LONG ISLAND SOUTH SHORE ESTUARY RESERVE
EASTERN BAYS PROJECT: NITROGEN LOADING, SOURCES, AND MANAGEMENT OPTIONS

Prepared for:
New York State
Department of State
Office of Planning and Development
on behalf of the South Shore Estuary Reserve Council

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Final Summary Report
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Cover Photo Credit: Algal bloom in Moriches Bay. Dr. Christopher Gobler, State University of New York at Stony Brook, School of Marine and Atmospheric Sciences.

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June 22, 2016

Nancy Rucks
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RE: Eastern Bays Project: Nitrogen Loading, Sources and Management Options

Dear Ms. Rucks,

The New York State Department of Environmental Conservation (DEC) appreciates the opportunity to have worked with the Department of State (DOS) in reviewing the DOS funded project "Eastern Bays Project: Nitrogen Loading, Sources and Management Options, Suffolk County" completed per Memorandum of Understanding (MOU# U1000184) between DOS and Stony Brook University.

This project utilized widely accepted methods to estimate nitrogen loads; assess the role of hydraulic flushing on waterbody health relative to the corresponding nitrogen load; provided a comparison of bay water quality to nitrogen loading rates; and evaluated various nitrogen mitigation scenarios. DEC views this work as complimentary to our efforts as we move forward with the Long Island Nitrogen Action Plan (LINAP) to address nitrogen pollution in Long Island's waters with the expectation that LINAP will be completed with an increased level of detail and complexity.

As we move forward with LINAP we are looking forward to having the DOS as a partner in this process.

Sincerely,

Kenneth A. Kosinski, P.E.
Chief, Water Implementation Section

C: L. Holdridge - NYSDEC
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Contents

1. Executive Summary 1
2. Introduction 3
3. Methods 6
3.1 Subwatersheds 6
3.2 Nitrogen Loading Model (NLM) 8
3.2.1 Atmospheric Deposition of Nitrogen 8
3.2.2 Wastewater 14
3.2.3 Fertilizer 15
3.3 Volumetric Flux Model (VFM) 16
3.3.1 Groundwater 16
3.3.2 Runoff 18
3.3.3 Streams 18
3.4 Further Modeling and Analyses 19
3.4.1 Nitrogen Load Comparisons 19
3.4.2 Estuarine Loading Model (ELM) 19
3.4.3 Evaluating Bay Water Quality Data and Eutrophication Vulnerability 20
3.5 Nitrogen Management Options 20
3.6 Determining the Relative Priority of Remedial Measures 21
4. Results 21
4.1 Nitrogen Loading Model (NLM) 21
4.2 Volumetric Flux Model (VFM) 25
4.3 Comparing Models 28
4.4 Estuarine Loading Model (ELM) 29
4.5 Marine Data 30
4.6 Nitrogen Mitigation Scenarios 37
5 Discussion 42
5.1 Nitrogen Sources and Transport Mechanisms 42
5.2 Comparability of Results 44
5.3 Effects of Marine Nitrogen Loading 46
5.4 Prioritizing Regions for Nitrogen Mitigation 47
5.5 Next Steps and Recommendations 50
6 References 53

1. Executive Summary

Since the 20th century, delivery of excessive nitrogen into Long Island's South Shore Estuary Reserve (SSER) waters has led to a host of environmental problems including: harmful algal blooms, hypoxic zones, bay water acidification, and habitat degradation and loss. Much of the nitrogen finding its way to SSER waters results from precipitation, stormwater (runoff from the land), atmospheric deposition, drainage, or seepage - including seepage from septic systems and cesspools. This type of pollution is referred to as nonpoint source pollution because it comes from many diffuse sources, unlike point source pollution that are a result of pipe discharge from sewage treatment and industrial plants[1]. The quantity and sources of nitrogen from non-point sources of pollution in the SSER have been largely unknown which has made determining the most effective course of water quality management a challenge.

This study was a first attempt to estimate the amount of nitrogen entering the Eastern Bays of the SSER - specifically Moriches, Quantuck and Shinnecock Bays - from non-point sources of nitrogen. Each bay watershed was broken down into subwatersheds using United States Geologic Survey (USGS) Hydrologic Unit Codes (HUC) 12 in order to narrow the geographic scope and thus obtain the most accurate estimates of nitrogen loading and sources. The study used two established nitrogen loading models - Nitrogen Load and Volumetric Flux, land use and population characteristics, information on fertilizer use, and the most current science in groundwater and surface water movement to estimate the relative contribution of wastewater from on-site septic systems or sewage treatment plants, fertilizer, and atmospheric deposition to nitrogen loads in each subwatershed. The models were also used to identify the primary nitrogen transport mechanisms including groundwater underflow to the bays, stream flow, and surface runoff. The study also modeled residence time of water within each bay. Longer residence time and less flushing can lead to greater rates of eutrophication of bay waters which is critical information in management decision making. Water quality data from Suffolk County and other regional sources were compared to nitrogen loading rates and water residence times determined in this study to assess the role of hydraulic flushing in influencing water quality. Lastly, this study evaluated several different nitrogen mitigation scenarios tailored to the land uses and residence times within each subwatershed. The effectiveness of various nitrogen mitigation scenarios, including changes in land use and improvements in wastewater handling, were assessed within the models.

Modeling determined that the nitrogen loads to the three bays, per hectare of surface water was moderate compared to other estuaries; but, was in the high range when the loads were assessed based on volume of water. The models determined that the relative contribution of land-based nitrogen load to the three bays was 65% wastewater, 20% fertilizer, and 15% atmospheric deposition. It was shown that groundwater was responsible for the transport of more than 90% of the nitrogen load in all subwatersheds except western Moriches Bay, while stream and stormwater runoff delivery of nitrogen to the subwatersheds was small. Modeling determined that the western portion of Moriches Bay, including the Forge River tidal estuary, and Quantuck Bay were two watersheds with the largest nitrogen loads on a per volume basis, the longest water residence times, and poorest water quality with regard to harmful algal blooms, dissolved oxygen, and water clarity. As

such, the key finding of this study is that extended water residence times are a key factor that, coupled with elevated nitrogen loads per unit volume of bay water, contributes toward poor water quality in the Eastern Bays of the SSER. Wastewater was the major source of nitrogen to the bays; therefore connecting onsite sewage disposal systems to sewage treatment plants, upgrading standard septic systems and antiquated cesspools to innovative septic systems with enhanced denitrification and nitrogen removal, and controlling future development were the key managerial efforts identified as methods of reducing nitrogen loading to the SSER's Eastern Bays, with the potential to reduce existing total nitrogen loads by up to 70%. Additionally, this report provides a series of recommended next steps for better understanding and managing nitrogen loads from land to surface within the Eastern Bays.

2. Introduction

Nitrogen found in coastal environments is derived from natural and anthropogenic sources such as septic systems and cesspools. As the human population, relying on septic systems, within a watershed grows so does the magnitude and proportion of anthropogenic nitrogen to coastal waters[2]. Eutrophication of a waterbody is a natural process that occurs over very long periods, it can become accelerated when there is an excessive input of anthropogenic nitrogen and is one of the most pressing contemporary environmental concerns in coastal areas. Microscopic marine plants, known as phytoplankton, are normally controlled by periodic nutrient limitation and grazing by a wide range of sea creatures, but in the face of nutrient overloading can become dense and pervasive.[3] Such algal blooms can attenuate light penetration through the water column, decreasing the depth at which benthic phototrophs, such as seagrasses, can survive.[4] Additionally, oxygen concentrations can decrease sharply beneath the surface of the water due to the respiration and decomposition of the excessive organic matter from decaying algal blooms. In this way, eutrophication often leads to hypoxia (very low levels of oxygen) or anoxia (zero oxygen), which can be deleterious to fish and benthic communities[5] living in and on the sea floor.

Harmful algal blooms (HABs) are also an environmental problem initiated by nutrient overload, which have increased in their geographic extent, intensity, duration, and diversity in recent decades.[6,7] There are clear linkages between increased loading of nitrogen in coastal waters and the presence and prevalence of HABs in many ecosystems.[8,7] In some coastal areas such as Long Island, HABs promoted by nitrogen have become annual occurrences. The phytoplankton that compose these HABs are diverse and can affect ecosystem conditions, commercial and recreational fisheries, and human health. For example, wastewater-derived nitrogen (i.e. from sewage) has been shown to support the proliferation of saxitoxin-producing blooms of *Alexandrium fundyense* that can cause paralytic shellfish poisoning.[9] Brown tides, caused by *Aureococcus anophagefferens* flourish when there are high levels of organic nitrogen and turbidity[10] and cause death to shellfish and eelgrass (*Zostera marina*).[11] Nitrogen also promotes toxic dinoflagellate blooms of *Cochlodinium polykrikoides* that cause fish kills.[12,13,14]

Since nitrogen limits primary production[3,15] (production by plants at the base of the marine food web) it is often the nitrogen delivery rate (weight of nitrogen delivered per land area or water body volume per year) coupled with hydraulic flushing that influences the prevalence of HABs, intensity of hypoxia, and the loss of seagrass beds.[16] On Long Island, the major sources of nitrogen to Long Island Sound and the Peconic Estuary are wastewater, fertilizer, and the atmosphere.[17,18] However, the relative importance of a nitrogen source can vary over even small geographic distances.[17,18] As a result, nitrogen loading models are required to predict the amount of nitrogen that various sources contribute to estuaries and how those spatial differences in nitrogen load relate to coastal land use.[19]

The SSER, which stretches more than 70 miles from the western boundary of the Town of Hempstead to the middle of the Town of Southampton, is comprised of a series of shallow interconnected bays separated from the Atlantic Ocean by barrier islands. Water

exchange between the Ocean and the bays is often minimal and occurs through inlets in the barrier islands.[20] Such bays are typically shallow, well mixed, and have long residence times.[20] Because of this, organic material from the watershed tends to accumulate in bays making them productive marine environments but also very susceptible to eutrophication.[21,22] The SSER watersheds are densely populated (1.5 million people) and economically and recreationally important. All of the SSER bays have been identified as impaired according to the *New York State Section 303(d) List of Impaired/TMDL Waters* since 2010 due to onsite wastewater disposal and algal blooms.[23]

Impairments of the SSER bays have been well-documented in recent decades and have included harmful algal blooms caused by multiple phytoplankton species and hypoxia. Since 1985 blooms of *Aureococcus anophagefferens*, 'brown tide', have become near annual occurrences.[11] *Cochlodinium polykrikoides*, a dinoflagellate, was first observed in the SSER in 2004[12] and has subsequently occurred every year through 2015.[13] A third toxic alga, *Alexandrium fundyense*, which causes paralytic shellfish poisoning, was first observed in the SSER bays in 2008 (C. Gobler, pers. obs.) and has since led to periodic closings of the Shinnecock Bay shellfish beds due to paralytic shellfish poison toxins (NYS DEC, 2011, 2012, 2015). In many years, all three harmful algal blooms occur in succession for May through September. Concurrently, shellfish populations have declined in Shinnecock Bay[24] and eelgrass coverage has decreased in both Shinnecock and Quantuck Bays.[25] These events are interrelated, as dense HABs can be both toxic to shellfish and can shade eelgrass.[11]

Eutrophication due to excess nitrogen has been a problem in the region since the 1950's.[26,27] Historically on Long Island there were more than 90 Long Island Peking duck farms. The 9,450 acre Forge River watershed, which discharges to western Moriches Bay, had eight duck farms at its peak in the 1960's. Peak duck production for Suffolk County occurred in 1959 with nearly eight million ducks produced. Approximately 10 ducks release the same amount of nitrogenous waste as one human.[27] The last of these duck farms in the Eastern Bays closed in 2011.

As duck farming in the Eastern Bays ended, the human population within the Forge River watershed has grown substantially from 10,000 in 1960 to 60,000 in 2005.[27] Swanson et al[27] determined that the increase in human population in recent decades has more than replaced the loss of nitrogen loading associated with the closure of duck farms. In parallel, since the 1980s the population of Suffolk County has grown to more than 1.5 million people with growth expected to continue in the coming decades.[28] Between 1990 and 2010 there was a 12% increase in the population of Suffolk County and the projected increase is 16% for 2035.[28] This population increase and the corresponding increase in anthropogenic nitrogen supply due to the growing population using septic systems with low nitrogen removal rates in sandy soils suggest that environmental conditions in these watersheds could worsen in the coming decades as groundwater originating from inland Long Island travels through the aquifer and discharges into surface waters. Additionally, the influx of summer residents and visitors is more than double the permanent population in eastern townships such as Southampton indicating that within eastern townships, seasonal nitrogen loads may increase substantially.[29] Finally, according to Southampton Town records, there

is a substantial amount of open space (more than 15% of land) that may still be developed, indicating a potential for increased nitrogen loading from wastewater.

Despite the prevalence of environmental problems within the Eastern Bays, the rates and sources of nitrogen loads to Moriches, Quantuck, and Shinnecock Bays have not been quantified prior to this study. This knowledge gap prohibits the formulation and evaluation of management plans to ameliorate nitrogen loads to these bays. Given the large costs associated with many nitrogen mitigation strategies, it is important to quantify the relative contribution of all of the major sources of nitrogen to the bays. This information can then be used to determine cost effectiveness of different strategies for reducing nitrogen loads. Quantifying the current nitrogen loads entering the Eastern Bays as well as quantifying how those loads would change under different nitrogen mitigation and land-development scenarios is a vital tool for proper water quality management.

Addressing the detrimental consequences of excess nitrogen loading to coastal waterbodies represents a significant challenge for local governments, Suffolk County, and New York State. However, the quality of surface waters and the health of the bays have an impact on the economy and quality of life in Suffolk County.[28] Tourism, commercial and recreational fisheries, and boating are collectively a multi-billion dollar industry in Suffolk County.[30] It is widely recognized that the future of smart economic development in Suffolk County will require upgrades to wastewater treatment infrastructure and local land-use policies.[30] Suffolk County is exploring options to reduce nitrogen such as new and expanded sewer districts and the installation of alternate on-site sewage disposal systems (denitrifying systems). The type of quantitative data generated by this research will assist in forecasting the value of proposed projects and actions in terms of how they will influence the nitrogen loads and the quality of bay waters.

This research quantified nitrogen loads to the SSER Eastern Bays and determined the major sources of nitrogen and transport mechanisms. The watersheds of Moriches, Shinnecock, and Quantuck Bays were broken down into multiple sub-watersheds. The study further assessed the spatial variability in water quality (HABs, dissolved oxygen, water clarity) and compared nitrogen loading rates from land to water residence time data to determine the eutrophication vulnerability of each subwatershed. How the various watershed management strategies would alter nitrogen loads to each subwatershed was assessed and from that, the relative priorities for remedial measures for nitrogen reduction were determined.

3. Methods

3.1 Subwatersheds

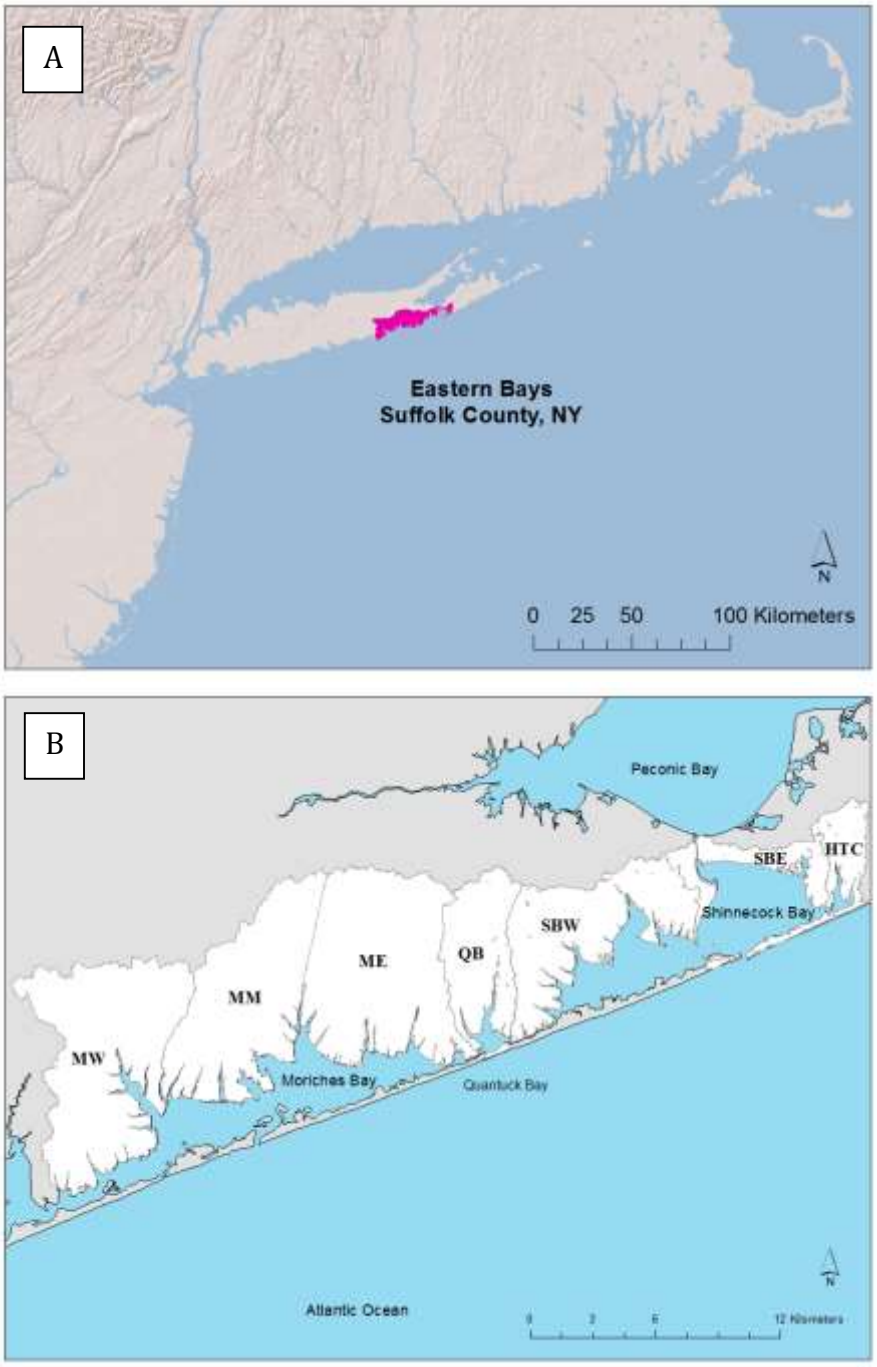
The areal extent of the Moriches, Quantuck, and Shinnecock Bay watersheds were determined using Suffolk County's LiDAR elevation data, topographical maps demarking the surface watersheds (HUC 12), and groundwater flow patterns, which have been previously found to generally follow hydraulic gradients established by surface topography.[31] The Moriches and Shinnecock Bay watersheds were separated into three subwatersheds in the same manner. Quantuck Bay, given its small size, was its own subwatershed. The resulting subwatersheds are more detailed than the Hydrologic Unit Code (HUC) 12 delineation (Figure 1). These seven small subwatersheds include:

Moriches West subwatershed (MW);
Moriches Middle subwatershed (MM);
Moriches East subwatershed (ME);
Quantuck Bay subwatershed (QB);
Shinnecock Bay West subwatershed (SBW);
Shinnecock Bay East subwatershed (SBE); and,
Heady/Taylor Creek subwatershed (HTC).

Throughout this report the term subwatersheds will be referencing these units while the term watersheds will be used more conceptually or when referencing the entire watersheds of the three bays: Moriches, Quantuck, and Shinnecock.

The United States Geological Survey (USGS) study of the SSER by Monti and Scorca[32] determined that a portion of Long Island groundwater flow bypasses the south shore bays and releases directly into the Ocean. This groundwater comes from the furthest upgradient portion of the watershed; therefore, Kinney and Valiela[19] used 20% as the underflow in their study of the Great South Bay (GSB) watershed. The Moriches Bay watershed abuts the GSB watershed study area, however, the Moriches Bay watershed is smaller latitudinally, indicating less land area upgradient, and the Moriches Bay shoreline is much more irregular than that of GSB. Because of this a 10% underflow rate was used for the Moriches and Quantuck Bay watersheds in this study. No underflow was used for the Shinnecock Bay watershed as there is significantly less upgradient area and little elevation gain in this watershed.

Figure 1. A. The study area. B. The subwatersheds: MW-Moriches West Subwatershed, MM-Moriches Middle subwatershed, ME-Moriches East subwatershed, QB-Quantuck Bay subwatershed, SBW- Shinnecock Bay West subwatershed, SBE-Shinnecock Bay East subwatershed, HTC-Heady/Taylor Creek subwatershed.



3.2 Nitrogen Loading Model (NLM)

The first model used to predict the total dissolved nitrogen input into the Eastern Bays was the Nitrogen Loading Model (NLM)[33] available through the Nitrogen load web-based modeling tool (nload.mbl.edu) described in Bowen et al.[34] and used in Bowen and Valiela[35] and recently in Kinney and Valiela,[36] among others. The NLM uses information about land use in a defined watershed to predict both the amount of nitrogen that is released into the watershed from various sources and how much of it ends up in a corresponding waterbody. This model requires accurate land-use and land cover information, such as area of agriculture, residential areas, and impervious surfaces as well as other environmental data gathered from scientific literature, GIS data, USGS reports, the Town of Southampton, and Suffolk County as described in Table 1.

The NLM is a good fit for watersheds such as the Eastern Bays that are a mix of residential, forested, and agricultural lands.[33] The NLM assumes that the primary transport mechanism for nitrogen entering the bays from each watershed is ground-water flow. This assumption is consistent with data available for the Eastern Bays as there is little inflow to the bay from streams and geologically, Long Island is composed of unconsolidated sands that allow for relatively easy transport of groundwater to coastal bays.[19] The NLM assumes that all nitrogen entering the bays from external sources originated from atmospheric deposition to the watershed, wastewater, or fertilizer. This study also includes atmospheric deposition directly to the surface water of the Bays. Valiela et al[33] validated this model by comparing its nitrogen load prediction to empirically measured nitrogen levels. They found the NLM's results to be statistically indistinguishable from measured concentrations and that a linear relationship exists between the percent contributions from wastewater that the NLM predicted and the stable isotope signature for wastewater expected from known isotopic nitrogen values of nitrate in groundwater. The NLM is one of the most inclusive nitrogen loading models regarding the transformation and transport of nitrogen as it travels from watershed to estuaries.[37]

The NLM utilizes multiple features within the model for each subwatershed; these features include: number of buildings; buildings within 200 meters of shore; surface area of the subwatershed; area of freshwater wetlands; agriculture; golf courses; parks and athletic fields; freshwater ponds; and, impervious surfaces, and were obtained from the Town of Southampton and Suffolk County. The model also includes a list of inputs assigned default values based on an extensive metadata analysis (Table 1).[33] These model defaults were changed when local and site-specific information were available. For example, a recent study by Young et al.[38] of denitrification in Long Island's aquifer, found that the 15% of nitrogen in groundwater is denitrified rather than the 35% assumed by Kinney and Valiela[39]. The NLM has a 12% standard error coefficient.[33]

3.2.1 Atmospheric Deposition of Nitrogen

Atmospheric nitrogen is delivered via precipitation (wet) or via dust (dry). Nitrogen that arrives in the SSER watershed through wet and dry deposition may have more or less of a contribution to bay water quality depending where the nitrogen lands. Nitrogen landing on vegetation has time to be assimilated by plants and organisms in the soils, and/or may be denitrified in the aquifer. Nitrogen that lands on impervious surfaces

can runoff directly into a stream, a bay, or a municipal separate stormwater sewer system (MS4), or eventually seep into sandy soils and discharge into coastal zones. In general, significantly less nitrogen is removed from atmospheric deposition when it lands on impervious surfaces.

The amount of nitrogen coming from atmospheric deposition is altered by land uses. Impervious surfaces offer no removal services and all deposited nitrogen on these surfaces makes its way to the bays through streams, municipal stormwater sewer system (MS4), or through groundwater close to estuaries. Conversely, nitrogen deposited on vegetated lands is removed by plants and organisms, and is denitrified in the aquifer. The atmospheric deposition of nitrogen is decreasing on Long Island (Figure 2) and the Northeast in general, a trend expected to continue due to changes in industrial atmospheric discharges in the Midwest

The land-use and land cover information used for the NLM was obtained from Suffolk County land-use GIS data for all watersheds and in the Shinnecock and Quantuck watersheds was supplemented with data from the Town of Southampton GIS Department. All inputs to the NLM and their sources are referenced in Table 1. The total area of impervious land cover was estimated from specifically assigned land use (for example, low-density residential ~ 10% impervious; Table 2). The parcel areas were then multiplied by a percentage of imperviousness as determined averaging values from several sources[40,41,42,43,44,45,46] (see Table 2). Roof area per building was quantified by calculating the average area of the footprint of buildings within the subwatershed. The road area as a percent of total subwatershed was calculated by using the length of actual roads in the subwatershed and a standard road width of 25 feet. The road area was divided by subwatershed area to determine percent.

Table 1: Inputs to NLM. The subwatersheds are labeled: MW-Moriches West subwatershed, MM-Moriches Middle subwatershed, ME-Moriches East subwatershed, QB-Quantuck Bay subwatershed, SBW- Shinnecock Bay West subwatershed, SBE-Shinnecock Bay East subwatershed, HTC-Heady/Taylor Creek subwatershed

	MW	**MM**	**ME**	**QB**	**SBW**	**SBE**	**HTC**	**Units**	**Source**
Number of buildings	21,800	8,330	5,340	1,760	8,410	2,680	1,400		Suffolk County building footprint dataset, 2006
Watershed area	5,740	4,770	5,790	2,330	4,040	1,320	892	ha	ArcGIS®
Area of freshwater wetlands	156	86	81	24	34	8	40	ha	Suffolk County NYS freshwater wetlands maps
Area of agriculture	171	573	148	5	125	0	34	ha	Suffolk County land use GIS maps
Area of golf courses	170	67	53	0	14	3	30	ha	Southampton GIS department (SB, QB) and Google Earth (MB)
Area of fertilized parks and athletic fields	103	277	43	0	297	63	43	ha	Suffolk County land use GIS maps
Impervious surfaces (commercial, industrial, and other)	1,420	631	703	654	841	330	199	ha	Suffolk County land-use maps with percentage imperviousness determined by averaging values.[47,48,49,50,51,52,53]
Area of freshwater ponds	9	13	26	6	2	2	6	ha	Southampton GIS department (SB, QB) and Google Earth (MB)
Buildings within 200m of shore	3,110	1,930	1,410	481	2,450	1,180	168		Suffolk County building footprint dataset, 2006
Average occupancy rate per residence	2.99	2.99	3.01	3.01	3.01	3.01	3.01	pp/house	2010 census + estimated seasonal population from Suffolk County.[29]
Percent of buildings with cesspools	48	48	48	39	57	54	44	%	Southampton GIS department (houses built before 1973 have cesspools) (SB, QB), estimate MB

	MW	MM	ME	QB	SBW	SBE	HTC	Units	Source
Percent of buildings with fertilized lawns	50	50	50	50	50	50	50	%	The Nature Conservancy, Cold Spring Harbor Office
Area of roof per building	147	204	186	243	181	177	215	m^2	Suffolk County building footprint dataset
Area of road as a percent of total watershed	3	3	5	3	5	6	4	%	Southampton GIS department multiplied the length of all roads in the sub-watershed by a standard road width (SB, QB) and Meta-analysis.[33] (MB)
Nitrogen inputs from wet and dry deposition	18	18	18	18	18	18	18	kg N ha^{-1} yr^{-1}	National Atmospheric Deposition Program (NADP) and the EPA's Clean Air Status and Trends Network (CASTNET) 2010-2011
Fertilizer applied to golf courses	232	232	232	232	232	232	232	kg N ha^{-1} yr^{-1}	Maximum amount allowed by Suffolk County law
Fertilizer applied to parks and athletic fields	117	117	117	117	117	117	117	kg N ha^{-1} yr^{-1}	[54,55]
Fertilizer applied to agriculture	171	171	171	171	171	171	171	kg N ha-1 yr-1	[54,55]
Denitrification in the aquifer	15	15	15	15	15	15	15	%	[38]

Table 2: Impervious percentages from various sources by land-use category.[56]
+ Only three residential categories are used by Suffolk County: Low Density (LD) is ≥ 1-acre lots, Medium Density (MD) is ¼ - ½ acre lots, and High Density (HD) is ≤ 1/8-acre lots. * Estimated. Sources: a. USDA 1986 b. Hoffman and Canace 2002 c. Mass GIS d. Kellogg et al. 1997 e. Center for Watershed Protection 2002 f. New York State Department of State 1999 g. Arnold and Gibbons 1996.

Land-use Category	**Percent Impervious**							**Average**
Sources	a	b	c	d	e	f	g	
LD Res (3-5 acre lot)		8		8				LD 10.3+
MLD Res (2 acre lot)	12	12		11	11			
MD Res (1 acre lot)	20	18	10	14	14	40		
MHD Res (1/2 acre lot)	25	27	13	25	21			MD 34.5+
MHD Res (1/3 acre lot)	30	34						
MHD Res (1/4 acre lot)	38	39	57	36	28	75		
HD Res (1/8 acre lot)	65	59		55	33	100		HD 62.3+
Multi-family residential			80		44			
Institutional	50			34	34			39.3
Agriculture								0
Vacant								0
Commercial	85		90	72	72		85	80.8
Recreational and open space								0
Industrial	72		75	54	53		75	65.8
Transportation	72		75	72	80		100	79.8
Utilities								75*
Waste Handling and mgmt.								75*
Surface waters								0

Nitrogen inputs from wet and dry deposition were determined using the National Atmospheric Deposition Program (NADP; wet) and the EPA's Clean Air Status and Trends Network (CASTNET; dry). Data from the closest NADP monitoring station is in Southold, NY, 10 miles from eastern Shinnecock Bay, were utilized. Two years (2010-2011) of monitoring from CASTNET's three closest monitoring stations (Washington Crossing, NJ, Claryville, NY, and Abington, CT) were averaged to determine the dry deposition input. Atmospheric deposition rates often only consider inorganic forms of nitrogen despite the fact that organic nitrogen contribution to atmospheric deposition can be considerable.[57] While direct measurements are not available, a 1:1 ratio of inorganic to organic deposition of nitrogen has been suggested by Cornell et al.[57] Hence, the value of wet and dry deposition was doubled to account for organic nitrogen loading from these sources. As a result total input for atmospheric deposition (kg N yr^{-1}) was quantified by: 3.135 kg N yr^{-1} (wet) + 5.64 9 kg N yr^{-1}(dry)+ 8.775 kg N yr^{-1}(organic) = 17.55 kg N yr^{-1}. A literature review determined that the atmospheric deposition value was comparable to prior studies (Table 3). Direct atmospheric deposition rate to the bays was added to the nitrogen load from wastewater and fertilizer for a final nitrogen load rate for each subwatershed.

Figure 2. Wet atmospheric deposition measurements by year from the three nearest CASTNET stations: Washington Crossing NJ, Claryville NY and Abington CT.

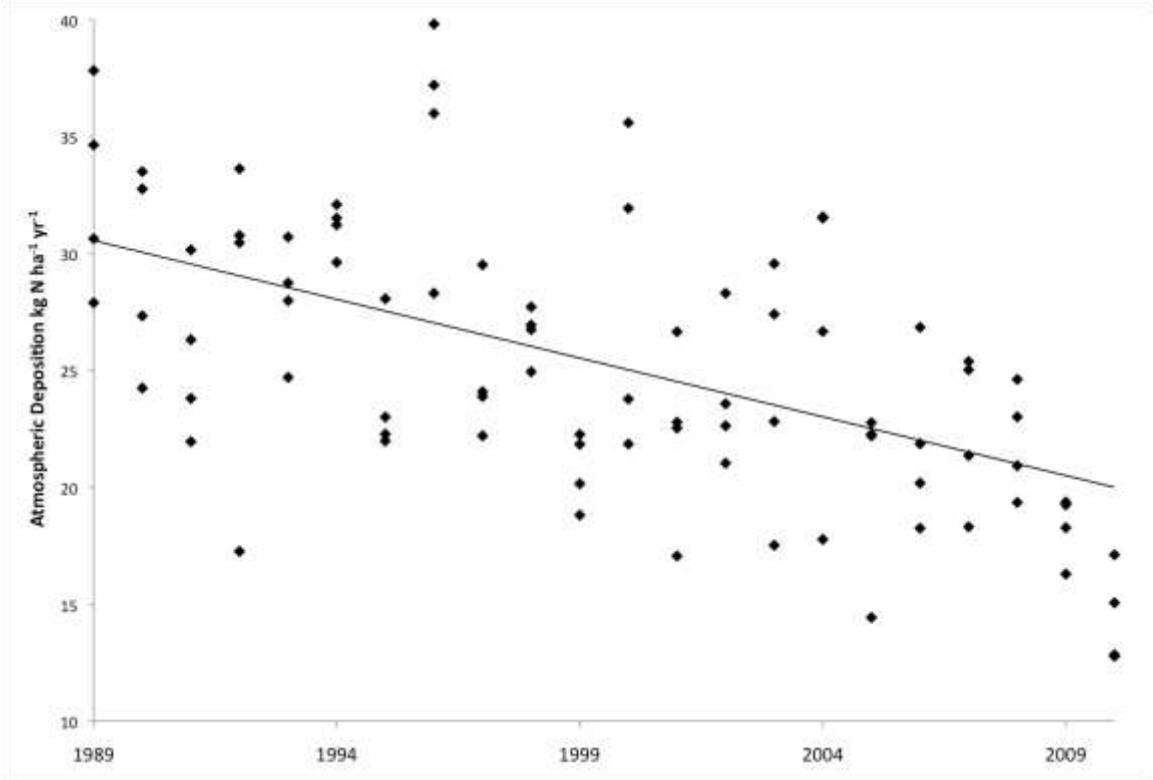


Table 3: Literature review of atmospheric deposition rates including this study. Not all studies included organic atmospheric deposition.

Source	Rate in kg^{-1} ha^{-1} yr	Location
Paerl 1993	2.8 - 14	Worldwide
Hu et al. 1998	6.64	Long Island, NY
Kinney and Valiela 2011	10	NE-NY area
Luo et al. 2002	9 - 23	CT coast
Bowen and Valiela 2001	12.26	MA
Valiela et al. 1997	15	Metadata analysis
This study	17.55	NY, CT, NJ
Peconic Estuary Program 2001	22	Peconic, NY

3.2.2 Wastewater

The contribution of nitrogen load to the bays from wastewater was calculated in the NLM by multiplying the nitrogen released per person by the housing occupancy rate and number of homes. Then differing levels of nitrogen were removed from this source depending upon the type of on-site sewage disposal system (septic or cesspool) and the system's distance from shore as there is significantly less nitrogen removed when septic tanks and cesspools are within 200 m of coastal waters.[19]

The average occupancy rate per home was determined from the 2010 Towns of Southampton and Brookhaven census results using data only for homes within the watershed study area. The occupancy rates of the owner-occupied homes and renter occupied homes were averaged to yield approximately three people per dwelling (with minor differences per watershed; Table 1). The seasonal population influx was determined by the Suffolk County Planning Department.[29] This estimated number of guests per year was divided by the number of houses. Since they are only seasonal visitors it was multiplied by 2/12ths (indicating an average stay of two months) to account for the fact that they are seasonal. This lead to an additional occupancy rate of less than 1 person per dwelling. Since the NLM model is based on annual averages, the timing of the occupancy would not alter the models.

Most residential and commercial uses have individual septic tank systems or cesspools, which differ in the fraction of nitrogen released to the underlying aquifer, with the less effective cesspools releasing about one-third more. In Suffolk County, a law was passed in 1973 requiring all newly constructed buildings to include a septic tank system instead of a cesspool. For this study, residential and commercial uses built before 1973 were assumed to have cesspools. The study area does not contain any municipal wastewater treatment facilities. There are several small, privately-owned sewage treatment facilities that were accounted for by removing the residential and commercial uses attached to the facility from the number of buildings within the NLM calculation and adding their NYS DEC State Pollutant Discharge Elimination System (SPDES) permitted nitrogen discharge amount as a "point source" that discharges to ground water (Table 4). Legacy duck farms were not included as a point source as there are no longer any in operation within the watersheds considered in this study. The remnant waste from this and other farms may remain in sediments and be a source of nitrogen as a benthic flux. Benthic fluxes were not considered in this study, however, as they represent a recycled rather than new nitrogen input that cannot be managed within the watershed.

Table 4: SPDES permitted nitrogen release amount from point sources in each subwatershed. Data from the private sewers Suffolk County GIS file.

Subwatershed	Permitted N load (kg N yr^{-1})
Moriches West	10,500
Moriches Middle	3,700
Moriches East	1,600
Quantuck Bay	2,800
Shinnecock Bay West	180
Shinnecock Bay East	0
Heady/Taylor Creek	1,100

The NLM breaks down the nitrogen removal in septic tank- and cesspool-based systems into four steps: removal in the tank, removal in leach fields, removal in septic plumes, and removal in the aquifer. Cesspools on Long Island are typicaly composed of cylinders arranged vertically, eliminating any traditional leach field and the associated nitrogen removal therein. Although there is a disposal pit associated with these vertically structured cesspools systems, it is unclear how much nitrogen is removed in this part of the system. Due to the lack of information, the cesspool removal percent was left at the default of 35%.

3.2.3 Fertilizer

The NLM considers fertilizer input from agricultural uses, golf courses, parks and athletic fields, and manicured lawns. The area of each land use was calculated using ArcGIS; an average lawn area was determined for each residential and commercial building. In 2009 Suffolk County passed a law limiting fertilizer use to the months between April and November and banning its use on County-owned property and parks. These parcels were removed from the parks and athletic fields category. Fertilizer application rates were obtained from three Long Island-based studies.[54,58,55] The residential rate used agrees with the fertilizer application standards suggested in Suffolk County's Healthy Lawns Clean Water program. The fertilizer application rate used for golf courses was the maximum allowed under the 2009 Suffolk County fertilizer legislation (Table 5).

Table 5: Average fertilizer application rates in Suffolk County as established by Cornell Cooperative Extension.

Fertilized Area	Application rates (kg N ha^{-1} yr^{-1})
Lawns	122
Parks and athletic fields	117
Agriculture	171
Golf courses	232

3.3 Volumetric Flux Model (VFM)

The Volumetric Flux Model (VFM) predicts nitrogen loads to the bays based on the volume of water that discharges from the watershed into the bay and the nitrogen concentrations in groundwater, streams, and runoff within the watershed. The VFM has been used successfully to predict nitrogen loads to several Long Island estuaries, bays, and harbors including West Neck Bay on Shelter Island, North Sea Harbor in Southampton, and Flanders Bay in Riverhead.[59,60,61] This model relies on the assumption that groundwater discharge to the bay is equal to the recharge of the aquifer[2]. In contrast to the NLM, which breaks down the nitrogen load into sources (i.e. wastewater v. fertilizer), the VFM differentiates nitrogen inputs by transport mechanism: stream flow, surface runoff, or groundwater flow.

Because the NLM and VLM determine nitrogen loading to estuaries via independent methods, they were used to complement each other as well as compare the accuracy of each model. The variance of the VFM model was determined to be 14% based on the mean relative standard deviation of the two primary factors used within the VFM, precipitation (19.7%) and nitrogen concentration (9%), noting that precipitation translates into the largest volumetric source of discharge, groundwater.

3.3.1 Groundwater

As described above, VLM divides freshwater flow from land to sea into stream flow, run-off, and groundwater flow. To determine the volume of groundwater that discharges into the Eastern Bays, subwatershed areas were multiplied by the annual average precipitation to obtain the volume of rain (Table 6). Annual precipitation on Long Island was ascertained by calculating the average amount of precipitation at the National Oceanic and Atmospheric Administration (NOAA) Long Island MacArthur Airport weather station, managed by the over the past decade (Table 6).

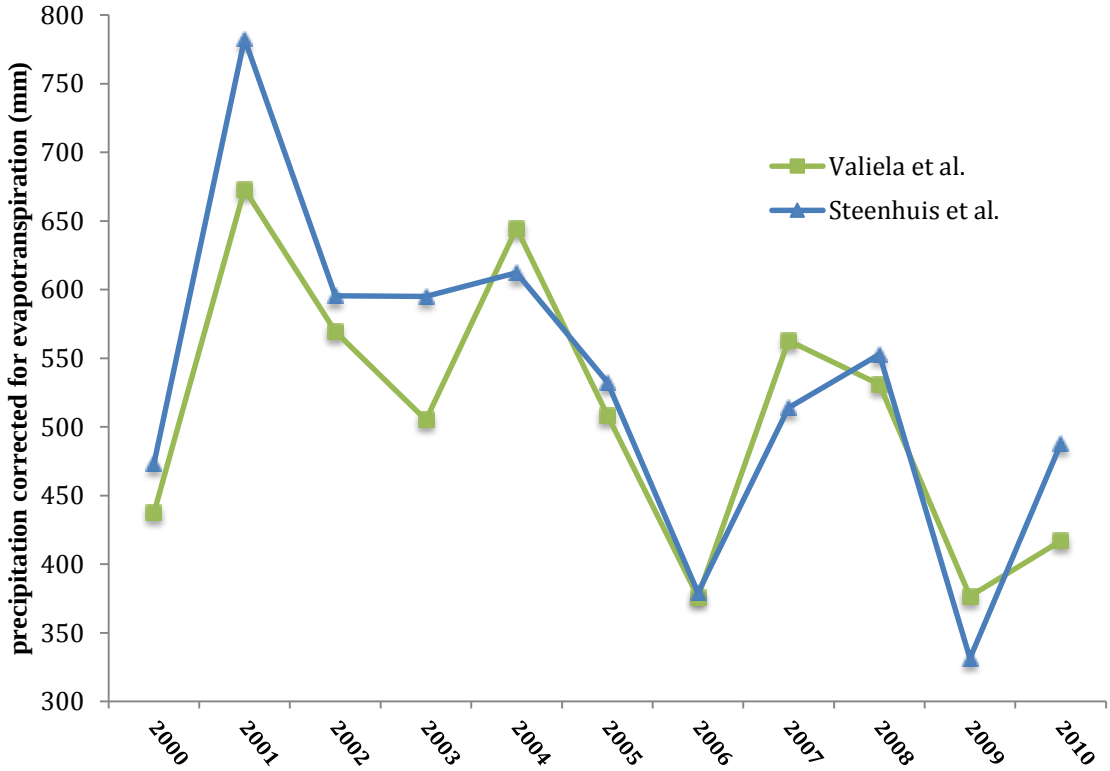
Table 6: Annual precipitation at Islip Airport weather station

Year	Precipitation (in yr^{-1})
2010	37.72
2009	58.03
2008	49.08
2007	43.55
2006	55.57
2005	43.84
2004	32.43
2003	48.52
2002	45.78
2001	32.47
2000	35.95
	Average = 43.90 (1.12 m)

The volume of rain was corrected for the volume of rainfall that comprises the stream flow, volume of runoff, and the fraction that does not recharge the aquifer (evapotranspiration percent). The recharge percent is the precipitation minus the evapotranspiration percent. The value used (45.67%) was the default provided by the meta-

analysis by Valiela et al.[33] However, to confirm that percentage, the results were compared to those of Steenhuis et al.[62] an eastern Long Island-based study that highlighted the strong seasonal nature of groundwater recharge in this region determining that the best measure of annual recharge percent is 75-90% of the precipitation from between 15 October and 15 May only. In some years the value determined by Valiela et al.[33] was slightly lower than the range given by Steenhuis et al.[60] and in some it was higher but the decadal averages were extremely similar (Figure 3).

Figure 3. Estimates of groundwater discharge using the methods of Valiela et al.[33] and Steenhuis et al.[60]



The resulting value for volume of groundwater was multiplied by groundwater nitrogen concentrations to determine nitrogen load to the bays. The Suffolk County Department of Health Services (SCDHS) regularly measures the nitrate, nitrite, and ammonium concentrations in hundreds of groundwater wells in the study area and provided measurements dating between 1990 and 2013. The nitrogen species (nitrate, nitrite, and ammonium) were summed for each given well sample and then averaged. In wells that showed an increasing trend in nitrogen concentration, only data between 2006 and 2013 was used. Additional groundwater well data was compiled from USGS wells' measurements between 1970 and 2006 and Suffolk County monitoring wells near the Forge River. All groundwater wells were shallow (< 30 m) and less than 4 miles from the shore and thus are assumed to contribute aquifer discharge to coastal waters. The groundwater nitrogen concentrations across all subwatersheds were interpolated and contoured using an inverse distance weighting (IDW) algorithm in ArcGIS permitting visual representation of the areas of the watershed that likely contribute the most nitrogen to the bays (see results).

The following equation summarizes the groundwater nitrogen load determined via the VFM: Groundwater nitrogen load (kg N yr^{--1}) = [(subwatershed area (m^2) x precipitation (m yr^{-1}) x recharge %) – stream flow volume (m^3 yr^{-1}) – runoff volume (m^3 yr^{-1})] * groundwater [N] (kg N m^{-3}).

3.3.2 Runoff

Most of the residences nearest to the Moriches, Shinnecock, and Quantuck Bays consists of older, larger homes that have little impervious cover. It was therefore assumed that most of the volume of runoff comes directly from the roads adjacent to the bays or through MS4s (Municipal Separate Storm Sewer Systems). The MS4 system is important because it brings stormwater that might otherwise drain off roads and seep into the groundwater through storm drains, pipes, and outfalls, directly into the coastal waters. In this study area, Brookhaven and Southampton Towns and Suffolk County have constructed MS4 structures. However, based on GIS files from the County and Town of Southampton showing locations of pipes and outfalls, the MS4 system in these subwatersheds is minimal. The average distance from a pipe to its outfall is 172 meters, so it has been assumed that all road runoff within 172 meters from shore could end up in the bay. As in the groundwater determination, the area of road in the runoff zone can be multiplied by the precipitation rate to determine the volume of the runoff. The volume of runoff was then multiplied by a nitrogen concentration of 0.00126 kg m^3 (measured stormwater nitrogen concentration[63]) to obtain the total nitrogen load contribution from runoff.

3.3.3 Streams

With the exception of the Forge River the streams that run into the Eastern Bays are small (<< 1 m^3 s^{-1}). The volume of precipitation that is captured in streamflow and discharged into the bays is not recharging groundwater, and thus was removed from the volume of groundwater discharging into the bays. Streamflow discharge was ascertained by field measurements, using a General Oceanics Mechanical Flowmeter to record velocity multiplied by the stream's measured width and depth. Water samples were collected from all freshwater creeks entering the study area's bays. At each stream, salinity was measured using a YSI85 sonde (Yellow Springs Inc®) to determine whether the stream was freshwater or a tidal creek. The latitude and longitude of each sampling location was recorded with a Garmin® GPS device. Water samples were collected by hand in 100 ml acid-and-distilled water-washed, polyethylene bottles that were rinsed and then filled with stream water. The samples were filtered with a 60ml polyethylene syringe coupled with a Swinnex filter holder holding a pre-combusted (2h at 450ºC) glass fiber filter (GFF, Pall®), then stored frozen until analysis. Filtered samples were colorimetrically analyzed for nitrate, nitrite, ammonium, and total nitrogen standard using wet chemistry and spectrophotometer methods.[64] Total nitrogen was used in the calculation of nitrogen load except in samples where the sum of the inorganic nutrients was greater than the total nitrogen. To enhance the representativeness of values, tributary streamflow volumes and nitrogen concentrations from the Forge River Nutrient Report[27] and the SCDHS Forge River water-quality monitoring program were also included when determining a mean nitrogen load for this tributary. Multiplying streamflow discharge by empirically measured nitrogen concentrations produced the annual nitrogen load from streams.

3.4 Further Modeling and Analyses

3.4.1 Nitrogen Load Comparisons

Once the nitrogen load was calculated for each subwatershed from both models (NLM and VFM) the values were compared. The resultant nitrogen load yields (nitrogen load divided by area of subwatershed), nitrogen loads per volume of estuary, sources of nitrogen from the NLM, and transport mechanisms from the VFM were compared on a subwatershed level. Finally, the nitrogen load to the bays was compared to other studies, including those using the NLM model, that have quantified the nitrogen load per area of estuary for different waterbodies: Great South Bay, NY;[19] Barnegat Bay, NJ;[34] Chincoteague Bay, VA;[22] and, West Falmouth Harbor and Pleasant Bay, MA,[65] among others.

3.4.2 Estuarine Loading Model (ELM)

Following the quantification of nitrogen load to the bays from the subwatersheds, an estuarine loading model was employed to determine the eutrophication vulnerability of the various bays. The Estuarine Loading Model (ELM[66]) as described in Bowen and Valiela (2004)[66] and Bowen et al. (2007),[34] is also available through the nitrogen-load modeling tool. The Estuarine Loading Model (ELM) calculates mean annual concentration of dissolved inorganic nitrogen (DIN) available to primary producers in shallow estuaries by considering how different processes modify pools of nitrogen provided by inputs and losses within components of the estuarine system.[66] The ELM, run on a subwatershed level, helped determine the sections of the bays with the highest predicted DIN concentrations. The ELM results were compared with empirically measured DIN concentrations provided by the SCDHS (1976-2010).[67] In some cases the results were favorable; see results on pages 32 and 33.

ELM is organized similarly to the NLM in that it is a web-based program requiring site-specific data for the bay. Some of the fields had default values[66] that were changed when more relevant or applicable data was available. Salt marsh area was calculated from the NY Department of State (DOS) GIS data available on the NY state GIS clearinghouse accessed in 2012. Eelgrass bed area was determined using DOS Submerged Aquatic Vegetation (SAV) GIS data. Coverage was broken down into patches that were continuous and discontinuous; the discontinuous patches were calculated at 50% of their area.

The depth of the Eastern Bays varies as ocean inlets in Moriches and Shinnecock Bays can be more than 10 meters deep, while much of the southern extent of the bays are less than 1 meter deep. The average depth for Quantuck Bay is 1.25 meter (Heerbrandt and Franson, unpublished 2003) and nearly all of Moriches and Shinnecock Bays are less than 2 meters deep. A mean depth of 1.25 meters was used for all bays for this study. Tidal range was available in most subwatersheds from NOAA (http://tidesandcurrents.noaa.gov/). Multiple tidal monitoring stations in Moriches and Shinnecock Bay were averaged per subwatershed. In two of the subwatersheds with no NOAA stations (Moriches East and Heady/Taylor Creek), the tidal ranges were estimated based on the adjacent stations and the distance to the nearest ocean inlet.

3.4.3 Evaluating Bay Water Quality Data and Eutrophication Vulnerability

The SCDHS has monitored bay water quality parameters at various locations since 1976. These data include total nitrogen, dissolved inorganic nitrogen, salinity, chlorophyll *a*, *A. anophagefferens*, secchi depth (April-October), and bottom dissolved oxygen (DO; April-October). In addition, Theresa Hattenrath-Lehmann who has quantified densities of *A. fundyense*, a toxic dinoflagellate, in the bays since 2008 using a molecular probe provided data[9] When averaging total nitrogen and DIN values for a monitoring station, all data points below their detection limit were used at half of the detection limit value. Water quality data was not available for the Heady/Taylor Creek section of the Bay but was estimated with data from adjacent sections of the Bay and Old Fort Pond, a similarly sized tidal tributary located 1 km northwest of Heady/Taylor Creek. The data sets were interpolated in ArcGIS using a standard Kriging algorithm to produce colored contour maps. DIN concentrations across all locations showed very little spatial variation and, thus, contouring this data set was not attempted.

Water residence times of the bays adjacent to each subwatershed were determined using a salt balance approach that assessed the bay volume, rates of freshwater flow, and the distribution of salinity across the estuarine region.[68] The following two equations were used to determine residence time in days: $tF = (f \times V) / R$ and $f = (SO - S) / SO$, where V equals the volume of the estuary (or section thereof), R equals the freshwater input, SO equals the salinity of the ocean water and S equals the salinity of the section of estuary.[68] The residence time for eastern Shinnecock Bay was modified to account for water flowing through the Shinnecock Canal from Great Peconic Bay. This influx (2×10^5 m^3 day^{-1})[69] was subtracted from the volume of the bay. Additionally the salinity within this basin was corrected for the salinity of the water entering from the Great Peconic Bay using the long term mean salinity data from Suffolk County, canal flow rates,[69] and known volumes of the basin. The extent to which marine data parameters were correlated to each other as well as with nitrogen loading rates and residence times was evaluated via a Spearman's rank order correlation matrix using SigmaStat within SigmaPlot 11.0. Flushing time and water quality were used to rank the subwatersheds into three qualitative tiers of eutrophication vulnerability based on relative differences in their flushing time (longer flushing times increases vulnerability) and nitrogen loads (larger nitrogen loads increase vulnerability; see Table 15; page 50).

3.5 Nitrogen Management Options

Nitrogen mitigation scenarios were assessed by making changes to the NLM. For example, tertiary sewage treatment plant facilities remove 93% of the nitrogen entering the plant[36] hence models were run reducing the wastewater water contribution by this amount and the resultant change in total nitrogen loading was determined. In sewage treatment plants with an ocean outfall, 100% of the nitrogen contributions from residential and commercial uses in a subwatershed were removed and the resultant change in nitrogen loading was determined. The large proportion of residential and commercial uses in this study site with cesspools (about 50%) were upgraded in the model to conventional septic tank systems by changing the percentage of residential and commercial uses with cesspools or to denitrifying systems by changing the default value of nitrogen-removal percentage (35%)[2] to an average percent nitrogen-removal for denitrifying systems (68%; Table 7).[70]

Table 7: Adapted from Maryland Department of the Environment Bay Restoration Fund best available technology for removing nitrogen from on-site septic systems.

Manufacturer	Field Verified Mean Effluent Concentration (mg L^{-1})
Advantex-RT	14
Advantex-AX20	17
Hoot BNR	21
RetroFAST	25
SeptiTech	27
Singulair TNT & Green	27
	Average = 20 or 68% Nitrogen removal

Septic systems and cesspools located closer to shore (200 m) are likely to release even more nitrogen into the bays in a very short period of time (< 2 years) as the wastewater effluent does not have time to recharge the aquifer and go through the ensuing denitrification process, before being transported through groundwater into the bay. Therefore, the nitrogen load effect of upgrading cesspools to septic tank systems closest to the shore was calculated. Finally, the model was run upgrading septic systems to denitrifying systems within high-density residential areas. The amount of fertilizer applied to lawns, agriculture, golf courses or parks and athletic fields was reduced in NLM to assess how this might alter nitrogen loading to the bay. Finally, 'Build out' scenarios were assessed by adding homes of differing lot-sizes to undeveloped areas and the change in nitrogen load to the subwatersheds was determined. The amount of undeveloped land per subwatershed was provided by the Town of Southampton and a Suffolk County report for the Town of Brookhaven.[71] The area of imperviousness value was updated, subtracting the area from naturally vegetated areas and adding it to medium or low density residential. The percent of cesspools was proportionally decreased to reflect that the new homes would have septic systems not cesspools. To expedite the process of running the NLM, a spreadsheet version of the NLM was created that was capable of running these scenarios within all of the subwatersheds at once.

3.6 Determining the Relative Priority of Remedial Measures

Consideration of priority areas for remediation were made by considering the eutrophication vulnerability, coupled with the efficacy of differing the nitrogen mitigation scenario. For example, if an area was particularly vulnerable to eutrophication but the nitrogen mitigation scenarios only removed a small percent of nitrogen, it would be a lower priority area. Conversely, if nitrogen mitigation in an area was particularly effective but that area of the bay is not vulnerable to eutrophication because it has a shorter residence time, that too, would be a lower priority for remediation.

4. Results

4.1 Nitrogen Loading Model (NLM)

The total nitrogen loads from the NLM modeling were: Moriches Bay 337,000 kg N yr-1, Quantuck Bay 20,600 kg N yr-1, and Shinnecock Bay 132,000 kg N yr^{-1}.Nitrogen

loads varied greatly over the subwatersheds of the bays (Table 8). The largest nitrogen load came from the Moriches West Subwatershed with 182,000 kg N yr^{-1} and the smallest nitrogen load was produced by the Heady/Taylor Creek subwatershed with 17,500 kg N yr^{-1}. Population density, land use, and land area can all influence the nitrogen load.

To best compare the nitrogen loading from the different subwatersheds, area-specific loading rates were quantified (kg N per ha of surface area). Moriches West Subwatershed had the largest yield with 31.7 kg N ha^{-1} yr^{-1}. Quantuck Bay Subwatershed had the smallest yield at 8.9 kg N ha^{-1} yr^{-1} (Fig 4).

Figure 4. Nitrogen load yield per subwatershed (kg N ha-1 yr-1) as determined by the NLM.

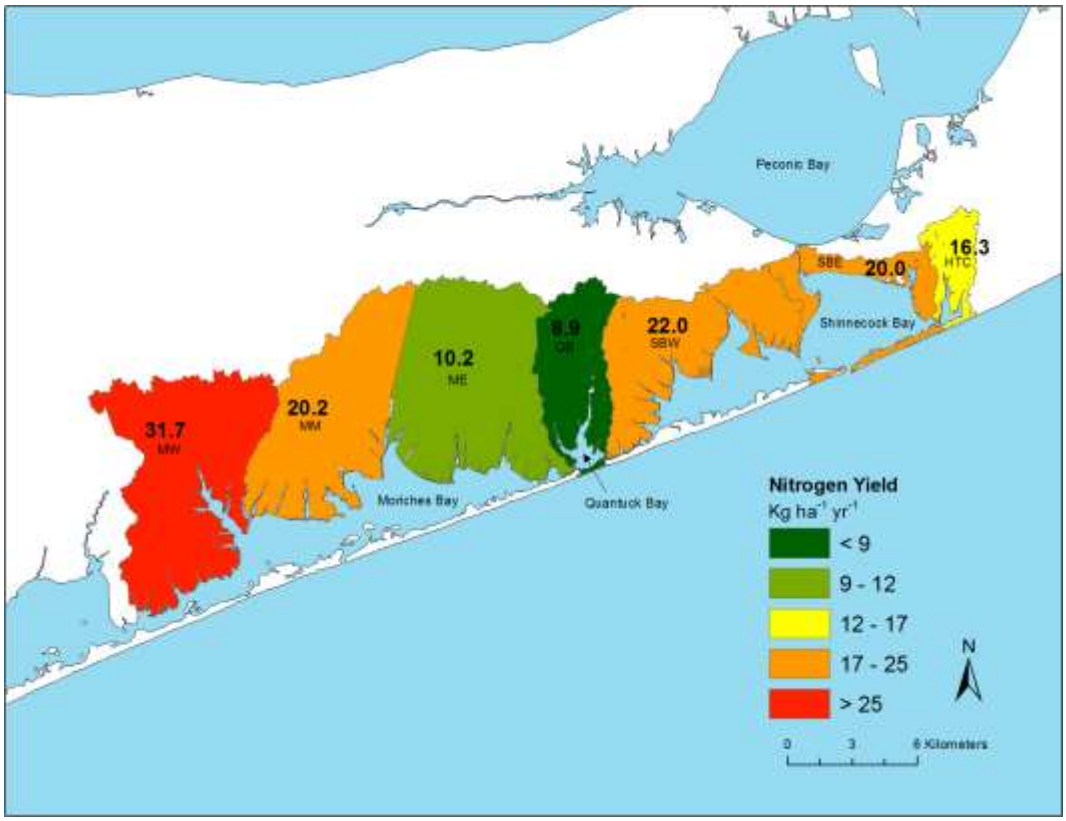


Table 8: Nitrogen loading results in kg N yr^{-1} from the nitrogen loading model (NLM) and volumetric flux model (VFM). These values include point sources (NLM only) and subtracting for groundwater underflow but do not include direct atmospheric deposition to the bays.

Subwatershed	NLM	VFM
Moriches West (MW)	182,000	133,000
Moriches Middle (MM)	96,400	99,800
Moriches East (ME)	58,800	77,800
Quantuck Bay (QB)	20,600	27,400
Shinnecock Bay West (SBW)	88,900	65,100
Shinnecock Bay East (SBE)	26,500	16,400
Heady/Taylor Creek (THC)	17,500	13,900

While the relative contribution of each nitrogen source varied across the subwatersheds, wastewater was consistently the largest nitrogen source for all subwatersheds. Exploring the diversity across the watersheds: Quantuck Bay had the highest percentage of nitrogen loading from atmospheric deposition to the land (32%); Moriches West Subwatershed had the highest percentage of nitrogen loading from wastewater water (76%); and the Moriches Middle Subwatershed had the highest percentage of nitrogen loading from fertilizer (31%; Fig 5).

The fertilizer nitrogen load was further segmented into agricultural uses, lawns, golf courses, and parks and athletic fields. The percent contribution by each sub-constituent of the fertilizer category was variable by subwatershed, but lawns and agriculture were the primary or secondary fertilizer contributor to all subwatersheds (Fig 6).

Throughout the entire study site, the relative total nitrogen loads from land contributions were: wastewater 65%, fertilizer 20%, and atmospheric deposition 15% (Fig 6). Adding atmospheric deposition directly to the bays changed the percentages thus: wastewater contributed 51% of the nitrogen load, direct atmospheric deposition to the water contributed 21%, fertilizer contributed 16%, and atmospheric deposition to the land contributed 12%.

Figure 5. Nitrogen load by source (NLM). Direct Atmospheric Deposition refers to atmospheric deposition to the bays and Atmospheric Deposition represents atmospheric deposition to the subwatershed (land). MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.

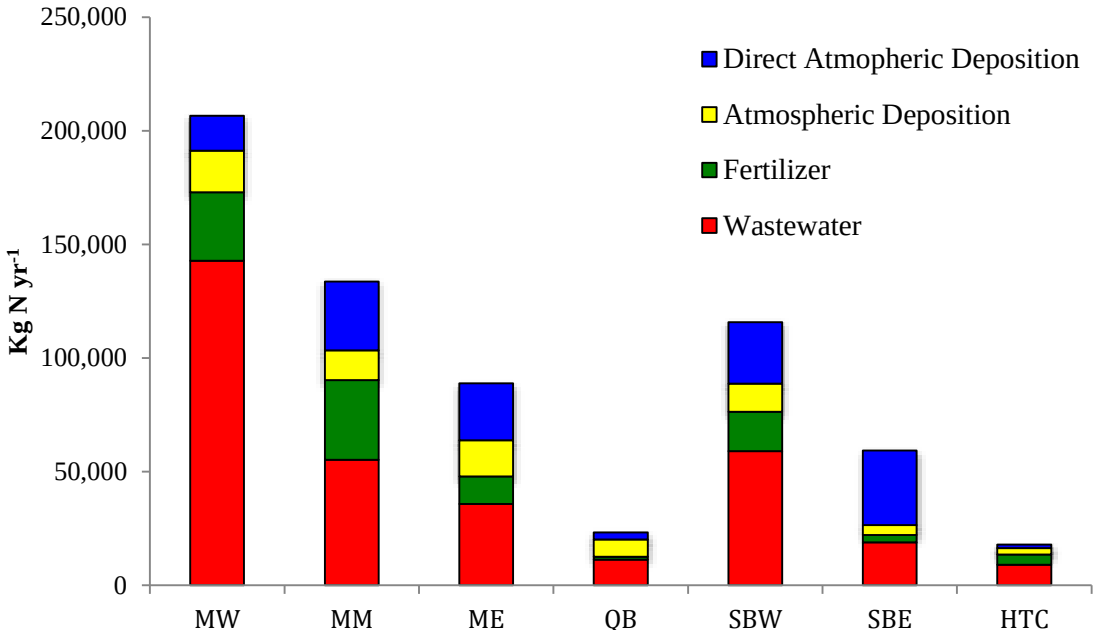
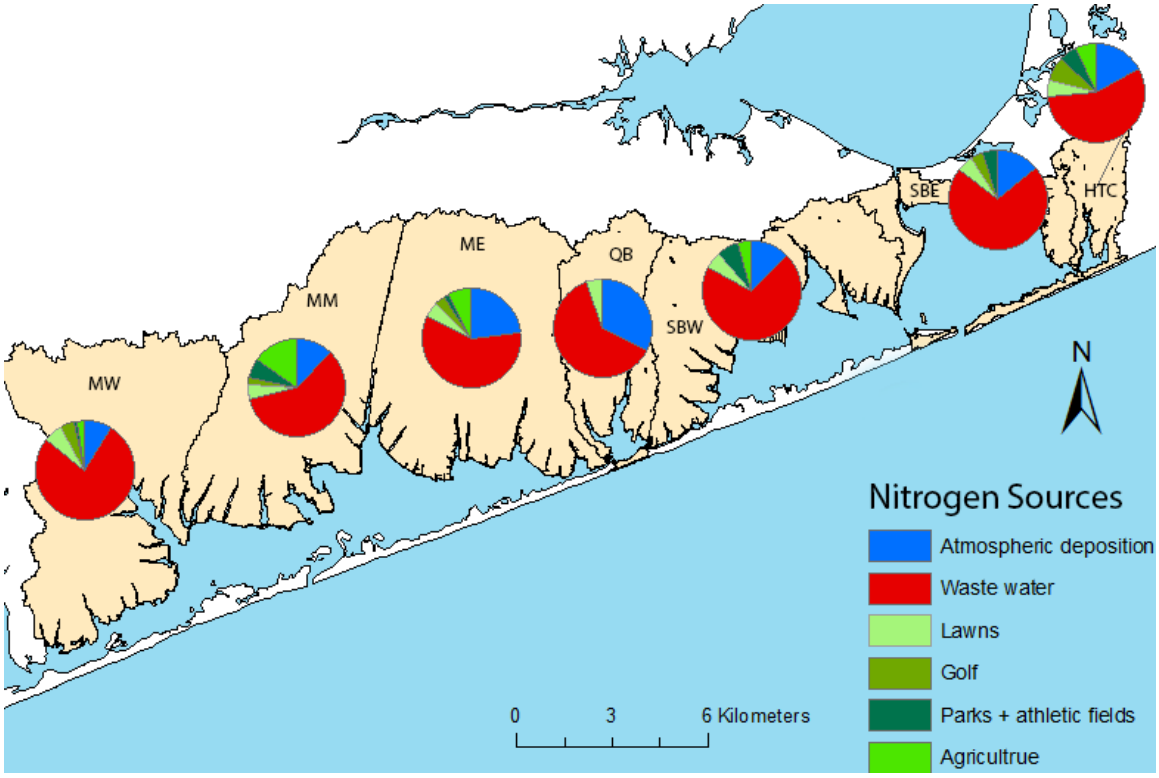


Figure 6. Relative contribution of all atmospheric deposition (to land and bays), wastewater and fertilizer to the total nitrogen load to subwatersheds with fertilizer sources broken down into lawns, golf courses, parks, and agricultural uses.



4.2 Volumetric Flux Model (VFM)

The VFM predicted that the largest nitrogen load came from the Moriches West Subwatershed with 133,000 kg N yr^{-1} and the smallest nitrogen load was produced by the Heady/Taylor Creek subwatershed with 13,900 kg N yr^{-1}. The total nitrogen loads in the VFM model were: Moriches Bay 310,600 kg N yr-1, Quantuck Bay 27,400 kg N yr-1 and Shinnecock Bay 95,400 kg N yr^{-1} (Table 8).

The VFM was used to divide nitrogen sources between groundwater, runoff, and streams. Mean groundwater nitrogen concentrations per subwatershed ranged from 2.4 mg L^{-1} in Shinnecock Bay East Subwatershed to 4.8 mg L^{-1} in Moriches West Subwatershed and Moriches Middle Subwatershed (Table 9). An interpolation of all groundwater nitrogen levels illustrated the widespread, high levels of nitrogen in groundwater in the Moriches West and Moriches Middle Subwatersheds, as well as regional 'hot spots' of high nitrogen levels in groundwater underlying other subwatersheds (Figure 7). Groundwater was responsible for over 90% of the nitrogen load contribution in all subwatersheds save Moriches West Subwatershed where it contributed 76% and streams contributed 23% (Figure 8). The volume of runoff ranged from 343,000 m^3 yr^{-1} in Shinnecock Bay East Subwatershed to 607,000 m^3 yr^{-1} in Moriches West Subwatershed but overall contributed less than 3% of freshwater flow in all subwatersheds. Stream discharge rates ranged from 36,900 to 6,020,000 m^3 yr^{-1} and average stream nitrogen

concentrations ranged from 0.139 to 6.09 mg L^{-1}. Many of the streams sampled had either low flow rates or low nitrogen concentrations (Table 10) and streams were a substantial nitrogen source (>5%) to the Moriches West (23%) and Moriches East (7%) subwatersheds only.

Table 9: Average groundwater nitrogen concentrations by subwatershed.

Subwatershed	Average GW N (mg L^{-1})
Moriches West	4.83
Moriches Middle	4.83
Moriches East	2.81
Quantuck Bay	2.78
Shinnecock Bay West	3.07
Shinnecock Bay East	2.35
Heady/Taylor Creek	2.96

Figure 7. Interpolation of total nitrogen concentrations from groundwater wells.

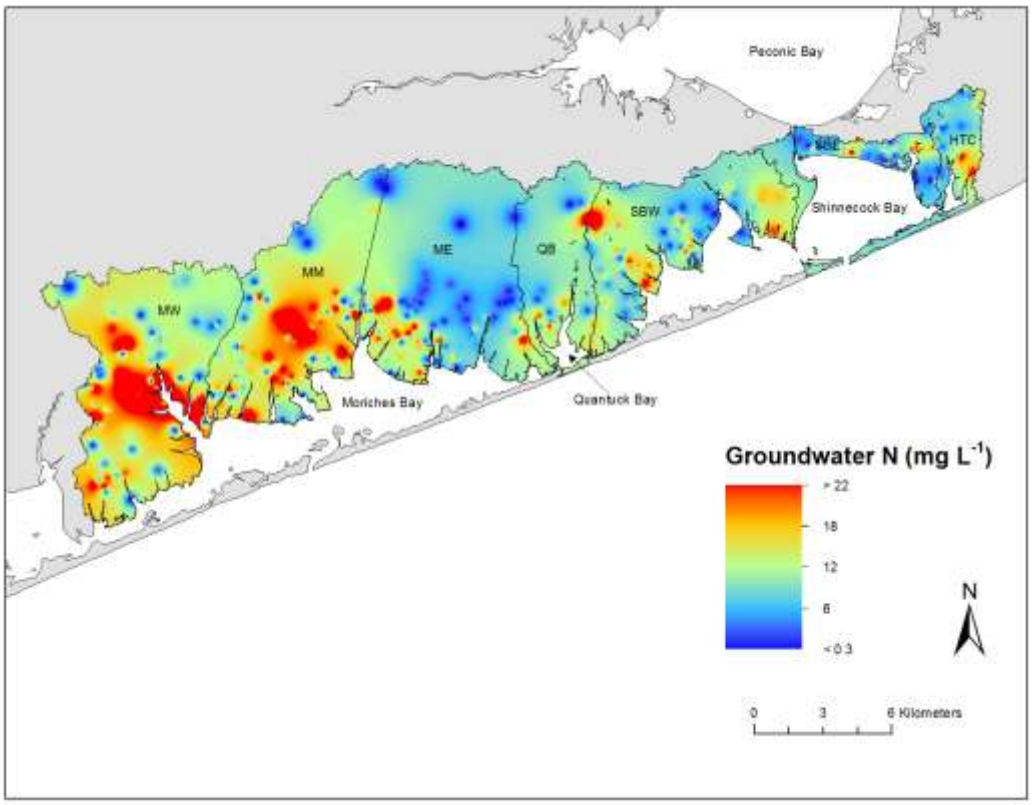


Figure 8. Nitrogen load of subwatersheds by source from the VFM. MW-Moriches West Subwatershed, MM-Moriches Middle, ME-Moriches East , QB-Quantuck Bay, SBW-Shinnecock Bay West, SBE-Shinnecock Bay East , HTC-Heady/Taylor Creek.

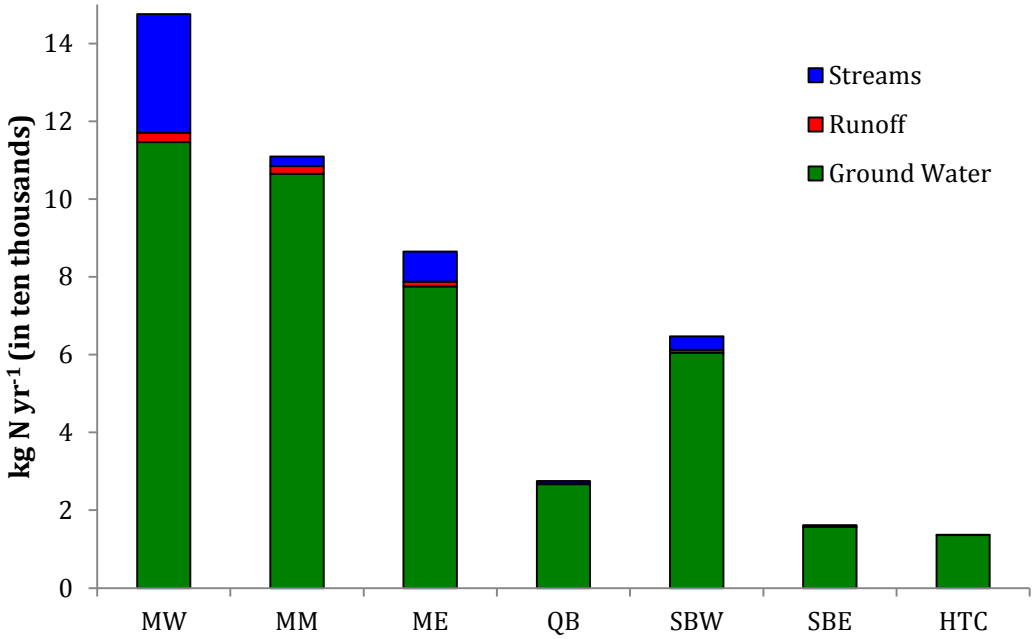


Table 10: Average volume, nitrogen concentration, and nitrogen loads from individual streams.

Stream Name	Subwatershed	Avg. Volume (m^3 yr^{-1})	Avg. N Concentration (kg m^{-3})	Av. N Concentration (mg/L)	N load (kg N yr^{-1})
Swift Creek	MW	1,790,000	0.006	6.1	10,900
East Millpond	MW	2,570,000	0.002	2.4	6,220
Forge River	MW	3,800,000	0.001	1.1	4,260
Seatuck Creek	ME	2,050,000	0.002	1.7	3,490
Lawrence River	MW	1,870,000	0.002	1.6	2,970
East River	ME	2,460,000	0.001	1.1	2,740
Terrell River	MM	6,020,000	0.000	0.4	2,310
Pattersquash Creek	MW	788,000	0.003	2.5	1,980
Poospatuck Creek	MW	357,000	0.004	4.5	1,600
Old Neck Creek	MW	1,741,000	0.001	0.9	1,550
Dave's Creek	SBW	488,000	0.003	3.0	1,460
Ely Creek	MW	439,000	0.002	2.2	969
Speonk River	ME	1,030,000	0.001	0.9	944
Philips Creek	SBW	1,160,000	0.001	0.7	823
Stone Creek	SNW	487,000	0.001	1.4	658
Weesuck Creek	SBW	282,000	0.002	1.8	501
Beaverdam Creek	ME	201,000	0.002	2.5	497

Quantuck Creek	QB	3,100,000	0.000	0.1	432
Aspatuck Creek	QB	108,000	0.001	1.1	116
Little Seatuck	MM	95,000	0.001	1.2	116
Wills Creek	MW	13,000	0.004	3.5	46
Unnamed (MM2)	MM	37,000	0.000	0.2	6

4.3 Comparing Models

For this research, two nitrogen loading models that are based on two different sets of assumptions and data sources were used to compare and cross-check the validity of each model. Overall, the two models produced similar nitrogen loading results for each subwatershed with differences between the models varying by 3 – 38% (Fig 9). There was a significant correlation between the amount of nitrogen predicted across the subwatersheds via the two models ($p<0.001$) and neither model was consistently lower or higher than the other (Fig 10). In the Moriches Middle Subwatershed the nitrogen load determined by NLM and VFM were nearly identical (103,000 v 111,000 kg N yr^{-1}; Fig 9). In contrast, the discrepancy was high (38%) in the Shinnecock Bay East Subwatershed.

Figure 9. Nitrogen loading results by subwatershed from both models. Error bars indicate the standard error of each model. The blue line indicates where both models produce the same result.

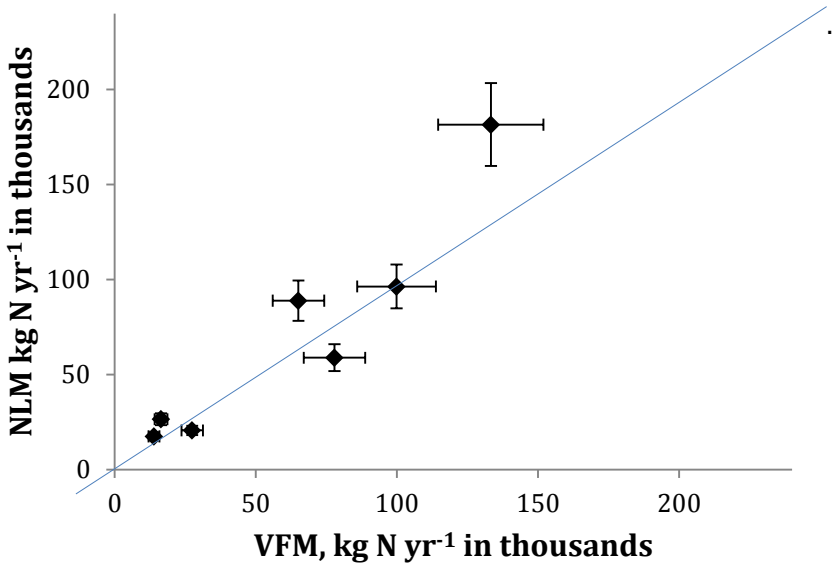
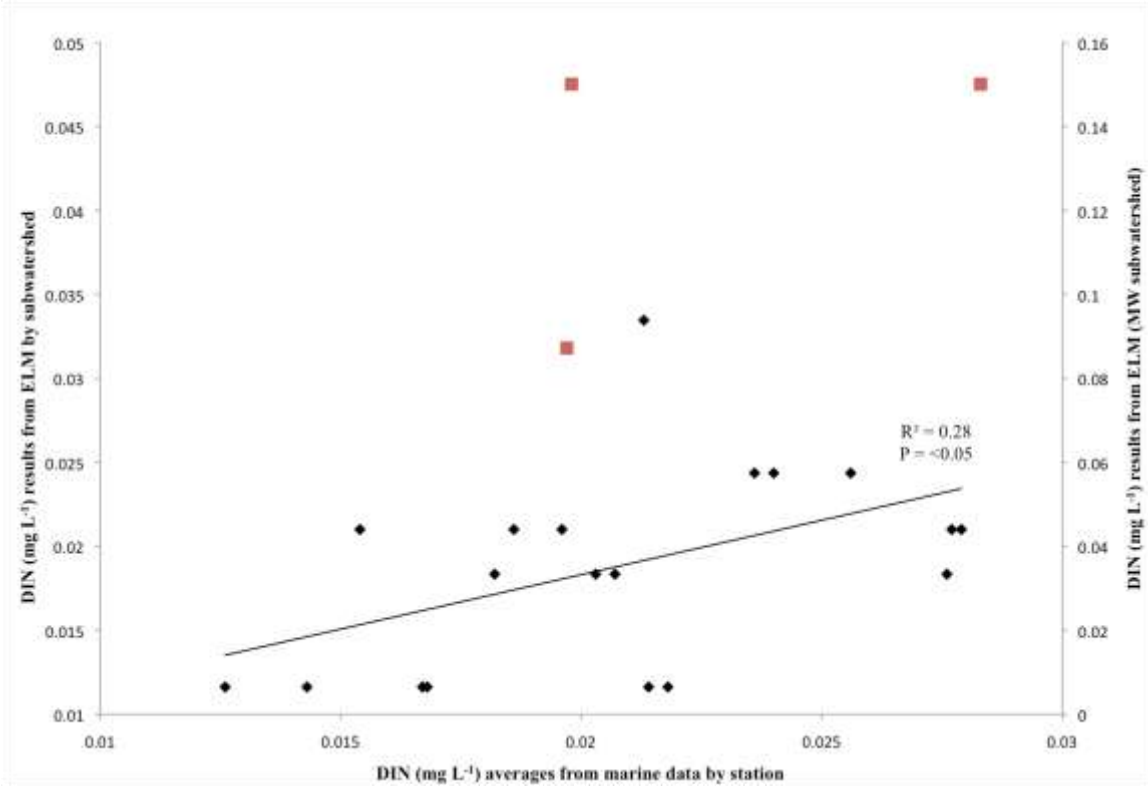


Figure 10. Results from the Estuarine Loading Model.[66] Red squares from the Moriches West Subwatershed are plotted on the left axis.



Direct atmospheric deposition to the bays, added to both models, was proportional to the size of the surface area of the bay segmented by subwatershed. Nitrogen load from direct atmospheric deposition ranged from 1,500 kg ha-1 yr-1 in Heady/Taylor Creek to 32,800 kg ha^{-1} yr^{-1} in Shinnecock Bay East. Because of the large size of the bay in the Shinnecock Bay East section, direct atmospheric deposition to the bay became the largest contributor to nitrogen load for this subwatershed.

4.4 Estuarine Loading Model (ELM)

For most of the bays, the ELM produced results that were similar to the measured dissolved inorganic nitrogen (DIN) concentrations within the bays by Suffolk County Department of Health Services. ELM's DIN concentrations ranged from 0.012 mg L^{-1} in Shinnecock Bay East Subwatershed to 0.15 mg L^{-1} in Moriches West Subwatershed with an average of 0.046 mg L^{-1}. With the exception of the Moriches West Subwatershed, the ELM prediction was within 0.01 mg L^{-1} of measured values and the correlation between the two data sets was statistically significant ($p<0.05$). However, the ELM predictions for marine water surrounding the Moriches West Subwatershed were more than 80% greater than measured values (Fig 10), suggesting the ELM is likely not a good fit for estuaries experiencing extreme nitrogen loads of the magnitude of this subwatershed. Such highly eutrophic systems can be highly dynamic and have large populations of both micro- and macro-algae that rapidly assimilate inorganic nitrogen delivered to these systems. Since

the ELM model does not account for nitrogen assimilation by macroalgae, and macroalgae grow densely within the estuarine waters of the Moriches West Subwatershed, these algae may partly account for differences in the modeled and actual DIN concentrations. Importantly, DIN is typically a small fraction of total nitrogen (TN)[66]. ELM is built to describe DIN only since it is the most biologically available form of TN. Concentrations of TN are typically five-to-ten-fold greater than TN values[66]. For many estuaries, such as the Peconic Estuary, water quality standards are made in terms of TN since they integrate both what is available (DIN) and what is already in plankton.

4.5 Marine Data

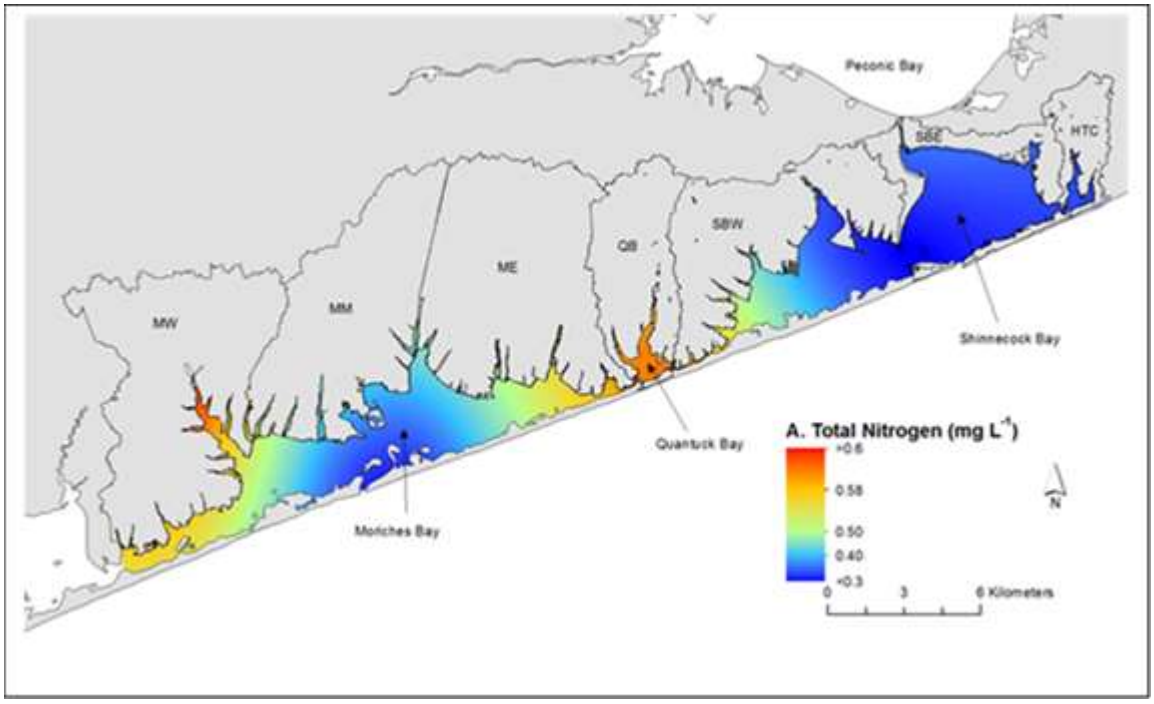
Mean salinity in the bays ranged from 26 - 31 PSU. Not surprisingly, the highest salinity was found near the inlets and the lowest was found in the Forge River (Moriches West Subwatershed) and Quantuck Bay. Total nitrogen ranged from 0.28 mg L^{-1} in the middle of Shinnecock Bay East and Shinnecock Bay West, to 0.58 mg L^{-1} in the Forge River and Quantuck Bay. Low bottom dissolved oxygen (DO) can be an indicator of eutrophication occurring within the waterbody. Seasonal (April – November) DO levels ranged from 6.5 mg L^{-1} in Moriches East to 8.5 mg L^{-1} near both inlets. Secchi depth is a measurement of clarity of the water; low Secchi depths may indicate higher phytoplankton biomass (more chlorophyll *a*) in the water column.[72] Seasonal (April – November) secchi depths ranged from 1 meter in Quantuck Bay to 3.1 m near the Shinnecock Bay Inlet. Chlorophyll *a* ranged from 3.2 µg L^{-1} near the Shinnecock Canal and in the middle of Shinnecock Bay East to 16.5 µg L^{-1} in Quantuck Bay. Mean densities of *A. anophagefferens* ranged from 6,800 cells mL^{-1} near the Shinnecock Canal to 200,000 cells mL^{-1} in Moriches East. Bloom densities were also very high in Quantuck Bay (200,000 cells mL^{-1}). Maximum *A. fundyense* bloom densities were the lowest in the Shinnecock Bay East and the highest in Weesuck Creek (Shinnecock Bay West). Dissolved inorganic nitrogen ranged from 0.013 mg L^{-1} near the Shinnecock Canal to 0.028 mg L^{-1} in the Forge River. Marine Data is summarized by bay in Table 11.

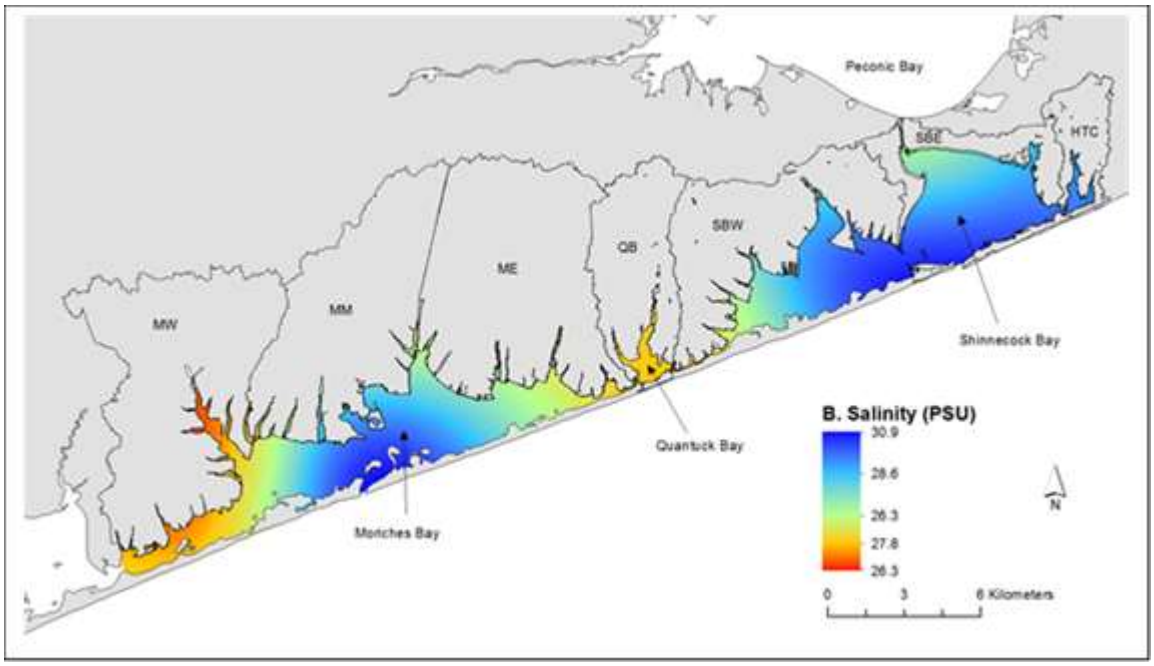
Table 11: Marine Data averages by bay for salinity, Chlorophyll *A*, total nitrogen, dissolved inorganic nitrogen, *Aureococcus anophagefferens*, dissolved oxygen, Secchi depth, and *Alexandrium fundyense*. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.

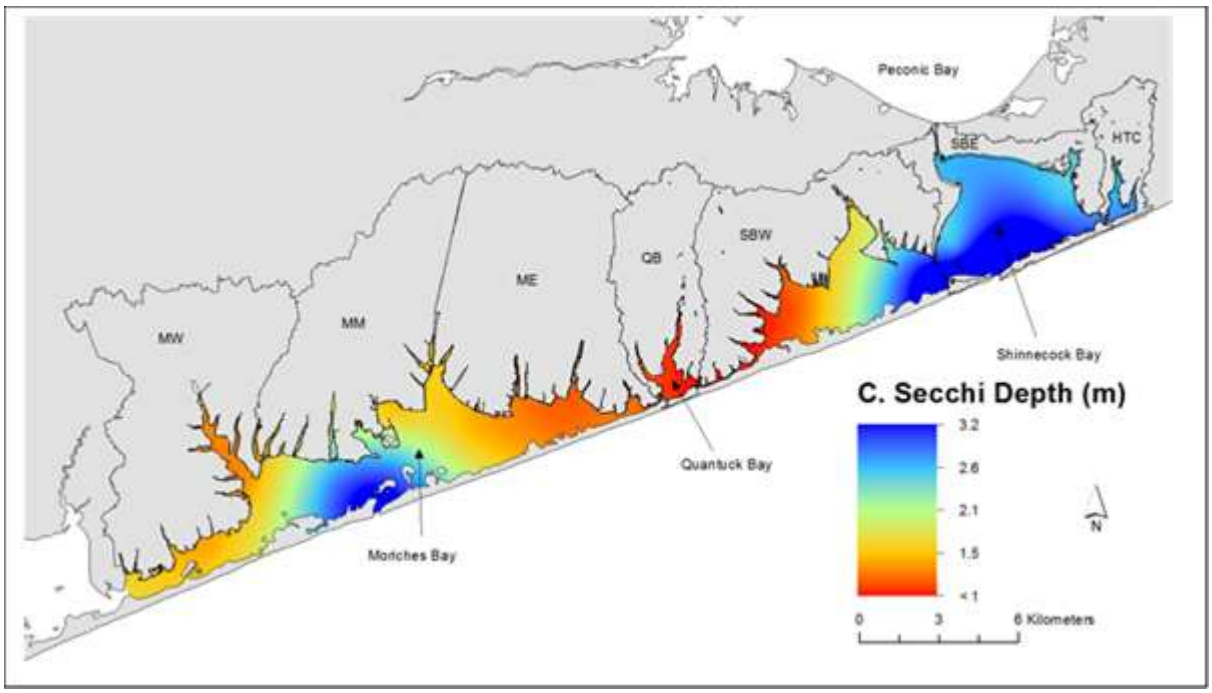
	Salinity (PSU)	**Chl. *a* (µg L^{-1})**	**Total Nitrogen (mg L^{-1})**	**Dissolved Inorganic Nitrogen (mg L^{-1})**	***A. anophagefferens* (cells mL^{-1})**	**DO (mg L^{-1})**	**Secchi Depth (m)**	***A. fundyense* (Log Max. cells)**
MW	27.3	8.93	0.525	0.023	26,000	7.67	1.44	4.04
MM	30.0	5.16	0.359	0.024	24,000	8.18	2.14	1.07
ME	28.4	12.33	0.490	0.022	129,000	7.10	1.32	1.77
QB	27.0	16.50	0.580	0.021	155,000	7.00	1.07	3.28
SBW	29.6	8.68	0.398	0.022	67,000	7.88	1.64	2.96
SBE	29.8	3.42	0.318	0.017	8,310	8.13	2.45	1.13
HTC	29.8	9.61	0.382	0.027	8,480	5.82	1.36	2.62

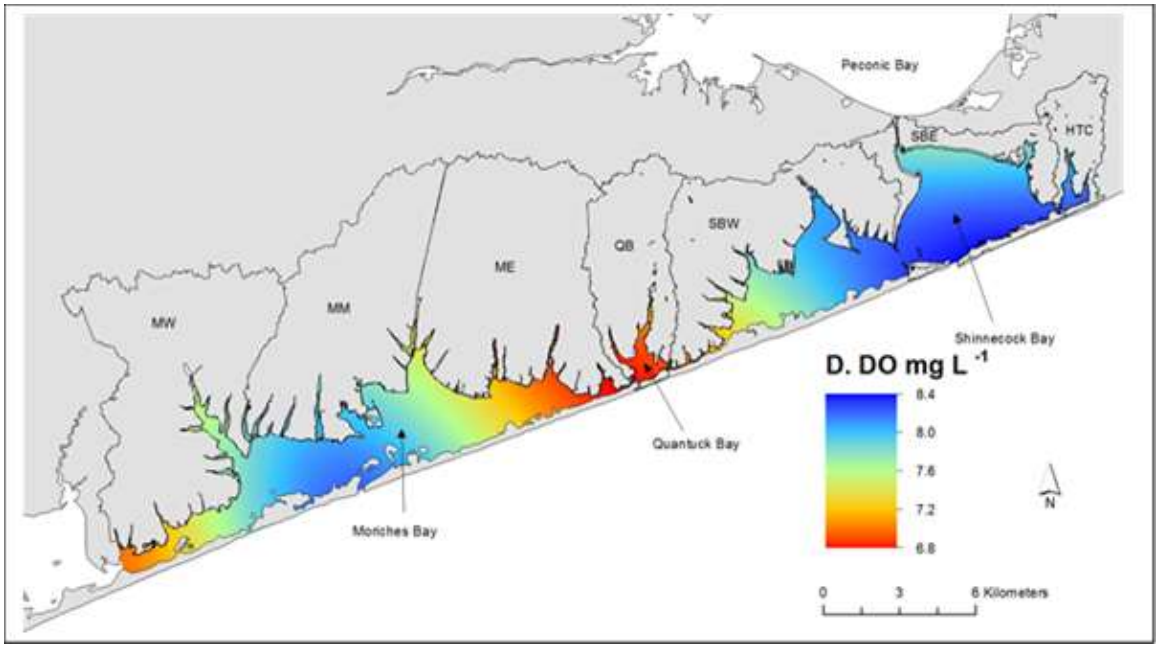
Contouring the marine data produced recognizable spatial patterns across the study area (Figure 11). Quantuck Bay and the Forge River, part of the Moriches West Subwatershed, stand out as the areas of the bays with the poorest water quality including signs of eutrophication (low DO and Secchi depth, high total nitrogen and chlorophyll *a*) and harmful algal blooms (high *A. anophagefferens* and *A. fundyense* densities). In contrast, bay areas near the ocean inlets had water with high DO, salinity, and water clarity (deep Secchi disc depth) and low total nitrogen, chlorophyll *a*, and harmful algae. Residence times of the bays ranged from 9 days for the Moriches Middle and Shinnecock Bay East areas near the ocean inlets to 26 days in Quantuck Bay (Figure 12). The mean residence time was 15 days. The bay areas with the poorest water quality had the longest residence times at 21 (Moriches West) and 26 (Quantuck Bay) days.

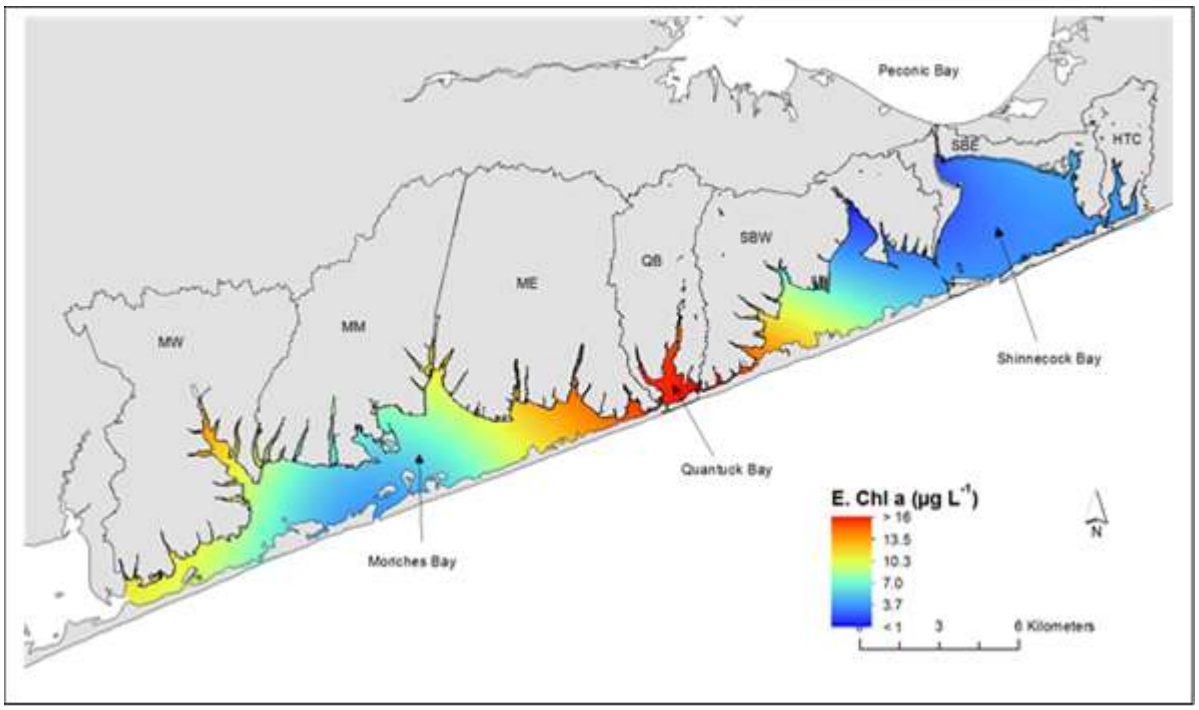
Figure 11. Contoured marine data from SCDHS including a. Total nitrogen b. Salinity c. Secchi depth d. Bottom dissolved oxygen e. Chlorophyll a f. *Aureococcus anophagefferens* g. *Alexandrium fundyense*.






Peconic Bay
SBE
HTC
SBW
QB
ME
MM
MW
Shinnecock Bay
C. Secchi Depth (m)
3.2
2.6
2.1
1.5
< 1
Quantuck Bay
Moriches Bay
N
0
3
6 Kilometers


Peconic Bay
SBE
HTC
SBW
QB
ME
MM
MW
Shinnecock Bay
D. DO mg L -1
8.4
8.0
7.6
7.2
6.8
Quantuck Bay
Moriches Bay
N
0
3
6 Kilometers


Peconic Bay
SBE
HTC
SBW
QB
ME
MM
MW
Shinnecock Bay
Quantuck Bay
Moriches Bay
E. Chl a (µg L-1)
> 16
13.5
10.3
7.0
3.7
< 1
3
6 Kilometers
N

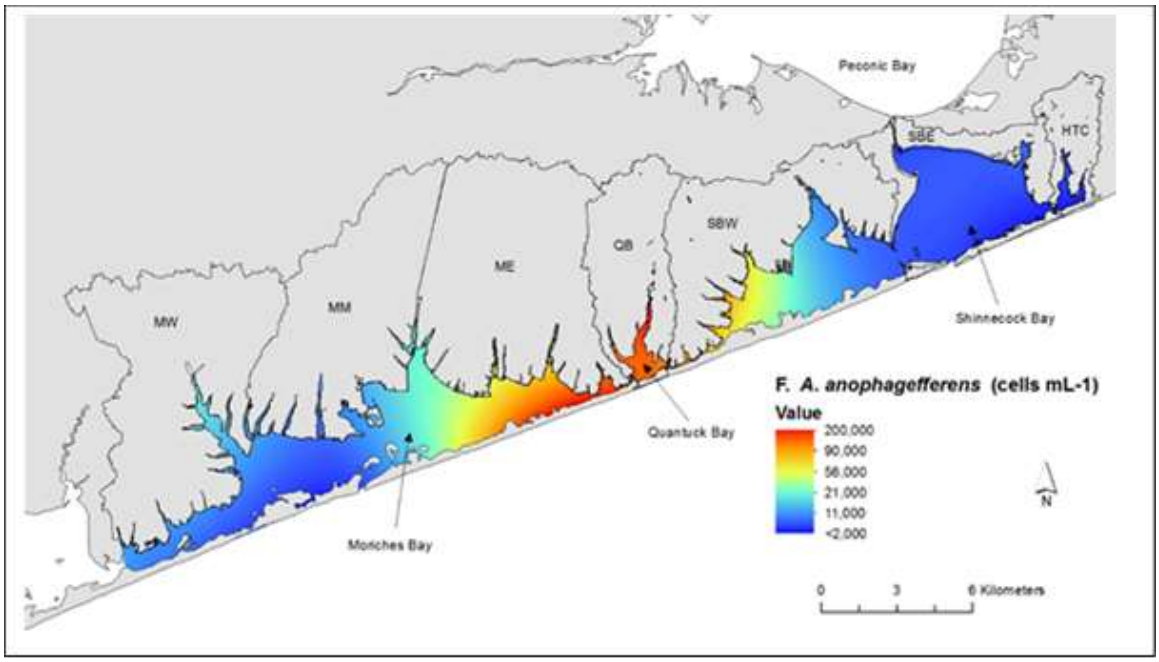
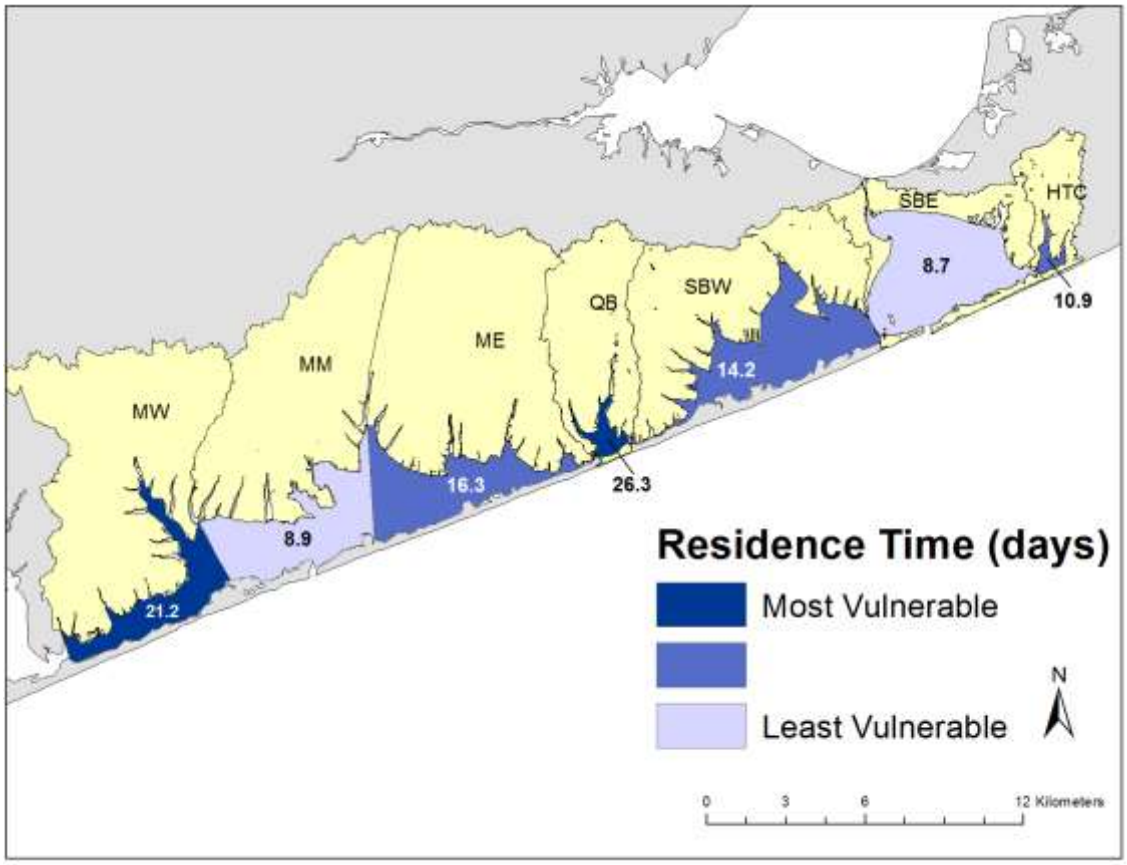

Peconic Bay
SBE
HTC
SBW
QB
ME
MM
MW
Shinnecock Bay
Quantuck Bay
Moriches Bay
F. A. anophagefferens (cells mL-1)
Value
200,000
90,000
56,000
21,000
11,000
<2,000
0
3
6 Kilometers
N

Figure 12. Flushing times for each section of Moriches, Quantuck and Shinnecock Bays. The three tiers of eutrophication vulnerability are indicated by color. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.



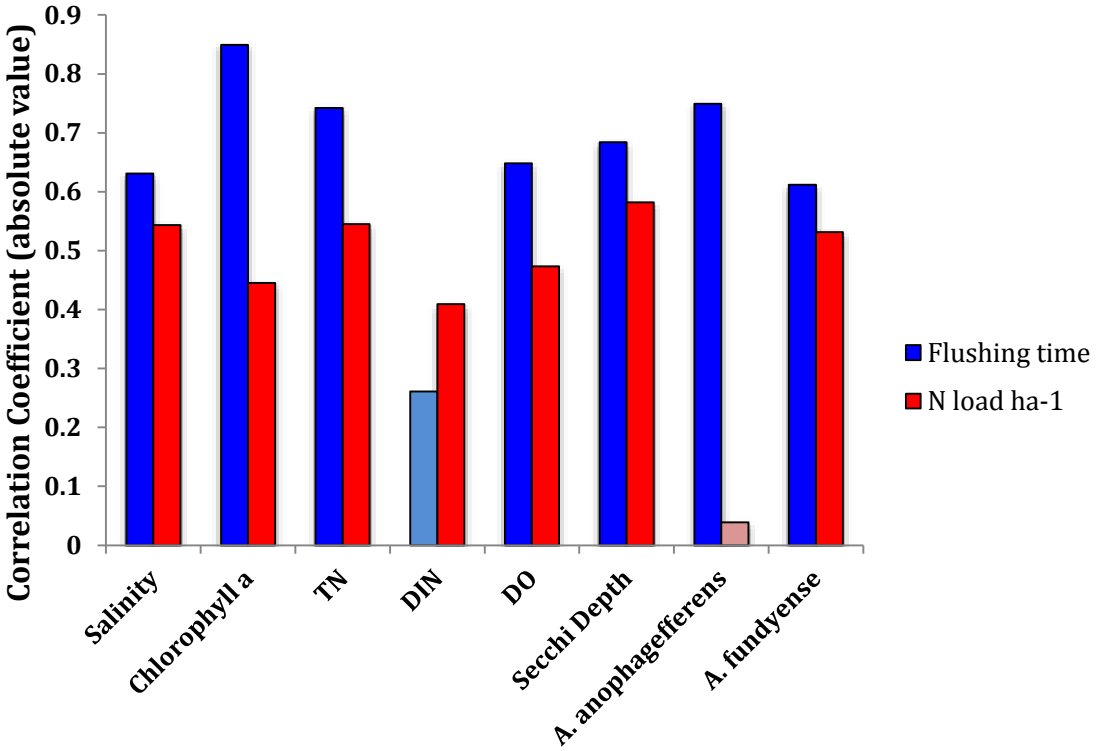
A correlation matrix including all marine data, residence times, and nitrogen loads showed that many of the parameters measured were significantly correlated. First, many marine parameters were correlated or inversely correlated with each other. Specifically, total nitrogen, chlorophyll a, *A. anophagefferens*, and *A. fundyense* were all significantly correlated with each other and were all significant, but inversely correlated with levels of DO, salinity, and Secchi disc depth (Table 12). The extent to which these trends were controlled by residence rates or nitrogen loads was then investigated. Residence time was significantly correlated with total nitrogen, chlorophyll *a*, *A. anophagefferens*, and *A. fundyense* and inversely correlated with Secchi depth, DO, and salinity (Figure 13). Nitrogen load per area of waterbody (N load ha^{-1}) was significantly correlated with total nitrogen, chlorophyll *a*, and *A. fundyense* and inversely correlated with Secchi depth, DO, and salinity although the correlation coefficients were generally lower than those with residence times (Figure 13). DIN did not correlate with any other marine parameter save for the nitrogen load ha^{-1}. As discussed above, DIN includes the most bioavailable forms of nitrogen (ammonium and nitrate) and is assimilated extremely rapidly by microbes, especially during summer when the entire pool of nitrate and ammonium can turnover in < 1 hour during summer month.[10] Hence, the highly transient nature of these compounds,

particularly during seasons when HABs are abundant, precludes their tracking of other water quality proxies.

Table 12: Correlations between the marine data, flushing time and nitrogen load. Correlation coefficients are shown numerically. Any values with * are statistically significant. P-values are represented as such: * indicates < 0.05, ** indicates <0.01, *** indicates < 0.001, **** indicates < 0.0001. N=7 (# of subwatersheds) for nitrogen load and flushing time, N=13 (# of sampling locations) for *A. fundyense*, and N=23 (# of marine stations) for all other marine data.

	Salinity	**Chl. *a***	**Total Nitrogen**	**Dissolved Inorganic Nitrogen**	**DO**	**Secchi Depth**	***A. anophagefferens***	***A. fundyense***	**N load ha^{-1}**
Flushing time	-0.631 ***	0.849 ****	0.742 ****	0.261	-0.648 ***	-0.684 ***	0.749 ****	0.612 *	0.491
Salinity		-0.659 ***	-0.897 ****	0.195	0.679 ***	0.729 ****	-0.564 **	-0.687 **	-0.543 **
Chl. *a*			0.851 ****	0.151	-0.767 ****	-0.873 ****	0.847 ****	0.713 **	0.445 *
TN				-0.071	-0.757 ****	-0.836 ****	0.708 ****	0.804 ****	0.545 **
DIN					0.143	0.0202	-0.031	0.0826	0.409 *
DO						0.778 ****	-0.609 **	-0.643 *	-0.474 *
Secchi Depth							-0.737 ****	-0.793 **	-0.582 **
A. anophagefferens								0.521	0.039
A. fundyense									0.531 *

Figure 13. Correlations between residence time, nitrogen load ha^{-1} and the water quality data. The correlation between residence time and dissolved inorganic nitrogen (DIN) and nitrogen load ha^{-1} and *A. anophagefferens* are not significant (p>0.05).



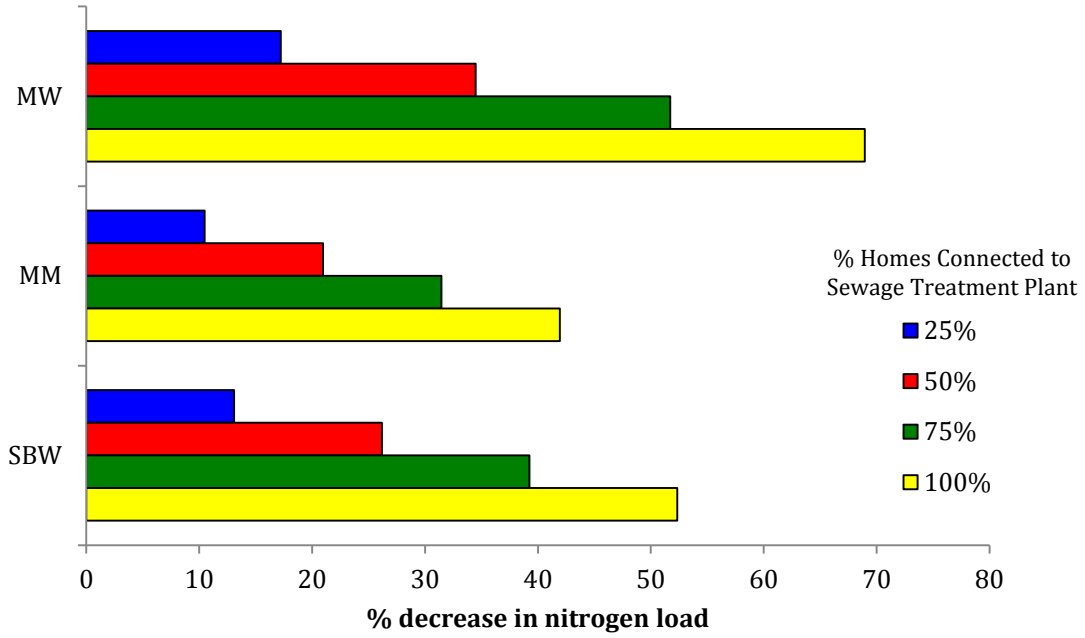
4.6 Nitrogen Mitigation Scenarios

Nitrogen mitigation scenarios were assessed for each subwatershed using the NLM models and Table 13 shows the results of each model run by subwatershed. Since wastewater contributes the majority of the nitrogen load for this area, connecting homes to sewage treatment plants is an obvious nitrogen mitigation option and produces a significantly larger nitrogen reduction than fertilizer use. The decrease in nitrogen load via the construction of sewage treatment plants varied from 8 – 69% depending upon the subwatershed, percent of the area covered by the sewer system, and where the outfall for the plant would be located (Fig 14). The smallest value (8%) was the connection of Shinnecock Bay East Subwatershed with an estuarine outfall whereas the largest nitrogen removal (69%) was the connection of Moriches West Subwatershed with an ocean outfall (Fig 14).

Table 13: NLM results of each model run by subwatershed. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.

Method of Nitrogen Mitigation	**% Reduction of TN per Subwatershed**						
	MW	**MM**	**ME**	**QB**	**SBW**	**SBE**	**HTC**
Fertilizer Reduction							
100% reduction in park and athletic fields fertilizer application	1.08	4.24	0.97	0.00	5.25	1.99	5.54
25% reduction in all fertilizer application	3.43	5.55	2.82	1.21	3.21	1.10	6.03
50% reduction in all fertilizer application	6.85	11.10	5.64	2.41	6.42	2.19	12.05
75% reduction in all fertilizer application	10.28	16.64	8.46	3.62	9.62	3.29	18.08
100% reduction in all fertilizer application	13.68	22.12	11.28	4.65	12.83	4.39	24.01
50% reduction in household fertilizer application	3.00	1.69	1.56	2.10	1.94	1.11	2.33
100% reduction in household fertilizer application	5.98	3.37	3.13	4.04	3.88	2.22	4.66
Connecting homes to sewer system by % of homes							
Sewering 25%, ocean outfall	17.24	10.49	9.99	13.02	13.08	8.28	12.91
Sewering 50%, ocean outfall	34.47	20.97	19.98	26.03	26.17	16.57	25.82
Sewering 75%, ocean outfall	51.71	31.46	29.96	39.05	39.25	24.85	38.72
Sewering 100%, ocean outfall	68.95	41.94	39.95	52.07	52.34	33.14	51.63
Sewering 25%, bay outfall	16.03	9.75	9.29	12.11	12.17	7.70	12.00
Sewering 50%, bay outfall	32.06	19.50	18.58	24.21	24.34	15.41	24.01
Sewering 75%, bay outfall	48.09	29.26	27.87	36.32	36.50	23.11	36.01
Sewering 100%, bay outfall	64.12	39.01	37.16	48.42	48.67	30.82	48.02
Upgrading the Nitrogen removal from onsite WW treatment							
Upgrade all septic tank systems and cesspools to alternative, denitrifying systems	41.99	25.55	24.34	22.23	32.56	20.47	31.11
Upgrade cesspools to alternative systems	27.57	16.79	16.00	12.90	23.63	14.38	19.33
Upgrade cesspools to septic tank systems	17.58	11.71	11.52	9.81	16.57	11.27	12.14
Upgrade all systems in HD res. areas to alternative systems	9.39	1.00	1.22	1.01	1.41	0.64	1.48
Upgrade all systems in 200m to alternative systems	7.99	7.24	7.75	7.35	11.27	9.56	5.09
Upgrade cesspools in 200m of shore to alternative systems	5.24	4.74	5.08	4.18	8.25	6.76	3.15
Upgrade cesspools in 200m of shore to conventional systems	2.47	2.23	2.39	1.97	3.88	3.18	1.48

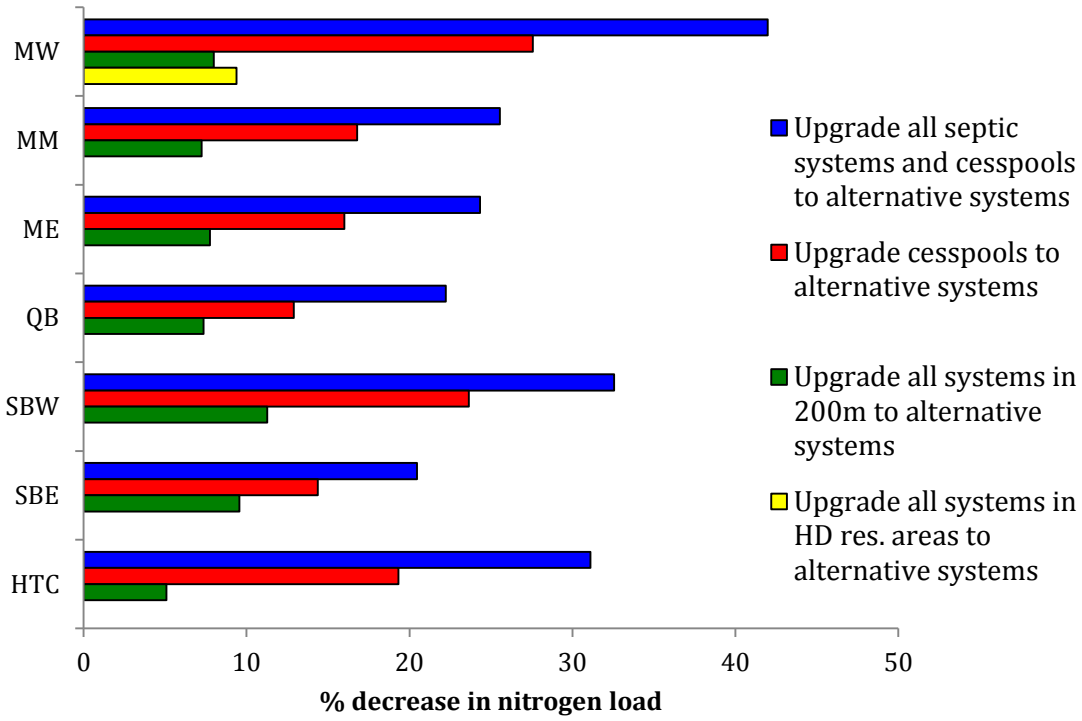
Figure 14. Percent decrease in Nitrogen load by connecting homes to a sewage treatment plant (25-100% of homes connected). Only the three subwatersheds with the greatest housing density are shown. Figure assumes that the sewage treatment plant has an ocean outfall. Results are slightly lower with an outfall in the bay. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, SBW- Shinnecock Bay West Subwatershed.



As the lower development density within several of the subwatersheds does not lend itself to sewage treatment plant construction, upgrading the large amount of cesspools to conventional septic tank systems is possible option. Upgrading all the cesspools in the study area had an effect of decreasing the nitrogen load by 10-18% (Figure 15). The nitrogen load effect of upgrading cesspools closest to the shore only produced a nitrogen load decrease of 1-4%. Quantuck Bay Subwatershed has the lowest percent of homes with cesspools therefore it showed the smallest response to upgrading.

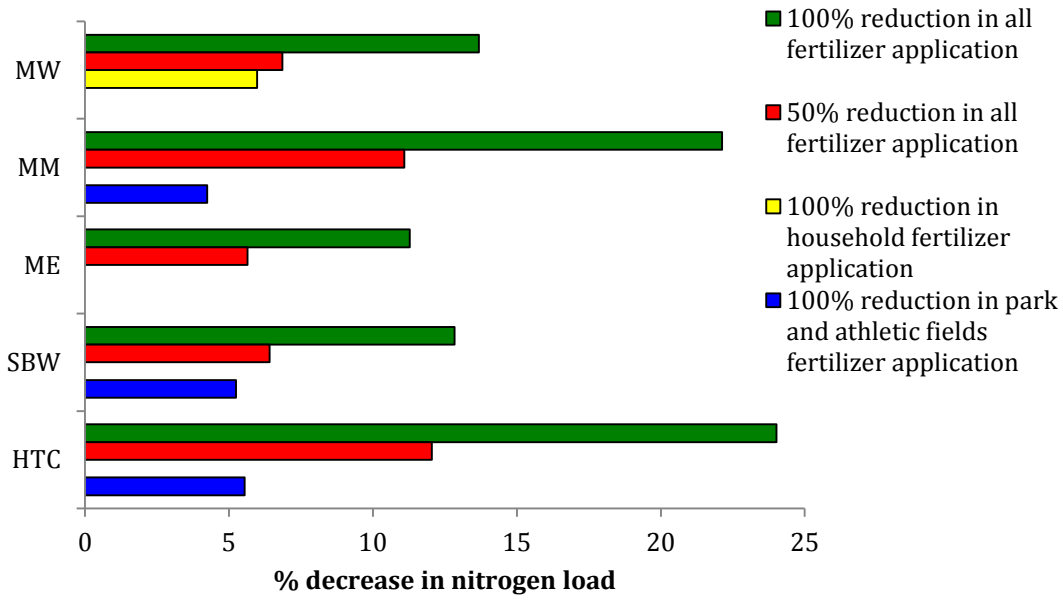
A promising solution to reducing wastewater nitrogen loads is converting existing septic tank systems and cesspools to denitrifying systems. Upgrading all homes within a subwatershed to denitrifying systems could decrease nitrogen loads by greater than 40% in some regions (Figure 15). Moriches West Subwatershed produced the best response (9% decrease) to upgrading systems in high-density residential areas whereas Shinnecock Bay East Subwatershed produced the largest response (11% decrease) to upgrading to denitrifying systems within 200 m of shore (Figure 15).

Figure 15. Percent decrease in Nitrogen load with different scenarios of upgrading cesspools and septic systems to denitrifying systems. Only scenarios that yielded reductions greater than 5% are shown. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.



Although fertilizer is a smaller source of nitrogen to these subwatersheds than wastewater, reductions in fertilizer may be simpler and more cost effective to implement than changes to traditional septic tank systems and cesspools. The reduction in nitrogen load from land to embayment was calculated as fertilizer use was decreased by 25, 50, and 75% across each watershed. A reduction in the percentage of fertilized lawns was also determined. In the current model, NLM assumes 50% of lawns are fertilized based on data from Cornell Cooperative Extension. Models were run again using 25% and 0% of lawns being fertilized as well as with a 100% reduction in fertilizer use in parks and athletic fields. Decreasing fertilizer use can lower the nitrogen load by as little as 1% if there is a 25% fertilizer reduction in Quantuck Bay Subwatershed, to as much as 24% if there is a 100% fertilizer reduction for Heady/Taylor Creek Subwatershed (Figure 16). The largest percent decrease in nitrogen load for an individual subwatershed with fertilizer reduction to lawns was found in Moriches West Subwatershed (6%) whereas Heady/Taylor Creek Subwatershed showed the strongest response (6% decrease) to reduction in non-County parks and athletic field fertilization.

Figure 16. Percent decrease in Nitrogen load with different fertilizer reductions. Only scenarios that yielded reductions greater than 5% are shown (Quantuck Bay Subwatershed and Shinnecock Bay East Subwatershed didn't have any). MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.

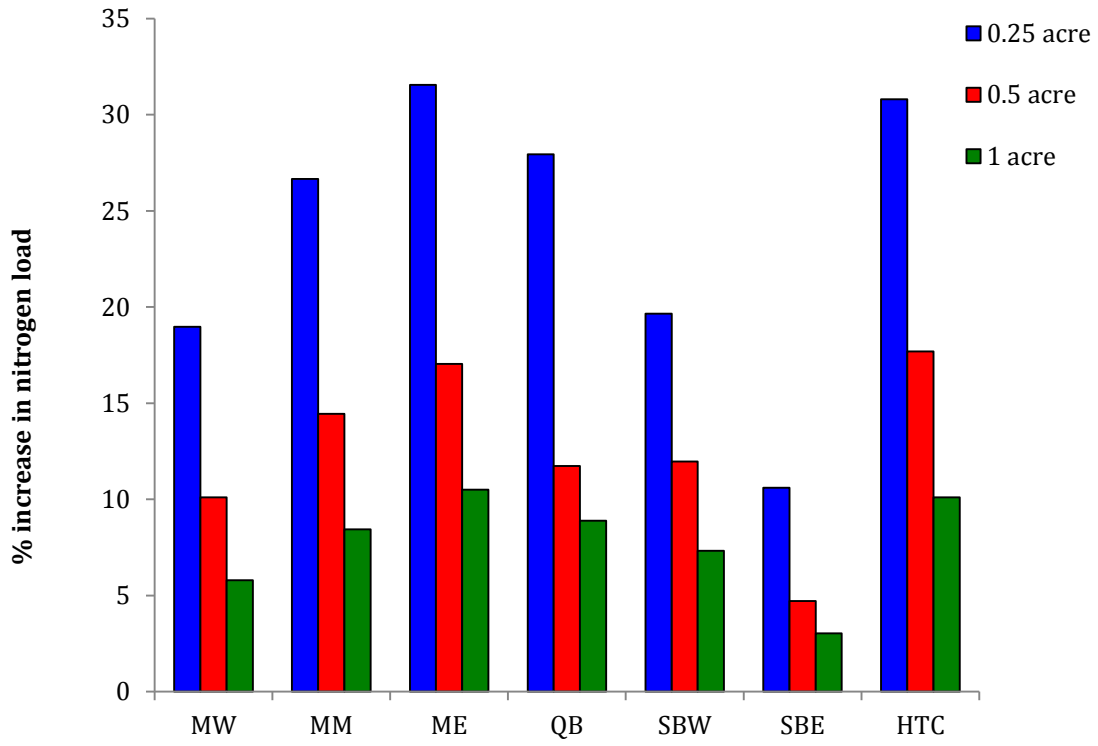


In regards to future increases in nitrogen loads due to future development, there is still undeveloped land within the study area, particularly in the Middle Moriches Subwatershed where 20% of the land area is undeveloped (Table 14). The most common lot sizes currently within these water shed (0.25, 0.5 and 1 acre) were used to calculate the additional buildings that would be added in a buildout scenario. Complete buildout for each individual sub-watershed (100%) would increase the nitrogen load across the entire study area by 3-11% for 1-acre lots, 5-18% for 0.5-acre lots, and 11-32% for 0.25-acre lots (Figure 17). However, these estimates may be slightly high as some of the undeveloped land may contain parcels too small to allow for development.

Table 14: Area of land available for development and the percent of the subwatershed that it represents. Moriches West Subwatershed and Moriches Middle data from 2009, all others from 2013.

Subwatershed	Area undeveloped (ha)	Percent of watershed
Moriches West	907	16
Moriches Middle	951	20
Moriches East	807	14
Quantuck Bay	178	8
Shinnecock Bay West	658	16
Shinnecock Bay East	137	10
Heady/Taylor Creek	144	16

Figure 17. Percent increase in Nitrogen load with complete buildout of differing lot sizes. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW- Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.

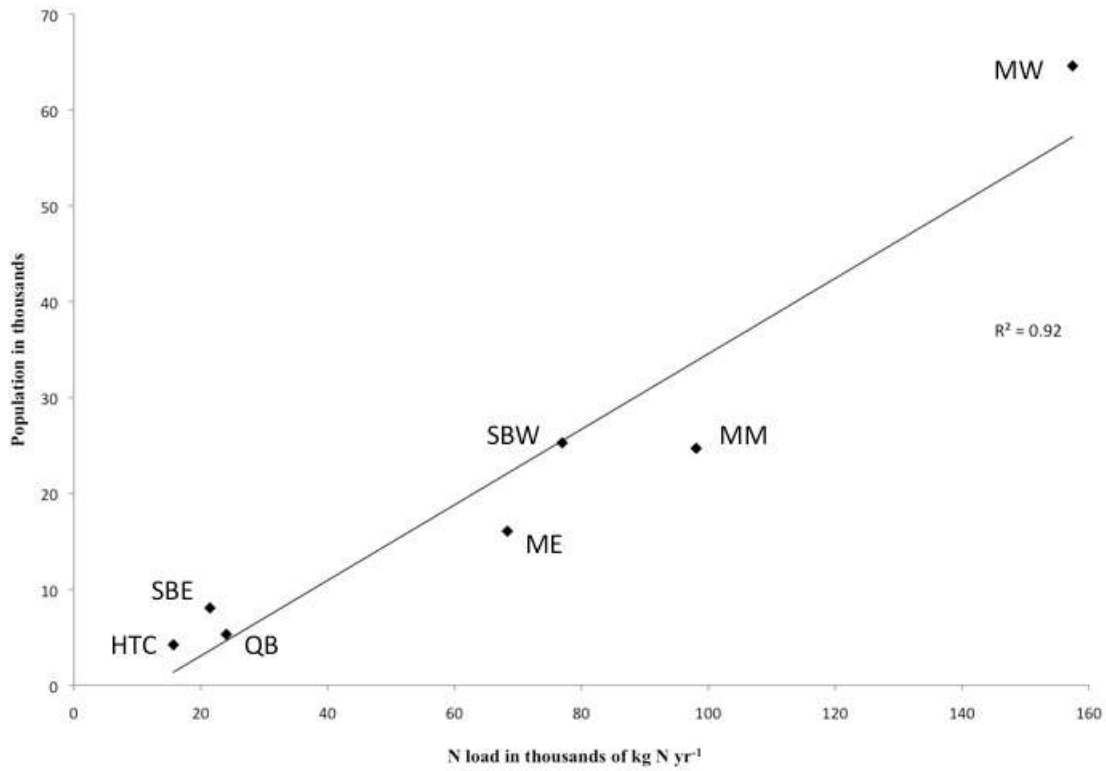


5 Discussion

5.1 Nitrogen Sources and Transport Mechanisms

Wastewater was the largest source of nitrogen in each of the three bays studied. There was a strong linear relationship between the population of a subwatershed and its nitrogen load (Figure 18). Given Great South Bay lies immediately to the west of Moriches Bay, Kinney and Valiela (2011)[19] is a good comparative study. Both studies found wastewater to be the largest nitrogen contributor and had the same overall percent contribution (51%). Atmospheric deposition to both the land and directly to the bay was less important in the Eastern Bays: (21% and 25%, land) and (12% v. 15% direct). Fertilizer was a larger source in the Eastern Baysstudy area, (16% v. 6%).

Figure 18. Nitrogen load of a subwatershed compared to population. Population estimated based on number of buildings and occupancy rate and Nitrogen load shown is the average of the NLM and VLM. MW-Moriches West Subwatershed, MM-Moriches Middle Subwatershed, ME-Moriches East Subwatershed, QB-Quantuck Bay Subwatershed, SBW-Shinnecock Bay West Subwatershed, SBE-Shinnecock Bay East Subwatershed, HTC-Heady/Taylor Creek Subwatershed.



The NLM model makes assumptions about how much nitrogen is removed during the various stages of wastewater being carried in groundwater underflow to the bays in household wastewater systems. The model defaults are as such: 7% removal in the septic tank system, 35% removal in the leaching field, and 35% in the septic plume as nitrogen is transported by groundwater. These reductions are followed by denitrification in the aquifer, for which 15% was used.[38] However, as mentioned previously, the septic tank systems in Suffolk County are primarily arrayed vertically, eliminating the leaching field, and the opportunity for nitrogen removal by microorganisms closer to the surface. Some nitrogen is likely removed in the disposal pits at the base of the septic tank structure but it is likely lower than the nitrogen removed by leach fields which are located near land surfaces and thus are within zones of more active biological nitrogen cycling. More research is needed to determine an average nitrogen removal percent for septic tank systems. Thus, the nitrogen loads from wastewater generated in these model runs are likely lower than in actuality. This underestimation should be considered in subwatersheds that have the strongest nitrogen load contribution from wastewater: Moriches West Subwatershed followed by Shinnecock Bay East Subwatershed and Shinnecock Bay West Subwatershed. Still, the similarlity between the VFM and NLM suggests that differences between actual

nitrogen removal by wastewater systems and removal modeled by NLM are within the collective variance of the models (<20%).

The VFM indicated that groundwater is by far the largest contributing transport mechanism for nitrogen loading to the bays. Previous studies in the Forge River region (Moriches West Subwatershed) determined that groundwater flow is twice as large as streamflow.[73] The Eastern Bays study corroborates this, finding groundwater flow in Moriches West Subwatershed was 1.75 times streamflow. In the other six subwatersheds, groundwater was more than three times greater than streamflow. Other recent studies in the region found groundwater to be the largest land-based transport mechanism for nitrogen and freshwater.[61],[19] Temporal variations in nitrogen concentrations within a confined region are not common[60] but the variability in precipitation does create variable groundwater discharge rates and therefore controls the flux of nitrogen to enclosed waterbodies on Long Island and within the SSER.[74] This variability in inorganic nitrogen flux has been shown to influence the phytoplankton assemblages of the bays allowing for conditions amenable for HABs.[74,59,60]

It is common to acknowledge a certain amount of uncertainty that occurs when making predictions using modeling studies. In this study, the standard percent errors for both nitrogen loading models (12% NLM and 14% VFM) were considered. However, with so many inputs to both models a true assessment of uncertainty is difficult to assess and a certain amount of error propagation may occur. However, having two models with such different approaches provides a level of robustness to the Nitrogen load results; there was a highly significant correlation ($p<0.001$) between the two models.

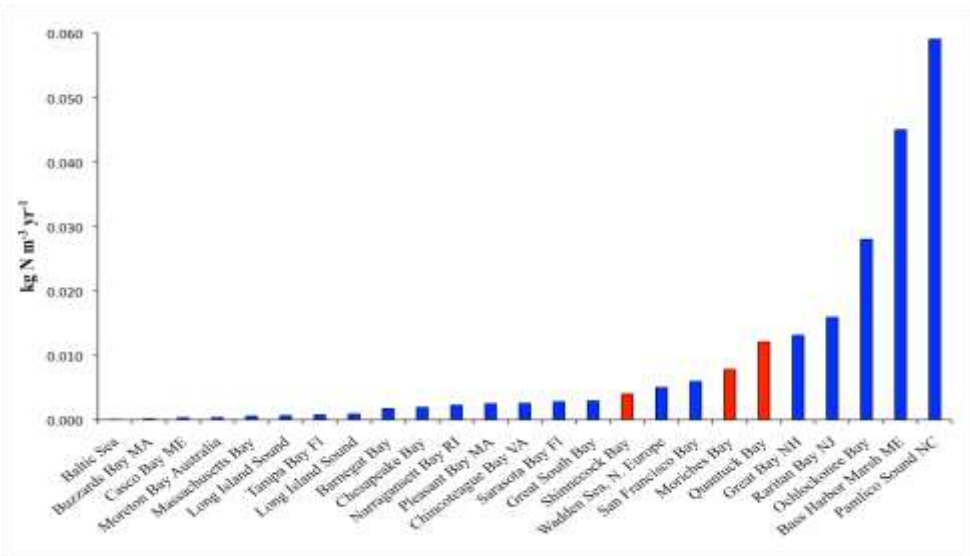
5.2 Comparability of Results

Nitrogen loads have been calculated for waterbodies all over the world that range in size by orders of magnitude. To facilitate comparisons among waterbodies of differing sizes, nitrogen loading among waterbodies can be normalized to the area of the receiving waterbody. Among a list of published nitrogen loads compiled by Bowen et al.,[34] Kinney and Valiea,[19] and Latimer and Charpentier,[75] the bays in this study had a moderate nitrogen load per area of waterbody with several bays having much higher loading rates, and many others being lower (Fig 19a). Of the three bays, Quantuck Bay had the highest yield, followed by Moriches and Shinnecock bays (Fig 19a). All three bays had higher nitrogen load yields than Great South Bay. Importantly, a number of factors contribute to how nitrogen loading affects a waterbody, including depth, water residence time, and aquatic vegetation.[35] For example, the Wadden Sea, in the southeastern part of the North Sea, has an average depth of 10 m and Moreton Bay, Australia, is 7 m deep, on average. Since the average depth of the Eastern Bays is 1.25 m, the volume of water is significantly smaller and thus the same areal nitrogen loading rate will have a larger impact on water quality. For example, the second highest nitrogen load per volume of water was found in Bass Harbor Marsh which had a mean depth of less than 1 m (Fig 19a). For the Eastern Bays subwatersheds, the second highest nitrogen load per volume of water was found in Moriches East Subwatershed (0.045 kg N m^{-3} yr^{-1}), the waterbody with the shallowest depth (0.5 m). The Eastern Bays have a comparable nitrogen loading rate per volume of water to the Wadden Sea and Great Bay, NH despite those two having a larger nitrogen load per area (Fig 19). In fact, given the differences in depth, the Eastern Bays have some

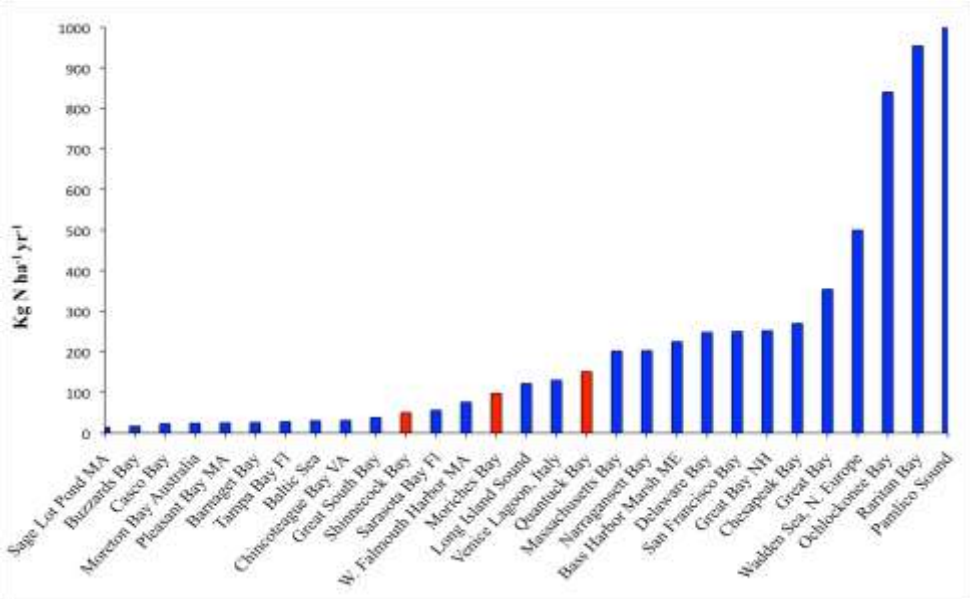
of the largest nitrogen loading rates per volume of water among the systems compared (Fig 19b). Additionally, while it was not considered in this study, shallow bays, such as those in this study area, receive a regular benthic flux of regenerated nitrogen from the sediments to the water column that have a larger impact on shallow systems. As such, benthic fluxes can be an important part of the nitrogen budget for shallow systems. For example, Gobler and Boneillo[60] found benthic flux contributed 28% of the nitrogen load to North Sea Harbor, NY. In contrast, deeper waterbodies are likely to have greater vertical stratification and benthic fluxes may have a smaller impact on the nitrogen concentration in the upper water column and phytoplankton in the euphotic zone.

Figure 19. Nitrogen loads for this study and comparables. a. Nitrogen loads are expressed in kg Nitrogen per *volume* per year and include direct atmospheric deposition to the bay. Chart adapted from Kinney and Valiela.[19] b. Nitrogen loads are expressed in kg Nitrogen per *surface area of the waterbody* per year

a.



b.



5.3 Effects of Marine Nitrogen Loading

Since 1985, the Eastern Bays have experienced multiple types of harmful algal blooms,[12,11,76,77] loss of eelgrass,[25] and declines in bivalve populations;[24] all occurrences with putative links to excessive nitrogen loading. While this study has assessed nitrogen loads in the bays, nitrogen concentrations in the bay water column, and the distribution of algal blooms in surface waterbodies are controlled not only by nitrogen load but also by biological (e.g. predation) and physical processes (flushing of water bodies). If total nitrogen is high in a given area of a bay, it may be due to an excessive nitrogen input, a small nitrogen export rate, or a combination of these factors. Total nitrogen was correlated with chlorophyll *a*, harmful algae (*A. anophagefferens, A. fundyense*) and inversely correlated with dissolved oxygen and water clarity and demonstrated that water quality impairments are associated with high nitrogen levels. All of these correlations are partly driven by autocorrelation as the harmful phytoplankton are blooming in regions where there are high levels of algal biomass that contain chlorophyll *a* and high levels of nitrogen, and these algal blooms shade the water and their demise leads to oxygen consumption. Bay water residence time had a primary influence on total nitrogen and a host of other parameters as it was strongly correlated with total nitrogen, chlorophyll *a*, harmful algae (*A. anophagefferens, A. fundyense*) and inversely correlated with dissolved oxygen and water clarity. This finding indicates that it is in the regions where algal biomass is retained (slow residence time), that water quality impairments (algal blooms, shading, low DO) are most likely to manifest themselves. Water depth is one of the largest sources of error in calculating residence time. As such, the residence time prediction may be more accurate when more specific average depths for each bay area are known.

Nitrogen load was significantly correlated with the majority of the water quality parameters but the r values were smaller than the correlations amongst the marine data and between residence time and the marine data. These findings suggest that the nitrogen loading rates in all subwatersheds are high enough to cause water quality problems and that extreme impairment is most likely when high nitrogen loads were combined with extended residence times.

Studies have associated low dissolved oxygen,[2,16,5] declining water clarity,[2,4] and harmful algal blooms[8,7] with excessive nitrogen loading. This Eastern Bays study demonstrates that water quality impairments in shallow bays with moderate to higher nitrogen loading rates are most likely to manifest themselves in bays with the longest residence time. In systems with high nitrogen loads and shorter residence times such as the Western Bays, the overgrowth of macroalgae is a more common water quality impairment since these organisms are typically able to attach to bottom substrates and thus are not subject to the same flushing action that microalgae are. In practical terms, while nitrogen loading rates are high enough to stimulate algal growth in most bays, strong tidal residence in zones near ocean inlets remove nitrogen supplies and/or algal biomass prior to it accumulating to high levels. Therefore, residence time was used as the primary factor in determining eutrophication vulnerability. The eutrophication vulnerability of the bays is best grouped into three tiers. Quantuck Bay and the Moriches West are in the first (most vulnerable) tier due to their extended residence time and clear signs of eutrophication including large HAB densities, low dissolved oxygen, water clarity, and nitrogen

concentrations that significantly exceed suggested ambient water quality targets identified in other studies such as the 1978 Long Island 208 Plan and the Cape Cod TMDLs as well as the management target of 0.45 mg L^{-1} set by the Peconic Estuary Program. Tier two is composed of the two embayments with the next slowest flushing times: Moriches East and Shinnecock Bay West. In addition to slower flushing, both of these embayments have higher averages for total N and Chlorophyll (Fig 11) and have a history of dense blooms of toxic algae dating to the 1950s and the 1980s, respectively.[67,9] The small Heady/Taylor Creek embayment is also included in tier two. While the flushing rate in the Heady/Taylor Creek section is faster than Moriches East and Shinnecock Bay West, Heady/Taylor Creek has a similar problem to Quantuck Bay in that a large watershed drains into a very small bay, which may compound the problem of nitrogen loading. Heady/Taylor Creek has also experienced large bloom densities of the toxic phytoplankton species *A. fundyense*.[9] The third eutrophication vulnerability tier is found in the bays with the fastest flushing times: Moriches Middle and Shinnecock Bay East. While these two bays may have adjacent subwatersheds with high groundwater nitrogen concentrations (Moriches Middle Subwatershed in particular, Fig 7), it is likely that the inlets and strong tidal influence keeps phytoplankton bloom densities low, which in turn creates areas of higher DO and light levels, and healthier conditions for marine habitats.

5.4 Prioritizing Regions for Nitrogen Mitigation

The multiple symptoms of eutrophication and impairments across the Eastern Bays have been shown to be interrelated in this study and have been shown to be closely tied to nitrogen loading rates from subwatersheds. Estuarine regions with harmful algal blooms also have low water clarity and low oxygen, collective impairments that will lead to the loss of seagrass and diminished fisheries. Given this, the final section of this report focuses on specific options for mitigating nitrogen loads from each watershed to the receiving waterbody. While nitrogen mitigation actions deemed likely to be most effective are presented (Table 15), all options should be considered since some, such as reduction in fertilizer use, are less costly than others. The Moriches West embayment is the most vulnerable to eutrophication and shows the largest proportion reduction in nitrogen load from different mitigation options (Table 15). The Moriches West Subwatershed is characterized by a high population density and degree of urbanization. The population density is approximately 11 people per hectare; almost double the next highest subwatershed of Shinnecock Bay West. Despite having 400 ha of high-density residential and 1,600 ha of medium-density residential land, there is no public sewage treatment available. Even if the septic tank systems and cesspools are working optimally, their density and the shallow depth to groundwater may be prohibitive for optimal biological nitrogen removal.[78,79] As such, it is likely that the best nitrogen mitigation scenario for the Moriches West Subwatershed a public sewer construction system. Construction of waste water treatment plants and sewers is not economically viable for much of the Eastern Bays watershed due to low population density, but the high densities in portions of the Moriches West Subwatershed may make this a feasible option. Constructing a sewer system with an ocean outfall for the Moriches West Subwatershed alone (100% of homes included) would lower the nitrogen load to the Moriches West part of Moriches Bay by 69% and as a whole by 32%. As excess nitrogen negatively affects the health of seagrass and salt marshes,[80,4] the decrease in nitrogen load coming from the Moriches West Subwatershed could allow

these ecosystems to recover, producing a positive feedback loop for the entirety of Moriches Bay.

Table 15. Nitrogen mitigation priority and most effective remedial measures by subwatershed for improving water quality.

Subwatershed	Eutrophication Vulnerability Tier	Mitigation Priority	Most Effective Mitigation Scenarios
Moriches West	1	1	Sewer construction; Upgrade septic tank systems and cesspools
Quantuck Bay	1	2	Upgrade septic tank systems and cesspools; Control build out
Shinnecock Bay West	2	3	Sewer construction; Upgrade septic systems; Control build out
Heady/Taylor Creek	2	4	Control build out; Upgrade septic tank systems and cesspools; Reduce fertilizer
Moriches East	2	5	Control build out; Upgrade septic tank systems and cespools
Moriches Middle	3	6	Upgrade septic tank systems cesspools; Reduce fertilizer
Shinnecock Bay East	3	7	Upgrade septic tank systems and cesspools within 200 m of shore

Quantuck Bay, site of severe eutrophication during the past several decades, is a very small but vulnerable bay. Both models predicted that the total nitrogen load from the Quantuck Bay Subwatershed is low. However, the nitrogen loading rate to Quantuck Bay on an aerial and volumetric basis exceeds both neighboring Moriches and Shinnecock bays. The anomaly of low nitrogen load and poor water quality may lead to a particular challenge in designing nitrogen mitigation efforts for Quantuck Bay. Unlike Moriches Bay, that would show an overall improvement in health with reduced nitrogen loading in any of its subwatersheds, Quantuck Bay, because of its isolation and slow residence time, is unlikely to benefit from any water quality improvements if nitrogen loading is reduced in the adjacent Moriches or Shinnecock bays. Due to the low housing density across much of the Quantuck Bay Subwatershed, nitrogen mitigation options are less likely to improve water quality than they would in an area that is more heavily populated such as the Moriches West Subwatershed. Excluding new sewers, as they are likely not economically feasible in this low densely developed area, the best nitrogen mitigation scenario is upgrading cesspools and septic tank systems to alternative, denitrifying septic systems. Many varieties of denitrifying systems exist, all of which have the goal of decreasing the concentration of nitrogen in their effluent. Denitrifying septic systems have the added benefit of being able to cover multiple dwellings within a half-acre,[81] which may be suitable for parts of the Quantuck Bay Subwatershed. Given that Suffolk County has estimated that more than two-thirds of homes in the County have cesspools[29], upgrading to denitrifying systems can reduce nitrogen loads by more than 20%. A similar amount of nitrogen load increase can be prevented by preserving under or undeveloped land. Although only 8% of the Quantuck Bay Subwatershed can be further developed, it showed a large increase in nitrogen load

with each build-out scenario compared to the other subwatersheds in the Eastern Bays. Complete build-out in this watershed could lead to the further degradation of Quantuck Bay.

Out of the tier three areas of eutrophication vulnerability, potential nitrogen mitigation changes in the Shinnecock Bay West Subwatershed produces the most effective reduction in nitrogen load across most of the nitrogen mitigation scenarios, making it the third priority area for nitrogen mitigation (Table 15). The NLM predicted that 67% of the land-based nitrogen load to Shinnecock Bay comes from the Shinnecock Bay West Subwatershed, making this subwatershed a good candidate for nitrogen mitigation efforts. Again, much like how changes to the Moriches West Subwatershed may positively reverberate in the entirety of Moriches Bay, the same may be true for Shinnecock Bay West Subwatershed and Shinnecock Bay. This subwatershed has the highest estimated percent of homes with cesspools (57%) when compared to the rest of the Eastern Bays subwatersheds. Therefore, the best nitrogen mitigation scenarios for the Shinnecock Bay West Subwatershed are related to controlling wastewater input by constructing sewage treatment facilities or upgrading septic tank systems and cesspools to denitrifying systems. Finally, there is a large amount of undeveloped land within the Shinnecock Bay West Subwatershed, development of which could enhance nitrogen loads to this subwatershed by 20%. Hence, efforts to preserve this remaining land are certainly warranted, especially within near-shore regions where nitrogen loads have the strongest effects on surface waters.

A small section of the eastern portion of Shinnecock Bay's the Heady/Taylor Creek Subwatershed is the fourth priority for nitrogen mitigation action (Table 15). As the Heady/Taylor Creek estuarine area flushes slower than the east and west sections of Shinnecock Bay, there is reason to believe that this area of the bay may be more vulnerable to eutrophication and HABs. Indeed, evidence of this can be found in the large blooms of *A. fundyense* that have occurred in this area.[9] This is the most effective subwatershed in which to lessen fertilizer nitrogen load to the entirety of Shinnecock Bay. Though small, this subwatershed contains golf courses, fertilized parks and athletic fields, agriculture and the some of the largest property sizes (Table 16), thus likely lawns sizes, among the subwatersheds. Upgrading septic systems and cesspools to denitrifying systems is the other good option for reducing the nitrogen load to the Heady/Taylor Creek Subwatershed. Controlling build-out will also prove important in Heady/Taylor Creek Subwatershed as it is the second most responsive subwatershed to increased development.

Table 16: Average property size in hectares per subwatershed.

Subwatershed	Ave property size (ha)
Moriches West	0.145
Moriches Middle	0.255
Moriches East	0.338
Quantuck Bay	0.344
Shinnecock Bay West	0.251
Shinnecock Bay East	0.285
Heady/Taylor Creek	0.338

The fifth priority area for nitrogen mitigation is the Moriches East Subwatershed as it lies within tier two of eutrophication vulnerability due to slower residence time and the presence of harmful algal blooms. The Moriches East Subwatershed is the most vulnerable to build-out, predicting a 32% increase in nitrogen load with build-out at 0.25 acre lots.

The areas in tier three of eutrophication vulnerability are the Middle Moriches and Shinnecock Bay East subwatersheds. Though both of these areas of the bay are well flushed, the nitrogen load for the Moriches Middle Subwatershed is roughly double that of Shinnecock Bay East Subwatershed, thus an important contributor to the overall nitrogen load to Moriches Bay. The Moriches Middle Subwatershed has the most area of agricultural acreage of any of the subwatersheds, over three times more land than the next highest subwatershed in the Eastern Bays; therefore focusing on reducing agricultural fertilizer in this area may be a key nitrogen mitigation step. However, the most effective nitrogen mitigation scenario, aside from constructing a sewage treatment plant is upgrading septic tank systems and cesspools to denitrifying systems.

Nitrogen mitigation measures for the Shinnecock Bay East Subwatershed is somewhat less of a priority, however, as there are fewer symptoms of eutrophication. Given that a large percentage of homes in this subwatershed are located close to shore, updating cesspools and septic tank systems within 200 meters of the shore would be a good first step in nitrogen load mitigation.

5.5 Next Steps and Recommendations

Through this study, priority regions and activities for nitrogen mitigation in the Eastern Bays have been identified. Below, a series of next steps and complementary activities that can be undertaken to improve water quality and ecosystem functions are outlined.

One next step would include refining the nitrogen loading models presented here (NLM, VFM) based on new and emerging information. For this study, watersheds were drawn to capture major waterbodies (e.g. Forge River, Quantuck Bay) and/or political boundaries (Towns of Southampton and Brookhaven). Presently, the USGS is redefining these subwatersheds by using state of the art hydrological models to ensure they best reflect groundwater and streamflow. Since future water quality management plans are needed on a watershed-by-watershed basis, it will be highly useful to apply the nitrogen loading models developed by this research project to the newly defined subwatersheds. Beyond the use of refined watershed boundaries, the water residence times in the surface water receiving bodies can be determined via multiple mechanisms. For this research project, the fraction of freshwater approach was used to determine residence times providing a first-order approximation of the flushing rates and residence times that are useful for bays. However, such models do not account for several important variables that can influence circulation in estuaries including complex geomorphology as well as differential tidal flow through ocean inlets and channels. Presently, Professor Robert Wilson at Stony Brook University has a highly sophisticated and refined hydrodynamic model that covers Great South Bay and western Moriches Bay. This model could be extended through the rest of

Moriches Bay and Shinnecock Bay. Alternatively, a hydrodynamic model is being constructed by Gobler in collaboration with the consulting firm, Anchor AEQ, for Shinnecock Bay and Moriches Bay. Applying either model to the receiving body of water for each new USGS-defined watershed will provide more precise residence times for each waterbody by which management decisions may be better informed. With this data, each new USGS watershed and receiving water body could be ranked with regard to nitrogen load per unit volume of estuarine water, residence time, and total water quality. The ranking of these factors could be used to assess the need for nitrogen load mitigation across the region with waterbodies experiencing the greatest nitrogen loads per volume of water, the longest residence times, and the poorest water quality representing the regions that are the highest priority for nitrogen load mitigation.

While efforts are being made to reduce the flow of nitrogen from land to sea, it will take decades for groundwater contaminated with nitrogen to flow through the aquifers and for groundwater with lower levels of nitrogen to begin to enter the bays. Hence, the implementation of multiple 'in the water' solutions for mitigating and removing nitrogen outlined below are desirable. There are several such projects that show promise. As they are implemented, their effectiveness in mitigating nitrogen loads to given systems could be determined quantitatively via the use of the models generated by this project.

One of the symptoms of heavy nitrogen loading to the Eastern Bays is the overgrowth of algae. Although microalgae and phytoplankton are the main primary producers in poorly flushed parts of estuaries, macroalgae or seaweeds overgrow estuaries in regions with faster flushing times causing a series of environmental problems including low oxygen and death of marine life.[2] In some areas, the seaweeds can cover the entire bottom of estuaries. For example, two species of macroalgae, *Ulva* and *Gracilaria* grow quickly and rapidly in the Forge River and eastern Shinnecock Bay during May through November. Preliminary calculations based on elemental analyses of these seaweeds suggest that their weekly removal from only 10% of Shinnecock Bay would remove up to 3,000 lbs of N per week, representing a potentially significant faction of the total nitrogen load to this system. Detailed research regarding precise yields of harvesting on a per pound of nitrogen basis is needed to better understand the feasibility of such an approach for mitigating nitrogen loads.

Finally, dense populations of filter-feeding bivalves have shown the potential to remove nitrogen from estuaries by consuming phytoplankton. There are presently efforts to replant natural populations of hard clams, *Mercenaria mercenaria*, as well as Eastern oysters, *Crassostrea virginica*, in different estuaries across Long Island, including Shinnecock Bay (www.shinnecockbay.org). In addition, there is also potential for using an aquaculture approach to grow and harvest bivalves and therefore remove nitrogen from embayments with high nutrient input. Sebastino et al (2015) recently determined that covering 2,500 acres of Great South Bay with aquacultured adult oysters from June through October would mitigate 75% of the nitrogen loads to this estuary. Given that Moriches Bay and Shinnecock Bay have smaller nitrogen loads and smaller volumes of water, a significantly smaller region of aquacultured oysters would be needed to mitigate nitrogen

loads in these systems. All these mitigation options must also take into account future climate change. Increased temperature and precipitation due to climate change may make nitrogen loading worse in bay systems and could offset mitigation efforts.[82] Further research should be conducted to determine the impact of climate change on nitrogen loading in these bays so that mitigations plan can be properly informed.

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