ECOSYSTEM ASSESSMENT AND NITROGEN MANAGEMENT IN WESTERN BAYS, NEW YORK

FINAL REPORT
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Prepared for
New England Interstate Water Pollution Control Commission

Prepared by

Thomas Gulbransen[1]
Scott Libby[1]
Roy Kropp[1]
Rosanna Buhl[1]

Larry Swanson[2], Katelin Willig[2]
Chris Gobler[2], Ryan Wallace[2]
Robert Wilson[2],
Dongming Yang,[2] Cassia Bauer[2]

Mike Owens[3],
Jeff Cornwell[3]
Robert J.Diaz[4]
Fred Mushacke[5]

1= Battelle Memorial Institute
2=Stony Brook University, School Of Marine & Atmospheric Science
3=Chesapeake Biogeochemical Associates
4=Diaz & Daughters
5=F.R.E.D. Environmental

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1.0 BACKGROUND OF THIS STUDY

(From New England Interstate Water Pollution Control Commission RFP)

Implementation of the Clean Water Act has allowed for significant improvement in the coastal water quality of the New York-New Jersey region, resulting in the restoration of many historical uses and functions of these waters. Despite this success, coastal water quality problems remain and additional actions are needed to fully restore coastal waters. Many waters experience periodic harmful algal blooms and declining trends in submerged aquatic vegetation and desirable fish and shellfish species.

Nutrient over-enrichment of coastal, shallow water embayments in New York and New Jersey leads to eutrophic conditions in those waters. Enriched nutrients in bay waters promote the growth of opportunistic organisms, such as certain species of phytoplankton and macroalgae. Due to the complex relationships between nutrient loads, ambient water column concentrations, and environmental fate and effects, the development of additional or alternative nutrient criteria and management scenarios is needed for these systems.

2.0 INTRODUCTION TO ISSUES ADDRESSED

The Battelle team addressed NEIWPCC's request through a four-part strategy focused on the Western Bays ecosystem on the south shore of Long Island, NY (Figure 1). Parts 1&2 of the strategy are a succession of studies to characterize the bays physically and biologically. Quantifying the ecosystem condition provides for both a one-time assessment of its state relative to designated uses, as well as a short-term evaluation of how well numerous environmental metrics describe the bays conditions from a regulatory perspective. Parts 3&4 of the strategy couple ecosystem condition with existing information about nitrogen loading. These investigations examine whether causal linkages can be found between nitrogen loads and biological responses which can manifest in impairments to designated uses of the bays. Integration of nitrogen load information and ecosystem characteristics enables the ultimate goal of this strategy which is to identify the potential effectiveness of alternative nutrient management actions.

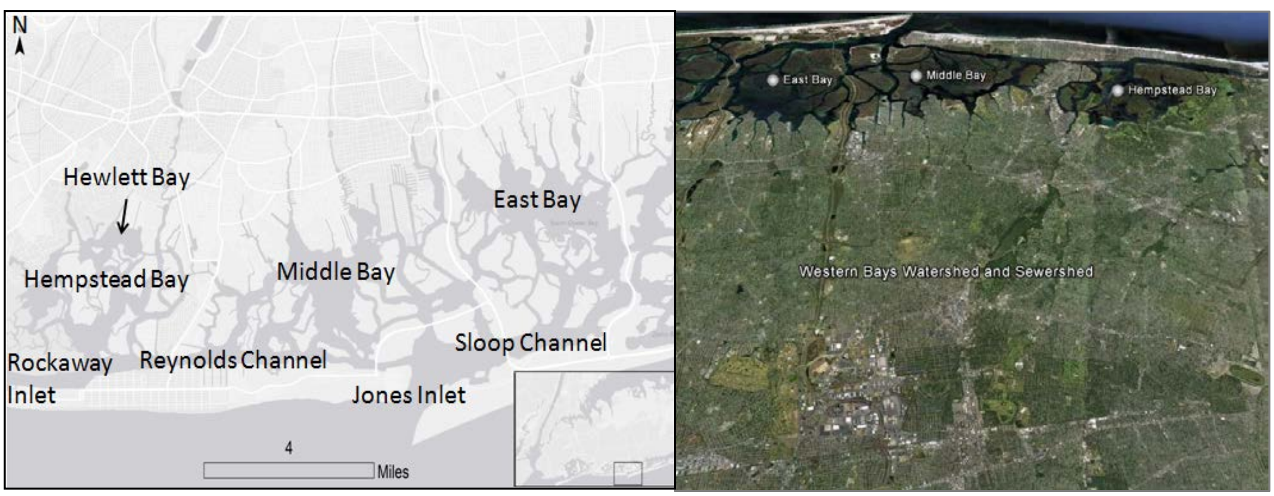

Figure 1. General extent of the Western Bays shallow embayments study area along with associated watershed and sewershed of ~500,000 people (2010 Census).

A conceptual model of the bays ecosystem unified the purposes and fit of the various investigations. These studies were initiated with a basic model which suggested distinct geomorphological regions of the bays, as well as major biogeophysical pathways for nutrient cycling (Figure 2). The conceptual model was not necessarily immutable or comprehensive, however it provided an initial framework from which to organize study topics and geography. Notionally, the conceptual model anticipated a gradient in water column characteristics from one end of the spectrum of poorly mixed headwaters dominated by freshwater inflows versus the other end of the spectrum of newly imported ocean waters. The conceptual model anticipated that benthic habitats would also differ substantially depending on the low energy depositional regions and the high energy, advective locations in the channels. Sampling locations were selected to ensure coverage of these varied conditions. The conceptual model did not anticipate the pronounced differences seen in segments of the Western Bays which were dominated by tidal mixing residence times and proximity to the Bay Park WWTP effluent discharge location. The conceptual model served well as the basis to compare and contrast the relative levels of ecosystem stressors and services.

Field studies demonstrated how varied the conditions are within a scale of hundreds of meters, thus the concept that channels differ from depositional bays remains true, however, the benthic habitats within each of these waterbody types also varied substantially. Field observations also updated this model based on the predominant influence of the Bay Park WWTP effluent. The initial conceptual model depicted nitrogen load generally entering the system from landward canals and submarine discharge (left side of diagram) with eventual mixing with oceanic water masses (right side of diagram). In fact, multiple lines of evidence demonstrated that ~79% of the nutrient load arrow should be directed to the SAV symbol because SAV distribution proved to be centric to the Bay Park effluent discharge location.

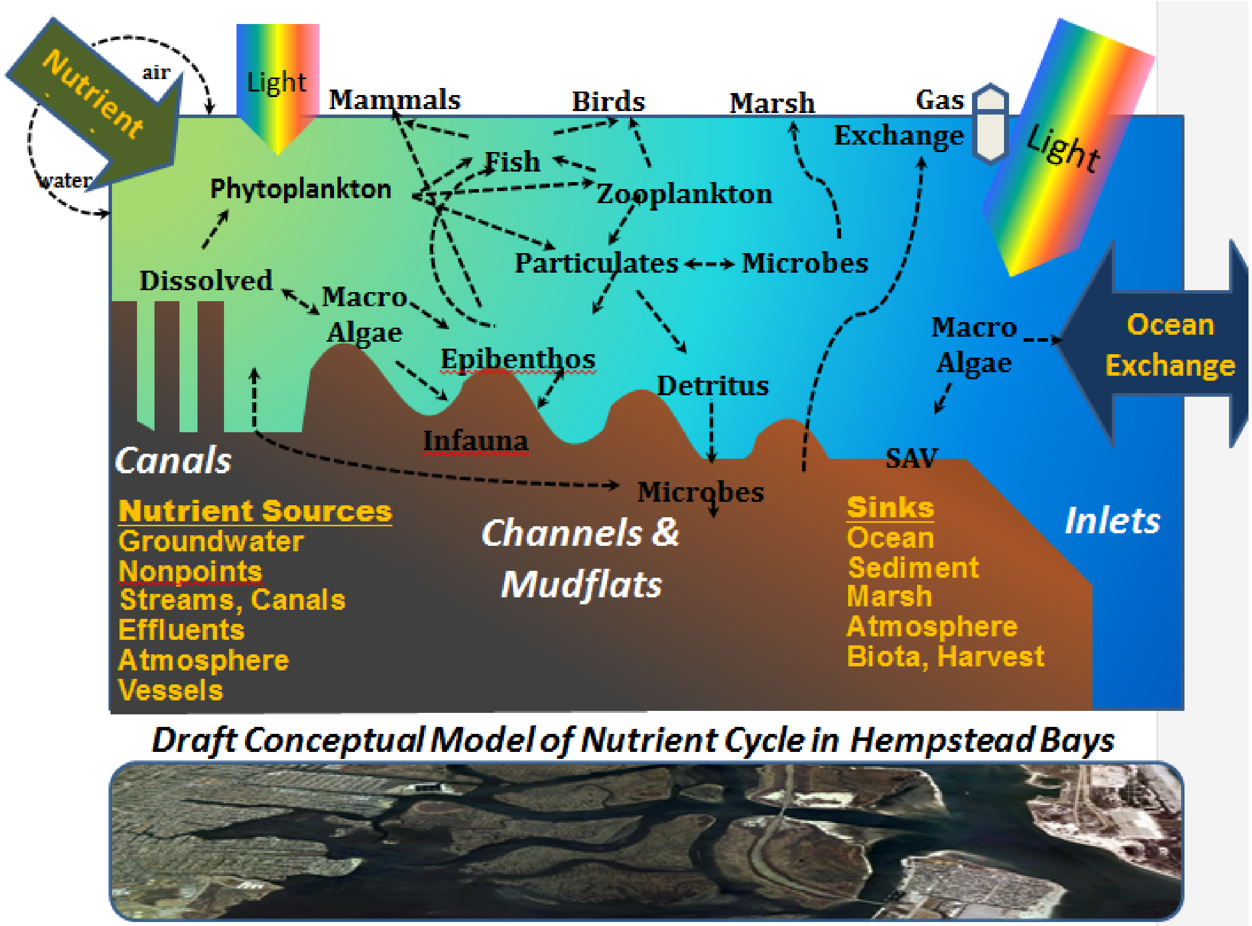

Figure 2. Draft Conceptual Model of nitrogen cycling through Western Bays' canals, channels, mudflats and inlets

Each of the investigations conducted were designed based on data quality objectives (DQO) which contributed to portions of the overall strategy. Data quality objectives reflected each unique component of the ecosystem being investigated. Each of the guiding DQOs shared at least of one the four main lines of investigation in this strategy (Table 1).

1. Can evidence of use impairments be observed consistently in areas of the Western Bays?
2. Which types of observations, if any, are most effective at quantifying use impairments?
3. Can apparent use impairments in Western Bays be attributed to excessive nitrogen loads?
4. Which nitrogen management options are most promising in the Western Bays setting?

Table 1 summarizes the step-wise DQO progression aligned in accordance with the project strategy and goals. Specific studies were deployed to achieve one or more steps in the progression. Some steps were addressed via lines of evidence from multiple studies. The details of each study were designed, documented, reviewed and refined by stakeholders in the form of Quality Assurance Project Plans (QAPPs) or Sampling and Analysis Plans (SAPs).

Table 1. Western Bays Ecosystem Assessment and Nitrogen Management DQO Steps.

DQO STEPS	WESTERN BAYS STUDY QUESTIONS (and supporting studies)			
	Is impairment evidence consistent?	**What are best endpoints to document impairment?**	**Is impairment linked to excess nitrogen?**	**Which nitrogen management options are most promising?**
1. State the Problem	303(d) listing is vague	Dissolved oxygen is not an adequate indicator	Conceptual Model of nitrogen is broad	Cannot afford to deploy all NMP options
2. Identify Decisions	Does impairment need TMDL?	Select alternative endpoints to track	Does impairment link with loads?	Prioritize NMP options by load size and cost (CJ)
3. Select Input	Find data that reveal trends (ABDEFG)	Evaluate varied data for utility (ABCDEFGHI)	Find bloom or habitat metrics (BCDEFGHI)	Scan literature and stakeholders
4. Set Boundaries	Focus on regions of bay and periods (ABCDEFGH)	When/where are data duly sensitive? (ABCDEFGHI)	Identify fate and transport paths (CEFGHJ)	Match NMP options with dominant sources (J)
5. Establish Rules	Agree on “if-then” triggers (GIK)	**Can data support regulations?** (ABCDEFGHIK)	Find thresholds and exceedances (ADEFGI)	**Can NMP options abate major nitrogen sources?** (JK)
6. Set Error Tolerance	Recognize variability (ACDEFGH)	Can uncertainty support regulations? (ABCDEFGHI)	**Weigh numerous lines of evidence** (CIJK)	Unique to each Stakeholder’s NMP
7. Design Sampling	**Fill in gaps with new samples** (CDEFGH)	Recommend monitoring		

<u>Niche Studies That Produced Ten Lines of Evidence</u>
A = Assessment of Historical Water Quality Sampling Data from Town of Hempstead (1980s-2010)
B = Delineation of Tidal Wetland Change (1974 and 2008)
C = Hydrodynamic Modeling of Western Bays
D = Infaunal Community Assemblage
E = Sediment Profile Imagery reconnaissance
F = Ulva Coverage Surveys
G = Ambient Water Quality Surveys
H = Sediment Nutrient Flux Experiments
I = Algal Growth Kinetics Experiments
J = Assessment of Nitrogen Loading Estimates and Management Options
K = Integration

Execution of the 4-part strategy through a sequence of QAPPs and SAPs allowed for a fair extent of adaptive management. Insights from historical data guided design of new water quality sampling. Hydrodynamic simulations were vital to both sampling designs and to improve input terms of the nitrogen loading budget.

3.0 INTEGRATION OF FINDINGS

This report is organized to highlight pertinent lines of evidence according to the four main questions posed. Results of each specialty study are presented in their entirety as appendices to this report. The body of this report focuses on how the findings produced inform key DQO elements of each question.

3.1 Can Evidence of Use Impairments Be Observed Consistently in Areas of the Western Bays?

NEIWPCC requested this study because "…the complex relationships between nutrient loads, ambient water column concentrations, and environmental fate and effects, the development of additional or alternative nutrient criteria and management scenarios is needed for these systems." The ecosystem services that are expected to occur within these waterbodies are clear based on their use designation classification as either SA, SB or SC waters (Figure 3). These classes include the following services:

a. Class SA –shellfishing for market purposes, primary and secondary contact recreation and fishing, fish propagation and survival
b. Class SB –primary and secondary recreation and fishing, fish propagation and survival
c. Class SC –fishing, fish propagation and survival. The water quality shall be suitable for primary and secondary contact recreation although other factors may limit the use for these purposes

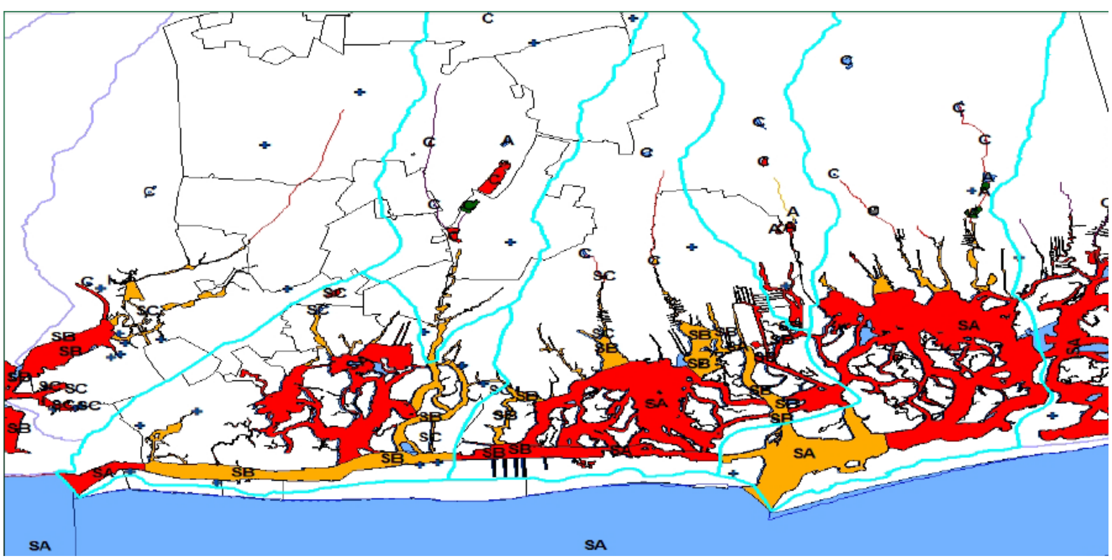

Figure 3. NYS DEC Waterbody use designations for segments of Western Bays.

A problem with these use designations is that questions persist about how best to determine whether these uses are being met in shallow coastal embayments such as the Western Bays. Stakeholders in Pleasant Bay on Cape Cod, MA and Great Bay in NH recently examined similar concerns about whether their water quality criteria were adequately protecting those shallow embayments. Impairments for some of uses are clearly articulated in regulations, for example, contaminants in edible fish tissue must remain below critical thresholds in order to protect human health from seafood consumption risks. And primary contact concentration thresholds exist for some priority pollutants in ambient waters. However, being allowed to eat the fish and being allowed to wade into the waters to fish, does not necessarily mean the waters are meeting the recreation, fishing and fish propagation designated uses. The waterbody may experience other degrading stresses which leave its habitat depauperate of flora and fauna to support lower food web prey species. This study intentionally focuses on possible impairments due to nutrients.

This investigation of possible impairments augments the current official status of the waterbodies:

- NYS DEC's 2012 303(d) list Part 1 of impaired waterbodies includes Hempstead Bay with nitrogen as the listed pollutant.

- NYS DEC's 2012 303(d) list Part 2c for waterbodies with categorically impaired shellfishing includes: East Bay, Middle Bay, East Rockaway Inlet, Reynolds Channel east, Hempstead Bay, and Woodmere Channel with pathogens as the listed pollutant.

This section of the report describes numerous lines of field sampling evidence pertinent to ecosystem services vital to each of the four designated uses: shellfishing for market purposes; primary and secondary contact recreation; fishing; fish propagation and survival. This evidence is initially presented to assess presence of possible impairments. In later sections, the measurements will be discussed relative to their ability to serve as consistent regulatory endpoints. Later sections will also examine the geospatial extent and applicability of the impairment evidence.

3.1.1 **Shellfishing for Market Purposes**

This study did not attempt to find evidence of impairments to the marketability of shellfish from the Western Bays. Many shellfish beds in Western Bays area are already closed based on administrative rules or the results of monitoring for pathogens. Those impairments are not considered pertinent to this study of nitrogen management concerns. The role of dissolved oxygen to support shellfish growth is discussed in the section below as it pertains to fish habitat quality.

Phytoplankton sampling performed by the State University of New York at Stony Brook School of Marine and Atmospheric Sciences (SoMAS) in 2010 prior to this study identified the presence of harmful dinoflagellates in Hewlett Bay and Middle Bay. They found *Alexandrium fundyense*, which is associated with paralytic shellfish poisoning, and *Dinophysis acuminata*, which is associated with diarrhetic shellfish poisoning (Figure 4). This could lead to an impairment determination based on ECL 47.4 Hazardous Shellfish for Use as Food for Human Consumption. Hewlett Bay has experienced three different HABs. The harmful raphidophyte, *Heterosigima akashiwo*, the paralytical shellfishing poison producing

Alexandrium fundyense, and the diarrheic shellfish poison causing *Dinophysis acuminata*. *Heterosigma* was observed over multiple years (2010 – 2012) at high levels. *Alexandrium* and *Dinophysis* were observed during purposeful sampling in 2010.

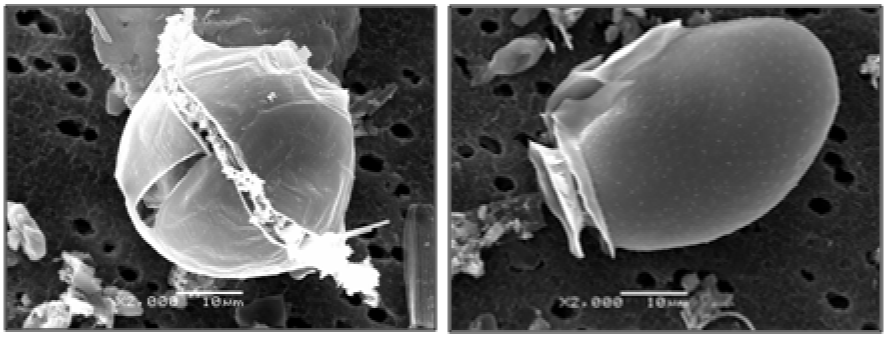

Figure 4. Harmful dinoflagellates *Alexandrium fundyense* and *Dinophysis acuminata* found in Hewlett Bay and Middle Bay 2010.

Nutrient levels in parts of the Western Bays appear similar to conditions observed elsewhere on Long Island where harmful algal blooms have been led to shellfish bed closures. It is not clear whether pre-emptive monitoring for these HABs occurs in the Western Bays segments which are closed to shellfishing.

3.1.2 **Primary and Secondary Contact Recreation**

The bacterial decomposition of Ulva mats can release hydrogen sulfide at substantial rates. H_2S release without adequate diffusion or ventilation may lead to malodorous conditions or harmful ambient concentrations. The NYS Public Health Protection threshold for 1 hour of exposure is 0.01 ppm H_2S. OSHA workplace safety standards are 1ppm H_2S for 15 minutes. (http://www.health.ny.gov/environmental/chemicals/hydrogen_sulfide/) This study did not attempt to sample and quantify ambient H_2S concentrations associated with decaying algal mats or from fugitive emissions.

Humans detect H_2S at levels ranging from 0.001 to 0.1 ppm. Thus, if recreational users of the bays, beaches or coastal communities ever smell H_2S, it is conceivable that such incidents could approach public health risk levels. If recreational users of the bays are deterred from recreational uses of the bay, either via primary or secondary contact, then it seems reasonable to conclude that this is evidence of a designated use impairment. It is not necessarily a public health risk, but it could be. Part 703.2 of the NYS CRR establishes that nitrogen shall not occur "…in amounts that will result in growths of algae, weeds, and slimes that will impair the waters for their best use." The linkage between nuisance levels of Ulva and nitrogen loading into the Western Bays is discussed directly in section 1.3 below.

3.1.3 **Fishing**

Evidence of impairments to general water quality criteria depends on locations within Western Bays and water depth. Water quality impairment evidence based on dissolved oxygen was found repeatedly in Hewlett Bay bottom waters. Certain locations in Hempstead Bay and Middle Bay were observed to have impaired benthic habitats (see Benthos section below) and reduced infaunal populations important as food sources to fisheries. Possible impairments due to harmful algal blooms, such as *Cochlodinium polykrikoides* which can kill fish and shellfish, have not been investigated in this study.

Conventional monitoring of water quality in Reynolds Channel by the Town of Hempstead in the summers between 2003-2009 suggests that dissolved oxygen levels near the surface do not drop below 5 mg/L, thus do not approach the acute criteria for impairment (3.0 mg/L NYS Acute standard)(Figure 5). Detailed summaries in Appendix F.

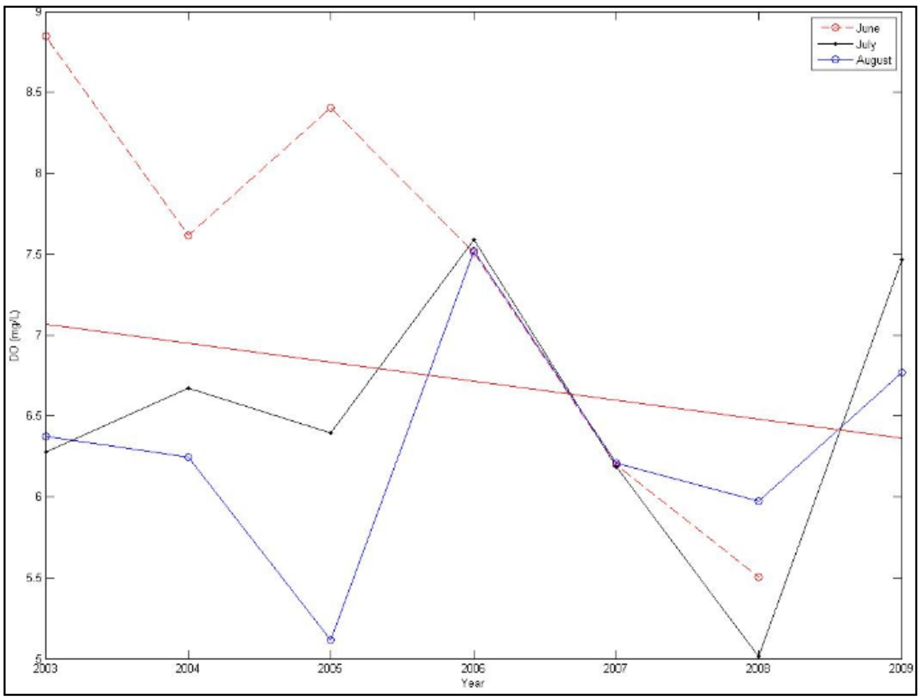

Figure 5. Average Dissolved Oxygen levels at the surface in Reynolds Channel during summers 2003-2009. (The solid line is the average of all measurements).

Discrete samples collected seasonally from bottom waters at 5 locations by SoMAS during 2012 reveal dissolved oxygen levels within the range of historical observations by the Town of Hempstead, although the Bay Park outfall which is closest to the Town of Hempstead Reynolds Channel site in August and September 2012 (Figure 6) did approach the saltwater acute water quality standard of 3.0 mg/L DO which is based on adult/juvenile survival (NYS DEC, 2008).

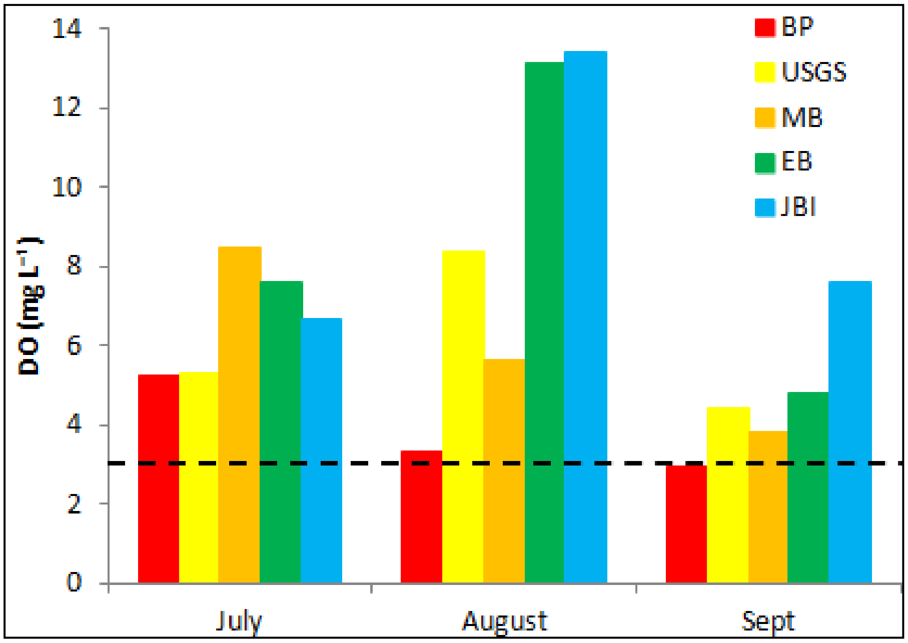

Figure 6. Dissolved Oxygen data from Bay Park outfall, Hog Island, Middle Bay, East Bay and Jones Inlet, as sampled by SoMAS 2012 relative to acute standard of never less than 3.0mg/L.

However, a closer look at data from locations within Hewlett Bay demonstrates that waterbody segment experiences impairments based on dissolved oxygen levels in bottom waters which were frequently well below the acute standard which is intended to support the larval recruitment criterion. (Figures 7, 2011 & Figure 8, 2012).

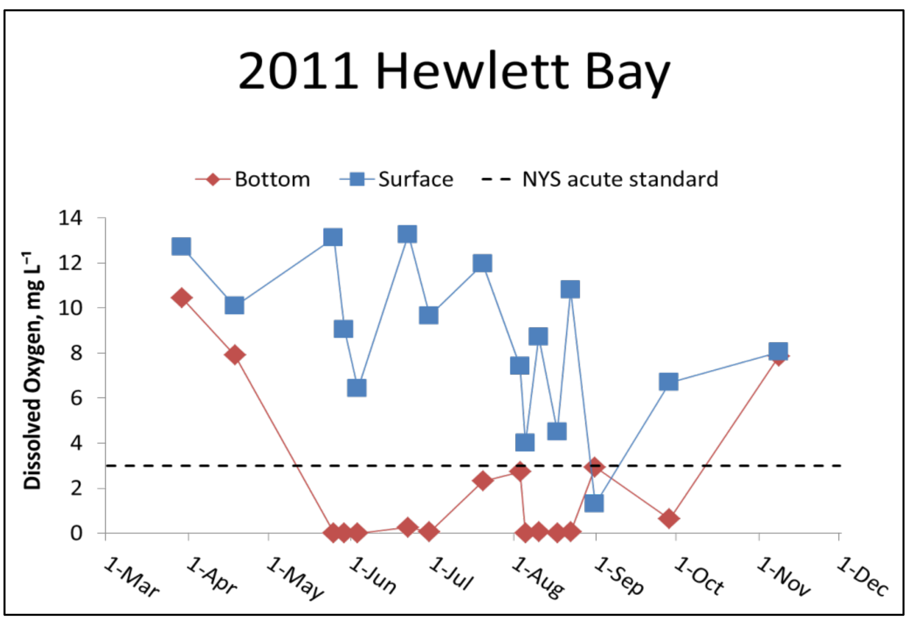

Figure 7. Dissolved Oxygen data from Hewlett Bay, as sampled by SoMAS 2011.

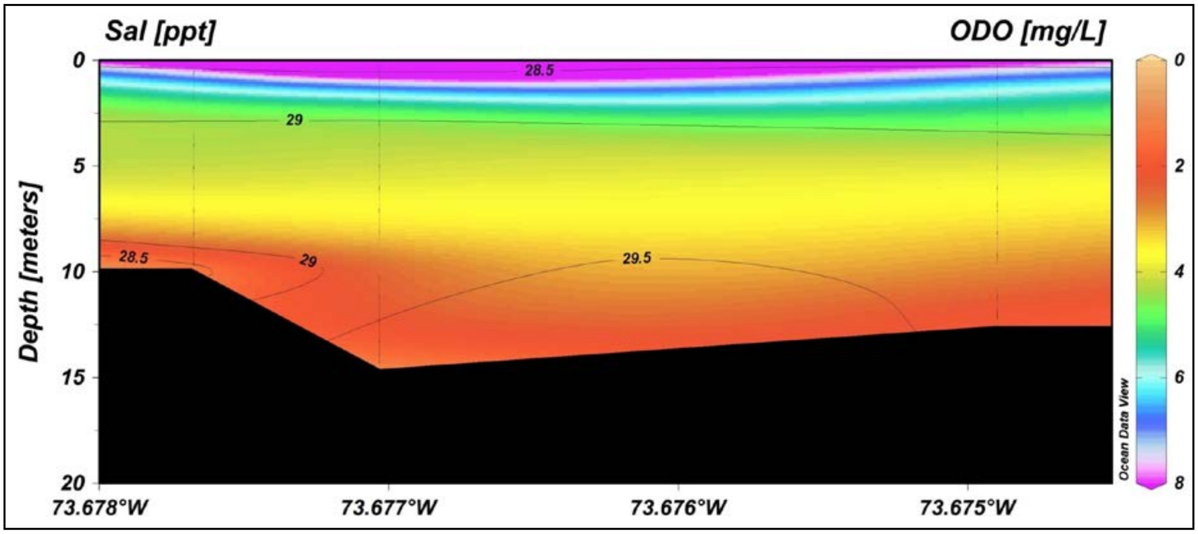

Figure 8. Vertical cross section profile of dissolved oxygen concentrations (mg/ L) with salinity (ppt) contour lines in Hewlett Bay during early September of 2012.

Physical evidence of this impairment would not necessarily manifest in fish kills due to motility and avoidance behavior.

3.1.4 Fish Propagation and Survival

Eutrophication fueled by nutrients appears, based on empirical observations of pH, to create conditions which could impair fish propagation. Anecdotal evidence of a long term decline in resident winter flounder young of year, even without embayment specific details, further suggests impairments to this designated use.

Excessive nutrient loading into coastal ecosystems such as the Western Bays promotes algal productivity and the subsequent microbial consumption of this organic matter lowers oxygen levels and contributes toward hypoxia. Microbial degradation of organic matter produces CO_2 and reduces pH. To assess the potential for eutrophication-driven acidification in Hewlett Bay, SoMAS mapped the pH levels in September of 2012. Measurements revealed that the pH levels in Hewlett Bay were below 7.1 in the bottom layer and below 7.6 through much of the water column (Figure 9). These low pH conditions could have been caused by tidal influx of treatment plant discharge and the subsequent enhanced bacterial decomposition. These levels of pH have previously been shown to yield elevated mortality in larval finfish and shellfish (Talmage and Gobler 2010; Baumann et al. 2012) suggesting that acidification, which has been intensified by climate change (Doney et al. 2009), may be currently altering the ability of Hewlett Bay to support robust fisheries.

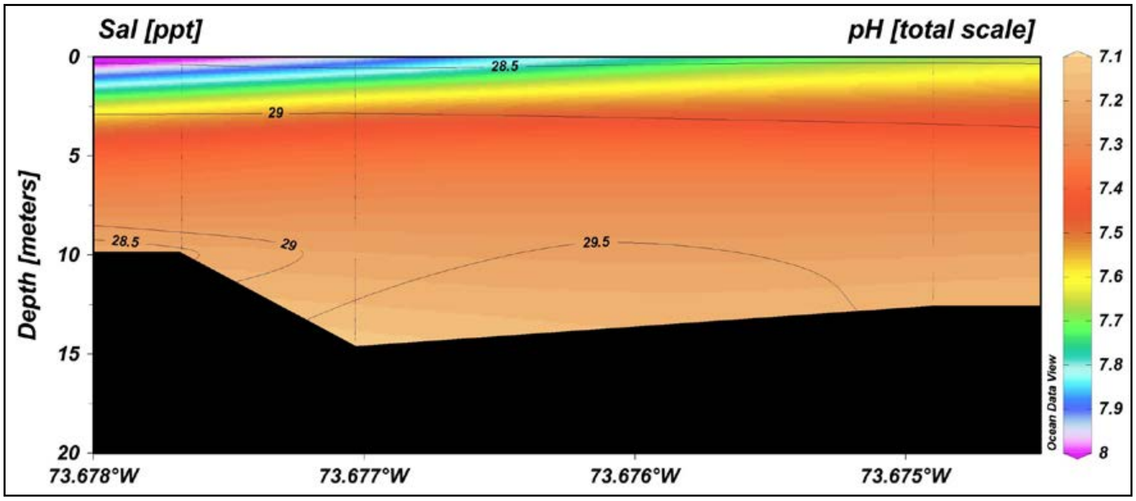

Figure 9. Vertical cross section profile of pH (total scale) with salinity (ppt) contour lines in Hewlett Bay during early September of 2012.

Empirical evidence of impairments to the fish propagation designated use was also observed during seining operations as part of an unrelated study by SoMAS. They were unable to catch a substantial amount of young of year winter flounder specimens for experiments (A. McElroy pers.comm. 2013). The apparent lack of young of year in Hempstead Bay for this commercially important species provides anecdotal evidence that conditions in 2012 were stressful enough overall to substantially decrease the population. Absence of specimens is not attributable to any one particular physical, chemical, or biological or ecological stressor. That young of year flounder were caught at sites in eastern portions of the Western Bays suggests that this endpoint may warrant higher spatial resolution sampling to assess whether levels show consistent trends.

3.1.5 Benthos Status

The relatively low abundance and low diversity of benthic infauna at some locations in the Western Bays indicate that benthic habitat provides varying levels of quality in ecosystem services (Figure 10) and that these quality levels are not easily distinguished simply according to regions. The infaunal sampling study was complemented by reconnaissance sediment profile images (SPI) at stations throughout the bays. The SPI results further informed the range of benthic conditions. The sediment nutrient flux experiments showed that substantial differences in bottom layer water quality can change even within scales of 10s of meters. No attempt was made to use or derive statistical values to classify locations as impaired or not impaired because such community statistics require inter-year replication beyond the scope of this one-time reconnaissance survey. The reconnaissance survey effectively answered the first two fundamental questions of this study pertaining to whether metrics could detect impairments and whether the metrics might be used consistently enough to support regulatory management of the waterbody. Initial findings indicate infaunal metrics, in association with normalizing factors, may be sensitive enough to detect impairments, however they will require substantial replication and spatial coverage to establish baseline community statistics which could support regulations.

The benthic infaunal community assemblages, both counts and biodiversity, at many stations throughout the Western Bays included a relative abundance of opportunistic species which are known to inhabit stressed environs less suitable for a wider diversity of more sensitive species. Differences in physical settings cannot be discounted regarding possible contributing factors to apparent impairments. The pattern of habitat factors displayed in figures of Appendix B demonstrate that percent fines co-occur along the separation between groups I and II more consistently than other factors such as nutrients.

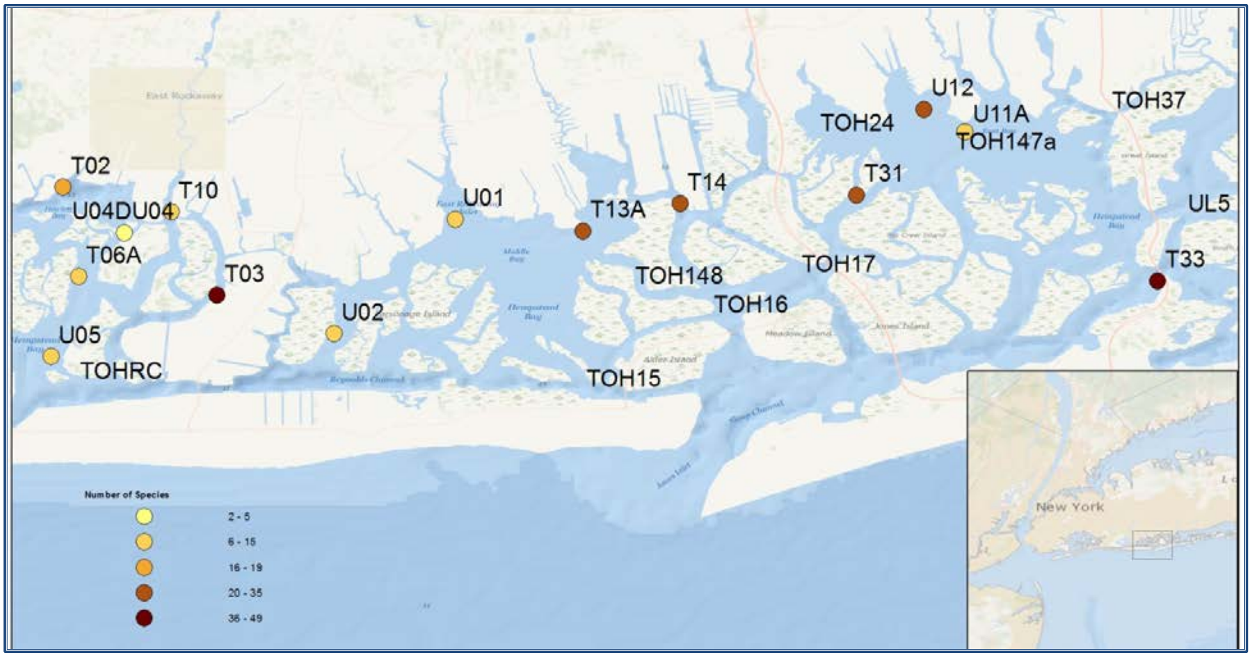

Figure 10. Number of benthic species observed in Fall 2012.

Statistical examination of multiple characteristics seen at infaunal sites can spread them out in two dimensions as shown in Figure 11. The Hempstead Bay infaunal community analysis identified that stations generally separated into two consistently dissimilar groups of eight stations (Group I) and six stations (Group II), with two outlier stations (ToH16, U04D) that had communities that were extremely dissimilar from all other stations (Figure 11). Examples of highlighted characteristics of these groups can be seen in Figures 12 and 13, as well as Appendix B.

In general, overall community similarity was relatively low among all of the stations; the two most similar stations had a similarity value of about 69%, which experience tells us is low. This overall dissimilarity is notable because one polychaete species, *Streblospio benedicti*, was the most or second most, abundant organism at all stations except the two "outlier" stations. Other annelid worms, such as *Capitella capitata* and unidentified oligochaetes, were among the other predominant taxa at many stations. These taxa are commonly referred to as "opportunistic" because they have the ability to rapidly colonize disturbed habitats. The predominance of these taxa indicates that the infaunal communities in the bay experienced some degree of environmental stress this year. A longer term study with similar ecosystem assessment goals relative to nutrient levels in Jamaica Bay (NYCDEP 2006) identified

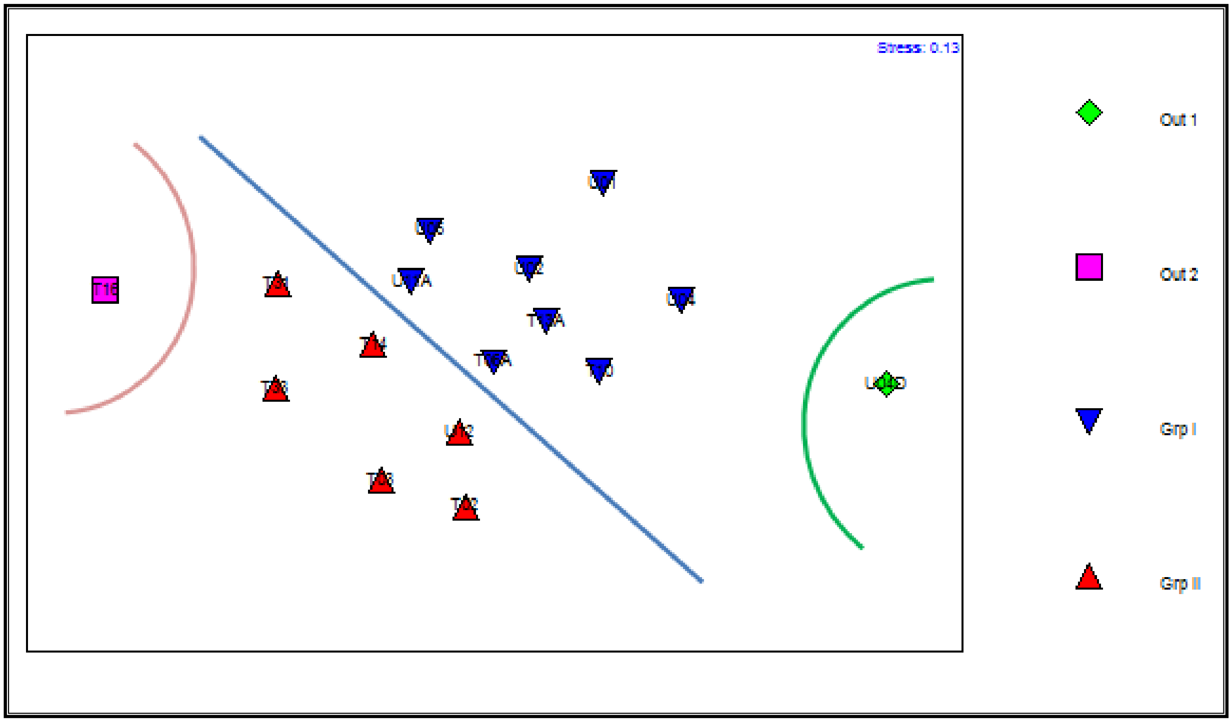

Figure 11. Non-metric Multi-dimensional Scaling plot of Bray-Curtis similarity. Outlier 1 and Outlier 2 are the least similar of the "groups".

numerous locations where benthic community structures would abruptly degrade after experiencing a protracted period of low dissolved oxygen and or high ammonia levels. Afterwards, within the same season, some of these locations would then see recruitment of opportunistic infauna from less stressed regions nearby, which would restart growth.

Of the two main infaunal community "groups" seen in the Western Bays, especially when considered along with the other environmental data collected, some sites within the bays are found to generally appear less stressed than others. Being "stressed" does not absolutely denote an impairment threshold relative to designated uses. Multiple years of such sampling results could establish baseline patterns from which to judge progress or impairments. The eight stations identified as "Group I" in this first year share ecological features such as low numbers of species, low abundance, and low species diversity. These features are often associated with somewhat impaired benthic communities.

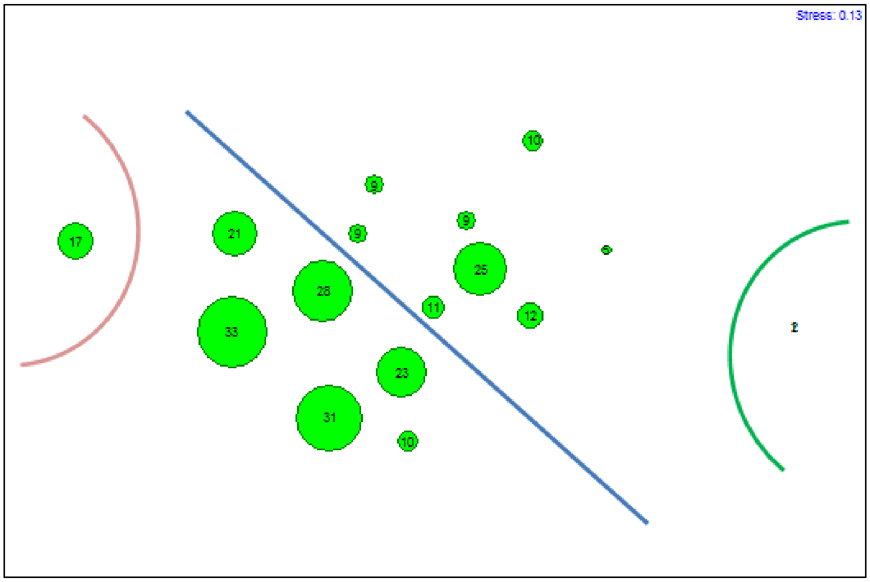

Figure 12. Number of macrobenthic species observed in Western Bays stations Fall 2012.

The worm *Streblospio benedicti* accounts for 50 to 75% of the organisms at these stations. Relative to Group I stations, the six stations identified as "Group II" share high numbers of species, high abundance, and greater diversity. *Streblospio benedicti* still predominates the fauna, accounting for 26 to 88% of the organisms. Regardless, the combination of higher species numbers (Figure 12), abundance, and diversity suggest that the Group II stations may encounter less stress than the Group I stations. A sediment profile image (SPI) measure, apparent Redox Potential Discontinuity (aRPD) depth, is slightly greater for Group II stations than for Group I stations. This metric is often influenced by infaunal community activity, with deeper aRPD depths indicating somewhat greater infaunal activity.

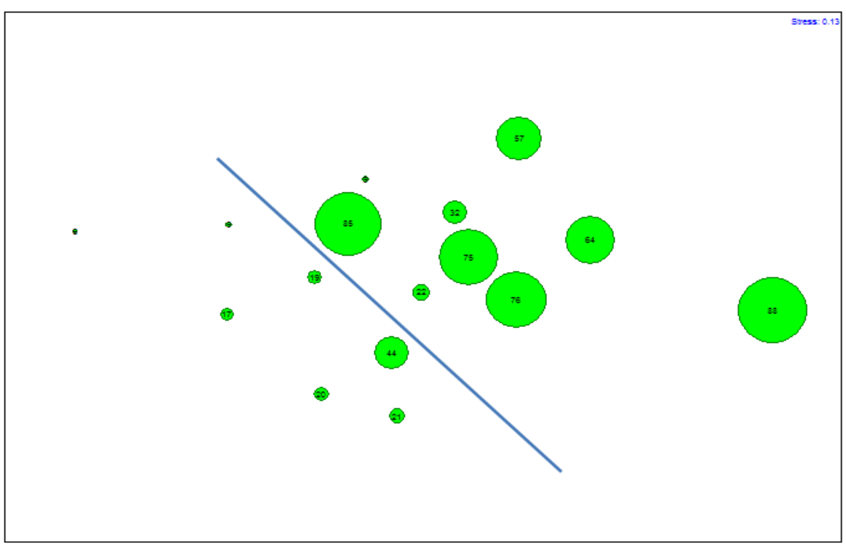

Figure 13. The percent of sediment classified as fine grained at each Western Bays station Fall 2012 (%; 100-(sand+gravel)).

The physical setting of each station correlates to some extent with the station groupings based on biodiversity. Group 1 stations generally have a much higher percentage of fine grained sediment (Figure 13) which can be indicative of a lower energy hydrodynamic regime and a depositional setting. Note that the 2012 benthic sampling station locations were located in order to cover regions of the Western Bays ecosystem, not specific bottom types (which can vary within hundreds of meters across the profile a channel or bay). Graphic depictions of the differences in sediment texture, i.e. sheer stress of bottom currents, that can occur within close proximity can be seen in Figure 14 where sandy oxygenated conditions at Ulva5 are contrasted by finer sediments with shallower apparent redox potential discontinuity horizon elsewhere in Hempstead Bay. Similar disparities are seen amongst stations with Middle Bay and East Bay as depicted in Appendix B. The SoMAS model discussed in Appendix E provides further evidence of the fine-scale spatial heterogeneity of bottom sheer stress.

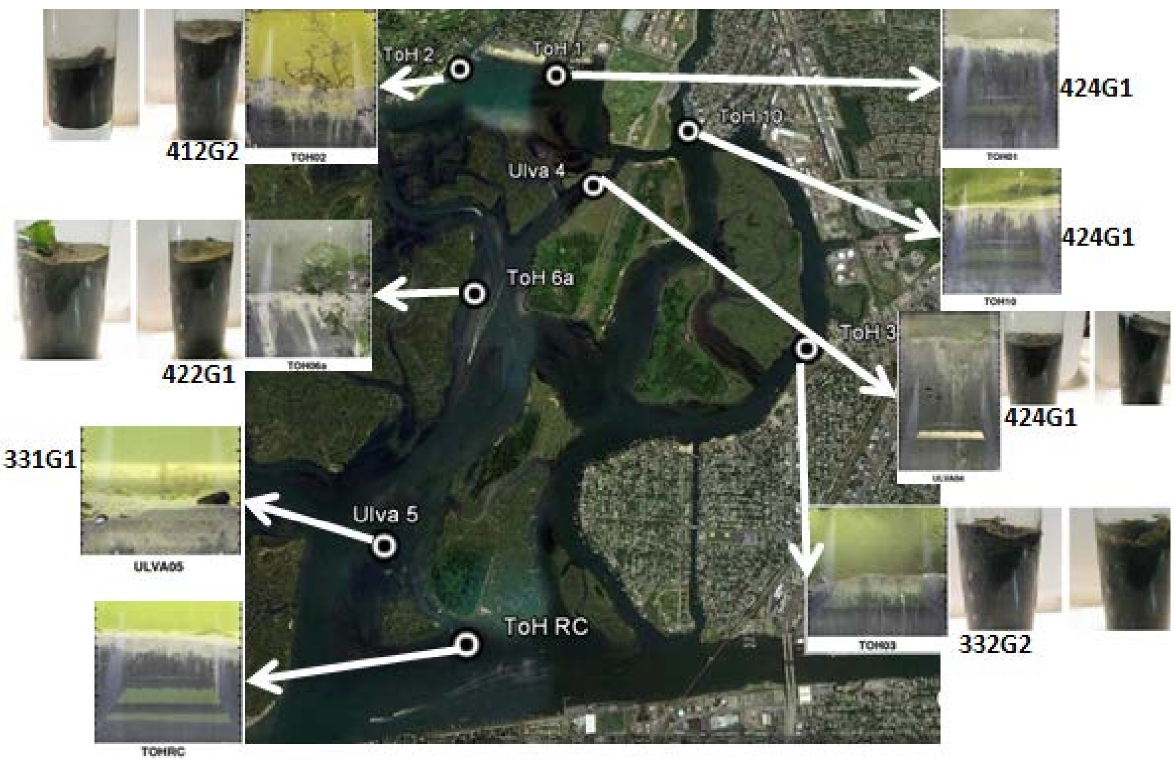

Figure 14. Hempstead Bay SPI images; core photos; quartiles of water quality, Ulva % cover, % fines; and infaunal similarity group #. Composite figures for Middle Bay and East Bay are available in Appendix C.

There does not seem to be a geographical division between the two community groups because they span all three bays, and often two stations in different community groups are located close to each other. The habitat data suggest that Group I stations may be in more depositional areas than Group II stations because they have a much lower proportion of sand than Group II stations. This supposition is supported by an SPI measure, prism penetration, that is deeper for Group I stations than Group II stations, indicating the presence of "softer" sediments at Group I stations. *Ulva* cover at the Group II stations was more than twice that found at the Group I stations; n.b. Ulva cover can be attached fronds growing in meadows as contrasted with dense mats deposited on the bottom.

Comparing infaunal similarity results with contour plots of more dynamic, advective water quality parameters for the bay shows that temperature, salinity, and photosynthetically active radiation are not closely related to the infaunal community groups (Appendix B). However, other water quality features generally appear to show some "relationship" to the primary infaunal community structure. The summed median values for water column nitrate, phosphate, ammonium, and silicate are all greater among Group I stations than Group II stations. The median ammonium values for Group I stations is almost six times greater than the median value for Group II stations. These general observations seem to suggest that the infaunal community may be more stressed in locations where water quality is relatively worse. However, they do not eliminate possible alternative explanations. For example, the finer sediment texture among Group I stations may contain higher levels of pollutants and total organic carbon (which were not measured) than areas with coarser sediments. Another caveat is that these general patterns do not always hold for each station, especially within one season of sampling. For example, the infaunal communities at stations ToH2 and ToH3 are aligned with the Group II stations, yet the water column nutrient levels at these stations are among the highest measured in the bay. The fauna at Station U11A aligns with Group I stations, but the water column nutrient levels there are relatively low.

Returning to the main questions of this study, the infaunal community structure metrics (species identities, ecological indices, similarity) suggests that there is detectable evidence of impairment in the bay. However, the study metrics do not allow the extent of impairment to be quantified definitively. The general comparison between nutrients and infaunal community structure hints at possible connections. However, other variables may confound the effect of those two ecosystem components. The more impaired community occurs in fine sediments indicative of a depositional area. That depositional area likely indicates an area of relatively poor water circulation, which also contributes to higher water column nutrient levels remaining in an area. That the two main infaunal community groups span all three of the sub-bays comprising the Hempstead Bay system suggests that factors affecting benthic habitats occur across all sub-bays, which supports management of the system as a single unit.

3.2 What are Suitable Indicators/Endpoints of Impairment?

3.2.1 Indicator/Endpoint Background

One of the primary objectives of this project is to evaluate development of additional or alternative nutrient criteria to improve understanding of the status of the ecosystem and attainment its designated uses (discussed above). The designated uses of the Western Bays range from shellfishing for market purposes, primary and secondary contact recreation and fishing, fish propagation and survival (figure 3) depending on site-specific considerations. NYS DEC's 305(b) assessment methodology describes numerous means for water quality examinations to determine the level of support of designated uses based on site-specific use restriction orders, numeric or narrative standards and criteria, or surrogate water quality indicators. These metrics are designed to protect against concerns for public health, aquatic life, wildlife, recreation or aesthetics. New York's Code of Rules and Regulations NYCRR Part 700 define multiple types of metrics which may pertain to regulation and enforcement of water quality. Some of the available metric types applicable to issues associated with nutrient loads examined in this study are listed alphabetically below:

1.0 *Biologically-based dose-response model* means a model that describes and quantifies the key events in the molecular, cellular, tissue, or organismal responses to a chemical or other toxic pollutant across a range of doses. Model parameters should represent biological phenomena rather than arbitrary statistically-derived values such as polynomial regression coefficients. Such models, if they accurately describe the relationship between dose and response within the range of experimental observation, may provide biological justification for predicted responses at doses below the range of observation.
2.0 *Effluent limitations* mean any restriction on quantities, qualities, rates and concentrations of chemical, physical, biological, and other constituents of effluents that are discharged into or allowed to run from an outlet or point source or any other discharge within the meaning of section 17-0501 of the Environmental Conservation Law into surface waters, groundwater or unsaturated zones.
3.0 *Groundwater effluent limitations* mean those effluent limitations that have been adopted in section 703.6 or developed in accordance with section 702.16(c) of this Title for protection of groundwater.
4.0 *Guidance value* means such measure of purity or quality for any waters in relation to their reasonable and necessary use as may be established by the department pursuant to sections 702.1 and 702.15 of this Title.
5.0 *Pathogenic organism* means any disease-producing organism.
6.0 *Pollution* means the presence in the environment of conditions and/or contaminants in quantities of characteristics that are or may be injurious to human, plant or animal life or to property or that unreasonably interfere with the comfortable enjoyment of life and property throughout such areas of the State as shall be affected thereby.
7.0 *Specific MCL* means a maximum contaminant level (MCL) included in 10 NYCRR 5-1.51, 5-1.52 or 5-1.55 for either an individual substance or group of substances. A Specific MCL does not include the 10 NYCRR Part 5 MCLs for principal organic contaminants or unspecified organic contaminants.
8.0 *Standards* mean such measures of purity or quality for any waters in relation to their reasonable and necessary use as may be established by DEC pursuant to section 17-0301 of the Environmental Conservation Law.
9.0 *Water quality-based effluent limitations* means effluent limitations for surface waters that are derived from water quality standards or guidance values.

This study produces evidence which can inform numerous types of metrics listed above. The algal kinetics experiments were intended to function as a dose-response model of how algal growth in the Western Bays changes with nitrogen levels. This study highlights the relative importance of groundwater discharge to the nitrogen budget, however, it also highlights how limits to groundwater are unable to be suggested due to the unknown extent to which groundwater nitrogen may be processed at the sediment water interface before entering the bays. Other lines of evidence may serve as input to guidance values or standards based on ecosystem condition relative to designated uses.

The terms metrics, indicators, ecosystem measures and endpoints are used as descriptive equivalents in this report even though their semantics can differ depending on their eventual regulatory application. Metrics are typically quantitative measures and may represent environmental phenomena ranging from stressor levels, such as pH, to ecosystem service outputs, such as seafood harvest. Numeric standards and threshold criteria are often expressed as metrics, e.g. dissolved oxygen levels. Standards may also be non-quantitative, such as nuisance determinations or subjective aesthetic complaints which can be indicators of a designated use impairment. Some metrics, standards or criteria, such as hypoxia or biodiversity, are observed at the end of a progression of ecosystem functions, thus they may also be referred to as endpoints.

Based on the historical data and results of the studies conducted as part of this project, this section of the report characterizes the potential effectiveness of both traditional and alternative metrics, i.e. surrogate water quality indicators, which may be able to augment the existing numeric or narrative standards and criteria to assess the status of the ecosystem relative to Western Bays designated uses. If determined to be effective pursuant to the New York State Nutrient Standards Plan of 2011, then these indicators could then be applied by resource managers as regulatory tools to restore the ecosystem if impairments are demonstrated to be due to anthropogenic stressors. Thus, the metrics also need to be evaluated for suitability based on their intended regulatory use and whether (and where) they can adequately assess waterbody impairment.

The conceptual model (Figure 2) serves as a context for assessing and prioritizing the most relevant stress and response components of the ecosystem to examine. Given the complexity of the conceptual model, it is not surprising that there are multiple levels of potential endpoints associated with the primary issue in the Western Bays – eutrophication. Figure 15 from Bricker et al. (2003) highlights an example of these multilevel endpoints associated with eutrophication in a typical coastal embayment. Additional examples from waterbodies across the country are provided to show how others have assessed and selected endpoints and how local geomorphology, hydrodynamics and knowledgebase can affect the utility of specific indicators.

In Chesapeake Bay, endpoints have been developed for dissolved oxygen (DO), water clarity, and chlorophyll *a* (US EPA 2003). DO criteria have been assigned to five different regions of the bay defined by uses and depth; water clarity criteria have been assigned to four different salinity regimes. For chlorophyll *a*, a narrative standard was established for the entire bay. The fact that criteria were able to be established for so many different regions of Chesapeake Bay is a testament to the extraordinary amount of research conducted in that area and the vast amount of data associated with those efforts. Investigators in this study crafted the initial conceptual model for Western Bays based on their experience using the Chesapeake Bay approach to define waterbody regions in Jamaica Bay.

In Long Island Sound (LIS), problems with seasonal hypoxia/anoxia in the western portion of the Sound led to establishment of the Long Island Sound Study in 1985. After 15 years of monitoring and related modeling and synthesis, a Total Maximum Daily Load (TMDL) for nitrogen loading to the Sound was approved by the EPA and the states of New York and Connecticut. That TMDL was established in order to meet DO water quality endpoints in LIS and a multiyear effort has been phased in by the States to meet the TMDL of a 58.5% reduction in nitrogen loading by 2014 (NYS DEC, CT DEP 2000). As was the case with Chesapeake Bay, the LIS DO criteria were established after many years of monitoring and data evaluation. The scale of hypoxia across varied geomorphologic regions of Long Island Sound presented a distinctly different context than the relatively small region of Western Bays where DO criteria are exceeded.

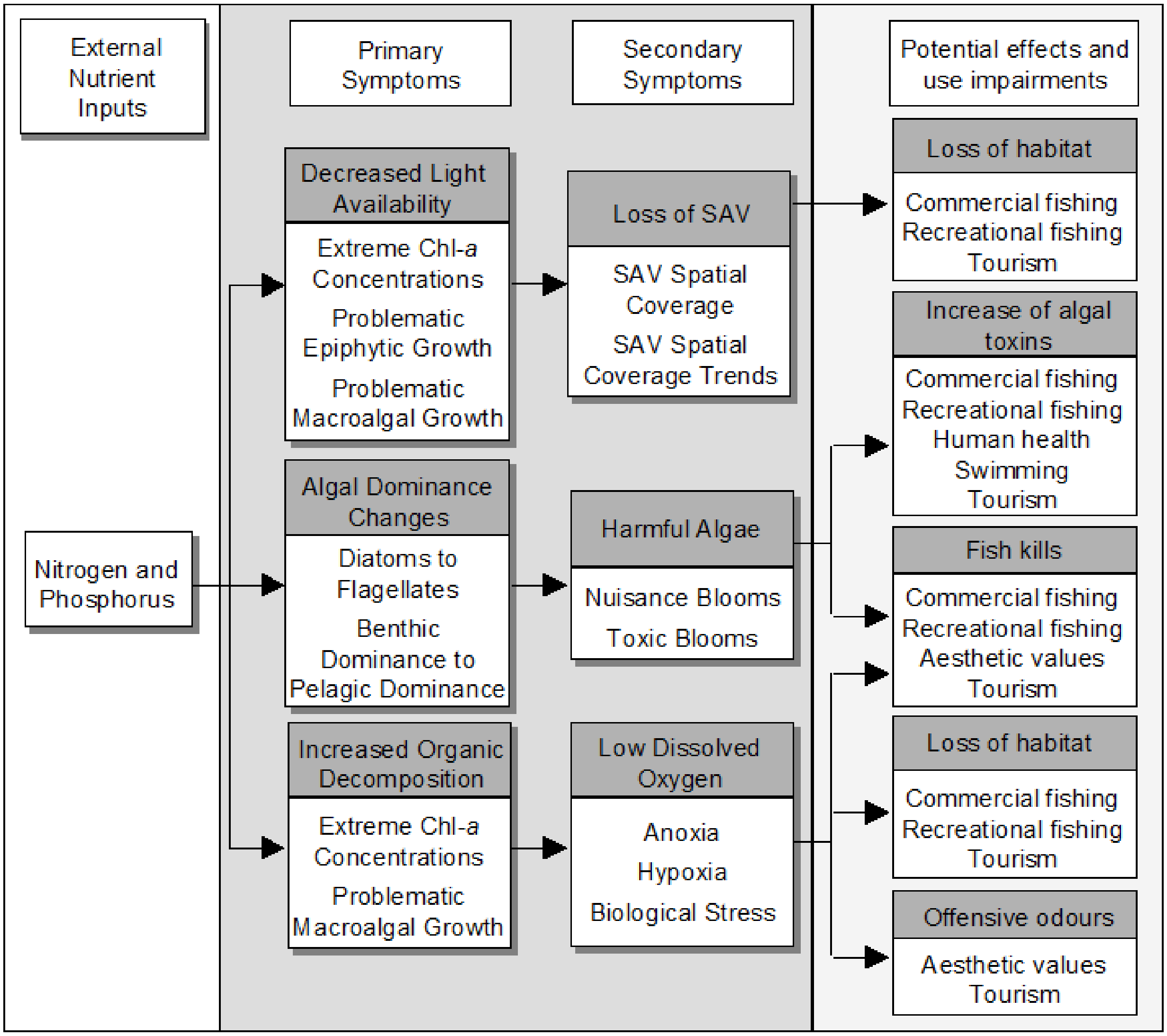

Figure 15. Example of multilevel endpoints associated with eutrophication (Bricker et al. 2003)

In Yaquina Estuary Oregon, existing data were used to examine spatial and temporal trends and a "weight of evidence" approach was used to develop criteria. Criteria were derived for the 'dry season' (May-October) and, given the estuarine nature of the system (~50% tidal), it was divided into two zones for criteria development. Zone 1 is highly influenced by offshore coastal water and nutrient loading from the ocean. Zone 2, in the upper estuary, is influenced by riverine and point source nutrient inputs. Overall, water quality conditions in the estuary were good and support the existing seagrass habitat (one of the goals for establishing criteria). Following the EPA guidance (EPA 2001), criteria were proposed using median values from the existing dataset for dissolved inorganic nitrogen (DIN), phosphate, chlorophyll *a*, and water clarity (Brown *et al*. 2007).

A weight of evidence approach was also used in Pensacola Bay (Hagy *et al*. 2008). The use of historical data to develop a reference condition was evaluated, but for this bay the historical condition was more nutrient enriched than the current state. Nutrient loading to the system had decreased since 1980 and present water quality was considered protective of the desirable uses. Hypoxic conditions appear to be the result of natural processes and a propensity toward low DO in the system and loss of seagrass in the bay were related to pre-1980 degraded water quality. Their goal was to keep water quality at its current levels and to avoid degradation as the regional economy and population continues to grow. As in Oregon, criteria were proposed for Pensacola Bay based on the relative freshwater and seawater influences along the salinity gradients with separate criteria for oligohaline (<5 PSU), mesohaline (5-18 PSU), and polyhaline (>18 PSU). The summer median levels were proposed as criteria for chlorophyll *a*, Secchi depth, DIN, phosphate, TN (<35 µM), and TP (Hagy *et al*. 2008).

In each of these cases, the endpoints were developed and evaluated in context of the system's physical attributes: geomorphology, freshwater inputs, salinity regimes, tidal flushing, and associated residence times. Consideration of these attributes is needed for any endpoint development and is discussed in context for the Western Bays in section 1.2.3. An extensive set of factors influencing susceptibility of waterbodies to eutrophication is presented in the EPA guidance manual (EPA 2001) as developed by the National Research Council (2000). Their list of factors ranges from physiographic setting to nutrient load to residence time/flushing to rates of denitrification. It is an ambitious list of measures for any monitoring program and not one that could be fully applied in the Western Bays, but this project has evaluated a number of them with modeling and field work as discussed in this report and associated appendices.

Established EPA guidelines were used to evaluate the efficacy of endpoints for the Western Bays (EPA 2000). The EPA broke the assessment process down into four steps:

1.0 ***Conceptual Relevance or Soundness***
2.0 ***Feasibility of Implementation (Current and Future)***
3.0 ***Response Variability***
4.0 ***Interpretation and Utility***

These phases describe an idealized progression for endpoint development that is followed for potential endpoints. The relative validity and usefulness of each endpoint in Western Bays is characterized in light of these four steps in order to provide a means for direct comparison across seemingly divergent or at least partially related endpoints.

3.2.2 Regulatory framework for consideration of traditional and alternative endpoints

In addition to examining whether potential endpoints will be effective, sensitive, representative indicators of ecosystem services within a given waterbody segment, the endpoints also need to be evaluated for suitability based on their intended regulatory use. Some endpoints can be agreed upon as absolute measures that can serve as the basis for quantitative permitting of acceptable pollutant loads. Some endpoints can be used to trigger immediate interventions based on exceedance of public health protection standards. Other endpoints can be used to inform status and trends of ecosystems. The endpoints being investigated in this study aim to fulfill the data quality objective of defining whether and where waterbody uses are impaired. This study seeks to identify alternative endpoints which can "…prioritize site-specific assessments needed to confirm the location of both high quality and impaired waters, and support control, restoration, and prevention actions" consistent with EPA's Guidance for 2006 Assessment, Listing and Reporting Requirements Pursuant to Sections 303(d), 305(b) and 314 of the Clean Water Act (EPA 2005).

This report also provides an overall perspective of nitrogen loading and ambient levels as a framework to inform the applicability of endpoints and potential corrective measures. As noted in DEC's method for regulating impaired or threatened waterbodies (excerpt below in Figure 16), "…for some water quality impairments and threats, actions other than TMDL development provide a more appropriate and effective response. Assignment of waters to this subcategory is based on the availability and appropriateness of other strategies that are expected to be more effective in addressing the impairments/threats than TMDLs…" Determination of appropriateness and effectiveness ultimately rests with Western Bays stakeholders. This report seeks to support this determination by identifying the multiple strategies that could be, or in many cases are already, implemented. Timing of corrective measures is vital to the waterbody's recovery from impairment, thus the role of an endpoint in accelerating corrective actions may be an element of its effectiveness. Sensitive, representative endpoints may be useful even if they are not optimally suited for use in a TMDL, such as if endpoints can demonstrate inadequate treatment facilities.

Impaired/Threatened Waters Not Requiring a TMDL

Waters assessed as *Impaired/Threatened Waters* are designated as either requiring a TMDL (IR Category 5) or not requiring a TMDL (IR Category 4). Waters assessed as *Impaired/Threatened Waters* but where TMDL development is not the most appropriate response to the water quality issue are assigned to IR Category 4 and are not included on the Section 303(d) List. (See also *Appendix: Comments on Listing/Delisting Decisions and Delisting Due to Other Required Control Measures*.) These *Impaired/Threatened Waters Not Requiring a TMDL* fall into one of the following three sub-categories.

Impaired/Threatened Waters where a TMDL is Developed and Being Implemented (IR Category 4a)
Once a TMDL has been developed and approved, the waterbody is no longer included on the Section 303(d) List. Progress regarding completion of TMDLs and the delisting of waters where TMDLs are in place will be evaluated with the development of each subsequent 303(d) List.

Impaired/Threatened Waters where Other Controls are More Suitable (IR Category 4b)
This sub-category recognizes that for some water quality impairments and threats, actions other than TMDL development provide a more appropriate and effective response. Assignment of waters to this sub-category is based on the availability and appropriateness of other strategies that are expected to be more effective in addressing impairments/threats than TMDLs. These strategies may include the correction of failing or inadequate treatment facilities, implementation of best management practices (BMPs) to specifically address impairments, zoning restrictions or other local initiatives. Progress and effectiveness of these strategies – relative to the development of a TMDL – will be evaluated during the development of each subsequent 303(d) List.

Figure 16. Excerpt from NYS DEC 2009b consolidated assessment and listing methodology.

Some of the endpoints examined for Western Bays, such as infaunal assemblages, function best as relative indicators which reveal can spatial and or temporal trends against baselines, instead of offering an absolute value above or below which a designated use impairment can be statistically triggered.

3.2.3 **Physical Representativeness of Western Bays**

In order to determine the potential efficacy of candidate endpoints for management and regulation of a waterbody, they need to be evaluated specifically for the ecosystem services associated with the designated uses and comparative baselines being investigated within the applicable waterbody segment. Endpoint evaluations for the Western Bays include consideration of questions such as:

- Is an endpoint measurement sensitive enough to indicate the status of a given ecosystem service, such as fish propagation?
- What waterbody area does an endpoint measurement represent, for instance the entire Western Bays or a distinct segment?
- What endpoint thresholds should trigger designated use impairment interventions, for example can a measurement be compared to an absolute value, to a relative baseline, or a trend?

Each of these questions represented unknowns at the beginning of this study and no specific metrics were agreed upon to indicate impairments. Further, no attempts had been made to address impairments within portions of the Western Bays instead of baywide and there was no agreement about current levels of water quality, sediment quality or ecosystem services, thus there was no ecosystem quality baseline or foundation from which to compare measurements over time.

The Western Bays pose a challenge to evaluating alternative regulatory endpoints potential efficacy due to the combination of the bays' complex geomorphology, their history of anthropogenic changes, and the associated impacts of these changes on ecological processes integral to designated uses.

The possible utility of waterbody segmentation is examined by this study in order to focus on operable units and watersheds, followed by impairment assessments within some segments. It is conceivable that an adjacent segment may have evidence of impairments or threats without having significant load sources originating within its subwatershed. Thus, the optimal regulatory approach and endpoints for a waterbody segment reflects its physical setting, ecological status and applicability of potential corrective measures. NYS DEC's Section 305(b) Assessment Methodology describes the relationship between a waterbody's impairment severity, level of documentation (i.e. certainty), and US EPA Integrated Reporting Categories. For waterbodies with impairments that have known or suspected level of documentation, DEC's approach allows for determination as to whether a TMDL is required will be made on a case-by-case basis (NYS DEC 2008). This report provides multiple lines of ecological evidence that are intended to inform this determination by the respective stakeholders.

This study provides lines of evidence about measuring the status of ecosystem services associated with designated uses, attributing measurements to geographic areas, and, to a limited extent, offers a snapshot in time of baseline conditions as of 2012. This section of the report is dedicated to an understanding of the physical characteristics that are fundamental to the fate and effects of nitrogen and associated metrics throughout the Western Bays. Whether measurements made in this study should be adopted as regulatory baselines is a decision that can only be made by regulators and stakeholders as they decide what quality levels can be expected in the Western Bays or waterbody segments therein.

3.2.3.1 ***Geomorphology of Western Bays***

According to the NYS DEC Assessment Methodology (2009a) estuary waters are defined by physical features and stream classification with less consideration of size. Homogeneity of the waters within a segment is a key consideration. Historically the Western Bays were geophysically very different than today (Figure 17) with respect to the depth regime and mixing with coastal ocean waters. A thorough description of the history of anthropogenic changes to this ecosystem is provided in Appendix D.

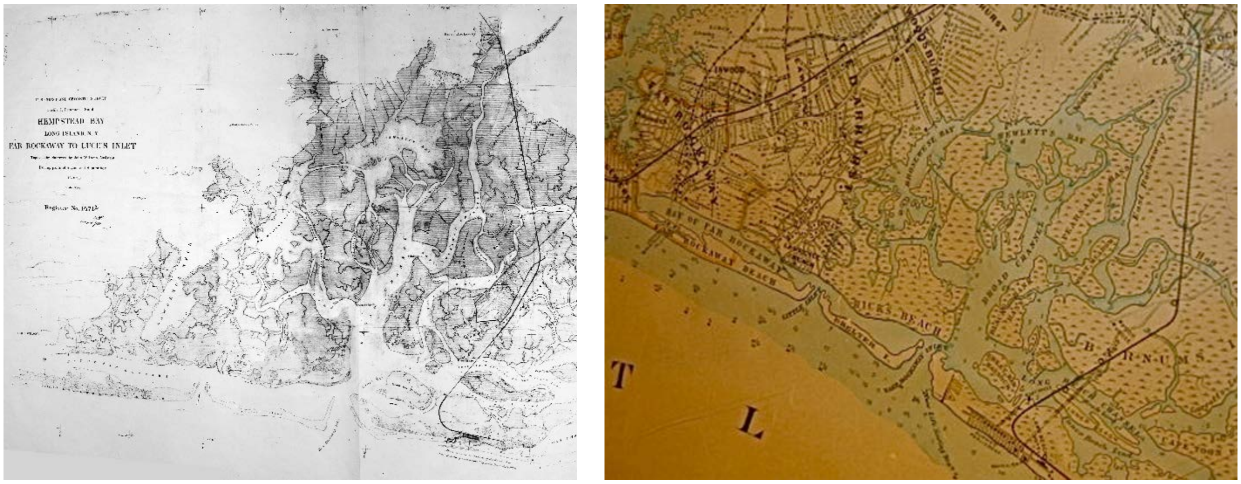

Figure 17. U.S. Coast and Geodetic Survey 1879-1880 topographic map (left) and 1897 Hyde and company map (right) showing shallow TIDAL wetlands of the Western Bays and coastal inlets.

The tidal wetlands, benthos and estuarine habitat functions that existed in the historical, natural configuration of the bays are not necessarily sustainable or recoverable if major perturbations have occurred. Numerous findings from studies in this program confirm that anthropogenic alterations to the geomorphology of the ocean inlets, Hewlett Bay borrow pit and navigation channels create circulatory patterns which differ dramatically from the historical condition (Appendix D). Data from a recent high-resolution bathymetry surface by SoMAS (Figure 18) highlights areas of sharp gradients in bathymetry which influence tidal exchange, advection and residence times. Tidal wetland within the Western Bays degraded measurably in recent decades. Changes in wetland distribution (average annual losses of over 19 acres of intertidal marsh), types (some gain in high marsh over former dredged spoils and more pannes), and fragmentation (worrisome increases) are summarized in Appendix H based on aerial image comparisons between 1974 and 2008.

Regulatory endpoints are intended to influence human behavior, but first the stakeholders who select regulatory endpoints need to decide which anthropogenic perturbations should be treated as part of the background/baseline conditions, and which anthropogenic perturbations should be regulated against monitoring endpoints. Portions of this study were dedicated to describing the physical, i.e. tidal mixing, forces in the bays. This section highlights how anthropogenic changes to the naturally formed bays can impact ecosystem services and the endpoints related to designated uses in the Western Bays.

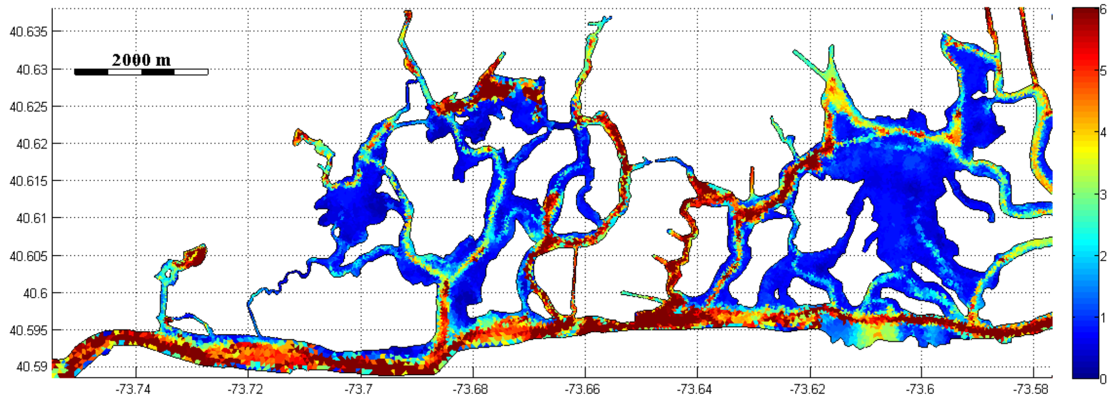

Figure 18. High resolution bathymetry (m) interpolated to refined model grid.

3.2.3.2 *Dredging role in baseline*

Past dredging actions directly impact multiple factors of primary importance to the designated uses of Western Bays waterbodies, including but not limited to: tidal mixing, salinity, temperature, pH, sediment budget of tidal wetlands, bottom sheer stress, sediment grain size, deposition and resuspension patterns, biological recruitment, and sea surface elevation. The impacts of past dredging are considered to be part of the baseline condition from which to evaluate the potential efficacy of alternative regulatory endpoints in the future. Environmental restoration dredging is included as possible nitrogen management plan option later in this study (Section 3.4).

Given the existing geomorphology as baseline, present day circulation offers an abiotic line of evidence to determine whether waterbody areas are homogenous. Identification of such areas can help determine if they can be characterized effectively by a common regulatory endpoint or if waterbody segments need to be characterized separately. Information about the existing baseline conditions also informs potential comparative thresholds.

3.2.3.3 *Salinity role in baseline*

Salinity distribution throughout the Western Bays (Figure 19 and Figure 20) helps to reveal waterbody characteristics. It is not possible to know with certainty whether there is a significant regional difference in total freshwater budget between each of the bays' inflows (surface water inflow, effluents and submarine groundwater discharges). Extensive, spatially resolved measurements of groundwater flow and surface water volumes, beyond the scope of this study, would be required to determine this in greater detail than the existing USGS gauging stations and groundwater modeling used by Monti and Scorca (2003). With regard to bay salinity distributions, this study assumes the freshwater inflows between bays are not different based on the essentially full extent of sewer hookups, homogenous upstream land uses, and development patterns. Thus, salinity distributions shown as model outputs are primarily a function of tidal exchange with the ocean, circulation impacts of dredging, and freshwater inflows.

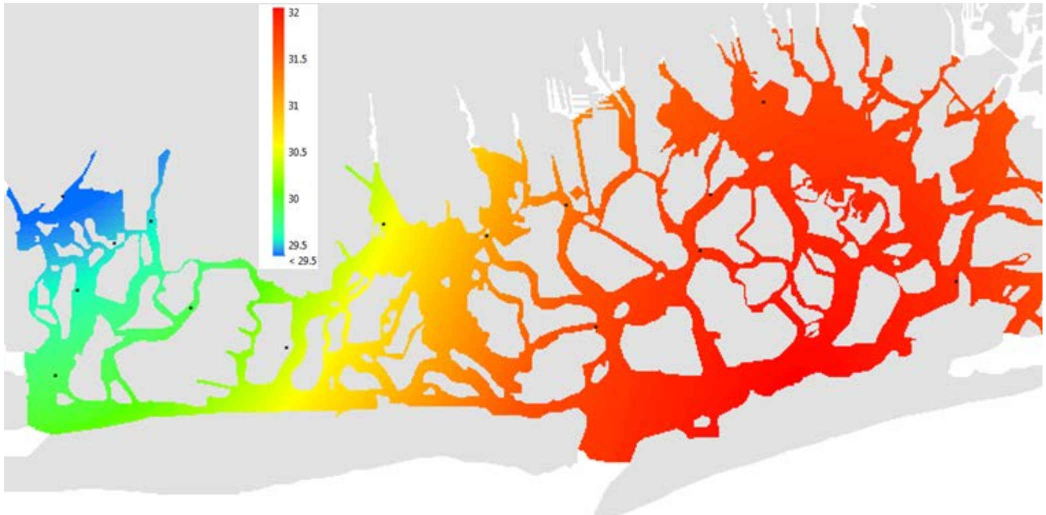

Figure 19. Surface salinity (PSU 29.5-32) measured in Western Bays September 2012.

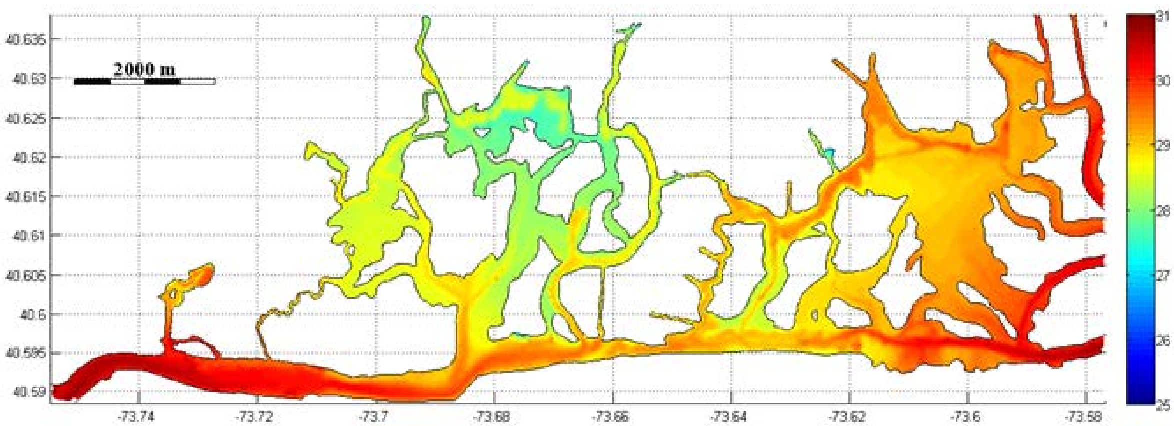

Figure 20. Tidally averaged bottom salinity (PSU) modeled within Hempstead Bay and Middle Bay.

Salinity is one indication of the degree of mixing with ocean waters. In fact, salinity was found to be of primary influence amongst the factors analyzed during the development of nutrient criteria for Great Bay in New Hampshire (Trowbridge 2009). SoMAS' use of hydrodynamic models of circulation (Appendix D) provide helpful metrics which reveal how such abiotic influences define baseline conditions in segments of the Western Bays. The salinity structure shown in Figure 20 is predicted to within 1.5 to 2 psu of observations available from Hempstead Town and further to the east from Suffolk County DHS. The model predicted salinity structure gives us confidence that both our surface river and stream flows and our groundwater inflow estimates have appropriate magnitudes. The salinity map based on sampling was developed using ordinary statistical interpolation with a Gaussian influence function with radius of influence chosen to minimize error variance.

3.2.3.4 *Residence time role in baseline*

Monsen et al. (2002) provide a clear and very readable summary of the hydrodynamic concepts of residence time and age for a waterbody. They emphasize that aquatic scientists often estimate retention time and compare it to time scales of inputs or biogeochemical processes to calculate mass balances or understand dynamics of populations and chemical properties. Boynton et al. (1995) argue that residence time is such an important attribute that it should be the basis for comparative analyses of ecosystem-scale nutrient budgets.

Residence time is the duration that a parcel, starting from a specified location within a waterbody, will remain in the waterbody before exiting. For this study, SoMAS estimated residence time in 3 dimensions using Lagrangian simulations based on the FVCOM output following the method described by Aikman and Lanerolle (2005). The waterbody boundary extended from -73.76^{o} to -73.57^{o}. Also following methods of Aikman and Lanerolle (2005), the simulations initially distributed particles uniformally throughout the waterbody and defined the residence time for a particle as the *first passage time* or the first time a water parcel exits. In this method particles are moving horizontally and vertically, with little difference revealed between residence time patterns for upper and lower parts of the water column throughout most of the domain. The results of these studies identified some discernable increase in the residence time for particles released in the bottom waters in Hewlett Bay. Within Reynolds Channel the residence time for particles released in surface and bottom waters is comparable because of increased vertical tidal mixing in the Channel. Further details about the model simulations and uncertaintes are presented in Appendix E.

The residence time pattern for particles released in the surface layer (Figure 21) emphasizes several important points. Residence in Hewlett Bay, Macy Channel and Mill River is relatively long. This simulation was only 250 hours and it is likely that the residence time is greater than 250 hours for some areas. It is surprisingly long in the central part of Reynolds Channel, presumably because of the tidal and residual current structure discussed above. It is also relatively long in the western part of the Channel. Residence time decreases towards the east, it is low in the eastern part of Middle Bay.

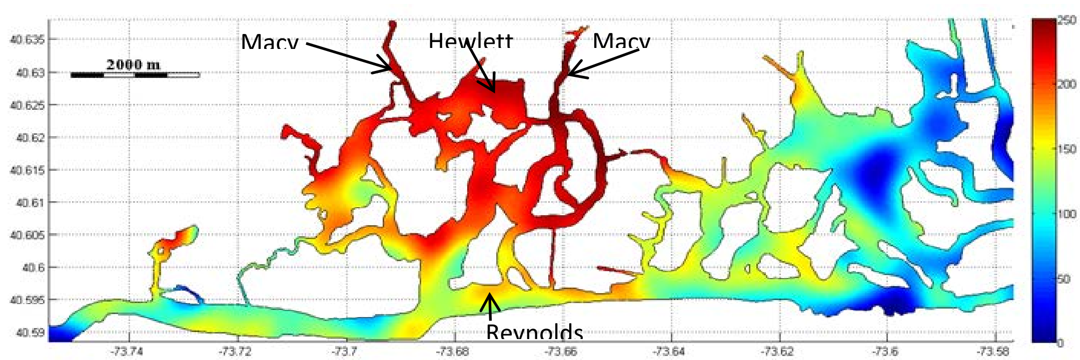

Figure 21. Residence time (hours) distribution in Hempstead Bay and Middle Bay.

The distinct hydrodynamics and salinity regime of Hempstead Bay are not homogenous with other waterbody segments in the Western Bays. The efficacy of potential regulatory endpoints to manage Hempstead Bay may or may not differ from their efficacy elsewhere in the Western Bays. The efficacy of endpoints may differ both because of how these waterbody segments behave physically and due to the possibility that comparative baseline conditions may differ between segments.

SUNY's Great South Bay modeling site is po.msrc.sunysb.edu/GSB/GSB_Model.htm. The model used is FVCOM, and both the hydrodynamic model and the particle tracking model which was used to derive residence time patterns are open source and readily available at fvcom.smast.umassd.edu/FVCOM.

3.2.4 Endpoint evaluation

This section of the report summarizes how well measurements in the Western Bays can fulfill the characteristics of optimal regulatory endpoints. Some of the suitability assessment presented below is based on recent or historical measurements in the Western Bays, as presented in the appendices and section1.1. Other parts of the suitability assessment are based on general knowledge of these measurements in ecosystems similar to the Western Bays. This section discusses the suitability of each candidate endpoint. As mentioned above, US EPA has established a four step process for evaluating potential indicators or endpoints (US EPA 2000). The expectation is that resource managers will seek to use optimal regulatory endpoints that best provide an early indication of the potential for impairment.

1.0 ***Conceptual Relevance***

- *1.1* Is the indicator relevant to Western Bays, assessment questions, and resources at risk?
- *1.2* Can the endpoint be used proactively to identify the impairment or potential for impairment before it cascades through the ecosystem?

2.0 ***Feasibility of Implementation***

- *2.1* Can the endpoints be measured cost effectively?
- *2.2* Are the methods for sampling and measuring the environmental variables technically feasible and appropriate for use in a monitoring program?
- *2.3* Will change in the endpoints occur within timeframes that can be acted upon?

3.0 ***Response Variability***

- *3.1* Do endpoints integrate both anthropogenic and natural factors – can the spatial and temporal variability of each factor be determined?
- *3.2* Are human errors of measurement and natural variability over time and space sufficiently understood and documented?
- *3.3* Is the endpoint sensitive enough for significant changes to be observed given variability & confounding factors?

4.0 ***Interpretation and Utility***

- *4.1* Will the endpoint convey information on resource conditions that is meaningful to environmental decision-makers?
- *4.2* Is the endpoint currently monitored or likely to be easily monitored in the future?

4.3 Are changed in the endpoint attributable to causes that can be acted on?

The endpoints evaluated during this project are listed in Tables 2 & 3 along with descriptions as to how they fulfill each US EPA evaluation step. Additional details for each of these endpoints are provided in the subsequent sub sections. This report focuses on the most appropriate endpoints for the Western Bays as determined based on the recent studies and best professional judgment.

Table 2. Summary of endpoints and general applicability (1 of 2).

Endpoint	**Variables to Measure**	**Type of indicator**	**Relevance** (Representative/Proactive)	**Feasibility** (Cost/Timeframe)	**Variability** (Natural/Sensitivity)	**Utility** (Meaningful/attributable)
Nitrogen	Nitrate, Ammonia, or Total Nitrogen	Primary	Point and non-point source nitrogen inputs are the primary factors in eutrophication. Ammonia standards exist.	Point sources are required to measure nutrients by permits. Most monitoring programs include these relatively inexpensive data	Analytical variability is minimal and known, although ELAP labs not accustomed to marine samples. Inputs are well known. Large natural variability in ambient waters each season.	Direct loading limits can be set & measured against baseline & post action changes. However, ambient levels difficult to interpret due to complex and dynamic cycling processes.
Dissolved oxygen	DO levels, hypoxic or anoxic events	Secondary	Integrator of many water quality processes and directly relevant to marine species. Standards exist, however use attainment questionable due to anthropogenic perturbations.	Easily measured. Among normal suite of measurements. Depth profiles and time series far better than grabs.	Analytical variability is minimal and known. Natural variability can be large. Seasonal, daily & vertical changes in DO minima detectable.	Interpretation of the data is straightforward (TOGS). Ancillary information on physical current structure and bathymetry is very helpful.
Other Water Quality Endpoints	Turbidity, Chlorophyll, pH	Primary & Secondary	Closely tied to changes in nutrient inputs. Impacts SAV. Can impact benthos as a secondary impact (light reduction, pH changes, increased organic load to sediments, etc.)	Easily measured and among normal suite of measurements	Analytical variability is minimal and known. Natural variability can be large, but seasonal changes are expected and can be accounted for. May require long-term monitoring before significant change can be recognized.	Interpretation of the data relatively straightforward. Provides context and bridges gaps between nutrient inputs and secondary endpoints (benthos, HABs, etc.).
Young of Year	Abundance, seasonality	Secondary	Direct indicator for recreational and commercially important species.	Fairly costly sampling to cover seasons, multiple sites, and to decide on reference areas.	Many confounding factors not limited to: stock, predation, pH, DO, NH3, sampling methods, timing, local settings, temperature.	Applicable to fish propagation service, but not directly attributable to nitrogen impacts alone.

Table 3. Summary of endpoints and general applicability (2 of 2).

Endpoint	**Variables to Measure**	**Type of indicator**	**Relevance** (Representative/Proactive)	**Feasibility** (Cost/Timeframe)	**Variability** (Natural/Sensitivity)	**Utility** (Meaningful/attributable)
Ulva	Distribution, density, accumulation, growth rates, isotopes, seasonality, sediment nutrient flux.	Secondary	Linked to nutrient inputs, higher growth rates in nitrogen rich waters, sediment nutrient fluxes and associated infaunal habitat stress..	Distribution can be mapped, though extensive transects repeated seasonally. Growth rates are a highly specialized analysis.	Distribution and growth rates are variable. Limited data available to quantify method variability vs. natural variability (spatial and temporal). Advective accumulation not tied to original growth sites.	Highly visible endpoint, public aesthetic issue, likely causes tertiary impacts such as H_2S offgassing and smothering of benthic habitats. Regional indicator only because geospatial accumulation separate from initial nutrient source exposure and growth.
Benthic infauna	Abundance, taxonomic ID, diversity, benthic indices, Redox depth	Secondary	Integral part of the ecosystem. Repository of much of the organic load and contaminants from anthropogenic inputs. Need to develop bay-specific linkages between stressors and benthic impacts. SPI helps extrapolate geophysical setting in between sentinel infauna assemblage stations.	Very expensive due to need for ongoing monitoring and replication plan. Slow feedback. Could monitor less frequently after critical periods identified. Can provide a clear indication of improvement or degradation.	Highly variable in Western Bays and dependent on sediments aside from nutrient influence. Focused studies can minimize relative temporal and spatial variability across the system, provided that confounding factors are accounted for such as grain size, TOC and pollutants.	Many variations of indices used selectively elsewhere. Primarily relative to baseline or reference sites. The more effort taken in selecting an appropriate index and consistently replicating sampling, the more useful the results will be. Critical in establishing 'baseline' conditions and for managers to assess conditions and mitigate impacts, including anthropogenic actions.
Harmful Algal Blooms	Frequency, duration, magnitude of *Alexandrium* blooms, PSP toxicity, Shellfishing closures	Secondary	Directly relevant to public health. Also can become aesthetic issue. Shellfish closures also a monetary incentive for monitoring these species. False positives unlikely, however false negative possible.	Often part of State monitoring programs or focus of local researcher with experience – otherwise can be very expensive.	Low analytical variability of counts and identifications made by experienced personnel. Natural variability can be large, unpredictable. Past data on shellfish closures & other public health records. Nutrient linkage partial.	Frequency of these blooms has increased – unclear from literature if effect of increase monitoring effort or result of natural or anthropogenic impacts to habitat such as, but not limited to, fish stocks, nutrients, climate (wind, temperature, precipitation).

Of course, all of the endpoints described above are also subject to variability due to major storm events.

3.2.4.1 *Nitrogen (Nitrate, Ammonia, and Total Nitrogen)*

Nitrogen is a macronutrient required by phytoplankton and macroalgae for growth. Excessive levels of nitrogen (and other macronutrients) can lead to eutrophic conditions where primary productivity is very high leading to high levels of biomass, harmful algal blooms, and potentially high rates of respiration/degradation of organic material and low dissolved oxygen. Dissolved inorganic nitrogen (DIN; nitrate+nitrite+ammonium) is the limiting nutrient in most coastal and estuarine waters and as such is the primary cause of eutrophication in the Western Bays. High DIN loading or elevated DIN concentrations can lead to and are indicative of eutrophication, but since it is readily utilized by phytoplankton and macroalgae, ambient concentrations alone are not an adequate endpoint for indication trophic conditions. However, monitoring efforts target DIN as well as total nitrogen (and other nutrients) because they are the primary causal agents of coastal eutrophication and their overall importance to understanding water quality conditions.

The nutrient measurements made under the auspices of this project were analyzed at both an ELAP certified laboratory and at Dr. Gobler's laboratory at SoMAS. The benefit of having samples analyzed by both laboratories was to compare the NYS DEC required ELAP methods and approved laboratory against those from SoMAS, which offered higher data resolution and a longer term data set. A comparison of nitrate (NO_3) data between these two laboratories and the USGS nitrate sensor indicates that there is good correlation across the datasets (Figure 22). Unfortunately, the methods used by the ELAP laboratory for NH_3 had a very high detection limit (MDL=23.5 µM) so those data were not comparable to the SoMAS results nor were they useful for this study. SoMAS utilizes accepted autoanalyzer seawater methods with much lower detection limits. In Figure 23, SoMAS NH_4 data are contoured across the Western Bays highlighting the gradient of decreasing concentrations from high levels (~50 µM) in Hempstead Bay to substantially lower levels (1-3 µM) to the east. The ELAP data would not have allowed these differences to be observed given the MDL of 23.5 µM. In subsequent sections, the SoMAS nutrient data are used to highlight the nutrient gradients across the bays for a number of reasons including their quality, longer time series, and higher spatial resolution.

Endpoint Utility and Recommendations: Nitrogen is a traditional indicator associated with eutrophication. Given the accepted nature of nitrogen as a traditional endpoint or at least an indicator of the potential for eutrophication, it is assumed that it will continue to be measured in the Western Bays as part of regulatory and research based monitoring programs. There are a number of data sources for nitrogen in the bays – Town of Hempstead monitoring efforts (Appendix F), the SoMAS water quality research programs (Appendix A), a USGS nitrate sensor in Hempstead Bay, and the studies conducted as part of this project. The data from these and future efforts will be interpreted to further describe current, baseline conditions and eventually to understand if future remedial actions are effective. There are clear gradients in nitrogen concentrations across the Western Bays with high levels to the west in Hempstead Bay to lower levels to the east in Middle and East Bays. These gradients are coincident with patterns seen for other parameters such as chlorophyll concentrations and Ulva distributions. Reductions in this primary indicator should lead to similar reductions in these secondary measures of impact.

Currently, TMDL efforts often focus on limiting the total amount of nitrogen loading into coastal ecosystems, even if the absolute amount of nitrogen in each source, sink and cycling process cannot be known fully at any given time across the entire ecosystem. US EPA is focused on the development of total nitrogen (TN) based nutrient criteria, including guidance on how such criteria can become specific to targeted ecosystems. State standards are not currently available for DIN or TN, but there are ammonia (NH_3) standards based on its toxicity. The continued measurement of DIN and TN in Western Bays cannot be overstated, along with concomitant biological indicators of designated uses, such as benthic infaunal habitat quality. The data are critical to both understanding current conditions and for establishing baseline or benchmarks against which to evaluate improvements to water quality due to mitigation efforts. The development of specific numeric thresholds for Western Bays designated use impairments due to nitrogen requires more data and sampling periods than were available within this project. Nitrogen data and patterns detected within this project can inform EPA's TN based criteria nationally, as well as Western Bays stakeholders'' follow through regionally.

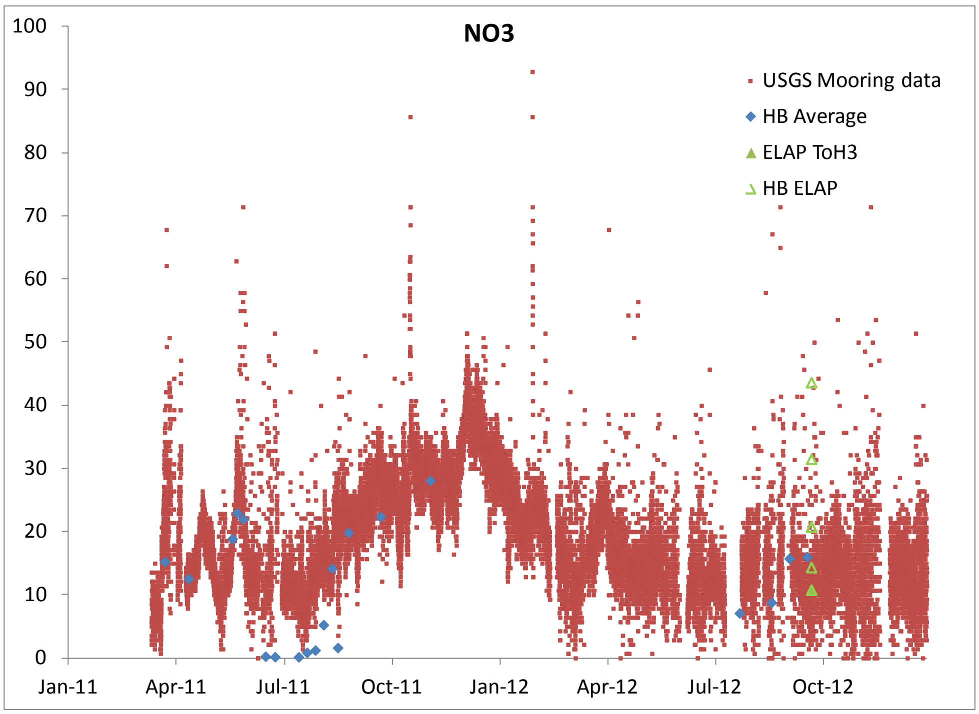

Figure 22. Nitrate concentrations (μM) for USGS mooring, SoMAS Hewlett Bay average, ELAP HB stations, and ELAP station ToH3 near mooring.

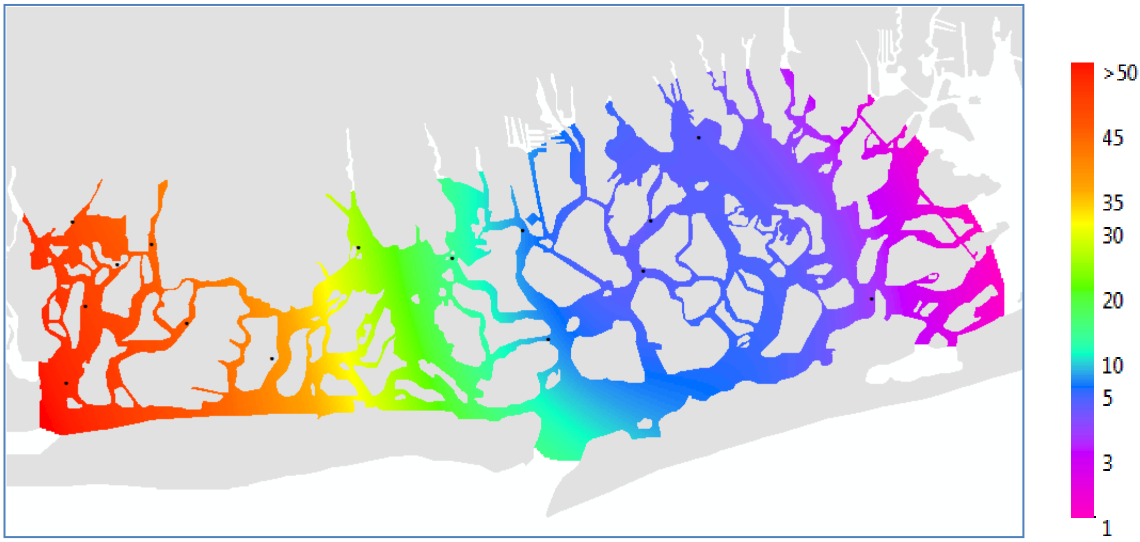

Figure 23. Dissolved ammonium concentrations (µM) across Western Bays from during late September 2012.

3.2.4.2 ***Dissolved Oxygen***

Dissolved oxygen (DO) is a primary indicator and integrator of water quality in coastal waters. As a basic necessity for aquatic life, DO levels directly affect ecosystem health. Low DO (anoxic or hypoxic) environments have existed historically, but their occurrence in shallow coastal and estuarine areas appears to be increasing and the cause seems most likely to be accelerated by human activities (Nixon, 1995). Given the relative ease, accuracy, and low cost of measuring DO (both in situ with sensors or in the laboratory with Winkler titrations), it is a mainstay of coastal water quality monitoring programs. Although most programs are moving towards in situ measurements, they are still calibrated against Winkler titrations and thus the recent measurements are directly comparable to the long historic record of DO data.

DO concentrations are directly related to physical conditions. Cooler waters have higher DO levels than warmer waters. More saline waters have less DO (at same temperature) than freshwaters. Thus, there is a strong seasonal signal in DO concentrations that is directly tied to seasonality of temperature and meteorology in temperate systems – highest in winter and lowest in summer. This cycle, however is further impacted by biological processes. It is the biological production and utilization of DO that is impacted by excessive nutrient loading to a system. Primary production by phytoplankton and macro algae increases with increasing nutrient availability (tied to nitrogen in the marine environment). To varying degrees, the higher the nitrogen load the higher the production of organic material and DO levels. In fact the surface waters of highly eutrophied bays often reach super-saturated levels of DO during the day, before crashing to low DO levels at night due to respiration. This is further exacerbated under stratified conditions when the organic material sinks to the deeper, bottom waters. Stratification serves to cut off communication between the bottom and surface waters and microbial degradation of the organic material can reduce DO to hypoxic or anoxic levels.

Endpoint Utility and Recommendations: DO concentrations in coastal waters like the Western Bays are dynamic on a variety of temporal levels from seasonal to diurnal and spatially over the bays due to proximity to freshwater sources or tidal mixing and changed bathymetry. There are clearly understood relationships between anthropogenic influences and biotic processes and ambient DO levels. So not only is DO inexpensive to measure accurately using highly resolved (spatially and temporally) methods, but it is also well understood as an integrator (daily to seasonally) of impact of nutrient loading to a system. Thus, DO is one of the most ubiquitous indicators used in water quality monitoring programs. Additionally, DO standards are in place for most State waters and thus DO is not only used as an indicator of ecosystem health in the Western Bays, but also as a regulatory standard. This study demonstrated DO measurements in Hewlett Bay are an effective metric of non-compliance, however, that non-compliance is not representative of conditions outside Hewlett Bay.

The interpretation of DO levels as a metric of the ecosystem status requires that measurements be interpreted within homogenous segments. Hydrodynamic modeling simulations and multiple water quality sampling surveys have demonstrated that the Hewlett Bay region of Hempstead Bay behaves as a distinctly different waterbody than all other regions of the Western Bays. Thus, DO measurements there are not indicative of the status of the ecosystem elsewhere. The low DO events observed in Hewlett Bay bottom (Appendix D, figure 17) trigger impairment determinations based on the never less than acute standard of 3.0 mg/L and on the basis of duration as per NYS DEC TOGS 1.1.6. However, further consideration may be warranted for other segments given the resilience seen in recruitment of some benthic infauna, and given the highly advective nature of some segments of the Western Bays. A couple of the issues with using DO as an endpoint include that it truly is an endpoint, as once those levels are reached there are clear impacts to the ecosystem – thus not very proactive or an early warning type of endpoint. The other issue is that since it is such an integrator of water quality it is not always that easy to ascribe the direct cause of low DO conditions – case in point in Hempstead Bays is it due to high TOC levels in depositional areas leading to DO consumption, or is it tied more directly to Ulva distributions and its ultimate decomposition. Regardless, DO levels are an easy, inexpensive, and critical component of ecosystem function and will remain an important endpoint for any monitoring program.

Some shallow segments of Western Bays, observed to have compliant DO levels, exhibited stressed infaunal communities. This suggests that although DO levels can be an effective indicator of non-compliance in deeper waterbodies, DO levels alone should not be relied upon to determine attainment of Western Bays designated uses.

3.2.4.3 *Water Quality (Clarity, Chl a, and pH)*

Water clarity and chlorophyll concentrations are primary response endpoints for eutrophication. As nutrient loads increase, phytoplankton production increases resulting in high concentrations of chlorophyll and decreasing the clarity of the water column. Both of these measures are made accurately and easily using in situ instruments and beyond the initial cost of the sensors, they are also relatively inexpensive to measure. Water clarity has historically been made using Secchi discs, but over the last few decades the use of turbidity meters or transmissometers have replaced the manual deployment of the Secchi disc. Laboratory analyses of chlorophyll continue to be made using fluorometers or spectrophotometers, but the advent of reliable and less expensive in situ fluorometers has made this another measurement that can be made over highly resolved spatial and temporal periods. As such, both of these measurements have become mainstays for most coastal monitoring programs.

During this project, historical data were examined and chlorophyll measurements were made during the water column/Ulva survey. The results (see Appendix A and Figure 25) suggest that chlorophyll and nitrate concentrations are well correlated (at least for the summer of 2012) and that ambient chlorophyll levels should be considered as a potential endpoint for the bay. However, given project constraints, we focused on the macroalgae issues rather than microalgae given their ecological importance and public's focus on the Ulva blooms/detritus. Hempstead Bay is on the 303(d) list as being impaired for excessive algal growth due to nutrients, but it is due to excessive growth of macroalgae not microalgae. Chlorophyll has been used as an endpoint in many other studies, but typically microalgae are the dominant primary producers in these systems so a more direct linkage can be made between nutrient loading and chlorophyll concentrations (which is really used as a proxy for primary production and biomass). For this work, we focused on Ulva – distribution, growth rates, and δ15N – for the important information they could provide and to explore as potential endpoints (see Section 3.2.4.5).

As discussed above, excessive nutrient loading into coastal ecosystems promotes algal productivity and the subsequent microbial consumption of this organic matter lowers oxygen levels and contributes toward hypoxia. A second, often overlooked consequence of microbial degradation of organic matter is the production of CO_2 and reduction in pH associated with that process. The overall acidification of the ocean has become a major focus of climate change research as the increasing CO_2 concentrations in the atmosphere have been somewhat mitigated by CO_2 being transferred into the oceans and decreasing pH levels. In coastal waters, this is further exacerbated under eutrophic conditions and has been linked to elevated mortality in larval finfish and shellfish (Talmage and Gobler 2010; Baumann et al. 2012) suggesting that acidification, which has been intensified by climate change (Doney et al. 2009), may be currently altering the ability of Hewlett Bay to support robust fisheries. In the coming years, pH measurements will become more and more prevalent in coastal monitoring programs due this issue and the ease and inexpensive cost of the in situ pH measurements warrants its use as a secondary indicator.

Endpoint Utility and Recommendations: As with total nitrogen, US EPA has included water clarity and chlorophyll as parameters in their nutrient criteria development program. Currently there are no regulatory levels associated with these parameters, but in the coming years US EPA plans to have criteria in place for estuarine waters (US EPA 2001). Both of these parameters, as well as pH, are relatively easy to measure and are typically part of most water quality monitoring programs. The expectation is that these parameters will continue to be measured and that they could be considered as potential endpoints, though as noted above, the current study focused more on Ulva than microalgae as measured as chlorophyll. There are many other reasons why chlorophyll was not more fully evaluated as an endpoint during this study – the most important of which is the knowledge that high chlorophyll levels are episodic in nature, closely associated with phytoplankton blooms. These biological events are certainly fueled by excess nutrients, but they are highly variable in severity and duration both seasonally and spatially. Like many other potential endpoints this one would necessitate many years of study to establish the typical ranges of values (rather than a couple of surveys during a single year) against which to compare these high chlorophyll events and or their geographic extent. In Chesapeake Bay for instance, which has been closely studied for decades, there are wide ranges of chlorophyll concentrations over different trophic and salinity regimes, as well as bathymetric zones (see Table V-8 in US EPA 2003). Even with decades of data, within different salinity regimes, the final chlorophyll criteria for Chesapeake Bay ended up being qualitative "*concentrations of chlorophyll a in free-floating microscopic aquatic plants (algae) shall not exceed levels that result in ecologically undesirable consequences—such as reduced water clarity, low dissolved oxygen, food supply imbalances, proliferation of species deemed potentially harmful to aquatic life or humans or aesthetically objectionable conditions—or otherwise render tidal waters unsuitable for designated uses*." To pursue chlorophyll's role impairing Western Bays designated uses may entail deriving water clarity goals for waterbody segments, i.e. bathymetric regions, which were known or expected to be viable habitats for sea grasses. This line of evidence was not investigated in the QAPPs and SAP agreed to.

Additionally, chlorophyll, unlike ammonia or some contaminants, does not have a recognized threshold value above or below which it is inherently deadly, harmful, toxic, or pathogenic to the ecosystem apart from the sea grass shading above. Certainly excess chlorophyll can stress ecosystems when excess phytoplankton settle out of the water column and create sediment oxygen demand above rates that can be supported by the water column dilution capacity. This stress, however, is better monitored through integrative endpoints of sediment and bottom water habitat suitability discussed elsewhere in this report, such as infaunal community statistics and bottom water DO, pH and NH_4 levels. Chlorophyll can serve as an effective indicator of water column growth, which can help reveal spatial and temporal patterns of growth in the Western Bays in response to nitrogen, but it is less suitable to serve as a regulatory trigger for designated use impairments. A long term record of chlorophyll levels in Western Bays may be able to be correlated with ecosystem impairments quantified through other endpoints. However, given the rapid cycling of nitrogen through ecosystem components, it would likely take many years to determine, even as a secondary endpoint, which blooms occur at sufficient levels, for sufficient duration, and during periods which are critically sensitive. Chlorophyll continues to be a useful measurement to make and clearly helps in our ecological understanding of the ecosystem, but it may not be as useful as a regulatory endpoint or water quality criterion.

3.2.4.4 *Young of Year*

Systematic sampling for young-of-year winter flounder should be considered as an endpoint line of evidence indicative of the waterbody's ability to support fish propagation. Variability in gear efficacy and seasonal differences associated with biological sampling dictate that such sampling plans rigorously apply DQO rules if intended for use in management decisions. While the presence or absence of young-of-year winter flounder can be a useful indicator of impairment, it likely reflects a combination of numerous stressors and ecological factors, thus should be examined in concert with other lines of evidence when determining pertinence of nitrogen loading to population data. For the Western Bays, young of year catches should be compared between bays to ensure there is a viable baseline population in the vicinity.

Endpoint Utility and Recommendations: This was not a focus for this study, or any of the QAPPs or SAP. Ancillary information became available and, as noted above, it should be evaluated further as a potential endpoint for the Western Bays system.

3.2.4.5 *Ulva (percent coverage, growth rates)*

Increased nutrient loading not only impacts phytoplankton growth, but also leads to increases in the growth of macroalgae. 6NYCRR 703.2 sets a narrative standard nitrogen as “None in amounts that will result in growths of algae, weeds and slimes that will impair the waters for their best usages.” In the Western Bays, dense accumulations of the macroalgae *Ulva* has been a persistent water quality issue. Unlike some endpoints, the presence of dense Ulva mats is a nuisance both aesthetically and potentially due to its impact on benthic habitats and bottom water DO levels when it dies and sinks to the bottom. For this study, the distribution, density and growth rates of Ulva were examined to both better understand the system and to assess the sensitivity of these potential endpoints. The characterization of Ulva density and percent cover are relatively straightforward and time-consuming, but not technically difficult. Measurements of Ulva growth rates, however, require highly specialized methods and trained researchers. If part of a research program, these measurements will be very useful, especially to compare spatial extents between time periods. Frequent monitoring of growth rate endpoints would be too expensive.

Endpoint Utility and Recommendations: Mapping the spatial and temporal distribution of a macroalgae species such as *Ulva*, however, would be a useful alternative endpoint for tracking ecosystem health and perhaps annual changes in spatial and temporal extent of accumulation across these shallow estuarine environments. This study did not derive absolute thresholds for *Ulva (*density or percent cover and duration) above which designated uses become impaired. Absolute values for air quality criteria which may be impaired by *Ulva* decay already exist. The *Ulva* metrics appear suitable to determine water quality trends over time and space. However, regulation and mitigation against *Ulva* accumulation would have to be normalized to many factors in addition to nitrogen levels, such as light, temperature, wind, and tidal advection.

Three lines of evidence supported the hypothesis that that excess nitrogen contributes to the excessive growth of Ulva. First, we performed multiple experiments that indicated that Ulva growth rates significantly increased with the addition of nitrogen. Next, we performed multiple experiments that indicated that Ulva growth rates significantly decreased when nitrogen was diluted from seawater. Finally, the distribution of Ulva in the bays spatially matched that of nitrate and ammonium. Other lines of evidence highlighted the complex and dynamic nature of ambient nutrient levels. Coupling these experiments with reconnaissance sampling results is not an advisable method to derive absolute values for ambient nitrogen concentrations useful to regulate Ulva growth. As noted in table 3, successive years of sampling Ulva, nutrient levels and infauna may enable an effects-based means to derive regulatory thresholds if all other normalizing factors and baseline assumptions can be agreed upon.

Ulva is a significant ecological problem in the Western Bays that also garners substantial public and thus political attention. Although these are not listed as attributes for assessing the validity of endpoints, they will certainly help the process of developing nutrient management options and funding an adequate monitoring program. This combines with the fact that Ulva is so important ecologically to the system and has been shown in the past and during this study to directly respond to nitrogen enrichment leads to the recommendation that it would be a useful alternative endpoint for the Western Bays.

3.2.4.6 *Benthic Infauna*

Although not expressly represented as any of the endpoints in Figure 15, the status of the benthic habitat is one of the ultimate impacts of eutrophication. Benthic infaunal lines of evidence proved very helpful, albeit relatively expensive and protracted, to understand the geographic status of designated uses in the Western Bays. Excessive nutrients lead to high rates of production of both phytoplankton and macroalgae adding to the organic load to the benthos as these organisms die and sink to the bottom. The sources of nutrients are often also sources of both natural and anthropogenic terrestrial organic material. This is especially true in the Western Bays where total organic carbon accumulation varies geographically (Figure 24) either from local loads, such as surface water runoff, or from deposition after water column blooms that occur locally or occur remotely and are transported into depositional areas through tidal circulation. Use of infaunal statistics in Western Bays should consider normalization to sediment TOC.

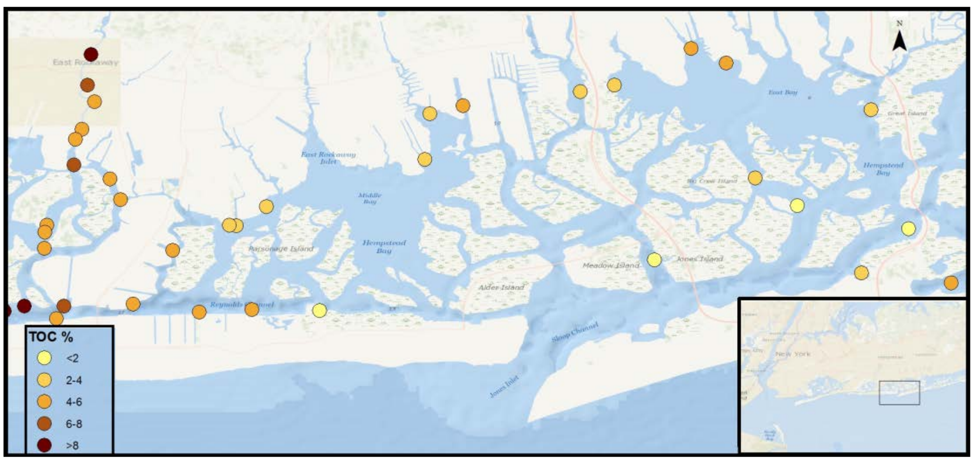

Figure 24. Total Organic Carbon (percent) as reported by SoMAS (Brownawell 2013)

Thus the benthos is subject to a one-two punch of multiple organic loadings in many eutrophic systems. There are a wide variety of metrics available to assess benthic habitat health and impacts. Direct analysis of potential contaminants is one approach that was not explored by this study, primarily due to the associated costs of analysis and the lack of any clear historical data indicating that it is a major issue in the Western Bays. However, if levels of organic and metal contaminants were to be used as endpoints there are clear EPA and State standards against which they could be assessed. For this project the benthos was examined via sediment profile imaging (SPI) (Appendix C), benthic infaunal identifications (Appendix B), and physical sediment characterization (grain size).

The physical characterization of the sediment is useful for normalizing the other measures and becomes even more useful if TOC content is also measured (TOC > 5% is considered organically rich and possibly indicative of anthropogenic impact). Grain size characterization helped separate infaunal station comparisons due to the varied geomorphology both across the Western Bays and even across proximate channel segments with varied hydrodynamics. SPI provides actual images of the sediments from which one can assess the relative condition of the biological communities within the sediments and measure the depth of the redox zone of surface sediments (Appendix C) – both measures can be used as metrics for assessing the status of the benthos. Infaunal statistics and SPI imagery are best used in concert, instead of one or the other, due to their metrics' representativeness and spatial coverage. Identification and enumeration of the benthic infauna provides a direct assessment of benthic community ranging from a diverse, abundant thriving community to a depauperate or low diversity, impacted community. Measurements of grain size, TOC, and SPI are relatively straightforward, but do require specialized equipment and training though the costs of these measurements are much less than those associated with collecting, counting and identifying benthic infauna, which can be daunting.

For this study, the infaunal data were used to calculate total abundance and total number of species at each station, which in turn were used to calculate standard ecological indices such as species diversity (H'), richness, evenness (J'), or dominance. The abundance metrics are agglomerative and often difficult to interpret because they consider only the total numbers of species or individuals, ignoring the identities of those organisms present. The index metrics also ignore the identities of those organisms present. In doing so, the metrics often suggest that two areas may be the same or very similar when in reality they encompass very different sets of organisms. Many ecological indices, such as Shannon Diversity (H'), may have inherent assumptions that are not met in a benthic sampling program. To rely on these metrics for regulatory decision making will require DQOs that define sampling replication requirements to account for variability due to seasonality, geophysics, water quality conditions, and laboratory methods. Also, ecological indices, and the suite of environmental indices that have followed them, are very difficult to interpret. For example, it is generally accepted that higher species diversity is "better" than lower diversity, but establishing a threshold separating good from less than good is not readily practicable.

Additionally, the infaunal communities in the Western Bays will be at different stages of development based on proximity to stressor sources, water depth, and seasonal shifts that can vary annually; these different stages cannot be detected by the usual ecological metrics or single annual snapshots. Even with multiple replicate surveys it can take years of consistently improving nutrients levels with comparable wind and rain and heat patterns for these agglomerative metrics and indices to provide quantitative discrimination to correlate infaunal community condition with water column nutrient loads and levels.

Endpoint Utility and Recommendations: Benthic infaunal analyses and indices, as well as sediment flux measurements, provide important information on benthic impacts and resilience relative to overall water quality conditions. The sediments are the ultimate repository of whatever is discharged into the system and as such provides relevant, sensitive, responsive, measurable long term integration of water quality and associated pelagic responses (e.g. phytoplankton blooms). It is this long term integrating quality that makes the benthic parameters both very important to measure and understand as well as difficult to ascertain and attribute to specific short-term changes in stressors. For the Western Bays, it would likely take more than three years of multiseason sampling with triplicates at three or more stations in each of the three embayments to create a baseline data set which could begin to relate infaunal assemblage patterns into waterbody-wide nitrogen concentrations below which regulators could declare the benthic habitat to be not impaired. This is difficult to derive because the infaunal communities respond throughout each season to many complex environmental factors, such as sediment texture, organic content, the presence of pollutants, freshwater inflows and salinity. Although informative and an area of active research and monitoring, benthic infaunal endpoints are both expensive and difficult to interpret and are recommended with budgetary cautions as potentially suitable endpoints for the Western Bays.

3.2.4.7 *Harmful Algal Blooms (Alexandrium, Dinophysis*, **others?*)***

Harmful algal blooms (HAB) and other nuisance blooms are the focus of researchers across the globe. Apparent increases in the frequency and magnitude of these blooms have been noted. It has been speculated that for open ocean blooms this may simply be a matter of more focused attention by researchers, but for near shore, coastal and estuarine blooms it appears that localized anthropogenic inputs of nutrients play a role in HAB development and occurrence (Anderson et al. 2008). Note that this continues to be an area of research and the relationship between HABs and eutrophic estuaries and coastal waters continues to be examined – specifically relationships associated with nutrient flux or loads, relative ration of nutrients, and the overall complexity of these systems, their nutrient loads and the response of HABs.

The importance of understanding these blooms and their associated toxicity continues to be stressed. As mentioned previously, in 2010 harmful species of the dinoflagellates *Alexandrium* and *Dinophysis* were observed in Hewlett Bay and Middle Bay (Figure 4). During the spring and late summer 2012, SOMAS observed blooms of *Heterosigma akashiwo*, which is a species known to be ichthyotoxic and has been associated with finfish kills worldwide (Lewitus et al. 2012). In addition to these species, which are known to cause paralytic and diarrhetic shellfish poisoning, respectively, it is likely that other HAB species, such as species of the pennate diatom Pseudonitzschia that produce domoic acid, could be a potential issue in the Western Bays.

Endpoint Utility and Recommendations: Sampling, analysis, and identification of phytoplankton species in general and HAB species in particular, can be expensive and require trained specialists. Typical taxonomic identification is laborious and between costs and time associated with the analyses, it is difficult to make these measurements in a highly resolved manner. Nonetheless, their measurement and the analysis of the toxins have become a necessity for States in order to protect consumers and guard the health of economically important fisheries. Technological advances, such as the use of fluorescent probes to identify *A. fundyense* (Anderson et al. 2005), are rapidly improving the ability to monitor these blooms, while decreasing the costs and speeding up the delivery of results. In the future, these new methods may allow HAB monitoring to become routine and thus increase its suitability to be a standard measurement and more possibly a useful endpoint. Ultimately, however, its use as an endpoint may rely more on researchers discovering a direct link between HABs and eutrophication than on easy presence/absence analysis. If this link cannot be made it has little use as a proactive endpoint. To consider it as a regulatory endpoint driving nutrient criteria, microalgal growth would need to be further investigated and could be the topic of a future study as it wasn't the sole focus of the current study.

3.3 Are Impairments Linked to Excess Nutrients?

3.3.1 Problematic phytoplankton blooms follow ambient nitrogen

The Western Bays experienced extremely dense phytoplankton blooms during the summer of 2012, with peak levels exceeding 50 µg chlorophyll *a* L^{-1} concomitant with elevated nitrogen levels (Figure 25). The temporal dynamics of phytoplankton biomass was similar across the entire Western Bays region, although intensity of these blooms was greatest in Hewlett and Middle Bays (Appendix A). Mean chlorophyll *a* concentrations in Hewlett Bay at station BP were >35 µg L^{-1} which exceeds levels found in almost all of New York State's South Shore Estuary Reserve (SCDHS 1976-2012). The sites with the second highest mean concentrations of chlorophyll *a* were Middle Bay and the USGS site at Hog Island (~20 µg L^{-1}) followed by East Bay and Jones Inlet (JBI) which generally had low levels of chlorophyll (<10 µg L^{-1}). These high density blooms in Hewlett Bay and to a lesser extent in Middle Bay and the USGS station occurred in the late spring and were maintained throughout the summer months, eventually decreasing in late September.

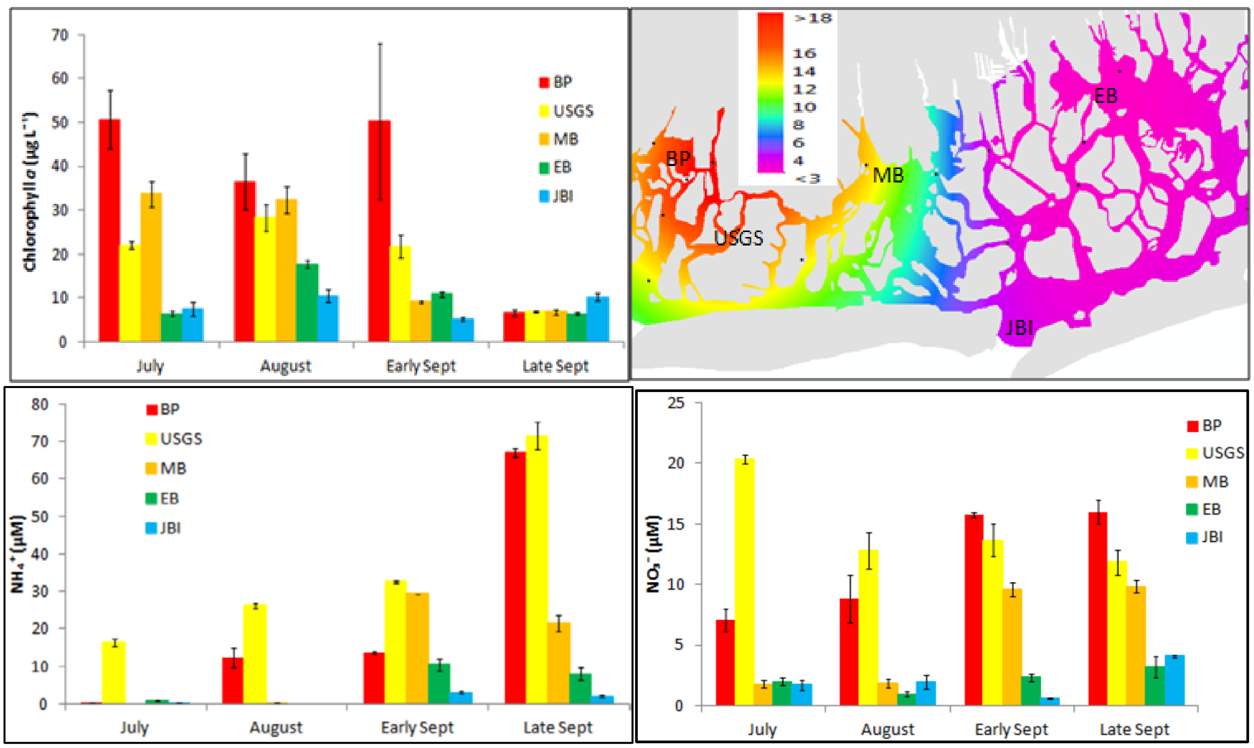

Figure 25. Chlorophyll a (upper left, ug/L); Dissolved Nitrate (upper right, uM); Dissolved Ammonium (lower left, uM);Dissolved Nitrate (lower right, uM) in Western Bays summer 2012.

Phytoplankton comprising these blooms typically included diatoms, dinoflagellates, autotrophic nanoflagellates, and raphidophytes (Figure 26). Phytoplankton blooms in Hewlett Bay during the spring and late summer 2012 were dominated by the harmful raphidophyte, *Heterosigma akashiwo*, with cell densities reaching extremely high levels (ranged from > $7x10^3$ to over 2.5 x 10^4 cells mL^{-1}). This species is known to be ichthyotoxic and has been associated with finfish kills worldwide (Lewitus et al. 2012). During this and prior studies, dense phytoplankton blooms occurred within Hewlett Bay and Middle Bay with chlorophyll values frequently exceeding 20 µg L^{-1}, a level USEPA considers eutrophic. As noted above, sometimes these blooms are comprised of harmful algae. The decay of these blooms contributes to hypoxia and acidification (Wallace et al 2014). As such, reductions in DIN that lead to reductions in phytoplankton and chlorophyll would limit the intensity of these impairments.

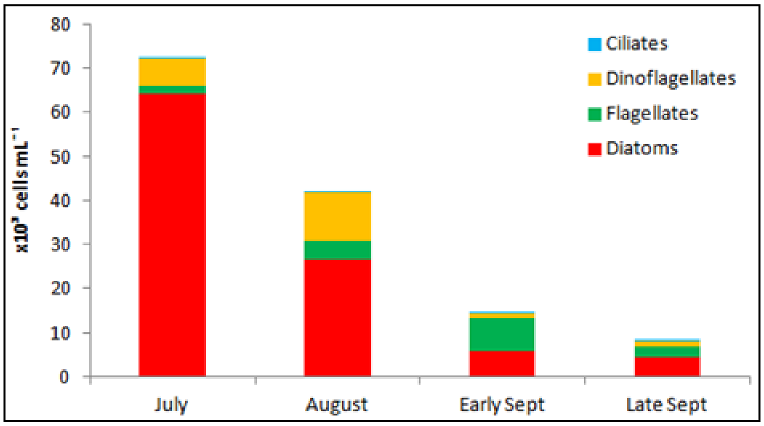

Figure 26. Plankton community composition (x10³ cells mL⁻¹) at station BP in Hewlett Bay in 2012.

3.3.2 **Algal growth decreases with reductions in dissolved inorganic nitrogen**

The algal growth kinetics experiments performed by SoMAS provide the best evidence available about how reducing nitrogen levels could reduce growth rates of stressful phytoplankton blooms. Phytoplankton blooms can result in depletion of bottom water dissolved oxygen after cells die and the organic detritus settles to the bottom in depositional areas. The blooms also reduce light penetration to the bay bottom which is vital to sea grasses which are no longer reported in benthic habitats of the Western Bays.

The nutrient dilution experiments, conducted by SoMAS and detailed in Appendix A, demonstrated the very strong limiting effect that dissolved inorganic nitrogen concentrations had on phytoplankton growth rates in Hewlett Bay and in Middle Bay (Figure 28). Specifically, during all experiments between May and September, decreasing DIN led to a linear reduction in phytoplankton growth rates.

In addition to the experimental evidence of phytoplankton growth rate changes with DIN concentrations, the water quality sampling results detected nutrient levels that can be related to phytoplankton growth. For example, occasionally during Summer months the ambient DIN levels in the Western Bays drop below the half-saturation constant of many phytoplankton species. Figure 27 shows seasonal ambient DIN:DIP and DIN:DSi ratios both indicative of nitrogen deficiency during summer months. These conditions were most clear in the eastern half of the system, but were also present in the western half.

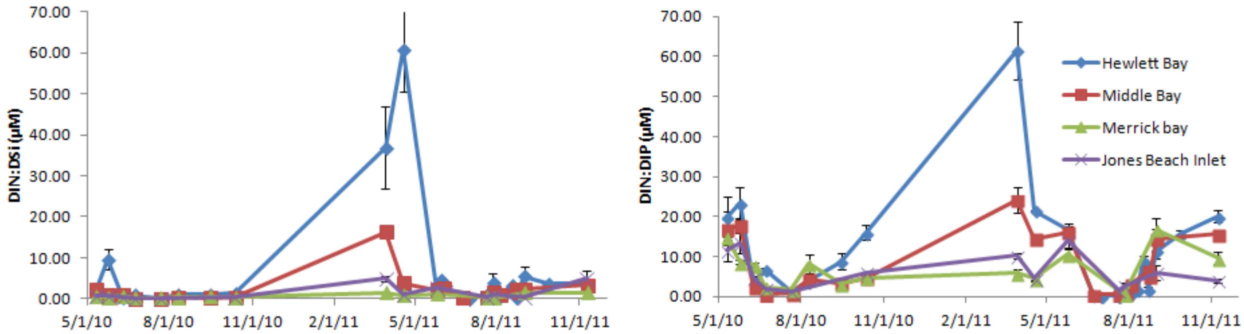

Figure 27. Nutrient ratios indicating Summer defecits in DIN relative to Si and P.

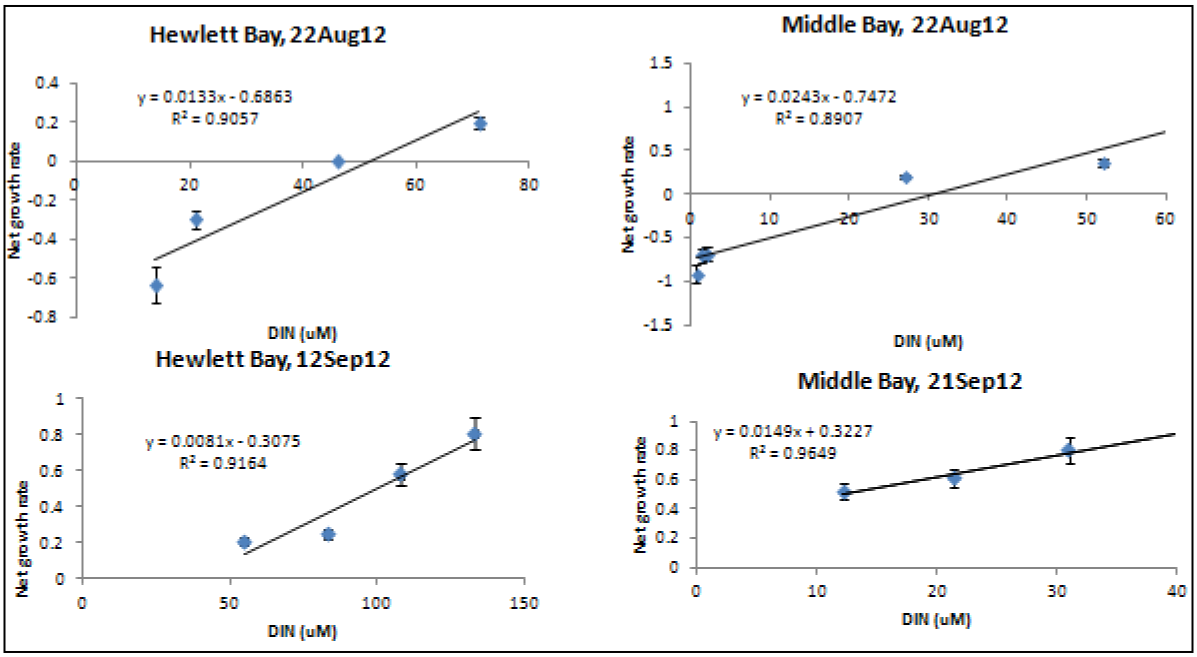

Figure 28. The relationship between dissolved inorganic nitrogen (DIN) and net growth rates of phytoplankton population per day in Hewlett and Middle Bay. Differing levels of DIN were achieved by enriching Hewlett or Middle Bay water with nitrate or mixing West Bay water with Atlantic Ocean water. Growth rates within dilutions were corrected for reduced zooplankton grazing rates.

Through the study, phytoplankton population net growth rates were reduced by 0.01 to 0.03 per day for every one micromolar reduction in concentrations of DIN (Table 4). The mean net growth rates of phytoplankton populations in Hewlett Bay and in Middle Bay over a three year period were found to be 0.073 and 0.14, respectively. Positive net growth rates result in large algal blooms. As such, a reduction in mean DIN concentrations in Hewlett Bay and Middle bay of 4 and 7 µM, respectively, may reduce the mean summer net growth rates to zero and would ostensibly lessen the intensity and impact of algal blooms in this ecosystem. Given the very large variability in the growth rates of phytoplankton (-0.7 to 1.2 per day), these small reductions would lower the intensity, but larger reductions would be required to more fully prevent algal blooms in this system.

Table 4. Net daily growth rates of phytoplankton communities in Hewlett Bay and Middle Bay, 2010-2012, as well as the calculated decrease in net growth rate achieved by a 1 µM decrease in DIN. Decreasing the mean net growth rates of phytoplakton in Middle Bay and Hewlett Bay to zero would require DIN reductions of 7 and 4 µM, respectively.

Hewlett Bay

date	Net growth rate	stdev
5/10/2010	0.485	0.036
5/24/2010	0.065	0.045
6/9/2010	-0.173	0.021
6/22/2010	-0.281	0.185
7/21/2010	-0.894	0.040
8/9/2010	0.369	0.015
9/14/2010	0.866	0.026
10/12/2010	0.307	0.133
3/29/2011	0.177	0.022
4/18/2011	0.254	0.029
5/27/2011	0.317	0.017
6/22/2011	-0.205	0.076
6/30/2011	0.118	0.144
7/27/2011	-0.710	0.100
8/31/2011	1.134	0.006
9/29/2011	0.192	0.054
7/28/2012	0.110	0.010
8/22/2012	-0.288	0.178
9/6/2012	-0.629	0.071
9/21/2012	0.254	0.054
Average	0.073	0.495

Middle Bay

date	Net growth rate	stdev
7/21/2010	-0.76	0.04
8/9/2010	0.04	0.15
9/14/2010	1.10	0.03
10/12/2010	-0.01	0.13
3/29/2011	0.56	0.01
5/27/2011	1.16	0.03
7/20/2011	-0.74	0.14
7/27/2011	-0.20	0.20
8/31/2011	0.22	0.02
8/22/2012	-0.68	0.07
9/21/2012	0.80	0.01
Average	0.14	0.07

Decrease in net growth per decrease in uM DIN

Date	Site	
5/10/2010	HB	0.030
6/9/2010	HB	0.009
6/22/2010	HB	0.050
7/28/2012	HB	0.020
8/22/2012	HB	0.013
9/6/2012	HB	0.005
9/21/2012	HB	0.008
8/22/2012	MB	0.024
9/21/2012	MB	0.015
Average		0.02

3.3.3 Nitrogen loads fertilize Ulva growth & hazardous deposition

Numerous possible impairments to designated uses of Western Bays are associated with dense meadows of Ulva growing at high rates and accumulating in dense mats after sloughing and being transported by currents. This study provides multiple lines of evidence pertaining to links between nitrogen loads and impairments associated with Ulva (Appendix A).

The general pattern of Ulva percent cover on the bottom appears to qualitatively mimic the geographic pattern of ambient nitrate levels. Figure 29 is a temporal mosaic of nitrate levels measured by SoMAS (uM) in 2012, and by Town of Hempstead (ug/L) 2000-2009, alongside Ulva percent bottom coverages for 2011 and 2012. The SUNY nitrate data was collected at the same sample locations as the Ulva. Higher Ulva percent cover appears positively correlated with higher nitrate concentrations. While ambient nitrogen levels are complicated by dynamic cycling processes, Ulva mat accumulations are also complicated by hydrodynamics which can result in accumulations in locations apart from those with stressful growth rates.

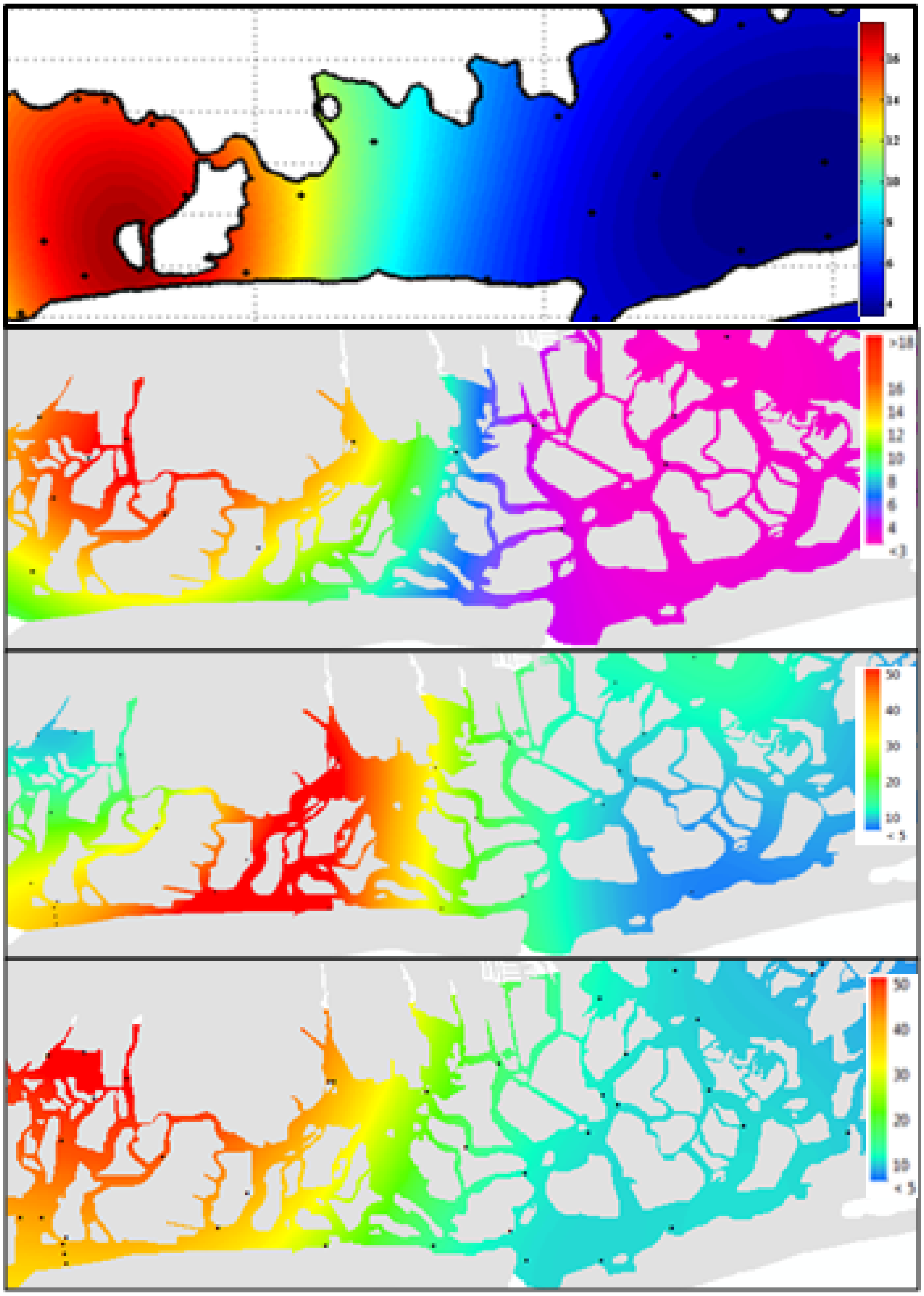

Figure 29. Mean summer surface Nitrate (ug/L) by Town of Hempstead 2000-2009 (top); Dissolved Nitrate (uM) by SoMAS 2012 (second); Percent bottom coverage of Ulva 2011 and 2012 (bottom).

Long Beach residents and ocean beach visitors know too well that even though the nitrogen loads into Western Bays fertilize rapid Ulva growth in shallow meadows miles away, and even though some Ulva settles to the bottom in Western Bays (Figure 29 above), a substantial portion of the malodorous, decomposing Ulva mats is also deposited in residential and recreational locations. The SoMAS model of Western Bays hydrodynamics explains some of these deposits based on tidal mixing and export through Jones Inlet. Figure 30 shows the trajectory of neutrally buoyant particles released at the Bay Park WWTP effluent location. Of course, Ulva particles are also released from meadows located throughout the Western Bays. As part of further research after this study was completed, Dr. Gobler's team collected and analyzed ulva that has been chronically washing up on the beaches in Long Beach and found it is highly enriched N-15 with a signature matching that of sewage.

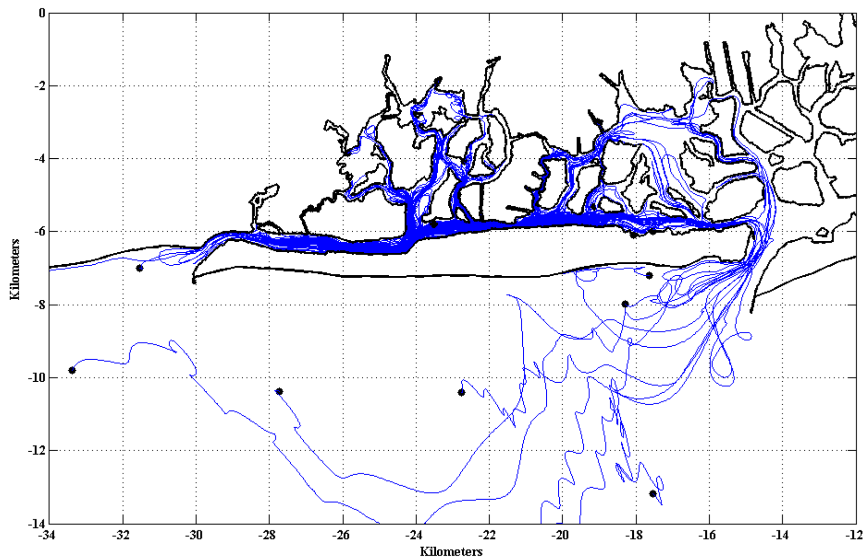

Figure 30. Lagrangian simulation of neutrally buoyant particles released from the Bay Park WWTP effluent location.

Algal growth kinetic experiments performed by SoMAS demonstrated that Ulva growth rates increased as nitrogen was enriched above the control of ambient levels in the Western Bays. These responses support the notion that Ulva growth appears to be limited by nitrogen, at least during the seasonal conditions mimicked experimentally. Note however that this does-response experiment was not able to derive a dilution level at which nitrogen levels would significantly decrease the Ulva growth rate within the experimental time period (Figure 31; details in Appendix A).

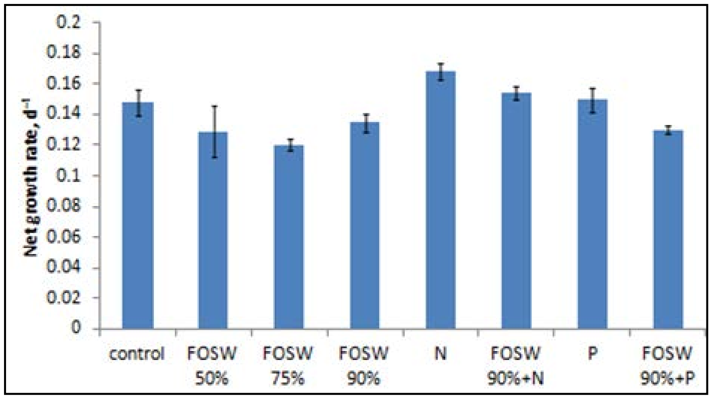

Figure 31. Effects of Ulva dilution experiments on Hewlett Bay sample.

One line of evidence presented counterintuitive patterns regarding the link between nitrogen loads and Ulva. SoMAS analyses of Ulva tissue revealed the signature of the $\delta^{15}N$ isotope, an indicator of nitrogen from anthropogenic origin. However the geographic trend increases with distance away from the largest anthropogenic load at the Bay Park WWTP effluent (Figure 32). $\delta^{15}N$ signatures have been be used elsewhere to identify anthropogenic nitrogen sources in many organisms including macroalgae (Cole et al. 2004). Tissue samples collected in the eastern portion of the Western Bays had heavier $\delta^{15}N$ signatures demonstrating that sewage effluent is the dominant N source to *Ulva* in this region (Valiela et al. 1992; Cole et al. 2004). Interestingly, lighter $\delta^{15}N$ signatures were found in western region of this ecosystem. These tissue samples were collected shortly after significant freshwater runoff associated with August storms and Hurricane Irene which significantly decreased salinity within Hewlett Bay. The western portion of the Western Bays has a number of golf courses and fertilizers have a very light $\delta^{15}N$ signature. As such, the mixed $\delta^{15}N$ signal may be due to this increased runoff during a period of intense rainfall. Additional temporal replication may clarify whether this metric could help nitrogen management.

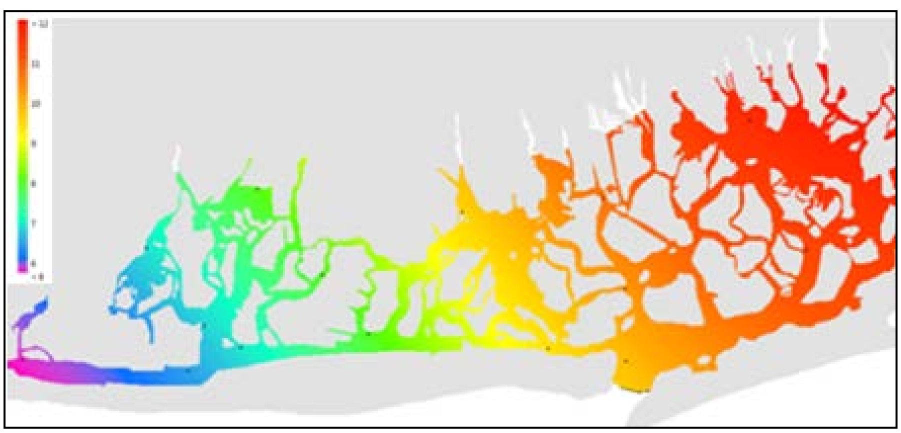

Figure 32. $\delta^{15}N$ signatures of Ulva sp. in Western Bays in the Fall of 2011.

3.3.4 Role of nitrogen in fish propagation

The fish propagation impairment finding (repeated below from section 3.1.4) provides a rational to link nutrients to the impairment. Experimental or in situ demonstration of these processes was beyond the scope of the QAPPs developed in this project.

Excessive nutrient loading into coastal ecosystems such as the Western Bays promotes algal productivity and the subsequent microbial consumption of this organic matter lowers oxygen levels and contributes toward hypoxia. Microbial degradation of organic matter produces CO_2 and reduces pH. To assess the potential for eutrophication-driven acidification in Hewlett Bay, SoMAS mapped the pH levels in September of 2012. Measurements revealed that the pH levels in Hewlett Bay were below 7.1 in the bottom layer and below 7.6 through much of the water column (Figure 9). These low pH conditions could have been caused by tidal influx of treatment plant discharge and the subsequent enhanced bacterial decomposition. pH levels in this range have previously been shown to yield elevated mortality in larval finfish and shellfish (Talmage and Gobler 2010; Baumann et al. 2012) suggesting that acidification, which has been intensified by climate change (Doney et al. 2009), may be altering the ability of Hewlett Bay to support robust fisheries.

3.3.5 Impairments without quantitative linkage to nitrogen

This study did not attempt to quantify all of the *in situ* nitrogen cycling processes. Nitrogen cycling processes in estuaries are dynamic both temporally and spatially (Figure 33). Quantification of all requisite nitrogen cycling processes within critical time periods and segments of the Western Bays exceeded the scope of this study, but the study design did incorporate a line of evidence focused on nutrient flux to/from sediment at multiple locations in Western Bays. The flux experiments demonstrated ammonium efflux from the sediment as another possible conceptual linkage between habitat impairments and nitrogen cycling (Appendix G). Sediment nutrient flux experiments highlighted the relative importance of this term due to its magnitude, as well as the spatial variability of sampling results.

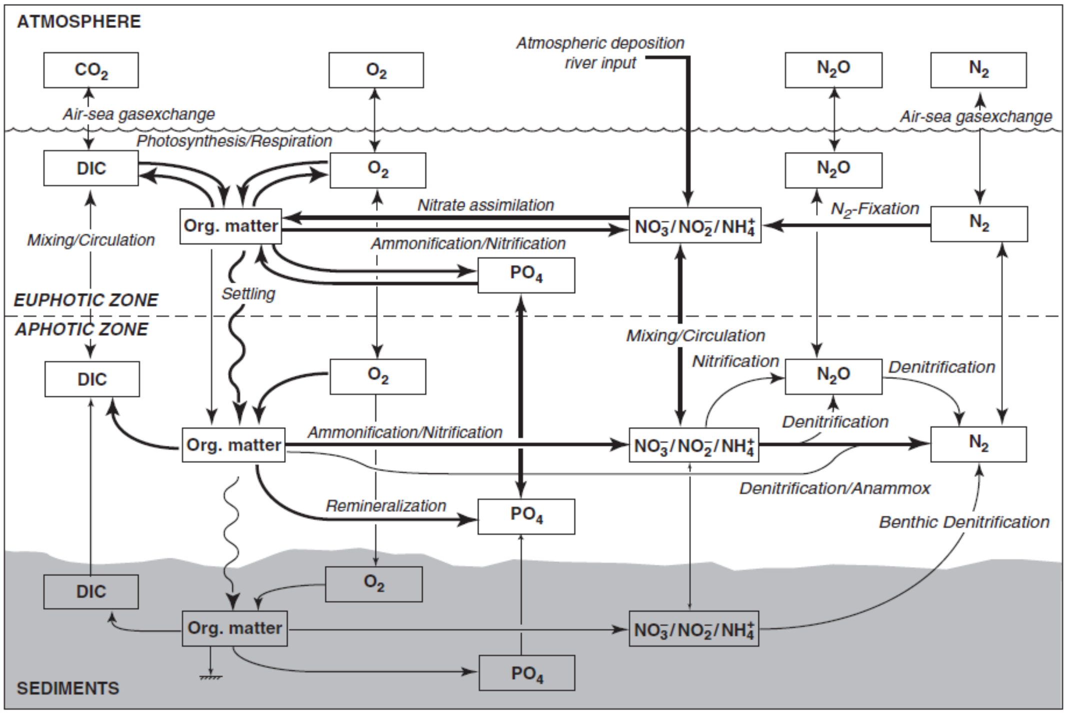

Figure 33. Marine nitrogen cycle (from Gruber 2008).

In lieu of attempts to quantify the full spectrum of nitrogen cycling processes, this study examined the sediment nutrient flux processes in lab experiments coupled with empirical evidence of ambient levels of nitrogen species at multiple sites in order to assess impairments relative to nitrogen load sources, ambient levels, and transport simulations.

Nitrification and remineralization of organic matter occur in processes which can deplete DO levels (Figure 33). These conceptual links define how nitrogen cycling with oxygen depleting processes, i.e. high sediment oxygen demand, could lead to impairments. However, empirically the DO impairments are not seen throughout the Western Bays, even though oxygen consumption was observed at all the sites sampled (Figure 34). DO impairments have only been seen consistently in waterbody segments which have poor circulation due to anthropogenic alterations (i.e. ToH 2 and Ulva 4). Thus ongoing nitrogen loads can contribute to impairments, but they may not be the only active cause for a given impairment event.

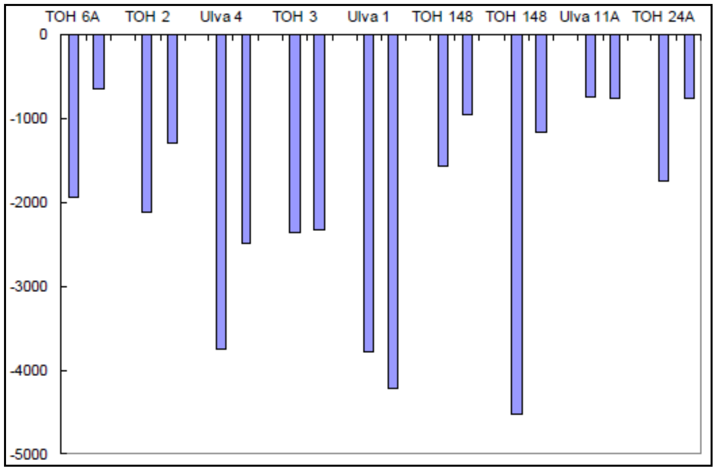

Figure 34. Sediment-water oxygen exchange (flux O_2 uM/m^2/hr). Negative indicates flux into sediment.

The sediment nutrient flux experiments performed in this study by Chesapeake Biogeochemical Associates identified that nutrient flux to/from the sediment occurs at high rates to varying degrees spatially (Appendix G). The wide range of flux rates observed, even within scales of 10 meters, underscores the difficulty that would accompany any attempt to define an absolute, bay-wide instantaneous nitrogen concentration to serve as an impairment measurement. Given the suspected impact of Ulva-based detritus on some of the high sediment nutrient flux rates, we would expect this variability to be fairly representative of these embayments and that a wide range in these rates should be expected at other depositional sites in the western Bays. It is not known whether the high rates results are representative of the parts of Western Bays which are not depositional.

The exchange of oxygen at the sediment-water interface occurred at rates typical of impacted coastal ecosystems. Rates > 2,500 µmol m-2 h-1 are relatively high; Five of 18 cores had high rates, with 5 having low rates below 1000 µmol m-2 h-1 (Figure 34). Replication was not especially good with these cores, reflecting differential inputs of organic matter in the Western Bays on the scale of 10 m or less. For example, the obvious macroalgal decaying biomass evident in TOH 6A core 1A led to higher rates of metabolism than observed in core 1B. Higher spatial variability is a general feature of shallow water ecosystems, with differential primary production of benthic microalgae and macroalgae, differential distribution of benthic animals, and variable organic matter deposition to the sediment-water interface.

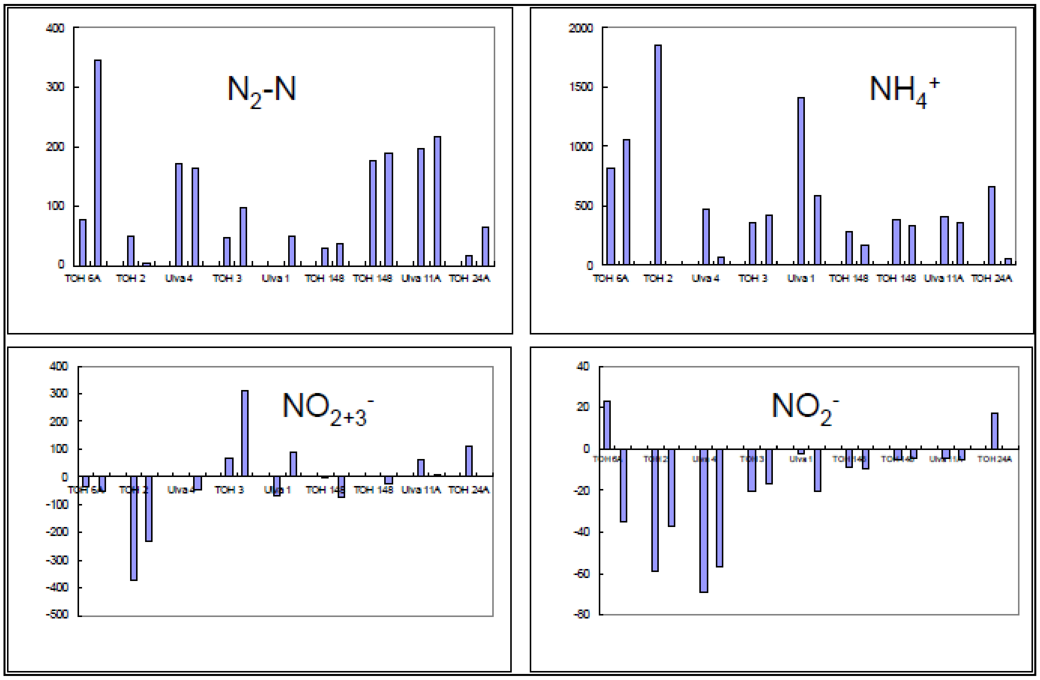

Figure 35. Sediment-Water nitrogen exchange. (N Flux uM/m^2/hr). Negative indicates flux into sediment.

The flux of ammonium was highly variable, with the highest rates at the high end of observations in coastal ecosystems (Boynton and Kemp 2008, Joye and Anderson 2008). The highest rates were > 1,000 µmol m-2 h-1 (Figure 35), higher even than rates for the anoxic deep Chesapeake Bay (Kemp et al. 2005). We have observed many rates in coastal ecosystems that were very low or actually representing an uptake, but the site to site and core to core variability in this ecosystem are unusual. Similarly, we observed both high rates of uptake of nitrate plus nitrite, as well as high rates of efflux (ToH 3). Most rates were relatively low. The rates of denitrification ranged from extremely low (< 25 µmol m-2 h-1) to high (> 300 µmol m-2 h-1). Denitrification rates calculated in this study's one-time snapshot (October 2012) of these Western Bays sites approach or exceed the ranges of most estuaries summarized by Seitzinger (1988). Calculation of annual rates would need further study to account for seasonality of confounding factors such as temperature, oxygen, ambient nitrogen, and infauna. For perspective only, extrapolation of these one-time values to median annual nitrogen removal from denitrification across the 4,000 hectares of Western Bays sediment, or 6,900 hectares of sediment plus intertidal wetlands, would range from 7,000 – 61,000 kgN/y or 12,000 – 110,000 kgN/y respectively.

Sediment flux results from the Western Bays were compared to findings from similar methods applied in Jamaica Bay (NYCDEP 2006). Relative to Jamaica Bay, sediment from the Western Bays sites consumes less oxygen, produces less N2-N from denitrification, has higher NH4+ effluxes, has lower NO2+3- fluxes, and has higher median soluble reactive phosphorus (SRP) effluxes (but fewer high SRP flux rates). Many of the Jamaica Bay rates were driven by high population of amphipods at the sediment surface; that animal community promoted nitrification that supplied the substrate for denitrification. Macrofauna communities like the tube dwelling benthic amphipod *Ampelisca* often found in Jamaica Bay were not observed in any the cores collected from the Western Bays in October 2012, however, they have been observed recently in abundance at other Western Bays sites (Brownawell pers. comm. 2013)

Empirical comparisons of sediment nutrient flux rates and core characteristics with results from other ecosystems indicate that these Western Bays sediments are impacted by eutrophication, with exceedingly high ammonium effluxes at a three sites. The dying and decomposing macroalgae observed at some sites suggest that cores were collected at a transition period, when shorter days and cooler temperatures may have resulted in decreased macroalgae production and increased rates of sediment metabolism. The rapid decomposition of macroalgae drives high rates of anaerobic metabolism causing very reducing conditions in near surface sediments even with near saturated bottom water oxygen. Reducing conditions in near surface sediments promote N and P recycling, inhibit coupled nitrification/denitrification, and drive the sediments to a state that is apparently inhospitable to most macrofauna. The importance of this conceptual linkage toward regulatory uses may deserve further study across seasons and embayments.

3.4 Which Nitrogen Management Practices Are Most Promising?

A nitrogen management measure is defined as most promising based on the relative merit of its sustainability as a cost-effective practice that significantly reduces the ecosystem stress of excess nitrogen. The following sections pursue answers to the question of which practices are most promising to reduce nitrogen stress in the Western Bays. A summary of the nitrogen loading sources forms the foundation to assess options. The four basic phases of the nitrogen cascade within the Western Bays ecosystem frame four strategies for options of when and how to manage nitrogen loads. Finally, a decision model is proposed as a flexible method to help Western Bays stakeholders examine the tradeoffs involved with selection amongst the many alternatives which can reduce nitrogen stress.

The level of significance of a candidate nitrogen management measure should be evaluated relative to its contribution to the overall load and the permanence of its impact. A description of the overall loads of nitrogen into the Western Bays follows in section 3.4.1 and 3.4.2. Identifying the relative merit of candidate nitrogen management measures requires a means to compare options systematically. A systematic comparison could integrate many considerations, such as agreed upon performance expectations and quantifiable criteria with ranges of acceptability. A proposed decision model to systematically compare nitrogen management measures in the Western Bays watershed is outlined in section 3.4.4 below. Sustainability in this context refers to how well the nitrogen management measures can be accomplished as part of the essential functions of residents, industry and government, e.g. energy production or consumption, food, transportation, commerce, public health or ecosystem services. Systematic treatment of cost-effectiveness is best accomplished within the bounds of a given funding source and its beneficiaries, i.e. sources of funds often dictate their application, thus cannot be systematically compared directly with the effectiveness of all other independent sources of funding which may address alternative timescales. The proposed decision model offers cost considerations based simply on scale and possibility of cost-sharing. The model does allow for a surrogate of cost-effectiveness by elevating the relative priorities of the cost criteria plus the criteria which reflect possible magnitude of nitrogen removal.

3.4.1 Nitrogen loads into Western Bays ecosystem

The nitrogen cascade (Galloway et.al. 2003) begins in the Western Bays ecosystem via atmospheric deposition, application of landscape fertilizer and import of food for humans and domestic animals. This section will summarize the fate of each source of imported nitrogen through the ecosystem. Due to the predominance of residential wastewater discharge into the Western Bays, an extra process will be addressed to supplement the conventional nitrogen pathways (Figure 36). Land surface characteristics and land use patterns are used to derive the fate of nitrogen deposited on land surfaces. Import of nitrogen as part of human food is addressed indirectly later on as part of WWTP effluent load to the waterbody. Import of nitrogen as part of domestic animal feed is addressed as a single total for dogs, cats and horses based on county population, number of households and industry coefficients for animals per household, nitrogen waste per species and portion of domestic pet waste disposed of subject to runoff.

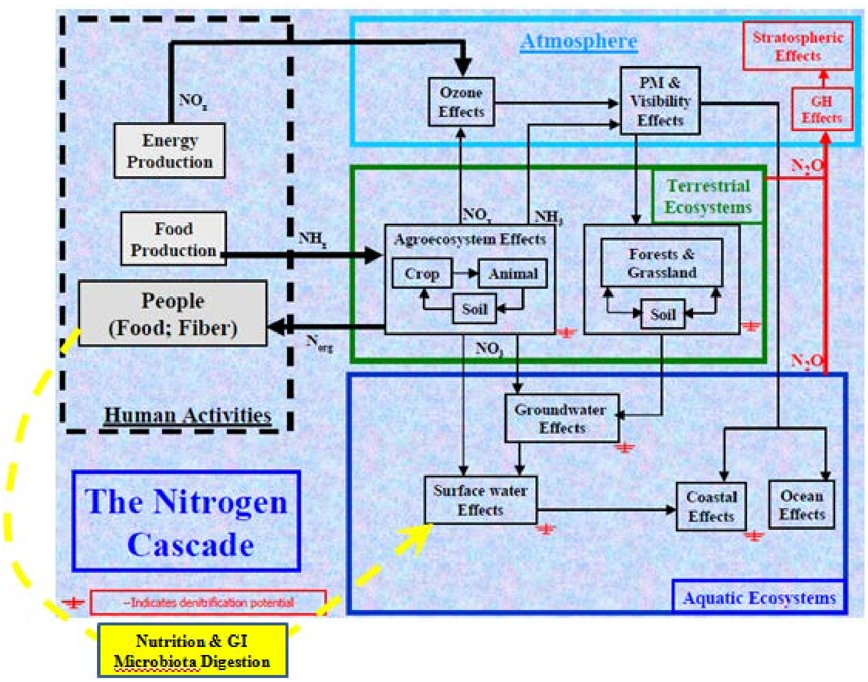

Figure 36 . The nitrogen cascade as conceived of by Galloway et.al. (2003) with an additional term (yellow) to represent nitrogen processing in human digestion prior to wastewater production.

After description of the nitrogen imports and their fate on the watershed, subsequent nitrogen loading into the Western Bays waterbody will be quantified as loads from atmospheric deposition, surface water, ground water and wastewater.

3.4.1.1 *Land surface Characteristics*

The land surface types on which the nitrogen is loaded are integral for estimating the fate of the nitrogen in relation to loading into the Western Bays. The amount of each land surface type is also integral to estimating the fate of imported nitrogen. Land surface amounts are discussed below. The fate of nitrogen is influenced by the residence time of imported nitrogen on a land surface, processes therein, and the transport distance – the distance that each land parcel is from the shoreline of the Western Bays. Geospatial modeling of transport time is not attempted within this estimation of Western Bays waterbody loading. General coefficients for nitrogen removal are provided to demonstrate that first order approximations of nitrogen import and nitrogen removal in the watershed do bound the waterbody loading terms used herein. The coefficients used were based on a combination of in situ measurements and hydrologic simulation models.

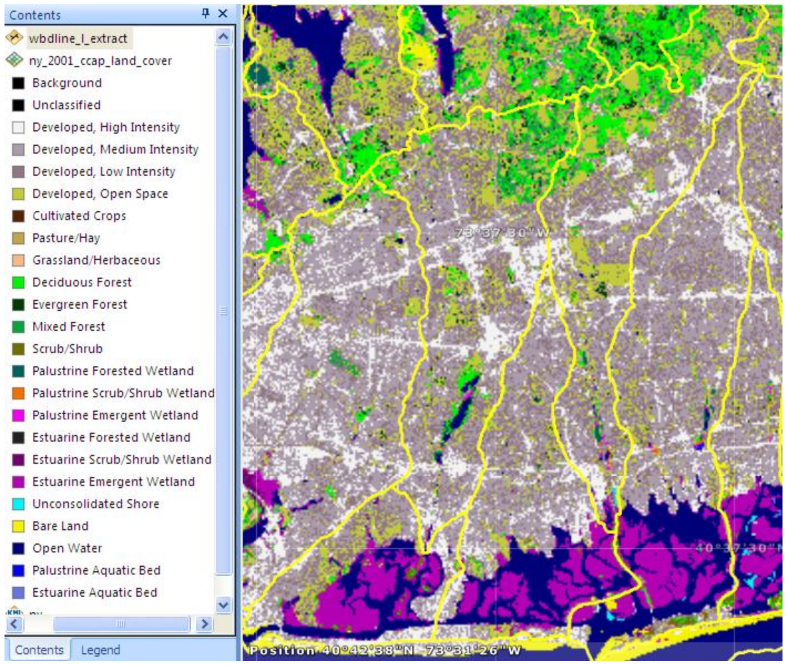

Figure 37. Subwatersheds of the Western Bays ecosystem delineated by USGS Hydrologic Units overlaid on land uses from NOAA Coastal Change Analysis Program

Land surface amounts were estimated based on USGS delineations of the area of watersheds which contribute surface water to the Western Bays (Figure 37). The total Western Bays watershed area is 39,000 hectares. Of this total area, the Western Bays waterbody was estimated via GIS manual delineation to be approximately 6,900 hectares. The F.R.E.D. Environmental study conducted for this program (Appendix H) determined that tidal wetlands make up 39% of Middle Bay and Western Bay segments. Extrapolation of this proportion to the total geographic extent of East Bay yields an estimated area of tidal wetlands to be 2,900 hectares as of 2008.

3.4.1.2 *Fate of Nitrogen from Atmospheric deposition*

Atmospheric deposition trends for nitrogen as a sum of dry and wet forms ($HNO_3 + NO_3 + NH_4$) were summarized in USEPA's CASTNET report in 2010. The trend of reductions in total nitrogen deposition in the eastern states continues (Figure 38). A total nitrogen deposition rate of 5 kg/ha/yr, reported for New York in 2010, was used for estimation of the loading rate into Western Bays watershed. The atmospheric deposition of nitrogen into Western Bays waterbody was treated as two terms, one for fate of the load deposited onto land and a second term for atmospheric deposition directly onto the waterbody.

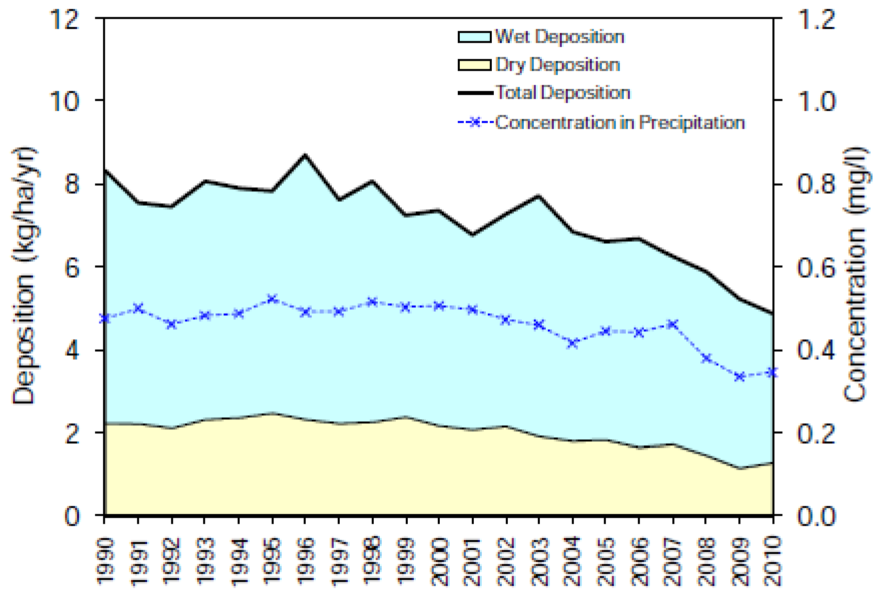

Figure 38. Ongoing trend of reductions in atmospheric deposition of nitrogen in eastern U.S.

3.4.1.3 *Fate of Nitrogen from Land Uses*

Nassau County's 2010 Master Plan offers a recent depiction of land uses county wide (Figure 39). That plan includes a breakdown of key land uses which impact the fate of imported nitrogen fate relative to loads into the Western Bays. The county-wide percent of land area in each major use as of 2009 were: residential 60%; retail 4%; office 2%; industrial 2%; institutional 2%; open space 17%; and vacant 4%. For purposes of this study, these land use percentages will be assumed to pertain equally to parcels throughout the Western Bays watershed. These major land uses were further consolidated into impervious surface or vegetated surface in order to employ nitrogen removal coefficients from Kinney and Valiela (2011). Retail, office, industrial and institutional uses were summed into the percent impervious surface.

The aerial percent impervious surface was increased further by adding 33% of the land classified as residential. 67% of the residential area was allocated to the vegetated surface total. This allocation of residential surfaces was made in accordance with the NY State Stormwater Management Design Manual (2011) and was confirmed through visual inspection (Figure 40) and quantification of the impervious surface cover from satellite images of a random collection of 15 residential parcels in the watershed.

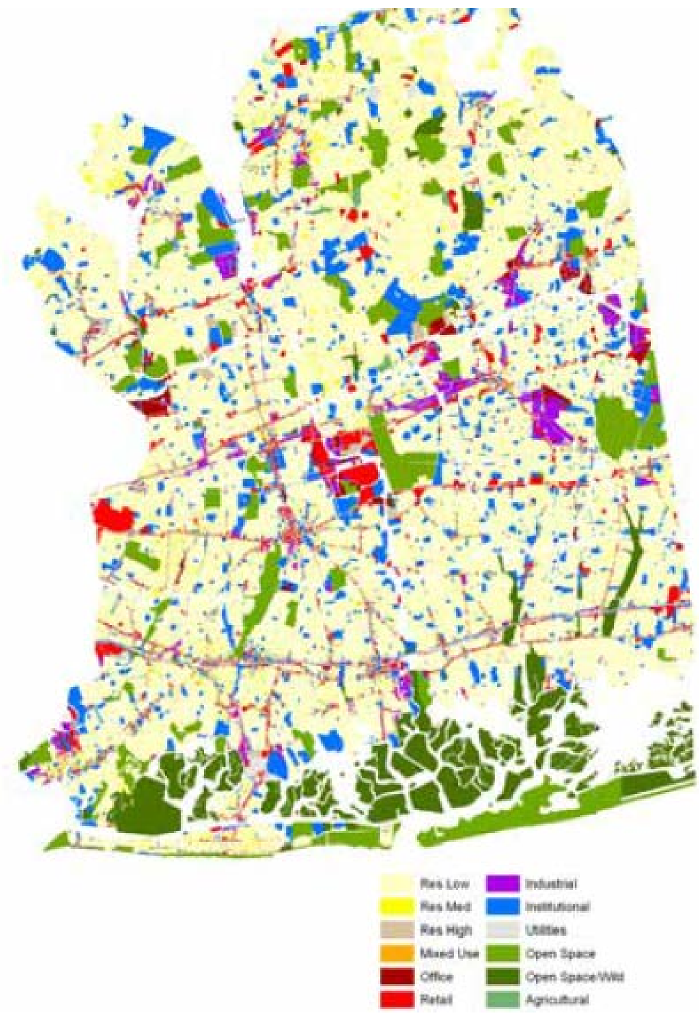

Figure 39. Distribution of land uses throughout Nassau County as presented in county's Master Plan 2010.

Figure 40. Residential land use assumed to be homogenous throughout the watershed and comprised of 33% impervious surface.

The county's residential land use statistic is also useful to predict the amount of fertilizer residents will introduce to the watershed. Law et.al. (2004) found through field studies that an accurate approximation of lawn fertilizer application can be made on the basis of the percent of land used as residential. An annual rate of 27 kg N / hectare residential area is used for Western Bays watershed estimates. Law et.al. (2004) found the variability of this rate was far less than fertilizer application rate estimates based on watershed area or total area of lawns in the watershed.

Commercial fertilizer sales statistics from NY State Department of Agriculture and Markets (NYS A&M FOIL #14-208) summarize county-wide total tons of fertilizer purchased for farm and non-farm use. The average tons of nitrogen purchased in Nassau County between 2011 and 2013 was 1,120 tons. The Western Bays watershed portion would be 590 tons of fertilizer nitrogen when allocated by its 52% portion of Nassau County's land area. This metric aligns well with the nitrogen load estimated based on land-use application rate estimates.

3.4.1.4 *Fate of Nitrogen from Domestic Pet Feed and Waste Disposal*

The introduction of nitrogen into the Western Bays watershed occurs when feed is imported into the county and it is excreted as waste onto land surfaces that will runoff onto the watershed. The number of dogs and cats are based on national average statistics for dogs (1.6) per households with dogs (37%), cats (2.1) per households with cats (30%), plus 50% of horses said to be stabled in Nassau county. The amount of waste these species produce is 2.0, 1.5 and 40 kg N / animal / year. The amount of dog and cat waste attributed to disposal onto watershed surfaces is 55% and 25% respectively. A guestimate of 50% of horse waste is said to be added to watershed land surfaces because a portion of their feed is from grazing watershed vegetation with nitrogen that was not imported. Total nitrogen waste added to the watershed from these domestic dogs (130,000 kg N/yr), cats (45,000 kg N/yr) and horses (32,000 kg N/yr) equals 210,000 kg N /yr. The nitrogen in this waste is then assumed to be 84% taken up by vegetation or percolation.

3.4.1.5 *Fate of Nitrogen from Golf Courses*

Golf courses within the Western Bays watershed were identified through a Google search atop the USGS 12 digit HUC perimeter. Sixteen golf courses were identified. The area of each golf course area was assumed to be 40 hectares (~100 acres). The annual rate of fertilizer application onto golf courses is estimated at 133 kgN/hectare of golf course land use (Buzzards Bay National Estuary Program, undated).

3.4.1.6 *Synopsis of Nitrogen Cascade Through Watershed*

Estimations of nitrogen being imported into the Western Bays watershed sum to an estimated maximum of 790,000 kg/yr (rounded to two significant figures) (Table 5). This estimate does not attempt to account for the import of nitrogen as human food products.

Table 5. Estimated budget and fate of nitrogen imported into the Western Bays watershed.

Nitrogen Source	Area	Nitrogen Import Rate	Annual Nitrogen Input to Watershed	Literature Values for *in situ* Nitrogen Removal	Estimated Annual Nitrogen Output from Watershed
Atmospheric Deposition Wet & Dry onto Watershed Land (CASTNET 2010)	32,000 hectares	5 kg/ha/y	180,000 kg/y 200 tons/y	90% in Vegetation 75% off Impervious Resid. 33% Imperv.	30,000 kg/y 33 tons/y
Residential Fertilizer Application Rate (adapted from Law et.al. 2004)	19,000 hectares	27 kg/resid. ha/y	520,000 kg/y 570 tons/y	84% in Vegetation	83,000 kg/y 91 tons/y
Domestic Pet Feed and Waste Disposal (AVMA 2012, Trowbridge et.al.2013)	110,000 dogs 120,000 cats 3,200 horses	2.0kgN/dog/y 1.5kgN/cat/y 40kgN/hrs/y	55% runoff 25% runoff 50% runoff	84% in Vegetation	32,000 kgN/y 35 tons/y
Watershed Golf Course Applications	650 hectares	133 kg/golf ha/y	86,000 kg/y 95 tons/y	84% in Vegetation	14,000 kg/y 15 tons/y
	Subtotal Watershed Loads		**790,000 kg/y 870 tons/y**	**(Budget below in Table 6 uses 590 tons/y estimates for streams & groundwater below)**	**160,000 kg/y 180 tons/y**

A portion of the 790,000 kg/yr of nitrogen imported into the Western Bays watershed becomes available to the downstream bays. Table 5 above employs a collection of estimates for rates of imported nitrogen cycling and removal within the watershed. The rates of nitrogen removal via vegetation uptake and denitrification, as well as the fate after deposition on impervious surfaces were adopted from the Kinney and Valiela (2011) simulation of nitrogen processes in Suffolk County, NY. The amount of nitrogen available to leave the watershed and enter the Western Bays could be as low as 160,000 kg/yr. Insertion of these loading terms into the nitrogen loading model as adapted from Latimer and Charpentier (2010) yields a comparable net load of 123,000 kg/yr available to enter the Western Bays.

3.4.2 Nitrogen loads into Western Bays waterbody

Selection of nitrogen management plan options should consider each measure's possible efficacy within the context of the cumulative load of nitrogen from all sources into the Western Bays. The bays receive nitrogen from direct atmospheric deposition, discharge of wastewater, and inflow of surface water and submarine discharge of groundwater. Wastewater discharge from plants was acquired from plant records. Wastewater leaching from septic systems was estimated by SoMAS (Appendix D). Wastewater discharge from vessels was not estimated in this budget. Atmospheric deposition rates are the same as applied to the upland watershed. Nitrogen loads from surface water and groundwater are tallied below based on in situ measurements combined with hydrological simulations. The budget and fate of nitrogen imported to the watershed offer high (790,000kgN/yr) and low (160,000kgN/yr) load ranges from which to evaluate the reasonableness of the measurements and simulations of how this load cascades en route to the bays.

To empirically estimate the nitrogen load into the bays from Western Bays subwatershed surface waters requires volumes of surface water inflows and their respective concentrations of nitrogen. Both sets of these measurements proved challenging to compile for the Western Bays during a synoptic period.

Surface water inflow volumes and concentrations were calculated by Monti and Scorca (2003). Their calculations examined a broader geographic range than the Western Bays. The pertinent section of their work integrated gauged stream inflows at 4 locations within the Western Bays watershed: Pines Brook, East Meadow Brook, Bellmore Creek and Massapequa Creek. Their study summarized decadal trends in total nitrogen levels measured in those streams. For Table 6 below, which is specific to Western Bays, surface water inflow nitrogen loading estimates were calculated for each individual stream. The total nitrogen median concentrations in those streams from 1971-1997 ranged from 3.0 – 7.4 mg/L annually. This nitrogen load source is estimated to be 160,000 kg/yr.

Monti and Scorca (2003) also calculated nitrogen load into the South Shore Estuary Reserve from submarine discharge of two types of groundwater: shallow and deep. Groundwater discharge data were not available for the exact Western Bays subwatersheds. To be conservative, i.e. to avoid underestimating nitrogen loads, this study adopts Monti and Scorca's groundwater discharge volumes and concentrations which were reported for the full length of the southern shoreline of mainland Nassau County. This region extends to the east approximately 4 miles beyond the 14.5 mile length of our study area. The total nitrogen median concentration in shallow and deep groundwater were 3.9mg/L and 0.15 mg/L respectively. These nitrogen load sources are estimated to be 160,000 kg/yr and 2,900 kg/yr respectively.

The Monti and Scorca estimates of raw nitrogen loads from watershed totals 320,000 kg/yr. This estimate is twice as large as the 160,000 kg/yr predicted when taking into account literature estimates for the generic nitrogen removal processes available for each watershed source.

The SoMAS hydrodynamic and hydrologic model of the Great South Bay, which was used in this study to predict residence times and circulation, offered another opportunity to check or refine the Monti and Scorca estimates. The SoMAS model incorporated both surface water inflows and submarine groundwater discharges in order to simulate forcing mechanisms for circulation and mixing throughout the Western Bays. The SoMAS model used surface water data from a broader range of locations (Figure 41) based on the USGS (http://nwis.waterdata.usgs.gov/usa/nwis/discharge) daily streamflow gauges at 18 locations along the north shore of the Western Bays. Annual inflow rates were assigned at model nodes adjacent to these sites. For the ungauged creeks, estimated freshwater inflow rates were assigned based on estimates for watershed area and mean annual precipitation. Some refinement of runoff coefficients was required to improve the agreement between available salinity observations and simulations. These inflows coupled with approximations of submarine groundwater discharge (http://po.msrc.sunysb.edu/GSB/forcing.htm) were used to formally calibrate the model for sea surface elevation and to informally verify salinity distributions. Submarine groundwater discharge flow was assumed to decrease exponentially away from the north shore of the Bays. The skill of the model's predicted salinity structure offers confidence that both the surface water stream flows and groundwater inflow estimates have appropriate magnitudes.

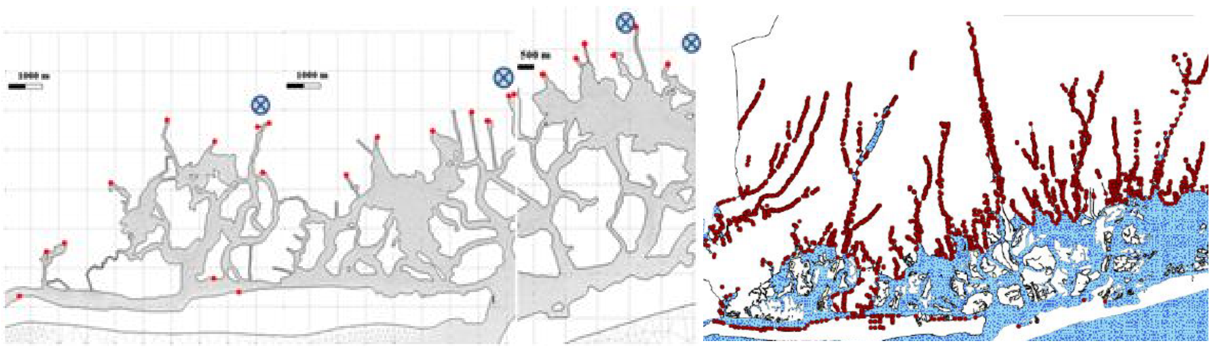

Figure 41. General location of 4 surface water gauged stations (blue) used by Monti and Scorca, 25 SoMAS model nodes (red) of surface water inflows, and stormwater outfall county inventory.

The SoMAS model's freshwater inflows (130,000,000 m3/yr) greatly exceeded the volumes used by Monti and Scorca (91,000,000 m3/yr). To be conservative, this study fills the gap by adding an additional surface water nitrogen load source term with a volume of 39,000,000 m3/yr with a conservative concentration of 5mg/L based on the range of median values Monti and Scorca observed in surface waters and groundwater. This nitrogen load estimate contributes 230 tons/y (200,000 kg/y). This adjustment source alone exceeds the entire 140 tons/y (130,000 kg/y) predicted when taking into account nitrogen removal processes all watershed sources would be subject to.

The average total nitrogen load entering the Western Bays from watershed land sources is estimated at 590 tons/yr (540,000 kg/yr). This total reflects nitrogen concentrations measured in surface water or ground water, and assumes zero removal of nitrogen from the surface water or the groundwater/sediment interface as it enters the Western Bays. This estimate is 4 times larger than the 180 tons/y (160,000 kg/y) predicted when taking into account some of the literature estimates for the generic nitrogen removal processes each watershed source could be subject to. This estimate also assumes 0% removal of submarine groundwater discharge nitrogen removal in estuarine sediments, even though Seitzinger (1988) found denitrification rates which could remove 20% to 50%.

Table 6 . Nitrogen loads into Western Bays waterbody. The only in situ nitrogen removal processes accounted for are vegetative uptake of atmospheric deposition and septic denitrification.

Nitrogen Source	Area or Flow	Nitrogen Load Rate	Annual N Input to Bays	Possible Source Load Nitrogen Removal	Annual N Load Into Western Bays
USGS Gauged Streams Pine Brook East Meadow Bellmore Creek Massapequa (Monti&Scorca,2003)	 3,100,000 11,000,000 7,800,000 8,700,000 m3/y	 3.0 3.9 5.7 7.4 mg/L		end of stream samples assume 0%	 9,300 kg/y 43,000 kg/y 44,000 kg/y 64,000 kg/y 180 tons/y
USGS Modeled Shallow GW Nassau (Monti&Scorca,2003)	41,000,000 m3/y	3.9 mg/L		sediment & marsh denitrification assume 0%	160,000 kg/y 180 tons/y
USGS Modeled Deep GW Nassau (Monti&Scorca,2003)	19,000,000 m3/y	0.15 mg/L		sediment & marsh denitrification assume 0%	2,900 kg/y 3.2 tons/y
Freshwater Inflows to Balance Salinity (SoMAS Appendix E)	39,000,000 m3/y	5 mg/L		assume 0%	200,000 kg/y 230 tons/y
Atmos. Dep. Wet&Dry onto W. Bays (CASTNET 2010)	6,900 Hectares	5 kg/ha/y	38 ton/y 35,000 kg/y	39% of bay wetland 90% in vegetation	21,000 kg/y 24 tons/y
Point Lookout Hamlet Septage (SoMAS Appendix D)			9.8 tons/y 8,800 kg/y	vadose denitrification assume 50%	4,400 kg/y 4.9 tons/y
Atlantic Beach STP Long Beach STP Bay Park STP Lawrence STP (SoMAS Appendix D)	690,000 6,800,000 69,000,000 1,700,000 m3/y	14 19-24 30 11-22 mg/L		NA	9,700 kg/y 150,000 kg/y 2,100,000 kg/y 29,000 kg/y 2,500 tons/y
				Total Load into Western Bays (with assumptions above)	**3,100 tons/y 2,800,000 kg/y**

The insertion of the additional nitrogen load attributable to the volume of surface water inflows beyond the Monti and Scorca surface water load term represents a substantial portion of the overall budget (Table 6 and Figure 42). The addition is 7.4% of the total nitrogen load budget (Table 6 and Figure 42).and, exceeds the magnitude of the original Monti and Scorca (2003) surface water nitrogen load estimate.

The nitrogen load estimates included in Table 6 reflect conservative assumptions at each incremental calculation. Retaining high estimates for these terms enables the nitrogen management plan to devote resources proportionally to the broadest set of possible contributing sources. However, efficacy of nitrogen management options should not be evaluated in terms of absolute values of nitrogen removed against this total load unless and until there is agreement on the assumptions and baseline conditions which are reflected in these nitrogen load estimates.

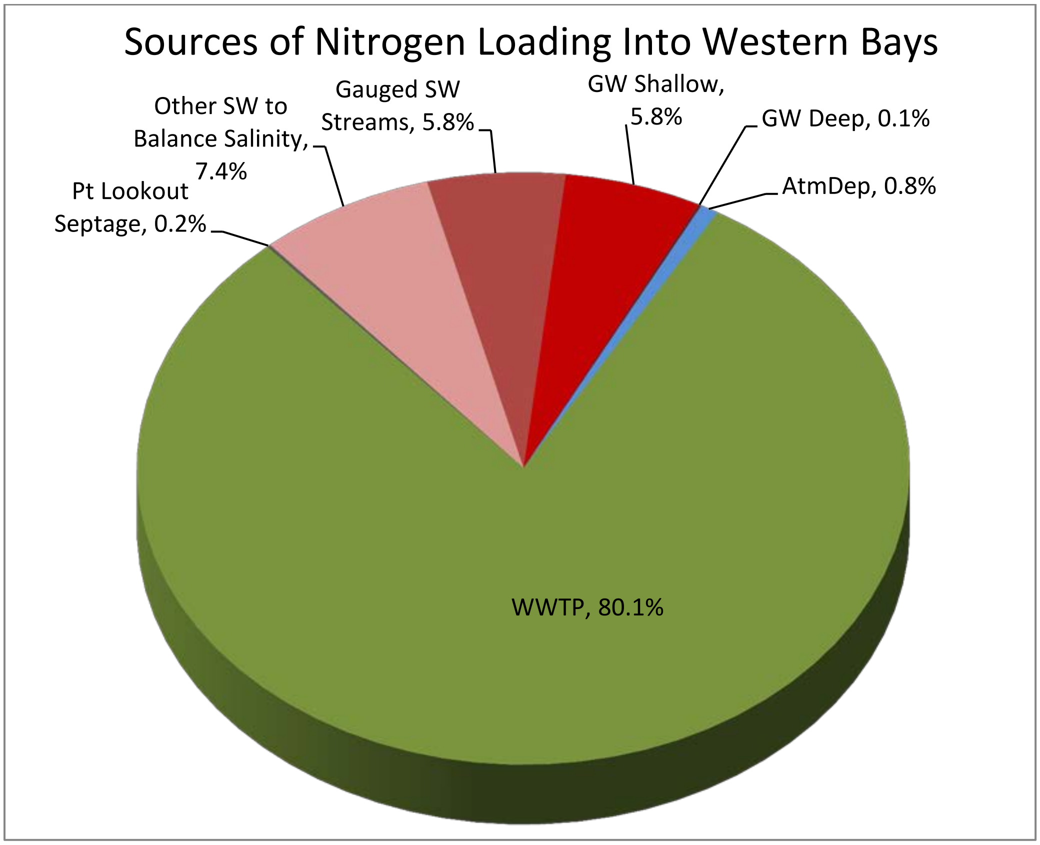

Figure 42. Relative amounts and sources of nitrogen loaded into the Western Bays.

The total load of 2,800,000 kg nitrogen into the 6,900 hectares that make up the Western Bays represents an average annual load of approximately 410 kgN/hectare of estuary. For perspective, Table 7 from Kinney and Valiela (2011) lists the annual load of 38 kgN/ha/yr for Great South Bay (just east of the Western Bays ecosystem) as within a wide range of such calculations made for estuaries worldwide. It is notable that if the entire nitrogen load contribution from the Western Bays WWTPs were removed from the annual total, the estuary may still receive up to an annual load of 61 kgN/ha/yr based on recent measurements. These present day loading rates are within the ranges of nitrogen loading rates reported for northeast U.S estuaries by Latimer and Charpentier (2010). This list is offered simply to describe the range of loading rates estimated in a variety of estuaries. No attempt is being made to suggest reference sites or to identify an optimal level of nitrogen loading into the Western Bays.

Any attempt to compare ecosystem health or eutrophication patterns between the Western Bays and other estuaries would need to account for a composition of factors well beyond nitrogen loading rates including, but not limited to, waterbody geomorphology, flushing or residence time, flora and fauna, age of the ecosystems and climate zone. Determination of reference conditions might consider pre-anthropocene loads as an alternative perspective to the loading derived from present day land-use. Howarth (2008) estimates that pre-industrial loading of nitrogen into northeast estuaries from watersheds was on the order of 1 kgN/ha/yr. Extrapolating for the 39,000 hectare watershed into the 4,000 hectares of Western Bays water would be a ratio of 9.8 kgN/ha/yr. Of course, pre-industrial would also mean the flushing and ocean exchange rates in Western Bays, i.e. dilution of the nitrogen load, were much higher than present.

The average loading rate of 410 kgN/hectare of estuary reflects numerous assumptions. This study sought to incorporate assumptions which consistently biased the loading rate toward higher values. While statistical confidence in this number is not expressed quantitatively, these assumptions make it more likely that the actual loading rate is lower than 410 kgN/hectare rather than higher. An extensive summary of the uncertainties associated with estimating estuarine loading rates is presented by Galloway et.al. (2004), wherein they highlight major uncertainties associated with estimating the rates of reactive nitrogen removal via estuarine sequestration and or denitrification.

Table 7. Total annual nitrogen load per area into estuaries (table from Kinney & Valiela 2011).

Estuary	N load (kg N ha^{-1} yr^{-1})	Reference
Sage Lot Pond, Massachusetts, USA	14	Valiela et al. (2000)
Moreton Bay, Australia	24	O'Donohue et al. (2000)
Barnegat Bay, New Jersey, USA	24.5-30.1	Bowen et al. (2007)
Pleasant Bay, Massachusetts, USA	25	Carmichael et al. (2004)
Tampa Bay, Florida, USA	28	Bianchi et al. (1999)
Chincoteague Bay, Virginia, USA	31	Boynton et al. (1999)
Great South Bay, New York, USA	**38**	**This study**
Sarasota Bay, Florida, USA	56	Bianchi et al. (1999)
West Falmouth Harbor, Massachusetts, USA	76	Carmichael et al. (2004)
Venice Lagoon, Italy	130	Sfriso et al. (1992)
Roskild Fjord, Denmark	204	Nienhuis (1992)
Bass Harbor Marsh, Massachusetts, USA	225	Kinney and Roman (1992)
Great Bay, New Hampshire, USA	252	Short and Mathieson (1992)
Quashnet River, Massachusetts, USA	350	Valiela et al. (2000)
Wadden Sea, Northern Europe	500	Nienhuis (1992)
Childs River, Massachusetts, USA	601	Valiela et al. (2000)

*Shallow coastal lagoons such as GSB (average depth <2 m) have significantly smaller volume of water per ha than some of the other embayment types listed in this table such as Wadden Sea and Roskild Fjord.

3.4.3 Nitrogen management plan (NMP) options

Strategies to reduce nitrogen in the Western Bays watershed ecosystem can be deployed at four levels: before import to the watershed; before release onto the watershed; before discharge into the bays; or after discharge into the bays. Each of these four levels offers a range of possible tactical measures as the nitrogen cascades through the ecosystem. Selection of which measures to invest in should reflect the funding organization's goals and priorities. The optimal choices can vary depending on many factors, such as whether one's goal is to invest all resources into one best possible measure, or if the goal is to have immediate results versus longer term approaches. Eleven suggested criteria are offered in Table 8, along with scoring thresholds, to portray inputs and outcomes of one scenario of a Western Bays nitrogen management plan decision model. This nitrogen management plan options model is offered as a tool for stakeholders to evaluate options and tradeoffs iteratively by supplying their own knowledge and priorities.

As mentioned in the previous section (noting the many caveats about the challenges of to making defensible comparisons), even if the Western Bays' WWTP effluent load were removed from the shallow bays ecosystem, the estimated remaining nitrogen loads may still occur at rates that would be higher than the areal loading rates of other estuaries where eutrophication issues exist. Thus, putting all NMP funds into only one effort may be ill-advised.

The following sections outline traditional, innovative, and in many case cost-effective, approaches to nitrogen load reduction. Many suggestions could be implemented as grass-roots efforts by an informed citizenry. The merits of any given NMP option needs to be evaluated relative to the complementary and or competing options that are available within the same geographic region and scale of interest. The numerous regions, scales and configurations of options can be evaluated through NMP model iterations.

3.4.3.1 ***Source Reduction - Reduce Nitrogen Import into Watershed***

1.0 reduce combustion emissions of NOx

US power utilities and the transportation industry continue to improve their emission controls in response to regulations. Consumer behaviors contribute further to this trend through purchase of non-fossil fuel power generation, lower emission vehicles and more energy efficient transportation.

2.0 improve nutrition and digestion to decrease WWTP influent

The amount of nitrogenous waste in WWTP influents depends in part on gastrointestinal efficiency in using the types and amounts of food consumed. Healthier food consumption choices and promotion of balanced gastrointestinal microbiota offer relatively new means to reduce nitrogen wasteload to the bays.

3.0 educate and incentivize lower nitrogen footprints

The public deserve greater awareness of the human health and ecological risks associated with the nitrogen cascade. Instead of a new, 3rd campaign, nitrogen considerations should be incorporated into ongoing education efforts about carbon, energy, waste and water footprints.

3.4.3.2 ***Waste Minimization - Reduce Nitrogen Release to Watershed***

- increase understanding of effective fertilizer use

Fertilizer use in Nassau County is almost entirely recreational, in contrast to monitored nitrogen application of agribusiness elsewhere on Long Island. For the approximately 2/3 of residents who succumb to industry advertising, it is essential that they fertilize lawns at proper rates and times.

- intercept runoff with bioengineered landscape

Denitrification can increase at the aerobic/anaerobic interface of soils as nitrogen transport through the watershed is slowed through runoff abatement and vegetative absorption. Availability of idle real estate for bioengineering is at a minimum in Nassau County. Where possible, contouring and planting existing parcels, could be cost-efficient, provided it does not exacerbate vector-borne disease control efforts.

- sweep streets

In addition to conveying excess lawn fertilizer runoff, our streets also concentrate flux of nitrogen in the forms of wet and dry atmospheric deposition, vegetative litter and animal waste. Collection of particulates may be suitable to augment other composting materials if pathogens and pollutant levels not a health risk.

- compost vegetative waste

Effective use of vegetation cuttings can decrease the need to import lawn fertilizer into the watershed. Mulching lawn mowers and composting of lawn clippings can produce valuable gardening substrate. A balanced amount of humic material from lawn mulching can decrease runoff of excess fertilizer.

3.4.3.3 *Enhance Processing - Reduce Nitrogen Load into Waterbody*

- denitrify runoff in stormwater infrastructure

While engineered stormwater infrastructure does not offer a prime opportunity to remove nitrogen from the watershed, there may be beneficial configurations which slow the passage of, or store, pulses of runoff to the bay and in doing so may allow the buildup of organic material to promote denitrification.

- upgrade denitrification at WWTPs

Given the sewershed is nearly 100% connected, there may be little chance to install denitrification processes upstream of the major WWTPs. New construction permits could examine onsite systems. There may be cost-efficiencies associated with central plant upgrade options beyond the scope of this study.

- upgrade onsite septic systems

A small number of residences still rely on septic or cesspool systems. However, some of these locations are close to the waterbody and not proximate to denitrifying tidal wetlands or mudflats so any nitrogenous waste releases may enter the waterbody. These residences are not new, so upgrades are likely warranted.

- increase use of food industry waste as alternative fuel

Nearly half of the family food budgets are spent eating out of the home. Aside from toilet wastewater, food preparation establishments may add nitrogenous waste to WWTP influents in the form of grease and oil. Harvesting trap waste as a biofuel source can reduce the WWTP nitrogen load.

- expand tidal wetlands

Creation of tidal wetlands, particularly along the fringing locations, can offer a last line of defense to process stormwater runoff, while also offering a first line of offense against marginal storm surge. Tidal wetlands beneficial ecosystem services role in habitat restoration increase their net cost-competitiveness.

3.4.3.4 *Remedial Measures - Remove Nitrogen from Waterbody*

- decrease residence time by recontouring Hewlett Bay

Filling in the anthropogenic borrow pit would alter the circulation patterns by decreasing the residence time and sedimentation rates in Hewlett Bay. Water masses would no longer remain over as many tidal cycles. Suspended material would settle less and decrease habitat stress of sediment oxygen demand.

- decrease residence time by dredging inlets

Existing bathymetry results in the existing flux of ocean waters through the inlets and channels yielding the residence times which do not dilute the nitrogen load quickly enough to avoid eutrophic events. Further channel deepening can increase flux, however it may also risk marsh erosion and sediment loss.

- decrease residence time by ocean pumping

Pumping ocean water into the northern bays would alter circulation. This was done in Gowanus Canal. Pumping was considered and dismissed in Jamaica Bay. Energy requirements alone would challenge sustainability.

- increase export of effluent to ocean by discharging during ebb tides

This measure would decrease the amount of effluent that is transported into Hewlett Bay which has the longest residence times. This option would obviously require holding tank capacity nearly 50% of time.

- relocate WWTP effluent discharge outside of the Western Bays

Ocean outfalls operate effectively in the NY Bight from the Long Island and New Jersey coasts. An ocean outfall nearby was investigated as an alternative to abate stressful WWTP discharges into Jamaica Bay. Relocating to an ocean diffuser would entail tradeoffs beyond the scope of this Western Bays study area.

- harvest macro algae

Detached macro algae at sufficient density can be harvested for removal. A program has been underway for 10 years in shallow estuaries in Delaware. http://www.dnrec.state.de.us/macroalgae/default.shtml While harvesting may be physically possible, a regional market will be needed to incentivize its use.

- cultivate and harvest bivalves

Bivalves have been used to improve water quality in Jamaica Bay, Long Island Sound and NY Harbor. However, they would need to be harvested in order to remove nitrogen from the ecosystem. A regional market for shellfish uses which is protective of human health risks.

3.4.4 **Western Bays nitrogen management plan options decision model**

At the beginning of this section it was stated that nitrogen management options should be evaluated and compared based on the relative merit of their sustainability as cost-effective practices that significantly reduce the ecosystem stress of excess nitrogen. The Western Bays NMP options decision model (Table 8) offers distinct categorical measures, within each of the four strategies (source control; waste minimization; in situ processing; and removal) which can be taken to reduce nitrogen from various loads which enter the Western Bays ecosystem. An approximation of the overall nitrogen load into the waterbody is associated with each candidate measure (in two leftmost columns). The eleven criteria across the top of the table are assigned a suggested level of priority between low (1) and high (5). Five score levels are described for each criterion. Suggested score thresholds range from least (1) to most favorable (5). Each measure was scored to a first order approximation based on knowledge of the Western Bays current situation. The rank amongst each candidate NMP option (column 3) reflects the magnitude of its criteria scores multiplied by the criteria's weights within this representative scenario.

The NMP is expected to be tailored to numerous site-specific scenarios by identifying a subset of available NMP options within a given geographic scale and pursuant to a given source of funding. NMP tailoring could entail adding or subtracting criteria and their relative importance, and or adjusting the applicable nitrogen load budget, and or adjusting threshold levels, and or refining the scores within each cell of the model based on specialized knowledge.

Each of the 11 recommended criteria is intended to focus on a unique consideration which highlights strengths or weaknesses that may apply to nitrogen management measures. Percent of applicable load reflects the amount of nitrogen load into the waterbody which that measure can impact. This criterion is intended to capture the influence of magnitude. The load, and percents of load, can be recalculated at finer geographic scales in order for the model to increase the granularity of tradeoffs in site-specific scenarios. The next two model criteria are closely related. Achievable reduction in load reflects the conceptual feasibility that measure could reduce nitrogen loads within the Western Bays ecosystem, i.e. could it work in Nassau County? Implementability is the practical likelihood of success, i.e. if selected, will the measure actually fulfill its design objective because stakeholders will alter behaviors in order to make it work? Practical knowledge of which measured have already been implemented, exhaustively or partially, by municipalities and residents will enable this criteria to more accurately reflect site-specific tradeoffs. Cost range of magnitude (ROM) reflects the basic level of funding that would be associated with the measure. Costs might be incurred centrally or distributed amongst stakeholders. Cost share is the amount to which added costs could be satisfied by sources of funding beyond watershed residents and businesses. The threshold values of cost should be adjusted to reflect stakeholders' choices and tolerances for site-specific scenarios. Ecological risk denotes the possibility that implementation could degrade existing natural resource conditions. Engineering risk reflects the level of complexity associated with construction, operation and maintenance of the measure. Response time is the period necessary to begin to realize demonstrable reductions in ecosystem stress due to nitrogen loads. Permanence is the amount of time that a measure will have a beneficial impact. Prevention versus end-of-pipe favors measures which address the issue at the source instead of after the excess nitrogen impacts have cascaded through the ecosystem. Relevance to sustainability indicates whether the measure is integral to activities which are of prime societal importance or whether the measure requires redirection of attention, energy and resources away from primary and most common societal investments.

It is important to note that the nitrogen management plan options decision model scenario shown in Table 8 contains fixed assumptions about nitrogen loading into the Western Bays. The first criterion is percent of load which is pertinent to each measure's effect. These percentage ranges were derived from the watershed scale, bay-wide nitrogen loading budget described in Tables 5 and 6 in section 3 above. That nitrogen loading budget reflects very conservative assumptions about the amount of nitrogen imported into the watershed and loss due to denitrification or mineralization processes while cascading through the watershed to enter the waterbody. Of the total estimated load into the watershed (870tons N/yr), as much as 710tons N/yr could be removed from the system through cycling prior to entering Western Bays. The decision model uses a value of 590tons N/yr as the watershed load into the Western Bays, which is based on empirical measures of groundwater and surface water concentrations of nitrogen, plus an assumption of 0% denitrification in sediment. Conservative assumptions were made in the nitrogen loading budget in order to identify the broadest range of possible nitrogen management measures which could help reduce loads into Western Bays. One effect of this conservative approach is that nitrogen load reduction measures may actually have lower achievability and implementability if the actual amount of nitrogen attributed to that pathway is lower than estimated. This consideration applies mostly to measures that would mitigate non-point source runoff loads into the Western Bays because, for instance, there may not be as much lawn fertilizer applied as assumed, and there may be far more denitrification from groundwater as it seeps through the highly reactive sediment-water interface prior to entering the waterbody.

There are some inherent constraints to be aware of when using the decision model to mimic stakeholders' tradeoff discussions. The decision model can flex in multiple ways in order to reflect stakeholders' consensus on how they think the decision should be influenced. Criteria can be turned on and off by setting their priority levels from zero to five. However, criteria reflect individual considerations that are summed together for ranking. Summation of criteria is unable to portray fatal flaws, e.g. even putting in a priority of zero or a score of zero does not stop the measure from achieving a rank, even though a single consideration might be sufficient to be a considered a deal breaker. Thus, measures should only be evaluated with this tool if they are free of any fatal flaws. The gradient of possible scores within each criterion can be adjusted by redefining the thresholds assigned to each score level. Stakeholders may want to alter the thresholds to be linear or exponential depending on acceptability of each performance level. The decision model delivered herein is a flat spreadsheet with straightforward sums and products. The decision model can be implemented in multi-criteria decision model (MCDA) software which can allow for further details such as alternative score threshold distributions and uncertainty profiles which can increase the precision of ranks and the sensitivity criteria influences on the final rankings. The spreadsheet approach to the model, or even MCDA implementations, are not well suited to depict a portfolio approach to nitrogen management measures. Each measure is evaluated herein on its own merit relative to the other measures. This tool does not attempt to calculate tradeoffs of combinations of measures which could improve or detract from the effectiveness or scale of other measures. For example, a significant change to the Bay Park effluent load into Reynolds Channel would alter the assumptions behind the efficacy of removing nitrogen by harvesting accumulations of Ulva. Likewise, nitrogen removal through bivalve mariculture may lose the extremely limiting constraint of administrative closures if the outfall is relocated.

Table 8 . Nitrogen Management Plan options decision model for Western Bays. The ranks shown reflect an initial scenario of priority weightings and suggested scores by criteria for each measure. Stakeholders are invited to adjust priority weightings, Pct Loads, thresholds, and or scores for each option.

Western Bays Nutrient Management Plan Options v2.0

Applicable Load (tons)	Pct of Load	Priority Rank		Percent of Load Pertinent	Achievable Reduction in Load Allocation	Implement ability (% likelihood to succeed)	Cost ROM	Cost Share	EcoRisk	TechRisk	Response Time	Permanence	Prevention or Source Control vs. End of Pipe or Cleanup	Relevance to Sustainability (criticality to Population, Energy, Commerce, Food...)
			relative priority (1=low,3=mid,5=high)	5	5	5	5	5	3	3	3	3	1	1
			Criterion Score Thresholds 1	<10	<10	<10	100,000,000	muni tax/fee	High	High	10yr	day	cleanup	none
			2	<20	<20	<20	10,000,000					month	in situ stabilization	remotely
			3	<50	<50	<50	1,000,000	st&fed grants	Moderate	Moderate	5yr	year	instream reduction	partially
			4	<90	<90	<90	100,000	marginal incrs				decade	in situ elimination	directly
			5	>90	>90	>90	10,000	net0 or re-alloc	Low	Low	1yr	one & done	source elimination	multiply
			Reduce Nitrogen Import into Watershed											
57	2%	13	reduce combustion emissions NOx (power & tran	1	2	3	1	3	5	4	3	4	5	5
2,500	81%	2	improve nutrition & digestion to lower influent N	4	1	1	4	5	5	4	3	2	5	5
590	19%	6	educate & incentivize lower nitrogen footprints	2	1	1	4	4	5	5	3	3	4	3
			Reduce Nitrogen Release to Watershed											
298	10%	4	increase understanding of effective fertilizer use	1	2	2	4	3	5	5	4	3	3	3
617	20%	8	intercept runoff with bioengineered landscape	2	2	4	2	3	4	3	3	4	3	3
617	20%	9	sweep streets	2	2	3	4	1	5	4	5	2	2	1
617	20%	9	compost vegetative waste	2	1	3	4	1	5	5	5	2	2	3
			Reduce Nitrogen Load into Waterbody											
617	20%	5	denitrify runoff in stormwater infrastructure	2	3	3	2	3	5	3	3	4	3	3
2,500	81%	3	upgrade denitrification at WWTPs	4	3	5	1	2	5	3	1	4	3	4
5	0%	7	upgrade onsite septic systems	1	2	2	3	4	5	3	3	5	2	4
120	4%	9	increase use of food industry waste for fuel	1	2	3	3	3	5	3	3	3	4	5
2	0%	15	expand tidal wetlands	1	1	2	3	3	5	3	3	5	3	5
			Remove Nitrogen from Waterbody											
1,000	32%	15	decrease residence time, recontour of Hewlett B	3	3	4	2	1	3	2	3	5	1	1
2,800	90%	15	decrease residence time, dredging inlets	5	2	1	2	3	3	2	4	4	1	1
1,000	32%	19	decrease residence time, ocean pumping	3	1	2	2	1	3	1	2	5	1	1
1,250	40%	18	increase effluent dilution by ebb tide discharge	3	2	2	3	1	2	3	4	4	1	3
2,400	77%	1	relocate WWTP effluent discharge	4	5	5	1	3	2	4	1	5	1	3
10	0%	9	harvest macro algae	1	2	2	4	3	5	4	5	2	1	2
1	0%	13	cultivate and harvest bivalves	1	1	3	3	3	4	4	4	3	3	5

3.4.5 Commentary on select nitrogen management plan options

3.4.5.1 *Relocate Effluent Discharge*

The Western Bays NMP options model examined the tradeoffs associated with nitrogen management options within the watershed and waterbody. Relocation of the effluent discharge outside of the Western Bays ranked as the top option within the initial scenario and assumptions. The decision model did not attempt to integrate some of the larger scale tradeoffs such as those evaluated recently when an ocean outfall was considered as an alternative to decrease nitrogen loads into Jamaica Bay (NYC DEP, 2006). Larger scale considerations may include, for instance, regional consolidation of WWTP upgrades or effluents, northeast coastal-shelf nutrient dynamics, regional air quality concerns or ongoing WWTP infrastructure upgrades underway. The decision model also did not include criteria for numerous topics which are vital to human health or ecosystem health, but are beyond the immediate steps in the Western Bays watershed nitrogen fate and transport. Other topics may include, for instance, regional water supply or sea-level rise mitigation. The sustainability criterion introduced in the model generally warrants greater levels of detail within any particular scenario in order to reflect the full complement of Nassau County and regional stakeholders that calculate sustainability as a triple bottom line of economic viability, social services and environmental health.

Upgrades to WWTP denitrification capacity ranks amongst the highest in the NMP Options decision model based on the amount of source load removal relative to costs and risks. The denitrification upgrade scenario is included in this watershed-wide model as a means to depict order of magnitude comparisons. Further details based on engineering specifications can increase the model's precision and accuracy of the WWTP infrastructure options.

3.4.5.2 *Nutrition and Digestion*

Numerous state and federal regulatory programs aimed at water quality improvements promote, if not require, development of drainage canals with flora to cultivate microbes that process nutrients in stormwater runoff. Breakthroughs in genetics and microbiology now enable, if not compel, an equivalent emphasis on programs that promote understanding of the health of our alimentary canals' flora to cultivate microbes that process nutrients in ingested food. This is not analogous; it is factual. A nutrient management plan for the Western Bays should consider the role of improving nutrition and digestion of residents as a means to decrease the load of nitrogenous waste entering our wastewater treatment systems.

"…Ten times more nitrogen is used to produce food than humans consume as protein, and not all the nitrogen in the food we eat is even used by our bodies—the excess enters the environment through human waste. And while most people require only 2 grams of nitrogen a day, the average American consumes 13 grams daily. The Recommended Daily Intake of protein is approximately 50 grams or less than 2 ounces, but the average American consumes 8 ounces of meat daily…" (Cho, 2011). The nutrients ingested per mass of protein consumed is analogous to vehicular miles per gallon. Some consumers switch to higher miles per gallon vehicles motivated by reducing transportation costs, not necessarily to reduce atmospheric deposition of nitrogen. Western Bays stakeholders should consider whether consumers' habits could switch to products that enable higher nutrition per protein ingested motivated by reducing food costs, regardless of the benefits to reducing nitrogenous wasteload.

Nutrition and digestion promptly impact fecal composition and production, and thus the amount of nitrogenous waste flowing to our WWTPs. The amount of nitrogen in feces is of utmost importance to this nutrient management plan option. Tilg and Kaser (2011) summarize numerous studies of animals and humans which describe how nutrition and digestion can impact alimentary microbiota (Figure 43) and the amount of nitrogenous waste produced.

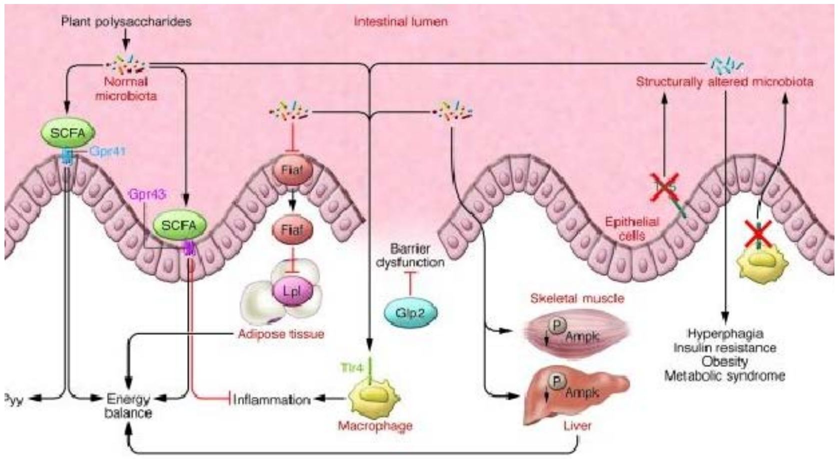

Figure 43 . Gastrointestinal microbiota changes can alter lipid and protein uptake (Tilg and Kaser, 2011) and thus nitrogenous waste content.

Symbiosis between microbiota and the host can be optimized by simple, over-the-counter pharmacological or nutritional interventions in the gut microbial ecosystem using probiotics or prebiotics (World Gastroenterology Organization, 2008). Efficient digestion and assimilation of proteins affects both our metabolism, health and fitness, as well as the nitrogenous wastes we supply to WWTPs or onsite sanitary systems. The Natural Resources Conservation Service of the US Department of Agriculture developed Feed Management Practice Standard 592 to increase the efficacy of animal nutrition and to reduce nitrogen and phosphorus in their waste.

We produce nine times more nitrogenous waste from our digestion, each of us, then from all of our other nitrogen loads into the Western Bays. And our personal ability to reduce nitrogen in our digestive waste is possibly greater than our ability to further improve elimination or interception of nitrogen beyond the existing non-point source control measures. Thus, on the basis of mass of nitrogen loaded into the Western Bays, consideration should be given to the benefits and costs of education campaigns aimed at improving residents' nutrition and digestion. "Changing our diets and changing our consumer habits could significantly reduce nutrient inputs, greenhouse gas emissions, agricultural land use and cause positive health impacts." (Costello, 2013)

Education campaigns to improve nutrition and digestion may be familiar to those of us pursuing non-point source outreach programs aimed at fertilizer application practices. Given that many active U.S. programs for water quality credit trading still struggle with quantification of the site-specific efficacy of non-point sources control measures, yet these programs award legally binding permits based on the principles of best management practice and best available technologies, the notion of investing in public education campaigns about nutrition and digestion as a means of reducing waste generation at its point of origin seem compelling, albeit novel and apart from the regulatory disciplines and auspices of the agencies which typically advance coastal waterbody protection efforts.

3.4.5.3 *Reduce Excessive Lawn Fertilizing*

Effective use of fertilizer is the 4th highest ranking NMP option in the initial model scenario. Optimal amounts and scheduling of lawn fertilizing, when done in concert with soil conditioning and proper watering, can yield growth without excessive nitrogen release due to runoff or leaching. A third or more of residents don't fertilize at all, and some who do fertilize may do so effectively based on existing public education materials. The next level of reduction in nitrogen load will need to target those who are apparently worry less about downstream consequences and more about green lawn aesthetics. However, passive outreach may need to be strengthened in light of ongoing pressure from horticultural industrial marketing. The top goal on page 2 of the Scotts company's 2012 annual report reads as follows:

> ***Increasing our advertising investment and introducing new marketing messages to consumers.*** We planned for a significant increase in our investment in paid media in the United States in order to support a more aggressive marketing effort. The changes we implemented included new advertising campaigns to support both the Scotts® and Miracle-Gro® brands. We also focused on significantly improving our digital advertising and marketing efforts. The analysis conducted throughout the season indicates the campaigns had a high level of consumer awareness and strong persuasion scores. Additionally, we made significant improvements in the amount of traffic driven to our web site and other digital content.

Figure 44 . Excerpt from Scotts Annual Report 2012.

Given that the Scotts company alone spent $170,000,000 on advertising in 2012, and given that their 2013 slogan shouts "Feed Yer Lawn, Feed it!" It seems imperative that a public education campaign help fertilizer buyers understand how to perform that feeding when the lawns won't drool.

3.4.5.4 *Bioengineered Landscape and Stormwater Infrastructure*

Nitrogen loading into the Western Bays can be reduced by enabling vegetative uptake and or denitrification. Vegetation buffers and porous soils enable these processes if they are located to intercept surface water runoff. Some stormwater conveyance equipment captures sediment and detritus in storage vaults which also can enable denitrification.

However, "…The effectiveness of conventional stormwater treatment Best Management Practices (BMP) is unclear, and can be particularly dependent on site conditions and maintenance. Thus, it is worthwhile to examine and test efficient and effective alternatives..." (Long Island South Shore Estuary Reserve, 2006). BMP effectiveness can be misleading if applied inappropriately, especially with regard to the pollutant targeted. Stormwater BMPs may offer little effect on nitrogen fate though they may be extremely effective at intercepting suspended sediment or eliminating pathogen transport. For example, in Nassau County's 2007 Stormwater Runoff Impact Analysis Procedures Manual, wherein Cashin & Associates summarized data from the Center for Watershed Protection's Urban Subwatershed Restoration Manual (Table 9), all engineered BMPs for stormwater management have been shown to be least effective at removing nitrogen in comparison higher efficiencies removing other targeted pollutants.

Table 9 . Percent efficiency of stormwater management practices to remove six types of pollutants (from Center for Watershed Protection's Urban Subwatershed Restoration Manual).

	TN	TSS	TP	Bacteria (F. Coliform)	Oil and Grease[2]	Trash[3]
Stormwater Dry Ponds	**25**	**50**	**20**	**35**	**70**	**80**
Quality Control Pond	5	3	19	-	-	-
Dry Extended Detention Pond	31	61	20	-	-	-
Stormwater Wet Ponds	**30**	**80**	**50**	**70**	**80**	**90**
Wet Extend Detention Pond	35	80	55	-	-	-
Multiple Pond System	N/A	91	76	-	-	-
Wet Pond	32	79	49	-	-	-
Stormwater Wetlands	**25**	**70**	**50**	**60**	**75**	**90**
Shallow Marsh	26	83	43	-	-	-
Extended Detention Wetland	56	69	39	-	-	-
Pond/Wetland System	19	71	56	-	-	-
Submerged Gravel Wetland	19	83	64	-	-	-
Filtering Practices	**30**	**85**	**60**	**40**	**85**	**90**
Organic Filter	41	88	61	-	-	-
Perimeter Sand Filter	47	79	41	-	-	-
Surface Sand Filter	32	87	59	-	-	-
Vertical Sand Filter	5	58	45	-	-	-
Bioretention	49	N/A	65	-	-	-
Infiltration Practices	**40**	**90**	**65**	**40**	**90**	**90**
Infiltration Trench	42	N/A	90	-	-	-
Infiltration Basin	50	90	70	90	75	
Pervious Pavement	83	95	65	-	-	-
Dry Ponds/Recharge Basin [1]	60	90	65	90		
Underground Dry Infiltration [1]	60	90	65	90		
W. Q. Channels/ Swales	**55**	**80**	**25**	**0**	**80**	**0**
Ditches	9	31	16	5	-	-
Grass Channel	NA	68	29	-	-	-
Dry Swale	92	93	83	-	-	-
Wet Swale	40	74	28	-	-	-

These efficiency estimated are incorporated into the scores for implementability (i.e. likelihood to succeed) criterion of the Western Bays nitrogen management plan decision model. A related criterion, achievable reduction in load allocation, reflects the extent to which each nitrogen management measure could be taken above and beyond the existing efforts and built-out condition of the watershed. These ratings can and should be adjusted when evaluating the tradeoffs between site-specific options.

3.4.5.5 *Nitrogen Footprint Amongst Our Other Footprints*

Who ever thought the public could be convinced to drive less? Not to buy more fuel efficient cars, simply persuaded to use their combustion engines less. In fact this trend is occurring and contributing to decreases in nitrogen deposition from vehicle emissions. Although it is unclear what influenced this behavioral change, it is yielding an apparently unintended beneficial side-effect of decreasing the inputs into the nitrogen cascade. Such changes in cultural behavior support the notion that water quality improvement goals can, and should, be advanced through public conversations, such as nitrogen footprint factors, which complement existing resources being dedicated directly to water quality permitting, regulation and litigation debates.

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APPENDICES

APPENDIX A: NUTRIENT ASSESSMENT AND MANAGEMENT IN SHALLOW COASTAL WATERS OF HEMPSTEAD BAY

Nutrient Assessment and Management in Shallow Coastal Waters of Hempstead Bay

DRAFT FINAL REPORT

Christopher J. Gobler, Ph.D., and Ryan Wallace, M.S.

Stony Brook University

School of Marine and Atmospheric Sciences

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EXECUTIVE SUMMARY:

This report describes the spatial (vertical and horizontal) and temporal (seasonal) variability of light, temperature, salinity, dissolved oxygen, chlorophyll, nitrogen, phosphorus, phytoplankton community composition, and the nutrient limitation of phytoplankton and *Ulva* in the Hempstead Bay. This project further describes the spatial (vertical and horizontal) variability of *Ulva*, its nitrogen isotopic content, and the factors that control its growth in Hempstead Bays. During 2012, there were substantial differences in water quality and algal ecology across the Western Bays. While the regions near the Jones Beach Inlet and East Bay were generally found to have good water quality, Middle Bay and West Bay and Hewlett Bay displayed multiple signs of eutrophication and impairment. Middle Bay, West Bay, and Hewlett Bay experienced extremely high levels of dissolved inorganic nitrogen and phosphorus and dense algal blooms (>50 µg chlorophyll a L^{-1}). These algal blooms were experimentally shown to be promoted by nitrogen. *Ulva* populations were found to be dense in West Bay and less abundant in East Bay. The isotopic composition of *Ulva* was matched that of sewage through much of the Bay, but was indicative of mixed nitrogen sources in some locations. The dilution of nitrogen from Bay water by ocean water proved capable of significantly reducing phytoplankton and *Ulva* growth rates. It is suggested that the reduction of standing DIN levels by 10µM would lessen the intensity of algal blooms in this ecosystem, although a larger reduction would be required to fully remediate this ecosystem. Collectively, these findings indicate that severe nitrogen loading from the Bay Park sewage treatment plant is likely the root cause of algal blooms, *Ulva* blooms and hypoxia in the Western Bays which in turn threaten multiple ecosystem services there.

BACKGROUND

Hempstead Bay is a eutrophic water body and is officially an impaired water bodies (303d) as declared by the NYS Department of Environmental Conservation. Dense accumulations of *Ulva* have been observed in many locations throughout the Western Bays with multiple negative ecosystem impacts. Sewage treatment plants are discharging directly into the Western Bays. Prior to this study, very little was known regarding hypoxia, HABs, and water quality in these systems. Almost nothing was known regarding the precise mechanisms responsible for the impairment of water quality in the Western Bays. This study addresses these issues.

ANALYTICAL METHODS

Phytoplankton Analyses

To maximize our understanding of the potential relationship between phytoplankton community composition, N cycling, and oxygen demand, we will strive to robustly characterize phytoplankton populations within this system. The distribution of phytoplankton biomass was estimated by the triplicate analysis of chlorophyll *a* using 0.2µm polycarbonate filters and standard fluorometeric techniques (Parsons et al. 1984). Selected duplicate 20 mL Lugol's - preserved plankton samples were settled in counting chambers and enumerated on an inverted light microscope (Hasle 1978). Settled microphytoplankton (> 10 µm) were generally grouped into major classes (i.e., diatoms, dinoflagellates, nanoflagellates, ciliates, etc.).

Nutrients Analyses.

Whole water samples were analyzed by an ELAP lab. In addition, whole water samples were filtered for nutrient analysis at SoMAS using pre-combusted (2 h @ 450°C) glass fiber filters (GF/F filters with nominal pore size of 0.7 µm). Filtrate was colorimetrically analyzed for a suite of inorganic nutrients (nitrate, nitrite, ammonium, orthophosphate, silicate) nutrients using standard methods (Parsons et al. 1984). Analysis of samples spiked with nutrient standard reference material will be incorporated into the sample processing procedures as a quality control measure.

Nitrogen sources assessment

Macroalgae was dried and encapsulated in tin discs and analyzed on a Europa 20/20 mass spectrometer, providing the precise nitrogen content and the ^{15}N signature for materials. Macroalgae isotopically resemble the source of nitrogen that they assimilate and algae with heavy N isotopic signatures can reflect their use of wastewater derived nitrogen **(**Cole et al. 2004). Other nitrogen sources yield differing signatures [e.g., fertilizer nitrogen has a very low signature (Cole et al. 2004). As such, examining the ^{15}N signature for macroalgae along spatial and temporal gradients in Hempstead Bay will provide an indication of the regions in Hempstead

Bay where sewage discharge has the greatest contribution to algal growth, as well as an indication of when primary producers are the most and least enriched in sewage-derived N

Nutrient enrichment effect on primary producers

A goal of the Hempstead Bay TMDL may be to reduce nutrient inputs in Hempstead Bay. However, there are dynamic events such as combined sewage overflow or stormwater runoff which can result in increases in nutrient levels to Hempstead Bay. In addition, sewage treatment plants may experience increases in nutrient loads or discharge during periods of construction, upgrading, and/or maintenance (Robert Nymann, USEPA, pers. comm.). To establish the effects of nutrient enrichment on primary producers in Hempstead Bay, nutrient amendment experiments with phytoplankton were conducted. To ascertain the extent to which nutrients may influence algal communities during this project, triplicate sets of Hempstead Bay water were amended with nitrate (50 µM), silicate (50 µM), phosphate (3 µM), or were left unamended as a control treatment. The concentrations of these additions are similar to previously observed increases of these nutrients in the water column of Hempstead Bay (NYCDEP 2007), and ratios are consistent with the Redfield stoichiometry. Experimental bottles (1-L, polycarbonate, acid-washed) were incubated for 48 h at near ambient temperature (achieved by incubating bottles in Shinnecock Bay) and ambient light levels (achieved by neutral density screening) at the Stony Brook-Southampton marine station. After 48 h, changes in levels of chlorophyll were determined. Significant differences in growth rates of all algal populations were assessed using a one-way ANOVA where nutrient source is the main treatment effect. Previous research has demonstrated that growth rates calculated from these proposed nutrient enrichment experiments should provide a clear indication of which nutrients are limiting the accumulation of phytoplankton biomass (Gobler et al. 2002; 2004; 2006).

Determination of nutrient reductions required to mitigate algal blooms

In hypereutrophic systems such as Hempstead Bay, there is often an excess of nutrients exceeding the requirements for algal growth. To estimate which nutrient could be limiting in this hypereutrophic system and to determine the levels of nutrient reduction required to reduce phytoplankton and macroalgal biomass, nutrient dilution bioassays were employed(Paerl and Bowles 1987). Nutrient dilution bioassays dilute out the major nutrients (nitrogen and phosphorus) to concentrations where phytoplankton and macroalgal growth will become limited or reduced compared to unamended controls (Paerl and Bowles 1987). The reduction in algal biomass relative to the nutrient reduction will allow the impact of future reductions in nutrient loads to the estuary to be assessed.

To create conditions similar to ambient conditions with respect to all variables other than nutrient concentrations, whole bay water was diluted with Atlantic Ocean water which contains very low levels of dissolved inorganic nitrogen. For initial assays, there were a series of treatments run in triplicate, whereby Hempstead Bay seawater will be progressively diluted with ocean in greater concentrations: 0%, 25%, 50%, 75%, 90%. A parallel set of bottles were established using filtered bay water to account for microzooplankton grazing. The precise series of dilutions were refined as experiments are repeated, allowing a more precise determination of the nutrient reduction needed to lower algal biomass. For macroalgae, parallel sets of modified bottles filled with filtered seawater will be used instead of whole seawater, and this was diluted with ocean water using the dilution series described above. Each bottle had a 3 cm x 3 cm square of *Ulva lactuca*, freshly obtained from Hempstead Bay and washed free of epiphytes added. Experimental bottles were incubated at near ambient temperature (achieved by incubating bottles in Shinnecock Bay) and ambient light levels (achieved by neutral density screening) at the Stony Brook-Southampton marine station. Phytoplankton experiments were incubated at ambient light and temperature conditions for 72 hours in Old Fort Pond at the Stony Brook Southampton Marine Science Center as described in Gobler et al., (2002), after which aliquots were filtered for levels of chlorophyll ***a***. Net growth rates of each population will be determined as follows: $\mu = \ln [N_t / N_o] / t$ where μ is the rate of population growth (d^{-1}), N_o and N_t are initial and final cell densities, and t is the duration of incubation in days. Microzooplankton grazing rates within Bay water were determined according to Landry et al (1995) and the growth rates of phytoplankton within each dilution were corrected for the reduced grazing rate at these dilutions. *Ulva* bottles were incubated for approximately 7 days, after which changes in area and biomass of *Ulva* were quantified. The effect of each nutrient on each population was analyzed with a one-way ANOVA with each nutrient considered a treatment effect ($\alpha = 0.05$). Post-hoc comparisons of significant impacts were elucidated with Tukey's multiple comparison tests. Experiments were performed in 2010 and 2012 to understand how increasing and decreasing concentrations of dissolved inorganic nitrogen in the Western Bays would alter phytoplankton growth rates.

RESULTS

Sea surface and bottom temperatures followed an expected temperate seasonal pattern with temperatures increasing to maximum values during the summer months (Figs 2 & 3). Sea surface temperatures during the summer of 2012 ranged from 25.86 °C at station MB in July to 19.36 °C at station EB in late September (Fig 2c & 3a). Bottom water temperatures ranged from 25.07 °C at station MB in July to 19.36 °C at station EB in late September (Fig 2c & 3a). The most intense temperature stratification occurred in Hewlett Bay during August (Figs 2a). Summer surface and bottom water temperatures were consistently higher at stations BP, USGS and MB than temperatures at stations EB and JBI whereas fall and winter temperatures were more uniform among sites (Figs 2 and 3). The summer temperature differences suggest that EB is well-flushed with cool, ocean seawater whereas BP, USGS and MB are not.

Surface and bottom salinity followed a trend of increasing salinity from west to east (BP < USGS < MB < EB < JBI; Figs 4 and 5), which can be attributed to tidal mixing with higher salinity ocean waters entering through Jones Beach Inlet. Given there are not any major tributaries within the northern basin, this salinity pattern suggests that Hewlett Bay is the poorest flushed basin, followed by Middle Bay, and then East Bay which has a salinity not substantially lower than Jones Beach Inlet, suggesting it is tidally well-flushed. During the summer of 2012, surface salinity ranged from 31.76 psu at station JBI to 29.07 psu at station BP in late September (Fig 4a & 5b). Bottom salinity ranged from 31.74 psu at station JBI in August to 29.43 psu at station BP in July (Fig 4a & 5b). The most intense salinity stratification occurred at station BP during the month of September (Fig 4a).

The depth that at least 1% incoming photosynthetically active radiation (PAR) penetrates in the water column can be considered the euphotic zone as most phytoplankton and macroalgae need at least 1% PAR for survival (Sand-Jensen 1988) whereas seagrasses need more (20%; Dennison et al. 1993). Light penetration is heavily controlled by phytoplankton biomass in the water column. The depth at which 1% light reached was consistently shallowest in Hewlett Bay and during our September cruise Hewlett Bay had less than 1% PAR reaching the benthic region (Fig 6). During this entire study, 1% PAR never reached the benthic region of Hewlett Bay. Hewlett Bay has been heavily dredged and is also the deepest station sampled. The combination of depth and high phytoplankton biomass precludes the growth of benthic macroalgae in this deep basin, confining the growth of macroalgae to shallower regions in the marsh and close to shore where > 1% PAR can penetrate to the bottom (Fig 6). Moreover, it demonstrates that Hewlett Bay does not have enough light to support the growth of seagrass beds, an important impairment within this water body. Middle Bay suffers from high density phytoplankton blooms, but the water column is shallow and as seen in the September cruise, > 15% PAR can penetrate to the bottom causing large macroalgal populations to proliferate throughout the year. Although regions surrounding Jones Beach inlet have lower turbidity than points west, the depth prohibits enough light from penetrating to the benthic region (Fig 6) and it is unlikely that large populations of benthic macroalgae will survive in this region. During the September cruise, which was conducted during a period of very low phytoplankton biomass throughout the entire Western Bays, there was no station in which > 20% PAR reached the benthic region (Fig 6) and this would inhibit the survival of any seagrass beds. Beyond light limitation, it is well known that seagrasses do not proliferate in estuaries with heavy nutrient loads (Taylor et al. 1999; Bintz et al. 2003) and this combination of factors are likely to prohibit the growth of seagrasses through the Western Bays.

The Western Bays experienced extremely dense phytoplankton blooms during the summer of 2012, with peak levels exceeding 50 µg chlorophyll *a* L^{-1} (Fig 7). The temporal dynamics of phytoplankton biomass was similar across the entire Western Bays region, although intensity of these blooms was greatest in Hewlett and Middle Bays (Fig 7). Mean chlorophyll *a* concentrations in Hewlett Bay at station BP were >35 µg L^{-1} which exceeds levels found in almost all of New York State's South Shore Estuary Reserve (SCDHS 1976-2012; Fig 7). The sites with the second highest mean concentrations of chlorophyll *a* were MB and USGS (~20 µg L^{-1}; Fig 7) followed by EB in East Bay and Jones Beach Inlet (JBI) which generally had low levels of chlorophyll (<10 µg L^{-1}; Fig 7). These high density blooms in Hewlett Bay and to a lesser extent in Middle Bay and the USGS station occurred in the late spring and were maintained throughout the summer months, eventually decreasing in late September (Fig 7). Phytoplankton comprising these blooms included diatoms, dinoflagellates, autotrophic nanoflagellates, and raphidophytes (Fig 8a-e). Diatoms dominated the phytoplankton community in the entire Western Bay region during the summer of 2012 (Fig 8a-e) and cell densities gradually declined into the early fall at stations BP, MB and USGS (Fig 8a-c). The highest diatom cell densities at stations EB and JBI occurred during August (Fig 8d & e). Station BP consistently had the highest diatom cell densities, which peaked during July, 2012 (> 60 x 10^3 cells mL^{-1}). Phytoplankton blooms in Hewlett Bay during the spring and late summer were dominated by the harmful raphidophyte, *Heterosigma akashiwo*, with cell densities reaching extremely high levels (> 2.5 x 10^4 cells mL^{-1}). In early September of 2012, *Heterosigma akashiwo* again dominated the phytoplankton community in Hewlett Bay (> 7x10^3

cells mL^{-1}; Fig 8a). This species is known to be ichthyotoxic and has been associated with finfish kills worldwide (Lewitus et al. 2012).

One consequence of heavy nitrogen loading, algal blooms, and the decay of these blooms can be low oxygen content of seawater, a condition also known as hypoxia. When algal blooms die, they sink below the euphotic zone and significant bacterial decomposition of algal biomass and oxygen consumption will occur. This will decrease dissolved oxygen within the lower water column and oxygen will not be resupplied until mixing occurs. Although surface dissolved oxygen concentrations were generally healthy in the Western Bays (Fig 9), hypoxia was frequently recorded within Hewlett Bay during this study. Bottom dissolved oxygen concentrations were below the New York State acute standard of 3 mg L^{-1} during September of 2012 (Fig 10 & 11). In previous years, hypoxic/anoxic conditions were observed in the bottom layer in Hewlett Bay throughout the entire summer. While Middle Bay experienced high density phytoplankton blooms, hypoxic conditions were never reached during the summer of 2012 (Fig 9 & 10). Middle Bay is much shallower than Hewlett Bay and thus ventilates more easily with the atmosphere and does not experience extensive hypoxia. However, oxygen levels < 4 mg L^{-1} were observed at station MB during September of 2012 (Fig 10). When surface waters begin to warm in the summer months, temperature stratification increases and this traps cooler water at the bottom of deeper regions within Hewlett Bay where microbial respiration consumes oxygen. Stations in East Bay and Jones Beach Inlet did not experience any hypoxic conditions at any point during this entire study (Fig 9 & 10). Vertical profiles of oxygen levels demonstrated that in the deeper portions of Hewlett Bay, hypoxic water persisted at depths below 7.5 m while surface waters had higher oxygen levels (Fig 11), Anoxic conditions were, on occasion, observed below 8 m depth (Fig 11). The low levels of oxygen found within Hewlett Bay were likely driven, in part, by rapid rates of microbial respiration.

Excessive nutrient loading into coastal ecosystems such as the Western Bays promotes algal productivity and the subsequent microbial consumption of this organic matter lowers oxygen levels and contributes toward hypoxia. A second, often overlooked consequence of microbial degradation of organic matter is the production of CO_2 and reduction in pH associated with that process. To assess the potential for eutrophication-driven acidification in Hewlett Bay, the pH levels were vertically mapped in September of 2012. Measurements revealed that the pH levels in Hewlett Bay were below 7.1 in the bottom layer and below 7.6 through much of the water column (Fig 12). These low pH conditions could have been caused by tidal influx of treatment plant discharge and the subsequent enhanced bacterial decomposition. These levels of pH have previously been shown to yield elevated mortality in larval finfish and shellfish (Talmage and Gobler 2010; Baumann et al. 2012) suggesting that acidification, which has been intensified by climate change (Doney et al. 2009), may be currently altering the ability of Hewlett Bay to support robust fisheries.

The concentrations of dissolved ammonium, nitrate, phosphate and silicate in the water column were dynamic throughout the entire study, but were generally lowest when phytoplankton biomass was high. In the summer of 2012, dissolved nutrient concentrations were highest in the western region of the Western Bays (stations BP, USGS, MB; Figs 13-16). East Bay and Jones Beach Inlet had consistently lower dissolved nutrient concentrations (Figs 13-16). Dissolved nitrate concentrations in July at station BP in Hewlett Bay were low during July (~7 μM) and increased to ~15 μM by late September (Fig 13). Ammonium concentrations in Hewlett Bay were < 1 μM in July and increased to ~70 μM by late September (Fig 14). Phosphate concentrations in Hewlett Bay increased from 2.5 to 3.5 μM from July to September (Fig 15) and silicate concentrations increased from 5 to 45 μM (Fig 16). Dissolved nutrient concentrations generally increased during the late summer and early fall as phytoplankton biomass decreased and dissolved nutrients were remineralized in the water column (Figs 13-16). To more accurately assess the spatial distribution of dissolved nutrient concentrations in the Western Bays, horizontal mapping of dissolved nutrient concentrations was conducted during September of 2012. Dissolved nitrate concentrations were highest in Hewlett Bay (> 18 μM) and decreased to the south and east (Fig 17). The lowest concentrations were in East Bay (< 3 μM). Ammonium, phosphate and silicate concentrations had a spatial distribution that was similar to nitrate (Figs 18-20). The highest ammonium concentrations were found in the western region (> 50 μM) and the lowest concentrations were in the east (< 3 μM; Fig 19). Phosphate concentrations ranged from < 1 μM in the east to ~4.5 μM in the west (Fig 19) and silicate concentrations ranged from > 45 μM in Hewlett Bay to < 5 μM in the east (Fig 20). High concentrations of nitrate, phosphate and ammonium were found in close proximity to the Bay Park sewage outfall and in areas with extended residence times. Regions to the east have shorter residence times and lower rate sewage derived nutrient input, a fact well illustrated by surface salinity and temperature distributions across the Western Bays (Fig 21 & 22). Salinity is lowest in Hewlett Bay and higher salinity ocean water is found east of Jones Beach Inlet (Fig 21). Highest temperatures are also found in Hewlett Bay and the lowest east of Jones Beach Inlet (Fig 22). High silicate concentrations in Hewlett Bay are likely the result of both water column and benthic remineralization of siliceous tests.

The experimental loading of nitrogen was found to significantly enhance phytoplankton growth rates in nearly all summer experiments performed within Hewlett Bay, Middle Bay, East Bay and Jones Beach Inlet ($p < 0.05$; Fig 23 & 24). The strongest growth response was generally found within locations with the highest levels of chlorophyll *a* and thus the largest nutrient demand (i.e. Hewlett Bay and Middle Bay). These results demonstrate that algal blooms within this region are controlled by nitrogen loading during summer. In the spring, it is possible that the phytoplankton community in Hewlett Bay is sometimes limited by the supply of silicate (Fig 25). This may occur when DIN is high and there is a diatom bloom in which silicate will be rapidly assimilated by the diatom community. By the end of the summer nitrogen increases and there was no significant response to any single treatment (Fig 26).

Another symptom of excessive nitrogen loading in the Western Bays is blooms of the macroalgae, *Ulva* sp. In the fall of 2011, the percent bottom coverage of *Ulva* across the Western Bays exceeded 60% within the region of the Bay Park sewage treatment plant discharge in Reynolds Channel and shallower areas in Hewlett Bay, but was lower to the east within the Western Bays (Fig 27). We found very little *Ulva* in East Bay and South Oyster Bay (~10%; Fig 27). In the summer of 2012, the percent bottom coverage of *Ulva* across the Western Bays was similar to the spatial coverage observed in 2011 (Fig 28). There was over 50% coverage in the Middle Bay area and east of Jones Beach Inlet there was less than 5% bottom coverage (Fig 28). The most striking difference between the two years is the lack of coverage within Hewlett Bay (Fig 28) a region with excessive nutrients. This can be attributed to depth in the water column. Hewlett Bay is heavily dredged and *Ulva* needs at least 1% PAR to grow (Sand-Jensen 1988).

Light does not reach the benthos within the deeper areas of the Bay. In 2011 cruise, stations in Hewlett Bay were in shallow regions; deeper stations were added in 2012 providing a more accurate interpretation of benthic macroalgal coverage.

$\delta^{15}N$ signatures can be used to identify anthropogenic nitrogen sources in many organisms including macroalgae (Cole et al. 2004). Tissue samples collected in the Western Bays had heavier $\delta^{15}N$ signatures (Fig 29) demonstrating that sewage effluent is the dominant N source to *Ulva* in this region (Valiela et al. 1992; Cole et al. 2004). Interestingly, lighter $\delta^{15}N$ signatures were found in western region of this system (Fig 29). These tissue samples were collected shortly after significant freshwater runoff associated with August storms and Hurricane Irene which significantly decreased salinity within Hewlett Bay. The western portion of the Western Bays has a number of golf courses and fertilizers have a very light $\delta^{15}N$ signature. As such, the mixed $\delta^{15}N$ signal may be due to this increased runoff during a period of intense rainfall.

Ulva sp. nutrient dilution experiments were conducted to determine if the growth rates of *Ulva* would change when incubated in progressively higher dilutions of filtered ocean seawater (FOSW). There was no significant growth difference between any of the dilution treatments (Figs 30-32), but *Ulva* growth rates always increased significantly when enriched with nitrogen. *Ulva* is known to harbor large stores of nutrients which may buffer their response to the different dilution treatments over short time scales (Steffensen 1976; Bjornsater and Wheeler 1990; Viaroli et al. 1996) perhaps accounting for the lack of response during these nutrient dilution experiments.

Algal kinetics and nutrient reduction experiments

Nutrient dilution experiments demonstrated the very strong control that dissolved inorganic nitrogen (DIN) concentrations had on phytoplankton growth rates in Hewlett Bay and in Middle Bay (Figs 33, 34). Specifically, during all experiments performed in these bays between May and September, decreasing DIN led to a linear reduction in phytoplankton growth rates (Figs 33, 34). Through the study, phytoplankton growth rates were reduced by 0.01 to 0.03 per day for every one micromolar reduction in concentrations of DIN (Table 1). The mean net growth rates of phytoplankton populations in Hewlett Bay and in Middle Bay over a three year period were found to be 0.08 and 0.14, respectively (Table 1). Positive net growth rates result in large algal blooms. As such, a reduction in mean DIN concentrations in Hewlett Bay and Middle bay of 4 and 7 µM would reduce the mean summer net growth rates to zero and would ostensibly lessen the intensity and impact of algal blooms in this ecosystem. Given the very large variability in the growth rates of phytoplankton (-0.7 to 1.2 per day; Table 1), these small reductions would lower the intensity, but larger reductions would be required to more fully prevent algal blooms in this system.

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Fig 1 Water quality sampling stations in Western Bays from 2010-2012.

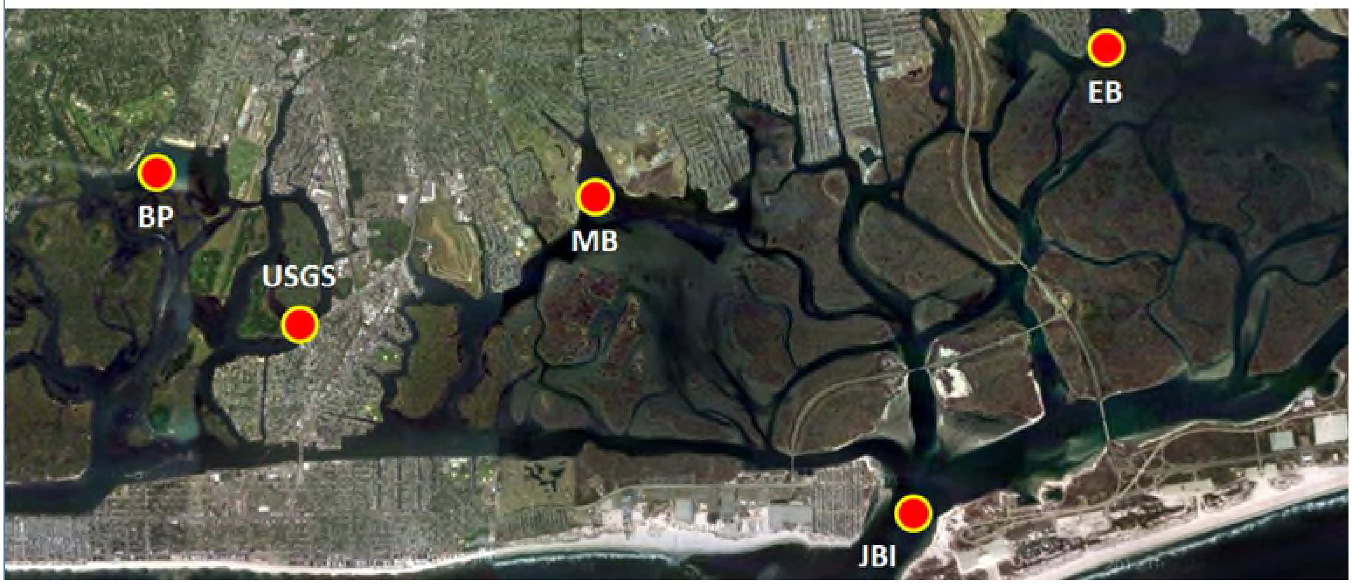


Fig 2 Sea Surface and Bottom temperatures (°C) at three stations in the Western Bays during the summer of 2012.

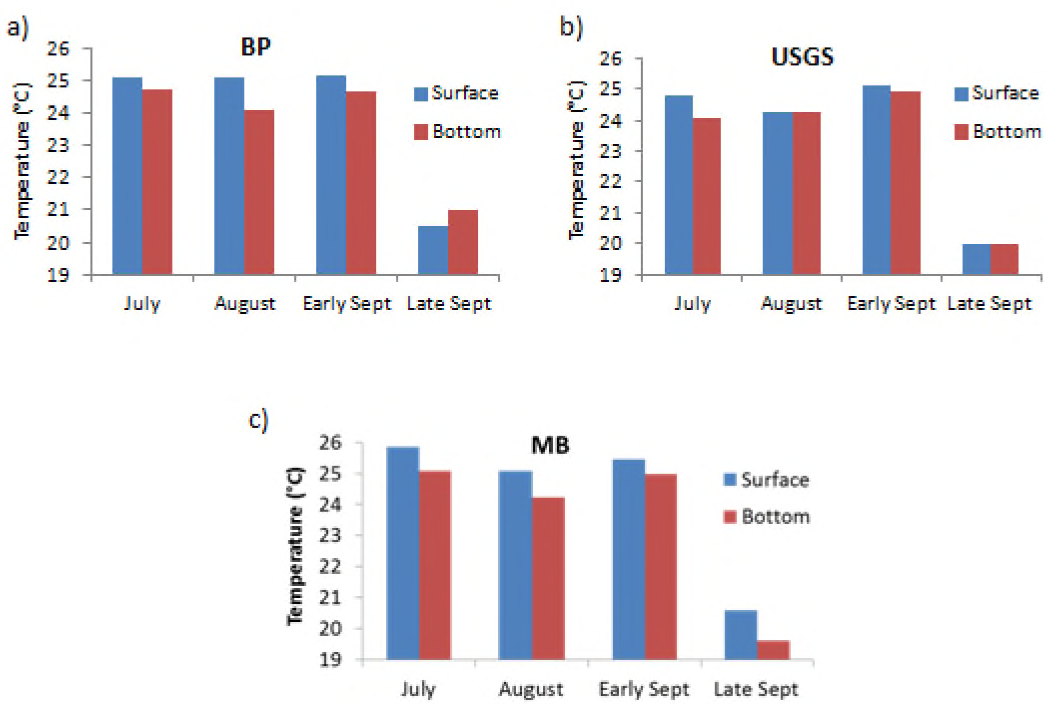


Fig 3 Sea Surface and Bottom temperatures (°C) at two stations in the Western Bays during the summer of 2012.

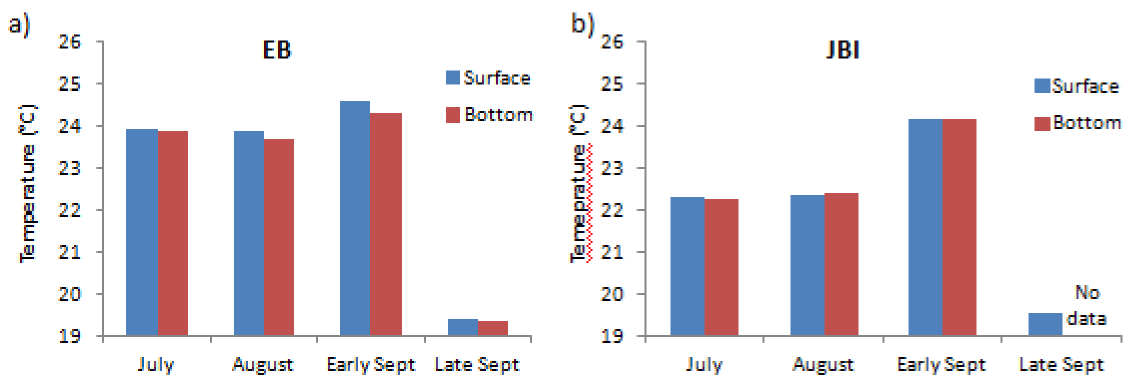


Fig 4 Sea surface and bottom salinity (psu) at three stations in the Western Bays during the summer of 2012.

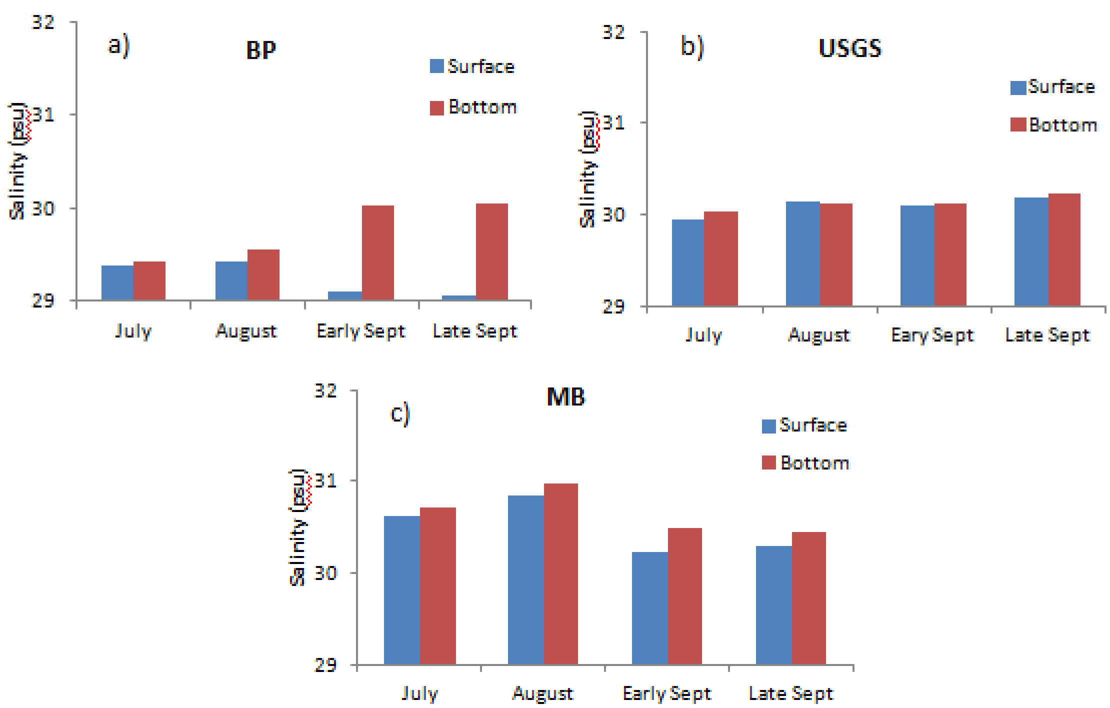


Fig 5 Sea surface and bottom salinity (psu) at two stations in the Western Bays during the summer of 2012.

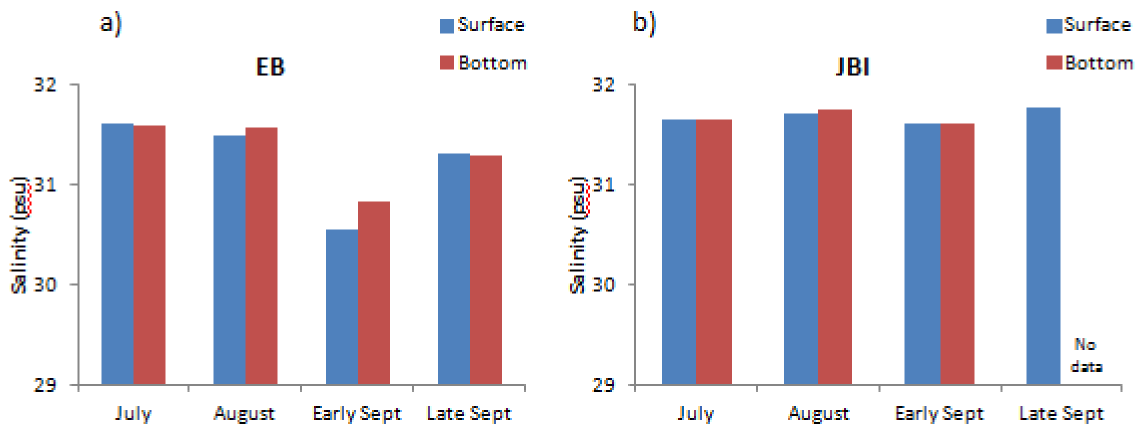


Fig 6 Percent incoming photosynthetically active radiation (PAR) which reaches the bottom during September of 2012.

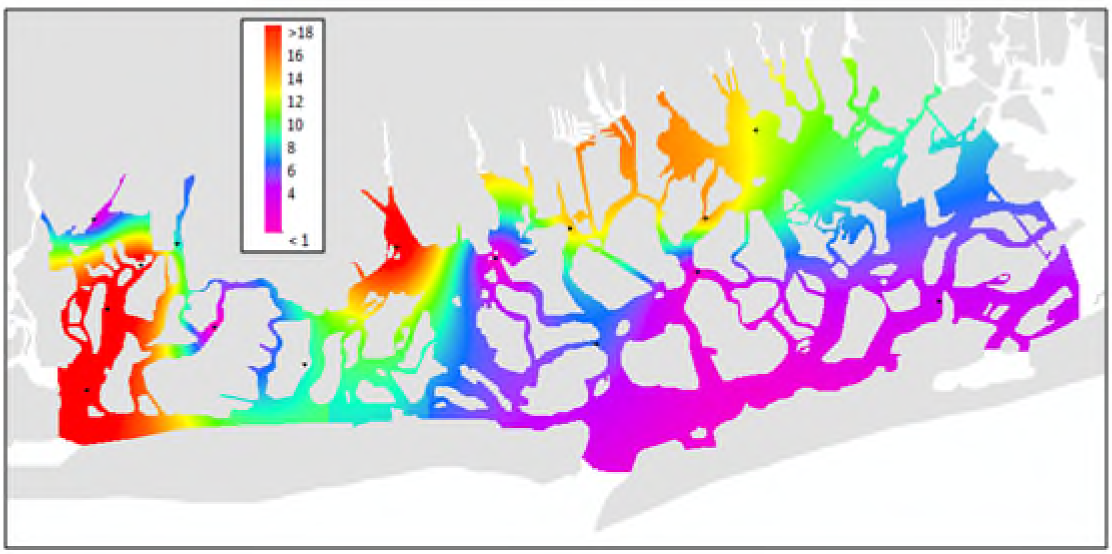


Fig 7 Total chlorophyll *a* concentrations ($\mu g\ L^{-1}$) at five stations across the Western Bays during the summer of 2012.

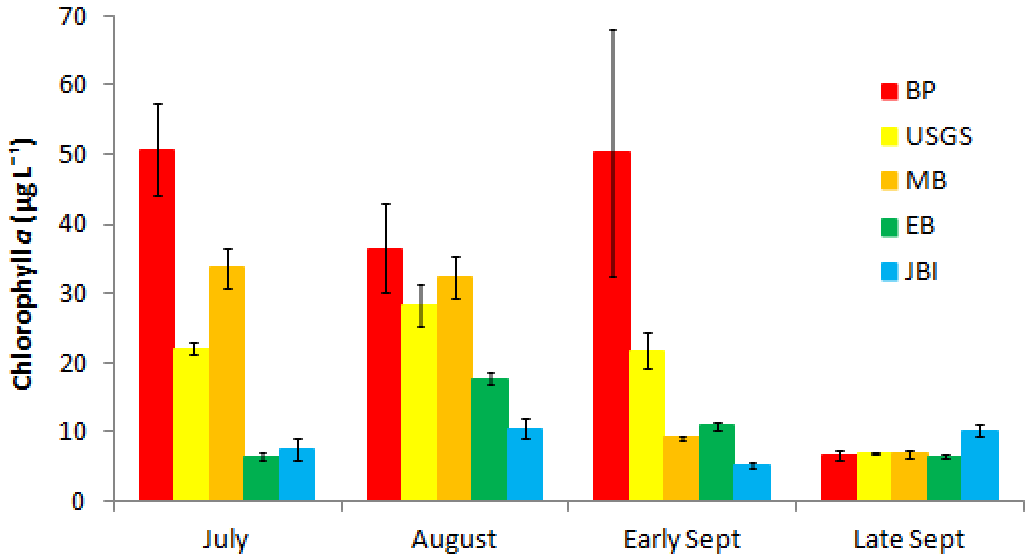


Fig 8a Plankton community composition ($x10^3$ cells mL^{-1}) at station BP in Hewlett Bay in 2013.

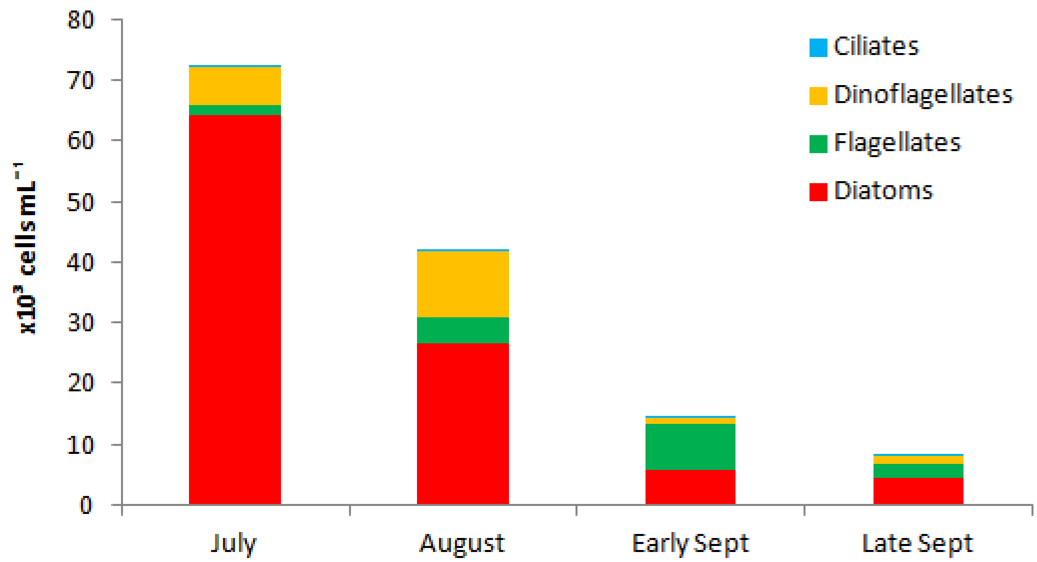


Fig 8b Plankton community composition ($x10^3$ cells mL^{-1}) at station USGS in 2013.

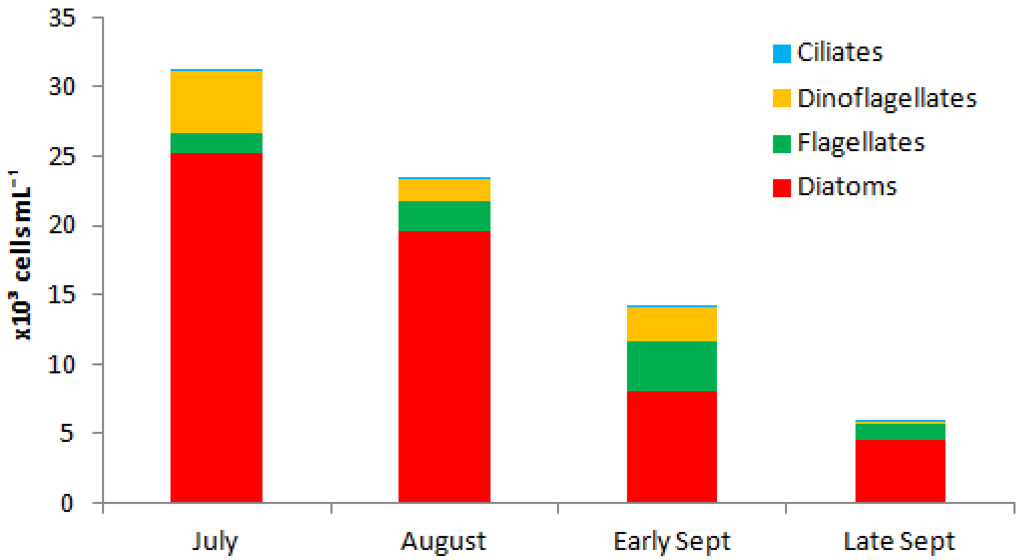


Fig 8c Plankton community composition ($x10^3$ cells mL^{-1}) at station MB in Middle Bay in 2013.

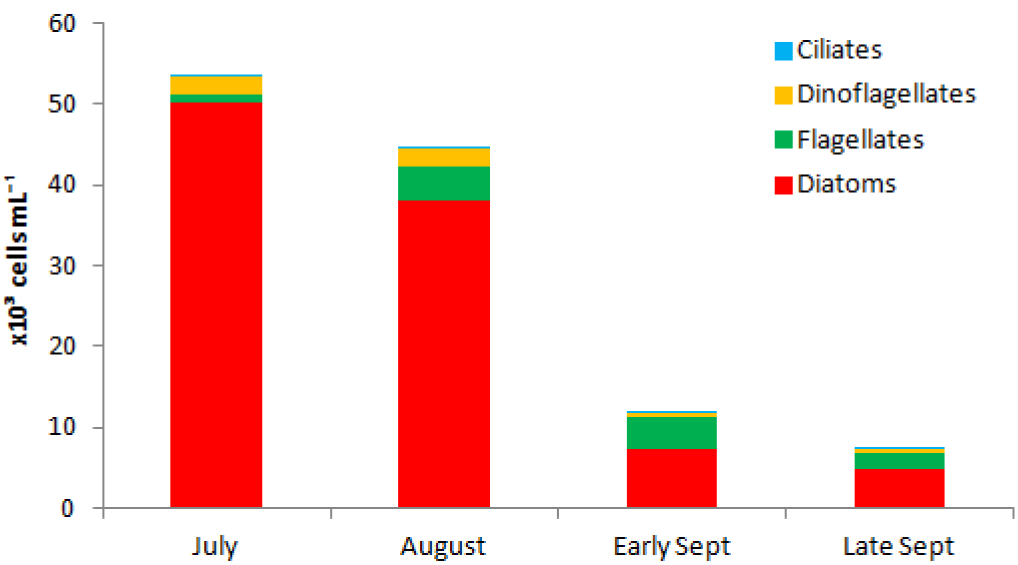


Fig 8d Plankton community composition (x10^3 cells mL^{-1}) at station EB in East Bay in 2013.

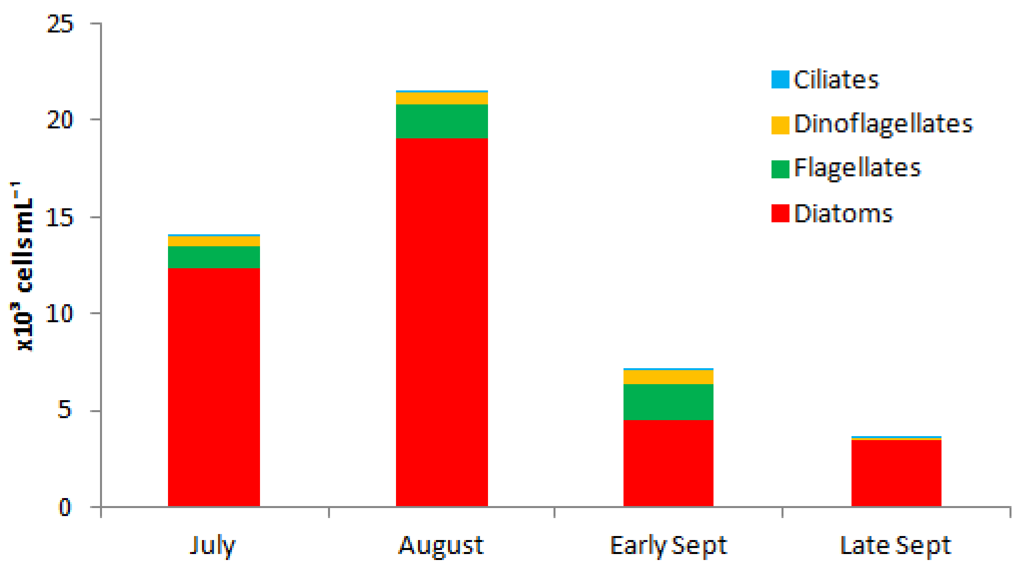


Fig 8e Plankton community composition (x10^3 cells mL^{-1}) at station JBI in Jones Beach Inlet in 2013.

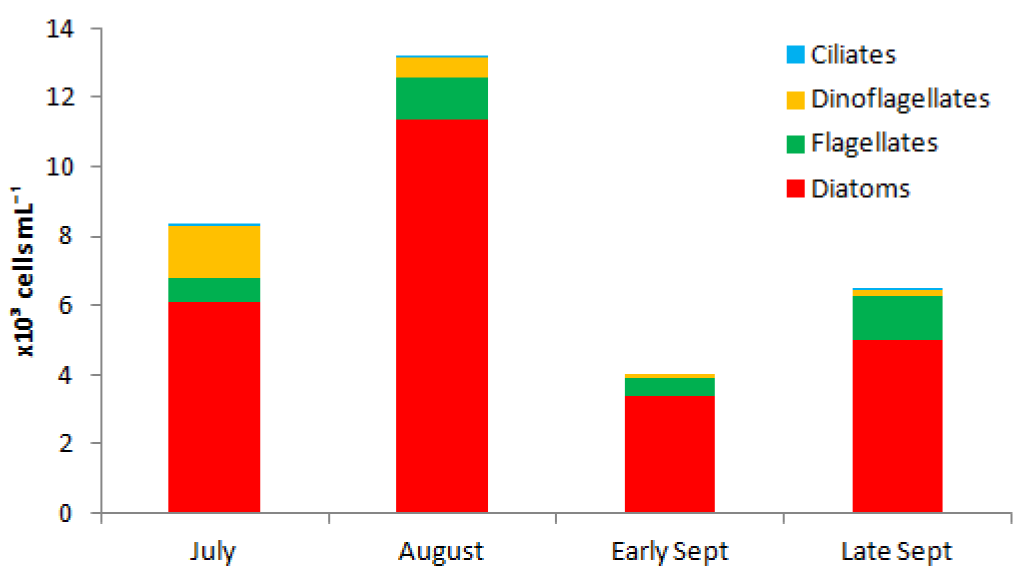


Fig 9 Surface dissolved oxygen concentrations (mg L^{-1}) at 5 stations BP in the Western Bays during the summer of 2012. Dashed line represents New York State acute standard for dissolved oxygen

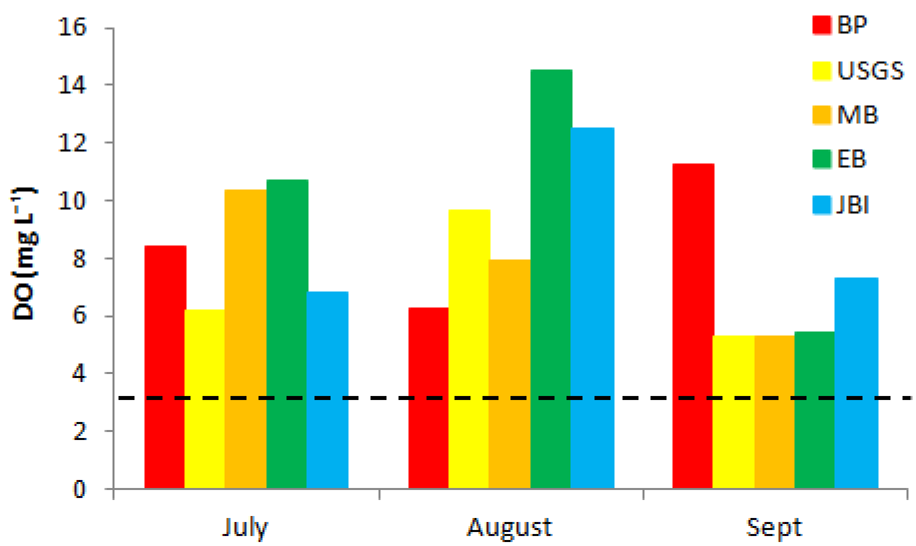
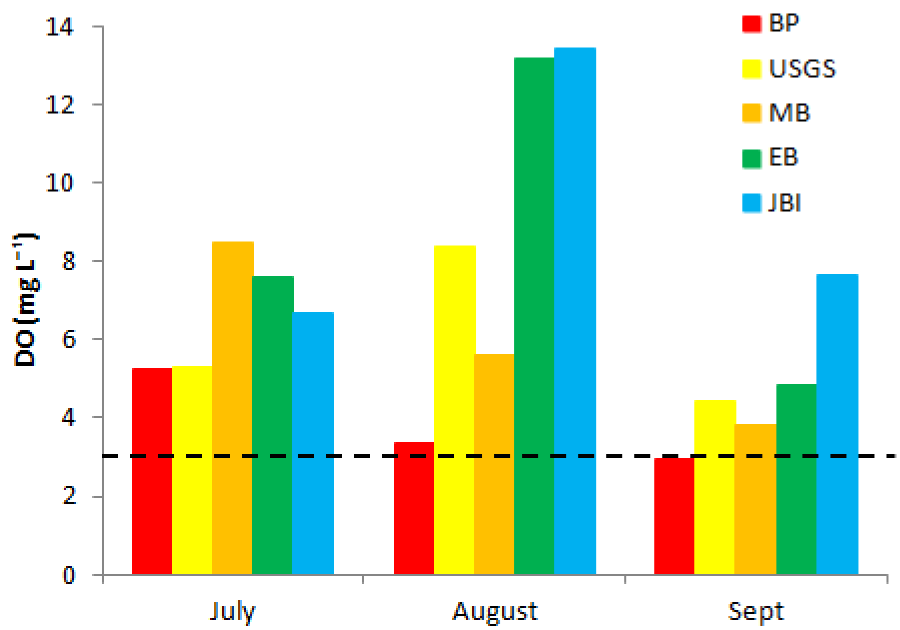


Fig 10 Bottom dissolved oxygen concentrations (mg L^{-1}) at 5 stations in the Western Bays during the summer of 2012. Dashed line represents New York State acute standard for dissolved oxygen



Appendix A

Fig 11 Vertical cross section profile of dissolved oxygen concentrations (mg L^{-1}) with salinity (ppt) contour lines in Hewlett Bay during early September of 2012.

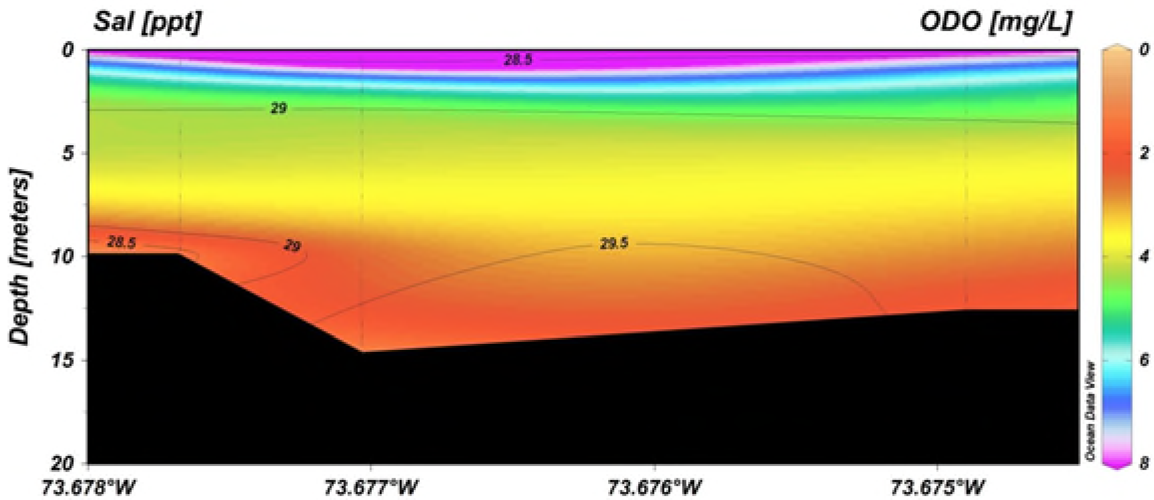
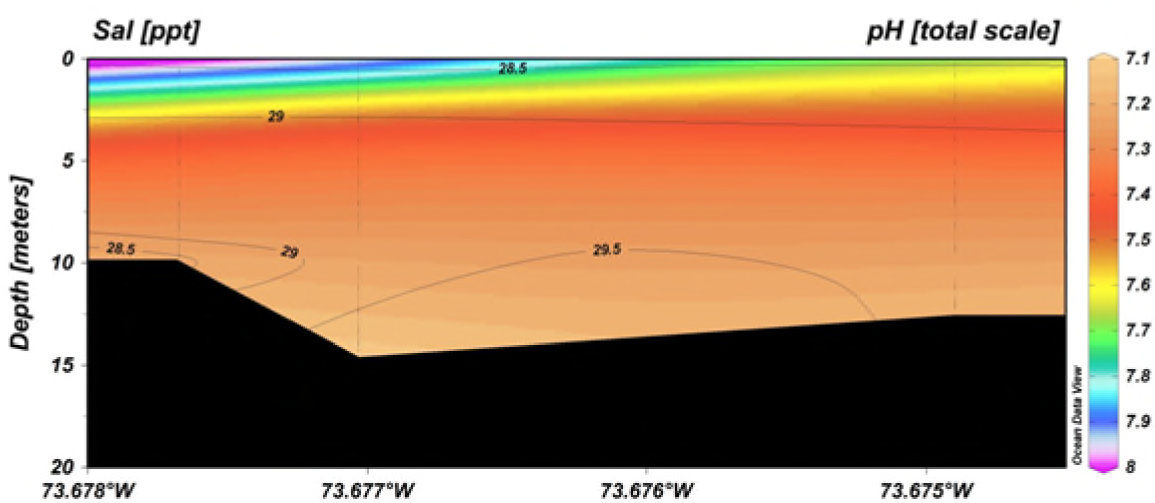


Fig 12 Vertical cross section profile of pH (total scale) with salinity (ppt) contour lines in Hewlett Bay during early September of 2012.



Appendix A

Fig 13 Dissolved nitrate concentrations at five stations (µM) across Western Bays from July through September, 2012.

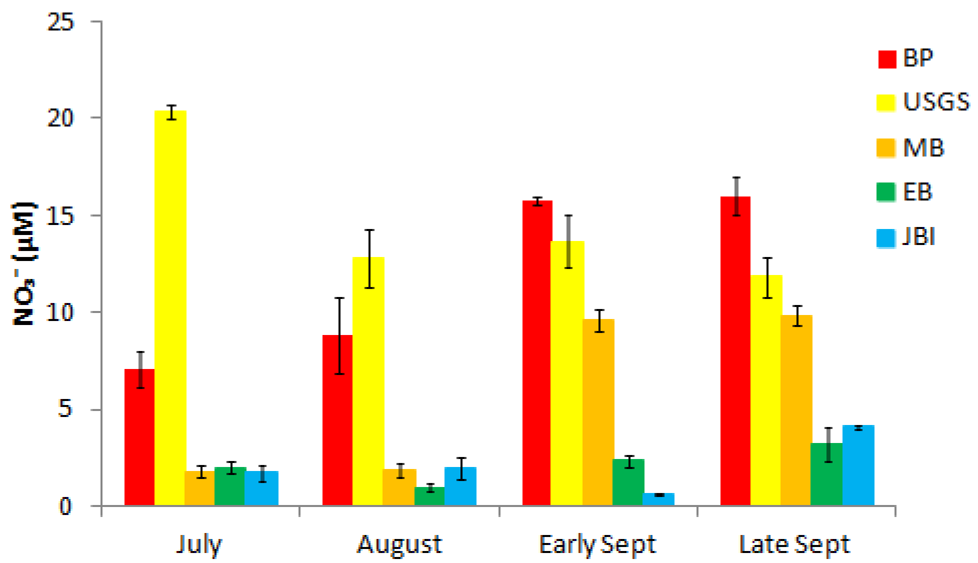


Fig 14 Dissolved ammonium concentrations at five stations (µM) across Western Bays from July through September, 2012.

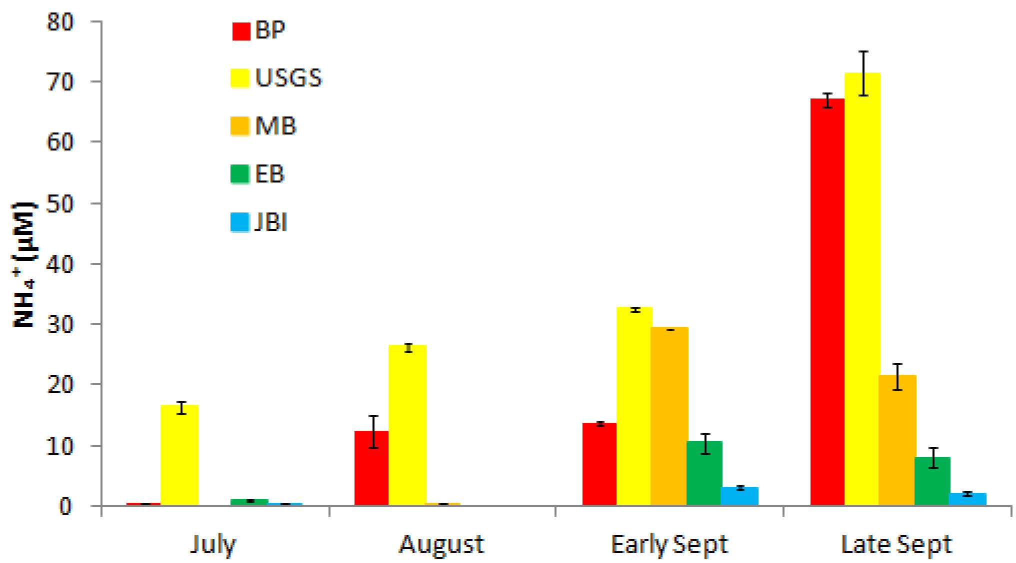


Fig 15 Dissolved phosphate concentrations (µM) at five stations across Western Bays from July through September, 2012.

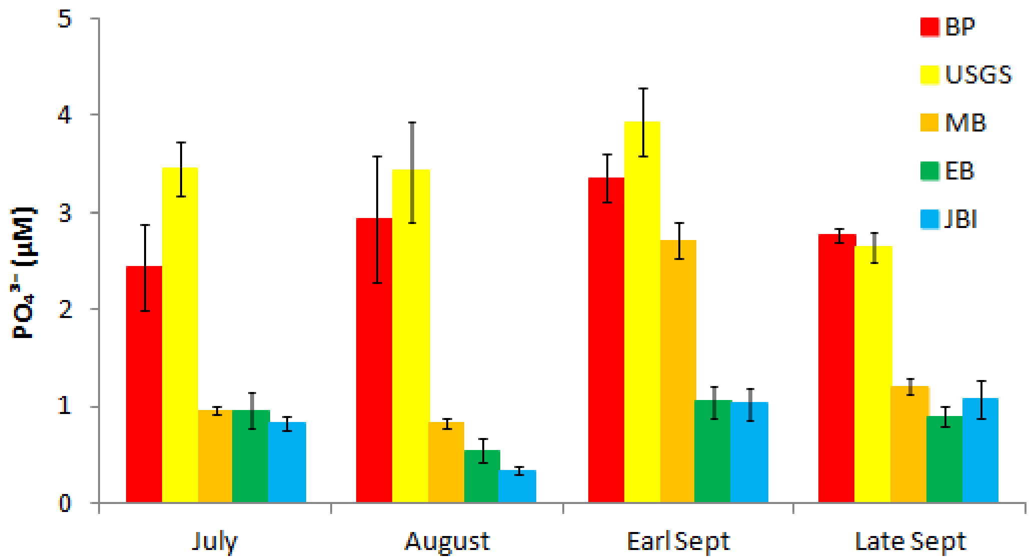


Fig 16 Dissolved silicate concentrations (µM) at five stations across Western Bays from July through September, 2012.

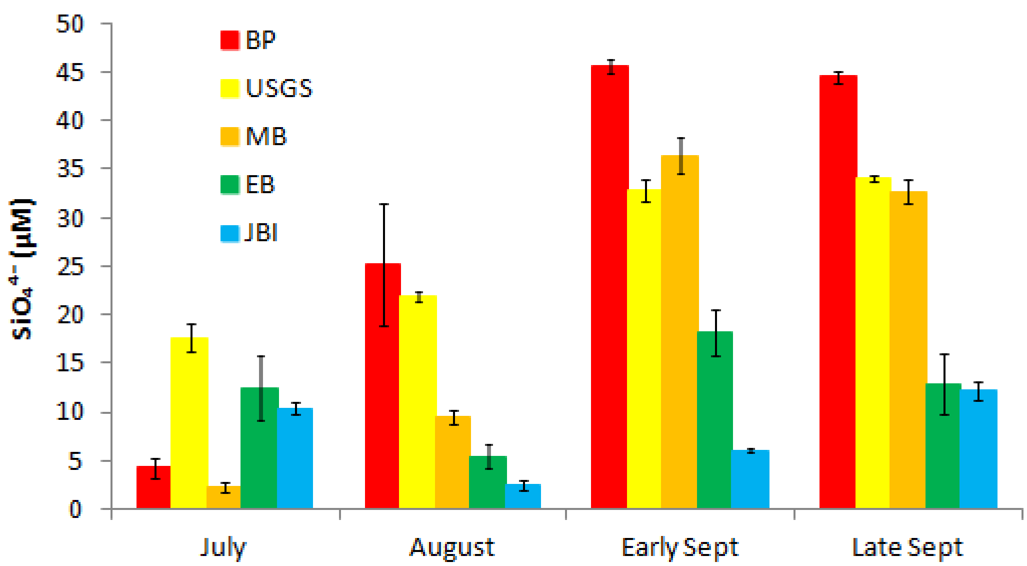


Fig 17 Dissolved nitrate concentrations (µM) across Western Bays during late September, 2012.

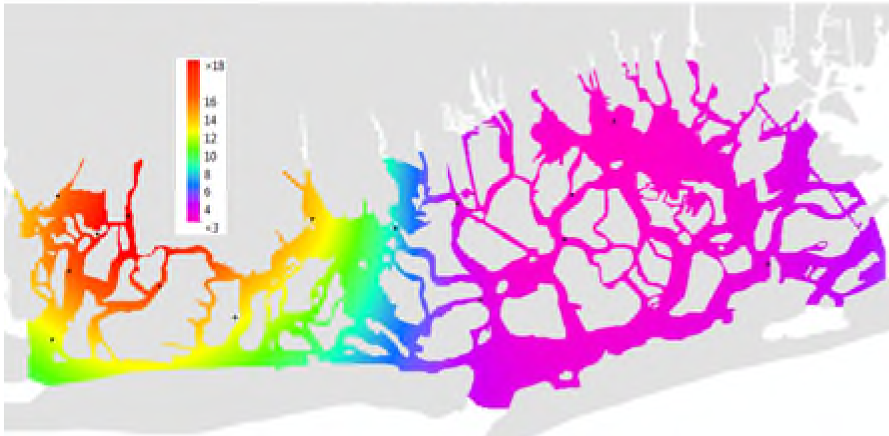


Fig 18 Dissolved ammonium concentrations (µM) across Western Bays from during late September, 2012.

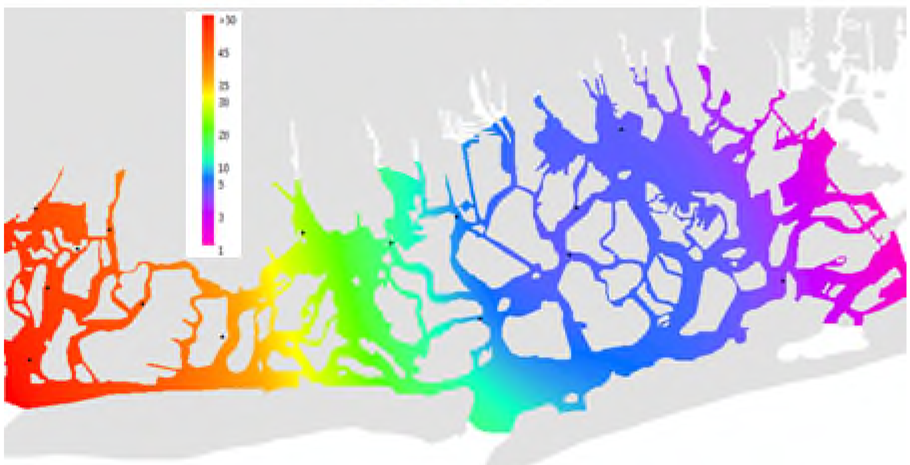


Fig 19 Dissolved phosphate concentrations (µM) across Western Bays in late September, 2012.

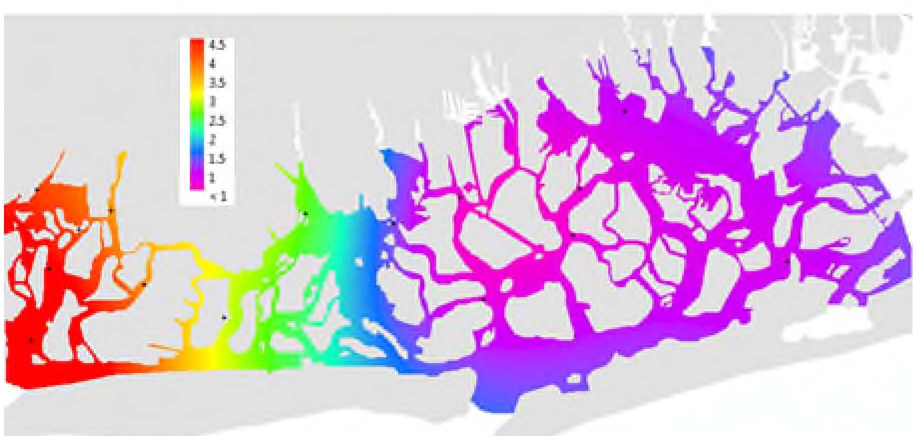


Fig 20 Dissolved silicate concentrations (µM) at five stations across Western Bays from July through September, 2012.

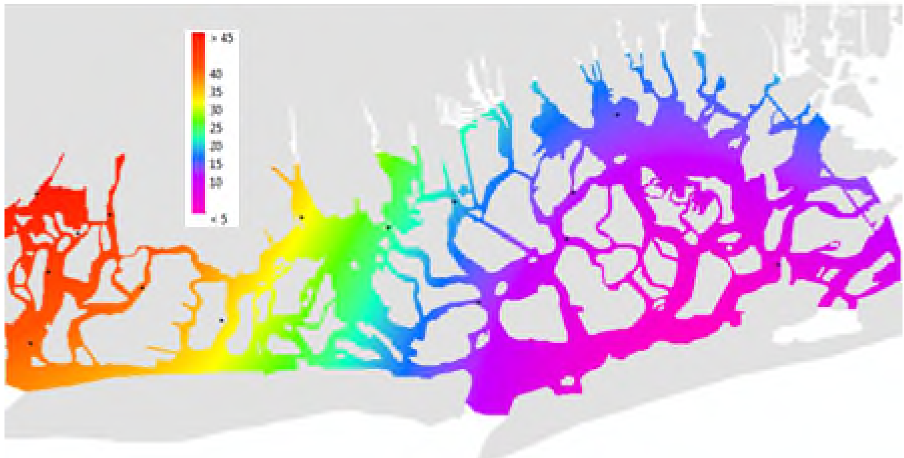


Fig 21 Sea surface salinity (psu) in the Western Bays during September of 2012.

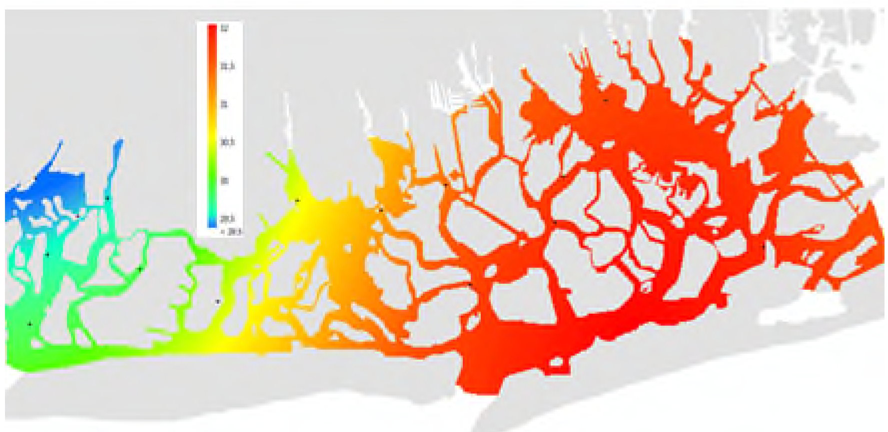


Fig 22 Sea Surface temperatures (°C) in the Western Bays during September of 2012.

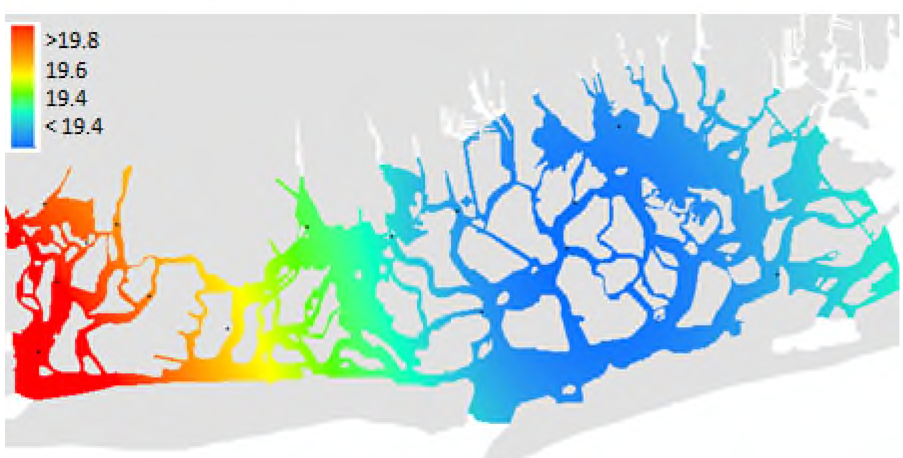


Fig 23 Effects of experimental nitrate, phosphate, and silicate loading on the entire phytoplankton population in Hewlett Bay in late May, 2010.

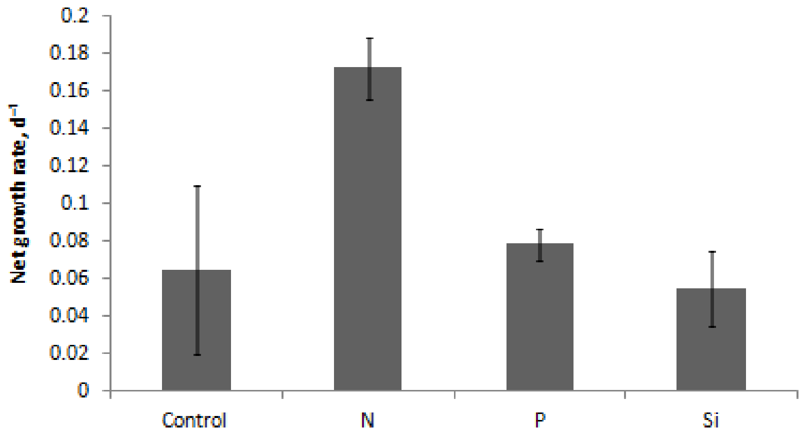


Fig 24 Effects of experimental nitrate, phosphate, and silicate loading on the entire phytoplankton population in Hewlett Bay in June, 2010.

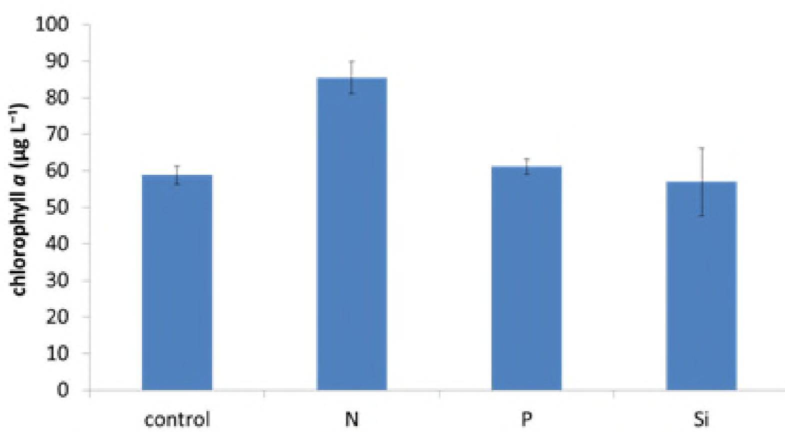


Fig 25 Effects of experimental nitrate, phosphate, and silicate loading on the entire phytoplankton population in Hewlett Bay in early May, 2010.

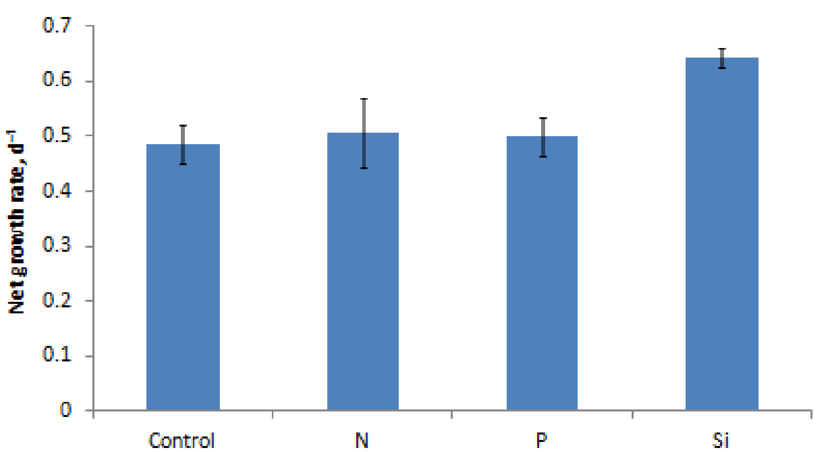


Fig 26 Effects of experimental nitrate, phosphate, and silicate loading on the entire phytoplankton population in Hewlett Bay in early August, 2010.

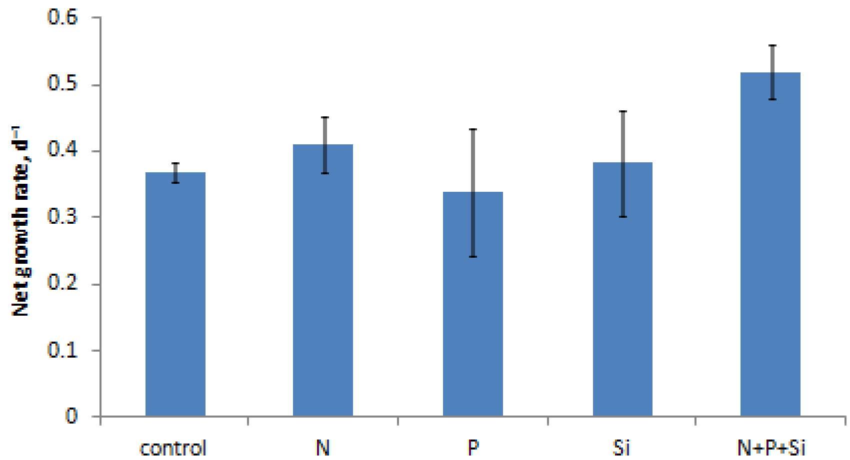


Fig 27 Percent bottom coverage of *Ulva* sp. in Western Bays in the Fall of 2011.

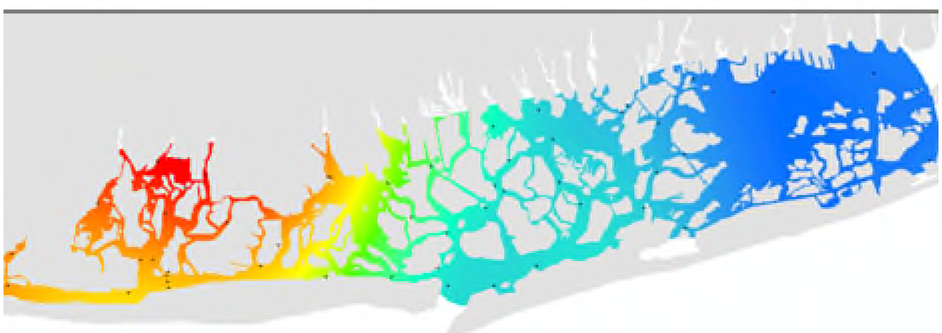


Fig 28 Percent bottom coverage of *Ulva* sp. in Western Bays in the Fall of 2012.

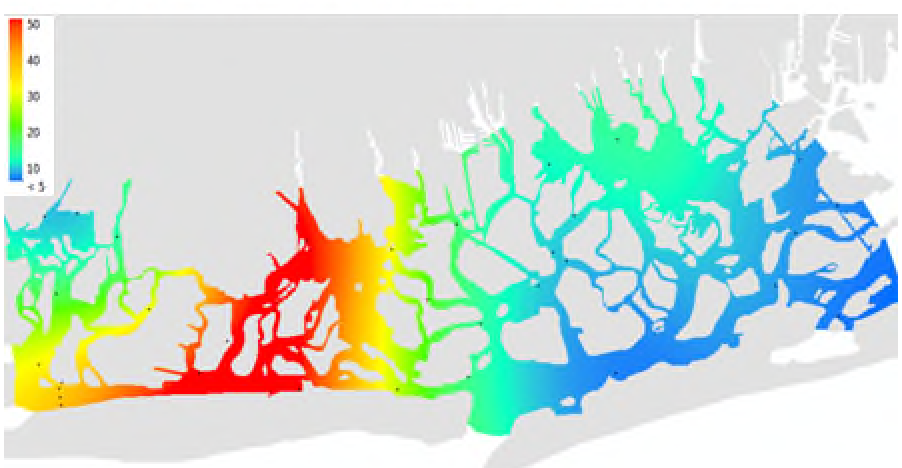


Fig 29 δ^{15}N signatures of *Ulva* sp. in Western Bays in the Fall of 2011.

Fig 30 Effects of *Ulva* dilution experiments in Hewlett Bay. FOSW=filtered ocean seawater

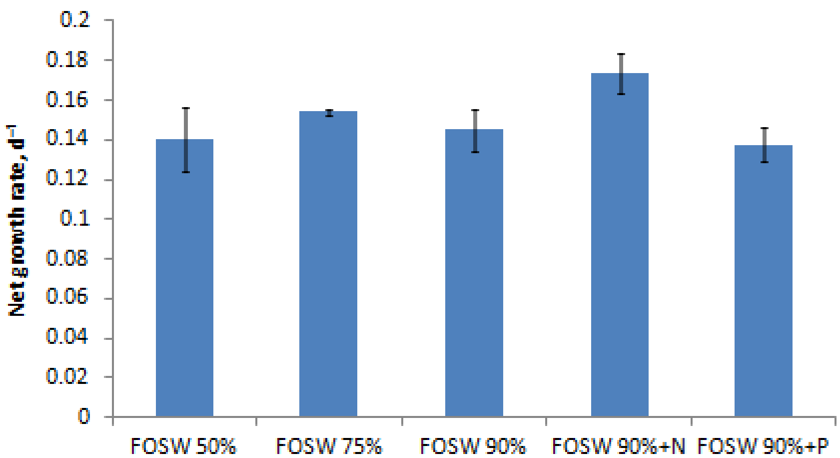


Fig 31 Effects of *Ulva* dilution experiments in Hewlett Bay. FOSW=filtered ocean seawater

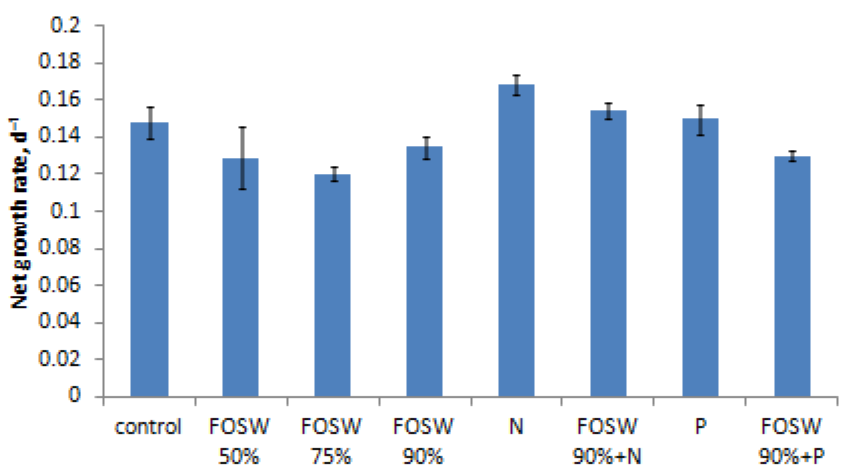


Fig 32 Effects of *Ulva* dilution experiments in Hewlett Bay. FOSW=filtered ocean seawater

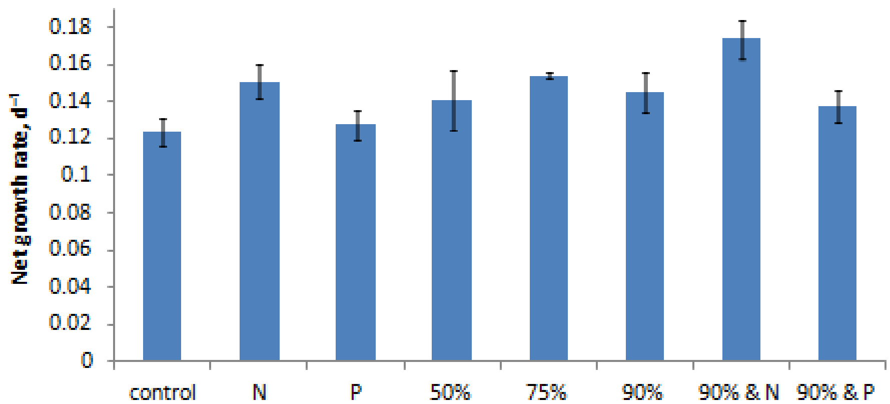


Figure 33. The relationship between dissolved inorganic nitrogen (DIN) and net growth rates of phytoplankton in Hewlett Bay. Differing levels of DIN were achieved by a enriching West Bay water with nitrate or mixing Hewlett Bay water with Atlantic Ocean water. Growth rates within dilutions were corrected for reduced zooplankton grazing rates.

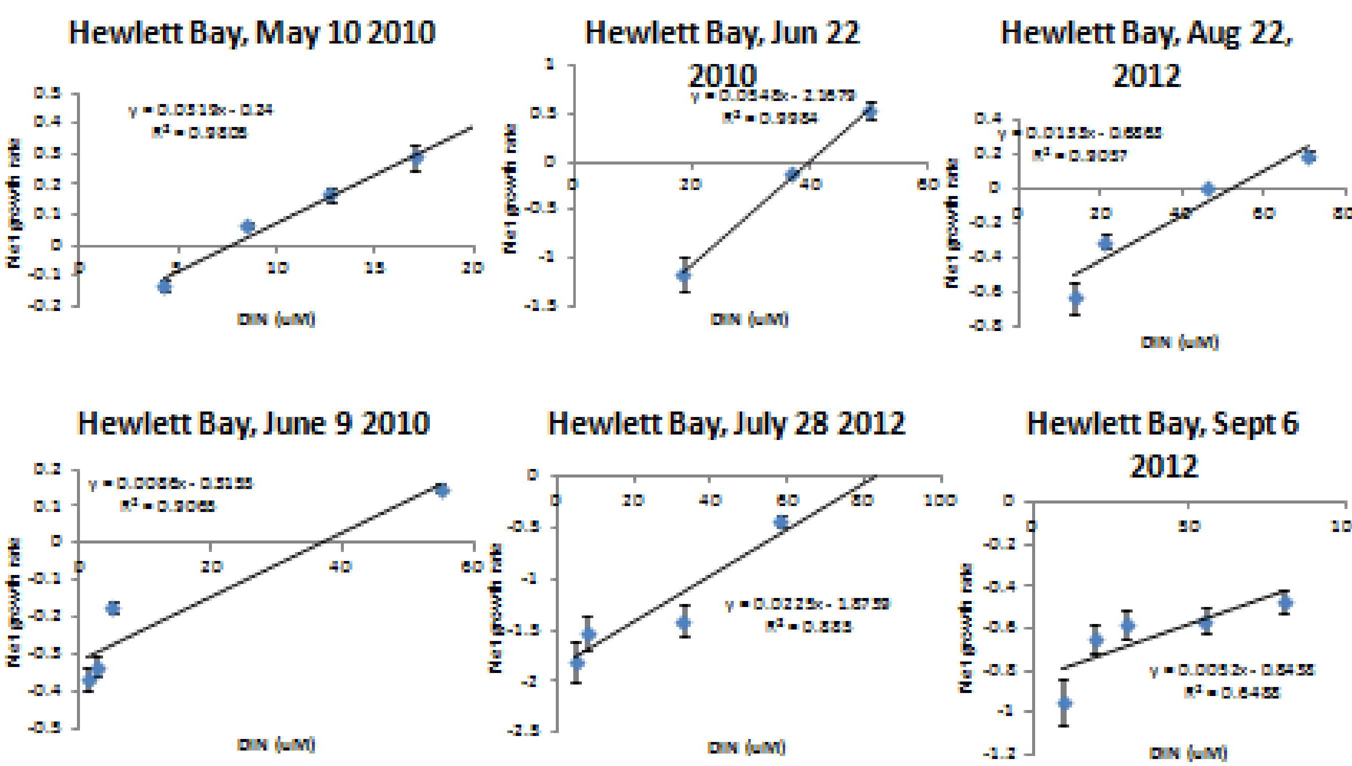


Figure34. The relationship between dissolved inorganic nitrogen (DIN) and net growth rates of phytoplankton in Hewlett and Middle Bay. Differing levels of DIN were achieved by a enriching Hewlett or Middle Bay water with nitrate or mixing West Bay water with Atlantic Ocean water. Growth rates within dilutions were corrected for reduced zooplankton grazing rates.

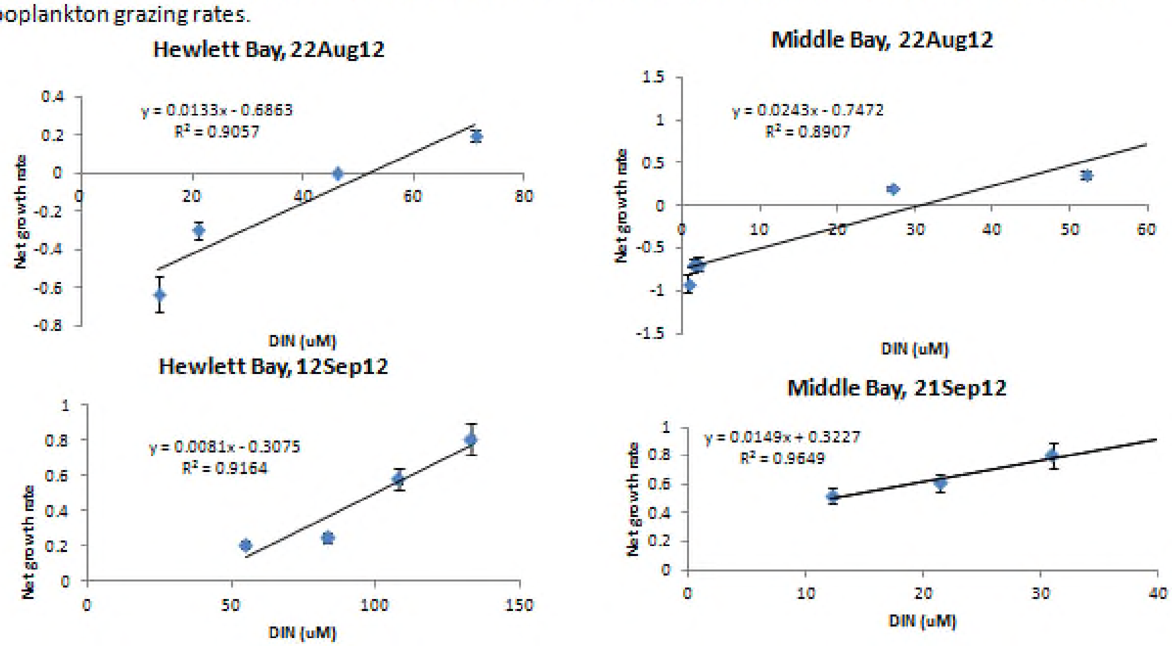


Table 1. Net growth rates of phytoplankton communities in Hewlett Bay and Middle Bay, 2010-2012, as well as the calculated decrease in net growth rate achieved by a 1 µM decrease in DIN. Decreasing the mean net growth rates of phytoplakton in Middle Bay and Hewlett Bay to zero would require DIN reductions of 7 and 4 µM, respectively.

Hewlett Bay

date	Net growth rate	stdev
5/10/2010	0.485	0.036
5/24/2010	0.065	0.045
6/9/2010	-0.173	0.021
6/22/2010	-0.281	0.185
7/21/2010	-0.894	0.040
8/9/2010	0.369	0.015
9/14/2010	0.866	0.026
10/12/2010	0.307	0.133
3/29/2011	0.177	0.022
4/18/2011	0.254	0.029
5/27/2011	0.317	0.017
6/22/2011	-0.205	0.076
6/30/2011	0.118	0.144
7/27/2011	-0.710	0.100
8/31/2011	1.134	0.006
9/29/2011	0.192	0.054
7/28/2012	0.110	0.010
8/22/2012	-0.288	0.178
9/6/2012	-0.629	0.071
9/21/2012	0.254	0.054
Average	0.073	0.495

Middle Bay

date	Net growth rate	stdev
7/21/2010	-0.76	0.04
8/9/2010	0.04	0.15
9/14/2010	1.10	0.03
10/12/2010	-0.01	0.13
3/29/2011	0.56	0.01
5/27/2011	1.16	0.03
7/20/2011	-0.74	0.14
7/27/2011	-0.20	0.20
8/31/2011	0.22	0.02
8/22/2012	-0.68	0.07
9/21/2012	0.80	0.01
Average	0.14	0.07

Decrease in net growth per decrease in uM DIN

Date	Site	
5/10/2010	HB	0.030
6/9/2010	HB	0.009
6/22/2010	HB	0.050
7/28/2012	HB	0.020
8/22/2012	HB	0.013
9/6/2012	HB	0.005
9/21/2012	HB	0.008
8/22/2012	MB	0.024
9/21/2012	MB	0.015
Average		0.02

APPENDIX B: BENTHIC INFAUNA

Benthic Infauna

When analyzed with the identities of the infaunal organisms intact, these new infaunal data from Hempstead Bay, in concert with the other lines of evidence, can tell us that the existing infauna, this year, across these specific sites, exhibited some patterns potentially informative to the study questions: Is there evidence of impairment? Does the impairment appear linked to nutrients? And can/should the bays be managed as one unit, or are the unique subwaterbodies with distinct regulatory characteristics?

The Bray-Curtis similarity method calculates the percent similarity between sets of samples by considering the identities and actual abundances of the organisms occurring in the samples. This method effectively allows sites with similar numbers of taxa or similar overall abundances to be distinguished based on the identities of those organisms present. The similarity output is used to create a branching diagram (a dendrogram) that sequentially links samples by closeness of similarity. The branching pattern can be transformed into a two-dimensional spatial plot of the samples on which various ecological and environmental parameters can be overlain. This allows us to visually estimate the characteristics of the main groups of stations and the possible correlation between environmental factors and the infaunal community.

The Hempstead Bay infaunal community analysis identified two stations (ToH16, U04D) that had communities that were extremely dissimilar from the remaining stations. Further, the remaining stations were separated into two very dissimilar groups of eight stations (Group I) and six stations (Group II). In general, overall community similarity was relatively low among all of the stations; the two most similar stations had a similarity value of about 69%, which experience tells us is low. This overall dissimilarity is notable because one polychaete species, *Streblospio benedicti*, was the most, or second most, abundant organism at all stations except the two "outlier" stations. Other annelid worms, such as *Capitella capitata* and unidentified oligochaetes, were among the other predominant taxa at many stations. These taxa are commonly referred to as "opportunistic" because they have the ability to rapidly colonize disturbed habitats. The predominance of these taxa indicates that the infaunal communities in the bay regularly experiences some degree of environmental stress.

By examining the characteristics of the two main infaunal community "groups," especially when considered along with the other environmental data collected, we find that some areas of the bay generally appear less stressed than others. The eight stations identified as "Group I" share ecological features such as low numbers of species, low abundance, and low species diversity. These features often associate with somewhat impaired benthic communities. The worm *Streblospio benedicti* accounts for 50 to 75% of the organisms at these stations. Relative to Group I stations, the six stations identified as "Group II" share high numbers of species, high abundance, and greater diversity. *Streblospio benedicti* still predominates the fauna, accounting for 26 to 88% of the organisms. Regardless, the combination of higher species numbers, abundance, and diversity suggest that the Group II stations may encounter less stress than the Group I stations. A sediment profile image (SPI) measure, apparent Redox Potential Discontinuity (aRPD) depth, is slightly greater for Group II stations than for Group I stations. This metric is often influenced by infaunal community activity, with deeper aRPD depths indicating somewhat greater infaunal activity.

There does not seem to be a geographical division between the two community groups because they span all three bays, and often two stations in different community groups are located close to each other. The habitat data suggest that Group I stations may be in more depositional areas than Group II stations because they have a much lower proportion of sand than Group II stations. This supposition is supported by an SPI measure, prism penetration, that is deeper for Group I stations than Group II stations, indicating the presence of “softer” sediments at Group I stations. *Ulva* cover at the Group I stations was more than twice that found at the Group I stations.

Comparing the water quality contour plots for the bay to the infaunal similarity results shows that temperature, salinity, and photosynthetically active radiation are not related to the infaunal community groups. However, other water quality features generally appear to show some “relationship” to the primary infaunal community structure. The estimated median values for water column nitrate, phosphate, ammonium, and silicate are all greater among Group I stations than Group II stations. The median ammonium values for Group I stations is almost six times greater than the median value for Group II stations. These general observations seem to suggest that the infaunal community may be more stressed in locations where water quality is relatively worse. However, they do not eliminate possible alternative explanations. For example, sediment texture is generally finer among Group I stations. Finer sediments usually indicate depositional areas, which may then contain higher levels of pollutants and total organic carbon than areas with coarser sediments, which were not measured. Another caveat is that the general patterns just identified do not always hold for each station. For example, the infaunal communities at stations ToH2 and ToH3 are aligned with the Group II stations, yet the water column nutrient levels at these stations are among the highest measured in the bay. The fauna at Station U11A aligns with Group I stations, but the water column nutrient levels there are relatively low.

Returning to the three main questions asked earlier, the nature of the infaunal community (species identities, ecological metrics, similarity) suggests that there is evidence of impairment in the bay. However, the metrics do not allow the degree of impairment to be quantitatively stated. The general comparison between nutrients and infaunal community structure hints at a possible connection. However, other variables may co-vary with those two ecosystem components. The more impaired community occurs in fine sediments indicative of a depositional area. That depositional area likely indicates an area of relatively poor water circulation, which also contributes to higher water column nutrient loads remaining in an area. That the two main infaunal community groups span all three of the sub-bays comprising the Hempstead Bay system suggests that factors affecting benthic habitats occur across all sub-bays, and that argues for management of the system as a single unit.

We do not, and likely after 5 to 10 years of monitoring may still not, have the ability to translate data about infaunal assemblages into waterbody-wide nutrient concentrations below which regulators could declare the benthic habitat to no longer be impaired. This is difficult to derive because the infaunal communities respond to many complex environmental factors, such as sediment texture, organic content, the presence of pollutants, and salinity. Often the initial infaunal data analyses calculate the total abundance and the total number of species at each station then may involve the calculation of standard ecological indices, such as species diversity (H'), richness, evenness (J'), or dominance. The former metrics are agglomerative and often difficult to interpret because they consider only the total numbers of species or individuals, ignoring the identities of those organisms present. The latter metrics also ignore the identities of those organisms present. In doing so, the metrics often suggest that two areas may be the same or very similar when in reality they encompass very different sets of organisms. Many ecological indices, such as Shannon Diversity (H'), may have inherent assumptions that are not met in a benthic sampling program. Also, ecological indices, and the suite of environmental indices that have followed them, are very difficult to interpret. For example, it is generally accepted that higher species diversity is “better” than lower diversity, but establishing a threshold separating good from less than good is not practicable.

Additionally, the infaunal community in Hempstead Bay, and elsewhere, will be at different stages of development based on proximity to stressor sources, water depth, and seasonal shifts that can vary annually, and these different stages cannot be detected by the usual ecological metrics. Even if we had multiple replicate surveys across years of consistently improving nutrients levels with consistent wind and rain and heat patterns, these agglomerative metrics and indices would not provide sufficient discrimination to correlate infaunal community condition with water column nutrient loads and levels.

Figure 1a. Names and types of benthic sampling sites.

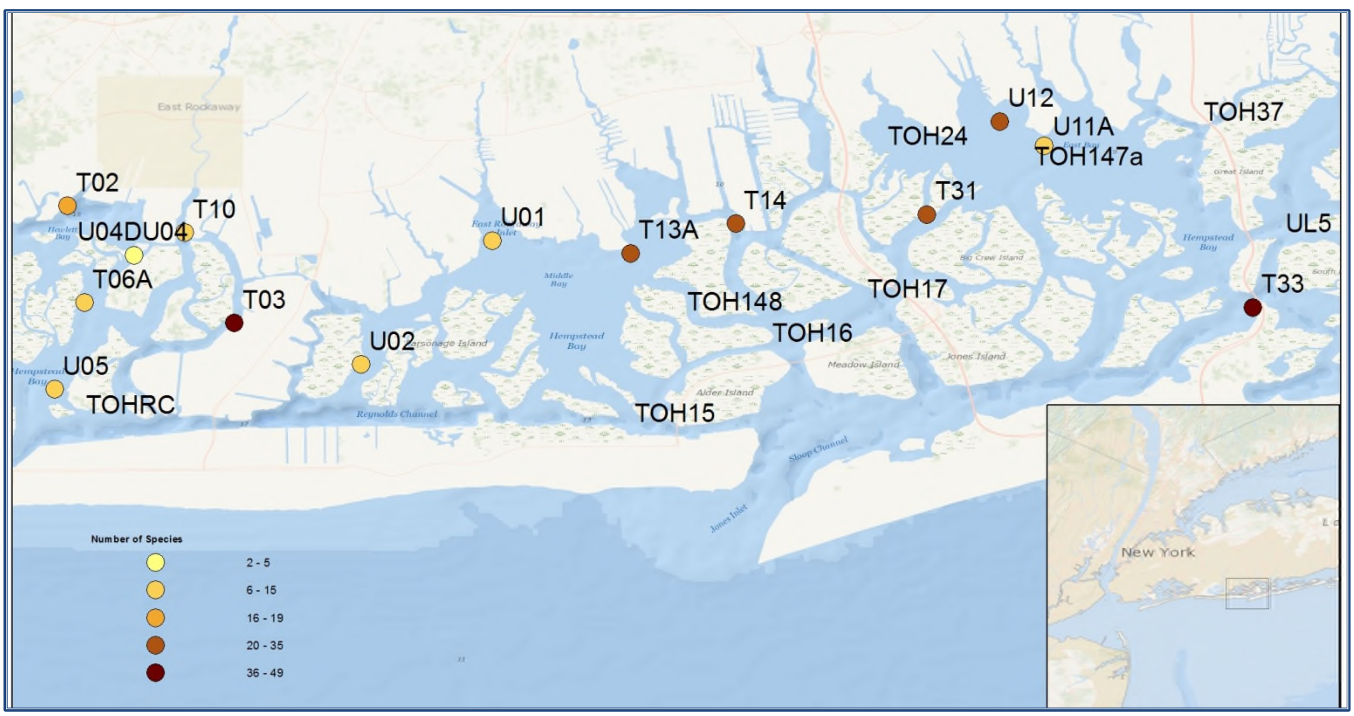


Figure 1b. Number of species.

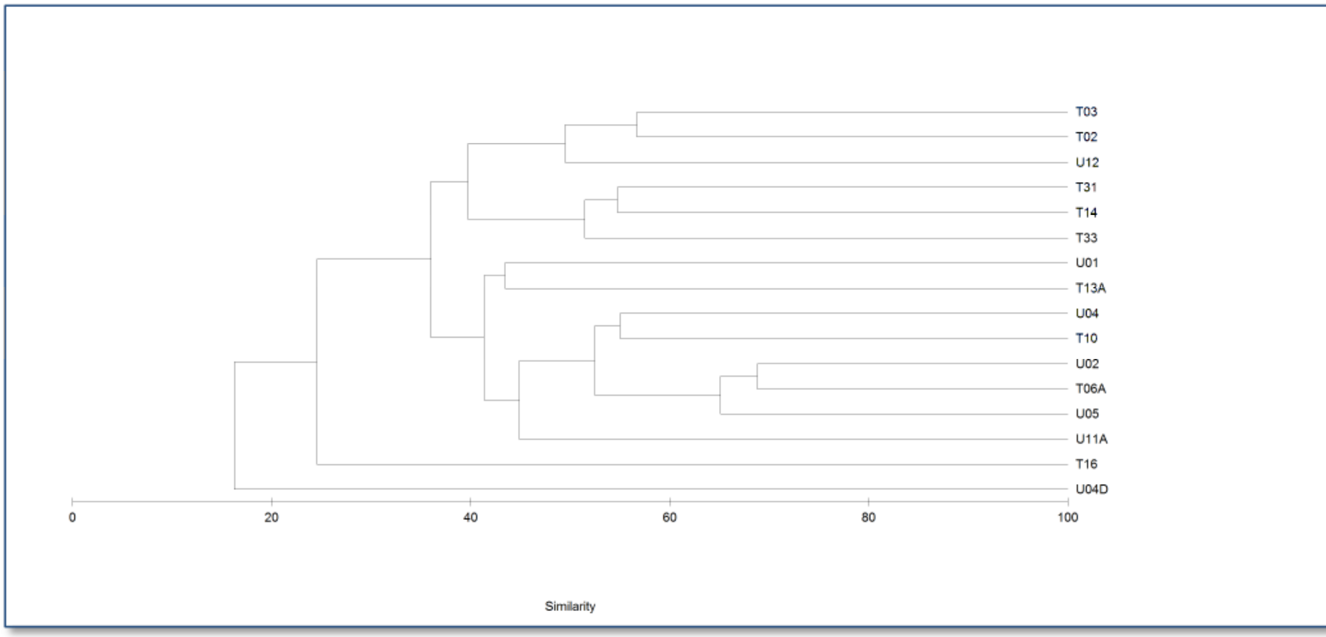


Figure 2. Bray-Curtis Similarity dendrogram showing two "outliers" and two major cluster groups of Hempstead Bay stations.

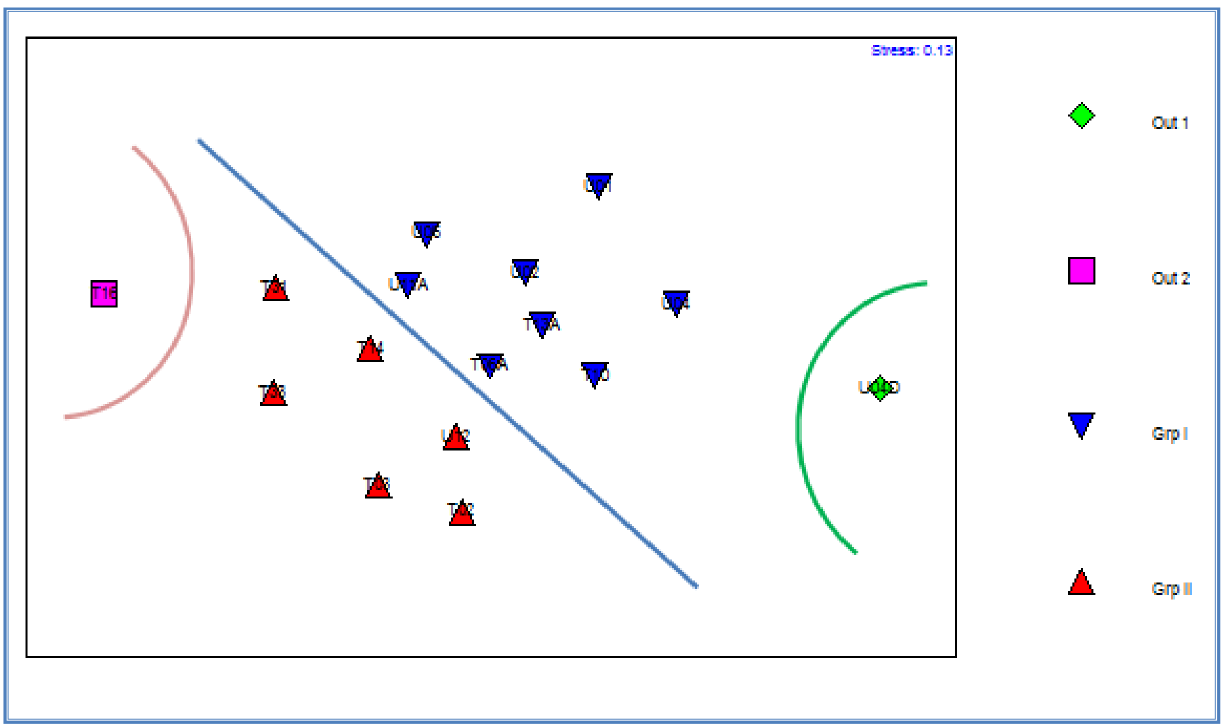


Figure 3. nMDS plot of Bray-Curtis similarity. Lines separate main cluster groups as shown in dendrogram. Out 1 and Out 2 are the least similar of the "groups".

Table 1. Infaunal habitat quality indicators measured at two distinct sites and the averages for two clusters of similar sites.

Discipline	Metric	U04D [1]	ToH16 [1]	Group I [2]	Group II [2]
Ecological	species	2	17	10	26
	Evenness (*J'*)	0.07	0.73	0.56	0.47
	Shannon Diversity (H'_2)	0.1	3.0	1.7	2.2
	Total Abundance	117	202	252	968
Sediment	Medium Sand (%)	1	5	1	6
	Fine Sand (%)	5	87	36	60
	Total Sand (%)	12	93	39	75
Water Quality	PAR	18	7	14.5	13.5
	Salinity (PSU)	29.8	31.5	30.3	31.8
	Nitrate (µM)	18	5	13.5	5
	PO4 (µM)	4	1	3.25	1.5
	NH4 (µM)	45	8	37.5	6.5
	SiO_2	14	5.5	12	9.8
	Ulva % Cover)	15	15	26	11
SPI	aRPD (cm)	0	1.4	1.7	2.6
	Prism Penetration (cm)	0	5	9.3	5.7
	Gas Voids (#)	0	0	0	0
Flux	O_2			-1293	-2339
	N_2-N			168	71
	NH_4			382	393
	NO23			0	-41
	PO_4			42	-1

[1] Actual value.
[2] Median value.

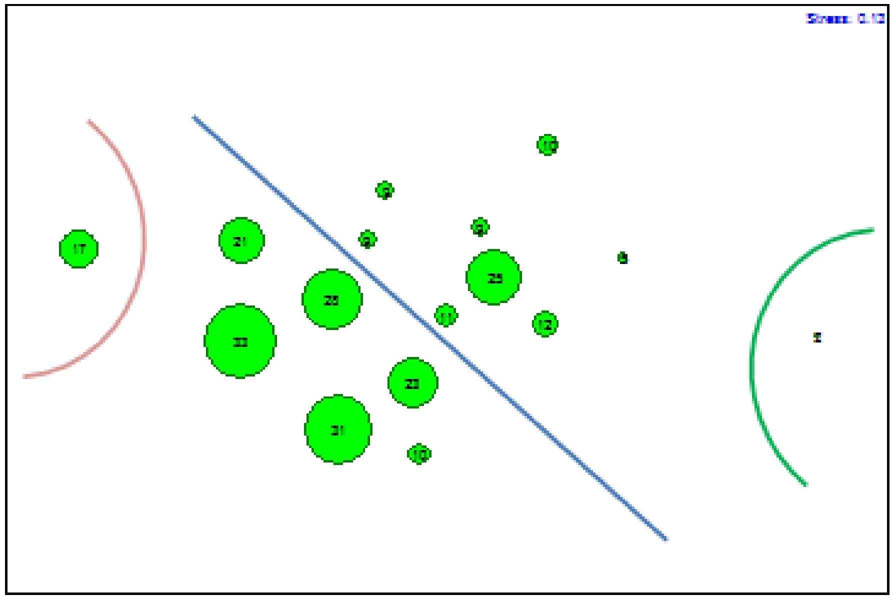


Figure 4. Number of Species

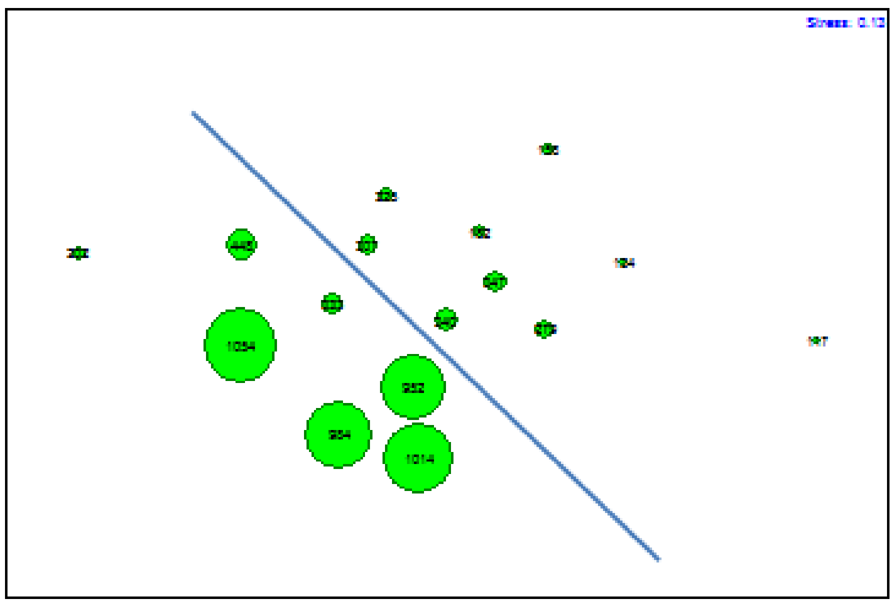


Figure 5. Total Number of individuals.

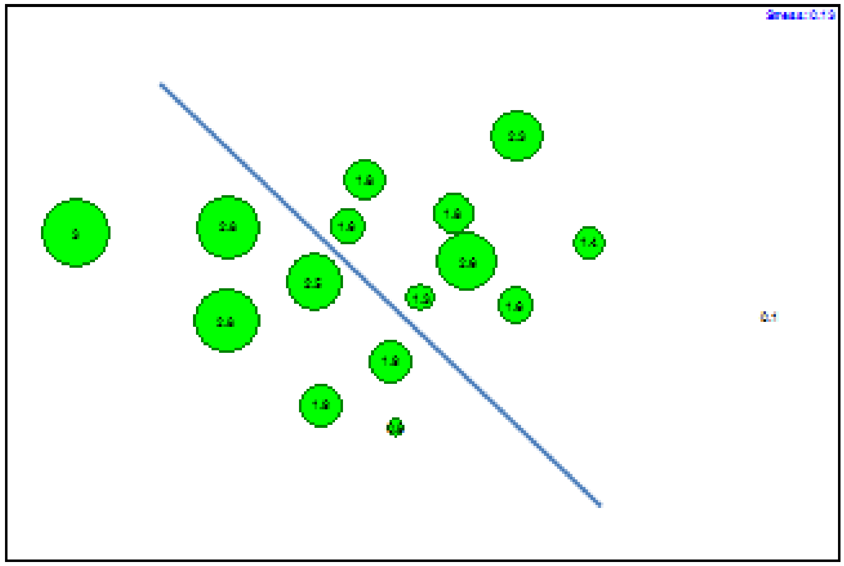


Figure 6. Shannon-Weiner Diversity ($\log_2$).

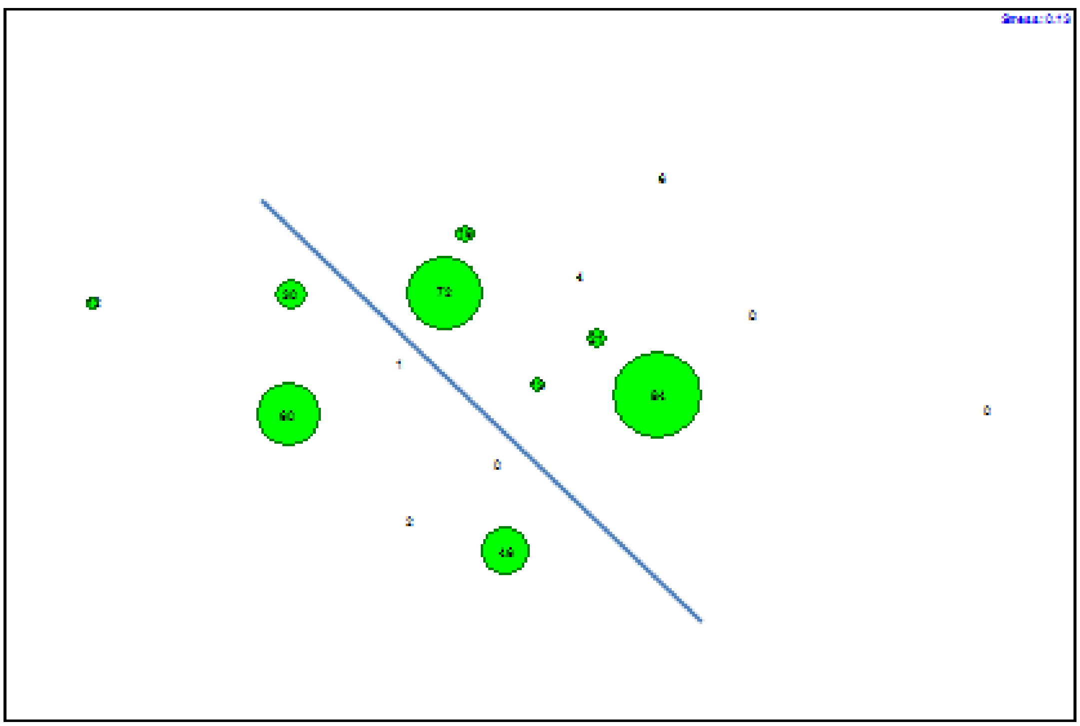


Figure 7. Number of Crustacea

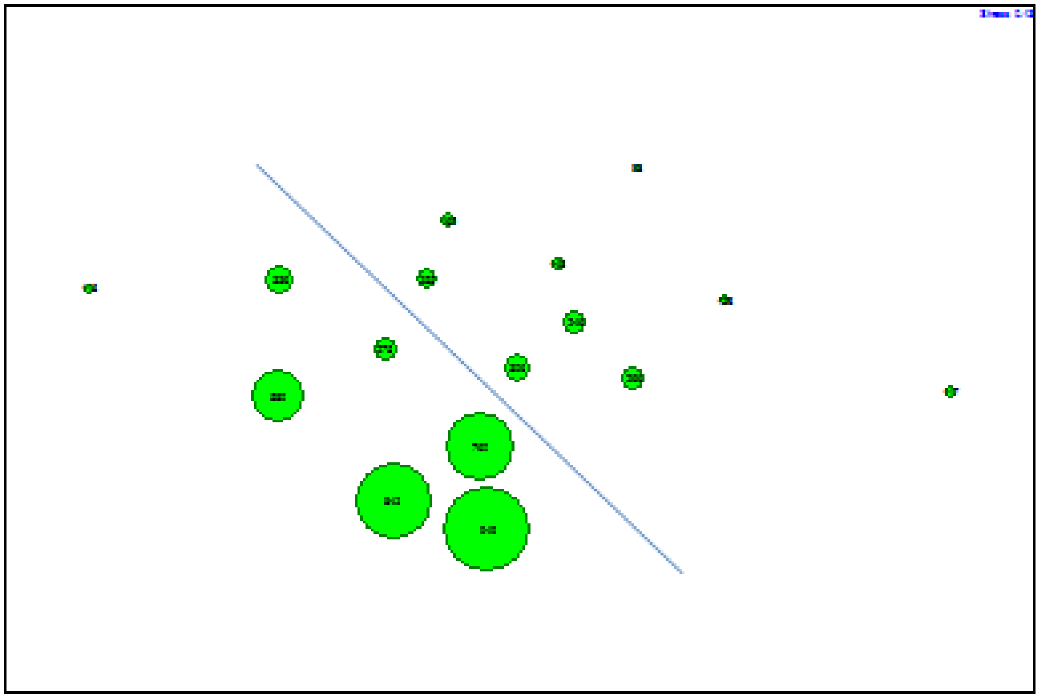


Figure 8. Number of Polychaeta

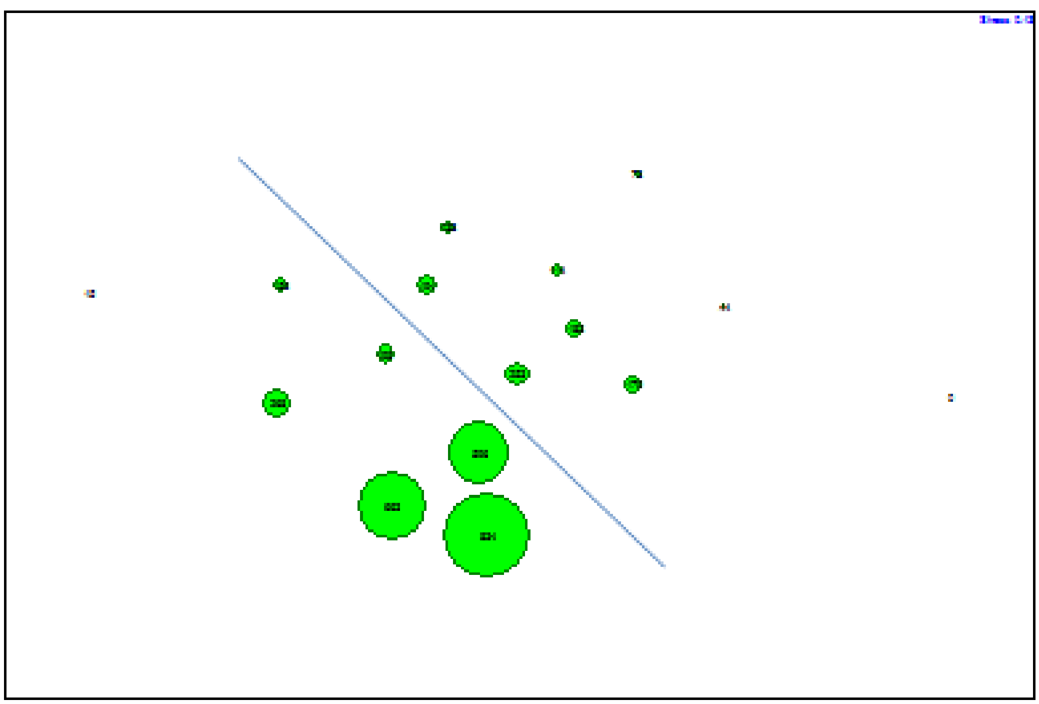


Figure 9. *Streblospio benedicti* abundance.

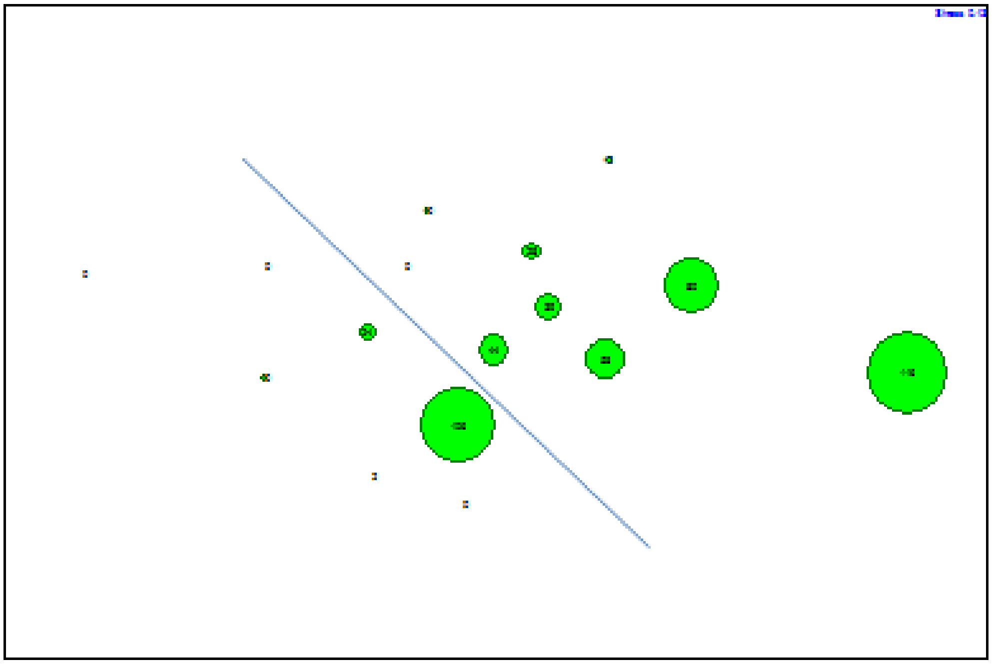


Figure 10. *Capitella capitata* abundance

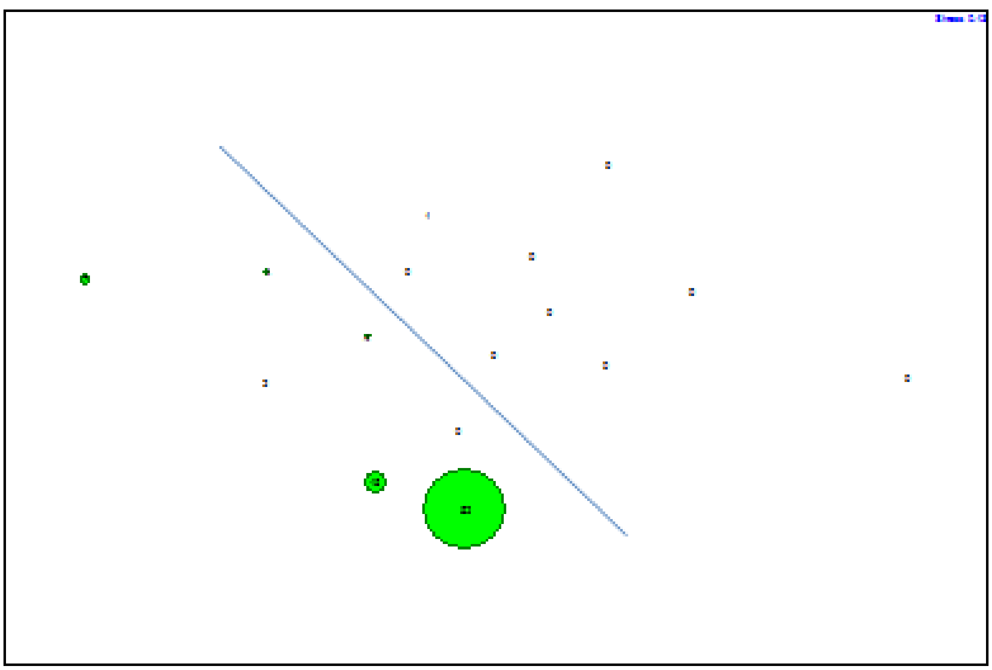


Figure 11. *Mercenaria* abundance.

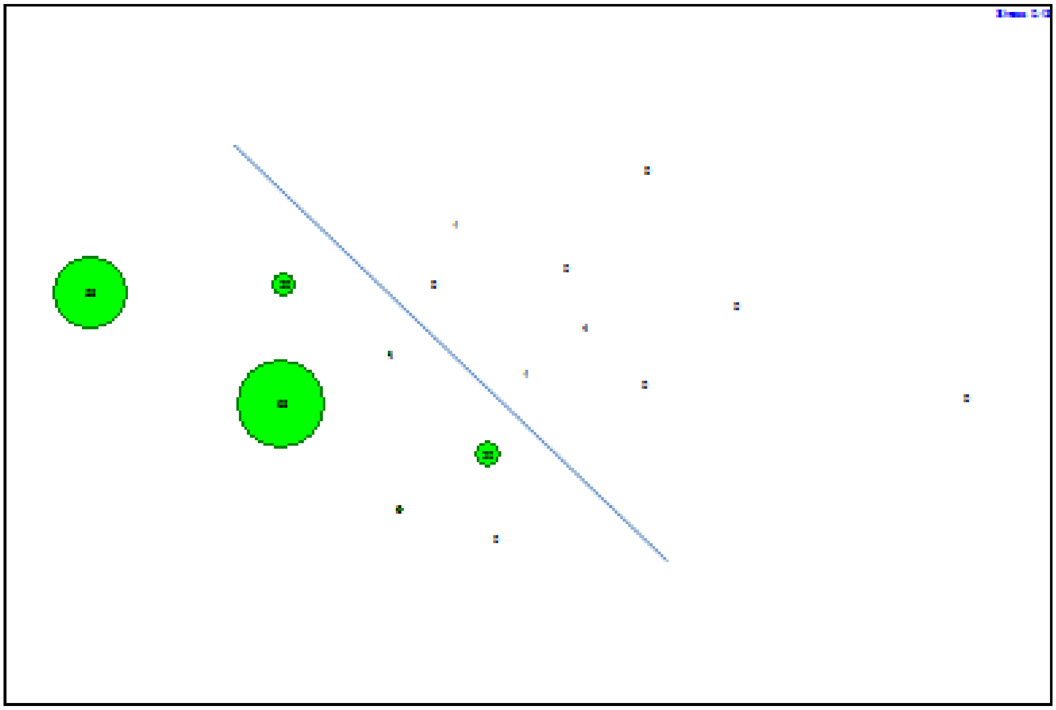


Figure 12. *Tellina agilis* abundance.

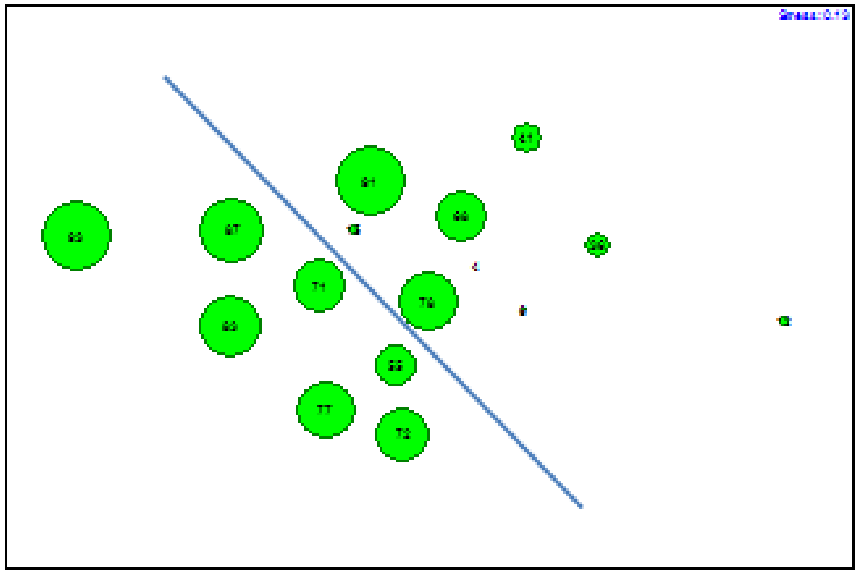


Figure 13. Total Sand (%)

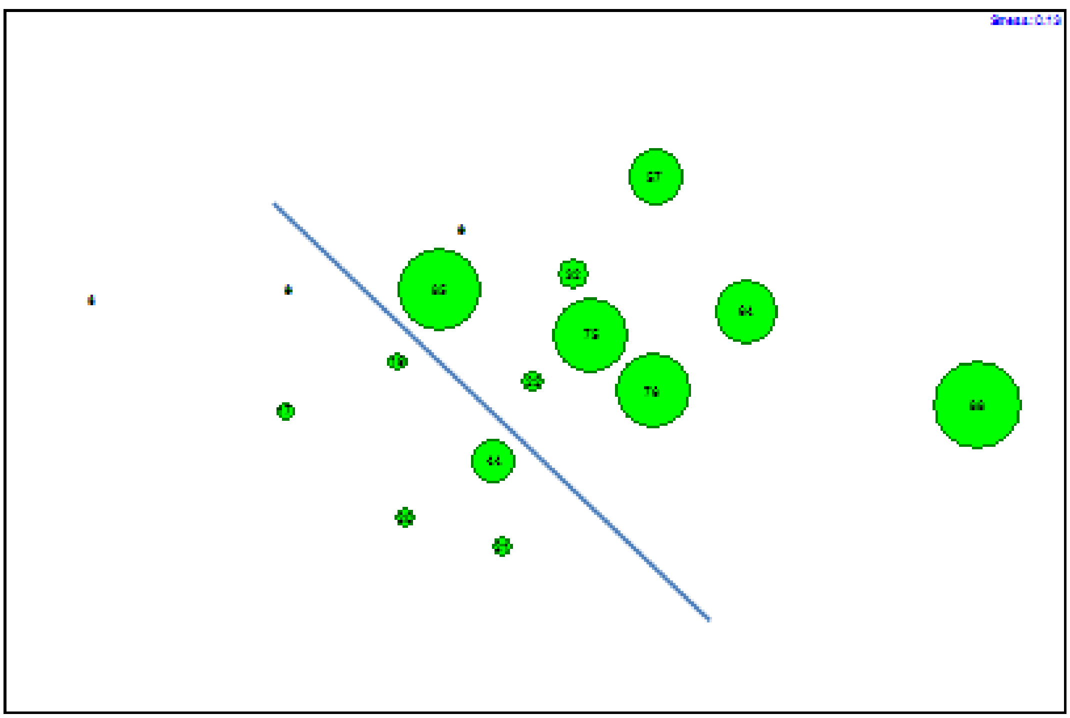


Figure 14. Calculated Fines {%; 100-(Sand + Gravel)}

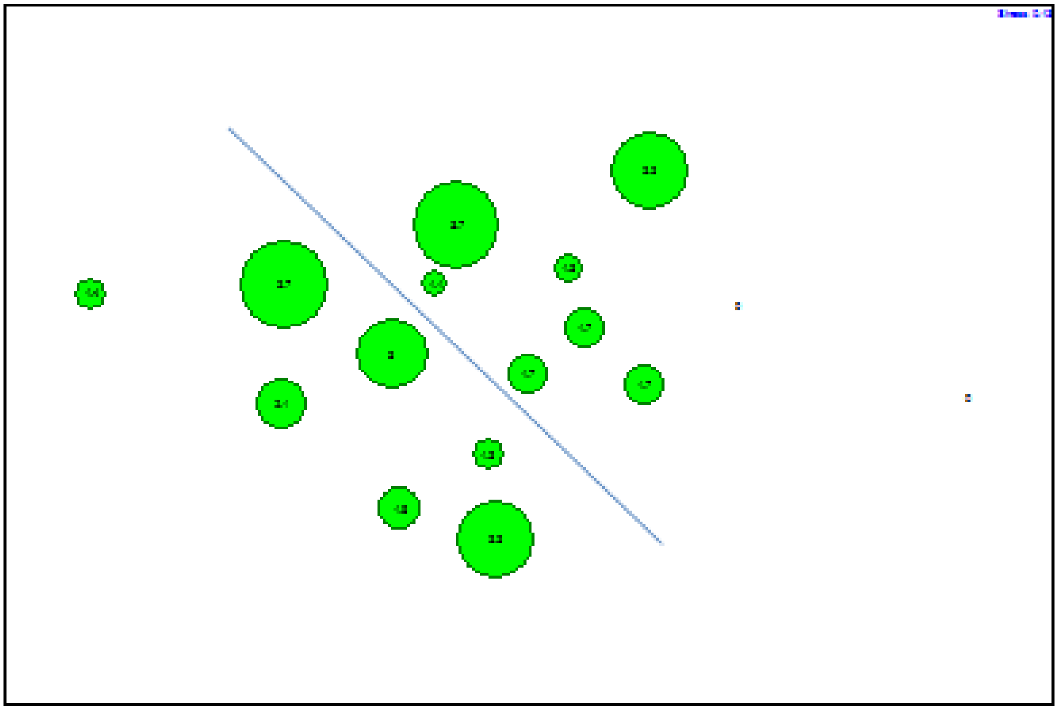


Figure 15. Sediment Profile Image derived apparent Redox Potential Depth (cm)

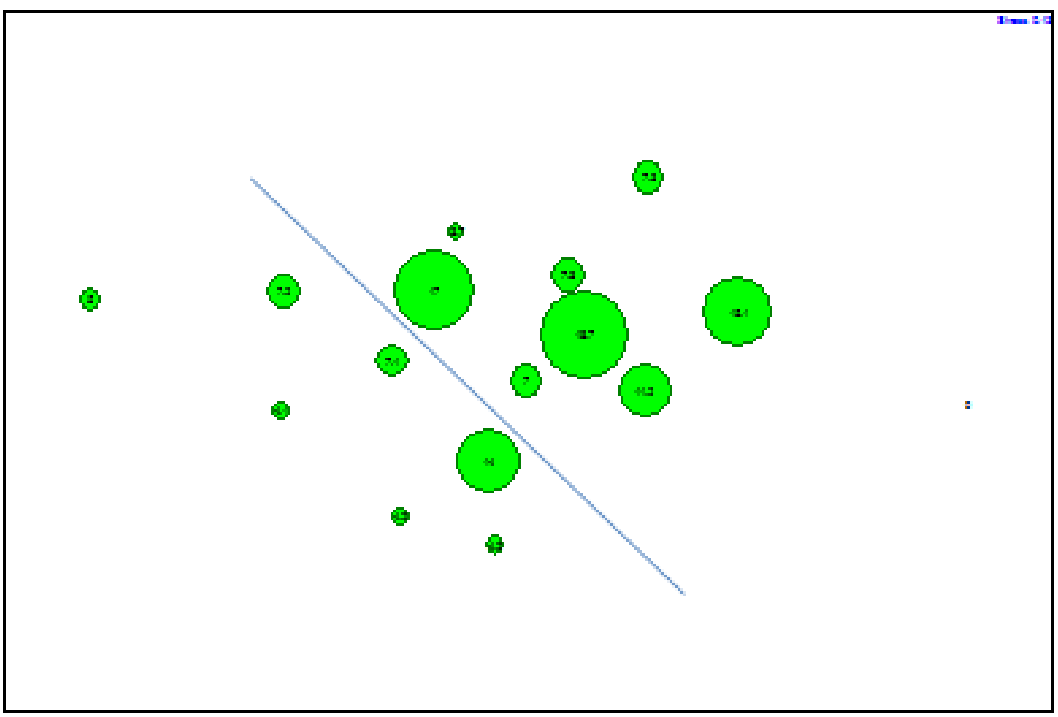


Figure 16. Sediment Profile Image camera Prism Penetration (cm)

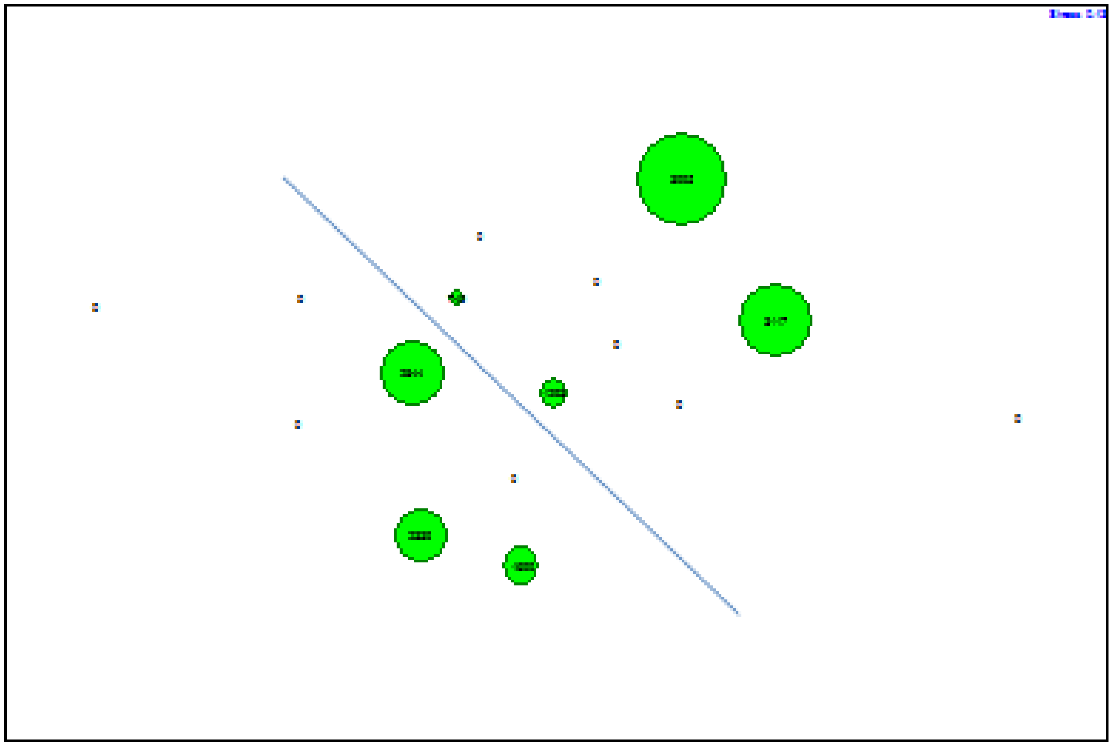


Figure 17. Sediment O_2 flux (note values are negative; "0" values = no data collected).

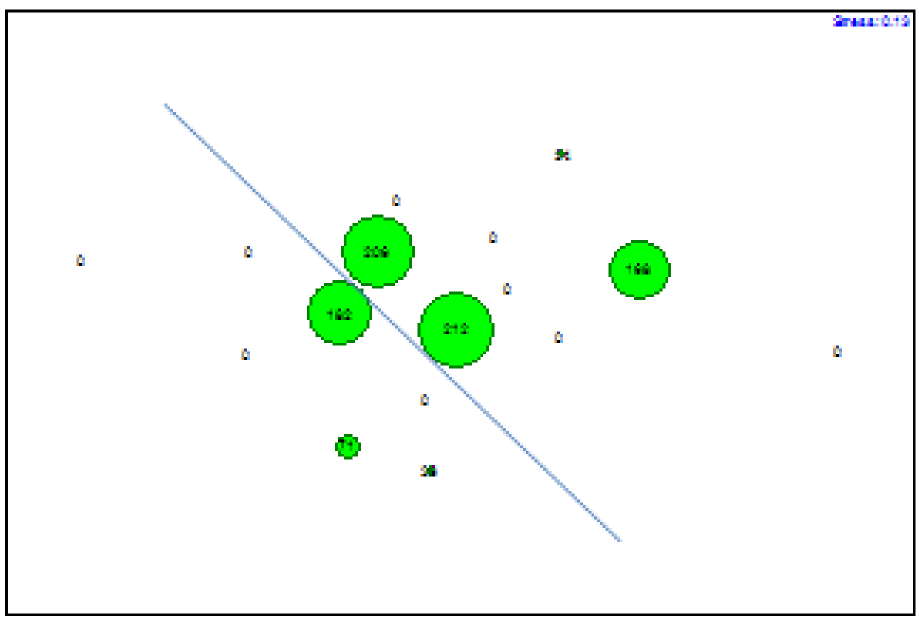


Figure 18. Sediment N_2-N Flux ("0" values = no data collected).

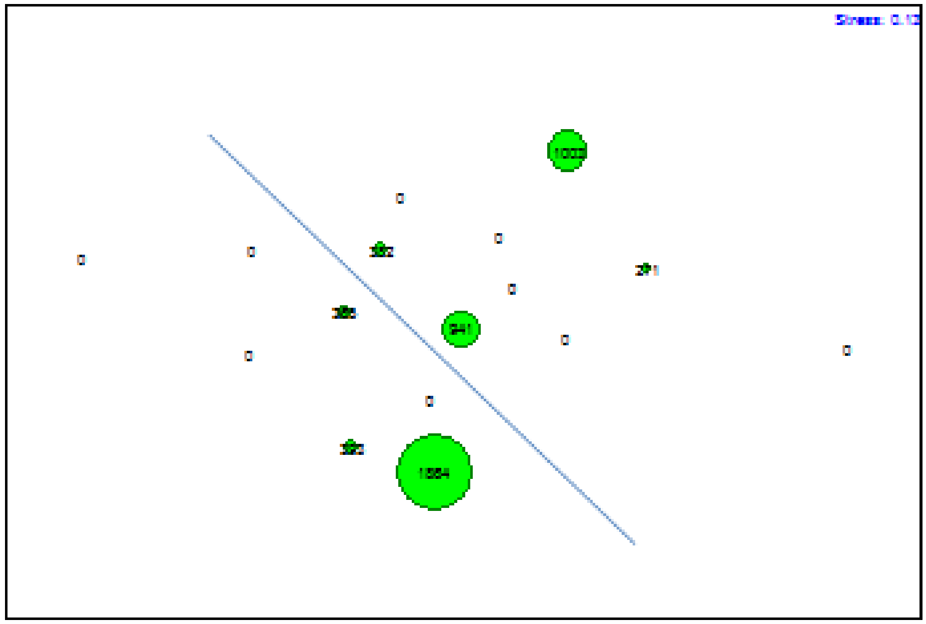


Figure 19. Sediment NH_4 Flux ("0" values = no data collected).

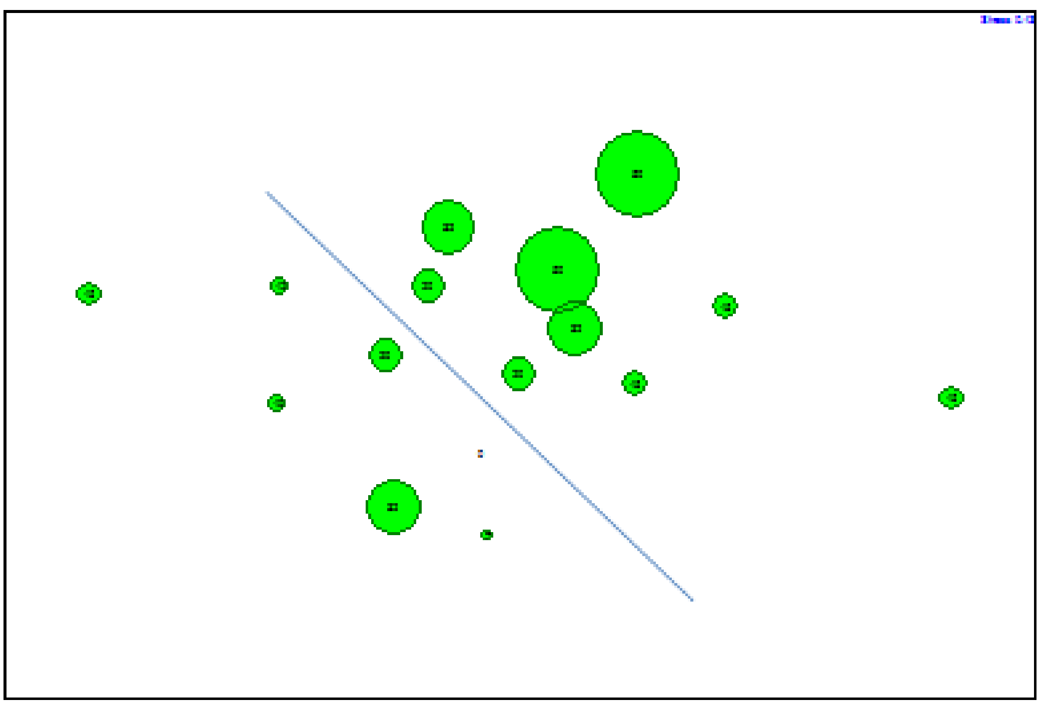


Figure 20. Ulva cover (%) approximated from contours maps.

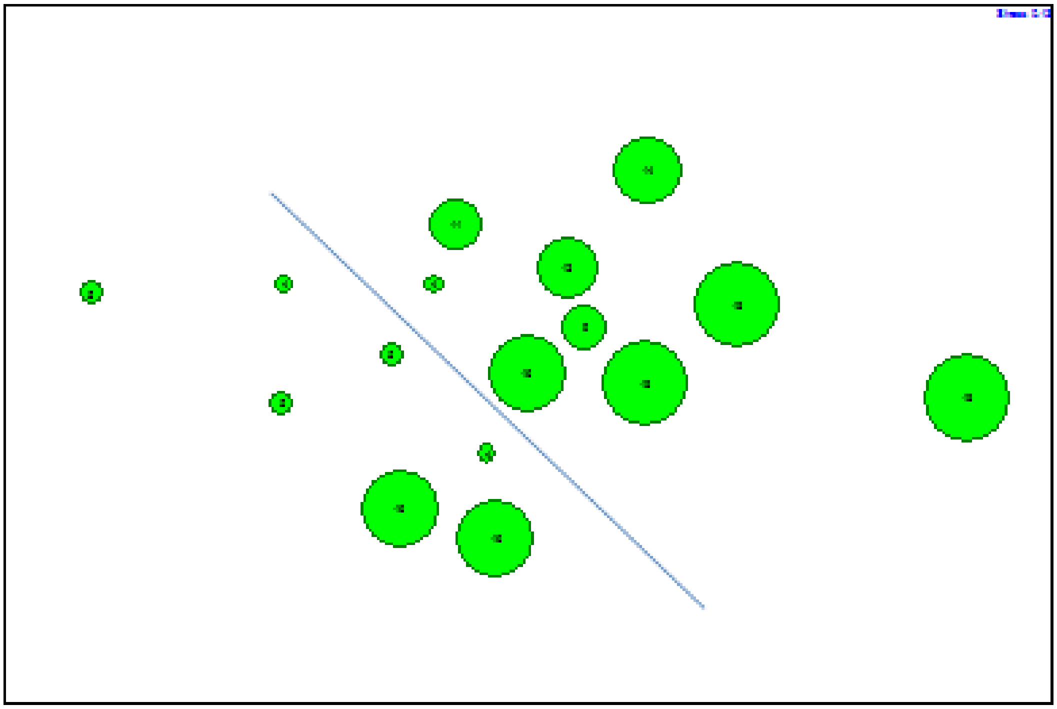


Figure 21. Ambient water column nitrate levels (µM) approximated from contours maps.

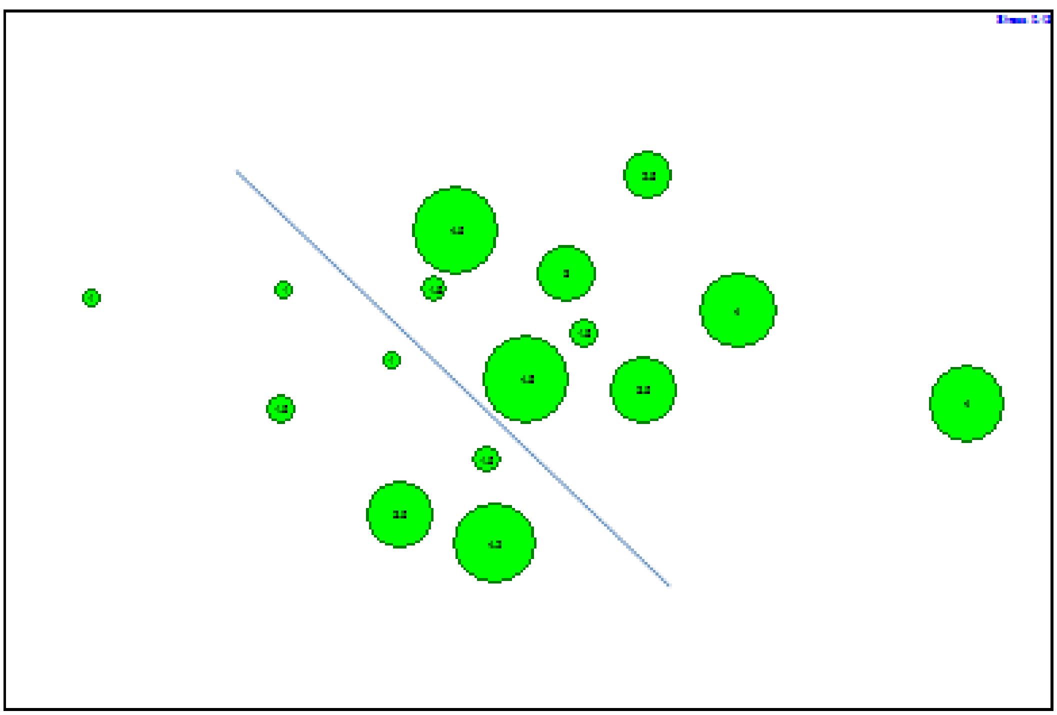


Figure 22. Ambient water column phosphate values (µM) approximated from contours maps.

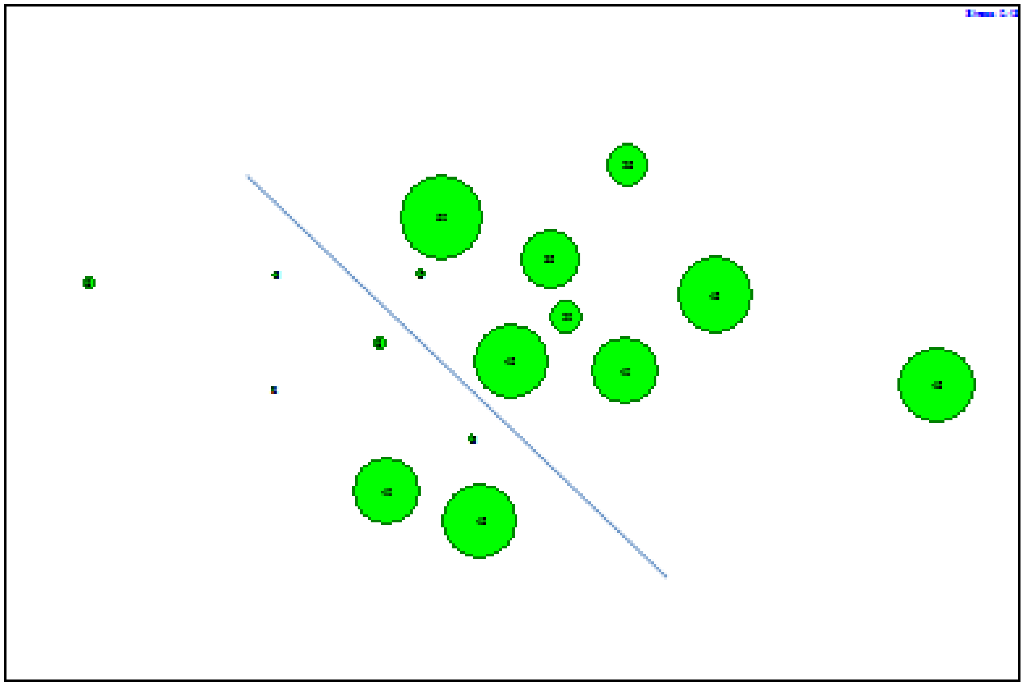


Figure 23. Ambient water column NH_4 values (µM) approximated from contours maps.

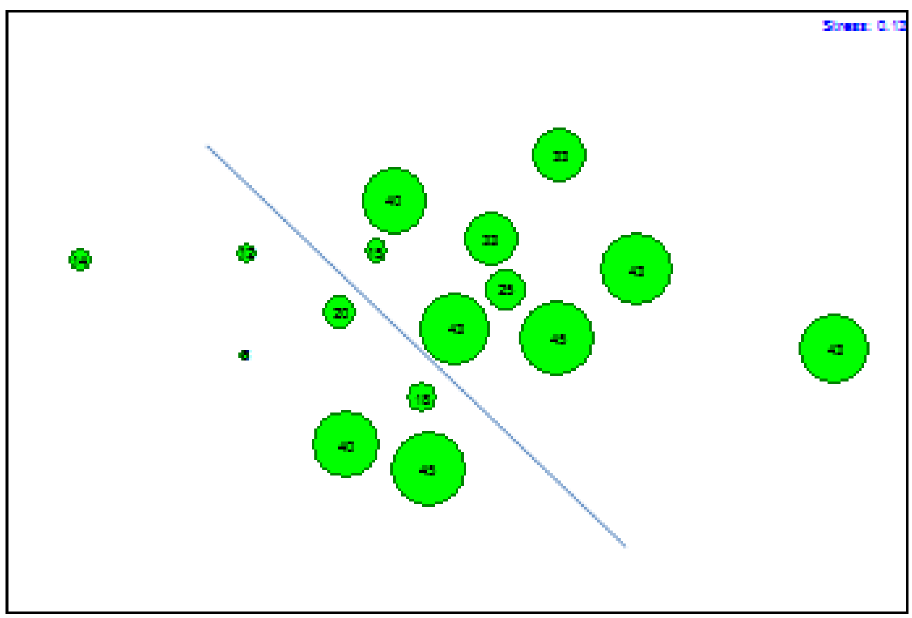


Figure 24. Ambient water column silicate values (µM) approximated from contours maps.

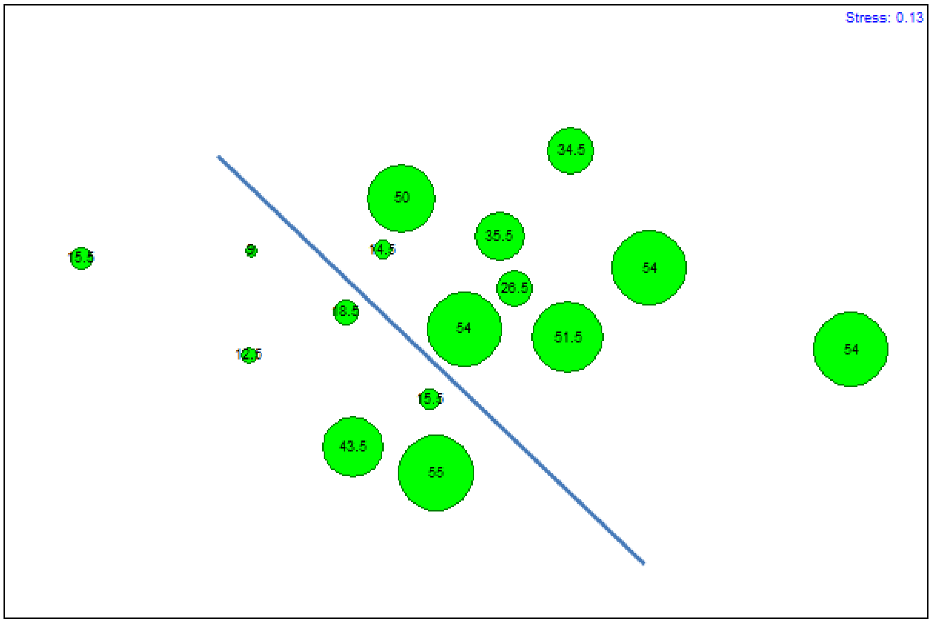


Fig. 25 Overall Water Quality (Sum of Ranks; lower scores = better WQ)

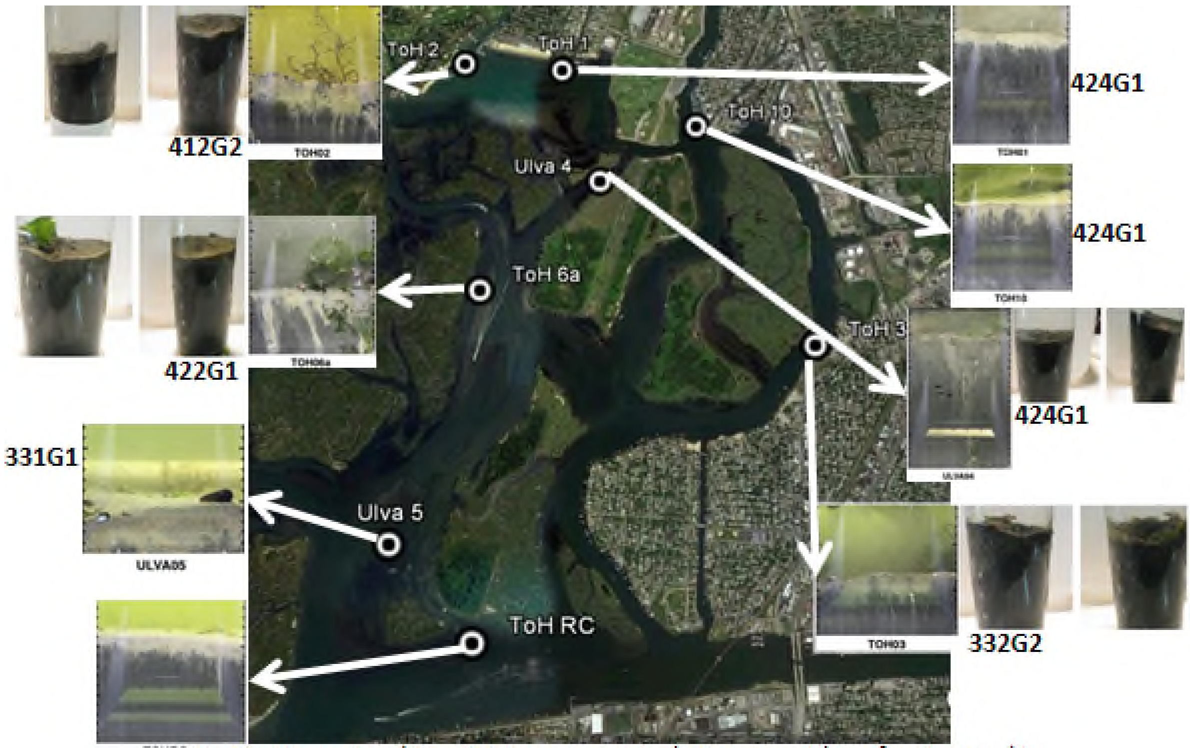


Fig 26. Hempstead Bay SPI images; core photos; quartiles of water quality, Ulva % cover, % fines; and infaunal similarity group #.

Fig 27. Middle Bay SPI images; core photos; quartiles of water quality, Ulva % cover, % fines; and infaunal similarity group #.

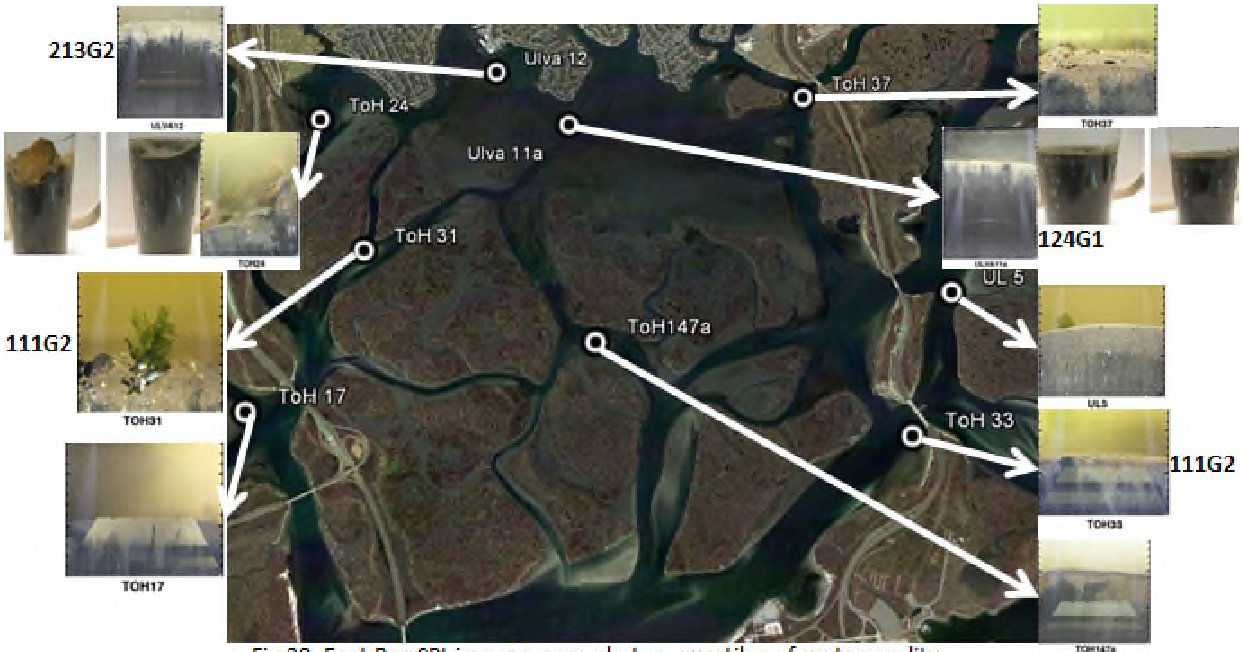


Fig 28. East Bay SPI images; core photos; quartiles of water quality, Ulva % cover, % fines; and infaunal similarity group #.

APPENDIX C: HEMPSTEAD BAY WATER QUALITY STUDIES: SEDIMENT PROFILE IMAGING

Hempstead Bay Water Quality Studies:

Sediment Profile Imaging

By

Robert J. Diaz
R.J. Diaz & Daughters
6198 Driftwood Lane
Ware Neck, VA 23178

To

Battelle
1212 North Country Road
Stony Brook, NY 11790

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Appendix C

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INTRODUCTION

Eutrophication continues to be a growing problem at all scales from local to global (Diaz et al. 2010). Within the US, national assessments have found chronic eutrophic conditions in many coastal systems (Bricker et al. 2007). Hempstead Bay is no exception with numerous observations indicating the presence of eutrophic conditions. Submerged aquatic vegetation is reported to be absent due to water column shading by macroalgae, microalgal blooms, or suspended solids. Bivalves and aquatic fisheries have declined in this area, which may be related to compromized benthic communities or stressful water quality levels. Decomposing algal mats cause localized hypoxia (dead zones) in shallow areas and have also been reported as aesthetic disturbances to recreational swimmers. For example, large canopies of *Ulva* spp. that accumulated in shallow areas of Jamaica Bay, to the west of Hempstead Bay, produced anoxic conditions on the bottom and sometimes in the water column in summer (Rhoads et al. 2001, Franz and Friedman 2002). The goal of the Hempstead Bay studies is to better characterize various aspects of the ecosystem and provide a more holistic understanding of eutrophication in the bay (Figure 1).

The overall objective of the Hempstead Bay water quality studies is to conduct a nutrient assessment specific to the Hempstead Bay ecosystem (Figure 2) that includes development of Nutrient Management Plan (NMP) options and a determination of the appropriate numeric criteria or endpoints for evaluating the status of the system and attainment of designated uses. The study seeks to determine if there are nutrient-related problems with Hempstead Bays' ability to fulfill its designated uses (shellfishing for market purposes, recreation, and fishing, as well as fish, shellfish, and wildlife propagation and survival). Details of this study can be found in Battelle (2012). This report uses sediment profile imaging (SPI) to characterize benthic habitat quality and provide input to the ecosystem Conceptual Model (CM) to ensure a consensus and understanding of the major components and functions in the Hempstead Bay ecosystem (Battelle 2012).

The sediment profile camera was developed by Rhoads and Cande (1971) to investigate processes structuring the sediment-water interface and as a means of obtaining in situ data on benthic habitat conditions. The technology of remote ecological monitoring of the sea floor (REMOTS) or sediment profile imaging (SPI) has allowed for the development of a better understanding of the complexity of sediment dynamics, from both a biological and physical point of view (Rhoads and Germano 1986, Diaz and Schaffner 1988, Valente et al. 1992, Bonsdorff et al. 1996, Nilsson and Rosenberg 2000, Rosenberg et al. 2001).

This approach to evaluating the environment can be easily combined with other methods to assess habitat conditions and impacts, providing scientists and managers with a more holistic ecosystem view (Solan et al. 2004, Diaz et al. 2009). The SPI results will be compared to other sites within Hempstead Bays during the same survey period. Additional analyses were conducted at a subset of SPI stations including benthic infauna identification and enumeration and benthic flux/porewater characterization. Results from this subset of stations will be extrapolated across the bays based on the SPI results.

The primary goal of the SPI survey was to evaluate whether the SPI method would be an effective means of characterizing shallow benthic habitats for infaunal activity and sedimentary conditions, and detecting evidence of impairment within Hempstead Bay. A secondary goal was a one-time assessment of the regional variability in habitat types and conditions. The conditions within benthic habitats can be used as an integrator of the ecosystem's response to varying levels of water quality.

MATERIALS AND METHODS

Field Methods

On 15 and 16 October 2012 sediment profile images were collected at 25 stations (Figure 3). The 25 SPI stations were identified based on historic nutrient gradients and bathymetry as well as targeting areas with heavy, light, and sparse *Ulva* spp. occurrence. Coordinates for each station are in Table 1 and each individual replicate image in Appendix A.

At each station, a digital sediment profile camera was deployed with a live video feed to the surface to monitor camera performance. The digital profile camera, an 18-megapixel Canon 7D enclosed in a pressure-housing with a 45-degree prism and a front surface mirror that reflects an image of the sediment to the camera. A strobe mounted inside the housing provides illumination. The prism was also equipped with a video feed that is used to send images to the surface via cable so that prism penetration can be monitored in real time. The camera/prism system is mounted in a cradle that is secured to a larger frame, which ensures that the prism penetrates the sediment at a 90° angle. For each camera deployment two to three images were taken as the camera operator monitored prism penetration from the surface vessel. Three replicate deployments were done at each station.

Image Analysis

Sediment profile images (SPI) were analyzed visually with data on all features observed recorded in a preformatted spreadsheet file. The least disturbed image at each station was analyzed digitally with a combination of NIH Image-J and Adobe Photoshop™ programs. Data from each image were sequentially saved to a spreadsheet file for later analysis. Details of how these data were obtained can be found in Diaz and Schaffner (1988) and Rhoads and Germano (1986). A description of each parameter measured and evaluated follows.

Prism Penetration - This parameter provided a geotechnical estimate of sediment compaction with the profile camera prism acting as a dead weight penetrometer. The further the prism entered into the sediment the softer the sediments, and likely the higher the water content. Penetration was measured as the distance the sediment moved up the length of the faceplate.

Surface Relief - Surface relief or boundary roughness was measured as the difference between the maximum and minimum distance the prism penetrated. This parameter also estimated small-scale bed roughness, on the order of the prism faceplate width (15.5 cm), which is an important parameter for predicting sediment transport and in determining processes that dominate surface sediments. The origin of bed roughness can be determined from visual analysis of the images. In physically dominated habitats, features such as bedforms and sediment granularity cause bed roughness. In biologically dominated habitats, bed roughness is a result of biogenic activity such as tube structures, defecation mounds, feeding pits, or epifaunal organisms such as hydroids.

Apparent Color Redact Potential Discontinuity (aRPD) Layer - This parameter is an important estimator of benthic habitat conditions, which relates directly to the quality of the habitat (Rhoads and Germano 1986, Diaz and Schaffner 1988, Nilsson and Rosenberg 2000). The aRPD provides an estimate of the depth to which sediments appear to be oxidized. The term “apparent” is used in describing this parameter because no actual measurement was made of the redox potential. It is assumed that given the complexities of iron and sulfate reduction-oxidation chemistry the reddish-brown sediment color tones (Diaz and Schaffner 1988, Rosenberg et al. 2001) indicate sediments are in an oxidative geochemical state, or at least are not intensely reducing (Bull and Williamson 2001). This is in accordance with the classical concept of RPD layer depth, which associates it with sediment color (Fenchel 1969, Vismann 1991). The apparent color RPD has been very useful in assessing the quality of a habitat for epifaunal and infaunal organisms from both physical and biological points of view. The depth of the aRPD layer from sediment profile images is a good indicator of the quality of the benthic habitat (Rhoads and Germano 1986, Nilsson and Rosenberg 2000).

Sediment Grain Size - Grain size is an important parameter for determining the nature of the physical forces acting on a habitat and is a major factor in determining benthic community composition (Rhoads 1974). The sediment type descriptors used for image analysis follow the Wentworth classification as described in Folk (1974) and represent the major modal class for each image. Maximum grain size was also estimated. For muddy to gravel sediments grain size was determined by comparison of collected images with a set of standard images for which mean grain size had been determined in the laboratory. For sediments larger than gravel, individual grains were measured. The following is provided as a means of comparing Phi scale sizes corresponding to sediment descriptors derived from SPI images:

Phi Scale	Upper Limit Size (mm)	Grains per cm of image	SPI Descriptor	Sediment Size Class and Subclass
-6 to -8	256.0	<<1	CB	Cobble
-2 to -6	64.0	<1	PB	Pebble
-1 to -2	4.0	2.5	GR	Gravel
1 to -1	2.0	5	CS	Coarse-sand
2 to 1	0.5	20	MS	Medium-sand
4 to 2	0.25	40	FS	Fine-sand
4 to 3	0.12	80	VFS	Very-fine-sand
5 to 4	0.06	160	FSSI	Fine-sandy-silt
5.5 to 4.5	0.06	160	FSSICL	Fine-sandy-silt-clay
6 to 5	0.0039	>320	SIFS	Silty-fine-sand
8 to 6	<0.0039	>320	SICL	Silty-clay
>8 to 7	<0.0039	>320	CLSI	Clayey-silt
>8	<0.0005	>2560	CL	Clay

Surface Features - These parameters included a wide variety of physical (such as bedforms) and biological features (such as biogenic mounds, shell, or tubes). Each contributes information on the type of habitat and its ability to support benthic organisms. The presence of certain surface features is indicative of the overall nature of a habitat. For example, bedforms are typically associated with physically dominated habitats, whereas the presence of worm tubes or feeding pits would be indicative of a more biologically accommodated habitat (Rhoads and Germano 1986, Diaz and Schaffner 1988). Surface features were visually evaluated from each image and compiled by type and frequency of occurrence.

Subsurface Features - Subsurface features included a wide variety of features (such as infaunal organisms, burrows, water filled voids, gas voids, or sediment layering) that reveal a great deal about physical and biological processes influencing the bottom. For example, habitats with grain-size layers or homogeneous color layers are generally dominated by physical processes while habitats with burrows, infaunal feeding voids, and/or visible infaunal organisms are generally dominated by biological processes (Rhoads and Germano 1986, Diaz and Schaffner 1988, Valente et al. 1992, Nilsson and Rosenberg 2000). The presence of gas voids, which are a mixture of nitrogen and methane from anaerobic microbial metabolism (Reineck and Singh 1975) and associated with high rates of microbial activity. Subsurface features were visually evaluated from each image and compiled by type and frequency of occurrence.

RESULTS AND DISCUSSION

Sediment grain-size ranged from soft fine-sand-silt at two stations, ULVA11a and ULVA12, to more compact medium-coarse-sand-gravel, which occurred at one station, TOH37. The majority of the stations were fine-medium-sand (56% or 14 of 25), followed by fine-sand that occurred at six stations (24%). Coarser sediments occurred only at stations TOH37 and UL5 (Table 2 and Appendix A). There was no evidence of sediment layering and sediment type at a given stations was uniform within the depth of prism penetration (Figures 4 and 5, and Appendix B). A thin layer of flocculent material was observed over the sediment surface at station TOH33. About two-thirds of the stations had varying amounts of shell or shell hash mixed into the sediments. Shell beds were seen at five stations (TOH15, TOH16, TOH24, TOH31, and TOH37). The sediment surface at all stations appeared to be dominated by physical processes.

The measured average station apparent color redox potential discontinuity (aRPD) layer depth ranged from 1.1 cm at station ULVA11a to 4.1 cm at station TOH148 (Table 2). At two stations prism penetration for all three replicate images was too shallow to see the aRPD layer. At station UL5 the aRPD was deeper than 7.5 cm. While the sediments were predominantly sands, there appeared to be moderate to high loading of organic matter to the sediments. This is based on the relatively shallow aRPD layer depths for sand sediments. In pure sand sediments with little organic loading the aRPD layer depths are much deeper and controlled by hydraulic pumping of oxygenated water into the sediments for turbulance created by tide and wave currents (Lohse et al. 1996). It appeared that organic content at UL5 was lower than other stations.

Surface and subsurface faunal activity was low to moderate and likely reflects the predominance of physical processes that shape surfical sediments. Shell and shell hash at the sediment surface and mixed in with the sediment was the most common biobenic feature being present at about two-thirds of the stations. The second most common surface biogenic feature was small tubes that occurred at 68% of stations (17 of 25). Density of tubes was low with 10 stations having few tubes (1 to 6 per image). Highest densities of tube was >25 per image at stations TOH10 and TOH24 (Table 2). Large tubes (>2 mm in diameter) were not common. What appeared to be tubes of the large polychaetes occurred at two stations, *Loimia* spp. at TOH06a and *Diopatra* spp. at TOH24. At station TOH10 what appeared to be tubes of the amphipod *Ampelisca* spp. were present. *Ampelisca* spp. are known to form dense tube mats in Jamaica Bay, to the west of Hempstead Bay, and are important prey items in the diet of juvenile flounders (Franz and Tanacredi 1992).

The number of infauna and burrows per image were low for all stations with 40% of stations having them. Two stations averaged more than one infauna per image (TOH14 and TOH31). Burrows were seen only at stations ULVA02. For pure sandy sediments these numbers for infauna and burrows are typical as unstable sand sediment favor smaller free burrowing species that do not form burrow structures. Sediment feeding voids were also uncommon for the same reason. Feeding voids were seen only at station TOH10. Gas filled voids, indicative of high levels of methanogenesis (Reineck and Singh 1975), were more common being present at 28% of stations (7 of 25). The presence of gas voids also point to high levels of organic loading to the sediments.

Ulva spp. and other algal species were widely distributed being present at 80% of stations (Table 2, Figures 4 and 5). At five stations the density of *Ulva* spp. was high enough to interfere with prism penetration (TOH06a, TOH11, TOH147a, ULVA01, and ULVA04). The density of *Ulva* spp. observed point to problems with nutrients and eutrophication (Fox et al. 2008). At the time of sampling there did not appear to be decomposition of the *Ulva* spp. or other macroalgal species seen in the SPI. But macroalgae decomposition is known to cause hypoxia and anoxia, and interfere with many ecosystem services provided by shallow water benthic habitat. Table 3 lists ecosystems from around the globe that are experiencing water quality and habitat degradation from excess macroalgal growth.

Overall, the shallow benthic habitats sampled by SPI in Hempstead Bay were characterized as fine to medium sands with what appears to be a high level of organic loading. The source of the organic matter may be the abundant *Ulva* spp. and other macroalgae, which in turn point to high nutrient loading to the water column to support the growth of algae. Infaunal activity was typical for dynamic sandy sediments, but the presence of gas voids indicated higher than expected levels of organic matter in the sandy sediments. Variability in benthic habitat types within Hampstead Bay was not great. End members were coarse-sand-gravel and fine-sand-silt, both of which were not common. Most habitats were either fine-sand or fine-medium-sand with varying amounts of shell or shell hash mixed in.

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Table 1. Coordinates for October 2012 SPI stations in Hempstead Bay. Check marks indicate other data collected at SPI stations.

Station	Latitude	Longitude	SPI	Ulva	WQ & Infauna	Flux/ Porewater
Ulva 1	40.6228	-73.6150	√	√	√	√
Ulva 2	40.6043	-73.6347	√	√	√	
Ulva 4	40.6206	-73.6688	√	√	√	√
Ulva 5	40.6006	-73.6807	√	√	√	
Ulva 11a	40.6370	-73.5322	√	√	√	√
Ulva 12	40.6406	-73.5389	√	√	√	
UL 5	40.6227	-73.4958	√	√		
ToH 1	40.6284	-73.6707	√	√		
ToH 2	40.6280	-73.6788	√	√	√	√
ToH 10	40.6240	-73.6612	√	√	√	
ToH 3	40.6105	-73.6537	√	√	√	√
ToH 6A	40.6135	-73.6762	√	√	√	√
ToH RC	40.5960	-73.6759	√	√		
ToH 11	40.5985	-73.6416	√	√		
ToH 15	40.5948	-73.5936	√	√		
ToH 16	40.6068	-73.5730	√	√	√	
ToH 17	40.6136	-73.5588	√	√		
ToH 31	40.6267	-73.5498	√	√	√	
ToH 37	40.6397	-73.5081	√	√		
ToH 24	40.6362	-73.5556	√	√		√
ToH 33	40.6128	-73.5009	√	√	√	
ToH 14	40.6253	-73.5785	√	√	√	√
ToH 147a	40.6203	-73.5316	√	√		
ToH 148	40.6115	-73.5857	√	√		√
ToH 13a	40.6209	-73.5943	√	√	√	

Table 2. Summary of SPI data from Hempstead Bay, October 2012. Quantitative variables are average of three replicates. Categorical variables are maximum that occurred for the three replicate images. Pen = Prism Penetration, SR = Surface Relief, Anaer = Anaerobic, - = Absent, + = Present, IND = Indeterminate, > = Deeper than prism penetration.

Station	Pen (cm)	SR (cm)	aRPD (cm)	Sediment Type	Shell or Shell Hash	Tubes	Algae	Surface Fauna	Infauna (#/image)	Burrows (#/image)	Oxic Voids (#/image)	Anaer Voids (#/image)	Gas Voids (#/image)
TOH01	12.1	0.8	1.8	FS	-	1-6	None		0	0	0	0	0
TOH02	4.2	2.1	3.3	FSMS	+	0	Green/Red	Shrimp	0	0	0	0	0
TOH03	4.2	0.8	1.8	FSMS	+	7-25	Green/Red	Snail	0	0	0	0	0.7
TOH06a	7.0	0.4	1.7	FS	+	1-6	Green	Mussels, Hermit Crab, Loimia Tube?	0	0	0	0	0
TOH10	11.2	0.7	1.7	FS	-	>25	None	Ampelisca tubes?	0.3	0	0.3	0.3	0.3
TOH11	2.8	0.4	IND	FS	-	0	Green/Red		0	0	0	0	0
TOH13a	18.7	1.7	1.7	FS	-	7-25	Green		0.3	0	0	0	5.0
TOH14	7.1	2.5	3.0	FSMS	+	1-6	Green/Red	Mussels	1.3	0	0	0	0
TOH15	8.6	1.4	1.4	FSMS	+	7-25	Green/Red	Mussels, Oyster set on mussel shell	0.3	0	0	0	3.0
TOH16	5.0	2.1	1.4	FSMS	+	1-6	Green/Red	Hermit Crab	0	0	0	0	0
TOH17	5.3	1.7	2.8	FSMS	+	0	None		0	0	0	0	0
TOH24	12.9	21.3	2.0	FSMS	+	>25	Red	Algae on Diopatra tube?	0.3	0	0	0	0
TOH31	7.2	1.0	3.7	FSMS	+	1-6	Green/Red		2.0	0	0	0	0
TOH33	4.1	0.8	2.1	FSMS	+	0	Green/Red	Flocc layer on surface	0	0	0	0	0.3
TOH37	6.3	1.8	2.2	MSCSGR	+	0	Green		0	0	0	0	0
TOH147a	8.7	0.8	2.9	FS	+	0	Green		0	0	0	0	0
TOH148	6.4	1.2	4.1	FSMS	+	0	Green		0	0	0	0	0
TOHRC	9.8	1.2	1.9	FSMS	+	1-6	None	Crab	0.3	0	0	0	0.3
UL5	7.5	0.6	>7.5	MSCS	+	0	Green		0	0	0	0	0
ULVA01	7.3	0.4	3.3	FSMS	-	1-6	Green		0.3	0	0	0	0
ULVA02	7.3	0.7	1.3	FSMS	+	7-25	Green		0	0.3	0	0	0
ULVA04	15.1	0.9	IND	IND	-	1-6	Green		0	0	0	0	18.3
ULVA05	3.7	0.9	>3.7	FSMS	+	1-6	Green	Snail, Hermit Crab	0	0	0	0	0
ULVA11a	17.0	0.9	1.1	FSSI	-	7-25	Green		0.3	0	0	0	0
ULVA12	14.0	1.5	1.3	FSSI	-	1-6	None		0.3	0	0	0	0

Table 3. Examples of ecosystems around the globe that suffer from the combined effects of excess nutrients, eutrophication, hypoxia, and excess macroalgal production.

Condition	Ecosystem	Country	Comment	Reference
Eutrophic	Great Barrier Reef	Australia	Parts of the reef (Burdekin and Fitzroy) have increased macroalgal cover. The increase in macroalgal cover is an important issue as macroalgae competes with coral species for cover and can displace them.	Furnas et al. 2005
Eutrophic	Princess Royal Harbour	Australia	Shellfish populations suffered collapse in the 1980s that was simultaneously with eutrophication and algal blooms.	Peterson et al. 1994
Eutrophic but now improved	Avon-Heathcote/Ihutai Estuary	New Zealand	Benthic fauna negatively impacted by decomposing macroalgae mats. Before the 1970s, blooms of *Ulva* spp. and other algae were common, but thanks to the release of pond effluents with the outgoing tide, most nutrients are taken out of the estuary reducing macroalgal growth.	Murphy 2006
Eutrophic	Jacobs River Estuary	New Zealand	During 2009, decomposing macroalgal mats and hypoxic sulphide-rich sediments were present in many parts of the estuary. Macroalgae was found to be at a higher density in sheltered embayments characterized by restricted water flushing.	Stevens and Robertson 2007
Eutrophic	Mar Menor	Spain	Since 1980s benthic layer has been dominated by high biomass of macroalgae *Caulerpa prolifera* restricting seaweed (*Cymodocea nodosa*) to small shallow patches.	Lloret et al. 2008
Eutrophic	Ghar El Melh	Tunisia	Nutrient-input is considered to have increased during the last few decades as indicated by a species shift from angiosperms to macroalgae.	Rasmussen et al. 2009
Hypoxic	Etang du Prévost	France	Mass mortality of benthos with annual recoloniaztion and reduced aquaculture production.	Guyoneaud et al. 1998
Hypoxic	Palude della Rosa	Italy	Mortality of benthos with annual recoloniazation, eutrophic saltwater pond within the Venice Lagoon that has experienced increased macroalgae and periodic and seasonal hypoxia.	Tagliapietra et al. 1998
Hypoxic	Venice Lagoon	Italy	Mass mortality of benthos. Macroalgal production is very high, and decomposition leads to hypoxia.	Ravera 2000
Hypoxic	Wadden Sea	Netherlands	Mortality of benthos and decrease in percentage of filter feeders in benthic biomass. Hypoxia has caused mortality and eutrophication increased macroalgae.	deJonge et al. 1994
Hypoxic	Mondego River	Portugal	Mortality of benthos with loss of species. Eutrophic conditions have led to macroalgae blooms and the extinction of sea grasses in shallow areas.	Flindt et al. 1997
Hypoxic	Narragansett Bay	US-Rhode Island	Impacts from excess organic loading, including increased incidence of opportunistic benthic species, destroyed benthic habitat, growth of macroalgae, and instances of low dissolved oxygen.	Deacutis et al. 2006

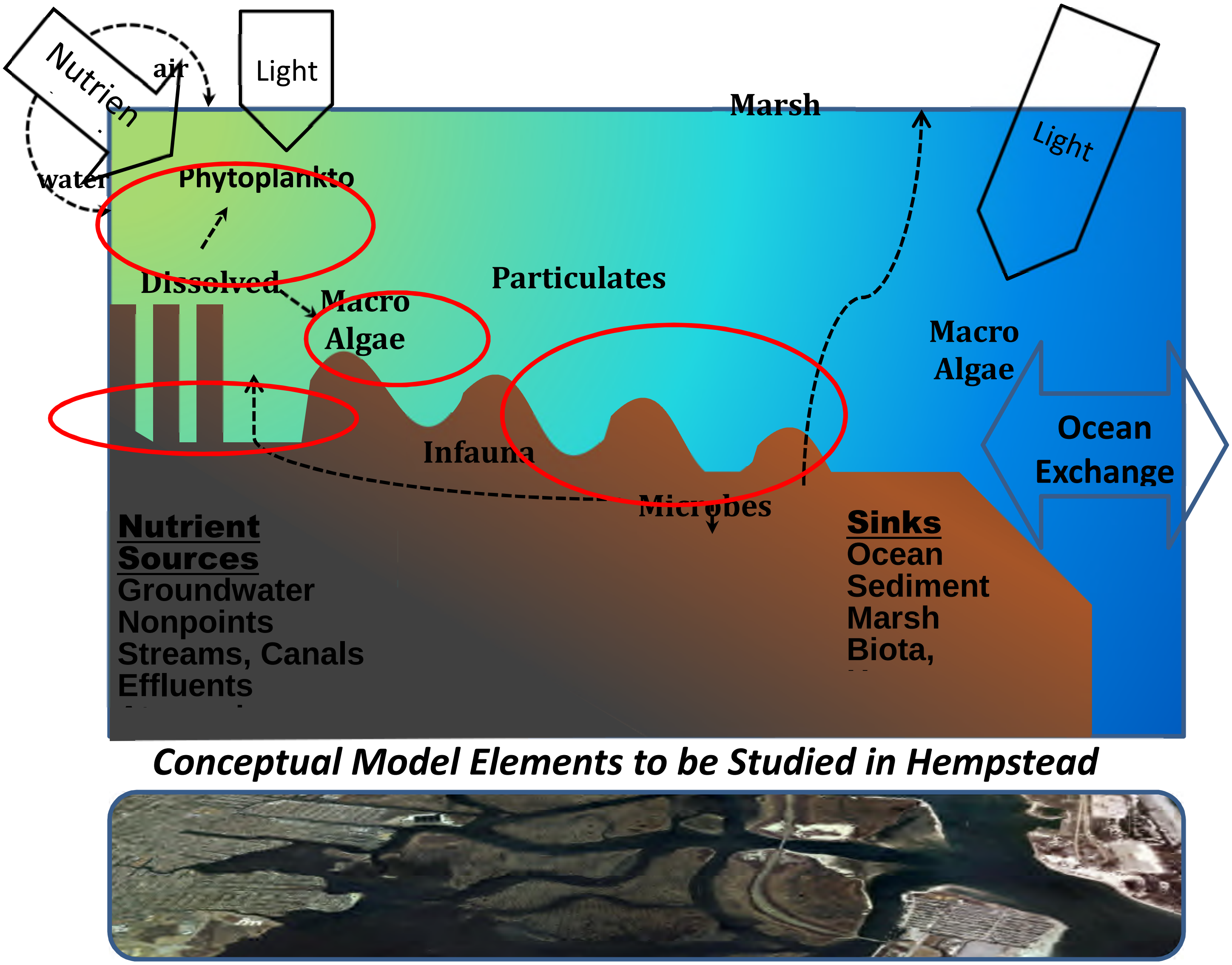

Figure 1. Major components of the Hempstead Bays ecosystem that will be studied under this SAP are highlighted in red.

Figure 2. Hempstead Bay study area.

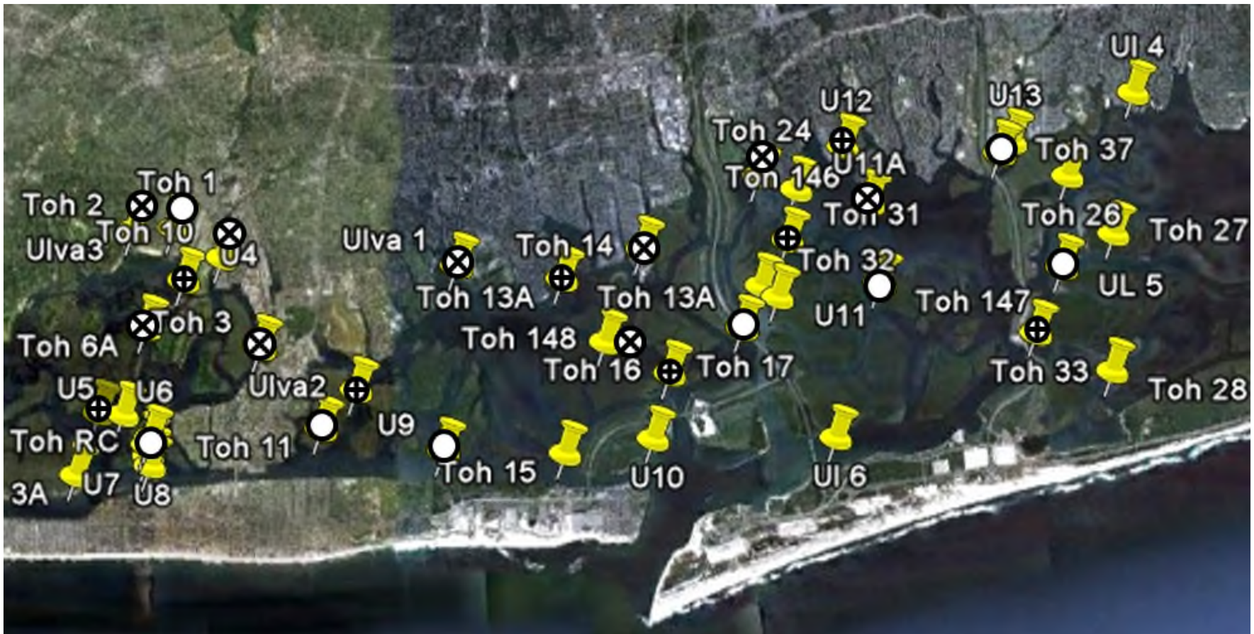

Figure 3. 2012 Hempstead Bay water quality stations. SPI was collected at 25 stations (circles ○). Water quality and Infauna were collected at 15 of the SPI stations (⊕) and benthic flux/porewater measurements at 9 stations (⊗). SoMAS re-occupied stations (yellow pins) for *Ulva* samples.

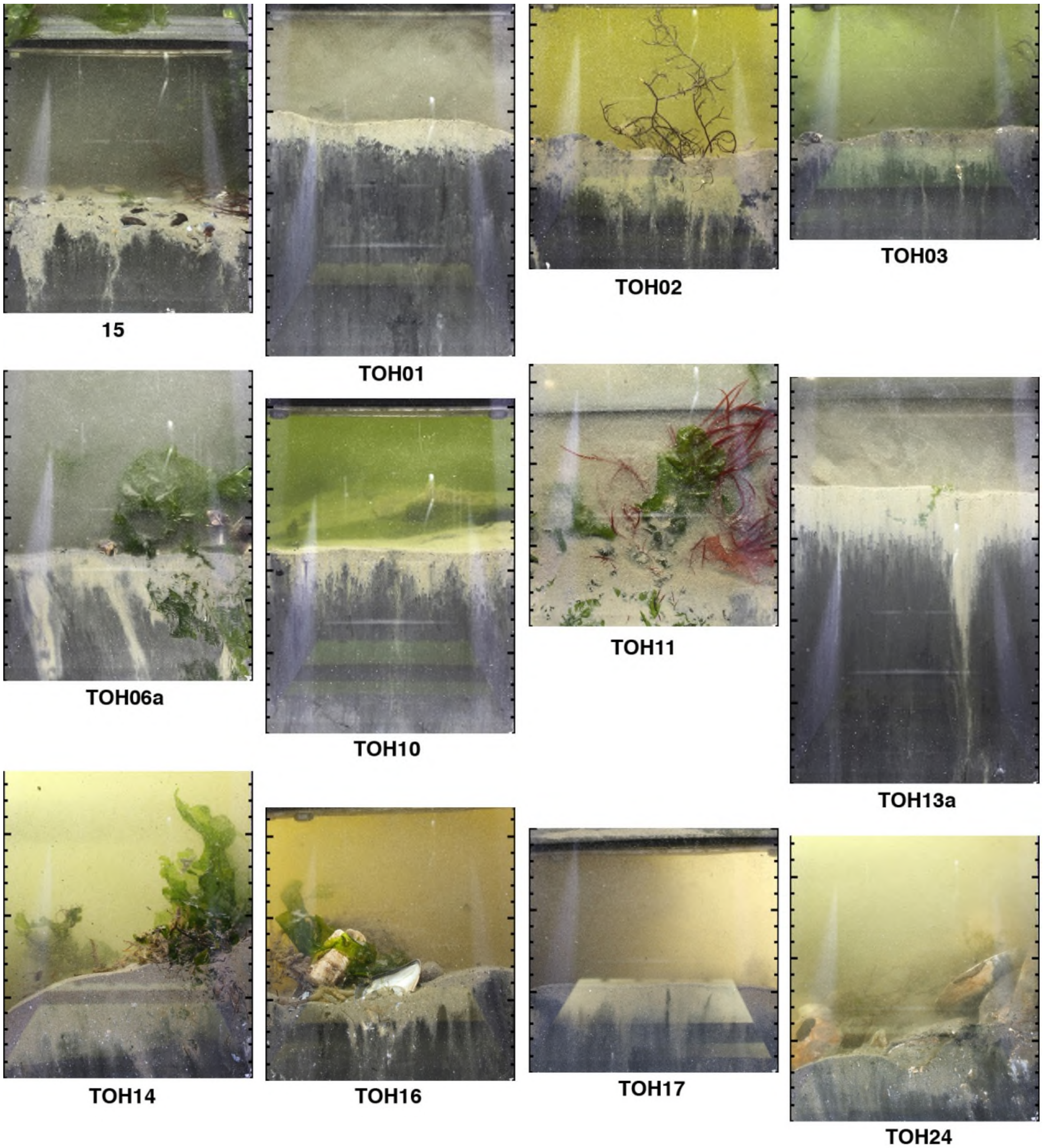

Figure 4. Mosaic of SPI from Hempstead Bay, October 2012. Scale on side of image is in cm. Reflections are from ambient light entering the prism.

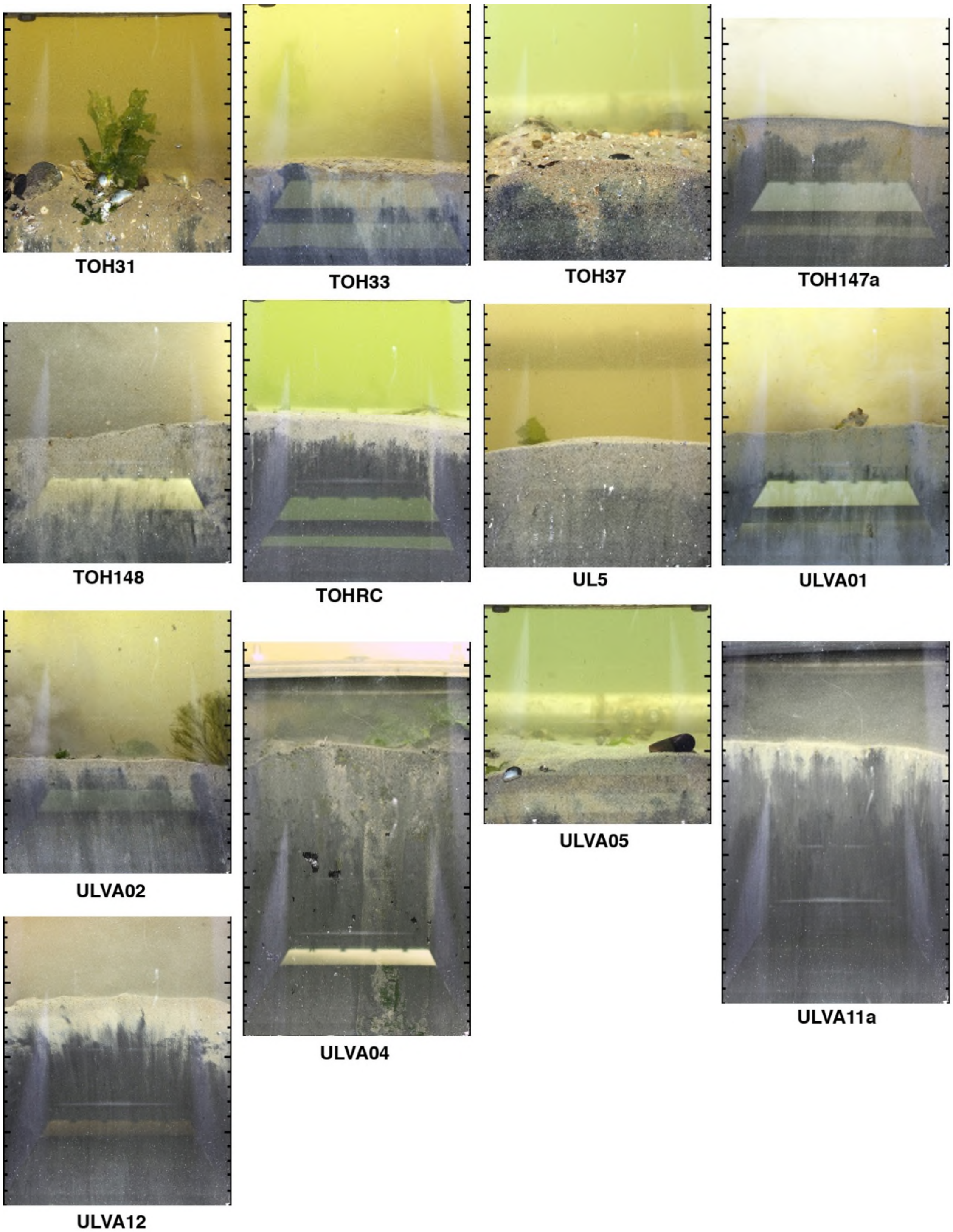

Figure 5. Mosaic of SPI from Hempstead Bay, October 2012. Scale on side of image is in cm. Reflections are from ambient light entering the prism.

APPENDIX D: A SYNTHESIS OF LOADINGS, MONITORING INFORMATION, AND IMPAIRMENTS IN THE WESTERN BAYS

A Synthesis of Loadings, Monitoring Information, and Impairments in the Western Bays

A Report for Battelle Memorial Institute and the New York State Department of State

R. Lawrence Swanson, Robert E. Wilson, Kaitlin Willig
School of Marine and Atmospheric Sciences
Stony Brook University
Stony Brook, NY 11794-5000

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Introduction

The Western Bays are on the New York State Department of Environmental Conservation's 303(d) list of impaired water bodies. Pathogens and nitrogen are thought to be the causes of the impairments.

The School of Marine and Atmospheric Sciences, Stony Brook University has been tasked by the New York State Department of State and Battelle to investigate causes contributing to the impairment(s) and to suggest means by which to alleviate them. This report synthesizes the research that has been undertaken and reported to the appropriate agencies and institutions over the past two years that is particularly relevant to determining causes of impaired uses.

While the study covers the Western Bays, this report primarily considers West Bay since this subset of the bays is clearly the most impacted and should be the focus of remedial attention. We review the relatively recent morphological changes to West Bay, the discharge of water and nitrogen from several sources, residence times of water, aspects of golf courses and the Oceanside landfill, and the major impairments to the bay along with potential remediation measures.

Background

The Western Bays are a subset of the South Shore Estuary Reserve, which includes the Long Island South Shore lagoonal system in both Nassau and Suffolk Counties and from the east, Shinnecock, Moriches, Great South, South Oyster, and Hempstead Bays. The Western Bays are South Oyster Bay and Hempstead Bay.

Hempstead Bay is geographically roughly subdivided from east to west into sub-embayments: East Bay (73.5°W to 73.6°W), Middle Bay (73.6°W to 73.65°W), West Bay (73.65°W to 73.7°W) and thence to East Rockaway Inlet. Reynolds Channel connects the western half of this longitudinal distance. This report focuses on West Bay.

These sub-embayments are marine lagoonal ecosystems and are typified by shallow depths, and by narrow, shoal and shifting connections to the ocean, and limited river runoff. They are influenced by groundwater discharge. Their circulation, while tidally driven, can be dominated by wind driven circulation on occasion -- particularly when there are strong east-west components in the wind direction.

Hempstead Bay, including the shoreline, despite its vast, healthy-appearing wetlands, has been heavily manipulated over the past century. Its ecosystem functioning has been altered by extensive dredging, filling, and bulkheading. Further, it experiences the adverse impacts of waste disposal, sewage effluent, leachate from landfilling, and industrial operations.

In the late 1800s, several barrier islands originally protecting West Bay from the ravages of the Atlantic Ocean were joined (Shelter Island joined to Long Beach in the vicinity of Broad Channel; Luce Inlet filled on Long Beach) (Figure 1). Fill from the then bay of Far Rockaway was used to further extend Long Beach to the present-day Atlantic Beach. East Rockaway Inlet was created some 1.2 miles to the west of the east end of Shelter Island.

In the early 20th century, William H. Reynolds pumped sand from what is now known as Reynolds Channel to sculpt the current Long Beach Island -- the channel was deepened and widened, wetlands filled, and south facing ocean beaches nourished. That portion of Island Park west of the bisecting channel was still marsh as indicated on the 1929 Coast and Geodetic Survey nautical chart No. 579. Certainly, by the late 1940s, the major channels (i.e., Broad Channel) within the bay had been dredged. These dredged materials were then used to fill and harden many of the marsh islands along with a considerable portion of the Bay's north shore.

Commencing as early as 1927, communities around the bays recognized the need for treating sewage. However, it wasn't until the 1950s, as the county population built out, that a significant portion of the region was connected by sewerage. But with these improvements, other problems developed including where to appropriately and economically discharge the treated effluent and how to treat and dispose the remaining sewage sludge. Additionally, domestic and industrial water was being withdrawn from the aquifer system, but the used water not returned as sewage effluent was discharged to the bays and ocean. Thus, the region began to experience reductions in stream flow and lowering of the groundwater table (Scorca, 1997), which would be accompanied by some salt water intrusion into the area.

Other than the creation of Reynolds Channel, the dredging of Hewlett Bay has been perhaps the most significant physical alteration in the area. It is here that severe hypoxia is often observed in part because it is poorly flushed. Figure 2 depicts the channel bathymetry throughout the Western Bays.

Ciappetta (undated) reports that between 1926 and 2004, some 2750 ha (6795 acres) of the 5461 ha (13,494 acres) of marsh existing in the entirety of Hempstead Bay in 1926 have been lost. He concludes that 47 percent of this loss took place by 1983 when Nassau County was extensively dredging and filling in the bay. More recently, the losses have been associated with erosion of the marsh fringes.

Sewage Effluent, Streamflow, Groundwater Discharge, and Golf Course Runoff

Sewage Effluent

Until the early 1950s, sewage for the most part was discharged in the Western Bay's drainage basin primarily via cesspools and septic systems. The Village of Freeport, in 1927, was the first Nassau County municipality to discharge treated sewage to marine waters in the Western Bays, according to Scorca (1997). This plant was connected to the Cedar Creek Plant in 1980 (Scorca, 1997). However, Nesbit (1933) states that in the same year (1927), a small Water Pollution Control Plant (WPCP) commenced operation for a population of about 500 in Atlantic Beach. A sewer district for the area was formed around 1930. The plant consisted of sedimentation and chlorination, and treatment was principally for the purpose of protecting shellfishing and swimming (Nesbit, 1933). During winter, with a reduced population, sewage sludge was pumped to the ocean. During peak summer flows, the discharge was around 0.5 MGD. In 1948, MacCallum (1948) recommended that the plant be upgraded to secondary treatment. The earliest that this could have been achieved was 1950. Today, the Greater Atlantic Beach WPCP operates as a secondary facility and still discharges about 0.5 MGD (IEC, 2008) to East Rockaway Inlet. The June 2012 reported total nitrogen (TN) effluent concentration was 13.85 mg/L (Alexander Michaelis, personal communication, July 24, 2012, The Greater Atlantic Beach Water

Reclamation District). Thus, the mass load of TN[1] to the bay from this facility is about 26 kg/d, corresponding to 0.03 ton/d or 11 ton/y (Table 1).

In 1951, the Long Beach WPCP was constructed as a secondary treatment facility. The disinfected effluent was discharged to Reynolds Channel at a rate of 6.4 MGD (Dvirka and Bartilucci, 2009). The mean discharge over the period of late 2010 to August 2012 was 4.9 MGD. Total nitrogen concentrations varied between 19.02-24.36 mg/L (William Knotholt, Long Beach WPCP). The estimated annual TN mass load is 157 tons/y.

The Bay Park WPCP was completed in 1952 as a secondary treatment facility. It initially discharged only 8.8 MGD (Scorca, 1997). Today, the discharge is about 49.8 MGD (Figure 3) as summarized from data supplied by Bay Park (Joseph Davenport, Chief Sanitary Engineer, Nassau County Department of Public Works). The volume of influent has increased as more of the county was connected to sewers, population grew, and small WPCP discharges were diverted to Bay Park. The outfall (the block) is located in about 10 ft (3 m) of water just south of South Black Banks Hassock. Between February 2007 and July 2010, this WPCP received over 50 notices of violation of its State Pollutant Discharge Elimination System (SPDES) permits. Forty of these effluent violations were for settleable solids and total suspended solids (Nassau County, 2010).

The Bay Park facility is also infamous for a most unfortunate accident occurring on June 2, 1976 when the two sewage sludge storage tanks on Pearsalls Hassock exploded as a consequence of unauthorized personnel playing with fireworks near the tanks. About one million gallons of sewage sludge spewed out onto the hassock and into East Rockaway Channel (Swanson et al., 1977). Nassau County dredged the area around the tanks over the next month, removing about 26×10^6 gal of sludge, sand, and mud. Evidence of sludge in sediment was found as far away as 800 ft from the tanks (Swanson et al., 1977).

In 2009 and 2010, the reported TN influent and effluent concentrations averages were 50.7 mg/L and 30.0 mg/L, respectively. Thus, there is about a 41 percent reduction in TN at this secondary treatment plant. The reported average effluent flow from Bay Park corresponding to these concentrations was 49.8 MGD. The TN load to the bay was 5585 kg/d or 6.16 tons/d (18,177 ton/y). The average concentration of TN in the water column at the block (station 4, Figure 4) over the period 2000-2010 was 5.53 mg/L. There is nearly a five-fold instantaneous dilution as the Bay Park effluent enters the mixing volume in Reynolds channel.

The Village of Lawrence operates a secondary WPCP that discharges about 1.2 MGD to the west end of Reynolds Channel (IEC, 2008). Quarterly TN data from 2008-2011 ranged from

[1] TN = Nitrate + Nitrite + Organic N + NH_4

11.47-22.16 mg/L (Joseph Davenport, Chief Sanitary Engineer, Nassau County Department of Public Works), thus delivering 30 ton/y to West Bay.

There is also a very small discharge into the channel near the west end of Jones Island from Jones Beach (0.04 MGD) (IEC, 2008). Long Island's first ocean outfall was constructed as part of the

Cedar Creek WPCP, which began functioning in 1975 (Scorca, 1997). The diffuser is about 2.2 nautical miles south of Jones Beach Island and some 5 nautical miles east of Jones Inlet.

There is approximately 56.4 MGD or 20,589 million gallons per year of secondarily treated sewage effluent discharged to West Bay. By way of comparison, Jamaica Bay received about 200 MGD of secondarily treated sewage effluent in 2008 (IEC, 2009).

The hamlet of Point Lookout (2010 population of 1219) on Jones Inlet is the only community around West Bay that is unsewered and using septic systems. Dean (2005) estimates that a human with a normal diet generates about 0.044 lbs/d (20 g/d) of nitrogen. At the indicated population, this equates to 9.8 ton/y. Assuming a 50 percent denitrification in groundwater, up to 4.9 ton/y could be discharging to the very eastern end of Reynolds Channel. See Table 1.

Stream and Groundwater Discharges

The U.S. Geological Survey (USGS) has estimated stream flow and groundwater flow along with nitrogen loadings for the south shore coast of Long Island (Monti and Scorca, 2003). In their analysis, they have subdivided the coast to include Sewer District II in Nassau County, which is essentially the north shore of West Bay (Figure 5).

Pines Brook and East Meadow Brook were estimated to have mean discharges over a number of years of 3.45 ft^3/s (25.8 gal/s) and 12.70 ft^3/s (95.0 gal/s), respectively. Groundwater flows were modeled and divided by county and also by shallow-water and deep-water discharges. For Nassau County, the simulated or modeled annual discharge was 40.5×10^6 m^3/y ($10{,}700 \times 10^6$ gal/y) for shallow, groundwater discharge (0-125 ft), while the deep, groundwater discharge (126-200 ft) was estimated to be 18.5×10^6 m^3/y ($4{,}900 \times 10^6$ gal/y) (Monti and Scorca, 2003).

Some 53 percent of the area in Sewer District II that the USGS considers the applicable groundwater contributing area is in West Bay. Thus, the estimated groundwater contributions to the north shore of West Bay are about 21.5×10^6 m^3/y (5670×10^6 gal/y) and 9.8×10^6 m^3/y (2600×10^6 gal/y) for shallow and deep discharges, respectively.

Based on the flows in Table 1, streamflows and groundwater flows are about 59 percent of that of sewage secondary effluent entering West Bay.

For the period 1992-1996, Monti and Scorca (2003) calculated TN loads from streams entering the Sewer District II area (Pines Brook and East Meadow Brook). The mean values for those years were 5.8 ton/y (5262 kg/y) and 5.5 ton/y (4990 kg/y), respectively. The area known as Sewer District II has had sewerage since the 1950s. As a consequence, the groundwater nitrogen concentrations have remained relatively stable since the area has not received large septic/cesspool loads (Monti and Scorca, 2003).

The median concentration for shallow ground water was 3.31 mg/L (minimum = 0.102 mg/L, maximum = 17.81 mg/L) over the period 1952-97. The deep wells had a median concentration of 0.14 mg/L (minimum < 0.01 mg/L, maximum 2.14 mg/L).

The total annual calculated TN discharge based on 1983 simulated discharge in Nassau County was 172 ton/y (156,000 kg/y) for shallow wells and 300 ton/y (2700 kg/y) for deep wells. Assuming Sewer District II is 53 percent of the contributing area, some 91 ton/y (82,555 kg/y) and 1.6 ton/y (1452 kg/y) enter West Bay from shallow and deep ground water along the north shore, respectively.

From all quantified sources used here, 96 percent of the TN loading is contributed by WPCPs. Eighty eight percent of the total is discharged by Bay Park. In 2002-2003, the four WPCPs in Jamaica Bay discharged some 6680 ton/y of TN (NYC DEP, 2007) or about 2.7 times that discharged by WPCPs in West Bay.

Golf Courses and the Oceanside Landfill

Five golf courses border West Bay (Figure 6). The Lido Golf Club (public, 18 holes, about 100 acres)[2] is on Reynolds Channel. The Seawane (private, 18 holes, 127 acres), and the Bay Park Golf Clubs (public, 9 hole, 96 acres) are along the northern edge of Hewlett Bay. The Lawrence Country Club is on Bannister Bay (semi-private, 18 holes, about 82 acres) and the Woodmere Club (private, 18 holes, 110 acres) is sited on Brosewere Bay. Circulation in these bays is limited, particularly in the latter. All these golf clubs (515 acres) contribute nitrogen to the local waterways as part of their fertilization of greens, fairways, and roughs. The Buzzards Bay National Estuary Program (undated) estimated in 2009 that some 133.6 kg/ha (119.2 lbs/acres) are spread on golf courses each year; of this, 26.70 kg/ha (23.83 lbs/acre) of TN per year leach to ground water (Buzzards Bay NEP, undated).

This rate yields an annual loading of 1.2 tons of TN to Reynolds Channel, 2.7 ton/y to Hewlett Bay, and 2.3 tons/y to the western end of West Bay. Some of the latter load is probably accounted for in the USGS groundwater loads.

The estimated total annual TN load contributed by golf courses to West Bay is 6.1 tons. This is small compared to those loads measured and tabulated in Table 1 since it was not determined directly.

New York State Environmental Conservation Law, Section 35-105, Article 17, Title 21) prohibits, beginning in 2012, lawn fertilizer within 20 ft of surface waters (with some

[2] With the exception of the Bay Park Club and Woodmere Club, acreages are scaled from topographic maps.

exceptions) and it may not be applied between December 1 and April 1. This may reduce the nitrogen contribution from golf courses and lawns in the future, but certainly not during the time of this study.

The closed 125 acre Oceanside Landfill is located just to the northeast of Island Park. It is bordered to the south, southeast, and east by Barnums Channel, Garrett Lead, and Doman Canal, respectively. The landfill was active between the early 1960s and May 1988. It was completely capped by 31 October 1998 (Weston Solutions, 2010).

An Environmental and Facility Monitoring Plan (EFMP) has been underway since 2009 and will continue through 2013. There are now nine groundwater monitoring wells on the landfill property, which are monitored once or twice per year. Ammonia (as N) is the only nitrogen species measured. Surface water measurements of ammonia are also obtained by the Town of Hempstead on about the same schedule at 10 sites around the landfill plus one in Middle Bay. Ammonia concentrations at all 11 sites continuously exceed the 2008 state standard of 0.035 mg/L.

As part of the monitoring endeavor, personnel seek out leachate seeps. If seeps are observed, they are to be sampled. None were found from 2008-2010. In the annual reports covering 2006-2010**,** a statement has been made based on this particular monitoring aspect that the landfill "does not appear to be negatively impacting the adjacent estuarine water quality."

There are no discharge data available in the monitoring reports, so it is not possible to estimate loadings to the surrounding waterways.

On 13 September 2012, SoMAS personnel undertook a radon reconnaissance survey that commenced in East Rockaway Channel (to the east of Hewlett Bay), thence to Hewlett Bay, thence south along East Rockaway Channel around Island Park up the channel west of Garrett Marsh to the vicinity of Oceanside Landfill (Figure 7). In this case, the concentrations were used in a relative sense -- not to quantify groundwater flows. Radon, a gas, is prevalent in ground water, but not surface waters. Thus, when detected, it is a good indicator of groundwater seepage (USGS, 2004).

In particular, radon has been used successfully to quantify fresh plus saline groundwater discharge in soils similar to those on Long Island. Mulligan and Charette (2006) demonstrated the technique in glacial outwash soils consisting of sand and gravel in the vicinity of Waquoit Bay on Cape Cod.

In West Bay, there was no evidence of significant groundwater seepage based on radon concentrations in the channel along the shore of the landfill, although it is doubtful that radon would be in landfill leachate. Water column nitrogen concentrations in these areas are likely derived from point and runoff sources (including the landfill), via advection and mixing.

Insights from Monitoring and Modelling

A time series analysis of surface observations obtained by the Town of Hempstead Department of Waterways and Conservation was completed at every station (Figure 8) for different time periods (depending on gaps in the record). The results of this analysis for the parameters of temperature, salinity, dissolved oxygen, and several species of nitrogen are briefly summarized and depicted below.

The temperature time series covered 1975 to 2010. Averaged over all stages of tide, the temperature increased at stations 1, 2, 3, 4, 11, 19, and 28. These stations are located at the southern end of the Western Bays. During low tide, the temperature significantly increased at all stations except for 11, 12, 13, 15, and 18, where no significant change was observed. All the stations that had no significant increase are located in the center of the Western Bays. During high tide, temperature only increased significantly at station 15 and decreased at station 10. Overall, 86 percent of the 28 stations analyzed have increased in surface temperature (mostly at low waters) from 1975-2010.

The time series for salinity for the 28 individual stations extended from 1989-2010. At all stages of the tide, salinity has increased significantly at stations 3, 6, 14, 16, 17, 21, 24, and 25. There was no significant change in salinity for any other station. Stations 3 and 6 are close to the Bay Park sewage outfall. In 1984, annual daily averaged discharge from the Bay Park WPCP reached a peak flow of 69.6 MGD. After 1984, the flow declined considerably. From 1989-2010, the flow averaged approximately 54.8 MGD, a nearly 15 MGD decrease from 1984. The increase in salinity at stations 3 and 6 during this time period may reflect the decline in flow from the Bay Park WPCP. Station 4, located at the Bay Park outfall, did not have a significant increase in salinity. However, the average salinity from 1989-2010 at station 4 was more than 1 psu lower than stations 3 and 6; 28.5 psu compared to 29.7 psu and 29.6 psu, respectively. The other stations that experienced increases in salinity are in a cluster north of Jones Inlet, probably reflecting changes in communication with the ocean. Modelled, tidally-averaged surface and bottom salinities throughout West Bay are plotted in Figure 9. Note the relatively low salinity near station 4, the Bay Park outfall.

Due to many gaps in the record, the dissolved oxygen time series is the shortest, 2003-2009. During this period, surface dissolved oxygen concentration decreased significantly at stations 18, 19 and 28, and remained the same at all other stations. Dissolved oxygen in surface waters does not appear to be an issue except possibly at night for which there were no records from the Town.

The three nitrogen species, nitrate, nitrite, and ammonia were observed from 2000-2010 by the Town of Hempstead, Department of Conservation and Waterways. Nitrate decreased at stations 1, 3, 6, 14, and 28, and remained the same at all other stations. Nitrite decreased at stations 16, 18, 20, and 24 but increased at station 4 (Bay Park outfall). Ammonia decreased at stations 1, 18, 19, 20, 21, and 28 and increased at stations 5, 6, 7, 8, 9, and 10. Overall, nitrogen species have increased at the Bay Park outfall and stations directly north of the Bay Park outfall. This suggests that some water discharged from the outfall is transported north instead of moving along Reynolds Channel and quickly flushed out of the ocean inlets.

The U.S. Geological Survey since 2010 has maintained a continuously operating water quality station (01311143) located in Hog Island Channel at the Masone Beach Pier in the Village of Island Park (http://nwis.waterdata.usgs.gov/nwis/UV?cb_62619=on&cb_62619=on). The sensor is 3-10 ft below the water surface depending on the tide. Tide elevation data are collected along with temperature, salinity, turbidity, dissolved oxygen, and nitrate data. Nitrate as nitrogen concentration seems to vary mostly between 0.15-0.6 mg/L. Nitrate concentrations are plotted (Figure 10) as a function of tidal elevation for summer and winter over the period June 2011 through May 2012. The mean of the mean summer and winter concentrations is about 27 percent of the long-term TN in the surface waters measured by the Town of Hempstead at this location (station 7).

Nitrate is a predominant species of nitrogen released in groundwater discharges. Thus, nitrate is used here to elucidate whether or not ground water is an important nitrogen source.

The data indicate that the summer nitrogen concentrations are independent of tidal elevation as the slopes of the regression lines are statistically zero. The winter slope of -0.0058 mg/L/ft is statistically significant at 5 percent. Practically it isn't. The mean concentration in summer is 0.24

mg/L and winter is 0.37 mg/L. The standard deviations for the two seasons are ± 0.095 and ± 0.130 mg/L, respectively.

Conventional wisdom is that any ground water and associated nitrogen discharge at the site would be greatest at low tide when the head of water is least (Mulligan and Charette, 2006). Maximum nitrogen concentrations would occur out of phase with the tide when discharge is greatest and mixing volume least.

The semidiurnal M_2 tidal constituent with a period of 12.42 h was fitted to the entire time series, after subjecting the data to a fifth order highpass Butterworth filter with a cutoff of 34 hours. The amplitude was 0.001 mg/L (less than 1 percent of the summer and winter mean, respectively) with an r^2 of 2.1×10^{-4} mg/L, indicating that the tidal signal is not significant. The semidiurnal nitrogen signal, even though exceedingly small, is out of phase with the tide (Figure 11).

Nitrogen-rich ground water is not entering West Bay at a significant rate at this location. The nitrogen concentrations are the same at high tide as at low. The tidal volume is not diluting a nitrogen source. Model simulations of particles released at the block during low tide stages indicate a considerable fraction transported northward through the web of channels including that by the USGS sensor at Island Park (Figure 12).

The circulation of West Bay has been altered considerably as its morphology has been modified over the past century or so. The creation of Long Beach changed the phasing of the tidal currents. Deepening of channels has done so as well. Hewlett Bay is now a deep depression with limited circulation. Even since many of the major physical changes were completed in West Bay, anthropogenic activities have occurred, resulting in an increase in the mean tide range of 0.3 m (1 ft) in Hewlett Bay over the period 1934 to 1975 (Progress Report #3, Task 1, 2011).

Reconfiguring islands, deepening channels, dredging inlets, hardening shorelines, and filling marshes have collectively changed how the waters of the New York Bight enter and fill West Bay. Maximum flood and ebb tidal current speeds are generally weak throughout West Bay (0.3 ms/ or less) with the exception of west Reynolds Channel (Figure 13). Ultimately, its residence time (time elapsed for a particle to exit the region of its initial position as determined from hydrodynamic modeling) has increased (see Figure 14 for an example of initial position domain). Even Reynolds Channel, other than at its east and west extremities, has residence times that fall between about 130-200 h. Note in particular the extremely low tidal current velocities (0.15 m/s on flood, 0.05 m/s on ebb) in Reynolds Channel near 73.65°W longitude coupled with the relatively long residence times (180 h).

Progressing northward through Broad Channel, East Rockaway Channel, and Hog Island Channel, the residence times increase exceeding 200 h. In Hewlett Bay, the time is 250 h or greater. It takes 10-20 tidal cycles to remove a material release in much of West Bay.

The long residence times obtained from modelling are substantiated by USGS salinity observations at Hog Island Channel at the Island Park site. Figure 15 shows that salinity fell abruptly from about 28 psu to 23 psu around August 14, 2011. The salinity didn't recover to 28 psu until nearly the end of November 2011. The National Oceanic and Atmospheric Administration (NOAA, 2011) recorded a rainfall of 19.8 cm (7.8 in.) on August 14 at John F. Kennedy (JFK) Airport. This rainfall event, with associated stormwater runoff, depressed salinity and it remained low with other downpours following on August 21, 26, and 27 (5.2 cm (2.06 in.), 6.8 cm (2.69 in.), and 5.9 cm (2.34 in.), respectively). Tropical storm Irene occurred on August 26 and 27. Overall, following August 14 and through the remainder of the month, JFK Airport, just 11 km from Hog Island Channel, experienced 40.7 cm (16.02 in.) of rain. Normal August precipitation is 11.79 cm (4.64 in.) (NOAA, 2011). September, October, and November rainfall was much closer to normal. It took 79 days (154 tidal cycles) for the salinity to rebound, following tropical storm Irene, to that prior to the extensive downpour in mid-August.

Quaternary Ammonium Compounds (QACs) used in commercial products such as fabric softeners, cosmetics, and hair care products are sewage specific residues. They are found in sewage sludge and waste water. Thus, their detection in sediment is an indication that the sediments and overlying waters have been exposed to sewage.

Measurements of QACs have been made throughout the Western Bays. The highest concentrations are located in muddy sediments within a radius of 4 miles of the Bay Park WPCP outfall (Figure 16). The majority of the high concentrations (> 20,000 ng/g) of total QACs are in West Bay, substantiating the notion that materials introduced there are not readily removed from West Bay due to the substantially long residence times.

Thus, from four different lines of evidence -- a hydrodynamic modeling approach, accumulation of sewage specific indicators in sediments, long-term monitoring by the Town, and continuous USGS water quality monitoring of salinity and nitrate -- it appears that water and materials are retained for long periods of time in West Bay and that rigorous tidal flushing does not occur. This West Bay-wide sluggishness is a major factor contributing to its impairments.

Hewlett Bay

Hewlett Bay experiences growth of extensive phytoplankton biomass relative to the rest of the Western Bays and in some instances growth of Harmful Algal Blooms (HABs). Using Hewlett Bay water, phytoplankton growth is stimulated by nitrogen relative to phosphorous and silicon (Gobler, 2013). During late spring and throughout the summer, some portions of this bay are hypoxic and even anoxic (Gobler, 2013) (Figure 17).

Hewlett Bay is bordered by Macy Channel on the west and East Rockaway Channel on the east. Hog Island Channel is the primary deep channel connecting Hewlett Bay to Reynolds Channel. The northern edge of the bay accommodates the Seawane Golf Club, the Bay Park Golf Club, as well as the Bay Park WPCP. As mentioned in the previous section, the two golf courses may contribute some 2.7 ton/y of TN to this bay.

The 1879-80 Coast and Geodetic Survey Topographic Sheet (1471A) depicts that the entirety of what is Hewlett Bay was a sand and gravel bottom, exposed at low water (Figure 18). In the early 20th century, a creek was dredged to create Macy Channel which was to provide a navigable waterway into, by then, the relatively deep passage along the northern edge of Hewlett Bay. According to the 1929 Coast and Geodetic Survey Nautical Chart (No. 579), there were depths in excess of 50 ft in Hewlett Bay, but much of the bay was still shallow and exposed at low water.

The 1950 edition of Chart No. 579 indicates that much of the shallows of the bay remained but that Broad Channel had been dredged (controlling depths about 5 ft). Chart No. 579 (1965) suggests that much of the shallow area had been dredged to 18 ft, Swift Creek had been cut to connect Hewlett Bay to Hog Island Channel. The maximum depth, as recorded by SoMAS in its 2010-2011 bathymetric survey, was 50 ft (15.3 m) relative to NAVD. Thus, a deep depression was created from a naturally shoal area to improve navigation, to provide access of West Bay north shore communities to the ocean, and develop fast land from marsh. This practice was somewhat common along the south shore of Long Island and has led to creation of several notorious polluted water bodies. Grassy Bay in Jamaica Bay is comparable, as is the northern portion of the Forge River in Moriches Bay.

Summer and winter average salinities in Hewlett Bay (stations 8, 9, 10) were the lowest (1989-2009) for all Town of Hempstead monitoring stations other than immediately adjacent to the Bay Park WPCP outfall. Summer average nitrate measurements (2000-2009) in Hewlett Bay at the same stations were high compared to other monitoring stations except at the Bay Park outfall. Winter nitrate was generally higher at stations 8, 9, 10 than most other locations and considerably greater than at the Bay Park outfall.

Exploratory radon measurements, an indicator of groundwater flow, were taken at depth near the bottom of Hewlett Bay (near stations 8, 9); no evidence of significant groundwater seepage was detected (Figure 7). However, the northern reach of East Rockaway Channel does have sources of ground water which, along with surface flows, are contributing fresh water to Hewlett Bay, and thus some nitrogen load as well. As indicated in Table 1, these loads are relatively small compared to WPCP loads.

Hydrodynamic modeling shows that neutrally buoyant particles released in Hewlett Bay have a long residence time there, exceeding 250 h. In fact, about 17 percent of the particles remained within the bay after 200 h. Further, particles released at the Bay Park WPCP are advected northward through the complex system of channels (Broad Channel, Hog Island Channel, Nums Channel, Ramscat Channel) into Hewlett Bay where they can accumulate due to its long residence time.

Thus, it appears that particles and nitrogen can be released, perhaps in relatively low concentrations, into Hewlett Bay, but because of its long residence time will accumulate before it can be utilized. It is not surprising that primary production (even some HABs) and biomass are generated within the bay wherein it tends to be trapped, contributing to oxygen depletion.

Impairments

Integrating the results of tasks associated with this Western Bays project, several geographic areas experience impaired waters -- those waters where designated uses are not fully supported. For West and Middle Bays, impairments include shellfishing closures (not discussed) and eutrophication. Hewlett Bay, a sub-region of West Bay, is an area of particular concern and is identified separately. It experiences the same general impairments but its altered seascape exacerbates the consequences.

The causes of eutrophication are excessive nitrogen inputs and limited flushing. Generally, the evidence for eutrophication includes excessive growth of the macroalgae *Ulva*, high concentrations of TN, sediments enriched in organic carbon, reduced light penetration, and harmful algal blooms.

West Bay

The primary impairment is the proliferation of *Ulva* or sea lettuce. Phytoplankton biomass is excessive; there are now occurrences of harmful algal blooms (HABs), and eutrophication-induced acidification, at least seasonally. West Bay can be considered eutrophic. Another significant impairment is that nearly all of the Western Bays are closed to shell fishing due to pathogenic pollution. The latter has not been considered in this assessment.

Ulva, which covered many of the channels of the Western Bays in the late 1960s, now appears to be more extensive, perhaps covering as much as 60 percent of the channels in portions of West Bay. Additionally, *Ulva* is growing throughout the year, not confined to the warmer spring and summer months. Udell et al. (1969) reported that in 1967 there were 198 hectares (490 acres) of *Ulva lactuca* in the Western Bays producing 7.84 tonnes per hectare (3.50 tons per acre) totaling 1553 tonnes (1715 tons). This *Ulva* coverage was about 10 percent of the 4800 acres (11,500 acres of aquatic environment -- 6700 acres of marshland) of open water in the Western Bay.

The issues associated with *Ulva* in West Bay include rafting in and clogging of channels, accumulating and decaying on some beaches, creating respiratory problems in some people at the sites of accumulation, and possibly negatively impacting some businesses due to foul odors. The extensive growth of *Ulva* also has interfered with the normal cycling of nutrients in the local ecosystem. Removing stranded *Ulva* is a financial issue for taxpayers.

Macroalgae, including *Ulva sp.,* has proliferated globally as nutrient enrichment associated with coastal development has increased (Valiela et al. 1997). Park (2007) identified a number of ecological consequences from *Ulva*. Included are adverse aesthetics and recreational issues from the fouling of beaches. Odors may result from the release of dimethyl sulfide creating public health issues. The *Ulva* mats can cover the bottom smothering benthic fauna or perhaps preventing settlement of fauna on the sea floor.

Gil (2005) found that *Ulva rigida* was effective in taking up secondary-treated sewage nitrogen in a laboratory setting. Ammonia was more rapidly assimilated than nitrate and nitrite. In the Western Bays, *Ulva* coverage nearly maps one to one with water column nitrate concentrations (Figure 19). It is also limited by light penetration requiring 1 percent of the irradiance at the water surface for survival (Gobler, 2013). Thus *Ulva* is restricted to the shallower portions of West Bay since phytoplankton shading often restricts the depth to which light can reach (Gobler, 2013).

West Bay, during summer months, experiences extensive phytoplankton biomass, which, based on experiments, appears to be driven by nitrogen. The HABs may also cause ecosystem disruption and have been observed in Hewlett Bay and Middle Bay. *Heterosigma akashiwe* (isolated in 2011) is a rapidophyte (eukaryotic algae) and under certain conditions has been found to be toxic to fishes. It is often found in coastal waters. *Peridinium spp*., a dinoflagellate, was also identified in 2011. It has caused fish kills in some locations and can be odiferous when in bloom conditions. Occurrences of both species in Hewlett Bay and Middle Bay took place in late spring through early fall. We are not aware of any recent fish kills or other consequences from these HABs in the Western Bays.

Eutrophication-induced acidification in summer and fall 2011 and 2012 yielded pH/CO_2 levels that have been shown to inhibit the survival of finfish and shellfish larvae in laboratory studies. We have no evidence that survival of finfish or shellfish larvae occurred in the Western Bays as a result of this phenomenon (Gobler, 2013).

Hewlett Bay

The predominant impairment in Hewlett Bay is low dissolved oxygen (Figure 17). However, the HAB and eutrophication-induced acidification issues pertain to the bay as well.

Hewlett Bay has been physically manipulated over the past 120 years so that many of the problems that are experienced there are compounded relative to the rest of West Bay. In the 1880s, the bay was essentially dry at low water according to the U.S. Coast and Geodetic Survey topographic map (Figure 18). Today, charted depths in Hewlett Bay are as much as 15 m (50 ft).

This artificial basin is a reservoir for oxygen demanding materials (organic matter) due to sluggish circulation. During some seasons of the year, the nitrogen concentrations in the water column are slightly elevated relative to other areas of West Bay. Preliminary radon measurements in the bay show that groundwater inputs, as calculated by the USGS (Table 1), are not uniformly distributed. Ground water does not appear to be a large freshwater source within Hewlett Bay; thus groundwater-derived nitrogen is not a big factor contributing to oxygen depletion there. However, the residence time of the bay is in excess of 250 h. It is most likely that nitrogen-rich water is advected into the bay from the northern reach of East Rockaway Channel and the channels to the south where it accumulates, stimulating primary production and ultimately oxygen depletion.

Impairment Causes and Remediation

Excessive nitrogen discharge, almost exclusively from WPCPs, is the general cause of the impairments in West Bay. See Table 1. Indirect assessments suggest that while groundwater nitrogen contributes to the overall eutrophication of the bay, it is not the primary cause of the observed impairments. The overwhelming nitrogen source is the Bay Park WPCP. This source should be removed from the bay.

However, any consequences of excessive nitrogen discharged to the bay are exacerbated in Hewlett Bay by its topography. The dredged depression is a sink for oxygen-demanding materials and water and material introduced to the bay are not readily flushed.

Several options to remediate the impairments are worthy of investigation. These include:

- constructing an ocean outfall for the Bay Park discharge. If this is done, it would probably be worth the investment to hook the Long Beach outfall to the same ocean discharge,

- implementing tertiary treatment (nitrogen removal) at the Bay Park WPCP,

- discharging the Bay Park effluent to groundwater.

In addition to reducing the major source of nitrogen to West Bay, the impaired conditions of Hewlett Bay would be greatly improved by filling Hewlett Bay to a depth where the residence time would be significantly reduced and the flushing improved.

Another approach to alleviating eutrophic conditions could be to decrease the residence times in West Bay by recreating a stabilized inlet in Long Beach. The best location might be near the old inlet near Broad Channel.

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	Flow galx10^6/y	TN Ton/y
Lawrence WPCP[1]	442	30
Greater Atlantic Beach WPCP[2]	182	11
Long Beach WPCP[3]	1788	157
Bay Park WPCP[4]	18,177	2247
Point Lookout septage	67 (estimated)	4.9
Pines Brook[5] (mean of 1992-1996)	814	5.8
East Meadow Brook[5] (mean of 1992-1996)	2996	5.5
Groundwater, Sewer District II[5]		
Shallow	5670	91
Deep	2600	3
Total	32,736	2555.2

Table 1. Discharge and nitrogen loading to West Bay.

[1] Quarterly data (November, February, May, August) 2008-2011.

[2] Spot data, Summer 2012.

[3] Monthly flow data and quarterly nitrogen data, December 2010-August 2012.

[4] Monthly data, 2010-2011.

[5] Monti, J. and M.P. Scorca. 2003.

Figure 1. Portion of the Hyde and Company map including the Western Bays published by Hyde and Company, Brooklyn, NY, 2nd Edition, July 1897. "Based upon recent U.S. Coast Surveys, together with local maps on file. Supplemented by Careful Territorial Observations."

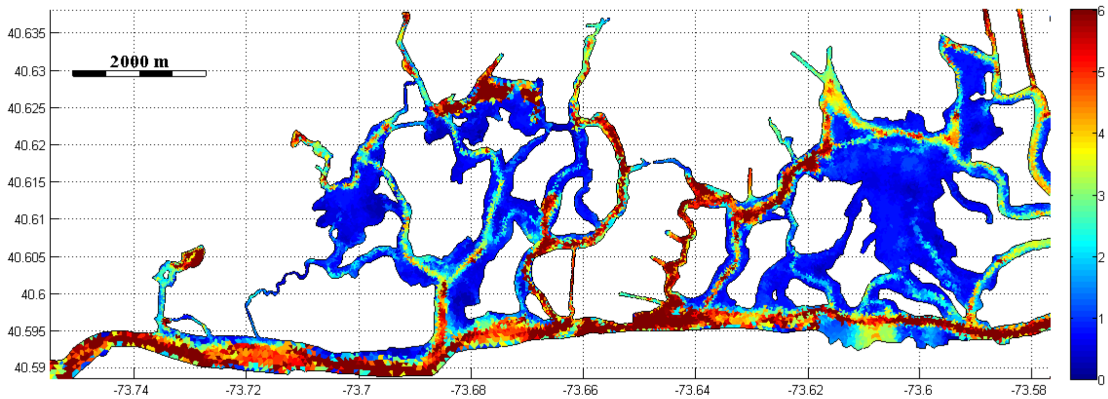


Figure 2. Model bathymetry (m) within Hewlett Bay.

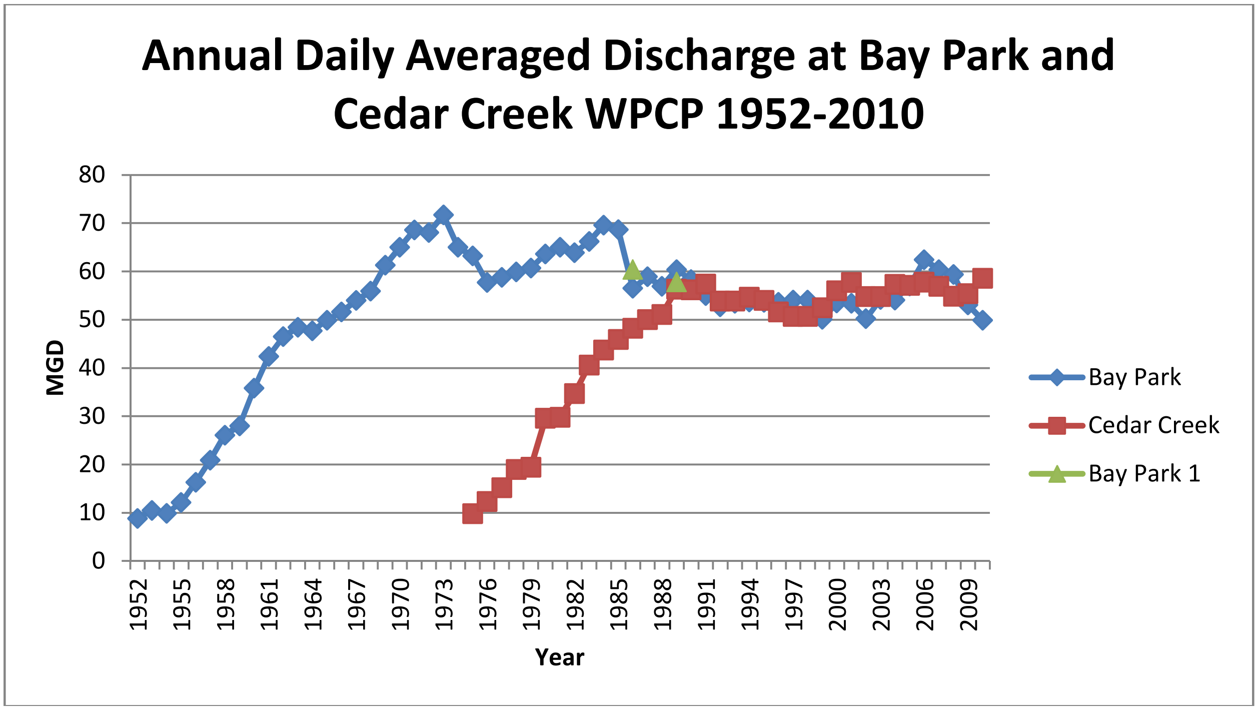


Figure 3. Annual daily averaged discharge at Bay Park and Cedar Creek WPCP 1952-2010.

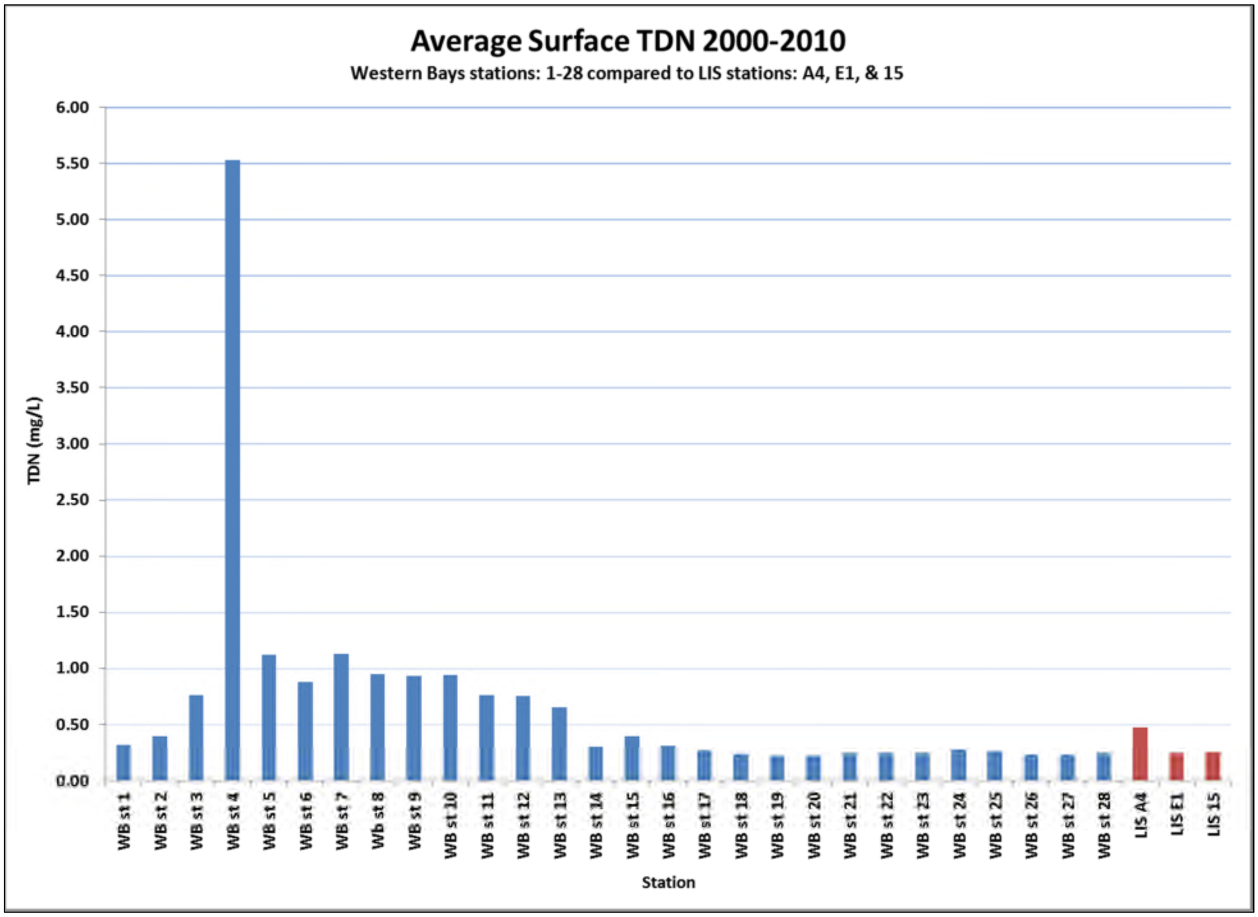


Figure 4. Average surface TDN from 2000-2010 at 28 Western Bays stations and three Long Island Sound stations.

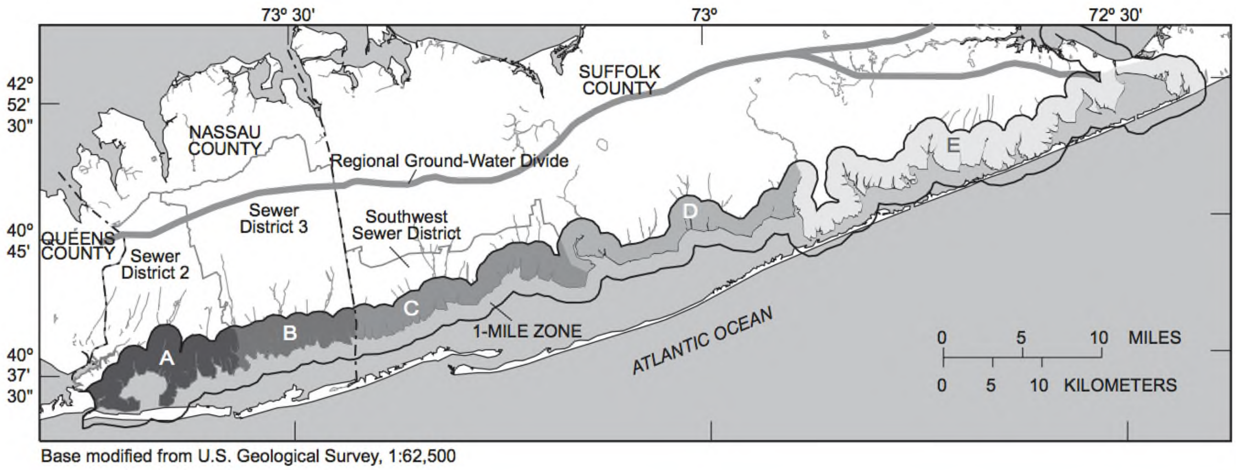


Figure 5. Location of geographic zones (including Sewer District II) used by the USGS to estimate nitrogen loadings.

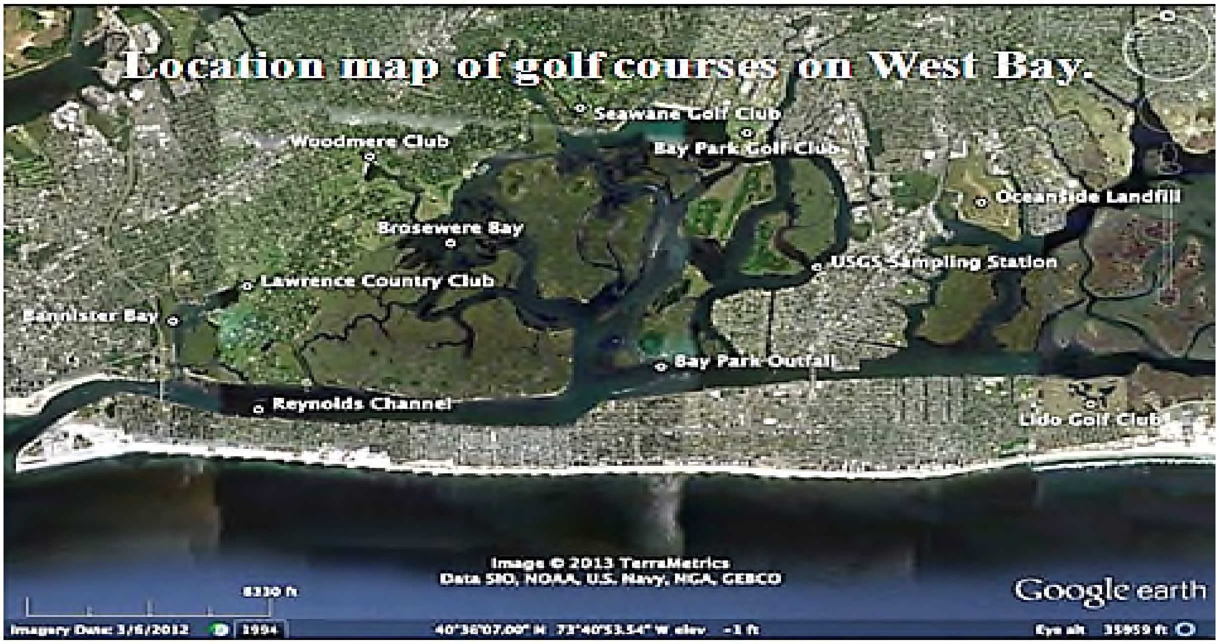


Figure 6. Golf courses located along West Bay.

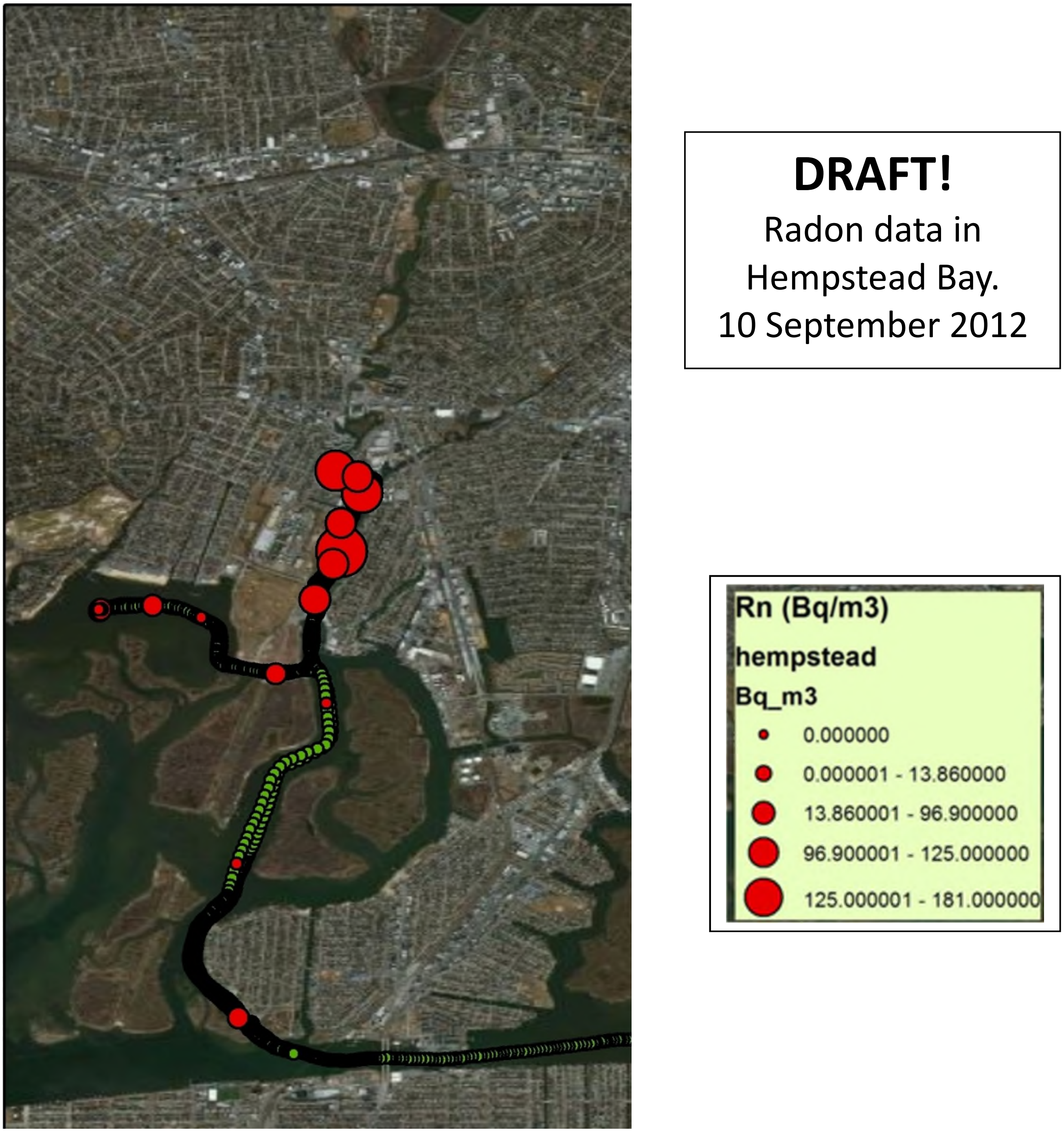


Figure 7. Radon concentrations in Hempstead Bay (Bq/m^3). 10 September 2012. From H. Bokuniewicz.

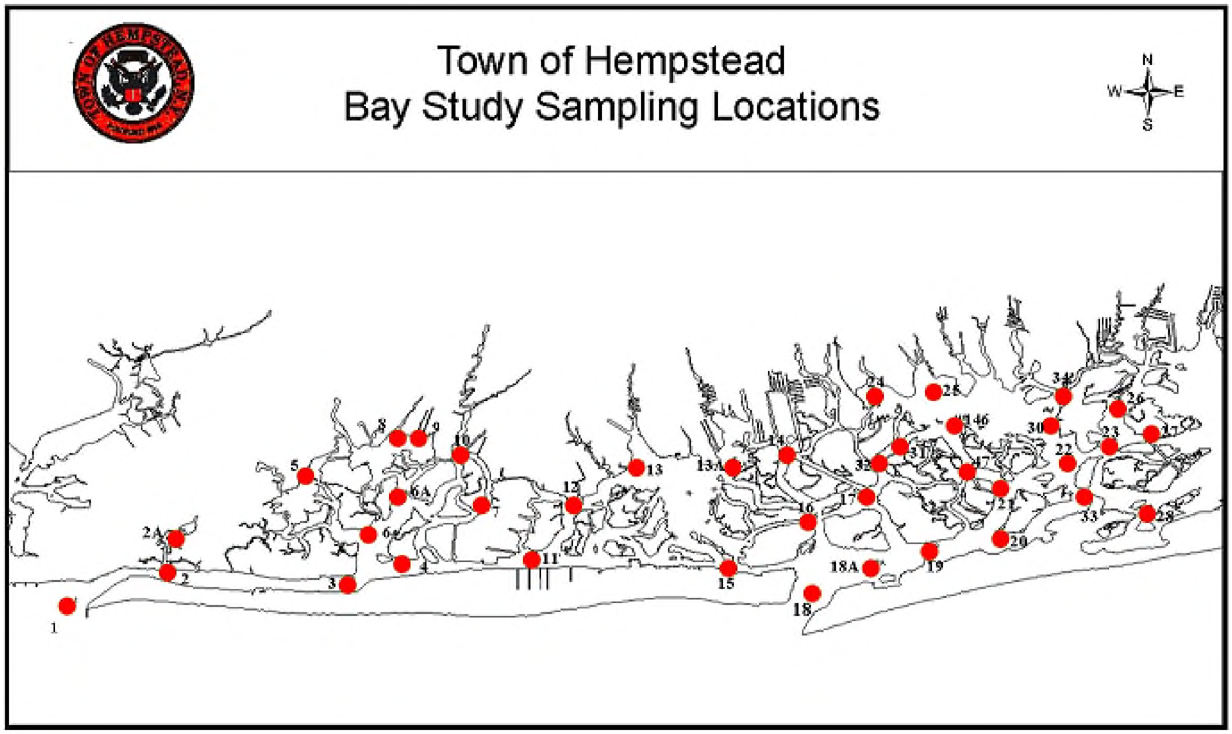


Figure 8. Map of Town of Hempstead sampling stations.

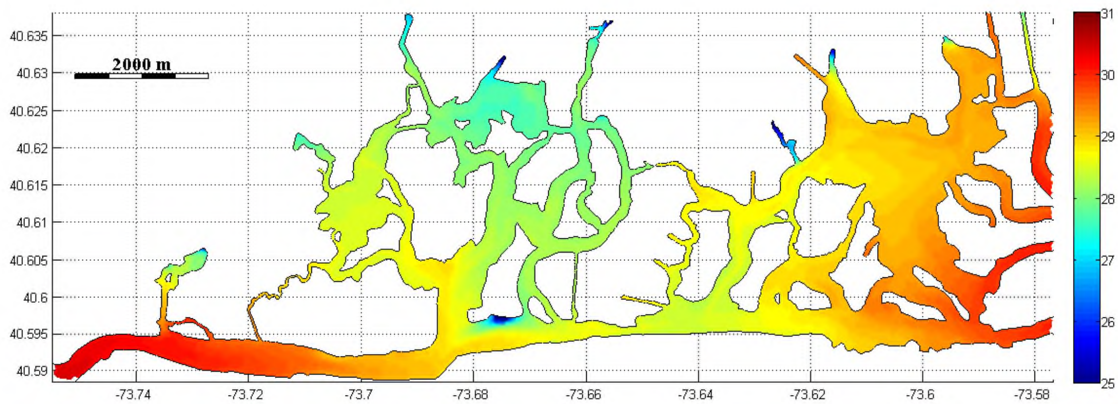


Figure 9a. Tidally averaged surface salinity (psu) within Hempstead Bay.

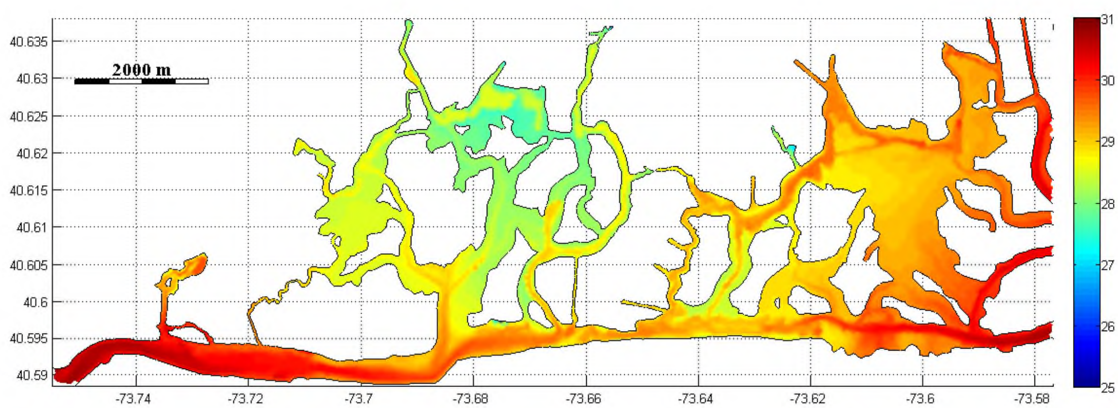
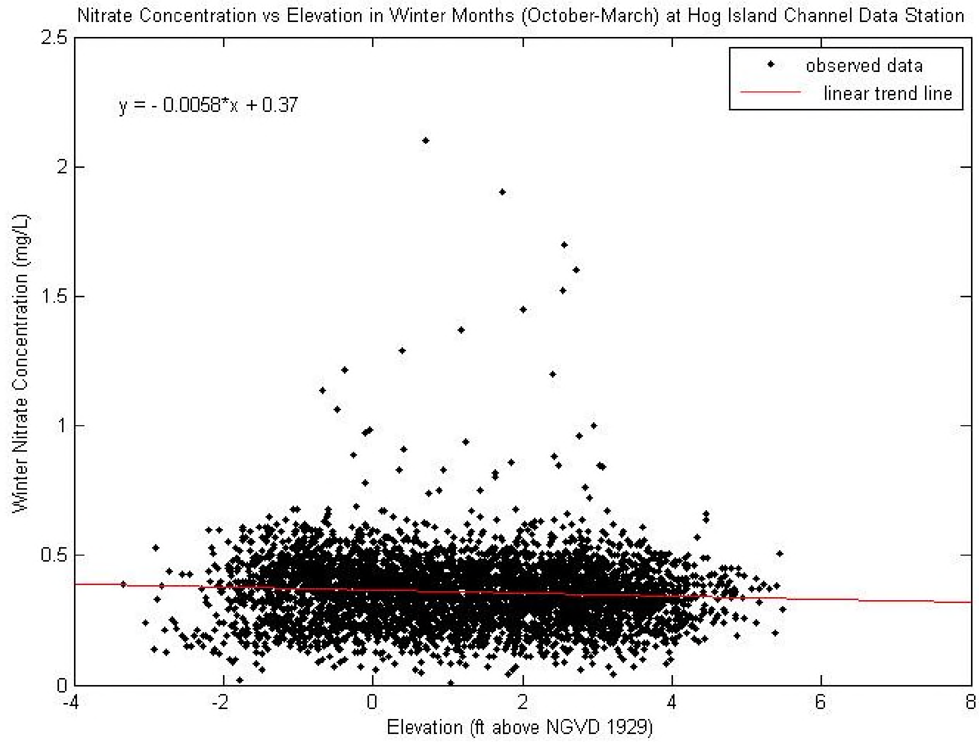


Figure 9b. Tidally averaged bottom salinity (psu) within Hempstead Bay.



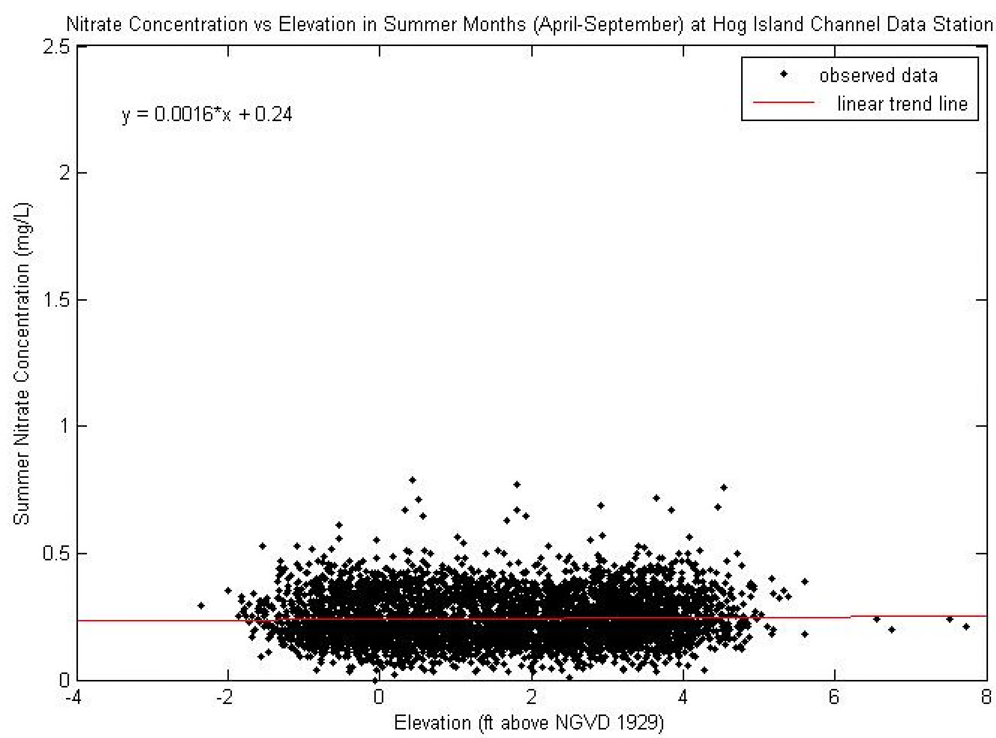


Figure 10. Summer and winter nitrate concentrations (mg/L) versus tidal height (feet) at USGS monitoring station, Hog Island Channel at Island Park, June 2011-May 2012.

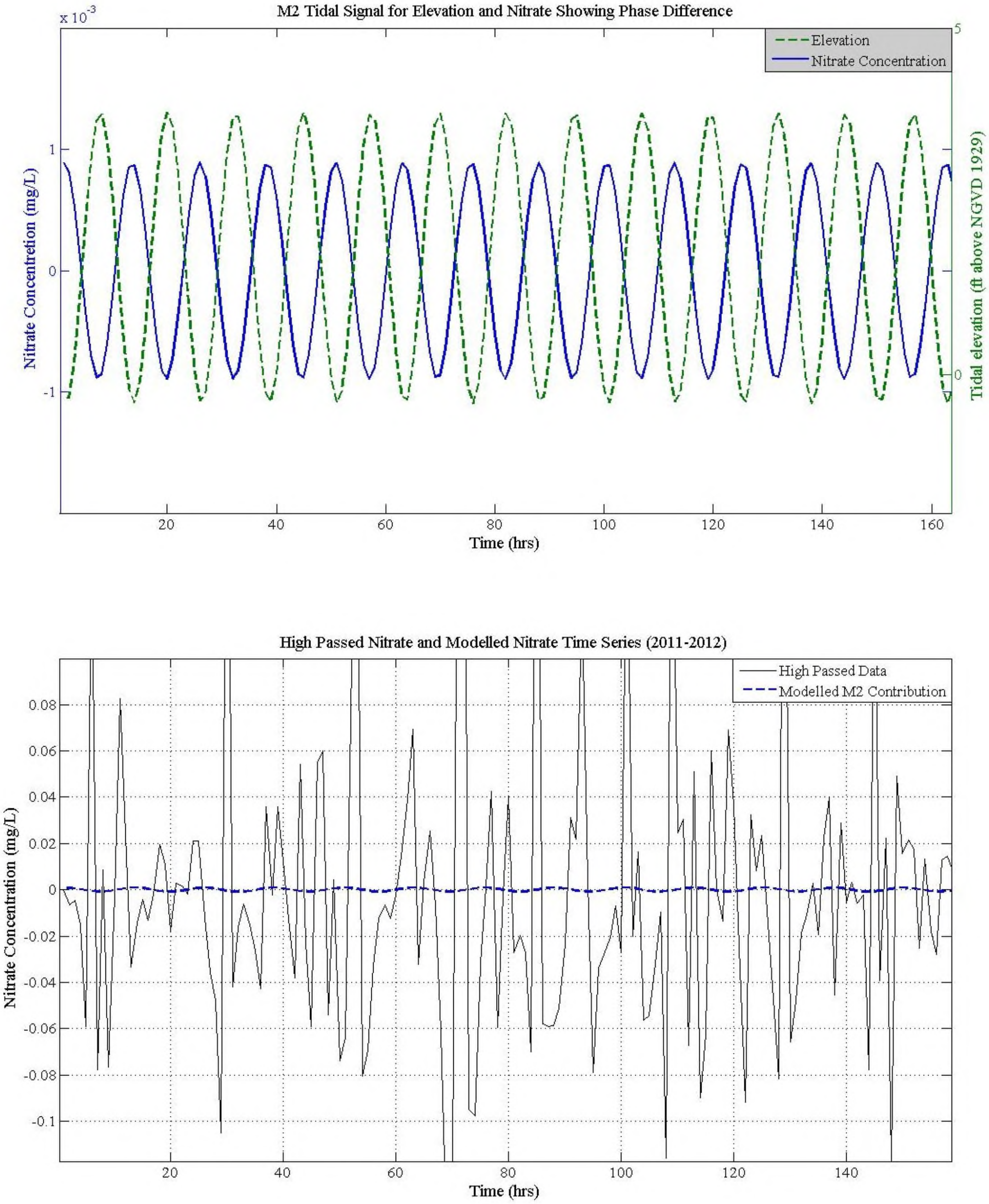


Figure 11. Top panel. Sample of time series of the semidiurnal tidal elevation signal (dashed line) along with that of NO^3 ($mg/Lx10^{-3}$). Data from USGS station at Island Park.

Bottom panel. Semidiurnal NO^3 signal (dashed line) plotted with the highpassed NO^3 concentrations.

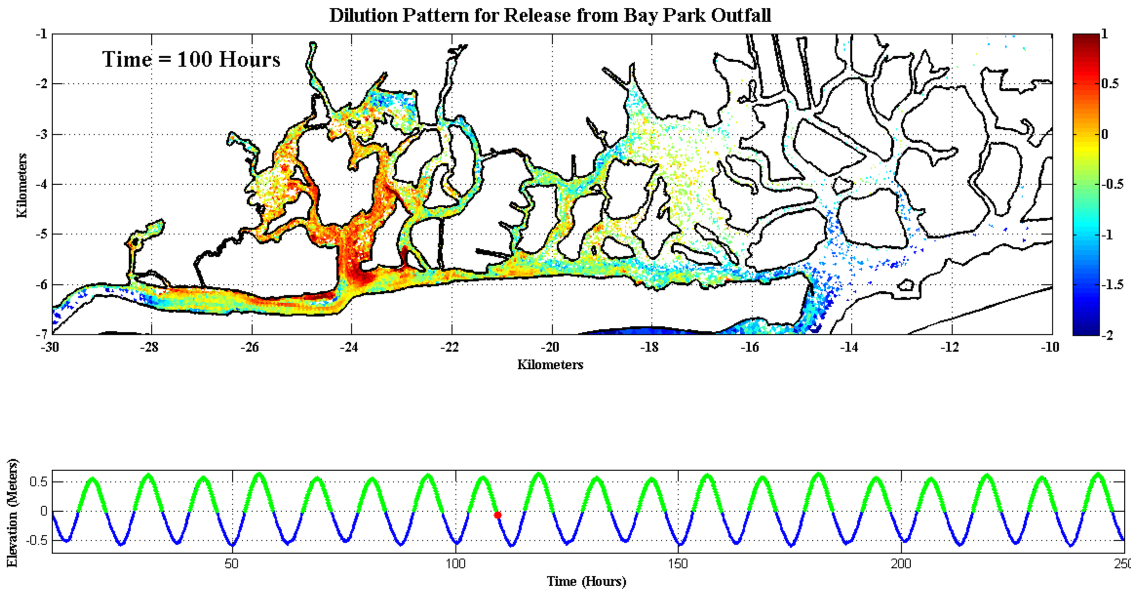


Figure 12. A dilution pattern for partial releases at the Bay Park Water Pollution Control Plant outfall in Reynolds Channel. The pattern is for releases in the low water portion of the tidal cycle and is about 110 h. following the initial release.

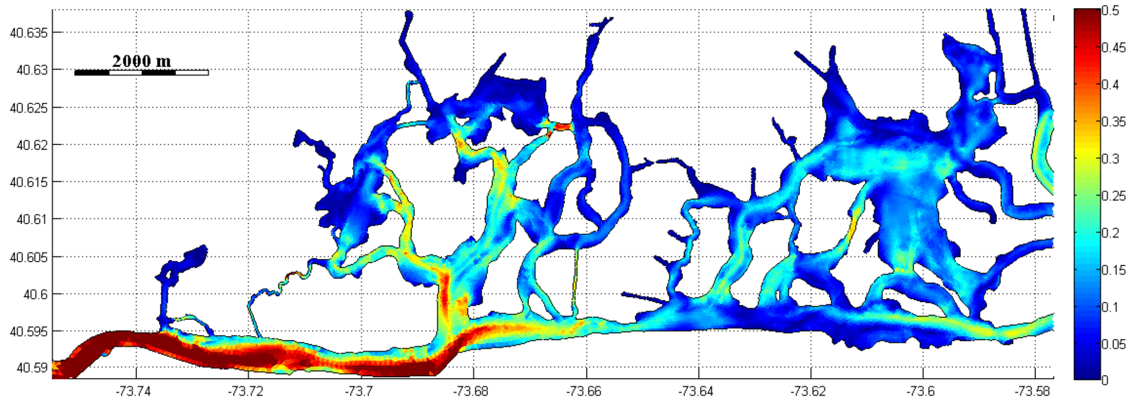


Figure 13a. Surface current speed (m/s) during maximum flood.

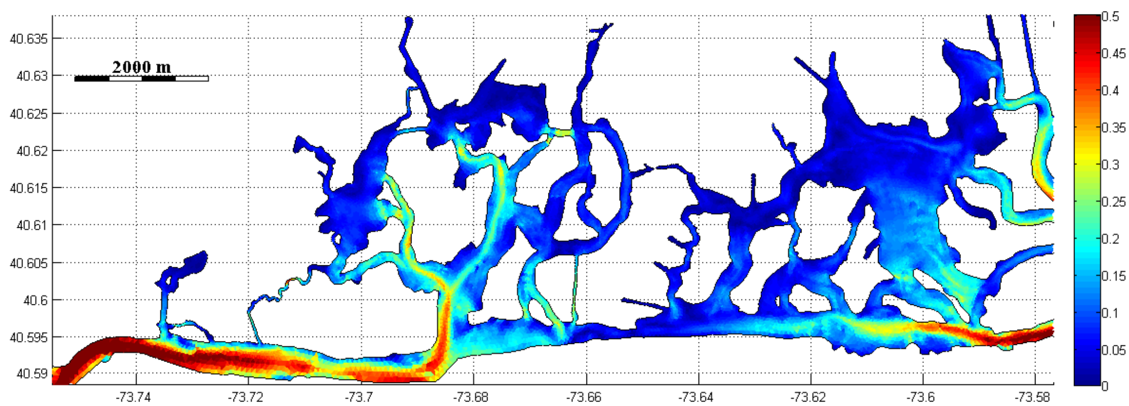


Figure 13b. Surface current speed (m/s) during maximum ebb.

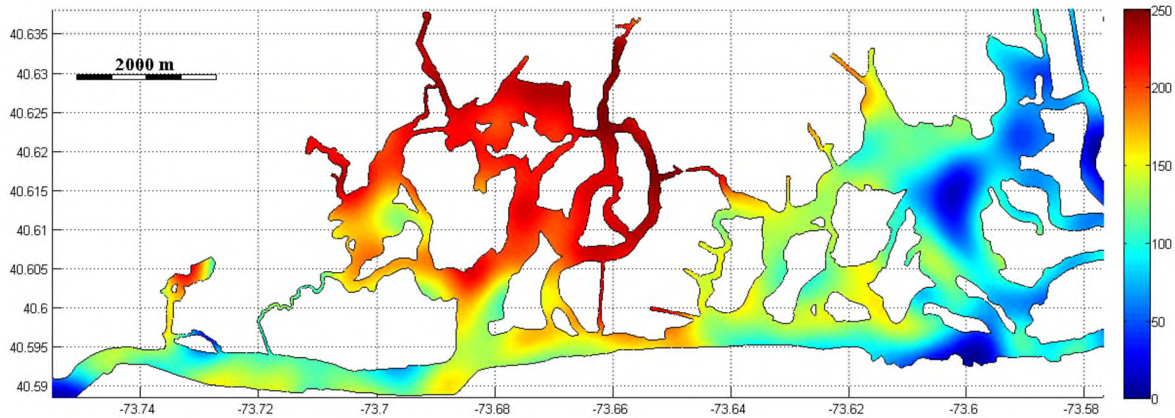


Figure 14. Residence time (hours) distribution in Hempstead Bay.

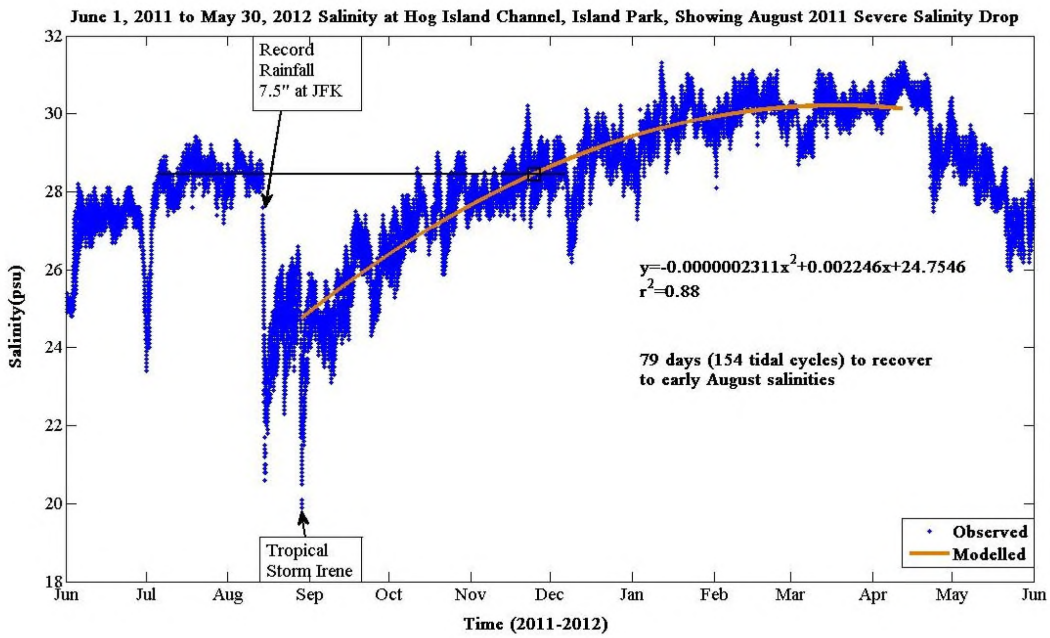


Figure 15. June 1, 2011 to May 30, 2012 salinity at Hog Island Channel, Island Park, showing August 2011 severe salinity drop. Data from USGS.

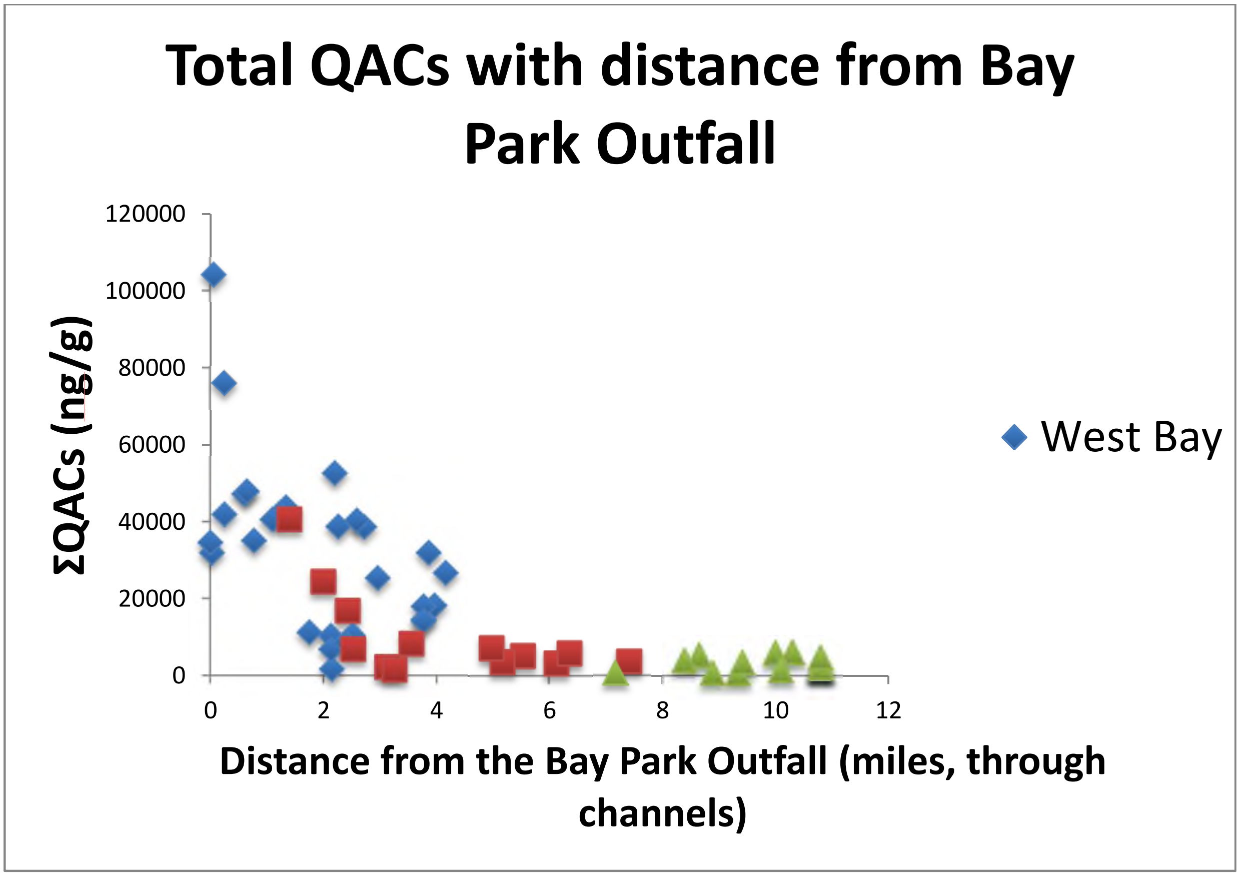


Figure 16. Total QACs in sediments with distance from Bay Park Outfall. From B. Brownawell.

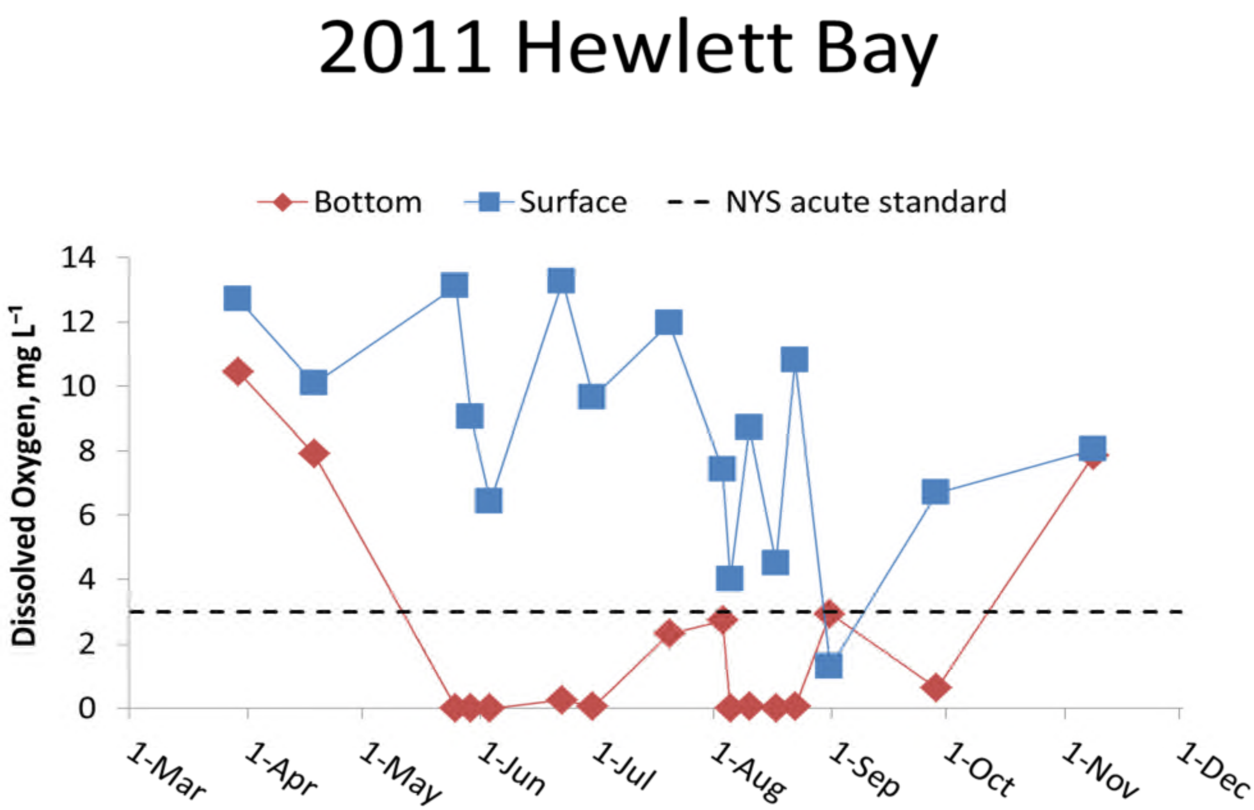


Figure 17. Dissolved oxygen concentrations (mg O_2/L) as a function of time in Hewlett Bay. From C. Gobler.

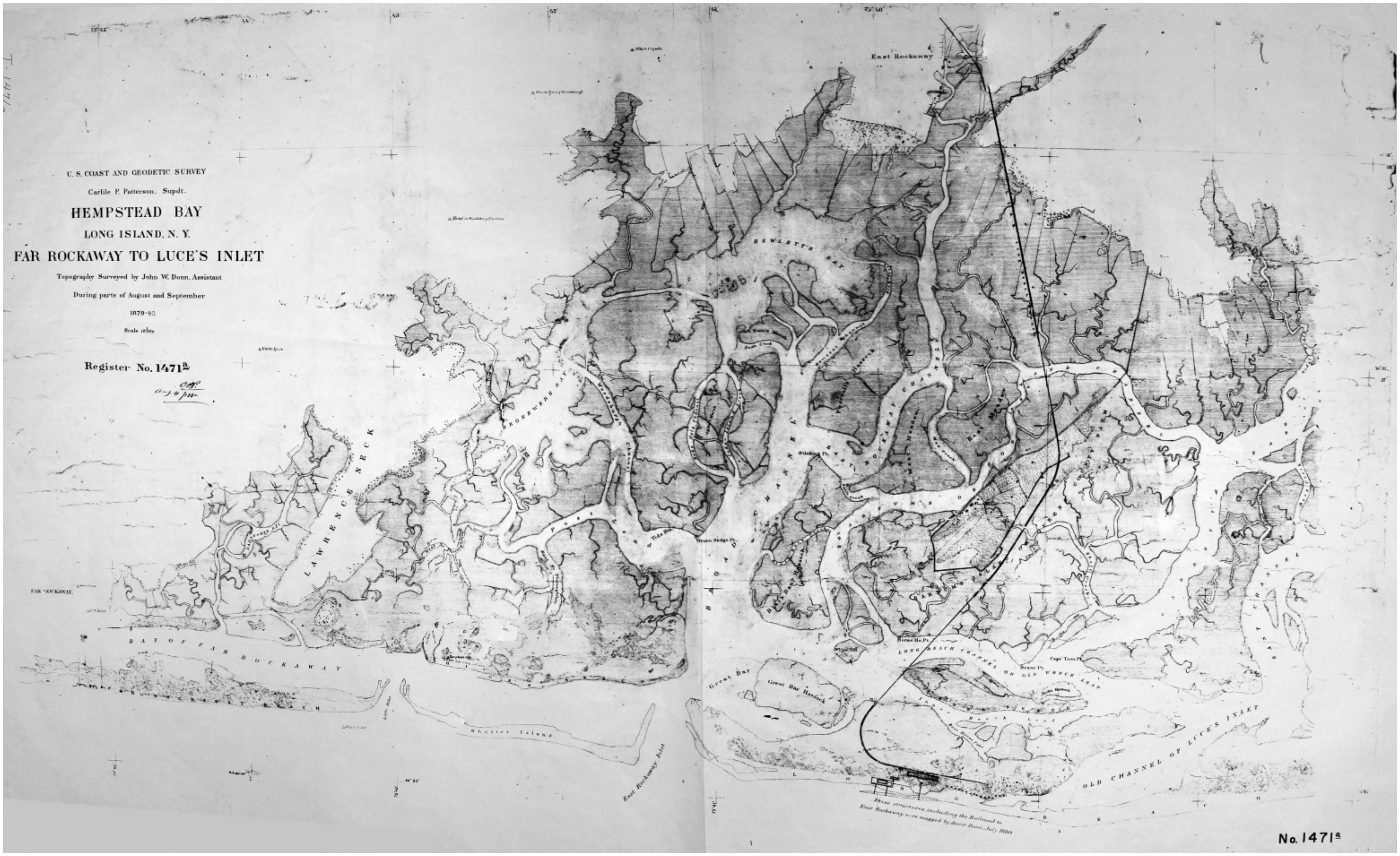


Figure 18. U.S. Coast and Geodetic Survey 1879-1880 topographic map of Hempstead Bay showing shallows of Hewlett Bay.

Comparison between *Ulva* growth and surface nitrate concentrations

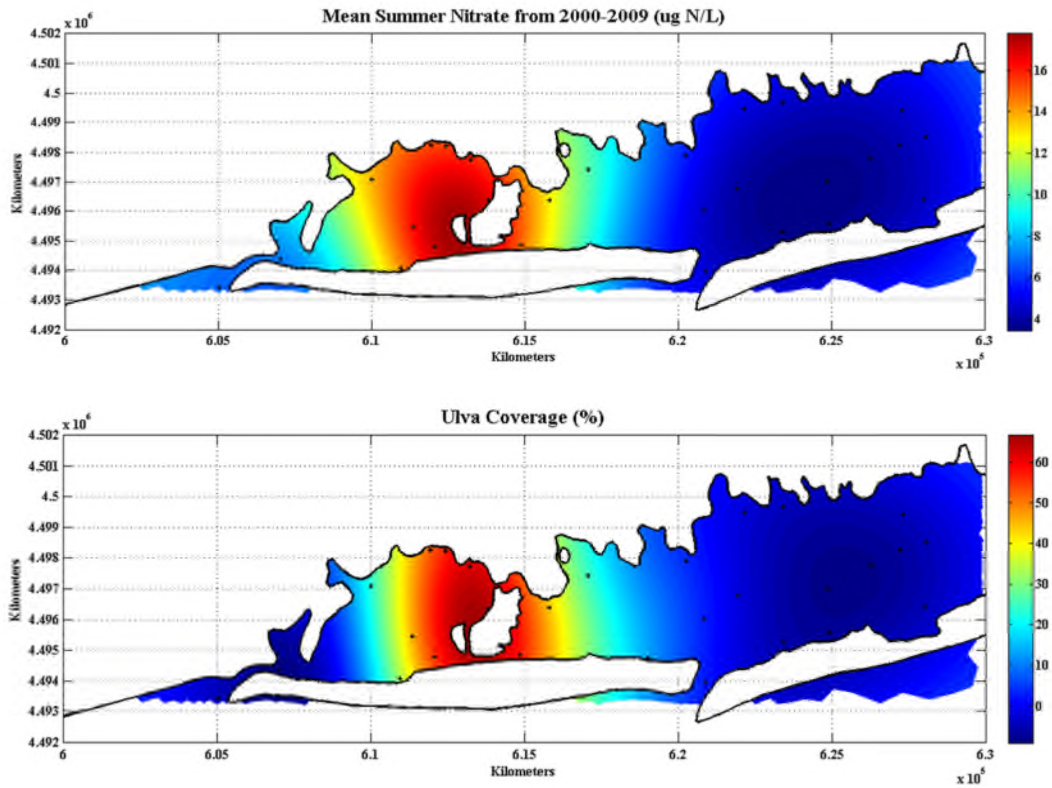


Figure 19. Comparison between *Ulva* growth and surface nitrate concentrations.

APPENDIX E: QUANTITATIVE DESCRIPTION OF THE SPATIAL PATTERNS IN RESIDENCE TIMES WITHIN THE HEMPSTEAD BAYS REGION

Quantitative Description of the Spatial Patterns in Residence Times within the Hempstead Bays Region

A Report for Battelle Memorial Institute and the New York State Department of State

Robert Wilson and Dongming Yang
School of Marine and Atmospheric Sciences
Stony Brook University
Stony Brook, NY 11794-5000

April 2013

1. Introduction

This represents the final interpretative report on contaminant distributions, water circulation, residence time patterns and contaminant age distributions. The report contains the following sections, each with its own sub-sections:

2. Grid Refinement and Incorporation of New Bathymetry
3. Model Forcing and Verification
  3.1 Model forcing
  3.2 Model verification
4. Tides and Currents
  4.1 M_2 amplitude and phase
  4.2 Tidal currents
  4.3 Tidally averaged currents
5. Residence Time and Age
  5.1 Residence time
  5.2 Age
6. References

2. Grid Refinement and Incorporation of New Bathymetry

The original Great South Bay (GSB) model is described in detail on the project web site http://po.msrc.sunysb.edu/GSB/). Under this new project we refined the model grid substantially from that pictured on the project web site with two objectives: improving grid quality improving resolution of the conveyance channels. The new grid for the entire south shore lagoon system contains 72,666 nodes and 131,434 elements; for comparison the old grid contains 59,531 nodes and 114,637 elements. Figures 1, 2 and 3 show three relevant segments of the grid to show gridded area and surface water input locations. Grid resolution is difficult to discern because of the small grid elements.

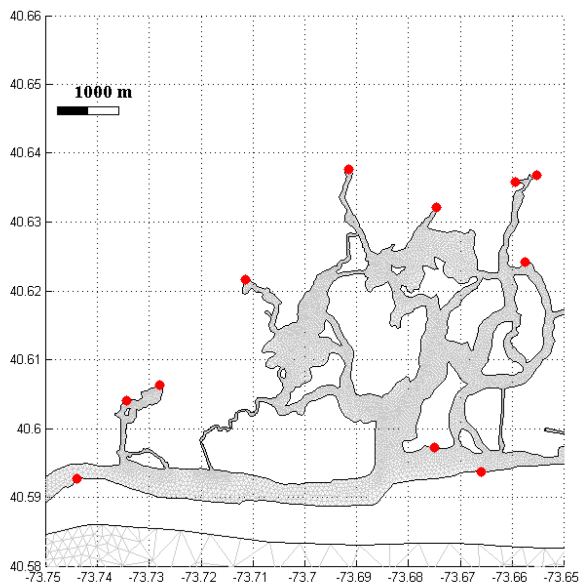


Figure 1. Refined grid segment for West Bay with surface water input locations.

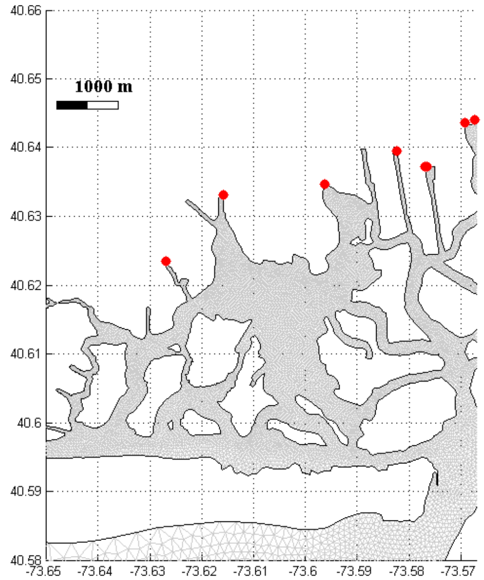


Figure 2. Refined grid segment for Middle Bay with surface water input locations.

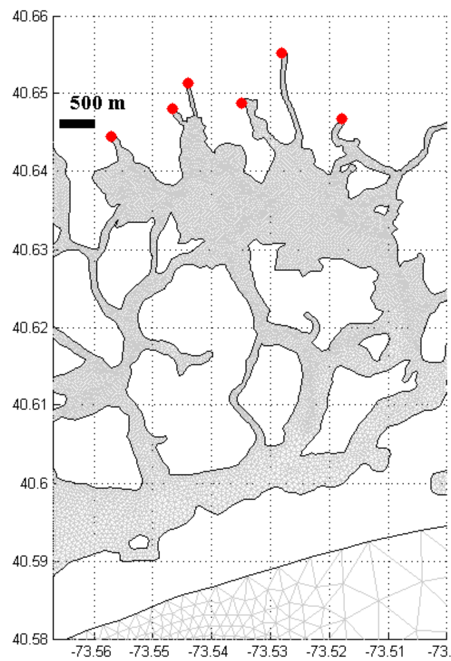


Figure 3. Refined FVCOM grid segment for East Bay including surface water input locations.

We also incorporated the new multi-beam bathymetry collected throughout Hempstead Bay under Task 2. Figure 4 shows this new bathymetry interpolated to the model grid; it emphasizes the importance of conveyance channels. This spatial resolution grid and the associated bathymetry require an external time step of 0.56 seconds with an internal time step which five times as long. Simulations are presently run on a Kraken supercomputer.

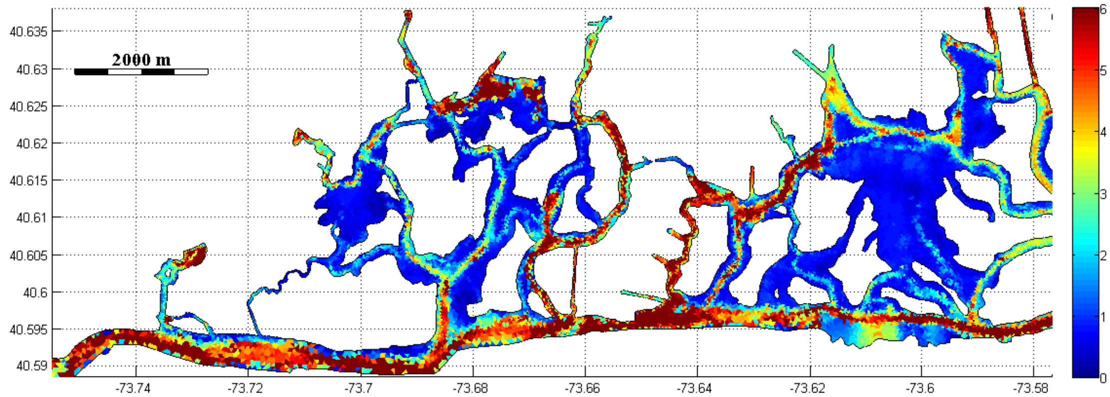


Figure 4. High resolution bathymetry (**m**) interpolated to refined model grid. Note that colorbar limits do not reflect the entire depth range in the domain.

3. Model Forcing and Verification

3.1 Forcing

The model was forced by specifying the amplitude and GMT phase for tidal constituents on the open ocean boundary nodes. Five major constituents were used: M_2, S_2, N_2, K_1, O_1. Open boundary constituents were extracted from the Oregon State global tidal model. The links for the model and download software are http://volkov.oce.orst.edu/tides and http://volkov.oce.orst.edu/tides/otis.html.

The model was forced by surface fresh water inflows distributed along the north shore of the domain and ground water as detail on the project web site http://po.msrc.sunysb.edu/GSB/. It was also forced by ***canonical*** summer wind and net surface heat flux. The canonical summer wind forcing was derived from wind observations at the GSB buoy (http://po.msrc.sunysb.edu/GSB/); it reflects the diurnal cycle in median wind speed and direction and so represents the summer sea breeze. The diurnal cycle in net surface heat flux was derived from analyses of observations by Duval (2008) of the components contributing to the net flux.

3.2 Verification

It is worth noting that this is a challenging model domain. It includes four tidal inlets whose effective cross-section and length controls tidal range in the interior. High velocity tidal flows within the inlets, especially Moriches Inlet, lead to numerical stability problems. There is a complex and dendritic network of channels and marshes which wet and dry (Figure 4).

In the modeling QAPP prepared for New England Interstate Water Pollution Control Commission, it was emphasized that water level is the most important diagnostic characterizing the depth mean circulation, and that in this shallow bay spatial variations in salinity can have only a secondary effect on the depth mean circulation. Comparisons of simulated salinity fields with observations would be made to ensure that they are reasonable, but because of large uncertainties in surface and groundwater inflow, no quantitative validation against salinity observations would be made. By reasonable, it was meant that simulations should reflect the large scale east to west and north to south salinity differences within the basin to within approximately 2 psu. In the modeling QAPP, the only quantitative verification required was against water level observations.

Figures 5 and 6 show tidally averaged surface and bottom salinity distributions. These reflect the distribution of fresh water inflow as well as the circulation patterns controlling intrusion of saline coastal waters. Comparisons with the historical Town of Hempstead salinity observations, which were analyzed under Task I and provide high spatial coverage, shows that these distributions are well with the 2 psu criterion.

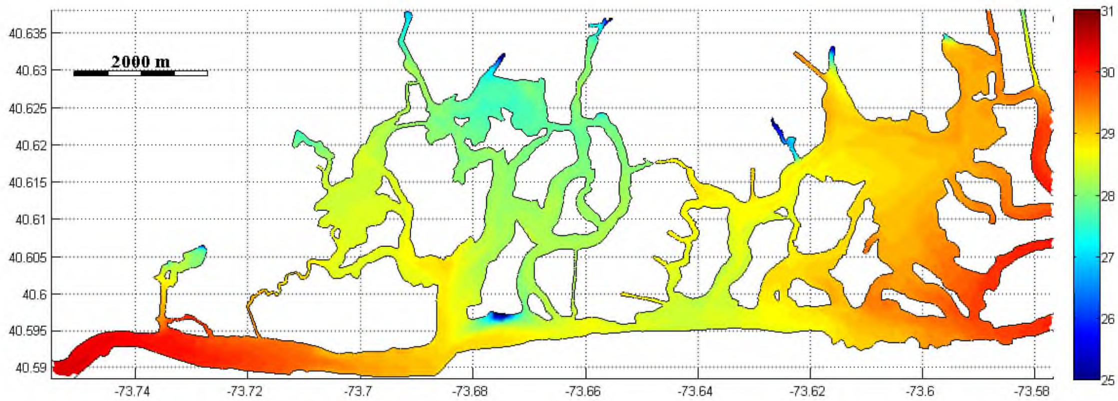


Figure 5. Tidally averaged surface salinity (psu) within Hempstead Bay.

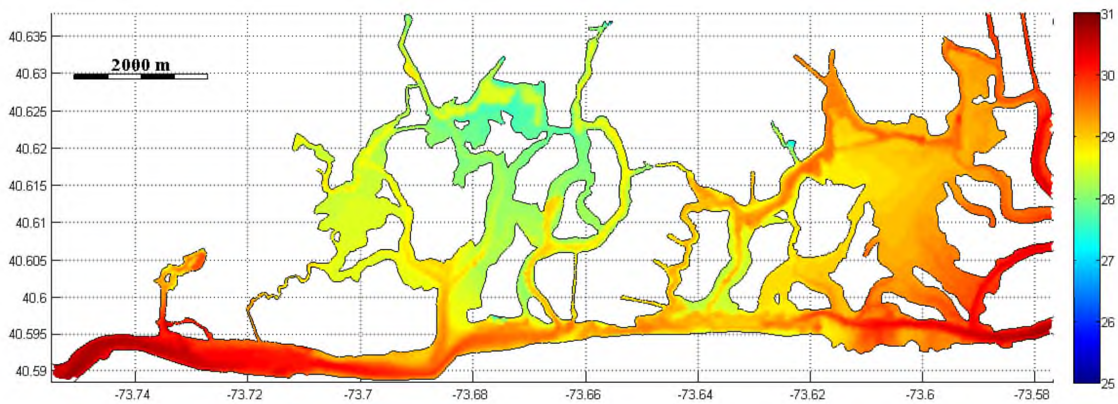


Figure 6. Tidally averaged bottom salinity (psu) within Hempstead Bay.

To verify water level throughout the domain, the community harmonic analysis program T-Tide (Pawlowicz, 2002) was applied to 30-day segments of model output and observational water level data from a total of seven interior stations. Results are reported in Table I. The precise positions of observational stations labeled as SeaBird Seacats are located on the project web site (http://po.msrc.sunysb.edu/GSB/). The station Forge River Entrance was a temporary station occupied by SoMAS during the period 03/21/2008-04/18/2008. The positions of the USGS stations can be found on the web page (http://waterdata.usgs.gov/ny/nwis/uv/?site_no=01311143&agency_cd=USGS&amp).

Considering the complexity of the morphology and bathymetry in this multi-inlet domain, and possible uncertainties in the amplitudes and phases in boundary constituents from the OSU model, results for the dominant semi-diurnal M_2 constituent are excellent. Note that model constituents are from an older run with elevated friction and will be updated with those from current run.

Table I. Comparison of tidal constituents from model with observations

USGS Point Lookout						
Model				USGS Observation		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.60	357.08		M2	0.59	0.19
S2	0.12	23.00		S2	0.15	42.21
N2	0.15	348.36		N2	0.16	350.78
K1	0.08	180.82		K1	0.07	185.05
O1	0.05	188.64		O1	0.03	200.84
Fire Island CG						
Model				SeaCat		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.27	13.29		M2	0.24	7.52
S2	0.05	35.61		S2	0.05	36.38
N2	0.06	3.15		N2	0.05	347.48
K1	0.05	200.86		K1	0.05	185.98
O1	0.03	213.14		O1	0.03	208.90
Forge River Entrance						
Model				Observation		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.26	41.48		M2	0.25	29.27
S2	0.04	77.52		S2	0.06	52.67
N2	0.05	41.14		N2	0.05	27.02
K1	0.04	220.49		K1	0.05	177.12
O1	0.04	230.38		O1	0.02	207.09
USGS Hog I. Channel						
Model				USGS Observation		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.56	16.47		M2	0.63	10.47
S2	0.10	49.50		S2	0.13	58.12
N2	0.12	12.22		N2	0.15	5.53
K1	0.08	195.36		K1	0.09	193.06
O1	0.05	202.34		O1	0.04	196.63
Barret Beach						
Model				SeaCat		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.14	111.22		M2	0.16	96.78
S2	0.02	143.98		S2	0.02	125.75
N2	0.02	96.08		N2	0.03	82.18
K1	0.02	263.55		K1	0.04	255.60
O1	0.02	275.97		O1	0.02	256.26
Tanner Park						
Model				SeaCat		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.21	69.88		M2	0.20	60.12
S2	0.04	99.38		S2	0.03	95.07
N2	0.04	66.48		N2	0.03	43.25
K1	0.04	241.94		K1	0.05	224.13
O1	0.03	252.44		O1	0.03	242.26
Bellport						
Model				SeaCat		
Constituent	Amplitude(M)	Phase(degree)		Constituent	Amplitude(M)	Phase(degree)
M2	0.15	115.29		M2	0.17	99.84
S2	0.02	148.63		S2	0.03	129.10
N2	0.03	99.89		N2	0.04	67.41
K1	0.03	292.39		K1	0.04	226.24
O1	0.02	278.42		O1	0.02	276.74

4. Tides and Currents

In this section we present basic features of tidal elevation and both tidal and residual currents in Hempstead Bay as they influence the movement of dissolved and suspended materials.

4.1 M_2 amplitude and phase

Figure 7 shows that for the M_2 constituent, elevation in Hewlett Bay lags that in the center of Reynolds Channel by approximately 15^o or approximately 30 minutes. The central part of Reynolds Channel near the Bay Park outfall lags Rockaway Inlet by approximately 15^o or approximately 30 minutes. In Reynolds Channel there is a maximum in phase lag to the east of the outfall: lagging Rockaway Inlet by approximately 18^o and Jones inlet by approximately 5^o. Overall, the region exhibits modest phase lags characteristic of a standing wave and so we expect currents to be in quadrature with the elevation. M_2 amplitude (Figure 8) shows maximum amplitude in Rockaway inlet, and west to east attenuation in Reynolds Channel and Middle bay. It also shows a reduction in amplitude within Hewlett Bay. It is important to note that the localized regions of low amplitude are not real; they indicate that the analysis program T-Tide did not function properly in areas of wetting and drying.

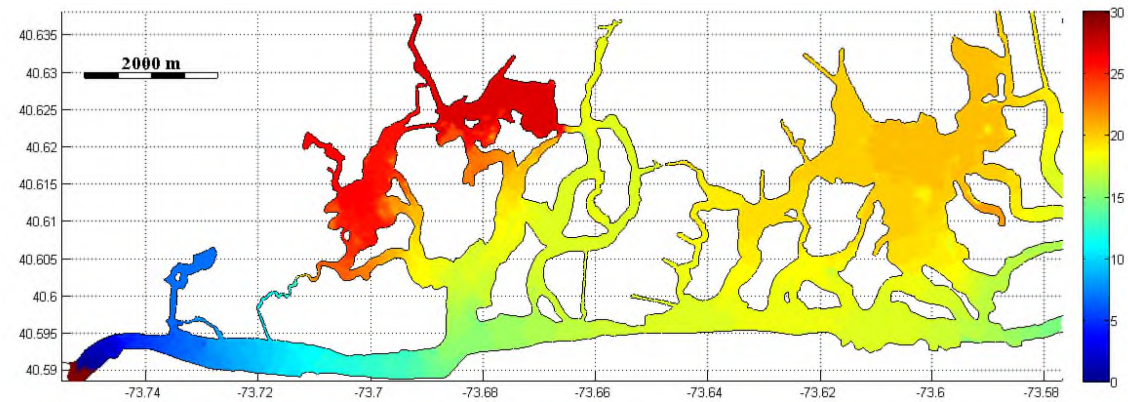


Figure 7. M2 elevation phase (**degrees GMT**) distribution within Hempstead Bay.

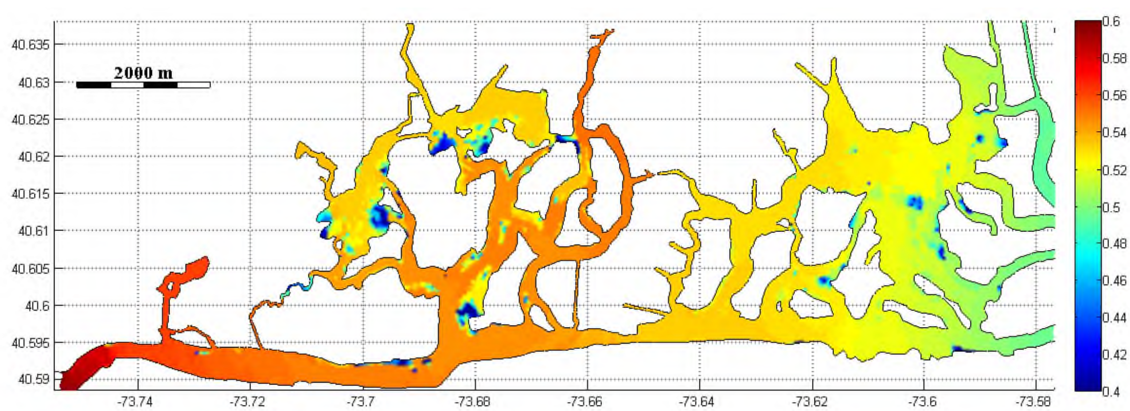


Figure 8. M2 elevation amplitude (**m**) within Hempstead Bay. Localized regions of low amplitude are anomalous and result from harmonic analysis in areas of wetting and drying.

4.2 Tidal currents

Tidal currents are of fundamental importance to dispersion processes, but they are difficult to present clearly in this complex basin. The conclusion reached above was that tides are essentiality standing wave in nature, and currents are in approximate quadrature with elevation. Maximum currents then occur approximately at the time of mean tide level: maximum flood currents occur at the time of mean tide level before high water, and maximum ebb currents at the time of mean tide level before low water. Figure 9 shows surface current speed during maximum flood. Very high velocity inflows in excess of 50 cm/s occur through Rockaway Inlet and in western Reynolds Channel. There are also high velocity flows northward through the channel network leading towards Hewlett Bay. At this phase of the tide there are weaker inflows through Jones Inlet and in eastern Reynolds Channel, and a confluence within Reynolds Channel at approximately -73.63^{o}. There are northward directed flows out of Reynolds Channel through the channel network leading to Middle Bay. During approximately maximum ebb (Figure 10), there are strong outflows through Rockaway and Jones Inlets and a divergence in Reynolds Channel at approximately -73.65^{o}. Hewlett Bay appears to drain into Reynolds Channels and out Rockaway Inlet. Middle Bay, on the other hand, drains into Reynolds Channel and out Jones Inlet.

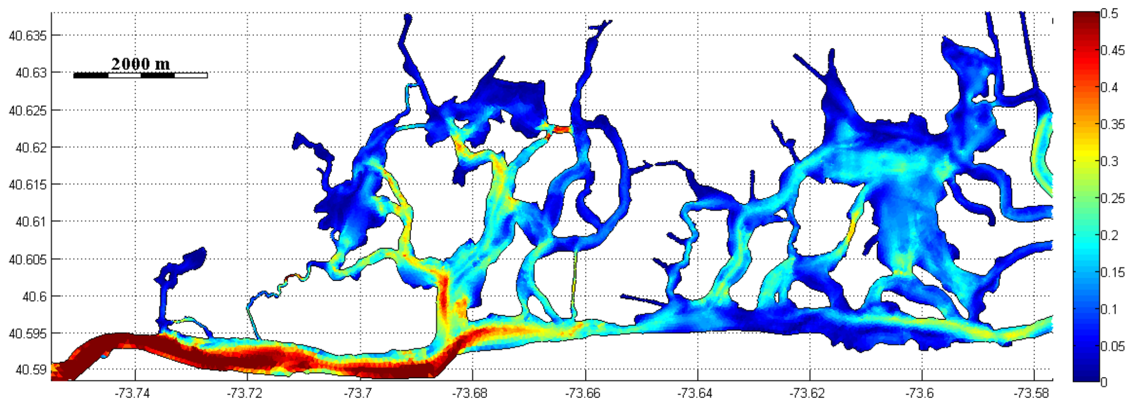


Figure 9. Surface current speed (**m/s**) during maximum flood.

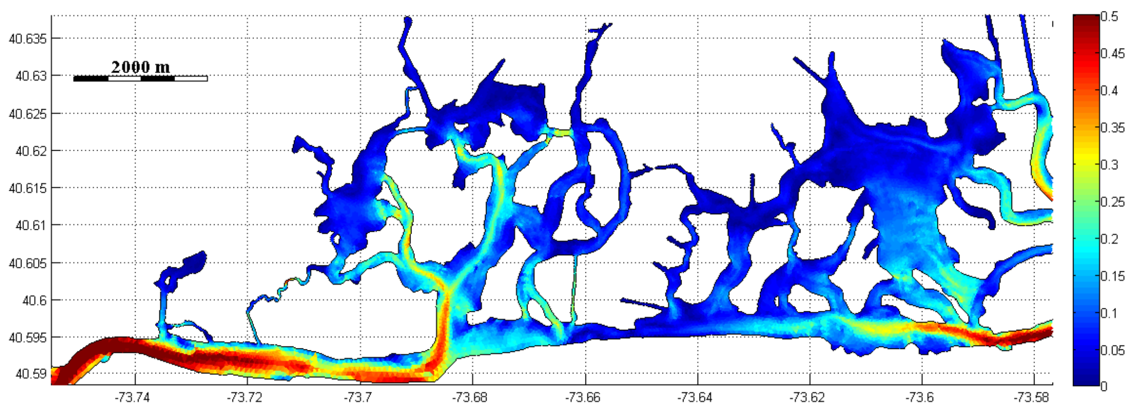


Figure 10. Surface current speed (**m/s**) during maximum ebb.

4.3 Tidally averaged currents

Tidally averaged currents (Figures 11 and 12) are spatially complex generally less than 10 cm/s with maximum amplitudes in Reynolds Channel. Because of the complex morphology the east and north components are presented; no attempt is made to present the along channel component.

In western Reynolds Channel, there is considerable lateral structure in the residual current. In eastern Reynolds Channel there is evidence of persistent eastward flow toward Jones Inlet. This Eulerian description of residual currents *does not* provide a good description of the residual movement of water particles. As we will see in the next section, a Lagrangian description obtained from particle tracking provides a far superior description of the patterns of movement.

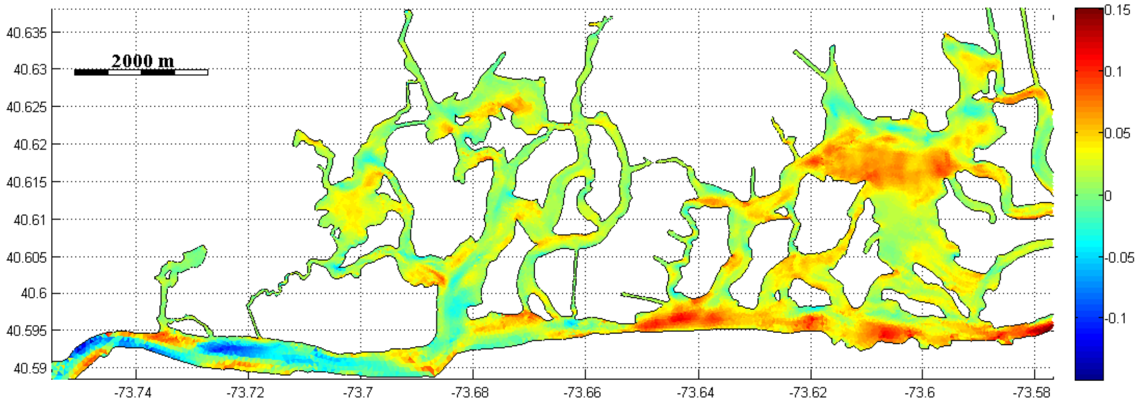


Figure 11. East component of residual current speed (**m/s**).

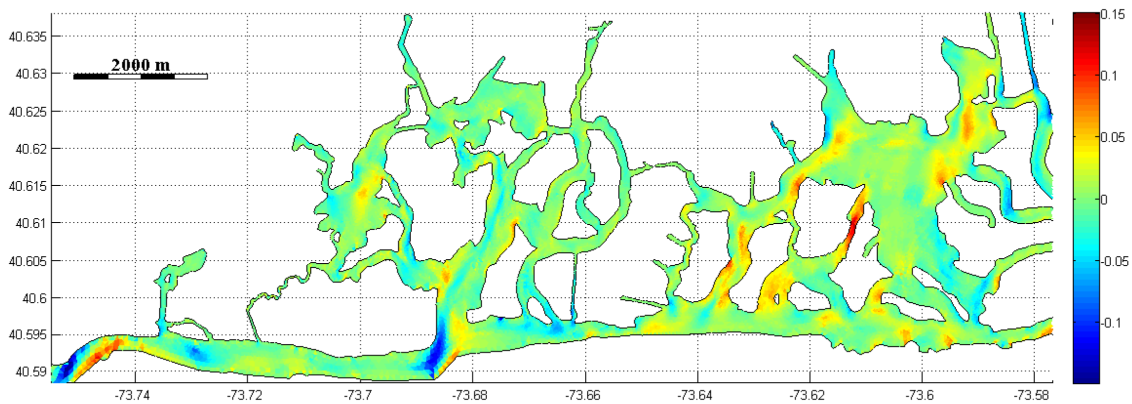


Figure 12. North component of residual current speed (**m/s**).

4.4 Lagrangian particle tracking description of movement and dispersion

Lagrangian particle tracking provides a far superior description of the patterns of movement and dispersion due to both tidal and residual currents. It is important to understand that the Lagrangian particle tracking algorithm in this model is fully 3D. Besides the 3D advective motion determined by the model's current field, vertical mixing is represented by a random displacement model (Visser, 1997) in which random vertical displacements are added, which are dependent on the model's vertical turbulent diffusivity.

Figure 13 shows results for a continuous release from the Bay Park outfall in Reynolds Channel. 10^3 particles were released into the element containing the discharge point every 1/2 hour. Because of the large number of particles, results are presented as dilutions within model elements as log to the base 10 of the percent. For example, for a dilution of 3 percent, we have log(3)=0.4771. Results show that particles from the Bay Park outfall are broadly dispersed. During flood tide particles move northward through the network of channels and into both

Hewlett Bay and Brosewere Bay. There is eventual movement of particles to the northeast through the channel network and into Middle Bay. Within Reynolds Channel, there is east-west dispersion of particles. To the west of the outfall this movement is limited to a tidal excursion. To the east of the outfall, there is clear evidence of residual transport towards the east and out of Jones Inlet.

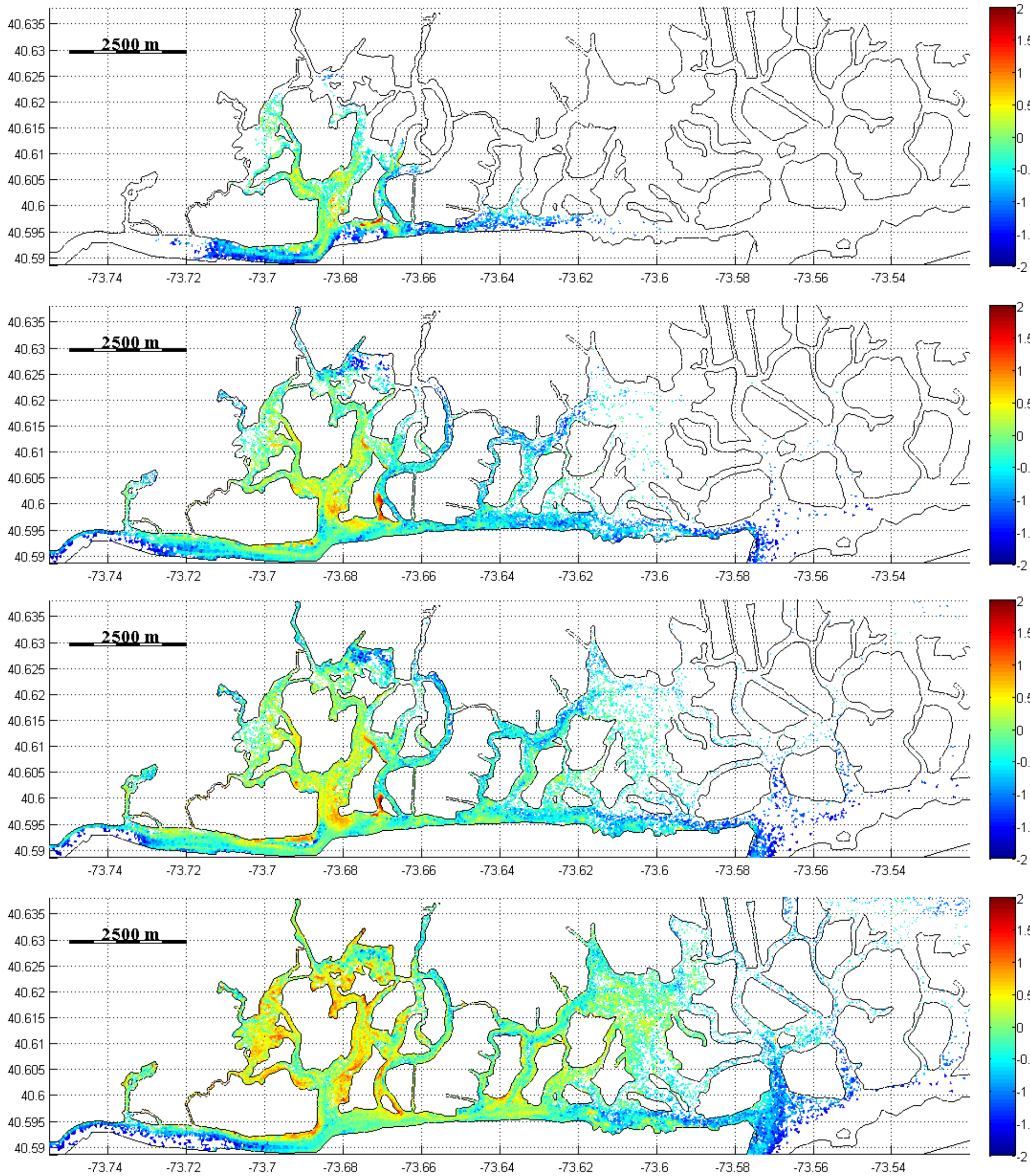


Figure 13. Lagrangian particle tracking results at 12, 50, 100 and 240 hours for a continuous release at the Bay Park outfall presented as **log to the base 10 of the percent dilution** at different times following initiation of the release.

For comparison, Figure 14 shows results for a continuous release from the Bay Park Plant site in Mill River. Note that this figure is for a more limited domain than that in Figure 13. There was an emergency release of sewage from this site following Hurricane Sandy. Results show that the dispersion of particles released from this north shore embayment site is limited when compared to dispersion associated with a Reynolds Channel release, although concentrations in Mill River can get very high. There is dispersion into Hewlett Bay. This is consistent with characteristics of dispersion for a release in Macy Channel. Figures 13 and 14 show that dilutions in East Bay are extremely low.

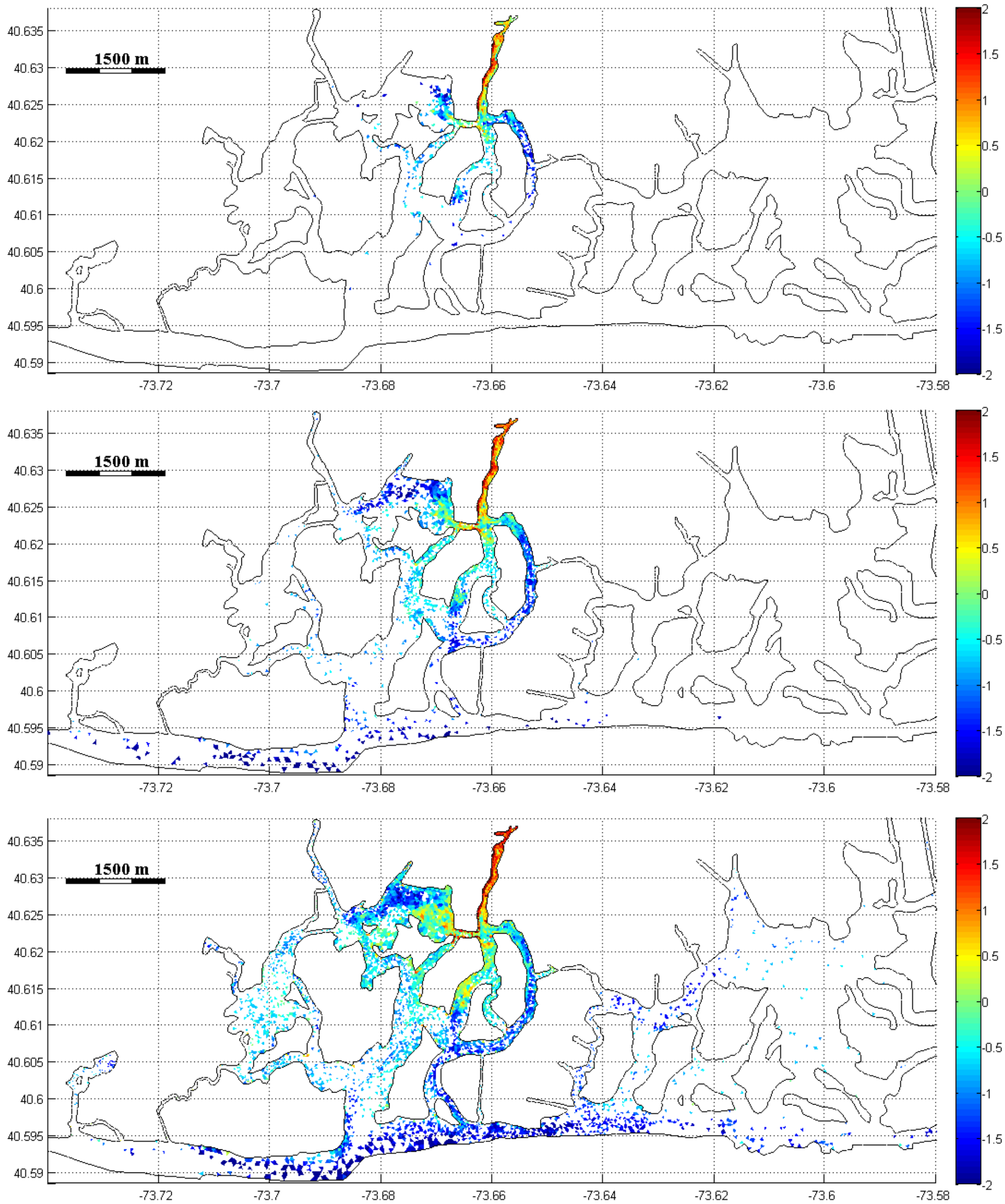


Figure 14. Lagrangian particle tracking results at 50, 100 and 240 hours for a continuous release at the Bay Park WWTP in Mill River presented as **log to the base 10 of the percent dilution** at different times following initiation of the release.

5. Residence time and age

5.1 Residence time

Monsen et al. (2002) provide a clear and very readable summary of the concepts of residence time and age for a waterbody. They emphasize that aquatic scientists often estimate retention time and compare it to time scales of inputs or biogeochemical processes to calculate mass balances or understand dynamics of populations and chemical properties. Boynton et al. (1995) argue that residence time is such an important attribute that it should be the basis for comparative analyses of ecosystem-scale nutrient budgets.

Residence time is the time it takes for a water parcel to leave a waterbody compartment through the compartment boundary. Restated: residence time is how long a parcel, starting from a specified location within a waterbody, will remain in the waterbody before exiting. Residence time is the complement to age: age is the time required for a water parcel to travel from a boundary or a specific location such as an outfall to a specified location within a waterbody. Age and residence time depend on the specification of the boundary, the measurement point of interest within the domain, and in tidal systems, the time of release.

Residence time was estimated using Lagrangian simulations based on the FVCOM output following the method described by Aikman and Lanerolle (2005). The NOAA Office of Coast Survey web site http://www.nauticalcharts.noaa.gov/csdl/residencetime.html discusses various methods and rationalizes that Lagrangian paticle tracking is the most effective method for defining residence time. We emphasize again that in our particle tracking we are using a fully 3D velocity field which includes random vertical displacements to represent vertical mixing; particles are moving both horizontally and vertically within the water colum.

For our residence time we defined the lagoon compartment extending from -73.76^o to -73.57^o. Also following Aikman and Lanerolle (2005), for our simulations we initially distributed particles uniformally throughout the compartment and defined the residence time for that particle as the *first passage time* or the first time a water parcel leaves a compartment. Two important points:

- Particles are moving not only horizontally but vertically and our simulations showed *relatively little difference between residence time patterns for upper and lower parts of the water column throughout most of the domain.* There was some discernable increase in the residence time for particles released in the bottom waters in Hewlett Bay; within Reynolds Channel the residence time for particles released in surface and bottom waters is comparable because of increased vertical tidal mixing in the Channel.
- As emphasized by Olivera and Bapista (1997) and Lipphardt et al. (2006), *there is uncertainity in the estimate for residence time in a particular region.* This can reflect the stochastic vertical motion and the dependence on tidal phase. For multiple simulations there is actually a distribution for the residence time estimate a particular point. We therefore present both the mean and the standard deviation for the residence time in order to provide a measure of this uncertainity.

The residence time pattern for particles released in the surface layer (Figure 15) emphasizes several important points. Residence in Hewlett Bay, Macy Channel and Mill River is relatively long. This simulation was only 250 hours and so there can certainly be areas where the residence time is greater than 250 hours. It is surprisingly long in the central part of Reynolds Channel, presumably because of the tidal and residual current structure discussed above. It is also relatively long in the western part of the Channel. Residence time decreases towards the east, it is low in the eastern part of Middle Bay.

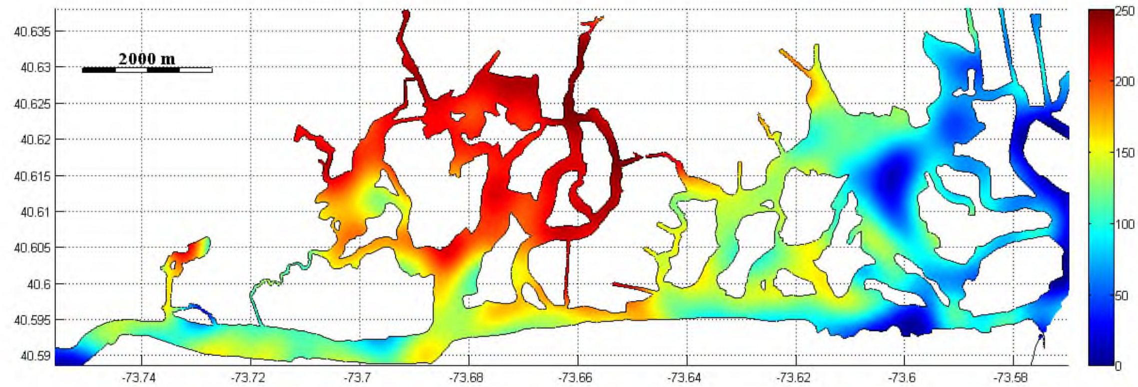


Figure 15. Mean surface layer residence time (**hours**) from multiple Lagrangian particle Tracking experiments for the domain defined in the text.

Figure 16 shows the pattern of the standard deviation in surface layer residence time and thereby an estimate of the uncertainty. Standard deviation decrease towards the north. It is relatively high (approximately 80 hours) presumably reflecting the influence of strong tidal currents. It is also high in the channel network connected to Reynolds Channel. It is low in Hewlett Bay.

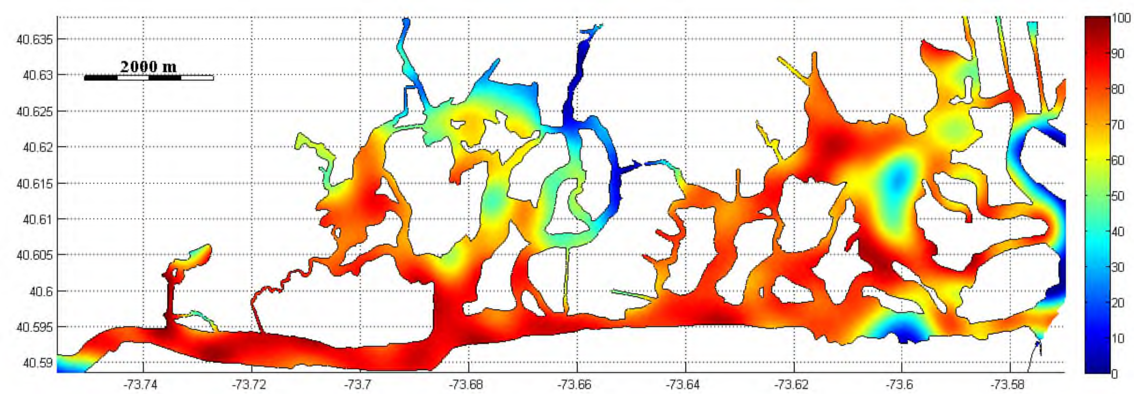


Figure 16. Standard deviation of surface layer residence time (**hours**) from multiple Lagrangian particle tracking experiments for the domain defined in the text.

Figures 17 and 18 provide a comparison with mean bottom layer residence time and the standard deviation in bottom layer residence time. Surface and bottom layer distributions are similar, but bottom layer residence time does show some discernible increase in both Reynolds Channel and Hewlett Bay.

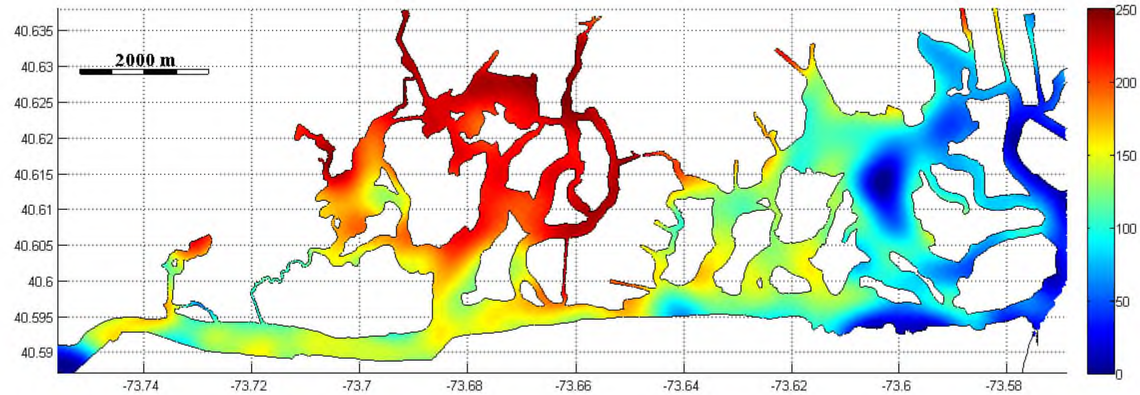


Figure 17. Mean bottom layer residence time (**hours**) from multiple Lagrangian particle Tracking experiments for the domain defined in the text.

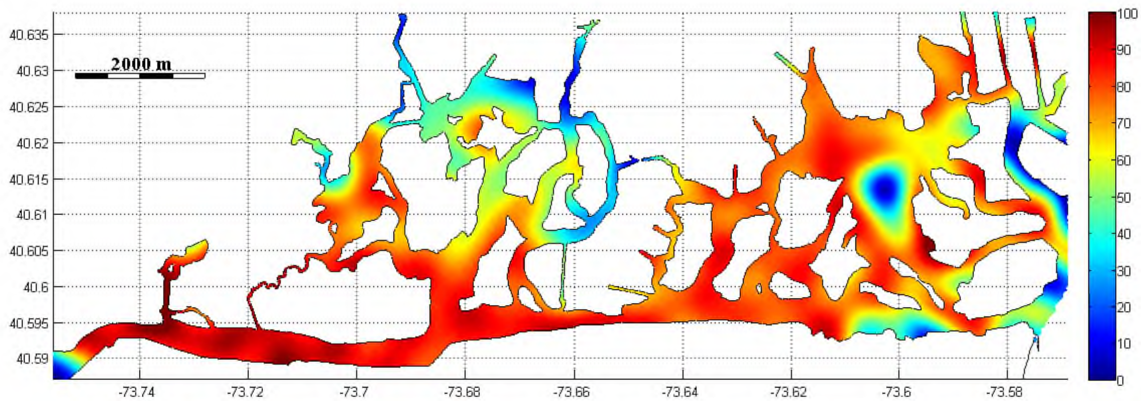


Figure 18. Standard deviation of bottom layer residence time (**hours**) from multiple Lagrangian particle tracking experiments for the domain defined in the text.

5.2 Age

The distribution of water particle age was estimated for water particles discharged from the Bay Park outfall in Reynolds Channel which is the major discharge into the Hempstead Bay network. We did this by using the 3D Lagrangian particle tracking described above for a continuous release of particles at the Bay Park site. The age distribution (Figure 19) is basically consistent with increasing age with distance from the source. It emphasizes that particles entering Hewlett Bay tend to remain. In Reynolds Channel, age increases to the east of the outfall. To the west of the outfall site there is evidence for 'new' ocean water entering through East Rockaway Inlet; there are a few elements containing particles with long age and this likely represents water which was re-circulated from the ocean. The dispersion pattern shown is consistent with the dilution patterns in Figure 13.

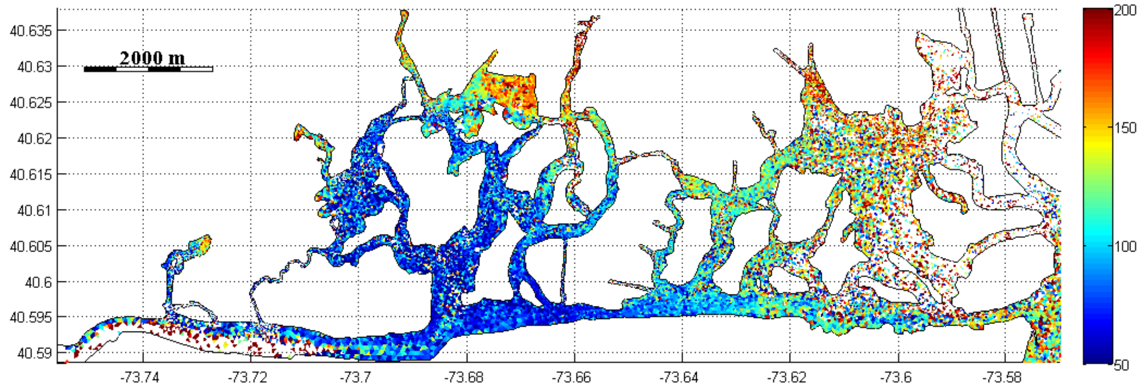


Figure 19. Distribution of water particle age (**hours**) from a Lagrangian particle tracking experiment for a continuous release at the Bay Park outfall site.

The distribution of water particle age for a continuous discharge at the Bay Park WWTP site is shown in Figure 20. The age distribution is again consistent with increasing age with distance from the source and it emphasizes that particles entering Hewlett Bay tend to remain. The very limited dispersion shown is consistent with dilution patterns in Figure 14. Because of the limited dispersion from confined north shore sites, age distributions from other discharge sites in this area are considered unenlightening and so are not presented. Also, consistent with dilution patterns in Figures 13 and 14, age distributions in Figures 19 and 20 show that dispersion into East Bay is limited.

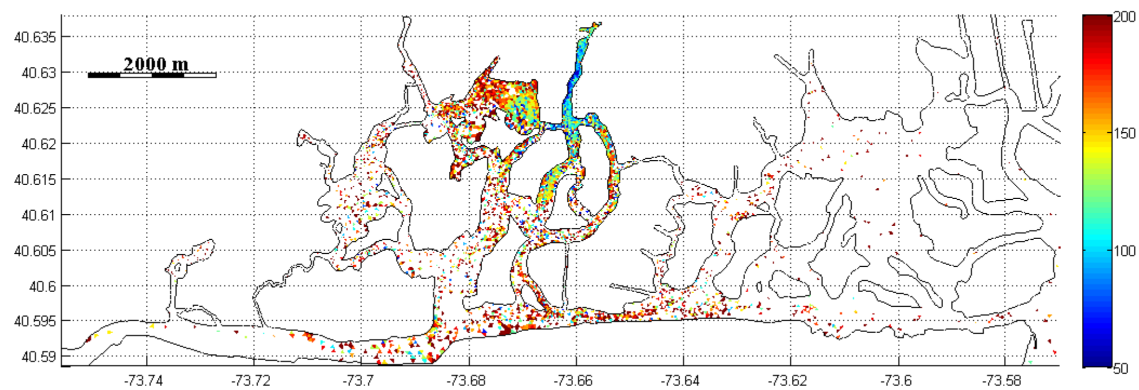


Figure 20. Distribution of water particle age (**hours**) from a Lagrangian particle tracking experiment for a continuous release at the Bay Park WWTP outfall in Mill River.

6. References

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APPENDIX F: HISTORICAL WATER QUALITY MONITORING DATA FOR WESTERN BAYS AS MEASURED BY TOWN OF HEMPSTEAD

Historical Water Quality Monitoring Data for Western Bays as Measured by Town of Hempstead

DATA SUMMARY

Cassia Bauer and Dr. Robert Wilson

Stony Brook University

School of Marine and Atmospheric Sciences

July 2011

Contours of mean summer and winter surface water values along with summer and winter PCA results for temperature, salinity, nitrite, nitrate, and ammonia are presented below. Due to gaps in the data record, I used temperature and salinity data from 1989-2010 and nitrite, nitrate, and ammonia data from 2000-2010.

For the mean summer values I averaged the data points from June, July and August for each year for 28 stations throughout the Bay (each station is represented as a diamond in the contours). The same was done for winter values using the months of December, January and February. I used optimal interpolation to make contours of the summer and winter averages.

The PCA results were determined by multiplying the mode 1 eigenvector by the standard deviation of the first principle component. For each parameter there are 4 contours of PCA results, two for summer and two for winter. For example, the 4 PCA contours for temperature are:

1) Summer temperature variability about the mean during a warm year

2) Summer temperature variability about the mean during a cold year

3) Winter temperature variability about the mean during a warm year

4) Winter temperature variability about the mean during a cold year

When looking at the contours, the highest variability occurs when the value is furthest from zero.

TEMPERATURE:

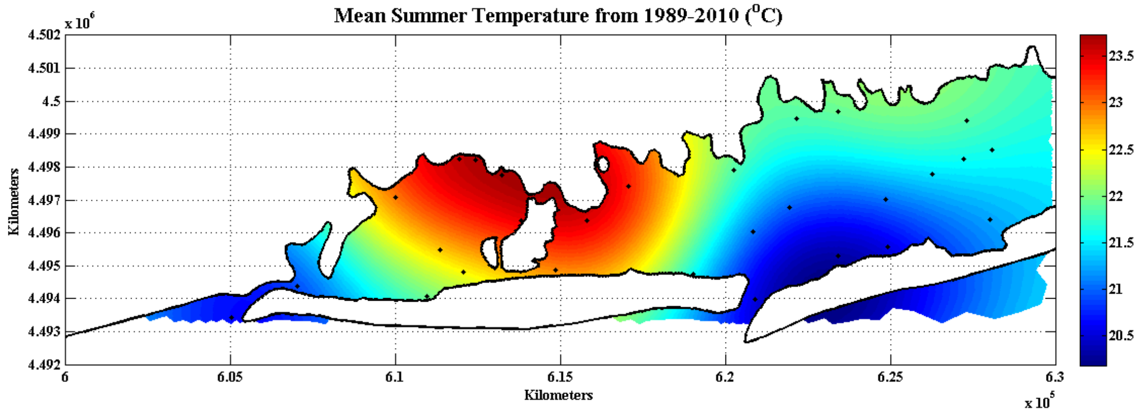

Figure 1: A contour of mean summer surface temperature from 1989-2010. The warmest summer temperatures occur on the northern part of Hempstead Bay, near the coast, and the coolest temperatures are seen near East Rockaway and Jones Inlet, which connect the bay to the Ocean.

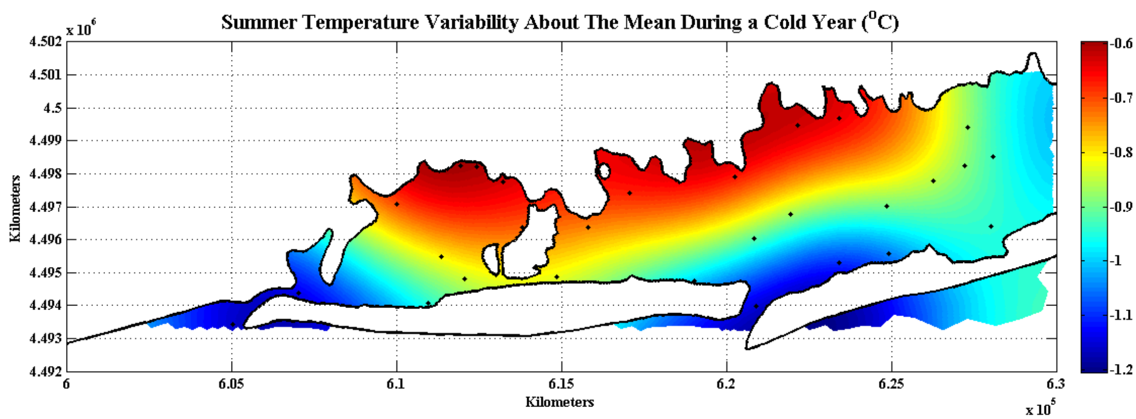

Figure 2: A contour of summer surface temperature anomalies during years when the temperatures were colder than the mean summer temperatures. The greatest negative anomalies are found near the inlets. The lowest negative anomalies are found in the northern part of the bay. This suggests that summer temperature anomalies are forced primarily by the ocean

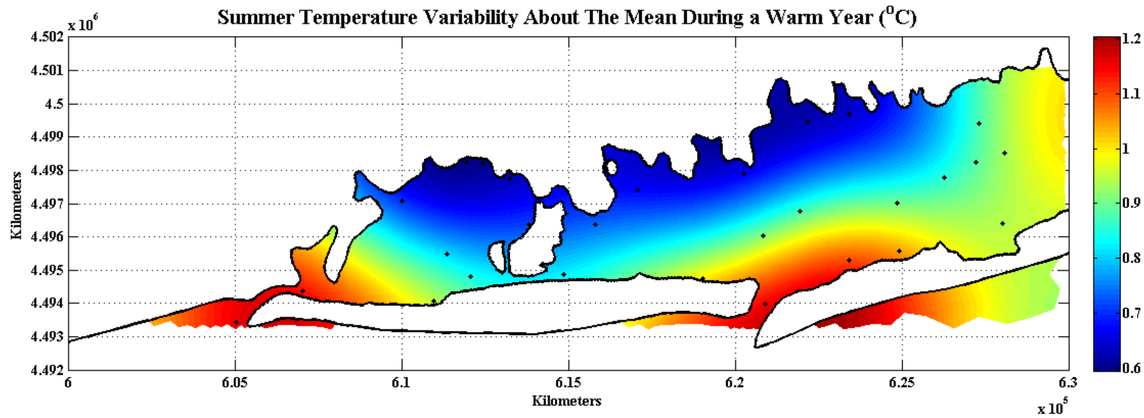

Figure 3: A contour of summer surface temperature anomalies during years when the temperatures were warmer than the mean summer temperatures. Similar to Figure 2, the greatest positive anomalies are found near the inlets and the lowest positive anomalies are found in the northern part of the bay. This again, suggests that summer temperature anomalies are forced primarily by the ocean

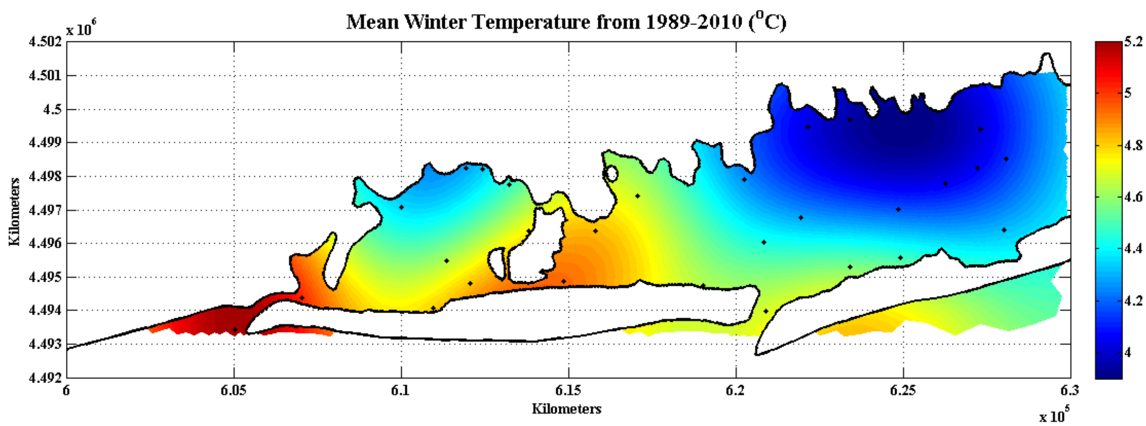

Figure 4: A contour of mean winter surface temperatures from 1989-2010. The warmest temperatures can be seen at the southern end of the bay, East Rockaway inlet being the warmest. The coolest temperatures are found of the northeast end of the bay.

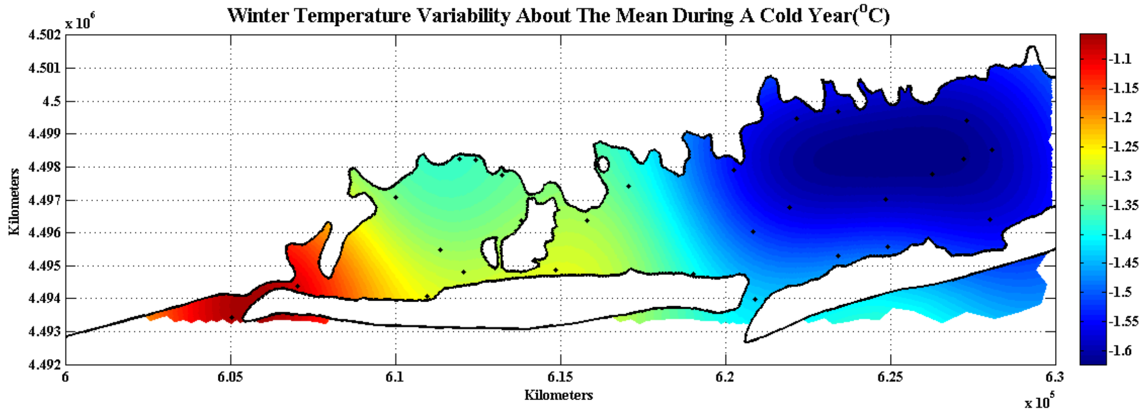

Figure 5: A contour of winter surface temperature anomalies when the temperatures are colder than the mean winter temperatures. The anomalies have a high east-west gradient, unlike summer anomalies which had a north-south gradient. The greatest negative anomalies are found on the eastern end of Hempstead Bay and the lowest are found to the west. Since the some of the lowest negative anomalies are found near the inlets, this suggests that winter temperature anomalies are forced primarily by heating/cooling inside the lagoon.

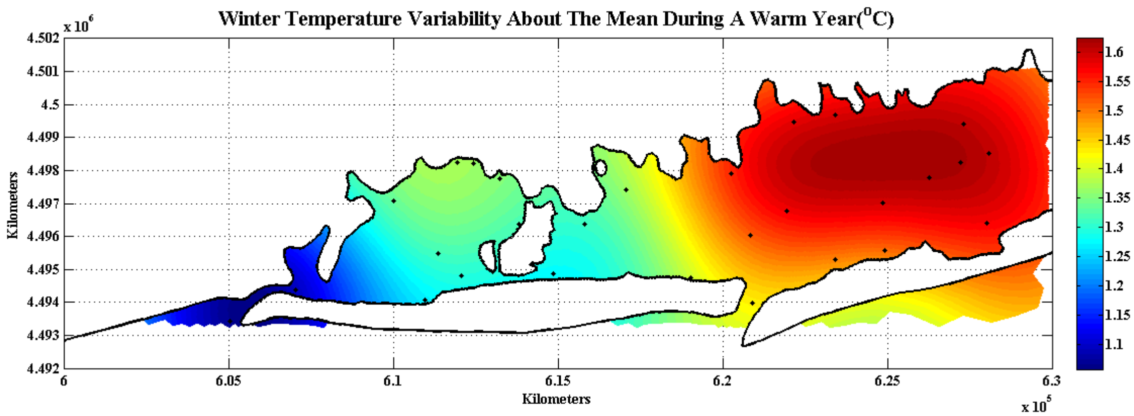

Figure 6: A contour of winter surface temperature anomalies when the temperatures are warmer than the mean winter temperatures. Again, we see an east-west gradient in temperature anomalies with the greatest positive anomalies being found to the east and the lowest found to the west. Similar to Figure 5, lower positive anomalies found near the inlets may suggest that winter temperature anomalies are forced primarily by heating/cooling inside the lagoon.

SALINITY:

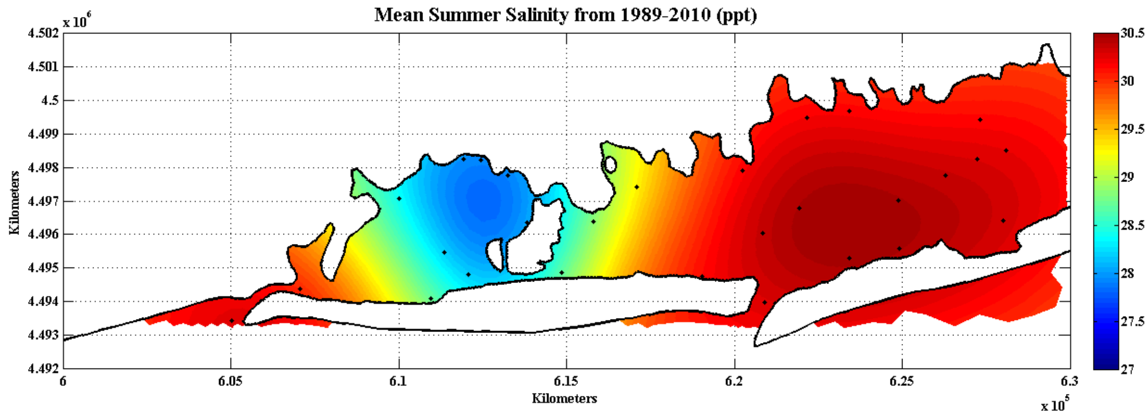

Figure 7: A contour of mean summer surface salinity from 1989-2010. The lowest salinity is seen at the northwest end of the bay. Higher salinity is observed near East Rockaway, Jones inlet and eastward of Jones inlet.

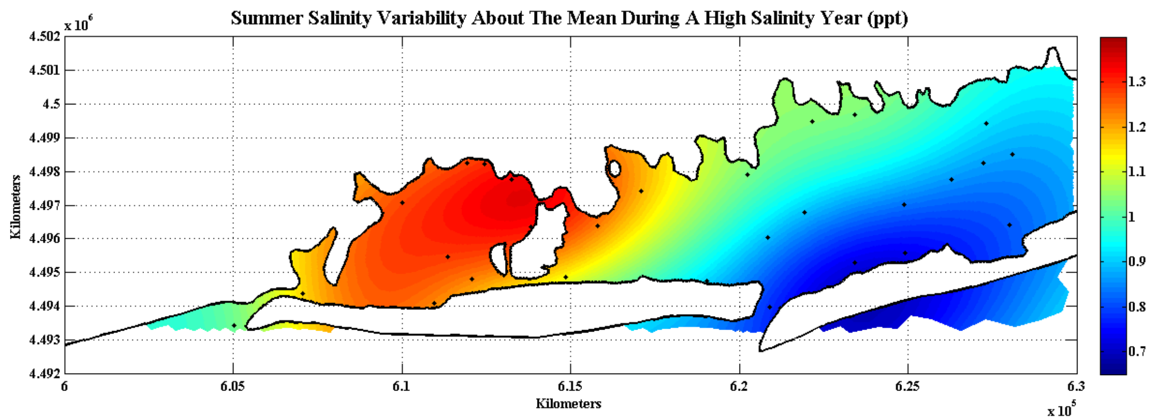

Figure 8: A contour of summer surface salinity anomalies when the salinity is higher than the mean summer salinity. The greatest positive anomalies are found closer to the coast, especially the northwest end of the bay and near the Bay Park outfall. The lowest positive anomalies are found near the ocean inlets. This suggests that summer salinity anomalies are forced primarily by the amount of freshwater entering the lagoon from tributaries and maybe more so from the Bay Park outfall.

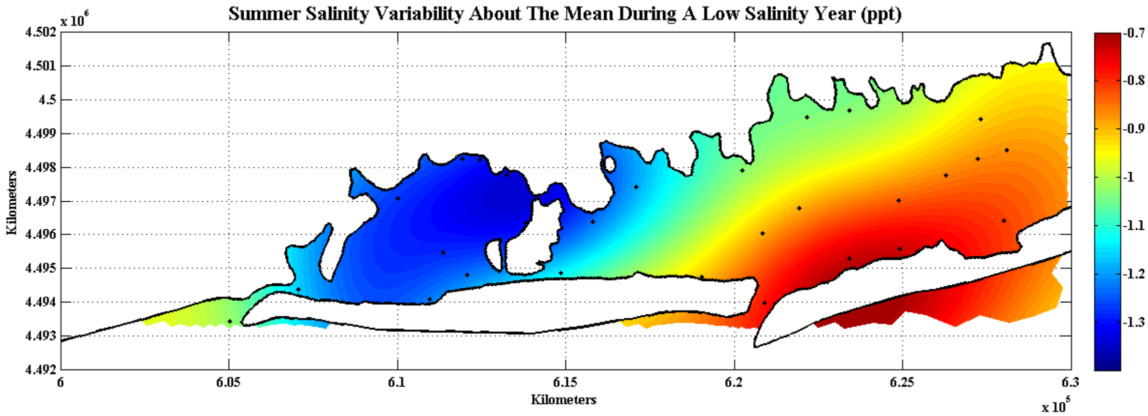

Figure 9: A contour of summer surface salinity anomalies when the salinity is lower than the mean summer salinity. Similar to Figure 8, the north end experiences the greatest negative anomalies and the ocean inlets experience the lowest negative anomalies in salinity.

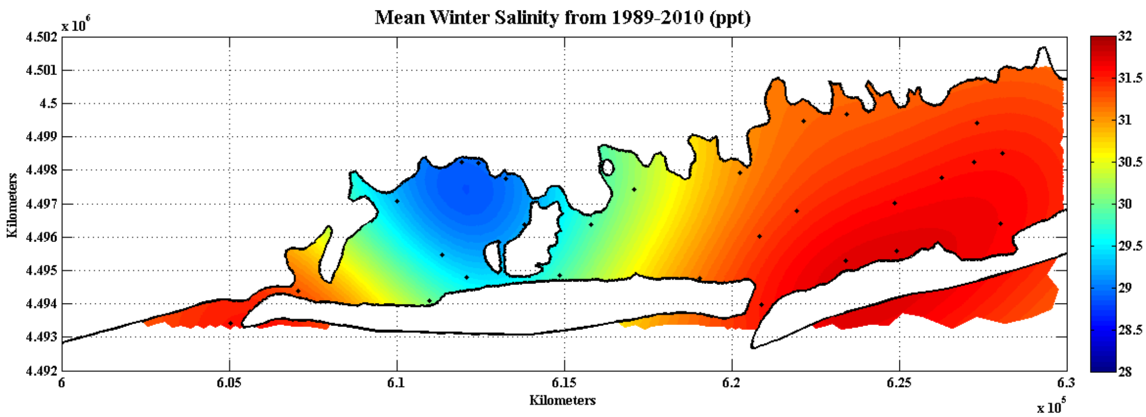

Figure 10: A contour of mean winter surface salinity from 1989-2010. Similar to mean summer salinity (Figure 7), the lowest salinity is seen to the northwest end of the bay and East Rockaway inlet, Jones Inlet, and east of Jones inlet experience the highest salinity.

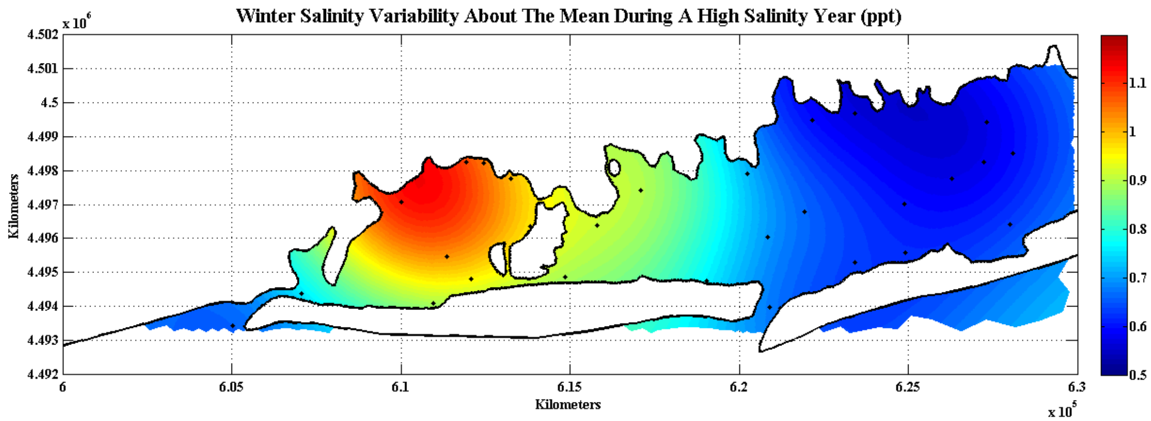

Figure 11: A contour of winter surface salinity anomalies when the salinity is higher than the winter mean salinity. The greatest positive anomalies are found on the northwest end of the bay while the two inlets and east of Jones Inlet experience the lowest positive anomalies. This suggests that winter salinity anomalies are forced primarily by the amount of freshwater entering the lagoon from the Bay Park outfall.

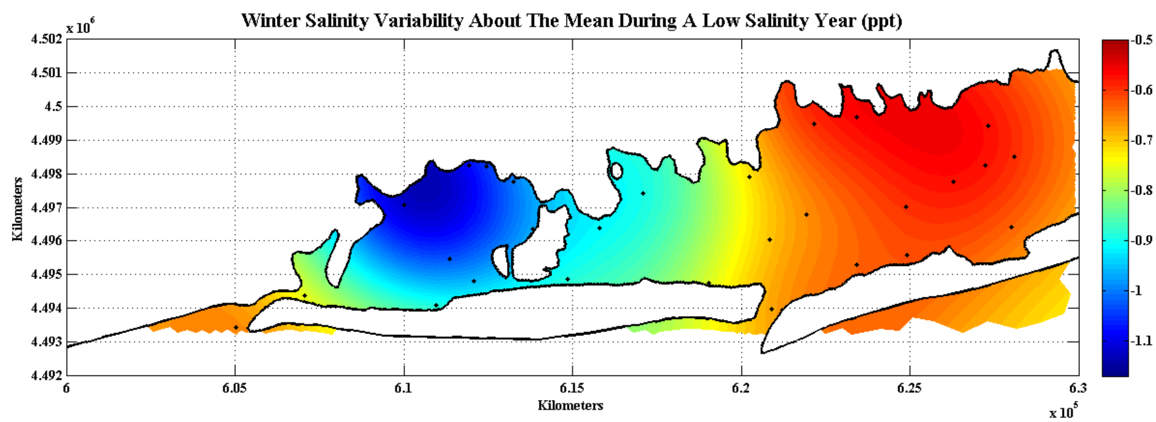

Figure 12: A contour of winter surface salinity anomalies when the salinity is lower than the mean winter salinity. Like Figure 11, the greatest negative anomalies are found in the northwest region of the bay and the lowest negative anomalies are found near the ocean inlets and east of Jones inlet.

AMMONIA:

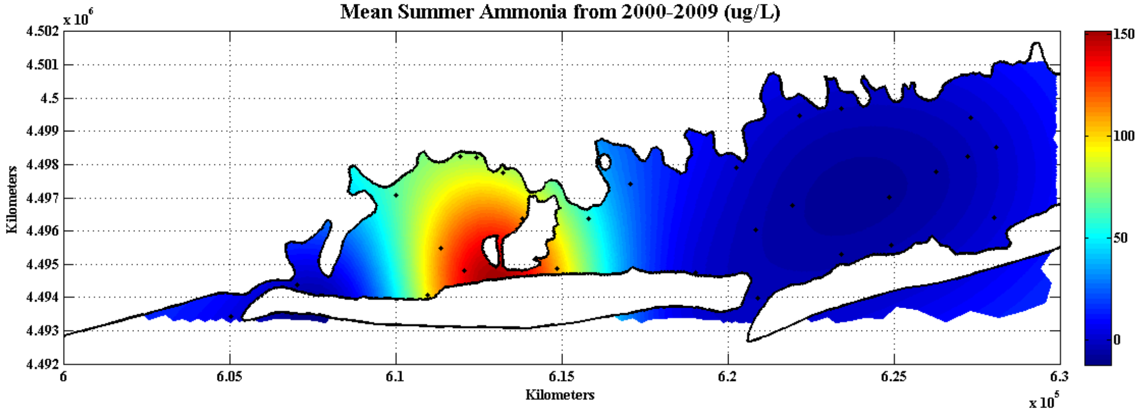

Figure 13: A contour of mean summer surface ammonia from 2000-2009. The highest ammonia is seen at the Bay Park outfall and northward of the outfall. The rest of the bay has much lower concentrations of ammonia.

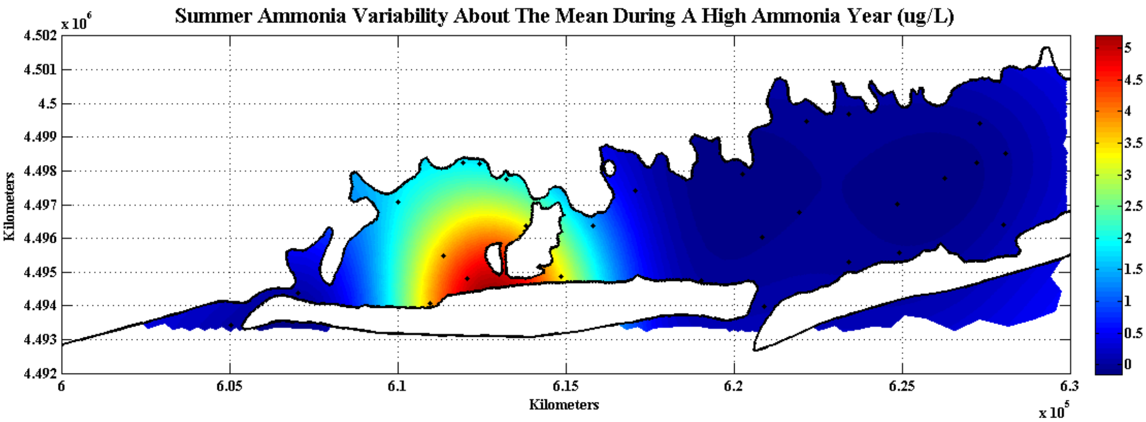

Figure 14: A contour of summer surface ammonia anomalies when the ammonia is higher than the mean summer ammonia. The greatest positive anomalies are found by the Bay Park outfall and north of the outfall. This suggests that summer ammonia anomalies are forced primarily by the concentration of ammonia entering the lagoon from the Bay Park outfall.

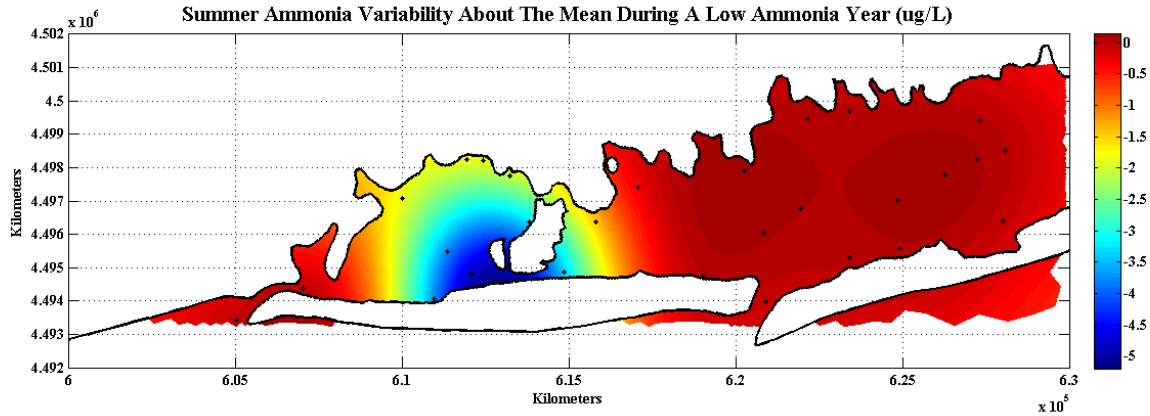

Figure 15: A contour of summer surface ammonia anomalies when the ammonia is lower than the summer mean ammonia. The greatest negative anomalies are found at and northward of the Bay Park outfall.

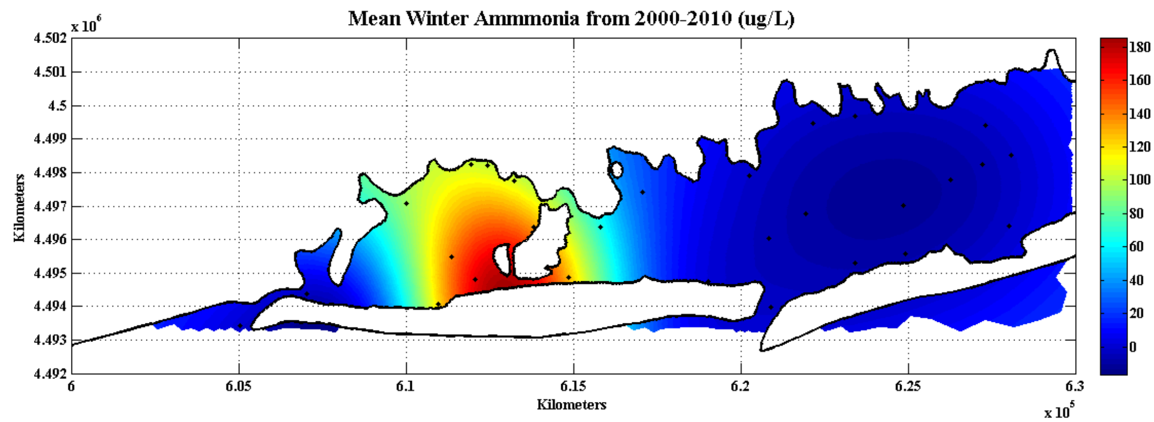

Figure 16: A contour of mean winter surface ammonia from 2000-2010. The highest ammonia concentrations are observed at and northward of the Bay Park sewage treatment plant outfall.

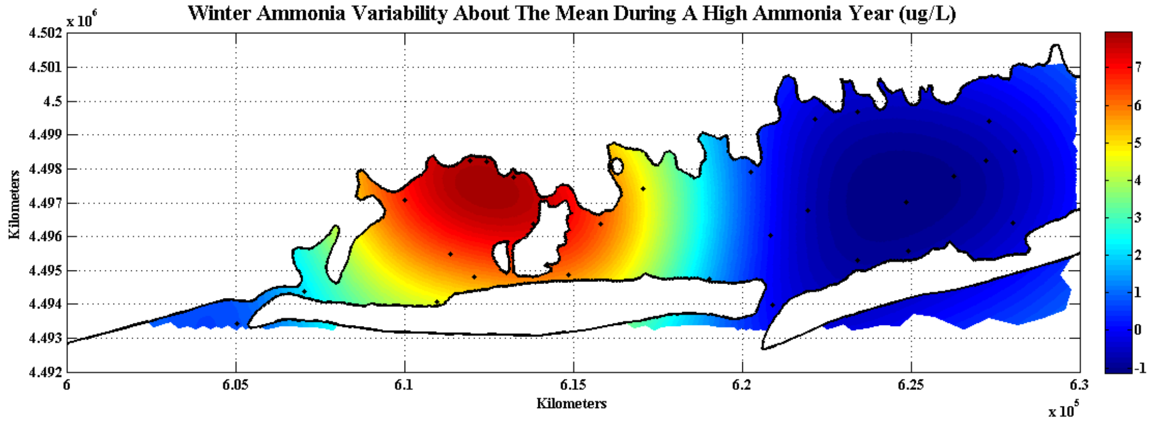

Figure 17: A contour of winter surface ammonia anomalies when the ammonia is higher than the mean winter ammonia. The greatest positive anomalies are found at and northward of the Bay Park sewage outfall. This suggests that winter ammonia anomalies are forced primarily by the concentration of ammonia entering the lagoon from the Bay Park outfall.

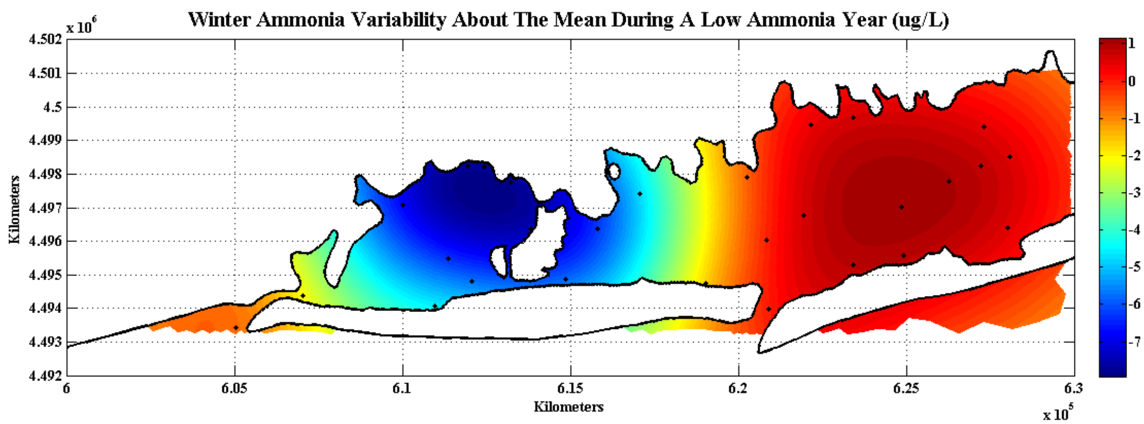

Figure 18: A contour of winter surface ammonia anomalies when the ammonia is lower than the mean winter ammonia concentrations. The greatest negative anomalies are found at and northward of the Bay Park outfall and the lowest negative anomalies are found by the two inlets and the east end of Hempstead Bay.

NITRATE:

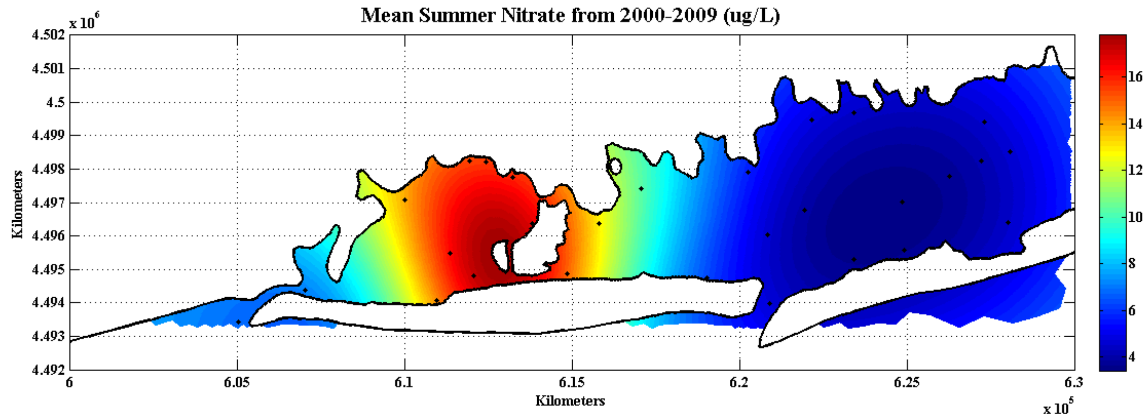

Figure 19: A contour of mean summer surface nitrate from 2000-2009. The highest concentrations of nitrate are seen at and northward of the Bay Park outfall.

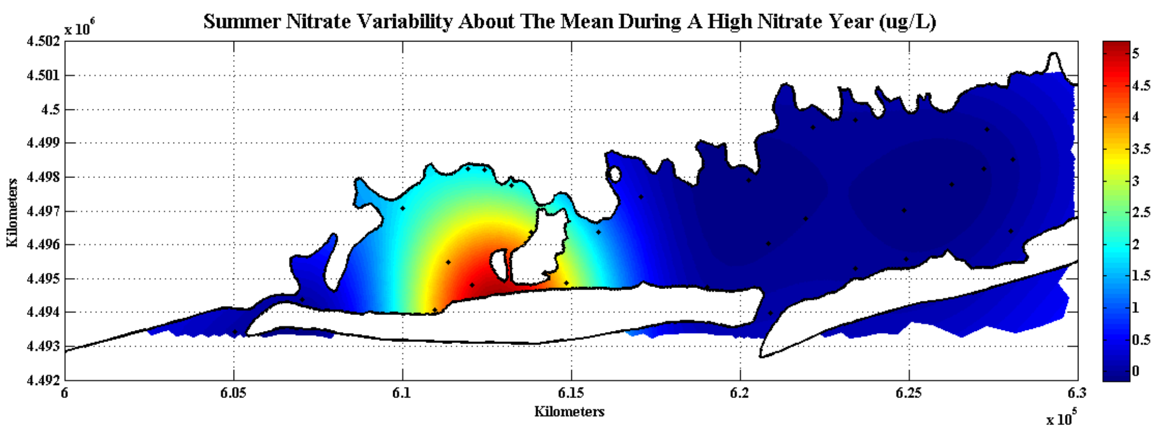

Figure 20: A contour of summer surface nitrate anomalies when the nitrate concentrations are higher than the mean summer nitrate concentrations. The greatest positive anomalies are found at the Bay Park sewage treatment plant outfall. This suggests that summer nitrate anomalies are forced primarily by the concentration of nitrate entering the lagoon from the Bay Park outfall.

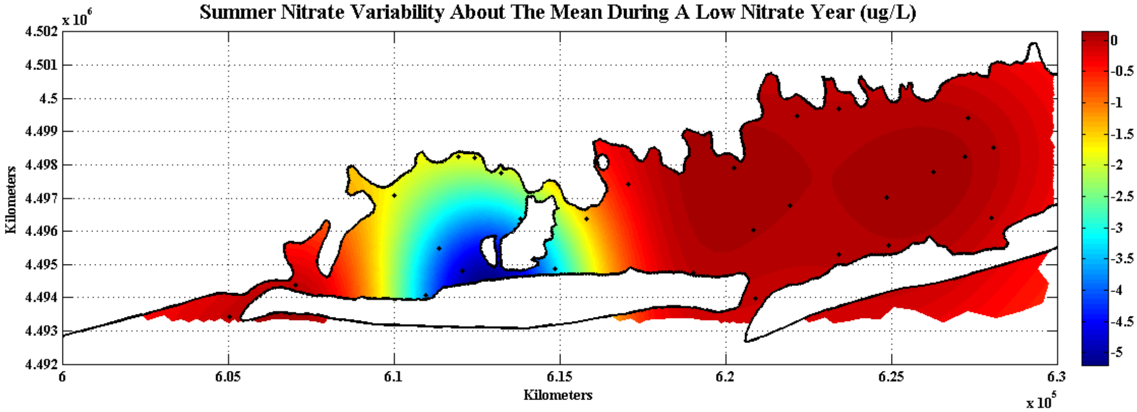

Figure 21: A contour of summer surface nitrate anomalies when the nitrate concentrations are lower than the mean summer nitrate concentrations. The greatest negative anomalies are found near the Bay Park outfall.

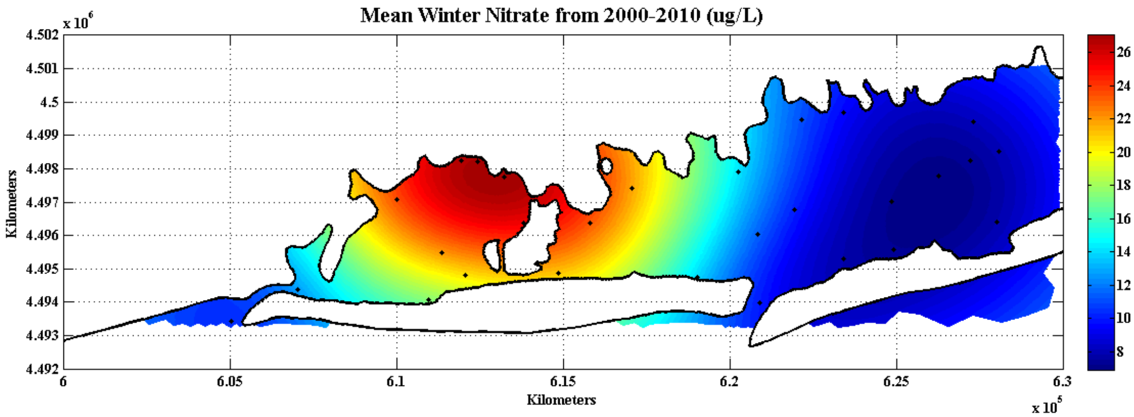

Figure 22: A contour of mean winter surface nitrate from 2000-2010. The highest concentrations of nitrate are seen at the northwest end of the bay, just north of the Bay Park outfall. The rest of the bay has relatively lower nitrate concentrations.

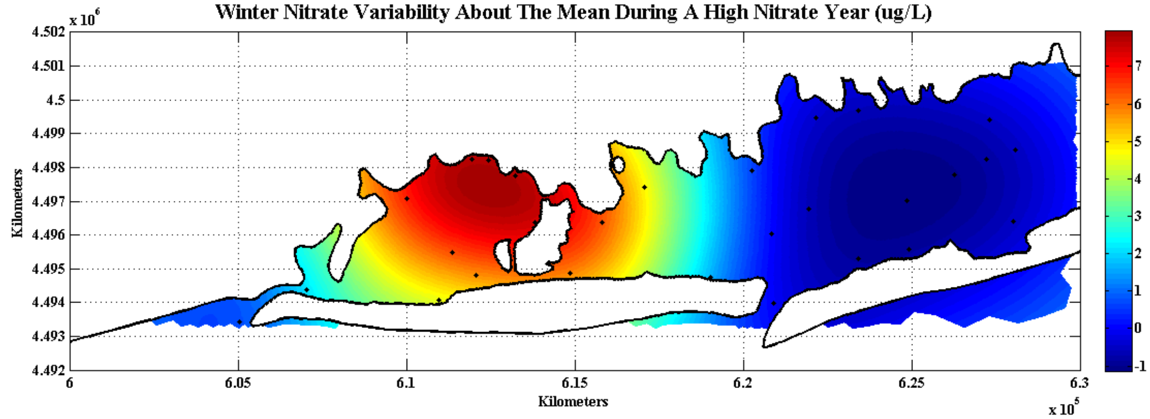

Figure 23: A contour of winter surface nitrate anomalies when the nitrate concentrations were higher than the mean winter nitrate concentrations. The greatest positive anomalies are found north of the Bay Park Sewage Treatment Plant outfall. This suggests that winter nitrate anomalies are forced primarily by the concentration of nitrate entering the lagoon from the Bay Park outfall.

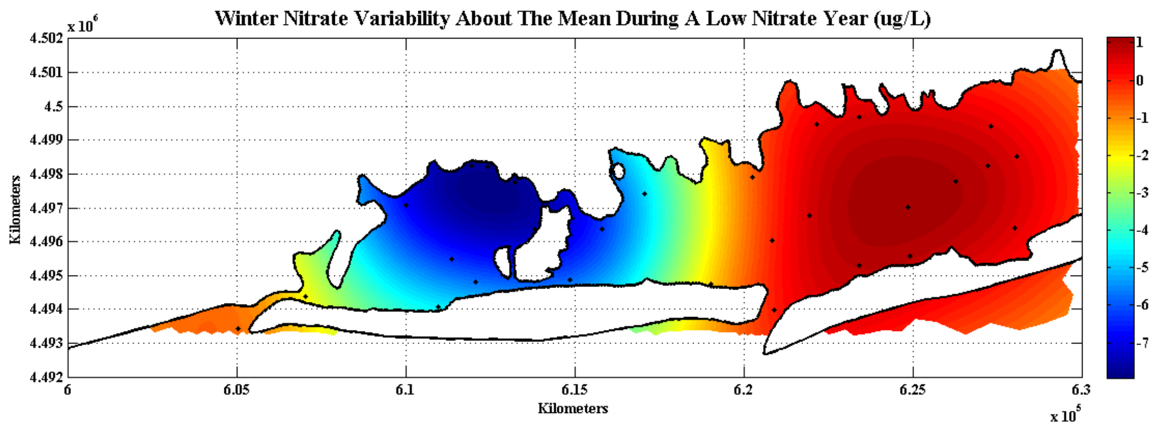

Figure 24: A contour of winter surface nitrate anomalies when then nitrate concentrations were lower than the winter mean nitrate concentrations. Again, the greatest negative anomalies are found north of the Bay Park outfall and the lowest positive anomalies are found by the inlets and to the east.

NITRITE:

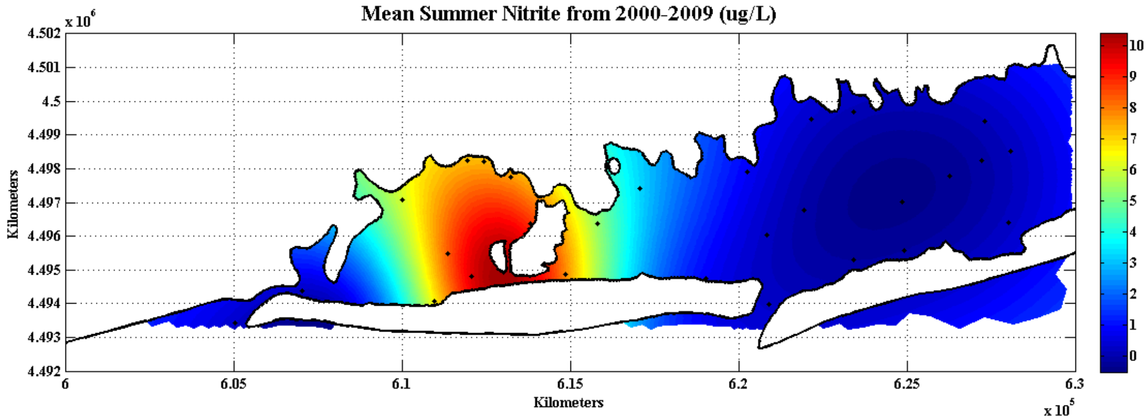

Figure 25: A contour of mean summer surface nitrite from 2000-2009. The highest concentrations of nitrite are at and northward of the Bay Park outfall.

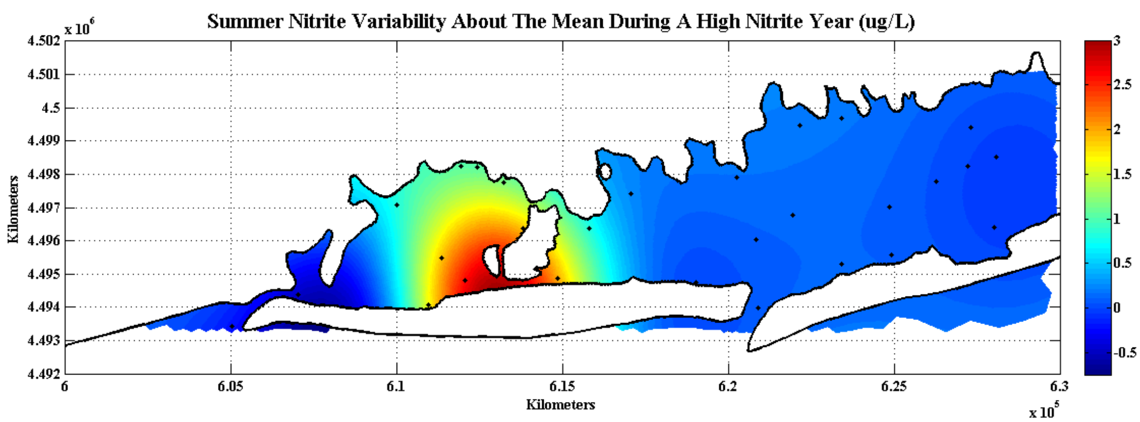

Figure 26: A contour of summer surface nitrite anomalies when the concentrations of nitrite are higher than the mean summer nitrite concentrations. The greatest positive anomalies are found near the Bay Park Sewage Treatment Plant outfall. This suggests that summer nitrite anomalies are forced primarily by the concentration of nitrite entering the lagoon from the Bay Park outfall.

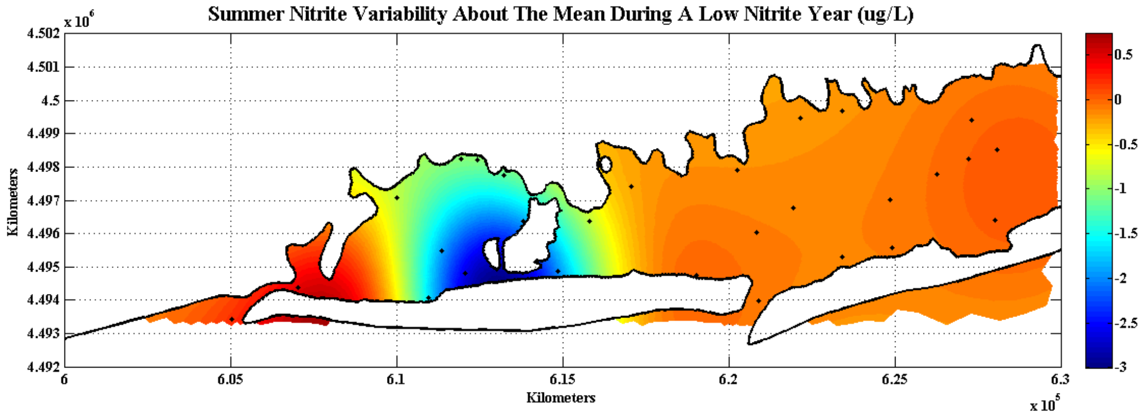

Figure 27: A contour of summer surface nitrite anomalies when the concentrations of nitrite are lower than the mean summer nitrite concentrations. Like Figure 26, the greatest negative anomalies are found near the Bay Park outfall.

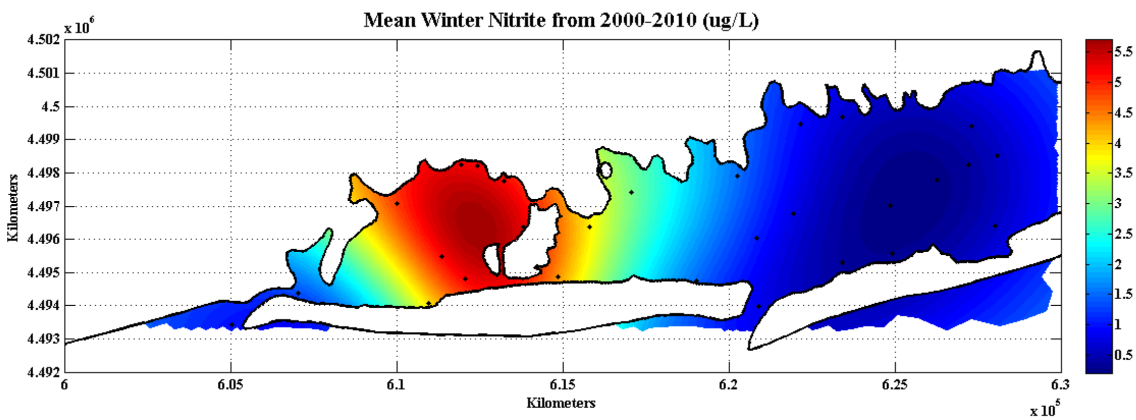

Figure 28: A contour of mean winter surface nitrite concentrations from 2000-2010. The highest concentrations of nitrite are at and to the north of the Bay Park outfall.

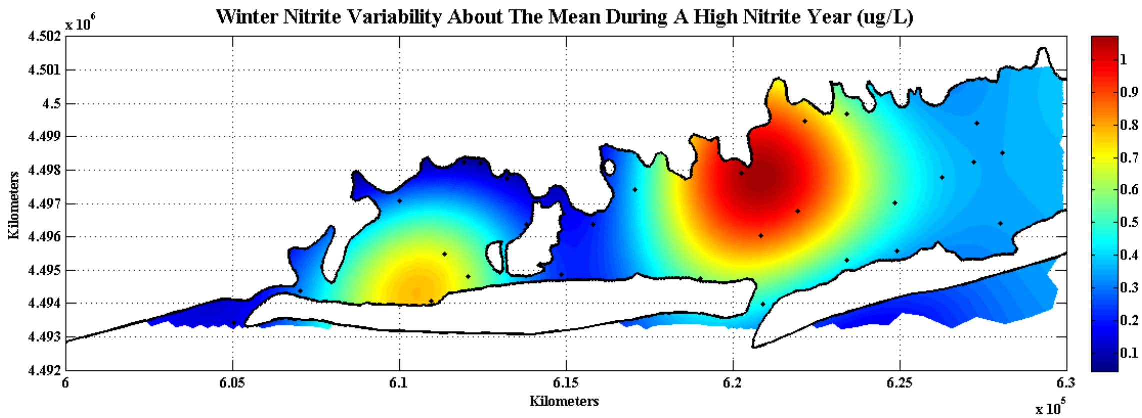

Figure 29: A contour of winter surface nitrite anomalies when the concentrations of nitrite are higher than the mean winter nitrite concentrations. The greatest positive anomalies are found north of Jones inlet and just east of East Rockaway inlet.

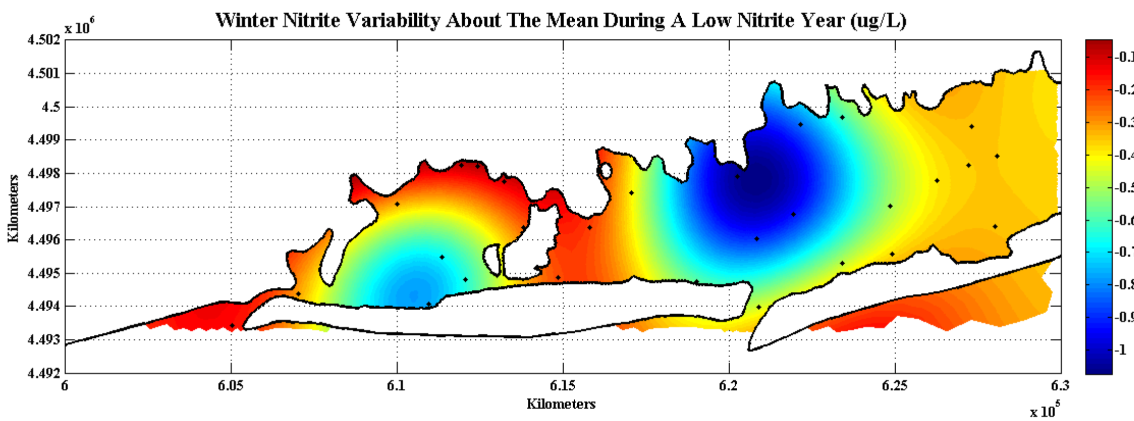

Figure 30: A contour of winter surface nitrite anomalies when the concentrations of nitrite are lower than the mean winter nitrite concentrations. Like Figure 29, the greatest negative anomalies are found to the north of Jones inlet and to the east of East Rockaway inlet.

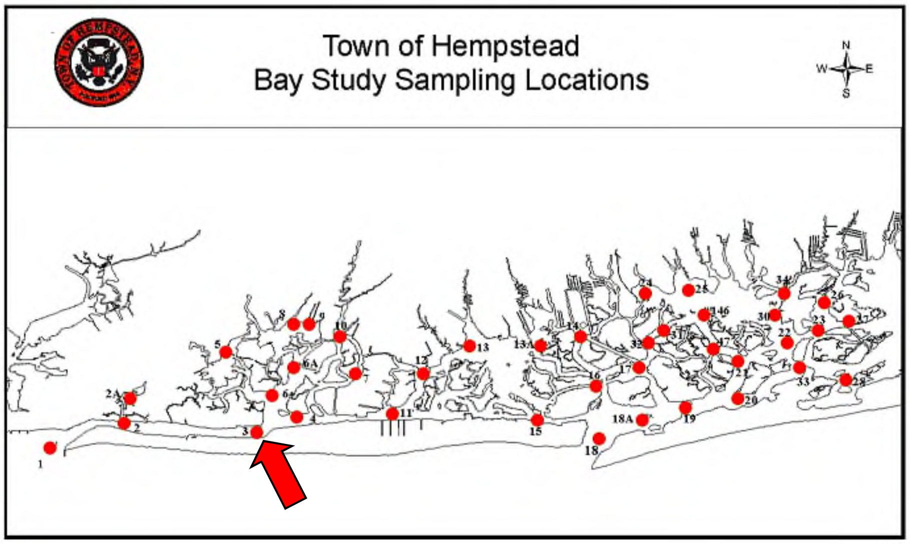

Figure31. Town of Hempstead station in Reynolds Channel used for decadal trend plots in figures below.

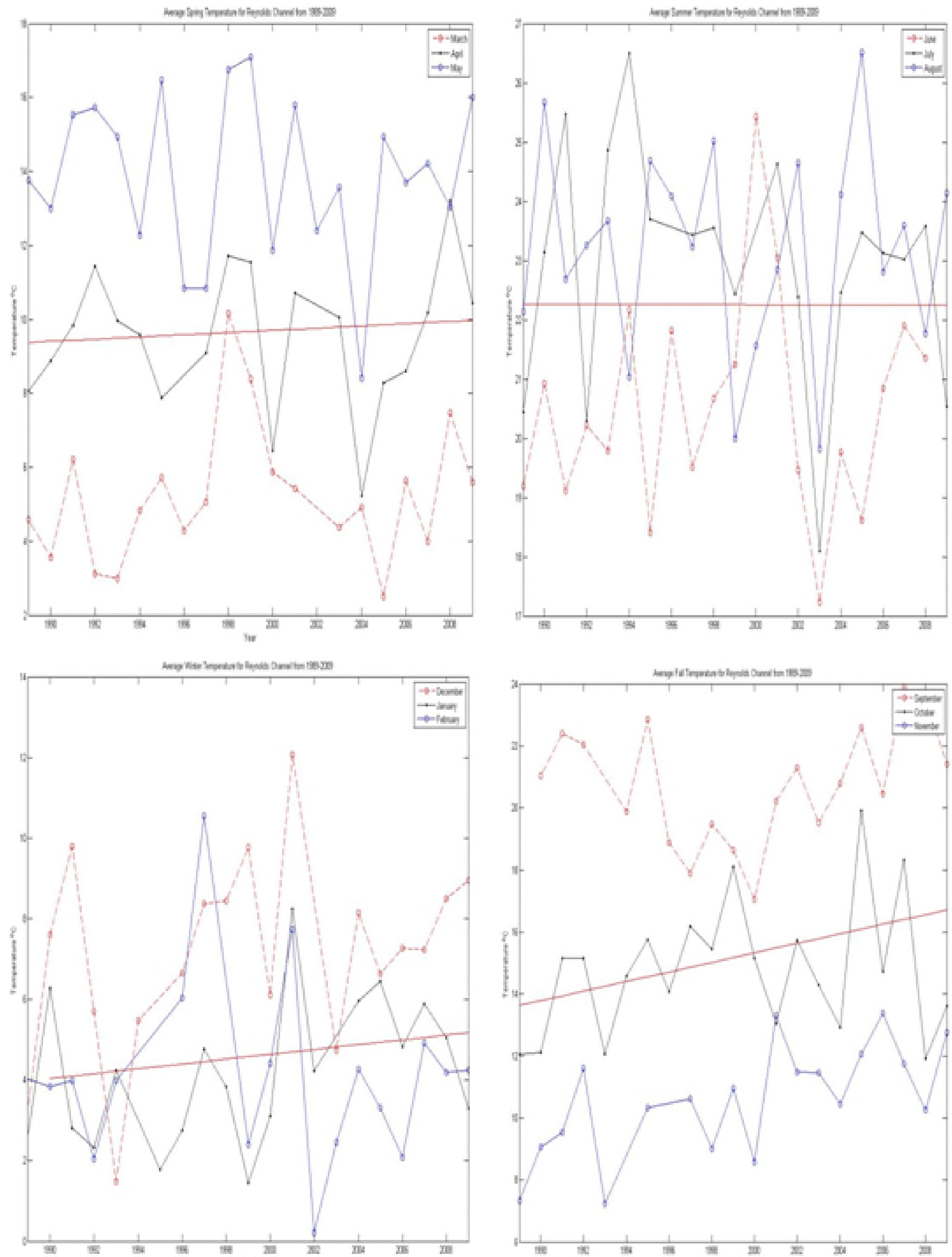

Figure 32. Average surface Temperature trends 2000-2010 (clockwise from upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station

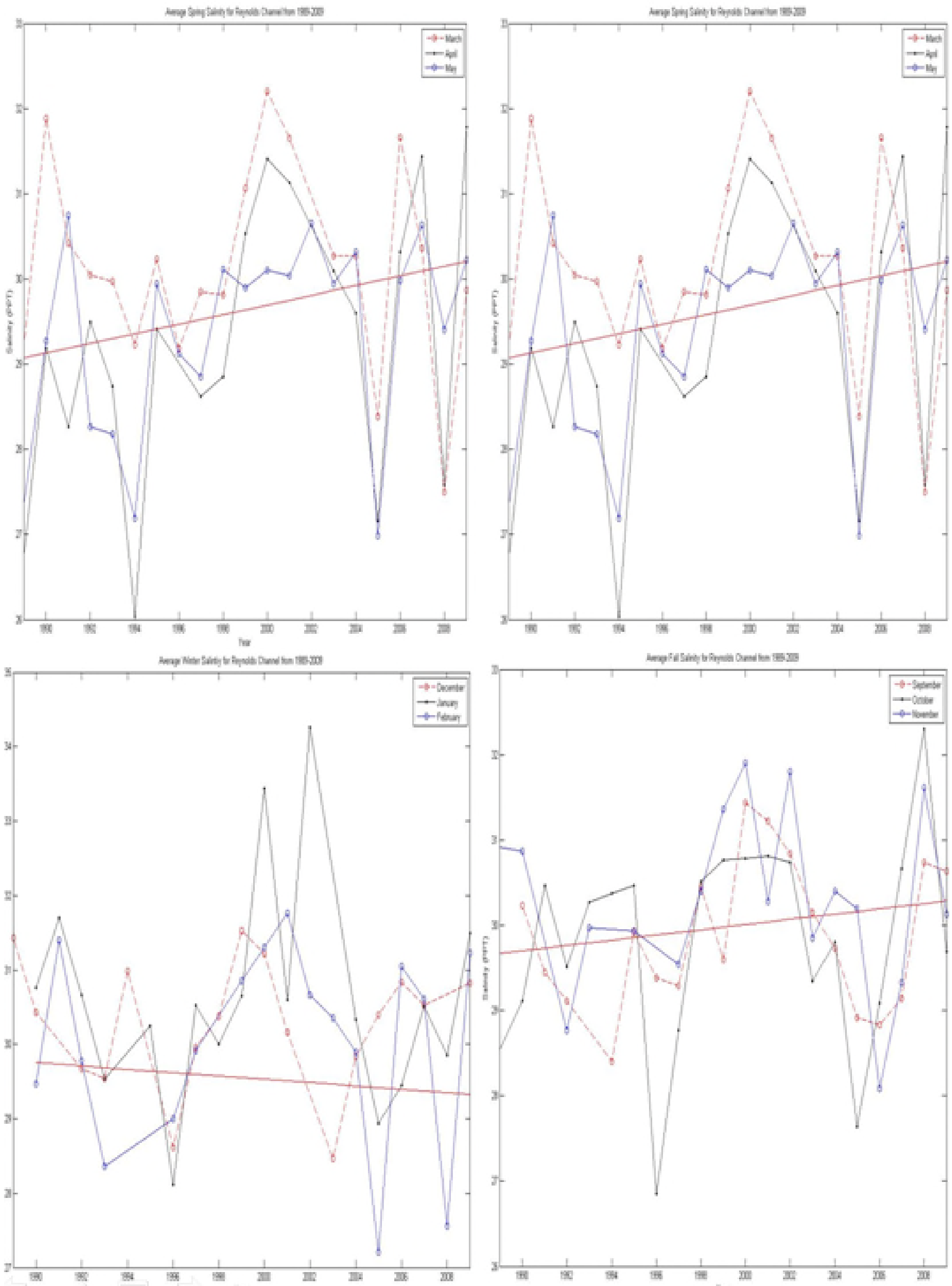

Figure 33. Average surface Salinity trends 2000-2010 (clockwise from upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station

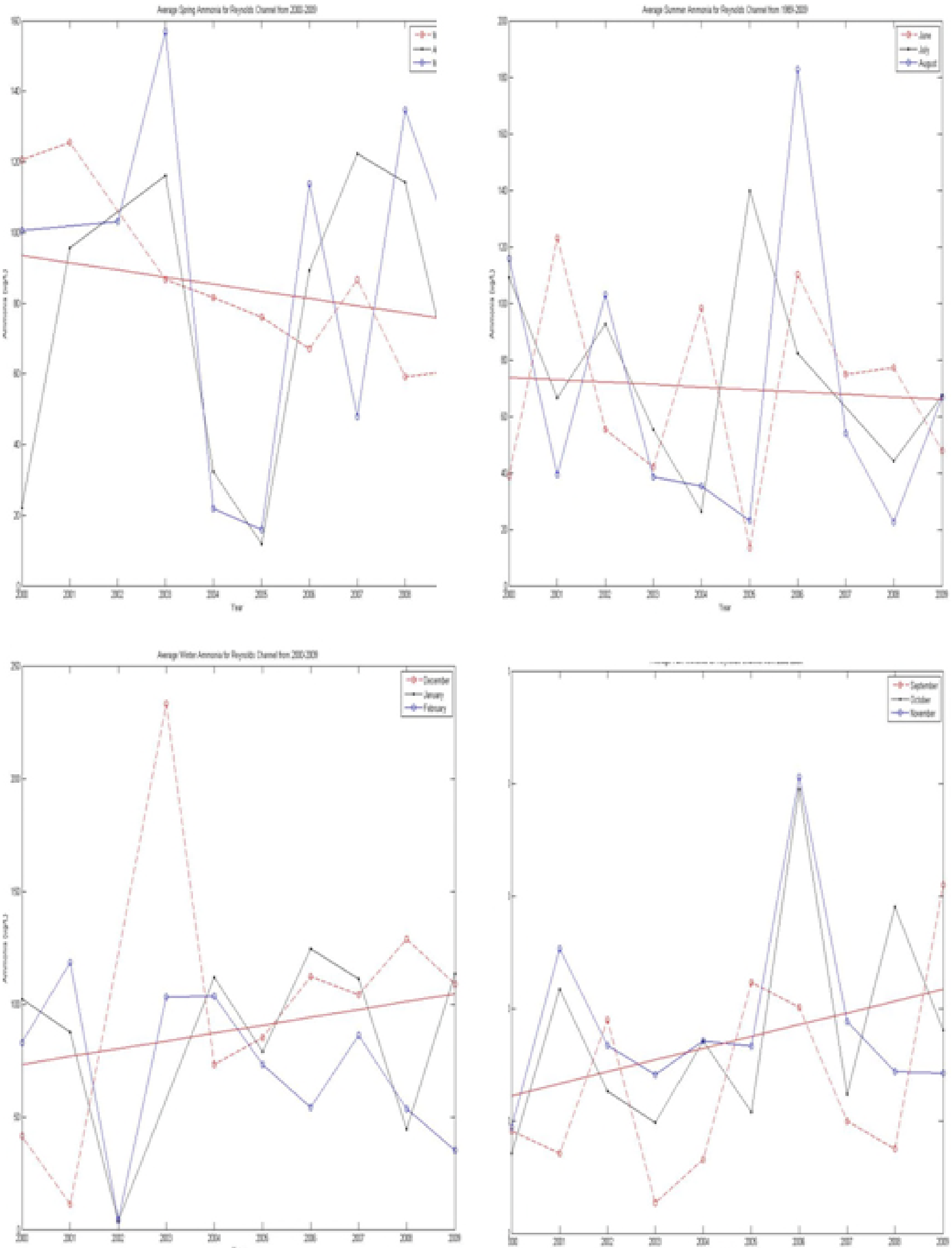

Figure 34. Average surface Ammonia trends 2000-2010 (clockwise from upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station

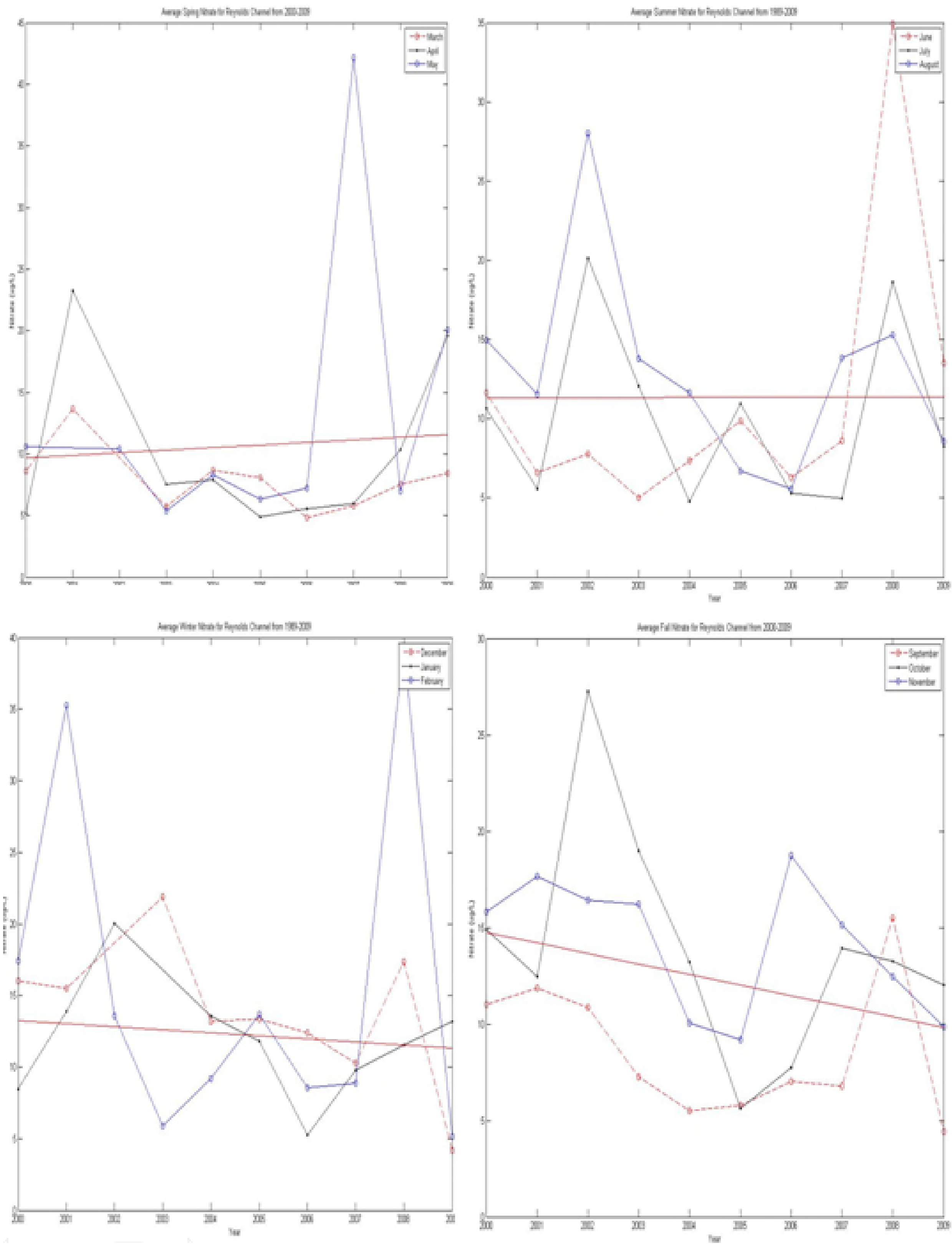

Figure 35. Average surface Nitrate trends 2000-2010 (clockwise from upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station

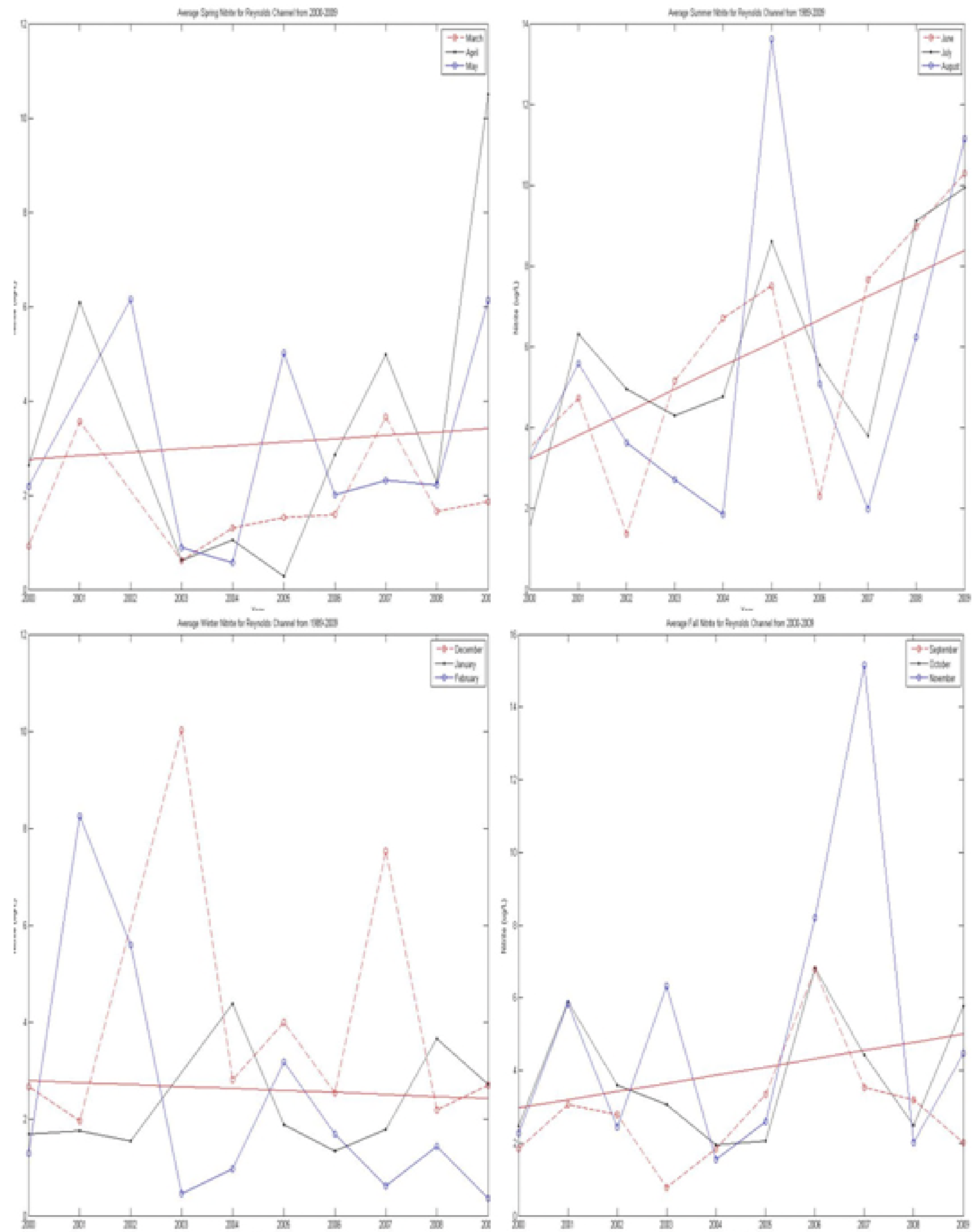

Figure 36. Average surface Nitrite trends 2000-2010 (clockwise from upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station

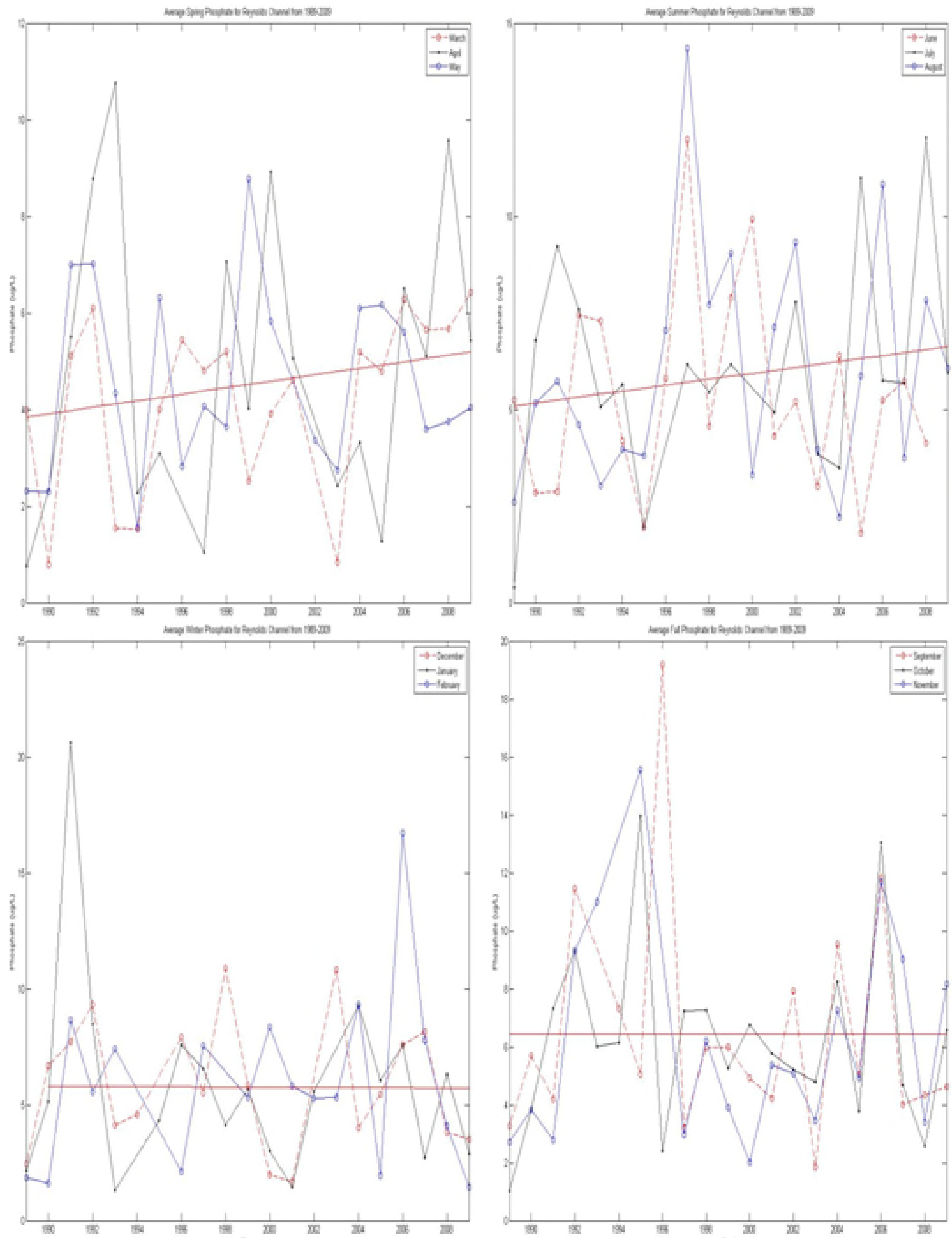

Figure 37. Average surface Phosphate trends 2000-2010 (clockwise from upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station

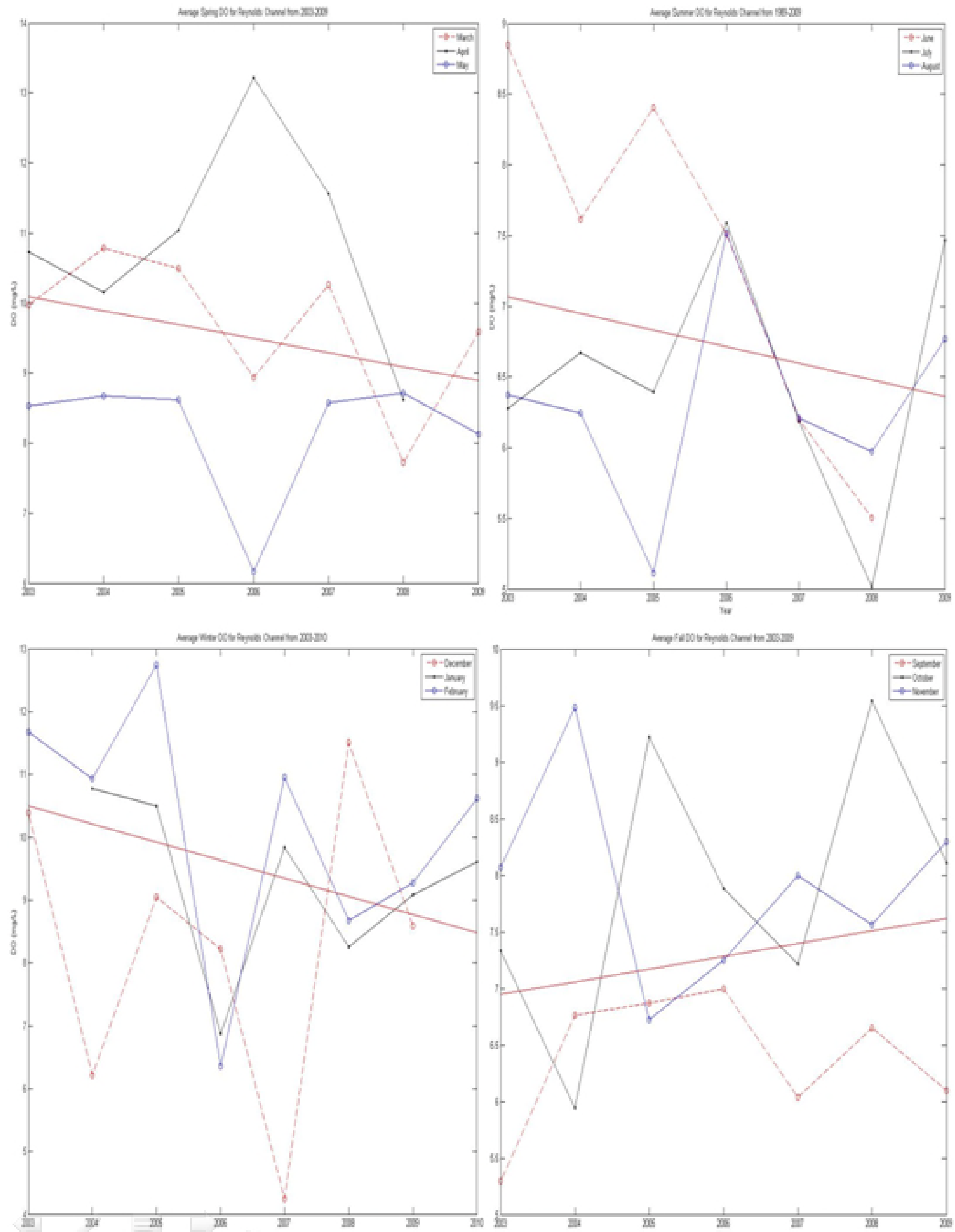

Figure 38. Average surface Dissolved Oxygen trends 2000-2010 (clockwise upper left spring, summer, fall, winter) as measured by Town of Hempstead at Reynolds Channel station.

APPENDIX G: HEMPSTEAD BAY SEDIMENT NUTRIENT FLUXES –FINAL REPORT

Hempstead Bay Sediment Nutrient Fluxes –Final Report

Michael S. Owens and Jeffrey C. Cornwell
Chesapeake Biogeochemical Associates
PO Box 167
Sharptown MD 21861

February 15, 2013

Prepared For:

Battelle Memorial Institute

Prepared By:

Michael Owens and Jeffrey Cornwell
Chesapeake Biogeochemical Associates
P.O. Box 167
Sharptown, MD 21861

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Acknowledgements

Summary

Measurement of the sediment-water exchange of gases (N_2, O_2) and nutrients (NH_4^+, NO_{2+3}^-, NO_2^-, soluble reactive P) were made in October 2012 at 9 sites in Hempstead Bay, New York. Cores were collected using a pole corer and a box corer and replicate sediment cores were incubated using standard procedures. The results showed variable but often high rates of oxygen uptake, ammonium efflux, and soluble reactive P release. Denitrification rates were variable and at several sites quite high.

The stoichiometry of nutrient efflux suggests that oxygen may be an imperfect measure of overall sediment metabolic rates. High ammonium effluxes and high SRP effluxes suggest either unmeasured metabolism or perhaps a day/night issue with nutrient uptake/release by benthic micro- or macro-algae. Overall, the benthic system in Hempstead Bay appears to have high rates of nutrient release that, depending on rates of wastewater/non-point source inputs and flushing, could support significant amounts of algal production.

Introduction

In shallow water aquatic environments, sediments can be important mediators of nutrient cycling. In the simplest case, the settling of suspended algae to the sediment-water interface results in the decomposition of organic matter within the sediments and the consequent remineralization of organic nitrogen and phosphorus. The remineralized nitrogen and phosphorus can be retained and buried within the sediment, incorporated in metazoan biomass through grazing or deposit feeding, flux back into the water column as bioavailable inorganic N and P forms, or in the case of N, denitrified. The more complex case occurs when benthic microalgae intercept remineralized N or P and either hinder or enhance denitrification. Key controls on these processes include the presence of oxygen in the water column and sediment, presence of benthic animals, overall rate of organic matter deposition.

The purpose of this measurement program was to provide sediment-water exchange measurements for gases (oxygen, di-nitrogen) and nutrients (soluble reactive P, ammonium, nitrate) for a transect of sediment sites in Hempstead Bay. Site selections were provided to Chesapeake Biogeochemical Associates and were chosen to provide geographic coverage.

Methods

Field and Incubation Techniques

Sediments were collected at all but one site using a pole corer which utilized a valve at the top end of the core to provide suction to keep the sediment in the core barrel upon withdrawal from the sediment. A Soutar light plastic box corer was used for sampling site ULVA 11A, with two of the acrylic core liners inserted in the box for sub-coring. Site locations are shown in Figure 1. Excellent quality cores were obtained using both devices.

Station water was collected in carboys from each site. Cores were kept in coolers with no air bubbles while on the ship; this minimized sediment disruption associated with boat motion. Upon return to land, the cores were placed in our incubators, a head space was added, and cores were transported to Maryland for incubation. This step is required to minimize oxygen stress on both infauna and bacteria. From previous work on Jamaica Bay, we observed that the best diagnosis of stress was the presence of sediment amphipods in the water overlying the sediment. Our transport to Cambridge, Maryland with this aeration succeeded with minimal stress (i.e. no migration of amphipods out of the sediments).

Table 2 shows the overall outline of how we do our sediment-water exchange work. The outside of the tank is double-walled and water from a refrigerated circulator is continuously pumped through the outside chamber to maintain temperature. Temperatures were adjusted to those measured *in situ* at each site. The central turntable has high performance magnets turned by a 12 v motor that is adjustable for various rotations per minute (rpm). Incubation cores were 10.2 cm id and 30 cm long. The cores are made of translucent PVC for the tube, clear acrylic for the top. The bottom cap is made of acrylic and uses o-rings for sealing the bottom.; care must be taken in coarse-grained sediment to ensure a good seal The tops consist of an o-ring cap, a down-rod with attached 1.5" magnetic stir bar, and two Luer through-fittings for attaching tubing to a replacement water port and a sampling port.

Upon return to the incubation facility, the open sediment cores were bathed for overnight hours in overlying water from the site. We used a bubble-lift system to flush the cores completely to maintain oxygen saturation. This pre-incubation provided the cores sufficient time to come to thermal equilibrium; although not critical for solute fluxes, we have observed a large improvement in the performance of our denitrification measurements with such pre-incubations. Gases in the plastics can have a second order impact on our measurements and the pre-incubation equilibration conditioning minimizes that effect. Incubations generally started with oxygen near saturation. Generally, oxygen levels were not allowed to drop below 50% saturation.

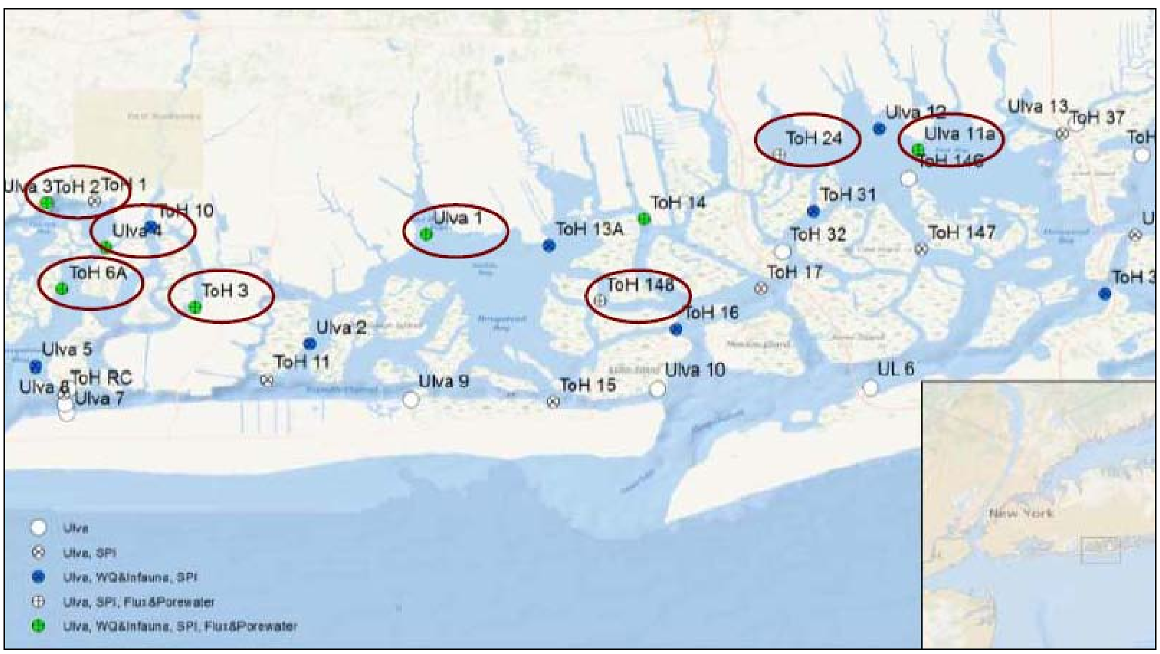

Figure 1. Station locations showing locations where cores were collected. Two sets of cores were collected at ToH 148, with the first set closer to the shoreling.

Table 1. Synopsis of sediment flux measurement approach.

Core Collection	Direct insertion of 4" id tube into sediment, capped underwater with a PVC insert with an o-ring seal. Cores were kept out of sun in cooler on deck. Cores were translucent PVC, allowing acid cleaning if needed.
Water collection	Whole water collected in 20 L carboys
Incubator	Stainless steel double wall construction, water circulated between the two outer walls to maintain water temperature. A pumping circulator was attached.
Pre-Incubation	Upon arrival at incubation site, cores were placed into the incubator. The circulator temperature was set to field temperatures. Filtered overlying water (nominally 1.0 μm) was added to the chamber to a level above the cores. T-shaped bubblers were added to pump water from cores into overlying water bath water, circulating and aerating the water. This promotes oxygen saturation and thermal equilibrium. The pre-incubation period was generally overnight, though periods as short as 2 h work well. Cores were not illuminated during the pre-incubation period.
Setting up the experiment	The cores were capped with o-ring sealed spinning tops; the spinners were Teflon-coated magnets. Care was taken to exclude bubbles. When the tops were in place, the input ports on the top were attached to tubes leading from the replacement water tank, whose bottom was placed ~1.5' higher than the core tops. A magnetic turntable was switched on to commence stirring. Between sample points, a black plastic sheet was put over the top to shield the cores from light.
Gas Sample Collection	At appropriate intervals, samples were collected for gas analysis. A 8" tube was attached to the outlet of the core, the replacement water valves were opened, and a 7 mL ground glass stoppered tube was filled to overflowing, from the bottom of the tube, using gravity to push water out of the core. Sample tubes were overflowed with approximately 2 sample volumes and the 3rd volume was the sample to be analyzed. 10 μL of 50% saturated $HgCl_2$ was added as a preservative. The samples were kept under water in a cooler until analysis; the temperature was ≤ ambient temperature. Samples were analyzed within 1 week of return to Maryland.
Solute Collection	A 20 mL syringe barrel was attached to the core outlet, the replacement water valves were opened, and the barrel filled to the top. The plunger was inserted and the syringe removed. All valves were closed. Samples were filtered through 0.4 μm pore size 25 mm diameter syringe filters. Samples were frozen immediately after collection and remained frozen until analysis.

Sediment core incubations consisted of 4 time points. Sediment core incubations were sampled for dissolved oxygen, di-nitrogen (N_2), argon (Ar), soluble reactive phosphorus (SRP), ammonium (NH_4),and nitrate + nitrite (NO_{2+3}). Samples for gas analysis of N_2, O_2 and Ar via mass spectrometry were collected in 7 ml glass stoppered tubes, preserved with mercuric chloride (Kana et al. 2006), stored under water at temperatures lower than the incubation temperature, and stored for < 1 week. Replacement water was fed by gravity into the tops of the cores; this pressure also pushes sample water into collection tubes. Syringe barrels were then attached to the cores and are filled by gravity. Syringe filters (0.4 μm) were used for filtration of dissolved components; samples were stored frozen in individual 7 mL polycarbonate vials until analysis.

At the end of the remote sediment core incubations, each core was photographed and overlying water column of each core was measured.

Pore Water Collection

For pore water collection, cores were placed into a glove bag filled with N_2 to minimize oxidation artifacts (Bray et al. 1974) and sectioned. The cores were sectioned into the following depth intervals: 0.0–0.5 cm, 0.5–1.0 cm, 1–2 cm, 2–3 cm, 3–5 cm, and 8–10 cm. Sediment was placed into 50-mL centrifuge tubes, spun at 3000 rpm for 10 minutes, and the supernatant water was filtered with 0.45-µm syringe filters. Water was collected in 7-mL vials and the pore water rapidly distributed into vials for hydrogen sulfide analysis. Based on previous work in nearby Jamaica Bay, pore water was diluted using calibrated Gilson pipettes, allowing a more rapid analysis at the nutrient laboratory. Nutrient samples were frozen for later analysis. Cores with little water content or coarse grain size yielded no water or insufficient water for chemical analysis.

Nutrient and Sulfide Analysis

Nutrient analyses (Table 2) were being carried out by the analytical services group at Chesapeake Biological Laboratory (CBL; http://www.cbl.umces.edu/nasl/index.htm). They also analyzed sulfate and chloride via ion chromatography. They have implemented a full QA/QC process within their laboratory, with complete details on their website.

Table 2. Solute Analysis.

Analyte	Reference	Description
soluble reactive P	(Parsons et al. 1984)	Automated colorimetric analysis (d.l. < 0.005 mg L^{-1})
NH_4^+	EPA 365.2	Automated Phenol/hypochlorite colorimetry
$NO_3^- + NO_2^-$	(Parsons et al. 1984)	Automated colorimetric analysis (d.l. < 0.03 mg L^{-1})

The hydrogen sulfide analysis followed the standard colorimetric procedure of Cline (1968) as described in Parsons *et al*. (1984).

Membrane Inlet Mass Spectrometry

Membrane inlet mass spectrometry (MIMS) will be used for analysis of oxygen. This mode of measurement is more precise than electrode or titration methods and requires small volumes of water that can be preserved for up to 3 weeks. A key advantage of this approach is that we simultaneously, at no extra charge, analyze for N_2 and Ar and can measure denitrification (Kana et al. 1994, Cornwell et al. 1999).

The sample is analyzed by pumping the water through a membrane tube situated inside the mass spectrometer vacuum. In practice, it requires ca. 1-2 minutes for a measurement to be completed. Generally, individual gas concentrations can be measured with a precision of 0.1-0.2% c.v. and gas ratios (e.g. N_2/Ar) can be measured with a precision of 0.02-0.03% c.v. Week to week repeatability for O_2 concentration has been determined to be better than 0.2% c.v.

Fluxes were calculated from the linear slope of the change in Ar-normalized gas ratios, which provide the highest resolution. Ar-normalized gas ratios were converted to gas concentrations by multiplying the gas ratio by the Ar concentration assuming Ar is in air equilibrium. Mass spectrometer discrimination for the gas of interest relative to Ar was determined from measurements of air equilibrated standards. The standards were prepared using deionized water held to a temperature tolerance of ±0.02 C with 100% relative humidity in the head space. Standards were stirred, not bubbled. Standards are calibrated against local barometric pressure with a precision (5-digit) pressure gauge. This technique has proven highly accurate and reproducible. We replicated analyses on the initial sampling (i.e. time zero) from each core.

Estimation of Sediment-Water Exchange Rates

Sediment-water exchange rates are 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 has a significant slope, the slope of the flux cores is adjusted accordingly.

Results

Field Observations

Bottom water measurements did not exhibit a large range (Table 3). Temperature ranged from 19.5-20.0°C, dissolved oxygen ranged from 4.6-7.6 mg L^{-1}, pH ranged from 7.5-7.8 and salinity range from 32.3-35.1 psu. The sites were generally shallow (< 3 m), with the exception of Ulva 11A which had depth of 9 m.

Table 3. Field-measured parameters at all stations. The temperature, dissolved oxygen (DO), pH and salinity measurements were from bottom water depths.

Sta	Time 10/03/2012	Temp	DO	Depth	pH	Salinity	Core Photo ID
		°C	mg L^{-1}	m		psu	
TOH 6A	8:48	19.47	5.92	1.25	7.64	33.04	1
TOH 2	9:14	19.61	5.42	1.9	7.56	32.27	2
Ulva 4	9:36	19.67	4.59	1.29	7.51	32.55	3
TOH 3	10:00	19.60	5.77	1.42	7.66	33.32	4
Ulva 1	10:54	19.76	6.80	1.68	7.71	33.97	5
TOH 148	11:15	19.61	7.17	2.88	7.84	34.80	6
TOH 148	11:38	19.75	6.57	2.58	7.74	34.51	7
Ulva 11A	12:34	19.63	6.49	9.0	7.73	34.75	8
TOH 24	13:04	20.00	7.63	0.4	7.80	35.14	9

The sediment cores collected in this study were of high quality; the sediment-water interface was collected intact without an excessive amount of resuspension. The appearance of the cores showed some heterogeneity, with cores 4B (Site TOH3) and 5A (Site Ulva 1) showing green decaying Ulva and dark decaying Ulva respectively (Figure 2). The whitish tinge to the Ulva 1 core is likely elemental sulfur generated by the oxidation of hydrogen sulfide by sulfur oxidizing species such as Beggiatoa.

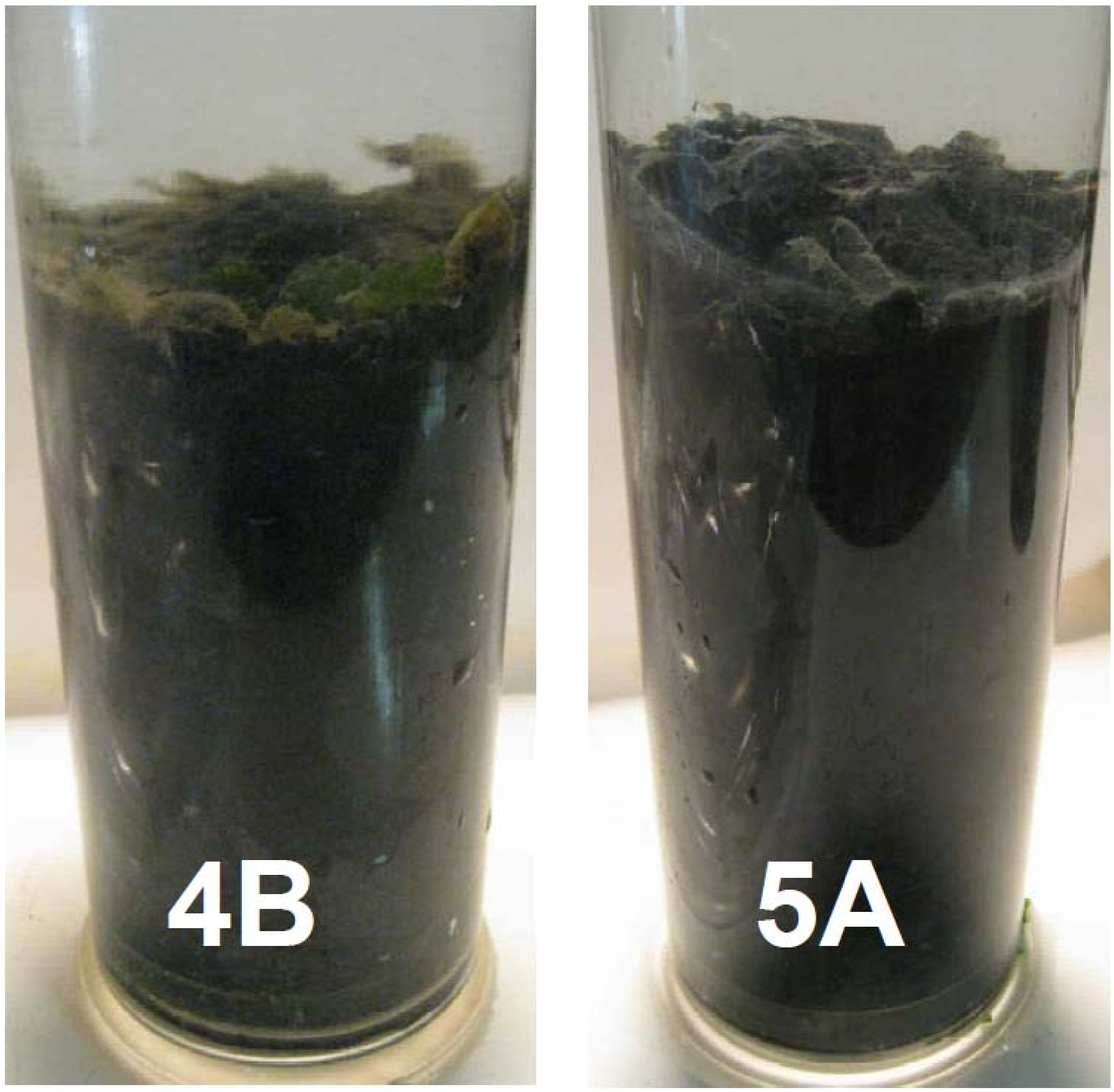

Figure 2. Core photos from the TOH3 site (4B) and from the Ulva 1 site (5A). Photos were taken immediately after incubation experiments.

The whole range of core appearances is shown in Figure 3. Most of the sites exhibited an aerobic-appearing surface layer, with reddish-brown coloration suggesting the presence of iron oxides. The dark black lower layer in some cores clearly indicates the presence of iron monosulfide minerals (aka acid volatile sulfur).

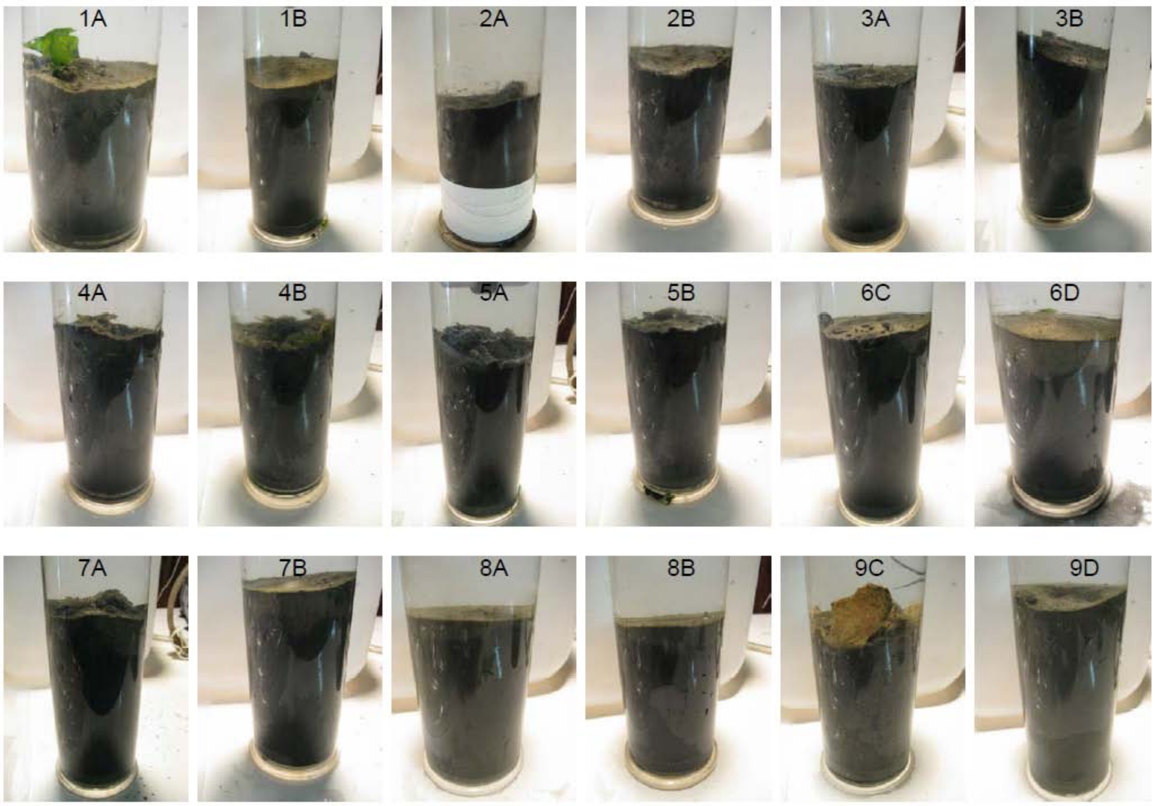

Figure 3. All flux cores from Hempstead 2012 sediment biogeochemical study. The A and B cores are replicates and the core numbers correspond to the locations in Table 3.

Sediment-Water Exchange Rates

In the course of this investigation, all of our sediment-exchange procedures were employed with no particular difficulties. The cores that were collected appeared to be of the highest quality (i.e. undisturbed) and the incubations went well.

The exchange of oxygen at the sediment-water interface occurred at rates typical of impacted coastal ecosystems. Rates > 2,500 μmol m^{-2} h^{-1} are relatively high; Seven of 18 cores had high rates, with 5 having low rates below 1000 μmol m^{-2} h^{-1} (Figure 4). Replication was not especially good with these cores, reflecting differential inputs of organic matter on the scale of 10 m or less. For example, the obvious macroalgal decaying biomass evident in TOH 6A core 1A lead to higher rates of metabolism than observed in core 1B. Higher spatial variability is a general feature of shallow water ecosystems, with differential primary production of benthic microalgae and macroalgae, differential distribution of benthic animals, and variable organic matter deposition to the sediment-water interface.

The flux of ammonium was highly variable (Figure 5), with the highest rates at the high end of observations in coastal ecosystems (Boynton and Kemp 2008, Joye and Anderson 2008). The highest rates were > 1,000 μmol m^{-2} h^{-1}, higher even than rates for the anoxic deep Chesapeake Bay (Kemp et al. 2005). We have observed many rates in coastal ecosystems that were very low or actually representing an uptake, but the site to site and core to core variability in this ecosystem are unusual. Similarly, we observed both high rates of uptake of nitrate plus nitrite, as well as high rates of efflux (at one site). Most rates were relatively low. The rates of denitrification ranged from extremely low (< 25 μmol m^{-2} h^{-1}) to high (> 300 μmol m^{-2} h^{-1}).

The fluxes of soluble reactive P (SRP or orthophosphate) were variable, but five of the sites had very high SRP effluxes (Figure 6). Thirteen of 19 cores had moderate to high SRP effluxes. One site (TOH 3) had modest and reproducible rates of SRP uptake. The highest rates observed in Hempstead Bay are similar to the highest rates observed in the Chesapeake Bay (Cowan and Boynton 1996).

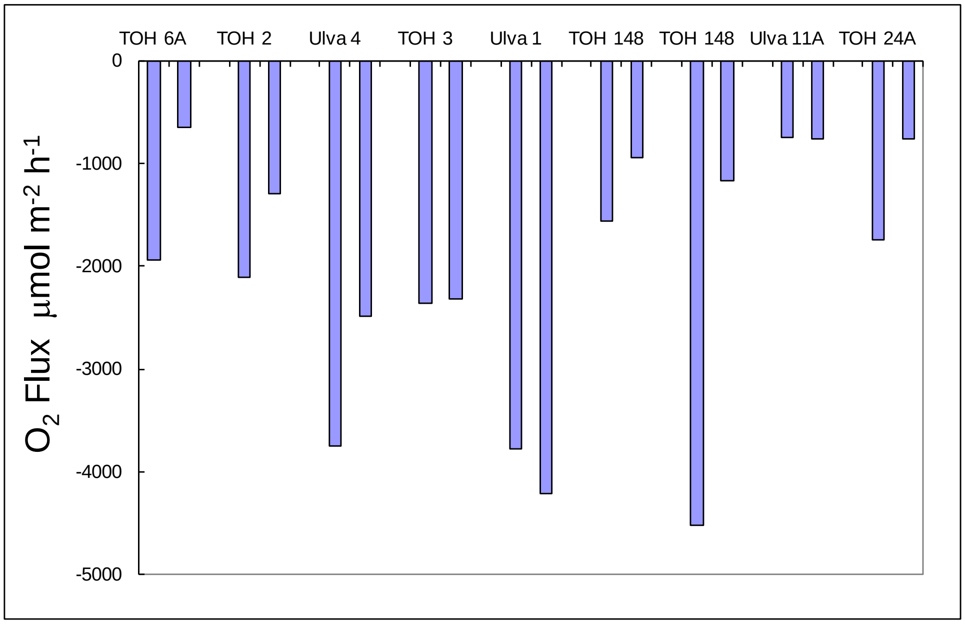

Figure 4. Sediment-water oxygen exchange. The adjacent cores are replicates from the same site. Negative values indicate flux into the sediment.

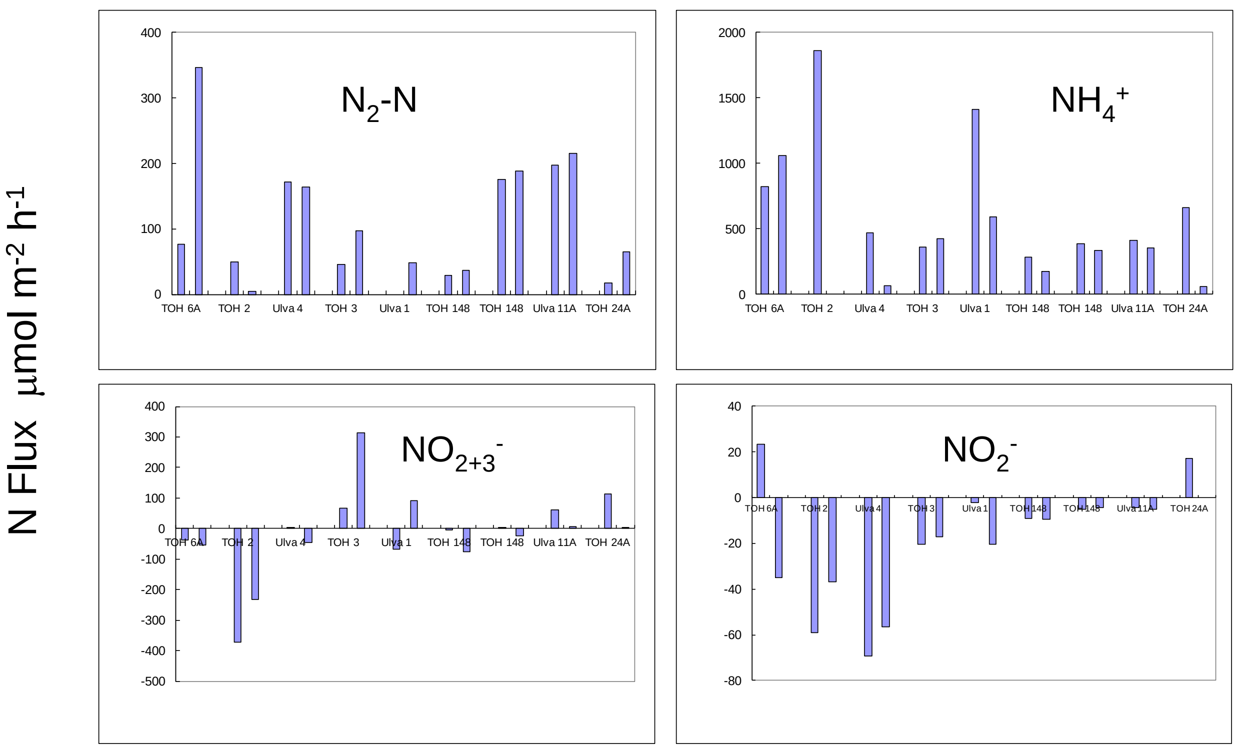

Figure 5. Sediment-water nitrogen exchange. The adjacent cores are replicates from the same site. Negative values indicate flux into the sediment.

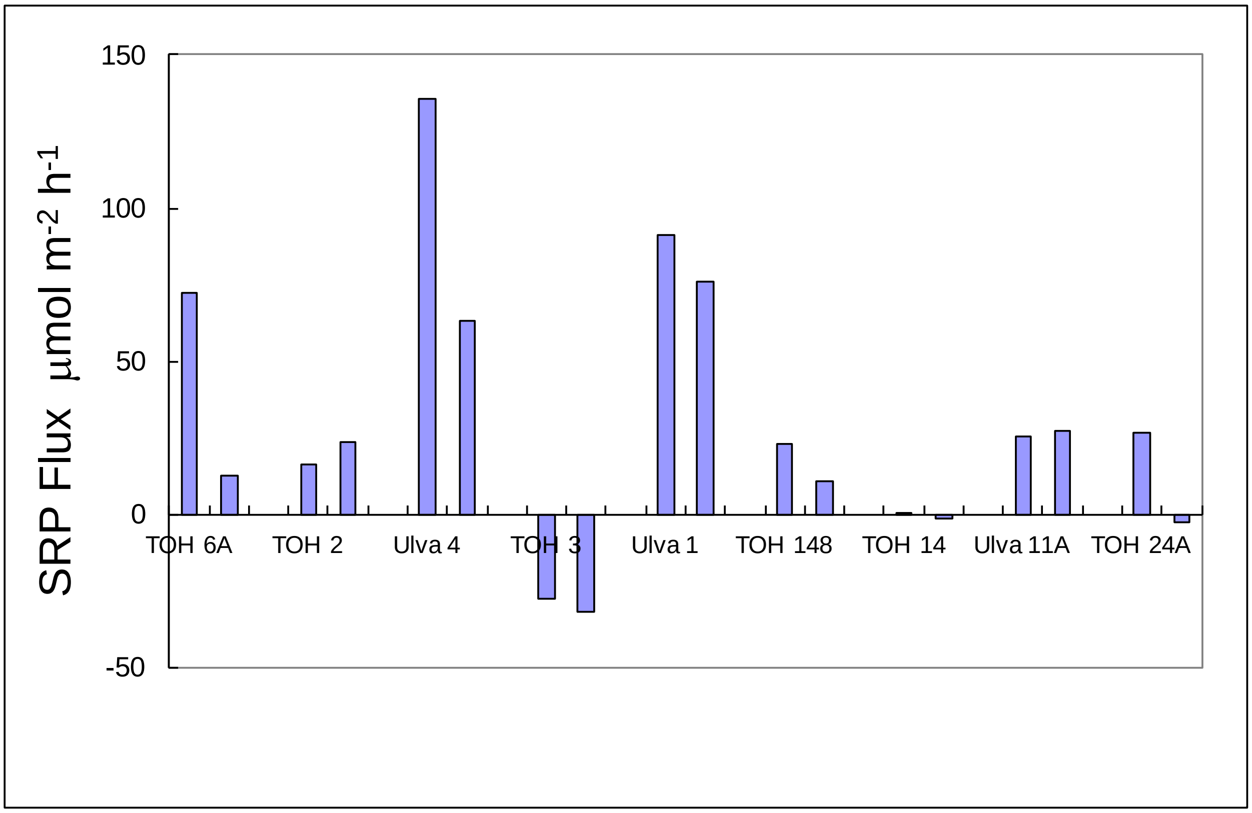

Figure 6. Sediment-water exchange rates of soluble reactive phosphorus (SRP). Negative values indicate flux into the sediment.

Pore Water Chemistry

Consistent with high rates of respiration and nutrient efflux, the pore water nutrient concentrations were very high. When considering these data, it must be recognized that the fine-grained sites are fully represented while coarse sediments that usually have lower concentrations are not.

Table 4. Concentrations of pore water nutrients and hydrogen sulfide. Not all sites are represented because of the lack of water during centrifugation from coarse-grained environments. Dewatered clays also limited pore water recovery. N.D. indicates not determined. Only pore water sections without H_2S present were analyzed for NO_{2+3}^- to avoid analytical interference.

SAMPLE ID	Interval cm	NH_4^+ µmol L^{-1}	NO_{2+3}^- µmol L^{-1}	SRP µmol L^{-1}	H_2S µmol L^{-1}
TOH 6A	0.0-0.5	374	0.4	156	0
	0.5-1.0	580	0.6	154	n.d.
TOH 2	0.0-0.5	269	0.6	75	10
	0.5-1.0	506	0.4	181	100
	1-2	797	0.6	268	235
	2-3	n.d.	n.d.	371	321
Ulva 4	0.0-0.5	329	0.6	284	7
	0.5-1.0	647	n.d.	344	28
	1-2	1040	n.d.	502	324
	2-3	1331	n.d.	506	372
	3-5	1622	n.d.	450	404
Ulva 1	0.0-0.5	459	1.1	163	282
	0.5-1.0	765	0.6	151	372
	1-2	1105	n.d.	151	382
	2-3	858	n.d.	170	387
	3-5	444	n.d.	118	351
TOH 148 A	0.0-0.5	145	0.8	11	13
	0.5-1.0	254	n.d.	35	35
	1-2	658	n.d.	79	282
	2-3	1163	n.d.	141	342
TOH 148 B	0.0-0.5	120	3.3	13	7
	0.5-1.0	185	0.5	50	3
	1-2	297	n.d.	87	35
	2-3	426	n.d.	87	185
Ulva 11A	0.0-0.5	195	0.3	33	139
	0.5-1.0	300	n.d.	48	33
	1-2	399	n.d.	68	220
	2-3	708	n.d.	118	344
	3-5	481	n.d.	180	396
	8-10	1102	n.d.	109	392

Near-surface (0.0-0.5 cm) ammonium concentrations ranged from 120-459 μmol L^{-1}, with sharp ammonium gradients observed in the top 2-3 cm. Near surface SRP concentrations ranged from 11-284 μmol L^{-1}; such surficial SRP concentrations result in the high observed effluxes. The low NO_{2+3}^- concentrations are typical in highly reducing sediments, with any overlying water NO_{2+3}^- rapidly reduced via denitrification or dissimilatory reduction to ammonium (An and Gardner 2002).

Hydrogen sulfide was immediately obvious during coring operations; the pore water concentrations here are high, but not as high as expected in such biogeochemically active sediments. The dark coloration of sediments suggest formation of iron sulfide minerals (Berner et al. 1979), likely creating a limit to the concentrations hydrogen sulfide in pore water (Morse et al. 1987).

Discussion

Data Inter-Relationships

When considering benthic fluxes and pore water chemistry, examining the relationship between key biogeochemical fluxes can provide insight into how nutrients are retained and released by sediments. The simplest view of these relationships comes from recognizing that in ecosystems driven by algal uptake and algal decomposition, there is a characteristic ratio of carbon, nitrogen and phosphorus, the so-called Redfield ratio (Redfield et al. 1963, Froelich et al. 1979, Nixon 1981):

Algae consists of 106C:16N:1P

With aerobic respiration, we generally observed 1 O_2 = 1 CO_2, so oxygen can serve as a proxy to C with some important caveats discussed below. The flux of all species of N measured in this study can be summed to make an estimate of the N remineralization rate (i.e. ΣN = rates of flux of NH_4^+, NO_{2+3}^- and N_2-N) and this can be compared to the fluxes of oxygen (Figure 7). The Redfield composition line is plotted; data that fall above the line are have N in excess. About 7 of 18 cores have near-Redfield fluxes, while the remainder or N-enriched. Similarly, SRP fluxes above the Redfield flux ratio line are P-enriched and those below are indicative of P retention. At higher rates of respiration/O_2 uptake, a number of very high SRP effluxes are observed.

The P-enrichment in the fluxes may arise, in part, from the desorption of Fe-bound P as iron oxides are reduced to iron sulfides (Krom and Berner 1981); this P release occurs more commonly as systems become eutrophic (Lehtoranta et al. 2009). High N releases are more difficult to explain; at least part of the discrepancy from Redfield fluxes comes from the lack of reoxidation of reduced species such as reduced iron, reduced sulfur and methane, resulting in the short or long-term “storage of electrons” that results in an imbalance between C respiration and oxygen uptake. The converse is also true; in summer, coastal sediments can build up high concentrations of iron sulfide minerals that can be reoxidized in fall and winter (Cornwell and Sampou 1995) resulting in excessive oxygen uptake relative to metabolism.

The imbalance between nutrient fluxes and oxygen uptake may represent a transient state in the estuary. Our visual observation of cores suggest that recent inputs of macroalgae may be driving the sediment metabolic rates; the literature documents large effects of viable and decomposing macroalgae on sediment N cycling processes and the productivity of benthic microalgae (Trimmer et al. 2000, Tyler et al. 2001, Garcia-Robledo et al. 2008).

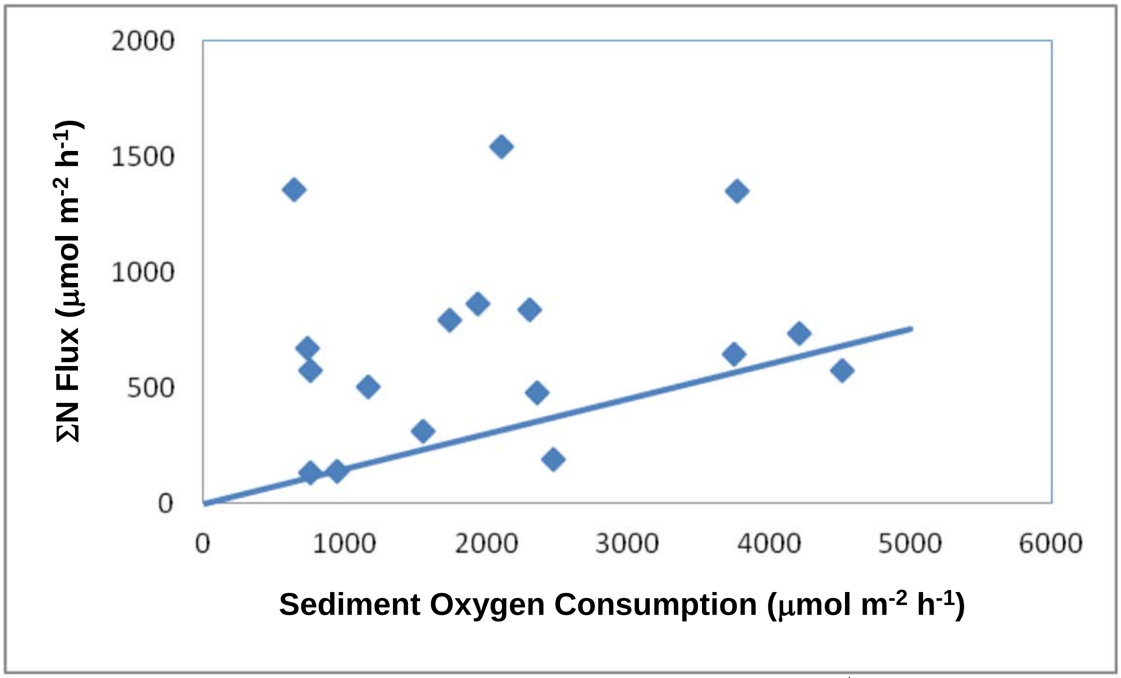

Figure 7. Plot of oxygen consumption versus the sum of N_2, NH_4^+ and NO_{2+3}^- fluxes. The line is the 106:16 line for Redfield stoichiometry.

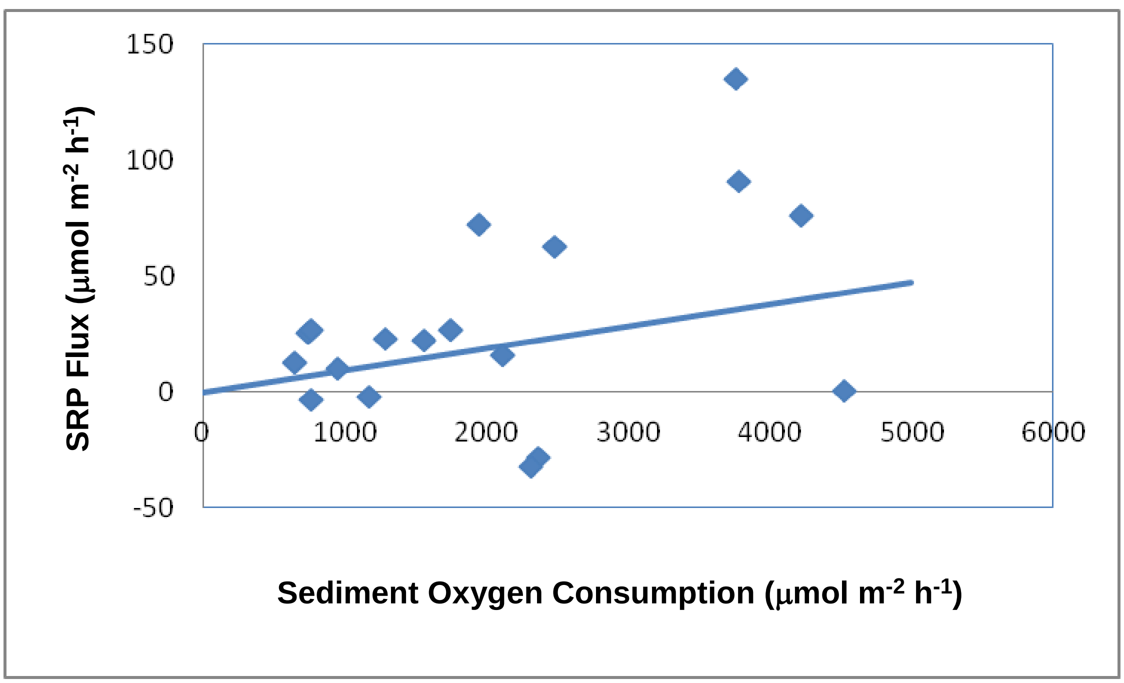

Figure 8. Plot of oxygen consumption versus the flux of soluble reactive P. The line is the 106:1 ratio for Redfield stoichiometry.

Comparison to Other Coastal New York Environments

Data from fall 2005 in Jamaica Bay, New York City is used as a point of comparison of rates observed in Hempstead Bay. The Jamaica Bay data were collected as part of a joint project with Battelle and represent a small subset of the data generated in that project. The field and laboratory procedures were identical to this study. Box plots are a good way to present the central tendencies of data sets (Figure 9); in this case both data sets consisted of flux rates from18 sediment cores.

Relative to Jamaica Bay, Hempstead Bay consumes less oxygen, produces less N_2-N from denitrification, has higher NH_4^+ effluxes, and has lower NO_{2+3}^- fluxes, and and has higher median SRP effluxes (but fewer high SRP flux rates). Many of the Jamaica Bay rates are driven by high population of amphipods at the sediment surface; that animal community promoted nitrification that supplied the substrate for denitrification. Macrofauna communities like the tube dwelling benthic amphipod Ampelisca often found in Jamaica Bay were not observed at any the cores collected in this study.

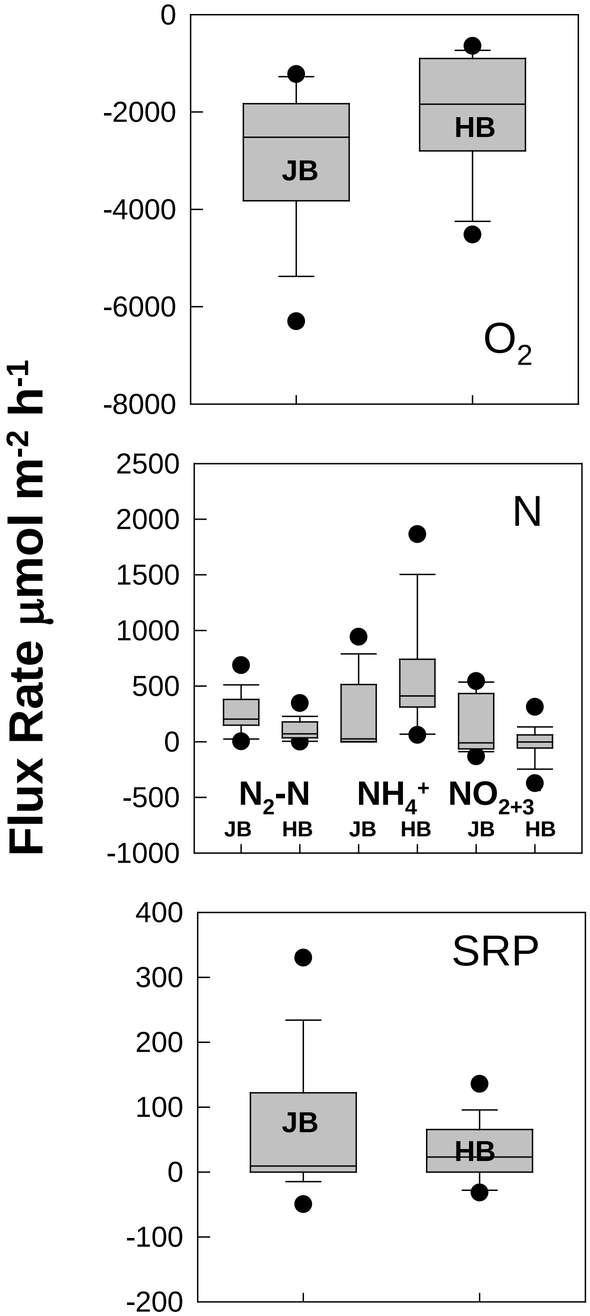

Figure 9. Box plots of sediment-water exchange data from Jamaica Bay in September 2005 and Hempstead Bay in October 2012. The horizontal line in the box is the median data, the box section above and below are the

Conclusions

This study represents an excellent first examination of the rates of oxygen and nutrient flux in Hempstead Bay. We would characterize these sediments as impacted by eutrophication, with exceedingly high ammonium effluxes at a number of sites. The dying and decomposing macroalgae suggest that we have collected cores at a transition period, when shorter days and cooler temperatures may have resulted in decreases in macroalgae production and increased rates of sediment metabolism. The rapid decomposition of macroalgae drives high rates of anaerobic metabolism causing very reducing conditions in near surface sediments even with near saturated bottom water oxygen. Reducing conditions in near surface sediments promote N and P recycling, inhibit coupled nitrification/denitrification, and drive the sediments to a state that is inhospitable to most macrofauna.

Two key questions remain:

- What is the role of illumination in these sediments? Observations using the same experimental setup, but with both dark and illuminated incubations, have shown that dark-only experiments can overestimate daily ammonium fluxes.
- What is the seasonality of benthic fluxes and do they help support water column production and biomass earlier in the season?

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APPENDIX H: REGARDING TIDAL WETLAND TRENDS IN WESTERN HEMPSTEAD BAY

Nutrient Assessment and Management in Shallow Coastal Embayments in New York and New Jersey

Regarding Tidal Wetland Trends in Western Hempstead Bay

Final Report

Prepared By

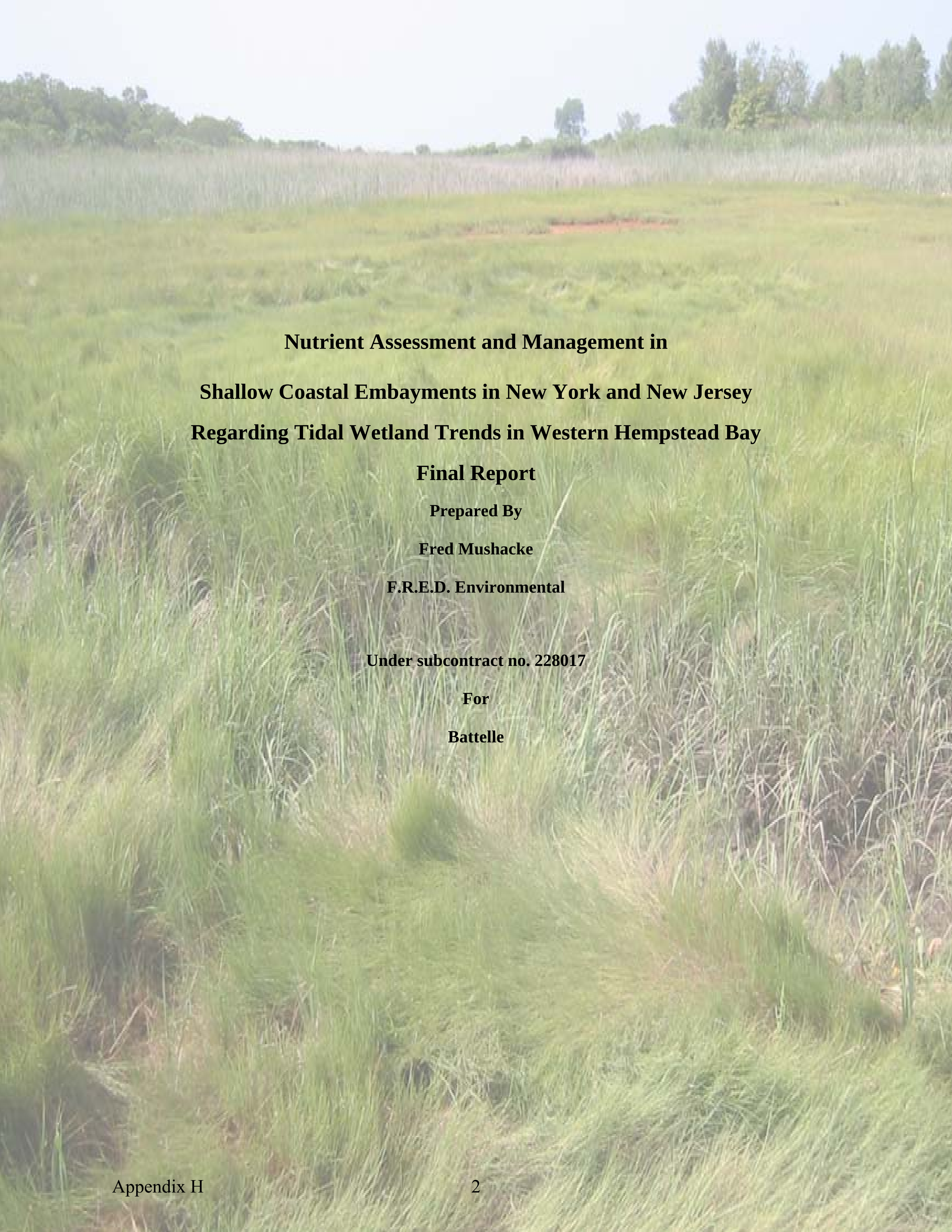
Fred Mushacke

F.R.E.D. Environmental

Under subcontract no. 228017

For

Battelle



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Special thanks to The Nature Conservancy on Long Island, for allowing use of the 2008 south shore infrared aerial imagery to conduct tidal wetland trends of a portion of the Hempstead Bay area.

Introduction

Tidal wetlands (TW) comprise one of the most productive, naturally occurring ecosystems on the northeast coast of the United States (Teal, 1969, Bertness, 1999). They produce up to four tons of organic material per acre per year (O'Conner and Terry, 1972, Udell et al., 1969). These tidal wetlands are vitally important as habitat and primary productivity to the surrounding ecosystems (Weinberg 2010). Approximately 60% of the finfish and shellfish depend on the tidal wetlands productivity in addition to wildlife benefits (Harmon, 1975).

Since their colonization, after the retreat of the last glacier some 20,000 years ago, tidal wetlands have evolved and migrated. They competed with invasive floral and faunal species throughout this period (Bertness, 1999) and established a roothold well landward of their present location. During the period 1954-1964 over 12,000 acres of vegetated tidal wetlands were lost to fill and build activities. In Nassau County 4,635 acres were lost to various filling and building activities; housing comprised the greatest, 1,885 acres, and miscellaneous fill was second, destroying 984 acres (Dept. of the Interior, 1965).

The passing of the Tidal Wetlands Act in 1973 and the subsequent Land Use Regulations, 6NYCRR Part 661, in 1977 stemmed the tide of filling and building activities (Fallon and Mushacke 1996, Mushacke and Picard, 2002). Recent trends, however, indicate that tidal wetlands loss within New York's Marine District (Tappan Zee Bridge south to Staten Island; east to Orient and Montauk points) are still occurring, approximately 1,500 acres since the 1974 inventory. Although losses are for the most part slower, there is no apparent smoking gun. Causes are suspected to be synergistic and/or discrete, anthropogenic and/or natural (Hartig, 2002, Mushacke, 2010).

Causes can be historic or contemporary and while not direct can have the same ultimate devastating effects. Some of the potential causes can be ice scour, wind and wave energy, boat wake, past and present dredging, nutrient loading, drought, sediment budget disruption and invasive floral and faunal species.

Under subcontract no. 228017, associated with the **Nutrient Assessment and Management in Shallow Coastal Embayments in New York and New Jersey** project, tidal wetlands were identified and quantified in the western portion of the Hempstead Bay complex from Meadowbrook Parkway west to the Nassau/Queens border (the AOI), (Figure 1). The assessment included all tidal wetlands found within the mainland canals and parks, the bay islands and barrier beach wetlands from Silver Point east to West End, Jones Beach (the southern clover leaf of Meadowbrook Pkwy). Anthropogenic impacts that are implicated, but not limited to, vegetative tidal wetlands loss are: high density upland development, impervious structures, bulkheading, dredging, past filling of tidal wetlands, nutrient loading from urban and storm runoff (NYS 2010 Section 303(d)) and demographic expansion within the watershed. The latter the apparent primary forcing behind these impacts (Deegn,2002, Pennings et al. 2002, Silliman and Bertness, 2004). In order to accommodate the increasing population and its required housing and infrastructural response, roads, canals, public transportation and sewerage treatment plants were constructed. This effort led to the immediate loss of tidal wetlands due to the filling and building mentality of the 50s and 60s and subsequent collateral impacts have transcended time to present day.

Material and Methods

Historic aerial infrared (IR) imagery acquired in 1973 was quantitatively and qualitatively evaluated using geographical information systems software (GIS) and compared to 2008 aerial imagery acquired by JW Sewall for The Nature Conservancy under a separate contract. Both sets of images were acquired within three hours of low tide at the height of the growing season, August through October, and at an altitude of 6,000 feet with a resultant image scale of 1:12,000. The 1974 imagery was scanned on an Epson scanner at 600 dpi and the 2008 imagery was scanned on a Vexcell 4000HT scanner at a 2400 dpi. The imagery sets were geo-referenced in Universal Transverse Mercator (UTM), North American Datum (NAD) 1983 coordinate system and mosaicked. “Heads up” digitizing was used; this is a method whereby features are identified by tracing a mouse over discrete image signatures forming vector polygons and creating a shapefile. Using ESRI's ArcGIS 9.3.1 software, a shapefile of biotic and abiotic features for each image set was delineated. The GIS software provided the ability to create spatial features via polygon digitization that can then be evaluated quantitatively in an attribute table. Biotic features delineated and identified include: intertidal marsh, (IM) tidally flooded on a daily basis, the primary vegetative species is *Spartina alterniflora*; High marsh, (HM) generally landward of IM, flooded during storms, new and full moon tides; primary species include, *Spartina patens, Distichlis spicata*. Additional species can also include seaside lavender, *Limonium carolinianum* and short form *S. alterniflora*. Upper limits of this zone often include black grass, *Juncus gerardi,* chairmaker's rush, *Scirpus sp*; marsh elder, *Iva frutescens*; and groundsel bush, *Baccharis halimifolia*.and Phragmites (PH) areas containing a dominance of the ditch reed *Phragmites sp*.

Abiotic features include: ponds (PD) deep marsh depression, generally always filled with water, un-vegetated with marsh species, may contain floating or rooted macro algal species; Pannes, (PN) shallow depressions in HM and/or IM, may hold water centimeters in depth, usually devoid of TW vegetation, may contain pioneer species *Salicornia sp*. Channels (CH) naturally occurring, meandering cuts through the marsh, whose bottoms may be exposed at low tide, may contain macro algal species, no HM/IM species and mosquito ditches (MD) Ditches dug to drain HM and /or IM for the prevention/reduction of mosquitoes. The abiotic features and Phragmites have been added to the TW inventory nomenclature because they were noted during previous trends to be persisting and /or expanding on the marshscape and presumed to have a significant impact on vegetative extent and associated intrinsic values and benefits provided by tidal wetlands; including primary food production, wildlife habitat, storm and flood control, absorption of silts and organic material, filtration of pollutants and esthetic values.

The attribute table, an integral part of the GIS, contains the type of features identified (TW_CAT), area of the marsh feature in (acres) and square meters (area) and a count (frequency) of the number of discrete polygons within each class. The tidal wetland categories and 2-letter labeling nomenclature are consistent with the inventory maps prepared by Earth Satellite Corp. and Mark Hurd Aerial Surveys and defined vegetatively in the Land Use Regulations 6NYCRR Part 661 section 661.4 (hh) (1-6). The methodology and vegetative classification used is described in the New York Wetlands Inventory, Final Report, sections 3.0 (3.1-3.5), Wetland Analysis: Techniques and Procedures (Martin et al. 1975) commonly called "the mapping conventions". The exception to these classes is the formerly connected (FC) category. In the past this category has led to confusion among managers, permit applicants and other parties of interest. The FC category has been re-identified by the signature of the vegetation that appears on the imagery or species found on the marsh. If a site was previously classified as FC in 1974, it will be noted in the Comments field of the attribute table. Typically this zone contains high marsh vegetation and can be infiltrated by Common reed, *Phragmites sp*. (Mushacke and Picard 2002, Mushacke, 2010).

A one-week site visit to the AOI was conducted during low tide in August 2010 to ground truth the remotely sensed images. Island marshes were inspected and access was gained by boat supplied by Jim Brown, Conservation Biologist III, Dept. of Conservation and Waterways, Hempstead. Sites marked on the imagery were inspected for signature validation; other sites were checked for class accuracy and qualitative density determination. Polygons were updated and assessed on the imagery and in the attribute table.

Quantitative results were placed in an Excel spreadsheet for trends analysis. The spreadsheet is composed of fields containing the two letter marsh categories (TW Type), 1974 and 2008 Acres, frequency (number of discrete polygons of each wetland type), total acre change, percent change, rate of change (acres/year), total frequency change, percent frequency change and rate of frequency change.

During groundtruthing, qualitative densities were taken at 10 IM tidal wetland island and mainland sites throughout the AOI. A 6" x 6" (15.24cm x15.24cm) PVC quadrat was tossed twice into the sample area and basal stalks were counted. In the IM two types of marsh were chosen based on their color signature. Typically, IM marsh dominated by *Spartina alterniflora* is rendered in colors on the IR imagery ranging from bright pink to reddish-brown to mottled brown and green (Martin et al. 1975). Predominate color signature on the IR imagery, in the AOI, was found to be as described by Martin. There were however additional zones with black signatures along the perimeter of the marshes and among tussocks of fragmenting IM, bordering and at the terminus of channels and mosquito ditches. These signatures depicted low density IM zones associated with organic substrates and macro algal species *Fucus sp*. and *Ascophyllum sp.*

Discussion

The AOI is dominated by low marsh mixed with high marsh zones at the landward boundaries and along mosquito ditches and channels. Marsh islands and mainland marshes contained intermittent dredge spoil containment areas that have contributed to marsh extent and Phragmites infiltration. In 1974 vegetative extent of (IM, HM and FM) within the AOI comprised 4459.66 acres, (Table 1, Figure2). In 2008 vegetative totals were 3,730.4 acres, a net loss of 729.26 acres and a 16.35% decrease (Table 2, Figure 3). The primary area of loss was the IM. In 1974 IM encompassed 3781.6 acres, and by 2008 acreage had decreased to 3,106.04 acres a 675.56 acre, 17.86% loss at a rate of 19.87 acres/year. Fresh marsh lost 100% of its 1.45 acres to Phragmites invasion. Overall, high marsh lost acreage as well. In 1974 HM consisted of 676.6 acres and in 2008 HM acreage dropped slightly to 624.36 acres, a loss of 52.24 acres, 7.72% even with a 20 acre contribution from dredge spoil island subsidence and re-vegetation (Table 3, Figure 4). From 1974 to 2008 polygon frequency increased except CH. Where frequency drops this may be an indicator of coalescence or loss due to vegetative shift or physical loss. When frequency increases it's generally an indicator of fragmentation. The Tidal Wetlands Act in1973 and the Land Use Regulations in 1977 presumably stopped the placement of spoils on additional wetland sites. The decrease in DS acres and slight increase in frequency indicates fragmentation of the DS site into (bona fide) vegetated IM and HM. The greatest increase in frequency was marsh pannes with a 6,619 gain (279.4%). In 2008 pannes ranged in size from 0.0001 – 1.7 acres with an average size of 0.024 acres, while 1974 PN had a maximum acreage of 2.14 acres. An examination of this phenomenon revealed that there was re-vegetation of the panne (Figure 5). In 1974 and 2008 high marsh was the vegetative class with the highest frequency; by 2008 an increase of 4,383 polygons. The majority of these HM polygons appeared along mosquito ditches and marsh channels. The former is possibly a result of ditch spoil placement, the latter possible wrack buildup (figure 6).

Eastern and western portions of the AOI were compared; Long Beach Rd. in Island Park/Barnum Island was the dividing point (Figure 7). In the eastern portion IM and HM lost acreage 302.65 and151.9 acres respectively. Frequency in both zones increased however, 237.9 % and 108.82 % respectively (Table 4; Figures 8 and 9). *Phragmites sp*. had the greatest vegetative change, an increase of 96.2 acres, but the lowest total frequency 112. Island and mainland marshes that experienced spoil placement and/or construction disturbance saw an invasion. Phragmites acreage may have been low in 1974 because many of the areas it invaded were recent post construction sites and Phragmites had not gotten a root hold and/ or existing plant colonies had not expanded enough to be identified on the imagery.

Marsh pannes increased from 52.79 to 134.39 acres respectively, and also had the highest frequency of abiotic features, 4,077. Panne frequency rate of change increased 3.7 times faster than IM and four times faster than HM (Table 4; Figures 8 and 9) indicating panne persistence and increase. PN acreage in the eastern portion (134.39 acres) was approximately 63% greater and had a 62 % higher frequency than PN in the western portion. Dredge spoil sites were the only areas that experienced a reduction in frequency at 13.33%. Cessation of marsh island spoiling, subsidence of spoil and re-vegetation by marsh species is apparent. Mosquito ditch area increased seven-fold from 5.5 acres to 35.91 acres, while CH increased nearly two fold; both increases a product of edge collapse and expansion. Pond acreage remained fairly stable while frequency increased.

In the western portion of the AOI, IM lost 379.91 acres while HM gained 99.65 acres. Total frequency increased for both vegetative features, 911 and 3,384 respectively (Table 5; Figures 10 and 11). *Phragmites sp* gained 67.57 acres as well as in number of occurrences 133. PH invasion completely invaded the only FM site 1.45 acres located in the upper reaches of the Mill River. Dredge spoil lost

82.03 acres, while gaining in number of occurrences 10, a result of fragmentation due to re-vegetation by IM/HM, *Phragmites sp* and formation/expansion of drainage ditches. Panne frequency gained 2,542 polygons and Frequency Rate of Change 74.76 acres/year was the highest of the abiotic features. Ponds decreased in this portion of the AOI 4.33 acres and channel acreage increased 61.46 acres. This state of environmental flux throughout the AOI is favoring the increase of abiotic features. While HM had an increase of 99.65 acres, Phragmites, PN, MD and CH all increased not only in acreage but in frequency as well. Part of the HM gain, nearly half, came from the contribution of the DS; the balance occurring along the banks of marsh channels and mosquito ditches and the majority occurring on the marshes in the western portion of the AOI, these generally narrow areas of HM are apparent higher elevations possibly due to historic ditch spoiling and further buildup, due to algal and wrack deposition.

Hicks Beach in the city of Lawrence (Figure 1, name in red) the largest IM-dominated island, experienced typical loss scenarios seen throughout the Hempstead Bay study area. Losses consisted of expansion of MD and CH (figure 12); perimeter loss along bay edge (Figure 13); and expansion/formation of marsh pannes and re-vegetation of internal PN with S. alterniflora and an HM shift to IM (Figure 14).

One of the largest losses of marsh vegetation was experienced on Alder Island found in the southeastern corner of the study area (Figure 1, name in red). In 1974, the eastern-most side consisted of 12.34 acres of low marsh; by 2008 11.34 acres had disappeared. The subsequent loss is now threatening the interior portions of the high and low marsh and has begun to take its toll on the southern marsh as well (Figure 15). Another area of loss involving not only island marshes but mainland ones as well are the Lido Beach marsh and islands directly north. Approximately 10.7 acres of perimeter marsh were lost (Figure 16). There were a number of small satellite island marshes ranging in size from just over an acre 1.17 to 0.007 acre that were completely or nearly completely lost between 1974 and 2008. Eight of these islands were found in the western portion of the AOI and seven were found in the eastern AOI. These marshes may have broken off from larger nearby adjacent marshes. One marsh island is the exception. This island is found in the approximate middle of Middle Bay (Figure 17). A 1903 USGS map displays the island clearly and amazingly, in 1974 the island was 1.1 acres indicating little change. In 2008, however the marsh had fragmented into two islands of 0.05 and 0.02 acre, respectively. Larger island marshes that are fragmenting and may be in danger of continued and potentially complete loss include Nums Island and island marshes to the southwest and southeast of Black Bank Hassock marsh (Figure 1, Table 6). The latter two are confronted by two impacting conditions. The southern portions of the islands border Reynolds Channel and are subject to erosive conditions of boat wake, tidal currents and ice shearing. Additionally, the historic indiscriminate dredging of the channel may have compromised the basal infrastructure of the marshes (Figures 18 and 19). The second condition is the multiple acre algal mats partially envaginating the island's low marsh vegetation along the northern perimeter and within the marsh proper. Algal mat acreage adjacent and within the southwest island comprises approximately 8.87 acres, while 2.58 acres infiltrate the southeast island and 2.12 acres lie just offshore to the northwest (Figure 20). The Black Bank Hassock southwest marsh island lost 33.21% and the southeast marsh lost 43.2% of its IM vegetation, frequency increased as well, indicating fragmentation. Approximately one acre of pans in 1974 have partially re-vegetated, although this is just a fraction of the 3.19 acres lost. Additionally, there are two channels that have developed connecting the small elongated ponded areas within the 1974 pan to the main water body. It's a matter of approximately six meters before the channelized ponds break through and fragments the island into three portions (Figure 18). A similar condition occurred on the SE Black Bank Hassock marsh (Figure 19). The marsh perimeter, however, has eroded, exposing a portion of the re-vegetated pan and that is beginning to erode as well. This island marsh has lost more than twice the IM of the southwest area island (7.53 acres) and 2.01 acres of HM as well.

While the northern Nums Marsh island complex is contending with algal mat concentrations (Figure 21) it is surrounded by SM which tends to moderate wind, wave and wake energy. Additionally, the algal mats and shallow SM may slow boaters, further reducing the negative effects of wake erosion. The algal mats, however, appear to be concentrating in this zone and may be having a negative effect. Nums Marsh (Table 6) lost 59.39 % of the IM and 68.66% of its HM.

Marsh island loss in the eastern portion of the AOI for IM is 238.16 acres and 75.84 acres for HM, 15.12% and 26% respectively; with a rate of 7acres and 2.23 acres loss per year for IM and HM respectively. Low marsh lost acres 319.15 acres in the western portion of the AOI as well, while HM gained 93.78acres, DS subsidence contributing to this gain. While the eastern AOI is losing IM at a rate of 7 acres a year, the western islands are losing IM at a rate of 9.38 acres per year (Table 7 and 8). These loss rates are considerably less (32% for eastern and 43% for western islands) than those experienced by Jamaica Bay islands which are losing overall island IM at a rate of 25.31 acres (GNRA, 2007).

Even though IM and HM are experiencing loss overall, the largest panne in 1974 (2.15 acres) within the study area, which was found on Cedar Island, has shown evidence of re-vegetation with S. alterniflora. The panne is now 0.2 acre (Figure 5). Another example is the re-vegetation of portions of the dredged spoil islands (Figure 22). In this example, HM species re-vegetated spoil out-wash areas of Pearsalls Hassock.

Densities of marsh grass *S. alterniflora* (IM) were taken at a total of 10 stations on mainland and island marshes, (Table 9, Figure 23; Dark (near black) IM signatures along the perimeter and within the marsh exhibited an approximate 38% decrease in stalk density compared to pink to dark red areas signifying typical stands of *S. alterniflora*. Average overall stalk density of IMs sampled equaled 667/m². Near black signatures were associated with low IM densities, organic sediments in the SM and macro algae *Fucus sp.* and *Ascophyllum sp.* and had an average density of 329stalks/ m². *Ulva sp.* was not observed at these sites during sampling. The near black to dark brown signature symbolizing low density *S. alterniflora* was then mapped throughout the AOI to determine area and frequency. Overall these low density zones encompassed 13.23 acres with a frequency of 218. Low density IM comprised only 0.4% of the overall low marsh extent and had a 7% occurrence within the overall study area (Figure 24). The largest area contained 0.9 acre and was found in the eastern portion of the study area on Alder Island. This is the same island that lost approximately 11.34 acres of TW from 1974 to 2008 (Figures 1 and 15). The AOI was divided using the same demarcation as previously mentioned. The eastern portion of the AOI contained the greater extent of low density IM, 12.43 acres and the higher frequency 200. The western portion of the AOI contained 0.1 acre with a frequency of 10 (Figures 25 and 26). The frequency of the western low density was too small to graph. The low density marshes were generally associated with marsh edge, whether it was outer marsh perimeter, along panne edges, or at the terminus of and or along marsh channels or mosquito ditches (Figure 27).

Floating algal mats (AM) (*Ulva sp*.) were observed during the August 2010 field trip. The mats were so thick in spots that access to the shore or marsh edge was nearly impossible without cleaning the prop several times. These mats and others were also observed on the 2008 IR imagery. The mats exhibited spectral red/deep purple to light pink signatures on the imagery. These signatures were mapped to estimate algal mat distribution and frequency at least on the day the imagery was acquired, August 26, 2008. Only apparent living macro-algal mats were mapped. White (desiccated) light gray (mud flats) and black (benthic) signatures were not included in the estimate. Overall extent of the mats equaled 298.36 acres (Figure 28 and 29). The higher acreage (246.23) and frequency (401) of these mats occurred in the western portion of the AOI (Figures 30 and 31). Algal mats ranged in size from 0.0012 to 28.68 acres. The western portion of the AOI also contained the four largest extents of algal mat acreage; 28.68, 22.79, 10.47 and 10.18 acres. The largest algal mat, 28.66 acres, was found in the western portion of the AOI and was associated with the Nums Island complex (Figure 21). A maximum size of only 2.8 acres was found in the eastern portion. The algal mat results were added to the vegetative marsh classes to determine a percent and frequency estimate of primary producers for the AOI (Table 9). Overall, algal mats comprise 7% and have the third highest frequency (665) of the autotrophic cover types. When compared with the eastern and western portions of the AOI, algal mats comprised the third highest percent and frequency. The mats were not found in open water but were associated with the SM, along sandy shores, marsh banks and in marsh channels. Only approximately 1.2 acres were found in marsh pannes and this occurred in the western portion of the AOI. Algal mats are associated with vegetative suffocation, the weight crushing stalks and acting like a mulch preventing growth. The mat’s presence on the marsh increases marsh peat temperature and prevents of O_2 absorption (Hartig and Gornitz, 2001). Additionally, the macro-algae may be carpeting the bottom preventing re-suspension of sediments

(GNRA, 2007). Breakdown and decay may contribute to anoxic and hypoxic conditions (Swanson et al. 2009).

Tidal wetlands and the habitat they provide are important resources and sentinels of overall ecosystem health and integrity (USEPA, 2000). Contributing factors within the AOI acting independently or synergistically may play a role in vegetative shift, loss, invasion, increase and expansion of biotic and abiotic features; to date there is no "smoking gun". Natural stressors associated with marsh trends include wind and wave energy, ice shear along banks and on the surface of the marsh (marsh rafting) (Bretness, 1992) wrack smothering (Deegan, 2002, Fischer et al., 2000, Brewer et al, 1998), ice scour (Erwin et al, 2004 and Ewanchuk and Bertness, 2004) and sea level rise (SLR) associated with crustal adjustment (Hartig et al. 2002). Anthropogenic stressors include cultural nutrient enrichment (Deegan, 2002), sediment budget disruption, (Fallon and Mushacke, 1995 and Mushacke, 2002), shoreline development and spoil placement. The former increasing nitrogen supply due to point and non-point source pollution which results in an increase and /or shift in vegetative species composition leading to a competitive imbalance to those vegetative species limited by nitrogen (Silliman and Bertness, 2004; Pennings et al. 2002). In nutrient rich systems, primary production and biomass increase (Paerl, 2006). The subsequent decay of excess plant organic matter results in an increase in biological oxygen demand, and can lead to hypoxia and anoxia (Deegan, 2002). This condition may exacerbate the accumulation of organic material, wrack formation and deposition on the marsh landscape forming pannes (Shumway and Bertness, 1994; Fischer, 2000). Marsh pannes had the greatest acreage and frequency change of abiotic features implicating this stressor. Pannes in the AOI were also found between HM strands and MD grid ditch systems, suggesting that the slightly higher elevation prevents or slows drainage encouraging waterlogging, panne formation and expansion (Ewanchuk and Bertness, 2004; Kennish, 2001). Peat structural integrity of the IM may be affected by nitrogen loading as well. More above ground than below ground biomass can potentially lead to a susceptibility of low marsh edge erosion; the lower root biomass reducing peat cohesion, sediment retention and entrapment (Wigand, 2007, Davey et al in press). Sediment budget disruption through dredging, bulkheading and inlet jetty construction may alter sediment accretion rates to the low marsh potentially creating susceptibility to SLR and a vegetative shift to SM. Dredge spoil placement, shoreline development, construction and nutrient loading acting alone or in concert, can also encourage the growth and infiltration of *Phragmites sp*. (Silliman and Bertness, 2004; Bertness et al 2002). Algal mats floating or deposited on marsh banks and the shore can have detrimental effects on marsh and benthic floral and faunal species diversity and richness (Mackenzie Jr., 2005).

Conclusions

Wholistically, the abiotic features are gaining ground in occurrence and acres (Table 4, Figure 8). The balance of vegetative loss occurring through perimeter loss (figures 13 and 15). While the western section gained 99.65 acres of HM, the eastern section lost 137.92 acres. The subsidence and re-vegetation of the DS sites tended to temper the loss of IM and HM areas. This phenomenon may continue, but there is considerable competition with Phragmites and surrounding elevations, although given SLR, marsh vegetative species may for a time begin vegetating the higher elevations and out competing Phragmites and terrestrial species. Additional tempering occurred by the re-vegetation of marsh pannes, ponds and SM. For example, in 1974 the largest panne, found on Cedar Island, an island in the western portion of the AOI, was 2.14 acres (Figure 5). By 2008 this panne had re-vegetated, gaining 1.76 acres of IM. Examination of the rest of the island revealed additional areas of gain and loss fluctuations. Considerable changes have and are taking place. As the low marsh begins to fragment the rate of loss increases, making the smaller vegetated tussocks /hummocks susceptible to erosional forces, ice shearing, wind and wave energy (Allen and Pye, 1992).

This trends analysis re-fined the 1974 inventory and attempted to place the historic inventory on equal spatial and temporal footing with contemporary imagery and features. The historic inventory was conducted primarily for regulatory purposes which were highly effective in protecting and preserving wetland values and benefits. Many of the features that have been quantified were not initially delineated during previous expedited trends.

Disturbances, whether natural or anthropogenic, can greatly affect floral and faunal species composition, diversity and community organization. In order to protect these marshes and the intrinsic values and benefits, known potential anthropogenic impacts should be reduced and or eliminated. The marsh's resilience can be seen by referenced examples and if impacts are reduced further, future recovery may occur.

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Appendix 1

Tables and Figures

Figure 1. Hempstead Bay study area (AOI) Nassau/Queens border – Meadowbrook Pkwy.

Table 1. 1974 TW categories, frequencies and area.

TWType	Frequency	Acres
IM	998	3781.60
HM	1378	676.60
PH	34	48.80
FM	2	1.45
DS	35	413.30
PN	2369	123.74
MD	224	7.90
PD	66	7.95
CH	328	115.32

Figure 2. 1974 TW acres and percent cover within the full extent of the Hempstead Bay AOI.

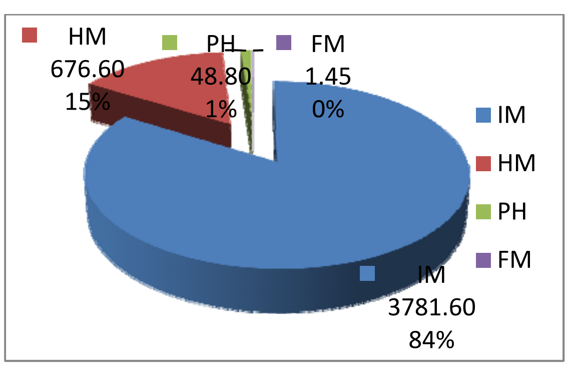


Table 2. 2008 TW categories, frequencies and area.

TWType	Frequency	Acreage
IM	3011	3106.04
HM	5761	624.36
PH	278	212.54
DS	43	310.87
PN	8988	218.60
MD	897	57.80
PD	151	4.85
CH	326	216.78

Figure 3. 2008 TW acres and percent cover within the full extent of the Hempstead Bay AOI.

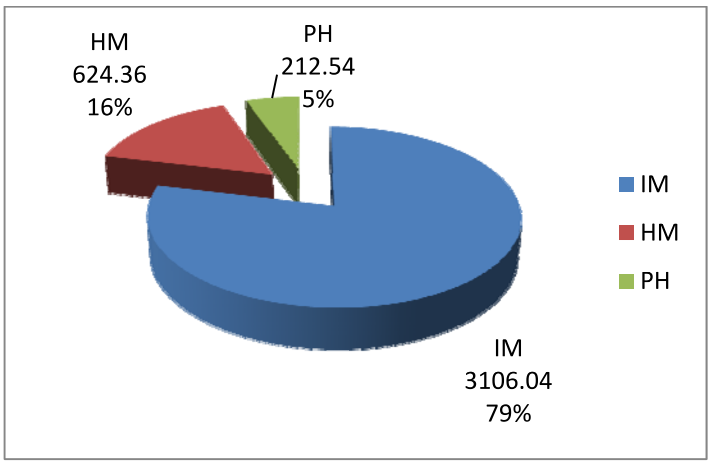


Table 3. 1974-2008 TW trends

TW Type	1974 acres	Frequency**	2008 acres	Frequency**	Total acre Change	Percent Change	Rate of change Acres/Yr	Total Frequency Change	Frequency Percent Change	Frequency Rate of Change
IM	3781.60	998	3106.04	3011	675.56	17.86	19.87	2013	201.70	59.21
HM	676.60	1378	624.36	5761	52.24	7.721	1.54	4383	318.07	128.91
PH	48.75	34	212.54	278	163.79	335.979	4.82	244	717.65	7.18
FM	1.45	2	•	•	1.45	100	0.04	•	•	•
DS	413.30	35	310.87	43	102.43	24.78	3.01	8	22.86	0.67
PN	123.70	2369	218.60	8988	94.9	76.72	2.8	6619	279.40	194.66
MD	7.90	224	57.80	897	49.9	631.65	1.47	673	300.45	19.79
PD	7.95	66	4.85	151	3.1	39	0.09	85	128.79	2.5
CH	115.32	328	216.78	326	101.46	87.98	2.98	2	0.61	0.06

Figure 4. 1974-2008 TW vegetative trends.

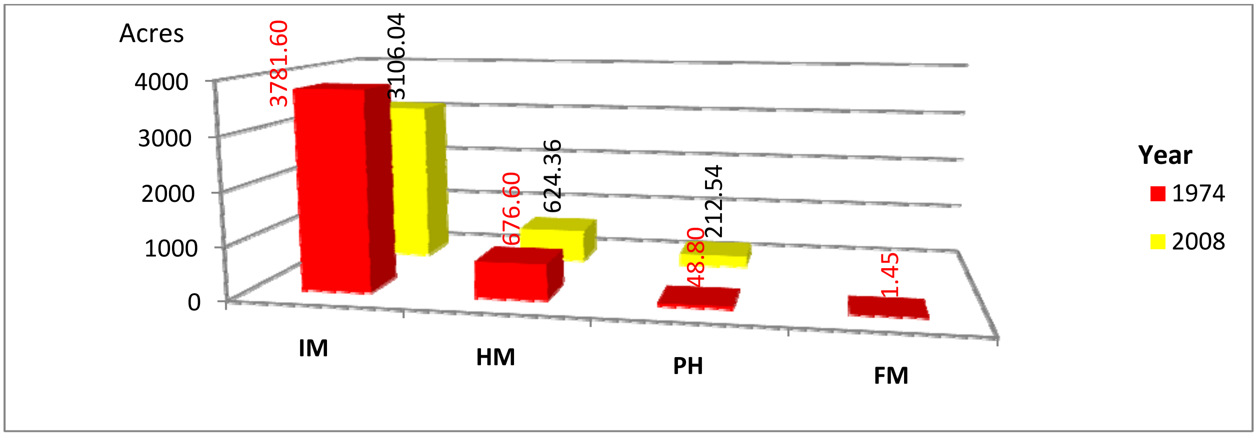


Figure 5. 1974 – 2008 largest panne re-vegetation. Red 1974 panne extent (A). 2008 imagery with 1974 panne extent overlay (B).

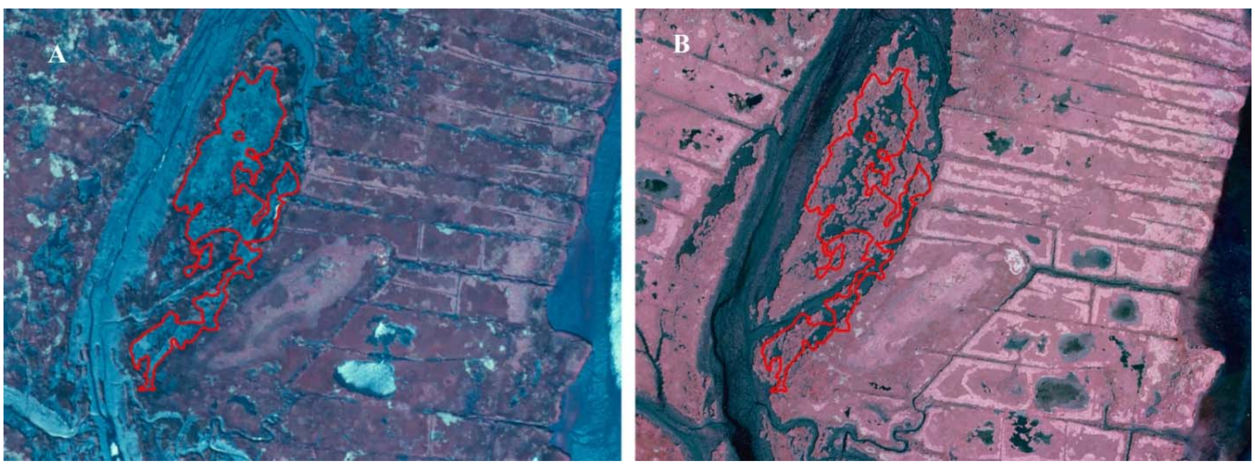


Figure 6. 2008 image SE South Green Sedge island HM (yellow) perimeter along mosquito ditch and channels.

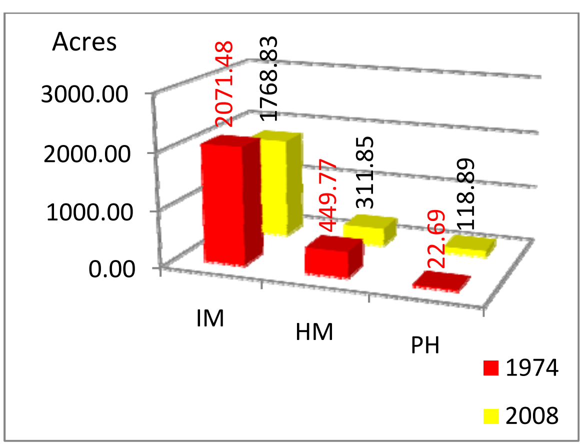
Figure 7. East and West AOIs within Hempstead Bay. Long Beach Rd. center line.

Table 4. East Hempstead Bay TW trends

TW Type	1974 acres	Frequency**	2008 acres	Frequency**	Total acre Change	Percent Change	Rate of change Acres/Yr	Total Frequency Change	Frequency Percent Change	Frequency Rate of Change acres/year
IM	2071.48	464	1768.83	1568	302.65	14.61	8.9	1104	237.931	32.47
HM	449.77	918	297.87	1917	151.9	33.773	4.47	999	108.824	29.38
PH	22.69	20	118.89	132	96.2	423.975	2.83	112	560	3.29
DS	143.09	15	122.72	13	20.37	14.24	0.6	2	13.33	0.06
PN	52.79	1221	134.39	5298	81.6	154.58	2.4	4077	333.91	119.91
MD	5.50	155	35.91	584	30.41	552.91	0.89	429	276.78	21.62
PD	3.11	37	4.34	148	1.23	39.55	0.04	111	300	3.26
CH	54.53	189	94.52	220	39.99	73.34	1.18	31	16.4	0.91

Figure 8. East Hempstead Bay biotic and abiotic TW trends.

Biotic Features **Abiotic Features**



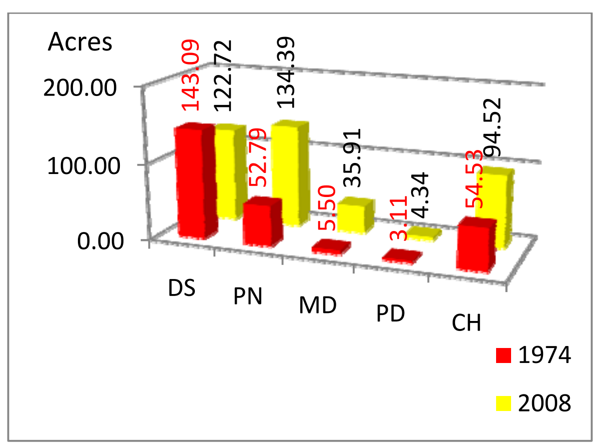


Figure 9. East Hempstead Bay biotic and abiotic frequency trends.

Biotic Features **Abiotic Features**

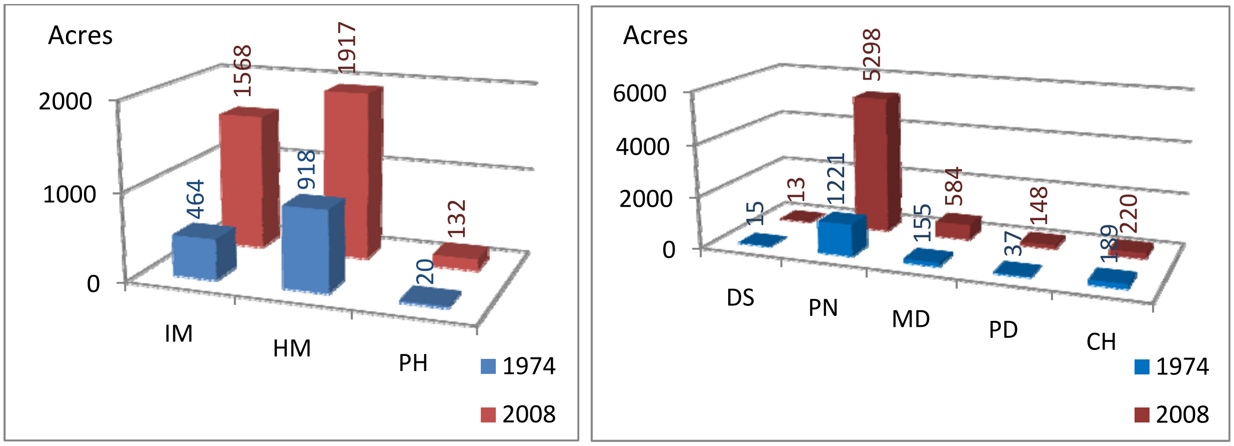


Table 5. West Hempstead Bay TW trends.

TW Type	1974 acres	Frequency**	2008 acres	Frequency**	Total acre Change	Percent Change	Rate of change Acres/Yr	Total Frequency Change	Frequency Percent Change	Frequency Rate of Change acres/year
IM	1717.13	534	1337.22	1445	379.91	22.13	11.17	911	170.6	26.8
HM	226.83	460	326.48	3844	99.65	43.93	2.90	3384	735.65	99.53
FM	1.45	2	*	*	1.45	100.00	0.04	*	*	*
PH	26.07	14	93.64	146	67.57	259.19	1.99	133	950	3.91
DS	270.18	20	188.15	30	82.03	30.36	2.41	10	50	0.29
PN	70.90	1148	84.21	3690	13.31	18.77	0.39	2542	221.43	74.76
MD	2.38	69	21.88	313	19.5	819.33	0.57	244	353.62	7.18
PD	4.83	29	0.50	3	4.33	89.65	0.13	26	89.66	0.76
CH	60.80	139	122.26	106	61.46	101.09	1.81	33	23.74	0.97

Figure 10. West Hempstead Bay vegetative TW trends.

Biotic Features **Abiotic Features**

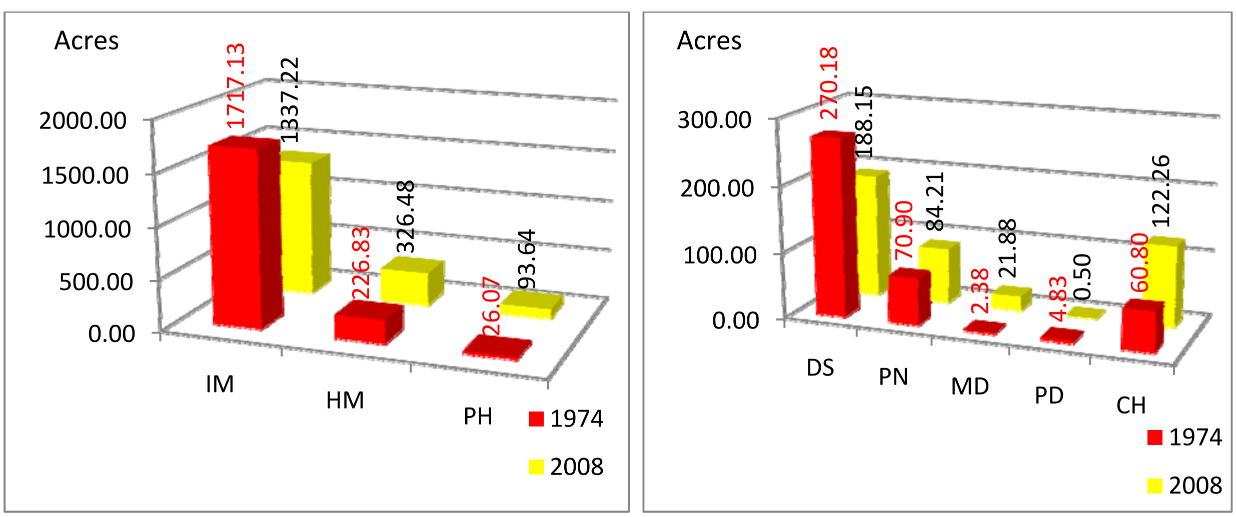


Figure 11. West Hempstead Bay frequency trends.

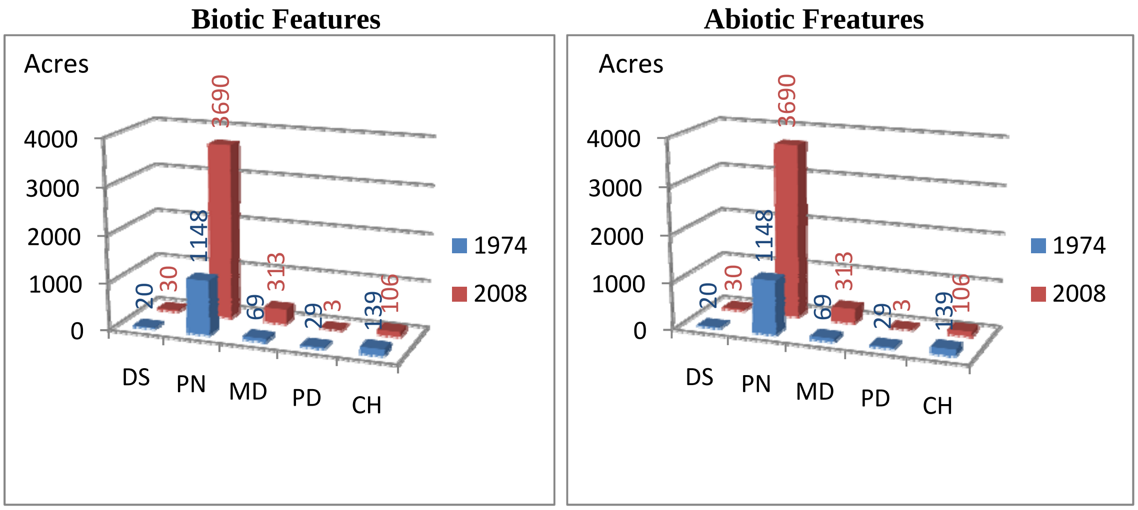


Figure 12. 1974 (left) - 2008 (right) images showing widening of channels and mosquito ditches.

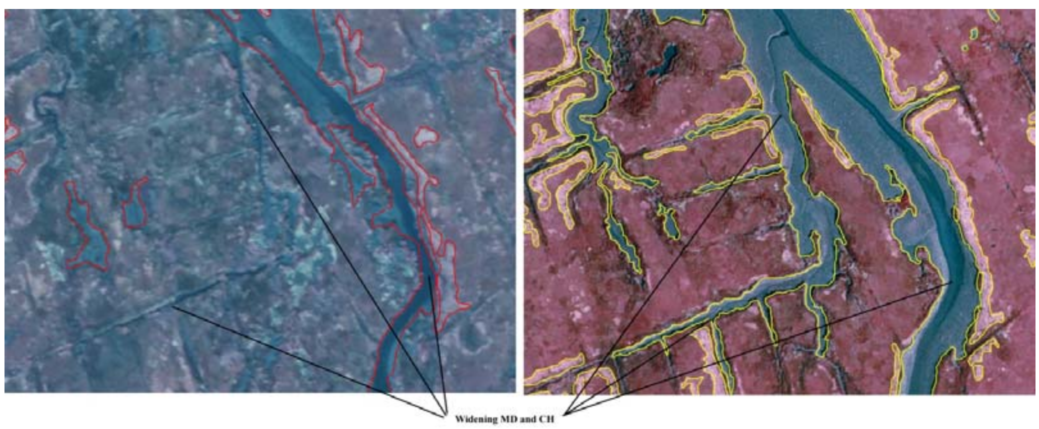


Figure 13. 1974 (left) - 2008 (right) images showing perimeter loss and CH expansion.

Figure 14. 1974 (left, red outline) - 2008 (right, black outline) images showing HM/IM shift (A), Panne formation (B) and Panne re-vegetation with *S. alterniflora* (C).

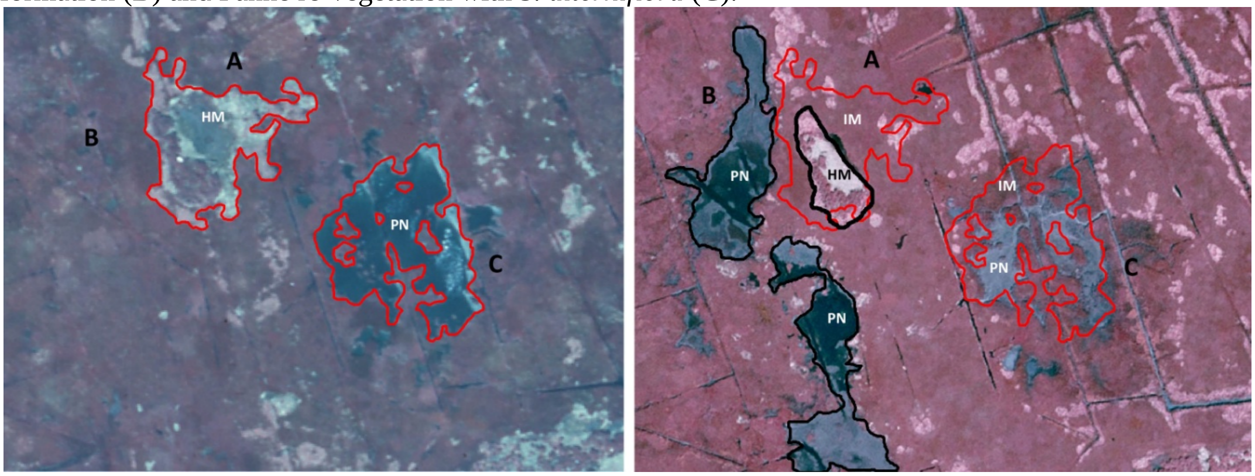


Figure 15. 1974(left) -2008 (right) imagery showing 11.34 acre IM loss on Alder Island (red).

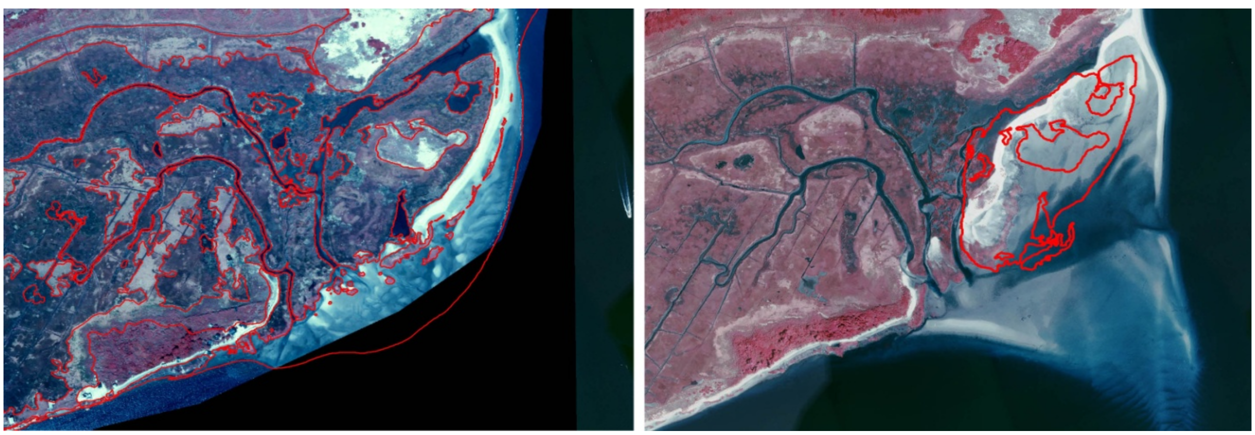


Figure 16. Perimeter loss along Lido Beach, mainland marsh and island marshes to the north. 2008 imagery, 1974 TWB (red polygons).

Figure 17. 1903USGS map (left) and 2008 IR (right) Middle Bay island change comparison.

Figure 18. Blackbank Hassock SW island marsh loss, fragmentation and re-vegetation, 1974 TWB (red); IR imagery background.

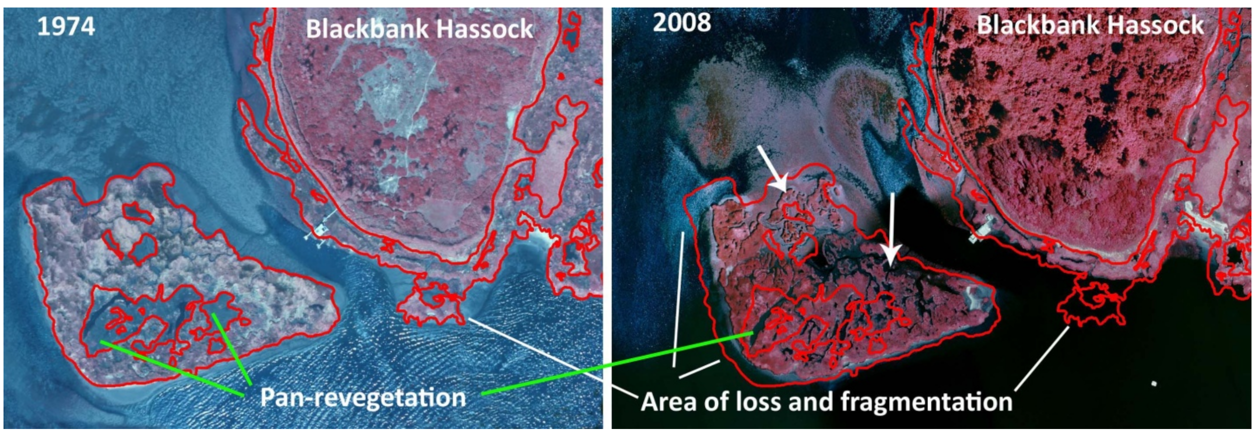


Figure 19. Blackbank Hassock SE island marsh loss and fragmentation and re-vegetation loss, 1974 TWB (red polygon)-IR imagery background.

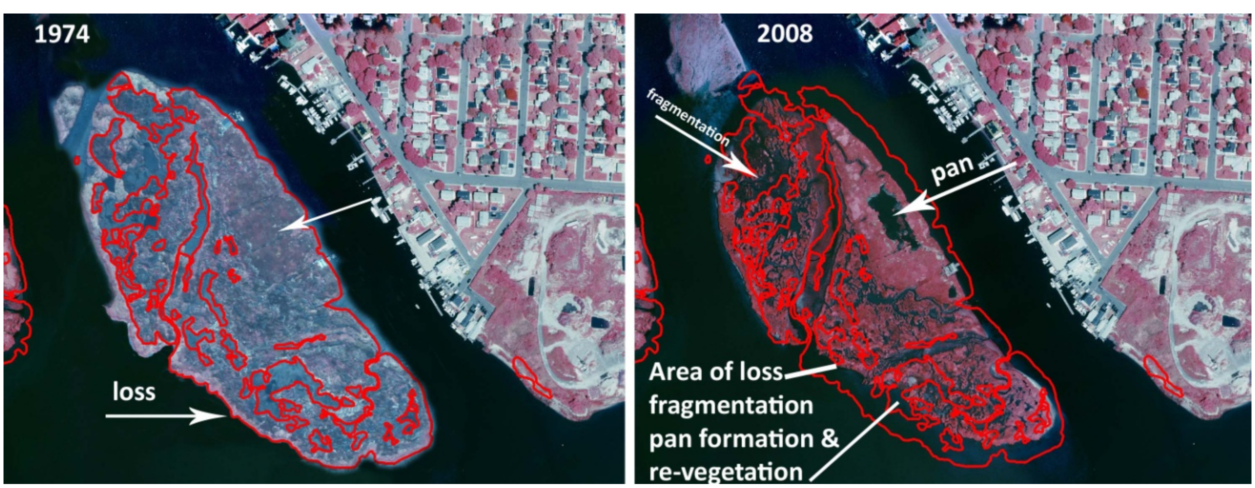


Figure 20. Algal mat distribution adjacent and within Blackbank Hassock SW and SE island marshes.

Figure 21. Algal mats surrounding portions of Nums Marsh island complex, 2008 imagery, algal mats (largest mat area, 28.68 acres) green polygon.

Table 6. Marsh island fragmentation.

Nums Island marsh										
					Total	Percent	Rate of Change	Total	Frequency	Frequency
TW Type	1974 Acres	Frequency	2008 Acres	Frequency	Acre Change	Change	A/Yr	Frequency Change	Percent Change	Rate of Change
IM	5.72	31	2.3230	44	3.40	59.39	0.1	13	41.94	0.38
PN	0.50	15	0.1567	18	0.34	68.66	0.01	3	20.00	0.09
CH	0.16	3	0	0	0.16	100.00	0.005	0	0.00	0
Black Bank Hassock SE Island Marsh										
					Total	Percent	Rate of Change	Total	Frequency	Frequency
TW Type	1974 Acres	Frequency	2008 acres	Frequency	Acre Change	Change	A/Yr	Frequency Change	Percent Change	Rate of Change
IM	17.43	9	9.90	95	7.53	43.20	0.22	86	955.56	2.53
HM	2.28	4	0.27	13	2.01	88.16	0.06	9	225.00	0.26
PN	3.00	29	0.54	79	2.46	82.00	0.07	50	172.41	1.47
CH	0.63	3	1.14	3	0.51	80.95	0.015	0	0.00	0
Black Bank Hassock SW marsh Island										
					Total	Percent	Rate of Change	Total	Frequency	Frequency
TW Type	1974 Acres	Frequency	2008 Acres	Frequency	Acre Change	Change	A/Yr	Frequency Change	Percent Change	Rate of Change
IM	9.60	5	6.4115	27	3.19	33.21	0.09	22	440.00	0.65
PN	2.01	9	0.1876	51	1.82	90.67	0.05	42	466.67	1.24

Table 7. 1974-2008 eastern island TW trends.

TW Type	1974 Acres	Frequency		2008 acres	Frequency	Total Acre Change	Percent Change	Rate of Change A/Yr	Total Frequency Change	Frequency Percent Change	Frequency Rate of Change
IM	1574.87	307		1338.98	1005	235.89	14.98	6.94	698	227.36	20.53
HM	291.75	622		200.91	1483	90.84	31.14	2.67	861	138.42	25.32
PH	1.49	2		49.82	24	48.33	3243.62	1.42	22	1100.00	0.65
DS	138.91	9		122.72	13	16.19	11.66	0.48	4	44.44	0.12
PN	35.49	845		104.94	4103	69.45	195.69	2.04	3258	385.56	95.82
MD	3.45	104		23.68	425	20.12	583.19	0,6	321	308.65	9.44
PD	2.33	22		0.80	68	1.53	65.67	0.05	46	209.09	1.35
CH	41.03	143		72.30	148	31.27	76.21	0.92	5	3.50	0.15

Table 8. 1974-2008 eastern island TW trends.

	1974 Acres	Frequency		2008 Acres	Frequency	Total Acre Change	Percent Change	Rate of Change A/Yr	Total Frequency Change	Frequency Percent Change	Frequency Rate of Change
IM	1445.21	354		1126.06	1083	319.15	22.08	9.38	729	205.93	21.44
HM	151.81	312		245.59	2719	93.78	61.78	2.76	2407	771.47	70.79
PH	7.12	5		49.78	76	42.66	599.16	1.25	71	1420.00	2.1
DS	268.28	19		188.15	30	80.13	29.87	2.36	11	57.90	0.32
PN	60.88	990		74.03	3144	13.15	21.60	0.4	2154	217.58	63.35
MD	1.58	53		15.16	190	13.58	859.49	0.4	137	258.49	4.03
PD	4.55	26		0.49	2	4.06	89.23	0.12	24	92.31	0.71
CH	51.51	105		109.33	87	57.82	112.25	1.7	18	17.14	0.53

Figure 22. Pearsalls Hassock DS (1974 left) re-vegetation to IM and HM 2008 right with 1974 overlay (red).

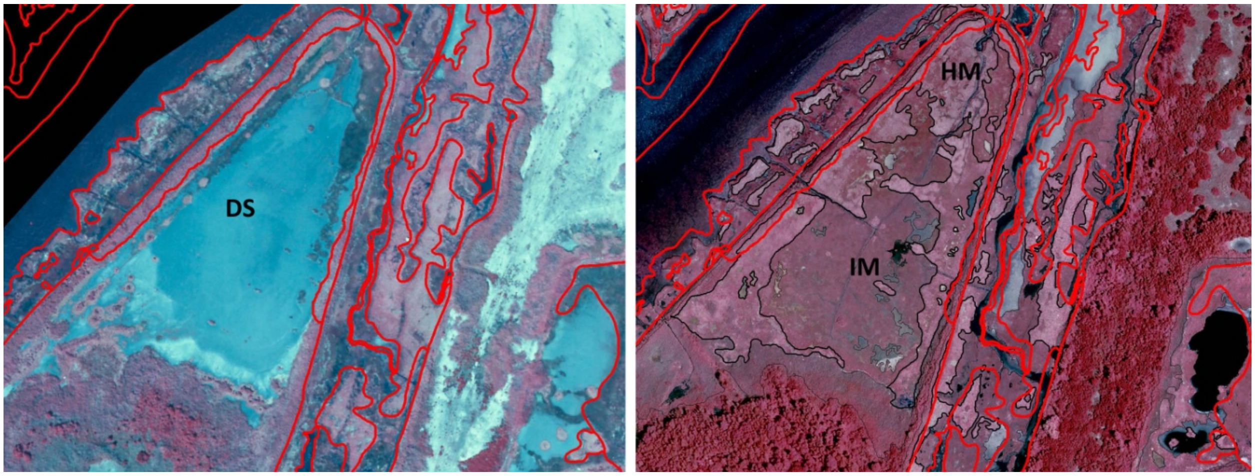


Table 9. 2008 IM densities

		Density		IM Avg Density (Stalks/Sq. M)	
Station No.	TW Sample sites	A	B	IM Peat	IM SM/Fucus
1	Long Meadow Island (IM)	288	396	342	
2	dark signature (IM)/Fucus	283	0		144
3	dark signature (IM)/Fucus	144	288		216
4	typical red signature (IM)	1296		1296	
5	Seadog Is. (IM) dark signature	180	324		252
6	Seadog Is. (IM)	1188	1332	1260	
7	Seadog Is. (IM)	1152		1152	
8	Seadog Is. (IM) dark signature	702			702
9	Recycle Ctr. West marsh (IM)	720		720	
10	Recycle Ctr. West marsh (IM)	720		720	
TOTALS		6673	2340	5490	1314
Avg. density (Mean)		667	858	915	329

Figure 23. Average IM densities from mainland and island marshes.

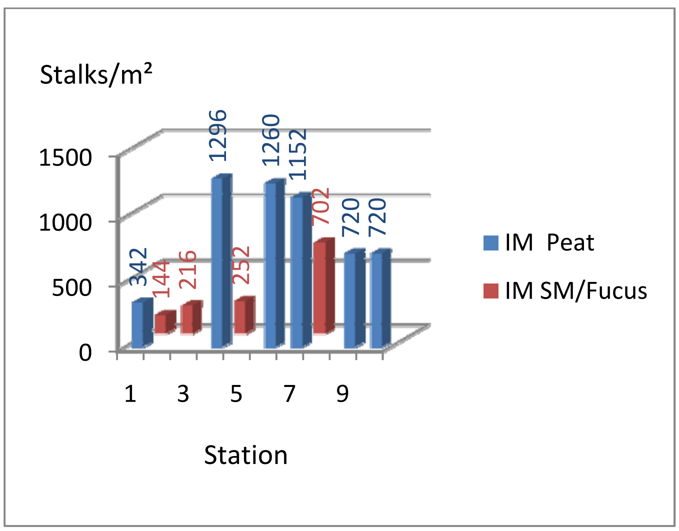


Figure 24. Percent low density and frequency of IM throughout the AOI.

Acres /Percent **Frequency/Percent**

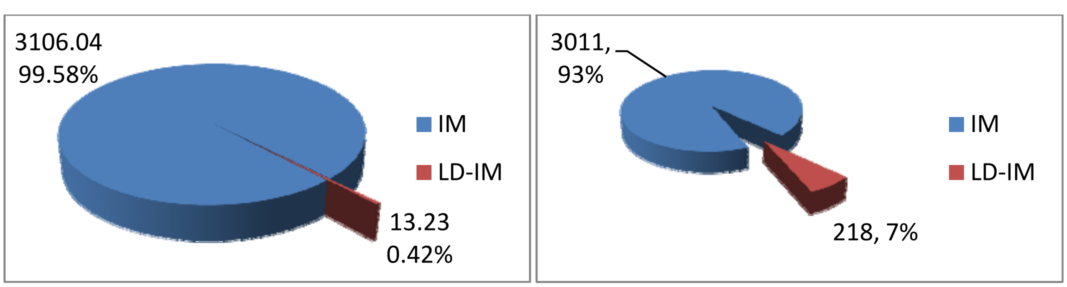
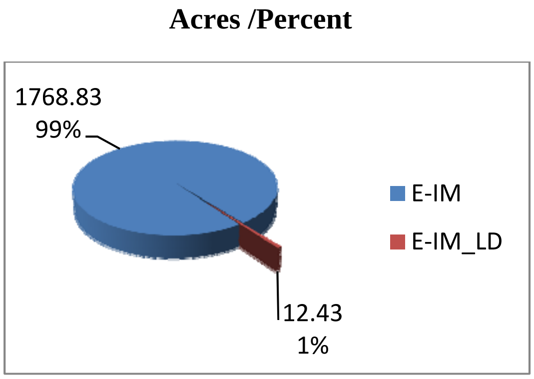


Figure 25. Percent low density and frequency of IM throughout the eastern AOI.



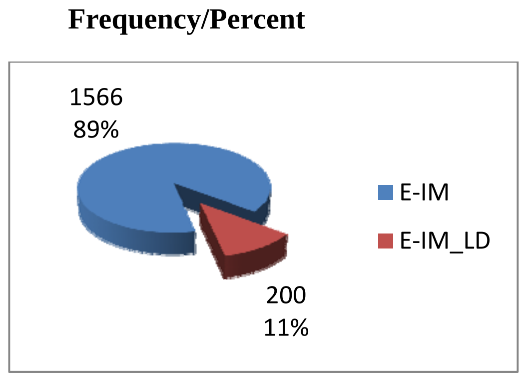


Figure 26. Percent low density and frequency of IM throughout the eastern AOI.

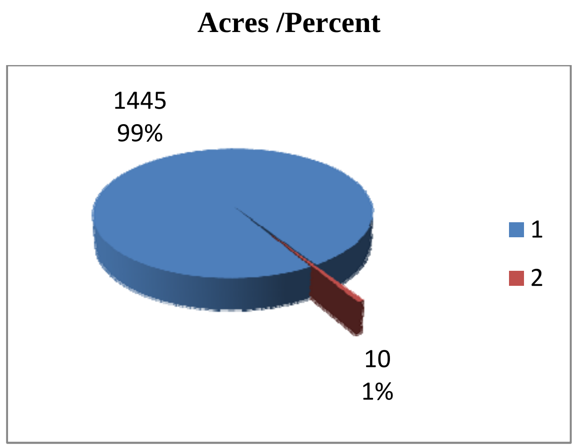


Figure 27. Extent of low density IM in the AOI (yellow), note greater frequency in eastern bay.

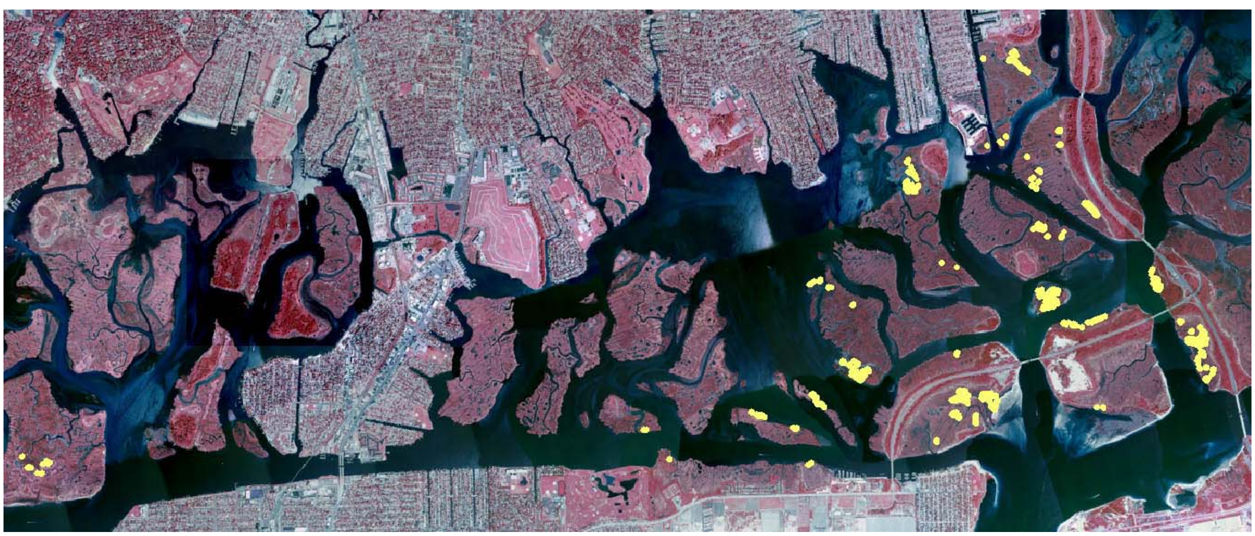
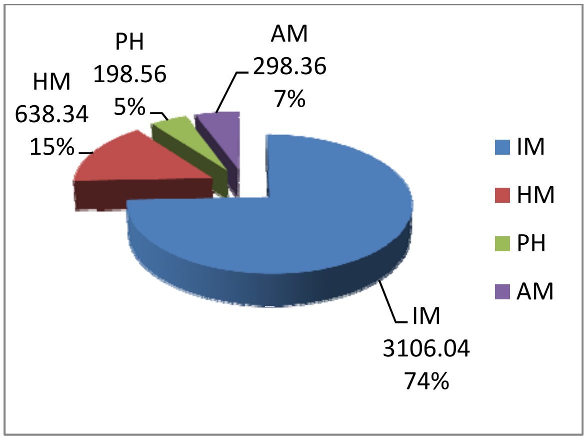


Figure 28. August 26, 2008 overall extent of algal mats within the AOI.

Figure 29. Percent and frequency of AM throughout the AOI.

Acres/Percent



Percent/Frequency

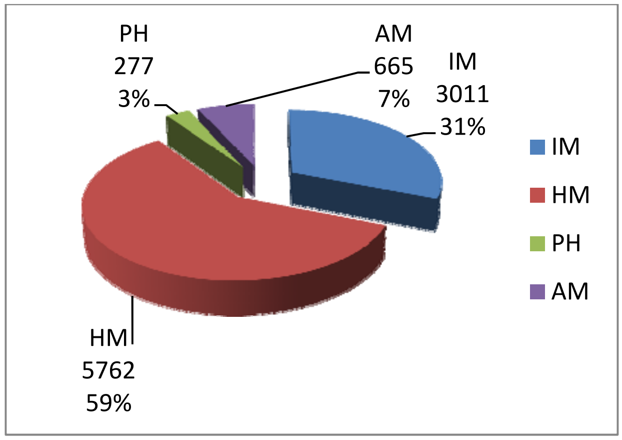
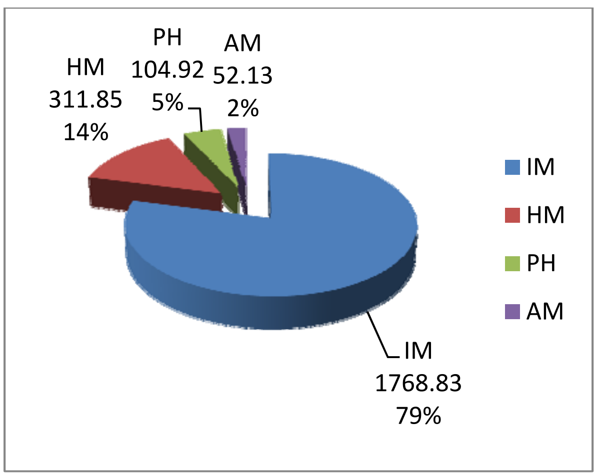


Figure 30. Percent and frequency of AM throughout the eastern AOI.

Acres/Percent



Percent/Frequency

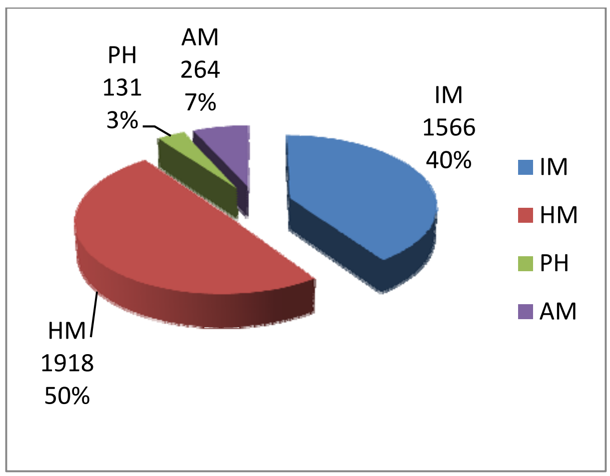
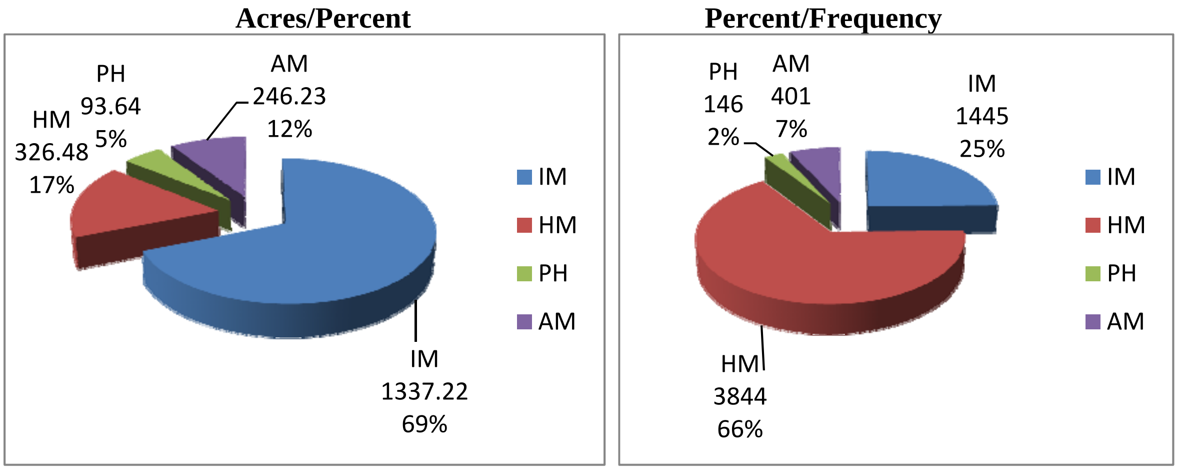


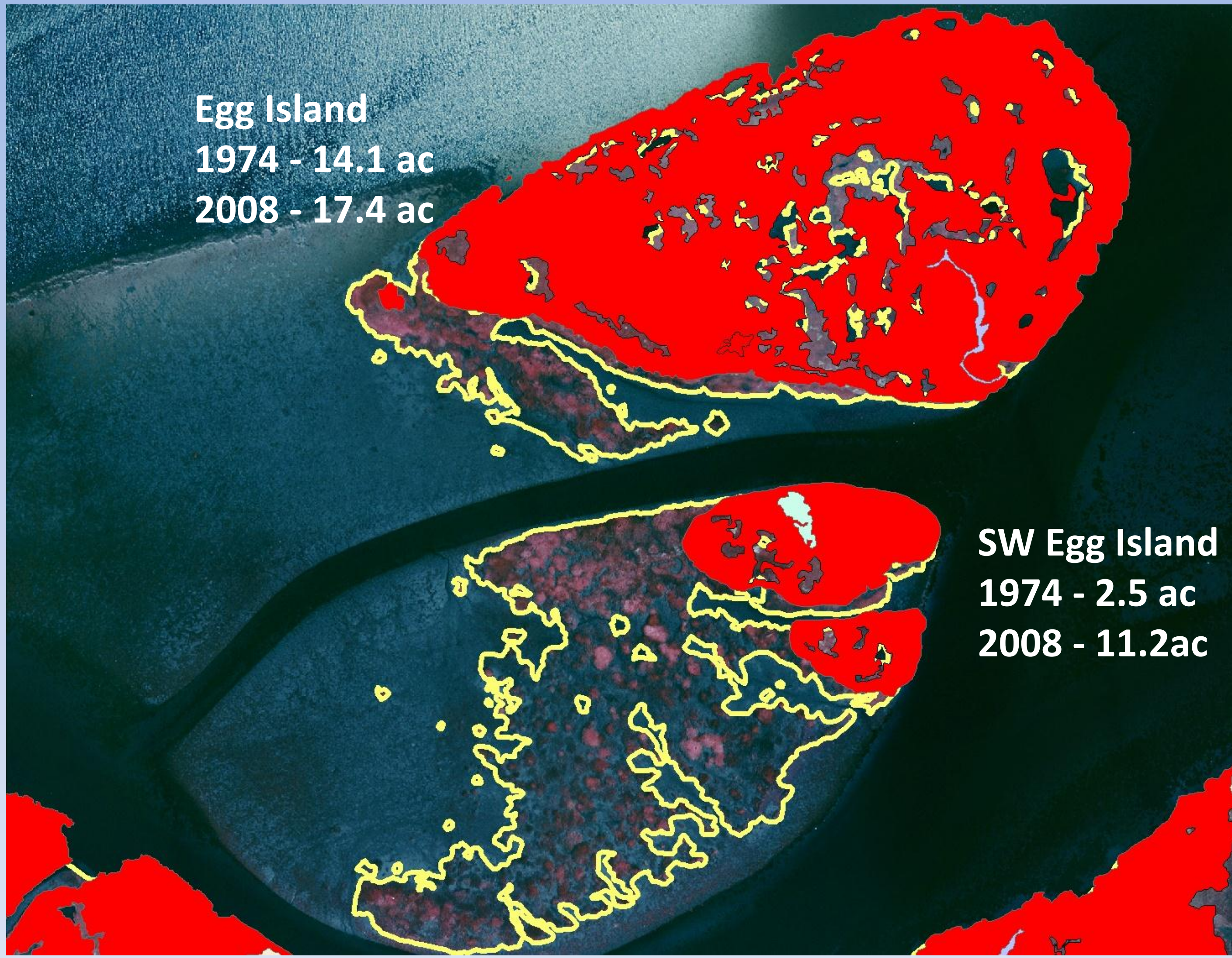
Figure 31. Percent and frequency of AM throughout the western AOI.



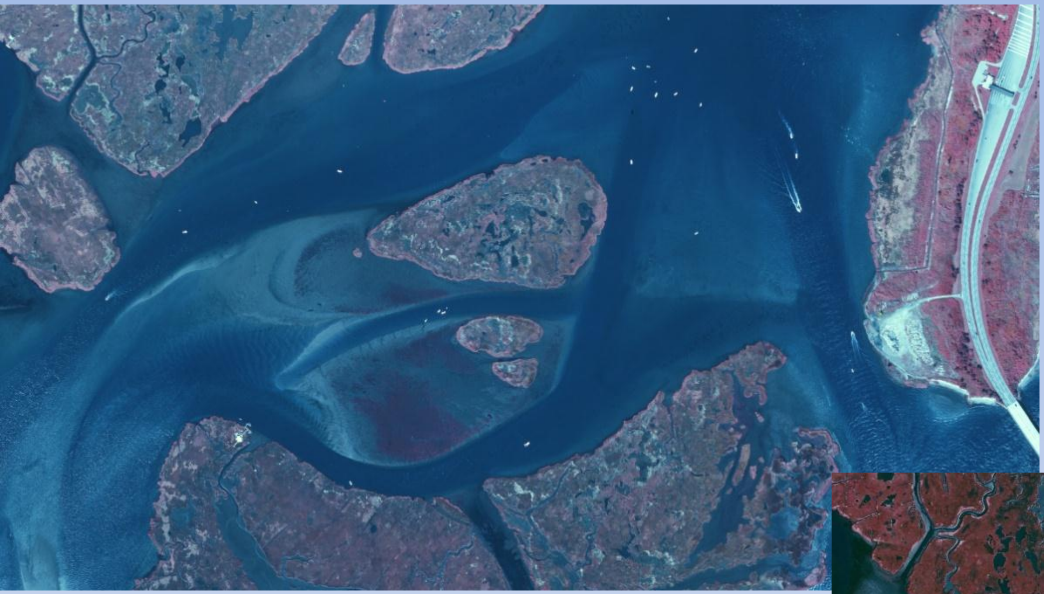
APPENDIX H

ATTACHMENTS

1974 – 2008 Comparison of Egg Island Complex 1974 (solid red); 2008 (yellow outline)



1974 IR image



2008 IR image