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David R. Wunsch, State Geologist

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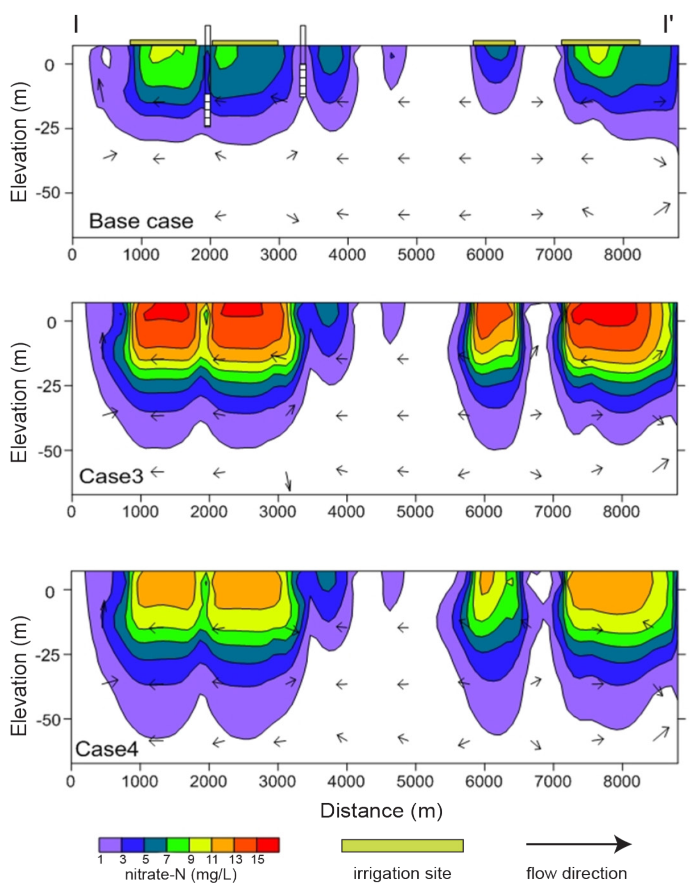
SIMULATION OF GROUNDWATER FLOW AND CONTAMINANT TRANSPORT IN EASTERN SUSSEX COUNTY, DELAWARE WITH EMPHASIS ON IMPACTS OF SPRAY IRRIGATION OF TREATED WASTEWATER

By

Changming He and A. Scott Andres



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SIMULATION OF GROUNDWATER FLOW AND CONTAMINANT TRANSPORT IN EASTERN SUSSEX COUNTY, DELAWARE WITH EMPHASIS ON IMPACTS OF SPRAY IRRIGATION OF TREATED WASTEWATER

ABSTRACT

This report presents a conceptual model of groundwater flow and the effects of nitrate (NO_3^-) loading and transport on shallow groundwater quality in a portion of the Indian River watershed, eastern Sussex County, Delaware. Three-dimensional, numerical simulations of groundwater flow, particle tracking, and contaminant transport were constructed and tested against data collected in previous hydrogeological and water-quality studies.

The simulations show a bimodal distribution of groundwater residence time in the study area, with the largest grouping at less than 10 years, the second largest grouping at more than 100 years, and a median of approximately 29 years.

Historically, the principal source of nitrate to the shallow groundwater in the study area has been from the chemical- and manure-based fertilizers used in agriculture. A total mass of NO_3^- - nitrogen (N) of about 169 kg/day is currently simulated to discharge to surface water. As the result of improved N-management practices, after 45 years a 20 percent decrease in the mass of NO_3^- -N reaching the water table would result in an approximately 4 percent decrease in the mass of simulated N discharge to streams. The disproportionally smaller decrease in N discharge reflects the large mass of N in the aquifer coupled with long groundwater residence times.

Currently, there are two large wastewater spray irrigation facilities located in the study domain: the Mountaire Wastewater Treatment Facility and Inland Bays Wastewater Facility. The effects of wastewater application through spray irrigation were simulated with a two-step process. First, under different operations and soil conditions, evaporation and water flux, NO_3^- -N uptake by plants, and NO_3^- -N leaching were simulated using an unsaturated flow model, Hydrus-1D. Next, the range of simulated NO_3^- -N loads were input into the flow and transport model to study the impacts on groundwater elevation and NO_3^- -N conditions.

Over the long term, the spray irrigation of wastewater may increase water-table elevations up to 2.5 m and impact large volumes of groundwater with NO_3^-. Reducing the concentration of NO_3^- in effluent and increasing the irrigation rate may reduce the volumes of water impacted by high concentrations of NO_3^- , but may facilitate the lateral and vertical migration of NO_3^-. Simulations indicate that NO_3^- will eventually impact deeper aquifers. An optimal practice of wastewater irrigation can be achieved by adjusting irrigation rate and effluent concentration. Further work is needed to determine these optimum application rates and concentrations.

INTRODUCTION

The population of Sussex County, Delaware, is expected to grow from just over 197,000 to more than 271,000 between 2010 and 2030 (Delaware Population Consortium, 2012). Groundwater is the sole source of drinking water for county residents and is a major source of fresh water for all other uses. Multiple large-scale projects have been constructed or proposed to dispose of treated wastewater via spray irrigation and rapid infiltration at rates of hundreds of thousands to millions of gallons per day (GPD). Collectively, spray irrigation and rapid infiltration basin systems are permitted and regulated as Large On-Site Wastewater Treatment and Disposal Systems (LOWTDS).

Not surprisingly, there are many concerns about the potential impacts of treated wastewater disposal on water quality. A better understanding of the hydrogeological system is essential for making proper and informed management decisions concerning groundwater use in this area.

Given the complexity of aquifer characteristics and development patterns, a numerical groundwater flow model can increase our understanding of the current groundwater flow system, and also provide a quantitative evaluation of groundwater level and quality changes under current and projected water use and wastewater disposal conditions.

Purpose and Scope

Similar to many flow model studies, the purpose of this work was to simulate groundwater levels, flow and quality, and to evaluate the completeness and suitability of existing hydrogeological data. This report documents the development of a simulation model, the results of model calibration, and analyses of water budgets, flow directions, and the impacts of existing LOWTDS.

This study focuses on a portion of eastern Sussex County (Fig. 1), a growing area with multiple existing and planned LOWTDS. By developing and analyzing a sub-regional groundwater flow model, we will address several issues related to these proposed projects:

1. Evaluating potential hydrological effects of existing and proposed wastewater reuse and LOWTDS.
2. Simulating the geochemical impacts of treated wastewater disposal on groundwater quality.
3. Evaluating the risks to both groundwater and surface water from the use of LOWTDS.

This study relies on the hydrogeological data generated by other projects and programs. No new field data were collected for this study.

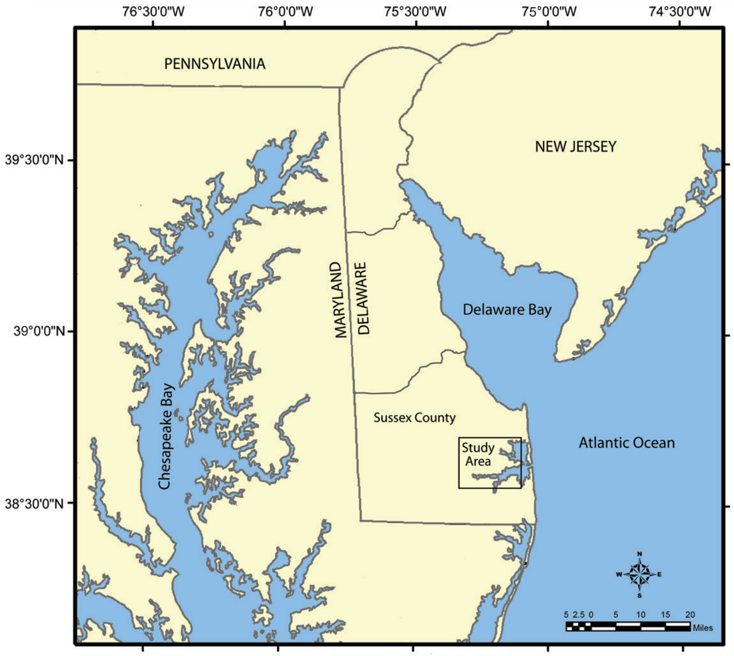

Figure 1. Location of study area and extent of the flow and transport model.

PERIOD	EPOCH	FORMATION	AQUIFER
QUATERNARY	HOLOCENE	*swamp, marsh, dune, upland, shoreline, and alluvial deposits*	*Unnamed confining units and sands*
	PLEISTOCENE	SCOTTS CORNERS	COLUMBIA
		LYNCH HEIGHTS	
NEOGENE	PLIOCENE?	BEAVERDAM	
	MIOCENE	BETHANY	POCOMOKE *and confining beds*
		b CAT HILL *a*	MANOKIN
		ST. MARYS	*Interbedded unnamed aquifers and confining units*
		CHOPTANK	
			MILFORD

Figure 2. Lithostratigraphy and hydrostratigraphy for the study area. This chart summarizes the names of the aquifers, the formations in which they occur, and their chronostratigraphic position. Areas shaded yellow are aquifers and areas shaded gray are confining beds.

Description of Study Area

The study area (Fig. 1), in Eastern Sussex County, Delaware, is bounded on the north, east, south and west by Love Creek, Rehoboth Bay, Indian River, and Cow Bridge Branch, respectively. The approximately 64 mi^2 region is part of the mid-Atlantic Coast Plain, and is comprised chiefly of unconsolidated deposits of gravel, sand, silt, and clay. The sediments are complexly stratified and form a sequence of aquifer and confining beds that reach more than 4,000 feet below the land surface to a sloping hard-rock floor (McLaughlin, et al, 2008). Since the purpose of this research is to study the impacts of wastewater reuse on the shallow aquifers, only the aquifers that overlie the St. Marys Formation are considered (Fig. 2). From shallow to deep, these aquifers are the Columbia, Pokomoke, and Manokin, and intervening confining beds. The lithostratigraphy and hydrostratigraphy for the study is illustrated in Figure 2 (Andres and Klingbeil, 2006).

SIMULATION OF GROUNDWATER FLOW

Groundwater flow was simulated using Visual MODFLOW (Schlumberger Water Systems, 2008), a 3D-finite-difference groundwater modeling program. This software is an implementation of Modflow-2000, developed by the U.S. Geological Survey (USGS).

Due to data limitations and the restricted goals of the modeling, model implementation was fairly simple. There were no long-term streamflow monitoring sites and the distribution of wells having water-level and water-quality observations extending over a sufficient length of time was sparse. Data from aquifer tests that could be used to determine the storage coefficient and storativity were limited. For these reasons, significant efforts were made to construct and calibrate a transient, groundwater flow model and to conduct contaminant transport simulations while understanding that data limitations have the potential to negatively impact the calibration of any model (see Results and Discussion).

Model Domain and Boundary Conditions

The dimensions of the model domain (Fig. 3) are 18,600 m (approximately 11.6 mi) in the east-west direction and 17,700 m (approximately 11.0 mi) in the south-north direction. A 372-by-354 node horizontal grid (50m by 50m, approximately 150 ft by 150 ft) oriented parallel with true north was constructed over the rectangular area. To form the 3D-finite-difference grid, the model domain was further subdivided into six layers of variable thickness. From top to bottom, layers 1 through 3 represent the Columbia aquifer, layer 4 represents a leaky confining layer, and layers 5 and 6 represent the Pocomoke and Manokin aquifers, respectively. Layer thicknesses were adapted from the grid products of Klingbeil and Andres (2006).

Boundary conditions define the manner in which water moves to or from the simulated groundwater system. The model used in this study employs recharge, no flow, constant head, river, and drain type boundaries (Fig. 3).

The northwest boundary is coincident with a surface-water watershed boundary (McKenna et al., 2007) and is simulated as a no-flow boundary. No-flow boundaries are also set at the base of layer 6 on the muddy beds of the St. Marys Formation, as well as at the eastern, northern, and southern limits of the model domain. Indian River and Rehoboth Bays and the tidal portions of creeks are represented as constant-head boundaries. The drain package is made up of smaller, non-tidal streams while the river package is made up of larger, non-tidal streams. Groundwater discharges to drains whenever the hydraulic head in the aquifer (i.e., the water-table elevation) rises above the drain bottoms. Math-

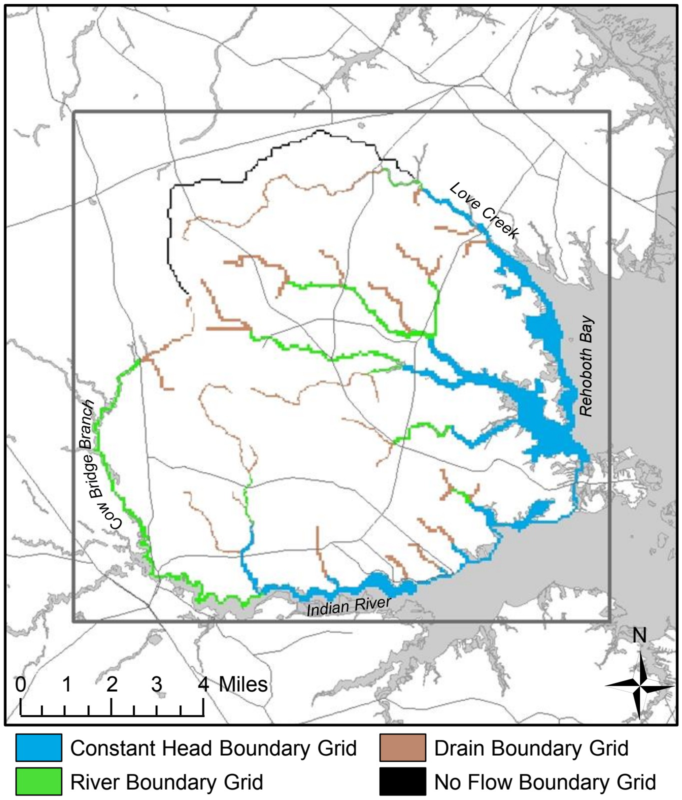

Figure 3. Map of model domain and boundary conditions.

assessments of aquifer materials, climatic water budget, and simulations. The magnitude of recharge was adjusted by an inverse simulation procedure that produced the best-fit, water-level observations in the surficial aquifer. We note that there is no significant difference in calibrated recharge rates between fair (green) and poor (red) recharge map units.

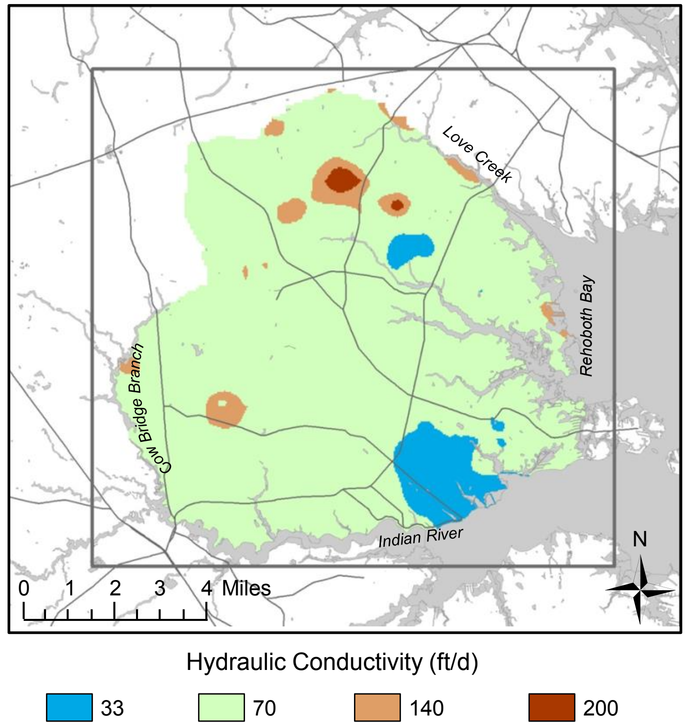

Figure 4. Calibrated hydraulic conductivity distribution map for layers 1-3 (Columbia aquifer).

ematically the rate of discharge is proportional to the difference between the water table and drain-bottom elevations, and a drain conductance term. Elevations for river and drain boundaries were determined in ArcMAP by intersecting hydrography with a Lidar DEM (USGS, 2005).

Hydraulic Properties

Andres and Klingbeil (2006) estimated the thickness and transmissivity of the unconfined aquifer of Sussex County, Delaware. We use their results to derive the hydraulic conductivity grid directly from the published thickness and transmissivity grids (Klingbeil and Andres, 2006) using equation 1:

$$K = \frac{T}{d} \qquad (1)$$

Where
K is hydraulic conductivity (ft/d);
T is transmissivity (ft^2/d);
d is thickness of the aquifer.

The resulting *K* grid was simplified by grouping hydraulic conductivity into four ranges, a common procedure for facilitating model calibration (Anderson and Woessner, 1992). The *K* value of each zone was adjusted during model calibration. The final calibrated results are shown in Figure 4.

Recharge

Recharge on the land-surface areas of the model was delineated by zone (Fig. 5). The initial values were assigned based on Andres (2004) and Andres et al. (2002) from

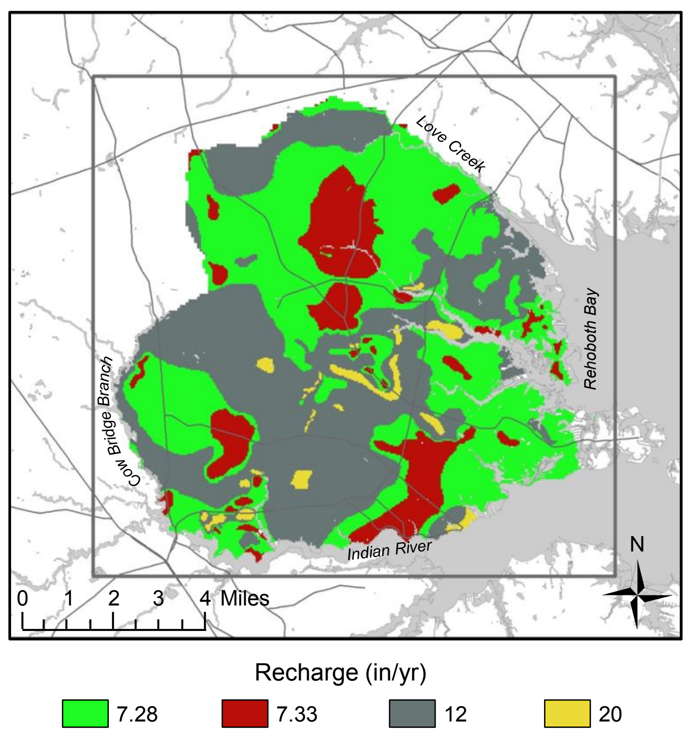

Figure 5. Calibrated recharge distribution map.

Pumping wells

Only pumping wells with allocation amounts greater than 100 gallons per minute (GPM) were considered, for a total of 10 agricultural wells and 15 public wells (Fig. 6). An estimated 7.4 million gallons of groundwater were withdrawn per day from the Columbia, Pocomoke, and Manokin aquifers.

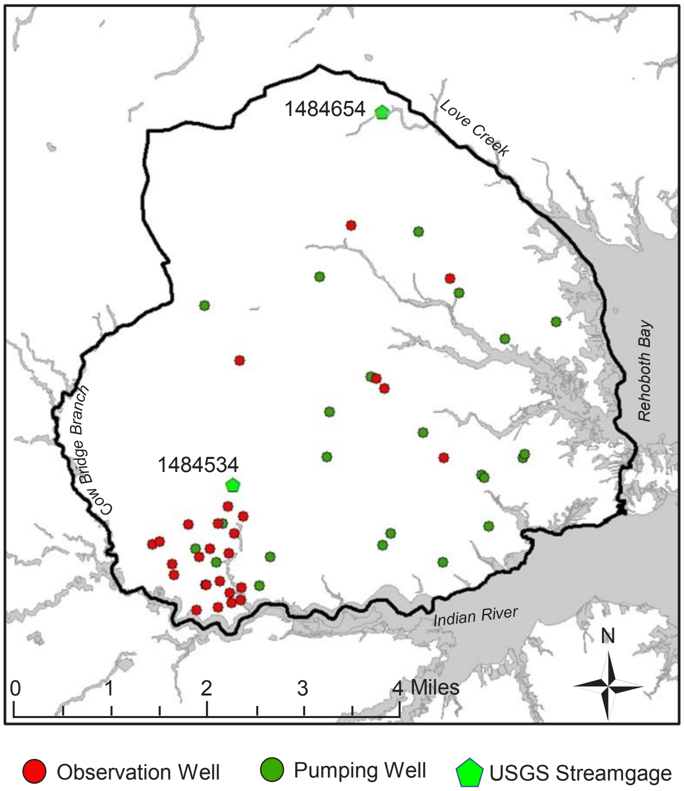

Figure 6. Locations of pumping wells, observation wells, and USGS streamgages in the study area. Model domain is outlined in black.

SIMULATION OF NITRATE TRANSPORT

Modeling approach

Simulation of nitrate (NO_3^-) transport was done with the program MT3DMS (Zheng and Wang, 1999), which uses the velocity field computed by MODFLOW to simulate the movement of dissolved chemical species in groundwater due to advection and dispersion. MT3DMS also was used to simulate the effects of chemical reactions on the concentration of NO_3^-.

The governing equation (equation 2) describes the fate and transport of aqueous- and solid-phase species in multi-dimensional saturated porous media (Zheng and Wang, 1999):

$$\frac{\partial(\theta C)}{\partial t} = \frac{\partial}{\partial x_i}\left(\theta D_{ij}\frac{\partial C}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(\theta v_i C) + q_s C_s + \sum R_n \tag{2}$$

Where:

C	concentration of dissolved species (N, P, et al.), M L^{-3};
θ	Porosity, dimensionless;
t	Time, T;
x_i	Distance along the respective Cartesian coordinate axis, L;
D_{ij}	Hydrodynamic dispersion coefficient tensor, L^2 T^{-1};
v_i	Seepage or linear pore water velocity, LT^{-1};
q_i	Volumetric flow rate per unit volume of aquifer representing fluid sources (positive) and sinks (negative), T^{-1};
C_s	Concentration of the source or sink flux for species, M L^{-3};
$\sum R_n$	Chemical reactions, M L^{-3} T^{-1}.

Different species have different reaction forms due to the biochemical properties of the specie itself and the aquifer.

Many studies of NO_3^- transformation in groundwater have concluded that the anion is quite soluble and is not significantly adsorbed onto clay-rich soils; NO_3^- may be reduced to N gas by multiple denitrification (DNF) processes, a group of interrelated biochemical processes through which microbes use NO_3^- in metabolic processes and transform NO_3^--N to other forms of N in anoxic conditions (Korom, 1992; Chowdary et al., 2004; Clement et al., 2002; Senzia et al., 2002). Transport model parameters are described in subsequent sections.

Nitrate loading

The major sources of NO_3^- to groundwater include point sources, such as municipal LOWTDS, and non-point sources, such as septic systems, poultry litter, horticultural and agricultural fertilizers, and atmospheric deposition (Robertson, 1977; Bachman, 1984; Ritter and Chirnside, 1984; Andres, 1991; Denver et al., 2004; Ator, 2008). Multivariate statistical analyses indicate that the presence of NO_3^- in groundwater is directly correlated with explanatory factors that describe land use (Greene et al., 2004). NO_3^- leaching from land surface to groundwater is achieved by assigning land-use-based NO_3^- concentrations (Table 1) in the recharge package of Visual MODFLOW. Land-use data are obtained from Sanborn, Inc. (2007), and the land-use-loading-rate values are from Horsley & Witten Inc. (1998).

Initial concentration

Hamilton et al. (1993) define natural groundwater as groundwater that is affected minimally by human activities, and is indicated by a threshold concentration of 0.4 mg/L NO_3^-. In this model, the pre-farming, initial NO_3^- concentration in the groundwater was assumed to be 0.4 mg/L.

Effective porosity

The advective velocity of groundwater is inversely proportional to the effective porosity (n_e) of the aquifer. The fac-

Table 1. Nitrate loading rates and recharge concentration (Source: Horsley and Witten, 1998).

Land Use	Percentage	Nitrate Loading Rate (lb/acre/yr)	Average Recharge Concentration (mg/L)
Cropland	27.80	38	15.8
Forest	24.70	0.22	0.11
Wetland/Surface water	23.00	0	0
Sewered Residential	10.30	9	4.37
Unsewered Residential	7.40	44	20.59
Range	1.50	0.22	0.11
Recreational Lands	1.20	39	21.27
Sewered Others	1.10	3	1.23
Unsewered Others	1.00	180	83.40
Farmsteads	0.80	26	12.82
Mixed Urban	0.80	70	35.23

tor n_e is typically less than the total porosity (n) because most groundwater flow occurs in a subset of n. There are very few published values of n_e in unconfined aquifers of the Atlantic Coastal Plain and therefore n_e is estimated to range from 20 percent to 40 percent (Dong et al., 2002; Sanford et al., 2012; Spayd and Johnson, 2003). In this study, a single effective porosity value (0.25) was assigned in the particle tracking and NO_3^- simulations.

Dispersivity

Dispersivity describes the variability of the velocity of a solute to be about the average groundwater velocity. In solute transport modeling, dispersivity is a lumped parameter that includes both molecular diffusion and mechanical dispersion (Anderson and Woesner, 1992). In MT3DS, diffusion and dispersion are represented by three dispersivity coefficients: one, the longitudinal dispersivity, or dispersion along the primary flow axis; two, the transverse dispersivity, or dispersion in the horizontal and vertical directions normal to the flow axis. In practice, dispersivity is scale dependent. In this study, the initial value for the longitudinal dispersivity was set to 50 m, representing heterogeneity within a one-grid cell, and the ratio of transverse and vertical dispersivity to longitudinal dispersivity was set to 0.1 and 0.01, respectively. Sensitivity analysis indicates that the results are not very sensitive to changes in dispersivity parameters.

Denitrification

In the well-oxygenated, low dissolved organic carbon (DOC) waters of the Columbia aquifer, N is stable and transported primarily as dissolved NO_3^--N (Ritter and Chirnside, 1984; Andres, 1991; Denver et al., 2004; Ator, 2008). Where the aquifer contains low concentrations (< 1 mg/L) of dissolved oxygen (DO) and affords adequate contact time between water and microbial organisms, NO_3^- concentrations are typically less than 1 mg/L (Andres, 1991, Denver, 1989). This indicates that NO_3^- may be removed from the system through DNF (Peterjohn and Correll, 1984; Spruill, 2000; Rivett et al., 2008).

The lack of data on the concentrations and biochemical characteristics of DOC, solid phase organic carbon, on the concentrations of dissolved gasses, and on the types and concentrations of organisms needed to characterize DNF systems, make it impractical to precisely model DNF (Colbourn, 1993). Nonetheless, there is a need to model NO_3^- transport in groundwater and so simplifying assumptions are commonly used (Heinen, 2003; Heatwole and McCray, 2007). Heinen (2003) reviewed more than 50 simplified models and found that more than 70 percent use first-order decay to describe DNF. The first-order decay model used in this study is

$$\sum R_n = \lambda \theta C \tag{3}$$

where λ is the first-order DNF coefficient with unit T^{-1}, and does not explicitly model microbial and chemical reactions and gaseous diffusion processes. Rather, the DNF coefficient incorporates all of these processes in a single, simplified term. As a result, the reported first-order DNF rates vary widely in the literature, and the choice of λ is always a challenge in modeling NO_3^- transport in groundwater. The modeling results must be viewed within these limitations.

The current study area also does not have data for the precise modeling of DNF and so indirect proxy evidence is used to approximate λ. Although there are no specific DO or DOC data, waters of the deeper, confined Pocomoke and Manokin aquifer have dissolved iron concentrations greater than 1 mg/L (Hodges, 1984; DGS internal records), which are typical of the anoxic conditions that would promote DNF. Land cover affects the amount of NO_3^- leaching to shallow aquifers; for example, the organic-carbon rich soils of forests and wetlands are more efficient at removing dissolved nutrients from infiltrating groundwater than soils under other land uses (Osborne and Kovacic, 1993). DOC leaching from soils to the water table also may lead to lower DO concentrations in the underlying aquifer, which promotes DNF (Peterjohn and Correll, 1984; Spruill, 2000). In this study, higher DNF rates are assigned to cells representing deeper aquifers and to cells in layer 1 under forested land cover (see Table 2)

Table 2. Denitrification rates used in the nitrate transport model.

Layers	Denitrification Rate (d^{-1})
Layer 1 (no forest)	0
Layer 1 (forest)	0.01
Layer 2	0
Layer 3	0.001
Layers 4, 5, 6	0.01

RESULTS AND DISCUSSION

Steady state flow calibration

The model was calibrated using hydraulic-head measurements from 29 wells (Fig. 6) and estimates of groundwater discharge to streams derived from streamgage data. Wells having more than 50 observations were selected. The head values were temporally averaged over all historical measurements as calibration targets. Only two streamgages, USGS 01484534, Swan Creek near Millsboro, DE, and USGS 01484654 Bundicks Branch at Robinsonville, DE (Fig. 6), are located within the model domain and had data suitable for estimating groundwater discharge. Daily mean base flows were estimated by hydrograph separation (Arnold et al. 1995, 1999). The annually averaged daily base flow, from the August 1998 to March 2000 period of record, was 3,717 m^3/d for Bundicks Branch and 1,109 m^3/d for Swan Creek, roughly 71 percent of total streamflow. The short period of record limits the extrapolation of this analysis to other streams.

During calibration, horizontal and vertical hydraulic conductivity, recharge, and streambed conductance were adjusted using a trial-and-error procedure to obtain the best fit between observed and simulated head and stream flux values. Figures 7 and 8 show that the calibrated model fits to field observed head observations had a root mean square error (RMSE) of 1.1 ft (0.36 m) with normalized RMSE of 3.95 percent, and fit flux observations with an error less than 3 percent. The simulated baseflow for Bundicks Branch and Swan Creek are 3,610 m^3/d and 1,086 m^3/d, respectively. Simulated base flow is approximately 70 percent of total streamflow, which is similar to values reported by Johnston (1976) for four other small watersheds in southern Delaware.

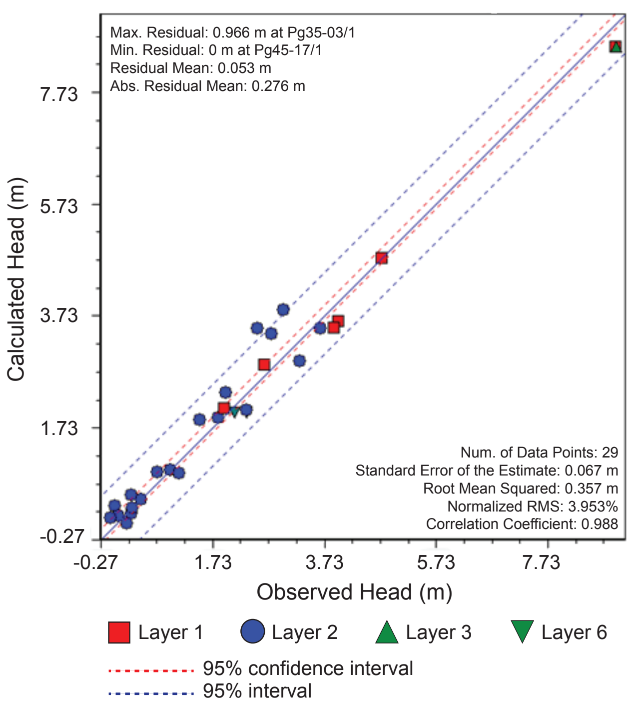

Figure 7. Simulated heads and field observed heads.

Groundwater residence time

The particle tracking module MODPATH is commonly used to compute groundwater flow velocities and particle pathlines (Pollock, 2012). By placing particles at the water table in each model cell, and using the forward-tracking feature, MODPATH can estimate and characterize groundwater residence times between recharge to the water table and discharge to cells representing surface water features (i.e., streams, creeks, and bays). The velocities and residence

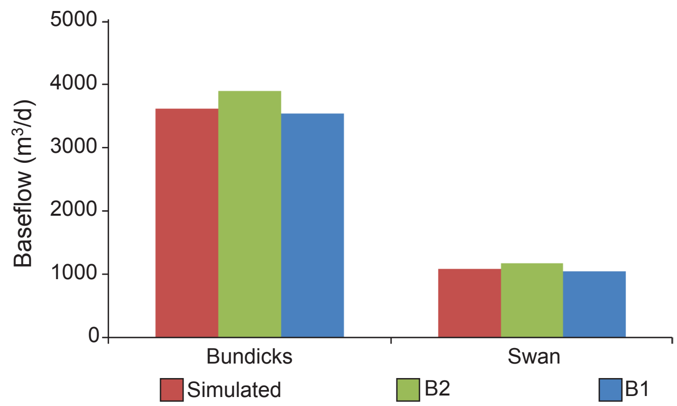

Figure 8. Simulated baseflow and field measured baseflow from 1998 to 1999. B1 and B2 are the baseflow that is estimated in the first pass and second pass, respectively (Arnold et al. 1995).

times computed by MODPATH are considered conservative since they do not take into account the physical and chemical processes that can alter the velocity of water. The particle tracking experiment was done under steady-state conditions.

Short duration flow paths are associated with areas adjacent to streams, as indicated by the simulated groundwater residence times (Fig. 9). These values display similar spatial patterns to those reported by Sanford et al. (2012) from a Delmarva-wide groundwater flow model and by Russoniello (2012) from a groundwater flow model of eastern Sussex County. This is noteworthy because these models used larger grid cell sizes and represented boundary conditions differently than in the current study, indicating that particle velocities are not sensitive to variations in boundary conditions and grid spacing.

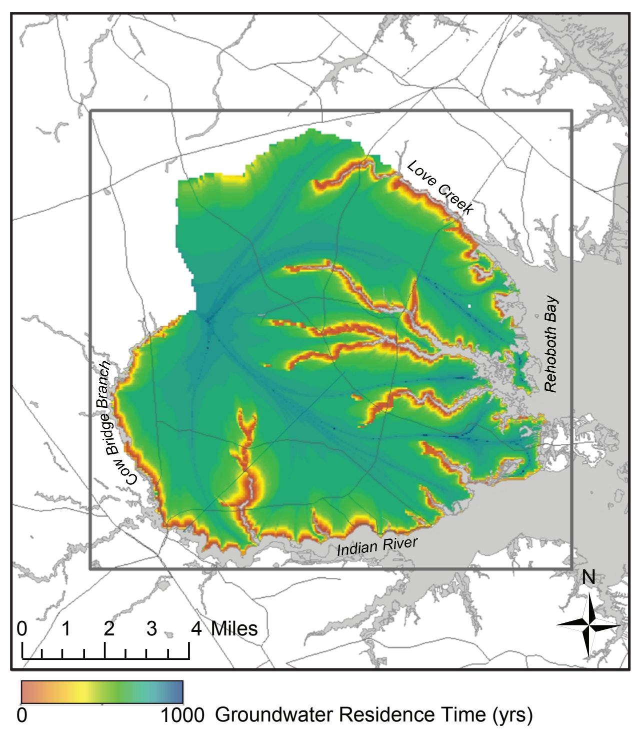

Figure 9. Simulated groundwater residence time in the study area.

A histogram of simulated groundwater residence times (Fig. 10) has a bimodal distribution, with residence times less than 10 years as the largest grouping and more than 100 years as the second largest grouping. This distribution (Fig. 9), along with the calculated median groundwater residence time of 29 years, indicate that several decades will be required to flush out the majority of the water now in the aquifer. Kasper et al. (2010) simulated flow for a single small sub-watershed within our model domain, and their particle tracking analysis found that a significant portion of the water recharging the aquifer discharged to a stream within 10 years. Though the Kasper et al. (2010) model simulated a maximum residence time greater than 100 years, it did not predict as large a portion of old water as was simulated by the current model study. Their results are most likely due to the smaller size, spatial location, and single sub-watershed characteristics of their model domain.

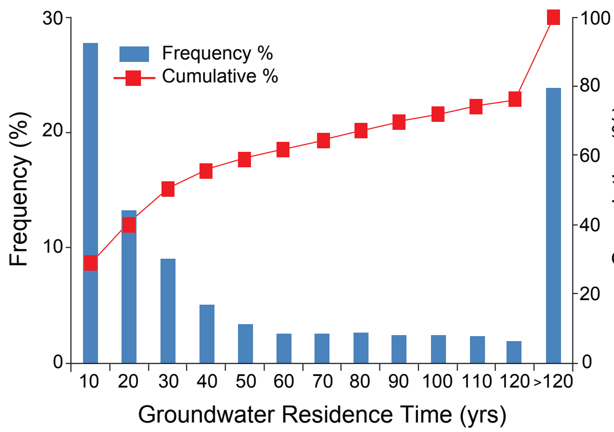

Figure 10. Histogram of groundwater residence time in the study area.

Nitrate transport simulation results

A NO_3^- transport simulation was run for 50 years under steady-state flow conditions. In the simulation, NO_3^- was added to the groundwater in recharge. Recharge rates and concentrations are described in previous sections. At the end of the simulation period for agricultural land, more than 82 percent of the NO_3^- concentrations in layer 1 were between 6 mg/L and 12 mg/L (Table 3, Fig. 11), and average concentrations were 7.6 mg/L, which is very close to the data from past studies (Sims et al., 1996, Ritter and Chirnside, 1984;

Table 3. Simulated nitrate concentrations in layer 1 for different land uses.

Land Use	Mean (mg/L)	Median (mg/L)	Minimum (mg/L)	Maximum (mg/L)	Standard Deviation (mg/L)
Agriculture	7.6	7.5	0.3	26	2.8
Forest	0.4	0.2	0	6.4	0.6
Residential*	3.7	3.6	0	15.3	1.9
Residential**	9.1	8.7	0	21.6	4.2

* Sewered
** Unsewered

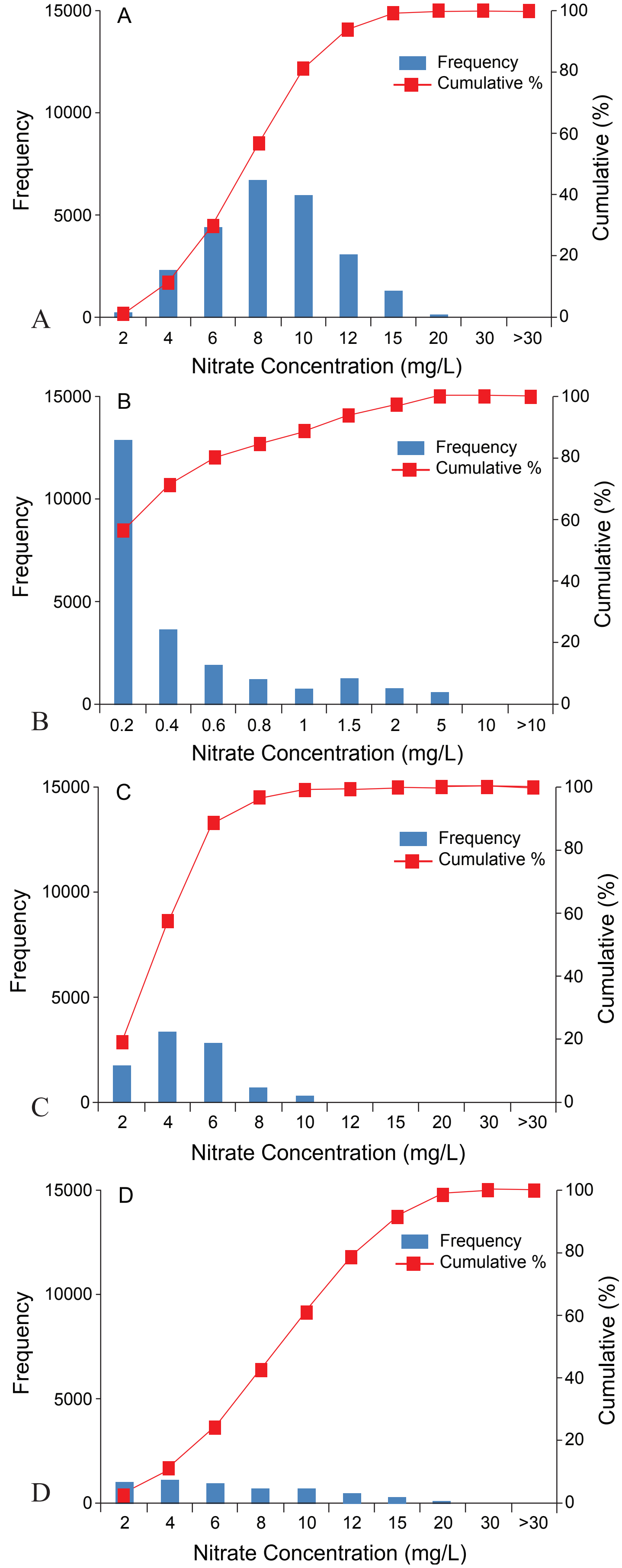

Figure 11. Histogram of simulated nitrate concentration beneath different land uses. A agriculture; B forest; C sewered residence; D unsewered residence.

Andres, 1991). Simulated NO_3^- concentrations were generally very low under forested lands because low concentrations of NO_3^- were applied to forested model cells and DNF occurs in cells underlying forested areas. Simulated NO_3^- concentrations under areas with unsewered residences are more than twice those of areas with sewered residences, primarily due to the greater input and leaching rates of NO_3^- from on-site wastewater disposal.

The model predicts that the highest NO_3^- concentrations will be in the Columbia aquifer (top three model layers), with very little NO_3^- reaching the Pocomoke and Manokin aquifers. This is similar to the results of groundwater quality studies (Robertson, 1977; Hodges, 1984; Ritter and Chirnside, 1984; Andres, 1991; Kasper and Strohmeier, 2007). Simulated NO_3^- concentrations decrease with increasing distance and depth due to dilution, and for model cells with a finite DNF rate, due to DNF. This model represents NO_3^- input as continuous recharge, so NO_3^- concentration contours show a smooth decrease downward from the land surface. In contrast, Andres (1991, 1995) shows isolated masses of water with high concentrations of NO_3^- at depth in the aquifer, due to discontinuous input of NO_3^- over time.

The total mass of NO_3^- that discharges to surface water was calculated by multiplying the groundwater flow rate and the groundwater NO_3^- concentration within the surface water boundary. Figure 12 shows that N discharge to surface water increased quickly in the initial 20 to 30 years of the simulation and then slowly but steadily increased. At the end of simulation, the annual mass of N that discharged to surface water was approximately 169 kg/d. This simulated pattern was very similar to that observed in surface waters of the Inland Bays watershed during the 1970s through 1990s (DNREC, 1998), indicating that observed increases in NO_3^- concentrations in surface water can be partly explained by groundwater transport processes. Although concurrent increases in NO_3^- inputs during that period (Sims et al., 1996) were not modeled, they would be expected to amplify the increase in NO_3^- flux to surface water during that period.

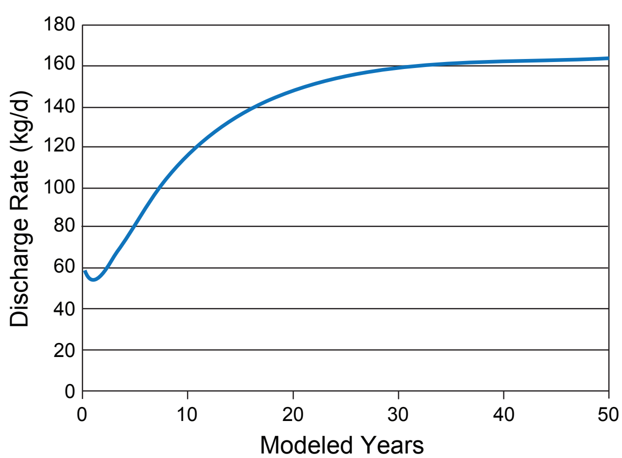

Figure 12. Simulated total nitrate discharge to surface water.

APPLICATION OF THE MODEL

Impacts of pumping on groundwater and surface water interactions

A comparison of results from simulations of steady-state flow without pumping to those with pumping show that pumping in the Angola Neck and Long Neck areas (areas circled in red on Fig. 13) caused some of the surface water cells to change from the expected condition of being discharge areas to being recharge areas that lose water to the underly-

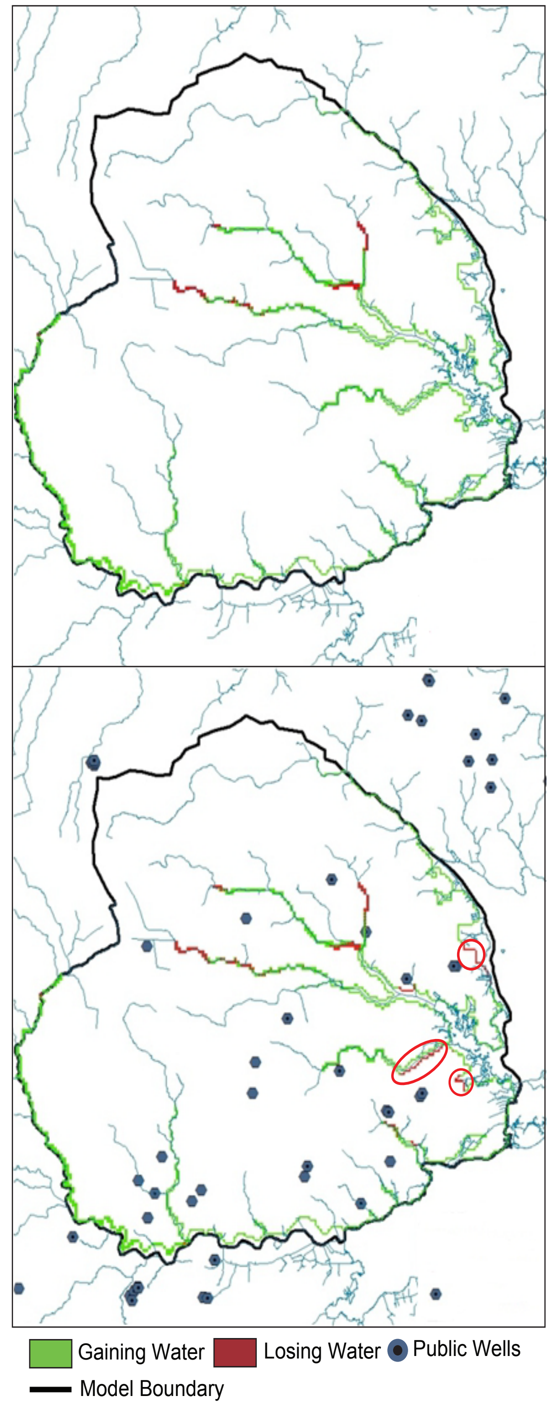

Figure 13. Comparison of groundwater-surface water flow directions due to pumping wells. Areas circled in red are areas that changed from being areas of discharge to being areas of recharge.

ing aquifer. In locations where the flow directions were not changed, pumping caused a reduction of the discharge to surface water (Fig. 13). The total change of groundwater flux to rivers and streams was approximately 4.7 percent of the amount that was pumped by wells.

Effects of improved nitrate management practices on groundwater quality

To assess the effects of improved NO_3^- management practices, the NO_3^- concentration in recharge on agricultural lands was reduced 20 percent, from a base of 15 mg/L (Table 1) to 12 mg/L after 5 years; the model then was run for an additional 45 years. The results show that NO_3^- discharge to surface water decreased from 169 kg/year to 162 kg/year, about a 4 percent reduction (Fig. 14).

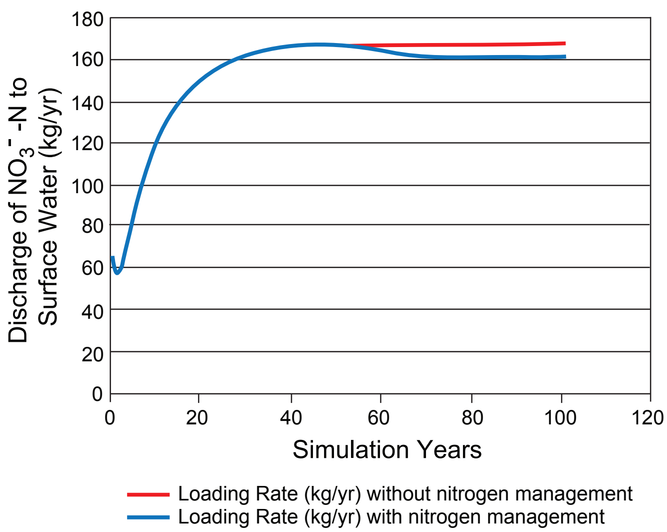

Figure 14. Sensitivity of reducing nitrate loading on agricultural land on the total groundwater discharge loading to surface water.

Impacts of wastewater spray irrigation on groundwater quality

Spray irrigation is a wastewater treatment technology that is widely employed in the United States (USEPA, 2006) and in the State of Delaware (Ritter et al., 2012). Municipal wastewater that has received secondary or higher treatment is applied to a vegetated soil surface so that the wastewater is further treated as it flows through the plant root/soil matrix (USEPA, 2006). The water may be taken up by plants by evapotranspiration, percolate down through the soil, and/or follow other pathways (USEPA, 2006).

The two largest wastewater spray irrigation facilities in the study domain (shown on page 11), Mountaire Wastewater Treatment Facility (MWTF) and Inland Bays Wastewater Facility (IBWF), are the focus of this simulation. Data for the amounts of wastewater applied via spray irrigation over time available from State sources were incomplete and acquiring data from the site operators was beyond the scope of this study. Therefore, a unit average application rate was approximated from flow information on MWTF (DNREC, 2010) and the size of the MWTF spray fields was estimated from aerial photography. This unit rate was used for both the MTWF and IBWF facilities. Average N concentrations (48 mg/L) in effluent from DNREC (2010) were also used in this simulation.

Monthly average precipitation is relatively constant between 3 inches and 4 inches except in August, when it rises to 5.46 inches. Potential evapotranspiration (PET) is larger during the growing season, from April to October (Table 4).

Table 4. Mean monthly precipitation and potential evapotranspiration data (1971-2000) recorded at the Georgetown Weather Station. This data can be obtained from the Office of the State Climatologist at the University of Delaware, Department of Geography: http://www.udel.edu/leathers/declim.html.

Month	Precipitation (in)	Potential ET (in)
JAN	3.67	0.00
FEB	3.15	0.00
MAR	3.99	0.62
APR	3.44	2.00
MAY	3.65	3.72
JUN	3.41	5.25
JUL	3.70	6.10
AUG	5.46	5.31
SEP	3.41	3.74
OCT	3.27	2.02
NOV	3.18	0.75
DEC	3.46	0.00

To study the impacts of the spray irrigation of wastewater on groundwater quality, we used a multi-step simulation. First, Hydrus-1D (Šimůnek et. al., 1998) was used to simulate the down flow of water and NO_3^- from the land surface to the water table. Next, simulated water flux and solute concentrations from Hydrus-1D were used as inputs to the three-dimensional MT3DMS transport model described earlier in this report. In Hydrus-1D, root water uptake was simulated using Feddes' model (Feddes et al, 1978), which is defined as follows:

$$S(h) = \alpha(h)S_p \qquad (4)$$

where $\alpha(h)$ is a root water uptake stress response function and S_p is the potential water uptake rate, which can be related to the potential transpiration, T_p , as follows:

$$S_p = b(x)T_p \qquad (5)$$

where $b(x)$ is the normalized water uptake distribution [L-1] over the root zone. In eastern Coastal Plain soils, the average root zone depth of corn, winter wheat, and soybeans has been reported to be 48 inches (approximately 1.2 m), with the effective rooting depth of 24 inches (Johnson, 2007). In this model, we assume that b(x) is constantly equal to one within the effective rooting depth and linearly decreases to zero at the bottom of the root zone. The simulated average annual recharge to groundwater, which includes precipitation, spray-irrigated water, and water uptake by roots, is approximately 54 inches.

Several assumptions were made to simulate N transport from the land surface to the water table:

1. All applied N (organic and NH_4^+) was converted to NO_3^- within the same time period during the application.
2. No additional artificial NO_3^- was applied to the irrigation sites besides what was added with treated wastewater.
3. NO_3^- in precipitation can be neglected compared to NO_3^- in treated wastewater; NO_3^- concentration from effluent was adjusted for dilution by precipitation.
4. There was no adsorption of NO_3^- to soil.
5. The removal of NO_3^- in natural agricultural soil was achieved through two main mechanisms: uptake by plants and DNF.

In this study, a total soil profile depth of 11.5 ft (3.5 m) was chosen (Fig. 15) to simulate NO_3^- leaching from the land surface through the vadose zone to the water table. Because soils in the study area are characterized as well- or excessively well-drained and having low concentrations (<0.1 %) of organic matter, usually limited to the uppermost 1 ft (0.3 m) of the soil profile (USDA-NRCS, 1974; Ator, 2008), we assumed that DNF only occurs in the upper 3.5 ft (1.1 m) of the soil profile. A first-order decay model was used to simulate DNF. Other model parameters, such as saturated hydraulic conductivity and porosity, were assigned the same values as those used in the saturated groundwater flow model.

To test the sensitivity of the model to uncertainties in DNF rates and effluent flow and concentration data, we conducted four 2-year simulations with varying DNF rates and wastewater N-concentrations (Table 5). In cases 1, 2 and 3, we assumed that a constant irrigation rate of 27.2 in/year, with NO_3^- concentration of 48 mg/L, was uniformly applied on the two irrigation sites; three DNF rate constants ranging from 0 (no DNF) to 0.004/day, from 0 (no DNF) to 0.025/day, and a median of literature reported value (McCray et al., 2005), were simulated. In case 4, we assumed that the N-concentration in the effluent decreased by 50 percent (24 mg/L) and the irrigation rate doubled (54.4 in/year) to keep the total mass of discharging NO_3^- unchanged. The DNF rate constant is same as case 3 (0.004/day).

The model was run for one year to initialize moisture content and NO_3^- concentrations in the soil profile. The model was run for a second year and the resulting flow and concentration values are summarized in Table 5. Model predicted NO_3^- concentrations are very sensitive to changes in DNF.

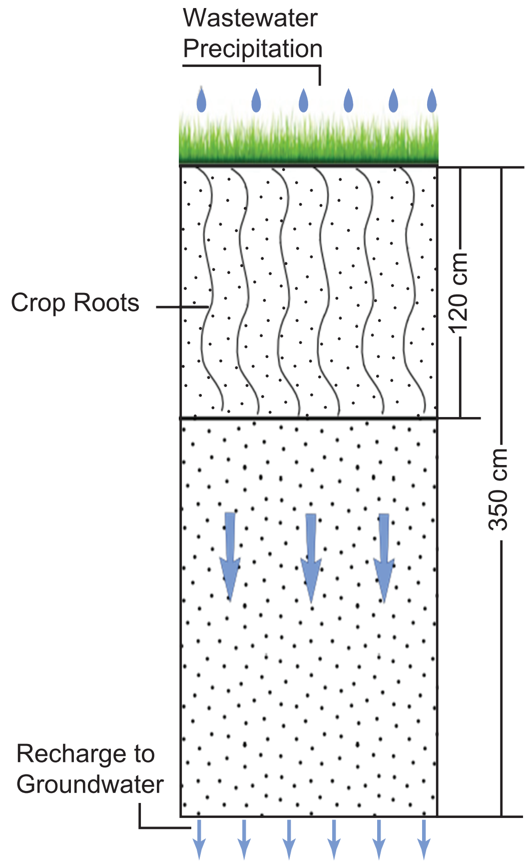

Figure 15. Illustration of water and nitrate movement in the vadose zone.

With increasing DNF, NO_3^- leaching groundwater was significantly decreased, while the plant uptake of NO_3^- was slightly reduced. Comparing case 3 to case 4, although the total mass of NO_3^- applied at the top soil was the same, the amount of leaching NO_3^- increased from 70.1 percent to 81.5 percent in the bottom layer. That is partially because the high volume of wastewater was beyond the plant water uptake capability, leaving a large amount of wastewater with lower concentration available to leach into the groundwater.

Simulated minimum discharge to groundwater generally occurs during the summer months when there is maximum water uptake by plants and maximum evaporation (Fig. 16). Exceptions occur in August when precipitation is highest. Differences in simulated NO_3^- concentrations (Fig. 17) between the top and bottom layers reflect the cumulative and relative impacts of NO_3^- loading, precipitation and infiltration, DNF and plant uptake. For example, lower NO_3^- concentrations

Table 5. Parameters used in the Hydrus-1D simulation cases.

Case	DNF Constant (d^{-1})	Spray Rate (in/yr)	Effluent Concentration (mg/L)	Recharge Rate (in/yr)	Average Concentration N Recharge to GW (mg/L)	DNF (%)	Plant Uptake (%)	Leaching Rate (%)
1	0	27.2	48	53.9	19.9	0	16.3	83.7
2	0.025	27.2	48	53.9	7.00	60.7	10.4	28.9
3	0.004	27.2	48	53.9	16.7	14.8	15.1	70.1
4	0.004	54.4	24	84.6	12.8	11.4	7.1	81.5

in the top layer during August result from dilution due to the greater volume of precipitation and maximum rates of plant uptake. The highest NO_3^- concentration in the bottom layer occurs in February when NO_3^- that was not removed by plants during the previous growing season has leached to the bottom layer. The average NO_3^- concentration and recharge rate in the bottom layer from cases 3 and 4 were used as inputs to the groundwater flow and contaminant transport model.

The impacts of N-leaching wastewater on groundwater quality were simulated for a 100-year period by comparing

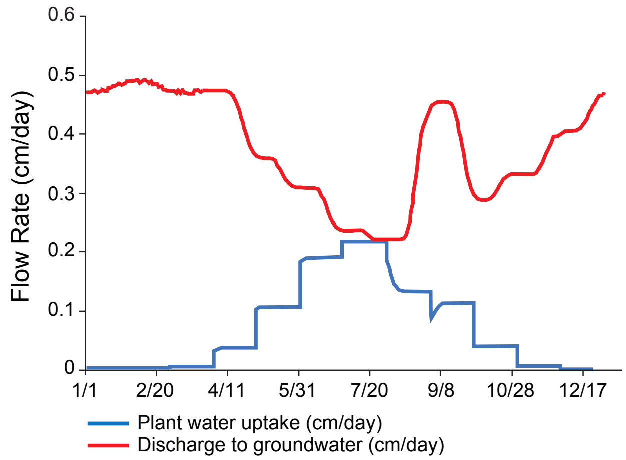

Figure 16. Simulated water uptake by plants and discharge to groundwater during one seasonal year (case 3).

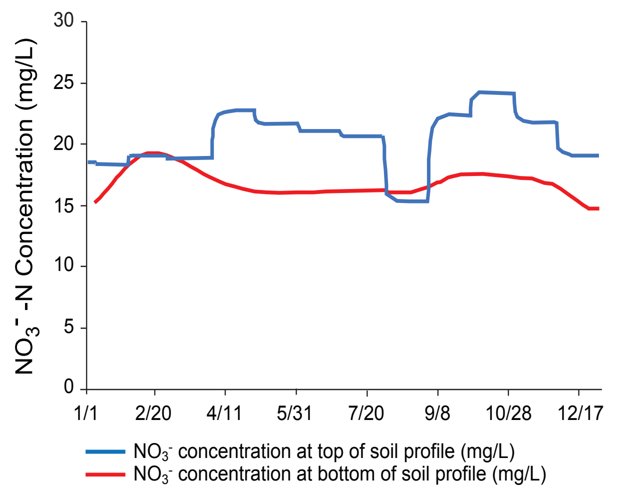

Figure 17. Simulated nitrate concentrations at the top and bottom of soil profile (case 3).

the results of two different wastewater irrigation practices (cases 3 and 4, Table 5) with a base case, no wastewater irrigation scenario. Simulation results from the vadose-zone NO_3^- transport model described above were used as inputs to the saturated flow and transport model. Simulated water-table elevations within spray fields (Fig. 18) increased between 1.5 m (case 3) and 2.5 m (case 4), causing a change in groundwater flow patterns around the facilities. Differences in NO_3^- concentrations between cases 3 and 4 and the base case (Fig. 19) show that changes in maximum NO_3^- -N concentration do not exceed 10 mg/L. However, a comparison of results from the high volume, low concentration practice (case 4) results in a larger volume of contaminated water, but a smaller volume of highly contaminated water (Figs. 19 and 20). The increased

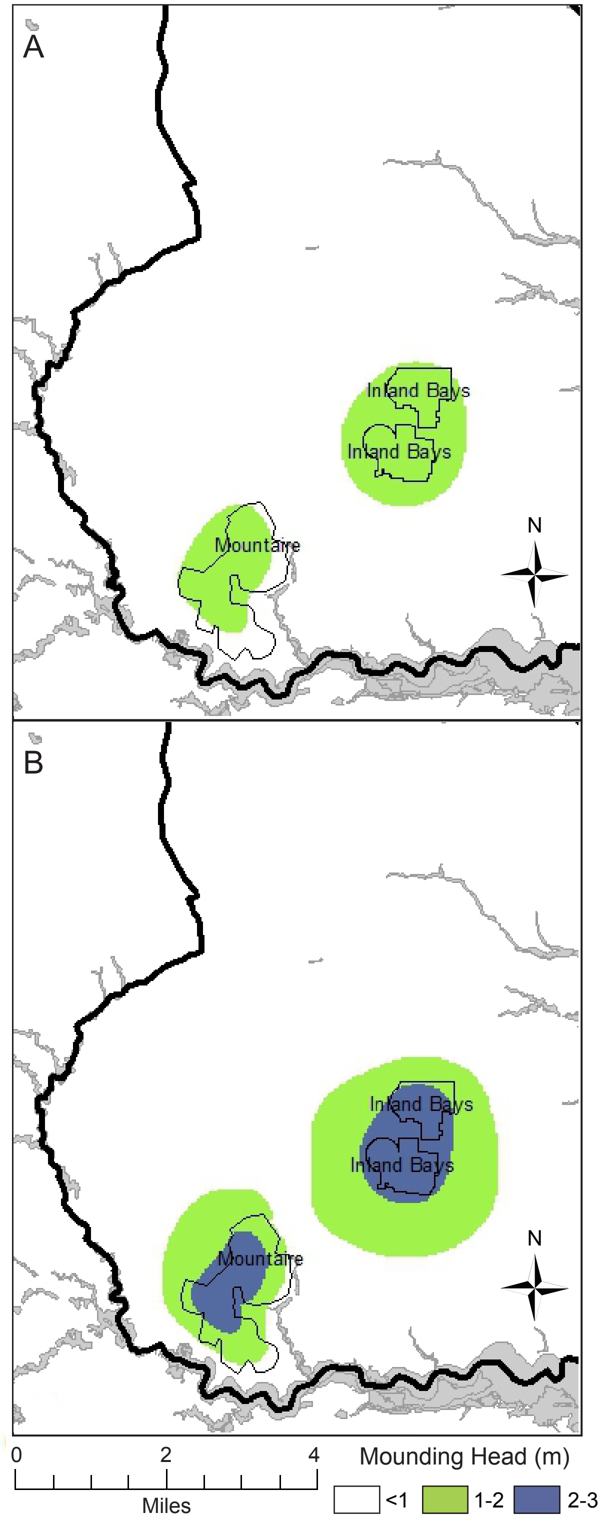

Figure 18. Water-table mounding beneath wastewater spray irrigation sites. A: case 3, irrigation rate is 27.2 in/yr and effluent nitrate concentration is 48 mg/L; B: case 4, irrigation rate is 54.4 in/yr and effluent nitrate concentration is 24 mg/L.

groundwater head gradient resulting from spray irrigation in both vertical and horizontal directions significantly facilitated the migration of NO_3^- to deeper aquifers.

SUMMARY AND CONCLUSIONS

We conducted a groundwater flow model around two major wastewater irrigation facilities located in Sussex County, Delaware. The model was constructed using Visual Modflow 2011 to simulate the groundwater flow and NO_3^- movement in the shallow aquifers. The application of wastewater irrigation was simulated using Hydrus-1D to incorporate both evaporation and plant-water uptake. The leached wastewater was then applied to the original flow and transport model to study the impacts on the groundwater conditions. Some conclusions based on modeling results are summarized below:

1. The water table closely mimics the land surface topography. The residence time of groundwater has a bimodal distribution, with times less than 10 years as the largest grouping. The spatial distribution of residence times along with a median groundwater residence time of 29 years indicate that several decades are required to flush the majority of the water now in the aquifer with new water.
2. The long-term operation of wastewater spray irrigation increases the water-table elevations. The amount of increase is dependent on different irrigation practices, and can significantly change flow patterns around the treatment facilities.
3. The DNF constant is a sensitive parameter for modeling NO_3^- movement and transformation in both the unsaturated and saturated zones. More field data such as DOC, solid phase organic carbon, concentrations of dissolved gasses, and types and concentrations of organisms should be collected to obtain an accurate estimate of the parameter.
4. Simply reducing the effluent concentration and increasing the irrigation rate may facilitate the migration of the NO_3^- plume to deeper aquifers and further down gradient, but the high concentration area of nitrate could be smaller. An optimal practice of wastewater irrigation can be achieved by adjusting the irrigation rate and effluent concentration.
5. The accuracy of model predictions is highly dependent on the accuracy and completeness of the input data to construct, calibrate, and validate the model. Several data deficiencies limit the predictive power of the model used in this study. Improved data for groundwater levels and streamflow would permit better model calibration. More complete records of water use would permit more realistic predictions of the hydrogeological effects of pumping wells. Better data on wastewater discharge rates and effluent would improve the accuracy of the model predictions.

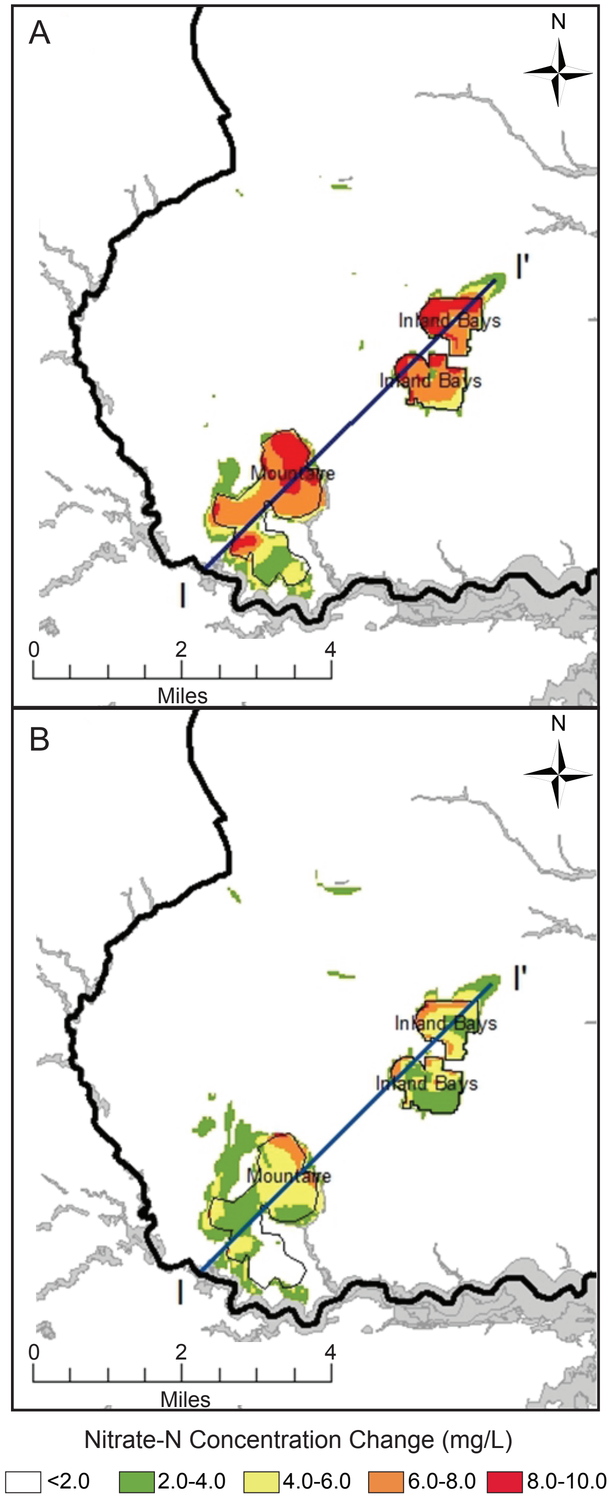

Figure 19. Changes in nitrate-N concentration after 100 years of spray irrigation of wastewater. A: case 3, irrigation rate is 27.2 in/yr and effluent nitrate-N concentration is 48 mg/L; B: case 4, irrigation rate is 54.4 inch/yr and effluent nitrate-N concentration is 24 mg/L.

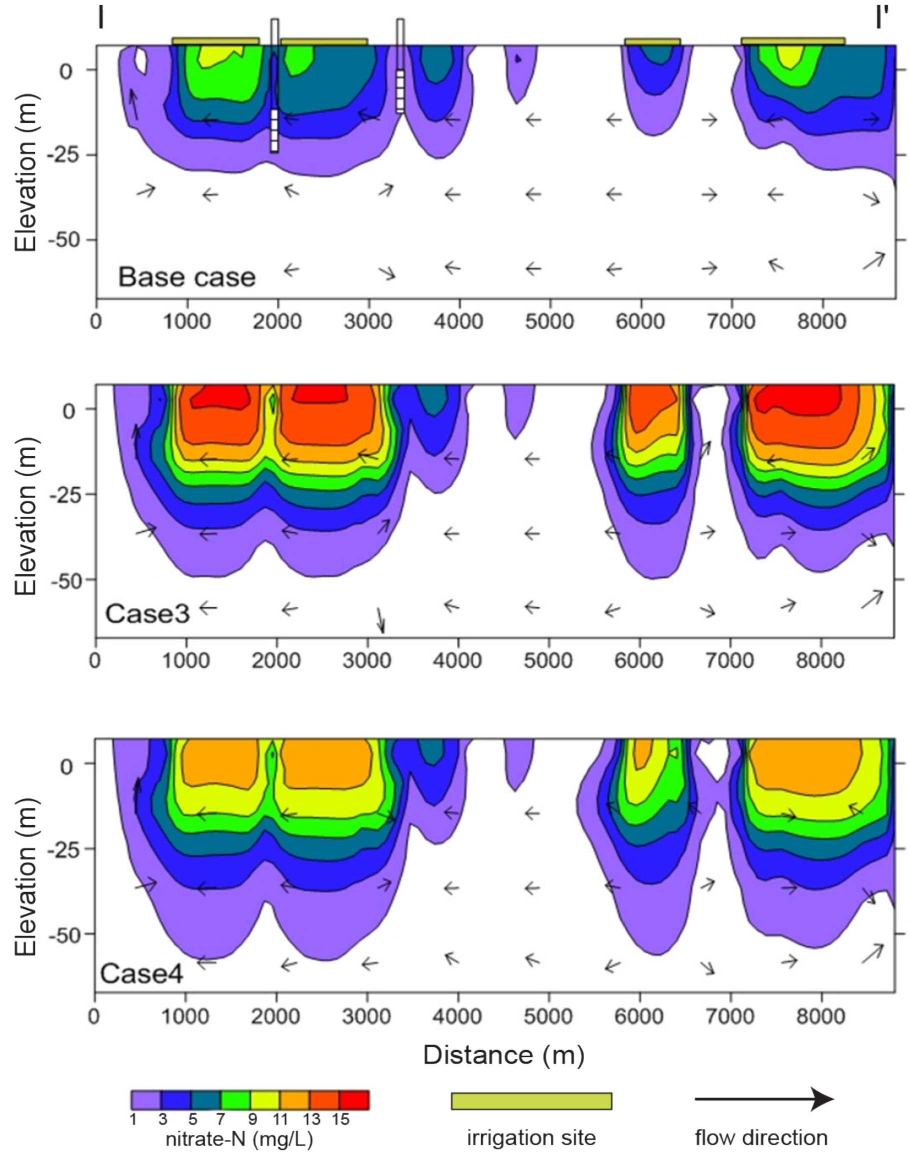

Figure 20. Simulated nitrate-N concentration for cross section I-I' shown in figure 19.

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Delaware Geological Survey
University of Delaware
Newark, DE 19716