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Nitrogen inputs to a marine embayment: the importance of groundwater

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Abstract. We examined the importance of nitrogen inputs from groundwater and runoff in a small coastal marine cove on Cape Cod, MA, USA. We evaluated groundwater inputs by three different methods: a water budget, assuming discharge equals recharge; direct measurements of discharge using bell jars; and a budget of water and salt at the mouth of the Cove over several tidal cycles. The lowest estimates were obtained by using a water budget and the highest estimates were obtained using a budget of water and salt at the Cove mouth. Overall there was more than a five fold difference in the freshwater inputs calculated by using these methods.

Nitrogen in groundwater appears to be largely derived from on site septic systems. Average nitrate concentrations were highest in the region where building density was greatest. Nitrate in groundwater appeared to behave conservatively in sandy sediments where groundwater flow rates were high ($> 1\ \mathrm{l/m^2/h}$), indicating that denitrification was not substantially reducing external nitrogen loading to the Cove.

Nitrogen inputs from groundwater were approximately $300\ \mathrm{mmol\text{-}N/m^3/y}$ of Cove water. Road runoff contributed an additional $60\ \mathrm{mmol/m^3/y}$. Total nitrogen inputs from groundwater and road runoff to this cove were similar in magnitude to river dominated estuaries in urbanized areas in the United States.

Introduction

The primary production of coastal marine waters and the lower reaches of most estuaries is limited by the supply of combined nitrogen (Boynton et al. 1982; Nixon & Pilson 1983; Howarth 1988). Nitrogen is supplied to estuarine and coastal waters by freshwater runoff, precipitation, nitrogen fixation, and sometimes offshore water. In addition to natural sources of nitrogen, large amounts of anthropogenic nitrogen may enter via the discharge of wastewater and runoff from fertilized land (Nixon & Pilson 1983).

Most studies on nitrogen cycling in coastal areas have focused on large estuaries. Much less is known about nitrogen dynamics in the restricted marine coves, bays and coastal lagoons that are common features of the coast of much of the Northeastern United States. These ecosystems differ from large river-dominated estuaries in morphometry and hydrology. Many are frequently quite shallow and many support fairly large communities of macrophytes. Most are separated from offshore waters by fairly restricted inlets which may exhibit strong time and velocity asymmetries between flood and ebb tide (Welsh et al. 1982). Because surface water runoff into these areas is small, a significant proportion of freshwater input is from groundwater (Bokuniewicz 1980; Capone and Bautista 1985).

Sewage derived nitrogen is a major source of new nitrogen in many river dominated estuaries. In a review of the nitrogen budget of 15 estuaries, Nixon and Pilson (1983) found that sewage inputs commonly made up half of the inorganic nitrogen input to urban estuaries. While wastewater is rarely directly discharged into coastal coves or lagoons, coastal development could still be having an impact on the nitrogen budget of these systems via groundwater inputs. Much of the Northeastern United States disposes of sewage in individual on-site septic systems (EPA 1980). In recent years it has become widely recognized that septic systems can contribute substantial amounts of nitrogen to the groundwater (Perskey 1986). This nitrogen can adversely affect the quality of the aquifer (Perskey 1986; Flipse et al. 1984) and may impact downstream ecosystems which receive this nutrient rich groundwater (Johannes 1980; Sewell 1982; Capone & Bautista 1985; Valiela & Costa 1988). Only recently have investigators begun to examine the importance of this nitrogen source to coastal ecosystems. In some cases nitrogen inputs from groundwater contribute more nitrogen than river discharge (Johannes 1980) and groundwater nitrogen inputs may contribute to eutrophication (Sewell 1982; Capone & Bautista 1985; D'Elia et al. 1981).

The objective of this study is to determine the relative magnitude of nitrogen derived from groundwater in Town Cove, Orleans, a small marine cove on Cape Cod, Massachusetts. We compared several commonly used methods of determining groundwater inputs to assess whether simple indirect methods are adequate to answer management questions. We also examined nitrogen processes in surficial sediments receiving groundwater inputs to determine if large amounts of nitrogen were lost via denitrification and thus, not contributing to primary production in the cove.

Methods

Site description

Town Cove, Orleans, is located on the outer arm of Cape Cod (Fig. 1). The Cove is connected to the Atlantic Ocean by a number of tidal channels which flow through an extensive area of mud flats and marshes. For our budgets and tidal studies we defined the Cove mouth as at a narrow restriction in the tidal channel just outside the deeper portions of the Cove. The average depth inside the Cove is 2.2 m with a maximum depth of 6 m. The area of Town Cove is $1.4 \times 10^6 \mathrm{m}^2$ and it has a volume of $3.13 \times 10^6 \mathrm{m}^3$.

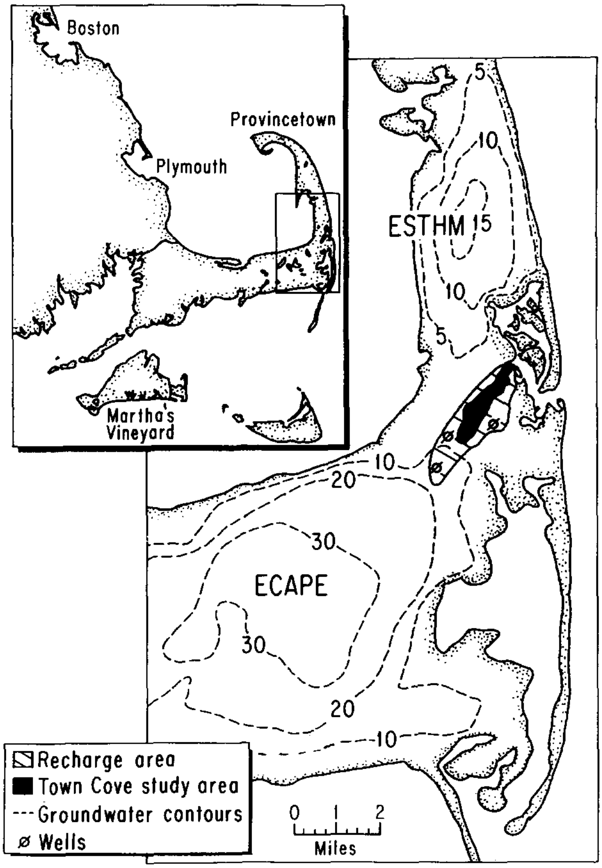


Fig. 1. A map of Town Cove, Orleans, Mass., showing the height of the groundwater table (redrawn from Guswa & LeBlanc 1981). The calculated recharge area is shown along with the location of the three groundwater wells used to calculate the slope of the water table.

Groundwater discharge

We used three independent methods of estimating groundwater discharge into Town Cove:

— seepage meters,
— a water budget of the watershed, and
— measurements of salinity and tidal displacements at the mouth of the cove.

Seepage meters

Ground water discharge was directly measured by inserting plastic chambers into subtidal sediment, and measuring the accumulation of water in a plastic bag attached to the chamber (Lee 1976; Bokuniewicz & Zeitlin 1980; McBride &

Pfannkuch 1975). Chambers were 45 cm in diameter and placed 3–5 cm into the sediment. A total of 29 seepage meters of this type were deployed along transects perpendicular to the shore in a number of locations, during June and July of 1983. Normally 2–4 measurements were made over a tidal cycle. In some cases the salinity of the water accumulating in the bag was also measured. Benthic chambers with bags attached were deployed in the deeper portions of the Cove four times during 1982 and 1983 and placed in shallow (< 5 m) sediments twice in the fall of 1982. The water inside of the benthic chambers was continuously stirred allowing us to measure changes in nutrients and salinity over time.

Water budget

Regional patterns of groundwater flow in the Orleans area have been mapped (Guswa & LeBlanc 1981; CCPEDC 1978, map 5.2 D.E.Q.E. Groundwater Atlas, unpublished) and were used along with surface contour data to delineate the watershed area of Town Cove. Precipitation data were obtained from the National Weather Service. Artificial recharge, from the piping of water into the recharge area from the Town wellfield adds to natural recharge. Data of Belfit (CCPEDC, pers. comm.) was used to estimate artificial recharge. We calculated the slope of the groundwater table over several time periods to determine if it was reasonable to assume that over a one year period discharge would equal recharge. We obtained seasonal data on water table elevation from 2 nearby USGS wells. We also installed 9 observation wells around Town Cove, 8 of which were located around the shore of Town Cove. These eight wells, referred to as Beach Wells, consisted of 10 foot long PVC pipes, with slotted ends wrapped with fine plastic mesh sand screen, jetted into the ground below a layer of resistant glacial till. The other well was located in the center of the Orleans business district. Absolute elevation of all wells was obtained by surveying back to USGS bench marks.

Discharge from the Cove mouth

Water samples were taken at the Cove mouth every half hour over two complete tidal cycles at four different times during the study. These samples were analyzed for salinity using an Autosal 6000 salinometer. A time series of tidal displacements, measured by Aubrey & Speer (1983), were used to estimate tidal fluxes of salt and their difference, the net freshwater tidal flux.

Nitrogen in groundwater

Nitrate concentrations in groundwater from shallow domestic wells were obtained primarily from Barnstable County Health Department. In addition we sampled 39 shoreside stations around the Cove which included our beach wells, active and inactive domestic wells and institutional wells, and seeps found around the Cove shoreline. In all, 116 wells were sampled between 1982 and 1983. A number of wells were sampled up to five different times to look for seasonal changes.

Surface runoff

Three storm sewer pipes enter Town Cove. These were sampled for nitrate, nitrite, and ammonium during 1982 and 1983 to calculate their contribution to nitrogen loading into the Cove. Only one of these pipes, from Gutter Pond, continuously discharges water. A 90° 'V' notch weir was installed to measure the discharge rate into the Cove. Discharge was estimated using the 'cone formula' (U.S. Department of Interior 1974).

Porewater measurements

Porewater measurements were taken on transects extending out from shore. Small diameter screened pipes were placed into the sediment and porewater removed with a syringe. Porewater was taken at 3–4 depths at each site. Samples were immediately filtered using 0.45 μm membrane filters. A subsample was taken for ammonium and the sample fixed with phenol-alcohol. Nitrate samples were frozen for later analysis. A third sample was kept for salinity determinations. Over the course of the study 195 measurements were made.

Analytical methods

Ammonium was analyzed using the method of Solorzano (1969). Nitrate was converted to nitrite using a cadmium reductant and nitrite was analyzed colorometrically using a modification of the procedure described in Strickland & Parsons (1972). Porewater samples were treated to remove interference before being analyzed for nitrate (Afghan & Ryan 1975).

Other nitrogen loading estimates

We independently assessed nitrogen loading by obtaining data on fertilizer sales from the hardware stores in the Town of Orleans. Estimates of domestic and commercial nitrogen inputs from septic systems has been estimated by Belfit (CCPEDC, pers. comm.). Nitrogen input from precipitation falling directly on the surface of Town Cove was estimated using data from a rain collection station located in South Truro (NADP 1982–3).

Results and discussion

Groundwater flow into the cove

Water budget

Although the Cape Cod aquifer has been designated by EPA as a 'sole source aquifer', the flow of groundwater can be represented as five separate principal groundwater systems (Guswa & LeBlanc 1981). Town Cove receives water from

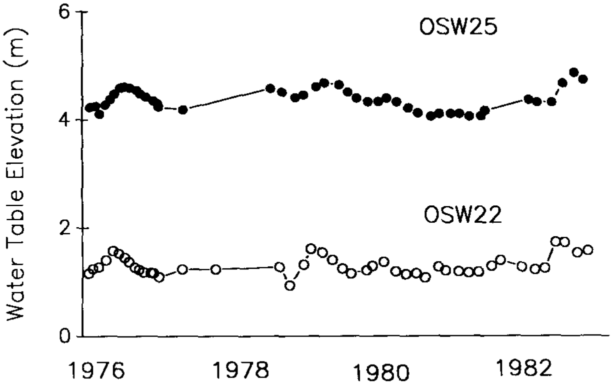


Fig. 2. The elevation of the water table at two USGS wells in the Orleans Town Cove recharge area (data from Belfit, CCPEDC, pers. comm.).

the 'ESTHM' system, centered in Eastham to the north, and the larger 'ECAPE' system, to the south and west (Fig. 1). We used this data, along with surface contours, to estimate a total recharge area of Town Cove as 7.9 × 10^6 m (Fig. 1).

Precipitation averages 119 cm/y (National Weather Service 1982–1983) of which 33 to 45 cm/y contributes to groundwater recharge (Horsley, CCPEDC, pers. comm.; Strahler 1972). Given the recharge area for Town Cove this is equivalent to 0.71 to 0.97 × $10^4\,m^3$ per day. Artificial recharge, from the piping of water into the recharge area from the Town wellfield, adds 0.23 × $10^4\,m^3$ to natural recharge (Belfit CCPEDC, pers. comm.).

Average monthly precipitation on the outer Cape shows considerable spread around 30 year averages. These variations in precipitation and recharge cause short-term fluctuations in the elevation of the groundwater table but over the long-term, discharge to the Cove can be expected to equal the recharge. The groundwater level on Cape Cod shows some seasonal fluctuation and also changes in response to annual differences in precipitation. For example, the unusually wet 1982–83 year resulted in an increase in recharge which produced a rise in the water table during 1983 (Fig. 2). However, in spite of this increase, the slope of the groundwater table during the study year was not appreciably greater than the twenty year average (Table 1).

Seepage meters

Groundwater flow in a homogeneous shallow unconfined aquifer, and freshwater flow into saline bodies of water should be restricted to a narrow zone along the shore (Hubbert 1940; review by Johannes 1980). Consistent with theory, and observations reported elsewhere (Bokuniewicz 1980; McBride & Pfannkuch 1975; Lee 1976), we found no flow into benthic chambers deployed in the deeper, muddy portions of Town Cove. We observed no decrease over time in the salinity of the water in benthic chambers deployed in these deeper

Table 1. Calculation of groundwater slope towards Town Cove using well heights for several time periods.

	Time of measurements			
	1963–1976	1976–1983	Sept. Sept 1982–1983	March Sept. 1983–1983
OSW22				
ht of Water table (m)	1.28	1.28	1.40	1.62
Slope	0.0032	0.0032	0.0034	0.0040
OSW25				
ht of Water table (m)	4.60	4.33	4.39	4.60
Slope	0.0034	0.0032	0.0032	0.0034
Town center				
ht of Water table (m)			3.20	
Slope			0.0070	

areas and we found no decrease in the salinity of porewater samples taken from 0 to 30 cm. From this we conclude that groundwater is not entering the deeper portions of the Cove.

Bags attached to seepage meters in shallow water rapidly accumulated water, suggesting appreciable rates of groundwater flow. The discharge was variable both in time and space (Table 2). Extreme values ranged from 0.4 to 15.7 liters/m^2/h, but when averaged over several deployments, flows typically lay between 1–3 liters/m^2/h. Discharge showed a tendency to decrease offshore from the intertidal areas to water depths of 1–2 m. Below 2 m we did not detect any flow, and the hydraulic gradient below 2 m was always zero. Discharge rates changed over time (Table 2), and we did not see a clear relationship between discharge rate and tidal stage, in contrast to the results of Lee (1976).

The large changes we observed in flow rates led us to suspect that not all of the water accumulating in the chambers was due to freshwater flow. Near the end of the study we examined the change in salinity in a small number of chambers. Salinity changes indicated that in several cases up to 65% of the water accumulating in the chambers was seawater. There was a tendency for the ratio of seawater/freshwater accumulation to be greatest in the intertidal zone. This may represent a true flow of seawater through the sediments or it may be an artifact. Shaw & Prepas (1989) have observed anomalous short-term influxes of water into seepage meters caused by expansion of the polyethylene bag attached to the chamber. We used a different bag than Shaw & Prepas (1989) and pre-filled our bags with some water as they have recommended; however, we cannot rule out a problem with these measurements. Because we did not measure salinity in our other chambers we cannot correct for seawater inputs. Therefore, the freshwater discharge we calculated using seepage meters is a maximum estimate. If all seepage occurred within 1 m of shore the maximum freshwater flow would be $4.3 \times 10^4 m^3/d$. If the flow extends to 2 m the flow would be $5.0 \times 10^4 m^3/d$.

Table 2. Water flow into seepage meters deployed in transects extending offshore. In one case, meters were deployed during a flood tide (F), allowed to remain in place through one and a half tidal cycles (E–F), and collected during the next ebb tide.

Distance from mean low water (m)

	−28		−16		0		3		8		12	
Tide	*t*	*q*	*t*	*q*	*t*	*q*	*t*	*q*	*t*	*q*	*t*	*q*
Ebbing	0.8	0.7	1.2	1.2	1.2	7.5	1.2	3.1	1.1	3.8	1.1	3.0
Ebbing	Exposed		Exposed		2.4	1.1	2.4	2.3	2.4	3.3	2.3	2.0
Low	Exposed		Exposed		Exposed		2.6	0.6	2.6	2.8	2.6	2.0
Flooding	Exposed		1.5	15.7	1.5	5.8	1.5	3.8	1.5	4.5	1.5	3.0
Mean		$\overline{0.7}$		$\overline{8.5}$		$\overline{4.8}$		$\overline{2.4}$		$\overline{3.6}$		$\overline{2.9}$
S.D.		–		10.2		3.3		1.3		0.7		0.7

Distance from mean low water (m)

	5		10		15		20		25	
Tide	*t*	*q*	*t*	*q*	*t*	*q*	*t*	*q*	*t*	*q*
Flooding	3.5	3.5	3.5	3.5	3.5	0.7	3.5	0.5	3.5	0.6
F-E-F-Ebbing	16.4	0.4	No data		16.4	0.7	16.4	0.5	16.4	0.3
Ebbing	2.3	5.4	2.3	1.1	2.3	0.7	2.3	0.5	2.3	0.6
Mean		$\overline{3.1}$		$\overline{1.6}$		$\overline{0.7}$		$\overline{0.5}$		$\overline{0.5}$
S.D.		2.5		1.7		0.02		0.04		0.15

(t = duration of measurements (h); q = discharge (l/m^2/h).

Table 3. Freshwater fluxes at the mouth of Town Cove, based on a salinity time series and tidal information.

Date	Tidal flux ($10^4\,m^3/d$)				Net daily discharge[*]
	Flood	Ebb	Flood	Ebb	
March 14–15, 1983	6.9	14	6.4	12	8.9
April 26–27, 1983	6.1	6.7	6.9	6.6	−0.2[†]
June 29–30, 1983	1.1	3.2	0.9	3.5	4.1
August 18–19, 1983	0.19	0.14	0.14	0.20	0.05[†]

[*] Corrected for net changes in tide level, and expressed over 24 h.

[†] Precipitation occurred during the tidal series.

Discharge from the Cove mouth

Freshwater fluxes past the Cove mouth were highly variable (Table 3). On rainy days (April and August) calculated freshwater fluxes out of the Cove were very small or negative. Evidently runoff from precipitation falling on the outer Nauset embayment can serve as a source of freshwater to the Cove. For short term time series this can mask groundwater discharge. Estimates of freshwater discharge on days when there was no precipitation ranged from 4.1 to 8.9 × $10^4\,m^3/d$. These estimates are not adjusted for net changes in salinity within the Cove, reflecting changes in freshwater storage in the water column.

Another way to evaluate these estimates is to compare them to estimates of freshwater inputs using Darcy calculations. However, Darcy calculations are difficult to make in coastal areas because the upward bending of the flow lines at the fresh-seawater interface constricts the groundwater flow into a narrow zone where flows are greatly increased. Therefore, the effective discharge area of the aquifer is greater than the 1–2 m we directly measured. This calculation does give us a lower bound upon which to check our other numbers. The slope of the water table to the south and east of the Cove are quite similar (Table 1). In the vicinity of Town Center slopes are nearly twice as great, perhaps as a consequence of the steeper surface topography. Hydraulic conductivity of soils in this area ranges from 69 to 91 m/d (Guswa & LeBlank 1981). If we assume the maximum slope (0.007) and the maximum discharge thickness (2 m), we calculate a discharge of 0.79 × $10^4\,m^3/d$. As expected, this is considerably lower than any of our other estimates indicating that we have not grossly underestimated groundwater flow.

Surface water inflow

There is a permanent flow of water from two small ponds which drain under the highway in a sewer pipe. Discharge from this pipe was estimated at 0.037 × $10^4\,m^3/d$ based upon 10 measurements. Runoff from other storm sewers occurred only during and shortly after precipitation events. Runoff from the soils of Cape Cod has been estimated as 7.1 to 22 cm/y (Strahler 1972), although paved roads and other surfaces can effectively increase this amount.

For our budget we calculated the estimated daily freshwater input from runoff assuming three runoff values; 7, 22, and 29 cm/y. Runoff volumes into the Cove were estimated to range from 0.15 to 0.62 × $10^4\,m^3/d$.

Nitrogen in groundwater and surface water

The dominant form of inorganic nitrogen in groundwater samples was nitrate. Ammonium and nitrate made up less than 5% of the total dissolved inorganic nitrogen (DIN) in all of the inland wells we sampled. Nitrate was also the dominant form of inorganic nitrogen in wells, seeps and springs around the Cove but ammonium concentrations were higher, making up an average of 13% of the total DIN. Ammonium concentrations were highest in wells and seeps near septic systems where ammonium made up as much as 50% of the DIN.

Nitrate concentrations in well water and seeps varied from 0.0 to 0.75 mM with widespread but sporadic values greater than 0.21 mM occurring in Orleans and Eastham. On average, nitrate concentrations were highest within a 1 km radius of Orleans Town Center (Fig. 3). Here nitrate concentrations averaged 0.21 mM (S.E. ± 0.05, $n = 17$). Half the wells in this area exceeded 0.16 mM.

In terms of the areal distribution of nitrate in domestic and other shallow wells, the data from the Orleans Quadrangle are similar to Cape Cod in general, in that comparatively high groundwater nitrate values are widespread but sporadic (Fig. 4). A frequency distribution of nitrate concentrations in Cape Cod groundwater indicates that water from more than half of the wells contained less than 0.03 mM nitrate, 90% contained less than 0.28 mM nitrate, and 4% exceeded the EPA standard of 0.71 mM (10 ppm). The frequency distributions of nitrate in wells from Orleans and Eastham were similar, with about half of the well containing less than 0.03 mM, 90% containing less than 0.24 mM nitrate and 1.7% exceeding the EPA standard.

Groundwater nitrate concentrations in the more densely populated area of Town Center, Orleans are characterized by an absence of low values more than by the presence of exceptionally high values. For example, 23% of the nitrate values for Cape Cod as a whole lie between 0.07 and 0.39 mM; for the Orleans Quadrangle the comparable value is 30%, and within Town Center area, 65% of the wells fell within this concentration range.

Dissolved inorganic nitrogen concentrations in storm sewers ranged from 0.01 to 0.24 mM. There was a trend toward decreasing nitrogen with increasing rainfall. Total DIN averaged 0.09 mM and was almost equally divided between nitrate and ammonium. This value is approximately half of the DIN concentration measured from 19 storm drains across the Cape (CCPEDC 1978).

Salinity and inorganic nitrogen in porewater

We found two patterns of nitrate and salinity in porewaters sampled in transects from the shore. In sandy areas where there were very low concentrations of

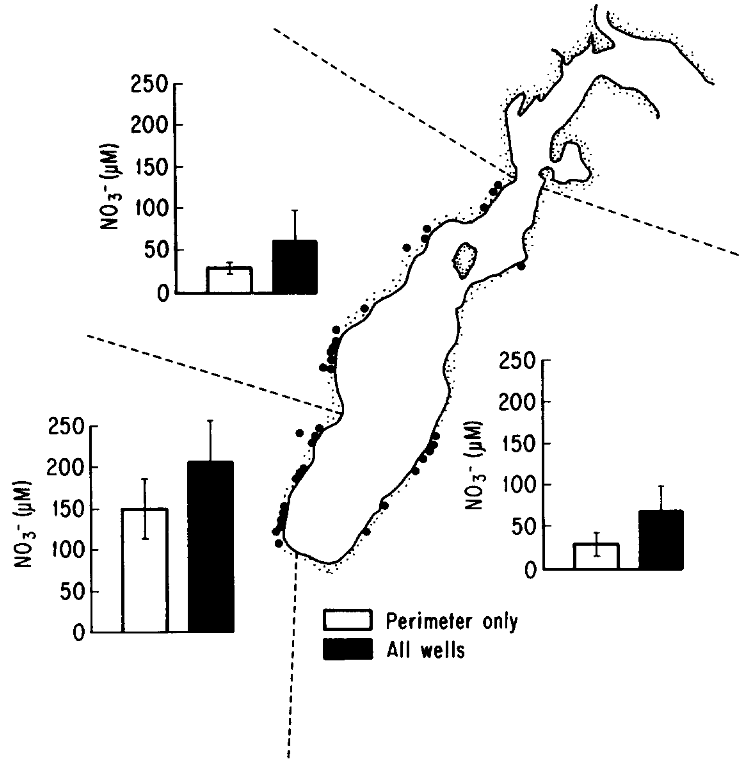
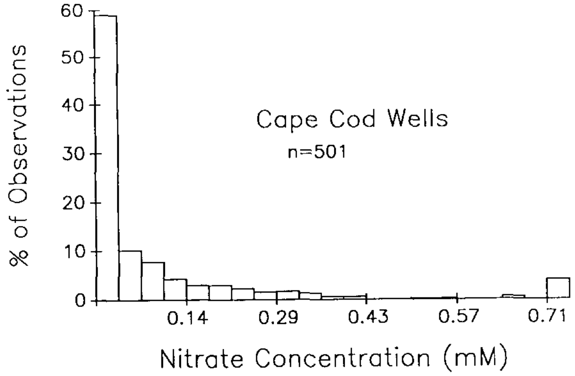


Fig. 3. The concentration of nitrate in groundwater around Town Cove, Orleans, Mass. The recharge area was divided into three zones, Eastham shore (top), Orleans Town Center (bottom left) and Tonset/Weeset shore (bottom right). The dots show the location of wells and seeps sampled around the perimeter of the cove. Nitrate concentrations were also measured in other wells within the recharge area. Nitrate concentrations (mean ± SE) for three areas are shown for both the perimeter, and for all wells within each of the three zones.

carbon in the sediments (0.01–0.2%) we found very steep salinity gradients in the porewater indicating high groundwater flow. In these areas, we observed an inverse relationship between salinity and nitrate concentration (Fig. 5 top). A plot of groundwater nitrate concentration vs. salinity from these sites revealed that most of the values fell on a conservative mixing line. This suggests that the major process influencing nitrate concentration in porewater was the dilution of high nitrate groundwater with low nitrate seawater.

In muddy areas around eelgrass beds, salinity gradients were less steep or absent, indicating rates of groundwater flow were very low or zero. At these sites we observed no relationship between porewater salinity and nitrate (Fig. 5 bottom), or a tendency for nitrate to increase with salinity (Fig. 6). A partial explanation for the lack of a decrease in nitrate with dilution can be seen when both nitrate and ammonium profiles are examined (Fig. 6). Along with the



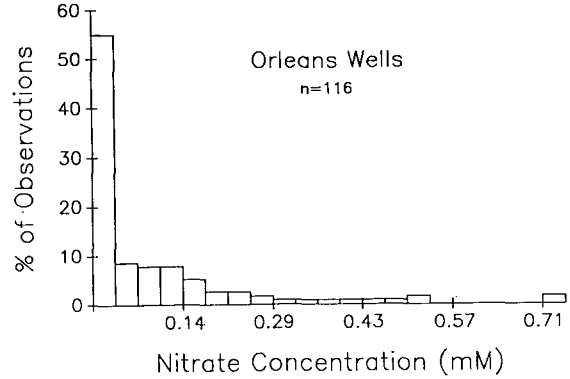


Fig. 4. The distribution of nitrate in groundwater samples from Cape Cod (CCPEDC 1978) and the Town Cove recharge area (this study).

increase in nitrate there is a decrease in ammonium suggesting that nitrate is being produced in the sediments by the oxidation of ammonium. Another possible reason for the lack of relationship between nitrate and salinity is the production of inorganic nitrogen by mineralization. When corrected for dilution by seawater, many stations showed an increase in the total DIN present in the sediments and an increase in phosphorus which indicates nutrients are being released from the sediments. This addition represents the recycling of nutrients already present in the Cove and does not represent a source of new nitrogen.

From these observations we see no evidence for significant denitrification in the sandy areas around the perimeter of the Cove where groundwater flow is high. The observation that nitrification is occurring in many areas would suggest that conditions in these sandy sediments are not suitable for denitrification.

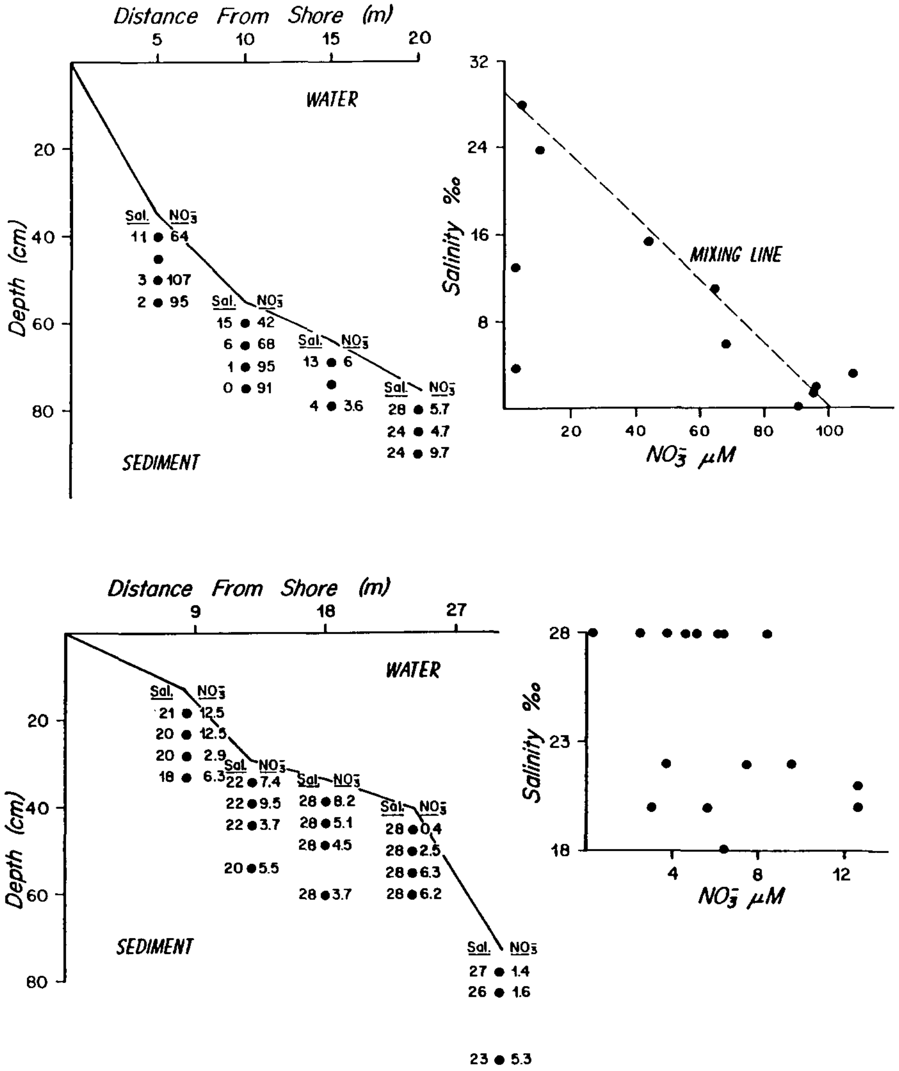


Fig. 5. Salinity and nitrate concentrations in the sediments of Town Cove. Samples were taken on a transect from inshore to offshore. Dots (on left) show the depth of samples (5 cm intervals) and their approximate distance offshore. *Top*–area of high groundwater flow. *Bottom*–area where groundwater flow was too low to measure (see text for details). Right hand graphs show the salinity nitrate relationship in porewater and the theoretical relationship (dotted line) if dilution alone is responsible for the observed concentration.

Nitrogen inputs into Town Cove from groundwater and surface water

We calculated a range of nitrogen inputs from groundwater to Town Cove using the freshwater inputs calculated from each of the three methods; the water budget, the seepage meters, and the tidal fluxes. Because we did not see evidence

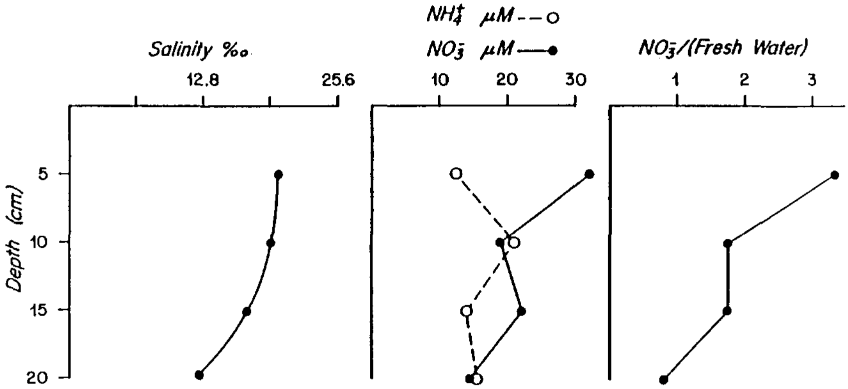


Fig. 6. Profiles of salinity, ammonia, and nitrate in sediment porewater. The ratio of nitrate/freshwater is defined as [nitrate/(30-salinity)].

for a large loss of nitrogen as groundwater passed through the sediments into the Cove we used groundwater nitrogen concentrations found in the wells and seeps (Table 4). For the purpose of these calculations we assumed that the discharge was uniform around the Cove. We did not include ammonium in our estimate of nitrogen input. A lower estimate of nitrogen inputs is obtained when the nitrate concentration in shore wells is used than when an average is taken of all wells (Table 4).

Nitrogen inputs from storm water flow contribute from 1.8 to 7.7 Kg N/d to

Table 4. Estimates of nitrogen loading from groundwater based upon different estimates of freshwater discharge and groundwater nitrate content.

	Estimates of nitrogen loading from groundwater	
	Nearshore wells	All wells
	Kg NO_3^--N/d	
Method		
Recharge		
33 cm/y	9.1	15
45 cm/y	11.6	19
Seepage meters		
1 m max. depth	41	68
2 m max. depth	49	80
Tidal flux		
All measurements		
26.5	42.7	
Dry days	68.6	110.8
Estimation of domestic and commercial use		
Fertilizer = 3 (hardware survey) to		9 (CCPEDC, 1983)
Septic systems = 24		24 (CCPEDC, 1983)
Total = 27		33

Town Cove. This is less than groundwater inputs, but large enough to make a significant contribution to the overall nitrogen budget of Town Cove.

Other calculations of nitrogen loading

Based upon fertilizer sales and interviews with local residents, our estimate of nitrogen loading from lawn fertilizer to Town Cove is 3.3 Kg N/d (assuming 60% leached). An independent estimate of fertilizer input by Belfit (CCPEDC, pers. comm.), based upon an average fertilizer use of 4 Kg/y per household, was 9.6 Kg/d.

Nitrogen loading from commercial and domestic septic systems was estimated by Belfit (CCPEDC, pers. comm.) using town water use records and population estimates taken from a 1974 survey and adjusted for growth. The nitrogen content of wastewater from commercial establishments was assumed to be 20 mg/l. The nitrogen contribution from domestic septic systems was estimated to be 6.8 Kg N/y per residence. The contribution of wastewater discharge was estimated as 24 Kg N/d for the entire recharge area and 9 Kg N/d for the Orleans Town Center area (Belfit CCPEDC, pers. comm.). Therefore the total nitrogen loading from groundwater is calculated to be from 27 to 33 Kg N/d (Table 4).

The volume weighted concentration of DIN in precipitation was 0.016 mM (NADP 1982–3). Precipitation during our study year was above average, 161 cm, as opposed to the long term average between 1951 and 1980 of 119 cm. We calculate the inorganic nitrogen input from precipitation directly falling into the Cove to be about 2 Kg N/d.

Uncertainties in calculating nutrient loading

Making calculations of nutrient loading from groundwater involves a number of uncertainties. One of our objectives was to evaluate which uncertainties would lead to large errors vs. those that only had minor effects on calculated loading rates. We identified four major areas of uncertainty:

1. The nitrogen concentration of the groundwater

While measuring nitrogen concentration is apparently straightforward, groundwater nitrate concentrations exhibit variation over several orders of magnitude. This makes it necessary to make a large number of measurements. However, we found that by sampling domestic wells we were able to obtain a sufficient sample size to make this a minor source of uncertainty (< ±20%). A more difficult problem is determining the appropriate population of wells to use to calculate inputs. By limiting our sampling to wells around the perimeter we would obtain a nitrogen concentration for groundwater that is only 60% the value measured for the recharge area as a whole. This difference may be due to a slow rate of groundwater movement relative to watershed development. Groundwater around the Cove is several years old and was formed when building density was lower. However, we cannot rule out the possibility that there has been some loss of nitrogen in the groundwater during travel.

2. Before the magnitude of nitrogen loading from groundwater can be calculated it is necessary to know the fate of groundwater nitrate as it passes through the sediment into the marine environment

If extensive amounts of the nitrate are denitrified then nitrogen concentrations in wells will not accurately represent the amount of nitrogen reaching the marine environment. Denitrification in groundwater has been documented (Trudell et al. 1986; Slater & Capone 1987); however, only a few studies have examined the fate of nitrate as groundwater passes through the sediment–water interface. Our data indicate that in sandy sediments, where groundwater flow rates range from 1 to greater than $10\,l/m^2/h$, nitrate appears to behave conservatively. Capone & Bautista (1985) have also observed that groundwater derived nitrate in sandy sediments appears to be governed largely by dilution with seawater. However, denitrification was occurring in deeper sediments (Slater & Capone 1987). We conclude that in Town Cove, Orleans, denitrification does not remove a major portion of the nitrate entering the Cove from groundwater. Therefore, calculations of groundwater nitrogen loading can be made using groundwater well data. We suspect that this will be the case for many areas where sandy sediments allow appreciable flows of oxic groundwater. In areas where flow rates are low, and sediments obtain appreciable quantities of organic carbon, denitrification may be an important removal mechanism. However, due to the low groundwater flow rates, nitrogen entering from groundwater is apt to be much less important to the overall nitrogen budget.

3. Groundwater flow rates must be determined

This involves two problems because, first, groundwater flow may actually vary seasonally or annually with precipitation patterns and, second, both the direct measurement, and the indirect calculation of groundwater flow requires making a number of assumptions which may not be valid. In our study we found that well elevations did vary but that this variation was nearly always less than ±35% of the mean, and less than 15% for seasonal differences. By far the greatest uncertainties came from estimating groundwater flow using different methods. Overall, there was a five fold difference between the minimum estimate of groundwater flow obtained by calculating flux using a water budget (and using minimum recharge estimates) and the maximum estimate obtained by measuring freshwater flux at the Cove mouth (using the higher estimates obtained on clear days). This illustrates the difficulty of determining groundwater flow. However, it is possible to contain the value somewhat by considering the possible errors and by determining which methods yield maximum and minimum estimates.

As previously discussed, our seepage meters represent maximum values because some of the measured flows included the trapping of seawater by the chambers as well as freshwater entering the chambers. In order to be used effectively seepage meters must be deployed over all stages of the tidal cycle and the salinity drop of the chambers must be measured. This adds considerably to the difficulty of making the measurements and requires that the chambers be stirred. Our very limited salinity data set indicates that our values for freshwater flow may be overestimated by as much as 50%.

Tidal flux calculated from tidal exchange and salinity data is subject to errors from changes in the total freshwater stored in the estuary. For another water body we are studying, changes in storage can result in a factor of two error in the discharge calculations. Therefore, our calculations made on dry days should be taken as maximum estimates. In addition, tidal exchange in Town Cove is complicated by the hydrodynamics of the inlet (Aubrey & Speer 1985). A better estimate could be obtained with a longer data set.

Fewer assumptions go into the calculations of freshwater inflow using groundwater recharge. The range of recharge value is reasonably well known for Cape Cod (Strahler 1972) so the major uncertainty is in determining the recharge area.

In spite of the uncertainties, the majority of the data suggests that the magnitude of groundwater nitrogen input into Town Cove is in the range of 15–50 Kg N/d. By disregarding the extreme values, we obtain an average of 35 Kg N/d. This is quite close to the estimate obtained by calculating nitrogen loading by estimating commercial and domestic nitrogen disposal in the watershed indicating that simple indirect measures may be adequate to answer management questions.

4. Both nitrate concentrations and groundwater flow may show strong spatial variations making loading difficult to assess from average measurements

The nitrogen concentration of the groundwater near the center of Town was significantly ($p > 0.05$) greater than in other areas around the Cove. This is consistent with other studies which have observed that groundwater nitrate concentrations in Cape Cod groundwater are related to building density (Perskey 1988). In addition the differences in groundwater slope around the Cove suggest that the flow of freshwater into the Cove is not likely to be uniform. The steepest slope to the groundwater table is in the vicinity of the center of Orleans, which is also the area of highest nitrogen concentrations. If groundwater flow from the center of the town of Orleans is twice as fast as in other areas around the cove our estimates of nitrogen loading will be low by 30%.

Importance of nitrogen in groundwater

Groundwater nitrate can represent a significant source of nitrogen to coastal embayments and estuaries. When adjusted for the volume of the ecosystem, the rate of nitrogen loading to Town Cove from groundwater exceeds the nitrogen loading from sewage discharge reported for many large river dominated estuaries (Table 5). The total inputs of nitrogen from freshwater sources to Town Cove (groundwater and runoff) total 361 mmol/m^3/y, higher than freshwater nitrogen loadings reported for the Pamlico River and San Francisco Bay (Table 5).

Table 5. A comparison of the nitrogen input from freshwater sources to a variety of estuaries. Orleans data from this study, all other data compiled by Nixon & Pilson (1983).

		Freshwater nitrogen inputs		
	Vol m^3 (10^9)	(mmol N m^3/y) Land	Sewage	Total
Chesapeake	74,000	50	30	80
Narragansett	2,200	60	40	100
Delaware	1330	70	70	140
Potomac	7150	80	60	140
Pamlico	980	250	< 5	250
S San Francisco		160	130	290
N		< 5	310	310
Raritan		50	280	330
New York		800	3750	4550
		Road runoff	Groundwater	
Orleans	0.003	57	304	361

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