

GROUND WATER DISCHARGE AND THE RELATED NUTRIENT AND TRACE METAL FLUXES INTO QUINCY BAY, MASSACHUSETTS

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Abstract. Measurement of the rate and direction of ground water flow beneath Wollaston Beach, Quincy, Massachusetts by use of a heat-pulsing flowmeter shows a mean velocity in the bulk sediment of 40 cm d^{-1} . The estimated total discharge of ground water into Quincy Bay during October 1990 was $1324\text{--}2177 \text{ m}^3 \text{ d}^{-1}$, a relatively low ground water discharge rate. The tides have only a moderate effect on the rate and direction of this flow. Other important controls on the rate and volume of ground water flow are the limited thickness, geographic extent, and permeability of the aquifer. Comparisons of published streamflow data and estimates of ground water discharge indicate that ground water makes up between 7.4–12.1% of the gaged freshwater input into Quincy Bay. The data from this study suggest the ground water discharge is a less important recharge component to Quincy Bay than predicted by National Urban Runoff Program (NURP) models.

The high nitrate and low nitrite and ammonia concentrations in the ground water at the backshore well sites and low nitrate and high nitrite and ammonia concentrations in the water flowing from the foreshore suggests that denitrification is active in the sediments. The low ground water flow rates and low nitrate concentrations in the foreshore samples suggest that little or no nitrate is surviving the denitrification process to affect the planktonic community. Similarly, oxidizing conditions in the aquifer and low trace metal concentrations in the ground water samples suggest that the metals may be precipitating and binding to sedimentary phases before impacting the bay.

1. Introduction

Ground water discharge has been shown to be an important source of dissolved chemical constituents in many shallow coastal environments (Valiela *et al.*, 1978; Johannes, 1980; Capone and Bautista, 1985). It is widespread in coastal settings (Pomroy *et al.*, 1983; Capone and Slater, 1990; Gaines *et al.*, 1983; D'Elia *et al.*, 1981; Valiela and Costa, 1988) and appears to occur anywhere an aquifer with a head above water level is hydraulically connected to that body of water through permeable bottom sediments. In contrast to surface runoff which occurs at discrete points and at high velocities, ground water discharge occurs more slowly, but over a much more extensive area along the shoreline. At least during the dry season, ground water seepage may often be the largest source of freshwater to many embayments (Bokuniewicz, 1980).

A recent computer search of the literature uncovered no references directly measuring the discharge of ground water into Boston Harbor, indicating that little is known about the rate and effect of ground water discharge on this estuary. The ground water in densely settled coastal areas may acquire elevated nutrient and trace metal concentration from a number of anthropogenic sources (Johannes, 1980). For example, sources potentially effecting nutrient concentrations in ground

waters include peaking septic/sewer systems, land fills, and fertilizer applications. This nutrient-loading of the ground waters can eventually stimulate the microbial activity in the soils through which the ground water flows promoting denitrification and possibly increasing water column productivity by planktonic and benthic algae after discharge into surface waters (Slater and Capone, 1987). This productivity can in turn lead to higher algal decomposition rates and significant O₂ deficits in both the bottom waters and sediments (Jorgenson and Sorensen, 1988).

The purpose of this study was to determine at what rate ground water and the associated dissolved chemical constituents are percolating through the Wollaston Beach sediments. Because the influence that ground water flow exerts in transporting contaminants into the Boston Harbor estuary has not been previously measured, the results presented here should be considered during implementation of any future environmental models.

2. Study Area

Quincy Bay is located southeast of the inner harbor of Boston, Massachusetts (Figure 1). The bay, which has a tidal range of about 2.90 m and an average depth of 2.09 m at mean low water, is bordered by tidal flats and beaches that are variably composed of tan to brown, siliciclastic muddy fine sand to gravel. These sediments are primarily reworked glacial drift that forms a thin layer covering a basement of Cambrian and older igneous and metamorphic rocks. Input from surface runoff is limited to seven storm drain pipes and two small streams: Blacks Creek (Furnace Brook) and Sachems Creek. The arcuate shape of the bay, offshore position of Long, Hangman, Rainsford, and Peddocks Islands, and gently sloping nature of the bay bottom combine to dissipate wave energy and allow the accumulation of fine sediments offshore.

Quincy Bay was selected as our study area for two reasons. First, although the oceanographic and sedimentological conditions here are generally similar to those of other embayments in the Boston Harbor estuary, these other bays are associated with large point-sources of freshwater. These point-sources include the Mystic, Charles, and Chelsea Rivers flowing into the Boston inner harbor, the Neponset River flowing into Dorchester Bay, and the Weymouth Fore, Weymouth Back, and Weir Rivers flowing into Hingham Bay. The lack of a major point-source of surface runoff into Quincy Bay enhances the relative importance of ground water discharge as a source of anthropogenic contaminants. Second, recent studies have shown that the sediments, shellfish, and winter flounder in Quincy Bay are contaminated with trace metals (Reimold *et al.*, 1988). Inasmuch as Wollaston Beach, which borders much of Quincy Bay, is an important recreational area during the summer months, examination of the ground water discharge around Quincy Bay is necessary to determine if ground water is a significant source of these contaminants.

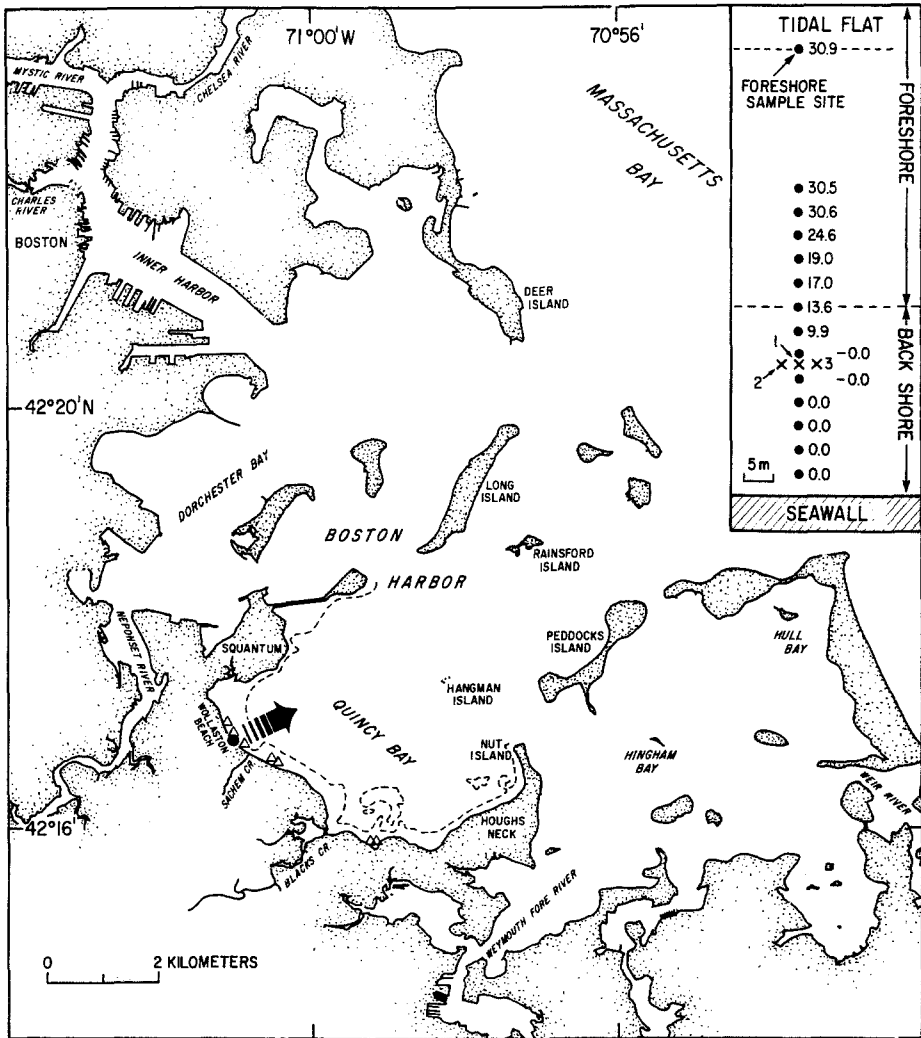


Fig. 1. Map of Quincy Bay showing location of the Wollaston Beach study site (solid circle), Blacks Creek, Sachems Creek, and the storm drains (open triangles). Arrow shows the average direction of ground water flow; dashed line shows the seaward edge of the tidal flat in Quincy Bay at mean low water. The distribution of the salinity stations (solid dots), well sites (x), and foreshore station are shown in the inset. Salinity values along the beach transect (‰) are from samples collected at low tide during a September 1990 beach survey prior to well emplacement.

3. Methods

Sampling was conducted along a transect perpendicular to the mean low waterline of Wollaston Beach located in south-western Quincy Bay (Figure 1). Site selection of the transect was based on lithologic, geographic, and topographic considerations. For example, sections of beach composed mostly of gravel were avoided because

this lithology would complicate well emplacement and the flow measurements. Ground water samples for salinity analyses were collected at both high and low tides from hand-augered wells spaced at 4 m intervals along this transect to locate the zone of diffusion and to determine if this zone of mixing between the fresh ground water and saline bay water migrated extensively with the changing of the tides. Samples were also collected from the foreshore at the boundary between the tidal flat and the beach face to examine the chemical constituents of the water flowing from the beach sediment at low tide.

Three shallow monitoring wells were hand-augered to bedrock using a 9.4 cm (3.75") O.D. bucket. The bedrock was encountered about 152 cm below the beach surface. Ten centimeter (4") I.D. PVC tubes were used as casings during the hand-augering to prevent caving and removed upon completion of the wells. These wells, which were used to determine the rate and direction of ground water flow and to sample for chemical constituents, were spaced 3 m apart in a line perpendicular to the transect at a point 12 m above mean high tide and 5 m above the shoreward extent of the zone of diffusion. The wells were completed with washed 5 cm (2") I.D. Schedule 40 PVC fitted with 76 cm of 0.010 gauge PVC well screen. A pack of 1 ϕ sand was used around the well screen in all wells; 52 cm risers brought the wells to an approximate depth of 30 cm below ground surface to conceal their locations during periods of sampling inactivity. Flow measurements were taken every 1–1.5 hours over a 12 hour period on October 3, 1990 using KVA Model 40 and Model 30 heat-pulsing flowmeters. During sampling the wells contained 71 cm of water; the bottoms of the flowmeter probes were positioned 60 cm below ground water level. The methods used for well construction, flowmeter calibration, and direct ground water bulk sediment flow velocity measurement are described by Kerfoot (1988). Earlier laboratory tests and field comparisons of heat-pulsing flowmeter results with results generated by more traditional chemical tracer and well pumping techniques have yielded similar flow velocities and directions (Melville *et al.*, 1985; Guthrie, 1986).

Each well was evacuated three times to eliminate stagnant conditions (Office of Water Data Collection, 1977) prior to collecting the samples for chemical analysis on November 20, 1990. Standard sampling and analytical methods for nitrate (Wood *et al.*, 1967), nitrite (Bendschneider and Robinson, 1952), phosphate (Murphy and Riley, 1962), silicate (Koroleff, 1976), and ammonia (Solorzano, 1969), were used to generate the nutrient data. These analyses were performed on a Technicon autoanalyzer. The water samples were analyzed for dissolved Cd, Cr, Cu, Fe, Pb, and Zn by graphite furnace atomic absorption spectroscopy, using a Perkin-Elmer (Model 5000) AA spectrophotometer. Three readings were obtained for each sample in order to determine signal reproducibility. One sample was analyzed using the system of standard additions as described in the Perkin Elmer analytical methods, in order to rule out matrix interferences which may suppress the signal. Samples were irradiated with a mercury vapor lamp in order to destroy organic complexes and surfactants which would interfere with the measurements

(Van Den Berg *et al.*, 1987). All spectroscopic analysis were carried out within linear response ranges and were calibrated using acidified commercial standards. Foreshore samples were analyzed using anodic stripping voltammetry (Van Den Berg *et al.*, 1987) because of the elevated salinities. Sampling equipment and containers for the trace metal analyses were prewashed, soaked overnight in 50% HNO₃, rinsed in deionized water, and pore water rinsed at each site. All sampling and analytical methods used to generate the trace metal data are approved by the U.S. Environmental Protection Agency (Kopp and McKee, 1979). Blanks were processed in the field and laboratory during the nutrient and trace metal analyses in order to assess contamination and analytical error. The results of the analyses on the blanks are given in Table II and show that field and laboratory contamination and analytical error are low. The nutrient and trace metal samples were immediately filtered and frozen in the field within a cooler containing dry ice. Subsequently, data obtained for each element were subjected to comparison with a pre-prepared standard curve for that specific element. Water samples from one of the wells and the foreshore were collected and analyzed in triplicate to check sampling and analytical variability.

Salinity measurements were made with a Endeco type 102 refractometer. Both pH and dissolved oxygen data were collected on February 4, 1991. The pH analyses were conducted and dissolved oxygen samples fixed in the field; the iodometric method (Winkler, 1888) was used to measure dissolved oxygen. Grain size analyses were performed on well samples from the aquifer using a Coulter Counter and rapid sediment analyzer.

4. Results

The results of the flowmeter survey and chemical analyses are presented in Tables I and II. The average bulk sediment flow velocity in the three wells is 40 cm d⁻¹ (1.32 ft d⁻¹), but individual flow measurements range between 21 cm d⁻¹ and 52 cm d⁻¹. These changes in the flow rates appear to be related to the tidal cycle. The lowest flows invariably occur within one half hour after high tide; the greatest flows are recorded within one half hour after low tide. The average net flow direction for the three wells is 65° True North; a flow to the northeast that is directly toward Quincy Bay. Although flow direction remained relatively constant and did not reverse during the tidal cycle, tidal changes deflect the flow directions. Incoming tides deflect the flows toward the east; flows during outgoing tides shift more toward the north. These tide-related shifts are between 1–5° from the mean flow direction in each of the monitoring wells.

Salinities of the ground water in the test wells are variable despite the results from salinity surveys that showed that any migration of the zone of diffusion was less than the 4 m sampling interval. Salinity of the ground water at the wellsites was at or near zero when the beach was surveyed during well emplacement (Figure 1) and < 2‰ during the nutrient and trace metal sampling (Table I). The low

TABLE I

Results of the textural, salinity (SAL), dissolved oxygen (DO), pH, and flowmeter analyses from Wollaston Beach wells and Quincy Bay. Mean flow rates and directions are based on ten measurements. Hyphens indicate no data; flow directions are in degrees from true North.

Sample	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	SAL (°/oo)	DO (mg/l)	pH	Max. flow (cm d ⁻¹)	Max. flow (deg.)	Min. flow (cm d ⁻¹)	Min. flow (deg.)	Mean flow (cm d ⁻¹)	Mean flow (deg.)
Well#1	10.76	75.73	12.49	1.03	1.1	11.16	6.01	52	89	40	99	46	99
Well#2	8.89	76.73	13.36	1.02	1.3	11.14	6.12	49	37	21	40	34	37
Well#3	13.73	75.16	9.78	1.32	-	11.62	6.50	52	51	34	52	43	52
Foreshore-1	-	-	-	-	30.9	11.98	7.08	-	-	-	-	-	-
Foreshore-2	-	-	-	-	31.0	-	-	-	-	-	-	-	-

TABLE II

Results of the dissolved nutrient and trace element analyses on Wollaston Beach wells and Quincy Bay. Hyphens indicate no data; the sample blanks are deionized water. Values marked with an asterisk were determined by the anodic voltametric method (Van Den Berg, 1987). Salt water interferences reduce the confidence levels for the iron values in the foreshore samples.

Sample	N-NO ₂ (μM l ⁻¹)	N-NO ₃ (μM l ⁻¹)	N-NH ₃ (μM l ⁻¹)	Silica (μM l ⁻¹)	Phosphate (μM l ⁻¹)	Cd (μg l ⁻¹)	Cr (μg l ⁻¹)	Cu (μg l ⁻¹)	Fe (μg l ⁻¹)	Pb (μg l ⁻²)	Zn (μg l ⁻¹)
Well#1 A	0.11	140.25	0.39	6.60	6.25	< 1	< 1	4.0	17.6	< 5	20.0
Well#1 B	0.10	146.29	0.53	6.32	6.69	< 1	< 1	5.1	16.8	< 5	15.2
Well#1 C	0.10	152.32	0.40	8.84	5.29	< 1	< 1	4.3	12.5	< 5	16.5
Well#2	0.48	136.44	1.33	6.39	15.41	< 1	< 1	3.9	17.2	< 5	15.2
Well#3	0.05	154.09	0.50	-	4.05	< 1	< 1	5.1	21.9	12	16.0
Foreshore-1	0.86	11.31	10.36	15.75	1.78	< 1*	< 1	4.0*	87.5	< 5*	5.0*
Foreshore-2	0.88	10.12	10.54	15.72	1.63	< 1*	< 1	4.5*	126.6	< 5*	5.2*
Foreshore-3	0.85	9.07	10.42	20.46	1.68	-	-	-	-	-	-
Blank-1	0.05	6.53	0.25	0.42	0.07	< 1	< 1	< 1	< 1	< 5	< 1
Blank-2	0.05	6.39	0.27	0.39	0.07	< 1	< 1	< 1	< 1	< 5	1.2
Blank-3	0.05	6.67	0.28	0.35	0.07	-	-	-	-	-	-

salinities recorded during the nutrient and trace metal sampling were consistent across the back shore and are therefore probably the result of sea spray produced under windy conditions.

Dissolved oxygen values in the ground water at the wellsites average 11.14 mg l⁻¹; a value of 11.98 mg l⁻¹ was determined for the water flowing from the foreshore (Table I). These data correspond with average surface water values reported for the Charles River basin during the same time of year of 12.9 mg l⁻¹ (Gadoury *et al.*, 1988).

Inasmuch as sea water is buffered, ground waters, especially those containing any CO₂ from rotting organics, are expected to be more acidic than seawater. This relationship is reflected by pH values that average 6.21 in the ground water and was 7.08 in the foreshore sample. These values are lower than those reported earlier for water samples collected from the center of the bay during October 1984 and October 1985 that averaged 7.8 (Massachusetts Division of Water Pollution Control, 1984; Massachusetts Division of Water Pollution Control, 1986).

A distinct pattern is present in the nutrient data (Table II). Nitrate and phosphate are enriched in the ground water at the wellsites relative to the water flowing from the foreshore. Nitrate averages 145.90 $\mu\text{M l}^{-1}$ in the well samples, but only 10.14 $\mu\text{M l}^{-1}$ in the foreshore samples. Phosphate concentrations in the ground water average 7.54 $\mu\text{M l}^{-1}$; PO₄ averages 1.70 $\mu\text{M l}^{-1}$ in the foreshore samples. Conversely, concentrations of ammonia, nitrite, and silica are lower in the ground water at the wellsites than in the foreshore samples. Ammonia averages 0.63 $\mu\text{M l}^{-1}$ in the well samples, but increases in the foreshore samples where it averages 10.44 $\mu\text{M l}^{-1}$. Nitrite averages 0.26 $\mu\text{M l}^{-1}$ in the ground water at the wellsites, and 0.87 $\mu\text{M l}^{-1}$ in the foreshore samples. However, the difference between the average ground water and foreshore nitrite concentrations would be even greater if the questionably high value (0.48 $\mu\text{M l}^{-1}$) of the well 2 sample were discounted.

Concentrations of dissolved trace metals in the ground water at the wellsites and foreshore samples are low (Table II). The Cd, Cr, and, with the exception of well 2, Pb contents are below the detection limits. The average concentrations of Cu, Fe, and Zn in the ground water samples are 4.7, 17.2, and 16.6 $\mu\text{g l}^{-1}$, respectively. Concentrations of Cu and Zn are greater in the ground water samples than in the foreshore samples; Fe concentrations are greater in the water flowing from the foreshore where it averages 107.1 $\mu\text{g l}^{-1}$. These levels are far below the maximum allowable limits for trace metals in natural waters as determined by the Environmental Protection Agency (American Public Health Association, 1980).

5. Discussion

The tides in Quincy Bay have only a moderate effect on the ground water flow beneath Wollaston Beach. As sea level drops 3 m during the change to low tide, the head driving the ground water flow is increased and the seaward migration of interstitial pore waters is enhanced. This effect is reflected by the higher flow rates at low tide. During the change to high tide, the head driving the ground water flow is decreased and the flow rates drop. Although these flow rates and directions change moderately with the tides, they do not reverse at the wellsites and the zone of diffusion does not migrate extensively across the beach. Factors limiting the influence of the tidal cycle on the flow rates and directions beneath Wollaston Beach probably include the fine-grained nature of the soil and the hydraulic gradient of the aquifer. The pore waters from the aquifer eventually mix with the Quincy Bay water permeating the beach sands and subsequently emerge from these sediments

as intertidal springs that coalesce into the wide dendritic drainage channels characteristic of the tidal flats and, presumably, subtidal seeps. Inasmuch as cobbles and finer-grained patches of sediment capable of deflecting the path of flow were encountered during well emplacement, the different base flow rates and directions measured in each well at any given time are probably due to the inhomogeneous sedimentary phases within the aquifer.

The total daily discharge of ground water into Quincy Bay can be roughly estimated if we assume that the bulk sediment flow velocity (40 cm d^{-1}) and thickness of the saturated water table (71 cm) measured at the wellsites remains constant along the entire length of beach bordering the bay. If Squantum Neck and Hough's Neck are ignored because numerous bedrock outcrops suggest that any ground water discharge from the thin, patchy alluvial aquifers there is negligible (Kaye, 1980), the length of beach bordering on Quincy Bay is approximately 4.63 km . Based on these numbers, the discharge of ground water into Quincy Bay during the month of October 1990 is about $1324 \text{ m}^3 \text{ d}^{-1}$ ($46770 \text{ ft}^3 \text{ d}^{-1}$). However, high-resolution seismic-reflection data (Rendigs and Oldale, 1990) suggest that the average depth to bedrock along the Quincy Bay shoreline may be slightly greater (as much as 50 cm) than at the wellsites. If the thickness of the water table is increased by this entire amount, than our estimate of total daily discharge would rise to $2177 \text{ m}^3 \text{ d}^{-1}$ ($76877 \text{ ft}^3 \text{ d}^{-1}$). In any case, the average October streamflow for Furnace Brook (Wandle, 1984) is about $17981 \text{ m}^3 \text{ d}^{-1}$ ($635000 \text{ ft}^3 \text{ d}^{-1}$). Therefore, ground water discharge makes up less than 7.4% of the gaged freshwater input of Quincy Bay based on the physical parameters present at the wellsites and no more than 12.1% when corrected for a greater water table thickness. These percentages represent the maximum because they do not include the ungaged runoff from Sachem Creek or the storm drain pipes. Earlier estimates of the relative input from ground water into southern Boston Harbor (Menzie and Cura, 1990), which were based on the National Urban Runoff Program (NURP) model, reported ground water to be about 27% of the river input. The data from this study suggest that the NURP model overestimates ground water discharge into Quincy Bay. Factors restricting the rate of ground water discharge into Quincy Bay include the limited geographic extent and thin nature of the aquifer and the presence of fine-grained glaciolacustrine and glaciomarine deposits (LaForge, 1932; Kaye, 1980) of low hydraulic conductivity.

The estimates of surface runoff and ground water discharge are dwarfed by tidal flushing. About 41870 m^3 of water is pumped in or out of Quincy Bay with each tidal change. Although unpublished tidal models show that much of the Quincy Bay water drained on an outgoing tide returns with the next high tide (R. Signell, USGS, verbal communication 1991) this tidal flushing probably mitigates any effect from the ground water input.

The inverse relationship between high nitrate and low nitrite and ammonia concentrations in the ground water at the well sites and low nitrate and high nitrite and ammonia concentrations in the water flowing from the foreshore has

been interpreted elsewhere (Pregall and Miller, 1988; Capone and Slater, 1990) as evidence for bacterial denitrification, the biogenic reduction of nitrate to nitrite and, ultimately, ammonia or gaseous nitrogen (N_2 or N_2O). Johannes (1980) has shown that this process occurs primarily at and seaward of the zone of diffusion. However, at least some portion of this change in nutrient concentrations could be related to mixing of the ground water with relatively higher nitrite- and ammonia-bearing bay water that percolates into the beach at high tide. No nutrient data is available on Quincy Bay water just offshore from the well sites, but nitrate and nitrite values from water samples, which were collected during September and November 1990 from the mouth of Quincy Bay just north of Hangman Island (Robinson *et al.*, 1992), are only slightly lower than at the foreshore sample sites and range from $5-7 \mu M l^{-1}$ and $0.6-0.7 \mu M l^{-1}$, respectively. In any case, because denitrification is more efficient during low flow conditions (Capone and Slater, 1990), low ground water flow rates and low concentrations of nitrate in the foreshore samples suggest that little of this nutrient is surviving the interstitial denitrification process to impact the planktonic communities of Quincy Bay.

Earlier estimates of PO_4 concentrations in ground water used by Menzie and Cura (1990) to estimate loadings in Boston harbor are low compared to the data from this study. The results from this study are more in line with those of Valiela and Costa (1988) who found ranges of phosphate up to $12.8 \mu M l^{-1}$ and a mean annual concentration of $3.3 \mu M l^{-1} PO_4$ in the ground water flowing toward Buttermilk Bay, Massachusetts.

The dissolved oxygen values for surface, mid-depth, and bottom water samples collected from the middle of Quincy Bay during October 1985 average $9.9 mg l^{-1}$ (Massachusetts Division of Water Pollution Control, 1986). Although dissolved oxygen values in surface waters range between 3 and $12 mg l^{-1}$, there is typically much less in ground water because the bulk of dissolved oxygen in recharge waters is consumed in the vadose zone by the decomposition of organic matter and microbial respiration or rapidly thereafter in the aquifer by various mineral-water and organic-oxidative reactions (Winograd and Robertson, 1978; Zafiriou *et al.*, 1984). The relatively high dissolved oxygen values reported in this study for the ground water beneath Wollaston Beach and in other shallow-water-table aquifers in Massachusetts (Frimpter and Gay, 1979) suggest that either the oxygen depleting processes are less intense than in many other aquifers or that elevated concentrations of oxygen in the source waters and the shallow, thin nature of the aquifer beneath Wollaston Beach combine to promote oxidizing conditions.

Many metals that are mobilized in ground waters precipitate as they migrate into more alkaline and oxidizing interstitial environments (Price, 1967). Because iron is abundant and readily precipitated, it commonly forms oxide and oxyhydroxide coatings and stains on the sediment grains. Evidence for the presence of these ferroan precipitates on Wollaston Beach is the tan-brownish color of the bulk surficial, vadose, and aquiferous sedimentary phases. Subsequently, these Fe-rich coatings form a favorable catalytic substrate for the nucleation and precipitation

of other metals (Burns and Burns, 1975). Earlier work has shown that Cd, Cu, Pb, Zn, and many other transition metals exhibit strong sorptive binding affinities for these oxidate crusts and stains (Gorham and Swaine, 1965; Davies-Colley *et al.*, 1984).

Elevated concentrations of the elements analyzed for this study were detected in nearshore and mid-bay bottom sediment samples from Quincy Bay (Reimold *et al.*, 1988) and in water samples from the Charles and Neponset-Rivers (Walker *et al.*, 1975; Brackley *et al.*, 1973). Therefore, it is unlikely that the low concentrations of dissolved trace metals in the ground water preclude greater concentrations of these elements in the aquifer beneath Wollaston Beach, but suggest only that greater concentrations of these metals, if present, are probably in solid phases. Whether these metals are remobilized during periodic reductions in Eh and pH, transported toward the bay, and eventually discharged into the bottom waters or sediments is uncertain.

6. Conclusions

This study has revealed several important aspects regarding the discharge of ground water into Quincy Bay.

(1) The average direction of ground water flow is toward Quincy Bay, with a mean bulk sediment flow velocity of 40 cm d^{-1} . The tides cause only minor deviations in the rate and direction of ground water flow.

(2) Ground water discharge makes up less than 12.1% of the gaged freshwater input into Quincy Bay, and appears to be a less important recharge component than predicted by NURP models.

(3) The decreasing concentrations of nitrate across Wollaston Beach, in conjunction with an increase in nitrite and ammonia, suggest that denitrification may be an important process affecting nutrient concentrations in the ground water discharge around Quincy Bay. Although utilization of the ground water-sourced nitrate by the biota may elsewhere affect interstitial and, possibly, benthic zonation and productivity, the low ground water flow rates and extensive tidal flushing probably mitigate any effect on the Quincy Bay planktonic community.

(4) The concentrations of dissolved Cd, Cr, Cu, Pb, and Zn in the ground water flowing into Quincy Bay are low. The color of the sediments in the aquifer and the elevated dissolved oxygen values indicate that oxidative precipitation affects the concentrations of dissolved trace metals in the ground water beneath Wollaston Beach.

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