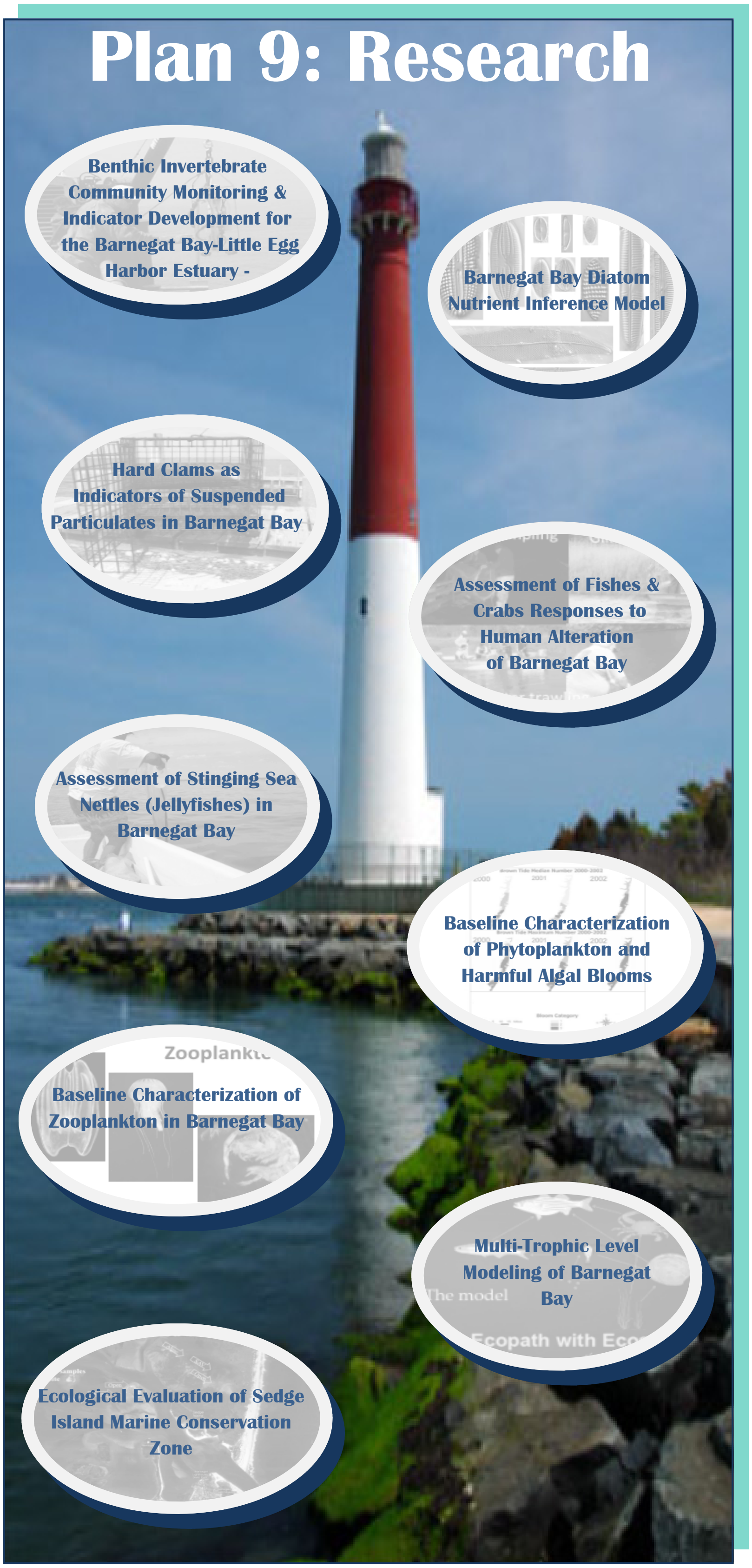
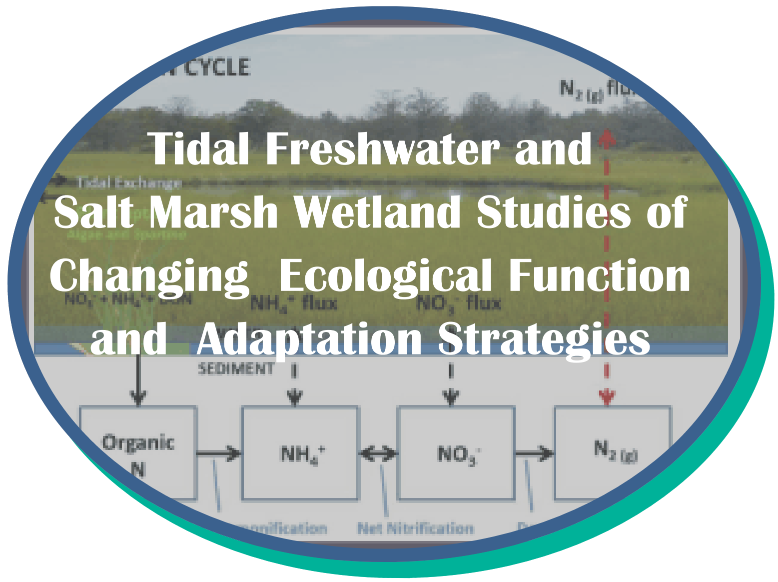


Barnegat Bay— Year 1



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

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FINAL REPORT

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Ecosystem Services of Tidal Wetlands in Barnegat Bay: Nitrogen removal

PCER Report No. 13-06

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Executive Summary

There is a debate going on about Barnegat Bay in New Jersey, namely whether nutrient-eutrophication, specifically nitrogen, is causing phytoplanktom blooms and increased macroalgae, and possibly secondary impacts (e.g., anoxia, loss of seagrass, increase in jelly fish, decreases in fish and crab populations, etc). The debate revolves around the fact that Barnegat Bay is poorly flushed, and that current eutrophication effects are only part of the natural conditions exacerbated by nitrogen loading along with some negative effects resulting from other stressors such as boat traffic, loss of connected wetlands, loss of freshwater flows and the withdrawal of cooling waters for the Oyster Creek Nuclear Generating Station.

Salt marshes play a large role in removing pollutants and nutrients from water. The main mechanisms of removal are 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 7.0 x 10^5 kg/yr N load. Our goal was to quantify N removal via denitrification in salt marshes of Barnegat Bay to contribute to our quanification of the Barnegat Bay N budget. A total of 18 soil cores were collected in 3 locations of the Bay in May, July and October 2012. While we know that the N load is relatively high, the concentration of dissolved N in the tidal creeks is relatively low. Nitrate + nitrite-N concentration ranged from not detectable to 2.0 µM and ammonium-N concentration ranged from 0.3 to 5.6 µM. Denitrification rates were similar among sites despite differences in creek water nutrients and salinity. Denitrification ranged from an average of 40 to 130 µmol/m^2/hr, depending on season. Denitrificaiton rates were generally highest in the mid-summer (July) with similar rates at each site during each season. During mid-summer and fall there was a significant relationship between ammonium and SOD fluxes in the marshes.

Salt marshes of Barnegat Bay are currently subject to a mosquito management technique, Open Marsh Water Management (OMWM), where interior vegetated marsh is converted to shallow ponds. Over 10,000 acres of salt marsh has been physically altered with OMWM in Barnegat Bay, thus making it important to measure the effect of OMWM ponds on denitrification. In July 2012, we collected five core in OMWM ponds, five cores in adjacent vegetated marsh, and five cores from vegetated marsh areas that have not been subject to

OMWM. OMWM pond sediments had lower and less variable denitrification rates than vegetated soils. Overall, salt marsh denitrification has the potential to remove approximately 13 to 33% of the incoming estimated N load entering the Bay (7.0 x 10^5 kg/yr). Both sediment burial and denitrification can sequester or remove between 91 and 111% 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.

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) 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. Studies using sediment deposition plates and ^{7}Be analysis would aid in this comparison.

Importantly, OMWM appears to lower the overall rate of denitrification during peak warm months. An areal 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.

A) Introduction

The environmental services provided by wetlands that fringe the coast are at risk from sea level rise. In this regard it is important to understand the current extent of services wetlands provide, such as nutrient cycling-retention, to better plan for the future and related environmental and land use changes. This was designed to enhance our understanding of the nitrogen uptake, burial and removal services provided by coastal wetlands in Barnegat Bay. By quantifying the proportion of the watershed's nitrogen load that is processed by vegetation, buried, and most importantly, removed by microbial denitrification per unit area, resource managers and policy makers will have needed information to evaluate, protect and enhance wetlands, while maintaining benefits for water quality, as well as for wildlife habitat, water flow and biodiversity. The objective of the study is to help inform resource managers of the value of wetland-watershed linkages, understand nutrient sinks, and how sea level rise may alter these critical environmental services.

A1: Background

In a study by Velinsky et al. (2010), four sediment cores from three marshes in Barnegat Bay were collected to assess long-term trends in nutrient levels and ecosystem change using diatom analysis. Although based on a small dataset, one outcome of that study was that a substantial portion of the nitrogen load into Barnegat Bay appeared to trapped and buried in the limited wetland area presently in the Bay. Another important N removal process is denitrification (**Figure 1**), a microbial process that transforms biologically available N (as oxidized NO_x) and releases it to the atmosphere as either N_2 or N_2O. Denitrification has been identified as an important removal mechanism for N being transported to coastal waters and estuaries, and has been shown to be an important aspect of N loadings to Barnegat Bay (Seitzinger 1987, 1988, 1992; Seitzinger and Pilling 1993, and Seitzinger et al. 2006). This ecosystem service can help remove nitrogen from the waters of the Bay. As nitrogen loading from the watershed has been shown to be a major source of the Bay's eutrophication problems (e.g., excessive algae growth, low dissolved oxygen, etc.), processes that can help remove nitrogen from various ecosystem compartments (e.g., wetlands, SAVs, and others) need to be better quantified and eventually enhanced and protected.

A2: *Objectives of Study*

The objective of this study was to estimate the removal of dissolved nitrogen via denitrification in the tidal wetlands of Barnegat Bay. In addition, we compared these removal rates to burial rates and inputs to the Bay from a previous study. To meet this objective we obtained sediment cores from three locations in Barnegat Bay during three seasons and determined the rate of denitrification and nutrient fluxes between tidal waters and the sediments. In addition, a preliminary study of Open Marsh Water Management (OMWM) ponds was undertaken to ascertain if these new features on the marsh are altering ecosystem services such as denitrification.

The proposed study 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).

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.

Changes in the Barnegat Bay-Little Egg Harbor Watershed and Nutrient Enrichment

The water quality of the Barnegat Bay is affected by persistent pollution impacts (i.e., high nutrient loads, algal blooms, eutrophication and low dissolved oxygen) from agricultural runoff and stormwater discharges, in addition to having somewhat restricted tidal flushing (BBNEP 2005, Kennish et al. 2007). Approximately 50-66% of the nutrient load is from surface waters with a substantial amount from atmospheric deposition (22-40%) and lesser amounts from groundwater inflow (~ 10%) (Hunchak-Kariouk and Nicholson 2001, Bowen et al. 2007, Wieben and Baker 2009). Wieben and Baker (2009) estimate that greater than 60% of the nitrogen load is from the Toms and Metedeconk rivers in the upper section of the Bay. Kennish et al. (2007), using the NLOAD model framework, estimated the land-derived nitrogen loading to the Bay of 7 x 10^5 kg N/yr and on an aerial basis of 3.9 kg N/ha-yr. Approximately, 15% of the nitrogen load to surface waters (and to groundwaters) is derived from the application of fertilizer in the watershed (Castro and Driscoll 2002, Ayars and Gao, 2007; Borgatti 2008). While development of the watershed has most likely increased loadings over time, a major change in the discharge of nutrients occurred in 1980. Prior to 1980, wastewater discharges and loadings of N and P were direct to the Bay, after which, between 1976 and 1979, the Ocean County Utilities effluent system was redirected to discharge wastewater approximately 2 km offshore of New Jersey. As such, there are few if any, point source discharges of nutrients to the Bay (not including stormwater runoff).

In general, there appears to be a lack of water quality (i.e., nutrient) monitoring data for the tidal Barnegat Bay prior to the early 1990s. Limited data from the 1970s by Durand (198) for the southern sections of Barnegat Bay and Great Bay show a range in nitrate concentrations between 6 to 70 µg N/L, while dissolved nitrate levels currently range from 5 up to 1000 µg N/L (NJ DEP, see www.state.nj.us/dep/bmw/ReportOcean.htm) throughout the entire Bay. There is a large seasonal change in nitrogen concentrations as well as a spatial gradient with higher concentrations in the northern sections of the Bay, i.e., from Barnegat Inlet to the Metedeconk River. The levels of nutrients result in elevated chlorophyll *a* concentrations in the Bay that can range up to 40 to 50 μg/L. Recurring phytoplankton blooms, including harmful algal blooms, have been shown over time, with brown tides (*Aureococcus anaphagefferans*) sporadically occurring since 1995 (Olsen and Mahoney 2001, Gastrich et al. 2004).

Overall, Barnegat Bay has shown increased development along with low freshwater inflow and flushing (i.e., high residence time of water) and high nutrient levels that result in eutrophic conditions. These conditions hinder the ecological and recreational benefits of Barnegat Bay and as such there are a number of management goals to restore the Bay. While restoration will most likely not bring the Bay back to "pristine" conditions (Duarte et al. 2009, Palmer and Filoso, 2009), information as to how ecosystems will respond, and respond over time, is needed to set reasonable restoration goals in the future.

B) Field and Laboratory Methods

B1: *Field Sampling*

Sediment cores (6.4 cm diameter x ~ 17 cm sediment depth) and overlying water were collected by hand from three salt marsh locations in May, July and October 2012.
Cores were collected from vegetated areas dominated by *Spartina alterniflora* and minor amounts of *Spartina patens* (**Figures 2-5; Table 1**). Cores from vegetated areas included sediment, roots and rhizomes. At each core collection site, elevation was determined using RTK GPS (Leica GX1230 GG) paired with a GNSS base station (AX1202 GG). For the OMWM study we collected five 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. Samples for this study were taken only in July 2012.

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.

B2: *Laboratory Methods*

Water and cores were transported, within 12 hr, to the Institute of Marine Science at University of North Carolina for incubation and nutrient fluxes. A portion of the water was retained at the Academy for filtration and nutrient analysis. Once incubations were completed, sediment cores were sectioned and placed into plastic baggies and water samples 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

Within 12 hr of sampling, water and sediment cores were transported on ice with site water overlying the headspace to the University of North Carolina Institute of Marine Sciences in Morehead City, NC (IMS). At IMS, cores were submerged in an aerated water bath in an environmental chamber (Bally Inc.) overnight at *in situ* temperatures in the dark (**Figure 6**). Continuous flow incubations of intact cores were used to determine the fluxes of nutrients and dissolved gases (Lavrentyev et al. 2000, McCarthy et al. 2007). The following morning, each core was capped with an air-tight Plexiglas top equipped with an inflow and outflow sampling port. Aerated and unfiltered water was passed over cores at a flow rate of 1 ml min^{-1}, which created a well-mixed water column within the chamber (Lavrentyev et al. 2000, Piehler and Smyth 2011).

Cores were acclimated in the continuous flow system for a period of no less than 18 hr prior to sampling to allow the system to reach equilibrium (Eyre et al. 2002a, 2002b). Water samples (5 ml) were collected from the outflow of each core at 18-, 24-, 36- and 48-hr increments, to ensure that steady-state conditions were present for analysis of dissolved gases. Inflow concentration was measured from a bypass line that flowed directly into the sample vials. Gas samples were analyzed for N_2, O_2 and Ar using membrane inlet mass spectrometry (MIMS; Kana et al. 1994, Kana et al. 1998). Once during the incubation (after 24 hr), 50-ml water samples were collected for nutrient analysis from the inflow line and outflow of each core.

Water was filtered through Whatman GF/F filters (25 mm diameter, 0.7 µm nominal pore size) and the filtrate was analyzed for NO_x^- and NH_4^+. Following each of the continuous flow experiments, three depth sections (approximately 0-4, 4-8 and 8-14 cm) of sediment cores were analyzed for percent organic matter determined by loss on ignition and organic carbon and total nitrogen concentrations.

Calculations

Flux calculations were based on the assumption of steady-state gradients that match *in situ* gradients and a homogenous water column. Benthic fluxes were calculated using the equation (C_{out} - C_{in}) x F/A, where C represents the concentration of analyte, C_{in} and C_{out} are the outflow and inflow concentration (μM), respectively, F is the peristaltic pump flow rate (l hr^{-1}), and A is the surface area of the core (m^2) (Miller-Way and Twilley 1996). N_2/Ar was used to calculate net N_2 fluxes, where the positive flux of N_2 out of the sediment was denitrification minus nitrogen fixation (Kana et al. 1994, An et al. 2001). O_2/Ar was used to calculate oxygen fluxes and sediment oxygen demand (SOD) was calculated as the flux of O_2 into the sediment (Kana et al. 1994, Smith et al. 2006). This method does not discern between the sources of $N_{2,}$ therefore denitrification refers to N_2 production from heterotrophic denitrification, anaerobic ammonium oxidation (anammoxa) and any other N_2-producing process. For NO_x^- and NH_4^+, a positive flux indicated production from the sediment to the water column and a negative flux indicated uptake from the overlying water. Individual measurements from each core over time were averaged to yield core-specific values. Denitrification data were extrapolated based on a 12-hr day to reflect our assumption of very low rates during the day due to both competition with benthic microalgae for N and increased oxygen concentrations (Tobias 2007, Hochard et al. 2010).

Direct denitrification was calculated by subtracting the denitrification rate from the absolute value of the nitrate flux. Coupled denitrification was calculated by the difference between measured denitrification and the estimated direct denitrification.

B2.2: Dissolved Nutrients

Water samples, both from the adjacent creek and core incubations, were collected for NO_x (NO_3^- + NO_2^-) and NH_4^+. 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.

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.

C) Results and Discussion

C1: *Creek Water and Sediment Properties*

C1.1: Water Chemistry

Water quality near the three sites showed some slight variations (**Table 2**). Temperature ranged from 18 to 21°C in May to approximately 26-27°C in July, decreasing substantially in October to 15-18°C. Salinity was lowest at Reedy Creek (~18-19 psu), highest at the mid-bay site (28 to 31 psu) and slightly lower downbay at Horse Point (26 to 28 psu). It is possible that the proximity and flow from Barnegat Inlet results in higher mid-bay salinities. Dissolved oxygen concentrations were lowest in the summer (2.4-6.7 mg O_2/L) and highest in the fall (8-12 mg O_2/L). Lower concentrations were measured at Reedy Creek in May and July with highest concentrations measured in the fall. pH ranged from 6.8 to 8.1 with slightly lower values at the upstream location.

Dissolved inorganic forms of N and P were measured in adjacent creek waters at all three sites during the study (**Table 3**). In each creek, five samples were collected to provide a snapshot of the water that would be flowing in and around the marsh where the cores were obtained for denitrification. Concentrations of dissolved ammonium+ammonia, ranged from 0.04 to 5.6 μM N with lowest concentrations at the IBSP site, while dissolved nitrate+nitrite concentrations ranged from 0.01 to 2.0 μM N with highest concentrations found at the Reedy Creek and Horse Creek locations; the most northern site. Soluble reactive phosphorus concentrations ranged from 0.01 to 1.2 μM P with highest concentrations at the downbay location. The dissolved inorganic N to P ratio (molar) ranged from 0.4 to 76 with an overall average of 12±24 (note: only 4 values out of 27 were above 10 and with those 4 removed, the average is 3.4±2.2). The highest values were observed at the Reedy Creek in July with values between 49 and 100. The average (and modified) ratio (which is < 16) suggests that the Bay is phosphorus limited. However, this data set is limited in time and space and only with a more expansive data set can this be fully determined.

C1.2: Sediment Organic Carbon and Total Sediment Nitrogen

After the water/gas exchange experiments the individual cores were sectioned into top-middle and bottom sections for organic carbon, total nitrogen and total phosphorus composition. These variables might influence microbial processing of water column nitrate and the production of pore water ammonium. **Table 4** presents the data for the May collection only for the three sites for organic carbon and total nitrogen.

Reedy Creek concentrations of sediment organic carbon and total nitrogen averaged 22.7% and 1.6% while IBSP average concentrations were 29.3% and 1.4%, respectively (**Table 4; Figure 7**). Horse Point, in the southern bay, exhibited significantly lower concentrations of 12.7% OC and 0.44% TN. The sediment concentrations at Horse Point were more variable overall with potentially two distinct groupings (**Table 4**) as noted by the OC. One grouping exhibited lower OC on average of 8% while another grouping had more than twice as much OC (19% on average). The sediment nitrogen concentrations were generally similar between groups and much lower than both Reedy Creek and IBSP.

C1.3: Seasonal Denitrification and Oxygen Fluxes

Denitrification rates at the three sites exhibited some variation with season (**Table 5, Figures 8 and 9**). Denitrification was similar among sites but varied seasonally with significantly higher rates in July than October across sites ($F_{2, 46} = 5.53$, $p = 0.0070$). Rates in 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 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).

C1.4: Nitrate and Ammonium Fluxes

Nitrate (+nitrite) fluxes ranged from -34 to +28 μmol N m^{-2} hr^{-1} across all sites and seasons. Fluxes were low and positive on average in May and July (i.e., movement of nitrate out of marsh sediments into overlying water) and slightly negative in October (**Figure 10**). Regardless of the direction of the nitrate flux it was generally 20 to 30 times lower than the N_2 production rate. The positive fluxes in May and July indicate that production (coupled ammonification-nitrification) of nitrate exceeded that which can be consumed during denitrification or autotrophic uptake. There were no relationships between nitrate fluxes and either N_2 production or sediment oxygen demand.

Ammonium-ammonia (i.e., ammonium) fluxes ranged from -39 to +500 μmol N m^{-2} hr^{-1} across all sites and seasons. Seasonally, fluxes were greatest, on average, in July and similar in May and October (**Figure 10**). Average fluxes (n=6) for each site (i.e, Reedy Creek, IBSP, and Horse Point) were generally positive (movement out of marsh) except for in October at the Island Beach State Park site in which the average flux was negative but small (-11.8±9.7 μmol N m^{-2} hr^{-1}). While there was no relationship between ammonium fluxes and N_2 production or nitrate flux, there was a significant relationship with sediment oxygen demand in the May and July time periods (**Figure 11**).

Both the nitrate and ammonium fluxes are similar to other studes in tidal salt marshes. 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). These data 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 ecasping 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). Lower oxygen availability would limit nitrification and the coupling to denitrification. Sediment oxygen demand ranged from -3400 to -240 µmol O_2 m^{-2} hr^{-1} from all sites, with highest rates measured in the July time period when temperatures were highest (note: negative rates indicate oxygen moving into the sediments from the overlying water). The relationship between sediment oxygen demand and N dynamics varied seasonally (**Figures 8 and 9**). In general, higher denitrification rates are associated with higher oxygen demand, allowing for greater nitrate production and indicating the importance of coupled nitrification – denitrification. Rates of SOD were significantly related to N_2 production in May and July and less so in October (**Figure 9**); with the strongest relationship observed in July (r^2 = 0.90, $p < 0.0001$) when temperatures were highest. Given the low water column concentrations of dissolved nitrate (**Table 3**), this suggests that external sources of nitrate would limit denitrification and that the coupled reaction is very important.

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, 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. In addition, ponds may be established in areas that have been previously grid ditched (**Figure 12**). 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 9000 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 (NO_x) production and denitrifying bacteria, yet the reduction of NO_x 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.

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} ,**Table 6** and **Figure 13**) which were similar to control locations (102±19 µmol N m^{-2} hr^{-1}). In addition, the average nitrate flux in the pond was low and directed into the sediment (-18±10 µmol N m^{-2} hr^{-1}) while the average ammonium flux was large (643±53 µmol N m^{-2} hr^{-1}) and directed out of the pond sediments into the overlying pond water. In all three locations, SOD was similar and averaged (-690±80 µmol O_2 m^{-2} hr^{-1}). A question for further study is why the pond sediments had substantially lower denitrification rates compared to the other adjacent areas. The hypothesis is 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. Further research is required to determine the magnitude and cause of this difference (see Year 2 program).

C3: Marsh Processing and Burial in Barnegat Bay

The rate of denitrification and nitrogen accumulation/burial (Velinsky et al. 2010) can be viewed in the context of inputs into Barnegat Bay. Nutrients can enter the Bay from river runoff, direct discharge, atmospheric deposition, and ocean exchange and can be exported or removed through burial (i.e., both in marshes and subtidal), ocean exchange, and importantly for nitrogen: denitrification (i.e., $NO_x \dashrightarrow N_2$). 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.

The current study expands on the study by Velinsky et al. (2010) 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. Removal or sequestration of nutrients by coastal marshes can be 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).

Denitrication rates, averaged across 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 nitirate 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 approxately 3 hr for their site in Chesapeake Bay. A water level data logger at IBSP and Horse Point was used to estimate the average amount of time the marsh was covered with water over the tidal cycle (**Figure 14**), as denitrification and its impact to the waters of the Bay will only take place during inundation periods. 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 7 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 rateas of N_2 production of between 12 and 30 μmol N m^{-2} hr^{-1} covering the time in the dark and inundated. The range in denitrification rate is most likely biased high 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.

C3.2 Conceptual Model: Ecosystem Services

An estimation of the area of tidal coastal wetlands fringing Barnegat Bay is 26,000 acres (1.1 x 10^8 m^2; Lathrop and Haag 2007; www.crssa.rutgers.edu/projects/lc/). Using this area and the depth-integrated and core-top rates for N accumulation yields current burial rates (gross rates; Velinsky et al. 2010) of 5.5 – 6.5 x 10^5 kg N/yr (**Table 7**). Similarly, using the Bay wetland area and the rates above yield a removal of N (assumed as nitrate) of between 0.9 – 2.3 x 10^5 kg N/yr (2.8±0.9 x 10^5 kg N/yr). This yields a total removal of N (as either particulate N, for burial, or nitrate in denitrification) of 6.4 – 8.7 x 10^5 kg N/yr (average of ~ 7.1 x 10^5 kg N/yr) within Barnegat Bay by the wetlands.

Using recent N load estimates for Barnegat Bay (Wieben and Baker 2009), coastal salt marshes can sequester 91 to 110% (average of 100%) of the nitrogen introduced into the bay (**Table 7**). The calculated removal percentages suggest that ALL 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. The load estimates from Wieben and Baker (2009) are currently being updated (Baker et al. [in review]). 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.

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 three marshes of Barnegat Bay over three seasons. In addition, inorganic nitrogen fluxes between sediment and overlying water were assessed as was sediment oxygen demand. Sites included Reedy Creek in the north, Island Beach State Park in the mid Bay and Horse Point in the southern Bay sampled in May, July and October 2012. Lastly, a preliminary study of the impact of Open Marsh Water Management (OMWM) on nitrogen processes in the marshes of Barnegat was undertaken. These data, along with sediment burial rate from tidal marshes obtained in an earlier study, help to provide a picture of the potential ecosystem services tidal wetlands in the Bay can provide.

Major findings of this study include:

- Denitrificaiton rates were generally highest in the mid-summer (July) with similar rates at each site during each season.
- During mid-summer and fall there was a significant relationship between ammonium and SOD fluxes in the marshes.
- OMWM pond sediments had lower and less variable denitrification rates compared to the other adjacent areas.
- Potentially, denitrification in the marshes can remove approximately 13 to 33% of the incoming estimated N load entering the Bay (7.0×10^5 kg/yr).
- Combined with sediment burial, denitrification can sequester/remov between 91 and 110% 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.

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) 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).

- OMWM appears to lower the overall rate of denitrification during peak warm months. An areal 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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Tables

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

Site	Sample	Latitude	Longitude
Reedy Creek	Soil core	40° 1'52.03"N	74° 5'6.01"W
Reedy Creek	Soil core	40° 1'47.52"N	74° 5'0.13"W
Reedy Creek	Soil core	40° 1'47.37"N	74° 5'4.51"W
Reedy Creek	Soil core	40° 1'44.62"N	74° 5'0.46"W
Reedy Creek	Soil core	40° 1'47.42"N	74° 4'48.54"W
Reedy Creek	Soil core	40° 1'44.67"N	74° 4'44.24"W
Reedy Creek	Water	40° 1'50.10"N	74° 5'4.53"W
Reedy Creek	Water	40° 1'48.60"N	74° 5'3.69"W
Reedy Creek	Water	40° 1'47.23"N	74° 5'0.88"W
Reedy Creek	Water	40° 1'45.73"N	74° 5'0.16"W
Reedy Creek	Water	40° 1'46.08"N	74° 4'45.23"W
IBSP	Soil core	39°47'58.21"N	74° 6'9.00"W
IBSP	Soil core	39°47'57.97"N	74° 6'9.36"W
IBSP	Soil core	39°47'56.82"N	74° 6'8.23"W
IBSP	Soil core	39°47'56.26"N	74° 6'8.15"W
IBSP	Soil core	39°47'56.20"N	74° 6'6.57"W
IBSP	Soil core	39°47'55.59"N	74° 6'6.38"W
IBSP	Water	39°48'3.17"N	74° 6'12.61"W
IBSP	Water	39°47'59.73"N	74° 6'15.51"W
IBSP	Water	39°47'54.84"N	74° 6'14.43"W
IBSP	Water	39°47'51.04"N	74° 6'12.69"W
IBSP	Water	39°47'46.60"N	74° 6'9.76"W
Horse Point	Soil core	39°37'51.11"N	74°15'26.45"W
Horse Point	Soil core	39°37'48.32"N	74°15'24.99"W
Horse Point	Soil core	39°37'47.86"N	74°15'27.18"W
Horse Point	Soil core	39°37'47.29"N	74°15'28.87"W
Horse Point	Soil core	39°37'47.34"N	74°15'30.77"W
Horse Point	Soil core	39°37'48.19"N	74°15'31.12"W
Horse Point	Water	39°37'49.88"N	74°15'25.99"W
Horse Point	Water	39°37'50.29"N	74°15'24.65"W
Horse Point	Water	39°37'50.53"N	74°15'22.82"W
Horse Point	Water	39°37'49.95"N	74°15'20.99"W
Horse Point	Water	39°37'50.13"N	74°15'18.87"W

Note: Sites are for May collection but are similar for other time periods.

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

	Temperature (°C)		
Site	**14-May**	**10-Jul**	**17-Oct**
Reedy Creek	20.1 ± 0.1	25.5 ± 0.1	14.8 ± 0.1
IBSP	18.4 ± 0.1	25.7 ± 0.1	17.1 ± 0.2
Horse Point	21.3 ± 0.3	27.2 ± 0.1	14.7 ± 0.1
	Salinity (psu)		
Site	**14-May**	**10-Jul**	**17-Oct**
Reedy Creek	17.6 ± 0.8[a]	17.8 ± 0.5[a]	19.2 ± 0.1[a]
IBSP	29.6 ± 0.1	28.4 ± 0.1	31.4 ± 0.2
Horse Point	27.5 ± 0.1	26.2 ± 0.1	27.1 ± 0.1
	Dissolved oxygen (mg/L)		
Site	**14-May**	**10-Jul**	**17-Oct**
Reedy Creek	4.6 ± 0.4	2.4 ± 0.2	11.7 ± 0.2
IBSP	7.4 ± 0.1[a]	6.7 ± 0.1[a]	10.2 ± 0.5
Horse Point	8.8 ± 0.2	6.4 ± 0.2	8.0 ± 0.1
	pH		
Site	**14-May**	**10-Jul**	**17-Oct**
Reedy Creek	6.9 ± 0.1[a]	6.8 ± 0.1[a]	7.8 ± 0.1
IBSP	7.9 ± 0.1[a]	8.0 ± 0.1[ab]	8.1 ± 0.1[b]
Horse Point	7.9 ± 0.1	7.5 ± 0.1	7.8 ± 0.1

Table 3. Nutrient concentrations in adjacent creek waters near each of the site (average±SE).

Site	May	July	October
Dissolved Ammonium+Ammonia (μM N)			
RC	3.6±0.4	0.4±0.04	1.5±0.25
IBSP	0.53±0.16	0.35±0.03	0.70±0.26
HC	1.1±0.35	0.40±0.03	5.6±0.12
Dissolved Nitrate+Nitrite (μM N)			
RC	1.5±0.33	ND	0.23±0.09
IBSP	0.09±0.02	ND	1.1±0.35
HC	0.08±0.03	0.09±0.01	2.0±0.04
Soluble Reactive Phosphorus (μM P)			
RC	0.18±0.02	0.15±0.01	0.09±0.01
IBSP	0.43±0.03	0.22±0.05	0.67±0.15
HC	0.5±0.13	0.34±0.04	1.2±0.01

Table 4. Concentrations of soil organic carbon and total nitrogen along with the C to N ratio for the three study sites in Barnegat Bay. Samples are from the May period only and are the average (±SE) for the six cores at three depths. Note: Horse Point cores were broken into two groups.

	Carbon		Nitrogen		C to N	
Site	**%**	**SE**	**%**	**SE**	**molar**	**SE**
Reedy Creek	22.7	0.7	1.60	0.07	17.0	0.9
Island Beach State Park	29.3	0.9	1.36	0.11	27.9	2.3
Horse Point	12.7	1.9	0.44	0.02	33.9	5.5
HP Cores 1-3	7.7	0.9	0.41	0.02	21.7	2.3
HP Cores 4-6	18.5	2.7	0.48	0.03	47.6	8.7

Table 5. Denitrification and sediment oxygen demand rates for each season at each location in Barnegat Bay.

		May		July		October	
Site	Core Number	N2 Flux ($\mu mol\ N\text{-}N_2\ m^{-2}\ hr^{-1}$)	Sediment Oxygen Demand ($\mu mol\ O_2\ m^{-2}\ hr^{-1}$)	N2 Flux ($\mu mol\ N\text{-}N_2\ m^{-2}\ hr^{-1}$)	Sediment Oxygen Demand ($\mu mol\ O_2\ m^{-2}\ hr^{-1}$)	N2 Flux ($\mu mol\ N\text{-}N_2\ m^{-2}\ hr^{-1}$)	Sediment Oxygen Demand ($\mu mol\ O_2\ m^{-2}\ hr^{-1}$)
		Dark		Dark		Dark	
Reedy Creek	1	12.4	-264.4	193.7	-2049.3	265.8	-2703.7
Reedy Creek	2	38.7	-583.6	62.5	-1131.1	-2.0	-293.6
Reedy Creek	3	16.0	-448.3	43.4	-688.6	0.0	-431.1
Reedy Creek	4	166.6	-1829.2	80.4	-1350.2	0.6	-244.4
Reedy Creek	5	-0.2	-295.4	215.2	-2777.6	22.1	-311.8
Reedy Creek	6	106.1	-1361.8	173.2	-2264.8	-4.5	-237.2
IBSP	1	72.9	-923.7	271.8	-3350.5	27.0	-326.1
IBSP	2	49.8	-637.5	72.5	-1412.7	16.7	-439.0
IBSP	3	211.9	-2254.9	41.0	-1105.9	13.1	-425.5
IBSP	4	121.1	-1241.5	104.6	-1252.0	251.6	-2612.6
IBSP	5	60.1	-916.8	53.5	-857.5	-18.4	-404.8
IBSP	6	58.2	-714.6	100.9	-1299.1	17.2	-717.9
Horse Point	1	133.4	-653.4	39.8	-384.2	64.9	-406.5
Horse Point	2	33.1	-358.7	51.9	-769.5	64.8	-625.2
Horse Point	3	49.6	-701.0	241.0	-3272.5	54.3	-610.4
Horse Point	4	111.7	-1182.6	65.4	-1180.4	72.4	-825.9
Horse Point	5	126.1	-1749.2	290.5	-3329.6	20.0	-484.1
Horse Point	6	119.9	-1506.8	83.3	-1291.3	13.5	-367.8

Note: Negative values indicates the flux is into the marsh from the overlying water

Table 6. Denitrification, sediment oxygen demand and inorganic N flux rates at the OMWM site in July 2012.

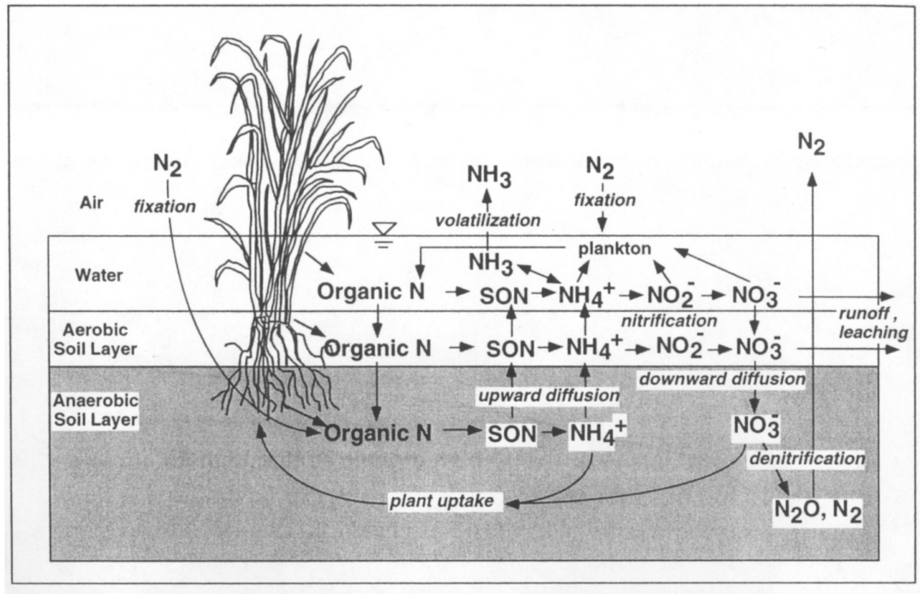
Site	Core Number	N2 Flux (μmol N-N_2/m^2- hr)	Sediment Oxygen Demand (μmol O_2 /m^2-hr)	NOx-N Flux (μmol N-N_2/m^2- hr)	NH4-N Flux (μmol N-N_2/m^2- hr)
		Dark			
Pond	1.0	60.2	-545.0	-38.5	699.8
Pond	2.0	64.8	-564.6	3.6	630.9
Pond	3.0	68.1	-578.4	-42.2	404.3
Pond	4.0	76.7	-588.6	18.4	654.1
Pond	5.0	78.0	-600.8	-11.9	786.3
Pond	6.0	88.2	-607.3	-35.6	685.2
Veg/OMWM	1.0	27.0	-230.7	0.00	-9.19
Veg/OMWM	2.0	164.2	-460.4	0.00	39.4
Veg/OMWM	3.0	76.9	-575.1	1.47	-43.0
Veg/OMWM	4.0	159.9	-1389.1	2.93	253.7
Veg/OMWM	5.0	100.3	-905.1	0.00	117.1
Veg/OMWM	6.0	152.2	-1157.7	0.00	18.52
Control	1.0	54.1	-274.3	-69.5	-451.4
Control	2.0	165.9	-336.8	-71.0	-484.7
Control	3.0	54.6	-585.9	-72.5	-357.2
Control	4.0	142.0	-1372.1	-66.6	-134.4
Control	5.0	85.5	-687.5	-64.3	-691.3
Control	6.0	110.8	-942.8	-56.2	-624.1

Table 7. Comparison of Barnegat Bay marsh nitrogen burial rates measured in this study to rates of nitrogen inputs to the Barnegat Bay.

	Nitrogen kg/yr X10^5
Nitrogen Inputs:	6.5±0.6
Marsh Burial :	
Core Top	6.5
Average over Core	5.5±0.6
Denitrification	
Low	0.9
High	2.3
Average	1.6±0.9
Total Removal as % of Inputs	
Low	91
High	110
Average	100

Nitrogen inputs ranged from 6.5 to 7.7 X10^5 kg/yr (Hunchak, 2001; Wieben and Baker, 2009; Kennish et al., 2007). Wetland area (26,000 acres, 1.1 X10^8 m^2) are obtained from www.crssa.rutgers.edu/ projects/lc/.

Figures



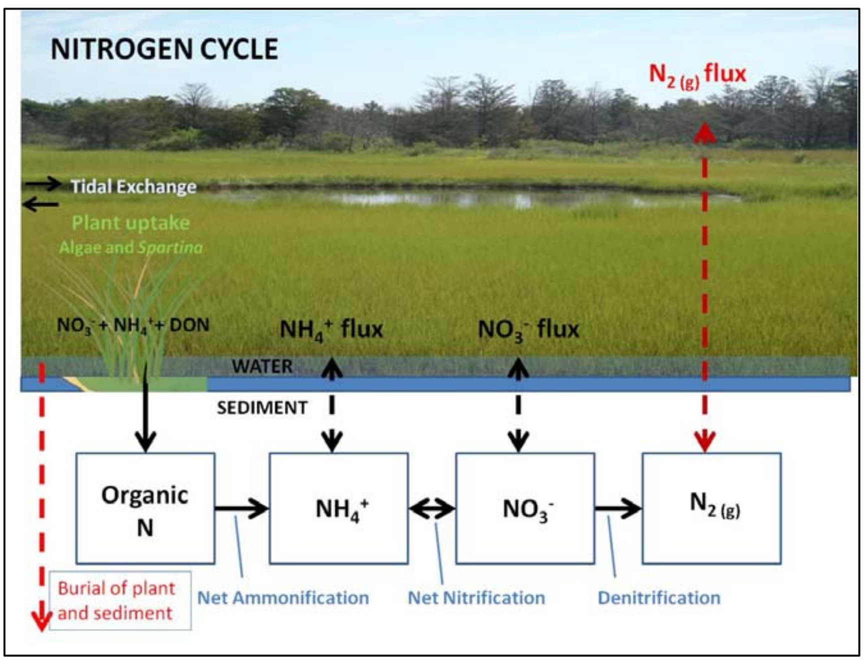


Figure 1. Generalized schematic of nitrogen cycling in wetlands (Mitch and Gosselink, 1993) along with conceptual model of cycling in Barnegat Bay.

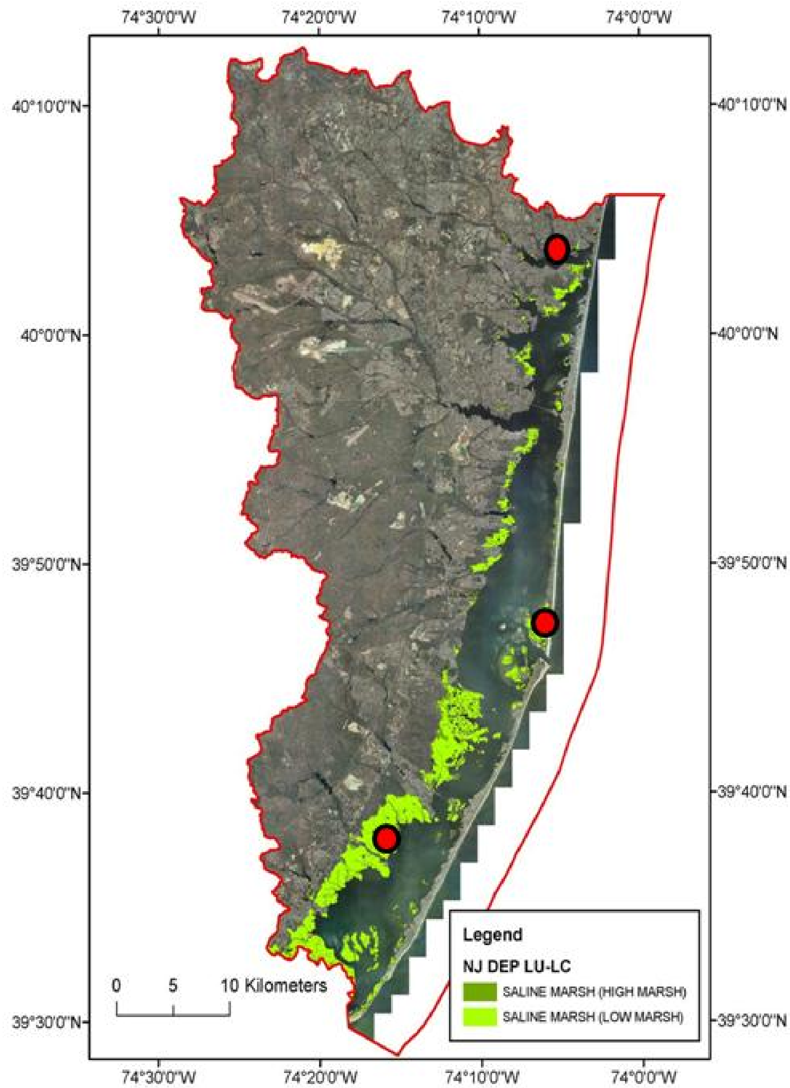


Figure 2. Barnegat Bay study area: Northern site (Reedy Creek) that has higher nutrient input and lower salinity (18±0.3 psu); Sedge Island in the mid-bay on barrier island and a gradient of salinity (30±0.3 psu) and nutrients and the southern site (Horse Creek) with lower nutrient inputs and higher salinity (27±0.3 psu).

Figure 3. Map of northern site at Reedy Creek.

Figure 4. Map of mid-bay site at Island Beach State Park (Sedge Island) .

Figure 5. Map of southern site at Horse Point.

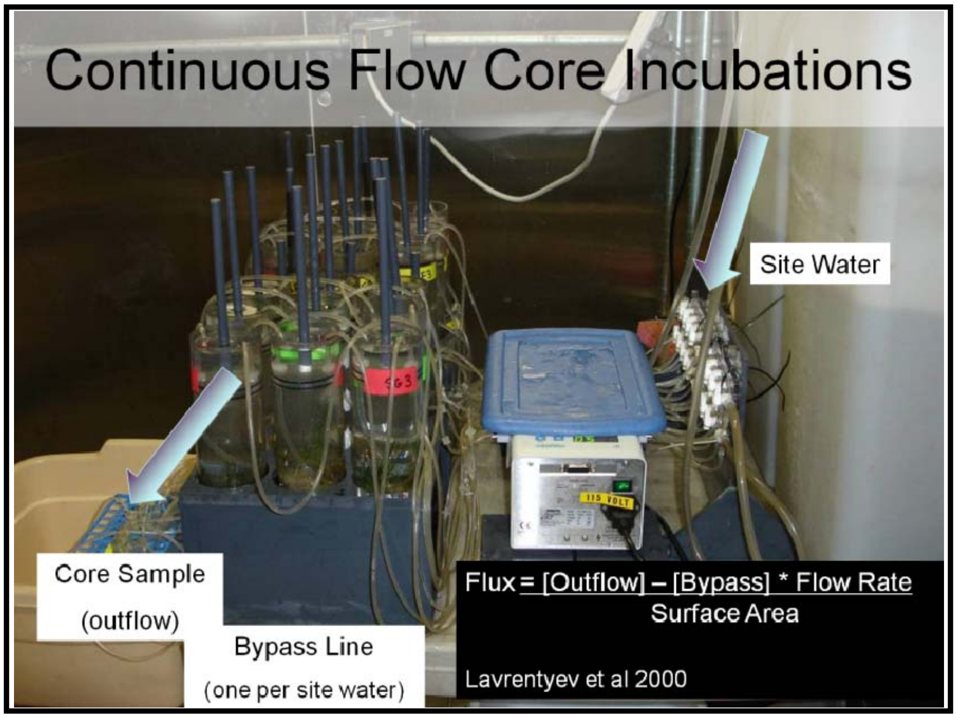


Figure 6. Experimental set up for denitrification and nutrient flux sampling. This is a continuous flow incubation method for water and gas sampling developed by Lavrentyev et al. 2000 and McCarthy et al. 2007.

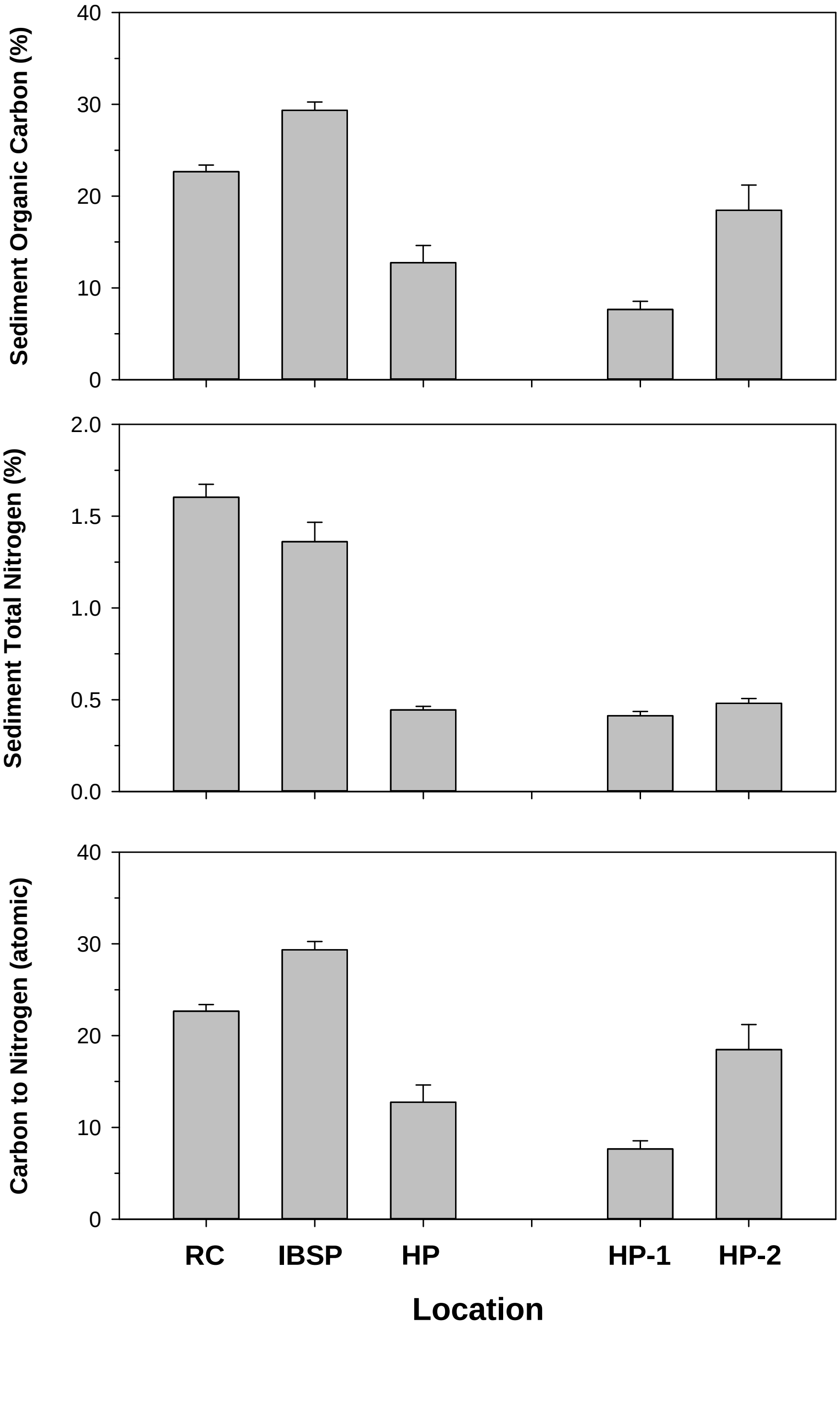


Figure 7. Sediment concentrations of organic carbon and total nitrogen along with the C to N ratio (atomic) for the May collection at the three study locations (RC – Reedy Creek, IBSP – Island Beach State Park and HP – Horse Point). HP was broken into two groupings based on organic carbon concentrations.

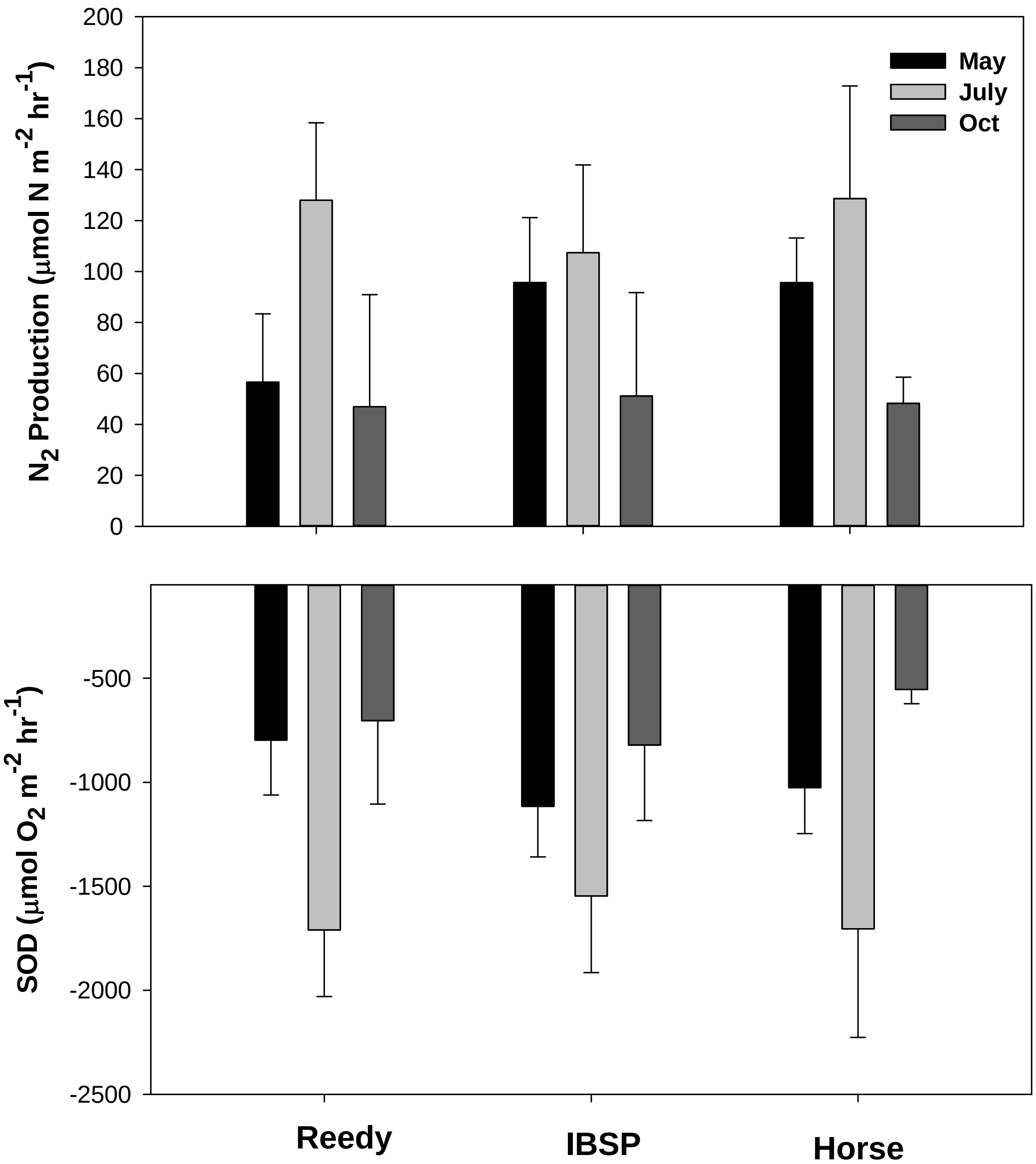


Figure 8. Nitrogen production rates and sediment oxygen demand for the three locations during the three periods. Rates are the average ±1SE (n = 6).

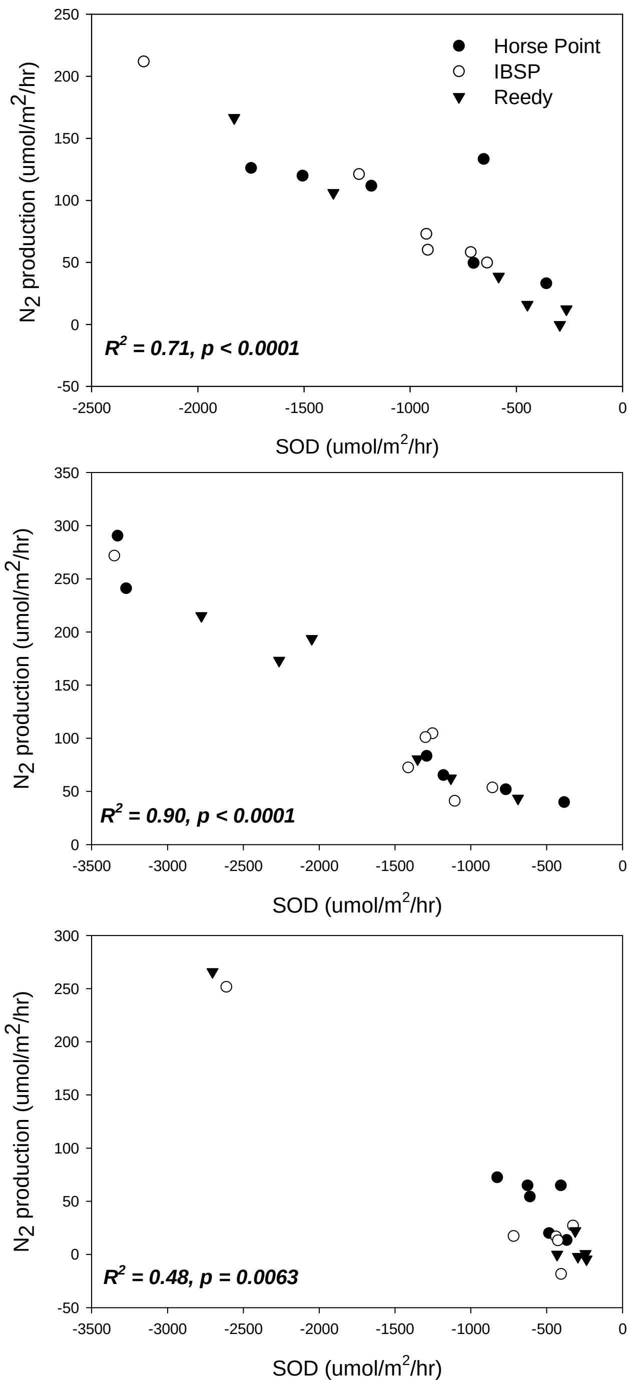


Figure 9. Nitrogen production rates versus SOD for the three time periods across all sites (top: May, Middle: July and Bottom October).

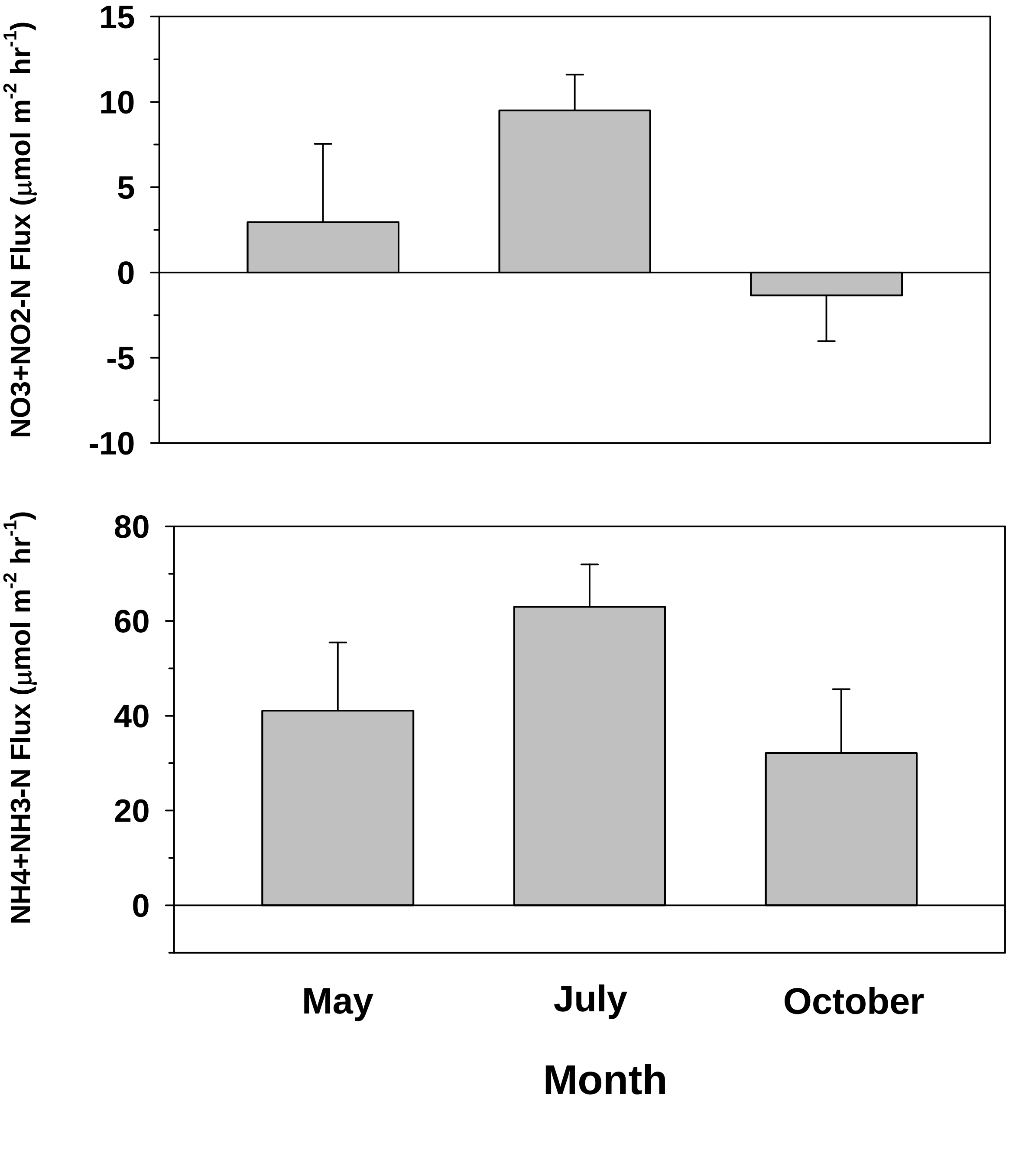


Figure 10. Average (±SE) NO_3+NO_2-N (top) and NH_4+NH_3-N (bottom) fluxes for the three sampling periods in Barnegat Bay.

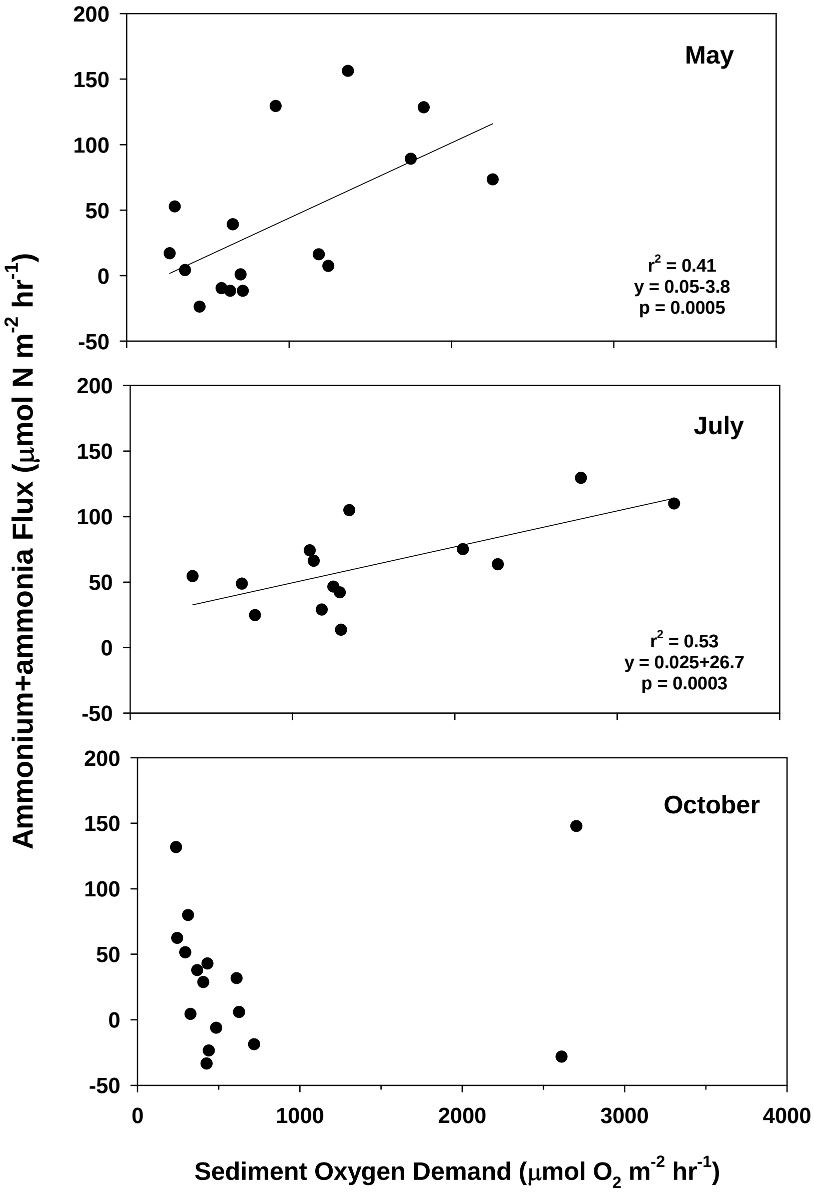


Figure 11. Sediment oxygen demand versus ammonium fluxes for the three time periods across all sites.

Figure 12. Open Marsh Water Management (OMWM) in Barnegat Bay, NJ (39°42'5 N, 74°11'5 W).

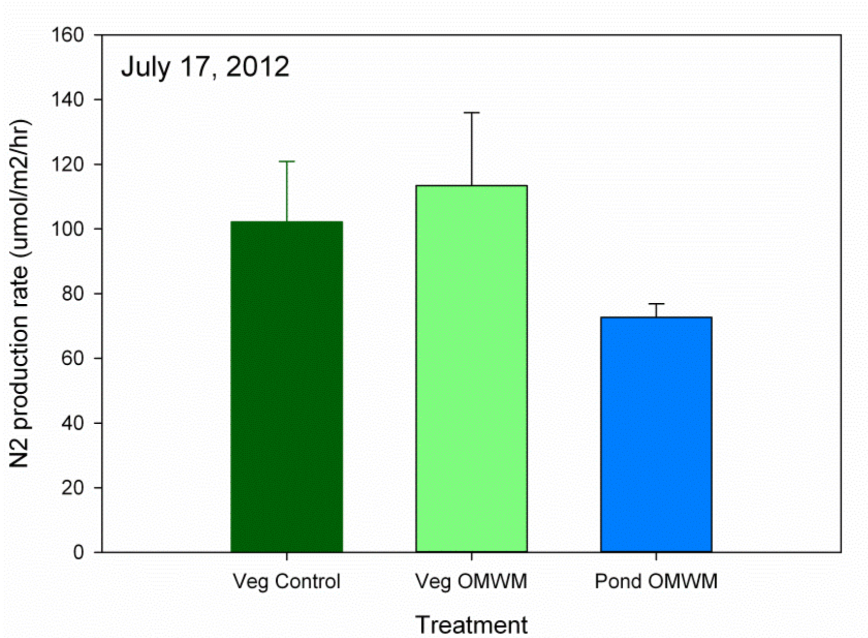


Figure 13. Denitrification rate in different treatment and control salt marsh areas in Barnegat Bay.

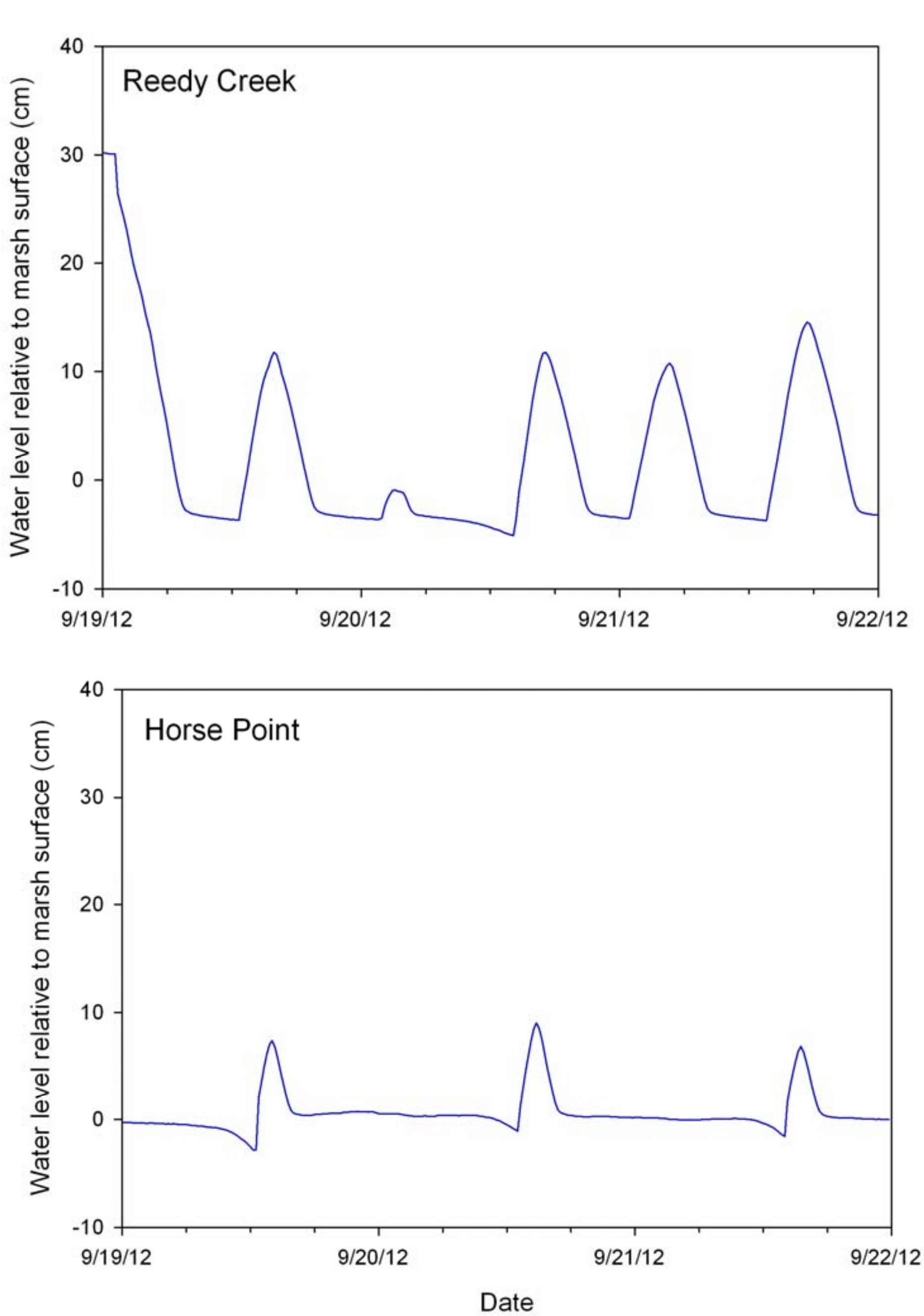


Figure 14. Water level data for Reedy Creek and Horse Point. The zero is the marsh surface.

Appendices

Excel File with Data and QA
(upon request)