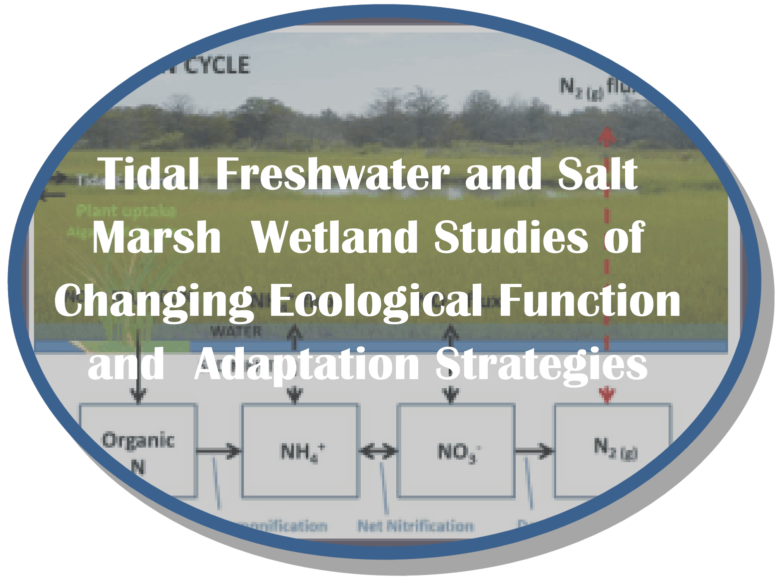
Barnegat Bay—Year 2



Tidal Freshwater and Salt Marsh Wetland Studies of Changing Ecological Function and Adaptation Strategies

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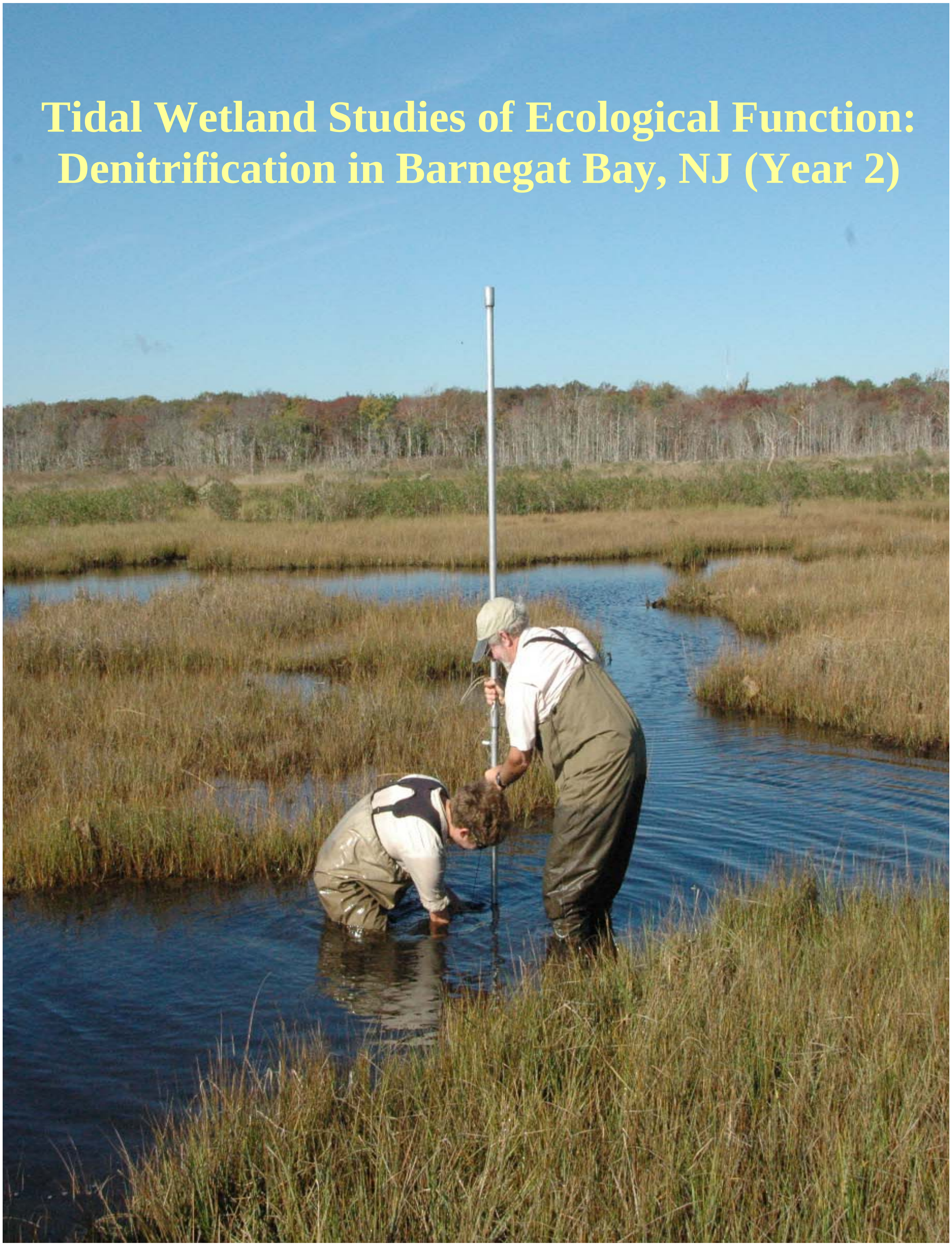
Tidal Wetland Studies of Ecological Function: Denitrification in Barnegat Bay, NJ (Year 2)



FINAL REPORT

To

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Tidal Wetland Studies of Ecological Function: Denitrification in Barnegat Bay, NJ (Year 2)

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Executive Summary

Barnegat Bay in New Jersey has been experiencing symptoms of nutrient-eutrophication such as phytoplankton blooms, but it is unknown whether nitrogen inputs are causing these blooms (e.g., HAB, harmful algal blooms), and possibly secondary impacts (e.g., anoxia, loss of seagrass, increase in jelly fish, decreases in fish and crab populations, etc.). Salt marshes play a large role in removing pollutants and nutrients from water. The main mechanisms of removal in marshes are plant uptake, burial in the soil and microbial denitrification. Based on nutrient and radiometric data collected from soil cores, burial in the 26,000 acres of salt marshes in Barnegat Bay removes over 90% of the estimated 7.7±0.7 x 10^5 kg/yr N load (average from 2005-2010). Our goal was to quantify N removal via denitrification in vegative and non-vegative (OMWM sites) areas salt marshes of Barnegat Bay to contribute to a more accurate quantification of a Barnegat Bay N budget. Specific to this project was to evaluate whether open marsh water management (OMWM) ponds alter the cycling of nitrogen via denitrification compared to non-OMWM wetted locations (e.g., ditches and creeks).

A total of 54 soil cores were collected at the AT&T site in Strafford, NJ in May, July and October 2014. Ponds and creek waters were analyzed for nutrients, nitrogen fluxes, and sediment oxygen demand. In addition, sediment quality (e.g., organic carbon, total nitrogen, chlorophyll *a*) were measured to help evaluate rate measurements. Lastly, pore water sulfide was measured at two depths as it was hypothesized that increased sulfide levels, in the ponds and wetted areas, would inhibit denitrification rates. Rates in the ponds and other areas in this study were only slightly lower than denitrication rates in previous studies indicating that ponds could impact overall nitrogen removal. While sulfide levels did not appear to impact denitrification rates at the levels measured other factors such as either low nitrate levels or high oxygen levels in the ponds could have inhibited denitrification.

Overall, salt marsh denitrification has the potential to remove approximately 13 to 33% of the incoming estimated N load entering the bay (median = 6.6 x 10^5 kg/yr). Both sediment burial and denitrification can sequester or remove ~85% of the incoming load. Tidal marshes are an important component of the Barnegat Bay ecosystem and help to remove a substantial amount of N entering the bay.

Recommendations for Future Research and Monitoring

This study determined that tidal marshes in the Bay can remove a substantial amount of N entering the system. On a seasonal time scale (over the course of the study) up to a third of the N is converted to nitrogen gas and possibly more through other nitrogen microbial processes. Burial works on longer time scales, years to decades, which may then bias the total removal to higher than expected values. In this regard, shorter term burial estimates need to be accomplished to make the two removal mechanisms more comparable. Studies using marker horizons, sediment deposition plates and ^{7}Be analysis would aid in this comparison.

Important to this study, OMWM appears to slightly lower or similar to overall rate of denitrification during peak warm months compared to other open water areas and the marsh surface. An accurate aerial survey of the extent of OMWM (i.e., acres of area) and rates within these sub-systems needs to be undertaken to better determine the impact on the overall budget of N to the Bay. Lastly, denitrification and burial in the sub-tidal waters of the Bay need to be assessed in order to make a complete determination of the removal processes in the Bay and the services they provide.

A) Introduction

Ecosystem services provided by wetlands that fringe the coast are at risk from human development, modifications, and sea level rise. In this regard it is important to understand the current extent of services wetlands provide, such as nutrient cycling-retention, to plan for the future and related environmental and land use changes. This study was designed to enhance our understanding of nitrogen uptake, burial and removal services provided by coastal wetlands in Barnegat Bay. Information on N load burial and removal (by vegetation and microbes; **Figure 1**) in the bay will assist resource managers for the development of water quality and biodiversity management plan.

Human management of salt marshes has been occurring for millennia. The fate of coastal wetlands with a rising sea level has prompted the interest to understand the natural marsh processes and its responses to the rising sea levels. These studies illustrate large variability in elevation change and accretion rates across spatial and temporal scales. Important factors that influence this variation include the availability of allocthonous sediment, hydrology, and plant processes including production and decomposition (Nyman et al. 2006; Mudd et al. 2009, D'Alpaos at el. 2011). A large gap in our understanding of marsh processes affecting accretion and elevation change is the effect of physical manipulations such as ditching, ditch plugging, and/or pool creation.

One of the most widespread of the physical manipulations of salt marshes is ditching, a historic method for controlling mosquito populations. Approximately 90% of tidal marshes between Maine and Virginia have been grid ditched (Bourne and Cottam 1950). Ditches have been shown to have negative consequences to marsh ecology including drainage of the marsh surface, a lower water table, and altered plant community structure and wildlife habitat (Daiber 1986). As an alternative to grid-ditching open marsh water management (OMWM) techniques (Ferrigno et al., 1975; Meredith et al., 1985; James-Pirri et al, 2015) were developed to create additional open water habitats. Small shallow pools and connecting ditches (radials) are excavated in high and low salt marsh areas (mainly salt hay cordgrass [*Spartina patens*] and smooth cordgrass [*Spartina alterniflora*]). While pools are natural salt marsh features, OMWM can increase the density of pools across a marsh and can locate pools in areas where natural pools may not have formed. In addition, pools and radials may be established in areas that have been previously grid ditched. Grid ditching has reduced the occurrence of natural pools

(Adamowicz and Roman 2005), but the effects of creating pools and radials at a high density in areas previously grid ditched is unknown (Elsey-Quirk and Adamowicz, 2015 *in press*). However, very few studies have examined the effect of various ecosystem processes such as denitrification and nitrogen removal from the physical manipulations of ditches and OMWM pools on the marsh surface. Concerns about wetland loss and resultant loss of ecosystem services (along with habitat alterations) have raised questions about the extent of constructed ponds on marsh surfaces needed to control mosquito. The overall objective of our studies (Velinsky et al, 2010; 2013) was to help inform resource managers of the value of wetland-watershed linkages in understanding nutrient sinks with regards to OMWMs in Barnegat Bay.

A1: Background

In order to quantify seasonal and spatial variation of denitrification rate in the salt marshes of Barnegat Bay, we collected five cores from three marshes in May, July and October 2012 (Velinsky et al, 2013). Sampling locations were representative of salt marshes across the bay including Reedy Creek in the north, Island Beach State Park on the Barrier Island, and Horse Point in the south (**Figure 2**). Denitrification rate averaged 84 ± 11 µmol $N\text{-}N_2$ m^{-2} hr^{-1} across sites and seasons (Velinsky et al., 2013). Denitrification rate did not differ between May and July at any of the sites, but was greater in July (121 ± 20 µmol $N\text{-}N_2$ m^{-2} hr^{-1}) than October (48 ± 19 µmol $N\text{-}N_2$ m^{-2} hr^{-1}) across sites ($F_{2,\,45} = 3.77$, $p = 0.0306$). There was no significant difference among sites in denitrification rate. The data indicated that the vegetated surfaces of salt marshes are a significant N sink.

In order to scale up meter square measurements of denitrification to the landscape scale of Barnegat Bay, other marsh features need to be examined. Open Marsh Water Management (OMWM). OMWM has been adopted in several Atlantic coastal states to control mosquitoes by excavating ponds and connecting ditches (pond radials) on the marsh platform. While ponds are natural salt marsh features, OMWM increases the density of ponds across the marsh and places ponds in areas where natural ponds may not have formed. Majority of OMWM ponds no longer connected via radials to flowing water and at only the higher tides have the potential to be flushed. In addition, ponds may be established in areas that have been previously grid ditched. Grid ditching has reduced the occurrence of natural ponds (Adamowicz and Roman 2005), but the effects of creating ponds at a high density in areas previously grid ditched is unknown. The mosquito control commissions operating in Barnegat Bay have been applying OMWM since the 1970's. Ocean County Mosquito has installed over 9,000 ponds across 12,000 acres in Barnegat Bay over the last ~30 years (OCM, per comm.). *It is unclear how a high density of ponds in areas that were once vegetated marsh will affect N removal.* Once established, the ponds are flushed with tidal water only during the highest of tides, thus they may become anoxic for extended time periods. The denitrification process requires both aerobic and anaerobic micro-zones as oxygen is required for nitrate and nitrite (NOx) production and denitrifying bacteria yet the reduction of NOx to N_2 gas (i.e., denitrification) is an anaerobic process. Pond sediments that are seldom flushed and re-oxygenated with tidal water and thus become anaerobic for periods of time may have a lower denitrification rate than the vegetated marsh. In a preliminary study, we collected 5 cores from OMWM ponds that were established in 2009, the adjacent vegetated marsh, where sediments from the pond excavation were deposited, and a nearby marsh that has not been subject to OMWM (Velinsky et al., 2013). We sampled these treatment and control sites one time in July 2012. Denitrification rates were lower, although not significantly in the OMWM ponds than the vegetated marsh sites, largely due to the variation in the vegetated marsh where roots and rhizomes can influence oxygen dynamics and therefore nutrient transformations.

A2: *Objectives of Study*

The overall objective of this study is to quantify the difference between marsh sites, using previous data, and OMWM ponds for N removal. *Our hypothesis is that oxygen and/or sulfide my limit denitrification in OMWM ponds, but further research is required to determine the magnitude and cause of this difference. In addition, we compared these removal rates to burial rates and inputs to the Bay from a previous study.*

The proposed study and objectives supports 1 of the 11 objectives to address research gaps for the protection of Barnegat Bay (http://www.nj.gov/dep/barnegatbay/plan-research.htm). Objective (9): "Tidal Freshwater and Salt Marsh Wetland Studies of Changing Ecological Function and Adaptation Strategies" is directly addressed in this research study and builds upon previous work that assessed nutrient (i.e., nitrogen and phosphorus) burial rates in wetlands in Barnegat Bay (Velinsky et al., 2010; 2013).

A3: *Study Area*

The Barnegat Bay-Little Egg Harbor estuary (BB; Barnegat Bay) is located along the central New Jersey coastline in the Atlantic Coastal Plain province (**Figure 2**). Barnegat Bay is a back-barrier lagoon-type estuary that extends from Point Pleasant south to Little Egg Inlet. The variety of highly productive shallow water and adjacent upland habitats found in this system include barrier beach and dune, submerged aquatic vegetation (SAV) beds, intertidal sand and mudflats, salt marsh islands, fringing tidal salt marshes, freshwater tidal marsh, and palustrine swamps.

The Barnegat Bay system is composed of three shallow bays (Barnegat Bay, Manahawkin Bay and Little Egg Harbor), is approximately 70 km in length, 2 to 6 km wide, and up to 7 m deep. The Bay watershed covers an area of approximately 1700 km^2 and has been extensively developed over the past 70 years. The tidal waters cover approximately 280 km^2 with a ratio of watershed area to water area of 6.1. The current land use (2006) of the watershed is agriculture (~1%), wooded/forest (~28%), tidal and non-tidal wetlands (~18%), urban areas (~20%) and open water (30%; Lathrop and Haag 2007). Importantly, watershed development (urban area) has increased over time. From 1986 to 2006 the amount of urban land cover increased from 15 to up to 21% of the land area, while forested land cover has decreased (NJ DEP, see www.state.nj.us/dep/bmw/ReportOcean.htm; Lathrop 2004). The population of the watershed has increased substantially from the 1940s (40,000) to over 570,000 year-round residents currently (US Census Reports). During the height of the summer season the population can rise to approximately 1,000,000.

B) Field and Laboratory Methods

B1: *Field Sampling*

To meet the objective of the study, canals, ditches and OMWM ponds were sampled during three seasons in 2014 to determine the rate of denitrification and nutrient fluxes between tidal waters and overlying waters (i.e., water within the pond or ditch) to see how these relatively new features on the marsh are altering ecosystem services such as denitrification. This was an expansion of our previous study (Velinsky et al., 2013) but with a focus on OMWM structures.

We obtained shallow sediment cores and overlying water from 18 randomly chosen locations in a mid-marsh tidal wetlands in Straffford County (NJ; ATT site; **Figures 2-3; Table 1**). We examined seasonal denitrification in OMWM ponds by comparing N_2 production rates in

OMWM ponds (n = 9), tidally flushed ditches with pond area (n=3), tidally flushed ditches in a non-OMWM area (n=3) and tidal creeks (n=3) in the spring (May), summer (July), and fall (October) of 2014. At each site one core (ca. ~20-25 cm in length, 12cm of sediment and ~400ml of water) was taken along with overlying water and meter readings of salinity, temperature, dissolved oxygen and pH. Approximately 30 liters of site water will also be collected for core incubations and water quality analyses in each of the sites.

Creek water (>20 L) was collected near each site during each sampling period in large pre-cleaned carboys. Adjacent creek water column temperature, dissolved oxygen, and conductivity near each site were measured using a handheld YSI Model 556.

B1.1 Line Transects

Three fixed line transects were established on either side of the main tidal channel (OMWM and control sections) for conducting plant community surveys (**Figure 4**). During peak biomass (July to September), marsh elevation and plant community changes are assessed using real-time kinetic (RTK) GPS (Leica GS14 GNSS receiver) that can attain centimeter scale accuracy for latitude, longitude and elevation measurements. Emergent macrophyte species within a square meter of the transect are identified. In conjunction, RTK GPS data points are taken when there is a surface feature (e.g. mosquito ditches, OMWM ponds, mudflats, etc.) or a change in the dominant plant species. RTK GPS points are taken at 25 m intervals if there are no changes to mark. The data from subsequent years will be used to assess these transects and document major changes in plant communities over time.

B2: *Laboratory Methods*

Water and cores were transported, within 12 hr, to the University of Maryland's Horn Point Environmental Laboratory for incubation and nutrient flux measurements (i.e., N_2, O_2, nitrate, ammonium and phosphate). A portion of the water was retained at the Academy for filtration and nutrient analysis. Once incubations were completed, sediment cores were sectioned into two depths and placed into centrifuge tubes (under N_2 atmosphere) for separation of water/sediments for H_2S analysis. The filtrate will be placed into small HDPE bottles and 1 mL of a concentrated zinc acetate preservative is added to each sample (immediately). The preserved samples were kept refrigerated at < 4^o C until analysis. Water samples for nutrient and related parameters were filtered and immediately frozen (see below). All samples were analyzed in the laboratories at the Academy of Natural Sciences of Drexel University.

B2.1: Denitrification and N Fluxes

Procedures for the sediment-water exchange incubations followed protocols used at the University of Maryland Center for Environmental Science (Kana et al. 2006; Cornwell and Owens 2011; Gao et al. 2012). A heating/refrigerating circulator was used to maintain bottom water temperature in the water-jacketed incubators. Water temperature was set at near-field temperatures. Cores were pre-incubated overnight with air lift pumps in each core used to prevent anoxia and to re-circulate the overlying water with a large reservoir water bath. This pre-incubation also ensured thermal equilibrium between surface sediments and the overlying water; this is critical to the measurement of denitrification but less critical for phosphorus fluxes. Incubations were started on the day after core collection, with addition of a sealed magnetic stirring unit to the top of the cores and insertion into the incubator. Control cores without sediment were used to correct for any water-column effects. A central magnetic stirring disk stirred cores in the annular chamber outside the disk. Inlet tube and outlet tubes were attached to the top of the flux cores and ambient replacement water in a carboy was attached to the inlet tube. On the outlet tube, a 2 way valve was attached for sampling. Stirring was carried out at a rate below that required for sediment resuspension.

For ambient fluxes, we sampled the cores 3 times at ~1.5 hour intervals after an initial sampling, for a total of ~4-5 hours incubation. Sampling consisted of opening a valve to the replacement water, opening the sampling valve, and sampling using gravity (the replacement water was placed 0.5 m above the cores). For gas analysis (i.e. N_2 and O_2), we added a small sample tube and filled 7 mL stoppered glass tubes to 2 times overflowing (i.e. like a BOD bottle fill). We add 0.010 mL 50% saturated $HgCL_2$ to each vial to preserve the sample; this preservation has worked well on the order of 3 weeks. After stoppers were added, vials are submerged under water and stored at a temperature lower than the incubation temperature; this minimized drying of the stopper/vial joint. For nutrient analysis, a 20 mL syringe was attached to the sampling valve without the plunger; when full, the plunger was added and the sample filtered through a 0.4 μm pore size 25 mm syringe filter. Upon completion of the sediment-water exchange measurements, the cores were extruded into two sections for porewater H_2S and the surface sediment for total C and N, and chlorophyll *a*. The water column height above the sediment was measured and used to calculate the water volume overlying the cores.

Calculations

Sediment-water exchange rates were calculated from the slope of the change of chemical constituent concentrations in the overlying water:

$$F = \frac{\Delta C}{\Delta t} * \frac{V}{A}$$

Where F is the flux (µmol m-2 h-1), $\Delta C/\Delta t$ is the slope of the concentration change in overlying water (µmol L^{-1} h^{-1}), V is the volume of the overlying water (L) and A is the area of the incubated core (m^{-2}). When the water-only control core had a significant slope, the slope of the flux cores was adjusted accordingly. We convert these rates to units more typically used in engineering studies (e.g. mg m^{-2} d^{-1}).

B2.2: Dissolved Nutrients

Water samples, both from the adjacent creek and core incubations, were collected for NO_x (NO_3^- + NO_2^-), NH_4^+, and soluble reactive phosphorus (SRP). Water was filtered and stored frozen in pre-cleaned bottles. Nitrate-nitrite and ammonium concentrations were determined using an Alpkem 300 segmented flow autoanalyzer with a detection limit of 0.006 and 0.005 mg/L for NO_x and NH_4, respectively, while SRP had a detection limit of 0.002 mg P/L.

B2.3: Sediment Total Organic Carbon and Total Nitrogen

Sediment total organic carbon and total nitrogen were measured using a CE Instruments, Flash EA 1112 Series following the guidelines in EPA 440.0, manufacturer instructions and ANSP-PC SOP. Samples were ground to a powder, pre-treated with fuming HCL to remove inorganic carbon, re-dried and ground. Samples were weighed into tin boats using a microbalance (in duplicate) and analyzed using the FLASH 1112 elemental analyzer.

Published laboratory clean-techniques were used throughout (US EPA 1997, APHA, AWWA and WEF 1995) using protocols as outlined in standard operating procedures (SOPs) at the

Academy of Natural Sciences of Drexel University. All materials coming in contact with the samples were precleaned plastic, glass or metal and were cleaned of any contaminants prior to use. Sample ID forms were used and each sample was given a unique laboratory number for sample tracking. Sediments and water were analyzed at laboratories operated by the Academy of Natural Sciences of Drexel University (Patrick Center) for dissolved nutrients, organic carbon, and total nitrogen, while denitrification and sediment oxygen demand rates were determined at the Institute for Marine Science at University of North Carolina.

B2.4. Porewater Sulfide

Porewater sulfide will be collected from the surface sections (0-5 cm and 5-10cm) of the marsh by centrifugation and filtration (operationally defined using a 0.45 µm Gelman Acrodisc syringe filters). Samples from selected cores will be placed into a pre-cleaned 250 mL HPDE centrifuge bottle in a N_2-purged glove bag (see for example Howes et al, 1985). The bottle will capped tightly and centrifuged for ~10 min using a high speed Dupont centrifuge (@8k rpm). The bottle will be placed back into a the N_2-purged glove bag and carefully decanted into a small beaker or directly into a glass or polypropylene syringe connected to a filter holder. The filtrate will be placed into small HDPE bottles and 1 mL of a concentrated zinc acetate preservative is added to each sample (immediately). The preserved samples were kept refrigerated at < 4o C until analysis. Prior to analysis the zinc acetate will be dissolved and the resultant solution will be analyzed using the iodometric titration method (SM 4500s-f) or the methlyene blue method of Cline (1969). Sulfide will be determined spectrophotometric analysis (Cline 1969) on a Shimadzu UV-1601 UV-Visible Spectrophotometer at 670 nm. A regression curve was used to determine the sulfide in the sample and the concentration of the standards was determined by an iodometric method. Sample concentrations were then corrected for dilution by the preservative.

C) Results and Discussion

C1: *Creek Water and Sediment Properties*

C1.1: Water Chemistry

Water quality showed some slight variations among the sites but were greater between seasons (**Tables 2 and 3; Figure C1**). Temperature ranged from 12 to 19°C in May to approximately 23-27°C in July, decreasing substantially in October to 18-21°C. Salinity was the lowest in May ranging from 1-16 psu (10±5 psu) to between 12-28 psu (25±4 psu) in July while October was the highest reduced (19-24 psu; 22±1 psu). Dissolved oxygen (DO) concentrations varied approximately 1 to 12 mg DO/L for all sites (**Tables 2 and 3**; note: July DO data for ditches near ponds and creek were not acceptable due to meter mis-calibration) and average concentrations were highest in May (7.7 mg DO/L) and similar in July and October (5.7 and 5.0 mg DO/L, respectively). The DO data was recalculated to %DOsat using temperature and salinity data and presented in **Figure 5**. The ponds exhibited a decrease in %DOsat from 104% in May to 65% in October. The ditches of various types and creek water exhibited similar % DO sat and were, on average, generally between 50 and 65%. pH values, on average, ranged from 5.4 in the Creek in July to 7 in the ponds in May. While the overall average for all sites and time periods was 6.9 the Creek sites and non-OMWM ditches has slightly lower pH values in May and July (**Table 2**).

Dissolved inorganic forms of N and P were measured in adjacent creek waters at all sites during the three time periods of this study (**Table 3abc; Figure 6**). Ranges of dissolved ammonium in May, July and October were 0.1-123, 4.0-60 and 9.2-55 μg N/L, respectively; while concentration ranges of dissolved nitrite+nitrate were 1.0-12.5, 1.0-8.3, 1.0-14.2 μg N/L, respectively. For dissolved ammonium and nitrate+nitrite, concentrations were lower in the ponds (except for ditches near OMWM in July) for all time periods. While somewhat variable, in non-pond locations, dissolved inorganic nitrogen concentrations were generally similar, again for the ditches near OMWM in July

C1.2: Sediment Quality Parameters

After the water/gas exchange experiments the individual cores were sectioned into two sections: 0-3 cm and 7-10cm for organic carbon, total nitrogen, total phosphorus, sediment chlorophyll-a and percent organic matter composition (**Table 4abc)**. These variables might influence microbial processing of water column nitrate and the production of pore water ammonium. **Figure 7** presents the data for the May, July and October collections only for the upper section.

Porewater sulfide concentrations were determined at two depth intervals in each core from each site (0-7 cm and 7-10cm). Concentrations of porewater sulfide were, on average, generally similar between the two depth intervals (**Table 3abc; Figure 8**) and ranged overall from 19 to 3,700 μmol/L in the 0-7cm interval and from 39 to 3,940 μmol/L.

C1.3: Seasonal Denitrification and Oxygen Fluxes

Denitrification rates did not differ among months of sites ($p > 0.05$, **Tables 5abc, Figure 9**). Rates of nitrogen gas production ranged from 0 to 96.3μmol/m^2/hr (32±6.6 μmol m^{-2} hr^{-1}, avg±SE) in May, 1.5 to 130 μmol m^{-2} hr^{-1} (49.8±7.9 μmol m^{-2} hr^{-1}) in July and 10.2 to 161.3 μmol m^{-2} hr^{-1} (55.1±10.0 μmol m^{-2} hr^{-1}) in October. Rates of oxygen gas consumption ranged from -3,170 to -775 μmol m^{-2} hr^{-1} (-1,890±160 μmol m^{-2} hr^{-1}, avg±SE) in May, -6,930 to -1,760 μmol m^{-2} hr^{-1} (-3,900±370 μmol m^{-2} hr^{-1}) in July and -5,140 to -1,570 μmol m^{-2} hr^{-1} (2,490±241 μmol m^{-2} hr^{-1}) in October (**Figure 9**). For SOD, there was a significant interaction between month and location ($F_{6, 42} = 3.02$, $P = 0.0151$). Pairwise comparisons show that SOD was significantly higher in July in ditches not near ponds (DNNP) than in all months and locations except for the ditches near ponds (DNP) in July. The SOD in DNP in July was higher than all treatment combinations except for October DNNP.

Velinsky et al. (2013) measured similar N_2 production rates in vegetative marshes in 2012: May averaged 83±14 μmol N m^{-2} hr^{-1} increasing to 121±20 μmol m^{-2} hr^{-1} in July and decreasing to 49±19 μmol m^{-2} hr^{-1} in mid-October. This trend follows the general water column temperature trend during 2012; i.e., 20°C, 26°C and 15.5°C in May, July and October, respectively. Valiela et al. (2000) reported rates of between 12 and 290 μmol m^{-2} hr^{-1} in marshes in the northeast United States, while Hopkinson and Giblin (2008) reported gross denitrification rates in vegetated marshes (variable vegetation) ranging from 36 to 4129 μmol m^{-2} d^{-1} with a median value of 1000 μmol m^{-2} d^{-1} (n=16).

In our preliminary study (Velinsky et al., 2013), denitrification rates were less variable in the OMWM ponds (72±4 μmol N m^{-2} hr^{-1}) than the vegetated marsh sites (113±23 μmol N m^{-2} hr^{-1})

which were similar to control locations (102±19 µmol N m^{-2} hr^{-1}). Rates of denitrification measured in vegetated stands on the marsh surface in year 1 (84 ± 11 µmol N-N_2 m^{-2} hr^{-1}) were significantly greater than those measured in aquatic habitats (i.e., creeks, ditches and ponds; 50 ± 5 µmol N-N_2 m^{-2} hr^{-1}) in year 2 ($t = 2.75$, $p = 0.0071$). However, it is not clear whether the results of this indirect comparison is related to annual differences in climate, flooding, etc., methodological differences in denitrification measurement, or actual differences in denitrification. It should be noted that many factors can impact the rates (including sulfide concentrations, see below) as well as methodological (i.e., method that determined N_2 production), which can make comparison problematic.

A question for this study is whether the pond sediments would have similar denitrification rates compared to the other aquatic habitat features of marshes. The hypothesis was that oxygen and/or sulfide may limit denitrification in OMWM ponds. These pools of water could become stagnant and depleted of oxygen during multiple tidal cycles enhancing anoxic conditions. In all ponds there was measureable concentrations of dissolved oxygen in the overlying waters. Across all aquatic habitats, denitrification was *positively* related to surface sediment sulfide concentration in May only ($r^2 = 0.39$, $p = 0.0307$; **Figure 10**).

C1.4: Nitrate and Ammonium Fluxes

Nitrate +nitrite (NOx) fluxes ranged from -125 to +49 µmol N m^{-2} hr^{-1} across all sites and seasons while ammonium-ammonia (i.e., ammonium) fluxes ranged from -116 to +1,590 µmol N m^{-2} hr^{-1} across all sites and seasons (**Table 4abc, Figure 11**). NOox uptake (negative; into the marsh) was sign greater in ditches not near ponds (DNNP), creek waters, and ditches near ponds (DNP) than in ponds in July (Month* location $F_{6,41} = 3.9$, $p = 0.0035$). Ammonium fluxes were significantly higher in DNNP in July than all other locations and months except October DNNP and May DNP (Month* location $F_{6,40} = 3.6$, $p = 0.0057$).

Both the nitrate and ammonium fluxes are similar to other studies in tidal salt marshes. Velinsky et al. (2013) measured fluxes in vegative marshes of Barnegat Bay in a previous study. Nitrate +nitrite fluxes ranged from -34 to +28 µmol N m^{-2} hr^{-1} across all sites and seasons while ammonium fluxes ranged from -39 to +500 µmol N m^{-2} hr^{-1} across all sites and seasons. Scudlark and Church (1989) measured fluxes from both porewater profiles and flux chambers in the Great Marsh (DE) over a year with ranges of -6 to 6 µmol N m^{-2} $hr^{-1}$and 5.1 to 206 µmol N

m^{-2} hr^{-1}, respectively. Similarly, Chambers et al. (1992), using flux chambers, measured ammonium fluxes of between 3 and 435 µmol N m^{-2} hr^{-1} in a tidal marsh in Virginia with highest rates in the mid-summer (August). Dissolved ammonium fluxes are generally out of the marsh into the overlying waters while nitrite+nitrate (NOx) fluxes are directed into the marsh sediment. On average, the magnitude of the NOx fluxes are not sufficient to support the level of N_2 production and that that a robust coupling of the oxidation of ammonium to nitrate within the sediments (i.e., SOD) makes up the difference (**Figures 9 and 11**). Nitrate flux was related to surface H_2S concentration in July ($r^2 = 0.31$, $p = 0.0193$).

Only in the warmer summer period was there a relationship between SOD and dissolved ammonium fluxes (**Figure 12).** These data also indicate that diagenesis of organic matter is producing dissolved ammonium in excess of that used in coupled nitrification-denitrification and that this remaining ammonium is escaping the marsh surface during tidal inundation.

C1.5: Oxygen Fluxes (Sediment Oxygen Demand)

Rates of oxygen demand (SOD) are an indicator of nitrification: the oxidation of ammonium to nitrite and eventually nitrate (Ward 1996) as well as organic carbon remineralization. Lower oxygen availability would limit nitrification and the coupling to denitrification. Sediment oxygen demand ranged from -6,930 (in July) to -774 (in May) µmol O_2 m^{-2} hr^{-1} in all sites, with the highest rates measured in July when temperatures were the highest (note: negative rates indicate oxygen moving into the sediments from the overlying water /oxygen consumption; **Figure 9**). These rates are higher than those reported in Velinsky et al. (2013) and are most likely due to the current locations being continuously wetted/flooded and potentially fresh organic matter from algal production. In the previous study (Velinsky et al. 2013), denitrification rates were associated with higher oxygen demand indicating tigher coupling of nitrification – denitrification. In the present study however, there was no relationship between sediment oxygen demand and nitrogen production, except for possibly the pond locations in July (**Figure 13**). Highest rates were measured in the warmer summer month (July) with low and no trends in the spring and fall time period. SOD was related to H_2S concentration in May ($r^2 = 0.46$, $p = 0.0027$), which may help to explain also the sign H2S and N2 prod in May. For the ponds, there was a significant relationship between H_2S and SOD across all months (r2 = 0.18, p = 0.0255) and between H_2S and nitrate flux (r^2= 0.18, p = 0.0292).

C2: Impact of Open Marsh Water Management (OMWM) on Nitrogen Processes

Salt marshes have a long history of management, such as diking, draining, salt hay farming, ditching, and more recently, OMWM. OMWM was introduced as a mosquito control technique that increases the number of shallow water pools on the marsh platform. The intention was to facilitate fish consumption of mosquito larvae and reduce the area of suitable habitat for salt marsh mosquito production (Ferrigno and Jobbins, 1968; Meredith et al., 1985). OMWM has been adopted in several Atlantic coastal states to control mosquitoes by excavating ponds and connecting ditches (pond radials) on the marsh platform. OMWM increases the density of ponds across the marsh and locate ponds in areas where natural ponds may not have formed. In addition, ponds may be established in areas that have been previously grid ditched (**Figure 2**). Grid ditching has reduced the occurrence of natural ponds (Adamowicz and Roman 2005), but the effects of creating ponds at a high density in areas previously grid ditched is unknown. The mosquito control commissions operating in Barnegat Bay have been applying OMWM since the 1970s. Ocean County Mosquito has installed over 9,000 ponds across 12,000 acres in Barnegat Bay over the last ~30 years (OCM, per comm.). It is unclear how a high density of ponds in areas that were once vegetated marsh will affect N removal. Once established, the ponds are flushed with tidal water only during the highest of tides, thus they may become anoxic for extended time periods. In addition, changes in vegetation, soil salinity, bird communities and water level can impact the marsh structure (James-Pirri et al. 2011; Elsey-Quirk and Adamowicz. 2015). The denitrification process requires both aerobic and anaerobic micro-zones as oxygen is required for nitrification and anaerobic condition requires for transformation of NO_x to N_2 gas by denitrifying bacteria. Pond sediments that are seldom flushed and re-oxygenated with tidal water and may become anaerobic for periods of time may have a lower denitrification rate than the vegetated marsh.

As mentioned earlier, in our preliminary study (Velinsky et al., 2013), denitrification rates were lower in the OMWM ponds (72±4 μmol N m^{-2} hr^{-1}) than the vegetated marsh sites (113±23 μmol N m^{-2} hr^{-1} ; Velinsky et al., 2013), which were similar to control locations (102±19 μmol N m^{-2} hr^{-1}). In the current study, four areas were sampled for measurement of denitrification that have difference submergence characterizes (OMWM ponds, OMWM ditches, non-OMWM ditchers and a nearby creek) within the AT&T site (**Figure 2**). Rates within the OMWM ponds ranged from 29±8, 45±8, and 50±12 μmol N m^{-2} hr^{-1} for May, July and October (**Table 4abc**,

Figure 9), respectively. These rates are slightly lower than the previous study for OMWM sites, and are lower than vegative marshes (Velinsky et al., 2013. For the other submerged areas, rates are either similar to the OMWM ponds or slightly higher, but on average are lower than vegetated sites as measured in the previous study. While pore water sulfide does not appear to limit denitrification in our sites, other factors such as meteorological condition, types of organic matter, DO and nitrite+nitrate concentration could affect the rate. As shown in **Figures 5 and 6**, the level of DO (as shown as %sat), while reduced is not near zero in these ponds and other sites and the amount of dissolved nitrate+nitrite is relatively low. The higher than expected levels of DO in the pond waters is mostly likely due to the active photosynthesis in the ponds by both suspended and benthic algae (**Figure 7**) In addition, the flux of nitrate+nitrite into the sediments in not sufficient to maintain the rate of N_2 production, again suggesting an active coupling of ammonification-nitrification and denitrification in the sediment.

C3: Marsh Processing and Burial in Barnegat Bay

The rate of denitrification and nitrogen accumulation/burial can be viewed in the context of inputs into Barnegat Bay (Velinsky et al. 2010). Nutrients can enter the Bay from river runoff, direct discharge, atmospheric deposition, and ocean exchange, whereas removed through burial (i.e., both in marshes and subtidal), ocean exchange, and importantly for nitrogen: denitrification (i.e., NO_x --→ N_2). Also, inputs of nitrogen can be from fertilizers, wastewater, urban runoff, livestock runoff, and other sources. Regardless, the increased inputs of nutrients (nitrogen and phosphorus) have caused a substantial change in the ecology of the Bay (see for example Kennish, 1984; Kennish et al., 2001, 2007).

The current study expands on the study by Velinsky et al. (2010; 2013) to understand the importance of the services tidal salt marshes perform for the Bay in terms of nitrogen removal. These data can help determine the potential for present-day tidal marshes to provide a service (i.e., ecosystem services) with regard to nutrient removal and, importantly, to help justify the protection and enhancement of tidal wetlands. In addition, the new data on OMWM sites can help to refine previous estimates of nitrogen removal. Removal or sequestration of nutrients by coastal marshes is important, and is dependent on many factors including the areal extent of marshes, accretion rate, nutrient inputs and biogeochemical processes. Nutrients taken up by plants and some heterotrophs are removed on a seasonal basis, but these can be remineralized and recycled to the water column (Mitch and Gosselink, 1993; **Figure 1**). Sediment profiles of nitrogen, as well as phosphorus and carbon, reflect many biogeochemical factors, including biological uptake and transformation (i.e., plant uptake and denitrification) and physical processes such as advection and diffusion of dissolved nutrients and sediment accretion.

C3.1 Sediment Burial Rates and Denitrification

In marsh sediments, many processes (autrophic growth and decomposition) are substantially active in the upper sections of a marsh (e.g., root zones; 0-15 cm) and removal, and especially accumulation rates in this interval may not reflect longer-term burial. For example, sediment concentrations of N in the surface sections are generally elevated compared to concentrations at depths greater than approximately 15 cm. This is due to remineralization processes and the release of dissolved forms of N as well as changes in nutrient loadings to the Bay.

For sediment burial, it is necessary to determine an average sediment concentration of N in each core in order to account for diagenetic changes as well as loading changes over time. For this, concentrations of N were multiplied by the dry sediment density (g/cm^3) at each interval and then divided by the total mass of sediment that represents the past 60 years (i.e., ~1950 to present). The average concentrations were then used along with the bulk accumulation rates (g/m^2-yr) derived from the constant-input-concentration model (CIC) to provide an average accumulation rate for the past 60 years (Velinsky et al. 2010). The depth-integrated rates for nitrogen ranged from 37 to 49 μmol N m^{-2} hr^{-1} (average = 42±6 μmol N m^{-2} hr^{-1}) and were slightly lower than those calculated for the surface section (27 to 62 μmol N m^{-2} hr^{-1}; Velinsky et al. 2010; note unit conversions).

In our pervious study, denitrication rates, averaged across all vegetated sites, ranged from 49±19 μmol N m^{-2} hr^{-1} in October, to 121±20 μmol N m^{-2} hr^{-1} in July, with May exhibiting a rate of 83±14 μmol N m^{-2} hr^{-1}. Two main factors need to be applied to these rates in order to compare them to burial rates. First, the incubations for this study were done under dark conditions to directly measure N_2 production from denitrification. Algal-plant uptake of nitrate and oxygen production in the surface sediments would limit denitrification. Therefore to scale our numbers up for the Bay we need to divide our rates by two assuming that half the time the cores or marsh are in the dark. Secondly, since the marsh is only flooded for a specific time period, each tide, the inundation period needs to be taken into account. Chambers et al. (1992)

assumed a tidal inundation period of approximately 3 hr for their site in Chesapeake Bay. Recent data from Wilson and DePaul (USGS, personal communication) indicate 3-4 hours of tidal inundation is appropriate for Barnegat Bay. In addition, once water overrides the marsh there may be a lag time before the onset of this process. In tidal freshwater marshes this lag may be an hour or two (Ensign et al. 2008). For this initial study we assume no lag time and use the average inundation time between the mid and lower Bay sites in which data are available. From these data from this location it appears that approximately 3 hr of tidal inundation per day is appropriate. Given these two assumptions and that the rate in May, July and October cover a third of the year each yields areal rates of N_2 production of between 9 and 23 μmol N m^{-2} hr^{-1} covering the time in the dark and inundated (note: reduced from our previous time by half given this new information; Velinsky et al., 2013). The range in denitrification rate may still be biased due to potential lag time in the onset of this process and the actual amount of wetlands that is inundated during a tidal cycle, a month and a year. It should be noted that these rates apply to vegetative areas within the bay and exclude the pond areas previously noted above. In our current study, we assumed that all sites are covered (inundated) with water for 24 hr with the same assumptions with light/dark. With these assumptions the rates in the OMWM ponds (only) range from 15 to 23 μmol N m^{-2} hr^{-1} similar to the vegative sites.

C3.2 Conceptual Model: Ecosystem Services a Mass Balance

The current estimate of the area of tidal wetlands in Barnegat Bay is 26,000 acres (1.05×10^8 m^2) with an estimate of OMWM pond area of 9,000 acres (3.6×10^7 m^2) (Lathrop and Haag 2007; www.crssa.rutgers.edu/projects/lc/;OCM, per comm). Using the pond area and the rates above, assuming constant submergence, yields a total N removal via denitrication from OMWM ponds of 0.24-0.37 X 10^4 kg N/yr (average±SD = 0.32±0.07 X 10^4 kg N/yr). For the remaining, non-OMWM marsh area (i.e., 26,000ac-9,000ac) and the N_2 production rates from Velinsky et al. (2013) results in a removal of 2.7-6.6 X 10^4 kg N/yr (average±SD = 4.6±2.0 X 10^4 kg N/yr) from the vegative marshes of Barnegat Bay (**Table 7**). For burial, we use the estimates derived in Velinsky et al. (2010; 2015 submitted). Using the marsh area of 1.05×10^8 m^2, the depth-integrated, and core-top rates for N accumulation current burial rates (gross rates; Velinsky et al. 2010) yield 36 – 42 x 10^4 kg N/yr (**Table 7**).

Using recent N load estimates from 2005 to 2011 for Barnegat Bay (Baker et al., 2014), coastal salt marshes can sequester ~85% of the nitrogen introduced into the bay (**Table 7**). The calculated removal percentages suggest that most of the nitrogen entering the Bay can be and is removed from the system via marsh burial and denitrification alone. Other inputs and removal terms such outwelling to the coastal ocean, biotic uptake/fishing, etc., are not taken into account. However, there are many areas of uncertainty in comparing the three flux areas (i.e., loads, burial and denitrification). Each source and removal function works on a different time scale; measured over many years. They include direct discharge from surface waters (base and storm), groundwater and atmospheric deposition and are, in summary, comparable to previous estimates. Burial rates are averaged over the past ~100 yrs (using ^{210}Pb dating) and therefore maybe biased low since there could be changes over time and changes in nitrogen processing over time, while denitrification rates have substantial spatial and temporal variation and depend on the hydroperiod, which was estimated. In addition, no estimates were made during the colder time periods of the year in which rates would be substantially reduced. Overall however, these calculations do show that marsh accumulation/burial and denitrification can sequester or remove a large portion of the N loads from the various sources (i.e., point sources and non-point sources). Further analysis of all the potential uncertainties of each load and removal term needs to be undertaken to better assess the importance of tidal wetlands in the Bay and the services they can provide.

In addition, sediment recycling of N and P (Berner 1980, Burdige 2006) are not accounted for in these estimates and will modify and most likely reduce these estimates (i.e., Burial – Recycling = Net Burial). The estimates provided above show that the marshes as well as subtidal areas (Seitzinger 1992), have a potential to trap N before being exported to the bay from the non-tidal watershed and highlight the importance of ecosystem services that marshes provide (i.e., water filtration) and the potential cost of water treatment if marsh areas are reduced by either land development or sea level rise.

D) Summary and Conclusions

This study involved the assessment of the denitrification rate in an area impacted by open marsh water management (OMWM). The site in Strafford, NJ has been studied in the past but not assessed for changes in marsh microbial processes with Barnegat Bay These data, along with previous marsh nitrogen cycling and sediment burial rate from tidal marshes obtained in an earlier studies, help to provide a picture of the potential ecosystem services tidal wetlands in the Bay can provide.

<u>Major findings of this study include:</u>

- Denitrification rates were generally similar across sites with slightly higher rates in non-OMWM ditches.
- Potentially, denitrification in the marshes (i.e., using all sites and areas) can remove approximately 4 to 9% of the incoming estimated N load entering the Bay (7.7×10^5 kg/yr; average from 2005-2011).
- Combined with sediment burial, denitrification can sequester/remove 85% of the incoming load.
- Tidal marshes within the Barnegat system are an important component of the ecosystem and help to remove a substantial amount of N entering the Bay.

<u>Recommendations for Future Research and Monitoring</u>

- This study determined that tidal marshes in the Bay can remove a substantial amount of N entering the system. On a seasonal time scale (over the course of the study) upwards of a third of the N is converted to nitrogen gas. Burial works on longer time scales, years to decades, and may then bias the total removal to higher than expected values. In this regard, shorter term burial estimates need to be accomplished to make the two removal mechanisms more comparable (e.g., studies using sediment deposition plates and ^{7}Be analysis would aid in this comparison).
- An aerial survey of the extent of OMWM and rates within these sub-systems needs to be undertaken to better determine the impact on the overall budget of N to the Bay.

- Denitrification and burial in the sub-tidal waters of the Bay need to be assessed in order to make a complete determination of the removal processes in the Bay and the services they provide.
- An uncertainty analysis needs to be undertaken to properly determine how the different source/removal terms can be compared.

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F) References

American Public Health Association, American Water Works Association and Water Environment Federation (APHA, AWWA and WEF). 1995. *Standard Methods for the Examination of Water and Wastewater*, 19th Edition. Washington, DC.

Adamowicz, S.C. and C.T. Roman 2005. New England salt marsh pools: A quantitative analysis of geomorphic and geographic features. Wetlands: 25: 279-288.

An, S.M. and W.S. Gardner. 2002. Dissimilatory nitrate reduction to ammonium (DNRA) as a nitrogen link, versus denitrification as a sink in a shallow estuary (Laguna Madre/Baffin Bay, Texas). Mar. Ecol. Prog. Ser. 237:41-50.

An, S. W.S. Gardner, and T. Kana. 2001. Simultaneous measurement of denitrification and nitrogen fixation using isotope pairing with membrane inlet mass spectrometer (MIMS) analysis. Applied and Environmental Microbiology. 67:1171-1178.

Ayars, J. and Y. Gao. 2007. Atmospheric nitrogen deposition to the Mullica River-Great Bay Estuary. Marine Environ. Res. 64: 590-600.

Baker, R.J., Wieben, C.M., Lathrop, R.G., and Nicholson, R.S., 2014, Concentrations, loads, and yields of total nitro- gen and total phosphorus in the Barnegat Bay-Little Egg Harbor watershed, New Jersey, 1989–2011, at multiple spatial scales: U.S. Geological Survey Scientific Investigations Report 2014–5072, 64 p., http://dx.doi.org/10.3133/ sir20145072.

Barnegat Bay National Estuary Program (BBNEP) 2005. 2005 State of the Bay Technical Report. (http://bbp.ocean.edu/Reports/2005-state_of_bay_tech.pdf)

Berner, R. 1980. *Early Diagenesis: A Theoretical Approach.* Princeton University Press, 241 pp.

Bonin P. 1996. Anaerobic nitrate reduction to ammonium in two strains isolated from coastal marine sediment: A dissimilatory pathway. FEMS Microbiol. Ecol. 19:27–38.

Bowden, W.B. 1986. Nitrification, nitrate reduction, and nitrogen immobilization in a tidal freshwater marsh sediment. Ecology 67 88-99.

Bowden, W.B., C.J. Vorosmarty, J.T. Morris, B.J. Peterson, J.E. Hobbie, P.A. Steudler, and B. Moore. 1991. Transport and processing of nitrogen in a tidal fresh-water wetland. Water Resources Research 27:389-408.

Bowen, J. L., J. M. Ramstack, S. Mazzilli, and I. Valiela. 2007. NLOAD: an interactive, web based modeling tool for nitrogen management in estuaries. Ecological Applications 17: S17–S30.

Borgatti, N. 2008 Nitrogen loading in the Barnegat Bay – Little Egg Harbor estuary and watershed: Developing a conservative model for determining the contribution to the total nitrogen load from lawn fertilizers and a review of existing data. Save the Barnegat Bay (*draft manuscript*).

Burgin, A.J. and S.K. Hamilton. 2007. Have we overemphasized the role of denitrification in aquatic ecosystems? A review of nitrate removal pathways. Front in Ecol. Environ. 5:89-96.

Burdige, D.J. 2006. Geochemistry of Marine Sediments, Princeton University Press.

Castro, M.S. and C.T. Driscoll. 2002. Atmospheric nitrogen deposition to estuaries in the mid-Atlantic and northeastern United States. Environmental Science and Technology, 36: 3242–3249

Chambers R.M., J. W. Harvey, and W.E. Odum. 1992. Ammonium and phosphate dynamics in a Virginia salt marsh. Estuaries 15: 349-359.

Christenson, P.B., L.P. Nielsen, J. Sørenson, and N.P. Revsbech. 1990. Denitrification in nitrate rich streams: Diurnal and seasonal variation related to benthic oxygen metabolism. Limnol. Oceanogr. 35:640-651.

Clarke, J.A., B.A. Harrington, T. Hruby and F.E. Wasserman. 1984. The effect of ditching for mosquito control on salt marsh use by birds in Rowley, Massachuetts. *J. Field Ornithol*. 55: 160-180.

Cornwell, J.C., M.W. Kemp, and T.M. Kana. 1999. Denitrification in coastal ecosystems: Methods, environmental controls, and ecosystem level controls, a review. Aquatic Ecol. 33:41-54.

Durand, J.B., 1984. Nitrogen distribution in New Jersey coastal bays. In: Kennish, M.J. and Lutz, R.A. (eds.), *Ecology of Barnegat Bay*, New Jersey. New York, Springer-Verlag, pp. 29-51.

Duarte, C.M., D.J.Conley, J. Carstensen, and M. Sanches-Camacho 2009. Return to Neverland: Shifting baselines affect eutrophication restoration targets. Estuaries and Coasts 32: 29-36.

Elsey-Quirk, T. and S. Adamowicz. 2015. Influence of physical manipulations on short-term salt marsh morphodynamics: Examples from the north and mid-Atlantic coast, USA. Estuaries and Coasts (accepted).

Engelhart, S.E., B.P.Horton, B.C. Douglas, W.R. Peltier, and T.E. Törnqvist. 2009. Spatial variability of late Holocene and 20th century sea level rise along the U.S. Atlantic Coast. Geology 37:1115-1118.

Ensign, S.H., M.F. Piehler, and M.W. Doyle. 2008. Riparian zone denitrification affects nitrogen flux through a tidal freshwater river. Biogeochemistry 91:133–150.

Eyre, B.D. and A.J.P. Feruson. 2002a. Comparison of carbon production and decomposition, benthic nutrient fluxes and denitrification in seagrass, phytoplankton, benthic microalgae- and macroalgae-dominated warm-temperate Australian lagoons. Mar. Ecol. Prog. Ser. 229:43-59.

Eyre, B.D., S. Rysgaard, T. Dalsgaard, and P.B. Christensen. 2002b. Comparison of isotope pairing and N-2:Ar methods for measuring sediment-denitrification-assumptions, modifications, and implications. Estuaries 25:1077-1087.

Fazzolari, E., B. Nicolardot, and J.C. Germon. 1998. Simultaneous effects of increasing levels of glucose and oxygen partial pressures on denitrification and dissimilatory nitrate reduction to ammonium in repacked soil cores. European Journal of Soil Biology 34:47-52.

Ferrigno, F., and D.M. Jobbins. 1968. Open marsh water management. Proceedings of the New Jersey Mosquito Extermination Association 55: 104–115.

Fernandes, S.O., P.C. Bonin, V.D. Michotey, N. Garcia, and P.A. LokaBharathia. 2012. Nitrogen-limited mangrove ecosystems conserve N through dissimilatory nitrate reduction to ammonium. Scientific Reports DOI: 10.1038/srep004419.

Gastrich, M.D.,J.A. Leigh-Bell, C.J. Gobler, O. R. Anderson, S.W. Wilhelm, M. Bryan. 2004. Viruses as potential regulators of regional brown tide blooms caused by the alga*, A. anophagefferens*. Estuaries 27: 112-116.

Gribsholt B., H.T.S. Boschker, E. Struyf, M. Andersson, A. Tramper, L. De Brabandere, S. van Damme, N. Brion, P. Meire, F. Dehairs, J.J. Middleburg, and C.H.R. Heip. 2005. Nitrogen processing in a tidal freshwater marsh: A whole-ecosystem ^{15}N labeling study. Limnol. Oceanogr. 50:1945–1959.

Gribsholt B., E. Struyf, A. Tramper, M.G.I. Andersson, N. Brion, L. De Brabandere, S. van Damme, P. Meire, J.J. Middelburg, F. Dehairs, and H.T.S. Boschker. 2006. Ammonium transformation in a nitrogen-rich tidal freshwater marsh. Biogeochemistry 80:289–298. doi:10.1007/s10533-006-9024-8

Gribsholt B., E. Struyf, A. Tramper, L. De Brabandere, N. Brion, S. van Damme, P. Meire, F. Dehairs, J.J. Middelburg, H.T.S. Boschker. 2007. Nitrogen assimilation and short term retention in a nutrient-rich tidal freshwater marsh—a whole ecosystem 15 N enrichment study. Biogeosciences 4:11–26

Groffman P.M., N.J. Boulware, W.C. Zipperer, R.V. Pouyat, L.E. Band, M.F. Colosimo. 2002. Soil nitrogen cycling processes in urban riparian zones. Environ. Sci. Technol. 36:4547–52.

Groffman, P.M., N.L. Law, K.T. Belt, L.E. Band, and G.T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393-403.

Heiri, O., A.F. Lotter and G.Lemcke. 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Paleolimnology*. 25: 101-110.

Hochard, S., C. Pinazo, C. Grenz, J.L. Burton Evans, and O. Pringault. 2010. Impact of microphytobenthos on the sediment biogeochemical cycles: A modeling approach. Ecological Modeling 221:1687-1701.

Homer, C., C. Huang, L. Yang, B. Wylie, and M. Coan. 2004. Development of a 2001 National Land Cover Database for the United States. Photogrammetric Engineering and Remote Sensing 70 (7):829-840.

Hopfensperger K.N., S.S. Kaushal, S.E.G. Findlay, J.C. Cornwell. 2009. Influence of plant communities on denitrification in a tidal freshwater marsh of the Potomac River, United States. J. Environ. Qual. 38:618-626.

Hopkinson, C.S and J.P. Schubauer. 1984. Static and Dynamic Aspects of Nitrogen Cycling in the Salt Marsh Graminoid *Spartina Alterniflora*. *Ecology* 65: 961-969.

Hopkinson C. and A. Giblin 2008. Nitrogen dynamics in salt marsh ecosystems, p. 977-1022. *In* D. Capone, D. Bronk, D. Mulholland and E. Carpenter (eds.) *Nitrogen in the Marine Environment*. Academic.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Jaworski, K. Lajtha, J.A. Downing, R. Elmgren, N. Caraco, and T. Jordan, F. Berendse, J. Freney, V. Kudeyarov, P. Murdoch, and Z. Zhao-Liang. 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Biogeochemistry 35:75-139.

Hunchak-Kariouk, K. and R.S. Nicholson. 2001. Watershed contributions of nutrients and other nonpoint source contaminants to the Barnegat Bay–Little Egg Harbor Estuary. *Journal of Coastal Research Special Issue* 32: 28–81.

SAS Institute Inc. 2008. *JMP® 8 Introductory Guide.* Cary, NC: SAS Institute Inc.

James-Pirri, MJ, R.M. Erwin, D.J. Prosser and J.D. Taylor. 2011. Responses of salt marsh ecosytems to mosquito control management practices along the Atlantic Coast (USA). Restoration Ecology (20): 395-404.

Joye, S.B. and J.T. Hollibaugh. 1995 Influence of sulfide inhibition of nitrification on nitrogen regeneration in sediments. Science 270:623–625.

Joye, S.B., S.V. Smith, J.T. Hollibaugh, and H. Paerl. 1996. Estimating denitrification rates in estuarine sediments: A comparison of stoichiometric and acetylene based methods. Biogeochemistry 33:197-215.

Kana, T.M., C. Darkangelo, M. Hunt, J. Oldham, G. Bennett, and J. Cornwell. 1994. Membrane inlet mass-spectrometer for rapid high-precision determination of N-2, O-2, and Ar in environmental water samples. Anal. Chem. 66:4166-4170.

Kana, T.M., M.B. Sullivan, J.C. Cornwell, and K.M. Groszkowski. 1998. Denitrification in estuarine sediments determined by membrane inlet mass spectrometry. Limnol. Oceanogr. 43:334-339.

Kauffman, G.J., A.R. Homsey, A.C. Belden, and J.R. Sanchez. 2011. Water quality trends in the Delaware River Basin (USA) from 1980 to 2005. Environ. Monit. Assess. 177:193-225.

Kauffman, G. and C. DeSisto. 2006. Delaware River State of the Basin Report 2006 Development of Environmental Indicators, Delaware Water Resources Center, University of Delaware, Newark, Delaware, 18 pages.

Kelly et al. 1985. Benthic-pelagic coupling and nutrient cycling across an experimental eutrophication gradient. Mar. Ecol. Prog. Ser. 26:207-219.

Kennish, M.J. 1984. Summary and conclusions. In: Kennish, M.J. and R.A. Lutz (eds). *Lecture Notes on Coastal and estuarine studies. Ecology of Barnegat Bay, New Jersey*. New York, Springer Verlag, pp 339-353.

Kennish, M.J. 2001. Characterization of the Barnegat Bay-Little Egg Harbor estuary and watershed. In: Kenish, M.J. (ed) Barnegat Bay-Little Egg Harbor, New Jersey: estuary and watershed assessment. *Journal of Coastal Research* 32:3-12.

Kennish, M.J., Roche, M.B., and T.R. Tatham. 1984. Anthropogenic effects on aquatic communities. In: Kennish, M.J. and R.A. Lutz (eds). *Lecture Notes on Coastal and estuarine studies. Ecology of Barnegat Bay, New Jersey*. New York, Springer Verlag, pp 318-337.

Kennish, M.J. et al. 2007. Barnegat Bay-Little Egg Harbor estuary: Case study of a highly eutrophic coastal bay system. *Ecological Applications* 17 (supplement): S3-S17.

Kemp, W.M., P. Sampou, J. Caffrey, M. Mayer, K. Henriksen, and W.R. Boyton. 1990. Ammonium recycling versus denitrification in Chesapeake Bay Sediments. Limnol. Oceanogr. 35:1545-1563.

Koop-Jakobsen, K.K. and A.E. Giblin. 2010. The effect of increased nitrate loading on nitrate reduction via denitrification and DNRA in salt marsh sediments. Limnol. Ocenanogr. 55(2): 789-802.

Lathrop, R.G. 2004. Measuring Land Use Change in New Jersey: Land Use Update to Year 2000**.** Rutgers University, Grant F. Walton Center for Remote Sensing and Spatial Analysis, New Brunswick, NJ, CRSSA Report#2004-01.

Lathrop, R.G. and S. Haag. 2007. Assessment of Land Use Change and Riparian Zone Status in the Barnegat Bay and Little Egg Harbor Watershed: 1995-2002-2006. Rutgers University, Grant

F. Walton Center for Remote Sensing and Spatial Analysis, New Brunswick, NJ, CRSSA Report#2007-04.

Lavrentyev, P.J., W.S. Gardner, and L. Yang. 2000. Effects of the zebra mussel on nitrogen dynamics and the microbial community at the sediment-water interface. Aquat. Microb. Ecol. 21:187-194.

Loomis, M.J. and C.B. Craft. 2010. Carbon sequestration and nutrient (nitrogen and phosphorus) accumulation in river-dominated tidal marshes, Georgia, USA. Soil Sci. Soc. Am. J. 74:1028-1036; DOI: 10.2136/sssaj2009.0171.

McCarthy MJ, Gardner WS, Lavrentyev PJ, Moats KM, Jochem FJ, Klarer DM 2007. Effects of hydrological flow regime on sediment–water interface and water column nitrogen dynamics in a Great Lakes coastal wetland (Old Woman Creek, Lake Erie). J Great Lakes Res 33: 219–231

McCormick, J. 1970. The natural features of Tinicum marsh with particular emphasis on the vegetation. In *Two Studies of Tinicum Marsh*. The Conservation Foundation, Washington DC.

McKeller, Jr., H.N., D.L. Tufford, M.C. Alford, P. Saroprayogi, B.J. Kelley, and J.T. Morris. 2007. Tidal nitrogen exchanges across a freshwater wetland succession gradient in the upper Copper River, South Carolina. Estuaries and Coasts 30:989-1006.

Merrill, J.Z. and J.C. Cornwell. 2000. The role of oligohaline marshes in estuarine nutrient cycling. *In* Weinstein, M., Kreeger, D.A. (eds.) *Concepts and Controversies in Tidal Marsh Ecology*. Kluwer Press, Dordrecht, pp. 425–441.

Meredith, W.H., D.E. Saveikis, and C.J. Stachecki. 1985. Guidelines for "Open Marsh Water Management" in Delaware's salt marshes-objectives, system designs, and installation procedures. Wetlands 5: 119–133.

Miller-Way, T. and R.R Twilley. 1996. Theory and operation of continuous flow systems for the study of benthic-pelagic coupling. Mar. Ecol. Prog. Ser. 140:257-269.

Mitsch, W. J. and Gosselink. J. G. (1993). Wetlands. Van Nostrand Reinhold. New York, 722 pp.

Mulholland, P.J., A.M. Helton, et al. 2008. Stream denitrification across biomes and its response to anthropogenic nitrogen loading. Nature 452:202-206. doi:10.1038/nature06686.

Neubauer S.C., I.C. Anderson, B.B. Neikirk. 2005. Nitrogen cycling and ecosystem exchanges in a Virginia tidal freshwater marsh. Estuaries 28(6):909–922. doi:10.1007/BF02696019.

Newell, R.I.E., J.C. Cornwell, and M.S. Owens. 2002. Influence of simulated bivalve biodeposition and microphytobenthos on sediment nitrogen dynamics, a laboratory study. Limnol Oceanogr. 47:1367-1379.

Nowicki, B.L. 1994. The effect of temperature, oxygen, salinity, and nutrient enrichment on estuarine denitrification rates measured with a modified nitrogen gas flux technique. Esturarine Coastal Shelf Sci. 38:137-156.

Nijburg J.W., M.J.L. Coolen, S. Gerards, et al. 1997. Effects of nitrate availability and the presence of Glyceria maxima on the composition and activity of the dissimilatory nitrate reducing bacterial community. Appl. Environ. Microb. 63:931–37.

Olsen, P.S. and J.B. Mahoney 2001. Phytoplankton in the Barnegat Bay—Little Egg Harbor Estuarine System: Species Composition and Picoplankton Bloom Development. Jour. Coastal Res. 32: 115-143.

Palmer, M.A. and S. Filoso. 2009. Restoration of ecosystem services. Science 325: 575-576.

Piehler, M.F. and A.R. Smyth. 2011. Habitat-specific distinctions in estuarine denitrification affect both ecosystem function and services. Ecosphere 2: art12. doi: 10.1890/ES10-00082.1.

Poe, A., M.F. Piehler, S.P. Thompson, and H.W. Paerl. 2003. Denitrification in a constructed wetland receiving agricultural runoff. Wetlands 23:817-826.

Racchetti, E., M. Bartoli, E. Soana, D. Longhi, R. Christian, M. Pinardi, and P. Viaroli. 2011. Influence of hydrological connectivity of riverine wetlands on nitrogen removal via denitrification. Biogeochemistry 103:335-354.

Ruttenberg, K.C. 1992. Development of a sequential extraction method for different forms of phosphorus in marine sediments. *Limnol. Oceanogr*. 37: 1460-1482.

Schlesinger W.H. 1997. *Biogeochemistry*. London: Academic Press.

Scudlark J.R. and T.M. Church. 1989. The sedimentary flux of nutrients at a Delaware salt marsh site: A geochemical perspective Biogeochemistry 7: 55-75.

Seitzinger S.P. 1994. Linkages between organic matter mineralization and denitrification in eight riparian wetlands. Biogeochemistry 25:19–39

Seiztinger 1987. Nitrogen biogeochemistry in an unpolluted estuary: The importance of benthic denitrification. Mar. Ecol. Progr. Ser. 37: 65-73.

Seitzinger, S.P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. Limnol. Oceanogr. 33:702-724.

Seitzinger, S.P. 1992. Nutrient loading in Barnegat Bay: Importance of sediment-water nutrient interactions (Year II). Final Report 92-24F. The Academy of Natural Sciences; Division of Environmental Research, Philadelphia, PA.

Seitzinger S.P., W.S. Gardner, A.K. Spratt. 1991. The effect of salinity on ammonium sorption in aquatic sediments: Implications for benthic nutrient recycling. Estuaries 14:167–174.

Seitzinger, S.P., S.W. Nixon, and M.E.Q. Pilson. 1984. Denitrification and nitrous oxide production in a coastal marine ecosystem. Limnol. Oceanogr. 29:73- 83.

Seitzinger, S.P. and I.E. Pilling 1993. Eutrophication and nutrient loading in Barnegat Bay: sediment-water phosphorus dynamics. Report No. 92-33 F, The Academy of Natural Sciences, Philadelphia, PA.

Seitzinger, S.P., J. Harrison, J. Bohlke, A. Bouwman, R. Lowrance, B. Peterson, C. Tobias, and G. Van Drecht. 2006. Denitrification across landscapes and waterscapes: a synthesis. Ecological Applications. 16: 2064-2090

Smith, C.J., R.D. DeLaune, and W.H. Patrick. 1985. Fate of riverine nitrate entering and estuary: Denitrification and nitrogen burial. Estuaries and Coasts 8:15-21.

Smith, L.K., M.A. Voytek, J.K. Böhlke, and J.W. Harvey. 2006. Denitrification in nitrate-rich streams: Application of N_2:Ar and ^{15}N-tracer methods in intact cores. Ecol App 16:2191-2207.

Sørensen J. 1987. Nitrate reduction in marine sediment: Pathways and interactions with iron and sulfur cycling. Geomicrobiol. J. 5:401-421

Thompson, S.P., H.W. Paerl, and M.C. Go. 1995. Seasonal patterns of nitrification and denitrification in a natural a restored salt marsh. Estuaries 18:399-408.

Tiedje, J.M. 1988. Ecology of denitrification and dissimilatory nitrate reduction to ammonium. In: Zehnder, A.J.B. (Ed). *Biology of Anaerobic Microorganisms*. New York, NY: John Wiley and Sons.

Tobias, C.R. 2007. Linking benthic microalgae to coupled denitrification. page 206. Estuarine Research Federation: Science and Management, Providence, Rhode Island, USA.

US EPA 1997. Methods for the Determination of Chemical Substances in Marine and Estuarine Environmental Matrices. 2nd edition. September 1997. (NSCEP or NTIS / PB97-127326).

US EPA 2005. National Coastal Condition Report II (NCCR II). (http://water.epa.gov/type/oceb/2005_downloads.cfm)

Valiela, I., M. Geist, J. McClelland, and G. Tomasky. 2000. Nitrogen loading from watersheds to estuaries: Verification of the Waquoit Bay nitrogen loading model. Biogeochemistry 49:277

Van Breemen et al. 2002. Where did all the nitrogen go? Fate of nitrogen inputs to large watersheds in the NE USA. Biogeochemistry 57/58: 267-293.

Velinsky, D.J., T. Quirk, M. Piehler, and A. Smyth. 2013. Ecosystem Services of Tidal Wetlands in Barnegat Bay: Nitrogen removal. Final Report submitted to NJ Department of the Environment. PCER Report No. 13-06The Academy of Natural Sciences of Drexel University, Philadelphia, PA.

Velinsky, D.J., C. Sommerfield, M. Enache, and D.Charles. 2010. Nutrient and Ecological Histories in Barnaget Bay, New Jersey. PCER Report No. 10-5; Final Report submitted to New Jersey Department of the Environment (Trenton, NJ).

Verhoeven, J.T.A., B. Arheimer, C. Yin, and M.M. Hefting. 2006. Regional and global concerns over wetlands and water quality. Trends Ecol. Evol. 21:96-103.

Vitousek P.M., J. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D. Tilman. 1997. Human alteration of the global nitrogen cycle: Causes and consequences. Issues Ecol. 1:1-14.

Ward, B.B. 1996. Nitrification and denitrification: probing the nitrogen cycle in aquatic environments. Microb. Ecol. 32: 247-261.

Wieben, C.M. and R.J. Baker. 2009. Contributions of Nitrogen to the Barnegat Bay-Little Egg Harbor Estuary: Updated Loading Estimates. United States Geological Survey and BBEP. Barnegat Bay Partnership. (http://www.bbep.org/studies.html.)

Whigham, D.F. and R.L. Simpson. 1976. The potential use of tidal marshes in the management of water quality in the Delaware River. Pages 173-186 *In*: J. Tourbier and R.R. Pierson, Jr. (eds). *Biological Control of Water Pollution*. University of Pennsylvania Press, Philadelphia, PA.

Tables

Table 1. Core and water locations and collection dates for Barnegat Bay field work.

ID	Lat	Long
Ponds		
P50	39°41'55.42"N	74°12'53.93"W
P46	39°41'53.07"N	74°12'49.67"W
P49	39°41'54.04"N	74°12'52.11"W
P16	39°41'53.26"N	74°12'54.77"W
P19	39°41'50.38"N	74°12'51.70"W
P12	39°41'46.38"N	74°12'53.68"W
P27	39°41'42.38"N	74°12'53.67"W
P31	39°41'44.41"N	74°12'47.06"W
P39	39°41'45.98"N	74°12'43.27"W
Ditches Near Ponds		
DO9	39°41'47.05"N	74°12'50.06"W
DO13	39°41'43.85"N	74°12'52.14"W
DO12	39°41'44.40"N	74°12'49.48"W
Ditches NOT near Ponds		
D9	39°41'55.93"N	74°12'38.90"W
D10	39°41'54.67"N	74°12'38.16"W
D16	39°41'51.13"N	74°12'38.17"W
Creek		
C1	39°41'55.68"N	74°12'50.85"W
C2	39°41'50.06"N	74°12'43.62"W
C3	39°41'45.61"N	74°12'35.37"W

Table 2. Basic water quality parameters from the three sites and time period

	Temperature (oC)	Salinity (ppt)	DO (mg/L)	DO sat %	pH
May					
Min	11.8	1.1	2.7	28.4	5.1
Max	19.4	16.4	12.3	134	8.4
Average	14.6	10.4	7.7	82.5	6.9
Std Dev	1.8	5.2	2.7	32.7	0.8
July					
Min	23.0	12.3	1.1	14.00	4.2
Max	26.9	28.0	12.0	169	8
Average	24.6	25.4	5.7	79.8	6.9
Std Dev	1.2	3.7	2.8	39.7	0.8
October					
Min	18.1	19.4	3.0	6.39	6.3
Max	21.0	24.1	9.9	87.3	7.2
Average	19.1	22.0	5.0	56.1	6.9
Std Dev	0.9	1.4	1.6	19.2	0.2

Table 3c. Summary surface water quality, nutrient and porewater sulfide concentrations in sampling areas in May 2014(avg±SE).

May 2014																
								Porewater -------------		Surface Waters--						
		Temperature	Salinity	DO	DO sat	pH		H2S (0-7cm)	H2S (7-10cm)	Ammonium	Nitrate+Nitrite	TDN	SRP	TDP	DOC
		(oC)	(ppt)	(mg/L)	%			umol/L	umol/L	ugN/L	ugN/L	ugN/L	ug P/L	ug P/L	ug C/L
	min	11.8	1.1	2.7	28.4	5.1		19.3	45.1	0.1	1.0	578.9	2.2	6.9	8.2
	max	19.4	16.4	12.3	134.0	8.4		3712	3944	122.5	12.5	837.3	18.5	35.6	27.2
	AVG	14.6	10.4	7.7	82.5	6.9		1381	1156	37.3	3.7	704.4	6.7	18.1	14.7
	SE	0.43	1.2	0.6	7.7	0.2		441	354	8.9	0.8	17.6	1.1	2.2	1.3
Ponds	**AVG**	15.0	13.7	9.7	104.2	7.4		388	559	6.9	1.2	655.4	2.8	9.9	11.2
	SE	0.4	0.7	0.5	5.6	0.2		130	148	3.4	0.2	16.9	0.2	1.1	0.8
Ditches Near Ponds	**AVG**	14.8	3.9	6.6	66.3	7.1		2172	1446	53.6	6.5	757.4	9.5	24.1	20.1
	SE	0.7	0.3	0.3	2.3	0.2		974.5	940.6	6.0	0.1	40.0	0.2	2.9	1.1
Ditches NOT near Ponds	**AVG**	15.1	12.3	5.5	65.0	6.4		3670	2457	94.5	7.2	750.8	13.2	28.0	13.4
	SE	2.1	3.0	2.4	34.1	0.1		41.9	1159	19.4	2.8	35.9	2.7	4.4	2.7
Creek	**AVG**	12.7	5.1	5.3	51.2	5.8		ND	ND	55.1	4.6	752.2	9.3	26.7	21.1
	SE	0.6	2.8	0.5	3.7	0.3		ND	ND	12.9	1.3	47.6	1.4	0.7	3.9

ND – Not detected; TBD – To be determined.

Table 3b. Summary surface water quality, nutrient and porewater sulfide concentrations in sampling areas in July 2014(avg±SE).

July, 2014							Porewater -------------		Surface Waters--					
		Temperature	Salinity	DO	DO sat	pH	H2S (0-7cm)	H2S (7-10cm)	Ammonium	Nitrate+Nitrite	TDN	SRP	TDP	DOC
		(oC)	(ppt)	(mg/L)	%		umol/L	umol/L	ugN/L	ugN/L	ugN/L	ug P/L	ug P/L	ug C/L
	min	23.0	12.3	1.1	14.0	4.1	74	108	4.0	1.0	373	5.7	15.7	4.0
	max	26.9	28.0	12.0	169.0	7.8	3222	2555	60.3	8.3	1301	20.8	44.9	14.1
	AVG	24.6	25.4	5.7	79.8	6.6	1003	919	16.8	2.5	890	9.6	25.5	9.2
	SE	0.3	0.9	0.80	11	0.3	247	293	14.5	2.1	249	4.2	6.9	2.6
Ponds	**AVG**	24.3	26.5	5.4	76.0	7.3	1618	1542	8.7	1.3	1081	6.9	21.6	11.3
	SE	0.4	0.2	1.0	14.6	0.1	356.9	535.9	1.1	0.1	59.3	0.4	0.7	0.6
Ditches Near Ponds	**AVG**	25.6	25.3	6.4	91.2	6.9	301	920	9.4	1.0	601	8.4	24.2	6.7
	SE	0.8	0.4	1.0	15.9	0.0	62.0	760.8	2.8	0.0	116	0.8	4.2	1.4
Ditches NOT near Ponds	**AVG**	25.1	22.5	NA	NA	6.0	414	295	40.5	5.1	791	12.1	26.7	7.4
	SE	0.8	5.1	NA	NA	0.9	78.2	45.4	10.8	1.7	37.2	1.1	2.0	0.5
Creek	**AVG**	24.1	25.2	NA	NA	5.0	173.2	296.7	24.6	4.9	704	16.3	37.0	7.6
	SE	0.6	2.3	NA	NA	0.9	99	132	4.0	0.5	20.5	2.8	4.1	0.9

ND – Not detected; TBD – To be determined.

Table 3c. Summary surface water quality, nutrient and porewater sulfide concentrations in sampling areas in October 2014(avg±SE).

October 2014								Porewater -------------		Surface Waters---					
		Temperature	Salinity	DO	DO sat	pH		H2S (0-7cm)	H2S (7-10cm)	Ammonium	Nitrate+Nitrite	TDN	SRP	TDP	DOC
		(oC)	(ppt)	(mg/L)	%			umol/L	umol/L	ugN/L	ugN/L	ugN/L	ug P/L	ug P/L	ug C/L
	min	18.1	19.4	3.0	36.8	6.3		25.8	38.7	9.2	1.0	488	3.1	2.6	5.2
	max	21.0	24.1	9.9	87.3	7.2		779.7	1256.6	54.7	14.2	687	17.3	28.8	11.0
	AVG	19.1	22.0	5.0	59.3	6.9		254.9	434.6	24.2	5.2	601	9.3	14.5	7.4
	SE	0.2	0.3	0.4	3.5	0.05		64.4	91.7	3.3	0.8	12.9	1.3	2.0	0.4
Ponds	**AVG**	19.6	21.0	5.6	65.4	6.9		405.3	662.1	11.9	3.6	630	4.0	8.1	8.5
	SE	0.3	0.3	0.6	5.2	0.1		83.7	124.2	1.2	1.4	13.2	0.6	0.8	0.5
Ditches Near Ponds	**AVG**	18.5	21.9	3.4	41.9	6.9		38.7	212.7	43.2	7.7	610	14.1	16.9	7.0
	SE	0.1	0.5	0.2	3.0	0.1		0.0	143.0	6.0	1.7	15.7	0.3	3.9	0.3
Ditches NOT near Ponds	**AVG**	18.5	23.1	4.2	52.2	6.9		125.7	386.7	36.1	6.2	569	13.1	24.0	6.0
	SE	0.1	0.1	0.2	3.1	0.02		87.0	206.2	1.6	0.3	14.6	0.7	0.9	0.1
Creek	**AVG**	18.6	23.8	5.3	65.9	7.1		34.4	81.6	30.5	6.2	504	16.5	25.2	5.2
	SE	0.3	0.1	0.2	2.8	0.1		5.7	33.8	0.9	0.3	16.4	0.5	3.5	0.1

ND – Not detected; TBD – To be determined

Table 4a. Summary surface sediment concentrations in sampling areas during May 2014(avg±SE).

May 2014									
		Surface Sediments (0-3 cm) ---							
		OC	TN	TP	Chlor a		solid	water	OM
		%	%	%	(ug/g wet)		%	%	%
	min	0.5	0.1	0.0	1.1		11.8	20.0	0.7
	max	23.1	1.4	0.1	48.4		80.0	88.2	45.4
	AVG	14.2	0.9	0.1	13.7		21.7	78.3	29.0
	SE	1.4	0.1	0.0	2.5		3.9	3.9	2.9
Ponds	AVG	17.1	1.0	0.1	12.8		15.9	84.1	35.0
	SE	1.5	0.1	0.0	1.0		1.1	1.1	3.0
Ditches Near Ponds	AVG	11.9	0.8	0.1	12.2		22.8	77.2	23.2
	SE	2.6	0.2	0.0	3.7		5.5	5.5	6.2
Ditches NOT near Ponds	AVG	9.4	0.7	0.1	21.3		37.6	62.4	20.8
	SE	4.5	0.3	0.0	14.1		21.2	21.2	10.0
Creek	AVG	11.5	0.6	0.1	8.3		22.1	77.9	22.9
	SE	0.2	0.0	0.0	5.3		4.3	4.3	3.0

Table 4b. Summary surface sediment concentrations in sampling areas during July 2014(avg±SE).

July 2014		Surface Sediments (0-3 cm) --							
		OC	TN	TP	Chlor a		solid	water	OM
		%	%	%	(ug/g wet)		%	%	%
	min	0.6	0.1	0.00	3.5		9.9	26.0	1.5
	max	22.4	1.3	0.1	36.4		74.0	90.1	49.6
	AVG	13.5	0.9	0.1	15.9		23.0	77.0	28.4
	SE	1.4	0.1	0.01	2.2		3.7	3.7	2.8
Ponds	AVG	16.4	1.0	0.1	15.2		18.7	81.3	34.8
	SE	1.9	0.1	0.01	2.0		3.8	3.8	3.8
Ditches Near Ponds	AVG	13.5	1.0	0.1	25.8		15.7	84.3	27.9
	SE	0.6	0.05	0.01	1.6		0.5	0.5	0.3
Ditches NOT near Ponds	AVG	8.9	0.7	0.1	17.6		36.1	63.9	19.4
	SE	4.2	0.3	0.03	9.6		19.0	19.0	9.0
Creek	AVG	9.5	0.5	0.1	6.4		30.0	70.0	18.8
	SE	2.6	0.1	0.01	1.9		4.0	4.0	3.3

Table 4c. Summary surface sediment concentrations in sampling areas during October 2014(avg±SE).

October 2014		Fluxes ----------						
		OC	TN	TP	Chlor a	solid	water	OM
		%	%	%	(ug/g wet)	%	%	%
	min	0.32	0.01	0.002	3.4	8.4	21.5	37.9
	max	24.3	1.4	0.1	63.2	78.5	91.6	79.3
	AVG	13.7	0.89	0.06	15.1	20.6	79.4	69.7
	SE	1.5	0.09	0.01	3.6	3.9	3.9	2.7
Ponds	AVG	17.7	1.06	0.06	13.9	13.9	86.1	73.6
	SE	1.7	0.10	0.01	3.6	1.5	1.5	1.5
Ditches Near Ponds	AVG	10.7	0.79	0.08	12.2	21.6	78.4	61.7
	SE	2.1	0.19	0.02	1.1	7.0	7.0	9.8
Ditches NOT near Ponds	AVG	8.6	0.74	0.08	31.6	34.4	65.6	64.9
	SE	4.1	0.37	0.04	17.4	22.1	22.1	13.5
Creek	AVG	9.9	0.59	0.04	4.8	25.8	74.2	70.9
	SE	3.1	0.13	0.02	0.73	5.7	5.7	1.1

Table 5a. Summary sediment flux data in sampling areas during May 2014(avg±SE).

May 2014						
Temp = 15.1 oC		Fluxes --				
		O2	N2	NH4	Nox	SRP
		umol/m2-hr	umol/m2-hr	umol/m2-hr	umol/m2-hr	umol/m2-hr
	min	-3171.2	4.8	-6.9	-55.2	-10.6
	max	-774.9	96.3	422.2	13.4	2.5
	AVG	-1890.2	44.4	134.3	-24.7	-1.8
	SE	157.6	6.0	24.0	5.6	0.9
Ponds	AVG	-1666.6	29.0	142.6	-31.5	0.2
	SE	200.1	7.7	39.4	7.8	0.7
Ditches Near Ponds	AVG	-2421.5	55.7	152.1	-11.3	-5.9
	SE	386.7	6.0	104.4	6.8	1.1
Ditches NOT near Ponds	AVG	-2457.4	61.3	101.3	-29.7	-3.3
	SE	406.0	18.1	54.0	14.9	3.7
Creek	AVG	-1462.5	45.7	130.3	-12.8	-2.5
	SE	58.7	2.6	12.6	18.0	1.0

Table 5b. Summary sediment flux data in sampling areas during July 2014(avg±SE).

July		Fluxes				
Temp = 26.0 oC		O2	N2	NH4	Nox	SRP
		umol/m2-hr	umol/m2-hr	umol/m2-hr	umol/m2-hr	umol/m2-hr
	min	-6926.5	1.5	-116.1	-125.9	-2.4
	max	-1758.0	130.2	1593.3	49.9	36.2
	AVG	-3895.8	49.8	162.1	-30.0	4.4
	SE	371.0	7.9	90.1	11.6	2.3
Ponds	AVG	-3127.6	45.2	-3.0	5.6	-0.2
	SE	220.2	7.7	23.8	10.0	0.4
Ditches Near Ponds	AVG	-5533.9	22.9	169.8	-68.2	4.1
	SE	635.5	10.8	61.9	30.3	2.8
Ditches NOT near Ponds	AVG	-6071.3	99.1	740.2	-64.2	17.3
	SE	429.8	22.0	435.0	24.0	10.7
Creek	AVG	-2387.0	41.1	71.4	-64.4	5.7
	SE	344.7	15.6	41.8	20.0	6.0

Table 5a. Summary sediment flux data in sampling areas during October 2014(avg±SE).

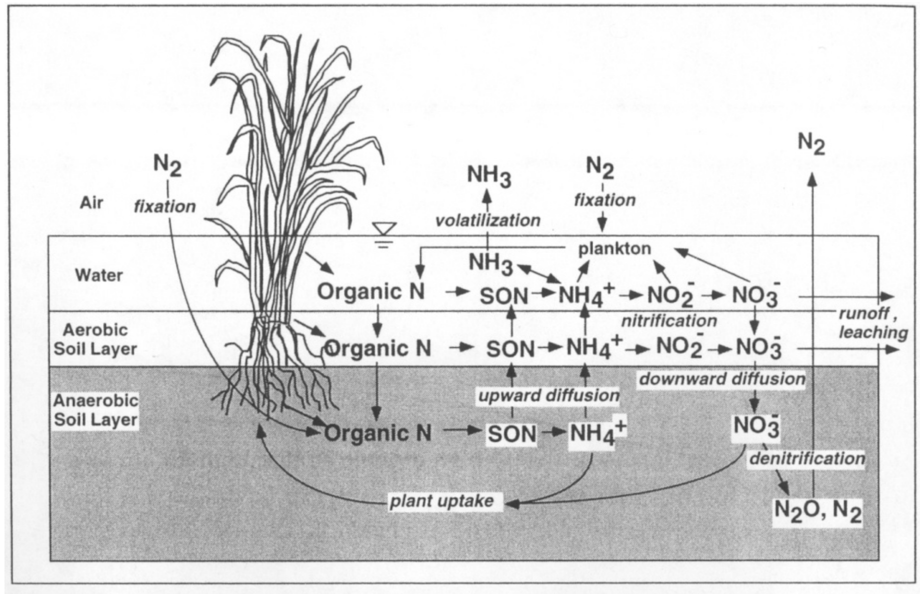
October		Fluxes --				
Temp = 18.9 oC		O2	N2	NH4	Nox	SRP
		umol/m2-hr	umol/m2-hr	umol/m2-hr	umol/m2-hr	umol/m2-hr
	min	-5143.2	10.2	-107.5	-43.9	-1.8
	max	-1565.8	161.3	343.3	10.2	39.6
	AVG	-2494.3	55.1	56.6	-18.0	2.1
	SE	240.5	10.0	31.3	3.0	2.2
Ponds	**AVG**	-2317.3	50.1	66.2	-12.2	-0.5
	SE	162.8	12.4	30.7	4.2	0.3
Ditches Near Ponds	**AVG**	-2925.3	55.0	-51.3	-25.0	0.8
	SE	951.9	20.1	55.9	3.4	1.7
Ditches NOT near Ponds	**AVG**	-3417.3	29.7	244.2	-24.3	13.2
	SE	905.3	6.0	72.4	10.1	13.2
Creek	**AVG**	-1671.5	87.0	-51.9	-22.0	ND
	SE	41.0	38.6	27.1	3.5	ND

Table 6. Comparison of Barnegat Bay marsh nitrogen burial rates measured in this study to rates of nitrogen inputs to the Barnegat Bay.

	Nitrogen kg/yr X 10^5
Inputs (2005-2011; Average)	7.7
Marsh Burial:	
Core Top Concentrations	6.1 ± 0.02
Avg Concentrations (50yrs)	5.2 ± 0.71
Burial as % of Inputs	
Core Top Concentrations	79%
Avg. Concentrations (50yrs)	67%
Marsh Denitrification:	
May	0.48
July	0.70
Oct	0.30
Avg ± SD	0.49 ± 0.20
Denitrification as % of Inputs (average)	6.4%
Total Removal as % of Inputs	73-85%

Nitrogen inputs ranged from 6.6 to 8.6 X10^5 kg/yr (Baker et al., 2014 from 2005 to 2011). Wetland area (26,000 acres, 1.1 X10^8 m^2) are obtained from www.crssa.rutgers.edu/ projects/lc/.

Figures



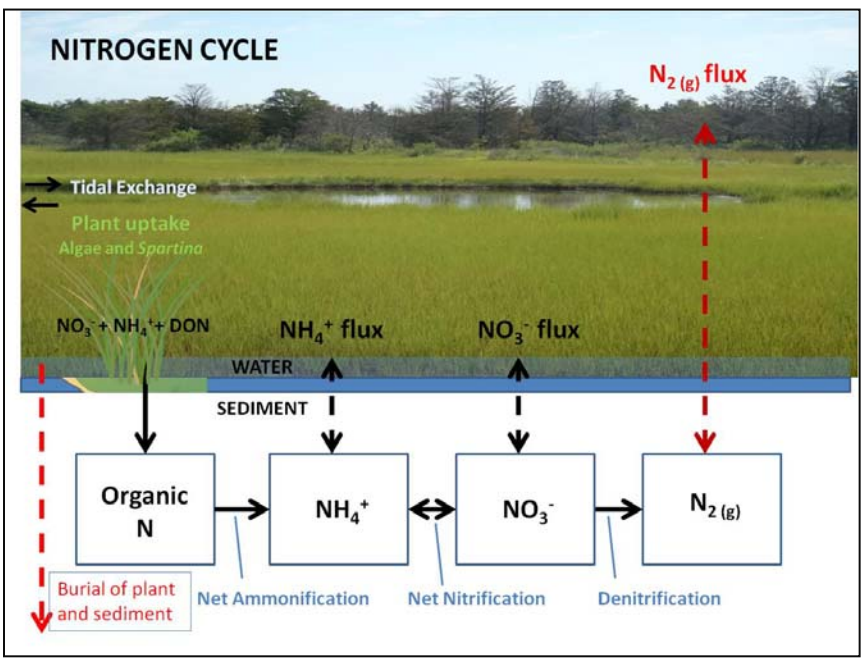


Figure 1. Generalized schematic of nitrogen cycling in wetlands (Mitch and Gosselink, 1993) along with conceptual model of cycling in Barnegat Bay.

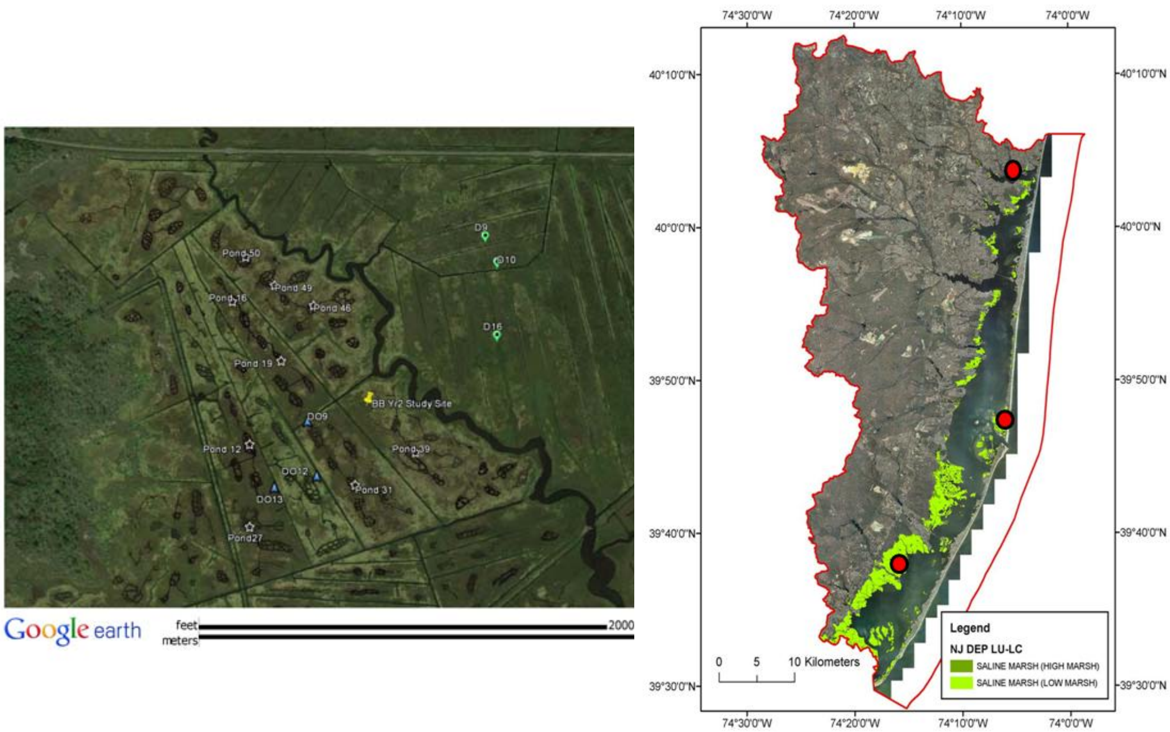


Figure 2. Barnegat Bay study area: The Red dots illustrate the marsh sites in Year 1 (Velinsky et al. 2013) while the triangle (and insert) show current study location (Stafford County, NJ; ATT Site).

Figure 3. Core collected from one of the ponds at the OMWM study area.

Figure 4. Established line transects on the OMWM and control sections of the AT & T site. On each side, transect 1 is red, transect 2 is yellow, and transect 3 is blue (oriented by proximity to the large tidal channel).

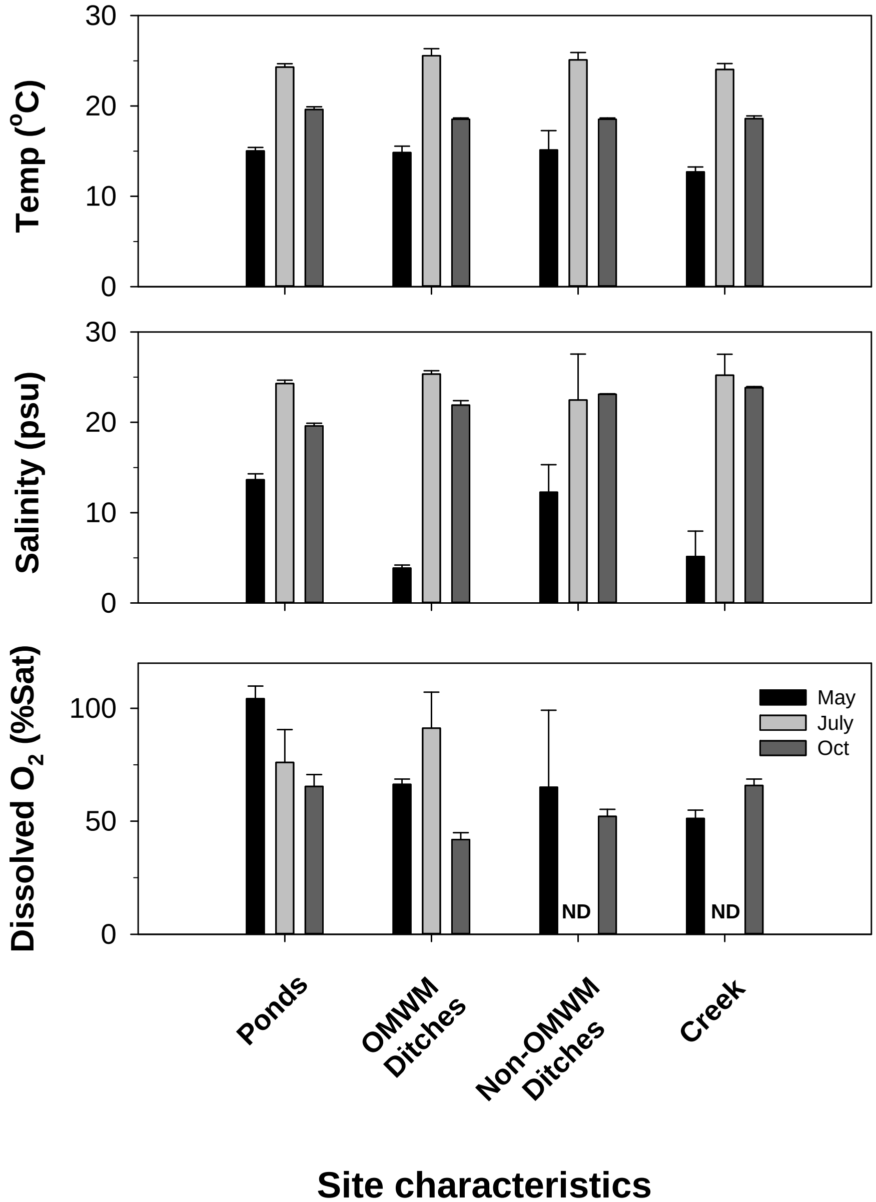


Figure 5. Basic water quality paramters for the AT&T site in May, July and October 2014. Concentrations are average ±1SE.

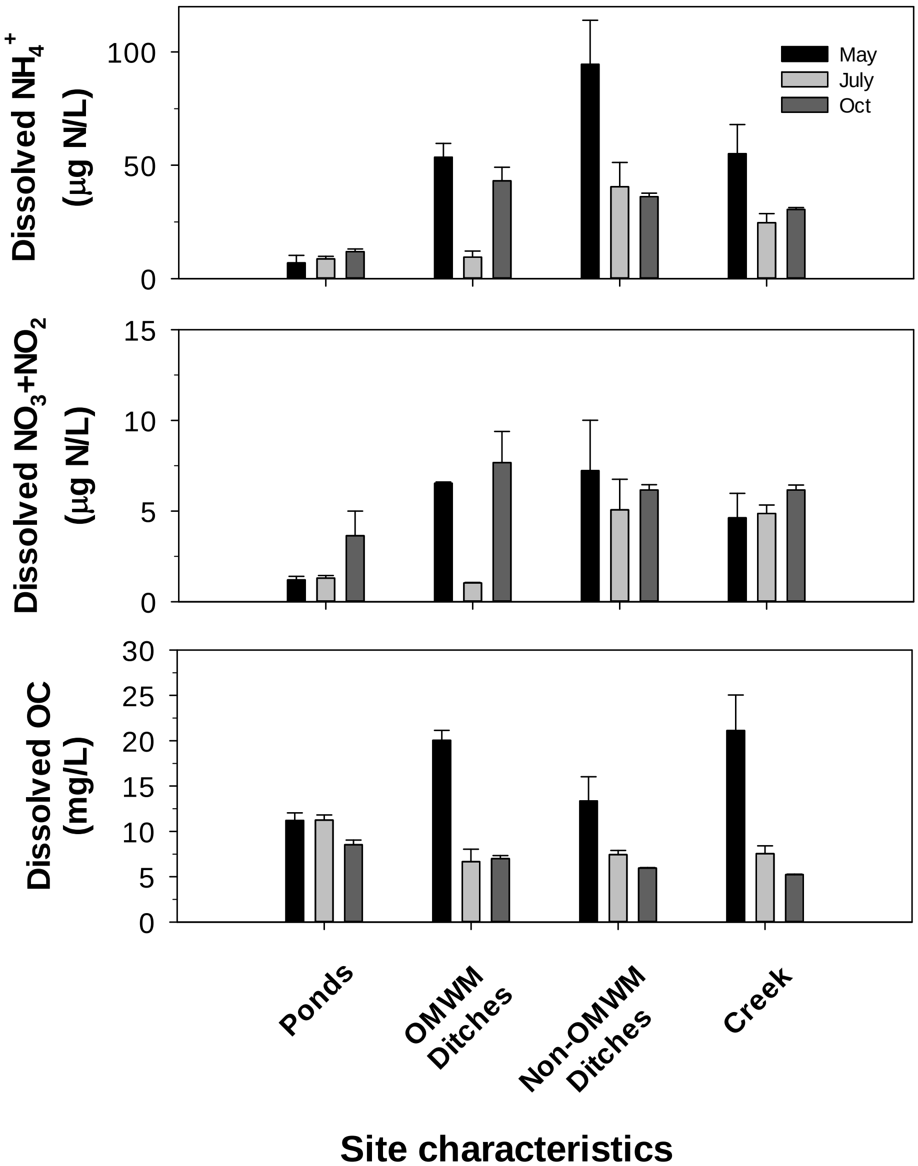


Figure 6. Dissolved inorganic N species and dissolved organic carbon in various sites in May, July and October of 2014. Concentrations are average ±1SE.

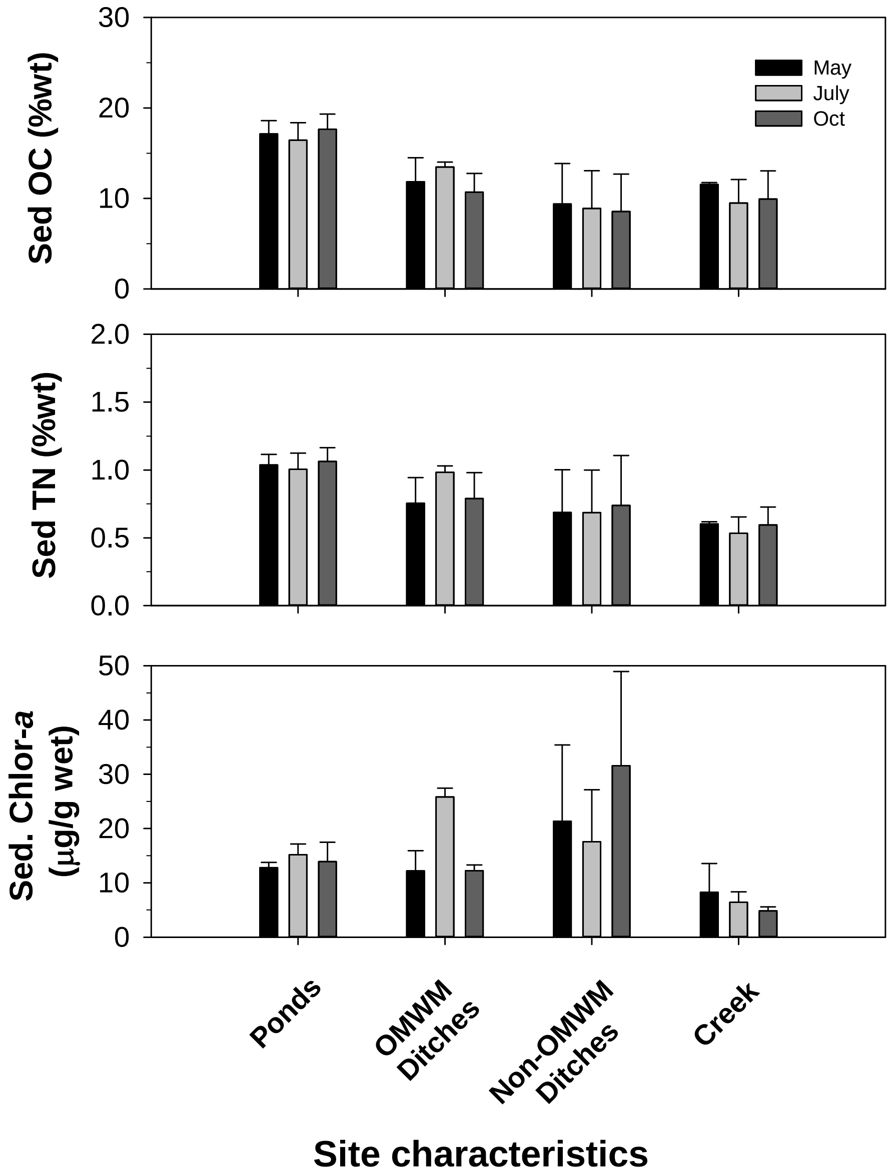


Figure 7. Sediment quality charactertics for the study sitesduring May, July and October of 2014. Concentrations are average ±1SE.

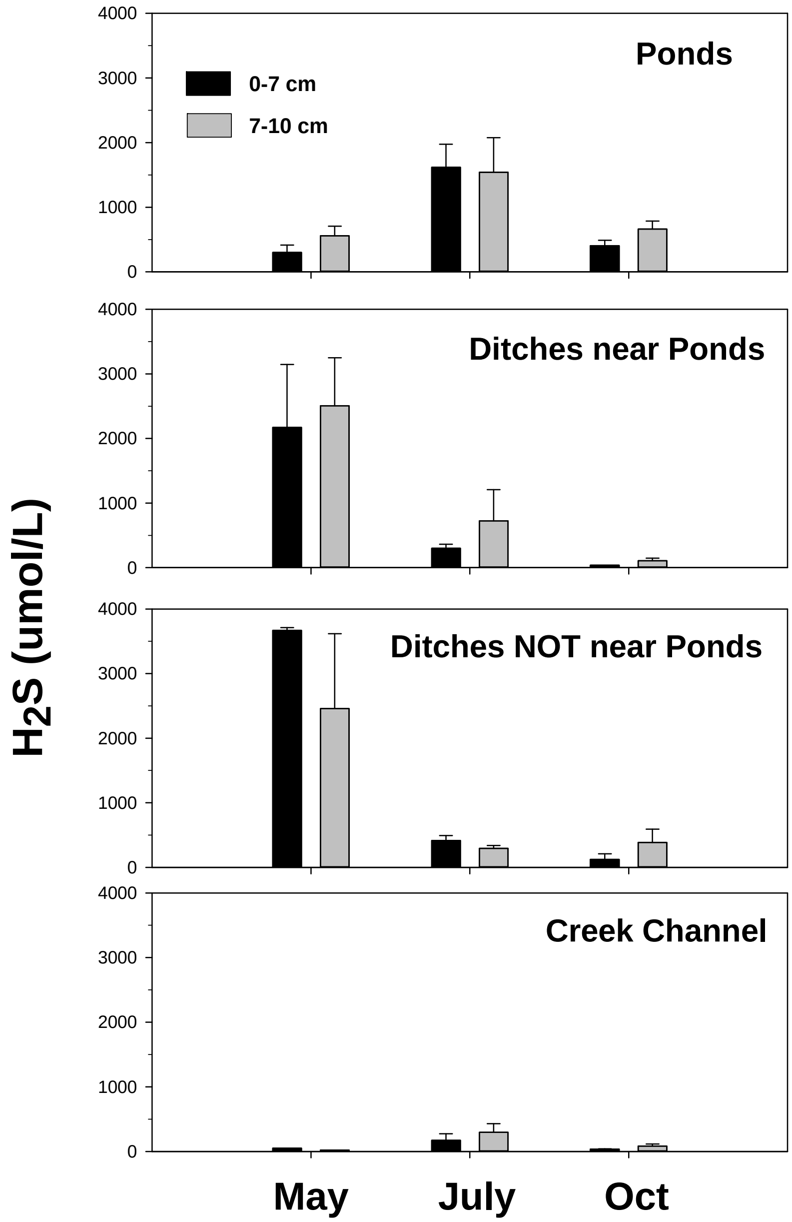


Figure 8. Average (±SE) dissolved hydrogen sulfide at the two depth intervals from the three sampling periods at the study sites.

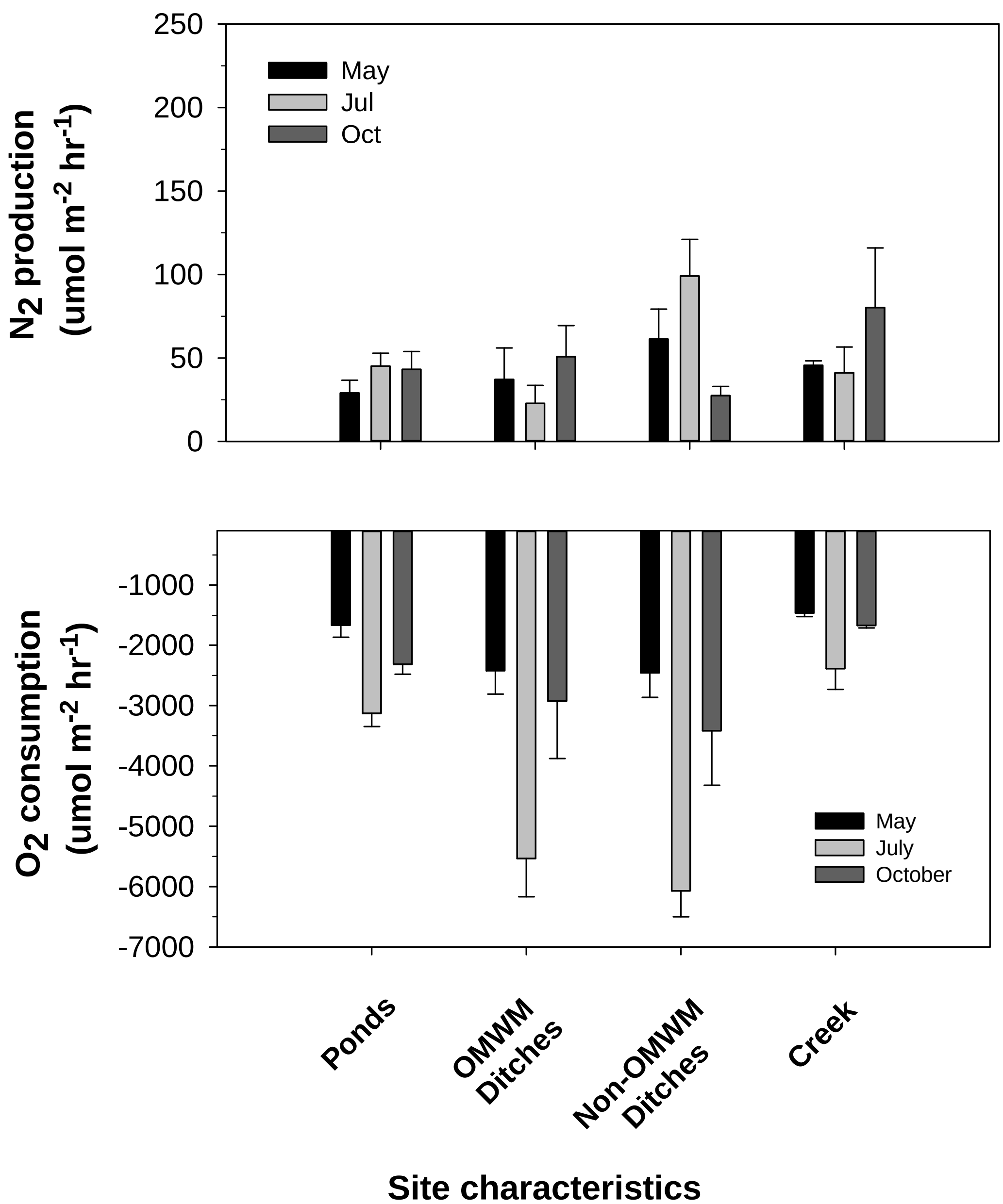


Figure 9. Nitrogen production and oxygen consumption rates for the study locations during the three periods. Rates are the average ±1SE (n = 6).

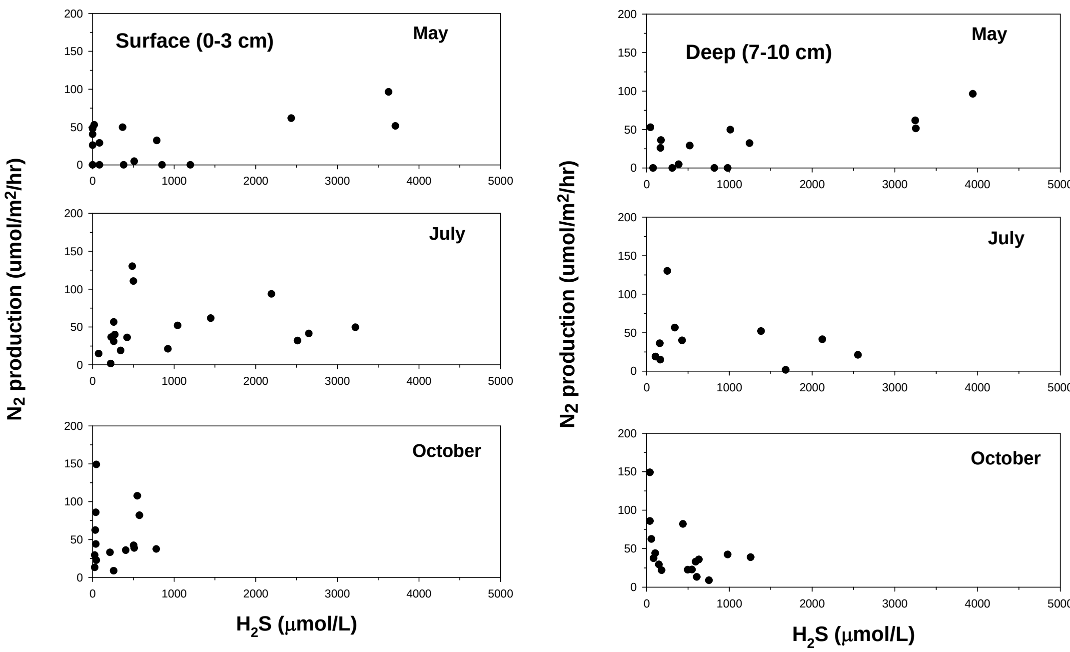


Figure 10. Relationship between porewater sulfide and N_2 production rates at the two depth intervals during the three sampling periods (May, July and October)

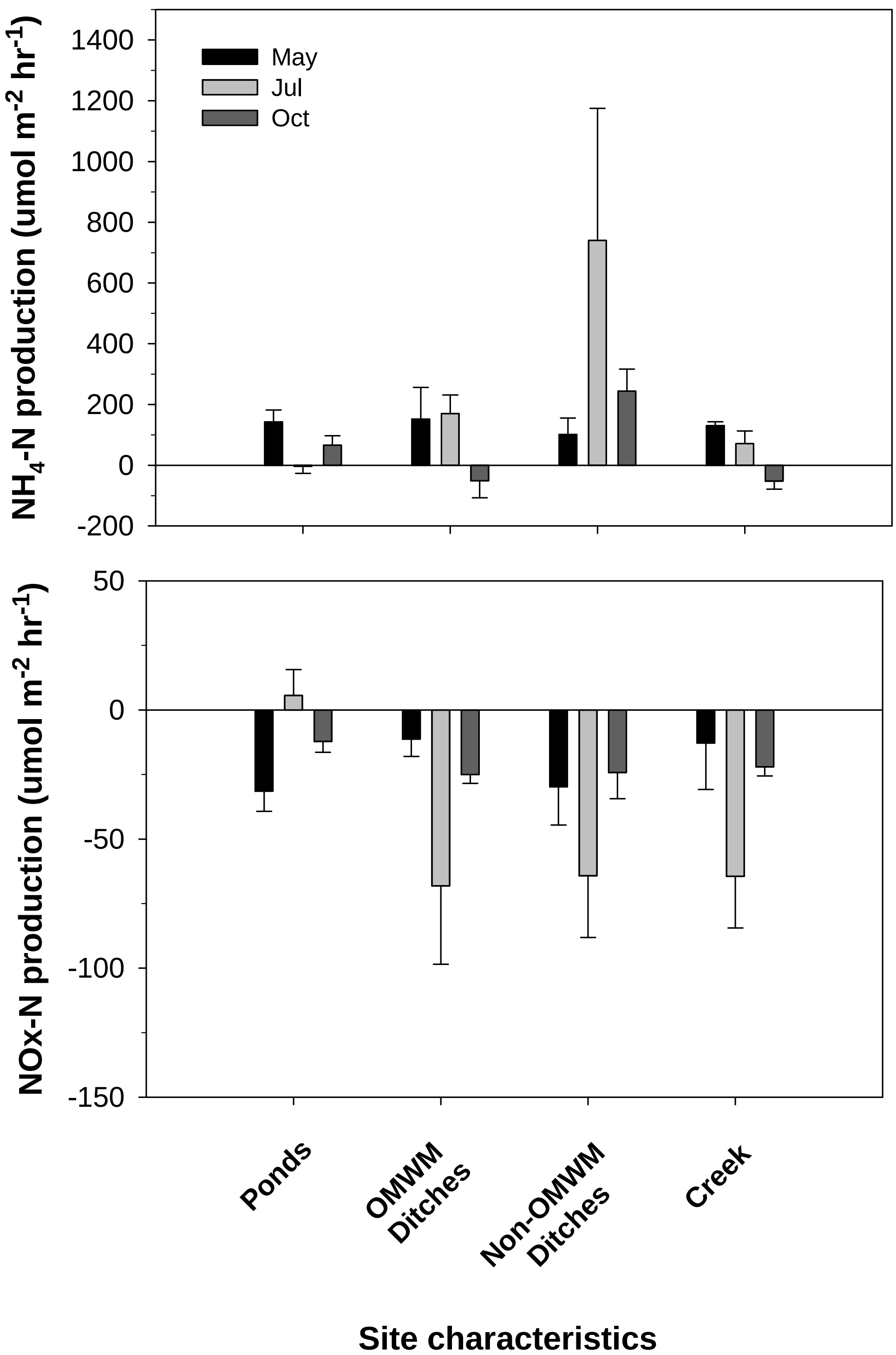


Figure 11. Dissolved ammonium and NOx fluxes for the study locations during the three periods. Rates are the average ±1SE (n = 6). Negative fluxes are into the marsh sediment.

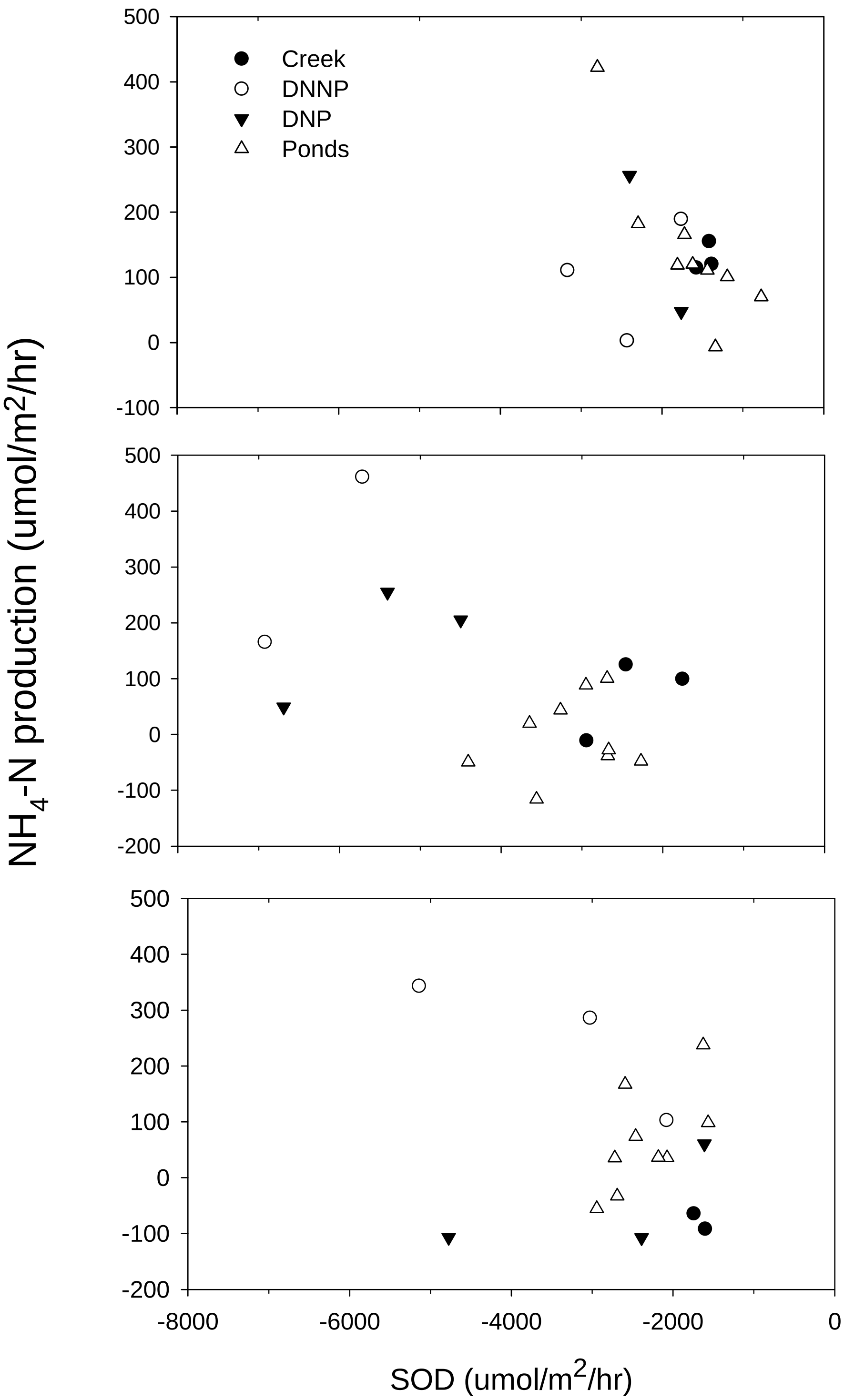


Figure 12. Ammonia-N production rates versus SOD for the three time periods across all sites and sampling locations for creeks, ditches not near ponds (DNNP), ditches near ponds (DNP) and ponds (top: May, middle: July and bottom October).

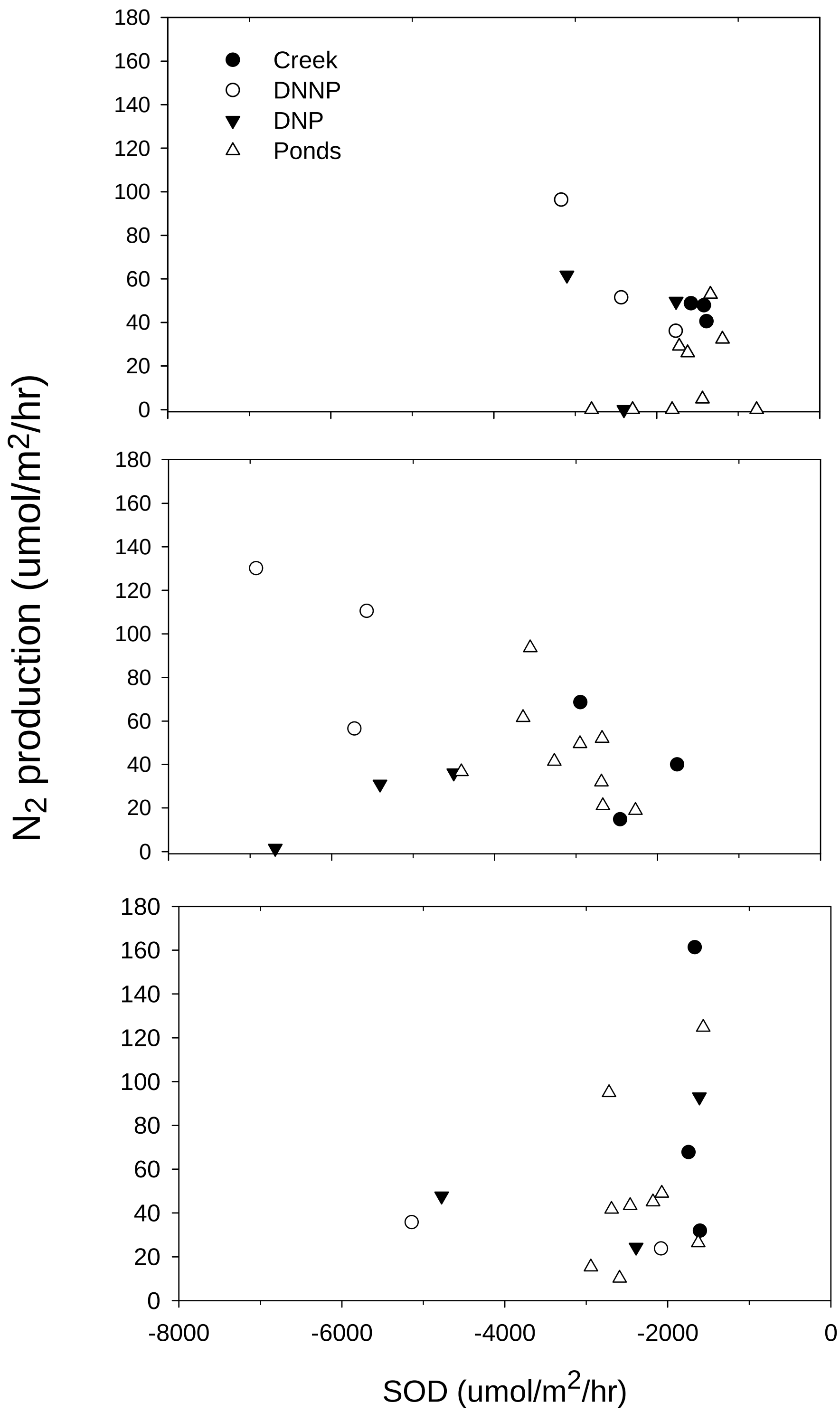


Figure 13. Nitrogen production rates versus SOD for the three time periods across all sites and sampling locations for creeks, ditches not near ponds (DNNP), ditches near ponds (DNP) and ponds (top: May, middle: July and bottom October).

Appendices

Excel File with Data and QA
(upon request)