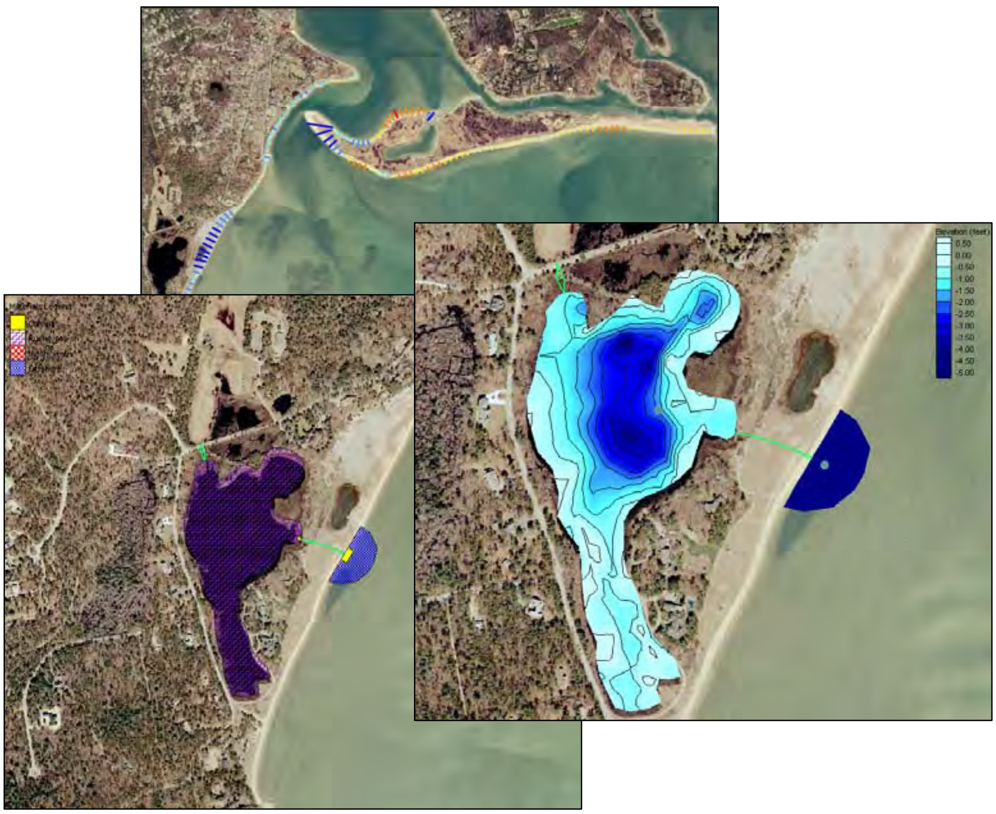
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Rushy Marsh, Barnstable, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

FINAL REPORT –APRIL 2006

Massachusetts Estuaries Project

LINKED WATERSHED-EMBAYMENT MODEL TO DETERMINE CRITICAL NITROGEN LOADING THRESHOLDS FOR RUSHY MARSH POND IN THE TOWN OF BARNSTABLE, MA

FINAL REPORT – APRIL 2006

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Executive Summary

1. Background

This report presents the results generated from the implementation of the Massachusetts Estuaries Project's Linked Watershed-Embayment Approach to the Rushy Marsh embayment system, a coastal embayment within the Town of Barnstable, Massachusetts. Analyses of the Rushy Marsh embayment system was performed to assist the Town with up-coming nitrogen management decisions associated with the Towns' current and future wastewater planning efforts, as well as wetland restoration, anadromous fish runs, shell fishery, open-space, and harbor maintenance programs. As part of the MEP approach, habitat assessment was conducted on the embayment based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. Nitrogen loading thresholds for use as goals for watershed nitrogen management are the major product of the MEP effort. In this way, the MEP offers a science-based management approach to support the Town of Barnstable resource planning and decision-making process. The primary products of this effort are: (1) a current quantitative assessment of the nutrient related health of the Rushy Marsh embayment, (2) identification of all nitrogen sources (and their respective N loads) to embayment waters, (3) nitrogen threshold levels for maintaining Massachusetts Water Quality Standards within embayment waters, (4) analysis of watershed nitrogen loading reduction to achieve the N threshold concentrations in embayment waters, and (5) a functional calibrated and validated Linked Watershed-Embayment modeling tool that can be readily used for evaluation of nitrogen management alternatives (to be developed by the Town) for the restoration of the Rushy Marsh embayment system.

Wastewater Planning: As increasing numbers of people occupy coastal watersheds, the associated coastal waters receive increasing pollutant loads. Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The elevated nutrients levels are primarily related to the land use impacts associated with the increasing population within the coastal zone over the past half-century.

The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities. The primary nutrient causing the increasing impairment of our coastal embayments is nitrogen, with its primary sources being wastewater disposal, and nonpoint source runoff that carries nitrogen (e.g. fertilizers) from a range of other sources. Nitrogen related water quality decline represents one of the most serious threats to the ecological health of the nearshore coastal waters. Coastal embayments, because of their shallow nature and large shoreline area, are generally the first coastal systems to show the effect of nutrient pollution from terrestrial sources.

In particular, the Rushy Marsh embayment system within the Town of Barnstable is at risk of eutrophication (over enrichment) from enhanced nitrogen loads entering through groundwater from the increasingly developed watershed to this coastal salt pond. Eutrophication is a process that occurs naturally and gradually over a period of tens or hundreds of years. However, human-related (anthropogenic) sources of nitrogen may be introduced into ecosystems at an accelerated rate that cannot be easily absorbed, resulting in a phenomenon known as cultural eutrophication. In both marine and freshwater systems, cultural eutrophication results in degraded water quality, adverse impacts to ecosystems, and limits on the use of water resources.

The Town of Barnstable has recognized the severity of the problem of eutrophication and the need for watershed nutrient management and is currently developing a Comprehensive Wastewater Management Plan, which it plans to rapidly implement. The Town of Barnstable has also completed and implemented wastewater planning in other regions of the Town not associated with the Rushy Marsh embayment system. The Town has nutrient management activities related to their tidal embayments, which have been associated with the MEP effort in Three Bays, Centerville River/Harbor and the Lewis Bay embayment systems. The Town of Barnstable and work groups have recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Town. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns' nutrient management groups to predict the impacts on water quality from a variety of proposed management scenarios.

Nitrogen Loading Thresholds and Watershed Nitrogen Management: Realizing the need for scientifically defensible management tools has resulted in a focus on determining the aquatic system's assimilative capacity for nitrogen. The highest-level approach is to directly link the watershed nitrogen inputs with embayment hydrodynamics to produce water quality results that can be validated by water quality monitoring programs. This approach when linked to state-of-the-art habitat assessments yields accurate determination of the "allowable N concentration increase" or "threshold nitrogen concentration". These determined nitrogen concentrations are then directly relatable to the watershed nitrogen loading, which also accounts for the spatial distribution of the nitrogen sources, not just the total load. As such, changes in nitrogen load from differing parts of the embayment watershed can be evaluated relative to the degree to which those load changes drive embayment water column nitrogen concentrations toward the "threshold" for the embayment system. To increase certainty, the "Linked" Model is independently calibrated and validated for each embayment.

Massachusetts Estuaries Project Approach: The Massachusetts Department of Environmental Protection (DEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool to communities throughout southeastern Massachusetts (the Linked Watershed-Embayment Management Model) for nutrient management in their coastal embayment systems. Ultimately, use of the Linked Watershed-Embayment Management Model tool by municipalities in the region results in effective screening of nitrogen reduction approaches and eventual restoration and protection of valuable coastal resources. The MEP provides technical guidance in support of policies on nitrogen loading to embayments, wastewater management decisions, and establishment of nitrogen Total Maximum Daily Loads (TMDLs). A TMDL represents the greatest amount of a pollutant that a waterbody can accept and still meet water quality standards for protecting public health and maintaining the designated beneficial uses of those waters for drinking, swimming, recreation and fishing. The MEP modeling approach assesses available options for meeting selected nitrogen goals that are protective of embayment health and achieve water quality standards.

The core of the Massachusetts Estuaries Project analytical method is the Linked Watershed-Embayment Management Modeling Approach, which links watershed inputs with embayment circulation and nitrogen characteristics.

The Linked Model builds on well-accepted basic watershed nitrogen loading approaches such as those used in the Buzzards Bay Project, the CCC models, and other relevant models. However, the Linked Model differs from other nitrogen management models in that it:

- requires site-specific measurements within each watershed and embayment;
- uses realistic “best-estimates” of nitrogen loads from each land-use (as opposed to loads with built-in “safety factors” like Title 5 design loads);
- spatially distributes the watershed nitrogen loading to the embayment;
- accounts for nitrogen attenuation during transport to the embayment;
- includes a 2D or 3D embayment circulation model depending on embayment structure;
- accounts for basin structure, tidal variations, and dispersion within the embayment;
- includes nitrogen regenerated within the embayment;
- is validated by both independent hydrodynamic, nitrogen concentration, and ecological data;
- is calibrated and validated with field data prior to generation of “what if” scenarios.

The Linked Model Approach’s greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing “what if” scenarios for evaluating watershed nitrogen management options.

For a comprehensive description of the Linked Model, please refer to the *Full Report: Nitrogen Modeling to Support Watershed Management: Comparison of Approaches and Sensitivity Analysis*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm. A more basic discussion of the Linked Model is also provided in Appendix F of the *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm. The Linked Model suggests which management solutions will adequately protect or restore embayment water quality by enabling towns to test specific management scenarios and weigh the resulting water quality impact against the cost of that approach. In addition to the management scenarios modeled for this report, the Linked Model can be used to evaluate additional management scenarios and may be updated to reflect future changes in land-use within an embayment watershed or changing embayment characteristics. In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries. Unlike many approaches, the Linked Model accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics and accommodates the spatial distribution of these processes. For an overview of several management scenarios that may be employed to restore embayment water quality, see *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm.

Application of MEP Approach: The Linked Model was applied to the Rushy Marsh embayment system by using site-specific data collected by the MEP and water quality data from the Water Quality Monitoring Program conducted by Three Bays Preservation in partnership with the Town of Barnstable, with technical guidance from the Coastal Systems Program at SMAST (see Chapter 2). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Town of Barnstable Planning Department, and watershed boundaries delineated by USGS. This land-use data was used to determine watershed nitrogen loads within the Rushy Marsh embayment system and each systems sub-embayments as appropriate (current and build-out loads are summarized in Table IV-3). Water quality within a sub-embayment is the integration of nitrogen loads with the site-specific estuarine circulation. Therefore, water quality modeling of this tidally influenced estuary included a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Once the hydrodynamics of the system was quantified, transport of nitrogen was evaluated from tidal current information developed by the numerical models.

A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the Rushy Marsh embayment system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates. Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic model was then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis. Boundary nutrient concentrations in Vineyard Sound source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Rushy Marsh embayment system was used to calibrate the water quality model, with validation using measured nitrogen concentrations (under existing loading conditions). The underlying hydrodynamic model was calibrated and validated independently using water elevations measured in time series throughout the embayments.

MEP Nitrogen Thresholds Analysis: The threshold nitrogen level for an embayment represents the average water column concentration of nitrogen that will support the habitat quality being sought. The water column nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition). The water column nitrogen concentration is modified by the extent of sediment regeneration. Threshold nitrogen levels for the embayment systems in this study were developed to restore or maintain SA waters or high habitat quality. High habitat quality was defined as supportive of eelgrass and infaunal communities. Dissolved oxygen and chlorophyll a were also considered in the assessment.

The nitrogen thresholds developed in Section VIII-2 were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Rushy Marsh system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for Rushy Marsh. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen reduction guidelines for nitrogen management of the Rushy Marsh embayment system in the Town of Barnstable. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. The MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of each embayment system. The concept was that since septic system nitrogen loads generally represent 90% of the controllable watershed load to the Rushy Marsh embayment system and are more manageable than other of the nitrogen sources, the ability to achieve needed reductions through this source is a good gauge of the feasibility for restoration of these systems.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout Rushy Marsh based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. At present, Rushy Marsh Pond is showing significantly impaired to severely degraded habitat quality. All of the habitat indicators are consistent with this evaluation of the whole of system (Chapter VII).

The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). The clear evidence of oxygen levels above atmospheric equilibration indicates that the Rushy Marsh System is eutrophic.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality within the estuary. Oxygen depletion was frequently to levels <4 mg/L (29 days) and periodically to < 3 mg/L (8 days). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of this estuarine system, although the nitrogen enrichment stems primarily from the restriction of tidal exchange. The frequent significant level of oxygen depletion coupled to the frequent phytoplankton blooms is clear evidence of that Rushy Marsh Pond is presently nitrogen over-loaded eutrophic embayment.

Currently, eelgrass is not present within Rushy Marsh Pond. Rushy Marsh Pond is functionally a basin with fringing wetland, and the sediments are currently soft muds rich in organic matter, which in some locations overlay medium to fine sands. The current lack of eelgrass beds is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within this system. In addition, it does not appear that eelgrass beds were present in the system in 1951, as well. It appears that the restriction of the tidal exchange starting circa 1900 (as discussed in Chapter V), resulted in an absence of eelgrass sometime prior to 1951. The restriction of tidal exchange has resulted in an enrichment of estuarine waters in nitrogen to the extent that the system is currently eutrophic. Restoration of tidal exchange will be needed for habitat restoration of this system, as watershed nitrogen inputs are relatively low.

Given that eelgrass has not been documented for this system, it is not clear that even when the system was much better flushed, it supported eelgrass beds. However, observations of brackish water submerged aquatic vegetation in the shallow region of the western channel suggest that eelgrass habitat might be sustainable under lower effective nitrogen loading rates (i.e. higher flushing). To the extent that conditions could be improved to the level of eelgrass colonization in this system, the acreage would likely range from 4-12 acres, most likely in the southern channel and the margins of the main basin.

The Infauna Study indicated that presently, habitat capable of supporting benthic infaunal communities is virtually absent in Rushy Marsh Pond (Table VII-3). The infaunal survey found that summer conditions apparently are sufficient to prevent a community from developing in the central basin. In the shallower southern channel region, again only very few individuals and species were found. The low numbers of species and individuals indicates that benthic infaunal habitat has been severely degraded throughout Rushy Marsh Pond. The conditions proximately result from the high level of nitrogen and organic matter enrichment and associated oxygen depletion of bottom waters. Ultimately, the cause is the highly restricted tidal exchange and very low flushing rate of Pond waters (system residence time ~48 d). However, restoration of infaunal animal communities should occur at the point that habitat can be restored.

3. Conclusions of the Analysis

The threshold nitrogen level for an embayment represents the average watercolumn concentration of nitrogen that will support the habitat quality being sought. The watercolumn nitrogen level is ultimately controlled by the integration of the watershed nitrogen load, the nitrogen concentration in the inflowing tidal waters (boundary condition) and dilution and flushing via tidal flows. The water column nitrogen concentration is modified by the extent of sediment regeneration and by direct atmospheric deposition.

Threshold nitrogen levels for this embayment system were developed to restore or maintain SA waters or high habitat quality. In this system, high habitat quality was defined as possibly supportive of eelgrass and supportive of diverse benthic animal communities. Dissolved oxygen and chlorophyll *a* were also considered in the assessment.

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Town of Barnstable Rushy Marsh embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 90% of the controllable watershed nitrogen load to the embayment was from wastewater.

A major finding of the MEP clearly indicates that a single total nitrogen threshold can not be applied to Massachusetts' estuaries, based upon the results of the Great, Green and Bournes Pond Systems, Popponesset Bay System, the Hamblin / Jehu Pond / Quashnet River analysis in eastern Waquoit Bay, the analysis of the adjacent Three Bays system and the Pleasant Bay and Nantucket Sound embayments associated with the Town of Chatham. This is almost certainly going to be true for the other embayments within the MEP area, as well.

The threshold nitrogen levels for the Rushy Marsh embayment system in Barnstable were determined as follows:

Rushy Marsh Threshold Nitrogen Concentrations

- Following the MEP protocol, since eelgrass has not been documented in Rushy Marsh Pond, restoration of infaunal habitat is the restoration goal. Infaunal animal habitat is a critical resource to the Rushy Marsh System and estuaries in general. Since there are virtually no infaunal animals remaining in the sub-tidal Rushy Marsh Pond sediments, comparisons to the muddy basins of other nearby estuarine systems were relied upon for setting the nitrogen threshold for healthy infaunal habitat at a nitrogen level of TN <0.5 mg TN L^{-1}. This level was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 were found supportive of high infaunal habitat quality in this system. Similarly, in the Three Bays System, healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay), with impairment in areas where nitrogen levels of TN >0.5 mg TN L^{-1} (North Bay), and severe degradation at nitrogen levels of TN >0.6 mg TN L^{-1}..

- The nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations were not attainable with 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the systems watershed. The limited circulation within the system prevents the threshold goals from be achieved. In order to meet the threshold concentrations in the system, alternative approaches beyond load reductions are required to increase circulation and water exchange with Nantucket Sound. .

It is important to note that the analysis of future nitrogen loading to the Rushy Marsh estuarine systems focuses upon additional shifts in land-use from forest/grasslands to residential and commercial development. However, the MEP analysis indicates that significant increases in nitrogen loading can occur under present land-uses, due to shifts in occupancy, shifts from seasonal to year-round useage and increasing use of fertilizers (presently less than half of the parcels use lawn fertilizers). Therefore, watershed-estuarine nitrogen management must include management approaches to prevent increased nitrogen loading from both shifts in land-uses (new sources) and from loading increases of current land-uses. The overarching conclusion of the MEP analysis of the Rushy Marsh estuarine system is that restoration will necessitate a reduction in the present (2004) nitrogen inputs and management options to negate additional future nitrogen inputs.

Table ES-1. Existing total and sub-embayment nitrogen load to the estuarine waters of the Rushy Marsh Pond estuary system, observed nitrogen concentration, and sentinel system threshold nitrogen concentration.

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
RUSHY MARSH POND SYSTEM										
Rushy Marsh Pond System Total	**0.08**	**0.10**	**0.35**	**0.00**	**0.45**	**0.20**	**-0.20**	**0.45**	**1.17-1.11**	**0.50[8]**

[1] assumes entire watershed is forested (i.e., no anthropogenic sources)
[2] composed of non-wastewater loads, e.g. fertilizer and runoff and natural surfaces and atmospheric deposition to lakes
[3] existing wastewater treatment facility discharges to groundwater
[4] composed of combined natural background, fertilizer, runoff, and septic system loadings
[5] atmospheric deposition to embayment surface only
[6] composed of natural background, fertilizer, runoff, septic system atmospheric deposition and benthic flux loadings
[7] average of 2002 – 2005 data.
Individual yearly means and standard deviations in Table VI-6.
[8] threshold for sentinel sites located at the mid point of Rushy Marsh Pond (RM2).

Table ES-2. Present Watershed Load, Threshold Load, and the percent reduction necessary to achieve the Threshold Load for the Rushy Marsh Pond system, Town of Cotuit, Massachusetts.						
Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed reductions needed to achieve threshold load levels
LITTLE POND SYSTEM						
Rushy Marsh Pond System Total	**0.45**	**0.09**	**0.20**	**-0.11**	**0.18**	**-79.1%**

(1) Composed of combined natural background, fertilizer, runoff, and septic system loadings.
(2) Target threshold watershed load is the load from the watershed needed to meet the embayment threshold concentration identified in Table ES-1.
(3) Projected future flux (present rates reduced approximately proportional to watershed load reductions).
(4) Sum of target threshold watershed load, atmospheric deposition load, and benthic flux load.

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The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the Rushy Marsh Pond Estuary. Without these stewards and their efforts, this project would not have been possible.

First and foremost, we would like to recognize the significant time and effort in data collection and discussion spent by members of the Three Bays/Rushy Marsh Water Quality Monitoring Program and SMAST "volunteers". These individuals gave generously of their time to collect nutrient samples from this system, without which the present analysis would not have been possible. Of particular note are Lindsey Counsell of Three Bays Preservation, Coordinator of the Monitoring Program, and John Jacobsen of the Town of Barnstable, Coordinator of the Town-wide Monitoring Program. Similarly, many in the Town of Barnstable helped in this effort, the Barnstable Department of Public Works, the Barnstable Conservation Division and the Barnstable Nutrient Management Committee. The technical team would like to specifically acknowledge the efforts of Three Bays Preservation and the Friends of Rushy Marsh for their energy and perseverance in the stewardship of this coastal resource.

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SUGGESTED CITATION

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I. INTRODUCTION

The Rushy Marsh Embayment System is a simple estuary located within the Town of Barnstable on Cape Cod, Massachusetts with a southern shore bounded by water from Nantucket Sound (Figure I-1). Rushy Marsh is situated on the coast between the larger estuarine systems of Popponesset Bay and Three Bays. The watershed to Rushy Marsh is fully within the Town of Barnstable, making Barnstable the sole municipal steward of this small estuary. Virtually all watershed freshwater and nutrients enter Rushy Marsh via groundwater seepage, as there are no significant surface inflows to this system. As a result, there is little opportunity for nitrogen removal during transport from watershed source to estuarine waters.

Rushy Marsh Pond is a simple estuary, with a single embayment and highly restricted tidal inlet. The open water area of ~15 acres, makes Rushy Marsh Pond a great salt pond, similar to Oyster Pond in Falmouth. The present configuration of the Rushy Marsh Estuary is relatively new in the coastal landscape, as the southern coast of Cape Cod in the vicinity of Rushy Marsh is a moderately dynamic region, where natural wave and tidal forces continue to reshape the shoreline (see Chapters II and V). All the while, Rushy Marsh was formed by the flooding of a kettle pond as a result of rising sea level following the last glaciation approximately 18,000 years BP. The growth of salt marsh deposits along the northeastern portion of its shore, further enclosed the system, thus its classification as a lagoonal type estuary. This system appears to have persisted until the 1890's. USGS maps from 1893 show Rushy Marsh as a fully tidal estuary with salt marsh along its eastern and northern shores. An island exists off shore, Gull Island, which disappeared in about 1896 (Coast & Harbor Institute and Robert L. Fultz Associates 2002). During the 1900's the tidal inlet became restricted due to sedimentation deposits and the formation of a barrier beach. During this period Popponesset Spit elongated, then breached, with the northern portion finally attaching to the shoreline just north of Rushy Marsh around 1960. This formed a cove to Nantucket Sound running the length of and parallel to Rushy Marsh Pond. Over the next two decades this cove was filled by overwash and today all that remains is a small pond in the barrier beach (Figure I-1). However, the process of barrier beach formation and then overwash resulted in a freshening of Rushy Marsh Pond, even with efforts to keep the system tidal (pipes, culverts). By the turn of the century, the system was a brackish salt pond.

While Rushy Marsh Pond presently has a relatively low nitrogen load from its watershed, due to its small size and proportionally large undeveloped areas, it is still significantly impaired by nitrogen enrichment and is clearly eutrophic. This apparent paradox results from its very low tidal exchange rate, resulting from barrier beach processes restricting the inlet to Nantucket Sound. The low rate of tidal exchange serves to greatly increase the nitrogen sensitivity of this system, such that lower nitrogen inputs cause eutrophic conditions. In recent years the inlet periodically became closed and the pond level rose (due to groundwater inflow) to exceed sea level in the adjacent sound. This also resulted in a further decline in salinity to <1 ppt. The Town of Barnstable (through Conservation Department) working with Friends of Rushy Marsh and Three Bays Preservation partially restored tidal exchange (temporary fix). However, the persistent restricted tidal exchange has caused significant ecological degradation of the Rushy Marsh System. Even with the low watershed nitrogen loading, the low rate of nitrogen removal through tidal flushing results in high nitrogen levels, large phytoplankton blooms and periodic anoxia of bottom waters. In addition, the freshening of the pond waters has resulted in a loss of salt marsh area and a significant expansion of the areal coverage by the common reed, *Phragmites*. It is clear that restoration of Rushy Marsh Pond will require addressing the tidal restriction as one of the principal components, especially as the system has historically operated as a tidal estuary and its proximity to Nantucket Sound prevents its management as a freshwater system due to periodic overwash of salt water (similar to Oyster Pond, Falmouth, see Howes et al., 2005).

Figure I-1. Study region for the Massachusetts Estuaries Project analysis of the Rushy Marsh Pond System. Tidal waters enter the Pond from Nantucket Sound. Freshwaters enter from the watershed primarily through direct groundwater discharge and direct precipitation. Note the small brackish pond in the barrier beach.

Although the nitrogen load to Rushy Marsh Pond is relatively low, nitrogen management should also be considered in the development of the restoration design. The Town of Barnstable has been among the fastest growing towns in the Commonwealth over the past two decades and does have a centralized wastewater treatment system located in Hyannis. However, the Rushy Marsh watershed is not connected to any municipal sewerage system, but relies on privately maintained septic systems for treatment and disposal of wastewater. As existing and probable increasing levels of nutrients impact Barnstable's coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

As the stakeholder to the Rushy Marsh Pond System, the Town of Barnstable and its citizens have been active in promoting restoration of this system. This local concern also led to the conduct of several studies (see Chapter II) to support restoration and the Town is presently willing to implement an appropriate plan. To this end, Friends of Rushy Marsh and Three Bays Preservation Inc. have been active in field data collection. One of the key projects undertaken by Three Bays Preservation was to establish, in 2002, a nitrogen related water quality monitoring program within Rushy Marsh Pond. The Three Bays/Rushy Marsh Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD and over the past several years has been incorporated into Barnstable's Town-wide embayment monitoring program. This effort provides the quantitative watercolumn nitrogen data (2002-2005) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the present study.

Since the initial results of the Water Quality Monitoring Program and the coastal processes and land-use studies indicated that parts of the Rushy Marsh Estuary are currently impaired by nitrogen enrichment, the Town of Barnstable and Three Bays Preservation undertook additional site-specific data collection to support MEP's ecological assessment and modeling project. The effort was part of the Town's Wastewater Facilities Planning effort and was aimed at restoration of the estuarine resources. As a result of these efforts and to facilitate the development and implementation of restoration, it was appropriate to complete the evaluation of the Rushy Marsh salt pond system at this time.

The common focus of the Barnstable effort has been to gather site-specific data on the current nitrogen related water quality throughout the Rushy Marsh Pond System and determine its relationship to tidal flushing and watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program, and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each major sub-embayment. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to develop and implement management alternatives needed by the Town of Barnstable for estuarine restoration/protection. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years, most notably within the Departments of Conservation and Public Works and from members of the local non-governmental organizations (NGO's), Three Bays Preservation and Friends of Rushy Marsh. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Barnstable to develop and evaluate the most cost effective management alternatives to restore this coastal resource.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over this assimilative capacity, which begins to cause declines in their ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities and the food chain which they support. At higher levels, enhanced nitrogen loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human uses. However like nutrients, bacterial contamination is frequently related to changes in land-use as watersheds become more developed. The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities.

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within Southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of Barnstable) are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries.

Municipalities are seeking guidance on the assessment of nitrogen sensitive embayments, as well as available options for meeting nitrogen goals and approaches for restoring impaired systems. Many of the communities have encountered problems with "first generation" watershed based approaches, which do not incorporate estuarine processes. The appropriate method must be quantitative and directly link watershed and embayment nitrogen conditions. This "Linked" Modeling approach must also be readily calibrated, validated, and implemented to support planning. Although it may be technically complex to implement, results must be understandable to the regulatory community, town officials, and the general public.

The Massachusetts Estuaries Project represents the next generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MA DEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts.

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the DEP and municipalities with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system will serve as the basis for the development of

Total Maximum Daily Loads (TMDLs). Development of TMDLs is required pursuant to Section 303(d) of the Federal Clean Water Act. TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an outline of an implementation plan. For this project, the DEP recognizes that there are likely to be multiple ways to achieve the desired goals, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, DEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly) through the Comprehensive Wastewater Management Planning process.

In appropriate estuaries, TMDL's for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). In these cases, the MEP (through SMAST) will produce a Technical Analysis and Report to support a bacterial TMDL for the system from which MA DEP develops the TMDL. The goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation.

In contrast to the bacterial program, the MEP nitrogen program also includes site-specific habitat assessments and watershed/embayment modeling approaches to develop and assess various nitrogen management alternatives for meeting selected nitrogen goals supportive of restoration/protection of embayment health.

The major MEP nitrogen management goals are to:

- provide technical analysis and supporting documentation to Towns as a basis for sound nutrient management decision making towards embayment restoration
- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of each of the 89 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment's model "alive" to address future municipal needs.

The core of the Massachusetts Estuaries Project analytical method is the Linked Watershed-Embayment Management Modeling Approach. This approach represents the "next generation" of nitrogen management strategies. It fully links watershed inputs with embayment circulation and nitrogen characteristics. The Linked Model builds on and refines well accepted basic watershed nitrogen loading approaches such as those used in the Buzzards Bay Project, the CCC models, and other relevant models. However, the Linked Model differs from other nitrogen management models in that it:

- requires site specific measurements within each watershed and embayment;
- uses realistic "best-estimates" of nitrogen loads from each land-use (as opposed to loads with built-in "safety factors" like Title 5 design loads);
- spatially distributes the watershed nitrogen loading to the embayment;
- accounts for nitrogen attenuation during transport to the embayment;
- includes a 2D or 3D embayment circulation model depending on embayment structure;
- accounts for basin structure, tidal variations, and dispersion within the embayment;
- includes nitrogen regenerated within the embayment;
- is validated by both independent hydrodynamic, nitrogen concentration, and ecological data;

- is calibrated and validated with field data prior to generation of “what if” scenarios.

The Linked Model has been applied for watershed nitrogen management in approximately 15 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach’s greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing “what if” scenarios for evaluating watershed nitrogen management options.

The Linked Watershed-Embayment Model when properly parameterized, calibrated and validated for a given embayment becomes a nitrogen management planning tool, which fully supports TMDL analysis. The Model facilitates the evaluation of nitrogen management alternatives relative to meeting water quality targets within a specific embayment. The Linked Watershed-Embayment Model also enables Towns to evaluate improvements in water quality relative to the associated cost. In addition, once a model is fully functional it can be “kept alive” and updated for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

Linked Watershed-Embayment Model Overview: The Model provides a quantitative approach for determining an embayment’s: (1) nitrogen sensitivity, (2) nitrogen threshold loading levels (TMDL) and (3) response to changes in loading rate. The approach is both calibrated and fully field validated and unlike many approaches, accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics (Figure I-2). This methodology integrates a variety of field data and models, specifically:

- Watercolumn Monitoring - multi-year embayment nutrient sampling
- Hydrodynamics -
  - embayment bathymetry
  - site specific tidal record
  - current records (in complex systems only)
  - hydrodynamic model
- Watershed Nitrogen Loading
  - watershed delineation
  - stream flow (Q) and nitrogen load
  - land-use analysis (GIS)
  - watershed N model
- Embayment TMDL - Synthesis
  - linked Watershed-Embayment N Model
  - salinity surveys (for linked model validation)
  - rate of N recycling within embayment
  - D.O record
  - Macrophyte survey
  - Infaunal survey

I.2 SITE DESCRIPTION

Rushy Marsh Pond is a simple estuary, with a single embayment and highly restricted tidal inlet. The open water area of ~15 acres, makes Rushy Marsh Pond a great salt pond. The Rushy Marsh Estuarine System presently exchanges tidal water with Nantucket Sound through a 2’ pipe running through the barrier beach in the general location of the historic natural inlet (see Chapter V). The present "inlet" was installed to lower pond levels and to restore tidal exchange that had virtually ceased in late 2002 or early 2003. As mentioned above, the severe tidal restriction of this system has resulted in nitrogen related habitat declines and shifts in wetland communities. For the MEP analysis, Rushy Marsh Pond is the principal estuarine basin in the modeling and thresholds analysis.

Rushy Marsh Pond is currently a brackish embayment with limited tidal exchange with adjacent Nantucket Sound. The basin consists of a drowned kettle pond and is relatively deep (>2m) compared to nearby typical drowned river valley estuaries (e.g. Green Pond, Falmouth, 1m). At present, the embayment is eutrophic and has periodic summer phytoplankton blooms and anoxia. MEP surveys found sediments consistent with eutrophication, i.e. very soft organic/sulfidic muds. The result is a system virtually devoid of benthic animals. The associated wetlands have also been altered as a result of the varying inlet and tidal exchange rates. Salt marsh is no longer found bordering the pond. Brackish wetland plants, principally Phragmites, now fringe the basin with freshwater marsh slightly inland in the northern region. The loss of salt marsh is likely the result of both the freshening of the system (periodically to <1ppt) and to the periodic rise in standing water (several feet), which would "drown" the salt marsh zone under severe tidal restriction or complete blockage.

As management alternatives are being developed and evaluated, it is important to note that the Rushy Marsh System is naturally a relatively dynamic system and has undergone significant alterations to its hydrologic and biological systems over the past 100 years. Within such dynamic systems, restoration alternatives need to be evaluated relative to the system's "maximum level of sustainable environmental health" in addition to traditional standards..

While the nutrient related health of the Rushy Marsh Estuary as it exists today is linked to changes wrought by natural processes and human activities, it is the physical structure of the system laid down by the retreat of the Laurentide Ice Sheet that still controls much of the Systems' tolerance to nutrient inputs. The physical structure, shape and depth of a coastal embayment plays a major role in its susceptibility to ecological impacts from nutrient loading. Physical structure (geomorphology), which includes embayment bathymetry, inlet configuration and saltwater reaches, when coupled with the tidal range of the adjacent open waters, determines the system's rate of flushing. System flushing rate is generally the primary factor for removing nutrients from active cycling within coastal bays and harbors like Rushy Marsh Pond. As a result maximizing system flushing is one of the standard approaches for controlling the nutrient related health of coastal embayments.

The present configuration of the Rushy Marsh Estuary is relatively new in the coastal landscape, as the southern coast of Cape Cod in the vicinity of Rushy Marsh is a moderately dynamic region, where natural wave and tidal forces continue to reshape the shoreline (see Chapters II and V). All the while, Rushy Marsh was formed by the flooding of a kettle pond as a result of rising sea level following the last glaciation, approximately 18,000 years BP. The growth of salt marsh deposits along the northeastern portion of its shore further enclosed the system resulting in the system's classification as a lagoonal type estuary. This system appears to have persisted until the 1890's. USGS maps from 1893 show Rushy Marsh as a fully tidal estuary with salt marsh along its eastern and northern shores. An island exists off shore, Gull Island, which disappeared in about 1896 (Gaines and Fultz 2005). During the 1900's the tidal inlet became restricted due to sedimentation deposits and the formation of a barrier beach. During this period Popponesset Spit elongated, then breached, with the northern portion finally attaching to the shoreline just north of Rushy Marsh around 1960. This formed a cove to

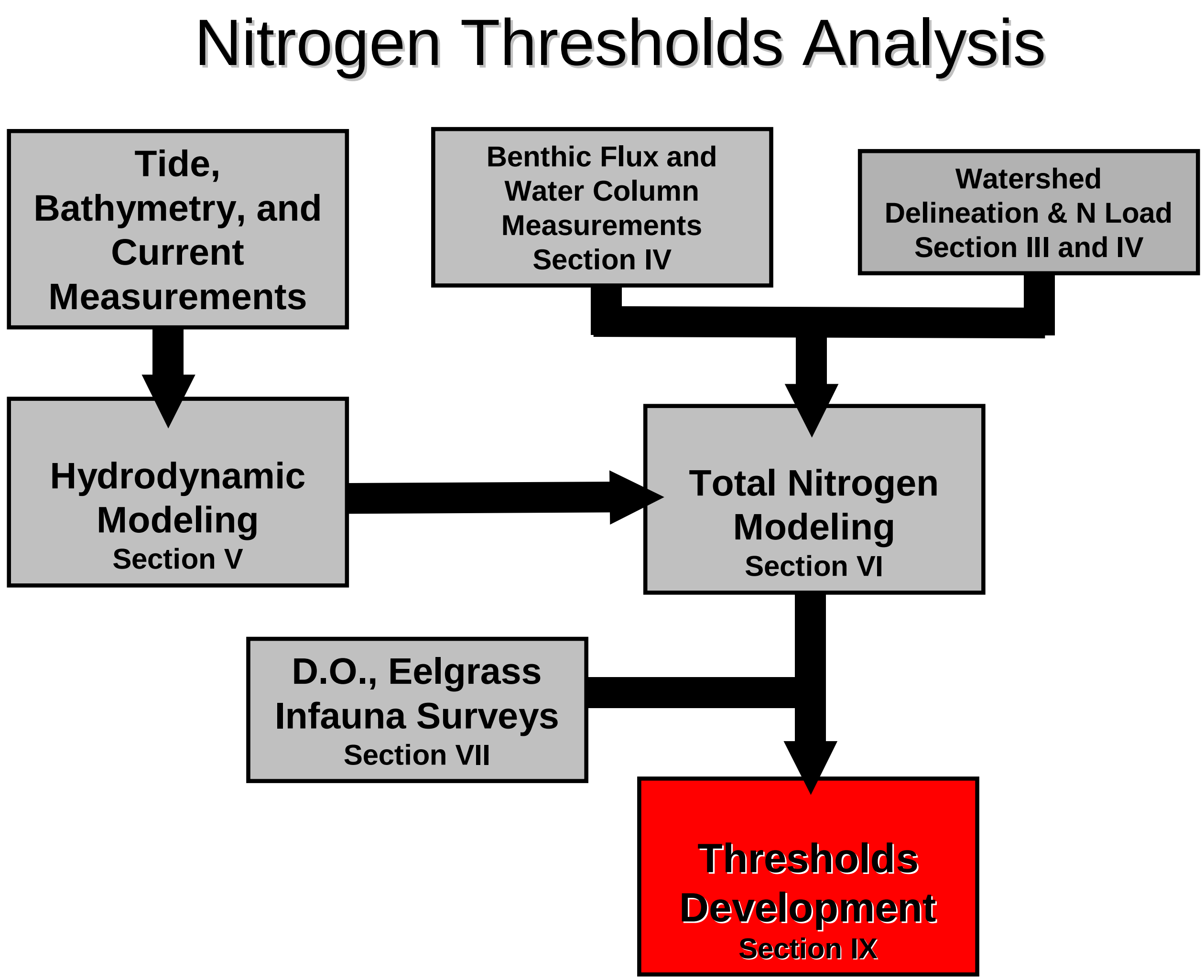


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

Nantucket Sound running the length of and parallel to Rushy Marsh Pond. Over the next two decades this cove was filled by overwash and today all that remains is a small pond in the barrier beach. However, the process of barrier beach formation and then overwash resulted in a freshening of Rushy Marsh Pond, even with efforts to keep the system tidal (pipes, culverts). By the turn of the century, the system was a brackish salt pond.

While Rushy Marsh Pond presently has a relatively low nitrogen load from its watershed, due to its small size and proportionally large undeveloped areas, it is still significantly impaired by nitrogen enrichment and is clearly eutrophic. This apparent paradox results from its very low tidal exchange rate, resulting from barrier beach processes restricting the inlet to Nantucket Sound. The low rate of tidal exchange serves to greatly increase the nitrogen sensitivity of this system, so that lower nitrogen inputs are needed to cause eutrophic conditions. In recent years the inlet periodically became closed and the pond level rose (due to groundwater inflow) to exceed sea level in the adjacent sound. This also resulted in a further decline in salinity to <1 ppt. The Town of Barnstable (through Conservation Department) working with Friends of Rushy Marsh and Three Bays Preservation partially restored tidal exchange (temporary fix). However, the persistent restricted tidal exchange has caused significant ecological degradation of the Rushy Marsh System. Even with the low watershed nitrogen loading, the low rate of nitrogen removal through tidal flushing results in high nitrogen levels, large phytoplankton blooms and periodic anoxia of bottom waters. In addition, the freshening of the pond waters has resulted in a loss of salt marsh area and a significant expansion of the areal coverage by the common reed, *Phragmites*. It is clear that restoration of Rushy Marsh Pond will require addressing the tidal restriction as one of the principal components, especially as the system has historically operated as a tidal estuary and its proximity to Nantucket Sound prevents its management as a freshwater system due to periodic overwash of salt water (similar to Oyster Pond, Falmouth, see Howes et al., 2005).

By far the greatest changes to the Rushy Marsh Pond watershed have also occurred during the last 100 years. The most obvious change has been the dramatic shift in land-use to residential housing during the last half of the 1900's. With this shift and the advent of fertilized lawns, has come an increase in the amount of nitrogen, which enters the estuary during a period where its sensitivity has increased due to reductions in tidal exchange rates. The previous large shifts in land-use, primarily from forest to agriculture did not have the same resultant enhancement in nitrogen loading, as agriculture generally recycled nitrogen (as opposed to commercial fertilizers) and the population was <10% of today.

The MEP analysis focused on determining the extent to which the environmental health of the Rushy Marsh System will be enhanced by restoration of tidal exchange with the high quality waters of Nantucket Sound, relative to the potential need to manage watershed nutrient loading. The goal of the MEP and the local stakeholders is to restore the estuarine habitats within Rushy Marsh to meet the high level of quality designated by the State Water Quality Standards for the benefit of both present and future generations.

I.3 NUTRIENT LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Rushy Marsh System, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer minerals (Weiskel and Howes 1992). Since even Cape Cod “rivers” are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1996, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidal reaches within Rushy Marsh Estuary presently follow this general pattern, where the primary nutrient of eutrophication in these systems is nitrogen.

Nutrient related water quality decline represents one of the most serious threats to the ecological health of the nearshore coastal waters. Coastal embayments, because of their enclosed basins, shallow waters and large shoreline area, are generally the first indicators of nutrient pollution from terrestrial sources. By nature, these systems are highly productive environments, but nutrient over-enrichment of these systems worldwide is resulting in the loss of their aesthetic, economic and commercially valuable attributes.

Each embayment system maintains a capacity to assimilate watershed nitrogen inputs without degradation. However, as loading increases a point is reached at which the capacity (termed assimilative capacity) is exceeded and nutrient related water quality degradation occurs. This point can be termed the “nutrient threshold” and in estuarine management this threshold sets the target nutrient level for restoration or protection. Because nearshore coastal salt ponds and embayments are the primary recipients of nutrients carried via surface and groundwater transport from terrestrial sources, it is clear that activities within the watershed, often miles from the water body itself, can have chronic and long lasting impacts on these fragile coastal environments.

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts has been the site of intensive efforts in this area (Eichner et al., 1998, Costa et al., 1992 and in press, Ramsey et al., 1995, Howes and Taylor, 1990, and the Falmouth Coastal Overlay Bylaw). While each approach may be different, they all focus on changes in nitrogen loading from watershed to embayment, and aim at projecting the level of increase in nitrogen concentration within the receiving waters. Each approach depends upon estimates of circulation within the embayment; however, few directly link the watershed and hydrodynamic models, and virtually none include internal recycling of nitrogen (as was done in the present effort). However, determination of the “allowable N concentration increase” or “threshold nitrogen concentration” used in previous studies had a significant uncertainty due to the need for direct linkage of watershed and embayment models and site-specific data. In the present effort we have integrated site-specific data on nitrogen levels and the gradient in N concentration throughout the Rushy Marsh System monitored by the Town of Barnstable/Three Bays Preservation Water Quality Monitoring Program, with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) utilized to “tune” general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Unfortunately, Rushy Marsh Pond is presently beyond its ability to assimilate additional nutrients without impacting their ecological health. This is in significant part due to the very restricted tidal exchange with Nantucket Sound waters. Nitrogen levels are elevated, eelgrass beds have not been observed within Rushy Marsh Pond for the past half century and there are large summer phytoplankton blooms and periodic anoxia of bottom waters. The result is that nitrogen management of the Rushy Marsh Pond system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed "eutrophication" and when the nutrient loading is primarily from human activities, "cultural eutrophication". Although the influence of human-induced changes has increased nitrogen loading to the systems and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within the Rushy Marsh System could potentially occur without man's influence and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a "pristine" system. In addition, to the impairment of Rushy Marsh Pond's sub-tidal habitats, there has been a loss of emergent salt marsh from the system stemming from the restricted tidal exchange in recent years. Restriction of the tidal inlet has resulted in freshening of the estuarine waters (sometimes to <1ppt) and an increase in both the mean tide level and during closures, the high water level. At present, the wetlands associated with Rushy Marsh are dominated by fresh and brackish water plants, with large areas of the common reed, *Phragmites*. It appears that the tidal restriction is affecting both the subtidal and intertidal resources, albeit through different mechanisms.

I.4 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important "boundary conditions" (e.g. watershed derived and offshore nutrient inputs) for water quality modeling of the Rushy Marsh System; however, a thorough understanding of hydrodynamics is required to accurately determine nitrogen concentrations within each system. Therefore, water quality modeling of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives. Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. The spread of pollutants may be analyzed from tidal current information developed by the numerical models.

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Rushy Marsh Pond under a variety of nitrogen input (loading) and hydrodynamic conditions. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations (both actual and projected under various inlet configurations) was employed. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all nitrogen entering the Rushy Marsh System is transported by freshwater, almost entirely through groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout Rushy Marsh Pond were taken from the Water Quality Monitoring Program (a coordinated effort between the Town of Barnstable, Three Bays Preservation and the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the Systems (2002-2005) were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Rushy Marsh System for the Town of Barnstable. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Nantucket Sound (Section IV). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section IV. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of the Estuary in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined Bay threshold for restoration. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative restoration options for this system. Finally, analyses of the Rushy Marsh System were relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of various inlet options to improve nitrogen related water quality (and wetland communities). The results of the nitrogen modeling for each scenario have been presented (Section IX).

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

In most marine and estuarine systems, such as Rushy Marsh Pond in the Town of Barnstable, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the eutrophication management approaches via the reduction of nitrogen loads has also generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2003).

Until recently, these tools for predicting loads and concentrations tended to be generic in nature and overlooked some of the site-specific characteristics associated with a given water body. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including Rushy Marsh Pond.

A number of studies have been performed regarding the south shore of Cape Cod geomorphology. Specific to Rushy Marsh Pond, two Woods Hole Oceanographic Institution reports: Aubrey and Gaines (1982) and Aubrey and Goud (1983) illustrated the historic processes governing tidal exchange over the past 100 years. For the Rushy Marsh estuarine system, the process of barrier spit elongation and breaching has had a significant influence on tidal exchange, even though the barrier spit is part of the adjacent Popponesset Bay System to the west. Over the past 100 years, the Popponesset Beach barrier elongation and breaching processes have governed the stability of the Rushy Marsh inlet. Between the late 1800s and 1950, the Popponesset barrier elongated past the Rush Marsh tidal inlet. Due to the influx of sediment associated with this barrier elongation, the natural tidal inlet to Rushy Marsh closed. As described in Aubrey and Goud (1983), the loss of nearly one-half of the Popponesset Beach barrier between 1954 and the early 1980s led to concerns regarding future barrier spit migration. According to Aubrey and Gaines (1982), the present spit length has been historically the stable configuration. It wasn't until after about 1860 that the spit began to grow past its present location.

As the barrier spit elongated between the early 1900s and the mid-1950s due to regional littoral drift, the inlet channel to Popponesset Bay became less efficient, where the tide height within Popponesset Bay decreased and the lag time between high tide in the estuary and Nantucket Sound increased. This increase in tidal attenuation was remedied in 1954, when a hurricane breached the barrier spit, creating an efficient inlet to Popponesset Bay in the vicinity of the present inlet. Once the spit had breached, the remnants of the spit east of the inlet gradually overwashed and rejoined the shoreline (primarily in the vicinity of Rushy Marsh). This inlet spit growth and breaching process has been documented extensively for the southeastern coast of Massachusetts (e.g. Fitzgerald, 1993).

The specific influence of barrier elongation and breaching upon the Rushy Marsh system was described in a report to the Town of Barnstable prepared by the Coast & Harbor Institute and Robert L. Fultz Associates (2002). Although the inlet to Rushy Marsh had naturally closed in the early 1900s, the influx of littoral sediment caused by the 1954 breach of the Popponesset barrier further widened the barrier beach system fronting Rushy Marsh. Efforts to maintain an effective inlet near the southern end of the Pond have been complicated by the unstable nature of the shoreline, the relatively weak littoral drift that continues to supply sediment to this region, and the small potential tidal prism exiting Rushy Marsh. Based upon limited sampling of water quality parameters (dissolved oxygen, salinity, and turbidity), Coast & Harbor Institute and Robert L. Fultz Associates concluded that the pond productivity is moderate or low, and the poor water clarity might be a result of tannins in the water, rather than algal accumulation.

As the stakeholder to the Rushy Marsh Pond System, the Town of Barnstable and its citizens have been active in promoting restoration of this system. This local concern also led to the establishment, in 2002, of a nitrogen related water quality monitoring program for Rushy Marsh by Three Bays Preservation. The program was an extension of the effort for the adjacent Three Bays Estuary. The Three Bays/Rushy Marsh Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD and over the past several years has been incorporated into Barnstable's Town-wide embayment monitoring program. The initial findings of the monitoring program are that Rushy Marsh Pond is currently impaired by nitrogen enrichment (i.e. eutrophic). As a result of the restriction of tidal exchange nitrogen entering the system is generally recycled rather being flushed out to Nantucket Sound. The result is elevated nitrogen levels, phytoplankton blooms, macroalgal accumulations and periodic depletion of dissolved oxygen in bottom waters. In addition, restriction of tidal exchange allows the pond to freshen and at present Rushy Marsh Pond is a brackish salt pond only periodically attaining ½ strength seawater salinities.

As part of on-going research and engineering efforts related to Rushy Marsh Pond, the geologic history of the pond was determined as was the recent history of shoreline change (See Section V). The short and long term trends in coastal processes as relate to the ecological health of Rushy Marsh Pond set an important background for the present restoration and management of this system. The Three Bays/Rushy Marsh Water Quality Monitoring Program provides the quantitative watercolumn nitrogen data (2002-2005) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the present study. In addition, for the MEP modeling analysis, the data from the previous studies were evaluated relative to the needs of the Linked Watershed-Embayment Model.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by to the USGS to organize and analyze the available data utilize up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, surface water/groundwater interaction, groundwater travel time, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Rushy Marsh embayment system. The Rushy Marsh Pond System and its watershed is fully located within the Town of Barnstable, Massachusetts and is situated between Popponesset Bay to the west and Three Bays to the east.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Rushy Marsh Pond system under evaluation by the Project Team. Unlike larger estuaries, Rushy Marsh did not require additional modeling to sub-divide the overall watershed into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from "new" development, which has not yet reached the receiving estuarine waters. The Rushy Marsh Pond embayment functions as a single horizontally mixed basin. There are no public water supply wells or "significant" streams or fresh ponds (e.g. >10 acres or which capture large amounts of groundwater) within its watershed. Furthermore, given the relatively small watershed, all of the recharged groundwater reaches the estuary in less than 10 years. Therefore a single watershed was used for the Linked Watershed-Embayment Management Model. Rushy Marsh Pond is similar to the larger Oyster Pond Estuary in Falmouth.

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by the land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water inputs and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to the stream and the portion of the groundwater system that discharges directly into the estuary as groundwater seepage. In the case of Rushy Marsh Pond, direct groundwater discharge was the sole pathway, although a stream may have existed prior to road construction. A field survey did not find any surface water inflow to the Pond of sufficient flow (generally ~0.0005 m^3 s^{-1} is required) to support a MEP stream gauge.

III.2 MODEL DESCRIPTION

Contributing areas to the Rushy Marsh system and local freshwater bodies were delineated using a regional model of the Sagamore Lens (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh et al., 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Rushy Marsh Pond basin.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. Layers 1-7 are stacked above NGVD 29 and layers 8 to 20 extend below. Layer 18 has a thickness of 40 feet and layer 19 extends to 240 feet below sea level. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens. Since the Rushy Marsh Pond watershed is relatively distant from the top portion of the Sagamore Lens (i.e. it is near the coast), most of the uppermost layers of the groundwater model are inactive in its delineation.

The glacial sediments that comprise the aquifer of the Sagamore Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Rushy Marsh watershed is situated in the midst of the very-coarse grained Mashpee Pitted Plain deposits (Masterson et al., 1996). Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water level and streamflow data collected in May 2002.

The model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the 15% consumptive loss and measured discharge at municipal treatment facilities, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within designated residential areas utilizing on-site septic systems. Since the watershed to Rushy Marsh is lacking municipal sewers, this area is part of the Barnstable residential area in the groundwater model.

III.3 RUSHY MARSH POND CONTRIBUTORY AREA

Newly revised watershed and sub-watershed boundaries for the Rushy Marsh Estuary were determined by the United States Geological Survey (USGS). Model outputs of MEP watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, and (c) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. This task was simplified for Rushy Marsh Pond as the contributing area is best represented as a single watershed and no great fresh ponds (>10 acres) are present. However, the USGS modeled output did require accounting for the grid spacing and incorporation of correct shoreline configuration (Figure III-1).

The daily discharge volume for the watershed was calculated by the groundwater model and the volume was used to assist in the salinity calibration of the tidal hydrodynamic models The MEP delineation determined that groundwater travel times were less than 10 yrs throughout the watershed.

The Rushy Marsh watershed from the USGS modeling effort is estimated to have an average annual freshwater discharge of 249,247 m^3 yr^{-1} to Rushy Marsh Pond. This estimate agrees well with estimates of freshwater input (watershed plus rainfall) calculated from a simple mixing model and measured dilution rates of Rushy Marsh Pond salinities (September 2002 through July 2003), when inputs of tidal water to the Pond were negligible. The freshwater estimate from the salinity dilution model was ~20% less than from the groundwater inflow and net rainwater inputs. However, this most likely results from salt diffusion from the pond sediments or slight amounts of tidal water entering over the sampling interval. The freshwater input estimates based upon salinity dilution were similar for both short (~1 month) and long (9 month) intervals. These data support the areal extent of the watershed delineation from the USGS groundwater model. The watershed delineation completed for the MEP project is the first for the Rushy Marsh Estuary.

The groundwater modeling approach to watershed delineation allows the Rushy Marsh delineation to be brought into congruence with adjacent watersheds and their supporting data. The evaluation of the Rushy Marsh watershed on the local and sub-regional scales (including the incorporation of new and old data) is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the downgradient estuary.

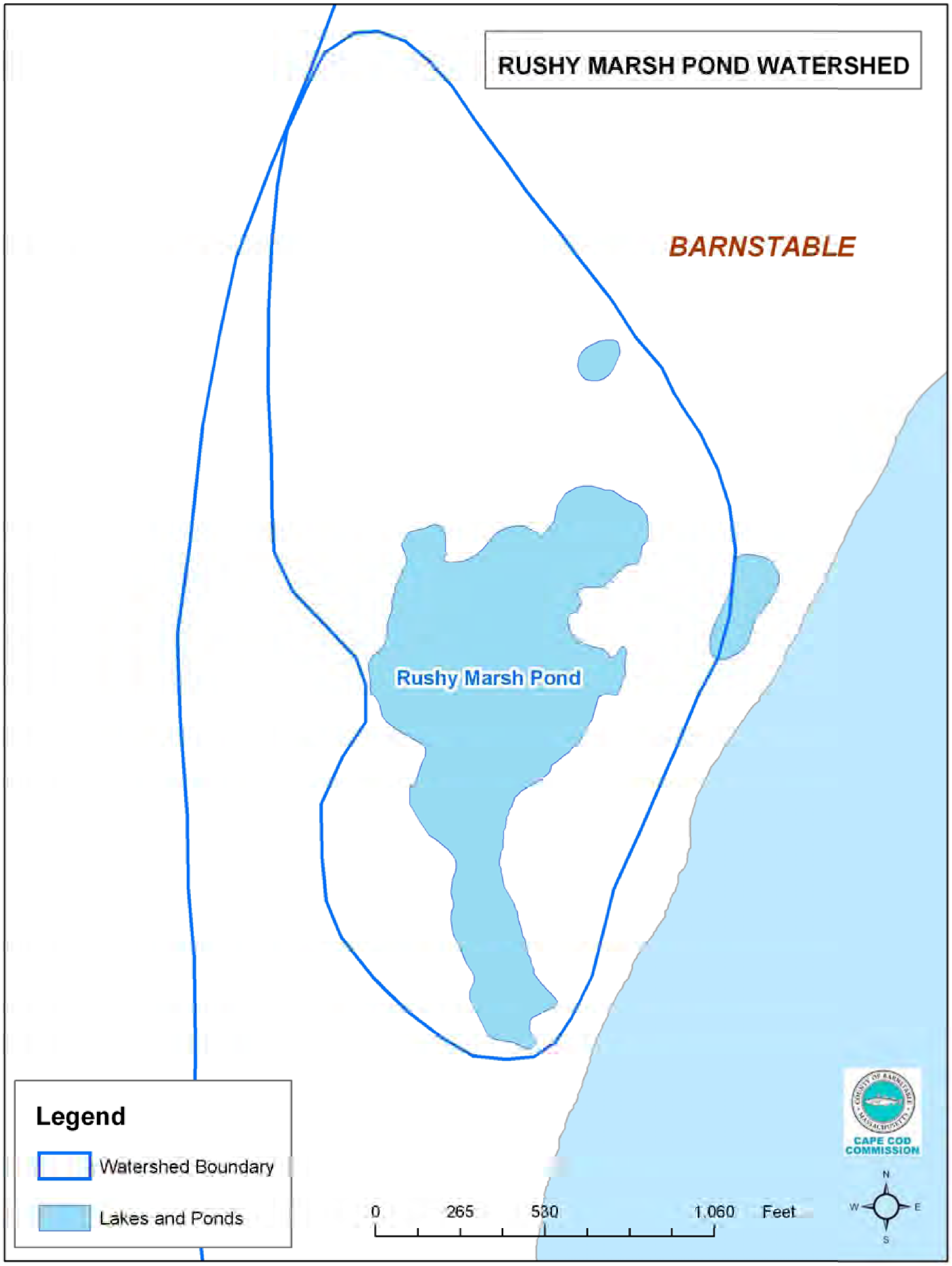


Figure III-1. Watershed delineation for the Rushy Marsh Estuary. All recharge reaches the estuary within ten years. A single watershed to the embayment was selected based upon the functional estuarine unit in the water quality model (see Chapter VI). The Popponesset Bay watershed is to the west.

IV. WATERSHED NITROGEN LOADING TO EMBAYMENT: LAND USE, STREAM INPUTS, AND SEDIMENT NITROGEN RECYCLING

IV.1 WATERSHED LAND USE BASED NITROGEN LOADING ANALYSIS

Management of nutrient related water quality and habitat health in coastal waters requires determination of the amount of nitrogen transported by freshwaters (surface water flow, groundwater flow) from the surrounding watershed to the receiving embayment of interest. In southeastern Massachusetts, the nutrient of management concern for estuarine systems is nitrogen and this is true for the Rushy Marsh Pond system. Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. This latter natural attenuation process results from biological processes that naturally occur within ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from sediments. Sediment nitrogen recycling results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if it is not included in determination of summertime nitrogen load.

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) to the Rushy Marsh embayment system (Section III). The Rushy Marsh contributing area is all within 10 years time-of-travel and includes only direct discharge to the estuary (e.g. no streams or great fresh ponds). The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to the embayment (see Chapter III).

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes and the time of groundwater travel provided by the USGS watershed model. As mentioned above, all of the watershed to Rushy Marsh is within 10 years worth of groundwater flow (Figure IV-1). After reviewing land use development records, and water quality modeling, it was determined that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries.

In order to determine nitrogen loads from the watershed detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed studies is applied to other portions of the watershed. The Linked Watershed-Embayment Management Model (Howes and Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land-uses and pre-determined nitrogen loading rates. For the Rushy

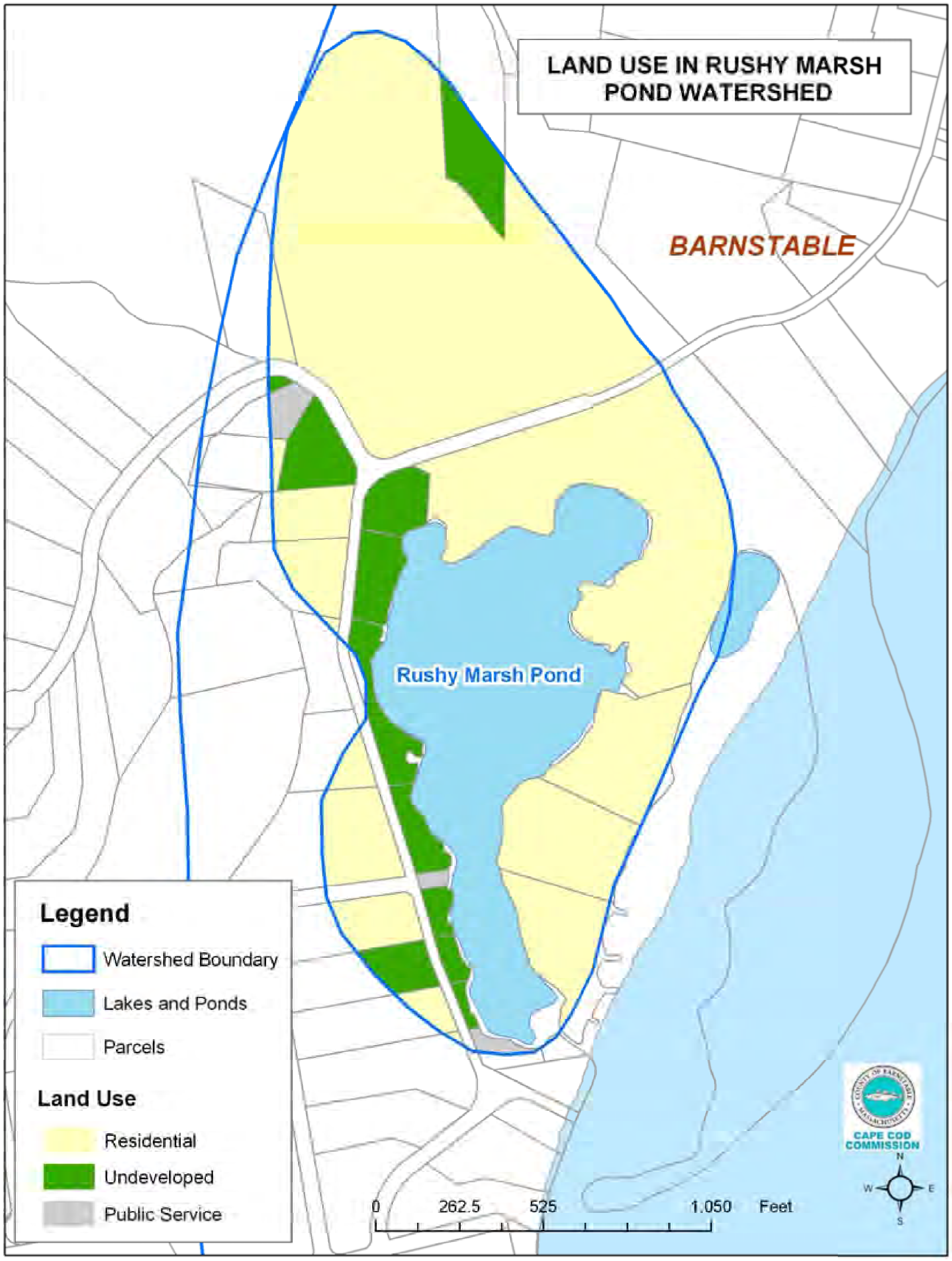


Figure IV-1. Land-use coverage in the Rushy Marsh watershed. Land use classifications are based on assessors' records provided by the Town of Barnstable. Note the shoreline overlaying the water of Nantucket Sound represents the shoreline of 19XX, while the present barrier beach shoreline is shown to the west (white/blue interface). The small pond in the barrier beach is all that remains today of the larger cove created by the spit.

Marsh embayment system, the model used Town of Barnstable specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local watershed-specific data (such as parcel by parcel water use). Determination of the nitrogen loads required obtaining watershed-specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the “potential” nitrogen load to each receiving embayment, since attenuation during transport has not yet been included.

Based upon the lack of streams or great ponds (>10 acres) within the Rushy Marsh Pond watershed, natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) was determined to be negligible. Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. However, even if attenuation of nitrogen is occurring during transport, given the distribution of the nitrogen sources and small ponds that do exist within the watershed, nitrogen loading to the estuary would only be slightly (~10%) overestimated. Based upon these considerations, the MEP Technical Team used the conservative estimate of nitrogen loading based upon direct groundwater discharge. Internal nitrogen recycling was also determined throughout the tidal reach of the Rushy Marsh embayment; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Estuaries Project staff obtained digital parcel and tax assessors data from the Town of Barnstable. Digital parcel and land use data are from 2004 and were obtained from the Town of Barnstable GIS Unit. These land use databases contain traditional information regarding land use classifications (Massachusetts DOR, 2002) plus additional information developed by the Town. The parcel coverages and assessors' database were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows land uses within the watershed area contributing freshwater and nitrogen to the Rushy Marsh estuarine waters. Land use in the Rushy Marsh Pond watershed is one of four land use types: 1) residential, 2) undeveloped, 3) agricultural, and 4) public service/government, including road rights-of-way. “Public service” is the land classification assigned by the Massachusetts Department of Revenue to tax exempt properties, including lands owned by government (e.g., wellfields, schools, open space, roads) and private groups like churches and colleges. Massachusetts Assessors land uses classifications (MADOR, 2002), which are common to all towns were are aggregated into these four land use categories.

In the Rushy Marsh watershed, the predominant land use based on area is residential, which accounts for 79% (70.1 acres) of the watershed area; undeveloped land is the second highest percentage of the watershed (11%). In addition, 38% of the parcels in the system watershed are classified as single family residences (MADOR land use code 101) and single family residences account for 97% of the residential land area. There are no properties classified as commercial or industrial in the Rushy Marsh watershed, and there are no municipal well areas. The remaining 9% of the land-use is divided between public service (roads, rights-of-way) and agriculture, 5% and 4%, respectively (Figure IV-2).

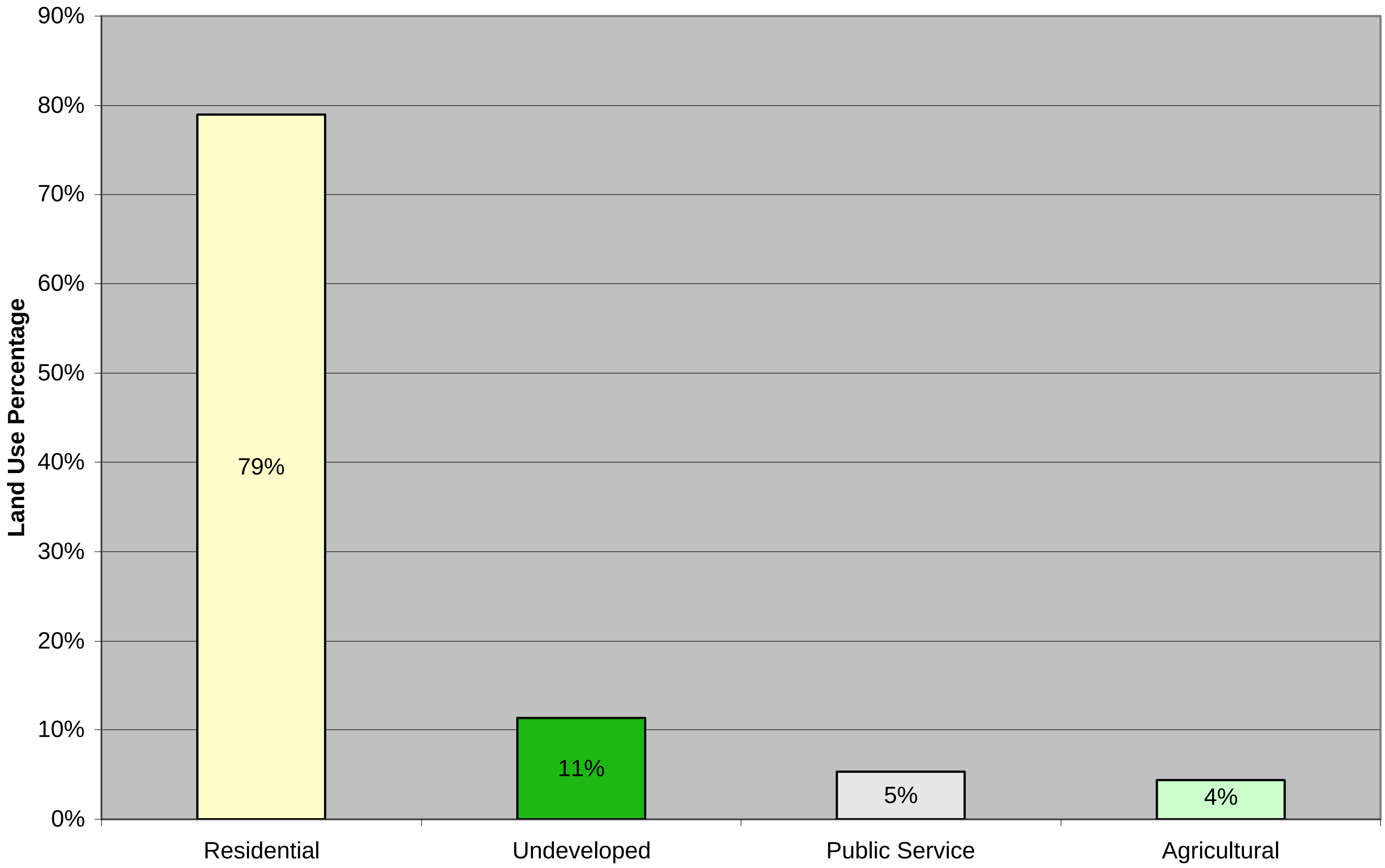


Figure IV-2. Distribution of land-uses within the Rushy Marsh watershed. Single family residential parcels make up 97% of the residential land-use.

In order to estimate wastewater flows within the Rushy Marsh Pond watershed, MEP staff also obtained parcel by parcel water use information for the Cotuit Water District from the Town of Barnstable GIS Unit. The Cotuit water use data is for one year (October 2002 through October 2003). Water use information was linked to the parcel and assessors data using GIS techniques. Water use for each parcel was converted to an annual volume for purposes of the nitrogen loading calculations. All parcels use on-site septic treatment and disposal of wastewater as there are no municipal wastewater treatment facilities (WWTFs) in the Rushy Marsh watershed.

As noted previously, all wastewater within the Rushy Marsh Pond watershed is returned to the aquifer through individual on-site septic systems. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration, and an assumed consumptive loss of water before the remainder is treated in a septic system. All wastewater in the Rushy Marsh watershed is returned to the aquifer through septic systems.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

Similar to many other watershed nitrogen loading analyses, the Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a per Capita Nitrogen load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon directly measured septic system and per capita loads determined on Cape Cod or in similar geologic settings (Nelson et al. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter et al. 1990, Brawley et al. 2000, Howes and Ramsey 2001, Costa et al. 2002). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}. However, given the seasonal shifts in occupancy in many of the watersheds throughout southeastern Massachusetts, census data yields accurate estimates of total population only in specific watersheds (see below). To correct for this uncertainty, the MEP employs a water-use approach. The water-use approach (Weiskel and Howes 1992) is applied on a parcel-by-parcel basis within a watershed, where usually an average of multiple years annual water meter data is linked to assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down-gradient in the aquifer. All losses within the septic system are incorporated. For example, information developed at the DEP Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa et al. 2002). Aquifer studies indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

In its application of the water-use approach to septic system nitrogen loads, the MEP has ascertained for the Estuaries Project region that while the per capita septic load is well constrained by direct studies, the consumptive use and nitrogen concentration data are less certain. As a result, the MEP has derived a combined term the effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per cubic meter) to nitrogen load (N grams). This term uses a per capita nitrogen load of 2.1 kg N person-yr^{-1} and is based upon direct measurements and corrects for changes in concentration that result from per capita shifts in water-use (e.g. due to installing low plumbing fixtures or high versus low irrigation usage, etc.).

The resulting nitrogen loads, based upon the above approach have been validated in a number of long and short term field studies where integrated measurements of nitrogen discharge from watersheds could be directly measured. For example, Weiskel and Howes (1991, 1992) conducted a detailed watershed/stream tube study that monitored septic systems, leaching fields and the transport of the nitrogen in groundwater to adjacent Buttermilk Bay. This monitoring resulted in estimated annual per capita nitrogen loads of 2.17 kg (as published) to 2.04 kg (if new attenuation information is included). The selected "effective N loading coefficient" also agrees with available watershed nitrogen loading analyses conducted on other Cape Cod estuaries. Aside from the concurrence observed between modeled and observed nitrogen concentrations in the estuary analyses completed under the MEP, analyses of other estuaries completed using this effective septic system nitrogen loading coefficient, the modeled loads also match observed concentrations in streams in the MEP region. Modeled and measured nitrogen loads were determined for a small sub-watershed to West Falmouth Harbor (Smith and Howes 2006) where a small stream drained the aquifer from a residential neighborhood. In this effort, the measured nitrogen discharge from the aquifer was within 5% of the modeled N load. A second evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The measured and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year and under the ecological situation (Samimy and Howes unpublished data).

While census based population data has limitations in the highly seasonal MEP region, part of the regular MEP analysis is to compare expected water used based on average residential occupancy to measured average water uses. This is performed as a quality assurance check to increase certainty in the final results. This comparison has shown that the larger the watershed the better the match between average water use and occupancy. For example, in the cases of the combined Great Pond, Green Pond and Bournes Pond watershed in the Town of Falmouth and the Popponesset Bay/Eastern Waquoit Bay watershed, which cover large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that from the water use approach. This comparison matches some of the variability seen in census data itself, census blocks, which are generally smaller areas of the towns, have shown up to a 13% difference in average occupancy from town-wide occupancy rates. These analyses provide additional support for the use of the water use approach in the MEP study region.

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy outwash aquifers; (b) has been validated in studies of the MEP Watershed "Module", where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Coefficients for stormwater, lawn fertilization, etc; (c) the MEP septic nitrogen loading coefficient agrees in specific studies of consumptive water use and N attenuation between the septic tank and the discharge site; and (d) the watershed module provides estimates of nitrogen attenuation by freshwater systems that are consistent with a variety of ecological studies. It should be noted that while points b-d support the use of the MEP Septic N Coefficient, they were not used in its development. The MEP Technical Team has worked out the septic system nitrogen load over many years, and the general agreement among the number of supporting studies has greatly enhanced the certainty of this critical watershed nitrogen loading term.

The independent validation of the water quality model (Section VI) and the reasonableness of the freshwater attenuation (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, it is also conservative in watersheds dominated by residential land-uses. Sensitivity analysis by MEP Technical Team showed that higher septic nitrogen loading factors (up to 33% larger), resulted in only slight changes in the required nitrogen removal (estimated at 1% to 5% lower)), to lower embayment nitrogen levels to a nitrogen target (e.g. nitrogen threshold, cf. Section VIII.

The independent validation of the water quality model (Section VI) and the reasonable assumption regarding the lack of the freshwater attenuation in the Rushy Marsh Pond watershed (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher loads that are generally used in regulatory situations.

Water use in the Rushy Marsh watershed is somewhat different than for many of the watersheds in the MEP region because of the comparatively low number of parcels. The average water use among the eight single family residences in the Rushy Marsh watershed is 426 gpd; further evaluation of this number found that the average water use per thousand square feet of building is 110 gpd. When this is multiplied by the average size single family residence determined for Mashpee in a previous MEP study (1,500 sq. ft.), the average flow is 166 gpd; this flow more closely approximates average single family residential flow in most previous MEP watershed analyses and is appropriate for use in determining flow from additional properties determined from the buildout assessment and the one residence without water use information in the Rushy Marsh watershed.

In order to provide an independent validation of the residential water use average within the study areas, MEP staff reviewed US Census population values. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so each person generates 55 gpd of wastewater. Average occupancy within the Town of Barnstable during the 2000 US Census was 2.44 people per household. If the Barnstable average of 2.44 is multiplied by 55 gpd, the average residential wastewater flow in the Rushy Marsh watershed would be 134 gpd. The adjusted flow in the Rushy Marsh watershed is a fairly good match with this simple check on the water use data.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized lawns and golf courses, with lawns being the predominant source within this category. In order to add this source to the nitrogen loading model for the Rushy Marsh system, MEP staff reviewed available information about residential lawn fertilizing practices.

Residential lawn fertilizer use has rarely been directly measured in watershed-based nitrogen loading investigations. Instead, lawn fertilizer nitrogen loads have been estimated based upon a number of assumptions: a) each household applies fertilizer, b) cumulative annual applications are 3 pounds per 1,000 sq. ft., c) each lawn is 5000 sq. ft., and d) only 25% of the nitrogen applied reaches the groundwater (leaching rate). Because many of these assumptions had not been rigorously reviewed in over a decade, the MEP Technical Staff undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the Watershed Nitrogen Loading Sub-Model.

The initial effort was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. The assessment accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20% resulted in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn for use in the nitrogen loading calculations. It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns were found to have the higher rate of fertilization application and hence higher estimated loss to groundwater of 3 lb/lawn/yr. Only residential fertilizer applications are included in the Rushy Marsh nitrogen loading.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture (Howes and Teal, 1995). Only the bog loses measurable nitrogen, the forested upland release only very low amounts. For the land-use N loading analysis, the areas of active bog surface are based on 85% of the total property area with cranberry bog land use codes. Factors used in the nitrogen loading analysis for the Rushy Marsh Pond watershed are summarized in Table IV-1.

Table IV-1. Primary Nitrogen Loading Factors used in the Rushy Marsh MEP analysis. General factors are from MEP modeling evaluation (Howes and Ramsey 2001). Site-specific factors are derived from Town of Barnstable data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

Nitrogen Concentrations:	mg/l
Wastewater (Septic System effluent)	35
Wastewater (pre-discharge from aquifer)	26.25
Road Run-off	1.5
Roof Run-off	0.75
Direct Precipitation on Embayments and Ponds	1.09
Natural Area Recharge	0.072
Wastewater Coefficient	23.63
Fertilizer:	
Average Residential Lawn Size (ft^2)*	5,000
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined by site-specific information	

Recharge Rates:	in/yr
Impervious Surfaces	40
Natural and Lawn Areas	27.25
Water Use/Wastewater:	
Rushy Marsh Residential Parcels Wo/water accounts and Buildout additions	166 gpd
For Commercial Properties wo/water accounts and Buildout additions:	18 gpd/1,000 ft^2 of building
For Parcels w/water accounts:	Measured annual water use
Average percentage of lot occupied by commercial dwelling = 28 percent	

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined individually and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each subwatershed and the sum of the area of the parcels within each subwatershed. The resulting “parcelized” watersheds to Rushy Marsh are shown in Figure IV-3.

The review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (condominiums, WWTFs, etc.) were also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to an overall estuary. However, in the case of Rushy Marsh, the limited number of parcels suggests that individual parcel assignment could have a significant impact on the total nitrogen loading. In order to ensure accurate assignment, MEP staff reviewed aerial photography to assess cleared locations close to houses likely to be the location of septic system leach fields. The assignment effort was undertaken to better define the sub-embayment loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives. As mentioned previously, the Rushy Marsh study area contains only one watershed (Figure IV-3).

For management purposes, the aggregated embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Rushy Marsh System, the major types of nitrogen loads are: wastewater, fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-2). The output of the watershed nitrogen loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-4). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model. However, as noted above, the Rushy Marsh watershed nitrogen input is not subject to attenuation within the watershed, as there are no associated streams or great fresh ponds.

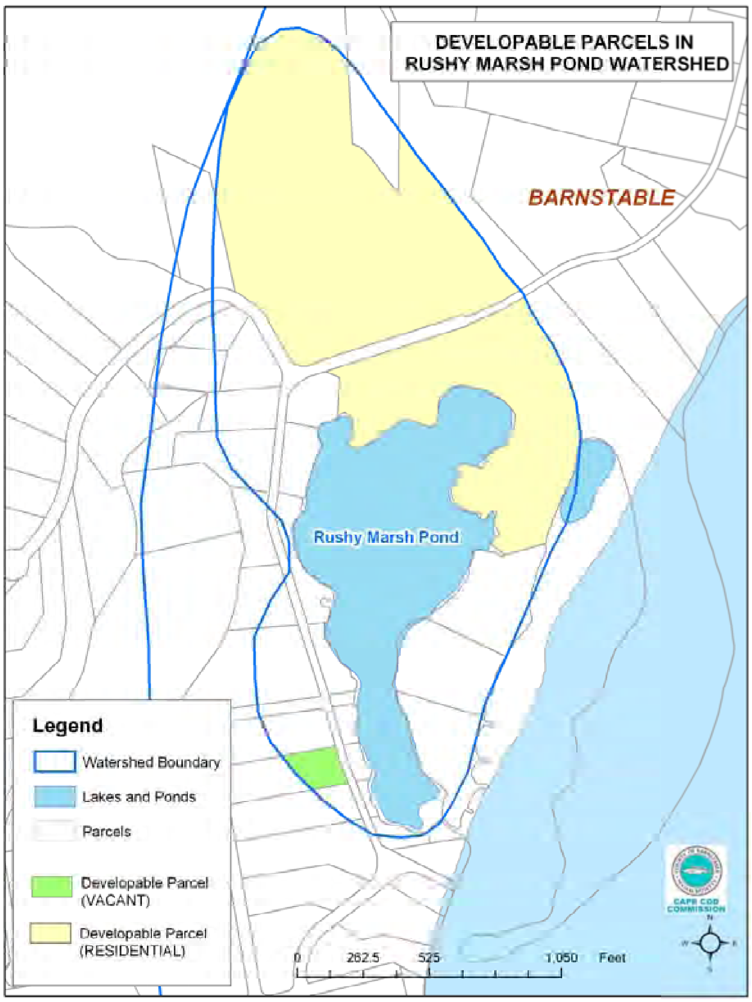


Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Rushy Marsh watershed.

Table IV-2. Nitrogen Loads to the Rushy Marsh Pond Estuary. There is negligible attenuation during transport to Rushy Marsh Pond and groundwater travel times are less than 10 years throughout the watershed. Wastewater represents on-site septic treatment and disposal from residential dwellings.

Values: kg N/yr	***Rushy Marsh Watershed N Loads by Input:***						***Present N Loads Without Atmos N to Estuary***			***Buildout N Loads Without Atmos N to Estuary***			***Atmos N to Estuary***
Name	**Wastewater**	**Lawn Fertilizers**	**Impervious Surfaces**	**Water Body Surface Area**	**"Natural" Surfaces**	**Buildout**	**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**
Rushy Marsh Pond	129	6	11	0	17	168	163	**0%**	163	331	**0%**	331	74

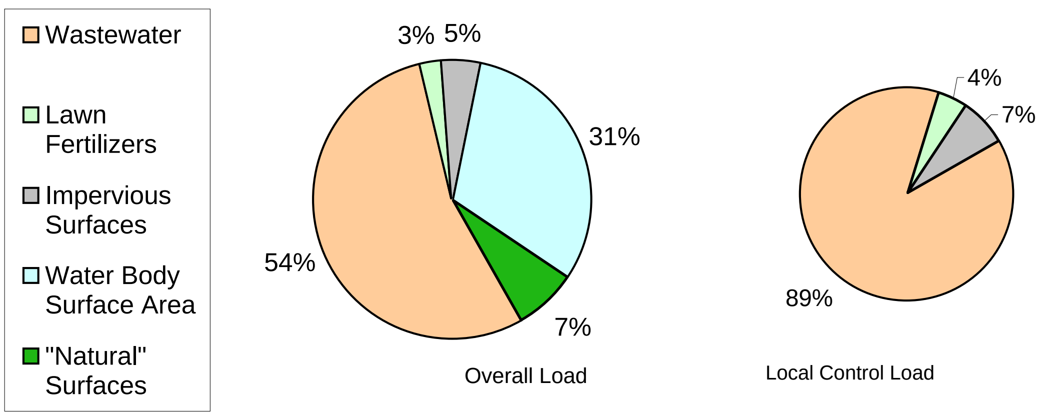


Figure IV-4. Land use-specific unattenuated nitrogen load (by percent) to the Rushy Marsh watershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control (e.g. excludes atmospheric deposition to estuarine surface).

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call “windows on the aquifer.” Groundwater typically flows into the pond along the upgradient shoreline, then lake water flows back into the groundwater system along the downgradient shoreline. Occasionally a Cape Cod pond will have a stream outlet or herring run too. Since the nitrogen loads flow into the pond with the groundwater, the relatively more productive ecosystems in the ponds incorporate some of the nitrogen, retain some of it in the sediments, and change it among its various oxidized and reduced forms. As result of these interactions, some of the nitrogen is removed from the watershed system, mostly through burial in the sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining, reduced loads flow back into the groundwater system along the downgradient side of the pond or through a stream outlet and eventual discharge into the downgradient embayment. The nitrogen load summary in Table IV-2 includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads. However, for Rushy Marsh Pond there are no great fresh ponds within the watershed and all watershed derived nitrogen enters the estuary through direct groundwater seepage, without attenuation.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watershed. For the Rushy Marsh modeling, MEP staff consulted with town planners to determine the parameters that would be used in the assessment.. The standard buildout assessment is to evaluate town zoning to determine minimum lot sizes in each of the zoning districts, including overlay districts (*e.g.*, water resource protection districts). Larger lots are subdivided by the minimum lot size to determine the total number of new lots. Staff also review developed properties with additional development potential; for example, residential lots that are twice the minimum lot size, but only have one residence. Parcels that are classified as developable residential (state class land use codes 130 and 131) but are less than the minimum lot size and are greater than 5,000 square feet are assigned an additional residence in the buildout; 5,000 square feet is a minimum lot size in some Cape Cod town zoning regulations. Commercial properties are not subdivided; the area of each parcel and the factors in Table IV-1 were used to determine a wastewater flow for these properties. Parcels included in the buildout assessment for the Rushy Marsh watershed are shown in Figure IV-3. A nitrogen load for each additional parcel included in the buildout analysis was determined for addition to the load determined for the existing development by using the factors presented in Table IV-1 and discussed above. A summary of total potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-2.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out or sewering analysis) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the watershed of the Rushy

Marsh System were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport is through groundwater in sandy outwash aquifers. The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. This is the case for the Rushy Marsh watershed. Unlike most watersheds in southeastern Massachusetts, nitrogen does not pass through a surface water ecosystem on its path to the adjacent embayment. It is in these surface water systems that the needed conditions for nitrogen retention and denitrification exist. As there were no streams or great fresh ponds within the Rushy Marsh watershed, the watershed loading approach considered that nitrogen reaching the water table was transported without attenuation in the groundwater system until discharge to the estuary.

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the Benthic Nutrient Flux Task was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters within the Rushy Marsh Pond basin. The mass exchange of nitrogen between watercolumn and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Rushy Marsh embayment predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the watercolumn (once it entered), predicting watercolumn nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayment from the surrounding watersheds it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton "particles". Most of these "particles" remain in the watercolumn for sufficient time to be flushed out to a downgradient larger waterbody (like Nantucket Sound). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen "load" become incorporated into the surficial sediments of the bays.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within small enclosed basins (e.g. Rushy Marsh Pond). To some extent, the settling characteristics can be evaluated by observation of the grain-size and organic content of sediments within an estuary.

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter (with its nitrogen content) that bioavailable nitrogen is returned to the embayment watercolumn for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that we have investigated, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for Determining Sediment-Watercolumn Nitrogen Exchange

For the Rushy Marsh Pond system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected in the Rushy Marsh Pond basin from 8 sites (Figure IV-5) in July 2004. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample. As part of a separate research investigation, the rate of oxygen uptake was also determined and measurements were made of sediment bulk density, organic nitrogen, and carbon content. These measurements were made by the Coastal Systems Program at SMAST-UMD.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by a small boat to the shoreside field “lab”. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. The number of core samples (see Figure IV-5) within Rushy Marsh central basin and channel were as follows:

Rushy Marsh System

- Station 1 – 1 core (Rushy Marsh – channel)
- Station 2 – 1 core (Rushy Marsh – channel)
- Station 3 – 1 core (Rushy Marsh – central basin)
- Station 4 – 1 core (Rushy Marsh – central basin)
- Station 5 – 1 core (Rushy Marsh – central basin)
- Station 6/7 – 2 cores (Rushy Marsh – central basin)
- Station 8 – 1 cores (Rushy Marsh – central basin)

Sampling was distributed throughout the embayment system and the results for each site combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-watercolumn exchange follow the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1995) for nutrients and metabolism. Upon return to the field laboratory (private residence on the shore of Rushy Marsh) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining sample frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

Chemical analyses were performed by the Coastal Systems Analytical Facility at the School for Marine Science and Technology (SMAST) at the University of Massachusetts in New Bedford, MA. The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

Figure IV-5. Rushy Marsh System locations (red diamonds) of sediment sample collection for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-3.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

Watercolumn nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (watercolumn and benthic), and uptake (e.g. photosynthesis). As stated above, during the warmer summer months the sediments of shallow embayments typically act as a net source of nitrogen to the overlying waters and help to stimulate eutrophication in organic rich systems. However, some sediments may be net sinks for nitrogen and some may be in "balance" (organic N particle settling = nitrogen release). Sediments may also take up dissolved nitrate directly from the watercolumn and convert it to dinitrogen gas (termed "denitrification"), hence effectively removing it from the ecosystem. This process is typically a small component of sediment denitrification in embayment sediments, since the watercolumn nitrogen pool is typically dominated by organic forms of nitrogen, with very low nitrate concentrations. However, this process can be very effective in removing nitrogen loads in some systems, particularly in salt marshes, where overlying waters support high nitrate levels.

In addition to nitrogen cycling, there are ecological consequences to habitat quality of organic matter settling and mineralization within sediments, which relate primarily to sediment and watercolumn oxygen status. However, for the modeling of nitrogen within an embayment it is the relative balance of nitrogen input from watercolumn to sediment versus regeneration which is critical. Similarly, it is the net balance of nitrogen fluxes between water column and sediments during the modeling period that must be quantified. For example, a net input to the sediments represents an effective lowering of the nitrogen loading to down-gradient systems and net output from the sediments represents an additional load.

The relative balance of nitrogen fluxes ("in" versus "out" of sediments) is dominated by the rate of particulate settling (in), the rate of denitrification of nitrate from overlying water (in), and regeneration (out). The rate of denitrification is controlled by the organic levels within the sediment (oxic/anoxic) and the concentration of nitrate in the overlying water. Organic rich sediment systems with high overlying nitrate frequently show large net nitrogen uptake throughout the summer months, even though organic nitrogen is being mineralized and released to the overlying water as well. The rate of nitrate uptake simply dominates the overall sediment nitrogen cycle.

In order to model the nitrogen distribution within an embayment it is important to be able to account for the net nitrogen flux from the sediments within each part of each system. This requires that an estimate of the particulate input and nitrate uptake be obtained for comparison to the rate of nitrogen release. Only sediments with a net release of nitrogen contribute a true additional nitrogen load to the overlying waters, while those with a net input to the sediments serve as an "in embayment" attenuation mechanism for nitrogen.

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-6).

Unfortunately, the tendency for net release of nitrogen during warmer periods coincides with the periods of lowest nutrient related water quality within temperate embayments. This sediment nitrogen release is in part responsible for poor summer nutrient related health. Other major factors causing the seasonal water quality decline are the lower solubility of oxygen during summer, the higher oxygen demand by marine communities, and environmental conditions supportive of high phytoplankton growth rates.

In order to determine the net nitrogen flux between watercolumn and sediments, all of the above factors were taken into account. The net input or release of nitrogen within a specific embayment was determined based upon the measured ammonium release, measured nitrate uptake or release, and estimate of particulate nitrogen input. Dissolved organic nitrogen fluxes were not used in this analysis, since they were highly variable and generally showed a net balance within the bounds of the method.

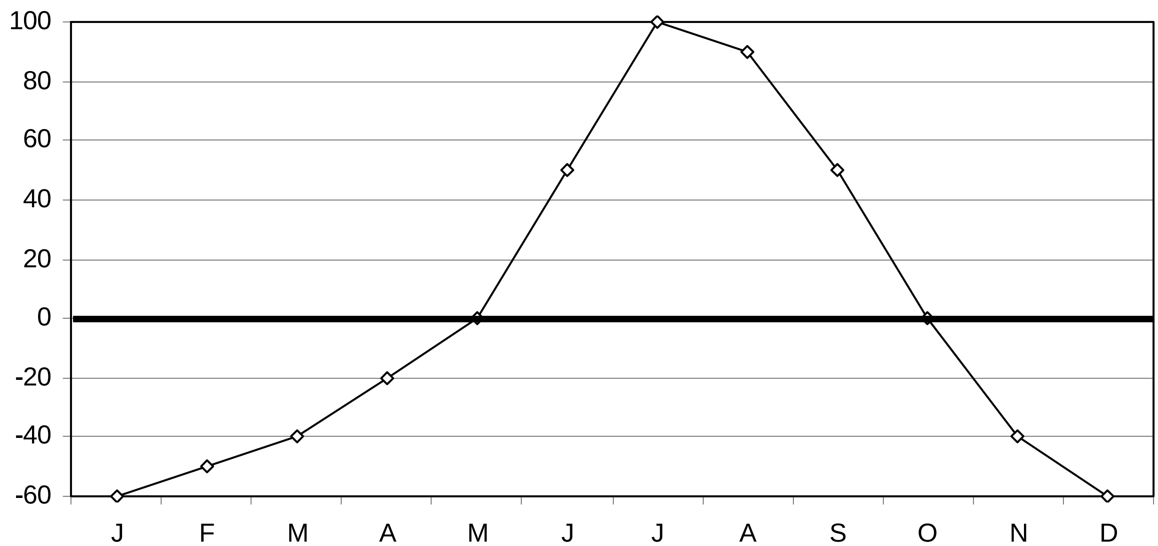


Figure IV-6. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment sampling was conducted throughout Rushy Marsh Pond in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-5). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and bulk density and an analysis of each site’s tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site and the average summer particulate carbon and nitrogen concentration within the overlying water. Two levels of settling were used. If the sediments were organic rich and a fine grained and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated a coarse grained sediments and low organic content and high velocities, then half this settling rate was used. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. This approach was validated in outer Cape Cod embayments (Town of Chatham) by examining the relative fraction of the sediment carbon turnover (total sediment metabolism) which would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins is driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments.

Net nitrogen release or uptake from the sediments within Rushy Marsh Pond for use in the water quality modeling effort (Chapter VI) are presented in Table IV-3. The Rushy Marsh Pond sediments are a slight sink for nitrogen in the central basin and a moderate strength source of recycled nitrogen within the shallow channel. The system is highly eutrophic with enriched total nitrogen concentrations, phytoplankton blooms and periodic oxygen depletion of bottom waters. As a result the sediments are generally very soft to fluid organic muds. Within the channel and deeper basin these organic muds generally had a clear oxidized surface layer, although, within the deeper basin, the surface oxidized layer was not uniform. There was also evidence of a sparse amphipod mat and previous colonization by a submerged aquatic macrophytes, presumably *Ruppia,* although no live plants were observed (see Chapter 7). Nitrogen enrichment of this system stems primarily from the lack of tidal flushing out of Pond waters to Nantucket Sound, rather than a heavy watershed nitrogen input.

Typically, embayments show relative small positive or negative net nitrogen fluxes (e.g. throughout adjacent Popponesset Bay). The pattern in Rushy Marsh Pond appears to result from the relatively high mass of particulate nitrogen settling within this system, due to the high phytoplankton production in the nitrogen rich embayment waters. A similar pattern was found in Oyster Pond (Town of Falmouth), also a highly tidally restricted embayment. However, Rushy Marsh Pond is only periodically oxygen depleted in its bottom waters, whereas the deep basins of Oyster Pond tend to be seasonally anoxic. However, in sediments above the deep basins (<3m depth) in Oyster Pond, the average net nitrogen flux was similar to that for Rushy Marsh. This consistency in sediment nitrogen cycling between embayments of similar hydrodynamics and configuration, serves as additional quality assurance for this parameter.

Table IV-3. Rates of net nitrogen return from sediments to the overlying waters of the Rushy Marsh Embayment System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent Summer rates. Note that the net nitrogen uptake by the central basin (drown kettle)

			Sediment Nitrogen Release		
Basin Region	Depth (m)	Stations	Mean mg N m^{-2} d^{-1}	S.E. mg N m^{-2} d^{-1}	N
Central Basin	1.2-1.7	3-8	-19.1	2.6	6
Channel	1.0	1,2	48.7	1.8	2

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of a hydrodynamic model for the Rushy Marsh Pond estuary system (Figure V-1). For this system, the final calibrated model offers an understanding of water movement through the estuary, and provides the first step towards evaluating water quality, as well as a tool for later determining nitrogen loading "thresholds". Nutrient loading data combined with measured environmental parameters within the system become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing estuarine water quality parameters, as well as determining the likely positive impacts of various alternatives for improving overall estuarine health, facilitating the understanding how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

In general, water quality studies of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives. Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. For example, the spread of pollutants may be analyzed from tidal current information developed by the numerical models.

Coastal embayments like Rushy Marsh Pond are the initial recipients of freshwater flows (i.e., groundwater and surfacewater) and the nutrients they carry. An embayment's shape influences the time that nutrients are retained in them before being flushed out to adjacent open waters, and their shallow depths both decrease their ability to dilute nutrient (and pollutant) inputs and increase the secondary impacts of nutrients recycled from the sediments. Degradation of coastal waters and development are tied together through inputs of pollutants in runoff and groundwater flows, and to some extent through direct disturbance, i.e. boating, oil and chemical spills, and direct discharges from land and boats. Excess nutrients, especially nitrogen, promote phytoplankton blooms and the growth of epiphytes on eelgrass and attached algae, with adverse consequences including low oxygen, shading of submerged aquatic vegetation, and aesthetic problems.

Rushy Marsh Pond is set in the southern shoreline of Barnstable. The layout of the Rushy Marsh Pond system is shown in Figure V-1. The central basin of the pond has a surface area of approximately 18 acres including areas of marsh fringing the main body of the pond. The pond is open to Nantucket Sound via a 12-inch culvert that is often blocked by littoral sediments. In the 1800s, a natural inlet connected Rushy Marsh to Nantucket Sound north of the existing culvert. Between the early 1950s and 1970s, a wooden flume near the southern end of the pond provided an ephemeral connection to Nantucket Sound.

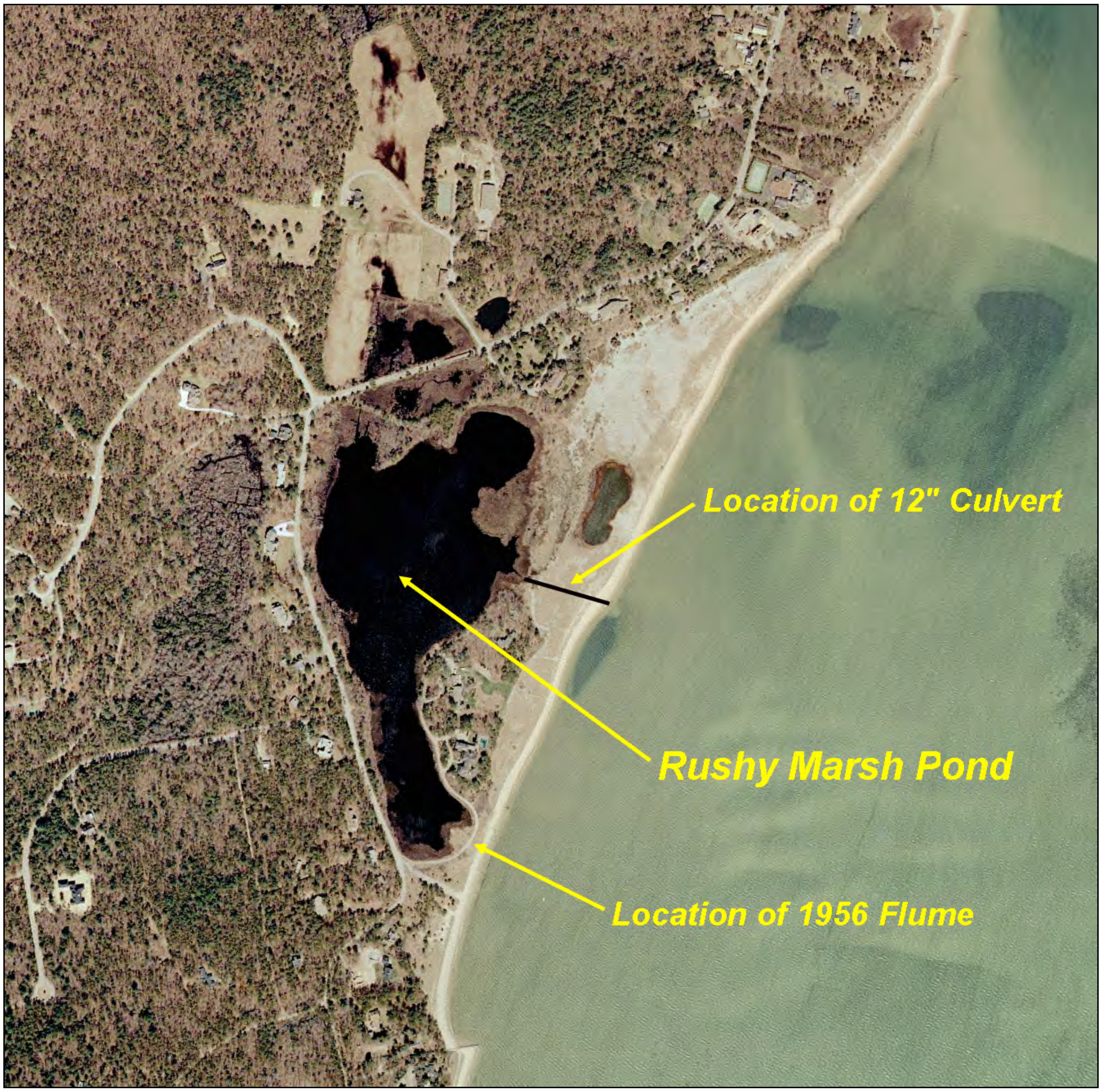


Figure V-1. Aerial photograph of Rushy Marsh in 2001, showing the location of the existing 12-inch culvert, as well as the 1956 flume location. Culvert is in location of natural inlet that existed prior to 1896.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE ESTUARINE SYSTEM

A general understanding of the hydrodynamic controls and coastal processes influencing estuarine dynamics provides the initial framework for the hydrodynamic analysis. In addition, both natural and anthropogenic changes to the estuarine system can guide the evaluation of effective alternatives to enhance tidal circulation and improve water quality.

The southern coast of Cape Cod in the vicinity of Rushy Marsh can be considered a moderately dynamic region, where natural wave and tidal forces continue to reshape the shoreline. Due to the protection afforded by the islands of Martha's Vineyard and Nantucket, the south shore of Cape Cod is protected from the influence of long period open ocean wave conditions. Similar to many portions of the Massachusetts coast, the available sediment supply influences the migration and/or stability of tidal inlets. Tidal inlets can become overwhelmed by the gradual wave-driven migration of a barrier beach separating the estuaries from the ocean. To balance the effect of gradual longshore sediment transport, episodic breaching of a barrier beach system creates new inlets that alter the pathways of water entering the estuary. Storm-driven inlet formation often leads to hydraulically efficient estuarine systems, where seawater exchanges more rapidly with water inside the estuary. As beaches elongate, the inlet channels to the estuaries often become long, sinuous, and hydraulically inefficient. Periodically, overwash from storm events will erode the barrier beach enough at a point to allow again the formation of a new inlet. It is then possible that the new inlet will stabilize and become the main inlet for the system, while the old inlet eventually fills in. Several examples of this process along the Massachusetts coast include Allen's Pond (Westport), New Inlet/Chatham Harbor (Chatham), and Nauset Inlet (Orleans). In addition to these natural coastal processes, man-made structures often can influence the stability of a shoreline/tidal inlet system.

V.2.1 Natural Coastal Processes

For the Rushy Marsh estuarine system, the process of barrier spit elongation and breaching has had a significant influence on tidal exchange. Over the past 100 years, the Popponesset Beach barrier elongation and breaching processes have governed the stability of the Rushy Marsh inlet. The USGS map from 1893 (Figure V-2) illustrates shows the Popponesset Beach spit in a condition where the barrier is elongated slightly beyond present day conditions and the flood shoal is emergent (Thatch Island). Between the late 1800s and 1950, the Popponesset barrier elongated past the Rushy Marsh tidal inlet. Due to the influx of sediment associated with this barrier elongation, the natural tidal inlet to Rushy Marsh closed. The 1943 USGS map (Figure V-3) of the area shows the condition of the system prior to the Popponesset Beach breach. As described in Aubrey and Goud (1983), the loss of nearly one-half of the Popponesset Beach barrier between 1954 and the early 1980s led to concerns regarding future barrier spit migration. Figures V-4 and V-5 illustrate changes to the barrier spit over the 30 year period between 1951 and 1981. According to Aubrey and Gaines (1982), the present spit length has been historically the stable configuration. It wasn't until after about 1860 that the spit began to grow past its present location.

As the barrier spit elongated between the early 1900s and the mid-1950s due to regional littoral drift, the inlet channel to Popponesset Bay become less efficient, where the tide height within Popponesset Bay decreased and the lag time between high tide in the estuary and Nantucket Sound increased. This increase in tidal attenuation was remedied in 1954, when a hurricane breached the barrier spit, creating an efficient inlet in the vicinity of the present inlet. Once the spit had breached, the remnants of the spit east of the inlet gradually overwashed and rejoined the shoreline (primarily in the vicinity of Rushy Marsh). This inlet spit growth and breaching process has been documented extensively for the southeastern coast of Massachusetts (e.g. Fitzgerald, 1993).

The specific influence of barrier elongation and breaching upon the Rushy Marsh system was described in a report to the Town of Barnstable prepared by the Coast & Harbor Institute and Robert L. Fultz Associates (2002). A schematic representation of barrier beach processes and their influence on Rushy Marsh is shown in Figure V-6. Although the inlet to Rushy Marsh had naturally closed in the early 1900s, the influx of littoral sediment caused by the 1954 breach of the Popponesset barrier further widened the barrier beach system fronting Rushy Marsh. Efforts to maintain an effective inlet near the southern end of the Pond have been complicated by the unstable nature of the shoreline, the relatively weak littoral drift that continues to supply sediment to this region, and the small potential tidal prism exiting Rushy Marsh.

Figure V-7 shows the historic shoreline change fronting Rushy Marsh and the Three Bays region between 1938 and 2005. Much of the accretion along the barrier beach separating Rushy Marsh from Nantucket Sound is a result of the Popponesset spit remnants joining the barrier beach fronting Rushy Marsh. After this spit welded onto the existing shoreline, the net west-to-east directed littoral drift has “straightened” the shoreline in this region, where slight erosion has been observed along the beach fronting the southwest end of the pond and significant accretion has been observed along the beach fronting the remainder of the pond. In general, the shoreline region between Rushy Marsh Pond and the Popponesset Bay entrance has been relatively stable since the 1950s.

Recent monitoring of beach profiles at Oregon Beach fronting Rushy Marsh Pond has been performed by volunteers overseen by Jim O’Connell of the Woods Hole Oceanographic Institution’s Sea Grant Program. Locations of transects at Oregon Beach are shown in Figure V-8. Initial monitoring results indicate a relatively stable shoreline and dune system. Elevation of the natural dunes varies from about 6.5 to 8 feet NGVD. The beach slope varies from about 1:10 to 1:20 (v:h) and levels off at about -1 ft NGVD to a tidal flat. Figures V-9 and V-10 show the inter-annual variation of transects OB#1 and OB#3, respectively.

V.2.2 Anthropogenic Changes Influencing Rushy Marsh Pond

Manmade coastal structures along the shoreline immediately west of Rushy Marsh Pond consist primarily of groins along this updrift shoreline. Based on site observations, most of these structures are not effective barriers to natural littoral drift; therefore, beach compatible material continues to supply the barrier fronting Rushy Marsh Pond. The volume of material transported along this shoreline stretch is relatively small, due primarily to the quiescent wave conditions within the protected waters of Nantucket Sound. The conclusion that the longshore sediment transport rate is relatively low is further supported by the stable shoreline northeast of Rushy Marsh Pond and the small maintenance dredging volumes required to maintain the entrance to Cotuit Bay (which receives littoral sediments from both the east and the west).

Although man has modified much of the coastline between the Popponesset Bay and Cotuit Bay entrances, most of the large-scale changes to the estuarine systems have been caused by nature. For example, the 1954 breach of Popponesset Beach created a much more efficient inlet channel to this system; however, the influx of littoral sediments associated with this breach caused a substantial increase in the barrier beach width fronting Rushy Marsh Pond. Relatively large-scale natural changes to the shoreline over the approximate 20-year period following the breach of the Popponesset spit hampered efforts to establish a stable inlet to Rushy Marsh Pond. The only existing connection between Rush Marsh Pond and Nantucket Sound is a 12 inch culvert that often is clogged at the seaward end.

The 1943 USGS map (Figure V-3) shows that no inlet existed to Rushy Marsh Pond prior to the 1954 breach of the Popponesset spit; therefore, release of freshwater from the pond has been performed mechanically for the past 60+ years. Following the breach, the influx of littoral sediments to the barrier beach system made it more difficult to establish and/or maintain manmade structures to enhance water exchange between Rushy Marsh Pond and Nantucket Sound.

Following the 1954 breach of Popponesset spit, in 1956 a wooden flume was constructed across Oregon Beach to link Rushy Marsh Pond to Nantucket Sound to provide emergency drainage from the pond (Coast & Harbor Institute and Robert L. Fultz Associates, 2002). This flume was approximately 8 feet wide, 2 feet deep, and 72 feet in length. Based on the 2002 analysis by Coast & Harbor Institute and Robert L. Fultz Associates, the base of the structure was at approximately the existing elevation of the pond, where only the highest tides could enter the pond. Before 1974, the flume became ineffective and was replaced by 24-inch culver near the location of the existing 12-inch culvert. Shortly after abandonment, the flume became buried by the beach and dune system. Hurricane Bob in 1991 uncovered the flume for a short period of time (Figure V-11).

Over the past 30 years, a variety of culverts have been utilized to provide freshwater drainage for Rushy Marsh Pond. Due to the relatively small watershed associated with the pond, freshwater flow through the culvert is not sufficient to keep the seaward end of the culvert open.

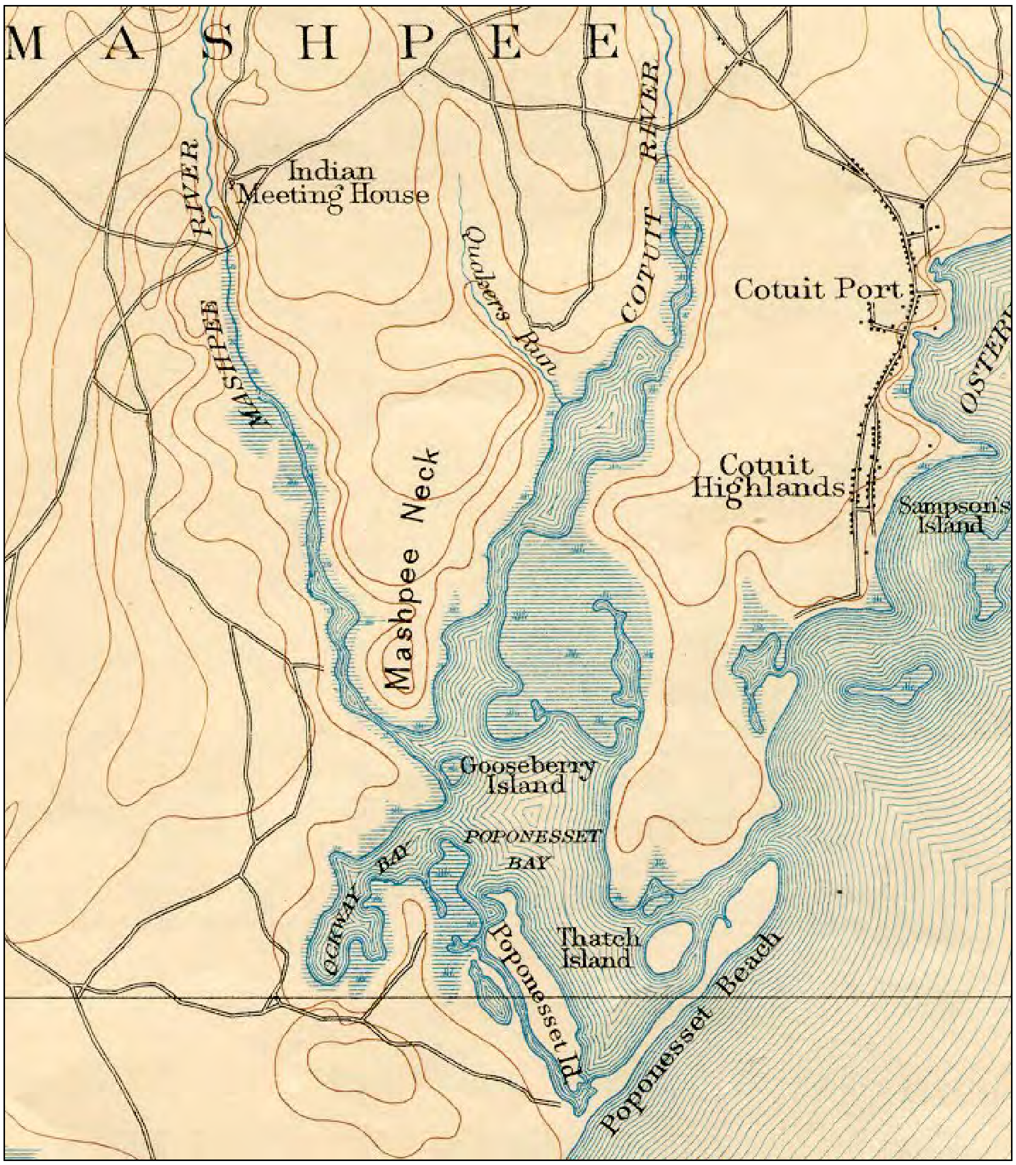


Figure V-2. Portion of the 1893 USGS topographic map (Cotuit Quadrangle) showing the position of the inlet at a similar location as the present inlet.

Figure V-3. Portion of the 1943 USGS topographic map (Cotuit Quadrangle) showing the elongation of Popponesset Beach and closed inlet conditions at Rushy Marsh.

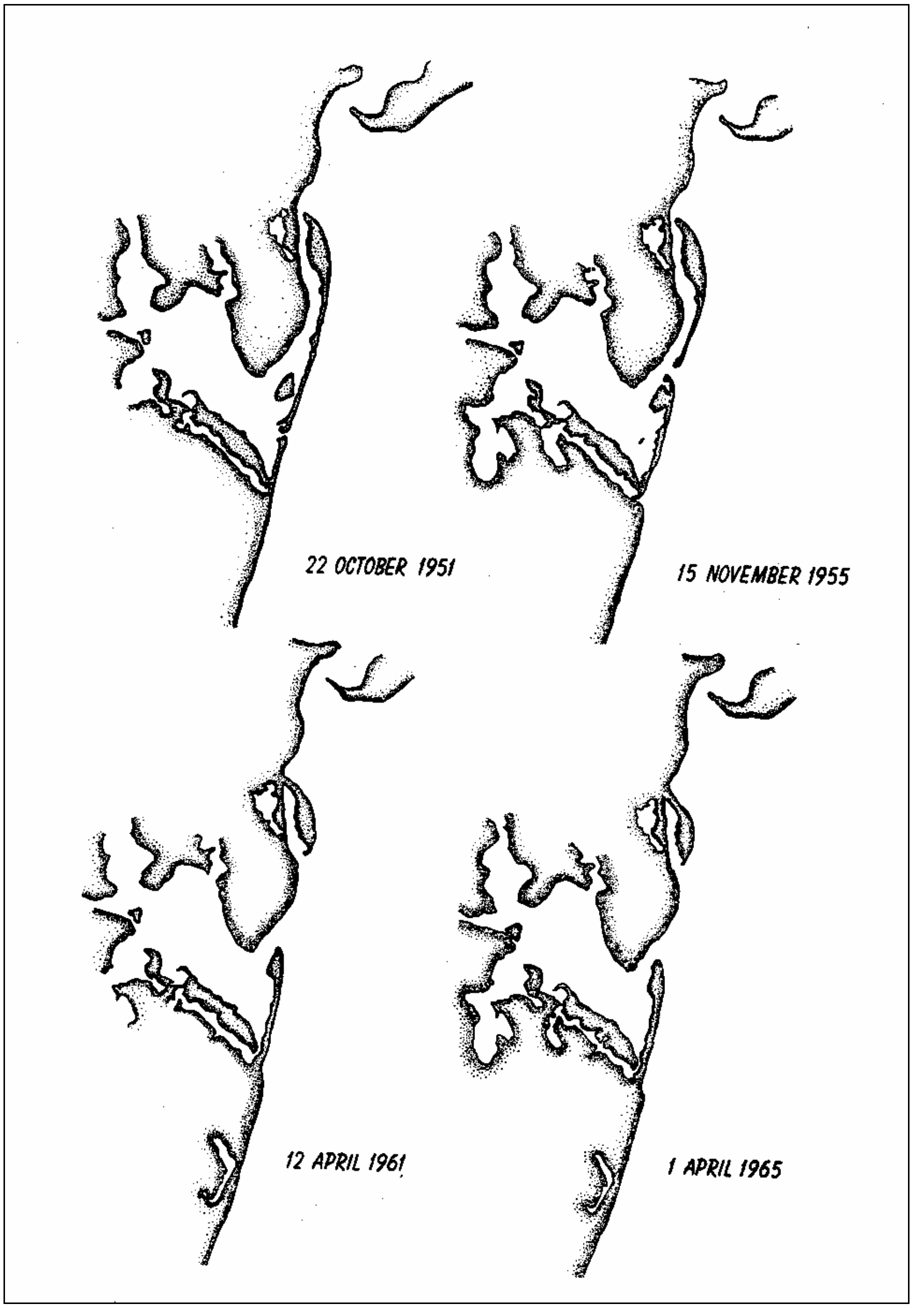


Figure V-4. Outlines of vertical aerial photographs illustrating stages of shoreline evolution in the Popponesset Spit region between 1951 and 1965 (from Aubrey and Goud, 1983).

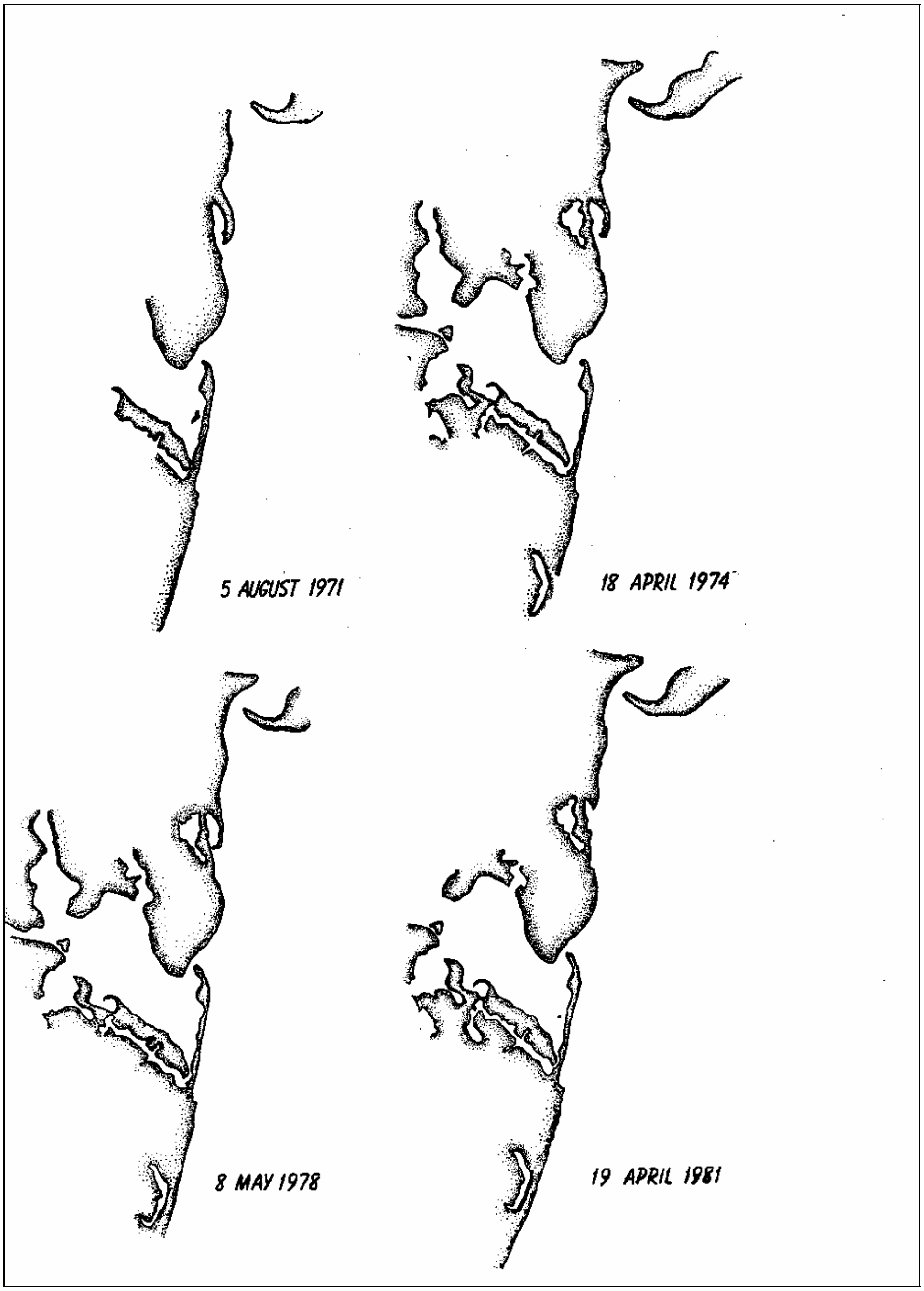


Figure V-5. Outlines of vertical aerial photographs illustrating stages of shoreline evolution in the Popponesset Spit region between 1971 and 1981 (from Aubrey and Goud, 1983).

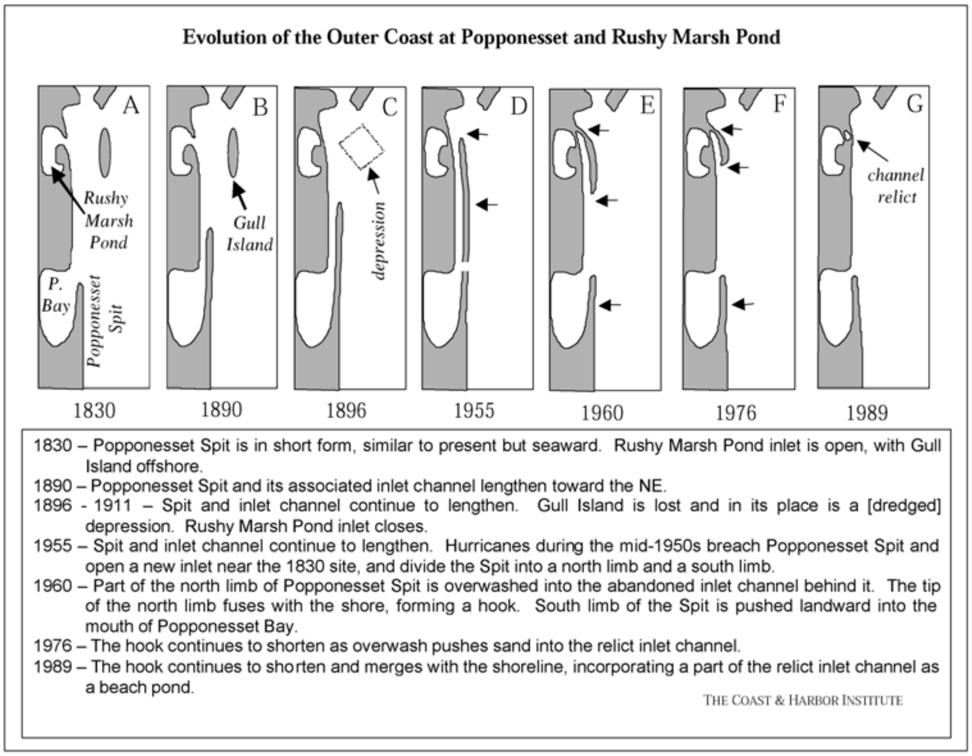


Figure V-6. A schematic description of barrier spit evolution in the vicinity of Popponesset Bay and Rushy Marsh Pond (Coast & Harbor Institute and Robert L. Fultz Associates, 2002).

Figure V-7. Observed shoreline change from 1938 to 2001/2005 for the shoreline area in the vicinity of Rushy Marsh Pond and Three Bays in Barnstable.

Figure V-8. Locations of beach transect monitoring stations along Oregon Beach.

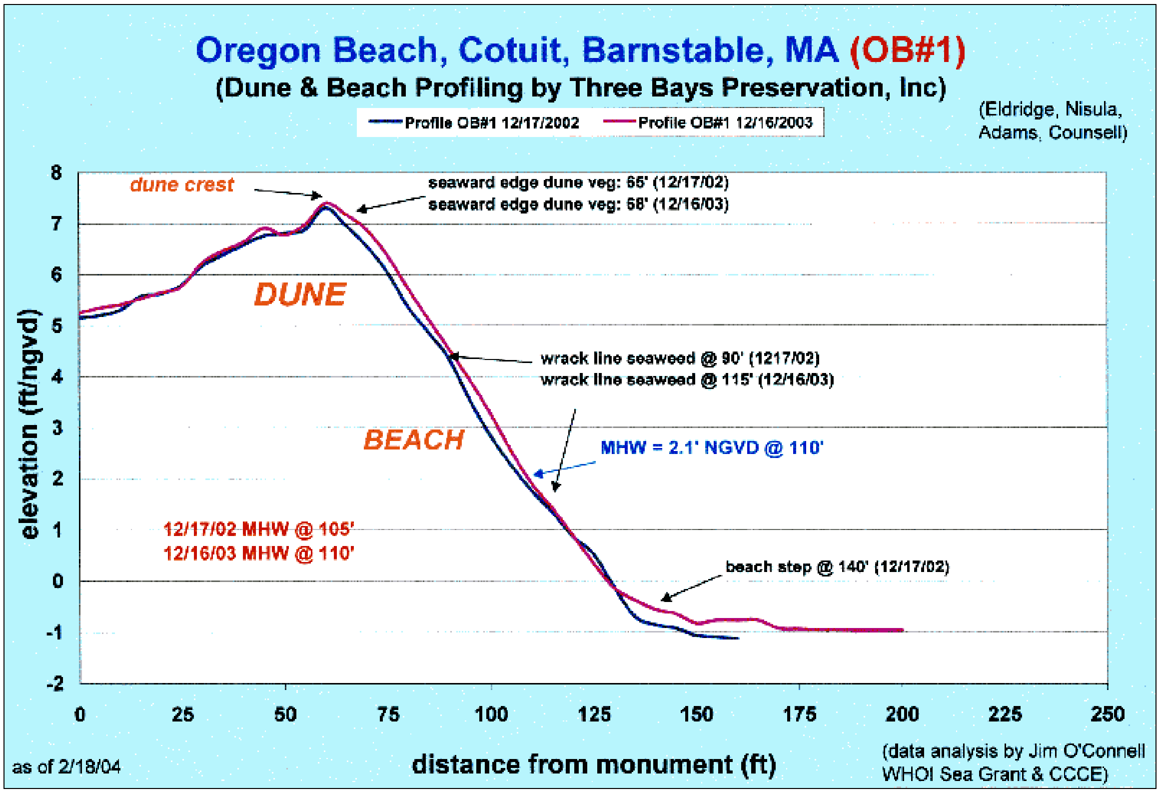


Figure V-9. Inter-annual changes to beach profile at Station OB#1 shown in Figure V-7.

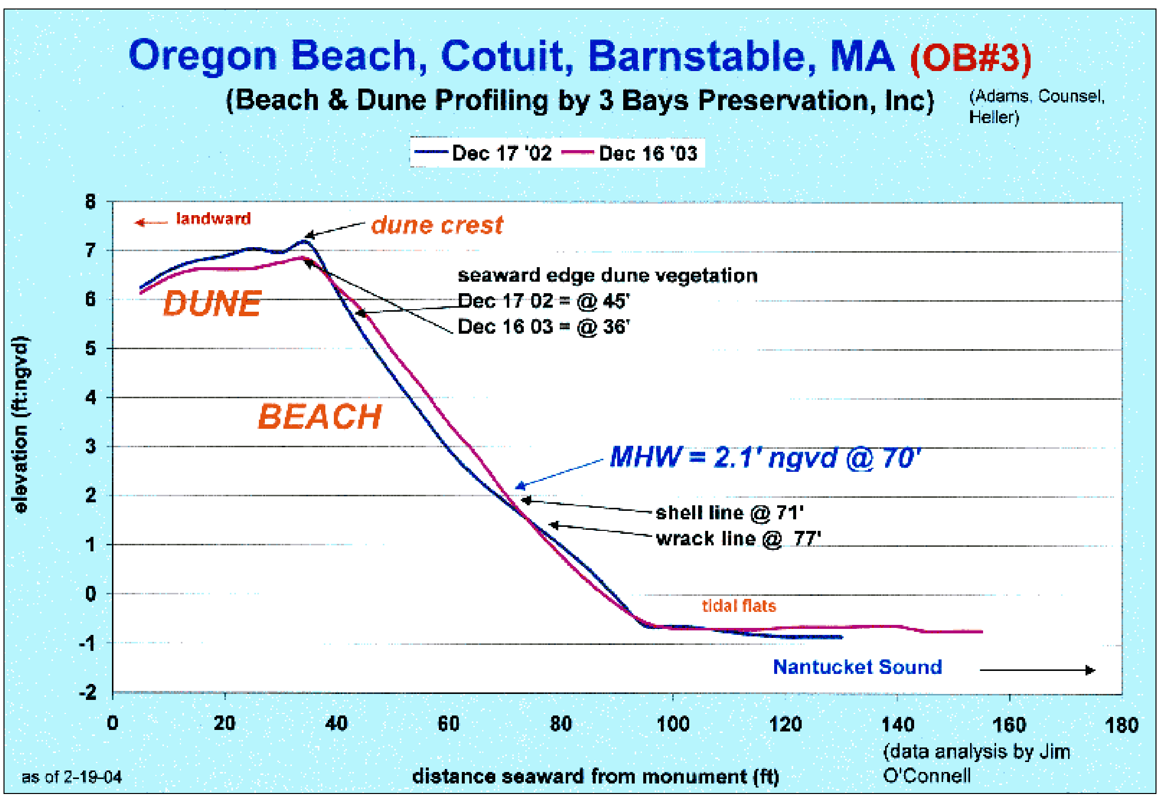
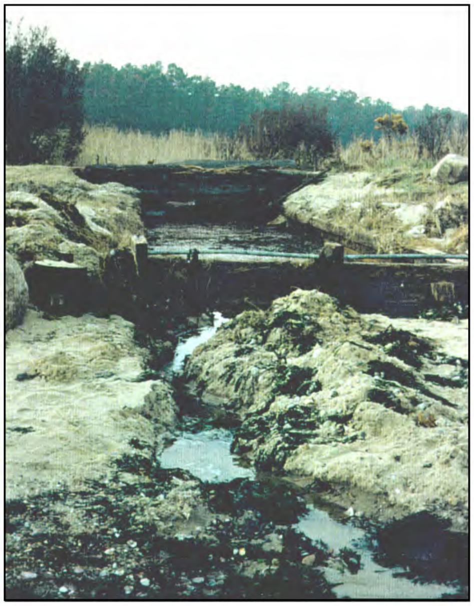


Figure V-10. Inter-annual changes to beach profile at Station OB#3 shown in Figure V-7.



The old wooden sluiceway (ca. 1956) exhumed in 1991 by hurricane Bob. View from Nantucket Sound toward Rushy Marsh Pond. The bulkhead in the foreground is a horizontally hinged gate. The bridge over the sluiceway is seen in the background (photo by E. Bragdon).

The old wooden sluiceway exhumed in 1991 by hurricane Bob. View from the bridge crossing the structure toward Nantucket Sound. The bulkhead in the background is a hinged gate (photo by R. Gatewood).

Figure V-11. The 1956 wood flume built to hydraulically connect Rushy Marsh Pond to Nantucket Sound (Coast & Harbor Institute and Robert L. Fultz Associates, 2002).

V.3 FIELD DATA COLLECTION AND ANALYSIS

A precise description of embayment geometries and hydrodynamic forcing processes is required for the development of numerical models. To support the MEP hydrodynamic and water quality modeling effort in Rushy Marsh, the embayment bathymetry and water elevation variations were measured.

Bathymetry data was collected throughout the main basin of Rushy Marsh estuary. Survey data was not collected above of the roadway embankment on the north side of Rushy Marsh, since the area was not be directly incorporated in the hydrodynamic model. Tidal elevation measurements were used for both forcing conditions and to evaluate tidal attenuation through the culvert into the system. Figure V-12 shows the location of the tide gauges.

Figure V-12. Rushy Marsh with tide gauge locations labeled as W1 (forcing tide) and W2 (Rushy Marsh tide)

V.3.1 Bathymetry

Bathymetry, or depth, of Rushy Marsh was measured during field a survey in May 2004. The survey was completed using a small vessel equipped with a precision fathometer interfaced to a differential GPS receiver. The fathometer has a depth resolution of approximately 0.1 foot and the differential GPS provides x-y position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder and GPS were logged to a data recorder.

GPS positions and echo sounder measurements were merged to produce data sets consisting of water depth as a function of x-y horizontal position (in Massachusetts Mainland State Plane, 1983). These data were combined with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NGVD 1929 vertical datum (NGVD29). The resulting xyz files were input to mapping software to calculate depth contours for the system shown in Figure V-13. The bathymetry was supplemented by existing data from NOAA collected in 1942 to define the offshore region.

V.3.2 Water Elevation Measurements and Analysis

Changes in water surface elevation were measured using internal recording tide gauges. These tide gauges were installed on fixed platforms (screw anchors) to record changes in water pressure over time. Variations in the water surface can be due to tides, wind set-up, or other low frequency oscillations of the sea surface. The tide gauges were installed in 2 locations for Rushy Marsh (Figure V-12) on April 10, 2004 and recovered on May 21, 2004. Data records span at least 29 days to yield an adequate time period for resolving the primary tidal constituents.

The tide gauges used for the study were Brancker XR-420 TG. Data recording was set for 10-minute intervals, with each observation resulting from an average of 60 1-second pressure measurements on 10-minute intervals. These instruments use strain gauge transducers to sense variations in pressure, with resolution on the order of 1 cm (0.39 inches) head of water. Each gauge was calibrated prior to installation to assure accuracy.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric readings were obtained from the NOAA buoy in Buzzards Bay (site BUZM3), interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in water pressure above the instrument. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (for example, feet of water above the tide gauge). Sensors were surveyed into local benchmarks to provide vertical rectification of the water level; these survey values were used to adjust the water surface to a known vertical datum. The result from each gauge is a time series representing the variations in water surface elevation relative to NGVD29. Figure V-14 present the water levels at each gauge location.

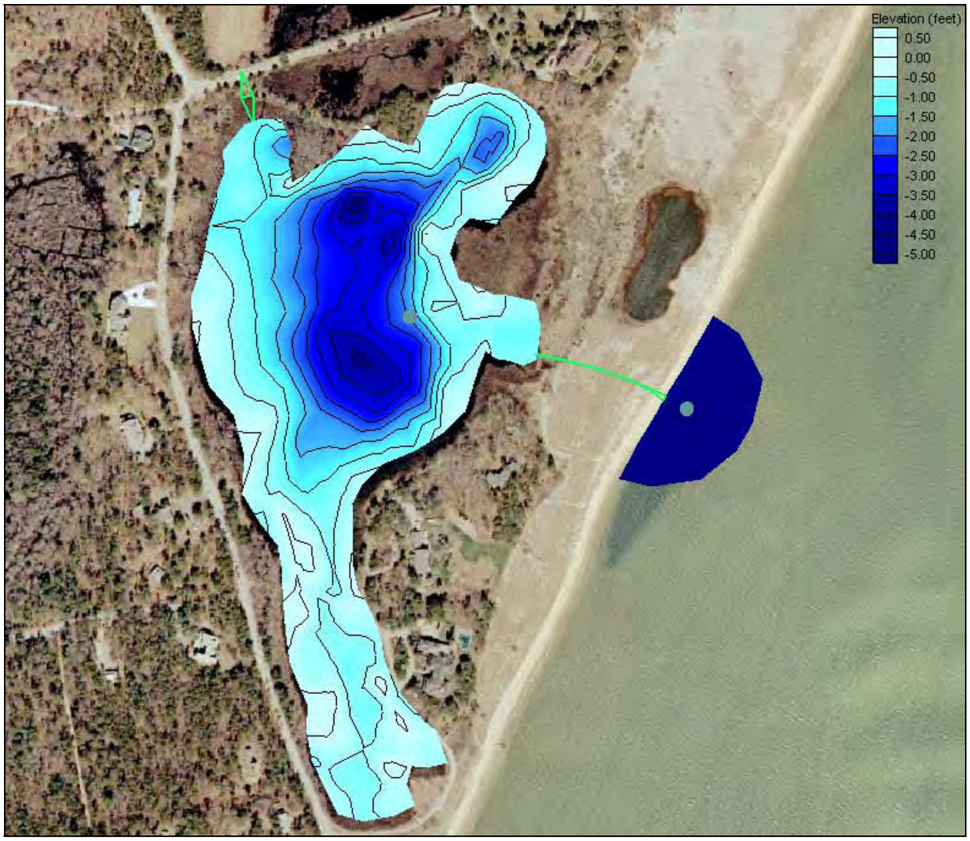


Figure V-13. Bathymetry showing depth contours of the numerical grid for the Rushy Marsh embayment at 0.5-foot contour intervals relative to NGVD29.

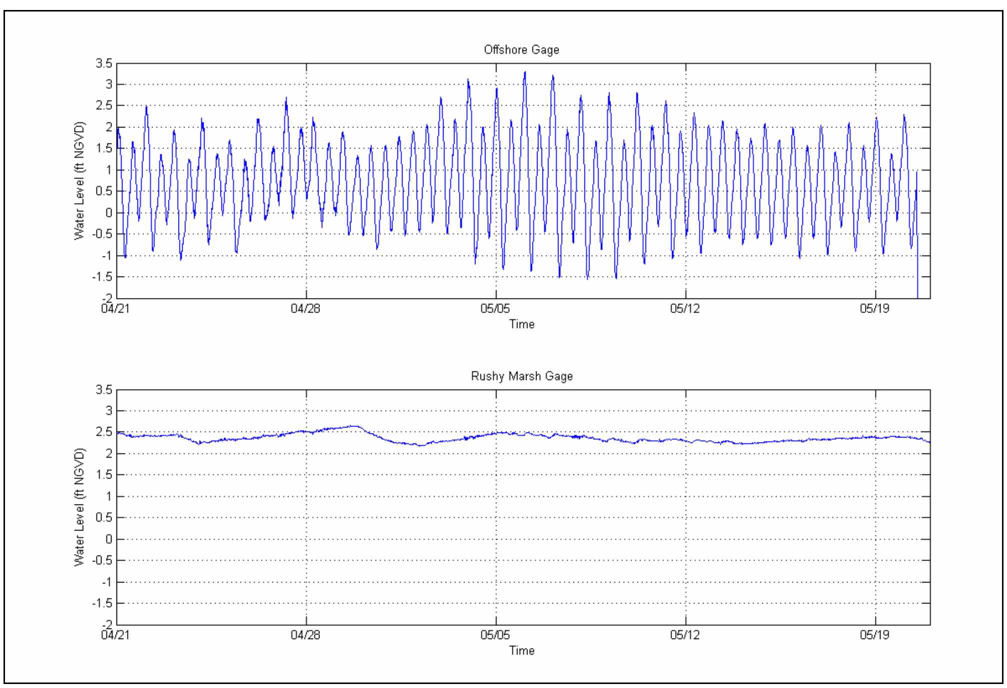


Figure V-14. Tidal elevation observations offshore Rushy Marsh relative to the Rushy Marsh embayment. (Upper) location W1 and (Lower) location W2 within Rushy Marsh. Locations are shown in Figure V-12.

Figure V-14 shows the tidal elevation for the period April 21 through May 20, 2004 at offshore gauge and in Rushy Marsh. The offshore curve has a predominant 12.42-hour variation around the lunar semi-diurnal (twice-a-day), or M_2, tidal constituent. Modulation of the lunar and solar tides, results in the spring-neap fortnightly cycle, typically evidence by a gradual increase and decrease in tide range. Offshore of Rushy Marsh the neap (or minimum) tide range was approximately 2.0 feet, occurring April 30. The spring (maximum) tide range was approximately 4.7 feet, and occurred on May 8. Inside Rushy Marsh the tidal signal very minimal due to the frictional damping caused by the culvert and sediment obstructing the inlet to the culvert. Figure V-14 illustrates that the fluctuations in water surface elevation resulting from non-tidal processes are greater than the tidal component. The tidal signal is less than 0.1 foot within Rushy Marsh.

Analyses of the tide data provided insight into the hydrodynamic characteristics of the system. Harmonic analysis of the tidal time series produced tidal amplitude and phase of the major tidal constituents, and provided assessments of hydrodynamic 'efficiency' of the system in terms of tidal attenuation. This analysis also yielded an assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of the offshore waters and the Rushy Marsh embayment.

Harmonic analyses were performed on the time series for each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-1 presents the amplitudes of the eight largest tidal constituents. The M_2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 1.18 feet at the offshore gauge. The range of the M_2 tide is twice the amplitude, or 2.36 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.30 feet. The N_2 (12.66-hour period) semi-diurnal tide, also contributes significantly to the total tide signal with an amplitude of 0.32 feet. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 is approximately 14% of the amplitude of the M_2 in the offshore gauge (about 0.16 feet). The M_6 amplitude is relatively small throughout the system (less than 0.06 feet). The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon. The observed astronomical tide is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency.

Table V-1. Tidal Constituents, Rushy Marsh, April-May 2004								
	AMPLITUDE (feet)							
Period (hours)	M2	M4	M6	S2	N2	K1	O1	Msf
	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Offshore	1.18	0.16	0.06	0.14	0.32	0.35	0.30	0.01
Rushy Marsh	0.01	0.00	0.00	0.00	0.00	0.01	0.01	0.02

Table V-1 also shows that the constituents within the marsh are very minimal due to the culvert restrictions. Generally, the constituents would vary as the tide propagates through the inlet into the marsh, however the frictional damping and restrictions are large enough to remove a majority of the tidal energy before it propagates into the system.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Nantucket Sound is a relatively shallow semi-enclosed basin, therefore the water surface responds readily to wind-forcing. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large. This analysis calculated the energy (or variance) of the original water elevation time series, and compared these energy values to that of the purely tidal signal (re-created by summing the contributions from the 23 known harmonic constituents). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. The results of this analysis for are presented in Table V-2.

Table V-2. Percentages of Tidal versus Non-Tidal Energy, Rushy Marsh, 2004.

	Total Variance ($ft^2 \cdot sec$)	Total (%)	Tidal (%)	Non-tidal (%)
Offshore	0.922	100	94.3	5.7
Rushy Marsh	0.010	100	4.1	95.9

The variability analysis shows that a majority of the change in water surface elevation in Nantucket Sound was due to tidal processes. However, in Rushy Marsh more than 90-percent of the energy was the result of non-tidal processes. The significant increase in non-tidal energy is due to tidal damping and frictional losses through the culvert.

In addition to the in-depth harmonic analysis of the offshore data records, a simpler analysis of the offshore tide record was undertaken to determine the elevation of standard tide datums. These computed datums are presented in Table V-3. The Mean Higher High Water (MHHW) and Mean Lower Low Water (MLLW) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW. The tides in Nantucket Sound are semi-diurnal, meaning that there are typically two tide cycles in a day. There is usually a small variation in the level of the two daily tides. This variation can be seen in the differences between the MHHW and MHW, as well as the MLLW and MLW levels.

Table V-3. Tide datums computed from tide record collected in Nantucket Sound. Datum elevations are given relative to NGVD 27.

Tide Datum	Offshore (feet)
Maximum Tide	3.29
MHHW	2.49
MHW	2.13
MTL	0.68
MLW	-0.76
MLLW	-1.01
Minimum Tide	-1.56

V.4 HYDRODYNAMIC MODELING

For the modeling of Rushy Marsh, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in these systems. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers.

V.4.1 Model Theory

In its original form, RMA-2 was developed by William Norton and Ian King under contract with the U.S. Army Corps of Engineers (Norton et al., 1973). Further development included the introduction of one-dimensional elements, state-of-the-art pre- and post-processing data programs, and the use of elements with curved borders. Recently, the graphic pre- and post-processing routines were updated by a Brigham Young University through a package called the Surfacewater Modeling System or SMS (BYU, 1998). Graphics generated in support of this report primarily were generated within the SMS modeling package.

RMA-2 is a finite element model designed for simulating one- and two-dimensional depth-averaged hydrodynamic systems. The dependent variables are velocity and water depth, and the equations solved are the depth-averaged Navier Stokes equations. Reynolds assumptions are incorporated as an eddy viscosity effect to represent turbulent energy losses. Other terms in the governing equations permit friction losses (approximated either by a Chezy or Manning formulation), Coriolis effects, and surface wind stresses. All the coefficients associated with these terms may vary from element to element. The model utilizes quadrilaterals and triangles to represent the prototype system. Element boundaries may either be curved or straight.

The time dependence of the governing equations is incorporated within the solution technique needed to solve the set of simultaneous equations. This technique is implicit; therefore, unconditionally stable. Once the equations are solved, corrections to the initial estimate of velocity and water elevation are employed, and the equations are re-solved until the convergence criteria is met.

V.4.2 Model Setup

There are three main steps required to implement RMA-2:

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using the shorelines within Rushy Marsh from 1994 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified along the offshore boundary to Rushy Marsh based on the tide gauge data collected near the entrance to Rushy Marsh in Nantucket Sound. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.4.2.1 Grid Generation

The grid generation process was aided by the use of the SMS package. A 1994 digital aerial orthophoto and the bathymetry survey data were imported to SMS, and a finite element grid was generated to represent the embayment and inlet. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of adjoining salt marshes. The bathymetry data was interpolated to the developed finite element mesh of the system. The completed grid consists of 934 nodes, which describes 307 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -6.0 ft (NGVD 29), along the offshore boundary to Nantucket Sound. The completed grid mesh of Rushy Marsh is shown in Figure V-15.

The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties in Rushy Marsh. Fine grid resolution and one dimension elements were required to simulate the inlet culvert which has a significant impact the estuarine hydrodynamics. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary.

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing around the culvert were designed to provide a more detailed analysis in these regions of rapidly varying flow. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically. Appropriate implementation of wider spacing of nodes and larger elements reduced computer run time with no sacrifice of accuracy.

V.4.2.2 Boundary Condition Specification

Two types of boundary conditions were employed for the RMA-2 model of the Rushy Marsh system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified at the inlet. Tide gauge (TDR) measurements provided the required data. The rise and fall of the tide in Nantucket Sound is the primary driving force for estuarine circulation in this system. For the boundary a dynamic (time-varying) water surface elevation condition was specified every model time step (10 minutes) to represent the tidal forcing.

Figure V-15. Hydrodynamic model grid mesh for Rushy Marsh.

V.4.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Rushy Marsh was calibrated. The calibration procedure ensures that the model predicts accurately what was observed in nature during the field measurement program. Numerous model simulations are required for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the hydrodynamic model requires a close match between the modeled and measured water surface elevations in the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured water surface elevations. Once visual agreement is obtained, the normal procedure is to calibrate the model based on dominant tidal constituents. However, since the tidal fluctuations within Rushy Marsh are minimal due to tidal dampening through the culvert, a RMS error analysis was used to assess the agreement between modeled and measured water surface elevations.

The calibration was performed for a seven-day period beginning April 30, 2004 at 0000 EDT. This representative time period included the spring tide range of conditions in Nantucket Sound.

The calibrated model was used to analyze existing detailed flow patterns and compute residence times. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire seven-day simulation. Other methods, such as dye and salinity studies, evaluate tidal flushing over relatively short time periods (less than one day). These short-term measurement techniques may not be representative of average conditions due to the influence of unique, short-lived atmospheric events.

V.4.2.3.1 Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficients between 0.020 and 0.07 were specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels and marsh plains with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy/silty bottom found in Rushy Marsh, versus the inlet culvert, which provides greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-4. The extents of each material type are shown in Figure V-16.

Table V-4. Manning’s Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-16.

System Embayment	Bottom Friction
Offshore	0.040
Culvert	0.065
Rushy Marsh	0.025
Ground Water	0.025

V.4.2.3.2 Turbulent Exchange Coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 50 and 80 lb-sec/ft^2.

Figure V-16. Hydrodynamic model grid material properties. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.4.2.3.3 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model predictions for the first tide gauge (TDR) deployment was achieved using the aforementioned values for friction and turbulent exchange. Figure V-17 illustrates the seven-day calibration simulation for Rushy Marsh. Modeled (solid line) and measured (dotted line) tides are illustrate the tide record. For this 200-hour simulation, the RMA analysis confirms that the model has very good correlation with the measured marsh elevation data, with a computed R^2 correlation coefficient of 0.63, and an rms error of 0.05 ft.

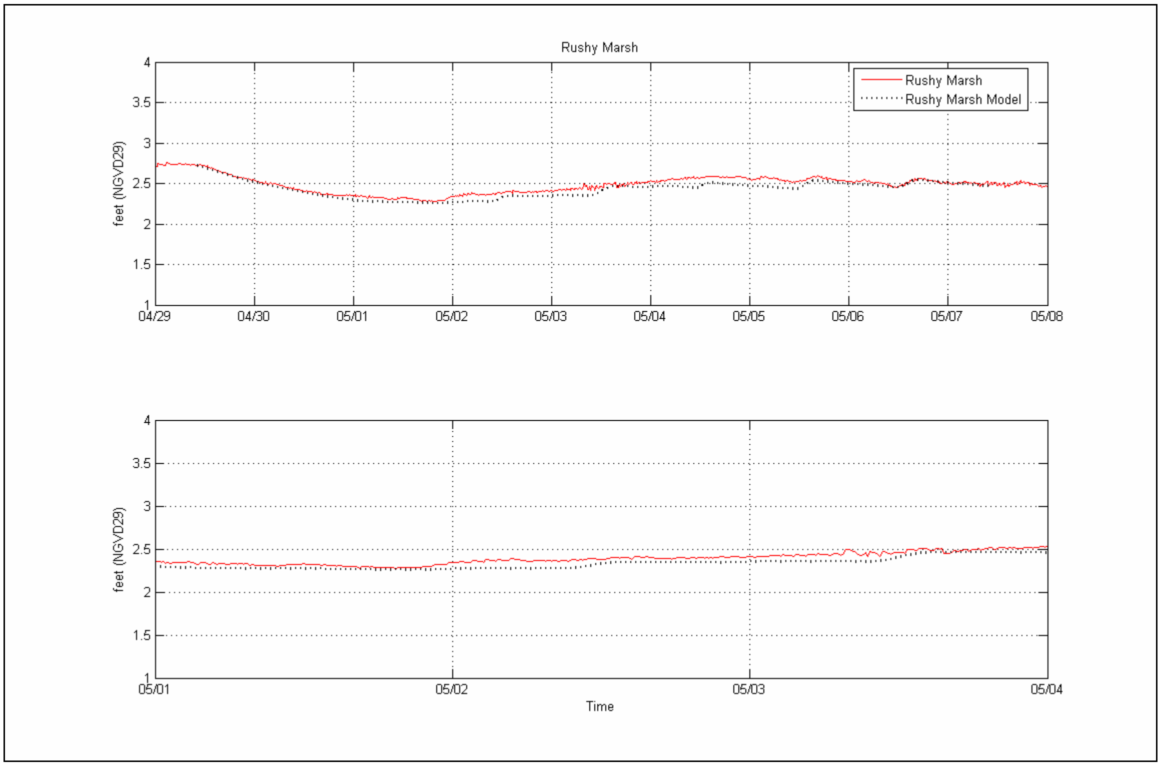


Figure V-17. Comparison of model output and measured tides for the gauge site in Rush Marsh. Differences between the observed (solid line) and modeled (dashed line) tidal elevation are generally imperceptible.

V.4.2.4 Model Circulation Characteristics

The final calibrated model serves as a useful tool in investigating the circulation characteristics of the system. Using model inputs of bathymetry and tide data, current velocities and flow rates can be determined at throughout the model domain. This is a very useful feature of a hydrodynamic model, where a limited amount of collected data can be expanded to determine the physical attributes of the system in areas where no physical data record exists.

Examining the results from the model run of Rushy Marsh shows the pond is minimally tidal and operates more as a salt water pond, with little to no current velocities within the main basin. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-18. The total flow rate of water flowing through a culvert was computed with the hydrodynamic model. Maximum flow rates occur during flood tides in this system, an indication that this estuary system is flood dominant, and likely a sediment sink (a system that accumulates sediment).

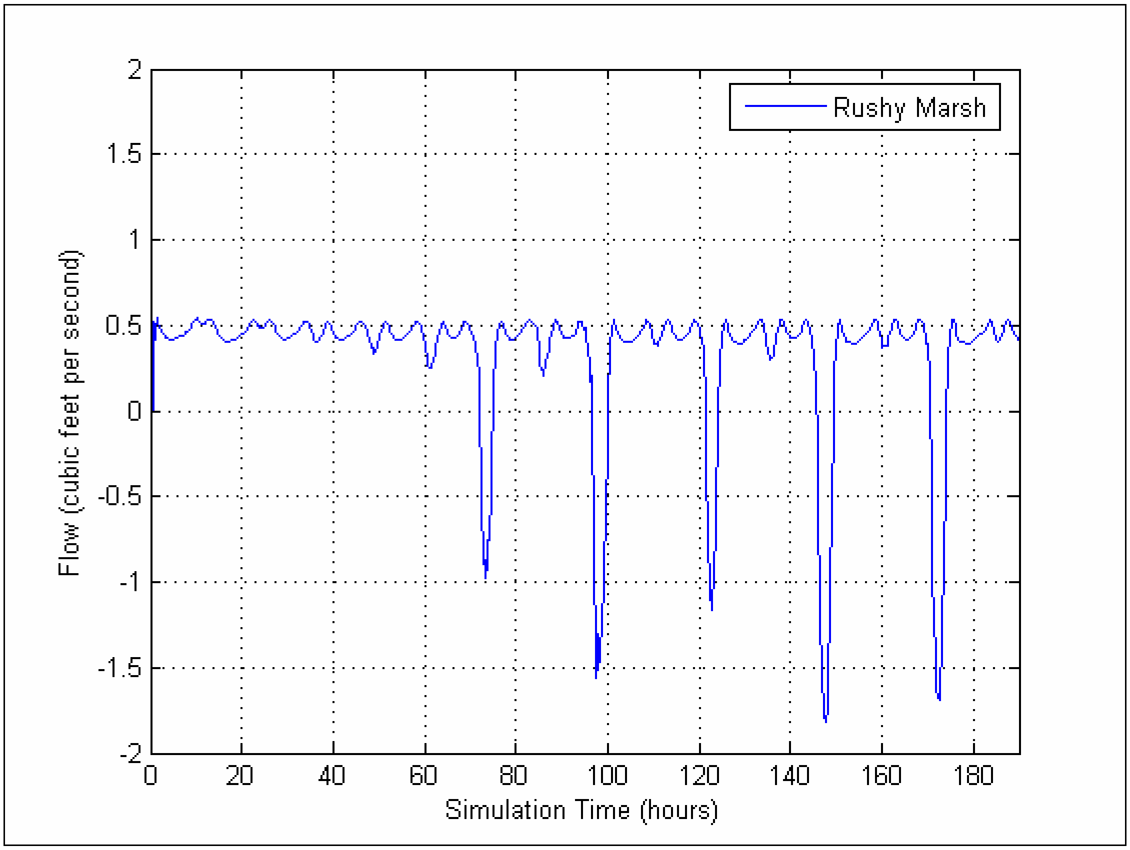


Figure V-18. Time variation of computed flow rate for inlet to Rushy Marsh. Model period shown corresponds to spring tide conditions, where the tide range is the largest. Positive flow indicated ebbing tide, while negative flow indicates flooding tide.

V.5 FLUSHING CHARACTERISTICS

Even with the highly restricted inlet, tidal exchange is much larger than the magnitude of freshwater inflow. The result is that the primary mechanism controlling estuarine water quality within the modeled Rushy Marsh system is tidal exchange. A rising tide offshore in Nantucket Sound creates a slope in water surface from the ocean into the modeled systems. Consequently, water flows into (floods) the system. Similarly, the estuary drains into the open waters of Nantucket Sound on an ebbing tide. This exchange of water between each system and the ocean is defined as tidal flushing. The calibrated hydrodynamic model is a tool to evaluate quantitatively tidal flushing of each system, and was used to compute flushing rates (residence times) and tidal circulation patterns.

Flushing rate, or residence time, is defined as the average time required for a parcel of water to migrate out of an estuary from points within the system. For this study, system residence times were computed as the average time required for a water parcel to migrate from a point within the each embayment to the entrance of the system. System residence times are computed as follows:

$$T_{system} = \frac{V_{system}}{P} t_{cycle}$$

where T_{system} denotes the residence time for the system, V_{system} represents volume of the (entire) system at mean tide level, P equals the tidal prism (or volume entering the system through a single tidal cycle), and t_{cycle} the period of the tidal cycle, typically 12.42 hours (or 0.52 days).

Residence times are provided as a first order evaluation of estuarine water quality. Lower residence times generally correspond to higher water quality; however, residence times may be misleading depending upon pollutant/nutrient loading rates and the overall quality of the receiving waters. As a qualitative guide, system residence times are applicable for systems where the water quality within the entire estuary is degraded and higher quality waters offshore provide the only means of reducing the high nutrient levels. This is a valid approach in the case of Rushy Marsh, since Nantucket Sound has relatively higher quality water then the Rushy Marsh.

The rate of pollutant/nutrient loading and the quality of adjacent offshore waters both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality will be obtained from the calibrated hydrodynamic model by extending the model to include pollutant/nutrient dispersion. The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the system (see Section VI).

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet.

The residence time was averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and is listed in Table V-6. The modeled time period used to compute the flushing rate was the modeled calibration period, and included the transition from spring to neap tide conditions. The model calculated flow crossing specified grid lines along the inlet to compute the tidal prism volume. Since the 7.25-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system embayment.

Table V-6. Embayment mean volume and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Rushy Marsh	2,848,800	30,936

Table V-7. Computed residence times for Rushy Marsh system.

Embayment	System Residence Time (days)
Rushy Marsh	47.7

The computed flushing rate for the Rushy Marsh shows that system takes approximately 48 days for the volume of the system to be exchanged. This suggests that the system has inadequate tidal flushing. This method assumes all the water in the system is exchanged, while in reality the water around the outlet of the culvert is most likely to be exchanged. Meaning that water quality continues to decrease in the remaining portions of the marsh.

Generally, possible errors in computed residence times can be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. In this study, the most significant errors associated with the bathymetry data result from the process of interpolating the data to the finite element mesh, which was the basis for all the flushing volumes used in the analysis. In addition, limited topographic measurements were available on the marsh plains. Minor errors may be introduced in residence time calculations by simplifying assumptions. Flushing rate calculations assume that water exiting the estuary does not return on the following tidal cycle. For regions where a strong littoral drift exists, this assumption is valid. However, water exiting a small sub-embayment on a relatively calm day may not completely mix with estuarine waters. In this case, the “strong littoral drift” assumption would lead to an under-prediction of residence time. Since littoral drift along the coast of Nantucket Sound typically is strong because of the local winds induce tidal mixing within the regional estuarine systems, the “strong littoral drift” assumption only will cause minor errors in residence time calculations. Based on our knowledge of estuarine processes, we estimate that the combined errors due to bathymetric inaccuracies represented in the model grid and the “strong littoral drift” assumption are within 10% to 15% of “true” residence times.

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort for the Rushy Marsh estuary system. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayment

Extensive field measurements and hydrodynamic modeling of the embayment were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated model output representing the transport of water within the system embayment. Files of node locations and node connectivity for the RMA-2V model grid were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 10-tidal cycle period in late April early May 2004. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model had reached a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayment

Three primary nitrogen loads to embayment are recognized in this modeling study: external loads from the watersheds, nitrogen load from direct rainfall on the embayment surface, and internal loads from the sediments. Additionally, there is a fourth load to Rushy Marsh, consisting of the background concentrations of total nitrogen in the waters entering from Nantucket Sound. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Four years of data (collected between 2002 and 2005) were available for stations monitored within Rushy Marsh by Three Bays Preservation in partnership with the Town of Barnstable, with technical assistance from the Coastal Systems Program, SMAST.

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Rushy Marsh estuary system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to

Table VI-1. Water Quality Monitoring Pond-Watcher measured data, and modeled Nitrogen concentrations for the Rushy Marsh system used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.

Sub-Embayment	monitoring station	2002 mean	2003 mean	2004 mean	2005 mean	data mean	s.d. all data	N	model min	model max	Model average
Rushy Marsh - north	RM1	1.174	1.411	0.991	0.898	1.111	0.271	28	1.098	1.106	1.102
Rushy Marsh - east	RM2	1.195	1.407	1.004	0.875	1.117	0.248	28	1.099	1.112	1.107
Rushy Marsh - west	RM3	1.317	1.447	0.974	0.812	1.123	0.306	22	1.103	1.112	1.108
Rushy Marsh - south	RM4	1.267	1.415	1.011	1.052	1.170	0.353	22	1.154	1.159	1.156

simulate the fluid dynamics of the Rushy Marsh. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other Cape Cod embayments, including systems in Falmouth (Ramsey *et al*., 2000); Mashpee, MA (Howes *et al*., 2004) and Chatham, MA (Howes *et al*., 2003).

The overall approach involves modeling total nitrogen as a non-conservative constituent, where bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. This modeling represents summertime conditions, when algal growth is at its maximum. Total nitrogen modeling is based upon various data collection efforts and analyses presented in previous sections of this report. Nitrogen loading information was derived from the Cape Cod Commission watershed loading analysis, Section IV.1 (based on the USGS watersheds, Chapter III), as well as the measured bottom sediment nitrogen fluxes (Section IV.3). Water column nitrogen measurements were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model of the system.

Figure VI-1. Estuarine water quality monitoring station locations in the Rushy Marsh system. Station labels correspond to those provided in Table VI-1.

VI.2.1 Model Formulation

The formulation of the model is for two-dimensional depth-averaged systems in which concentration in the vertical direction is assumed uniform. The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in the modeled sub-embayments. The governing equation of the RMA-4 constituent model can be most simply expressed as a form of the transport equation, in two dimensions:

$$\left(\frac{\partial c}{\partial t}+u\frac{\partial c}{\partial x}+v\frac{\partial c}{\partial y}\right)=\left(\frac{\partial}{\partial x}D_x\frac{\partial c}{\partial x}+\frac{\partial}{\partial y}D_y\frac{\partial c}{\partial y}+\sigma\right)$$

where c in the water quality constituent concentration; t is time; u and v are the velocities in the x and y directions, respectively; D_x and D_y are the model dispersion coefficients in the x and y directions; and σ is the constituent source/sink term. Since the model utilizes input from the RMA-2 model, a similar implicit solution technique is employed for the RMA-4 model.

The model is therefore used to compute spatially and temporally varying concentrations c of the modeled constituent (i.e., total nitrogen), based on model inputs of 1) water depth and velocity computed using the RMA-2 hydrodynamic model; 2) mass loading input of the modeled constituent; and 3) user selected values of the model dispersion coefficients. Dispersion coefficients used for each system sub-embayment were developed during the calibration process. During the calibration procedure, the dispersion coefficients were incrementally changed until model concentration outputs matched measured data.

The RMA-4 model can be utilized to predict both spatial and temporal variations in total for a given embayment system. At each time step, the model computes constituent concentrations over the entire finite element grid and utilizes a continuity of mass equation to check these results. Similar to the hydrodynamic model, the water quality model evaluates model parameters at every element at 10-minute time intervals throughout the grid system. For this application, the RMA-4 model was used to predict tidally averaged total nitrogen concentrations throughout Rushy Marsh.

VI.2.2 Water Quality Model Setup

Required inputs to the RMA-4 model include a computational mesh, computed water elevations and velocities at all nodes of the mesh, constituent mass loading, and spatially varying values of the dispersion coefficient. Because the RMA-4 model is part of a suite of integrated computer models, the finite-element meshes and the resulting hydrodynamic simulations previously developed for the Rushy Marsh system was used for the water quality constituent modeling portion of this study.

Based on groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include the latest estimate of surface water flow into Rushy Marsh. The groundwater recharge has a measure flow rate of 0.28 ft^3/sec (682 m^3/day).

For the model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (28 day) spin-up period. At the end of the spin-up period, the model was run for an additional 5 tidal-day (125 hour) period. Model results were recorded only after the initial spin-up period. The time step used for the water quality computations was 10 minutes, which corresponds to the time step of the hydrodynamics input for the Rushy Marsh system.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, 3) summer benthic regeneration. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed direct atmospheric deposition load for Rushy Marsh was evenly distributed at grid cells that formed the perimeter of the embayment. Benthic regeneration load was distributed among another sub-set of grid cells which are in the interior portion of each basin.

The loadings used to model present conditions in Rushy Marsh estuary systems are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV.1. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV.3. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, some sub-embayments have almost twice the loading rate from benthic regeneration as from watershed loads. For other sub-embayments, the benthic flux is relatively low or negative indicating a net uptake of nitrogen in the bottom sediments.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Vineyard Sound was set at 0.280 mg/L, based on SMAST data from the Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Nantucket Sound.

Table VI-2. Sub-embayment and surface water loads used for total nitrogen modeling of the Rushy Marsh, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions.**

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Rushy Marsh	0.447	0.203	-0.197

VI.2.4 Model Calibration

Calibration of the total nitrogen model proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in Section V. Observed values of *E* (Fischer, *et al.*, 1979) vary between order 10 and order 1000 m^2/sec for riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents (from tides or atmospheric forcing). Generally, the relatively quiescent Rushy Marsh embayment system require values of *E* that are lower compared to the riverine estuary systems evaluated by Fischer, et al., (1979). Observed values of *E* in these calmer areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). Final values of *E* used in each sub-embayment of the modeled systems are presented in Table VI-3. These values were used to develop the "best-fit" total nitrogen model calibration. For the case of TN modeling, "best fit" can be defined as minimizing error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Rushy Marsh estuary system.

Embayment Division	E m^2/sec
Main Basin Rushy Marsh	1.0
Southern Arm Rushy Marsh	1.0
Culvert	0.7
Nantucket Sound	8.0

Comparisons between model output and measured nitrogen concentrations are shown in plots presented in Figure VI-2. In these plots, means of the water column data and a range of two standard deviations of the annual means at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the SMAST monitoring stations.

For model calibration, the mid-point between maximum modeled TN and average modeled TN was compared to mean measured TN data values, at each water-quality monitoring station. The calibration target would fall between the modeled mean and maximum TN because the monitoring data are collected, as a rule, during mid ebb tide.

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for the system. The model fit is exceptional for the Rushy Marsh model, with rms error of 0.01 mg/L and an R^2 correlation coefficient of 0.72.

A contour plot of calibrated model output is shown in Figure VI-3 for Rushy Marsh. In the figure, color contours indicate nitrogen concentrations throughout the model domain. The output in the figure show average total nitrogen concentrations, computed using the full 5-tidal-day model simulation output period.

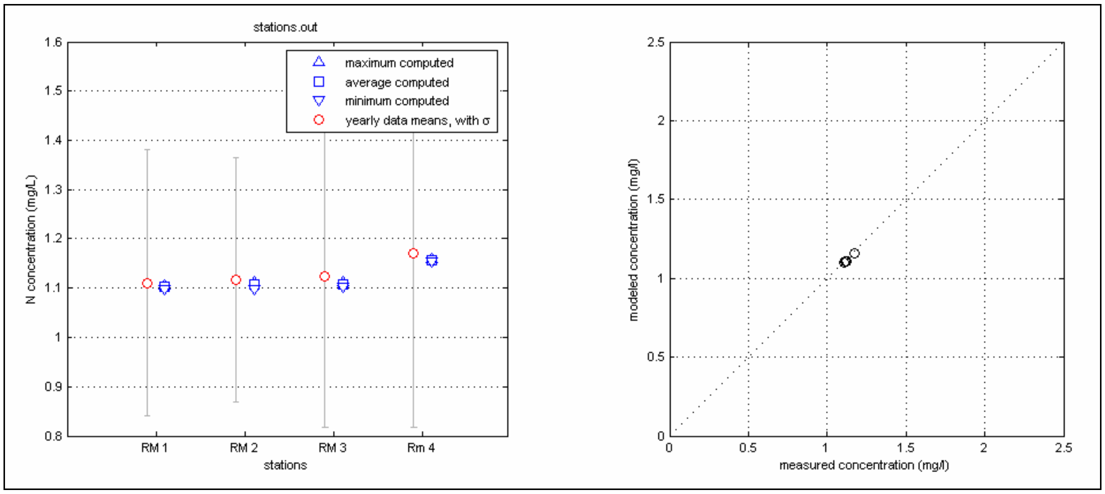


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Rushy Marsh. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

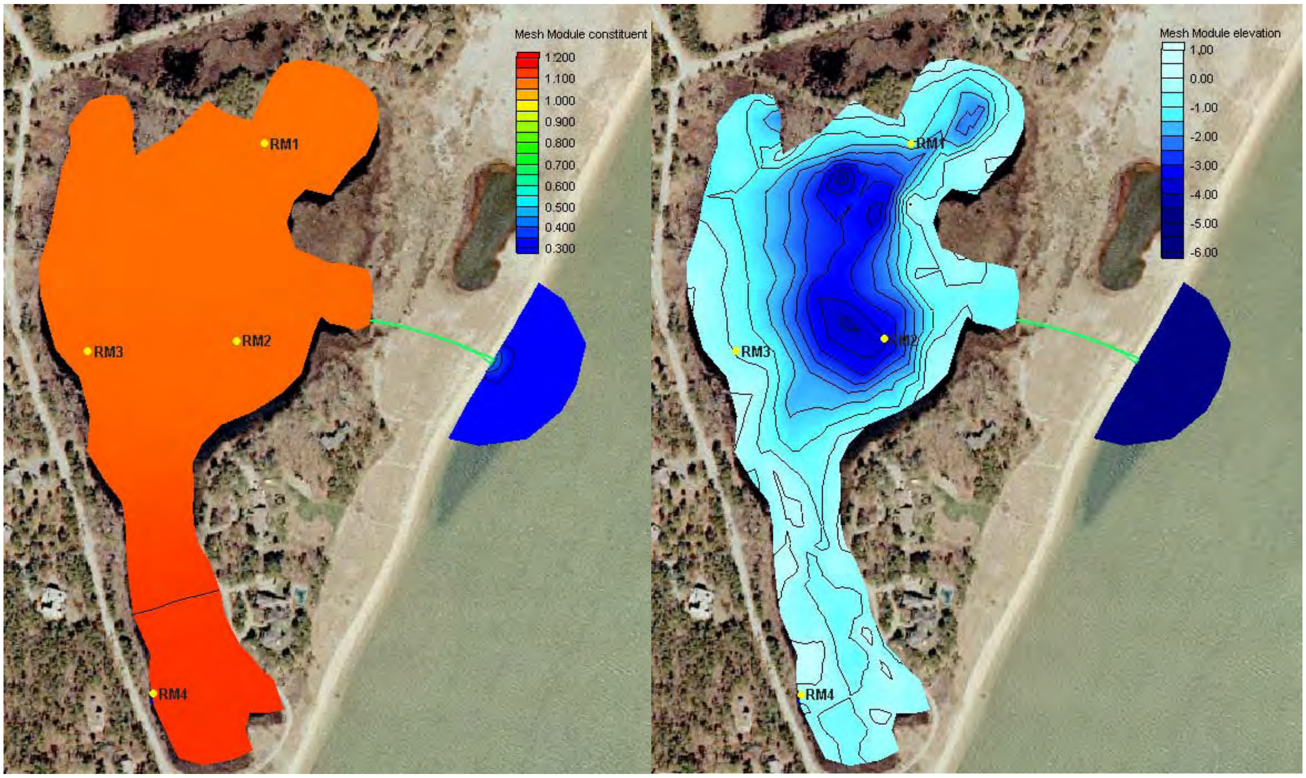


Figure VI-3. Contour plots of average total nitrogen concentrations from results of the present conditions loading scenario and the bathymetry, for Rushy Marsh. The approximate location of the sentinel threshold station for Rushy Marsh (RM2) is shown.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Rushy Marsh system using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary and groundwater inputs. The open boundary salinity was set at 29.6 ppt. For groundwater inputs salinities were set at 0 ppt. Groundwater input used for the model was 0.28 ft^3/sec (682 m^3/day). Groundwater flows were distributed evenly in each model through the use of several 1-D element input points positioned along each model's land boundary.

Comparisons of modeled and measured salinities are presented in Figure VI-4, with contour plots of model output shown in Figure VI-5. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in Rushy Marsh. The rms error of the models was 1.3 ppt, and correlation coefficient was 0.95. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical systems.

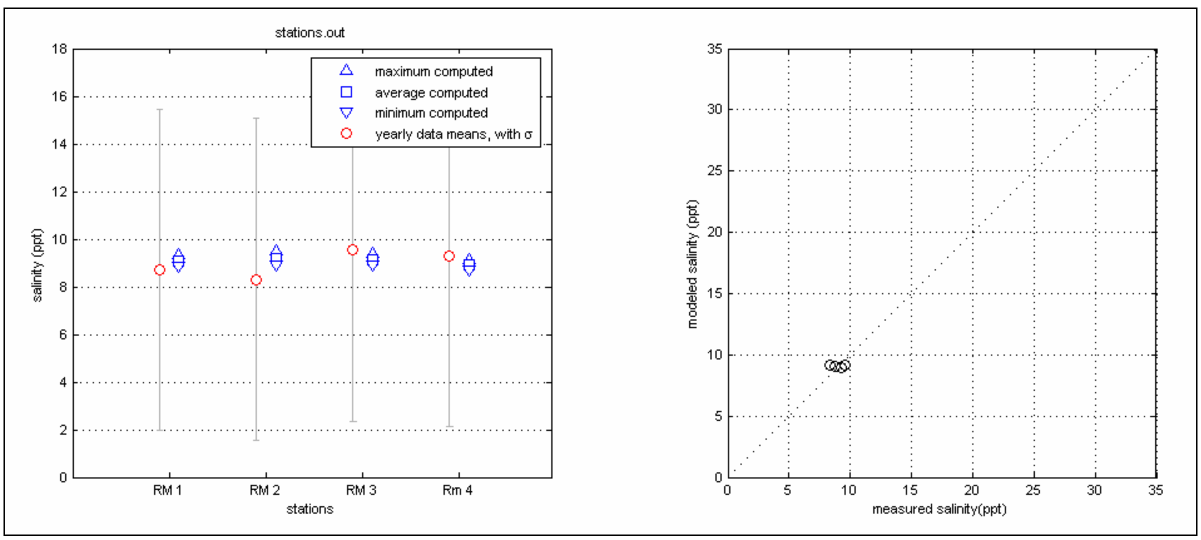


Figure VI-4. Comparison of measured and calibrated model output at stations in Rushy Marsh. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

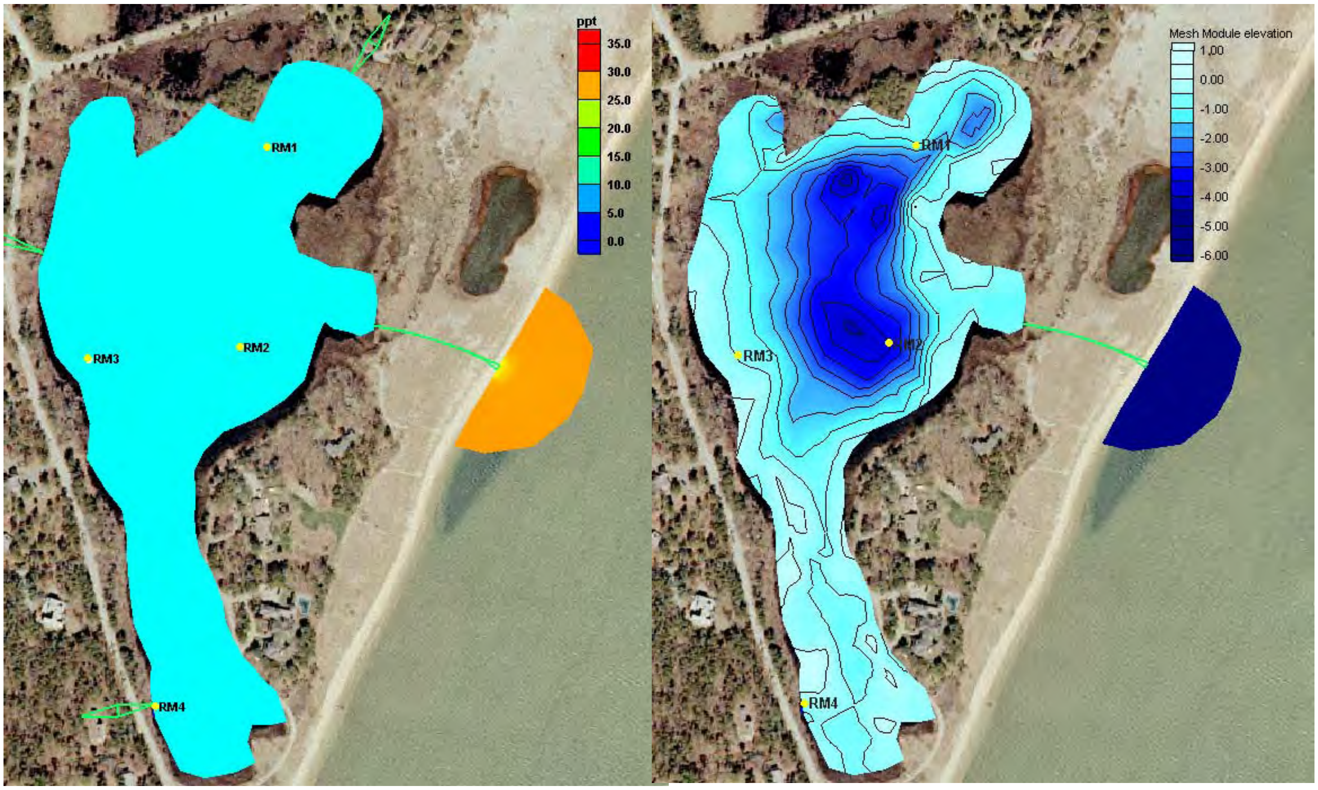


Figure VI-5. Contour plots of modeled salinity (ppt) and bathymetry in Rushy Marsh.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

To assess the influence of nitrogen loading on total nitrogen concentrations within the embayment system, two standard water quality modeling scenarios were run: a "build-out" scenario based on potential development (described in more detail in Section IV) and a "no anthropogenic load" or "no load" scenario assuming only atmospheric deposition on the watershed and sub-embayment, as well as a natural forest within each watershed. Comparisons of the alternate watershed loading analyses are shown in Table VI-4. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability.

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Rushy Marsh system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	build out (kg/day)	build-out % change	no load (kg/day)	no load % change
Rushy Marsh Pond	0.447	0.907	+103.1%	0.077	-82.8%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be more than a 100% increase in watershed nitrogen load to Rushy Marsh as a result of potential future development. For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions by over 80%.

For the build-out scenario, a breakdown of the total nitrogen load entering the Rushy Marsh embayment is shown in Table VI-5. The benthic flux for the build-out scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase in benthic flux (i.e., a positive change in the absolute value of the flux), and *vise versa*.

Projected benthic fluxes (for both the build-out and no load scenarios) are based upon projected PON concentrations and watershed loads, determined as:

$$(Projected\ N\ flux) = (Present\ N\ flux) * [PON_{projected}]/[PON_{present}]$$

where the projected PON concentration is calculated by,

$$[PON_{projected}] = R_{load} * \Delta PON + [PON_{(present\ offshore)}],$$

using the watershed load ratio,

$$R_{load} = (Projected\ N\ load) / (Present\ N\ load),$$

and the present PON concentration above background,

$$\Delta PON = [PON_{(present\ flux\ core)}] - [PON_{(present\ offshore)}].$$

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Rushy Marsh, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Rushy Marsh	0.907	0.203	-0.310

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Rushy Marsh was run to determine nitrogen concentrations at each measurement station (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. Total N concentrations increased evenly across the system due to the limited marsh circulation. Color contours of model output for the build-out scenario are present in Figure VI-6. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-3, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Rushy Marsh system. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Rushy Marsh - north	RM1	1.102	1.237	+12.2%
Rushy Marsh - east	**RM2**	**1.107**	**1.245**	**+12.5%**
Rushy Marsh - west	RM3	1.108	1.246	+12.5%
Rushy Marsh - south	RM4	1.156	1.320	+14.2%

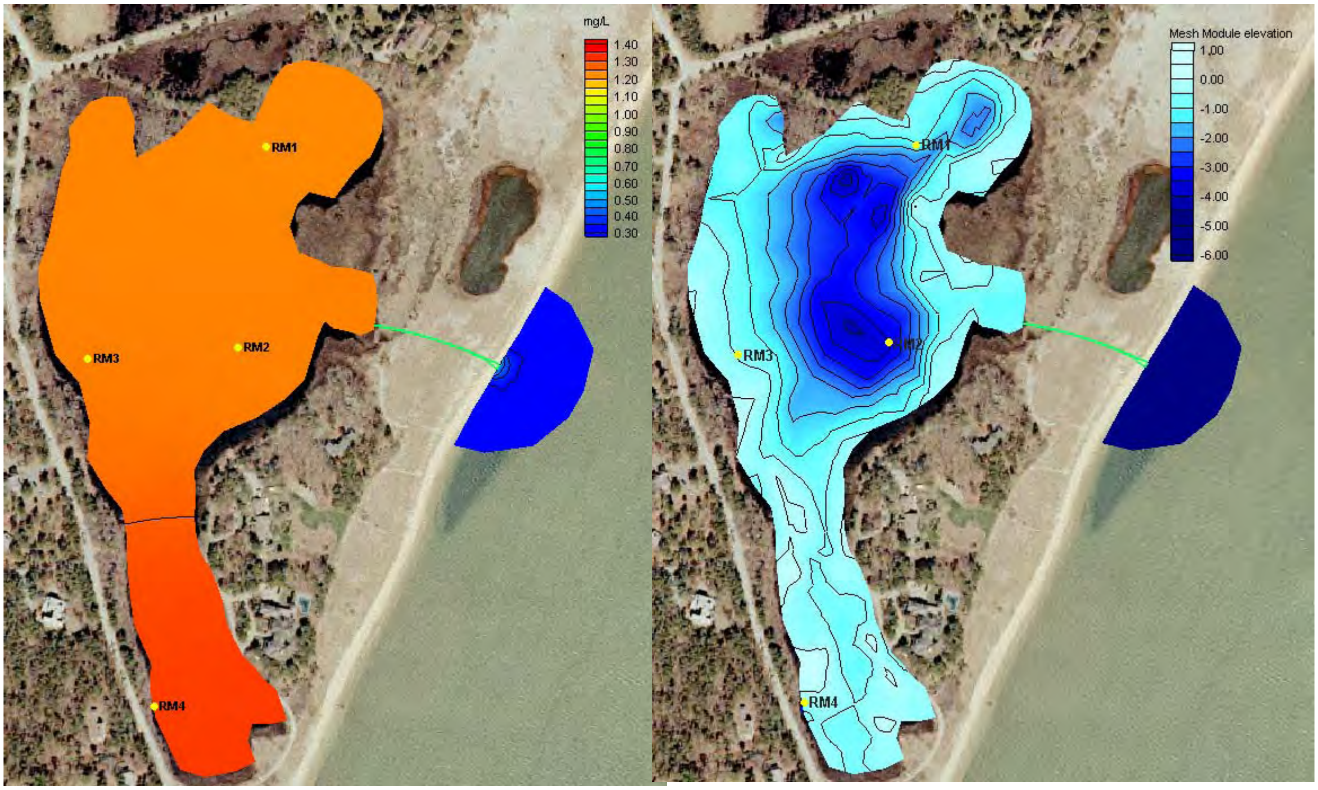


Figure VI-6. Contour plots of modeled total nitrogen concentrations (mg/L) in Rushy Marsh, for projected build-out loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Rushy Marsh (RM2) is shown.

VI.2.6.2 No Anthropogenic Load

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load ("no load") scenario is shown in Table VI-7. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in §VI.2.6.1). Compared to the modeled present conditions and build-out scenario, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of Rushy Marsh, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Rushy Marsh	0.077	0.203	-0.110

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations at each recording station. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions greater than 12% occurring the all portions of the systems. Results for each system are shown pictorially in Figure VI-7.

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for the Rushy Marsh system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Rushy Marsh - north	RM1	1.102	0.968	-12.2%
Rushy Marsh - east	**RM2**	**1.107**	**0.970**	**-12.4%**
Rushy Marsh - west	RM3	1.108	0.971	-12.4%
Rushy Marsh - south	RM4	1.156	0.996	-13.8%

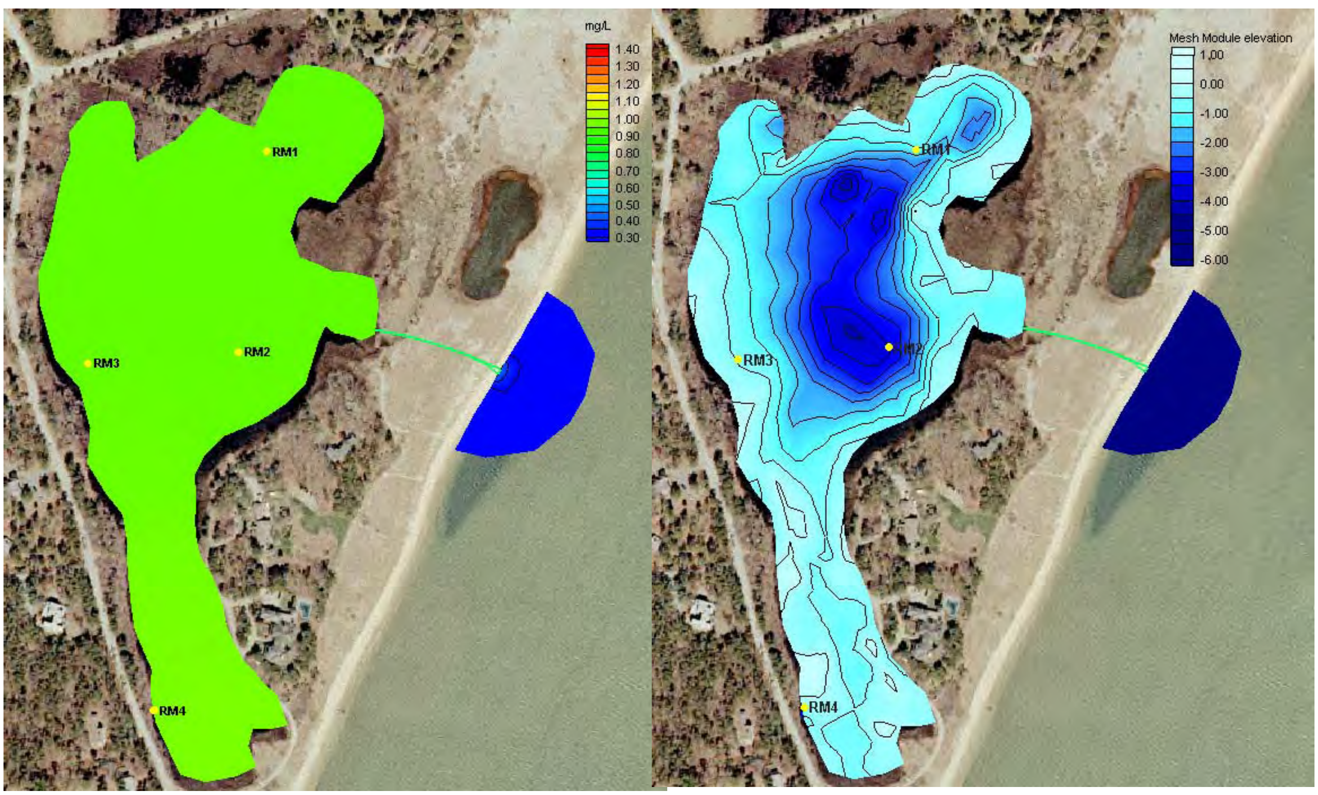


Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Rushy Marsh, for no anthropogenic loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Rushy Marsh (RM2) is shown.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Rushy Marsh embayment system, our assessment is based upon data from the water quality monitoring database and our surveys of eelgrass distribution, benthic animal communities and sediment characteristics, and dissolved oxygen records conducted during the summer of 2004. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

There are a variety of indicators that can be used in concert with water quality monitoring data for evaluating the ecological health of embayment systems. The best biological indicators are those species which are non-mobile and which persist over relatively long periods, if environmental conditions remain constant. The concept is to use species which integrate environmental conditions over seasonal to annual intervals. The approach is particularly useful in environments where high-frequency variations in structuring parameters (e.g. light, nutrients, dissolved oxygen, etc.) are common, making adequate field sampling difficult.

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within Rushy Marsh Pond (2004) to record the frequency and duration of low oxygen conditions during the critical summer period, July-September.

The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Assessment of eelgrass beds, past and present, within the Rushy Marsh System was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. Within Rushy Marsh Pond present and/or historic eelgrass distribution provides guidance for setting habitat restoration goals and associated nitrogen thresholds.

In areas that do not support eelgrass beds, benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H’) and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 3.8 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Rushy Marsh Pond System, as marine waters, are listed under this Classification as SA. It should be noted that the Classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within the central basin of the Rushy Marsh basin (Figure VII-2). The sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during the 3 month deployment, July – September.

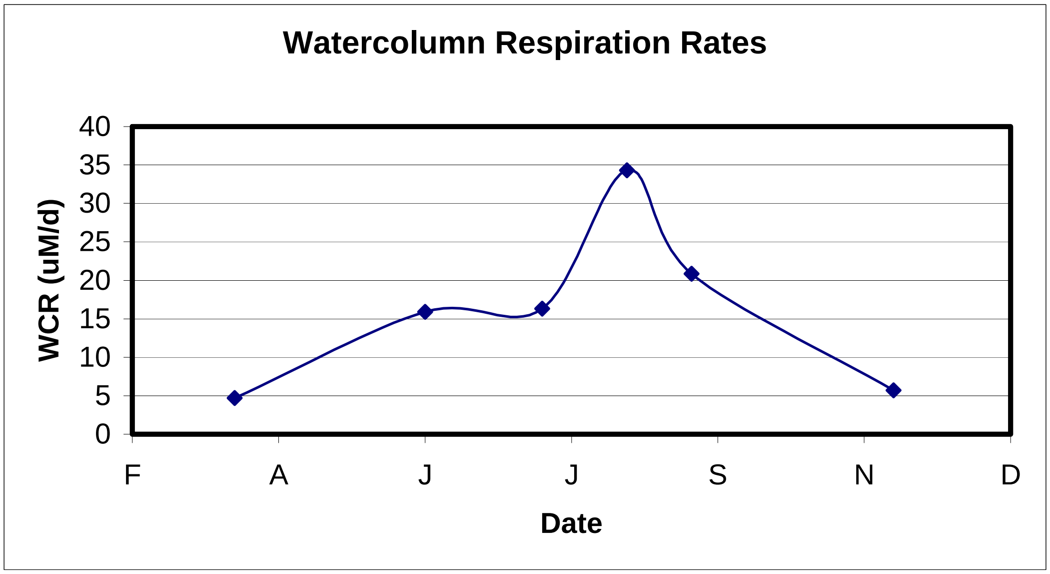


Figure VII-1. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Similar to other embayments in southeastern Massachusetts, the Rushy Marsh system evaluated in this assessment showed high frequency variation, apparently related to diurnal and sometimes tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration the each mooring site, underscores the need for continuous monitoring within these systems.

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.
The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality within the estuary (Figures VII-3 through VII-4). Oxygen depletion was frequently to levels <4 mg/L (29 days) and periodically to < 3 mg/L (8 days). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of this estuarine system, although the nitrogen enrichment stems primarily from the restriction of tidal exchange. The frequent significant level of oxygen depletion coupled to the frequent phytoplankton blooms is clear evidence of that Rushy Marsh Pond is presently nitrogen over-loaded eutrophic embayment.

Figure VII-2. Aerial Photograph of the Rushy Marsh embayment system in the Town of Barnstable showing locations of the Dissolved Oxygen mooring deployment conducted in the Summer of 2004.

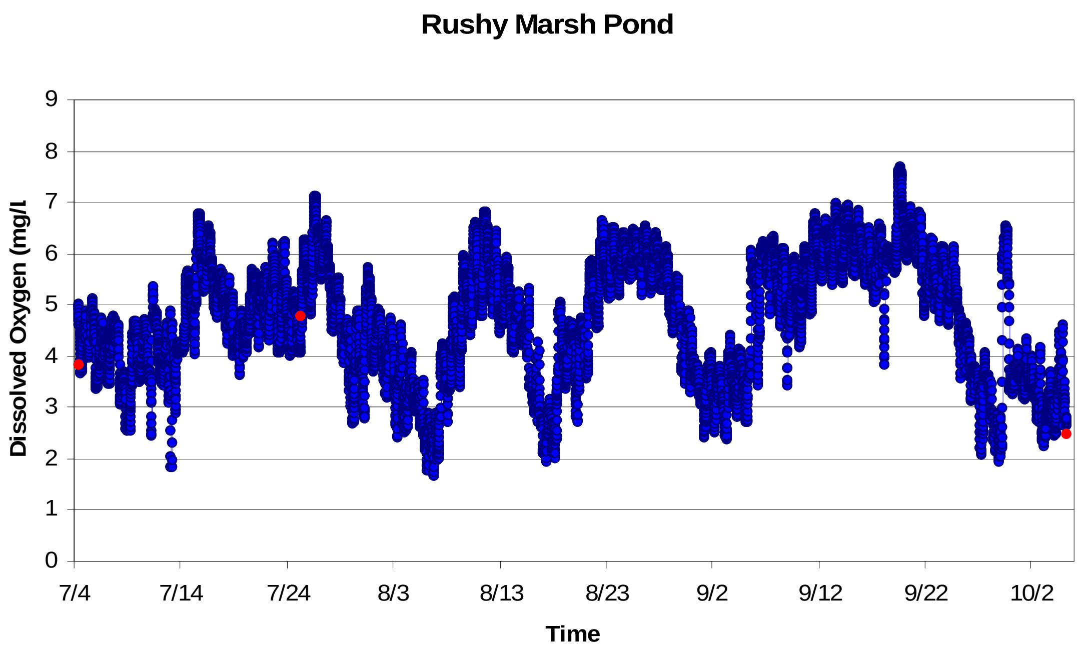


Figure VII-3. Bottom water record of dissolved oxygen in the Rushy Marsh Pond station, Summer 2004. Calibration samples represented as red dots

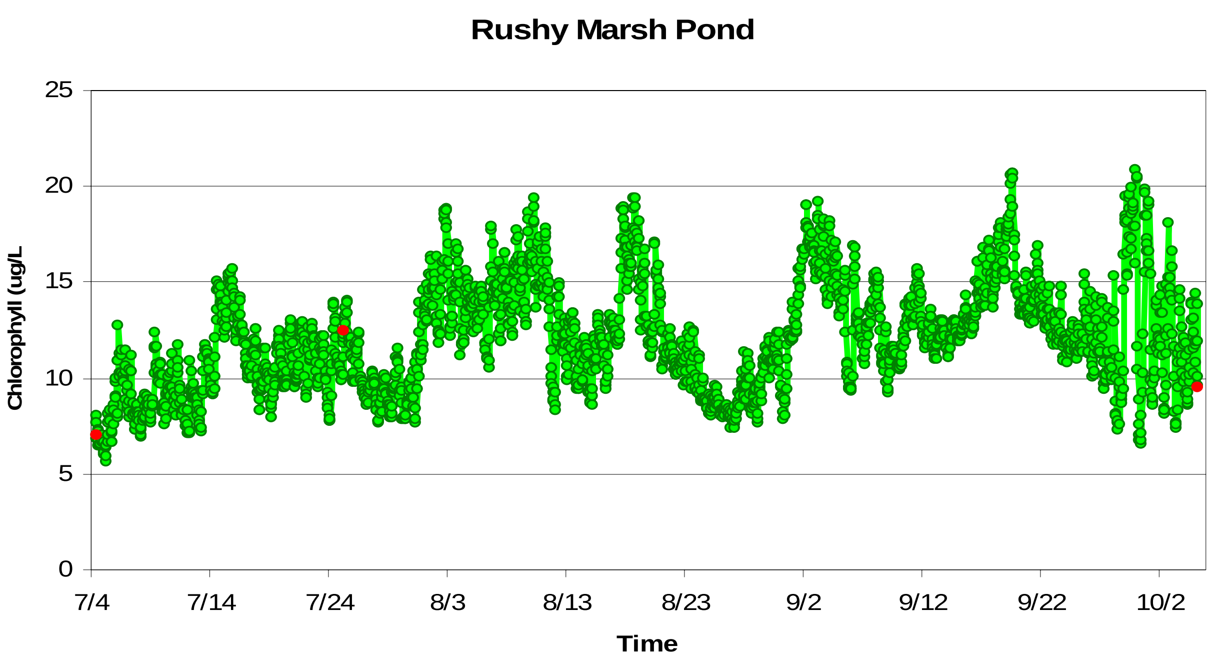


Figure VII-4. Bottom water record of Chlorophyll-*a* in the Rushy Marsh Pond station, Summer 2004. Calibration samples represented as red dots

Table VII-1. Bottom water dissolved oxygen levels within the principal sub-embayments to the Rushy Marsh Estuary. Percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels during July – September 2004.

Sub-Embayment	Dissolved Oxygen: Continuous Record, Summer 2000- 2002				
	Deployment Days	**< 6 mg/L (% of days)**	**< 5 mg/L (% of days)**	**< 4 mg/L (% of days)**	**< 3 mg/L (% of days)**
Rushy Marsh	93.0	80%	53%	31%	9%

Table VII-2. Duration (% of deployment time) that chlorophyll a levels exceed various benchmark levels within the embayment system. "Mean" represents the average duration of each event over the benchmark level and "S.D." its standard deviation. Data collected by the Coastal Systems Program, SMAST. The mean in the final column is the average level over the deployment.

Sub-Embayment	**Start Date**	**End Date**	**Total Deployment (Days)**	**> 5 ug/L Duration (Days)**	**> 10 ug/L Duration (Days)**	**> 15 ug/L Duration (Days)**	**> 20 ug/L Duration (Days)**	**> 25 ug/L Duration (Days)**	**Mean Chl a Level (ug/L)**
Rushy Marsh Pond	7/4/2004	10/5/2004	93.0	**99.9%**	**99.1%**	**73.3%**	**52.7%**	**15.6%**	
		Mean		23.3	1.4	1.03	1.02	0.30	12.0
		S.D.		24.4	0.8	2.67	0.58	0.41	

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted for the Rushy Marsh Estuary by the DEP Eelgrass Mapping Program as part of the MEP Technical Team by other members of the Team. Field survey data Surveys was conducted in 2004, in concert with the MEP benthic recycling and infaunal animal sampling. Additional analysis of available aerial photos from 1951 were examined to determine eelgrass distribution under conditions of lower watershed nitrogen loading. The 1951 data were only anecdotally validated, while the 2004 information was by direct observation. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of the data sets can provide a view of temporal trends in eelgrass distribution from 1951 to 2004; the period in which watershed nitrogen loading increased (although present level is only moderate to low). This temporal information can be used to determine the stability of the eelgrass community.

At present, eelgrass is not present within Rushy Marsh Pond. Rushy Marsh Pond is functionally a basin with fringing wetland, and the sediments are currently soft muds rich in organic matter, which in some locations overlay medium to fine sands. The current lack of eelgrass beds is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within this system. In addition, it does not appear that eelgrass beds were present in the system in 1951, as well. It appears that the restriction of the tidal exchange starting circa 1900, resulted in an absence of eelgrass sometime prior to 1951 (Chapter V). The restriction of tidal exchange has resulted in an enrichment of estuarine waters in nitrogen to the extent that the system is currently eutrophic. Restoration of tidal exchange will be needed for habitat restoration of this system, as watershed nitrogen inputs are relatively low.

Other factors which influence eelgrass bed loss in embayments are not at play in Rushy Marsh Pond. However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss does not seem to be directly related to mooring density, as the Bay supports no boat moorings. Similarly, pier construction and boating pressure may be adding additional stress in nutrient enriched areas, but are not a factor in Rushy Marsh as there are no docks. Similarly, there is no present effect of shellfishing on benthic habitat, as there is no shellfishing harvesting.

Given that eelgrass has not been documented for this system, it is not clear that even when the system was much better flushed, it supported eelgrass beds. However, observations of brackish water submerged aquatic vegetation in the shallow region of the western channel suggest that eelgrass habitat might be sustainable under lower effective nitrogen loading rates (i.e. higher flushing). To the extent that conditions could be improved to the level of eelgrass colonization in this system, the acreage would likely range from 4-12 acres, most likely in the southern channel and the margins of the main basin.

The eelgrass information is consistent with the results of the benthic infauna analysis and the observed eelgrass loss is typical of nutrient enriched shallow embayments (see below).

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 7 locations throughout the Rushy Marsh Pond Estuary (Figure VII-5). In some cases multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, portions of the Three Bays System are clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records and eelgrass coverage, have the highest diversity (generally >3) and evenness (~0.7). The converse is also true, with poorest habitat quality found where diversity is <1 and evenness is <0.5.

The Infauna Study indicated that presently, habitat capable of supporting benthic infaunal communities is virtually absent in Rushy Marsh Pond (Table VII-3). The infaunal survey found that summer conditions apparently are sufficient to prevent a community from developing in the central basin. In the shallower southern channel region, again only very few individuals and species were found. The low numbers of species and individuals indicates that benthic infaunal habitat has been severely degraded throughout Rushy Marsh Pond. The conditions proximately result from the high level of nitrogen and organic matter enrichment and associated oxygen depletion of bottom waters. Ultimately, the cause is the highly restricted tidal exchange and very low flushing rate of Pond waters (system residence time ~48 d). However, restoration of infaunal animal communities should occur at the point that habitat can be restored.

Figure VII-5. Aerial photograph of the Rushy Marsh embayment system showing location of benthic infaunal sampling stations (green symbol).

Table VII-3. Benthic infaunal community data for the Rushy Marsh embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2).

Location	ID	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)	Infaunal Indicators
Rushy Marsh Pond							
Channel	RMP-1	5	9	N/A	1.94	0.83	
	RMP-1a	1	1	N/A	0	N/A	
	RMP-2	3	10	N/A	1.33	0.84	
	RMP-2a	1	2	N/A	0.50	N/A	
	Average	2.5	6	N/A	0.94	N/A	***SD***[1]
Main Basin	RMP-3	0	0	N/A	N/A	N/A	
	RMP-4	0	0	N/A	N/A	N/A	
	RMP-5	0	0	N/A	N/A	N/A	
	RNP-6	0	0	N/A	N/A	N/A	
	RMP-8/9	0	0	N/A	N/A	N/A	
	Average	0	0	N/A	N/A	N/A	***SD***

H – Healthy, MI – Moderately Impaired, SI – Significantly Impaired, SD – Severely Degraded. A system is severely degraded if the infaunal community shows both low numbers of individuals and species or is dominated by stress indicator species.

VIII. CRITICAL NUTRIENT THRESHOLD DETERMINATION AND DEVELOPMENT OF WATER QUALITY TARGETS

VIII.1 ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were collected to support threshold development for the Rushy Marsh Pond System by MEP Team and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the long-term baseline Water Quality Monitoring Program conducted by Three Bays Preservation in partnership with the Town of Barnstable, with technical guidance from the Coastal Systems Program at SMAST. At present, Rushy Marsh Pond is showing significantly impaired to severely degraded habitat quality. All of the habitat indicators are consistent with this evaluation of the whole of system (Chapter VII).

Eelgrass: At present, eelgrass is not found within Rushy Marsh Pond. The current lack of eelgrass beds is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within this system. In addition, it does not appear that eelgrass beds were present in the system in 1951. It appears that the restriction of the tidal exchange starting circa 1900, resulted in an absence of eelgrass sometime prior to 1951. The restriction of tidal exchange has resulted in an enrichment of estuarine waters in nitrogen to the extent that the system is currently eutrophic. Restoration of tidal exchange will be needed for habitat restoration of this system, as watershed nitrogen inputs are relatively low.

Given that eelgrass has not been documented for this system, it is not clear that even when the system was much better flushed, it supported eelgrass beds. However, observations of brackish water submerged aquatic vegetation in the shallow region of the western channel suggest that eelgrass habitat might be sustainable under lower effective nitrogen loading rates (i.e. higher flushing). To the extent that conditions could be improved to the level of eelgrass colonization in this system, the acreage would likely range from 4-12 acres, most likely in the southern channel and the margins of the main basin.

The eelgrass information is consistent with the results of the benthic infauna analysis and the observed eelgrass loss is typical of nutrient enriched shallow embayments (see below).

Water Quality: Rushy Marsh Pond currently exhibits seasonal oxygen stress, consistent with nitrogen enrichment (Tables VII-1, VII-2). That the cause is eutrophication is supported by the high levels of chlorophyll a, 15 ug/L to >20 ug/L (Table VII-2). Oxygen conditions and chlorophyll a levels indicated nutrient related stress throughout the Pond

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality within the estuary (Figures VII-3, VII-4). Oxygen depletion was frequently to levels <4 mg/L (29 days) and periodically to < 3 mg/L (8 days). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of this estuarine system, although the nitrogen enrichment stems primarily from the restriction of tidal exchange. The frequent significant level of oxygen depletion coupled to the frequent phytoplankton blooms is clear evidence of that Rushy Marsh Pond is presently nitrogen over-loaded eutrophic embayment. The chlorophyll a, dissolved oxygen and total nitrogen within Rushy Marsh Pond are consistent with the observed eelgrass losses (above) and the significantly impaired infaunal animal communities (below).

Infaunal Communities: The Infauna Study indicated that all areas are presently severely degraded and that habitat capable of sustaining benthic infaunal animals is virtually absent in Rushy Marsh Pond (Table VII-3). The infaunal survey found that summer conditions apparently are sufficient to prevent a community from developing in the central basin. In the shallower southern channel region, again only very few individuals and species were found. The low numbers of species and individuals indicates that benthic infaunal habitat has been severely degraded throughout Rushy Marsh Pond. The conditions proximately result from the high level of nitrogen and organic matter enrichment and associated oxygen depletion of bottom waters. Ultimately, the cause is the highly restricted tidal exchange and very low flushing rate of Pond waters (system residence time ~48 d). However, restoration of infaunal animal communities should occur at the point that habitat can be restored.

The infaunal community based classification throughout Rushy Marsh Pond is fully supported by the water quality and eelgrass data discussed in the text above.

Table VIII-1. Summary of Nutrient Related Habitat Health within the Rushy Marsh Pond Estuary on the south shore of Barnstable, MA., based upon assessment data presented in Chapter VII.

Health Indicator	Estuary: Rushy Marsh — Main Basin	Estuary: Rushy Marsh — Channel
Dissolved Oxygen	SI/SD[1]	--
Chlorophyll	SI	SI
Macroalgae	SI/SD[2]	SI
Eelgrass	SI/SD[3]	SI[4]
Infaunal Animals	SD[5]	SD
Overall:	**SD**	**SD**

1 – periodic oxygen depletions to <3 mg/L and frequently <4 mg/L.
2 – macroalgal accumulations on bottom.
3 – no eelgrass in pond presently and likely for over 60 years.
4 – no eelgrass, but strands of brackish submerged aquatic vegetation, possibly *Ruppia*.
5 – virtual absence of infaunal animal community in 2004 survey.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment; SD = Severe Degradation
-- = not applicable to this estuarine reach

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout and embayment system, is to first identify a sentinel location within the embayment and second to determine the nitrogen concentration within the water column which will restore that location to the desired habitat quality. The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels. Once the sentinel site and its target nitrogen level are determined, the Linked Watershed-Embayment Model is used to sequentially adjust nitrogen loads until the targeted nitrogen concentration is achieved.

Within the Rushy Marsh Estuary the most appropriate sentinel station was about in the center of the basin at Station RM2 in Figure VIII-1. This location was selected because restoration of nitrogen conditions supportive of eelgrass or infauna at this location will necessarily result in similar quality conditions throughout the basin. As is shown below and in Chapter IX, concentrations at the Sentinel Station (RM2) approximate concentrations throughout the pond waters (i.e. it is representative or higher than the other pond locations).

Following the MEP protocol, since eelgrass has not been documented in Rushy Marsh Pond, restoration of infaunal habitat is the restoration goal. Infaunal animal habitat is a critical resource to the Rushy Marsh System and estuaries in general. Since there are virtually no infaunal animals remaining in the sub-tidal Rushy Marsh Pond sediments, comparisons to the muddy basins of other nearby estuarine systems were relied upon for setting the nitrogen threshold for healthy infaunal habitat at a nitrogen level of TN <0.5 mg TN L^{-1}. This level was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 were found supportive of high infaunal habitat quality in this system. Similarly, in the Three Bays System, healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay), with impairment in areas where nitrogen levels of TN >0.5 mg TN L^{-1} (North Bay), and severe degradation at nitrogen levels of TN >0.6 mg TN L^{-1}.

Given the low watershed nitrogen load to Rushy Marsh Pond, reconstruction of the tidal inlet will be required to meet the nitrogen threshold level and achieve restoration of this system. In addition, restoration of tidal exchange (i.e. tide range) will allow the restoration of fringing salt marsh in this system, which has lost its salt water wetlands.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Rushy Marsh. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for Rushy Marsh. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations were not attainable with 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the systems watershed. The limited circulation within the system prevents the threshold goals from be achieved. In order to meet the threshold concentrations in the system, alternative approaches beyond load reductions are required to increase circulation and water exchange with Nantucket Sound. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Rushy Marsh system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.			
sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Rushy Marsh	0.353	0.000	-100.0%

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. Removal of 100% of the septic load from the watershed of Rushy Marsh results in an 79% reduction in total nitrogen load. Table VIII-4 shows the breakdown of threshold sub-embayment loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent 'worst-case' summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Nantucket Sound.

Table VIII-3. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Rushy Marsh system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Rushy Marsh	0.447	0.093	-79.1%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Rushy Marsh system, with total watershed N loads, atmospheric N loads, and benthic flux			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Rushy Marsh	0.093	0.203	-0.113

Comparison of model results between existing loading conditions and the selected loading scenario attempting to achieve the target TN concentrations at the sentinel station is shown in

Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station, a different structural approach to increase circulation in the system is required for Rushy Marsh.

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled potential threshold scenario, with percent change, for the Rushy Marsh system. This threshold scenario (100% of septic, Table VIII-2) failed to meet the infaunal threshold (<0.5 mg N/L) at the sentinel station (bold print)

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Rushy Marsh - north	RM1	1.102	0.988	-10.4%
Rushy Marsh - east	**RM2**	**1.107**	**0.990**	**-10.6%**
Rushy Marsh - west	RM3	1.108	0.991	-10.6%
Rushy Marsh - south	RM4	1.156	1.018	-11.9%

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

IX.1 FLUSHING IMPROVEMENTS TO RUSHY MARSH BY RECONSTRUCTION OF THE INLET

Water quality improvements may be possible by improving tidal exchange in an estuary. It is clear from the nitrogen loading (Chapter IV), tidal flushing (Chapter V) and habitat assessment (Chapter VII) that Rushy Marsh Pond supports only significantly impaired to severely degraded sub-tidal habitats as a result of highly restricted tidal exchange with Nantucket Sound waters. In addition, the brackish nature of the Pond waters, the minimal tidal range and prolonged elevation of water levels in previous years during periods of inlet blockage have resulted in the virtual absence of salt marsh within this estuarine system. It is clear from the MEP analysis that Rushy Marsh could benefit from flushing improvements.

At present, tidal attenuation is through the existing culvert is very high, resulting in an average tide range within Rushy Marsh Pond less than 1% that of the offshore range. Attenuation in this system is primarily caused by an undersized inlet and sedimentation. In contrast, for the adjacent estuaries, Three Bays and Popponesset Bay, tide attenuation is near zero, respectively, compared to the range offshore in Nantucket Sound.

Historically, Rushy Marsh was connected to Nantucket Sound through a jettied inlet. The inlet was located at the southern end of the marsh situated between the existing stone groins. Once the inlet closed, rather than reopening the inlet, a culvert was constructed to connect the marsh with the sound. Reconstruction of an inlet and channel provide the most appropriate approach for improving the tidal flushing in Rushy Marsh to achieve habitat restoration. The tidal restriction is the predominant source of the nutrient related habitat degradation in Rushy Marsh Pond. Removal of all of the watershed nitrogen load, would still result in an estuarine system that is significantly impaired. The reason stems from the relatively low present watershed load and the very restricted tidal flushing, which allows the build-up of eutrophying constituents within the pond. A more detailed analysis of inlet stability, maintenance requirements, and potential environmental impacts is required to fully assess inlet reconstruction. To quantitatively assess inlet improvements, two model simulations were executed to simulate Rushy Marsh hydrodynamics with a new 4-ft wide inlet and a new 10-ft wide inlet, each located at the southern end of the marsh.

Hydrodynamic model results for existing and improved inlet conditions are presented in Figure IX-1. In the top plot, tide attenuation is apparent by the lack of a tidal signature in the water surface elevations in the marsh. The middle plot shows a clear tidal signature in the marsh, tidal attenuation is still present with higher elevation of the low tides, and also by the time delay of the tide signal inside the marsh. In the bottom plot of this figure, tidal attenuation is reduced for the proposed 10 ft-wide inlet.

Based on model output, the average tidal prism increases by 6 fold with the improved 4-foot inlet and 20 fold with the 10-foot inlet. Average volumes of Rushy Marsh Pond for existing conditions and for a 20ft-wide inlet scenario are presented in Table IX-1. As a result of the increased tidal prism volume and the reduced mean tide volume of the system, the computed system residence time decreases from 47.6 days for existing conditions, to 4.9 days and 1.6 days for the 4-foot and 10-foot inlets, respectively. The restoration of tidal conditions should also encourage the re-establishment of fringing salt marsh habitat, an important additional benefit of estuarine restoration in this system.

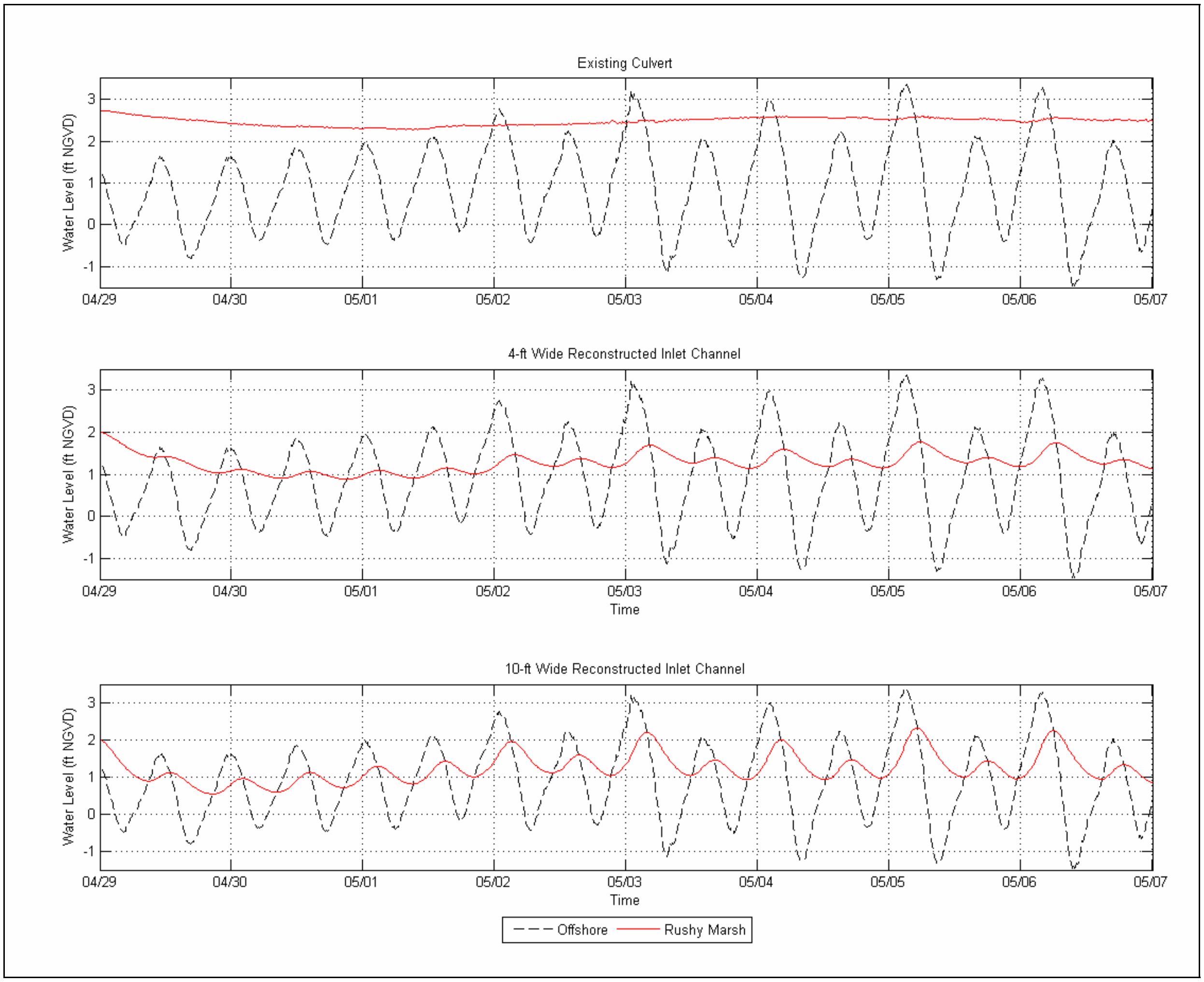


Figure IX-1. Plots showing a comparison of typical tides for modeled existing conditions (top plot), proposed reconstructed 4 ft-wide inlet (middle plot), and proposed reconstructed 10 ft-wide inlet (bottom plot) to Rushy Marsh.

Table IX-1. Average mean tide volumes, mean tide prism, and residence times for Rushy Marsh, for existing inlet conditions, and for the proposed inlet configurations.

	existing inlet	4 ft-wide inlet	10 ft-wide inlet
Mean Volume (ft^3)	2,848,800	1,950,600	1,882,400
Mean Prism Volume (ft^3)	30,936	204,625	629,120
Residence Time (days)	47.65	4.93	1.55

Water quality model runs were performed using the hydrodynamic model output of the proposed reconstructed 4-foot and 10-foot wide inlets. First, present loading conditions were modeled with the reconstructed inlets. Results from the existing loading conditions with the improved hydrodynamics of the reconstructed inlets are presented in Tables IX-2 and IX-3, and plotted in Figures IX-2 and IX-3. The TN concentrations are significantly reduced with the new inlets (i.e., up to an 62% reduction in the northern portion of the marsh), the reduction is large enough to meet the threshold limits set for Rushy Marsh (TN of 0.50 mg/L at water quality monitoring station RM2). Potential environmental and regulatory implications exist for reconfiguration of the inlet; therefore, a complete analysis of the costs, benefits, and impacts of this strategy would be required prior to further consideration of this option. From an engineering cost perspective alone, it likely is cheaper to modify the inlet than to sewer a large portion of the upper watershed, especially as sewering alone will not achieve the nitrogen threshold levels in Rushy Marsh Pond. In contrast, given the low watershed nitrogen load, reconstruction of the tidal inlet alone, will achieve the nitrogen threshold level and restoration of this system. In addition, restoration of tidal exchange (i.e. tide range) will allow the restoration of fringing salt marsh in this system, which has lost its salt water wetlands.

Table IX-2. Comparison of model average total N concentrations from present loading and the reconstructed 4 ft-wide inlet scenario with present loading, with percent change.

Sub-Embayment	monitoring station	present (mg/L)	Channel mod, present (mg/L)	% change
Rushy Marsh - north	RM1	1.102	0.417	-62.2%
Rushy Marsh - east	**RM2**	**1.107**	**0.414**	**-62.6%**
Rushy Marsh - west	RM3	1.108	0.414	-62.6%
Rushy Marsh - south	RM4	1.156	0.374	-67.7%

Table IX-3. Comparison of model average total N concentrations from present loading and the reconstructed 10 ft-wide inlet scenario with present loading, with percent change.

Sub-Embayment	monitoring station	present (mg/L)	Channel mod, present (mg/L)	% change
Rushy Marsh - north	RM1	1.102	0.336	-69.5%
Rushy Marsh - east	**RM2**	**1.107**	**0.333**	**-69.9%**
Rushy Marsh - west	RM3	1.108	0.333	-69.9%
Rushy Marsh - south	RM4	1.156	0.315	-72.7%

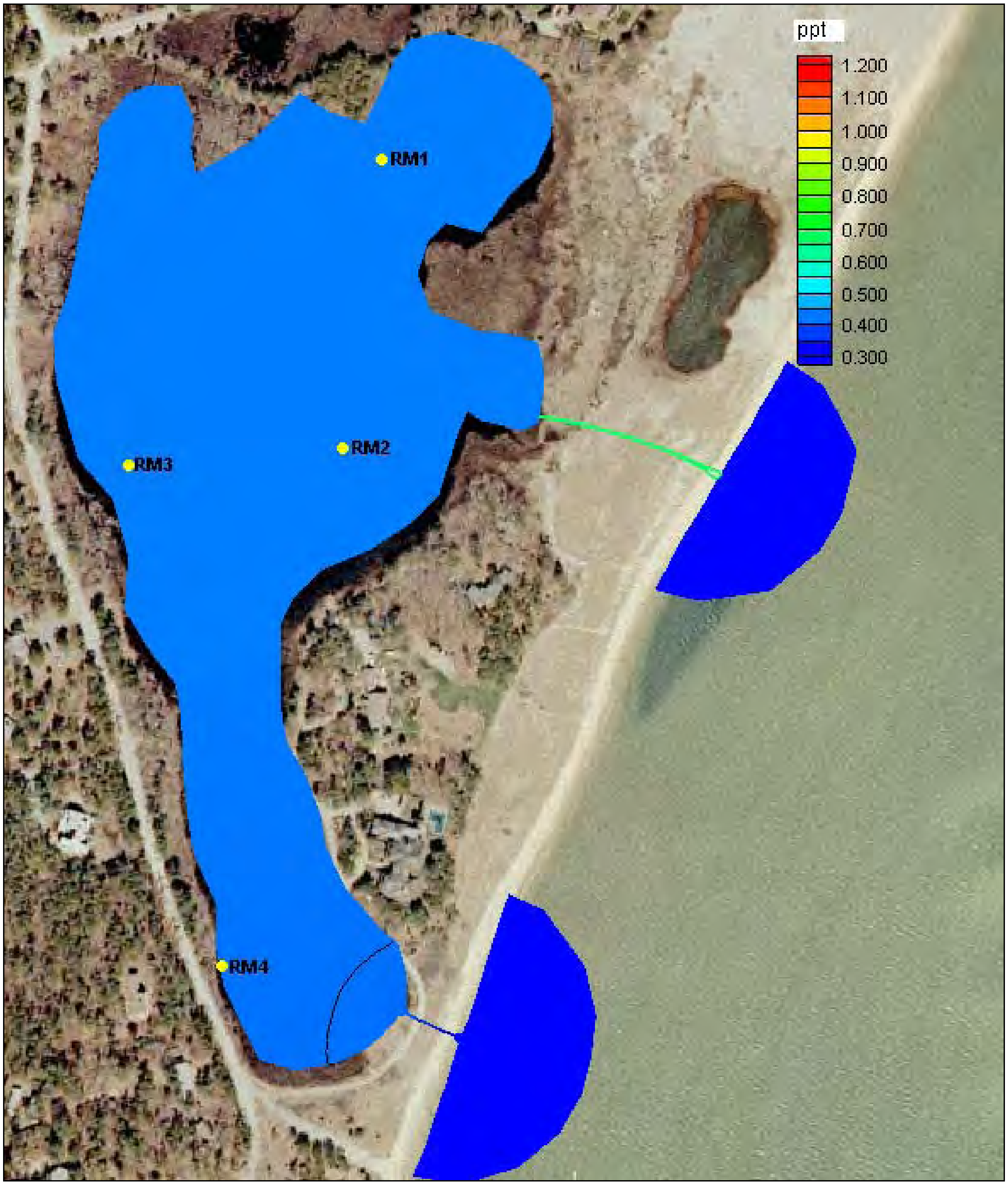


Figure IX-2. Contour Plot of modeled total nitrogen concentrations (mg/L) in Rushy Marsh, for present loading conditions, and reconstructed inlet channel (4 ft), with existing culvert. The approximate location of the sentinel threshold station for Rushy Marsh (RM2) is shown.

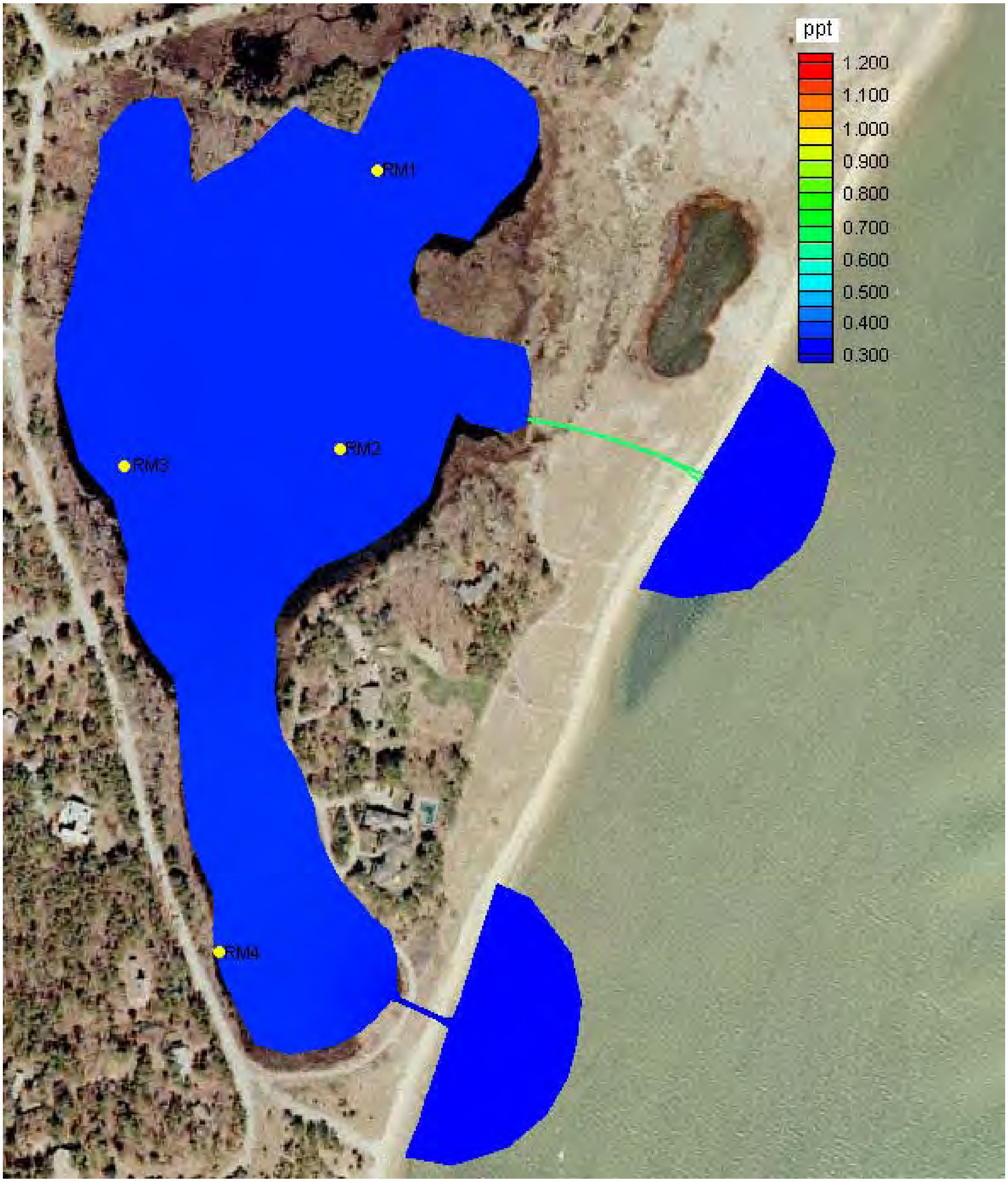


Figure IX-3. Contour Plot of modeled total nitrogen concentrations (mg/L) in Rushy Marsh, for present loading conditions, and reconstructed inlet channel (10 ft), with existing culvert. The approximate location of the sentinel threshold station for Rushy Marsh (RM2) is shown.

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