

Estimation of Ground-Water and Nutrient Fluxes to the Neuse River Estuary, North Carolina

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## Abstract

A study was conducted between April 2004 and September 2005 to estimate ground-water and nutrient discharge to the Neuse River estuary in North Carolina. Largest ground-water fluxes were observed to occur generally within 20 m of the shoreline. Ground-water flux estimates based on seepage-meter measurements ranged from  $2.86 \times 10^8$  to  $4.33 \times 10^8 \text{ m}^3$  annually and are comparable to estimates made using radon, a simple water-budget method, and estimates derived by using Darcy's Law and previously published general aquifer characteristics of the area. The lower ground-water flux estimate (equal to about  $9 \text{ m}^3 \text{ s}^{-1}$ ), which assumed the narrowest ground-water discharge zone (20 m) of three zone widths selected for an area west of New Bern, North Carolina, most closely agrees with ground-water flux estimates made using radon ( $3\text{--}9 \text{ m}^3 \text{ s}^{-1}$ ) and Darcy's Law (about  $9 \text{ m}^3 \text{ s}^{-1}$ ). A ground-water flux of  $9 \text{ m}^3 \text{ s}^{-1}$  is about 40% of the surface-water flow to the Neuse River estuary between Streets Ferry and the mouth of the estuary and about 7% of the surface-water inflow from areas upstream. Estimates of annual nitrogen (333 tonnes) and phosphorus (66 tonnes) fluxes from ground water to the estuary, based on this analysis, are less than 6% of the nitrogen and phosphorus inputs derived from all sources (excluding oceanic inputs), and approximately 8% of the nitrogen and 17% of the phosphorus annual inputs from surface-water inflow to the Neuse River estuary assuming a mean annual precipitation of 1.27 m. We provide quantitative evidence, derived from three methods, that the contribution of water and nutrients from ground-water discharge to the NRE is relatively minor, particularly compared with upstream sources of water and nutrients and with bottom sediment sources of nutrients. Locally high ground-water discharges in the NRE do occur, however, and could help explain the occurrence of localized phytoplankton blooms, submerged aquatic vegetation, or fish kills.

## Introduction

Ground-water inflow to estuaries has been, until recently, one of the least studied and, consequently, one of the least understood sources of nutrients to estuaries. In the late 1970s, Lee (1977) measured ground-water seepage using a simple seepage meter in lakes and estuaries in various areas of the United States, including a coastal sound in North Carolina. In the early 1980s, Bokuniewicz (1980) estimated ground-water inflow to the Great South Bay in New York using a seepage meter of the same design used by Lee (1977). Later, Capone and Bautista (1985) used ground-water flux estimates by Bokuniewicz (1980) to conclude that more than 20% of the nitrogen contributed by surface runoff to Great South Bay was derived from ground water. After recognizing the potential importance of ground water in contributing nutrients to coastal ecosystems, others began investigating the phenomenon in a variety of coastal settings (Reay et al. 1992; Staver and Brinsfield 1996; Portnoy et al. 1998; Corbett et al. 1999). A recent collaborative paper between more than 20 scientific institutions worldwide focused on investigating the role of ground water in marine systems by multiple assessment methodologies (Burnett et al., 2006). This latter paper demonstrates a universal recognition by environmental scientists and managers of the need to quantify ground water in coastal areas for implementation of effective management strategies.

Although several of these previous studies have addressed ground-water discharge in estuaries of the mid-Atlantic and northeastern United States, few studies have been directed toward understanding the role of ground water in estuarine systems of the southeastern United States until recently. In addition, no studies in the published literature have attempted to measure ground water and associated nutrient inflows to an estuarine watershed sufficiently to quantify ground-water contribution in the context of a water and nutrient budget.

In this paper, we quantify groundwater discharge and associated nutrient delivery to the Neuse River Estuary (NRE), an estuary located in the Coastal Plain of the southeastern United

States that suffers from severe eutrophication impacts. We describe the results of a study conducted between April 2004 and September 2005 to measure ground-water discharge in the NRE by using seepage meters and piezometers and to integrate the results with those of other geophysical and geochemical techniques employed in parallel studies in order to better understand the importance of ground water in the NRE. Specific study objectives were to (1) estimate the discharge of ground water and nutrients to the NRE, (2) broadly characterize spatial variability of ground-water discharge and nutrient flux to the NRE, (3) compare estimates of nutrient flux from ground water with benthic nutrient flux estimates by other researchers, and (4) compare ground-water flux to the NRE measured by conventional seepage meters (Lee 1977; Bokuniewicz 1980) with ground-water fluxes estimated by other common methods, including a water budget developed previously for the Coastal Plain of North Carolina (Wilder et al. 1978) and a Darcy's Law estimate based on general hydrogeologic and environmental characteristics of the NRE. The relatively simple hydrologic principles that we demonstrate in our analysis here of the NRE, coupled with currently available field technologies, can be used effectively to estimate reasonable regional ground-water and nutrient fluxes in many other estuarine systems.

#### Site Description and Background

The Neuse River basin (Fig. 1) is one of four major river basins that drain into the Albemarle-Pamlico estuarine system in North Carolina. The Albemarle-Pamlico estuarine system is the second largest estuarine system in the United States and provides critical habitat for numerous fish and shellfish species along the mid-Atlantic region of the United States (Paerl et al. 1995). The Neuse River estuary (NRE) extends from approximately 10 km west and upstream from New Bern, North Carolina, to about 45 km east and downstream from New Bern, where it enters Pamlico Sound (Fig. 1). Effects of eutrophication have been observed and documented by many researchers since the late 1970s. Principal problems associated with eutrophic conditions in the NRE include the occurrence of nuisance algal blooms (Paerl 1983) and fish kills and toxic

dinoflagellate outbreaks during the 1990s (Bowen and Hieronymus 2000; Burkholder 2001). Research in the 1980s identified excess nitrogen and phosphorus loads into the estuary as a major cause of eutrophication (Tedder et al. 1980; Paerl 1983; Paerl 1990), although during the mid-1990s, nitrogen was recognized as the primary growth-limiting nutrient in the middle and lower parts of the estuary (Paerl et al. 1995; Luettich et al. 2000). More recently, researchers have reported that control of both nutrients will be necessary to effectively manage water quality in the entire NRE (Paerl et al. 2004; Burkholder et al. 2006).

Chemicals entering estuarine waters, including nutrients from a watershed, originate from either diffuse or point sources. Although previous studies indicate that both point and nonpoint pollution sources contribute nutrients, nonpoint sources compose the larger portion of nutrient inputs to the river basins draining into the Albemarle-Pamlico estuarine system (Dodd et al. 1992; McMahon and Woodside 1997). Excess nitrogen and phosphorus can cause eutrophication of streams and estuaries in the Albemarle-Pamlico drainage basin (Spruill et al. 1998), although nitrogen has been identified as a principal limiter of phytoplankton growth and primary production in many estuarine environments, including the NRE (Paerl et al. 1995). An estuary receives water containing nutrients and other chemicals from a variety of environmental sources, including precipitation falling directly onto the estuary, seawater moving periodically landward from the ocean as a result of lunar or wind-driven tides, freshwater draining from watersheds upstream into the estuary, water draining coastal plain tributary watersheds to the estuary, nutrients remobilized from bed sediments, and finally, ground water moving directly from shallow aquifers into the estuary from areas adjacent to the estuary, such as streambeds or along the estuarine shoreline.

In the most recent work conducted in the NRE (Luettich et al. 2000), which addressed strategies to improve water quality in the Neuse River and estuary, ground water was considered a possible source of nutrients to the estuary. The investigators concluded that ground water is

not a significant source of nutrients (Luettich et al. 2000) and that nutrients enter the water column primarily by diffusion from the bed of the estuary. This conclusion was reached based on information from nine 30-cm sediment cores collected from mid-channel in the estuary upstream from New Bern to near Minnesott Beach (Fig. 2) where pore water was extracted and salinity was measured. The lack of significant dilution of pore water with depth in all but one (near New Bern) of the nine cores indicated to the researchers that fresh ground water was not discharging in substantial quantities to the estuary and, therefore, was not a significant source of nutrients (Luettich et al. 2000).

Luettich et al. (2000) concluded that diffusion of nutrients from the bed of the estuary into the water column was the primary source of nitrogen to the NRE. Estimates of average nitrogen flux (as ammonium) from the bed of the estuary into the water column ranged from 0.9  $\text{mmol m}^{-2} \text{d}^{-1}$  to 5.4  $\text{mmol m}^{-2} \text{d}^{-1}$ , or an average of 3.2  $\text{mmol m}^{-2} \text{d}^{-1}$  based on six separate studies (Table 1). In one of the studies, Fisher et al. (1982) found that the highest rates of ammonium regeneration occurred in an estuary with fine-grained sediment and a high content of organic material, whereas ammonium flux was zero in an estuary with coarse-grained sands and a low content of organic material. Based on sediment-type distribution in the NRE (Fig. 2), about 50–70% of the area has fine-grained material (sediments having less than 40% sand) in the center of the estuary and sandy material (sediments with more than 40% sand) confined to nearshore areas. Luettich et al. (2000) reported that about 7,000 tonnes of nitrogen are contained in the upper 2 cm of sediment in the NRE. Thus, assuming that half of the estuary surface area of  $4.55 \times 10^8 \text{ m}^2$  (Buzzelli and Christian 2001) is covered with nitrogenous organic debris and serves as a source of nitrogen, between approximately 1,000 and 6,200 tonnes of nitrogen annually could be regenerated from sediment to the water column based on benthic sediment-flux estimates for the NRE (Table 1). This amount ranges from about 25% to more than 150% of

the total annual average riverine input of nitrogen, assuming a surface-water flux to the NRE of about 4,000 tonnes for an average year based on several published annual nitrogen loads — 4,400 tonnes reported for 1992–96 (North Carolina Department of Environment and Natural Resources 2001), 4,400 tonnes reported for 2000, (Spruill et al. 2005), or 1,900 tonnes for 1986 (a dry year) and 6,170 tonnes for 1987 (a wet year) estimated by Buzzelli and Christian (2001). Previously published phosphorus fluxes from the bed sediments were approximately 940 tonnes  $\text{yr}^{-1}$  (Rizzo and Christian, 1996; Table 1), 850 tonnes  $\text{yr}^{-1}$  (Fisher et al. 1982; Table 1), and almost 250 tonnes  $\text{yr}^{-1}$  (Matson et al. 1983; Table 1). These amounts are 220% and 65%, respectively, of the average riverine input of phosphorus based on a transported load of 385 tonnes for 2000 (Spruill et al. 2005).

Although Luettich et al. (2000) did not find that ground-water inflow was significant in the NRE, other researchers have measured ground-water inflow in other estuarine systems and determined that it ranged from less than  $0.01 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$  in the Chesapeake Bay (Reay et al. 1992) to more than  $0.23 \text{ m}^{-3} \text{ m}^{-2} \text{ d}^{-1}$  in Biscayne Bay, Florida (Halley et al. 2005; Table 2). However, none of the published research attempted to quantify ground-water inputs in the context of a total estuarine water budget. Ground-water discharge to estuaries typically is greatest along the shoreline and decreases with increasing distance from the shoreline (Bokuniewicz 1980). In saltwater environments, dense saline water restricts the movement of fresh ground water offshore because of salinity-related density gradients (Bear 1979), which form a saltwater-freshwater interface (Fig. 3). The conclusions of Luettich et al. (2000), which are consistent with what is known about ground-water flow into estuaries, were that ground-water flow was not significant in the center of the Neuse River estuary at nine study sites. Their investigation did not include the shoreline area, however, where ground water would be expected to discharge. Therefore, it is possible that their analysis underestimated the amount that ground

water contributes to the nutrient budget of the estuary and the relative importance of ground water to other nutrient sources in the NRE.

A variety of approaches beyond benthic-flux measurements and pore-water salinity profiling have been developed in recent years to study ground-water processes. These include integrated marine seismic and marine electrical resistivity surveys to establish submarine ground-water salinity distribution in a hydrogeologic framework (Krantz et al. 2004; Manheim et al. 2004); coring and sampling in offshore wells to analyze nutrients, stable isotopes, and ground-water age tracers, including tritium, tritium/helium, sulfur hexafluoride, and chlorofluorocarbons (Bratton et al. 2004); and natural ground-water tracer studies using radon gas emanation to evaluate ground-water location, rates of discharge, and time variability over tidal cycles (Crusius et al. 2005). These and many other useful techniques for measuring and understanding ground water in coastal regions, and incorporated in our study of the NRE, are presented in Burnett et al. (2006). Because of the large spatial and temporal variability of ground-water discharge in coastal areas (Burnett et al., 2006), the use of several techniques will typically be necessary to reasonably quantify ground-water discharge.

In order to understand the role that ground water plays in the ecology of any estuary, it is prerequisite to understand the regional hydrogeology. The principal shallow geologic formations of the NRE between Fort Barnwell, located upstream and west of Streets Ferry and defining the upper limit of the NRE, and the Cherry Point Marine Corps Air Station (section A-A' in Fig. 1) are shown in Fig. 4. The surficial aquifer, composed of Pleistocene and recent sediments (Winner and Coble 1996), covers most of the study area except where the sediments have been eroded by the river. These sediments, ranging from 10 to 30 m in thickness and thickening toward the east, provide the bulk of the ground water draining to the estuary. Hydraulic conductivity of the surficial sediments is estimated to be about  $10 \text{ m d}^{-1}$  (Winner and Coble



1996). Based on information presented in Giese et al. (1992), ground water in the surficial aquifer discharges to the estuary along flow paths generally perpendicular to the central axis of the NRE.

The Yorktown confining unit, which is at least 3–10 m thick in the vicinity of Cherry Point (Eimers et al. 1994), underlies the surficial aquifer and generally averages 8 m in thickness in the Coastal Plain (Giese et al. 1992). The Yorktown confining unit serves as the bed of the NRE near New Bern and eastward (Floyd 1969; Fig. 4) to the mouth of the estuary. Very little ground water likely discharges to the NRE from the Yorktown aquifer except where buried paleochannels, such as one located near Cherry Point (Daniel et al. 1996; Wrege and Jen 2004), locally allow water to move between aquifers through the breached confining units. The local absence and discontinuity of confining units near Cherry Point, which were later determined to be the result of incising by the paleochannel, were first noted by Lloyd and Daniel (1988). The Yorktown confining unit merges with the Pungo River confining unit near New Bern (Fig. 4). It does not appear from the information shown in Fig. 4 that the Pungo River aquifer is connected to the Neuse River directly. Most of the ground-water discharge to the NRE probably is derived primarily from the surficial aquifer. West of New Bern and near Streets Ferry, however, parts of the upper Castle Hayne aquifer, composed largely of limestone, along with the surficial aquifer may form part of the streambed of the upper NRE. The hydraulic conductivity of the limestone may be as high as  $60 \text{ m d}^{-1}$  (Winner and Coble 1996). Hydraulic conductivity of deposits underlying the NRE west of New Bern is conservatively estimated to be  $20 \text{ m d}^{-1}$ , a combination of hydraulic properties of the Castle Hayne and surficial aquifers.

Bottom sediments of the NRE are composed of material ranging from coarse sands along the shoreline and estuary west and upstream from New Bern to fine silts and clays that cover most of the axial basin of the open estuary beyond about 8 km downstream from New Bern

(Luettich et al. 2000). Fine-grained material with substantial organic carbon (Luettich et al. 2000) settles out where the NRE widens downstream from New Bern, and such sediments dominate the lithology of the bed material throughout the deeper parts (> 2 m) of the middle and lower estuary (Fig. 2). These fine organic sediments rest on the Yorktown or Pungo River confining units from New Bern eastward, forming very low permeability material in the central part of the estuary. Medium to coarse sands that compose the surficial aquifer have been eroded and removed by the main channel of the Neuse River (Wells 1989), but the sands left behind form the banks and grade into the bed along the shoreline, except along the south bank between Broad Creek, a tributary on the north bank where the Neuse widens, and Flanner Beach (Fig. 2).

### Field Methods

Measurements of ground-water and nutrient flux were made at 36 sites in the Neuse River estuary, North Carolina (Table 3; Fig. 2). Seven of these sites (NR34-40, Table 3; Fig. 2) were selected to evaluate the relations between distance from the shoreline and discharge and head difference between surface and ground water. Head measurements of ground water and surface water were made using a potentiometer (Winter et al. 1988), and ground-water discharge was measured using a conventional seepage meter (Lee 1977) at all sites between April 2004 and September 2005. Minipiezometers were driven to a depth ranging between 0.3 and 1.2 m beneath the streambed for head measurements. Seepage was measured by placing the meter (having an area of  $0.064 \text{ m}^2$ ) with an empty collapsed plastic collection bag on the bottom of the estuary within about 2 m from the minipiezometer and leaving the meter in place for 8 to 1,330 minutes during the April and June 2004 sampling periods. Meters penetrated the bottom sediments about 0.18 m with a head space of approximately 6-12 mm. During September 2005, seepage meters were left in place 20–30 minutes before recording seepage volumes. Seepage was measured along the shoreline at sites where the surface-water depth was less than about 1 m. The

volume of water that seeped into the collection bag during the sampling period was then measured, using a 100-mL or 500-mL graduated cylinder, and recorded. Where surface-water depths were greater than 1 m, only head measurements were made. Where head differences were not detectable, seepage was assumed to be minimal and assigned a value of  $< 0.01$ . Unit ground-water fluxes presented in Table 3 are derived from one to three measurements made at each site. Head measurements, recorded in triplicate, were considered accurate to within  $\pm 9$  mm; the median values are given in Table 3. Surface-water specific conductance and temperature recorded at each site are presented in Table 3.

Water-quality information was obtained from 15 of the 36 sites to estimate nutrient flux to the estuary from ground-water discharge (Table 4). Water samples were collected from the minipiezometer placed between 0.3 and 1.2 m beneath the bed of the estuary to characterize the water quality of ground water discharging to the estuary; a peristaltic pump was used to collect water from the minipiezometers at each site. Nutrient samples were filtered through a 0.45-micron filter into sample bottles, chilled, and sent to the U.S. Geological Survey National Water Quality Laboratory in Denver, Colorado, for analysis. Laboratory analysis for Ortho-P,  $\text{NH}_4\text{-N}$ , and  $\text{NO}_3\text{+NO}_2\text{-N}$  were determined using colorimetry or automated segmented-flow spectrophotometry as described in Fishman (1993). Total N determined using alkaline persulfate extraction as described in Patton and Kryskalla (2003). Analytical methods used and the associated reporting levels for each nutrient are given in Table 4. Median concentrations of dissolved N and dissolved P were multiplied by areal ground-water fluxes for both reaches east and west of New Bern to estimate total ground-water flux to the estuary. Nutrient fluxes for each sampling site were derived using unit ground-water fluxes and concentrations measured at each of the 15 sites (Table 4). Nutrient fluxes in milligrams  $\text{m}^{-2} \text{d}^{-1}$  were computed by multiplying unit ground-water flux ( $\text{m}^3 \text{m}^{-2} \text{d}^{-1}$ ) times concentration in  $\text{mg L}^{-1}$  times 1,000 (conversion from  $\text{m}^3$  to L). Phosphorus in millimoles  $\text{m}^{-2} \text{d}^{-1}$  was computed by dividing flux in  $\text{mg m}^{-2} \text{d}^{-1}$  by the

atomic weight of phosphorus (30 g). Nitrogen in millimoles  $\text{m}^{-2} \text{d}^{-1}$  was computed by total inorganic N (equivalent to  $\text{NH}_4\text{-N}$ ) divided by the atomic weight of nitrogen (14 g).

## Results and Discussion

### Annual Ground-Water Flux to the Neuse River Estuary using Seepage-Meter Measurements

A major factor that is critical to reasonably estimating annual ground-water flux to an estuary using measured unit fluxes is the width of the ground-water outflow face or discharge zone (the area between the shoreline and offshore; Fig. 3). Fresh ground water discharging to a saltwater estuary is restricted by heavier saline water that acts as a density barrier, creating an interface between fresh ground water and saltwater in the estuary and preventing discharge toward the center of the estuary (Fig. 3). Fresh ground-water discharge is restricted to a narrow zone next to the shoreline when the estuary contains more saline water (Fig. 3A). When the estuary contains less saline water and no or low density differences are present (Fig. 3B), fresh ground water can discharge in a wider zone to areas further offshore. Based on data from previous investigations, the discharge zones range from 10 to 1,000 m (Table 2), with decreasing flux offshore (Bokuniewicz (1980); Reay et al. (1992); Staver and Brinsfield (1996). In order to estimate the discharge area to the NRE, theoretical widths based on the Dupuit-Ghyben-Herzberg model (Fetter 2001) and aquifer and salinity characteristics of the NRE coupled with actual measurements and spatial patterns of ground-water flux made as part of our investigation were used as discussed below.

Equations derived by Glover (1964) from the Dupuit-Ghyben-Herzberg model of one-dimensional flow in unconfined coastal aquifers (Fetter 2001) also can be used to estimate discharge zone width in the saline part of the aquifer (the part of the aquifer that contains saltwater from the ocean or estuary; Fig. 3). The width of the discharge zone depends on the variables shown in the following equation:

$$X_0 = -Gq'/2K \text{ (from Fetter 2001),} \quad (1)$$

where

$X_0$  = the distance of the freshwater-saltwater interface from the shoreline;

$G = \rho_w/(\rho_s - \rho_w)$ , and  $\rho_w$  is the density of freshwater and  $\rho_s$  is the density of saline water;

$q'$  = discharge from the aquifer at the coastline per unit length of shoreline ( $L^3 T^{-1}$ )  $L^{-1}$  (Fetter 2001); and

$K$  = hydraulic conductivity of the aquifer in  $L T^{-1}$ .

The variables in eq. 1 must be known or estimated to calculate an estimate of the width of the discharge zone from the shoreline. Before the discharge per unit of shoreline length ( $q'$ ) can be computed, the dimensions of the aquifer (length and width) must be known in order to apply the Dupuit-Ghyben-Herzberg model. The first variable,  $q'$ , was computed by approximating the average width of the aquifer along the shoreline and assuming 20% of the average annual precipitation for the Coastal Plain estimated by Wilder et al. (1978) to recharge the shallow aquifer. The total precipitation recorded at New Bern for 2004 and 2005 during this investigation was 1.24 and 1.35 m (Southeast Regional Climate Center 2006), with a mean of 1.29 m, which resulted in aquifer recharge of about 0.26 m. The estimated width of the aquifer was obtained by dividing the total land area by the total length of shoreline:

$$W_a = WS_{\text{area}} / S,$$

(2)

where

$W_a$  = average width (m) of aquifer along the shoreline (distance from ground-water divide to shoreline);

$WS_{area}$  = total area of NRE watershed (light-shaded area, Fig. 2) minus the estuary water-surface area (dark-shaded area, Fig. 2) in  $m^2 = 1.42 \times 10^9$ ; and

$S$  = total shoreline length (m) =  $8.384 \times 10^5$ .

The variable  $G$  in eq. 1 is the ratio of freshwater density to the difference between saltwater density and freshwater density (Fetter 2001). The most saline water is in the lower part of the NRE at the mouth nearest Pamlico Sound (Fig. 1), where the salinity can be as high as 22 parts per thousand (ppt; Harned and Davenport 1990; Luettich et al. 2000). The estimated mean salinity is about 10 ppt (density =  $1.0046 \text{ g/cm}^3$ ). Water temperature during the investigation ranged from  $18.2 \text{ }^\circ\text{C}$  during April 2004 to  $30.7 \text{ }^\circ\text{C}$  during June 2004 (Table 3). Thus, using eq. 1 and the density of water at  $25 \text{ }^\circ\text{C}$  with mean salinity of 10 ppt, mean  $K$  of  $10 \text{ m d}^{-1}$  for the surficial aquifer, and annual discharge along the shoreline of  $1.21 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}$ , the discharge zone width ( $X_0$ ) in the area east and downstream from New Bern is

$$\begin{aligned} X_0 &= -Gq/2K \\ &= -(0.9972 \text{ g cm}^{-3} / (1.0046 \text{ g cm}^{-3} - 0.9972 \text{ g cm}^{-3}) * 1.21 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}) / (2 * 10 \text{ m d}^{-1}) \\ &= -(135 * 1.21 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}) / (20 \text{ m d}^{-1}) \\ &= -8.16 \text{ m (or about } -10 \text{ m as a general approximation in the middle to lower NRE)} \end{aligned}$$

Eq. 1 is less helpful in determining the width of the discharge zone in the freshwater part of the NRE west and upstream from New Bern, however. In this part of the estuary where

average salinity is 0.5 ppt or less, the width of the discharge zone could theoretically be 150 m or more, a value larger than previously reported in most studies of estuaries. This result is demonstrated by applying eq. 1 and assuming the salinity of the estuary is 0.5 ppt (density = 0.9974) at 25 °C with  $K = 20 \text{ m d}^{-1}$  and a discharge of  $1.21 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}$ :

$$\begin{aligned}
 X_0 &= -Gq'/2K \\
 &= -(0.9972 \text{ g cm}^{-3} / (0.9974 \text{ g cm}^{-3} - 0.9972 \text{ g cm}^{-3}) * 1.21 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}) / (2 * 20 \text{ m d}^{-1}) \\
 &= -(4,986 * 1.21 \text{ m}^2 \text{ d}^{-1}) / (2 * 20 \text{ m d}^{-1}) \\
 &= -151 \text{ m}
 \end{aligned}$$

Thus, depending on relative densities of freshwater and saltwater at particular locations and times in the estuary, the width of the discharge zone may vary and decreases as the estuarine water becomes more saline.

Although the width of the discharge zone in the middle and lower estuary is constrained by relatively high salinity, the width of the discharge zone in the freshwater part is not and therefore is more dependent on the hydraulic characteristics of the aquifer and the recharge to the aquifer. Previous estuary studies did not appear to include freshwater parts of the estuaries, and discharge zones generally were reported to be 50 m or less (Table 2), with the exception of the carbonate aquifer system in Florida studied by Halley et al. (2005). Reay et al. (1992) reported the widest zone in Cherrystone Inlet in the Chesapeake Bay estuarine system where discharge decreased with increasing distance from shore (up to 50 m). Even though the discharge zone theoretically could be wider, much of the upper NRE channel is only about 300–400 m wide; thus, the possible maximum width for the discharge zone is approximately 150–200 m (assuming

the estuary receives ground-water discharge from both banks, which leaves about half the total channel width for the discharge zone).

Measurements made during this investigation were used to estimate the discharge zone in the estuary by evaluating the measured unit ground-water flux with distance from the shoreline. Data collected during this study (Fig. 5; Table 3) indicate that most ground-water flux occurred within 20 m of the shoreline, and the highest fluxes were generally within 10 m of the shore, particularly east of New Bern. Measured fluxes were more uniform with distance from shore west of New Bern (NR35–37; Fig. 5) with  $0.04 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$  being the lowest flux recorded in this reach (Table 2). At three transects measured west of New Bern, fluxes appeared to be somewhat higher and generally more uniform between the shoreline and 10 m, with no apparent decrease with distance from the shore (Fig. 5; Table 3). Water depth was too deep to place seepage meters beyond 10 m at these three transects, so measurements could not confirm whether any decrease in flux with increasing distance from shore occurred in this reach. Unit ground-water fluxes measured using seepage meters in the NRE ranged from near zero to  $0.43 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$ , with a median flux for the NRE of  $0.07 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$  and a mean flux of  $0.10 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$  (Table 3; Fig. 2). These measurements generally fall within the range of previous ground-water flux measurements reported for other estuaries along the east coast of the United States (Table 2). Ground-water fluxes in the NRE were similar both east and west of New Bern (medians of  $0.07$  and  $0.08 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$ , respectively), indicating that ground-water discharge from the bed of the estuary within the discharge zone could be considered to be somewhat uniform.

Ground-water flux to the NRE was computed by using a constant width of 10 m east of New Bern, three estimated discharge-zone widths west of New Bern, and a typical discharge of  $0.09 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$  (a value between the mean and median given in Table 3) for both reaches. The



similar ground-water fluxes derived from measurements imply similar hydrogeologic and lithologic characteristics of the NRE, both upstream (west) and downstream (east) from New Bern. However, based on available hydrogeologic information (Floyd (1969); Winner and Coble (1996)), the two reaches are different and these differences are addressed when applying Darcy's Law in the next section. This apparent discrepancy suggests that more hydrogeologic study may be necessary in this section of the NRE. Three discharge-zone widths were assumed (Table 5): (1) a minimum discharge-zone width of 20 m; (2) an estimate of 50 m, the limit past which most other researchers did not find detectable seepage in estuarine environments; and (3) 160 m, the approximate maximum width calculated using the Dupuit-Ghyben-Herzberg model (Fetter 2001) that would be possible in much of the upper NRE, as discussed above. Thus, three different scenarios with increasing widths (20, 50, and 160 m; Table 5) were used to provide ground-water discharge estimates for this reach to evaluate which estimated seepage rates of the three appeared to best agree with ground-water discharge estimates established by other methods or with data reported in the literature. The total annual ground-water discharge to the entire estuary during 2004–05, with an average rainfall of 1.29 m, is estimated to range between  $2.86 \times 10^8 \text{ m}^3$  or  $9.07 \text{ m}^3 \text{ s}^{-1}$  and  $4.33 \times 10^8 \text{ m}^3$  or  $13.73 \text{ m}^3 \text{ s}^{-1}$  (Table 5).

The width of the discharge zone between New Bern and the mouth was assumed to be approximately 10 m, because the computed zone is limited by generally high salinity ( $> 10$  ppt), thus restricting the zone to 10 m or less. In general, most measurable unit ground-water fluxes occurred within 10 m of shore in this reach (based on fluxes shown in Table 3; Fig. 5); thus, 10 m seems to be a reasonable approximation for the middle and lower parts of the NRE. Also, in this part of the NRE, discharge of ground water throughout the middle of the estuary was assumed to be near zero based on previous research, indicating no significant ground-water discharge over most of the central portion of the estuary (Luettich et al. 2000) and on marine

resistivity surveys conducted in parallel with this study (Fig. 6). From these surveys (Cross et al. 2006), fresh (high resistivity) ground-water discharge areas are restricted to areas parallel to the shoreline downstream from New Bern, with most fresh water contained in the confined Yorktown aquifer beneath the NRE bed, as indicated in the marine resistivity profile (Fig. 6), as well as in several other profiles shown in Cross et al. (2006). Measured fluxes that occurred near a buried paleochannel near Cherry Point, North Carolina (Daniel et al. 1996; Wrege and Jen 2004), were typical of fluxes observed at other sites sampled in the NRE. Highest observed unit ground-water fluxes ( $> 0.1 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$ ) were measured in the area between New Bern and Minnesott Beach (Fig. 2). This approach assumes that all ground-water discharge originates (is recharged) within the NRE basin and that the principal aquifer discharging ground water to the NRE (the surficial aquifer) is unconfined. Water from underlying or confined systems, if there is a breach in the confining unit, for example, could contribute water that is not accounted for in this analysis.

#### Annual Ground-Water Flux using a Standard Conceptual Water Budget and Darcy's Law

The ground-water flux estimates derived from our measurements also were compared with estimates derived by using a standard conceptual water budget for the Coastal Plain and general aquifer characteristics using Darcy's Law. A water budget method using 0.26 m of recharge to the surficial aquifer (about 20% of the average rainfall based on information in Wilder et al. (1978)) was applied to the land area ( $1.42 \times 10^9 \text{ m}^2$ ; Fig. 2) of the watershed surrounding the NRE to estimate ground-water flow to the NRE ( $3.69 \times 10^8 \text{ m}^3$ ; Table 5) equal to a daily flux of about  $12 \text{ m}^3 \text{ s}^{-1}$ .

Darcy's Law (eq. 3) was also used to estimate ground-water discharge using typical aquifer characteristics for the area:

$$Q = KIA, \quad (3)$$

where:

$Q$  = discharge, in  $m^3$ ;

$K$  = hydraulic conductivity of the aquifer, in  $m\ d^{-1}$ ;

$I$  = change in hydraulic head over distance, in  $m/m$  (dimensionless), based on average gradient obtained from previous studies (Giese et al. 1992; Eimers et al. 1994); and

$A$  = area of the aquifer face discharging to the estuary, in  $m^2$ , based on average length and thickness of the aquifer adjacent to the estuary.

Basic conceptual information for the hydrogeology of the NRE was provided by Floyd (1969), Giese et al. (1992), Eimers et al. (1994), and Winner and Coble (1996). A hydraulic conductivity of  $10\ m\ d^{-1}$  (the median for the entire Coastal Plain in Winner and Coble (1996) and an estimated average aquifer thickness of 18 m (based on the midpoint of 10–21 m range in thickness at Cherry Point reported by Eimers et al. (1994) and assuming some thickening of deposits eastward (Giese et al. (1992), where no well data are available) were assumed for the surficial aquifer east of New Bern. The average gradient for the surficial aquifer was derived from Giese et al. (1992) and was estimated to be about 8 m/1,500 m or 0.005 (Table 5). For the area west of New Bern,  $20\ m\ d^{-1}$  was used as the average hydraulic conductivity for the combined surficial and Castle Hayne aquifer estimated from the average hydraulic conductivity given by Winner and Coble (1996) for the Castle Hayne aquifer. The thickness of upper

weathered portion of the Castle Hayne west of New Bern was estimated to be 20 m (Floyd, 1969). For our analysis, the surficial aquifer was assumed to be in direct contact with the Castle Hayne aquifer and also to behave as an unconfined aquifer in this reach. We assumed that the average gradient for the combined surficial and Castle Hayne aquifer was influenced by surficial aquifer characteristics and was slightly lower (0.004; Table 5) due to a lower gradient in the Castle Hayne aquifer (estimated as 0.002 from information published in Giese et al. 1992). Application of Darcy's Law using average hydraulic characteristics of aquifers in both reaches of the NRE (Table 5) yielded an estimate of  $2.84 \times 10^8 \text{ m}^3 \text{ yr}^{-1}$  or about  $9 \text{ m}^3 \text{ s}^{-1}$  for the entire NRE.

#### Annual Nutrient Fluxes

Concentrations of nitrogen, which occurred primarily as ammonium in samples collected from the minipiezometers that were screened between 0.3 and 1.2 m beneath the bed of the estuary, ranged from 0.04 to 14.5  $\text{mg L}^{-1}$ , with a median concentration of 1.05  $\text{mg L}^{-1}$  (Table 4). Using the median nitrogen concentration as the best estimate of ground-water concentration and assuming a 10-m discharge zone between New Bern and the mouth of the estuary and a 20-m discharge zone between New Bern and Streets Ferry (Table 5) as the best approximation of annual flux to the NRE (as explained in the following section), we estimated the median flux of nitrogen to the estuary to be about 333 tonnes  $\text{yr}^{-1}$ , or about 4% of the annual nitrogen flux from all sources and about 8% of upstream surface-water nitrogen flux to the estuary for a typical year (Table 6). Concentrations of dissolved phosphorus, primarily from orthophosphate, ranged from 0.01 to 4.22  $\text{mg L}^{-1}$ , with a median of 0.21  $\text{mg L}^{-1}$  (Table 4). The median flux to the estuary,

using the same assumption as above, was about 66 tonnes of phosphorus  $\text{yr}^{-1}$  or about 17% delivered by surface-water inflow and about 5% of all sources of phosphorus to the estuary (Table 6). The largest nitrogen fluxes estimated for this study ( $> 18 \text{ mmol m}^{-2} \text{ d}^{-1}$ ; Table 4) occurred in areas along the north bank near Minnesott Beach and on both the north and south banks between New Bern and near Broad Creek (Fig. 2; Table 4). Highest estimated phosphorus fluxes ( $> 0.5 \text{ mmol m}^{-2} \text{ d}^{-1}$ ) occurred in ground water collected along the north bank from NR4 near Minnesott Beach and from the upper portion of the NRE near Streets Ferry (Fig. 2; Table 4).

#### Importance of Ground-Water Discharge in the Neuse River Estuary

The importance of ground water and its role in the functioning of an estuary, particularly as it relates to primary productivity, has not been extensively studied. Over the last 30 years, however, interest has increased in determining the importance of ground water in estuarine systems. Reference has been made to Luetlich and others (2000) regarding their conclusion that ground water was an insignificant source of nutrients to the NRE. Buzzelli and Christian (2001), estimated that ground water contributed about 5% of freshwater inflow of the NRE, also implying that the contribution of ground water was small with respect to nutrients. Neither of these studies included the shoreline area of the estuary where ground-water discharge is most likely. The purpose of our study was to provide several documented methods to derive ground-water flux estimates to the NRE and, subsequently, using basic water-quality concentration data, to provide estimates of nutrient flux from ground water. The following discussion assesses the validity of our ground-water and nutrient flux estimates and, ultimately, the importance of ground water in affecting primary productivity and ecology of the NRE.

The question, then, is how reasonable are our estimates of ground-water flux to the NRE? Our measurements of ground-water discharge agree fairly well with gross estimates of discharge derived from both Darcy's Law and general knowledge of hydrology in the Coastal Plain of North Carolina. Ground-water seepage measurements coupled with model estimates of discharge zone width, produced ground-water discharge estimates in the NRE that ranged from 7 to 11% of the annual average freshwater riverine inflow to the estuary ( $3.9 \times 10^9 \text{ m}^3$  (Robbins and Bales 1995; Buzzelli and Christian 2001)).

How does the use of seepage meters, which were not pre-filled with water, affect the estimates? Shaw and Prepas (1989) found that the use of empty bags deployed for times of less than 45 minutes, as was done in our study, can produce elevated flux values compared to bags which are pre-filled with 1 L of water. On the other hand Belanger and Montgomery (1992) found in tank experiments that seepage meter rates, derived from using 1 L pre-filled bags, were about 77% of the actual seepage and 57% of the calculated rate based on interstitial composition of tank sediments, and thus yielded seepage rates lower than actual seepage measured in the experimental tanks. These results show that both methods can bias seepage rates, one positively and one negatively, suggesting that the use of unfilled bags for short time periods of deployment may not severely overestimate seepage rates. The median and mean seepage rates ( $0.07$  and  $0.1 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$ ) of all measurements made during our study of the NRE (table 3) represent seepage within about 10-50 m of the shoreline and were within the range of seepage rates reported for other studies shown in table 2. Although the variability of ground-water flux in the NRE is large ( $<0.01$ - $0.43 \text{ m}^3 \text{ m}^{-2} \text{ d}^{-1}$ ), it is similar to variability observed in other estuaries shown in table 2 and indicates that more detailed mapping of discharge zones in the NRE is warranted for more precise flux estimation at local scales.

Probably one of the best practical ways to assess the validity of the estimates presented in this paper is to compare whether the amount of ground water discharged to the estuary agrees with a typical ratio of ground water to total streamflow, which is commonly expressed as a base-flow index (BFI)—the BFI reported for streams in the Coastal Plain ranges from 40% to more than 70% (Harned and Davenport 1990; McMahon and Lloyd 1995). The total discharge from the Neuse River to Pamlico Sound has been estimated to be  $4.73 \times 10^9 \text{ m}^3$  (Giese et al. 1985; Kim 1990) for an average year. Inflow to the NRE during years having average precipitation (about 1.27 m) has been estimated to be  $3.9 \times 10^9 \text{ m}^3$  (Robbins and Bales 1995). The difference in streamflows between upstream and downstream is about  $7.3 \times 10^8 \text{ m}^3$  and is, therefore, the approximate gain in flow between Streets Ferry and the mouth of the NRE during an average year. Based on these estimates, the base flow or ground-water contribution derived from measurements made for this study (as well as Darcy's Law and the water budget method) is between  $2.86 \times 10^8 \text{ m}^3$  and  $4.33 \times 10^8 \text{ m}^3$  annually and produces a BFI between about 40 and 60% of the surface-water inflow to the NRE between Streets Ferry and the mouth, which is in agreement with previously reported BFI for coastal streams in the Albemarle-Pamlico drainage basin. Estimates which were much below 40% would imply lower actual recharge or losses to either evapotranspiration or leakage to confined aquifers that far exceed rates that are currently considered reasonable. Estimates greater than 70% would suggest an additional source, such as significant discharge from a confined aquifer or from an aquifer where the hydraulic characteristics were estimated incorrectly too low.

Our ground-water flux estimates also can be evaluated by comparing them with estimates derived using radon measurements. Using radon data from a parallel study (J. Crusius, U.S. Geological Survey, written communication, June 15, 2006), the estimated ground-water contribution to the area west of New Bern was  $1.9 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}$  of shoreline, somewhat higher

than the  $1.21 \text{ m}^3 \text{ d}^{-1} \text{ m}^{-1}$  for the entire estuary using a recharge of  $0.26 \text{ m yr}^{-1}$ , as shown in Table 5 for the water budget method. This estimate indicates that the discharge could be higher in the 16-km reach west of New Bern than in the reach to the east, possibly reflecting a difference in hydrogeology or some other factor. Based on available information, however, the estimate derived from measurements using a 20-m discharge zone width best agrees with estimates using radon ( $3\text{--}9 \text{ m}^3 \text{ s}^{-1}$  reported by J. Crusius (written communication, June 15, 2006) and estimates generated by using Darcy's Law (about  $9 \text{ m}^3 \text{ s}^{-1}$ ; Table 5)—an annual ground-water flux of  $2.86 \times 10^8 \text{ m}^3$  appears to be the most reasonable estimate based on our analysis. The estimated flux to the NRE of nearly  $12 \text{ m}^3 \text{ s}^{-1}$ , using the water budget method, is higher than the estimated flux using radon or Darcy's Law and suggests that the recharge to the surficial aquifer ( $0.26 \text{ m yr}^{-1}$ ) could be somewhat high for this part of the Coastal Plain. However, recent work by Coes et al. (2007) indicated that annual recharge rates measured using hydrograph separation in four areas of the Coastal Plain in 2004 ranged from 0.15 to 0.34 m, with three of the four sites 0.28 m or greater, providing further evidence that the recharge rate of 0.27 m is reasonable. Evapotranspiration or subsurface leakage may be somewhat greater than rates estimated by Wilder et al. (1978).

With respect to the importance of nutrients, ground-water flux appears to contribute less than 6% of both nitrogen and phosphorus from all sources directly to the estuary (Table 6). The estimated ground-water inflow, however, provides about 8% of the nitrogen and 17% of the phosphorus transported in surface water from upstream areas of the NRE. As indicated previously, the major potential source of nitrogen (7,000 tonnes) in the NRE is sediment (Luettich et al. 2000). Even though large quantities of nutrients are available to the water column from sediment, some question remains as to the availability of these nutrients to phytoplankton in the upper 0.6 m of the estuarine water column where most primary production occurs.



Although the relative contribution of nutrients in ground water appears minor in relation to the quantity potentially available from sediment, nutrients in ground water could be important because they are released into the water column through advection, as compared to simple molecular diffusion from the bed. This is particularly true when advective forces overcome the density gradients that may occur during summer stratification. This phenomenon can be seen in thermal infrared photographs of the Nauset Marsh near Eastham, Massachusetts, and published in Portnoy et al. (1998), in which cold low-salinity ground-water plumes can be seen moving into the warm estuarine water away from the shoreline toward the center of the estuary. Diffusion of nutrients from bottom sediments typically are trapped by density gradients and confined to the denser saline bottom waters, where typically lower primary production takes place during summertime stratified periods (Luettich et al. 2000) or after periodic rainfall events (Cloern 1991).

Even though the mass of nutrients derived from ground water seems relatively small—only 4 to 5% of the total nutrient pool at the scale of the entire NRE—ground water in some localized areas in the NRE can have high fluxes and concentrations (Fig. 2), which could affect the distribution of biota—both micro- and macro-fauna and flora. Although the median ground-water concentration of total inorganic nitrogen was  $1.05 \text{ mg L}^{-1}$  (derived almost entirely from ammonium), much higher concentrations and fluxes of ammonium were measured at some sites ( $7.6 \text{ mg L}^{-1}$  and  $23 \text{ mmol m}^{-2} \text{ d}^{-1}$  at NR4 and  $1.27 \text{ mg L}^{-1}$  and nearly  $40 \text{ mmol m}^{-2} \text{ d}^{-1}$  at NR34, Table 4; Fig. 2), which could affect primary production at times and places in the estuary where primary production is otherwise nitrogen limited. It also may be important that ground water provides about 8% of the nitrogen and almost 17% of the phosphorus surface-water load from upstream areas, which is forced through advection into the upper part of the water column where

nutrients are available to primary producers, even under conditions when the estuary normally is stratified, particularly during the summer months.

It should be noted that our nutrient flux estimates to the NRE were based on ground water collected from the piezometers and not from the seepage meters. Because of sediment-water interface reactions that can take place before ground water discharges into the estuary, nutrient concentrations in ground water collected from piezometers may be different from concentrations collected from seepage meters. Further research focused on this possibility would be beneficial in providing more definitive ground-water flux estimates.

Although the ground-water flux estimates from our study appear generally reasonable, more accurate and precise estimates of ground-water discharge in the NRE will require additional study. Some researchers (Reay et al. 1992; Staver and Brinsfield 1996) found decreasing head or discharge with distance from shoreline; decreasing discharge with increasing distance from shore was observed in our study, although a quantifiable relation between head and discharge was not, partly because of the difficulty in making accurate head measurements in the NRE, which frequently is windy and turbulent. Another reason for our inability to discern a correlation between discharge rate and head, or either of these variables with distance from shoreline, could be the highly heterogeneous bed-sediment material and locally variable hydraulic properties in the NRE. Variable hydraulic conductivity of the bed could obscure such relations, and more detailed work defining hydraulic properties will be necessary to determine if the expected relations are discernible in this system. Effects of tidal variations on discharge need to be evaluated, which would require continuous discharge and head monitoring. The lack of reach-specific hydrogeology, particularly in the area west of New Bern, is needed to better ascertain ground-water discharge to the estuary. Additional measurements downstream from Cherry Point to the mouth of the NRE also are needed to verify our estimates, particularly

because estimates of ground-water discharge in this study were assumed to represent downstream areas that were not sampled.

There are some notable differences related to findings from our study compared to findings from studies in other areas along the east coast of the United States. Nitrate was not detected ( $< 0.06 \text{ mg L}^{-1}$ ; Table 4) in any of the samples collected during this study, even though high nitrate concentrations ( $> 10 \text{ mg L}^{-1}$ ) were detected in studies conducted in the Chesapeake Bay (Staver and Brinsfield 1996) and Massachusetts (Portnoy et al. 1998) in the nearshore areas with decreasing concentrations offshore. This result also was found at other sites in the Chesapeake Bay by Speiran (1996), where water having oxygen and oxidized forms of nitrogen can move through clean sands with little nitrate-reducing effect of riparian buffers. Based on previous work in the Albemarle-Pamlico drainage basin, the surficial aquifer, which is composed largely of fine- to medium-grained sands and provides ground water to Coastal Plain streams and estuaries, including the NRE, contains water with substantial amounts of dissolved organic carbon, characteristic of ground water in the North Carolina Coastal Plain (Spruill et al. 1998; Tesoriero et al. 2004). Shallow ground water in this area does not have the high nitrate concentrations found in many other coastal areas of the east coast of the United States—low or absent nitrate in ground water from this area occurs because of generally reducing conditions, which are prerequisite for denitrification. Consequently, the primary form of biologically available nitrogen in ground water discharging to the NRE occurs primarily as ammonium, which may be used by phytoplankton (Paerl et al. 1995). High concentrations of ammonium were observed by Paerl et al. (1995) in the Neuse River at New Bern, which could be due to discharging ground water. Results from our study support the assessment of Paerl et al. (1995), as measurements made on the northeast bank just downstream from New Bern (NR34; Fig. 2) resulted in a flux of almost  $40 \text{ mmol m}^{-2} \text{ d}^{-1}$  of ammonium (Table 4). High concentrations of

ammonium ( $> 0.2 \text{ mg L}^{-1}$ ) have been observed in discharging ground water in other areas of the Coastal Plain in the Albemarle-Pamlico drainage basin (Spruill et al. 1998). It is interesting that the piezometer at site NR34 is located adjacent to an extensive wetland that is not apparently affected by human activities along the northeast bank. Although no nitrogen was detected as nitrate in ground-water samples collected during this study, it is possible that nitrogen as nitrate could occur locally in ground water in clean sands extending into the estuary next to nitrogen-source areas (for example, cultivated fields) but were not included in the relatively limited sampling effort for this study. Additional water samples from piezometers placed along transects perpendicular to the shoreline at various locations in the estuary, particularly in areas where agricultural activity occurs or residential or urban land is present and is known to be a likely source of nitrate, could be collected and evaluated to determine if nitrate occurs in ground water discharging to the NRE. Nevertheless, nitrogen occurring either as ammonium or nitrate can fuel primary productivity in a nitrogen-limited estuarine system. Similarly, locally high concentrations and fluxes of phosphorus in ground water also could fuel primary productivity in areas of the NRE that are often phosphorus limited, such as the upper part of the estuary (Paerl et al. 2004). Some of the highest phosphorus fluxes occurred in this area (Fig. 2).

Our study shows, based on quantitative information on ground-water flux distribution, that to accurately estimate ground-water flux into the NRE, sampling must be concentrated within 10-20 m of the shoreline, where most discharge occurs, or calculated fluxes will likely be biased low. Ground water does contribute measurable nutrients and water to the NRE, although similar to conclusions from previous work, ground water appears to be a minor source of nutrients and water to the NRE relative to watershed and benthic inputs. Results from our study support the view that nutrient management focused in areas upstream of the NRE will be most effective in protecting estuarine water quality.

There are, however, possible effects of large locally derived inputs of nitrogen or phosphorus on primary productivity and biotic distribution in the NRE that deserve further research. As pointed out by Johannes (1980), even though submarine ground-water discharge (SGD) may be generally much smaller than surface water discharge, SGD may still be of great ecological significance. Ground-water discharges have been linked to the occurrence of nuisance algal blooms including brown tide in Long Island (Laroche et al., 1997) and red tide in Korea (Lee and Kim, 2006). Stimulation of aquatic plant growths by inorganic nitrogen in submarine groundwater discharges have also been reported by Maier and Pregnall (1990) and LaPointe (1997). Table 1 shows that relatively large unit inputs of nitrogen and phosphorus where ground water discharges to the NRE compared to inputs of these elements through benthic release into the water column—unit fluxes of both nitrogen and phosphorus in ground water are almost twice that of benthic sources. This observation is particularly relevant in controlling mass loads into an estuary for mitigation of various phenomena considered to be problems—areas that may have algal blooms or fish kills could be a feature of localized conditions, including nutrient fluxes from discharging ground water, and not necessarily from overall loading to the system from upstream sources, as is assumed for total maximum daily load (TMDL) programs to control and improve water quality.

## Conclusions

Measurements, using seepage meters in conjunction with other theoretical and quantitative methods to estimate annual ground-water flux to the NRE, were compared with estimates derived from radon measurements, Darcy's Law and published hydraulic and aquifer characteristics, and a simple water budget method. Largest ground-water fluxes generally

occurred within 20 m of the shoreline. The lowest of the estimates derived from seepage meter measurements (equivalent to about  $9 \text{ m}^3 \text{ s}^{-1}$ ) which ranged from  $2.86 \times 10^8$  to  $4.33 \times 10^8 \text{ m}^3$  annually from measurements made in the NRE between April 2004 and September 2005, best approximates ground-water flux estimates derived from radon measurements ( $3\text{--}9 \text{ m}^3 \text{ s}^{-1}$ ) and flux estimates derived from Darcy's Law and aquifer characteristics based on values published in the literature (about  $9 \text{ m}^3 \text{ s}^{-1}$ ). This estimate is about 40% of surface-water inflow to the NRE between Streets Ferry and the mouth of the estuary and about 7% of the surface-water inflow from areas upstream from the NRE. All methods used indicate that ground water discharged to the estuary ranges from 40 to 60% of the total annual surface-water inflow to the reach between Streets Ferry and the mouth of the NRE. Estimates of nitrogen and phosphorus fluxes from ground water to the estuary, based on this work, are less than 6% of the total input of these elements derived from all sources, and between approximately 8% for nitrogen and 17% for phosphorus from surface-water inflow upstream from the NRE in an average precipitation year. We provide quantitative evidence that the contribution of water and nutrients from ground water discharge to the entire NRE is relatively minor, particularly compared with upstream sources of water and nutrients and with bottom sediment sources of nutrients. Locally high ground-water discharges in the NRE do occur, however, and could help explain the occurrence of localized phytoplankton blooms, submerged aquatic vegetation, or fish kills.

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