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The Effects of Tidal Export from Salt Marsh Ditches on Estuarine Water Quality and Plankton Communities

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Abstract Salt marshes are an important transition zone between terrestrial and marine ecosystems, and in their natural state, they often function to cycle or trap terrestrially derived nutrients and organic matter. Many US salt marshes were ditched during the twentieth century, potentially altering their functionality. The goal of this 4-year study was to assess the impact of water from ditches within seven salt marshes on estuarine water quality and plankton communities within four estuaries on Long Island, NY, USA. We found that concentrations of inorganic nutrients (ammonium, phosphate), dissolved and particulate organic nitrogen and carbon (POC, PON, DOC, DON), and total coliform bacteria were significantly enriched in salt marsh ditches compared to the estuaries they discharged into. In addition, concentrations of ammonium and DON became more enriched in ditches as tidal levels decreased, suggesting these constituents were generated in situ. Quantification of nitrogen sources in Flanders Bay, NY, suggested salt marsh ditches could represent a substantial source of N to this estuary during summer months. Experimental incubations demonstrated that water from salt marsh ditches was capable of significantly enhancing the growth of multiple classes of phytoplankton, with large diatoms and dinoflagellates displaying the most dramatic increases in growth. Experiments further demonstrated that salt marsh ditchwater was capable of significantly enhancing pelagic respiration rates, suggesting discharge from ditches could influence estuarine oxygen consumption. In summary, this study demonstrates that tidal draining of salt marsh ditches is capable of degrading multiple aspects of estuarine water quality.

Keywords Salt marsh · Estuary · Salt marsh ditch · Phytoplankton · Respiration · Nutrients · Organic matter · Coliform bacteria

Introduction

Salt marshes line the coastlines of many near shore environments and are the dominant intertidal habitat along the East and Gulf coasts of the USA (Reinold 1977; Mitsch and Gosselink 1993). Salt marshes serve as an important habitat for a variety of animals as multiple marine, terrestrial, and migratory species utilize these systems for food, shelter, and nurseries (Ursin 1972; Turner 1987; Leonard et al. 1999). Marine marshes can also serve as a buffer between the land and the adjacent marine ecosystem whereby terrestrially derived nutrients and organic carbon may be retained and re-mineralized, potentially minimizing the effects of these constituents on the local marine environment (Heinle and Flemer 1976; Valiela et al. 1978; Valiela and Teal 1979; Dame et al. 1992). Alternatively, older salt marshes or salt marsh systems with small tidal ranges can export labile dissolved and particulate organic matter and nutrients to deeper water, a concept known as the "outwelling hypothesis" first proposed by Odum (1968, 1969) and since then tested by a series of element flux studies (Axelrad 1974; Heinle and Flemer 1976; Haines 1976; Woodwell et al. 1977, 1979; Valiela et al. 1978 and Dame et al. 1986) as reviewed by Childers (1994).

The expansion of human populations along coastlines during the past century has led to the alteration and degradation of many salt marsh habitats, a process which,

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in turn, can impact estuaries. The draining of the marshes for land use, construction of bulkheads, or homes on salt marshes, and the creation of fish or salt ponds are all processes which have altered, degraded, or destroyed salt marsh habitats (Kennish 2001; Valiela 2006). The cumulative effects of these processes has left a majority (50–75%) of salt marshes in the Chesapeake and Delaware Bays to be degraded or eliminated (Stevenson et al. 2002). Such marsh loss can have adverse secondary effects. Valiela and Cole (2002) found that the area of intact wetlands along estuaries generally correlates positively with seagrass production and negatively with seagrass habitat loss. This is presumably due to the removal of nutrients by marshes which discourages estuarine plankton blooms, thus enhancing pelagic light penetration and encouraging benthic productivity (Valiela et al. 2000).

A prominent form of salt marsh alteration is ditching. Salt marsh ditching has been practiced for many centuries but was made more prevalent in the USA by public work programs in the 1930s (Bourn and Cottam 1950; Daiber 1986; Kennish 2001). This was partly prompted by an increase of mosquito-borne illness as ditches were designed to eliminate standing water in marshes, a required habitat for mosquito larvae (O'Meara 1992). It has been estimated that 90% of salt marshes in the northeast USA were ditched during the twentieth century as a means of reducing mosquito populations (Daiber 1986). An unintended consequence of this practice was the lowering of the marsh soil salinity, speculated by some to be responsible for the establishment of the invasive *Phragmites* over the native *Spartina* (Chambers et al. 1999) as well as the destruction of habitat for small fish which may prey on mosquito larvae. The potential impacts of salt marsh ditching on adjacent estuarine ecosystems have not been well studied.

The goal of this study was to assess the impact of water from salt marsh ditches on estuarine water quality. Working in salt marshes on the eastern and southern shores of Long Island, NY, USA, we established the chemical and biological composition of the ditchwater and the estuaries which receive salt marsh ditch drainage. Experiments were conducted to examine the effects of this ditchwater on the plankton community composition, biomass, and metabolic processes. We also constructed a nitrogen input budget for an estuary which included ditches as a nitrogen source.

Methods

Study Sites

Seven salt marshes along four estuaries on Long Island, NY, USA, were sampled from 2004 to 2007 during this study (Fig. 1; Table 1). Because the goal of this study was to assess the impacts of salt marsh ditches on estuarine water quality, we sampled during summer and fall months (May–October) when estuarine water quality issues (i.e., algal blooms, hypoxia, pathogenic bacterial loads, seagrass growth) are of greatest concern. Multiple ditches ($n \geq 4$) within five salt marshes along the south shore of Long Island, which were typical of salt marshes in this region, were sampled monthly (June to September; $n=4$) in 1 year to characterize the chemical (dissolved inorganic and organic nutrients) and biological (total coliform bacteria) features of water contained in the region's salt marsh ditches. These included salt marshes in East Islip, NY (40° 42′50 N, 73°11′01 W on Great South Bay; Fig. 1, Table 1), Bellport, NY (40°43′15 N, 73°08′34 W; on Great South Bay; Fig. 1, Table 1), Mastic, NY (40°44′55 N, 72°51′41 W; on Moriches Bay; Fig. 1, Table 1), Westhampton, NY (40°48′08 N, 72°38′0 W; on Moriches Bay; Fig. 1, Table 1), and Hampton Bays, NY (40°50′22 N, 72°29′30 W; on Shinnecock Bay; Fig. 1, Table 1).

Two other salt marshes, representative of systems on eastern Long Island, were sampled biweekly to monthly from May to October over a 3-year period (2005–2007) to establish a detailed understanding of the composition of ditchwater in these systems and to approximate the potential impacts of the ditchwater on receiving estuaries.

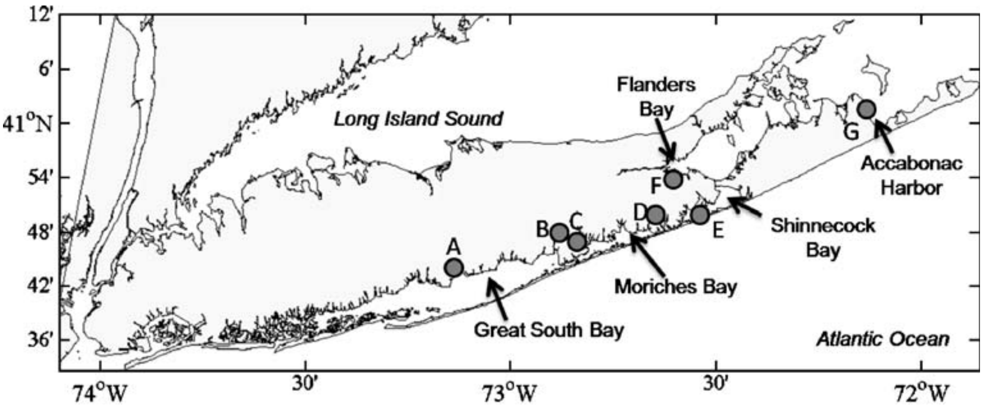

Fig. 1 Salt marsh and estuarine study sites across Long Island, NY, USA: *A* East Islip, *B* Bellport, *C* Mastic, *D* Westhampton, *E* Hampton Bays, *F* Flanders Bay, *G* Accabonac Harbor. Estuaries receiving tidal discharge from salt marshes are also depicted

Table 1 Salt marshes sampled during this study including information regarding location, their receiving estuary, size, number of ditches present, mean ditch length, ditches per square kilometer, ditch length, and general pattern of ditches present

Site	Adjacent bay	Latitude	Longitude	Area (km^2)	Number of ditches (km^{-2})	Mean ditch length (m)	Ditch pattern
East Islip	Great South Bay	40°42′50 N	73°11′ 01W	1.5	27	90±6	Parallel
Bellport	Great South Bay	40°43′15 N	73°08′34 W	0.6	122	140±6	Gridded
Mastic	Moriches Bay	40°44′55 N	72°51′41 W	0.3	143	170±1	Parallel
Westhampton	Moriches Bay	40°48′08 N	72°38′00 W	0.2	95	220±30	Gridded
Hampton Bays	Shinnecock Bay	40°50′22 N	72°29′30 W	0.1	100	60±10	Parallel
Flanders Bay	Flanders Bay	40°54′40 N	72°36′02 W	1.9	77	82±10	Parallel
Accabonac Harbor	Accabonac Harbor	41°01′31 N	72°08 44 W	1	125	137±23	Gridded

Flanders Bay of the Peconic Estuary is lined by ditched salt marshes along its southern shore (40°54′36.6 N, 72°36′2.4 W, Fig. 1, Table 1). More than 90% of the roughly 150 ditches on Flanders Bay are open and subject to full tidal flushing twice a day while approximately one dozen ditches have been plugged or filled. The western portion of this salt marsh system lies adjacent to a densely populated neighborhood with individual cesspools whereas the southern and eastern portions border on undeveloped woodlands. While the entire system was surveyed during this study, four ditches on western developed portion were compared to four on the undeveloped eastern side for more detailed investigation. In Accabonac Harbor, East Hampton, NY (41°01′31 N, 72°08′44 W, Fig. 1), six sites within ditches were sampled. Mid-bay sites within Flanders Bay and Accabonac Harbor were also sampled on a biweekly basis during summer and fall. Ditch dimensions and volumes in both sites varied greatly. The average length of the ditches in the Flanders Bay salt marsh system was found to be 82±10 m with a mean volume of 66±8 m^3, while the average ditch length of the survey area in Accabonac Harbor was 137±23 m with a mean volume of 198±37 m^3.

Field Measurements

To characterize water which typically entered neighboring estuaries, ditches were sampled at their mid-point during mid-ebb tide. An exception to this trend was intensive sampling through tidal cycles which occurred in Flanders Bay. During each sampling, measurements for temperature and salinity were obtained with a YSI 556 sondes. Ditchwater was collected from 20 cm below the surface in acid-washed polycarbonate bottles which were immediately transferred to the laboratory. Duplicate particulate organic carbon and nitrogen (POC and PON) and triplicate chlorophyll *a* (Chl *a*) samples were collected on precombusted GF/F glass fiber filters and stored frozen. DOC, DON, and nutrient samples were filtered with acid-cleaned, polypropylene capsule filters (0.2 µm; MSI Inc.). Within 2 h of collection, DOC samples were acidified with quartz-distilled nitric acid and frozen along with DON and nutrient samples. DOC samples were analyzed in duplicate by high-temperature catalytic oxidation using a Shimadzu TOC-5000 Total Organic Carbon Analyzer (Benner and Strom 1993). Duplicate POC and PON samples were dried at 60°C before analysis on a Carlo Erba NA 1500 NCS system (Cutter and Radford-Knoery 1991). Total dissolved N (TDN) was analyzed in duplicate by persulfate oxidation techniques (Valderrama 1981) and DON was calculated by subtracting levels of nitrate, nitrite, and ammonium from concentrations of TDN. Chl *a* was analyzed in triplicate by standard fluorometric methods (Parsons et al. 1984). Nitrate, nitrite, ammonium, and phosphate were analyzed in duplicate by standard spectrophotometric methods (Parsons et al. 1984; Jones 1984). Measurements of J. Sharp's (U. Delaware) intercalibration DOC samples were within 5% of the consensus value. Recoveries (mean ± 1 SD) of SPEX Certi-Prep^{INC} standard reference material at environmentally representative concentrations were 93±7% for nitrate and 94±6% for total nitrogen. Blanks for DOC, DON, POC and PON were <10% of the lowest sample. In 2005 and 2007, coliform bacterial abundances in ditches across several marshes and in their receiving waters were quantified using standard methods. In 2005, these measurements were made using the Most Probable Number Technique which involves incubating a serial dilution of ditchwater with nutrient media broth specific for coliform bacteria and estimating abundances based on physiological activity (gas production) at each dilution (Crescerl et al. 1998). In 2007, coliform bacteria were quantified via strict adherence to International Standard Organization's protocol for the detection and enumeration coliform bacteria via the membrane filtration technique (Crescerl et al. 1998; ISO 9308-1). Briefly, ditchwater samples were filtered onto 0.45-µm-gridded cellulose filters placed in lactose triphenyltetrazolium chloride tergitol nutrient media (Crescerl et al. 1998; ISO 9308-1). The number of coliform bacteria per 100 ml was estimated by using an inverted microscope to quantify the number of yellow colonies formed following

24 h incubation at 37°C. In both years, ditches were sampled monthly during June through September and collected in a manner identical to other field samples (as described above). Analysis of samples using both techniques yielded results which were statistically identical (*t* test, $p>0.05$).

For the determination of percent organic matter of sediment, surface sediments (upper 5 cm) were collected in duplicate from each of the salt marsh ditch (at mid-ditch locations) and from each estuary surveyed using a PVC core (5 cm diameter; 15 cm depth). Samples were dried to constant mass at 70°C for several days and then combusted at 500°C for 5 h to burn off organic matter (Carroll et al. 2008). The percent organic content of the sediments was determined to be that constituent lost on ignition at 500°C and was calculated as the dry weight minus the ashed weight divided by the dry weight times 100. The mean percentage for all of the ditches of Accabonac Harbor and Flanders Bay were compared to those of each estuary.

Nitrogen Budget

The volume of ditches in Flanders Bay was obtained by measuring width and depth at mean high tide at three points within each ditch. Length was directly measured or estimated using computer satellite images. Comparison of these two techniques gave statistically identical results. In cases where calculated volume discharge were compared to actual volume drained as measured with flow meters, net volume discharges were not significantly different. Total ditch volume for the Flanders Bay salt marsh was estimated to be 9,910 m^3. Since all observed ditches ($n>100$) were tidally emptied of standing water at the end of low tide, fluxes were calculated by accounting for the complete tidal draining twice a day (as observed in the field).

The supply of total dissolved nitrogen to Flanders Bay during summer was estimated to establish the relative contribution of various sources to these systems. Nitrogen input fluxes considered included precipitation/atmospheric deposition, land run-off, diffusive benthic fluxes, groundwater seepage, rivers, tidal exchange, and salt marsh ditches. Groundwater discharge rates and N concentrations were obtained from Montlucon and Sañudo-Wilhelmy (2001). Atmospheric deposition was calculated in four ways. Precipitation data obtained from the National Weather Service were applied to concentrations of TN in precipitation, which were obtained from Cornell et al. (1995). Atmospheric deposition fluxes (wet and dry) were also calculated according to Paerl (1993) and Hu et al. (1998). Direct inorganic N fluxes were also obtained from US EPA National Atmospheric Deposition Program and were doubled assuming that the organic N content of rain would equal that of DIN (Cornell et al. 1995). Since these four approaches yielded similar flux rates (relative standard deviation=35%), a mean of these calculations was utilized. Runoff volumes (5% of precipitation to the watershed) and estimates of DIN concentrations in runoff were obtained from a study of a similarly sized, enclosed embayment on eastern Long Island (Gobler and Boneillo 2003). Summer diffusive benthic fluxes of N measured in Flanders Bays were obtained from Howes et al. (1994), who determined these fluxes using 24 h sediment core incubation at in situ light and temperature conditions as described by Howes et al. (1986). Peconic River fluxes were obtained from USGS annualized summer flow rates and mean N concentrations measured in the river during summer (USGS 1955–2005). Tidal exchange volumes into Flanders Bay were calculated using a salt balance (Fischer et al. 1979) between Flanders Bay and the next basin in the Peconic estuary, Great Peconic Bay (Fig. 1) according to the following equation: $Q_{\mathrm{T}} = \left(Q_{\mathrm{GW}} + Q_{\mathrm{RO}} + Q_{\mathrm{precip}} + Q_{\mathrm{river}}\right)/[1 - (S_{\mathrm{FB}}/S_{\mathrm{GPB}})]$ where Q represents the water fluxes from net tidal exchange (T), groundwater (GW), surface runoff (RO), precipitation (precip) and the Peconic River (river), and S is the mean annual salinities for Flanders Bay (FB) and Great Peconic Bay (GPB). Salinities and total dissolved nitrogen concentrations in these bays were obtained from Suffolk County (NY) Department of Health Services (SCDHS 2004–2006), Office of Ecology, which sampled each basin twice monthly from May through October of 2004 and 2006, three of the 4 years which are representative by this study (2004–2007). Dissolved N levels in water emanating from salt marsh ditches were corrected for the nitrogen content of bay water before volume discharge rates were applied for ditch N fluxes.

Phytoplankton Amendment Experiments

During June through October of 2006 and 2007, monthly experiments were conducted to estimate the impact of ditchwater on receiving estuarine plankton communities in Flanders Bay and Accabonac Harbor (Fig. 1). For all experiments, water from multiple ditches per site was pooled. To ensure the water used for experiments was representative of water which would enter estuaries, experimental water (ditch and bay) was collected during the mid-ebb tides and brought back to the lab in 20-l carboys. Each type of water (ditch and bay) was then filtered through a 0.2-μm capsule filter to remove particulate matter. Filtered ditch or bay water was added to experimental bottles as 25% of the total experimental volume, mimicking the approximate ratio of ditchwater volume to the volume of enclosed tributaries which receive ditchwater in both Flanders Bay and Accabonac Harbor (Fig. 1). As such, 250 ml of filtered treatment water was added to 750 ml of whole bay water in 1-l, acid-washed polycarbonate bottles. Treatments consisted of a control (75% bay water + 25% filtered bay water), ditchwater additions (75% bay water + 25% filtered ditchwater from each site), and an inorganic nutrient enrichment treatment (75% bay water + 25% filtered bay water + 20 μM nitrate and 1.25 μM phosphate) which allowed comparison of nutrient enrichment from salt marsh ditches to nutrient enrichment from inorganic nitrogen and phosphorus only. Triplicate bottles for each treatment were then incubated under neutral density screening (50% ambient light, mimicking ambient light attenuation) for 2 days in Old Fort Pond at the Stony Brook-Southampton Marine Station (Gobler et al 2005). Incubations were terminated by filtering of the samples onto glass fiber (whole community) and 5 μm polycarbonate filters (>5 μm community) for subsequent chlorophyll *a* analysis and for high performance liquid chromatography (HPLC) pigment analysis (DiTullio and Geesey 2002). Five photopigments quantified by HPLC which are found exclusively in single classes of phytoplankton were selected to represent five major algal groups. Peridinin was used as an indicator of dinoflagellates, alloxanthin was analyzed as a proxy for cryptophytes, lutein indicated chlorophytes, zeaxanthin represented cyanobacteria, and fucoxanthin was used for diatoms (DiTullio and Geesey 2002). Net growth rates were estimated for each group based on changes in pigment concentrations using the formula: $\mu = \ln\left([\text{pigment}]_{\text{final}}/[\text{pigment}]_{\text{initial}}\right)/\text{incubation time}$ where $[\text{pigment}]_{\text{final}}$ was the concentration of a given photopigment in bottles at the end of experiments, $[\text{pigment}]_{\text{initial}}$ was the concentration of photopigment at the beginning of the experiment, and the incubation time was in days and was generally two. For each phytoplankton group, differences in growth rates among treatments were statistically examined using one-way analyses of variance (ANOVA) followed by Tukey's multiple comparison tests.

Photosynthesis/Respiration Measurements

During the 2007 field season, phytoplankton amendment experiments were modified to assess the impact of the ditchwater on rates of photosynthesis and respiration in the water column. Six clear and six black, acid-washed, borosilicate, 300 ml biological oxygen demand bottles were filled with 225 ml of estuarine water and 75 ml of filtered bay or salt marsh ditchwater, providing the same ratio of whole and filtered water as described above. Initial dissolved oxygen readings for each bottle were made using a YSI 5100 oxygen meter, which contains a Clark-type electrode and was calibrated daily according to the manufacturer's instructions. The 12 bottles were incubated under neutral density screening (50% ambient light, mimicking ambient light attenuation; Gobler et al. 2005) in Old Fort Pond as described above. Experiments were terminated by making final dissolved oxygen readings after 24 h. Changes in oxygen levels in dark bottles were interpreted as pelagic respiration rates, whereas changes in oxygen levels in light bottles, corrected for respiration rates, were considered rates of gross photosynthesis. Differences between treatments and controls were assessed using Student's *t* tests.

Results

Nutrient Profiles of Salt Marsh Ditches and Adjacent Estuaries

During the study, nutrient concentrations of most constituents collected over the sampling period were significantly higher in salt marsh ditches than the adjacent estuary (Fig. 2). One exception to this was nitrate (NO_3^-) concentrations which did not differ between ditches and the bay of both systems (Fig. 2a). Ammonium (NH_4^+) concentrations in both Peconic Estuary study areas were more than an order of magnitude higher in the ditches (12.6±5.40 μM for FB and 7.61 μM for AH) than in the bays (0.48±2.25 for FB and 0.30±2.24 μM for AH, $p<0.001$, *t* test). In the Flanders Bay system, ortho-phosphate concentrations (PO_4^{-3}) were found to be 4-fold higher in the ditches (4.01±1.41 μM) than the bay (1.06±1.79 μM, Fig. 2a; $p<0.001$, *t* test), although differences were not significant in Accabonac Harbor (ditches=2.20±0.54 μM; bay=1.15±1.76 μM, Fig. 2a). Levels of organic matter were also enriched in salt marsh ditches relative to receiving estuaries. For example, levels of dissolved organic nitrogen (DON) were enriched in ditches relative to the estuary for both FB (38.18±8.94 vs. 19.24±4.91 μM, $p<0.001$, *t* test, Fig. 2b) and AH (30.7±7.69 μM vs.20.9±4.24 μM, $p<0.001$, *t* test, Fig. 2b). Similarly, FB ditches were significantly enriched in particulate organic nitrogen (PON; 21.94±2.46 μM) when compared with FB (12.6±0.77 μM, Fig. 2b; $p<0.001$, *t* test), as were AH ditches (7.24±0.92 μM) compared to the surrounding harbor (4.88±0.36 μM; Fig. 2b; $p<0.001$, *t* test). Both dissolved and particulate organic carbon (DOC, POC) were found at far higher concentrations in the ditches than in the bays of both Flanders Bay and Accabonac Harbor. DOC concentrations found in FB (229±22.6 μM) were 2-fold lower ($p<0.001$, *t* test) than those found in the surrounding ditches (577±38.9 μM, Fig. 2b). Concentrations of DOC in AH (211±6.03 μM) were also significantly ($p<0.001$, *t* test) lower than those in the surrounding salt marsh ditches (324±81.7, Fig. 2b). POC in the FB ditches was higher than in FB (236±31.6 and 109±8.16 μM, respectively, $p<0.001$, *t* test, Fig. 2b). AH ditches were also enriched in POC with respect to the bay (84.4±10.1 and 55.1±2.98 μM, respectively, $p<0.001$, *t* test, Fig. 2b). In contrast to the differences

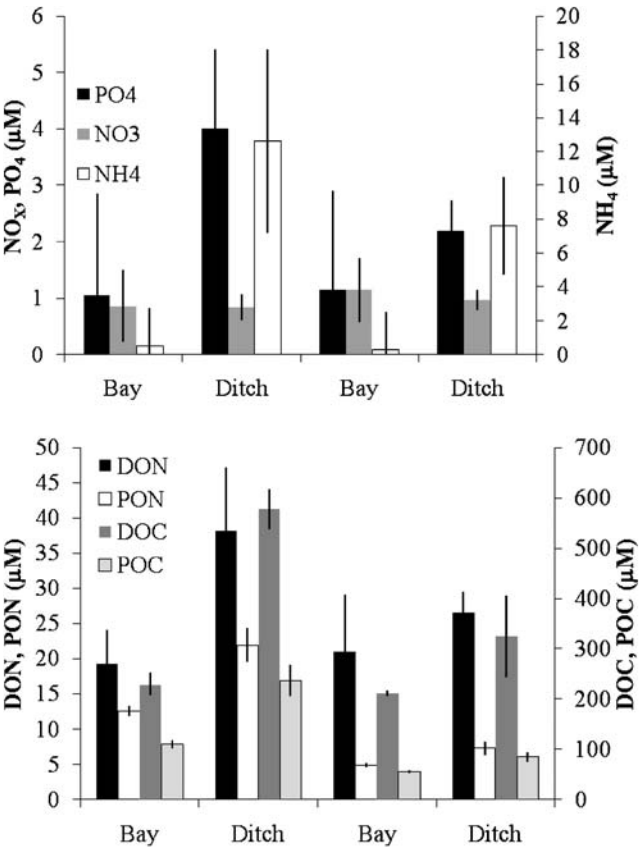

Fig. 2 Levels of inorganic (**a**) and organic (**b**) nutrients for the Flanders Bay and Accabonac Harbor salt marsh systems. Concentrations represent means for all biweekly to monthly samples collected during the June through October sampling period of 2005, 2006, and 2007. *NO_x* nitrate+nitrite, *NH_4* ammonium, *PO_4* ortho-phosphate, *DOP* dissolved organic phosphorus, *DON* dissolved organic nitrogen, *PON* particulate organic nitrogen, *DOC* dissolved organic carbon, *POC* particulate organic carbon

between ditches and estuarine water, there were no significant differences in the concentrations of dissolved inorganic (NO_3^-, NH_4^+, PO_4^{-3}) and dissolved organic (DON, DOP, DOC) constituents between the bays of the two study sites (FB and AH), and dissolved inorganic nutrients and DOP and DON did not vary significantly between the ditches of the two systems (Table 2, Fig. 2a). PON and POC were found at higher ($p<0.001$, *t* test) concentrations in FB than AH (Table 2, Fig. 2b), and ditches in FB contained higher levels of PON ($p<0.001$, *t* test), DOC ($p<0.001$, *t* test), POC ($p<0.001$, *t* test, Table 2, Fig. 2b) than AH.

Surveys of three ditched salt marshes on Long Island's south shore (Bellport, Mastic, East Islip; Fig. 1) displayed similar results to our long-term study (Table 2). Ditches in all sites were found to be significantly enriched in DON (5-fold enrichment; *t* test, $p<0.001$ for all) and NH_4^+ (order of magnitude higher; *t* test, $p<0.001$ for all; Table 2) relative to their receiving estuaries. However, the levels of nitrate were similar between ditches and adjacent estuaries (Table 2).

Tidal Influence on Salt Marsh Ditch Nutrients and Organic Nitrogen

Surveying multiple sites over multiple tidal cycles showed that nutrient concentrations in salt marsh ditches changed with tidal state. Except for nitrate, all other nutrients measured (NH_4^+, PO_4^{-3}, and DON) increased in concentration with decreasing tidal state (Fig. 3). Nitrate concentrations were significantly different ($p<0.001$, *t* test) between high and mid tide (1.94±0.83 and 1.20±0.77 µM) and mid- and low tide (0.32±0.16 µM, Fig. 3), with concentrations decreasing with tidal height. In contrast, ammonium concentrations increased significantly ($p<0.001$, *t* test) from high (5.77±0.59 µM) to mid- (13.2±2.02 µM) and low tide (23.2±4.56 µM, Fig. 3). This was also the case for PO_4^{-3} which increased in concentration from 1.33±0.17 to 3.89±0.66 to 8.02±1.30 µM during the ebbing tide ($p<0.001$, *t* test, Fig. 3). DON concentrations rose significantly from 34.59±2.65 to 56.32±4.47 to 79.22±10.86 µM as the tide receded ($p<0.001$, *t* test, Fig. 3).

Coliform Bacteria Abundance

Total coliform bacteria abundances in ditches were significantly higher ($p<0.05$) than adjacent estuaries for all three sites surveyed (Fig. 4). The highest concentrations of coliform bacteria were found in the western sites of the Flanders Bay salt marsh (808±278 bacteria 100 ml^{-1}, $n=22$). These concentrations were much greater than those found in adjacent Flanders Bay (42±12 bacteria 100 ml^{-1}, $n=70$). The eastern ditches of the Flanders Bay salt marsh hosted much lower coliform bacteria levels, although the densities were still significantly higher than the bay (93±24 bacteria 100 ml^{-1}, $n=20$, Fig. 4). Ditches in the Moriches Bay salt marsh system were found to have levels of coliform bacteria (316±64 bacteria 100 ml^{-1}, $n=26$, Fig. 4) which were much higher than those found in adjacent Moriches Bay (60±20 bacteria 100 ml^{-1}, $n=29$, Fig. 4). In contrast, ditches of the southern shores of Shinnecock Bay had the lowest bacterial densities (39±8 bacteria 100 ml^{-1}, $n=25$, Fig. 4), although these levels were still significantly higher than nearby Shinnecock Bay which had significantly lower bacterial concentrations than the other estuaries studied (15±2 bacteria 100 ml^{-1}, $n=28$, Fig. 4).

Relative Contribution of N Loading from Salt Marsh Ditches to Flanders Bay

Our surveys of the salt marshes lining the southern half of Flanders Bay showed the total discharge volume of the ditches to be 9,910 m^3, twice a day. Mean summer total dissolved nitrogen (N) concentrations over the study period for the Flanders Bay ditches were shown to be 52 µM.

Table 2 Mean nutrient concentrations during summer months (June–September) within five sampling sites

	Years	*n*	NO_x	NH_4	PO_4	DOP	DON	PON	DOC	POC
Flanders Bay, Flanders, NY										
Bay	2005–2007	20	0.86 (0.63	0.48 (2.25)	1.06 (1.79)	0.92 (2.50)	19.24 (4.91)	12.61 (0.77)	229.23 (22.59)	109.52 (8.16)
Ditch	2005–2007	75	0.84 (0.24)	12.62 (5.40)	4.01 (1.41)	2.69 (1.81)	38.18 (8.94)	21.94 (2.46)	577.40 (38.94)	236.31 (31.58)
Accabonac Harbor, East Hampton, NY										
Bay	2006–2007	7	1.15 (0.57)	0.30 (2.24)	1.15 (1.76)	0.94 (2.43)	20.98 (4.24)	4.88 (0.36)	211.25 (6.03)	55.07 (2.98)
Ditch	2006–2007	48	0.98 (0.18)	7.61 (2.90)	2.20 (0.54)	2.42 (1.26)	30.68 (7.69)	7.24 (0.92)	324.09 (81.70	84.38 (10.10)
Havens Point, East Moriches, NY										
Bay	2004	31	2.40 (2.23)	1.63 (1.22)	–	–	19.62 (8.50)	–	–	–
Ditch	2004	16	1.60 (5.69)	15.31 (6.97)	–	–	74.95 (17.60)	–	–	–
John's Neck, Mastic, NY										
Bay	2004	62	2.15 (2.44)	1.47 (1.51)	–	–	27.32 (16.53)	–	–	–
Ditch	2004	16	3.43 (4.27)	25.54 (14.10)	–	–	140.07 (57.50)	–	–	–
Timber Point, East Islip , NY										
Bay	2004	57	6.66 (13.92)	2.15 (2.69)	–	–	33.76 (14.59)	–	–	–
Ditch	2004	16	5.12 (11.69)	14.41 (3.28)	–	–	81.07 (17.50)	–	–	–

Four ditches within the salt marshes of Havens Point, John's Neck, and Timber Point were sampled monthly from June to September ($n=16$). Multiple ditches within Flanders Bay and Accabonac Harbor were sampled biweekly to monthly, June through October from 2005 to 2007. Values are means and SD in parenthesis

NO_x nitrate+nitrite, *NH_4* ammonium, *PO_4* ortho-phosphate, *DOP* dissolved organic phosphorus, *DON* dissolved organic nitrogen, *PON* particulate organic nitrogen, *DOC* dissolved organic carbon, *POC* particulate organic carbon

After correcting for the N originating from the bay water (see Table 2), the resulting total daily flux of N into Flanders Bay was 770 mol. Our mass balance estimates indicate that the ditches along Flanders Bay represent ~5% of the N flux to the estuary during summer and ~18% of the N flux to the more pristine southern portion of Flanders Bay (Table 3). The bulk of N enters Flanders Bay via groundwater (>59%) followed by the Peconic River which delivers about 20% of the bay's N (Table 3). For the entire system, the ditches are comparable in importance as a N source to the atmosphere (7%) and benthic fluxes (8%) and an order of magnitude greater than surface runoff and tidal exchange (<1% for each, Table 3). For the southern portion of Flanders Bay, the outflow from the ditches constituted the second largest source of N (18%) after groundwater (54%) and was larger than N input from the atmosphere (13%) and benthic fluxes (14%).

Phytoplankton Community Composition Response to Ditchwater Additions

Flanders Bay In additions of water from the western ditches and inorganic nutrient (N+P) additions yielded increased net growth rates ($p<0.001$, Fig. 5) in all size fractions of the phytoplankton community (<5 µm, >5 µm, and total). Pigment-based net growth rates indicated that net growth rates of dinoflagellates, diatoms, and cryptophytes were also enhanced by western ditchwater and inorganic nutrients ($p<0.001$, $p<0.001$ and $p<0.004$, respectively, Fig. 5). In contrast, cyanophytes and chlorophytes responded to the addition of the ditchwater from the western ditches only ($p<0.05$ for both, Fig. 5).

In August, all treatments yielded significant increases in net growth rates in the total and large (>5 µm) phytoplankton community ($p<0.001$ for both, Fig. 5), while only the additions of N+P significantly enhanced the net growth rates of smaller size algae (<5 µm, $p<0.05$). All five of the main groups of phytoplankton displayed significantly enhanced net growth rates from the addition of water from the western ditches ($p<0.01$, Fig. 5). Diatoms, dinoflagellates, and chlorophytes also experienced a significant

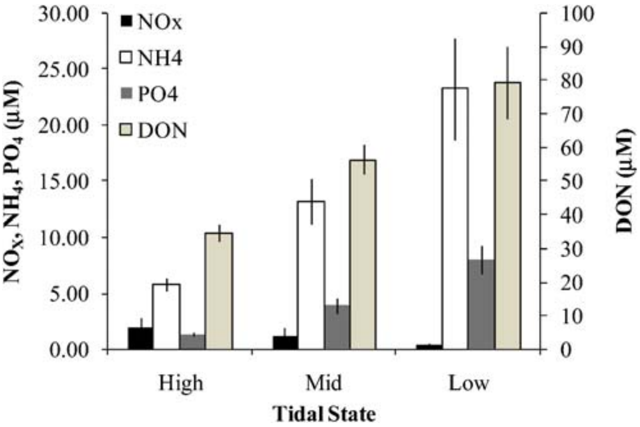

Fig. 3 Nutrient concentrations at different tidal states in Flanders Bay. Values are means of four ditches sampled monthly, June through September, during 2005. *NO_x* nitrate+nitrite, *NH_4* ammonium, *PO_4* ortho-phosphate, *DON* dissolved organic nitrogen

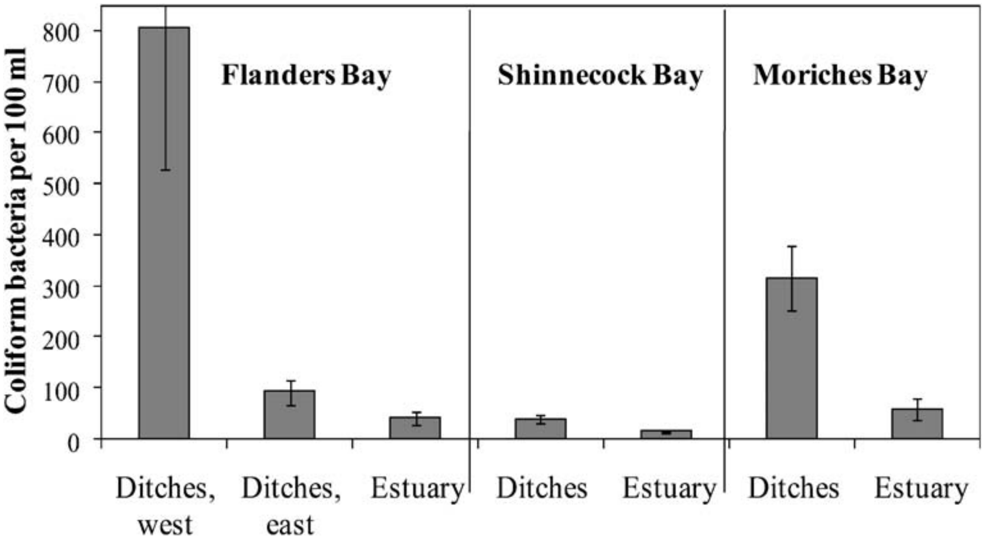

Fig. 4 Coliform bacterial concentrations in ditches and surrounding bays. Values are means from monthly samples (June–August) collected during summers of 2005 and 2007. *Error bars* are the standard errors of those values

increase in net growth rate ($p<0.001$) with the addition of N+P.

In September 2006, all phytoplankton size classes (<5 µm, >5 µm, and total) experienced increased net growth rates with the three treatments ($p<0.001$, Fig. 5). Diatoms and dinoflagellates net growth rates were significantly increased relative to controls for all treatments ($p<0.01$), whereas chlorophytes and cryptophytes net growth rates were significantly enhanced by the addition of water from the western ditches and inorganic nutrients ($p<0.05$). Cyanophyte net growth rates were not stimulated by any of the treatments.

Accabonac Harbor For July, the addition of N+P significantly increased the net growth rate of all phytoplankton size classes over the control ($p<0.03$, Fig. 5). Pigment analysis was not performed for this experiment. In August, all treatments resulted in enhanced net growth of all size classes ($p<0.001$, Fig. 5). Diatoms, cyanophytes, and chlorophytes were also all stimulated by all treatments ($p<0.05$). In contrast, cryptophytes experienced a significant increase in net growth rate when water from the southern ditches or inorganic nutrients was added ($p<0.05$). An absence of dinoflagellates prohibited peridinin pigment analysis for the August and October experiments. In October, the addition of N+P significantly increased the net growth rates of all size classes relative to the control ($p<0.01$, Fig. 5). Water from ditches enhanced phytoplankton net growth relative to the bay water addition for the larger size fraction only ($p<0.01$, Fig. 5). Diatoms and chlorophytes also experienced significantly increased net growth rates relative to the control during this experiment ($p<0.05$, Fig. 5).

Effects of Ditchwater on Photosynthesis and Respiration

Flanders Bay The addition of filtered salt marsh ditchwater significantly increased rates of pelagic respiration beyond those of the filtered bay water controls (Fig. 6). In June,

Table 3 Summer dissolved nitrogen fluxes from various sources into Flanders Bay, Peconic Estuary, NY, USA

	Flanders Bay		Southern half only	
	mol N day^{-1}	Percent of total	mol N day^{-1}	Percent of total
Peconic River	3,100	20	0	0
Atmosphere	1,100	7	550	13
Groundwater	9,000	59	2,300	54
Benthic fluxes	1,200	8	580	14
Surface runoff	55	<1	27	<1
Tidal exchange	99	<1	45	1
Ditches	770	5	770	18
Total	15,324	100	4,272	100

Details of calculations are provided in the “Methods” section. Note: Groundwater discharging from the undeveloped southern half of Flanders Bay is pristine compared to the groundwater discharging from the northern half which is enriched with nitrogen from intense farming in that region

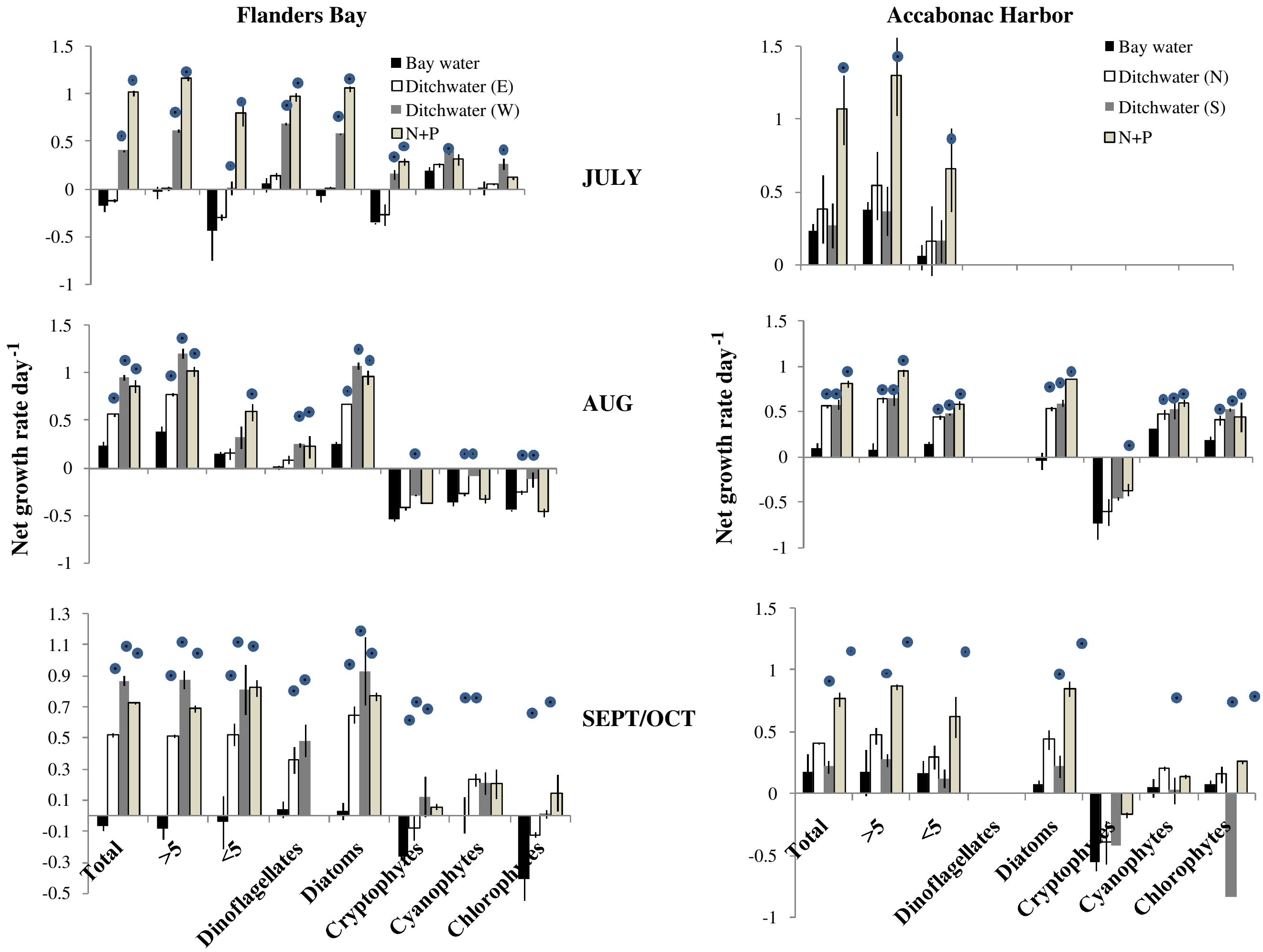

Fig. 5 Net growth rates for different phytoplankton groups during nutrient, ditchwater, and bay water experiments. Experiments were conducted monthly in 2006 with the addition of inorganic nutrients (N+P), filtered salt marsh ditchwater, or filtered bay water (as a control) from Flanders Bay (*left pane*) and Accabonac Harbor (*right pane*). Statistically significant ($p<0.05$) differences in growth rates over the controls are marked with *dots*

respiration rates were significantly higher ($p<0.001$, *t* test) in the ditch treatment bottles than in the control (1.00±0.06 vs.0.84±0.07 mg O_2 l^{-1} day^{-1}, Fig. 6) while gross photosynthesis was not affected (1.97±0.1 mg O_2 l^{-1} day^{-1} in the treatment vs. 1.86±08 mg O_2 l^{-1} day^{-1} in the control, Fig. 6). In July, filtered ditchwater nearly doubled respiration rates from 0.86±0.09 to 1.65±0.10 mg O_2 l^{-1} day^{-1} ($p<0.001$, *t* test, Fig. 6), but had no effect on gross photosynthesis. Additions of filtered ditchwater in August more than tripled respiration rates from 0.15±0.06 to 0.48±0.05 mg O_2 l^{-1} day^{-1} ($p<0.001$, *t* test), again without having an effect on the gross photosynthesis (Fig. 6). October deviated from this pattern in that, despite significantly higher respiration rates in the ditchwater treatments (0.88±0.10 mg O_2 l^{-1} day^{-1}in the treatment; 0.25±0.02 mg O_2 l^{-1} day^{-1} in the control; $p<0.001$, *t* test), gross photosynthesis was also significantly higher in the treatment (0.97±0.14 mg O_2 l^{-1} day^{-1}) relative to the controls (0.71±0.07 mg O_2 l^{-1} day^{-1}; Fig. 6; $p<0.002$, *t* test).

Accabonac Harbor All experimental enrichments with filtered salt marsh ditchwater conducted in Accabonac Harbor yielded significantly increased gross photosynthetic rates relative to filtered bay water amendment (Fig. 6). In June, gross photosynthesis was significantly enhanced by the addition of water from the ditches to 1.16±0.42 mg O_2 l^{-1} day^{-1} compared to 0.59±0.27 mg O_2 l^{-1} day^{-1} in the filtered bay water (Fig. 6; $p<0.05$, *t* test). Similarly, in July, gross photosynthesis in the ditchwater treatments (2.45± 0.18 mg O_2 l^{-1} day^{-1}) was significantly higher than in the controls (1.72±0.18 mg O_2 l^{-1} day^{-1}, $p<0.001$, *t* test, Fig. 6). Only the August experiment yielded enhanced ($p<$ 0.001, *t* test) respiration rates in the treatments (1.40± 0.28 mg O_2 l^{-1} day^{-1}) compared to the controls (0.09± 0.09 mg O_2 l^{-1} day^{-1}, Fig. 6). The increase in respiration,

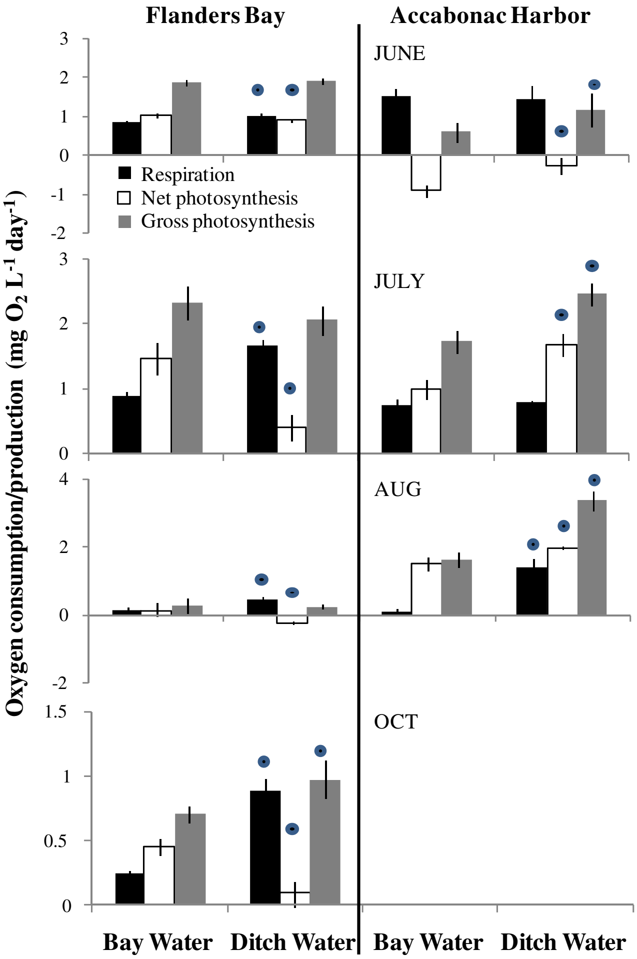

Fig. 6 Respiration, net, and gross photosynthesis rates for light/dark bottle experiments during summer 2007. Experiments were conducted using salt marsh ditchwater or bay water (as a control) from Flanders Bay (*left pane*) and Accabonac Harbor (*right pane*). Significant differences between treatments and controls are marked with *dots*

however, was the largest one observed in any of the experiments for both Accabonac Harbor and Flanders Bay (Fig. 6). Gross photosynthetic rates were also significantly higher in the filtered salt marsh ditchwater addition (3.37± 0.28 mg O_2 l^{-1} day^{-1}) when compared to the filtered bay water control (1.62±0.21 mg O_2 l^{-1} day^{-1}, Fig. 6; $p<0.001$, *t* test).

Discussion

Salt marshes play a critical role in land–estuary interactions. During this study, we demonstrated that water within salt marsh ditches on Long Island, NY, USA, is enriched in nutrients (ammonium, phosphate), organic matter (DON, DOC, DOP, PON, POC), and coliform bacteria relative to receiving estuarine waters. We also established the ability of salt marsh ditchwater to significantly alter the growth, composition, and metabolism of estuarine plankton communities. Our nitrogen budget for Flanders Bay demonstrated that, during summer months, salt marsh ditches can be a substantial N source to this estuary. In sum, these results provide new perspective on the potential impact of salt marsh ditches on estuarine water quality

Fluxes of Nutrients and Dissolved Organic Matter from Salt Marsh Ditches to Estuaries

Several hypotheses exist as to whether salt marshes function as sources or sinks for estuarine nutrients and organic matter. Teal (1962), Odum (1971), and others have long argued that salt marshes export nutrients which are then transported into the coastal ocean. This “outwelling” hypothesis has since been challenged by other investigators who have argued salt marshes serve as a sink for such constituents. A number of prominent studies of salt marshes (Heinle and Flemer 1976; Haines 1976; Woodwell et al. 1977; Valiela et al. 1978, and Dame et al. 1986) have shown that marshes export dissolved organic carbon (DOC), dissolved organic nitrogen (DON), and phosphate (PO_4^{-3}). Several marshes have also been shown to be a source of ammonium (NH_4^+), particularly during summer (Axelrad 1974; Heinle and Flemer 1976; Woodwell et al. 1977). In contrast, the majority of these studies have concluded marshes import more oxidized forms of dissolved inorganic nitrogen (NO_3, NO_2^-; Axelrad 1974; Valiela et al. 1978; Woodwell et al. 1979). Consistent with these prior investigations, POC, PON, DOC, DON, NH_4^+, and PO_4^{-3} concentrations occurred at significantly higher concentrations in salt marsh ditches compared to adjacent estuaries (Fig. 2, Table 2), suggesting these ditches are a source of organic matter and nutrients to estuaries.

Prior reviews of the import or export of nutrients from salt marshes have considered tidal range and the state of marsh vegetation as predictors of marsh function. For example, the “marsh–estuarine continuum theory” Dame et al. (1992) and Childers’ (1994) review of marsh–water column interactions concluded that tidal range and marsh maturity dictate whether a salt marsh system behaves as a source or sink for nutrients. In salt marshes with a relatively large tidal range (>1 m), active marsh plant growth is expected to assimilate nutrients which enter when the marsh is inundated by high tides and thus nutrients are generally not exported (Wolaver et al. 1985; Whiting and Childers 1989; Childers et al. 1993, 1994). Similarly, younger marshes with robust plant growth are also known to serve as a net sink for nutrients due to active uptake by vegetation (Dame et al. 1992). It is likely that the effect of ditches on salt marsh vegetation and flooding patterns influences the ability of marshes to assimilate nutrients. For example, within the Long Island salt marshes examined for this study, ditches hosted no vegetation, eliminating uptake of nutrients by plants in these zones. Moreover, ditches observed during this study filled and emptied twice daily by tidal flushing, which may, in part, circumvent tidal inundation of marshes, an occurrence which was extremely rare in all seven systems observed during this study. This may be partly related to tidal amplitude, which is below Dam's critical threshold of 1 m (0.2–0.7 m on the south shore of Long Island and 0.7 m in the Peconic Estuary). As such, these salt marshes would be considered 'well-flushed salt marsh category III' by Odum et al. (1979) and would be a 'low tidal range salt marsh' by Childers (1994), both categories of salt marsh which have been shown to be sources of constituents to the estuary. The model of Childers et al. (1993) shows that at low tide, these types of marshes release NH_4^+, NO_3^-, PO_4^{-3}, total nitrogen (TN), total phosphorus (TP), DOC, and DON.

Beyond the absence of plant biomass to assimilate nutrients, the chemical, geological, and morphological nature of ditches likely make them rich sources of nutrients and organic matter. Prior research has shown that the organically enriched bottoms of marsh channels are a source of reduced nitrogen and DOC (Whiting and Childers 1989; Chambers and Odum 1990). Consistent with this concept, our analysis of sediment in the ditches and bays (Flanders Bay and Accabonac Harbor) found that ditch sediments contained 31±6.2% organic matter whereas bay sediments contained 4.0±1.6%. The microbial degradation of the rich organic matter in salt marsh ditches during warm summer months is likely to generate copious amount of nutrients (NH_4^+, PO_4^{-3}) and dissolved organic matter (DOC and DON; Boynton and Kemp 1985), partly accounting for their enrichment. This hypothesized robust microbial degradation of organic matter in salt marsh ditches is supported by our observation that ditchwater stimulates high rates of microbial respiration (Fig. 6) and most salt marsh ditches have low oxygen levels and often become anoxic at night (data not shown). These low oxygen levels may foster active denitrification which would account for the lower levels of nitrate found in salt marsh ditches compared to estuarine waters (Nixon 1980; Table 2; Figs. 2 and 3). In addition, as water empties from ditches during outgoing tides, interstitial water stored in the porous, organic-rich bottom of the ditches (Mitsch and Gosselink 1993) is likely leach into the ditches as the hydraulic pressure of the overlying water is diminished. This process would further contribute to the enrichment of nutrients and organic matter within ditches as tidal height decreases (Fig. 3). Finally, the substrate surface area (sides and/or bottom) to volume ratio of the salt marsh ditches is much larger than it is for the bays and even for tidal creeks, which are generally deeper and wider. Combined with the shallow nature of the ditches (<1 m), this will lead to the influence of benthic processes being maximal, further promoting the observed enrichment of nutrients and organic matter. As such, ditched marshes would be expected to contribute more nutrients and organic matter than marshes with an equal volume of creek water because the surface area to volume ratio of ditches is larger and the depth is shallower.

Our results demonstrate that ditches may serve as a source of dissolved nitrogen to Flanders Bay, contributing 5% to the total system and 18% to the southern half of the bay (Table 3). Some aspects of the morphology of Flanders Bay and its salt marsh likely maximize the influence of salt marsh ditches on this system. Specifically, the entire southern shore of Flanders Bay is comprised of dense salt marshes, all of which are extensively ditched. Because Flanders Bay is shallow (mean depth=2 m), the ratio of ditched salt marsh to estuarine volume is maximized for this estuary, particularly, along its southern extent. Moreover, the influence of salt marsh ditchwater on estuaries is likely to be maximal during the summer months considered in this budget, as this is when nutrient content of ditchwater and salt marshes seems to be maximal (Kaplan et al. 1979; Woodwell et al 1979; this study) and freshwater flow from rivers and groundwater is minimal (Gobler and Sañudo-Wilhelmy 2001). Interestingly, despite the highly eutrophic nature of the western extent of Flanders Bay, which receives water from the Peconic River and its sewage treatment plant (Breuer et al. 1999), salt marsh ditches still had a discernable impact on the entire estuary's N supply.

Pathogenic Bacteria in Salt Marsh Ditches

During this study, levels of total coliform bacteria were significantly higher in all salt marsh ditches compared to adjacent estuaries. Fecal coliform bacteria likely originate from the feces of the many warm-blooded animals which may inhabit the marsh such as terrestrial mammals, waterfowl, domestic pets, and humans via failing septic systems and storm water runoff. The New York Department of Environmental Conservation (NYDEC) requires total coliform bacterial levels to be below 70 MPN per 100 ml to keep marine waters open for shellfishing in NY. Our data show levels of coliform bacteria in the ditches of two of the surveyed estuarine systems exceeded this level (Moriches Bay and Flanders Bay, Fig. 4) which could impact the microbial loads of shellfish in regions adjacent to these salt marshes. The higher levels within Moriches Bay and Flanders Bay ditches may be partly related to the poor flushing of these bays, the enrichment of coliform bacteria in salt marshes ditches surrounding them, and the complex gridded ditch network which feeds into tributaries prior to entering the estuary. Perhaps acting in a manner similar to our ditches, Mallin et al. (1999) found that small creeks with high concentrations of coliform bacteria were a source to the main tributaries of North Carolina estuaries. In addition, flooding and ebbing of salt marsh channels has been shown to increase coliform abundance in salt marsh creeks since coliform bacteria are often concentrated in sediments and are subsequently reintroduced to the water column by tidal currents (Davies et al. 1995; Burkholder et al. 1997). This impact would be maximized in salt marsh ditches which are generally shallower than salt marsh channels and tributaries and thus possibly more likely to resuspend bottom material via tidal currents. In contrast to Moriches Bay and Flanders Bay, the salt marsh ditches along Shinnecock Bay are smaller and shorter, and thus tidally discharge quickly and directly into the bay, and perhaps contributing to the lower levels of coliform bacteria there (Portnoy and Allen 2006).

Salt Marsh Ditchwater Impact on Estuarine Phytoplankton Populations

Phytoplankton populations in the bays of the Peconic Estuary have been shown to be strongly nitrogen-limited during summer months (Gobler and Boneillo 2003). Ditchwater from Flanders Bay and Accabonac Harbor was highly (25-fold) enriched in ammonium compared to bay water and proved capable of stimulating the net growth rates of the majority of phytoplankton groups we examined in our experiments. During the August, September, and October experiments, the growth responses from ditchwater were nearly equivalent to those due to the additions of 20 μM nitrate, despite a substantially lower amount of inorganic nutrients being added from the ditchwater (<5 μM final DIN concentration; Table 2). This suggests other components of the ditchwater may have also provided nutrition to algae such as the copious amounts of DON and DOC (Table 2). In Flanders Bay, dinoflagellates were one of the groups which displayed the strongest net growth response to filtered ditchwater (Fig. 5), perhaps because they grow well on organic forms of nitrogen and carbon (Anderson et al 2002) providing them an advantage over autotrophic estuarine phytoplankton (Smayda 1997; Anderson et al 2002; Fig. 5). Ditchwater from FB was enriched in DON and DOC over bay water, 2- and 2.5-fold, respectively (Table 2), providing ample organic substrate for mixotrophic algae. In contrast, Accabonac Harbor ditchwater had significantly lower concentrations of DON and DOC and seemed to favor the growth of diatoms (Fig. 5).

In general, most of the algal response from the experimental addition of nutrients came from larger phytoplankton (Fig. 5). During experimental additions of salt marsh ditchwater, net growth rates of phytoplankton >5 μm typically outpaced those of smaller phytoplankton, and net growth rates of dinoflagellates and diatoms, which are typically large cells, were greater than algal classes which are generally composed of small cells such as cyanobacteria, chlorophytes, and cryptophytes (Fig. 5). Small algal cells have high cell surface to volume ratios and thin surface diffusive boundary layers, permitting higher nutrient uptake rates per unit of biomass, and thus making them well adapted to low nutrient environments (Raven and Kubler 2002). As such, elevated nutrient levels associated with salt marsh ditchwater discharge to estuaries is likely to stimulate the growth of larger algae (diatoms, dinoflagellates), which may settle quicker than smaller phytoplankton (Smetacek 1985; Ducklow et al. 1986; Kiorboe et al. 1996), and thus may be more likely to contribute to bottom oxygen demand (Justic et al. 2002).

Salt Marsh Ditchwater Impact on Microbial Metabolic Rates

Plankton communities in tidal salt marsh waters are known to be heterotrophic (Ragotzkie 1959; Turner 1978). Experiments conducted in Flanders Bay demonstrate that filtered ditchwater enhanced respiration rates of plankton communities relative to filtered bay water (Fig. 6). Heterotrophic bacteria are likely responsible for some of this respiration since the ditchwater in Flanders Bay was enriched in NH_4^+, PO_4^{-3}, DON, and DOC, all compounds known to stimulate bacterial respiration (Keil and Kirchman 1991; Kirchman 2000). In addition, the stimulation of heterotrophic activity from mixotrophic dinoflagellates may also be responsible for this respiration since this group of phytoplankton was always stimulated by the addition of filtered ditchwater and they are known to be able to utilize organic compounds (Anderson et al. 2002). Gross photosynthesis was not enhanced in FB during three of four experiments (Fig. 6), despite the increase in pigment-specific growth rates in other experiments conducted there (Fig. 5). These results again suggest that the nutrients from the ditchwater additions stimulated mixotrophic phytoplankton (Fig. 6). Hypoxia is a symptom of eutrophic marine environments (Parker and O'Reilly 1991; Boesch et al. 2001). Our results suggest that in Flanders Bay, export from salt marsh ditches may contribute to oxygen consumption in this system and could exacerbate hypoxia which occurs in the western extent of this basin (Breuer et al. 1999).

The role of bacteria and mixotrophic phytoplankton in influencing metabolic rates is further supported by results from AH where ditchwater always stimulated gross photosynthesis but rarely changed respiration rates. The general lack of an enhancement in respiration rates may be partly attributed the substantially lower levels of DOC in the ditchwater in AH (320 μM; Table 2), which were nearly half of the levels found in FB ditchwater (580 μM; Table 2). In addition, there was an absence of dinoflagellates at AH

(Fig. 5), which we suspect contributed to high respiration rates in Flanders Bay via heterotrophic pathways (see above). Despite substantially lower DOC levels in AH, ditchwater there was enriched 25-fold in ammonium over bay water there (Table 2), which likely supported strong, autotrophic growth of diatoms and chlorophytes during experiments and contributed toward the large and consistent increases in gross photosynthesis in the ditchwater treatments there (Figs. 5 and 6).

Conclusion

Salt marsh ditches are capable of impacting estuarine water quality. During this study, the salt marshes studied did not inundate with water, but rather, unvegetated salt marsh ditches filled and drained tidally, exporting nutrients, organic matter, and pathogenic bacteria to estuaries. As such, ditching may short circuit the processes by which undisturbed salt marshes absorb and cycle nutrients and organic matter. Beyond the delivery of reactive chemical and biological constituents, salt marsh ditchwater is likely to stimulate algal growth and estuarine respiration. The management practice of filling or plugging salt marsh ditches may mitigate their impacts on estuarine water quality.

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