), the data set is divided into poultry farming and non-poultry farming area groups. The poultry farming area includes all of the sub-basins in the Indian River Bay basin except for the Long Neck south and east sub-basins (see fig. 3). A number of wells surrounding the poultry farms in the Fairmount area are also included in the group. The non-poultry farming area includes all of the Rehoboth Bay sub-basins plus the Long Neck south and east sub-basins (see fig. 4). The area covered by the non-poultry farming data set does include a few scattered poultry farms and sites of former poultry farms.

Because a major hydrologic boundary, the Indian River, bisects the poultry farming area, the poultry farming area data group is further divided into two sub-groups, north of Indian River and south of Indian River. The north of Indian River sub-group includes data

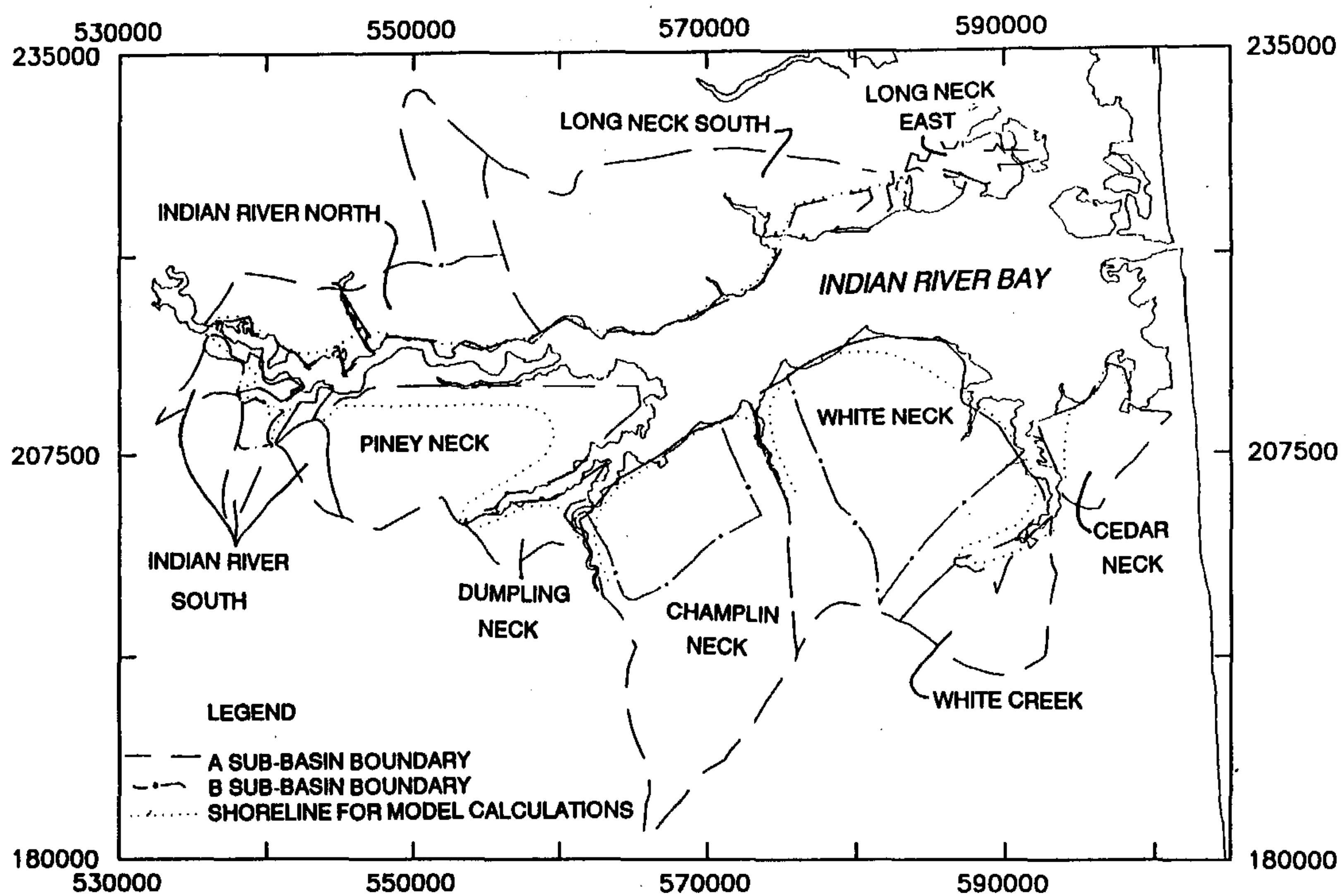


Figure 3. Map of Indian River Bay drainage basin. Coordinates are Delaware State Plane Coordinates (1927) in feet.

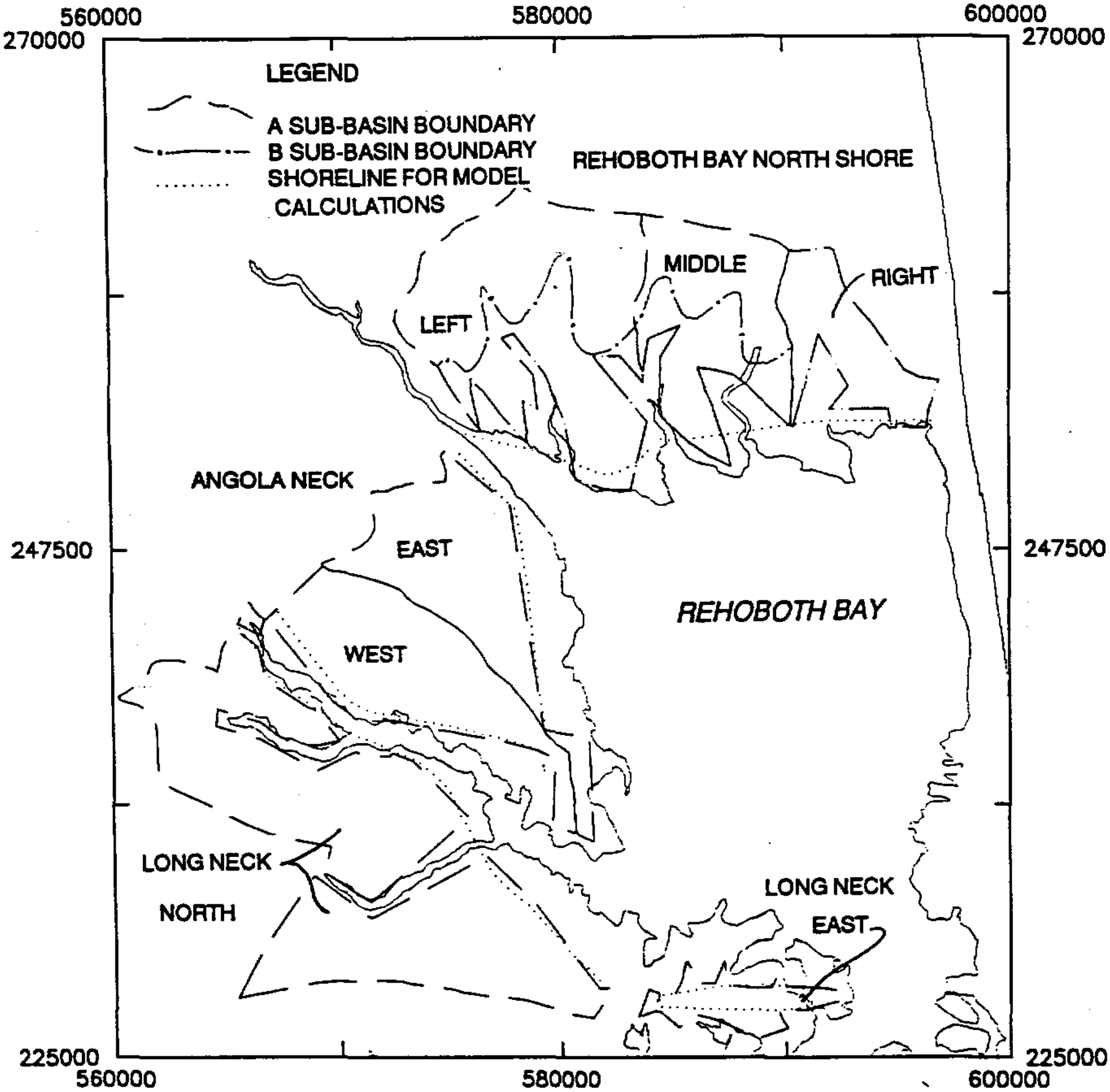


Figure 4. Map of Rehoboth Bay drainage basin. Coordinates are Delaware State Plane Coordinates (1927) in feet.

points from the Indian River north sub-basin and the Fairmount area. The Indian River north sub-basin data point coordinates are shifted 10,000 ft north to keep these points from being compared to the data points located south of Indian River. The coordinates of the points in the Fairmount area were transposed to correct for the predominately west to east ground-water flow direction in the Fairmount area as opposed to north to south or south to north ground-water flow directions in the other sub-basins. Following these manipulations the sub-groups were recombined for variogram modeling.

For kriging, data sets were created for each sub-basin (except the Indian River south sub-basin) containing all points within 10,000 ft of the basin boundaries. Because there are no data points in the Indian River South basin, the Robertson (1977) data (37 samples) were used for this sub-basin.

Outlier screening. Outliers, which can be extremely high or low values, present difficulties in variogram modeling because they can greatly skew variogram results. The means to solve the problems caused by outliers is the subject of much debate and research (Armstrong, 1984; Omre, 1984; Shafer and Varljen, 1990). In this study very simple methods of detecting and treating outliers are employed. Outliers were detected by examining the results of histogram, variogram, and pair-comparison computations. Simple methods of treating outliers are to remove them by limiting the maximum value to be used in variogram calculation or by removing the data points that cause the largest differences in the variogram calculations. The first screening procedure is used only on the poultry farming area data set, and in that case over 90 percent of the data set is retained for the variogram calculations. Unfortunately, removing data points by this procedure also changes the statistical characteristics of the resultant data set and causes some spatial variability information to be lost. For these reasons a variogram model was also made for the poultry farming area data set by removing only a few of the data points that caused the largest differences in the variogram calculations.

Ground-Water Flux Estimation

The other main research tool of this study is a ground-water flux model. One of the two models used for ground-water flux calculation (fig. 5) is the same as in Andres (1987b), that is, a relatively simple ground-water flow-net model based on Darcy's Law:

$$Q=KiA$$

where:

Q = flux, length^3 per time l^3/t
K = hydraulic conductivity, l/t
i = gradient, dimensionless
A = area, l^2, product of shorelength and aquifer thickness.

This simple Darcian model was chosen over a numerical model because of the time and budgetary constraints of the project. The Darcian model will also be referred to as a flow-net model. The flux calculations are actually dependent on a number of complex subsidiary models that represent the hydrogeologic conditions of the lands draining into the bays. These subsidiary hydrogeologic models are based upon an extensive data set consisting of borehole, aquifer test, ground-water level, stream discharge, and geochemical data.

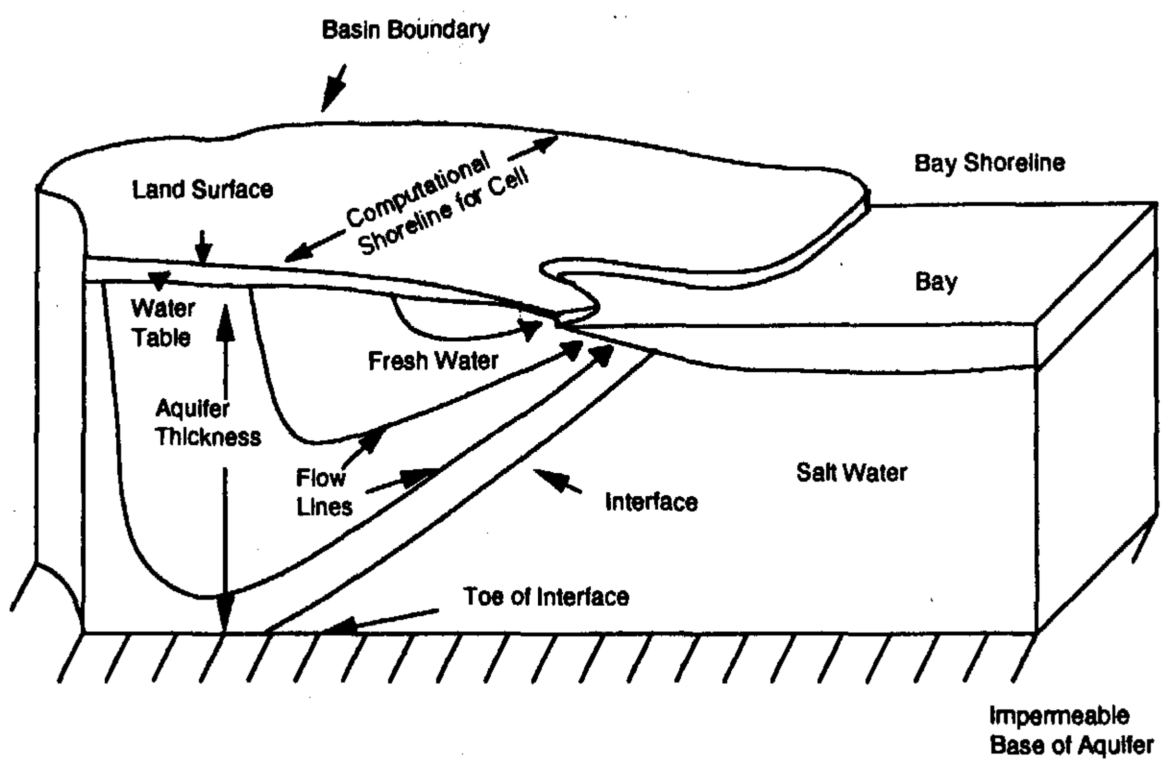


Figure 5. Illustration of ground-water flux model. Modified from Andres (1987b).

The Rehoboth Bay (fig. 3) and Indian River Bay (fig. 4) shorelines and sub-basin boundaries were discretized following Andres (1987b). This discretization was based on analysis of water-table elevation contour maps prepared by Boggess et al. (1964), Boggess and Adams (1964), Adams, Boggess, and Davis (1964), and Adams, Boggess, and Coskery (1964). The state-plane coordinates of shorelines and sub-basin boundaries were then digitized and stored on disk for later use. A shoreline was also defined and digitized for the Long Neck east sub-basin. This shoreline is used to calculate a unit ground-water flux and nitrate discharge.

Average ground-water flux estimates from Andres (1987b) were then reevaluated based on review and analysis of new aquifer test, ground-water level, and lithologic log data (Andres 1991b, 1991c; unpublished data in DGS files). This evaluation included construction of additional cross sections for evaluation of bulk aquifer transmissivity and analysis of new piezometric surface maps for determination of average and seasonal hydraulic gradients. Bulk aquifer transmissivity is the product of hydraulic conductivity and thickness and was determined as follows.

- Cross sections that approximately followed sub-basin shorelines were constructed.
- On each cross section geologic materials were classified as sand, silty sand, mixtures of sand and gravel, and mud using a procedure similar to that of Andres (1991c).
- Mean thicknesses of these materials were computed for the cross sections.
- The material thicknesses were multiplied by the corresponding hydraulic conductivities shown in Table 2.

Table 2. Hydraulic conductivities used in ground-water flux calculations. Materials categories and hydraulic conductivity values are modified from procedures and unpublished data summarized in Andres (1991c).

Material	Hydraulic Conductivity (ft/d)
Mud	0
Silty sand	16
Sand	80
Sand and gravel mixtures	290

The other model used to estimate ground-water flux is a water-budget model. This model uses the sub-basin areas and the range of average annual areal ground-water recharge rates determined by Johnston (1976) to calculate ground-water flux. Fluxes determined by this model represent long-term averages.

Estimates of average ground-water flux and seasonal flux variations are based on evaluation of variations in ground-water levels and gradients in the Rehoboth Bay North Shore left (data from Andres, 1991b) and Indian River north (unpublished DNREC permit compliance monitoring reports) sub-basins. Two general ground-water conditions were evaluated, a recharge period and a discharge period. An average of two recharge and two discharge periods was also evaluated. Gradients were calculated by dividing the elevation difference between hand-drawn piezometric-surface contour lines by the distances between the contour lines.

Nitrate Flux Estimation

Nitrate fluxes to the bays for each sub-basin were calculated by a computer program (Grace, unpublished). The methodology is illustrated in fig. 6. The program multiplies the estimated nitrate concentration determined by kriging in each grid cell closest to the shoreline by the ground-water flux for the length of shoreline in that grid cell and sums those results for a sub-basin. The results were converted into areal loading rates, shoreline loading rates, and areal relative loadings using a spreadsheet program. Program accuracy was checked by comparing results computed manually with those produced by the program. Seasonal variations in nitrate flux are inferred from observed seasonal variations in ground-water flux.

A qualitative measure of the error in the computed nitrate flux due to estimation of nitrate concentrations is derived from the distribution of block kriging standard deviations (KSD). KSD is computed by GEOEAS along with the block-centered nitrate concentrations. KSD is the square root of the weighted sum of the semi-variances for the distances of the data points to the location of the estimate (Davis, 1986). KSD is not a measure of the absolute accuracy of the estimate, but can be used as a measure of the relative reliability of the estimate. An index of the amount of error is found by computing the ratio of the mean estimated nitrate concentration and the median KSD.

RESULTS

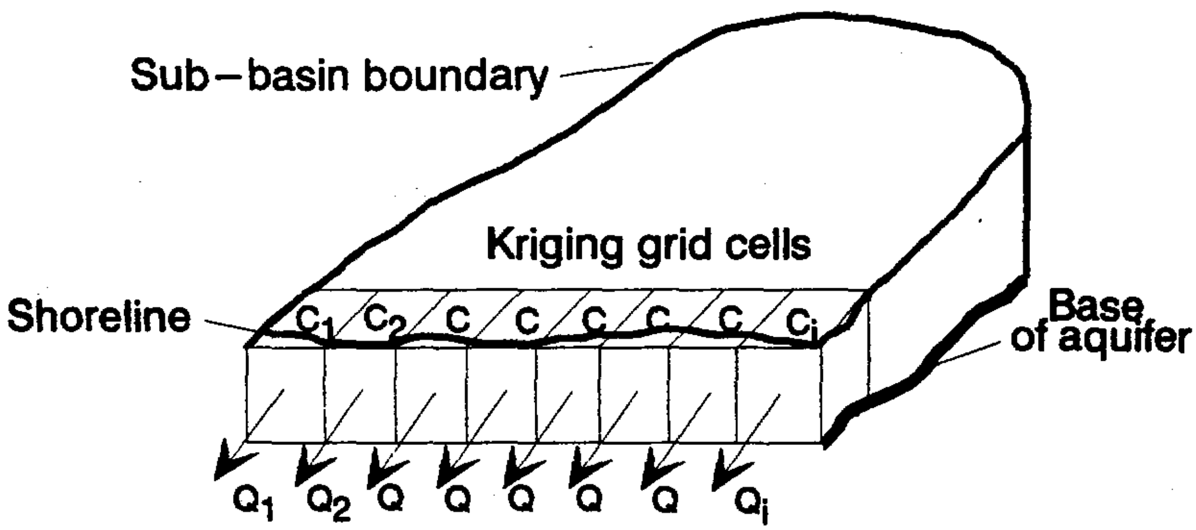
Ground-Water Flux

Geologic and geochemical data consisting of lithologic logs and ground-water level, aquifer test, and ground-water quality data acquired since the Andres (1987b) study, were used to reevaluate the previous hydrogeologic models and recalculate ground-water fluxes. Table 3 summarizes the results of the ground-water flux reevaluation and also the results of the Andres (1987b) calculations. The total direct ground-water flux computed in this study ranges from 20.6 to 39.2 million gallons per day (mgd) (7.83×10^4 to 1.48×10^5 cubic meters per day [m^3/d]). The range for Rehoboth Bay is 6.10 to 9.01 mgd (2.31×10^4 to 3.41×10^4 m^3/d), and for Indian River Bay it is 14.5 to 31.0 mgd (5.49×10^4 to 1.17×10^5 m^3/d). The lesser fluxes are calculated for the smaller (B) size sub-basins using the 500,000 gallons per day per square mile recharge rate water-budget model. The greater fluxes are calculated using both the Darcian and 600,000 gallons per day per square mile recharge rate water-budget models with the larger size (A) shorelines and sub-basins. The total fluxes are similar to those presented in Andres (1987b). There are, however, significant differences in the sub-basin flux rates between the two studies (see Table 3).

As discussed by Andres (1987b), fluxes from A sub-basins and shorelines represent the fluxes expected if the small streams draining some of the sub-basins do not intercept significant amounts of ground water. Conversely, fluxes from B sub-basins and shorelines

C_i – Estimated NO_3–N concentration in cell i

Q_i – Ground–water flux through cell i.



Total NO_3–N flux = $Q_1C_1 + Q_2\,C_2 + \ldots + Q_i\;C_i$

Figure 6. Illustration of nitrate flux calculation method.

represent the fluxes expected if the small streams do intercept significant amounts of ground water. To determine which sub-basin size better represents field conditions a limited number of low-flow stream discharge measurements reported by Talley and Simmons (1988) were analyzed. The period of record used in this analysis is not quite long enough to make any significant conclusions but the data allow some inferences. To summarize:

- The mean area-normalized low-flow discharge rate observed on the Indian River are in the range of Johnston's (1976) long-term ground-water recharge rates.

- The mean area-normalized discharge rates for two streams draining the White Neck and Champlin Neck sub-basins are less than one-half the magnitude of those observed on the Indian River at Millsboro.

This indicates that some, but not all, of the ground-water recharge in the White Neck and Champlin Neck A sub-basins discharges to Indian River Bay. It is also possible that some ground-water recharge enters the underlying Pocomoke aquifer.

Fluxes calculated with the Darcian model indicate that ground-water flux is not equally distributed through the study area. Area normalized fluxes indicate that the Piney Neck, Dumpling Neck, Cedar Neck, Rehoboth Bay North Shore middle, and Angola Neck east sub-basins contribute substantially more ground water in proportion to their drainage basin areas than the other sub-basins. The unequal distribution of ground-water fluxes and area-normalized fluxes are partially attributable to differences in aquifer thickness, permeability, and hydraulic gradient between sub-basins. The larger area-normalized fluxes observed in some sub-basins are also partially due to the larger ratios of shorelength to basin area observed in those basins.

Gradient Analysis and Seasonal Variation in Ground-Water Flux

Ground-water flux varies seasonally and directly with ground-water recharge. On an average annual basis, most ground-water recharge (and the greatest ground-water discharge rates) occurs in the winter and spring and is minimal during summer and autumn (Talley, 1988b). Under these conditions, maximum ground-water flux occurs in winter and spring and is minimal during summer and autumn.

Gradient analysis is the basis for evaluating seasonal flux variations because gradient is the only term with significant time variation in the Darcian model. Evaluation of the gradient data takes into account a drier than normal period (1988 through February 1989) during the usual recharge season (Andres, 1991a). Rehoboth Bay North Shore sub-basin gradients determined from 1988 to 1990 monitoring average 1.2×10^{-3} and range from 8.8×10^{-4} to 1.6×10^{-3}. Indian River north sub-basin gradients range from 1.6×10^{-3} to 2.0×10^{-3} and average 1.8×10^{-3}.

In the Rehoboth Bay North Shore area, average gradient is about one-third the value of the gradient estimated from data of Boggess et al. (1964). It appears that the average depth to water versus land-surface elevation method used by Boggess et al. (1964) to derive water-table contours overestimated the water-table elevation in the Rehoboth Bay North

Table 3. Ground-water fluxes. See figs. 3 and 4 for sub-basin locations.

	DARCY'S LAW MODEL							WATER BUDGET MODEL 1 (RECHARGE= 500 GPD x MI²)			WATER BUDGET MODEL 2 (RECHARGE =600K GPDxMI²)	
BASIN OR SUB-BASIN NAME	SHORE-LENGTH (FT)	1987 UNIT G-W FLUX (FT²/D)	1987 G-W FLUX (MGD)	1991 UNIT G-W FLUX (FT²/D)	1991 G-W FLUX (MGD)	SHORE-LENGTH NORMALIZED G-W FLUX (FT²/D)	AREA NORMALIZED G-W FLUX (FT/D)	BASIN AREA (MI²)	UNIT G-W FLUX (FT²/D)	G-W FLUX (MGD)	UNIT G-W FLUX (FT²/D)	G-W FLUX (MGD)
REHOBOTH BAY												
REHOBOTH BAY NORTH SHORE												
BASIN "A"												
LEFT	9093	20.00	1.36	19.68	1.34	1.41	0.74	3.31	24.33	1.66	29.20	1.99
MIDDLE	6032	27.90	1.26	25.40	1.15	1.83	1.19	1.76	19.50	0.88	23.40	1.06
RIGHT	5976	29.00	1.30	5.53	0.25	0.40	0.46	0.94	10.51	0.47	12.62	0.56
TOTAL	21101	24.81	3.92	17.31	2.73	1.24	0.83	6.01	19.04	3.01	22.85	3.61
BASIN "B"												
LEFT	9093	20.00	1.36	19.68	1.34	1.41	1.43	1.39	10.22	0.70	12.26	0.83
MIDDLE	6032	27.90	1.26	25.40	1.15	1.83	1.95	0.67	9.64	0.44	11.57	0.52
RIGHT	5976	29.00	1.30	5.53	0.25	0.40	0.39	0.94	10.51	0.47	12.62	0.56
TOTAL	21101	24.81	3.92	17.31	2.73	1.24	1.26	3.20	10.14	1.60	12.16	1.92
BASIN "A" = BASIN "B"												
ANGOLA NECK												
WEST	13524	10.88	1.10	10.88	1.10	0.78	1.08	1.85	9.14	0.92	10.97	1.11
EAST	13351	11.10	1.11	13.60	1.36	0.98	1.22	2.02	10.11	1.01	12.14	1.21
LONG NECK												
NORTH	23763	15.40	2.74	16.52	2.94	1.19	1.09	4.90	13.77	2.45	16.53	2.94
EAST	7449	2.06	0.12	2.06	0.12	0.15	0.91	0.23	2.06	0.12	2.46	0.14
TOTAL A [1]	79206	15.16	8.98	13.92	8.24	0.96	0.96	15.01	12.67	7.51	15.20	9.01
TOTAL B [2]	79206	15.16	8.98	13.92	8.24	0.96	1.15	12.20	10.30	6.10	12.35	7.32
INDIAN RIVER BAY												
BASIN "A"												
INDIAN RIVER												
NORTH	23902	18.80	3.36	16.71	2.99	0.80	0.78	5.50	15.38	2.75	18.46	3.3
SOUTH	13840	15.40	1.59	15.40	1.59	0.73	1.18	1.94	9.37	0.97	11.24	1.164
CEDAR NECK	15482	20.20	2.34	20.20	2.34	0.96	2.14	1.67	6.78	0.79	8.13	0.942
WHITE CREEK	6962	23.40	1.22	23.40	1.22	1.11	0.83	2.12	20.30	1.06	24.36	1.272
WHITE NECK	37106	16.50	4.58	13.31	3.69	0.63	0.57	9.26	16.68	4.63	20.02	5.556
CHAMPLIN NECK	23936	22.30	3.99	15.26	2.74	0.73	0.44	8.96	25.06	4.49	30.09	5.388
BASIN "B"												
INDIAN RIVER												
NORTH	23902	18.80	3.36	16.71	2.99	0.75	0.93	3.61	10.10	1.80	12.11	2.166
SOUTH	13840	15.40	1.59	15.40	1.59	0.69	2.60	0.69	3.33	0.35	4.00	0.414
CEDAR NECK	12653	20.20	1.91	20.20	1.91	0.90	1.37	1.57	8.29	0.79	9.95	0.942
WHITE NECK	18234	16.50	2.25	13.31	1.81	0.60	0.37	6.46	20.02	2.73	24.02	3.276
CHAMPLIN NECK	11140	22.30	1.86	15.28	1.27	0.68	0.52	2.75	16.50	1.38	19.80	1.65
BASIN "A" = BASIN "B"	/											
LONG NECK												
SOUTH	28677	26.00	5.58	27.57	5.91	1.31	1.05	8.13	18.95	4.07	22.74	4.876
EAST	7449	2.06	0.12	2.06	0.12	0.10	0.72	0.23	2.06	0.12	2.48	0.138
PINEY NECK	26816	40.50	8.12	38.06	7.73	1.83	2.00	5.54	13.81	2.77	16.57	3.324
DUMPLING NECK	12874	27.50	2.65	27.50	2.65	1.31	3.07	1.24	6.44	0.62	7.73	0.744
TOTAL A [1]	197068	22.76	33.55	21.02	30.98	0.95	1.28	44.51	15.10	22.25	18.12	26.70
TOTAL B [2]	155567	23.56	27.44	22.33	25.99	0.91	1.40	29.22	12.55	14.61	15.06	17.53
REHOBOTH AND INDIAN RIVER BAYS TOTALS												
TOTAL A [1]	276276	20.58	42.53	18.98	39.22			59.52	15.65	29.76	18.78	35.71
TOTAL B [2]	234795	20.74	36.42	19.49	34.23			41.42	11.79	20.71	14.15	24.85

Notes:
[1] Total "A" computed from "A" shorelines plus "A = B". Within basin normalized fluxes are averages for that basin.
[2] Total "B" computed from "B" shorelines plus "A = B". Within basin normalized fluxes are averages for that basin.
[3] Units: FT=feet, D=day, MGD=million gallons per day, MI=mile, G-W=ground-water.

Shore sub-basins. Some of the difference may be related to errors in estimation of land-surface elevation by Boggess et al. (1964).

The ratio of maximum to minimum gradients indicates that the maximum flux rates are about 1.8 and 1.3 times greater than the minimum rate in the Rehoboth Bay North Shore and Indian River north sub-basins, respectively. The maximum rates are about 1.3 and 1.1 times greater than the average flux rates in the Rehoboth and Indian River north sub-basins, respectively. The effect of the wastewater spray irrigation that is occurring in the Indian River north area on seasonal gradient variations is uncertain. It is possible that the seasonal gradient variations observed in the Rehoboth area are more representative of typical hydrologic conditions than the seasonal gradient variations observed in the Indian River north area. If the maximum to minimum gradient ratio of 1.8 is typical, then under normal hydrologic conditions, about two-thirds of ground-water flux occurs during winter and spring.

Nitrate Flux Estimation

How to Use the Results

The results can be used to evaluate how much nitrate-nitrogen each sub-basin potentially discharges to Rehoboth and Indian River bays. However, the results should be evaluated in the context of the assumptions used to formulate the models as well as the following factors:

- Ground-water discharge rate and nitrate concentration estimation methods tend to smooth local scale (sub-sub-basin scale) variations in discharge rate and nitrate flux.

- The models do not account for sub-bay-bottom hydrogeologic and geochemical processes that could affect the locations, rates, and chemical composition of ground-water discharge.

- There is an estimation error associated with calculated nitrate concentrations and nitrate fluxes.

As a result of the above factors, it would not be proper to compare the results of this study with the results small-scale field studies that make a few measurements of bay-bottom ground-water discharge rates and nitrate concentrations.

As stated in the methods section, a qualitative nitrate flux error is derived from the ratio of the mean estimated nitrate concentration and the block kriging standard deviations (KSD) for each sub-basin. For a given kriging run, larger KSD values occur where there are fewest data points near the estimation location and where there is the greatest variation among data values. The magnitude of the KSD is smallest where there are the most data points and where there is little variation in the data values. The best nitrate flux estimates are for those sub-basins where the ratios of the estimated nitrate concentration to the median KSD are largest.

Statistical Summary

Figure 7 shows simple summary statistical plots for the entire data set and for the poultry farming and non-poultry farming data subsets. The plots illustrate the larger mean, median, and variance of the poultry farming area data set. Chi-squared tests indicate that neither the entire data set nor the subsets are normally or log-normally distributed, even after the sample points with non-detectable nitrate concentrations are removed. Additional statistical analyses of the data are presented by Andres (1991a).

Variogram and Kriging Models

Several variogram and kriging search models were formulated and tested using the VARIO and XVALID programs of GEOEAS (Englund and Sparks, 1988). The sample variograms (figs. 8 and 9) show a scatter of values (gamma squared on figs. 8 and 9) at distances shorter than the range (distance at which samples are uncorrelated) but, at the same distances also exhibit overall increases in values with increasing distance. Both poultry farming area theoretical variograms have large nuggets (the variogram value at zero distance) and sills (upper limit of theoretical variogram) compared to the non-poultry farming area variogram. This is a proportional effect and has been observed in other cases where contamination levels are much higher than background levels (Englund and Sparks, 1988). In the Inland Bays region, the proportional effect is attributed to the significant range and variance in ground-water nitrate concentrations in poultry farming areas (Ritter and Chirnside, 1982, 1984; Andres, 1991a). This is thought to occur because nitrogen has been added to the land surface at much lower rates in non-poultry farming areas than in poultry farming areas.

The sample and theoretical variograms for poultry farming areas are shown in fig. 8. The poultry farming area sample variogram derived from the data set that was restricted to a maximum value of 25 mg/L nitrate-nitrogen has the smaller scatter. The theoretical variogram from this data set generally produced the best results from the XVALID program, and has a smaller nugget and sill than the data set created by removing a few samples. This variogram is used for kriging. Both sample variograms show a relatively large maximum value in the vicinity of the 2000-ft sample distance. This feature could be related to comparison of samples that are affected by different contaminant sources and/or to the spacing of strong contaminant sources. Anisotropic variograms were modeled, but the degree to which they better fit the data was judged to be insignificant. The data point search methods used by the KRIGE program (Englund and Sparks, 1988) to compute kriging weights are shown in Table 4.

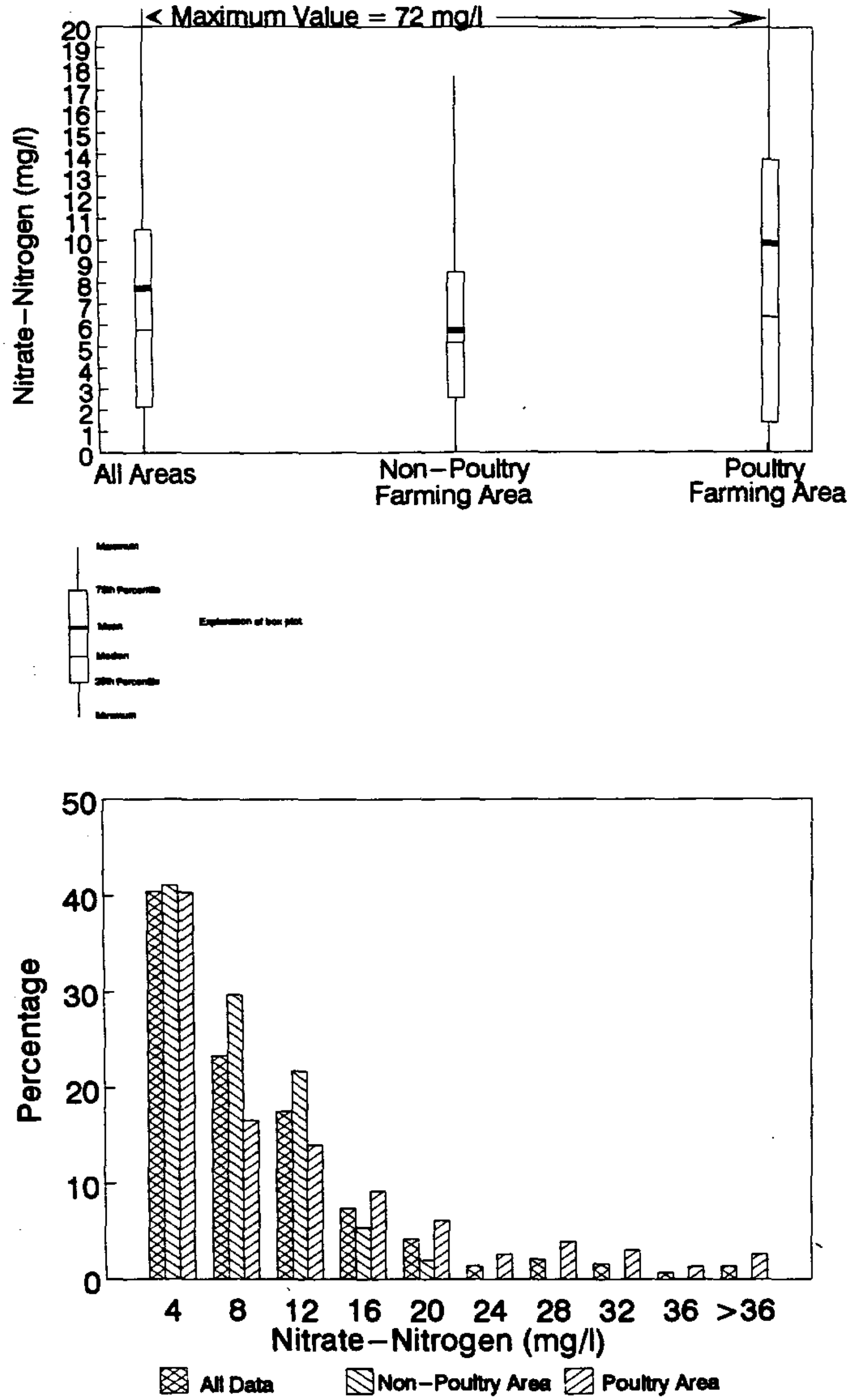


Figure 7. Summary statistical plots of the data sets.

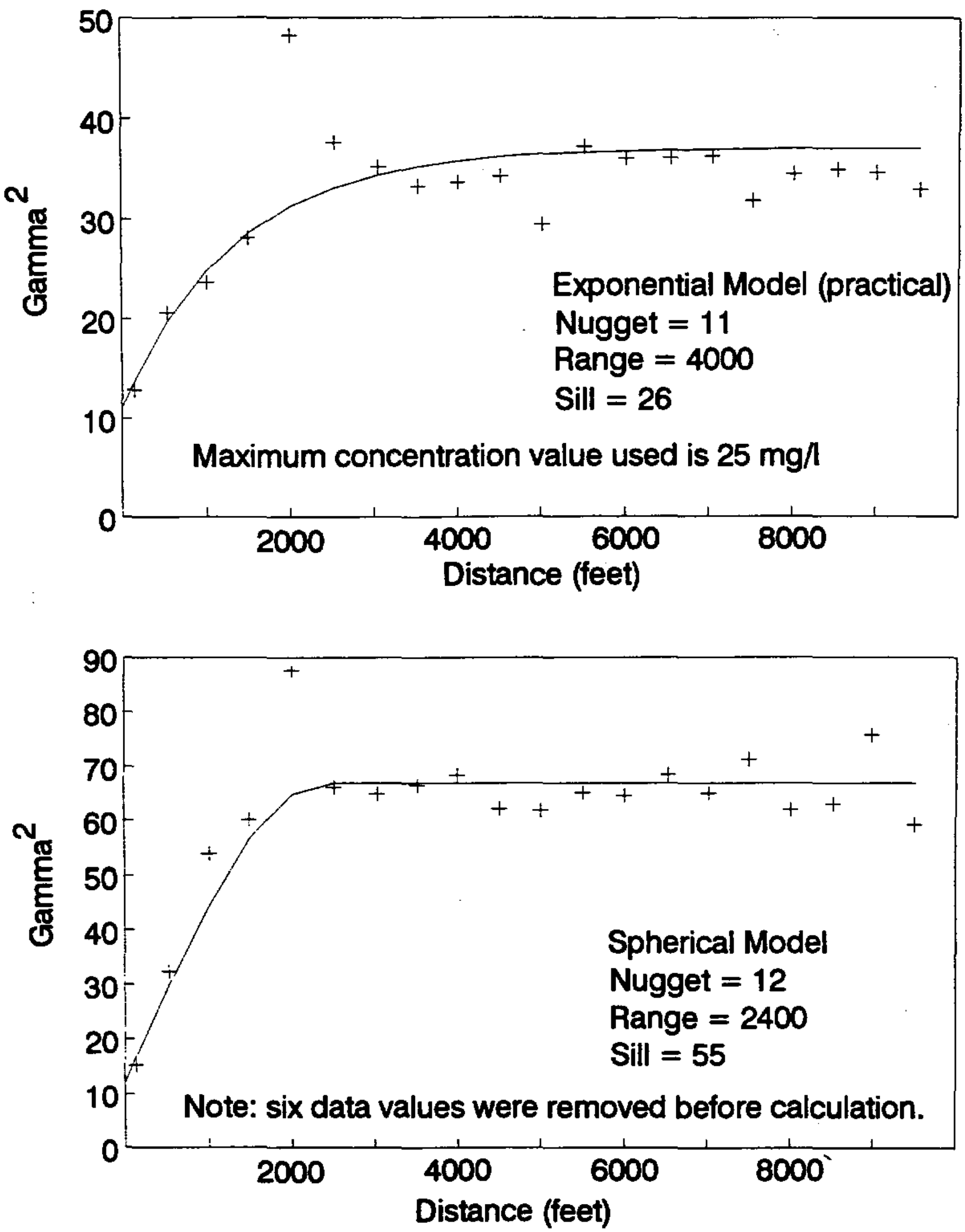


Figure 8. Variograms for poultry farming area. Upper variogram is used for kriging. The first lag distance is 250 ft, subsequent lag intervals are 500 ft. See text for additional discussion.

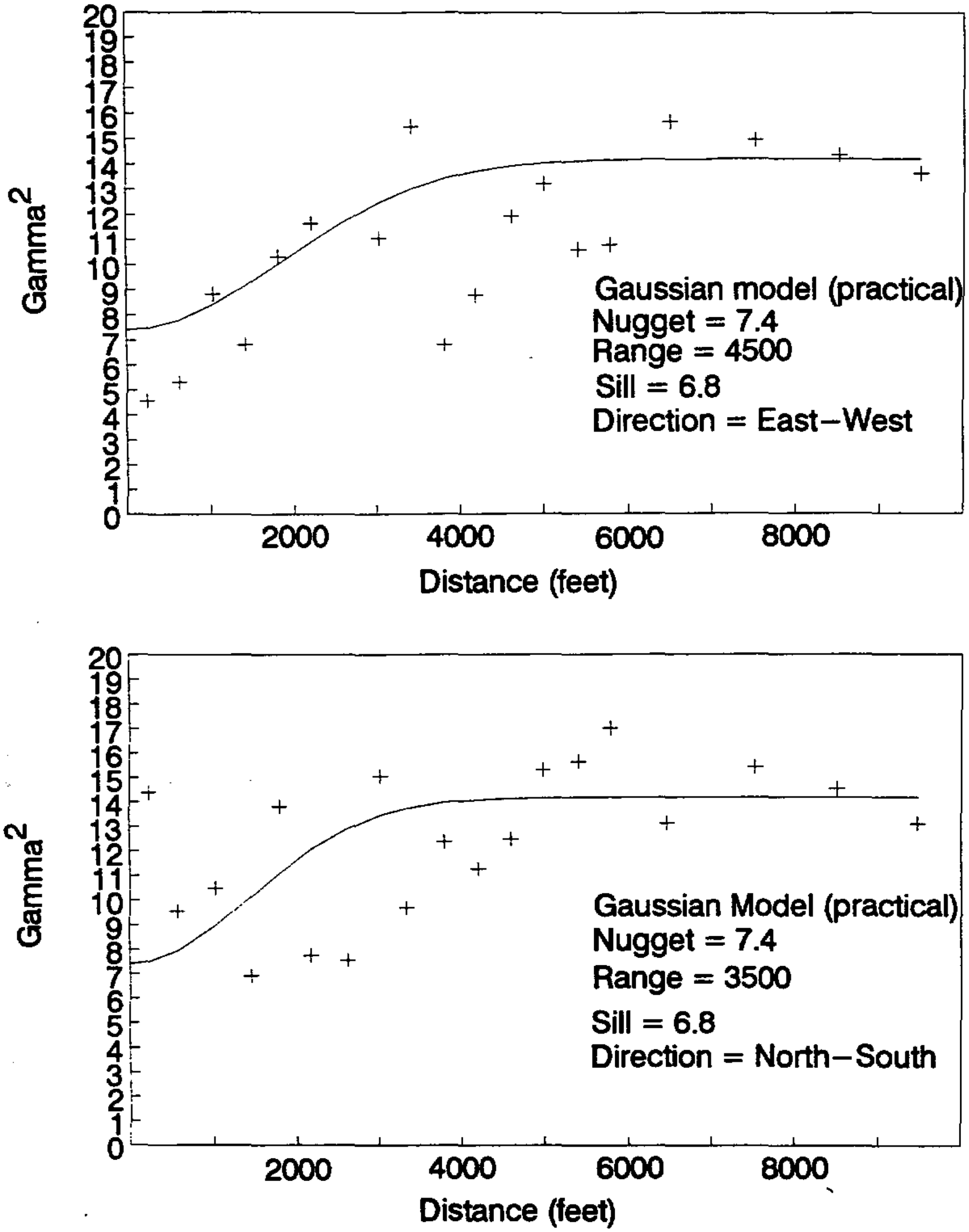


Figure 9. Variograms for non-poultry farming area. The first lag distance is 250 ft, subsequent lag intervals are 500 ft.

Table 4. Data point search methods used in kriging. The minimum search distance is used except for the Piney Neck and Long Neck east sub-basins.

Area	search method	search distance (ft)	minimum/maximum no. of neighbors per sector
Poultry Farming	quadrant 4 sectors	7500 - 10000	1/4
Non-poultry Farming	radial 1 sector	8000 - 10000	1/24

The non-poultry farming area data set sample variograms (fig. 9) show lower values than the poultry farming area variograms. As a result, the corresponding theoretical variograms have smaller nuggets and sills than the poultry farming area variograms. The correlation range is greater. The non-poultry farming area variograms do not show the relatively large maximum value near the 2000-ft sample distance. The greater range and lack of the maximum value at the 2000-ft distance may be related to the lack of strong point sources of contamination in the non-poultry farming area (proportional effect).

An anisotropic variogram model generally produces somewhat better results from the XVALID program (Englund and Sparks, 1988) than an isotropic model (fig. 9) and is used for kriging. In the anisotropic model the east-west component has a larger range than the north-south component. The north-south sample variogram also shows substantial scatter and larger differences at small distances. Based on an analysis of overall ground-water flow directions in the area covered by the non-poultry farming area data set, it is possible that the anisotropy reflects lesser nitrate concentration variability in the predominantly easterly ground-water flow direction. The data point search methods used by the KRIGE program (Englund and Sparks, 1988) to compute kriging weights are shown in Table 4.

Potential Nitrate-Nitrogen Flux and Nitrate-Nitrogen Loading Rates

Table 5 shows that the potential flux of nitrate-nitrogen through direct ground-water discharge into Rehoboth and Indian River bays is in the range of 236 to 356 and 1068 to 2180 pounds per day (lb/d), respectively (107 to 161 and 484 to 989 kilograms per day [kg/d]). Basin-wide areal loading rates range from 11.0 to 14.8 pounds per acre-year (lb/ac-yr) (12.4 to 16.6 kilograms per hectare per year [kg/ha-yr]) and 21.0 to 37.7 lb/ac-yr (23.5 to 42.3 kg/ha-yr) for Rehoboth and Indian River bays respectively. Loading rates are given in kg/ha-yr to allow for comparison with work by Ritter (1986). Table 6 shows the results for the individual sub-basins.

The results shown in tables 5 and 6 demonstrate that the potential flux of nitrate-nitrogen and areal nitrate-nitrogen loading rates from sub-basins in poultry farming areas are greater than that from sub-basins in non-poultry farming areas. Two-tailed t-tests on areal loading rates from flow-net and water budget derived ground-water fluxes show that the mean loading rates for poultry farming and non-poultry farming areas are different at the five percent level. Statistically significant differences in areal loading rates remain after equivalent unit ground-water fluxes are used to calculate loading rates. This indicates that the differences in areal loading rates between land-use types are not attributable to differences in ground-water fluxes. Instead, the higher areal loading rates are attributable to the higher ground-water nitrate concentrations usually observed in poultry farming areas. In addition, there is significant variation in potential nitrogen flux between sub-basins within the two major groupings. This variation is due to both differences in ground-water flux and ground-water nitrate concentrations.

Figures 10 and 11 illustrate the relative contributions of ground water (determined by the Darcian model) and nitrate from each sub-basin (A sub-basins only) to Rehoboth and Indian River bays and the relative sizes of the sub-basins. The relative sizes of the sub-basins also show the relative amounts of ground water, determined by the water-budget models, contributed by each sub-basin. The column labeled "Area Normalized NO_3-N Flux" in Table 6 shows which sub-basins contribute more (values greater than 1) or less (values less than 1) than the average amount of nitrogen to the bays. For both total and areally normalized nitrogen fluxes the Indian River north, Indian River south, and Piney Neck sub-basins contribute the most nitrogen to surface water. These sub-basins contribute almost 50 percent of the total nitrogen load through direct ground-water discharge to the Indian River Bay drainage basin, but only comprise about 30 percent of the land area.

Because of the temporal averaging inherent in the ground-water nitrate concentration estimation model, seasonal variations in potential nitrate-nitrogen flux are directly related to variations in ground-water flux. Given the results of gradient analysis, under normal hydrologic conditions about two-thirds of the total annual loading would normally occur during the winter and spring.

The indicators of nitrate flux errors due to estimation of nitrate concentrations are shown in fig. 12. The observed nitrate flux errors are a result of the spatial variation in ground-water nitrate concentrations and the spacing of data points. Differences in the ratios of mean nitrate concentration to median kriging standard deviation (KSD) show, in a relative way, which basins have better or poorer estimates. Because the computed KSDs are not independent and normally distributed, there is no simple way to compute nitrate-flux confidence intervals or to statistically compare the KSDs in one basin to another.

Comparison with Previous Work

One of the objectives of this study is to reevaluate the results of Ritter's (1986) loading study. In that study, he assigned nitrogen and phosphorus loading rates to different land-use types based on an extensive literature review and on the results of streamflow water-quality monitoring in several small drainage basins in the Inland Bays region and elsewhere. The result of his research was that similar nitrogen loading rates were applied to the Indian River and Rehoboth Bay drainage basins. With this approach the differences

Table 5. Total potential nitrate-nitrogen fluxes.

AREA NAME	DARCIAN MODEL: GROUND-WATER FLUX (MGD)	DARCIAN MODEL: NO_3-N FLUX (LB/D)	DARCIAN MODEL: AREA NORMALIZED NO_3-N FLUX	DARCIAN MODEL: LOADING RATE (LB/AC-YR)	AREA NAME	WATER BUDGET MODEL 1 (RECHARGE = 500,000 GALLONS PER DAY PER SQUARE MILE): GROUND-WATER FLUX (MGD)	WATER BUDGET MODEL 1: NO_3-N FLUX (LB/D)	WATER BUDGET MODEL 1: AREA NORMALIZED NO_3-N FLUX	WATER BUDGET MODEL 1: LOADING RATE (LB/AC-YR)	WATER BUDGET MODEL 2 (RECHARGE = 600,000 GALLONS PER DAY PER SQUARE MILE): GROUND-WATER FLUX (MGD)	WATER BUDGET MODEL 2: NO_3-N FLUX (LB/D)	WATER BUDGET MODEL 2: AREA NORMALIZED NO_3-N FLUX	WATER BUDGET MODEL 2: LOADING RATE (LB/AC-YR)
Total					Total								
A Sub-basins	39.22	2497		24.02	A Sub-basins	29.76	1896		18.24	35.71	2276		21.89
B Sub-basins	34.23	2234		30.93	B Sub-basins	20.71	1303		18.05	24.85	1564		21.66
Non-poultry farming area					Non-poultry farming area								
A Sub-basins	14.27	676	0.69	16.67	A Sub-basins	11.57	539	0.73	13.28	13.88	647	0.73	15.94
B Sub-basins	14.27	676	0.61	18.97	B Sub-basins	10.17	478	0.74	13.41	12.20	574	0.74	16.10
Poultry farming area					Poultry farming area								
A Sub-basins	24.95	1821	1.20	28.73	A Sub-basins	18.08	1357	1.17	21.41	21.69	1629	1.17	25.70
B Sub-basins	19.96	1557	1.38	42.58	B Sub-basins	10.43	825	1.25	22.56	12.52	990	1.25	27.07
Rehoboth Bay sub-basins					Rehoboth Bay sub-basins								
A Sub-basins	8.24	317	0.50	12.04	A Sub-basins	7.51	296	0.62	11.27	9.01	356	0.62	13.52
B Sub-basins	8.24	317	0.48	14.82	B Sub-basins	6.10	236	0.61	11.02	7.32	283	0.61	13.22
Indian River Bay sub-basins					Indian River Bay sub-basins								
A Sub-basins	30.98	2180	1.17	28.08	A Sub-basins	22.25	1600	1.13	20.61	26.70	1920	1.13	24.73
B Sub-basins	25.99	1917	1.22	37.71	B Sub-basins	14.61	1068	1.16	21.01	17.53	1281	1.16	25.21

Units: MGD = million gallons per day; LB = pounds; AC = acre; YR = year
Note: small differences amongst totals are due to rounding.

Table 6. Potential nitrate-nitrogen fluxes for individual sub-basins. See figs. 3 and 4 for sub-basin locations.

	FLOW-NET MODEL				WATER BUDGET MODEL 1 (RECHARGE= 500 GPD x MI[2])				WATER BUDGET MODEL 2 (RECHARGE =600K GPDxMI [3])			
BASIN OR SUB-BASIN NAME	GROUND-WATER FLUX (MGD)	NO_3-N FLUX (LB/D)	AREA NORMALIZED NO_3-N FLUX[4]	LOADING RATE (LB/AC-YR)	GROUND-WATER FLUX (MGD)	NO_3-N FLUX (LB/D)	AREA NORMALIZED NO_3-N FLUX[4]	LOADING RATE (LB/AC-YR)	GROUND-WATER FLUX (MGD)	NO_3-N FLUX (LB/D)	AREA NORMALIZED NO_3-N FLUX[4]	LOADING RATE (LB/AC-YR)
REHOBOTH BAY												
REHOBOTH BAY NORTH SHORE												
BASIN "A"												
LEFT	1.34	57	0.81	9.79	1.66	70	1.07	12.10	1.99	84	1.07	14.52
MIDDLE	1.15	52	1.39	16.71	0.88	40	1.14	12.83	1.06	48	1.14	15.40
RIGHT	0.25	13	0.67	8.02	0.47	25	1.35	15.26	0.56	30	1.35	18.31
TOTAL	2.73	122	0.96	11.54	3.01	135	1.14	12.81	3.61	162	1.14	15.37
BASIN "B"												
LEFT	1.34	57	1.57	23.31	0.70	29	1.10	12.10	0.83	35	1.10	14.52
MIDDLE	1.15	52	2.28	33.81	0.44	20	1.16	12.83	0.52	23	1.16	15.40
RIGHT	0.25	13	0.54	8.02	0.47	25	1.38	15.26	0.56	30	1.38	18.31
TOTAL	2.73	122	1.46	21.67	1.60	74	1.20	13.23	1.92	89	1.20	15.87
BASIN "A" = BASIN "B"												
ANGOLA NECK												
WEST	1.10	36	0.92	11.07	0.93	30	0.83	9.30	1.11	36	0.83	11.16
EAST	1.36	31	0.73	8.75	1.01	23	0.58	6.50	1.21	28	0.58	7.80
LONG NECK												
NORTH	2.94	121	1.17	14.10	2.45	101	1.04	11.75	2.94	121	1.04	14.10
EAST	0.12	7	1.51	18.19	0.12	7	1.61	18.19	0.14	9	1.61	21.83
TOTAL A[1]	8.24	317		12.04	7.51	296		11.27	9.01	356		13.52
TOTAL B[2]	8.24	317		14.82	6.10	236		11.02	7.32	283		13.22
INDIAN RIVER BAY												
BASIN "A"												
INDIAN RIVER												
NORTH	2.99	450	1.66	46.70	2.75	415	2.09	42.99	3.30	497	2.09	51.58
SOUTH	1.59	146	1.53	42.95	0.97	89	1.27	26.13	1.16	107	1.27	31.36
CEDAR NECK	2.34	140	1.81	50.74	0.79	47	0.83	17.03	0.94	56	0.83	20.43
WHITE CREEK	1.22	51	0.49	13.73	1.06	44	0.58	11.91	1.27	53	0.58	14.29
WHITE NECK	3.69	204	0.45	12.54	4.63	255	0.76	15.72	5.56	306	0.76	18.87
CHAMPLIN NECK	2.74	180	0.41	11.44	4.49	296	0.91	18.78	5.39	355	0.91	22.53
BASIN "B"												
INDIAN RIVER												
NORTH	2.99	450	1.89	71.15	1.81	272	2.05	42.99	2.17	327	2.05	51.58
SOUTH	1.59	146	3.20	120.75	0.35	32	1.24	26.13	0.41	38	1.24	31.36
CEDAR NECK	1.91	110	1.06	40.14	0.79	45	0.78	16.48	0.94	54	0.78	19.78
WHITE NECK	1.81	113	0.31	11.83	2.73	170	0.85	17.80	3.28	204	0.85	21.36
CHAMPLIN NECK	1.27	87	0.48	18.03	1.38	94	0.93	19.47	1.65	113	0.93	23.37
BASIN "A" = BASIN "B"												
LONG NECK												
SOUTH	5.91	352	0.91	25.42	3.95	235	0.82	16.98	4.74	282	0.82	20.37
EAST	0.12	7	0.65	18.19	0.12	7	0.88	18.19	0.14	9	0.88	21.83
PINEY NECK	7.73	463	1.70	47.68	2.77	168	0.84	17.30	3.32	202	0.84	20.76
DUMPLING NECK	2.65	187	3.06	86.03	0.62	44	0.98	20.14	0.74	53	0.98	24.17
TOTAL A[1]	30.98	2180		28.08	22.25	1600		20.61	26.70	1920		24.73
TOTAL B[2]	25.99	1917		37.71	14.61	1068		21.01	17.53	1281		25.21

Notes:

[1] Total "A" computed from "A" shorelines plus "A = B"

[2] Total "B" computed from "B" shorelines plus "A = B"

[3] Units: Ft=feet, D=day, KG=kilogram, MI=mile, YR=year, HA=hectare

[4] Area normalized fluxes are computed relative to either Rehoboth or Indian River Bay.

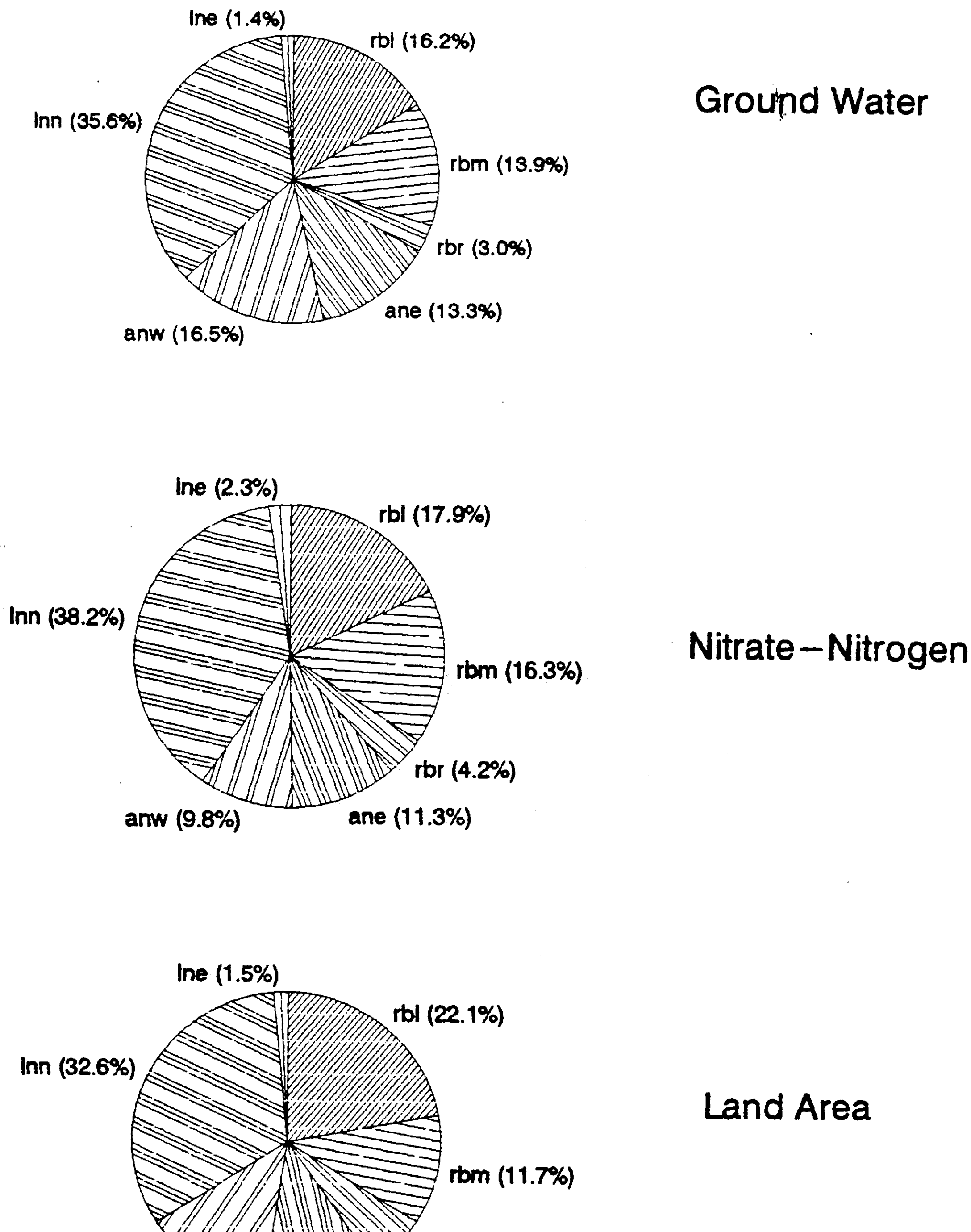


Figure 10. Relative contributions of ground water, nitrate, and land area for A sub-basins in the Rehoboth Bay drainage basin. rb* = Rehoboth Bay North Shore, l = left, m = middle, r = right; an* = Angola Neck, e = east, w = west; ln* = Long Neck, n = north, e = east. Relative ground-water and nitrogen concentrations determined by Darcian model.

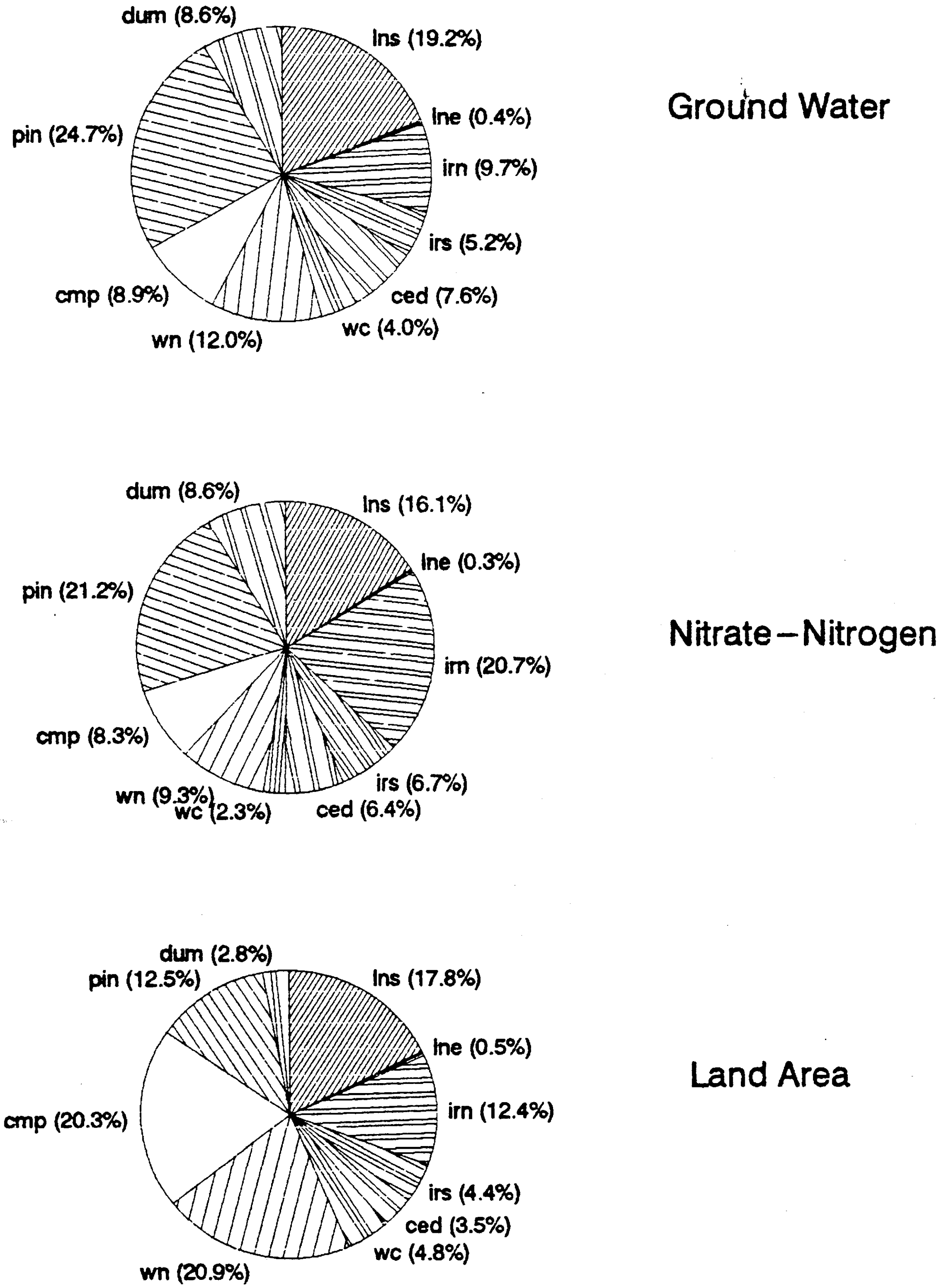


Figure 11. Relative contributions of ground water, nitrate, and land area for A sub-basins in the Indian River Bay drainage basin. ln* = Long Neck, s = south, e = east; ir* = Indian River, n = north, s = south; ced = Cedar Neck; wc = White Creek; wn = White Neck; cmp = Champlin Neck; pin = Piney Neck; dum = Dumpling Neck. Relative ground-water and nitrogen contributions determined by Darcian model.

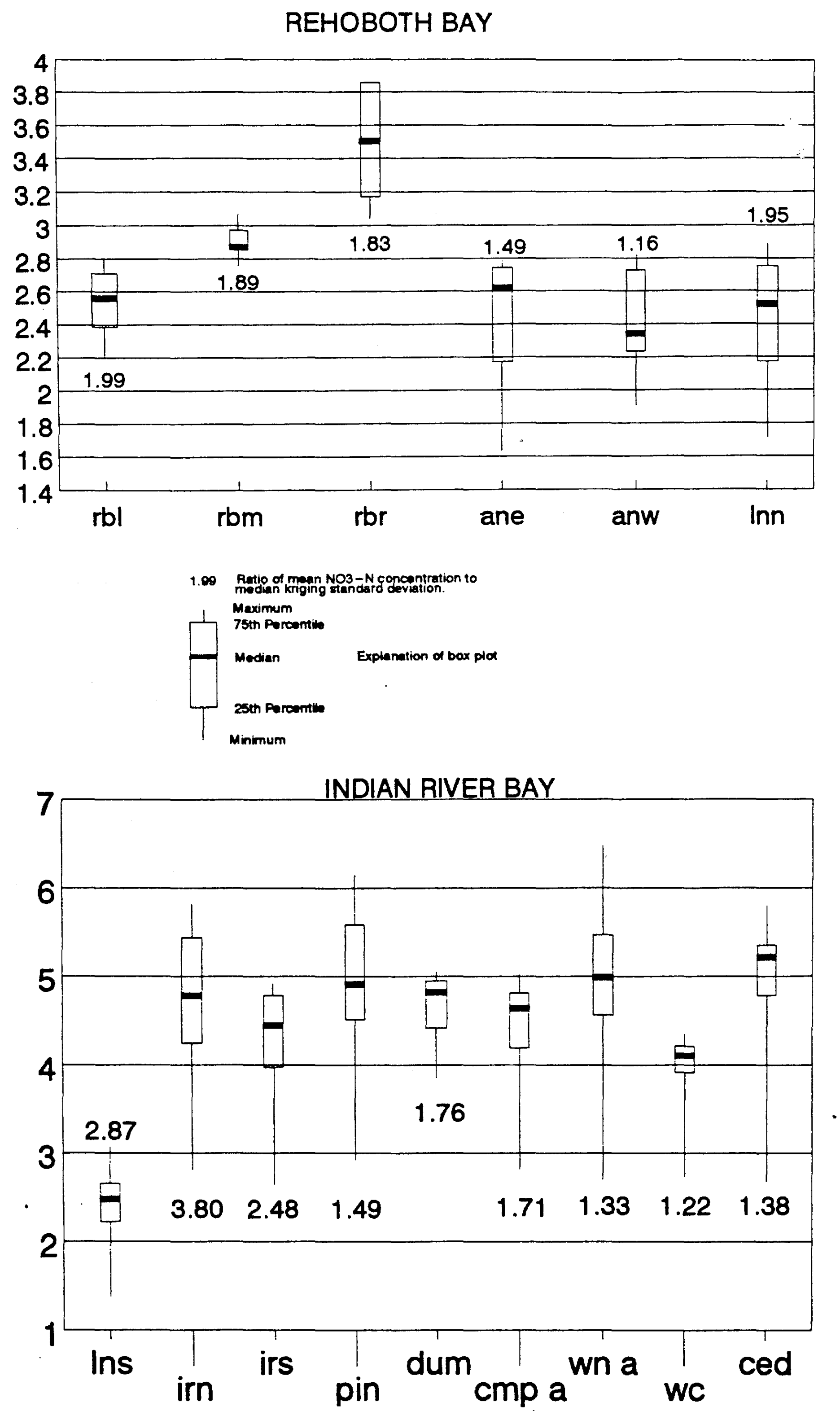


Figure 12. Box plots of kriging standard deviations for Rehoboth Bay and Indian River Bay A sub-basins and ratios of mean estimated nitrate concentrations and median kriging standard deviations. rb* = Rehoboth Bay North Shore Left, middle, and right; an* = Angola Neck east and west; ln* = Long Neck north and south; ir* = Indian River north and south; pin = Piney Neck; dum = Dumpling Neck; cmp = Champlin Neck; wn = White Neck; wc = White Creek; ced = Cedar Neck.

observed in the nitrogen loads and loading rates in sub-basin areas between this and Ritter's (1986) studies are expected because of the different methods used. However, the differences should average out over the study area if the loading rates used by Ritter (1986) are reasonable and if it is assumed that all of the potential nitrate-nitrogen load gets into surface water.

The basin-wide nitrogen loading rates for the Rehoboth Bay basin (tables 5 and 6) calculated by Darcy's Law and water-budget models average 12.7 lb/ac-yr (14.2 kg/ha-yr). Ritter's (1986) average areal loading rates for streams draining into Rehoboth Bay average 11.2 lb/ac-yr (12.5 kg/ha-yr). A two-tailed t-test shows that these means are not different at the five percent level. The basin-wide average areal nitrogen loading rates for the Indian River Bay basin calculated by Darcy's Law and water-budget models is 28.2 lb/ac-yr (31.6 kg/ha-yr). A one-tailed t-test (alpha = 0.5) shows that this value is greater than the maximum loading rate 20.2 lb/ac-yr (22.6 kg/ha-yr) for agricultural and urban lands used by Ritter (1986) and indicates that Ritter's agricultural lands loading rate is too low for the Indian River Bay basin.

Direct comparison of nitrogen loads calculated in this study and those from Ritter (1986) cannot be done effectively for individual sub-basins because of differences in drainage basin definitions. However, some of the drainage basin areas from this study are similar to those in Ritter (1986) and some observations can be made regarding direct ground-water discharge nitrogen areal loading rates.

- The Rehoboth Bay North Shore left A sub-basin occurs within the northern part of the Rehoboth Bay direct drainage area of Ritter (1986). Loading rates computed in this study range from 11.2 to 19.4 lb/ac-yr (12.6 to 21.7 kg/ha-yr) while loading rates calculated from Ritter's (1986, p. 44) loads average 11.8 lb/ac-yr (13.2 kg/ha-yr) and range from 5.89 to 13.3 lb/ac-yr (6.6 to 14.9 kg/ha-yr).

- Most of the area of the Rehoboth Bay North Shore middle A and right A sub-basins occurs within Ritter's (1986) Lewes-Rehoboth Canal basin. Loading rates computed in this study range from 8.02 to 18.3 lb/ac-yr (8.99 to 20.5 kg/ha-yr). Ritter's (1986, p. 43) figures yield loading rates that average 12.2 lb/ac-yr (13.7 kg/ha-yr) and range from 6.11 to 13.7 lb/ac-yr (6.85 to 15.4 kg/ha-yr).

- The Champlin Neck sub-basin covers much of Ritter's (1986) Blackwater Creek basin (Indian River Bay basin). Loading rates computed in this study are the lowest for any of the poultry farming area sub-basins. They range from 11.4 to 23.4 lb/ac-yr (12.8 to 26.2 kg/ha-yr). Loading rates calculated from Ritter's (1986, p. 31) figures average 11.7 lb/ac-yr (13.1 kg/ha-yr) and range from 5.89 to 13.2 lb/ac-yr (6.6 to 14.8 kg/ha-yr).

These figures show that Ritter's (1986) loading rates are reasonable for the Rehoboth Bay basin, but are too low for the Indian River Bay basin, again indicating that Ritter's (1986) loading rate for agricultural land use is not representative of poultry farming areas.

CONCLUSIONS

The results of ground-water flow and geostatistical modeling show that the potential flux of nitrate-nitrogen through direct ground-water discharge into Rehoboth and Indian River bays is in the range of 236 to 356 and 1070 to 2180 lb/d (107 to 161 and 484 to 989 kg/d), respectively. Basin-wide areal loading rates range from 11.0 to 14.8 pounds per acre-year (lb/ac-yr) (12.4 to 16.6 kilograms per hectare per year [kg/ha-yr]) and 20.6 to 37.7 lb/ac-yr (23.1 to 42.3 kg/ha-yr) for Rehoboth and Indian River bays respectively. Nitrate-nitrogen loading rates for the individual contributing sub-basins range from 6.5 to 120 pounds per acre-year (7.3 to 135 kilograms per hectare-year). The nitrogen loading is a direct result of land-use practices of the past 30 to 40 years.

There is substantial variation in ground-water flux, potential nitrate-nitrogen flux, and areal nitrate-nitrogen loading rates between the sub-basins making up the Rehoboth and Indian River Bay basins. These variations are attributed largely to differences in nitrate concentrations in ground water, although differences in ground-water fluxes account for some of the variation. Differences in nitrate concentrations in ground water are primarily due to land-use characteristics. Differences in estimated ground-water fluxes are due to spatial variation in the natural characteristics of the aquifer and to modeling assumptions.

Nitrate-nitrogen fluxes and loading rates generally are highest for sub-basins that have historically had intensive poultry production. Within this group, the Indian River north, Indian River south, and Piney Neck sub-basins have the largest nitrate-nitrogen fluxes and loading rates. These sub-basins contribute almost 50 percent of the total nitrogen load through direct ground-water discharge to the Indian River Bay drainage basin, but only comprise about 30 percent of the land area.

Areal nitrogen loading rates calculated in this study are similar to those used by Ritter (1986) for the Rehoboth Bay drainage basin. Areal loading rates calculated for the Indian River Bay basin and for the poultry farming area sub-basins as a whole are larger than the maximum rate used by Ritter (1986). This indicates that the loading rates used by Ritter (1986) are reasonable for all land-use types and areas except those that have had intensive poultry production in the past.

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