Limnol. Oceanogr., 52(1), 2007, 240–247
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Direct determination of total and fresh groundwater discharge and nutrient loads from a sandy beachface at low tide (Cape Henlopen, Delaware)

Rebecca L. Hays and William J. Ullman[1]
College of Marine and Earth Studies, University of Delaware, Lewes, Delaware 19958

Abstract

Total groundwater discharge from a sandy seepage site to the Delaware Bay at Cape Henlopen, Delaware, was determined together with the associated nutrient loads by trapping the discharge from a length of the shoreline in a tidal pond at low tide and measuring the discharge of the trapped water through a weir at steady state. Salinity was used to parse the total groundwater discharge from the beachface at low tide into a "recycled" estuarine component and a "new" fresh groundwater component. Based on 16 measurements over 18 months, average total discharge and average freshwater discharge at low tide were found to be 2.76 ± 1.08 and 0.87 ± 0.43 L min^{-1} m^{-1} of shoreline, respectively. The estuarine component of discharge varied with the maximum height of the immediately preceding high tide. Owing to the control of freshwater discharge by the average upland hydraulic gradient away from the beach, individual measurements of freshwater discharge at low tide represent a good estimate of this discharge component over time scales of hours to days. Based on the nutrient concentrations in the discharge waters, nutrient loads from the beach to the adjacent estuary at low tide were 148 ± 79, 7.5 ± 5.1, and 197 ± 101 μmol min^{-1} m^{-1} of shoreline for total dissolved N, P, and Si, respectively. Groundwater discharge and these nutrient fluxes contribute significantly to the unusually productive benthic communities that inhabit the seepage and nearby intertidal zones.

Estuaries receive their primary inputs from fluvial discharges and exchange with the ocean, but it is increasingly evident that atmospheric deposition and direct groundwater discharges from uplands also play a significant role in estuarine biogeochemical processes (Paerl 1997; Burnett et al. 2003). Although groundwater discharge is thought to represent only a small fraction of the total freshwater discharge to estuaries and the ocean, there is increasing evidence that nutrients and contaminants associated with this discharge may play a more significant ecological role than implied by the relatively small discharge (Valiela et al. 1992; Burnett et al. 2003; Shaw 2003). This is particularly true in coastal plain settings where nutrient and contaminant inputs to the groundwater from domestic, industrial, municipal, and agricultural practices may be high, the hydraulic conductivity of soils and aquifer sediments is large, and the topographically driven groundwater flow to shallow upland streams is low.

Submarine groundwater discharge in many settings appears to be associated with productive benthic communities (Johannes and Hearn 1985; Bussmann et al. 1999; Zipperle and Reise 2005) and may influence sediment–water exchange through the imposition of hydraulic gradients, the modification of subbottom flow paths, and the alteration of diagenetic reactions and rates (Moore 1999; Li et al. 1999; Slomp and Van Cappellen 2004). These processes, in turn, may influence local biodiversity, ecosystem function, and potentially the transfer of nutrients and contaminants to marine and terrestrial (avian) food chains (Miller and Ullman 2004; Zipperle and Reise 2005). It is therefore important to document the magnitude of total groundwater and associated nutrient or contaminant discharges and the effect of these discharges on nutrient and contaminant budgets over a range of temporal and spatial scales.

There are a large number of methods that may be used to quantify the total and fresh groundwater and associated nutrient discharges to estuaries and the ocean. Over large (up to global) regions, hydrological balance between precipitation inputs and outputs from surface water runoff, evapotranspiration, and groundwater discharge may be useful. These methods, however, suffer from the accumulation of uncertainty from each of the components in the discharge balance and from the variability and uncertainty in nutrient concentrations over these spatial scales (Burnett et al. 2003). Radiochemical or stable isotope methods may also provide useful estimates of groundwater discharge over large areas, although these measurements are subject to some complexity and large uncertainties as well (Moore 1996; Li et al. 1999; Gramling et al. 2003). Seepage meters have proved to be useful devices for determining areal (m^3 m^{-2} d^{-1}) groundwater discharges, but these devices require a great deal of replication to account for the observed heterogeneity in discharge on centimeter to meter scales both perpendicular and parallel to the shoreline (Robinson et al. 1998; Dale 2006). These areal measurements must also be converted to linear units (per unit length of shoreline) to be applied to larger (decameter to kilometer) scales.

In this paper, we propose and test a new method for directly determining total and freshwater discharges and

[1] Corresponding author (ullman@udel.edu).

Acknowledgments

This work was supported by a grant from the Delaware Sea Grant College Program (R/ME-31) to W.U. and D. Miller. R. Bialas (Dartmouth College), an NSF REU intern, assisted in the testing and development of the field methods in summer 2003. J. Brown, R. Dale, D. Miller, K. Savidge, J. Scudlark, and J. Volk assisted in the field and laboratory. We also thank two anonymous reviewers whose comments substantially improved the presentation of our work.

associated nutrient loads from a sandy estuarine shoreline at low tide, a period when the discharge associated with recycled seawater is at a minimum and therefore freshwater discharge may be measured with greatest certainty. This method takes advantage of the local topographical characteristics of our study site. Groundwater discharge is impounded between the beach, the first offshore sand bar, and a constructed weir. Discharge from the pond and the associated nutrient concentrations are then determined at the weir at steady state as an estimate of groundwater discharge and nutrient loads. The beachface, intertidal topography, and subsurface Ghyben-Herzberg zone serve to constrain most of the discharge to the receiving pond. Total discharge is measured directly at the weir; freshwater discharge is determined from total discharge and the salinity balance. The associated nutrient loads are calculated from the product of total discharge and nutrient concentrations measured at the weir. This method yields direct measurements of groundwater and associated material discharges from the unconfined surficial aquifer per unit length of the shoreline at low tide, is appropriate for shoreline lengths up to tens of meters, and integrates discharge over the small-scale heterogeneity parallel to the shoreline. Offshore seepage, submarine springs, and flow due to sediment compaction (Burnett et al. 2003; Slomp and Van Cappellen 2004) cannot be determined by this method. Our method was applied to measure discharges and nutrient loads at Cape Henlopen, Delaware, at the southwest margin of Delaware Bay, a sandy beachface environment with documented freshwater discharge from upland groundwaters (Ullman et al. 2003; Miller and Ullman 2004).

Methods

Sampling site—Cape Henlopen is a sandy and active beach–spit complex located at the intersection of Delaware Bay and the Atlantic Ocean (Fig. 1). The estuarine waters at the Cape are well mixed, and the typical salinities are similar to those of the local coastal ocean (nearby estuarine salinities range from 27 to 30; Kawabe et al. 1988). At low tide, extensive sand flats with diverse and productive benthic communities are exposed (Ullman et al. 2003; Miller and Ullman 2004 and references therein).

The beach sands at this site are predominantly medium to fine sands with occasional minor gravel. Tides are semidiurnal with a diurnal range of 1.42 m (National Oceanic and Atmospheric Association Center for Operational Oceanographic Products and Services (CO-OPS) Tidal Sta. 8557380, Lewes, Delaware; 1 km west of Cape Henlopen site). The potentiometric surface on the beachface at low tide was previously found to range from mean high water high on the beachface to mean low water and outcropped between the beach and first sandbar (Ullman et al. 2003). Discharge from the beach through runnels perpendicular to the shoreline is visible at this site at low tide and during experiments.

Fresh groundwater seepage was identified at the Cape Henlopen site from thermal infrared images at low tide and benthic surveys (Miller and Ullman 2004). Thermal

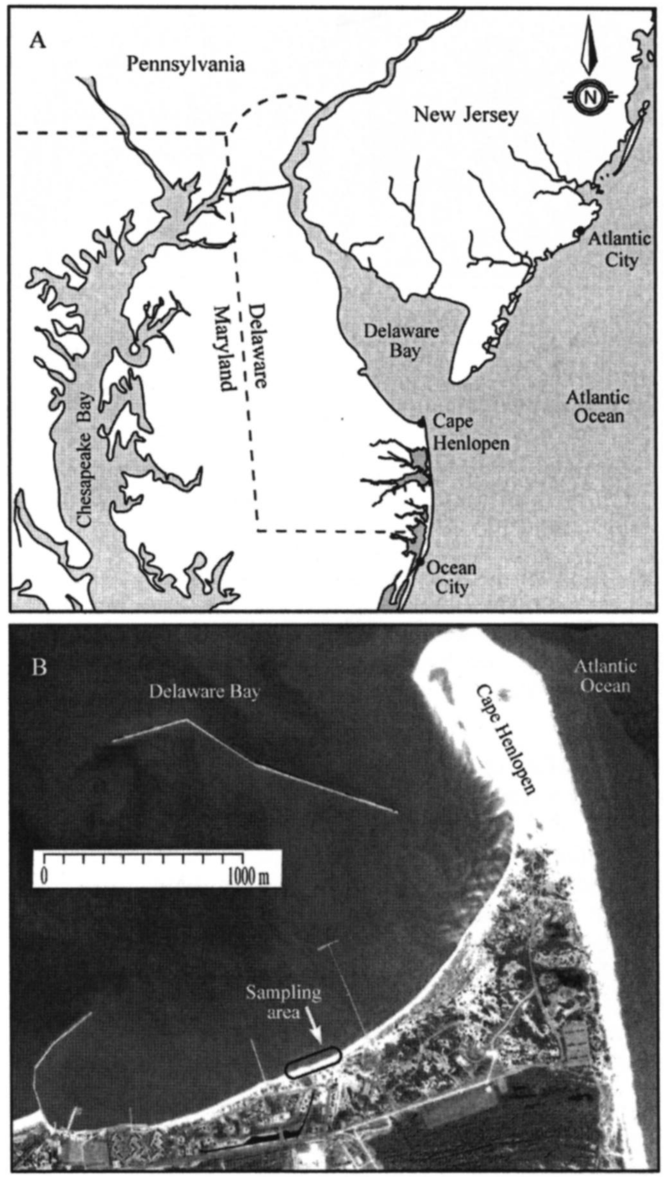

Fig. 1. (A) Cape Henlopen is located at the juncture of the Delaware Estuary and the Atlantic Ocean. (B) The beachface sampling area. Linear features in the estuary are breakwaters and piers.

imagery during the winter months (when there is high thermal contrast between ground and surface waters) demonstrated that seepage is widespread, but discontinuous, along this shoreline and focused in the base of the beachface (Miller and Ullman 2004). Surveys of this site also indicated that there is a productive community consisting of benthic diatoms, an oligohaline polychaete, *Marenzelleria viridis*, and the mud snail *Illyanassa obsoleta* associated with areas of high freshwater discharge. Groundwater discharge moderates the salinity and temperature of *M. viridis* burrows and permits this oligohaline species to survive in this otherwise euhaline–polyhaline environment (Miller and Ullman 2004; Dale 2006). Areas of high groundwater seepage may also be identified by the accumulation of distinctive orange–red iron-hydroxide floc

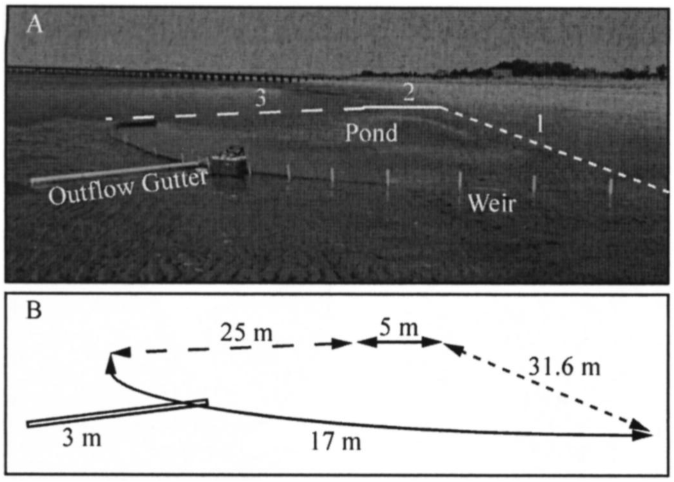

Fig. 2. (A) The sampling pond at Cape Henlopen on 04 May 2004 is bounded by (1) the shoreline, (2) a topographic hydrological divide perpendicular to the shoreline, (3) the first offshore sandbar, and a Plexiglas weir. The outlet of the pond is a section of roofing gutter used for measuring discharge. (B) The length of the sampled shoreline and approximate dimensions of the remaining pond boundaries. The pond area is not disturbed during sampling.

on the surface of the sediment and native sulfur films on the standing water surface at low tide.

Sampling—Discharge could only be collected and sampled during significant low tides (below mean low water) when the Cape Henlopen beachface was fully isolated from the adjacent estuary and the first offshore sand bar was exposed, acting as a hydrological barrier for groundwater flow. Sampling was initiated following the isolation of a section of a shoreline-parallel seepage zone between the beachface and the first offshore bar. A temporary Plexiglas weir constructed between the beachface and the sandbar and a natural hydrological divide between the bar and beachface (identified using dye) served as the remaining boundaries of the receiving pond (Fig. 2). The sections of the weir were embedded a minimum of 10 cm into the sand and linked with plastic hardware. The weir was monitored for leakage during each experiment.

The length of the sampled shoreline (17.7 to 72 m; Table 1), the length of the weir (maximum length 17 m), and the area and maximum depth of the sampling pond depended on the morphology of the beach and adjacent sandbar found on the sampling day. The maximum water level in the pond was kept below 12 cm (range 3.8 to 11.9 cm) to minimize leakage across the weir and disruption to the near-field hydrology.

A 3-m section of roofing gutter provided the controlled cross section for discharge determination. Once the dam was completed, discharge was determined and water samples were collected just above the weir near the discharge outlet for the determination of salinity and nutrient concentrations. Samples were collected every 15 min for up to 3.0 h (4 to 12 samples per experiment, average of 8) until the tide rose and flooded the pond. One or more samples of nearshore estuarine water were taken in order to determine the salinity of the estuarine water (S_{SW}) that recycles through the beachface water because of tidal processes and wave swash. All samples for nutrient analysis were vacuum-filtered through Whatman GF/F 47-mm filters into polyethylene bottles and put on ice in the dark for storage until returned to the laboratory for analysis.

Discharge from the sampling pond was determined from the product of flow velocity and the cross-sectional area of flow through the roofing gutter/discharge channel attached to the sampling weir. Flow velocity was determined by adding dye to the flow stream and measuring the interval of time that the dye took to flow a distance of 2.5 m marked on the discharge channel. The cross-sectional area of flow

Table 1. Steady-state discharge, nutrient concentrations, and nutrient loads from the beach at Cape Henlopen to Delaware Bay at low tide (June 2003 to November 2004).

Date	Shoreline length (m)	Q^*_T (L min^{-1} m^{-1} of shoreline)	Q^*_{SW}	Q^*_{FW}	TDN (μmol L^{-1})	TDP	Si	O^*_{TDN} (μmol min^{-1} m^{-1} of shoreline)	O^*_{TDP}	O^*_{Si}
10 Jun 03	72	1.85±0.09	1.08±0.05	0.77±0.04	25±4	4.0±0.3	74±4	46±8	7.4±0.7	137±10
12 Jun 03	65	1.56±0.08	1.01±0.05	0.55±0.03	31±3	3.4±0.2	49±2	48±5	5.3±0.4	76±5
11 Jul 03	57.5	2.14±0.08	1.68±0.13	0.46±0.07	31±3	3.1±0.1	44±4	66±7	6.6±0.3	94±9
15 Jul 03	57	1.41±0.12	0.92±0.06	0.49±0.06	29±2	4.0±0.1	46±1	41±4	5.6±0.5	65±6
27 Aug 03	24.3	2.80±0.11	1.75±0.11	1.04±0.01	79±7	4.7±0.6	51±4	221±21	13.2±1.8	141±13
26 Sep 03	28	4.78±0.11	3.52±0.14	1.27±0.03	40±3	4.8±0.6	60±0.3	191±15	22.9±2.9	287±7
24 Oct 03	31.3	3.34±0.08	2.53±0.06	0.81±0.15	45±9	2.6±0.7	62±4	150±30	8.7±2.3	207±14
21 Nov 03	24	4.28±0.16	2.44±0.02	1.84±0.18	63±4	2.2±0.8	85±10	270±20	9.4±3.4	364±45
22 Dec 03	41.3	1.76±0.03	0.86±0.29	0.90±0.32	98±5	1.9±1	82±8	172±9	3.3±1.8	144±14
18 Mar 04	20.7	3.32±0.01	2.27±0.13	1.05±0.12	87±2	3.4±0.3	103±4	289±7	11.3±1.0	342±13
04 May 04	31.6	3.38±0.13	1.90±0.13	1.48±0.06	36±1	2.6±0.1	62±2	122±6	8.8±0.5	210±11
02 Jun 04	23.7	2.31±0.25	1.53±0.09	0.78±0.08	46±2	1.7±0.3	65±3	106±12	3.9±0.8	150±18
01 Jul 04	27.2	2.01±0.06	1.47±0.09	0.54±0.08	47±2	0.8±0.1	74±1	94±5	1.6±0.2	149±5
30 Jul 04	20.6	2.95±0.07	2.56±0.09	0.39±0.02	77±8	1.2±0.3	75±1	227±24	3.5±0.9	224±6
27 Sep 04	19	4.44±0.22	3.21±0.03	1.23±0.05	39±3	1.1±0.3	89±2	173±16	4.9±1.4	395±21
11 Nov 04	17.7	1.80±0.11	1.50±0.14	0.30±0.04	83±8	2.2±0.5	94±0.6	149±17	4.0±0.9	169±10
Mean annual average		2.8±1.1	1.9±0.8	0.9±0.4	53±24	2.7±1.3	70±18	148±79	7.5±5.1	197±101

was determined from the shape of the discharge channel and the water height in the channel. Three replicate determinations of velocity were made and averaged for each interval.

Samples of pond water were collected and pond discharge was determined throughout each experiment to monitor the approach to steady-state discharge. After steady state was confirmed from discharge and salinity, the final two to four discharge measurements and samples were used to determine groundwater discharge and nutrient loads from the beach to the estuary at Cape Henlopen. The average steady-state discharge and nutrient loads from the beach were then normalized to the length of the sampled shoreline for subsequent calculations of linear discharge and loads.

Analytical methods—Salinity was measured at the time of sample collection with a YSI-30 salinometer (Yellow Springs Instruments). Nutrient concentrations were determined using an O/I Analytical Flow Solution IV analyzer (OI Analytical). Total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) were determined as nitrate (reduction to nitrate and then nitrite by the sulphanilamide/N[1-napthyl] ethylene diamine method) and phosphate (phospho-molybdenum blue method) following oxidation by multiply reprecipitated potassium persulfate in an autoclave (D'Elia et al. 1977; Solorzano and Sharp 1980*a,b*). Silicate (Si) was determined by the silico-molybdenum blue method after complexation of phosphate by oxalic acid (Strickland and Parsons 1972; Sharp et al. 1982).

Mass balance—The measured discharge (Q_T; L min^{-1}) includes contributions from new freshwater (Q_{FW}) and recycled estuarine water (Q_{SW}):

$$Q_T = Q_{\mathrm{SW}} + Q_{\mathrm{FW}} \tag{1}$$

(Destouni and Prieto 2003). The relative importance of these two sources can be distinguished using the salt balance (Destouni and Prieto 2003):

$$S_T Q_T = S_{\mathrm{SW}} Q_{\mathrm{SW}} + S_{\mathrm{FW}} Q_{\mathrm{FW}} \tag{2}$$

where S_T is the salinity (or any other conservative property of water) of the water discharging from the beachface (i.e., at the discharge gutter), S_{SW} is the salinity of the near-shore estuarine water, and S_{FW} is the salinity of the fresh groundwater. Since S_{FW} is always much less than either S_{SW} or S_T, it can be neglected, and therefore Eq. 2 can be simplified as

$$S_T Q_T \cong S_{SW} Q_{SW} \quad \text{or} \quad Q_{\mathrm{SW}} \cong \left(\frac{S_T}{S_{\mathrm{SW}}}\right) Q_T \tag{3}$$

Q_{FW} is then determined as the difference between Q_T and Q_{SW}:

$$Q_{\mathrm{FW}} = Q_T - Q_{\mathrm{SW}} \tag{4}$$

Discharge (Q_i; L min^{-1}) is normalized to shoreline length (Q_i^*; L min^{-1} m^{-1} of shoreline) for comparison of measurements made over the course of the study.

The linear nutrient flux at steady state (O^*; μmol min^{-1} m^{-1} of shoreline) is determined from the product of linear discharge, Q_T^*, and nutrient concentration (C_T) measured at the discharge gutter:

$$O^* = C_T Q_T^* \tag{5}$$

Uncertainties—True experimental and sampling replication was impossible in this study, since only a single experiment could be conducted during a low tide period and the morphology of the beachface changed with every tidal cycle. As a result, different regions and lengths of the same beach (Fig. 1) were sampled on each sampling date. On two occasions (June and July 2003), however, two experiments were conducted during the same spring tidal period, in the same area of the beach, separated by two to four days. These two pairs of results serve as the best example of overall experimental uncertainty of the sampling and analytical protocols.

Results and discussion

We measured discharge and the composition of the discharge water at Cape Henlopen 16 times over a period of 18 months from June 2003 to November 2004 (Table 1). Sampling was undertaken approximately once a month, except when periods of winter ice-cover prevented sampling. A detailed description of the results of one experiment is given below, followed by a summary of the results of the remaining 15 experiments.

Example experiment—On 27 August 2003, salinity, discharge, and nutrient (TDN, TDP, and Si) concentrations were determined at the weir, behind which beachface seepage waters from a 24.3-m length of shoreline were collected (Fig. 3). The salinities of nearshore estuarine water (S_{SW}) and upland groundwater (S_{FW}; 1.5 m below the beachface above the berm) were found to be 27.25 ± 0.07 (n = 2) and 0.13 ± 0.06 (n = 3), respectively.

The salinity of the discharge pond water varied little (S_T = 16.3 to 17.1) over the course of the experiment (Fig. 3B) and was significantly less than the salinity of the estuarine water, confirming the importance of freshwater dilution and discharge at the Cape Henlopen site. Q_T was initially high but then dropped to a fairly steady level after 30 min and remained constant, within the uncertainty of the measurements, for the remainder of the experiment (Fig. 3A). The steady-state discharge (Q_T = 68 ± 3 L min^{-1}) was determined from the average of the final three discharge measurements before the receiving pond was flooded by the rising tide. With the measured salinities (S_T), discharges (Q_T), and Eqs. 3 and 4, Q_{SW} and Q_{FW} at steady state were found to be 43 ± 3 and 25.4 ± 0.3 L min^{-1}, respectively. Q_T^*, Q_{SW}^*, and Q_{FW}^* were found to be 2.80 ± 0.11, 1.75 ± 0.11, and 1.04 ± 0.01 L min^{-1} m^{-1} of shoreline, respectively, and the fraction of fresh upland groundwater in the steady-state discharge was 37% ± 1%.

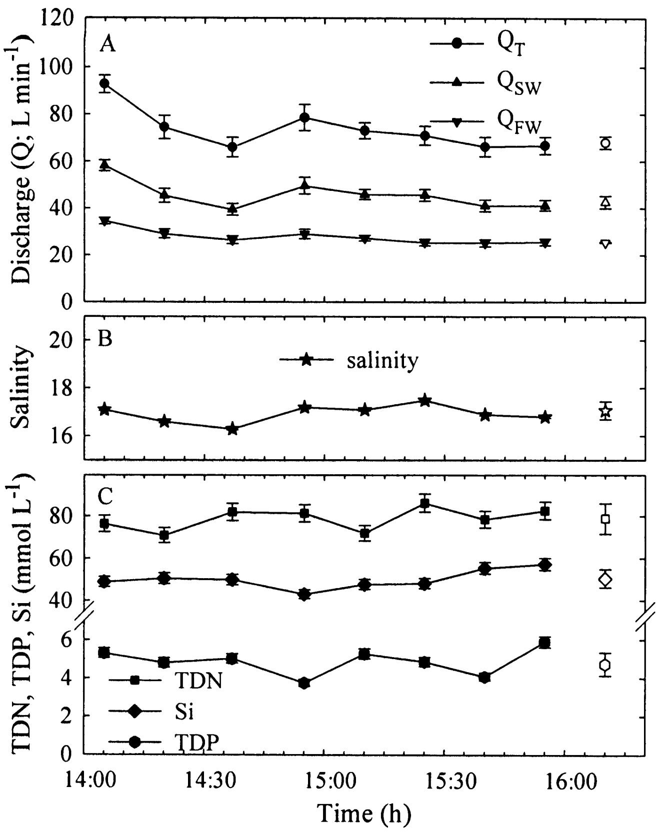

Fig. 3. Results of one experiment conducted on 27 August 2003. Experimental data are solid symbols. Steady-state averages for (A) discharge, (B) salinity, and (C) nutrients used in subsequent calculations (open symbols) are based on the means of the last three experimental measurements at steady-state. Error bars indicate 1 σ uncertainties.

Nutrient concentrations in the discharge pond often varied significantly during a single experiment, possibly reflecting incomplete mixing in the beachface (*see* Ullman et al. 2003) or some nutrient processing within the pond itself. Nonetheless, a period of relatively stable concentration that corresponded to the period of stable salinity could be identified for each experiment. In the 27 August 2003 experiment (Fig. 3C), the concentrations of TDN, TDP, and Si rose at the end of the sampling period, possibly because of an incipient response to the rising tide; thus, the three concentrations used to calculate the steady-state concentration were the fifth, sixth, and seventh nutrient samples collected. The last sample collected during an experiment was not always included in the determination of steady-state nutrient concentrations for this reason.

The steady-state TDN concentration in the pond water was 79 $\pm$ 7 μmol L^{-1} (Fig. 3C; Table 1). In this experiment, $NO_3^- + NO_2^-$ were the dominant forms of dissolved nitrogen, but NH_4^+ and dissolved organic nitrogen (DON) could dominate at other times of the year (Hays 2005). TDP concentrations (Fig. 3C) are somewhat more variable than the TDN concentrations, with a steady-state concentration of 4.7 $\pm$ 0.6 μmol L^{-1}. Orthophosphate was the dominant form of phosphorus discharging from the sampling pond. The average steady-state concentration of Si was 51 $\pm$ 4 μmol L^{-1}. On the basis of the steady-state data, the linear fluxes of TDN, TDP, and Si (O^*) from the beachface at Cape Henlopen on 27 August 2003 were 221 $\pm$ 21, 13.2 $\pm$ 1.8, and 141 $\pm$ 13 μmol min^{-1} m^{-1} of shoreline, respectively.

Seasonal patterns of discharge—Experiments identical to the one performed on 27 August 2003 were performed on an additional 15 days between June 2003 and November 2004. The results of these experiments are summarized in Table 1. The average total discharge (Q_T^* = 2.8 $\pm$ 1.1 L min^{-1} m^{-1} of shoreline) is within the ranges reported in previous studies: 1.2–42 L min^{-1} m^{-1} of shoreline (Cable et al. 1997), 0.5–2.5 L min^{-1} m^{-1} of shoreline (Ullman et al. 2003), 1.5–16 L min^{-1} m^{-1} of shoreline (Boehm et al. 2004), 2.3 L min^{-1} m^{-1} of shoreline (Bokuniewicz et al. 2004), and 4.2–8.9 L min^{-1} m^{-1} of shoreline (Boehm et al. 2006).

There is a large variation in total discharge (Q_T^*) during the 18 months of sampling (1.5 to 5 L min^{-1} m^{-1} of shoreline; Table 1 and Fig. 4A). Relative minima occur during the summer and winter and relative maxima during the spring and fall. Q_T^* is the sum of two components with contrasting origins and controls. We expected that Q_{SW}^*, the estuarine seawater component, would respond to the extent of wave and tide replenishment of the beachface groundwater during preceding tidal cycles and that Q_{FW}^*, the fresh groundwater component, would reflect longer term seasonal changes in water balance and upland water levels. Consistent with this expectation, there is a direct and apparently linear dependence of Q_{SW}^* on preceding high tidal height (Fig. 5). Although Fig. 4A shows that there are relative minima in Q_{FW}^* in both sampled summers and a broad maximum from fall 2003 to spring 2004, the present results are insufficient to confirm any seasonality in Q_{FW}^*. No correlation between Q_{FW}^* and recent rainfall was found (Hays 2005). Future studies should monitor nearby groundwater levels in order to confirm the dependence of Q_{FW}^* on groundwater hydrology.

The fraction of freshwater discharge observed in this study (32% $\pm$ 19% of total groundwater discharge) is consistent with the higher values observed in other studies. In Tannowa, Osaka (Japan), the fresh groundwater component of the submarine groundwater discharge ranged from 4% to 29% (Taniguchi and Iwakawa 2004). In Huntington Beach, California, between 0% and 26% of the measured total discharge resulted from freshwater input (Boehm et al. 2006). A study by Destouni and Prieto (2003) suggests an upper value of 30% for the quotient of fresh groundwater to total submarine groundwater discharge. The uniformly high fraction of freshwater discharge in our measurements is the result of our experimental protocol, which focuses on measurements made during a period (low spring tides) when there is limited or no seawater recharge to the beach that contributes to discharge. Since Q_{SW}^* is directly and Q_T^* is therefore indirectly dependent on preceding tidal height, the fractional amount of Q_{FW}^* must be inversely dependent on preceding tidal height (Eq. 1).

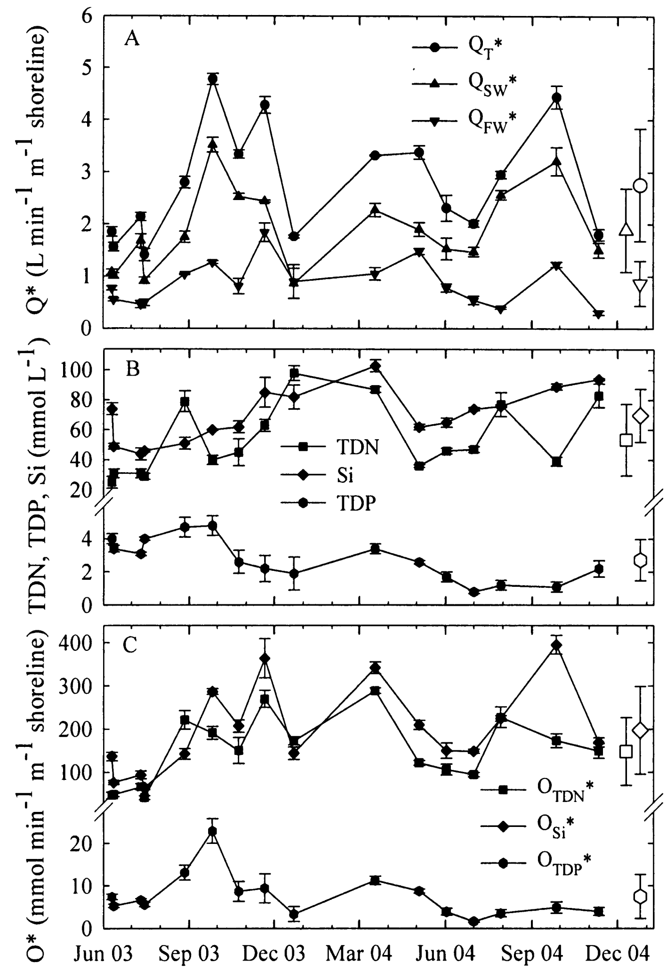

Fig. 4. (A) Patterns of steady-state discharge, (B) nutrient concentrations, and (C) nutrient loads from the Cape Henlopen to Delaware Bay from June 2003 to November 2004. Open symbols at the right of each diagram indicate average annual values. Linear discharge (Q^*) and linear nutrient loads (O^*), in each case, are normalized to the length of the sampled beachface.

Average daily freshwater discharge—Although Q^*_{FW} was determined only during low spring tides, these measured values serve as a good estimate of average daily freshwater discharge over the entire tidal cycle. This assumption is based on evidence that the effect of tidal change on upland water tables is rapidly damped out in the aquifer with distance from the beach. This results in a constant hydraulic gradient and therefore constant discharge from the upland toward the beachface at all times of the tidal cycle (Ataie-Ashtiani et al. 1999; Li et al. 1999; Uchiyama et al. 2000). Our method of discharge determination, however, cannot rigorously confirm this hypothesis.

The maximum uncertainty in the average daily freshwater discharge based on the measured values and the assumption of constant Q^*_{FW} is a factor of two. If at high tide Q^*_{FW} approaches zero, and at low tide it is at its maximum as determined in our study, then the average discharge (at a constant mean tide) should be approximately one half of the measured maximum. If, however, the

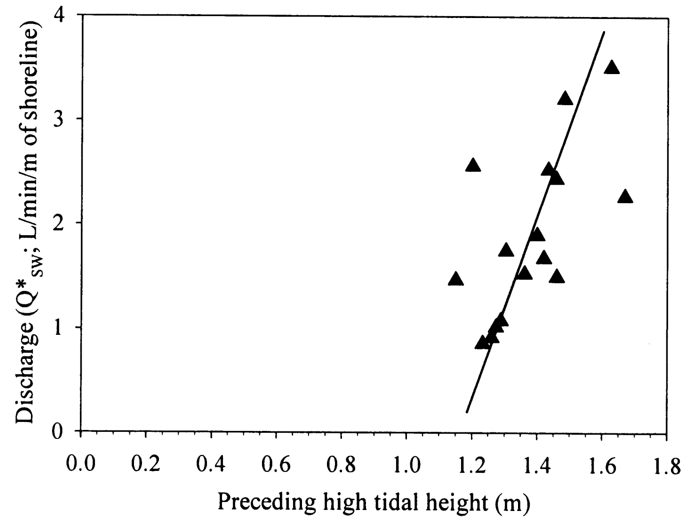

Fig 5. The apparently linear relationship between the discharge of recycled seawater, Q^*_{SW}, and the height of the immediately preceding high tide. This component of the total discharge reflects, as expected, the tidal behavior of the estuary and the replenishment of the beach by tidally driven water.

Q^*_{FW} at high tide is any value greater than zero, then the average daily discharge of freshwater will be closer to the measured value of Q^*_{FW}.

Nutrient loads at low tide—Steady-state nutrient concentrations and linear nutrient loads (O^*; Eq. 5) from the Cape Henlopen beachface to Delaware Bay at low tide are reported in Figs. 4B and 4C and Table 1.

The average load of TDN from the beachface to the estuary at low tide was 148 ± 79 μmol min^{-1} m^{-1} of shoreline. These loads are similar to that reported by other investigators for the entire tidal cycle: 198 μmol min^{-1} m^{-1} of shoreline (Uchiyama et al. 2000), 69–420 μmol min^{-1} m^{-1} of shoreline (Ullman et al. 2003), and 80 μmol min^{-1} m^{-1} of shoreline (Boehm et al. 2006). Although seasonal patterns are unclear, the minimum measured loads are found in the summers of 2003 and 2004 and broad maxima occur during the fall and winter (Fig. 4C). The minimum loads correspond to minima in the total discharge from the beach to the estuary. $NO_3^- + NO_2^-$, NH_4^+, and DON all contribute substantially to the measured nitrogen load to the estuary, although there is variation in the relative amount of these species (Hays 2005). However, no seasonal trends in species dominance were found.

The average annual TDP flux from the beachface to the estuary at low tide was 7.5 ± 5.1 μmol min^{-1} m^{-1} of shoreline, similar to P loads for the entire tidal cycle from groundwater to estuaries found in previous studies: 11 μmol min^{-1} m^{-1} of shoreline (Uchiyama et al. 2000), 12–28 μmol min^{-1} m^{-1} of shoreline (Ullman et al. 2003), and 3.5 μmol min^{-1} m^{-1} of shoreline (Boehm et al. 2006). The TDP fluxes broadly follow the seasonal patterns of discharge and of TDN loads (Fig. 4C). The similarity in the two patterns may result from the common dependence of these loads on discharge and the relatively small variation in measured TDN and TDP concentrations compared to discharge. There is, however, a distinct maximum in the measured TDP load in fall of 2003 that is not observed in the TDN load. TDP loads were always dominated by orthophosphate, with dissolved organic phosphorus contributing only a small additional load (Hays 2005).

The average silica flux from the beachface at low tide was 197 $\pm$ 101 μmol min^{-1} m^{-1} of shoreline. Si loads follow the seasonal patterns of discharge and TDN loads (Fig. 4C).

It is intriguing to speculate on the origin of the nutrients discharging at the Cape Henlopen shoreline. There are at least three possible sources of nutrients: upland freshwaters that represent a new addition of dissolved nutrients to the intertidal ecosystem (Paerl 1997), seawater that represents a recycled source of dissolved nutrients, and diagenesis in the beachface, a source of "regenerated" nutrients to the intertidal. Generally, owing to the low concentrations of nutrients in seawater compared to upland freshwaters (Ullman et al. 2003; Denver et al. 2004; Hays 2005), recycled seawater should be a minor contribution to the measured loads. The diagenetic contribution is likely to be the most variable since sediment water exchange can be both a source and a sink for dissolved nutrients. Preliminary measurements suggest that diagenetic contributions to nutrient loads may be comparable in magnitude to the contributions from upland freshwaters (Hays 2005).

It is also not clear whether the diagenetic source should be considered a new or regenerated source of nutrients to the intertidal ecosystem. There are certainly mineral sources of P and Si in the beach sediments and mineral–water reactions that could release these nutrients to beachface waters by diagenesis, but it is not clear whether these processes are important in this setting. It is also possible, however, that the bulk of diagenetic nutrient contribution is the result of the degradational release from biogenic particles that infiltrate into the beachface with recycled seawater and are subsequently released. The N : P (26 $\pm$ 18) and N : Si (0.77 $\pm$ 0.29) ratios of the measured nutrient fluxes at Cape Henlopen compare favorably to the composition of marine organic matter (N : P of 16 and N : Si of 1; Redfield et. al 1963; Brzezinski 1985) and give credence to the possibility that regeneration of nutrients from marine biogenic particles is an important source of the nutrients found in discharge waters at this site. Further work is needed to quantify these various sources and their relative contributions to nutrient loads to Delaware Bay.

Groundwater discharge and estuarine nutrient balance— The steady-state impoundment technique precisely determines groundwater discharge across coastal and marine shorelines for a short period of time at low tide. Salinity further distinguishes between contributions of upland freshwater and recycled seawater to total discharge. Freshwater discharge determined by this method is a good estimate of discharge over longer time scales and is determined with sufficient precision to be useful in determining new nutrient inputs to estuaries from upland groundwater sources.

Direct groundwater discharge is an often ignored pathway of nutrient transport to coastal and estuarine receiving waters and should be better integrated into estuarine mass-balance models. There are a number of possible nutrient sources (and potential sinks) that contribute to the nutrient loads from the beachface to the estuary, and the present results are insufficient to speculate on the relative magnitudes of these contributions. Total observed loads cannot be used as an estimate of the new loads through the groundwater pathway. However, fresh groundwater discharge, determined by the impoundment method, together with relevant nutrient concentrations in upland groundwaters have the potential to provide sufficiently precise estimates of the new nutrient loads for inclusion in estuarine mass-balance models.

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Received: 29 March 2006
Accepted: 9 August 2006
Amended: 29 August 2006