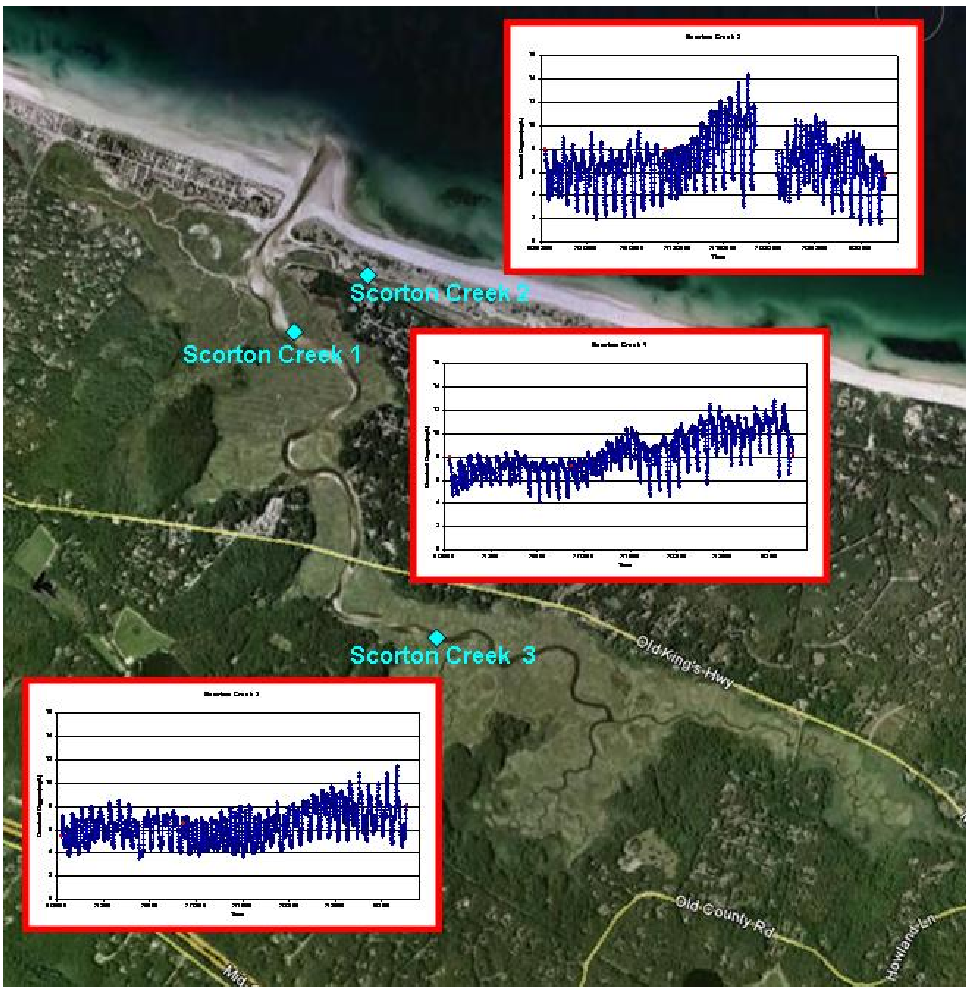
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Threshold for the Scorton Creek Estuarine System Town of Sandwich, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

REVISED FINAL REPORT – MAY 2015

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Massachusetts Estuaries Project

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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 Scorton Creek embayment system, a salt marsh dominated coastal embayment within the Town of Sandwich, Massachusetts. Analyses of the Scorton Creek embayment system was performed to assist the Town of Sandwich with ongoing nitrogen management decisions associated with the current and future wastewater planning efforts of the Town, 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 Sandwich 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 Scorton Creek 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 protection/restoration of the Scorton Creek 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 Scorton Creek embayment system (although dominated by salt marsh) within the Town of Sandwich is at risk of eutrophication (over enrichment) from enhanced nitrogen loads entering through groundwater from the increasingly developed watershed to this coastal system. 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 Sandwich 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 the Town plans to implement upon its completion. The Town of Sandwich has been working with the MEP Technical Team to also complete a nutrient threshold analysis for the Sandwich Harbor system such that nutrient management can be undertaken in a unified manner across the entire town. The Town of Sandwich with associated working groups has 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 Towns across southeastern Massachusetts, Martha's Vineyard and Nantucket Islands. 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 Scorton Creek embayment system by using site-specific data collected by the MEP and water quality data that was collected as a collaboration between the Coastal Systems Program (UMD-School for Marine Science and Technology) and the Town of Sandwich. The Town of Sandwich was able to implement the appropriate water quality monitoring program in order to develop the necessary baseline to join the MEP program through the support of the Commonwealth of Massachusetts 604(b) grant program. Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Town of Sandwich Planning Department, and watershed boundaries delineated by USGS. This land-use data was used to determine watershed nitrogen loads within the Scorton Creek embayment system and the 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 Scorton Creek 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 Cape Cod Bay source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Scorton Creek 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 from the watershed in order to maintain the health of the salt marsh dominated Scorton Creek system. In more open water dominated systems, a reduction is required for restoration of eelgrass and infaunal habitats, however, such is not always the case in systems dominated by salt marshes (e.g. Namskaket Marsh, Little Namskaket Marsh in the Town of Orleans as well as Cockle Cove Creek in the Town of Chatham). The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout an 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 (threshold nitrogen level). 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 adjust nitrogen loads sequentially until the targeted nitrogen concentration is achieved. For the Scorton Creek system, the restoration target should reflect both pre-degradation habitat quality and be reasonably achievable. The presentation in this report of nitrogen loading limits aims to establish the general degree and spatial pattern of loading that will be required for protection of this healthy salt marsh dominated embayment system.

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen loading guidelines for future nitrogen management in the watershed to the Scorton Creek embayment system. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in changes to nitrogen loading (increase or decrease) to the embayment. These scenarios should be developed in coordination with the Town of Sandwich and its consulting engineer such that potential town-wide nitrogen management strategies can be considered in order to effectively examine the effect of load increases/reductions on water column nutrient concentrations in both the Scorton Creek system as well as the adjacent Sandwich Harbor system.

It is important to note that contrary to most other estuarine systems evaluated as part MEP, the threshold concentration for Scorton Creek (similar to the adjacent Sandwich Harbor Marsh) was set higher than present conditions, meaning that the system would be allowed to have a higher load than present while still being able to meet the threshold. Therefore, watershed nitrogen loads were sequentially raised in the model until the nitrogen levels reached the threshold level at the sentinel station (SC-8) chosen for Scorton Creek. It is important to note that load increases could be produced by increasing of any or all sources of nitrogen to the system. The load increases presented in this report represent only one of a suite of potential approaches that need to be evaluated by the community. The current presentation is to establish the general degree and spatial pattern of loading that will be allowable for this system.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout the Scorton Creek system based upon available water quality monitoring data, historical changes in eelgrass distribution (as appropriate), time-series water column oxygen measurements, and benthic community structure. The Scorton Creek estuarine system is showing high habitat quality throughout its salt marsh reach. The upper reach appears to be a fully functional tidal salt marsh with deeply incised narrow creeks surrounded by extensive emergent marsh. This reach is typical of New England salt marshes, with smaller tidal creeks and a marsh plain dominated by low marsh and high marsh plant communities with patches of fringing brackish marsh vegetation. The lower reach of the marsh supports a large wetland area to the west along with larger tidal creeks. The lower portion of the system is also heavily influenced by sand transport via nearshore coastal processes associated with adjacent Cape Cod Bay. Plant communities in the lower reach are similar to the upper reach except that there is less fringing brackish water species and the marsh grades to barrier beach/dune vegetation near the tidal inlet. All of the key habitat indicators are consistent within Scorton Creek, and particularly its tidal creeks, supporting high quality habitat in line with the system's salt marsh structure and function (Chapter VII).

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 each of the sub-embayment systems in this study were developed to restore or maintain SA waters or high habitat quality. In these systems, high habitat quality was defined as supportive of diverse benthic benthos animal communities. Dissolved oxygen and chlorophyll-*a* were also considered in the assessment.

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Scorton Creek embayment system were comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 78% of the controllable watershed nitrogen load to the embayment was from wastewater.

As a salt marsh dominated estuary, Scorton Creek, like nearby Namskaket Marsh in the Town of Orleans, does not support eelgrass habitat. As a result, threshold development for protection/restoration of this system focuses on infaunal habitat quality. The primary mechanism for infaunal habitat quality decline in salt marsh creeks of this type is through stimulation of macroalgal production and accumulation.

Determination of the critical nitrogen threshold for maintaining high quality habitat within the Scorton Creek estuarine system is based primarily upon: 1) the systems structure and function as a salt marsh, 2) macroalgal distribution, 3) current benthic community indicators and 4) nitrogen levels. Given the database it is possible to develop a site-specific threshold, which is a refinement upon general threshold analysis frequently employed.

The Scorton Creek system is presently supportive of high quality salt marsh infaunal habitat throughout its tidal reach. While there is periodic summertime oxygen depletion of creek waters, the levels are consistent with unimpaired New England salt marsh systems. At present, significant macroalgal accumulations do not occur within this macro-tidal estuary at tidally averaged total nitrogen levels of 0.677 mg N L-1 (headwaters) to 0.310 mg N L-1 (tidal inlet).

The threshold nitrogen levels for the Scorton Creek embayment system in the Town of Sandwich were determined as follows:

Scorton Creek Threshold Nitrogen Concentrations

- Scorton Creek salt marsh is presently below the level of nitrogen loading that would cause impairment to its infaunal habitats (i.e. below its nitrogen threshold level), therefore, a conservative estimate of the threshold was established. The threshold was based upon site-specific data and comparison to other similar systems on Cape Cod where detailed nitrogen threshold studies have been completed (e.g. Namskaket Marsh, Little Namskaket Marsh, Cockle Cove Creek). The inter-estuarine comparison focused upon similar salt marshes which are presently experiencing higher nitrogen levels, with and without impairment.

- A principal component of the high tolerance of salt marsh systems to nitrogen inputs from groundwater and surface water inflows is that unlike embayments, creek waters cannot accumulate nutrients over multiple tidal cycles as embayments do. In addition, increasing the nitrogen concentration in the tidal waters that flood the marsh plain will have a negligible or possibly a stimulatory effect on marsh primary and likely secondary production (i.e. an enhancement of habitat). In addition, since the inflowing fresh waters flow down gradient through the marsh creek and out to the adjacent offshore waters, the nitrogen level will never exceed the inflowing freshwater nitrogen level.

- A detailed nitrogen threshold analysis of Cockle Cove Creek (Chatham), a similarly configured salt marsh to Scorton Creek and Namskaket Creek, has recently been completed (SMAST 2006). In addition to having similar structures, Cockle Cove Creek and Namskaket Creek both support similar benthic communities, macroalgal accumulations are sparse to absent in both systems and tidal velocities within the central creek are similar. In addition, the infaunal habitats within Namskaket and Cockle Cove Marsh are similar in composition and diversity (dominated by polychaetes and crustaceans, with some mollusks). The dominant species (*Leptocheirus, Paranais*) was also observed in a study of a healthy salt marsh, Great Sippewisset Marsh on Cape Cod.

- Putting all the assessment elements together, it appears that for Scorton, the critical values are a total nitrogen level of 2 mg N L^{-1} in the headwaters and a level of 1 mg N L^{-1} in the mid upper reach of the main channel (Station SC-8). This station is associated with the upper marsh infauna habitat and is not significantly diluted by direct freshwater inflows (like stations SC-3 or SC-7) and therefore (SC-8) was selected as the sentinel station for this system. The threshold (tidally averaged) total nitrogen level of 1 mg N L^{-1} was determined to be appropriate for the sentinel station (SC-8). It should be noted that the tidally averaged total nitrogen level at the middle marsh station in Cockle Cove Creek is currently 1.378 mg N L^{-1} and the tidal inlet station shows concentrations of 0.472 mg N L^{-1}, consistent with 1 mg N L^{-1} at the sentinel station in Scorton Creek. This threshold applies as long as the tidal creek maintains its present hydrodynamic characteristics (flushing and velocity). The nitrogen threshold for Scorton Creek is intentionally conservative based upon all available data from comparable systems. While not intuitive, the threshold in Scorton Creek is more restrictive of nitrogen loading than for Namskaket Marsh, since the station if further upgradient in the system than in Namskaket Marsh.

- As presented in Chapter VIII, the threshold set for this system would allow up to a 96% increase in the present nutrient concentration at the sentinel station (SC-8) with an associated increase in septic system related watershed loading of 390% for Scorton Creek as a whole. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

The overarching conclusion of the MEP analysis of the Scorton Creek estuarine system is that protection of this currently healthy salt marsh system will allow for increased nitrogen loading from a variety of watershed sources, however, limits to nitrogen loading have been determined as detailed further in the report. This requires careful long term monitoring of conditions in the marsh system and watershed based management of present and future nitrogen inputs such that nitrogen concentration thresholds specified as supportive of health marsh habitat are not exceeded in the future.

It is recommended that if significant new information is obtained by the Town or the Town's consultant regarding matters pertaining to the content of this threshold report, the MEP Technical Team be contacted to assess the need for re-running the system specific linked models. This is particularly applicable to any major land use changes that occur in the watershed resulting in significant increases in nutrient loading to the estuary from within the watershed or imported from areas external to the watershed. This also applies to any new quantitative information related to beach dynamics (longshore transport) or sea level rise.

Table ES-1. Existing total and sub-embayment nitrogen loads to the estuarine waters of the Scorton Creek estuary system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations.

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)
SYSTEMS										
Scorton Creek	3.885	8.690	23.532	0.181	32.403	0.395	-0.065	32.733	0.29-1.21	1.0
Surface Water Sources										
Long Hill Creek	0.241	1.029	3.468	0.088	4.584	-	-	4.584	--	--
Jones Lane	0.162	0.704	2.049	--	2.753	-	-	2.753	--	--
System Total	**4.288**	**10.423**	**29.049**	**0.269**	**39.740**	**0.395**	**-0.065**	**40.070**	**0.29-1.21**	**1.0**

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 2005 – 2007 data, ranges show the upper to lower regions (highest-lowest) of an sub-embayment. Individual yearly means and standard deviations in Table VI-1.
8 Threshold for sentinel site located in Scorton Creek is located along the main channel at Station SC-8.

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Scorton Creek estuary system, Town of Sandwich, 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
SYSTEMS						
Scorton Creek	32.403	124.244	0.395	0.614	125.253	+73.9%
Surface Water Sources						
Long Hill Creek	4.584	23.537	-	-	23.357	+80.5%
Jones Lane	2.753	10.290	-	-	10.290	+73.2%
System Total	**39.740**	**158.071**	**0.395**	**0.614**	**158.900**	**74.9%**

(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.

ADDENDUM

to

Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Scorton Creek Embayment System, Sandwich, Massachusetts

At the request of the Town of Sandwich, select responses to comments are provided for the reader to clarify specific aspects of the MEP nutrient threshold analysis for Scorton Creek.

MassDEP / Town of Sandwich Comments (MEP Technical Team Response in bold italics):

1) The report should comment on whether the mix of *Spartina* and *Phragmites* would significantly or meaningfully change based on the increased freshwater discharge associated with the projected build-out with no change in wastewater management approach (i.e. septic systems). MEP should also comment on the same question with build-out being addressed by centralized collection, treatment and disposal at a flow rate of 1.3-mgd at 8-mg/l effluent total nitrogen.

   Changes in the freshwater regime of salt marshes can, in some circumstances, affect the distribution and types of marsh plants. However, small changes such as projected by buildout will typically be offset by ongoing local sea level increases. In addition, increased freshwater will only occur if buildout results in net import of potable water. Furthermore, most of any increase in net imported water will tend to increase stream flows which will not influence the Spartina/Phragmites mix. As a result any shift in the Spartina/Phragmites mix will likely be small, but to confirm this requires site specific information not available at this time. Note that the MEP Technical Team focuses on nitrogen effects. Also the reviewer is requesting the Technical Team to make a prediction with no real data, and the MEP does not do this.

2) The report should address the following items:

   a. Impacts of inlet migration, expansion or contraction on the reported results.
   b. Impacts of climate change and sea level rise on the reported results.
   c. Areas of relative certainty and areas of certainty in the modeling effort

The requested items have never been part of the MEP analysis and hence not included within the report. Impacts of inlet migration or contraction can be assessed as scenarios and the impacts of climate change and sea level rise on the reported results would be an entire study in and of itself. However, it should be noted that local sea level rise has not been accelerating significantly over the past 30 years. For an analysis of the modeling effort one might review the report of the Barnstable County Review of the MEP.

3) Discussion of sea level rise (even if to state that it has not been accelerating significantly) is appropriate to address recent questions.

As previously mentioned, the impacts of climate change and sea level rise on the reported results would be an entire study in and of itself as it would require extensive research on how climate change may affect precipitation, recharge, groundwater levels, watershed delineations, N-loads from the watershed, N-loads from atmospheric deposition etc. Moreover, it would be necessary to do extensive research on how sea level rise would affect the geomorphology of the overall marsh system as well as the inlet, its migration over time and the consequence on the efficiency of flushing and residence time etc. We re-iterate that this is well beyond the scope of the MEP as originally conceived but could be researched exhaustively as a separately funded project.

4) p. 25, last para.: Did USGS modeling account for the Long Hill Creek discharge which bypasses the estuary? Furthermore, USGS maintains that it does not have a record of delineating specific Long Hill Creek subwatersheds. The data disks seem to reference these subwatersheds as Quaker Meetinghouse. Was there a name change? Please reconcile or explain.

The regional USGS model generally does not have the benefit of the MEP streamflow information. Based on the measured flow information, the presence of the weir, and the size of the watershed, some portion of the watershed flow was assumed to discharge out of the watershed through the barrier beach. As explained in the text, this seems reasonable given the regular ponding observed in the bog / pond upstream of the gauge. In other subwatersheds, where water could not flow out through the barrier, the gauged flow and modeled flows agreed, adding support to the outflow through the barrier beach argument. Further analysis necessary to further evaluate the flow in this area is beyond the scope of the MEP, although it could be conducted if funds were available. The MEP data disk correctly labels these subwatersheds as Long Hill Creek.

ACKNOWLEDGMENTS

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 critical coastal resources of Sandwich and drove for the completion of the Linked Watershed-Embayment Model to Determine the Critical Nitrogen Loading Threshold for the Scorton Creek Embayment System. Without these stewards and their efforts, this project would not have been possible.

First and foremost we would like to recognize and applaud the commitment shown by the Town of Sandwich in carrying forward with the Massachusetts Estuaries Project and the protection/restoration of all the estuaries of the Town. Significant time and attention has been dedicated to this effort by Mr. David Mason, whose efforts where instrumental to completion of these reports. Mr. Mason, Town of Sandwich Health Agent, provided animal husbandry information and was critical to establishing the Town of Sandwich Estuarine Water Quality Monitoring Program and dedicated much personal time and effort towards organizing and participating in sample collection to determine the nutrient related water quality for both of the Town's estuaries (Sandwich Harbor and Scorton Creek). Without this baseline water quality data the present analysis would not have been possible.

Critical support was also provided by the Sandwich Water District, Superintendent Dan Mahoney, who provided the water use data and site specific fertilizer usage data provided by Dave Polidor, Golf Course Superintendent of Sandwich Hollows Golf Club, Richard Dalrymple of Riverview School, Alan Hall, Director of Facilities Sandwich Schools. We would also like to acknowledge the field support provided to the MEP by the staff of Riverview School who gave us unrestricted use of their facility to complete critical field tasks.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by Paul Niedzwiecki and Tom Cambareri of the Cape Cod Commission. We are also thankful for the long hours in the field and laboratory spent by technical staff (Jen Benson, Michael Bartlett, Sara Sampieri and Dahlia Medieros), interns and students within the Coastal Systems Program at SMAST-UMD.

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I. INTRODUCTION

Scorton Creek is an 43.4 hectare (107.3 acre) salt marsh on the northwestern coast of Cape Cod (Figure I-1), with an adjacent similar salt marsh system, Sandwich Harbor (also within the Town of Sandwich). The Scorton Creek Estuary is a tidal salt marsh and is located within the Town of Sandwich on Cape Cod Massachusetts. This estuary is among the larger salt marshes on Cape Cod and exchanges tidal water with Cape Cod Bay (Figure I-2). Scorton Creek Marsh does have a small, deeper channel immediately inside its inlet to Cape Cod Bay. However, in both Scorton Creek and Sandwich Harbor, these deeper water regions do not retain significant water volume at low tide. Scorton Creek is predominantly tidal salt marsh with a central tidal creek, smaller tributary tidal creeks and extensive emergent vegetated areas. Scorton Creek is a classic New England Salt Marsh like Sandwich Harbor and Great Sippewissett Marsh (Falmouth). The watershed contributing nitrogen to the waters of Scorton Creek is distributed between the Towns of Sandwich and Barnstable making protection or restoration of the system complex as restoration of any degraded habitats within the estuarine salt marsh system will depend upon coordinated efforts of two municipalities and their citizens.

The Scorton Creek Estuary is functioning as a typical New England salt marsh dominated by a central tidal creek and emergent marsh colonized by low marsh flora (*Spartina alterniflora*) and high marsh flora (*Spartina patens, Distichlis spicata*) with some more brackish marsh plants found in the upper most regions and limited bordering patches of *Phragmites*. Tidal exchange with the high quality waters of Cape Cod Bay is high, given the ca. 10 foot (3 m) tide range, which has also resulted in tidal creeks which are deeply incised, with near complete drainage at low tide. The result is the type of coastal system which has a relatively high tolerance for nitrogen inputs. Observations by the MEP Technical Team indicate a healthy functioning New England pocket salt marsh. Scorton Creek Marsh provides both wildlife and shellfish habitat and serves as a nursery area for offshore fisheries and as important habitat for commercially important migratory fish, as well as serving as a storm buffer and nutrient sink for watershed derived nitrogen.

Scorton Creek Marsh is a moderately sized estuary behind a barrier beach formed by coastal processes associated with nearshore Cape Cod Bay. Salt marshes require a relatively quiescent depositional environment away from the force of breaking ocean waves, where sandy and muddy sediments can accumulate and become vegetated. With rising sea level salt marshes, such as Scorton Creek, tend to grow seaward, filling the shallow basin formed between the upland and barrier beach complex. Salt marshes persist as long as vertical accretion of trapped sediments and organic matter raise the marsh surface to keep up with sea level rise.

The Scorton Creek Estuary is a relatively “young” coastal feature that required significant post glaciation sea-level rise and the formation of the barrier beach, occurring on the order of 2500-4000 years b.p. Similar to other salt marsh dominated systems on the northern coast of Cape Cod (e.g. Namskaket Marsh and Little Namskaket Marsh, Barnstable Great Marshes), Scorton Creek Marsh is a shallow estuary dominated by emergent salt marsh, tidal creeks and tidal flats, as well as being located within a watershed that includes glacial outwash plain and ice contact deposits (Figure I-3). These subsurface formations consist of material deposited after the retreat of the Laurentide Ice sheet ~15,000 years ago. These deposits, which form the present aquifer soils, are highly permeable and vary in composition from well sorted medium sands to course pebble sands and gravels (Oldale, 1992). As such, direct rainwater run-off is typically rather low and most freshwater inflow to these estuarine systems is via groundwater discharge or groundwater fed surface water flow.

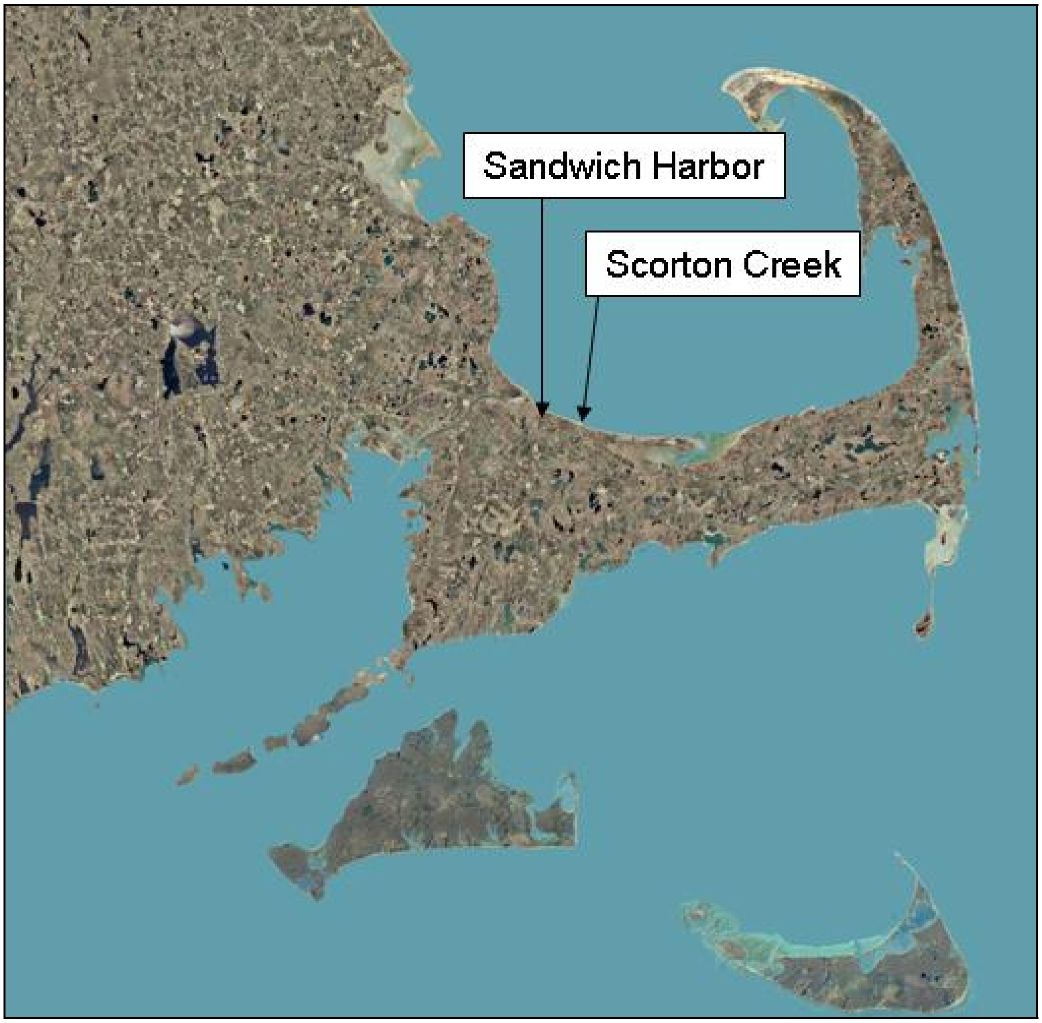


Figure I-1. General location of the Sandwich Harbor and Scorton Creek salt marshes in the Town of Sandwich, MA. on the northwestern shore of Cape Cod, exchanging tidal waters with Cape Cod Bay.

Scorton Creek Marsh acts as a mixing zone for terrestrial freshwater inflow and saline tidal flow from Cape Cod Bay. Salinity levels vary with the volume of freshwater inflow as well as the effectiveness of tidal exchange. Given the large tidal flows and volumetric exchange, there is presently only minor dilution of salinity throughout most of the estuary at high tide (31 ppt versus 31-24 ppt), except in the uppermost tidal creeks with significant freshwater inflow where the mid-low tide salinity is 8-12 ppt. However, the system is a salt marsh, and as such, the elevation of the tidal creek bottoms is generally higher than the low tide elevation in the adjacent Bay (e.g. the creeks drain nearly completely at low tide). The result is that at low tide, the salinity of the out flowing Scorton Creek water can be fresh to brackish, due to the dominance of the freshwater inflow in the absence of the tidal waters. As a result salinity variations of the creek waters in the upper marsh are very large with the range decreasing toward the tidal inlet. Organisms associated with these creeks have developed strategies for dealing with these large salinity variations.

Figure I-2. Study region for the Massachusetts Estuaries Project nitrogen threshold analysis for the Scorton Creek Estuary. Tidal waters enter the system through a single central tidal inlet from Cape Cod Bay. Freshwaters enter the head of the system from the watershed primarily through 2 surface water discharge points (Long Creek and Scorton Creek-freshwater reach), as well as direct groundwater discharge.

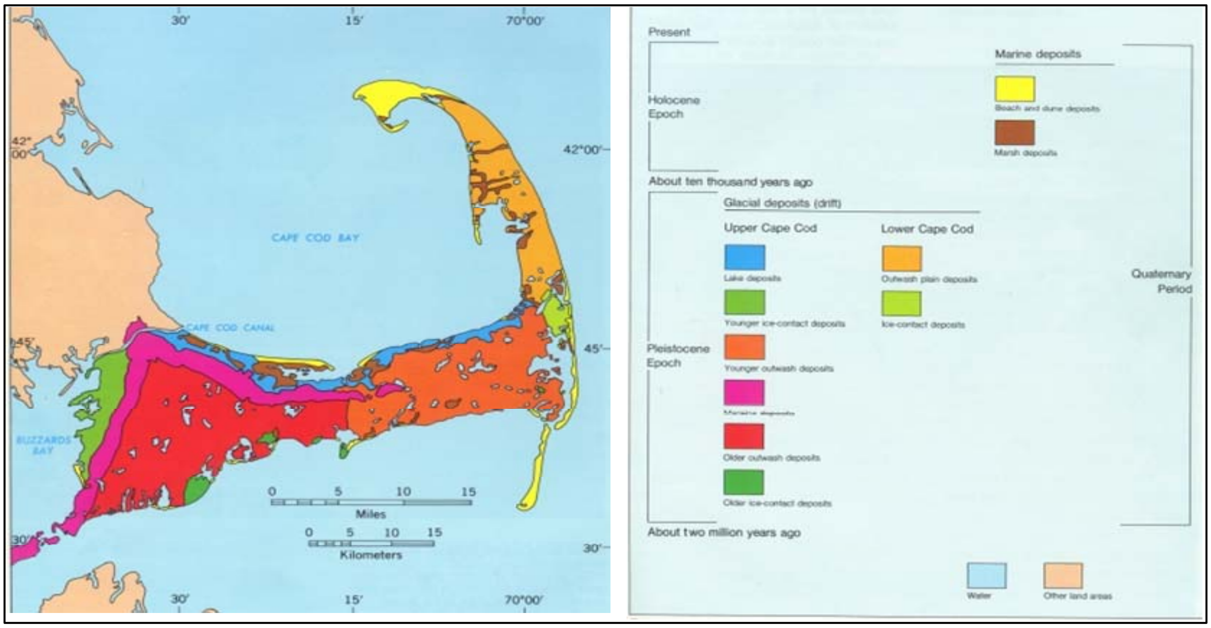


Figure I-3. Geologic map of Cape Cod (generalized from detailed mapping by K. F. Mather, R. P. Goldthwait, L. R. Theismeyer, J. H. Hartshorn, Carl Koteff, and R. N. Oldale). Scorton Creek Estuary can be seen as "marsh deposits" emplaced in post-glacial lake deposits overlain by outwash and bordered to the south by moraine deposits.

At present, “streams” flowing into Scorton Creek are relatively small and discharge only a small portion of the aquifer recharge to the estuary. One surface flow enters the marsh system from the west (passing under Ploughed Neck Road) and the second is the head of Scorton Creek passing under Jones Lane and entering the system from the east. Generally, the streams are not well formed, with almost all freshwater entering through groundwater seepage at the margin with upland areas or directly to tidal tributary creeks through creek bottom seepage.

Tidal exchange with Cape Cod Bay is through an approximately 20 to 50 meter wide inlet (depending on tidal stage) which empties into the nearshore waters off east Sandwich Beach. The inlet was previously armored with groins to the east and west, however, sediment transport has essentially changed the structure of the inlet making the groins only marginally effective for maintaining the main inlet channel. The beach and the inlet are very dynamic geomorphic features, due to the influence of littoral transport processes. These processes may periodically affect the health of this estuary through changes in hydrodynamics wrought by inlet dynamics (see Chapter V). To the extent that the inlet becomes restricted and tidal flushing is reduced, nitrogen loading impacts will be magnified over present conditions. Equally important, to the extent that tide range may become reduced, the health and productivity of the extensive emergent salt marsh would be reduced. Any long term habitat management plan for the Scorton Creek Marsh System must recognize the importance of inlet dynamics and include options to maintain tidal exchange.

A primary ecological threat to Scorton Creek Marsh resources is degradation resulting from nutrient enrichment by increased nitrogen loading and/or reduced tidal exchange. Loading of the critical eutrophying nutrient (nitrogen) has been increasing over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other outer Cape estuaries (such as Namskaket Marsh, Little Namskaket Marsh and

Nauset Harbor in the Town of Orleans), like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater or disposal of treated effluent from municipal treatment facilities. The Town of Sandwich has been among the fastest growing towns in the Commonwealth over the past two decades and does not have centralized wastewater treatment, but relies almost entirely on wastewater treatment and disposal by on-site septic systems. As levels of nitrogen loading to coastal systems continue to increase, concern has grown in towns across Cape Cod over associated nutrient impacts.

Fortunately for the resource protection of Scorton Creek Marsh, as is the case for other marsh systems like Sandwich Harbor Marsh and Namskaket Marsh in Orleans, its function as a tidal salt marsh makes it more tolerant of watershed nitrogen inputs than more open-water coastal embayments, like nearby Three Bays, Town Cove or Pleasant Bay on outer Cape Cod. The greater sensitivity of embayments versus wetlands results from the lower rates of tidal flushing in an embayment, the fact that there is limited to no exposure of the sediments to the atmosphere at low tide (e.g. marsh plain), and the fact that open water systems have evolved under much lower levels of productivity and organic matter loading than salt water wetlands. For example, the organic carbon content of vegetated sediments of a New England salt marsh can frequently reach 20% (w/w), while embayment sediments are generally in the 1%-5% range. Yet another difference between system types is that oxygen depletion in the tidal creeks of pristine wetlands can naturally occur on summer nights, while embayment bottom waters become hypoxic generally as a result of eutrophic conditions.

Some additional insight into the nitrogen response by salt marshes can be garnered from long-term chronic nitrogen addition experiments. These have been conducted at multiple sites along the Atlantic coast and specifically in a nearby New England salt marsh, Great Sippewissett Marsh (West Falmouth, MA). This latter project was started by WHOI scientists in 1970 and has been overseen solely by current SMAST Staff since 1985. These studies reveal that nitrogen additions to low marsh (*Spartina alterniflora*) and high marsh (*Spartina patens, Distichlis spicata*) areas, typically results in increased plant production and biomass and secondary production as well. Nitrogen dynamics have been quantified, which show that as nitrogen is added the initial increased nitrogen available is taken up by the plants, but this plant demand is rapidly satisfied and any additional load is denitrified *in situ* by soil bacteria. In the Great Sippewissett Marsh fertilization experiments, the denitrification capacity of the sediments has not been exhausted in 30 years of N additions and at nitrogen input levels about 7 times the natural background N input (75.6 g N m^{-2} each growing season). Consistent with these observations, studies examining nitrogen loss from the marsh surface find negligible loss of added nitrogen, with only a small loss generally being associated with plant senescence and decay during the fall.

Salt marsh creek bottoms and creek banks (such as those found in Scorton Creek, Sandwich Harbor, Namskaket Marsh, Little Namskaket Marsh) have developed under nutrient and organic matter rich conditions, as have the organisms that they support. It is the creek bottoms rather than the emergent marsh, which are the primary receptors of increased watershed derived nitrogen in Cape Cod salt marshes. Watershed nitrogen predominantly enters these salt marshes through groundwater or small headwater streams, as is the case in the Scorton Creek or Long Hill Creek, with some groundwater entry through creek bottoms. Both surface and groundwater entry focuses on the tidal channels. Even groundwater entry through seepage at the upland interface is channeled to creek bottoms. As the tide ebbs in New England salt marshes (like Scorton Creek and Sandwich Harbor) the freshwater inflow "freshens" the waters and the nitrogen levels in the tidal creeks increase due to the nitrogen entry from the watershed. At low tide the nitrogen levels in the tidal creeks are dominated by watershed inputs.

Since the predominant form of nitrogen entering from the watershed is inorganic nitrate, the effect on the creek bottom is to stimulate denitrification, hence nitrogen removal. For example, in a salt marsh in West Falmouth Harbor, Mashapaquit Creek, ~40% of the entering watershed nitrogen is denitrified by the creek bottom sediments on an annual basis. This stimulation of denitrification does not negatively affect the salt marsh, but does result in a reduction of nitrogen loading to the adjacent nitrogen sensitive coastal waters. However, analysis by MEP Staff of salt marsh areas receiving wastewater discharges indicates that at very high nitrogen loads (inputs relative to tidal flushing), macroalgal accumulations can occur. These accumulations are generally found in the creek bottoms and flats and also may drift and settle on the creek banks. Large macroalgal accumulations in tidal creeks can cause impairment of benthic animal communities. In the latter case, negative effects on creek bank grasses can occur, which may lead to bank erosion and negative effects on organisms. A part of the focus of the present MEP analysis of the Scorton Creek Marsh Estuary relates to potential macroalgal issues.

The Town of Sandwich is the primary stakeholder to the Scorton Creek Marsh Estuary, the focus of this report. The Town of Sandwich was among the first communities to partner with the MEP to determine any existing and potential future degradation of its estuaries. As such, the Town of Sandwich (via the Health Department) undertook a town-wide estuarine monitoring effort, inclusive of Scorton Creek and Sandwich Harbor Marsh. Through funding from the Commonwealth of Massachusetts 604(b) Grant Program, a comprehensive water quality monitoring program was developed to acquire the necessary background water quality monitoring data from the two embayment systems in the Town of Sandwich such that the MEP Linked Watershed-Embayment Management Modeling Approach could be applied to the development of nutrient thresholds. Moreover, water quality data generated by the Town of Sandwich Water Quality Monitoring Program is consistent with that generated by other Towns engaged in the MEP (including the Towns of Barnstable and Orleans which support salt marsh systems very similar to those in Sandwich) making the data for all the systems in this area cross comparable.

The common focus of the water quality monitoring efforts undertaken by the Town of Sandwich has been to gather site-specific data on the current nitrogen related water quality throughout the Scorton Creek and Sandwich Harbor salt marsh systems. These data were then utilized to determine the relationship between observed water quality 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 water quality data set developed by the Town of Sandwich Water Quality Monitoring Program in collaboration with the SMAST-Coastal Systems Program at the University of Massachusetts form a baseline from which to gage long-term changes as watershed nitrogen management moves forward. The Sandwich Water Quality Monitoring Program efforts allowed the MEP to prioritize both of the Sandwich systems for the next step in the restoration/protection and management process. The MEP effort builds upon the efforts of the water quality monitoring program and includes higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Scorton Creek Estuary.

The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning and nitrogen management alternatives development needed by the Town of Sandwich for protection/restoration of estuarine habitats. 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. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Sandwich to develop and evaluate the most cost effective nitrogen management alternatives to protect/restore the valuable coastal resources of Sandwich that have been affected by nitrogen overloading.

Overall, salt marsh which comprises the Scorton Creek Estuary presently appears to be functioning as a productive healthy New England salt marsh as noted by the MEP Technical Team's field survey. Nitrogen levels within the tidal creeks are elevated and were evaluated as to any negative effects on salt marsh resources. A primary ecological concern for this estuary is degradation resulting from nutrient enrichment by nitrogen loading and/or reduced tidal exchange. Loading of the nitrogen has been increasing over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other outer Cape estuaries, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater or disposal of treated effluent from municipal treatment facilities. However, as noted above, salt marshes are relatively tolerant of nitrogen inputs from the surrounding watershed. This results from the structure of the salt marsh within the upland hydrologic system and the natural nitrogen processing by these systems. In addition, the plants and animals within salt marshes have adapted to the high organic matter levels within the marsh sediments and associated waters and the associated biogeochemical effects. Critical to the MEP analysis of this estuary is consideration of the structure and resources within the different components of this salt marsh dominated estuary.

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. At its higher levels, enhanced 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 related to changes in land-use as a watershed becomes 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 environment 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 Sandwich) 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 estuarine basins, 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 (MassDEP), 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 municipalities and MassDEP 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 outlines an implementation plan. That plan must identify, among other things, the required activities to achieve the allowable load to meet the allowable loading target, the time line for those activities to take place, and reasonable assurances that the actions will be taken.

The major Project goals are to:

- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of 70 of the embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment model available through the MEP Technical Team to address future regulatory 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 ca. 55 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 suggests “solutions” for the protection or restoration of nutrient related water quality and allows testing of “what if” management scenarios to support evaluation of resulting water quality impact versus cost (i.e., “biggest ecological bang for the buck”). In addition, once a model is fully functional it can be “kept alive” and corrected 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 fully field verified and unlike many approaches, accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics (Figure I-4). This methodology integrates a variety of field data and models, specifically:

- 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

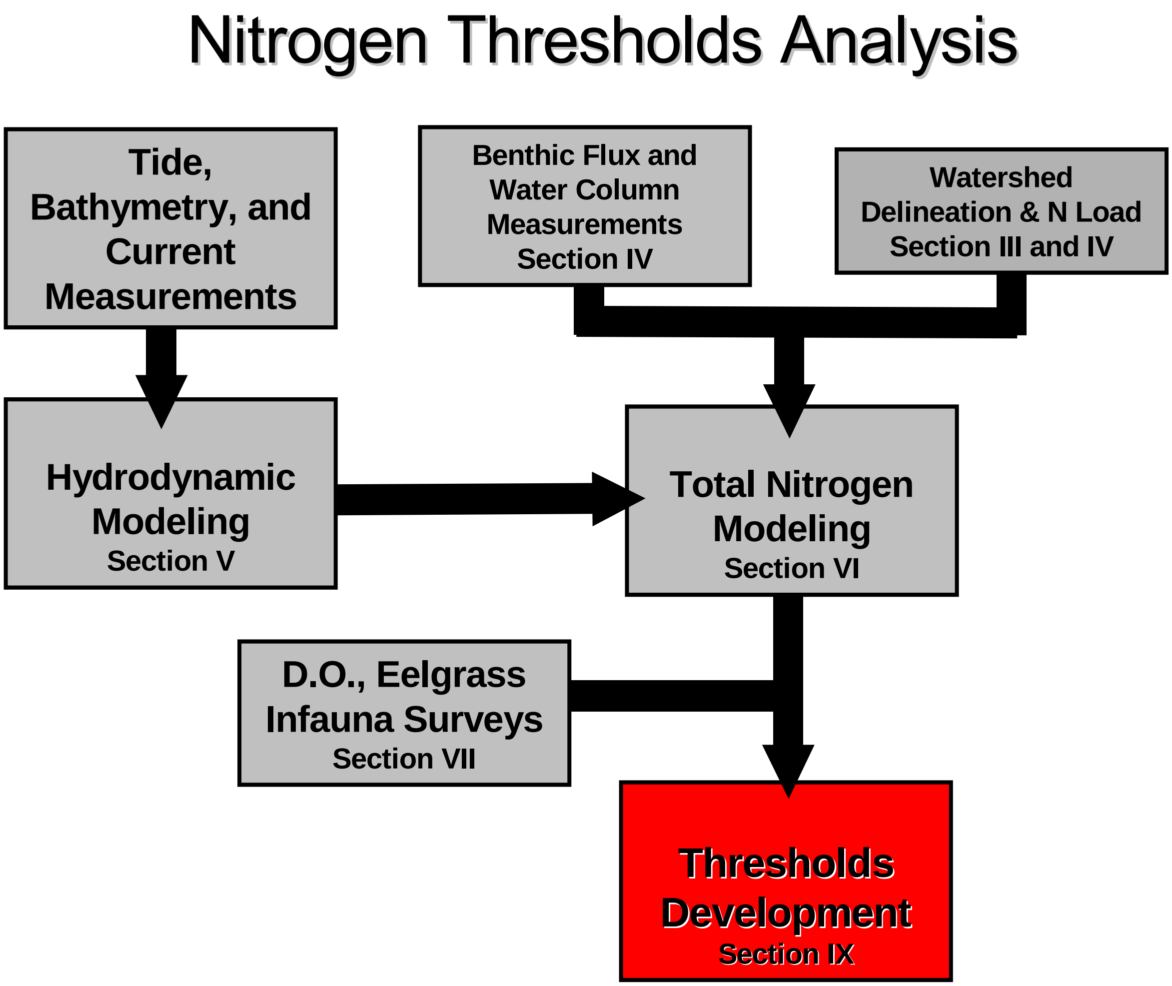


Figure I-4. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach.

I.2 NITROGEN 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 aquifers derived from glacial deposits, such as in the watershed to the Scorton Creek embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (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 1998, 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).

Nutrient related water quality decline represents one of the most serious threats to the ecological health of 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 estuarine 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. As 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, 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 the MEP approach integrates site-specific data on nitrogen levels and the gradient in N concentration throughout the marsh systems monitored by the Town of Sandwich Water Quality Monitoring Program with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) to "tune" general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

I.3 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important "boundary conditions" for water quality modeling of the Town of Sandwich marsh systems; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within the systems. 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 marsh (Scorton Creek) and all of the component tidal tributaries. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the system. Once the hydrodynamic properties of the estuarine systems 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 Sagamore model for sub-watershed areas designated by MEP. Almost all nitrogen entering the marsh systems is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Cape Cod Bay source waters and the salt marsh systems themselves was taken from the water quality monitoring program run by the Town of Sandwich (associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of these marsh systems were used to calibrate and validate the water quality model (under existing loading conditions).

I.4 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Scorton Creek Marsh Estuary in the Town of Sandwich. A review of existing studies related to habitat health or nutrient related water quality is provided in Chapter II with a more detailed review of prior hydrodynamic investigations in Chapter V. The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Chapters III and IV. In addition, nitrogen input parameters to the water quality model are described. Since nitrogen recycling associated with the 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 (Chapter IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Cape Cod Commission and Town Planning Department data and offshore water column nitrogen values were derived from an analysis of monitoring station data on the flooding tide just inside the inlets to the marsh systems (Chapter 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 Chapter VI. Results of hydrodynamic modeling of embayment circulation are discussed in Chapter 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 Chapter 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 each embayment was performed that included a review of existing water quality information and the results of a benthic analysis (Chapter VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Chapter VIII. Additional modeling is occasionally conducted and presented in Chapter XI to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration in a given estuarine system. This latter assessment may also include examining hydrodynamic options for increasing flushing of a system and only represent some of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the marsh system.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient related habitat quality. Effects include excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, organic matter enrichment of waters and sediments with the concomitant increased rates of oxygen consumption and periodic depletion of dissolved oxygen, especially in bottom waters, and the limitation of the growth of desirable species such as eelgrass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic animal communities (e.g. "infauna", animals living in the sediments). This habitat change causes a shift in infaunal communities from high diversity deep burrowing forms (which include economically important species), to low diversity shallow dwelling organisms. This shift alone causes significant degradation of the resource and a loss of productivity to both the local shell fisherman and to the sport-fishery and offshore fin fishery, which are dependent upon these highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. In addition, the diverse avian fauna which feed upon infauna or fish communities are also affected and their numbers and diversity decline. This overall nutrient driven process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and ponds, it is not a necessarily a part of the natural evolution of a system.

In most marine and estuarine systems, such as the Scorton Creek Estuary, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if the level of nitrogen enrichment is controlled, then eutrophication is controlled. As a result, there has been significant effort to develop tools for predicting how modification of watershed nitrogen loads and changes in tidal flushing quantitatively cause changes in the concentrations of water column nitrogen in the receiving estuary. Further development of these approaches 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. 2002).

These tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. In contrast, some approaches can be tailored for each individual estuary of interest, but require large amounts of site-specific information and therefore are not generally applied. The present Massachusetts Estuaries Project (MEP) effort uses one such site-specific approach. The assessment 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 within individual estuaries. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the prediction process for the specific conditions in each of the coastal embayments of southeastern Massachusetts, including the Scorton Creek System. As the MEP approach requires substantial amounts of site-specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality and unique features.

Few studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Scorton Creek Salt Marsh System over the past two decades to help inform the MEP process. Directly supporting the present Massachusetts Estuaries Project effort to develop a nitrogen threshold for Scorton Creek Estuary was the Scorton Creek Nutrient

Related Water Quality Monitoring. This previous work provided quantitative information on water column parameters over multiple summers (including nitrogen) was used in the calibration and verification of the MEP water quality model for this estuary. Available studies are summarized below.

Scorton Creek Nutrient Related Water Quality Monitoring: The MEP analysis requires high quality water quality data in order to complete its assessment and modeling approach. The Town of Sandwich Water Quality Monitoring Program collected data on nutrient related water quality throughout the estuarine systems of the Town of Sandwich (specifically Scorton Creek and Sandwich Harbor). The Town of Sandwich Water Quality Monitoring Program has collected the principal baseline water quality data necessary for ecological management of each of the estuaries of the Town. The Town of Sandwich monitoring program was a citizen-based water quality monitoring program run by the Town of Sandwich Health Department (D. Mason, Town of Sandwich Health Agent and Monitoring Program Coordinator) with technical and analytical assistance from the Coastal Systems Program at SMAST-UMD. During the period the monitoring program was actively collecting water samples, the program had a USEPA and MassDEP approved Quality Assurance Project Plan (QAPP), which was operational over the entire period of 2005-2007 (data period for this MEP analysis).

In order to initiate the needed data collection for the Scorton Creek and Sandwich Harbor Estuarine Systems to support a full evaluation of future nutrient management options, the Town of Sandwich initiated the water quality monitoring program in the summer of 2005. While the initial effort was funded solely by the Town, the Town of Sandwich sought and received DEP 604(b) funding support for collection, processing and analyses of water samples from the overall embayment system. In total, two grants were obtained allowing water quality data collection at the estuarine stations throughout each summer, 2006 and 2007. Samples and field data were collected from a total of 28 marine sample stations (Sandwich harbor + Scorton Creek respectively), with 15 stations associated with the Sandwich Harbor Estuarine System and 13 stations with the Scorton Creek System. During the course of the three year sampling program, water samples were collected from each station during 6 sample rounds from June through mid-September, 2005-2007, in order to target what is typically the period of poorest nutrient related water quality that is the focus of managing these systems. Marine stations were sampled at approximately two-week intervals during the falling tide (targeting the 2 hours before and after mid-ebb) during the early morning hours (6-9 A.M.).

The common focus of the Town of Sandwich Water Quality Monitoring Program effort has been to gather site-specific data on the current nitrogen related water quality throughout Scorton Creek and Sandwich Harbor to support evaluations of observed water quality and habitat health. The Sandwich Water Quality Monitoring Program in the Scorton Creek and Sandwich Harbor Embayment Systems developed a data set that elucidated the long-term water quality of this system (Figure II-1). The monitoring undertaken was a collaborative effort with the Town of Sandwich Health Department (David Mason) coordinating the field effort and chemical assays being completed by the SMAST Coastal Systems Analytical Facility. The Coastal Systems Analytical Facility is located in the School for Marine Science and Technology UMASS-Dartmouth, 706 S. Rodney French Blvd, New Bedford, MA, and the laboratory Points of Contact are Sara Sampieri 508-910-6325 (ssampieri@umassd.edu) or Mike Bartlett (mbartlett@umassd.edu). Use of the SMAST Analytical Facility ensured sufficient sensitivity and accuracy of the analytical protocols and that proper QA/QC procedures were followed to allow incorporation of the data into the MEP analysis. The baseline water quality data were a

Figure II-1. Town of Sandwich Water Quality Monitoring Program for Scorton Creek. Estuarine water quality monitoring stations sampled by Town of Sandwich Staff and Volunteers and analyzed by SMAST staff during summers 2005, 2006, 2007.

prerequisite to entry into the MEP. Implementation of the MEP's Linked Watershed-Embayment Approach necessarily incorporates the quantitative water column nitrogen data (2005-2007) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff.

Since the results of the long term Water Quality Monitoring Program (2005-2007) and the above studies indicate that portions of the Scorton Creek system could be threatened by the combination of land-derived nitrogen inputs and intermittent restriction of tidal exchange, the Town of Sandwich undertook participation in the Massachusetts Estuaries Project to complete ecological assessment and water quality modeling for the development of nutrient thresholds for protection of the Scorton Creek salt marsh system.

Preliminary Survey of the Hydrodynamics and Water Quality of Scorton Creek (W.H.Grp and W.H.O.I.). A preliminary survey of Scorton Creek to assess potential water quality and flushing issues was conducted in the late 1990's. Water quality assays were conducted by scientists at the Woods Hole Oceanographic Institution, now at SMAST. A fully functional water quality model was not developed as part of this effort. Water quality sampling took place 5-6 times over an annual cycle, so there was limited summer sampling. However, the data indicated freshwater inflows at the terminal ends of major creeks and tidal variations in salinity and nitrogen, with lowest salinity and highest nitrogen observed at low tide. Information from this preliminary survey was used to guide development of the extensive, multi-year Town Water Quality Monitoring Program (2005-2007) used to calibrate and verify the MEP water quality model.

Regulatory Assessments of Scorton Creek Resources - In addition to locally generated studies, Scorton Creek is part of the Commonwealth's environmental surveys to support regulatory needs. The Scorton Creek Estuary contains a variety of natural resources of value to the citizens of Sandwich as well as to the Commonwealth. As such, over the years surveys have been conducted to support protection and management of these resources. The MEP gathers the available information on these resources as part of its assessment, and presents some of them here for reference by those providing stewardship for this estuary and some in Chapter 7 to support the nitrogen thresholds analysis. For the Scorton Creek Estuary available agency surveys include:

- Designated Shellfish Growing Area – MassDMF (Figure II-2)
- Shellfish Suitability Areas – MassDMF (Figure II-3)
- Area of Critical Environmental Concern - ACEC (Figure II-4)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-5)
- Mouth of Coastal Rivers – MassDEP Wetlands Program (Figure II-6)

The MEP effort builds upon earlier watershed delineations (Section III) and land-use analyses and water-use data (Section IV.1), historical eelgrass surveys (Section VII) and water quality surveys discussed above and in Sections below. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Scorton Creek Estuary. The MEP has incorporated appropriate and available data from pertinent previous studies and available Town databases to enhance the determination of nitrogen thresholds for the Scorton Creek System and to reduce costs to the Town of Sandwich.

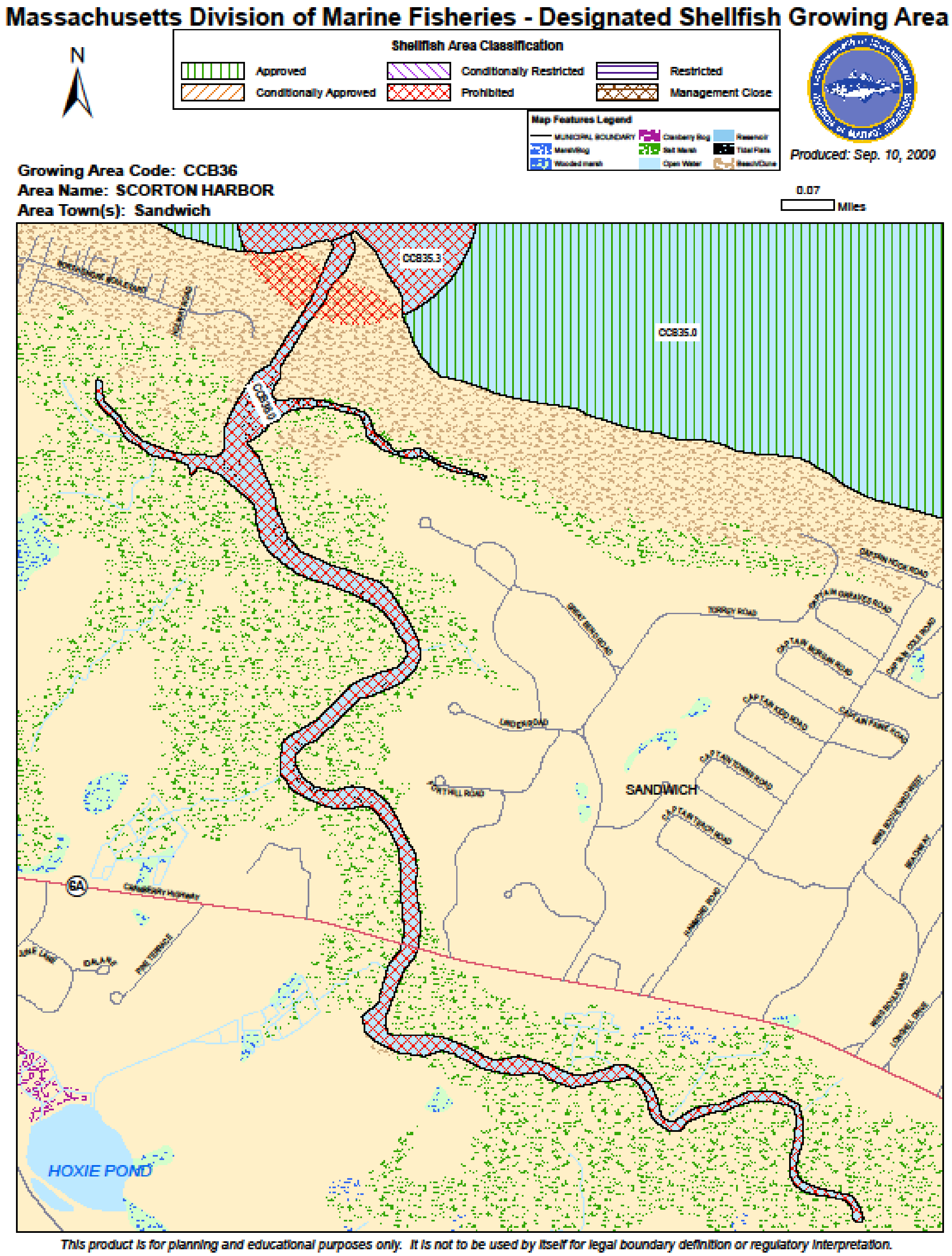


Figure II-2. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas. However, areas dominated by wetlands with persistent fecal coliform levels >14 cfu per 100 mL may be prohibited to shellfishing until the cause of the contamination (frequently wildlife and birds) is documented.

Figure II-3. Location of shellfish suitability areas within the Scorton Creek Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence". The denoted shellfish areas are generally associated with the lower tidal reaches of the major tidal creeks, which support sediments comprised of medium to fine sands.

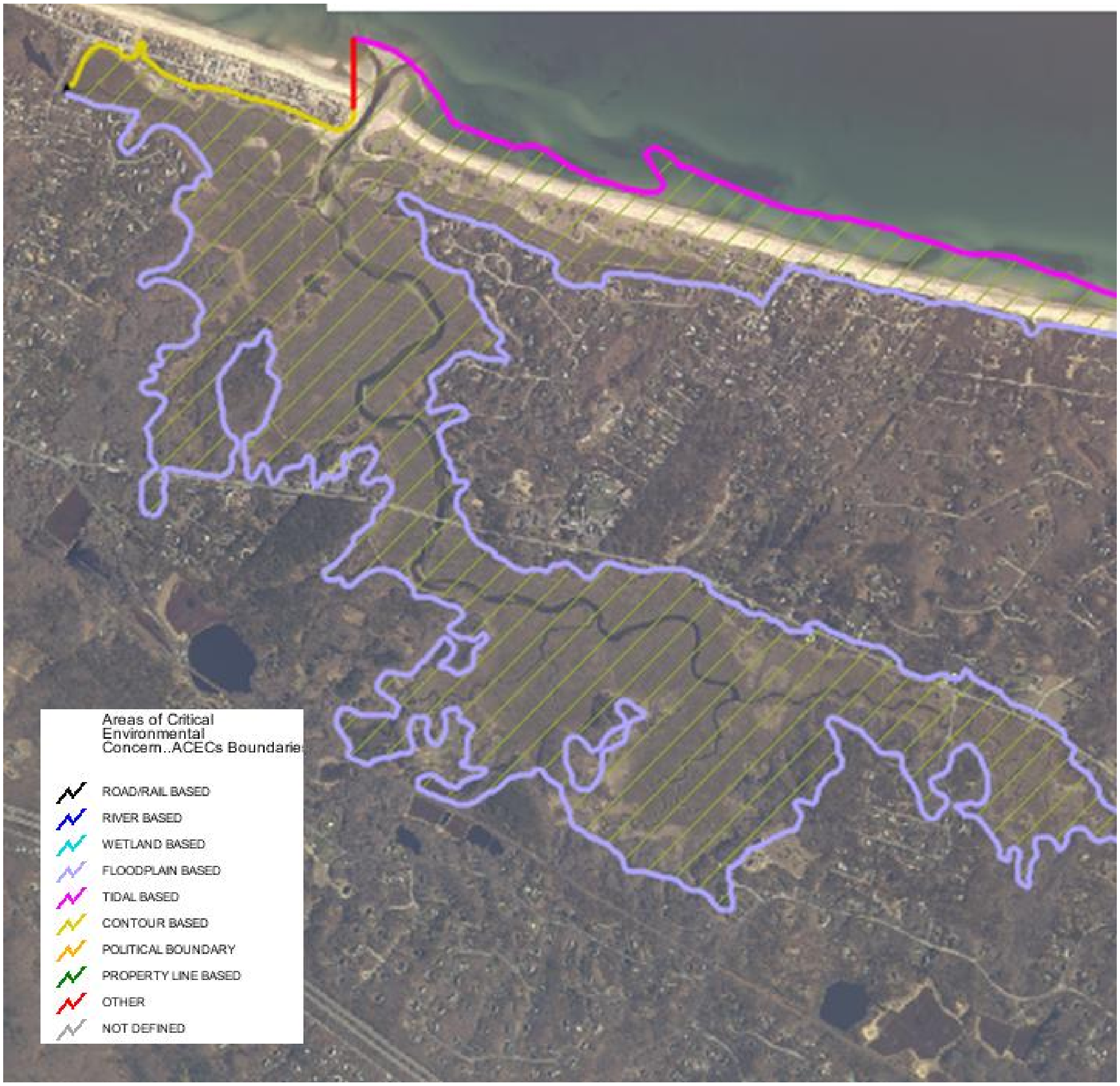


Figure II-4. Areas of Critical Environmental Concern (ACEC) within the Scorton Creek Estuary extending the Howland Lane / Route 6A interconnection to the Barnstable Harbor Great Marshes system.

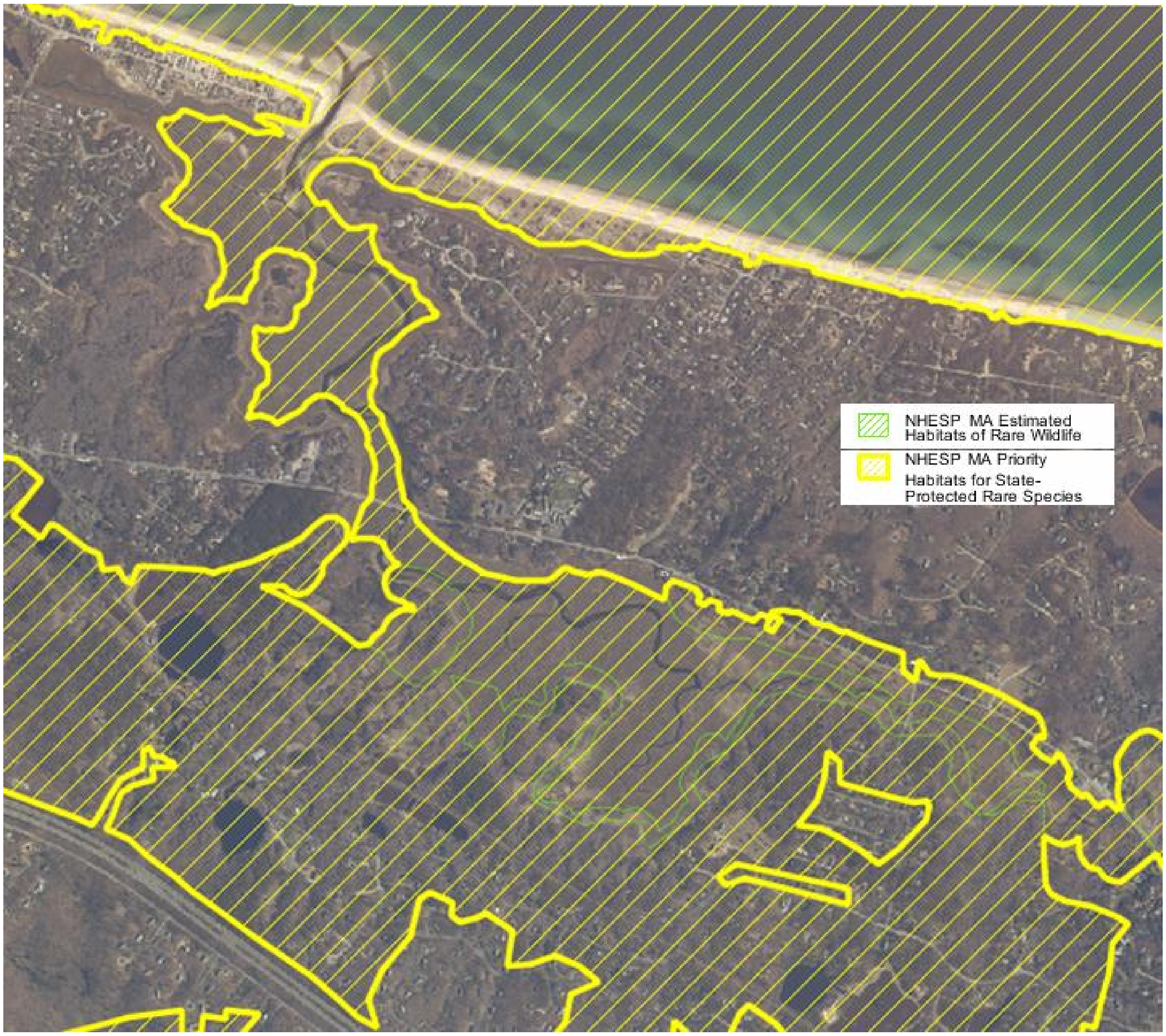


Figure II-5. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Scorton Creek Estuary as determined by - NHESP.

Figure II-6. Mouth of Coastal Rivers designation for Scorton Creek as determined by – MassDEP Wetlands Program. Estuaries on the down gradient reach of coastal rivers and major streams are part of the river system, and as such the mouth of the "river" is at the tidal inlet.

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 the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation. These questions include surface water/groundwater interactions, groundwater travel times, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Scorton Creek estuary. The Scorton Creek watershed is mostly within the Town of Sandwich with a small area along its eastern boundary that is shared with the Town of Barnstable. The Scorton Creek estuary is composed of tidal salt marsh creeks and emergent marsh plain throughout its tidal reaches. The dominant vegetation throughout is *Spartina alterniflora* and *Spartina patens*. It currently functions as a classic New England pocket marsh. As such, freshwater discharge is primarily to the tidal creeks with some groundwater discharge through the creek bottoms and a small amount through the marsh fringe where it meets the upland.

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 Scorton Creek embayment system under evaluation by the Project Team. The Scorton Creek estuarine system is a shallow, extensively branched, marsh-dominated estuary with a single inlet to Cape Cod Bay. Watershed modeling was undertaken to sub-divide the overall watershed to the Scorton Creek system 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 attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These travel distributions within each sub-watershed are used as a procedural check to gage the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the Sagamore flow cell on Cape Cod; the Scorton Creek watershed is located within the Sagamore groundwater lens. Model assumptions for calibration of the Scorton Creek Estuary hydrodynamic and water quality models included surface water discharges measured as part of the MEP stream flow program (2006 to 2007).

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 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 streams and the portion of the groundwater system that discharges directly into an estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the Scorton Creek system and its various sub-watersheds, such as Hoxie and Nye Ponds and Long Hill Creek, were delineated using the regional model of the Sagamore Lens flow cell (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 Scorton Creek system and its sub-watersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

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. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 519 feet below NGVD 29 in the Sagamore Lens). The maximum thickness of layer 20 is 279 feet (Sagamore model). In the Scorton Creek watershed area, bedrock generally is approximately 150 feet below NGVD 29 (Walter and Whealan, 2005). In the groundwater flow model, this means that the lowest model layer is inactive throughout most of the watershed area. 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 also varies in elevation depending on the location within the lens.

Direct rainwater run-off in these Cape Cod aquifer materials is typically rather low. Lithological 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 in the groundwater models 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 stream flow data collected in 1989-1990 as well as 2003.

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 southern portions of the Scorton Creek system watershed are located in the Mashpee Outwash Plains Deposits, in which 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. Glacial collapse structures caused by melting of remnant ice blocks form the kettlehole depressions that are now freshwater ponds. As one moves north toward Cape Cod Bay, the watershed is crossed by the Sandwich Moraine, which was created during a re-advance of the regional lobes of the continental ice sheet that excavated and piled up previously deposited materials. Along the northern edge of the watershed, including most of the marsh areas, are Lake Cape Cod Bay Deposits, which tend to be fine sands, silts, and clay. These materials, which underlie most of the marsh systems along the northern edge of Cape Cod, were deposited as lake bottom deposits for a large lake that formed in the southern portion of Cape Cod Bay between the moraines to the south and a relatively stable continental ice sheet position to the north (Walter and Whealan, 2005).

Although these glacial materials vary, modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that groundwater flowpaths are largely unaffected by the transitions between outwash and moraine materials (*e.g.*, Masterson, *et al*., 1996). Most of the lake bottom deposit areas along the northern portion of Cape Cod are covered by saltwater marshes, but the presence of extensive streams at margin of the marshes suggests that large portions of the upgradient aquifer is discharging along this margin. This is largely supported by the good agreement between modeled Scorton Creek stream watershed flows and measured MEP streamflows on the gaged streams (see Section IV.2). This agreement is also consistent with similar comparisons in other marsh-dominated systems along the northern coast of Cape Cod (*e.g.*, Howes, *et al*., 2007).

The Sagamore Lens groundwater 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 consumptive loss, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within residential areas designated as using on-site septic systems.

III.3 SCORTON CREEK SYSTEM CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Scorton Creek embayment system, including Hoxie and Nye Ponds, and the two large streams gauged by the MEP and sub-watersheds with direct groundwater discharge to the estuary (Figure III-1) were determined by the United States Geological Survey (USGS). Model outputs of the watershed boundaries are usually presented as “saw toothed” lines that reflect the movement of modeled particles between the grid cells that make up the groundwater model and how those cells are organized to reflect natural features, such as pond shorelines, river segments, and contributing areas to public water supplies. In order utilize the guidance provided by the model, these modeled lines were “smoothed” to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (d) to more closely match the sub-estuary segmentation of the tidal hydrodynamic model and (e) to address streamflow measurements collected as part of the MEP. The smoothing refinement process was a collaborative effort developed between the USGS and the rest of the MEP Technical Team. The smoothing simplification of watershed delineations/recharge areas lines and other model outputs usually involved visual curve fitting and checking model outputs against aerial maps of ponds, streams, and wetlands. The MEP sub-watershed delineation includes 10-yr time-of-travel boundaries. Overall, 13 sub-watershed areas were delineated within the Scorton Creek study area.

Table III-1 provides the daily freshwater discharge volumes for the sub-watersheds as calculated from the groundwater model; these volumes were used in the salinity calibration of the MEP tidal hydrodynamic model and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to the directly measured surface water discharges. The overall estimated freshwater flow into the Scorton Creek system from the MEP delineated watershed is 46,032 m3/d. This flow includes corrections for inflow from Lawrence and Spectacle Ponds, which straddle the boundary of the Scorton Creek system watershed and the adjacent Three Bays MEP watershed (Howes, *et al*., 2006), as well as Long Hill Creek watershed discharge that bypasses the estuary (see Section IV.2). Lawrence and Spectacle ponds are located on the regional groundwater divide. As has been done in all previous MEP reports, the watersheds to ponds in these circumstances are defined and the outflow is divided among the subwatersheds that intersect the downgradient shoreline of the pond. This approach is reasonable and consistent with the USGS modeling.

The MEP watershed delineation is the second watershed delineation completed in recent years for the Scorton Creek System. Figure III-2 compares the delineation completed under the current effort with the delineation completed by the Cape Cod Commission as part of the Coastal Embayment Project (Eichner, *et al*., 1998). The CCC delineation was largely based on local and regional water table measurements collected from available well data over a number of years and normalized to average conditions. The Commission's delineation was incorporated into the Commission's regulations through the three versions of the Regional Policy Plan (CCC, 1996, 2001, and 2009).

The MEP watershed area for the Scorton Creek system as a whole is 6% larger than 1998 CCC delineation (7,353 acres vs. 6,917 acres, respectively). These areas include the estuary surface area. Although the overall area is similar, the spatial distribution of the 2 watershed delineations are different. The most significant differences are the western boundary of the watershed, which is located further to the west in the MEP watershed, and the location of the regional groundwater divide/southern watershed boundary, which was located further to the south in the CCC delineation. The eastern watershed boundary is essentially the same in both the CCC and MEP delineations. The MEP watershed delineation also includes delineation of interior sub-watersheds to various components of the Scorton Creek estuarine system, such as selected ponds and streams that were not included in the CCC delineation. The inner subwatershed delineations show the connections between adjacent watersheds and the complexities of flow paths. These refinements are another benefit of the update of the regional groundwater model (Walter and Whealan, 2005).

The evolution of the watershed delineations for the Scorton Creek system has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model and the use of this model 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 down gradient estuary. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Scorton Creek system (Section V.1).

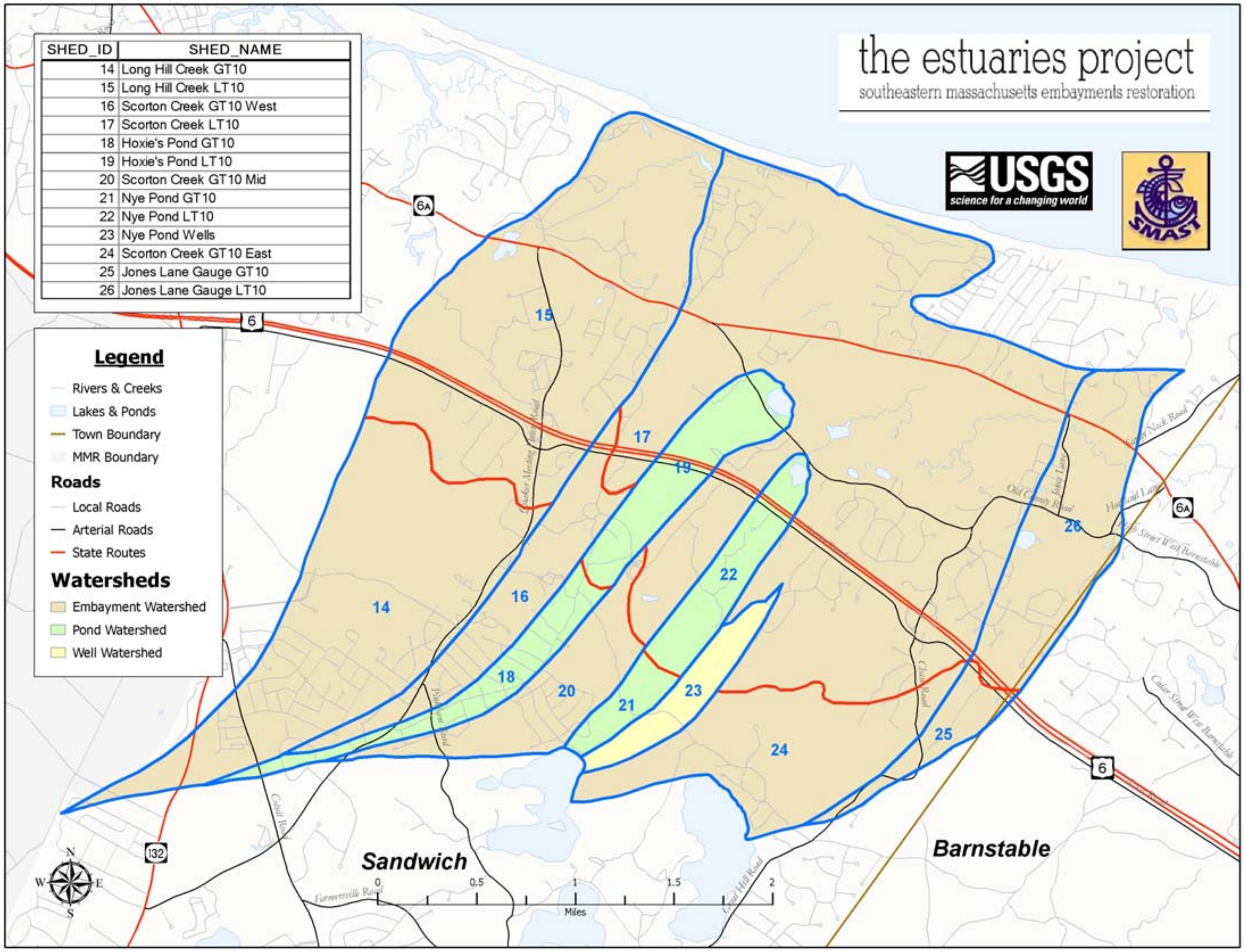


Figure III-1. Watershed delineation for the Scorton Creek estuary system, which exchanges tidal waters with Cape Cod Bay. Subwatershed delineations are based on USGS groundwater model output with modifications to better address pond and estuary shorelines and MEP stream gage measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a “10” in the watershed names. Recharge from the Lawrence Pond and Spectacle Pond watersheds are shared with the Three Bays MEP watershed (Howes, *et al*., 2006).

Table III-1. Daily groundwater discharge from each of the sub-watersheds in the watershed to the Scorton Creek system estuary, as determined from the regional USGS groundwater model.

Watershed	#	Watershed Area (acres)	% contributing to Estuaries	Discharge	
				m^3/day	ft^3/day
Long Hill Creek GT10	14	975	40%	2,974	105,017
Long Hill Creek LT10	15	1,219	40%	3,715	131,203
Scorton Creek GT10W	16	293	100%	2,245	79,289
Scorton Creek LT10	17	2,411	100%	18,499	653,288
Hoxie Pond GT10	18	172	100%	1,318	46,542
Hoxie Pond LT10	19	228	100%	1,748	61,718
Scorton Creek GT10 Mid	20	268	100%	2,056	72,597
Nye Pond GT10	21	69	100%	527	18,626
Nye Pond LT10	22	172	100%	1,320	46,617
Nye Pond Wells	23	124	100%	953	33,671
Scorton Creek GT10E	24	477	100%	3,657	129,152
Jones Lane Gage GT10	25	104	100%	799	28,214
Jones Lane Gage LT10	26	439	100%	3,366	118,873
Lawrence Pond	3Bay	403	54%	1,679	59,291
Spectacle Pond	3Bay	308	50%	1,176	41,522
TOTAL SCORTON CREEK SYSTEM				46,032	1,625,620

Notes:
1) discharge volumes are based on 27.25 inches of annual recharge on adjusted watershed areas (total watershed areas are shown);
2) Lawrence Pond and Spectacle Pond are shared with Three Bays MEP watersheds (Howes, *et al*., 2006), percentage of flow from these ponds to each estuary watershed is determined by length of downgradient watershed boundary,
3) As indicated, only 40% of the Long Hill Creek watershed flow discharges within the Scorton Creek system. A flapper valve at the point of freshwater Creek discharge to the estuary causes a ponding of water behind it and stream gauging at this point shows that the valve only allows 40% of the watershed flow to discharge into the estuary. The remainder of flow is thought to discharge back into the downgradient aquifer system and directly into Cape Cod Bay following the artificially induced hydraulic gradient. This is discussed in Section IV.2.
4) listed flows do not include precipitation on the surface of the estuary
5) totals may not match due to rounding.

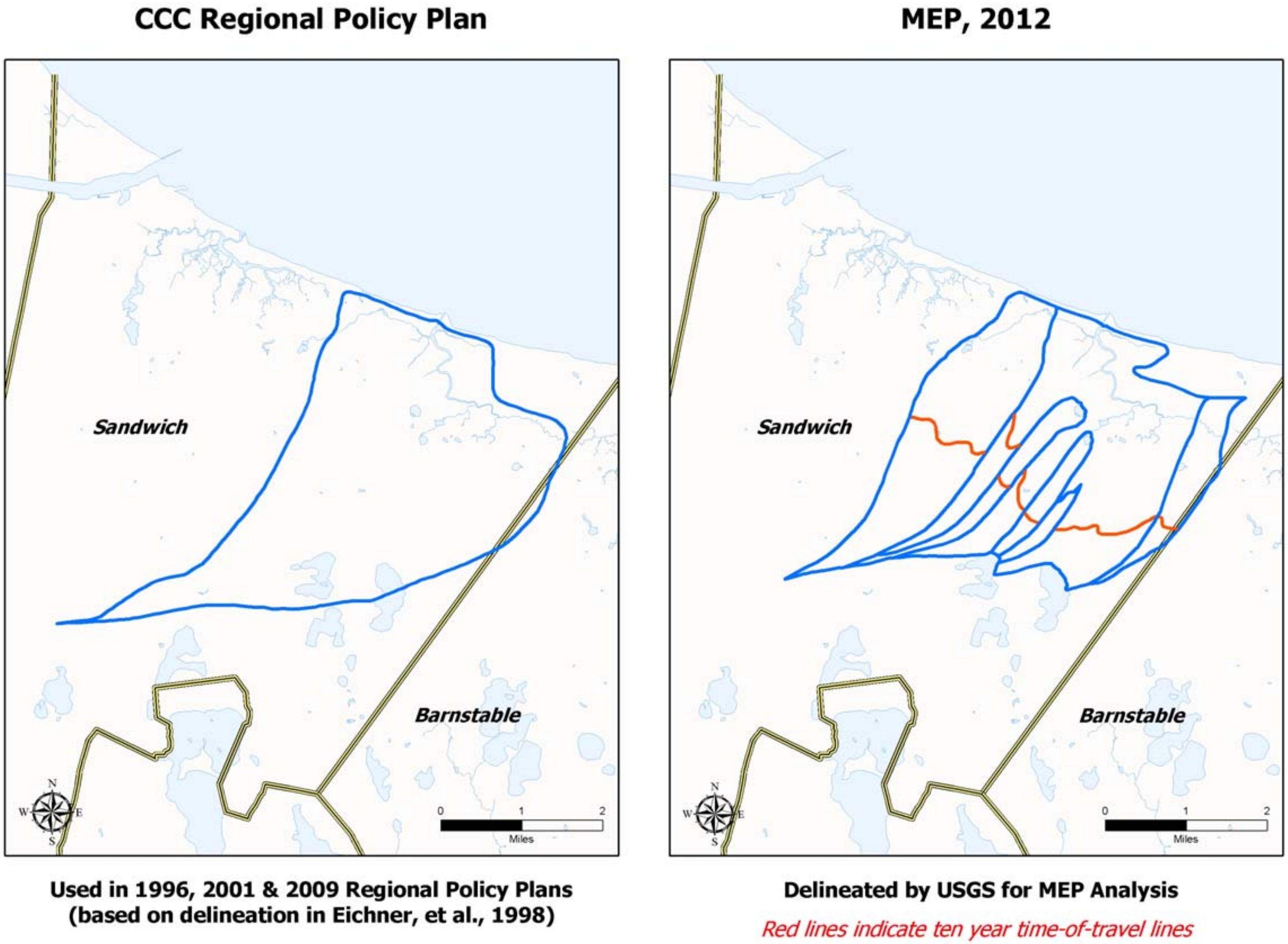


Figure III-2. Comparison of MEP Scorton Creek watershed and sub-watershed delineations used in the current assessment and the Cape Cod Commission watershed delineation (Eichner, *et al*., 1998), which has been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009). The MEP watershed area for the Scorton Creek system as a whole is 6% larger than 1998 CCC delineation. Scorton Creek exchanges tidal waters with Cape Cod Bay to the north.

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 Scorton Creek estuary system. Determination of watershed nitrogen inputs to this embayment system requires: (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 prior to reaching the estuary. This latter natural attenuation process results from biological processes that naturally occur within these 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 estuarine 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. Permanent burial of nitrogen in the sediments is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters or in some cases a sink for nitrogen reaching the bottom. Failure to include the nitrogen balance of estuarine sediments and the watershed attenuation generally leads to errors in predicting water quality, particularly in determination of summertime nitrogen load to embayment waters.

In order to determine watershed nitrogen loading inputs to the Scorton Creek estuary system, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of the estuary and its watersheds (Section III). The Scorton Creek watershed was sub-divided to define contributing areas or subwatersheds to each of the major inland freshwater systems and to each major portion of the estuary. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches estuary waters in less than 10 years or greater than 10 years. A total of 13 subwatersheds were delineated in the overall Scorton Creek watershed, including watersheds to Hoxie and Nye Ponds and the two MEP gaged streams (see Section III). The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each portion of the estuary. The Scorton Creek watershed is located between the Sandwich Harbor and Barnstable Harbor watersheds.

The initial task in the MEP land use analysis is to gage whether or not nitrogen discharges to the watershed have reached the estuary. This involves a temporal review of land use changes, the time of groundwater travel provided by the USGS watershed model, and review of data at natural collection points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. Ten-year time of travel subwatersheds in the Scorton Creek watershed have been delineated for ponds, streams and the estuary itself. Review of less than and greater than watersheds indicates that 54% of the unattenuated nitrogen load from the whole watershed is within less than 10 year travel time to the estuary (Table IV-1). This review includes refinements for flow leaving the watershed from ponds along its outer boundary. If the attenuated loads are reviewed, the percentage increases to 58% of the watershed load is within a 10 year time of travel. MEP staff also reviewed the year-built information in the town assessor's database and found that most of the average year-built in the greater than 10 year subwatersheds is more than 20 years (range is 13 to 31 years). The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuary (after accounting for natural attenuation, see below) and that the distinction between time of travel in the subwatersheds is not important for modeling existing conditions. Overall and based on the review of all this information, it was determined that the Scorton Creek estuary is currently in balance with its watershed load.

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed site-specific studies is applied to other portions. 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 based on regional analyses. For the Scorton Creek estuary system, the model used land-use data from the Towns of Sandwich and Barnstable which was transformed into nitrogen loads using both regional nitrogen loading factors and local watershed-specific data (such as parcel by parcel water use and alternative septic system monitoring). 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" or unattenuated nitrogen load to each receiving embayment, since attenuation during transport is included at a later stage.

Natural attenuation of nitrogen during transport from land-to-sea within the Scorton Creek watershed was determined based upon a site-specific study of streamflow and assumed attenuation in the upgradient freshwater ponds. Streamflow was characterized at the Jones Lane crossing of the upper portion of the marsh and Long Hill Creek at Ploughed Neck Road. Subwatersheds to these stream discharge points allowed comparisons between field collected data from the streams and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is generally estimated based on available information. Attenuation through the ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data is reliable enough to calculate a pond-specific attenuation factor. Streamflow and associated surface water attenuation is included in the MEP's nitrogen attenuation and freshwater flow investigation, presented in Section IV.2.

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. In the present effort, two freshwater ponds have delineated subwatersheds within the Scorton Creek watershed. If smaller aquatic features that have not been included in this MEP analysis were providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly (~10%) overestimated given the distribution of nitrogen sources within the watershed.

Table IV-1. Percentage of unattenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Scorton Creek.

WATERSHED		LT10	GT10	TOTAL	%LT10
Name		kg/yr	kg/yr	kg/yr	
Long Hill Creek GT10	14		4,188	4,188	0%
Long Hill Creek LT10	15	4,404		4,404	100%
Scorton Creek GT10W	16		1,733	1,733	0%
Scorton Creek LT10	17	6,528		6,528	100%
Hoxie Pond GT10	18		1,339	1,339	0%
Hoxie Pond LT10	19	413		413	100%
Scorton Creek GT10 Mid	20		1,367	1,367	0%
Nye Pond GT10	21		148	148	0%
Nye Pond LT10	22	172		172	100%
Nye Pond Wells	23	64		64	100%
Scorton Creek GT10E	24		519	519	0%
Jones Lane Gage GT10	25		341	341	0%
Jones Lane Gage LT10	26	1,556		1,556	100%
Lawrence Pond	3Bay		976	976	0%
Spectacle Pond	3Bay		525	525	0%
Scorton Creek Whole System		13,282	11,136	24,417	54%

Notes:

a) loads have been corrected to 1) include division of portions of nitrogen load from ponds and wellhead protection areas to downgradient subwatersheds, 2) exclude nitrogen loads that are discharged outside of the Scorton Creek system watershed from ponds or wellhead protection areas on the system watershed boundaries, 3) exclude atmospheric loading on the estuary surface waters; if these are included the percentage of load within a less than 10 year time-of-travel increases to 58%.

b) Review of year-built information for single-family residences (the predominant parcel type in the watershed) within the GT10 subwatershed show that the average year-built in most of the subwatersheds is greater than 20 years (ranging between 13 and 31 years). This information suggests that average nitrogen loads from the GT10 subwatersheds are currently reaching the estuary and estuarine water quality data is in balance with watershed nitrogen loading.

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary without flowing through one of these interim pond and stream measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Scorton Creek Estuarine System; 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

Since the watershed to Scorton Creek is almost entirely included in the Town of Sandwich, Estuaries Project staff obtained digital parcel and tax assessor's data from the town to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data from Sandwich are from 2010. Using GIS techniques, this data was linked to three years (2007-2009) of individual account water use data from the Sandwich Water District. This unified database also contains traditional information regarding land use classifications (MassDOR, 2012) plus additional information developed by the town. It is also the database that the town is using for its current wastewater planning effort. The database efforts were completed with the assistance from GIS staff from the Cape Cod Commission (CCC).

Figure IV-1 shows the land uses within the Scorton Creek estuary watershed. Land uses in the study area are grouped into nine land use categories: 1) residential, 2) commercial, 3) industrial, 4) agricultural, 5) mixed use, 6) undeveloped, 7) public service/government, including road rights-of-way, 8) freshwater, and 9) properties without assessor's land use codes. These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2012). "Public service" in the MADOR system is tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, open space, roads) and private groups like churches and colleges.

Public service land uses are the dominant land use type in the overall Scorton Creek watershed and occupy 47% of the watershed area (Figure IV-2). Examples of these land uses are lands owned by town and state government (including golf courses, open space, and wellhead protection lands), housing authorities, and churches. Residential land uses occupy the second largest area with 33% of the watershed area. It is notable that land classified by the town assessor as undeveloped is 10% of the overall watershed area. Public service lands generally are located in a swath south of Route 6 that extends across the middle of the watershed and includes Sandwich Hollows Golf Club, the Maple Swamp Conservation Area, and Oak Ridge School.

In all the subwatershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type. Residential parcels are 78% of the parcels in the Jones Lane Gage subwatershed, 75% of parcels in the Long Hill Creek subwatershed, and 76% of all parcels in the Scorton Creek system watershed. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 87% to 100% of residential parcels in the individual subwatersheds and 95% of the residential parcel area throughout the Scorton Creek system watershed.

In order to estimate wastewater flows within the Scorton Creek study area, MEP staff also obtained parcel-by-parcel water use data from the Sandwich Water District. Three years of water use (years 2007, 2008 and 2009) was obtained (personal communication, Dan Mahoney, Superintendent, 3/11). The water use data were linked to the town parcel database. These data are also the basis for the town's current Watershed Nitrogen Management Plan (Wright-Pierce, 2012).

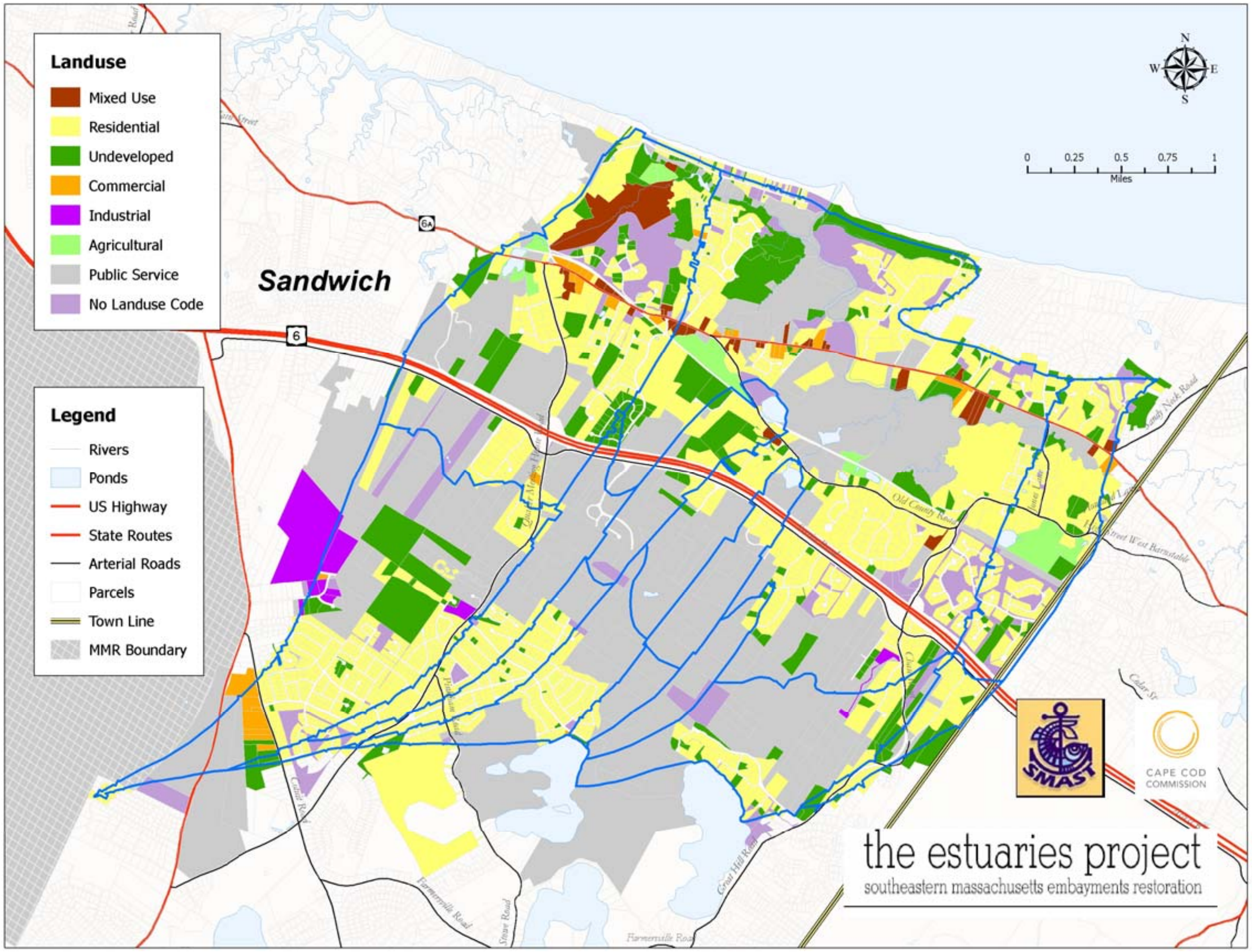


Figure IV-1. Land-use in the Scorton Creek system watershed and subwatersheds. Watershed is almost completely within the Town of Sandwich (a small area of Barnstable is included along the eastern border). Land use classifications are based on town assessor classifications and MADOR (2012) categories. Base assessor and parcel data are from the year 2010.

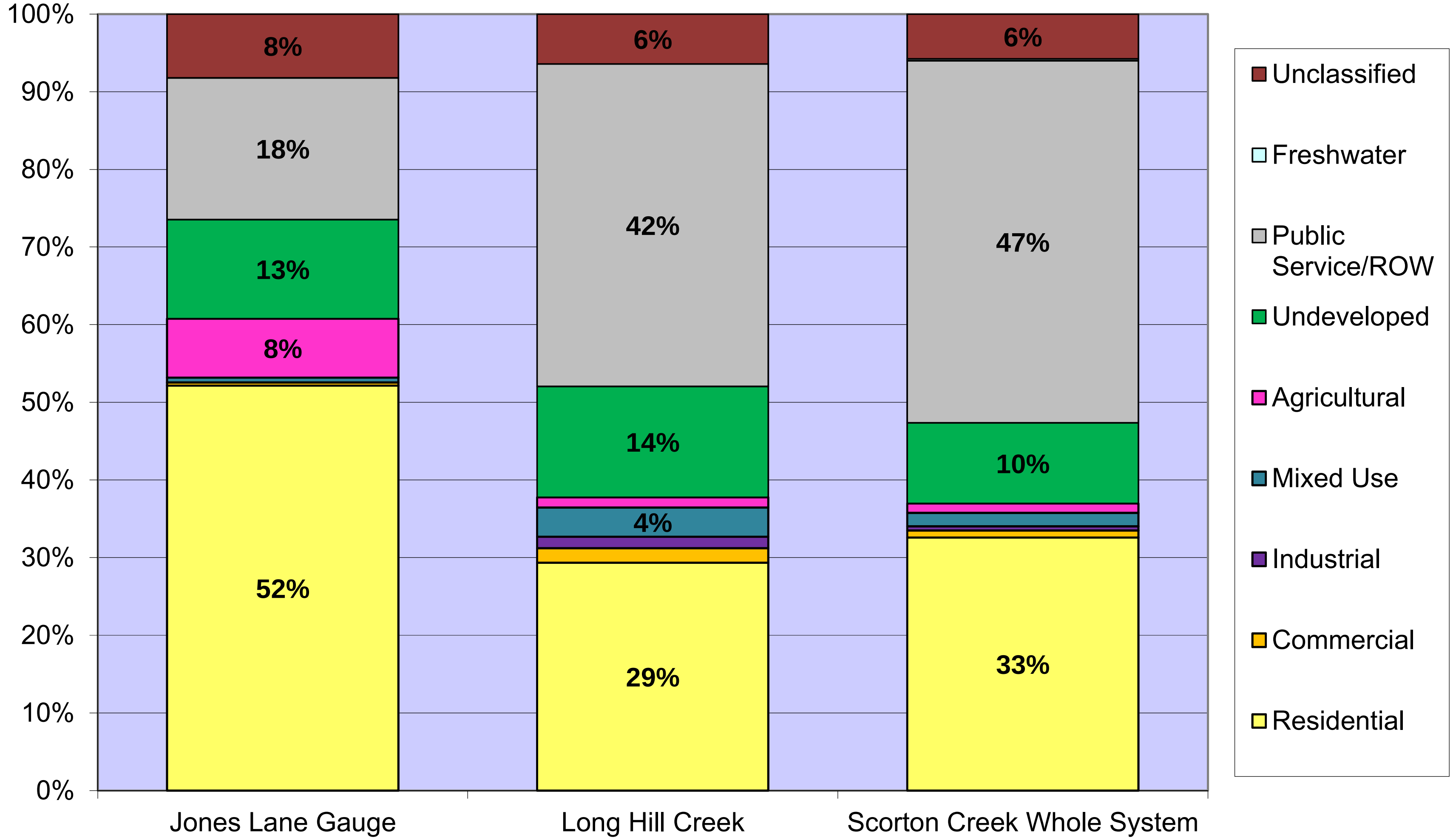


Figure IV-2. Distribution of land-uses by area within the Scorton Creek system watershed and two component subwatersheds. Land use categories are generally based on town assessor's land use classification and grouping recommended by MADOR (2012). Unclassified parcels do not have an assigned land use code in the town assessor's databases. Only percentages greater than or equal to 4% are shown.

Measured water use is used to estimate wastewater-based nitrogen loading from the individual parcels; average water use for each parcel is used for parcels with multiple years of data. The final wastewater nitrogen load for each parcel is based upon the measured water-use, wastewater nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2). All parcels are assumed to use on-site septic systems unless additional information is available.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

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 a number of studies and additional information that directly measured septic system and per capita loads 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 2000, Costa *et al.* 2001). 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}.

Given the seasonal shifts in occupancy and rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use approach is applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to assessor's 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 that reaches the aquatic receptors downgradient in the aquifer.

All nitrogen losses within the septic system are incorporated into the MEP analysis. For example, information developed at the Massachusetts 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. 2001). Downgradient studies of septic system plumes in similar soils 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, MEP staff 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, MEP staff has derived a combined term for an effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per volume) to nitrogen load (N mass). This coefficient 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).

The nitrogen loads developed using this 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. 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). Further, modeled and measured nitrogen loads were determined for a small sub-watershed to Mashapaquit Creek in West Falmouth Harbor (Smith and Howes, manuscript in review) where measured nitrogen discharge from the aquifer was within 5% of the modeled N load. Another 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 modeled and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year in this type of 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 covers 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 any given town, 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 soils and 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 with other MEP Nitrogen Loading Coefficients (*e.g.*, stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees with specific studies of consumptive water use and nitrogen 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 developed 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, 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 for septic systems in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%) needed to lower embayment nitrogen levels to a nitrogen target (*e.g.*, nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to provide an independent validation of the average residential water use within the Scorton Creek watershed, MEP staff reviewed US Census population values for the Town of Sandwich. 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 for the purposes of Title 5 each person generates 55 gpd of wastewater. Based on data collected during the 2000 US Census, average occupancy within Sandwich is 2.75 people per housing unit with 84% of year-round occupancy of available housing units; 2010 Census results are roughly the same: 2.66 and 82%, respectively. Average water use for single-family residences with municipal water accounts in the Scorton Creek MEP study area is 195 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the study area wastewater average flow for a single-family residence is 175 gpd.

In order to provide a check on the measured water use, Sandwich 2000 and 2010 Census average occupancies were used to estimate wastewater flows. Multiplying the Census occupancies by the state Title 5 estimate of 55 gpd of wastewater per capita results in an average estimated water use per residence of 151 gpd and 146 gpd, respectively. Correction for minor summer occupancy increase (2X for seasonal residences) these estimated flows to 189 gpd and 183 gpd, respectively. This analysis suggests that population and water use information are in reasonable agreement and that the average water use is reasonably reflective of average wastewater estimates.

At the outset of the MEP, project staff decided to utilize the water use approach for determining residential wastewater generation by septic systems because of the inherent difficulty in accurately gauging actual occupancy in areas impacted by seasonal population fluctuations such as most of Cape Cod. The above analysis suggests that water use, on average, is a reasonable estimate of wastewater generation within the study area.

Water use information exists for 64% of the 2,477 developed parcels in the Scorton Creek watershed. Parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have a listed account in the water use databases. Of the 896 developed parcels without water use accounts, 853 (95%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and are assigned the MEP Sandwich study area average water use of 184 gpd in the watershed nitrogen loading modules. Since the analysis of the Scorton Creek and Sandwich Harbor watersheds were being completed at the same time, MEP staff decided to use the average for both systems for various residential parcels groups (single family, two family and multi-family residences) without water use and for buildout estimates. Another 21 developed parcels without water use are parcels classified as other types of residential properties (e.g., multi-family or condominiums). These parcels are assumed to utilize private wells and are assigned the MEP Sandwich study area average water use of 191 gpd and 236 gpd for parcels with the 104 (two family residences) and 109 (multi-family residences) land use codes, respectively.

Wastewater Treatment Facilities

When developing watershed nitrogen loading information, MEP project staff typically seek additional information on enhanced wastewater treatment in the project study area. This information is reviewed and if judged reliable is included in the watershed nitrogen loading model.

MEP staff reviewed whether large wastewater treatment facilities discharge within the Scorton Creek watershed. Three state Groundwater Discharge Permits (GWDPs) are listed within the Scorton Creek watershed: Sandwich High School, Oak Ridge School, and the Riverview School (personal communication, Brian Dudley, MassDEP, 2/12). MEP staff requested three years-worth of monitoring data and received data for 2009, 2010, and 2011. A GWDP is required under MassDEP regulations for wastewater treatment systems with design flows greater than 10,000 gallons per day. Among these three schools, Sandwich High and Riverview had both flow and effluent total nitrogen concentration, so site-specific wastewater nitrogen loads could be determined. Since Oak Ridge School did not have site-specific wastewater system performance data, the wastewater nitrogen loading from the school was treated as a standard Title 5 septic system. These loads were incorporated into the watershed nitrogen loading model.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of watershed nitrogen loading to estuaries is usually fertilized areas: lawns, golf courses, and cranberry bogs. Residential lawns are usually the predominant source within this category. In order to add this source to the watershed nitrogen loading model for the Scorton Creek system, MEP staff reviewed available regional information about residential lawn fertilizing practices and incorporated site-specific information for the Sandwich Hollows Golf Club. An estimated nitrogen load is also included for the cranberry bogs and agricultural areas in the watershed. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts.

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 in this assessment was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. This assessment, which was completed prior to the start of the MEP, 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 nitrogen leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn; these factors are used in the MEP 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 in the three town survey were found to have the higher rate of fertilizer application and hence higher estimated annual contribution to groundwater of 3 lb/yr.

MEP staff obtained course- and turf-specific, nitrogen fertilizer application information for the Sandwich Hollows Golf Club (personal communication, Dave Polidor, Golf Course Superintendent, 2/12). Golf courses usually have different fertilizer application rates for different turf areas, usually higher annual application rates for tees and greens (~3 to 4 pounds per 1,000 square feet) and lower rates for fairways and roughs (~2 to 3.5 pounds per 1,000 square feet). As has been done in all MEP reviews, MEP staff reviewed the layout of the golf course from aerial photographs, classified the various turf types, and, using GIS tools, assigned these areas to the appropriate subwatersheds. The golf course-specific nitrogen application rates were then applied to the respective turf areas, a standard MEP 20% leaching rate was applied, and annual load for the portion of each golf course within each subwatershed was calculated.

MEP staff also contacted Riverview Schools and Sandwich Schools for nitrogen fertilizer applications at both of these sites. Riverview School only uses fertilizer for newly seeded areas and for young trees; only ~9 kg of N was used during 2011 (personal communication, Richard Dalrymple, Business Manager, 2/12). Sandwich Schools provided fertilizer application data for 2006 to 2008 including monthly fertilizer application rates and nutrient mixes (personal communication, Alan Hall, Director of Facilities, 2/12). These data indicated that Sandwich Schools regularly fertilize 26 acres at three facilities: the High School, the Oak Ridge School, and the Forestdale School. MEP staff digitized the athletic field areas at the two schools within the Scorton Creek watershed (the High School and the Oak Ridge School) and applied the monthly-specific nitrogen application rates to determine the loads at each school and the distribution of the loads among the subwatersheds where the field areas are located.

Nitrogen loads were also added for site-specific agricultural land uses. Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on an enhanced review of nitrogen export from cranberry bogs in southeastern Massachusetts (Howes and DeMoranville, 2009; Howes and Teal, 1995). Based on these studies, only the bog loses measurable nitrogen, the forested upland releases only very low amounts. For the watershed nitrogen loading analysis, MEP staff obtained a MassDEP GIS coverage that is maintained for Water Management Act purposes (personal communication, Jim McLaughlin, MassDEP SERO, 1/13). This GIS coverage includes the surface areas of the eight cranberry bogs in the Scorton Creek watershed.

MEP also received farm parcel-specific animal counts from the Town of Sandwich (personal communication, Dave Mason and Tori Hall, Sandwich Health Department, 4/13). These counts indicated the presence of the following farm animals within the Scorton Creek watershed: horses, burros, sheep, chickens, and ducks. Species-specific nitrogen loads were developed based on USDA and other species-specific research on nitrogen manure characteristics, including leaching to groundwater. Loads were assigned to individual farm lots based on the animal counts.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Scorton Creek assessment are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the CCC's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and MassDEP'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). Factors used in the MEP nitrogen loading analysis for the Scorton Creek watershed are summarized in Table IV-2.

Table IV-2. Primary Nitrogen Loading Factors used in the Scorton Creek MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Sandwich-specific data.

Nitrogen Concentrations:	**mg/l**
Road Run-off	1.5
Roof Run-off	0.75
Natural Area Recharge	0.072
Direct Precipitation on Embayments and Ponds	1.09
Wastewater Coefficient	23.63
Fertilizers:	
Average Residential Lawn Size (sq ft)[1]	5,000
Residential Watershed Nitrogen Rate (lbs/lawn)[1]	1.08
Leaching rate	20%
Cranberry Bogs nitrogen release – flow through bogs (kg/ha/yr)	23.08
Cranberry Bogs nitrogen release – pump on/pump off bogs (kg/ha/yr)	6.95
Nitrogen Fertilizer Rate for golf courses, determined from site-specific information; other areas assumed to utilize residential application rate; vegetable crop nitrogen fertilizer applications based on loads determined in other MEP assessments	

Recharge Rates:	**in/yr**
Impervious Surfaces	40
Natural and Lawn Areas	27.25
Water Use/Wastewater:	
Existing developed single-family residential parcels wo/water accounts and buildout residential parcels:	184 gpd[2]
Existing developed parcels w/water accounts:	Measured annual water use
Commercial and Industrial Buildings without/WU and buildout additions[3]	
Commercial	
Wastewater flow (gpd/1,000 ft2 of building):	36
Building coverage:	12%
Industrial	
Wastewater flow (gpd/1,000 ft2 of building):	16
Building coverage:	10%
Average Single Family Residence Building Size from watershed data (sq ft)	1,713

Notes:
1) Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.
2) Based on average measured flow in the Sandwich MEP study area
3) Based on characteristics of similarly classified properties with the Town of Sandwich

Road areas are based on GIS information developed by the Massachusetts Executive Office of Transportation, which provides road, sidewalk, and road shoulder widths for various road segments. MEP staff utilized the GIS to sum these segments and their various widths by subwatershed. Project staff also checked this information against parcel-based rights-of-way.

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information is linked to the parcel coverages, parcels are assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel is located within a respective subwatershed. Following the assigning of boundary parcels, all large parcels are examined individually and are 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. This effort results in "parcelized" watersheds that can be more easily used during the development of management strategies.

The review of individual parcels straddling watershed boundaries includes corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Building footprints, for example, are based on available information contained in the Sandwich assessor's database. Project staff used the average single-family residence building footprint based on available properties in the MEP study area (1,713 sq ft) for any residential units without footprint information. Commercial and industrial footprints for properties without building footprint information are also based on average building coverage of individual lots with similar land uses within the town. Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) is 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 the Scorton Creek estuary. The assignment effort is undertaken to better define sub-estuary loads and enhance use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

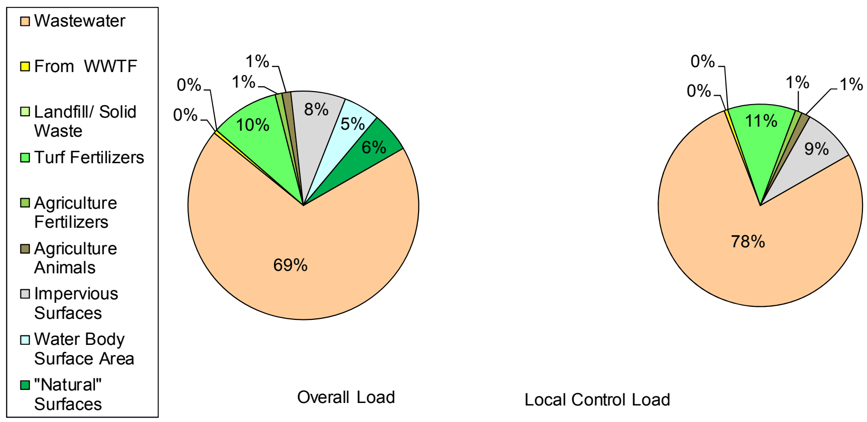
Following the assignment of all parcels, subwatershed modules were generated for each of the 13 subwatersheds in the Scorton Creek study area. These subwatershed modules summarize, among other things: water use, parcel area, frequency, private wells, and road area. All relevant nitrogen loading data is assigned to each subwatershed. Individual sub-watershed information is then integrated to create the Scorton Creek Watershed Nitrogen Loading module with summaries for each of the individual 13 subwatersheds. The subwatersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Scorton Creek study area, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), wastewater treatment facilities, fertilizers (including contributions from agriculture and golf courses), impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-3). 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-3). 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.

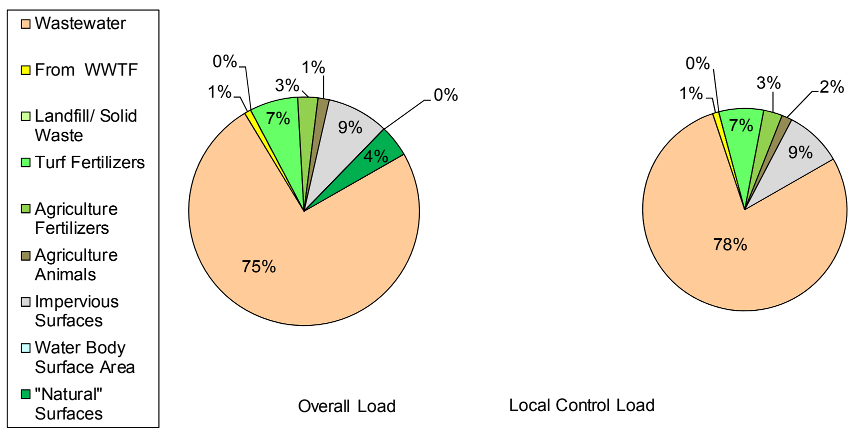
One of these attenuation adjustments occurs in the freshwater ponds. Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond is used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment is based on the percentage of discharging shoreline bordering each downgradient sub-watershed. In the Scorton Creek study area, this occurs for the ponds located along the outer boundary of the watershed (Lawrence Pond and Spectacle Pond). For example, at Spectacle Pond 39% of its downgradient shoreline discharges into the Scorton Creek GT10 Mid subwatershed and 11% discharges to the Nye Pond GT10 subwatershed. The remainder of the shoreline discharges to the subwatersheds in the upper portion of the Three Bays MEP watershed (Howes, *et al.*, 2006). This breakdown of the water discharge from Spectacle Pond means that 39% of the accompanying attenuated nitrogen load that leaves the pond reaches Scorton Creek and the

Table IV-3. Scorton Creek Watershed Nitrogen Loads. Unattenuated nitrogen loads are a sum of all sources within the watershed without including natural nitrogen attenuation during transport through surface freshwater systems. Attenuated nitrogen loads are based on measured and assigned attenuation factors for upgradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2), while pond attenuation factors are assigned a standard MEP nitrogen attenuation of 50% attenuation based on MEP data review, including water quality monitoring from the Cape Cod Pond and Lake Stewards program. All nitrogen loads are kg N yr^{-1}.

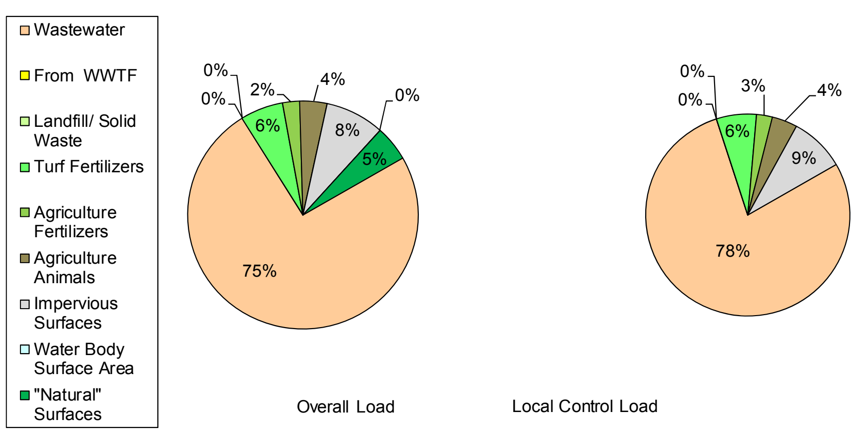
		Scorton Creek N Loads by Input (kg/y):										% of Pond Outflow	Present N Loads			Buildout N Loads		
Watershed Name	shed ID#	Wastewater	From WWTF	Landfill/ Solid Waste	Turf Fertilizers	Agriculture Fertilizers	Agriculture Animals	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Scorton Creek System		**13,325**	**98**	**-**	**1,852**	**183**	**237**	**1,480**	**977**	**1,086**	**6,166**		**19,239**		**14,650**	**25,405**		**18,756**
Scorton Creek GT10W	16	1,414	-	-	177	-	-	93	-	49	74		1,733		1,733	1,807		1,807
Scorton Creek LT10	17	4,633	66	-	637	24	110	620	-	437	1,897		6,528		6,528	8,425		8,425
Scorton Creek GT10 Mid	20	1,020	-	-	212	-	-	92	-	43	7		1,367		1,367	1,373		1,373
Nye Pond Wells	23	30	-	-	4	-	-	5	-	25	13		64		64	78		78
Scorton Creek GT10E	24	362	-	-	29	-	-	35	-	93	222		519		519	741		741
Lawrence Pond	LP	337	-	-	21	-	-	18	547	53	735	54%	976	50%	346	1,711	50%	659
Spectacle Pond	SPP	203	-	-	11	-	-	5	168	23	251	39%	411	50%	206	663	50%	331
Hoxie Pond TOTAL		1,153	-	-	341	17	-	135	43	63	266	100%	1,752	50%	876	2,018	50%	1,009
Hoxie Pond GT10	18	1,034	-	-	178	-	-	101	-	25	51		1,339		1,339	1,389		1,389
Hoxie Pond LT10	19	119	-	-	163	17	-	34	43	38	215		413		413	628		628
Nye Pond TOTAL		209	-	-	74	-	-	25	75	51	70	100%	434	50%	188	504	50%	206
Nye Pond GT10	21	93	-	-	34	-	-	9	-	12	-		148		148	148		148
Nye Pond LT10	22	60	-	-	37	-	-	14	29	32	-		172		172	172		172
Spectacle Pond	SPP	56.32	-	-	3	-	-	1	47	6	70	11%	114	50%	57	184	50%	92
Long Hill Creek Gauge TOTAL		2,551	32	-	231	97	54	296	-	153	2,242	40%	3,414	51%	1,673	5,655	51%	2,771
Long Hill Creek GT10	14	3,414	-	-	223	-	-	380	-	172	4,300		4,188		4,188	8,488		8,488
Long Hill Creek LT10	15	3,008	80	-	359	243	135	364	-	214	1,343		4,404		4,404	5,746		5,746
Jones Lane Gauge TOTAL		1,413	-	-	114	45	73	158	-	95	389		1,897	47%	1,005	2,286	47%	1,212
Jones Lane Gauge GT10	25	206	-	-	17	-	73	26	-	19	87		341		341	428		428
Jones Lane Gauge LT10	26	1,206	-	-	97	45	-	132	-	75	302		1,556		1,556	1,858		1,858
Scorton Creek LT10 Estuary Su	17								144				144		144	144		144



A. Scorton Creek Whole System



B. Scorton Creek: Long Hill Creek subwatershed



C. Scorton Creek: Jones Road Gauge subwatershed

Figure IV-3. Land use-specific unattenuated nitrogen loads (by percent) to the a) whole Scorton Creek watershed, b) Long Hill Creek subwatershed, and c) Jones Road Gage subwatershed. "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.

11% flows to Nye Pond and is subject to additional natural nitrogen attenuation. Similar pond-specific calculations were completed wherever pond flows and nitrogen loads were divided among a number of downgradient receiving subwatersheds. The two ponds completely within the Scorton Creek watershed (Nye Pond and Hoxie Pond) discharge directly to the Scorton Creek LT10 subwatershed, so their flow and attenuated nitrogen load is not subdivided among downgradient subwatersheds.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally watershed sites of natural nitrogen reduction (or attenuation) prior to the watershed nitrogen reaching an estuary. These ponds are generally kettle hole depressions of the land surface 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 also have a stream outlet, which is often a herring run, that also acts as a discharge point. Since the nitrogen loads usually flow into a pond with the groundwater, the relatively more productive pond ecosystems incorporate some of the nitrogen, retain some nitrogen in the sediments, and change the nitrogen among its various oxidized and reduced forms. As result of these interactions, some of the nitrogen in the pond watershed is removed from the estuary watershed system, mostly through burial in pond sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining (attenuated) loads flow back into the groundwater system along the downgradient side of the pond and eventual discharge into the downgradient embayment or through a stream outlet directly to the estuary. The nitrogen load summary in Table IV-3 includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads.

Nitrogen attenuation in freshwater ponds has generally been found to be at least 50% in MEP analyses, so a conservative attenuation rate of 50% is generally assigned to all nitrogen from freshwater pond watersheds in the watershed model unless more detailed pond monitoring or studies are available. Detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported a 50% attenuation factor as a reasonable, somewhat conservative rate. However, in some cases, if sufficient monitoring information is available, a pond-specific attenuation rate is incorporated into the watershed nitrogen loading modeling [*e.g.*, 87%, Mystic Lake; 40%, Middle Pond; and 52%, Hamblin Pond in the Three Bays MEP Report (Howes, *et al.*, 2006)]. In order to review whether a pond-specific nitrogen attenuation rate other than 50% should be used, the MEP Technical Team reviews the available data on each pond, including available nitrogen concentrations, impacts of sediment regeneration, temperature profiles, watershed freshwater volumetric inputs and bathymetric information.

Bathymetric information is generally a prerequisite for determining enhanced attenuation, since it provides the volume of the pond and, with appropriate pond nitrogen concentrations, a measure of the nitrogen mass in the water column. Combined with the watershed recharge, this information can provide a residence or turnover time that is necessary to gage nitrogen attenuation.

In addition to bathymetry, temperature profiles are useful to help understand whether temperature stratification is occurring in a pond. If the pond has an epilimnion (*i.e.*, a well-mixed, relatively isothermic, warm, upper portion of the water column) and a hypolimnion (*i.e.*, a deeper, colder layer), the stability and volume of these two layers must be accounted for in the nitrogen attenuation calculations. In these stratified lakes, the upper epilimnion is usually the primary discharge for watershed nitrogen loads; the deeper hypolimnion generally does not interact with the upper layer. However, deep lakes with hypolimnions often also have significant sediment regeneration of nitrogen and in lakes with impaired water quality this regenerated nitrogen can impact measured nitrogen concentrations in the upper epilimnion and this impact should also be considered when estimating nitrogen attenuation.

Many ponds on Cape Cod have been sampled through the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and the initiative of local volunteer pond sampling programs. The PALS Snapshots are regional, one-time, volunteer pond samplings undertaken during the summer and supported for the last twelve years by SMAST and the Cape Cod Commission, with free laboratory services and technical oversight provided by the Coastal Systems Program Laboratory at SMAST. Sampling protocols developed through the PALS program (Eichner *et al*., 2003) have been used for more extensive pond sampling programs in many communities on Cape Cod. Sampling under these protocols has included field collection of temperature and dissolved oxygen profiles and sampling of standardized depths that include some evaluation of the impact of sediment nutrient regeneration. PALS water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, total chlorophyll-*a*, alkalinity, and pH. In some cases town programs have generated sufficient sampling data collected throughout a number of summers that modified MEP nitrogen attenuation rates can be reliably assigned to freshwater ponds.

Within the Scorton Creek study area, there are four freshwater ponds with delineated watersheds: Lawrence, Spectacle, Hoxie, and Nye. Lawrence and Spectacle have been sampled multiple times during the 12 years of PALS Snapshots that have been conducted, but Hoxie and Nye have not been sampled through the PALS Snapshots. Among these ponds, Lawrence, Spectacle, and Hoxie have available pond-wide bathymetric data (Eichner, *et al*., 2003). For the two ponds with both bathymetry and water quality sampling data, neither has had sufficient sampling outside of the PALS Snapshots to assign a pond-specific nitrogen attenuation rate. This data review supports the use of the standard MEP pond 50% attenuation rate for all four ponds in within the Scorton Creek study area.

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development and accompanying nitrogen loads within the study area watersheds. The MEP buildout is relatively straightforward and is generally completed in four steps: 1) each residential parcel classified by the town assessor as developable is identified and divided by minimum lot sizes specified in town zoning and the resulting number of new residential units is rounded down, 2) parcels classified as developable commercial and industrial parcels by the town assessor are identified, 3) residential, commercial and industrial parcels with existing development and areas greater than twice zoning's minimum lot size are identified, divided by the minimum lot size and the resulting number of new units is rounded down, and 4) results are discussed with town staff and/or planning board members and the analysis results are modified based on local knowledge.

It should be noted that the initial MEP buildout approach is relatively simple and does not include any modifications/refinements for lot line setbacks, wetlands, road construction, frontage requirements, parcel shape requirements, or other more detailed zoning provisions. The MEP buildout approach also does not include potential impacts associated with the higher densities usually associated with 40B affordable housing projects. The fourth step, including the discussions with town planners, and, occasionally, town planning boards and wastewater consultants, usually leads to additional insights on developments that are planned, especially developments planned on government or public service parcels, and updates to assessor classifications, including lands purchased by the town as open space. This final step may lead to removal and/or additions to the number of parcels initially identified as developable and may include application of more detailed zoning provisions.

As an example of how the MEP approach might apply, assume an 81,000 square foot lot is classified by the town assessor as a developable residential lot (land use code 130). This lot is divided by the 40,000 square foot minimum lot size specified in town zoning and the result is rounded down to two. As a result, two additional residential lots would be added to the subwatershed in the MEP buildout scenario. This addition could then be modified during discussion of town staff.

Other provisions of the MEP buildout assessment include undevelopable lots, commercial and industrial properties, and lots less than the minimum areas specified by zoning. Properties classified by the Town of Sandwich assessors as "undevelopable" (*e.g.*, MassDOR codes 132, 392, and 442) are not assigned any development at buildout (unless revised by the town review). Commercial and industrial properties classified as developable are not subdivided; the area of each parcel and the factors in Table IV-2 are used to determine a building size and wastewater flow for these properties. Pre-existing lots classified by the town assessor as developable are also treated as developable even if they are less than the minimum lot size specified in zoning; so, for example, a 10,000 square foot lot classified by the town assessor as a developable residential property (130 land use code) will be assigned an additional residential dwelling in the MEP buildout scenario even though the minimum lot size required by the zoning in the area is 40,000 square feet. Most town zoning bylaws have a lower minimum lot size for pre-existing lots (usually 5,000 square feet) that will minimize instances of regulatory takings. Existing developed residential properties that are larger than zoning's minimum lot sizes are also assigned additional development potential only if enough area is available to accommodate at least one additional lot as specified by the zoning minimum.

Following the completion of the initial buildout assessment for the Scorton Creek watersheds, MEP staff reviewed the results with town officials. MEP staff reviewed the preliminary watershed buildout results with Nathan Jones, Sandwich Town Planner in June 2012. Provisions were included in the final buildout for the proposed development associated with Community Green and South Sandwich Village. The configuration of these developments were based on the most current proposal at the time, which were contained in the Draft Environmental Impact Report (DEIR)(HWG, 2012). The proposal includes 62 units for Community Green and a community wastewater treatment facility (WWTF) with a total Title V design flow of 140,000 gpd. The proposed WWTF would discharge within subwatershed #14 (Long Hill Creek GT10) with a 10 mg/L TN discharge. The DEIR also mentions connection of additional development to the WWTF, which would increase the flow to 320,000 gpd, but it was not clear at the time of the DEIR submittal where the additional connections would occur. Impacts of alternative buildout scenarios involving South Sandwich Village could be evaluated in subsequent scenarios using the MEP linked model for Scorton Creek estuary. All other suggested changes from Sandwich staff based on the initial review were incorporated into the final buildout for Scorton Creek.

All the parcels with additional buildout potential within the Scorton Creek watershed are shown in Figure IV-4. Each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for wastewater and impervious surfaces. Residential

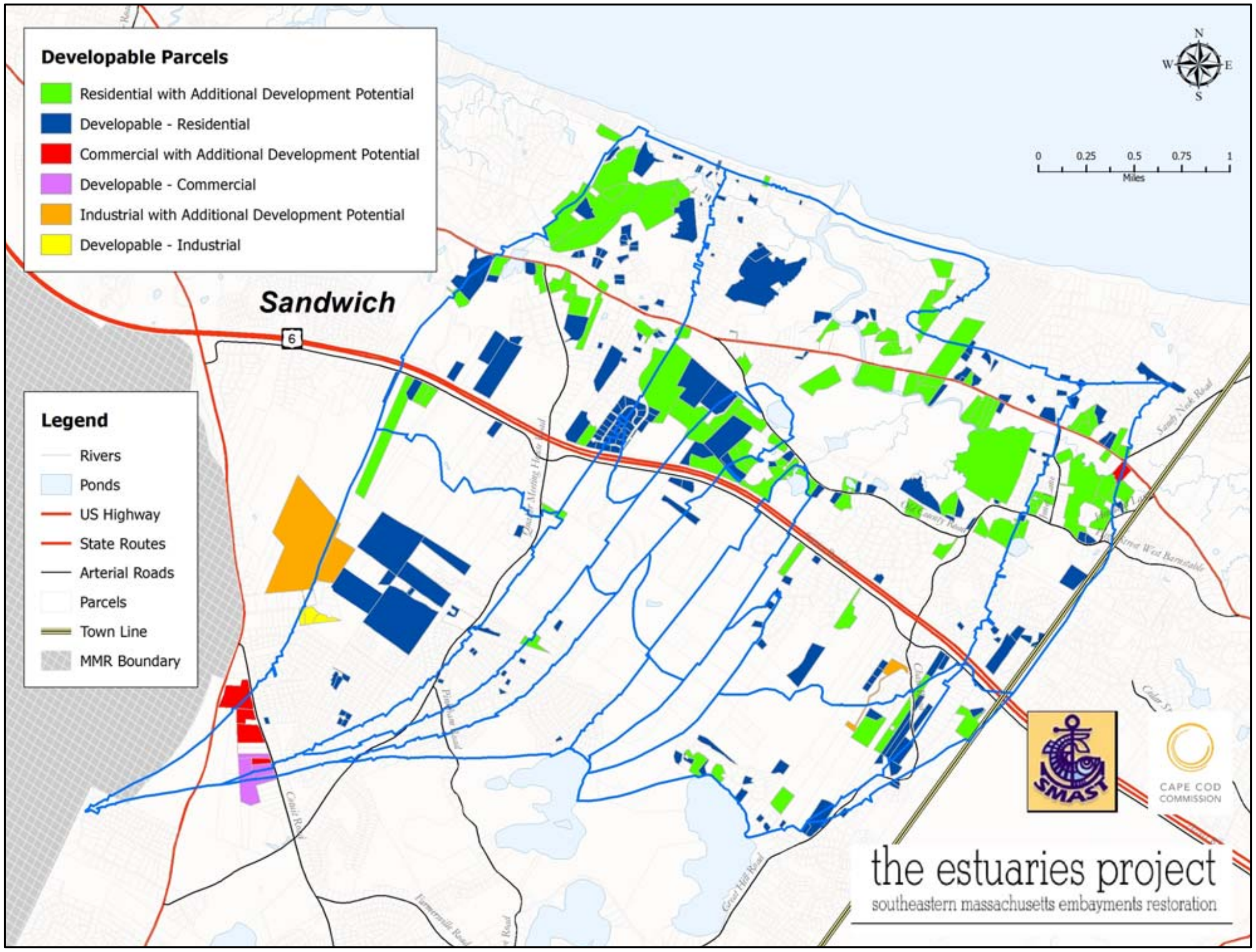


Figure IV-4. Developable Parcels in the Scorton Creek watershed. Parcels colored red, orange, and green are developed parcels (residential, commercial and industrial, respectively) with additional development potential based on current zoning, while parcel colored blue, light orange, and light green are corresponding undeveloped parcels classified as developable by the town assessor. Parcels along watershed boundaries are assigned to subwatersheds to 1) minimize the splitting of properties for future management purposes and 2) achieve a match of area with the modeled watersheds of 2% or less. Developable parcels are based on town assessor classifications and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. Buildout scenario also includes proposed development of South Sandwich Village as detailed in the DEIR (HWG, 2012), which was the most current proposal at the time. All buildout results were reviewed with town staff.

additions also include lawn fertilizer nitrogen additions. All wastewater loads are assumed to come from standard on-site septic systems. Cumulative unattenuated buildout loads are indicated in a separate column in Table IV-3. Buildout additions within the Scorton Creek watersheds will increase the unattenuated loading rate by 32%.

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, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) 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 sub-watersheds of the Scorton Creek Embayment System being investigated under this nutrient threshold analysis was 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 from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such as the developed regions of the Scorton Creek watershed). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the watershed for the Scorton Creek Estuary, a portion of the freshwater flow and transported nitrogen passes through two main surface water systems prior to entering the estuary (e.g. Long Creek discharging into a tidal tributary creek to Scorton Creek and flow from Scorton Creek near to its headwaters at the Jones Lane road crossing). These surface water systems provide the opportunity for reductions in nutrient loading to the estuary, primarily through nitrogen attenuation.

Failure to determine the attenuation of watershed-derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another, the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2000). Similarly, MEP analysis of the Quashnet River indicates that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach in the Scorton Creek embayment system. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the perimeter of the embayment system in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted in the 2 major surface water flow systems in the Scorton Creek watershed, 1) flow from Long Creek into the estuarine portion of Scorton Creek and 2) freshwater from the head of Scorton Creek flowing into the estuarine portion of the marsh system.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater stream discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area up gradient from the various gaging sites. Flow and nitrogen load were determined at two gage locations for 16 months of record (Figure IV-5). During the study period, a velocity profile was completed at each gage positioned in each of the creeks every month to two months. The summation of the products of creek subsection areas of the channel cross-section and the respective measured velocities represent the computation of instantaneous flow (Q) through a given creek.

Determination of flow at the gages on the two main surfacewater inflows to Scorton Creek Marsh was calculated and based on the measured values obtained for cross sectional area of each creek as well as creek specific velocity. Freshwater discharge was represented by the summation of individual discharge calculations for each channel subsection for which a cross sectional area and velocity measurement were obtained. Velocity measurements across the entire channel cross section were not averaged and then applied to the total creek cross sectional area.

The formula that was used for calculation of stream flow (discharge) is as follows:

$$Q = \Sigma(A * V)$$

where by:

Q = Stream discharge (m^3/s)
A = Stream subsection cross sectional area (m^2)
V = Stream subsection velocity (m/s)

Thus, each stream subsection will have a calculated stream discharge value and the summation of all the sub-sectional stream discharge values will be the total calculated discharge for the stream.

Periodic measurement of flows over the entire "stream" gage deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gages. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given creek. These hourly stages values were then entered into the stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. A complete annual record of flow in the creeks (365 days) was generated for the surface water discharge flowing into the head of the two tidal creeks flowing into the main estuarine channel of the Scorton Creek system and emanating from a network of bogs to the west of the system (Long Creek) and the wetland at the head of Scorton Creek.

The annual flow record for the surface water flow at each of the gages was merged with the nutrient data set generated through the weekly water quality sampling performed at the gage locations to determine nitrogen loading rates to the Scorton Creek Estuary. Nitrogen discharge from the two small creeks was calculated using the paired daily discharge and daily nitrogen concentration data to determine the mass flux of nitrogen through the specific gauging sites. For each of the stream gage locations, weekly water samples were collected at low tide as both gage location were tidally influenced, in order to obtain nutrient concentrations from which nutrient load was determined. In order to pair daily flows with daily nutrient concentrations, interpolation between weekly nutrient data points was necessary. These data are expressed as nitrogen mass per unit time (kg/d) and can be summed in order to obtain weekly, monthly, or annual nutrient load to the embayment system as appropriate. Comparing these measured nitrogen loads based on flow in the creeks and water quality sampling to predicted loads based on the land use analysis allowed for the determination of the degree to which natural biological processes within the watershed to the gaged creeks currently reduces (percent attenuation) nitrogen loading to the overall embayment system.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Scorton Creek at Jones Lane discharging to Scorton Creek Estuary

Located up gradient of the gage site on the freshwater portion of Scorton Creek is a small wetland and like many of the freshwater ponds and wetlands on Cape Cod, this small wetland system (source water to Scorton Creek) has a surface water discharge rather than draining solely to the aquifer. This outflow may serve to decrease the wetlands ability to attenuate nitrogen, however, it also provides for a direct measurement of the nitrogen attenuation taking place in the wetland and associated riparian zones as well as the streambed of the freshwater portion of the Creek. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Creek above the gage site and the measured annual discharge of nitrogen to the tidally influenced estuarine portion of Scorton Creek, Figure IV-5.

At the gage site (situated immediately up-gradient of the culvert passing under Jones Lane), a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater creek discharging from the wetland and associated nitrogen load to the lower portion of Scorton Creek. The gage was located as far down gradient along

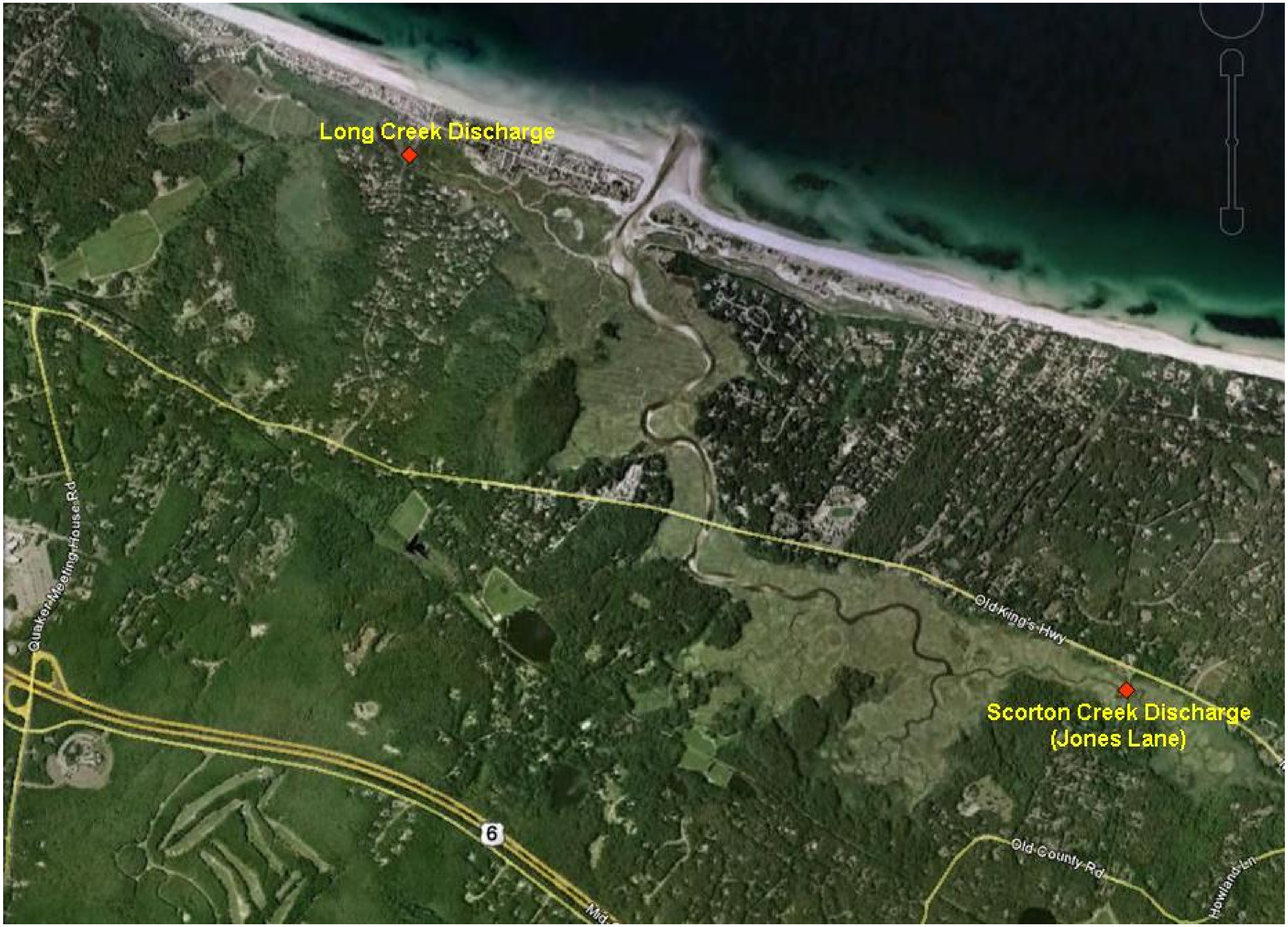


Figure IV-5. Location of Stream gages (red symbol) in the Scorton Creek embayment system.

the Creek reach as possible such that freshwater flow could be measured at low tide while also capturing as much contribution of flow and load from the up-gradient watershed. Based on the stage record, the location of this specific gage was tidally influenced, however, stage at low tide was reflective of freshwater. Salinity measurements on the weekly water quality samples collected from the gage site yielded an average salinity over the entire gage deployment period of 5.6 ppt. The gage location was deemed acceptable for making freshwater flow measurements at low tide, however, a small salinity adjustment was made to adjust for saltwater in the flow measurements. Calibration of the gage was checked approximately monthly and a flow measurement obtained. The gage on the Creek was installed on June 20, 2006 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until September 18, 2007 for a total deployment of 15 months.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gage site based upon these flow measurements and parallel measurements of water level. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to the estuarine portion of Scorton Creek. This measured attenuated mass discharge is reflective of the biological processes occurring in the headwater wetland of Scorton Creek as well as the stream channel and riparian zone contributing to nitrogen attenuation (Figure IV-6 and Tables IV-4 and IV-5). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the measurement site.

The annual freshwater flow record for the discharge from the headwater wetland as measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the wetland was 5% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September and ending in August (low flow to low flow) was 4,392 m^3/day compared to the long term average flows determined by the USGS modeling effort (4,165 m^3/day).

The slight difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Creek was considered to be negligible. The small difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Creek discharging from the headwater wetland would indicate that the Creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Creek outflow from the headwater wetland to Scorton Creek were moderate, 0.627 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 2.75 kg/day and a measured total annual TN load of 1,005 kg/yr. In the Creek (freshwater), nitrate was a small portion of the total nitrogen pool (26%) and the low concentration of inorganic nitrogen (0.228 mg N L^{-1}) in the out flowing creek waters, indicates that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland and the creek was largely taken up by plants and transformed to organic nitrogen within the wetland / creek ecosystem. This is further supported when considering that dissolved and particulate organic nitrogen constitutes 64 percent of the total nitrogen load discharging to the estuarine

Table IV-4. Comparison of water flow and nitrogen discharges from the major surfacewater systems (freshwater) discharging to the Scorton Creek system. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Scorton Creek at Jones Ln. Discharge[(a)] Scorton Creek	Long Creek at Ploughed Neck Rd. Discharge[(a)] Scorton Creek	Data Source
Total Days of Record	365[(b)]	365[(c)]	(1)
Flow Characteristics			
Stream Average Discharge (m3/day) **	4,392	6,689	(1)
Contributing Area Average Discharge (m3/day)	4,165	6,734	(2)
Discharge Stream 2006-07 vs. Long-term Discharge	5.17%	-0.67%	
Nitrogen Characteristics			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.160	0.115	(1)
Stream Average Total N Concentration (mg N/L)	0.627	0.680	(1)
Nitrate + Nitrite as Percent of Total N (%)	26%	17%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	2.75	4.55	(1)
TN Average Contributing UN-attenuated Load (kg/day)	5.2	9.35	(3)
Attenuation of Nitrogen in Pond/Stream (%)	47%	51%	(4)

(a) Flow and N load to creeks discharging to the estuarine portions of the Scorton Creek Marsh system and includes apportionments of Pond contributing areas.
(b) September 1, 2006 to August 31, 2007.
(c) September 1, 2006 to August 31, 2007.
** Flow is an average of annual flow for 2006-2007 in both surfacewater systems

(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the creeks to Scorton Creek;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-5. Summary of annual volumetric discharge and nitrogen load from the freshwater discharge from Long Creek passing under Ploughed Neck Road to the Scorton Creek Estuary as well as Scorton Creek passing under Jones Lane flowing to the estuarine portion of Scorton Creek Marsh. Flows and loads based upon the data presented in Figures IV-X and Table IV-4. Note that the modeled watershed flow was adjusted for the direct discharge to Cape Cod Bay from the Long Hill Creek impoundment.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Long Creek @ Ploughed Neck Road Discharge to Scorton Creek Estuary (MEP)	**September 1, 2006 to August 31, 2007**	**2,441,514**	**280**	**1661**
Long Creek @ Ploughed Neck Road Discharge to Scorton Creek Estuary (MEP)	Based on Watershed Area and Recharge	**2,458,056**	--	--
Scorton Creek @ Jones Lane Discharge to Scorton Creek Estuary (MEP)	**September 1, 2006 to August 31, 2007**	**1,603,081**	**256**	**1005**
Scorton Creek @ Jones Lane Discharge to Scorton Creek Estuary (CCC)	Based on Watershed Area and Recharge	**1,520,225**	--	--

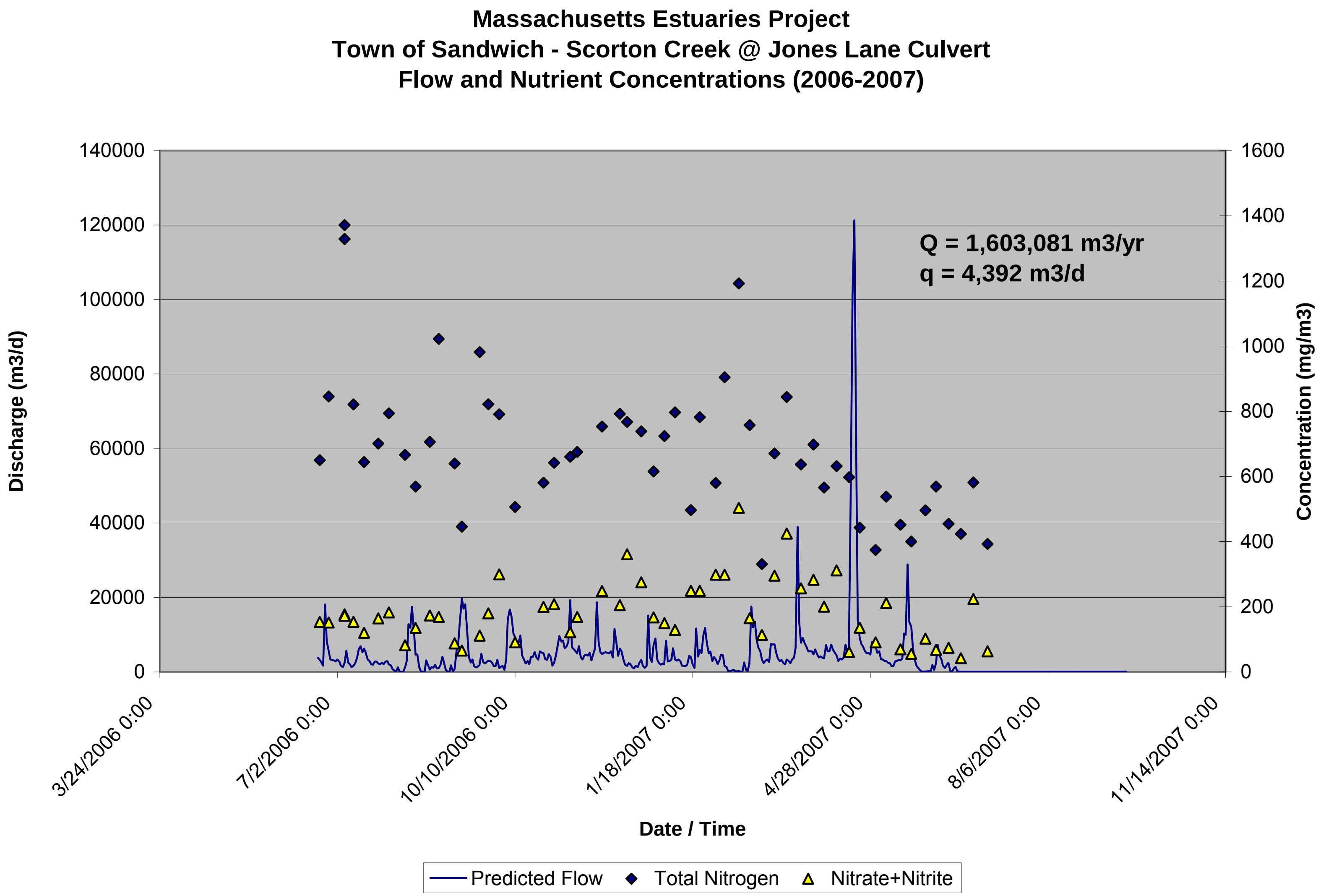


Figure IV-6. Freshwater flow from head of Scorton Creek directly into the estuarine portion of Scorton Creek Marsh (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (blue diamond) concentrations for determination of annual volumetric discharge and nitrogen load from the upper eastern sub-watershed to Scorton Creek Marsh (Table IV-4).

portion of Scorton Creek below Jones Lane. In addition, the low nitrate level (0.160 mg N L^{-1}) suggests that increasing uptake by freshwater systems upgradient from the gage location is small in this system. Inorganic nitrogen appears significantly attenuated already. Opportunities for enhancing nitrogen attenuation elsewhere in the watershed to the overall Scorton Creek system could be considered, keeping in mind there is not likely to be much more natural attenuation to be gained from the sub-watershed to the headwater wetland of Scorton Creek.

From the measured nitrogen load discharged by freshwater from the Creek at Jones Lane and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon lower total nitrogen load (1,005 kg yr^{-1}) discharged from the freshwater portion of Scorton Creek compared to that added by the various land-uses to the associated watershed (1,897 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 47% (i.e. 47% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams/creeks evaluated under the MEP is expected given the hydrologic and biogeochemical characteristics of the up-gradient wetland capable of attenuating nitrogen. The directly measured nitrogen loads from the Creek at Jones Lane was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.4 Surface water Discharge and Attenuation of Watershed Nitrogen: Long Creek at Ploughed Neck Road flowing into Scorton Creek Estuary

Located up gradient of the gage site on Long Creek discharging into a tidally influenced tributary channel to Scorton Creek is a small freshwater impoundment that captures flow from a network of upgradient bogs. This small impoundment (no recorded date of construction) has a surface water discharge that passes under Ploughed Neck Road rather than draining solely to the aquifer along its down-gradient shore. This outflow through a highly controlled culvert (up-gradient end has weir boards and down gradient end of culvert has a flapper valve) passing under the road may serve to decrease the attenuation of nitrogen taking place within the impoundment. However, it also provides an opportunity for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within associated wetland areas, riparian zones and streambed associated with the Creek flowing into the impoundment. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Creek and impoundment above the gage site and the measured annual discharge of nitrogen to the tidal creek down gradient of Ploughed Neck Road flowing into the Scorton Creek Marsh system, Figure IV-5.

At the gage site (situated in the impoundment immediately up-gradient of the Ploughed Neck Road crossing), a continuously recording vented calibrated water level gage was installed to yield the level of water in the impoundment spilling over a weir and into the culvert to Scorton Creek. As Long Creek is strongly tidally influenced down gradient of the flapper valve at Ploughed Neck Road, the gage had to be in the impoundment such that freshwater flow could be measured at low tide. To confirm that freshwater was being measured the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples. Despite the down gradient flapper valve and the up gradient weir, the stage record did indicate slight tidal influence in the impoundment, however, average low tide salinity was determined to be 1.2 ppt. Given the low average salinity, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked approximately monthly and a flow measurement obtained.

The gage on the Creek was installed on June 5, 2006 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until December 14, 2007 for a total deployment of 18 months.

Flow (volumetric discharge) leaving the freshwater impoundment considered Long Creek was measured at the weir every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Long Creek site based upon these flow measurements and parallel measurements of water level. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to Scorton Creek estuary. This measured attenuated mass discharge is reflective of the watershed source loading and the biological processes that intercept the nitrogen on its way to the estuary, e.g. in the up-gradient bogs, the small impoundment as well as the stream channel and riparian zone (Figure IV-7 and Tables IV-4 and IV-5). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the gage site.

The complex water control structures at the mouth of Long Creek resulted in an increase in water levels in the upgradient "impoundment" (artificial pond). This increase in water levels appears to have altered the hydraulic gradient from what had been the unimpounded situation. The result is that some of the flow that would have drained into the Scorton Creek Estuary, currently discharges by subsurface flow from the impoundment to Cape Cod Bay, through the barrier beach. This can be seen in both fresh pond hydrology and in coastal ponds that are periodically opened (e.g. Edgartown Great Pond, Tisbury Great Pond). The remainder of the flow discharges to the estuary, with its associated nitrogen load. The annual freshwater flow record for Long Creek discharging to Scorton Creek as measured by the MEP accounts for 40% of the total discharge from the Long Hill Creek watershed, with the rest discharging directly to Cape Cod Bay (Table III-1). The average daily flow based on the MEP measured flow data for one hydrologic year beginning September and ending in August (low flow to low flow) was 6,689 m^3/day, compared to the partitioned long term average flow determined by the USGS modeling effort (6,734 m^3/day).

Total nitrogen concentrations within the Creek outflow were moderate, 0.680 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 4.55 kg/day and a measured total annual TN load of 1,661 kg/yr. In the Creek (freshwater), nitrate was a minor portion of the total nitrogen pool (17%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the creek was largely taken up by plants and converted to organic nitrogen forms within the small up-gradient pond or stream ecosystems. This is further supported when considering that dissolved and particulate organic nitrogen constitutes 80 percent of the total nitrogen load discharging to the Scorton Creek system from Long Creek. The relatively low concentration of inorganic nitrogen (0.139 mg N L^{-1}) and nitrate (0.115 mg N L^{-1}) in the out flowing creek waters suggests that inorganic nitrogen is well attenuated by the small up-gradient impoundment and associated creek bed, riparian zone and wetlands in this sub-watershed and that the possibility for additional uptake by freshwater systems up-gradient from the gage location is limited. Opportunities for enhancing nitrogen attenuation elsewhere in the watershed to the overall Scorton Creek Marsh system could be considered. However, there is not likely to be additional natural attenuation to be gained from the sub-watershed to Long Creek. This would not be the case if nitrate levels were high. The

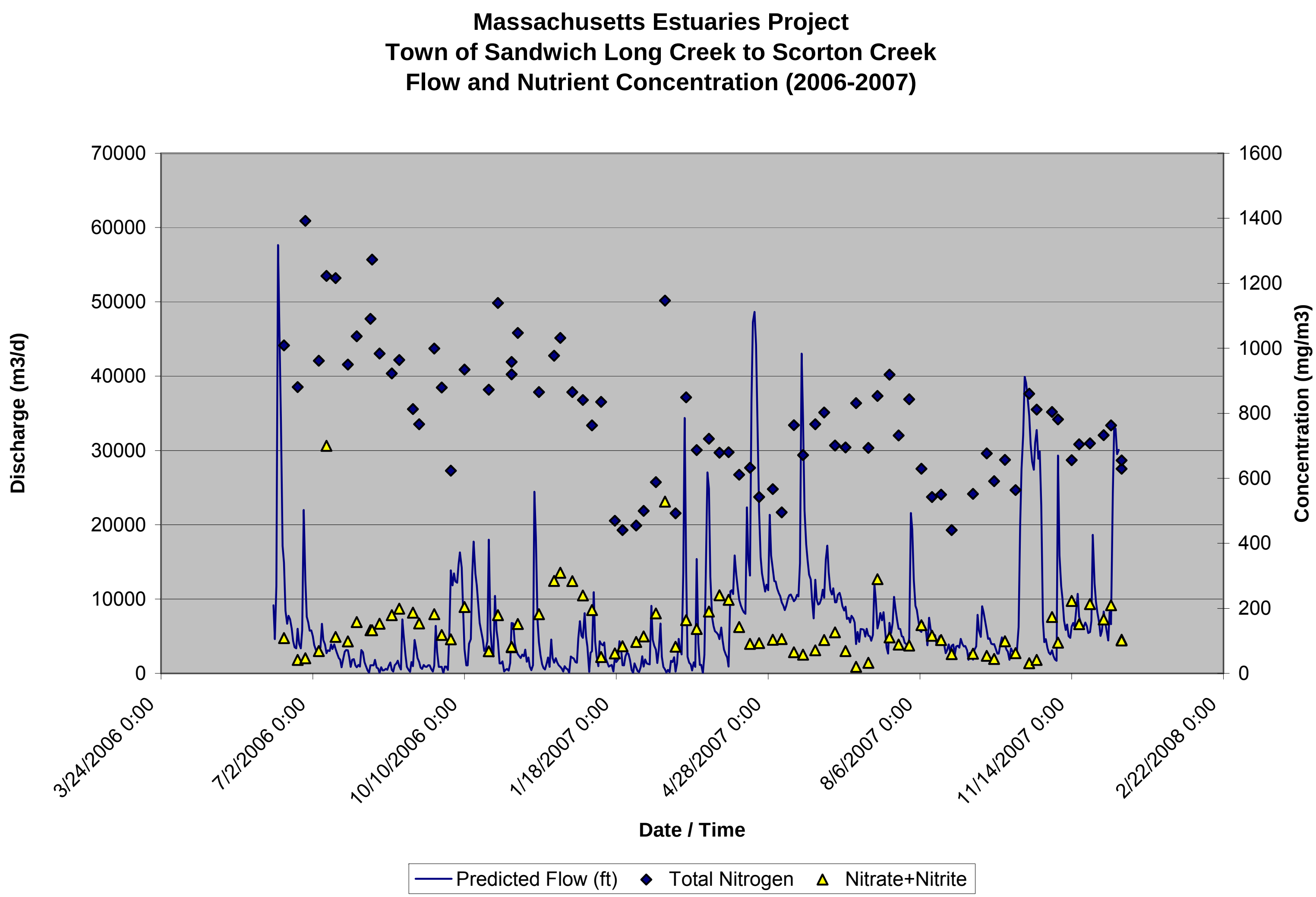


Figure IV-7. Long Creek flowing directly into the Scorton Creek Marsh system (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (blue diamond) concentrations for determination of annual volumetric discharge and nitrogen load from the upper western sub-watershed to Scorton Creek Marsh (Table IV-4).

shifting of flow directly to Cape Cod Bay is providing a significant nitrogen reduction, so maintenance of the water control structure appears to be important to the nitrogen balance of the Scorton Creek Estuary.

From the measured nitrogen load discharged by Long Creek to the Scorton Creek estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is attenuation of nitrogen being transported from the upper watershed to the estuary. Based upon lower total nitrogen load (1661 kg yr^{-1}) discharged from the freshwater Creek compared to that added by the various land-uses to the associated watershed (3,414 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 51% (i.e. 51% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other stream/pond complexes evaluated under the MEP is expected given the hydrologic and biogeochemical characteristics of the up gradient pond(s) and wetland areas capable of attenuating nitrogen. The directly measured nitrogen loads from Long Creek were used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the benthic nutrient flux surveys was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters throughout the Scorton Creek System. The mass exchange of nitrogen between water column 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-Water column Exchange of Nitrogen

As stated in the above section, 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 Scorton Creek system predominantly in highly bio-available forms from the surrounding upland watersheds and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the water column (once it entered) then predicting water column 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 bio-available form nitrate. This nitrate and other bio-available 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 water column for sufficient time to be flushed out to a down gradient larger water body (like Cape Cod Bay). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals and deposited on the bottom. 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 embayments.

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 enclosed tributary basins, particularly if they are deeper than the adjacent embayment. 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 bio-available nitrogen is returned to the embayment water column 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 have been investigated by SMAST and the MEP, 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. In contrast in some systems, with deep depositional basins or salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer (e.g. Namskaket Salt Marsh (lower reach), Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh or Sesachacha Pond on the Island of Nantucket). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, for example in the margins of the main basin to Lewis Bay (Town of Barnstable, Cape Cod). In contrast, most embayments show low rates of nitrogen release throughout much of a basins area and, in regions of high deposition the anoxic sediments show high release rates during summer months. The consequence of high deposition rates is that the basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

Failure to account for the site-specific nitrogen balance of the sediments and its spatial variation from the tidal creeks and embayment basins will result in significant errors in determination of the threshold nitrogen loading to the Scorton Creek system. 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-Water column Nitrogen Exchange

For the Scorton Creek Embayment 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. In the Scorton Creek system, sediment samples (9 cores) were collected from 9 sites (Figure IV-6) in July-August 2006, focusing on obtaining an areal distribution that would be representative of nutrient fluxes throughout the system but also considering tributary “basins” such as the narrow tidal creeks that extend landward off the main embayment channel/basin. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

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 small boat to a shore side 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 from each site (Figure IV-8) during the Scorton Creek incubation are as follows:

Figure IV-8. Scorton Creek Marsh System locations (green symbols) of sediment sample collection for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-6.

Scorton Creek System Benthic Nutrient Regeneration Cores

• SC-1	1 core	(Main Channel, above 6A)
• SC-2	1 core	(Main Channel, above 6A)
• SC-3	1 core	(Tributary Channel, above 6A)
• SC-4	1 core	(Main Channel, above 6A)
• SC-6	1 core	(Main Channel, below 6A)
• SC-7	1 core	(Main Channel, below 6A)
• SC-8	1 core	(Tributary Channel, below 6A)
• SC-9	1 core	(Tributary Channel, below 6A)
• SC-10	1 core	(Tributary Channel, below 6A)

Sampling was distributed throughout the salt marsh system such that the results for each site could be combined to calculate the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-water column exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1998) for nutrients and metabolism. Upon return to the field laboratory (Brewer Fiddlers Cove Marina), 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 orthophosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining samples 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 [*508-910-6325*]. The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

Water column nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (water column 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 water column 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 water column 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 streams, ponds and 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, these relate primarily to sediment and water column oxygen status. However, for the modeling of nitrogen within an embayment it is the relative balance of nitrogen input from water column 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 levels of organic matter within the sediments, whether the sediments are oxic or 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 lowering nitrogen levels.

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-9).

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 water column 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 total dissolved nitrogen uptake or release, and estimate of particulate nitrogen input.

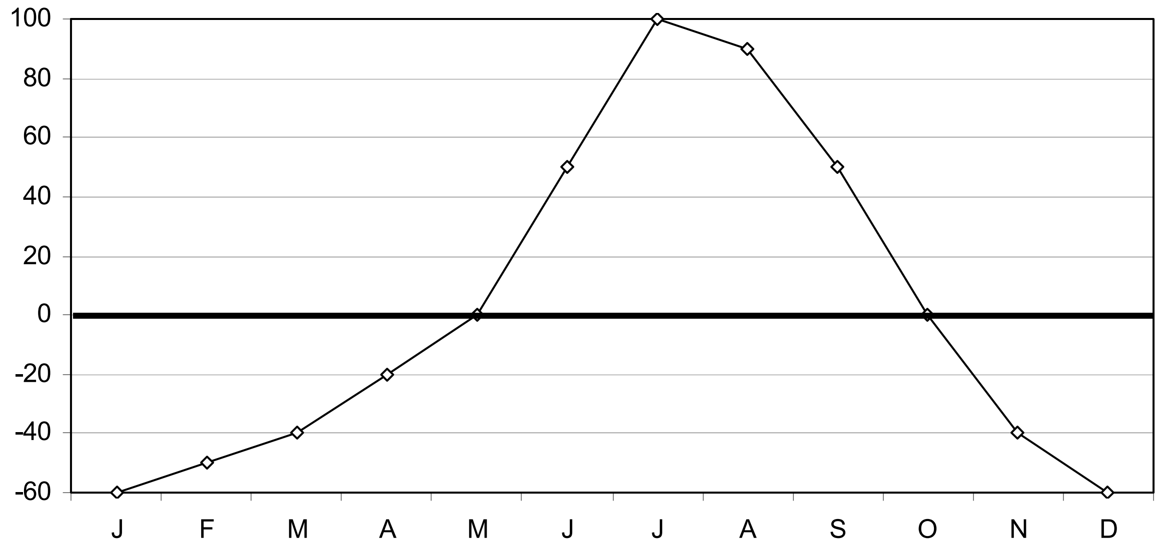


Figure IV-9. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment Nitrogen Release by Standard Core Approach: Sediment sampling was conducted throughout the main tidal channel of the Scorton Creek embayment as well as the major tidal creeks tributary to the main channel of the estuary. 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, as well as sediment type and an analysis of each site's tidal flow velocities. As expected flow velocities are generally low in the uppermost reaches of the tidal creeks and high in the lower portions of the system situated closer to the inlet to the system. 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, the average summer particulate carbon and nitrogen concentration within the overlying water and the tidal velocities from the hydrodynamic model (Chapter V). Two levels of settling were used. If the sediments were organic rich and 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 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 has been previously validated in outer Cape Cod embayments (Town of Chatham embayments) 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. Additional, validation has been conducted on deep enclosed basins (with little freshwater inflow), where the fluxes can be determined by multiple methods. In this case the rate of sediment regeneration determined from incubations was comparable to that determined from whole system balance.

Net nitrogen release or uptake from the sediments within the Scorton Creek Estuary were comparable to other salt marsh dominated systems on Cape Cod with similar configurations and flushing rates (e.g. Namskaket Marsh, Little Namskaket Marsh). The spatial distribution of nitrogen release/uptake by the sediments of Scorton Creek ranged from net release in the organic rich upper tidal reaches, 25.5 mg N m^{-2} d^{-1} to net uptake throughout the lower sandy tributary creeks to Long Hill and Scorton Shore, -19.9 and -12.7 mg N m^{-2} d^{-1}, respectively, with the high velocity larger tidal creeks with little organic matter accumulation showing little net release, -3.0 mg N m^{-2} d^{-1}. The same pattern was observed in another Cape Cod Bay salt marsh, Namskaket Creek, where the upper marsh creeks also showed a net release, 45.4 mg N m^{-2} d^{-1}, and the lower creek areas net uptake, -21.2 mg N m^{-2} d^{-1}. Again in Little Namskaket Marsh, the upper marsh creeks showed a net release, 64.5 mg N m^{-2} d^{-1}, and the lower creek areas net uptake, -7.8 mg N m^{-2} d^{-1}.

Within the Scorton Creek Estuary rates of uptake were also similar to other salt marsh systems on Cape Cod. For example, net nitrogen uptake in the lower salt marsh creeks (-3.0 to -19.9 mg N m^{-1} d^{-1}) was similar to that observed for the salt marsh areas in the Centerville River System (-4.5 to -13.2 mg N m^{-1} d^{-1}) and Cockle Cove Salt Marsh, Chatham (MEP Centerville River Final Nutrient Technical Report 2006, MEP Cockle Cove Technical Memorandum-Howes et al. 2006) and the lower basin of the Little River marsh system (-3.1 mg N m^{-1} d^{-1}). The net release rates from the upper tidal reaches are similar and comparable to other systems. For example, the upper reaches of the Herring River wetland system (9.7-10.5 mg N m^{-1} d^{-1}), Wild Harbor River (1.4 mg N m^{-1} d^{-1}) and salt marsh dominated portions of the Back River (Bourne) and the Slocums and Little River Estuaries (Dartmouth) support similarly small net release rates of 6.5 mg N m^{-2} d^{-1} and 4.6-9.0 mg N m^{-2} d^{-1}, respectively.

The sediments within the Scorton Creek Estuary showed nitrogen fluxes typical of similarly structured systems with low to moderate watershed nitrogen loading in the region and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the low moderate nitrogen loading to this system and it relatively high flushing rate.

Net nitrogen release rates for use in the water quality modeling effort for the component sub-basins of the Scorton Creek Salt Marsh System (Chapter VI) are presented in Table IV-6. There was a clear spatial pattern of sediment nitrogen flux, with net release from the sediments of the upper reach of the tidal creek and net uptake by the sediments of the lower tidal creek region. The sediments within the Scorton Creek System showed nitrogen fluxes typical of similarly structured systems with low to moderate watershed nitrogen loading and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the low nitrogen loading to this system and it relatively high flushing rate.

Table IV-6. Rates of net nitrogen return from sediments to the overlying waters of the Scorton Creek Embayment System. These values are combined with the basin areas to determine total nitrogen mass release/uptake in the water quality model (see Chapter VI). Measurements represent July - August rates.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1}) Mean	S.E.	N	Station i.d. *
Scorton Creek Embayment System				
Upper Marsh Creeks	25.5	11.7	4	1,2,3,4
Long Hill Tidal Creek	-19.9	8.0	1	8
Scorton Shores Channel	-12.7	0.2	2	9,10
Lower Main Channel	-3.0	1.3	2	6,7

* Station numbers refer to Figure IV-6.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of a hydrodynamic model for the Scorton Creek estuary system. 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 the Scorton Creek system 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.

A hydrodynamic study was performed for the Scorton Creek system. A section of a topographic map in Figure V-1 shows the general study area. The entire Scorton Creek system has a surface coverage of 450 acres, and is comprised of marsh plain and marsh creeks. Circulation in the Scorton Creek system is dominated by tidal exchange with Cape Cod Bay. The Creek is connected to the Bay through a ~90-foot-wide inlet that is armored on the east and west side and that empties onto a broad tidal flat over East Sandwich Beach to the west of the inlet and Scorton Shores to the east.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry and tide data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Scorton Creek was performed to determine the present variation of the main Harbor basin and channel depths throughout the system. These tide data were necessary to run and calibrate the hydrodynamic model of the system.

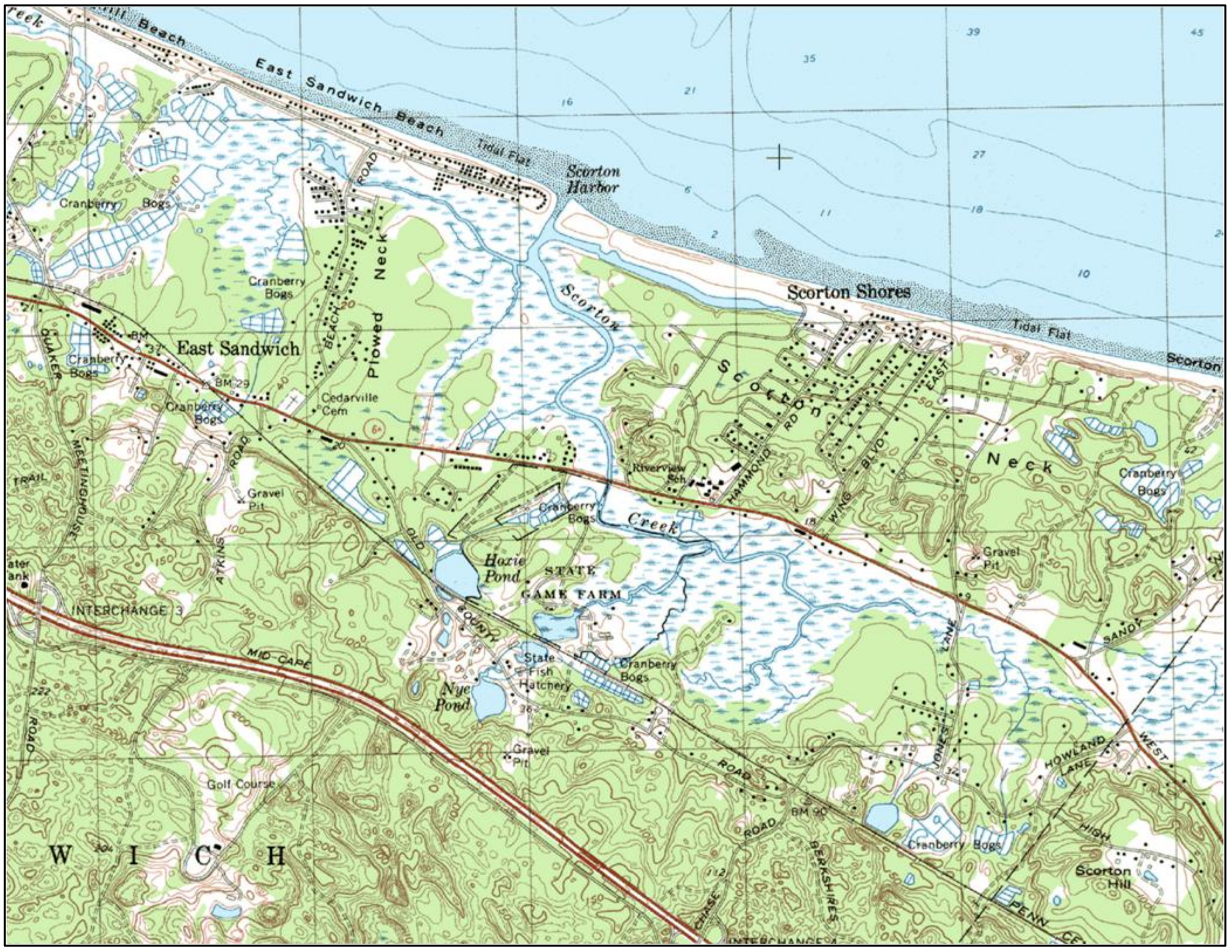


Figure V-1. Topographic map detail of the Cape Cod Bay embayment of Scorton Creek, Sandwich, Massachusetts.

A numerical hydrodynamic model of the Scorton Creek system was developed in the second portion of this analysis. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from offshore, in Cape Cod Bay, were used to define the open boundary conditions that drive the circulation of the model at the system inlet, and data from the TDR station within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

The calibrated computer model of Scorton Creek and the associated tidal creek system was used to compute the flushing rates of selected sub-sections. Though water quality in an embayment cannot be directly inferred by use of computed flushing rates alone, they can serve as useful indicators of embayment flushing performance relative to other areas in the same system. The ultimate utility of this hydrodynamic model is as input into a constituent transport model, where water quality constituents like nitrogen are modeled to determine the real water quality dynamics of a system.

V.2 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of the Scorton Creek estuary. Bathymetry were collected throughout the system so that it could be accurately represented in the computer hydrodynamic model and water quality model of the system. In addition to the bathymetry, tide data were also collected at offshore and inshore locations (within the marsh system), to run the circulation model with real tides, and also to calibrate and verify its performance.

V.2.1 Bathymetry Data Collection

Bathymetric data was collected in the Sandwich salt water embayments (Scorton Creek and Sandwich Harbor), as part of a larger town-wide effort, over the course of two separate surveys, occurring in the Summer of 2008. The surveys employed an Odem HydroTrac fathometer mounted on a 16 foot motor skiff. Positioning data were collected using a differential GPS. The position data from the GPS and the depth data from the fathometer were recorded digitally in real time using the Hypack hydrographic survey software package. Where practical, predetermined survey transects were followed at regular intervals. Marsh channels in the upper portion of the marsh areas were also surveyed, where depths allowed the passage of the survey boat. Collected bathymetry data was tide-corrected to account for the change in water depths as the tide level changed over the survey period. The tide-correction is performed using tide data collected while the survey was run.

Marsh plain topography data on the marsh plane were also collected in the Scorton Creek system using a level and stadia rod. The topography data were necessary to set the elevation of the marsh plain included in the hydrodynamic model. Additional topographic and bathymetric data was gathered from NOAA's Coastal Services Center which provide the 2007 US Army Corps of Engineers (USACE) Topographic and Bathymetric Lidar Dataset for the region. The complied elevation dataset is shown in Figure V-2.

V.2.2 Tide Data Collection and Analysis

Tide data were collected for this study and in the summer 2008. Tide data were collected by Temperature Depth Recorders (TDR), which are strain-gage instruments that record pressure and ambient temperature. Each gage was leveled to a standard survey vertical datum (NAVD88).

The instruments deployed in this study were set to record data at 10-minute intervals, for at least a 29-day period. This period of time is necessary to capture the bi-monthly variation between the spring and neap ranges of the tides in the Sandwich embayments. Although a one-month record likely does not include extreme high or low tides, it does provide an accurate basis for typical tidal conditions governed by both lunar and solar motion. For numerical modeling of hydrodynamics, the typical tide conditions associated with a one-month record are appropriate for driving tidal flows within the estuarine system.

The deployment locations for the Scorton Creek embayment gages are shown in Figure V-3. Tides were measured offshore and at stations inside of the Scorton Creek estuary system. The total Cape Cod Bay deployment spanned late July to early September. For comparison, the complete tide data record is plotted in Figure V-4, from each gage deployed in the Scorton Creek.

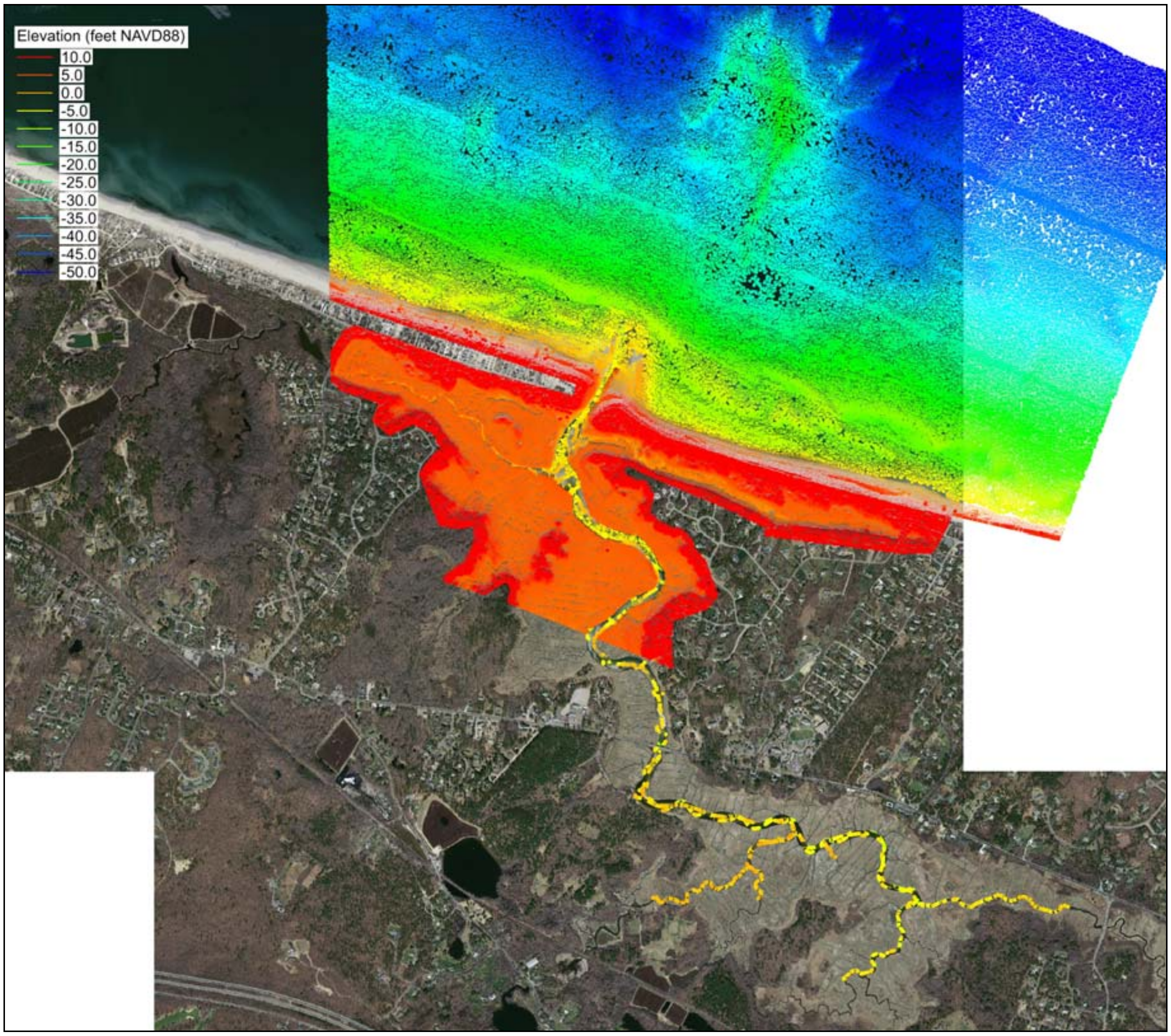


Figure V-2. Bathymetry data used with the RMA-2 hydrodynamic model. Points are colored to represent the bottom elevation relative to NAVD. The data sources used to develop the grid mesh are the 2008 bathymetry survey, and the NOAA Lidar data.

Plots of the tide data from the gages are shown in Figure V-4 for the entire 43-day deployment. The spring-to-neap variation in tide range is discernible in these plots. The data record during a period of neap tides shows a minimum range of approximately 0.7 feet; occurring on August 11. Between July 30 and August 5, there is a period of spring tides, where the maximum range of 12.9 feet occurs on August 2.

A visual comparison between tide elevations offshore and at the different stations in the marsh system shows that the tide amplitude does significantly change between stations. This reduction in tidal amplitude throughout the system, described as tidal attenuation, indicates a frictional damping along the upper marsh channels of Scorton Creek and tidal restrictions through some of the culvert/bridge structures within the system.

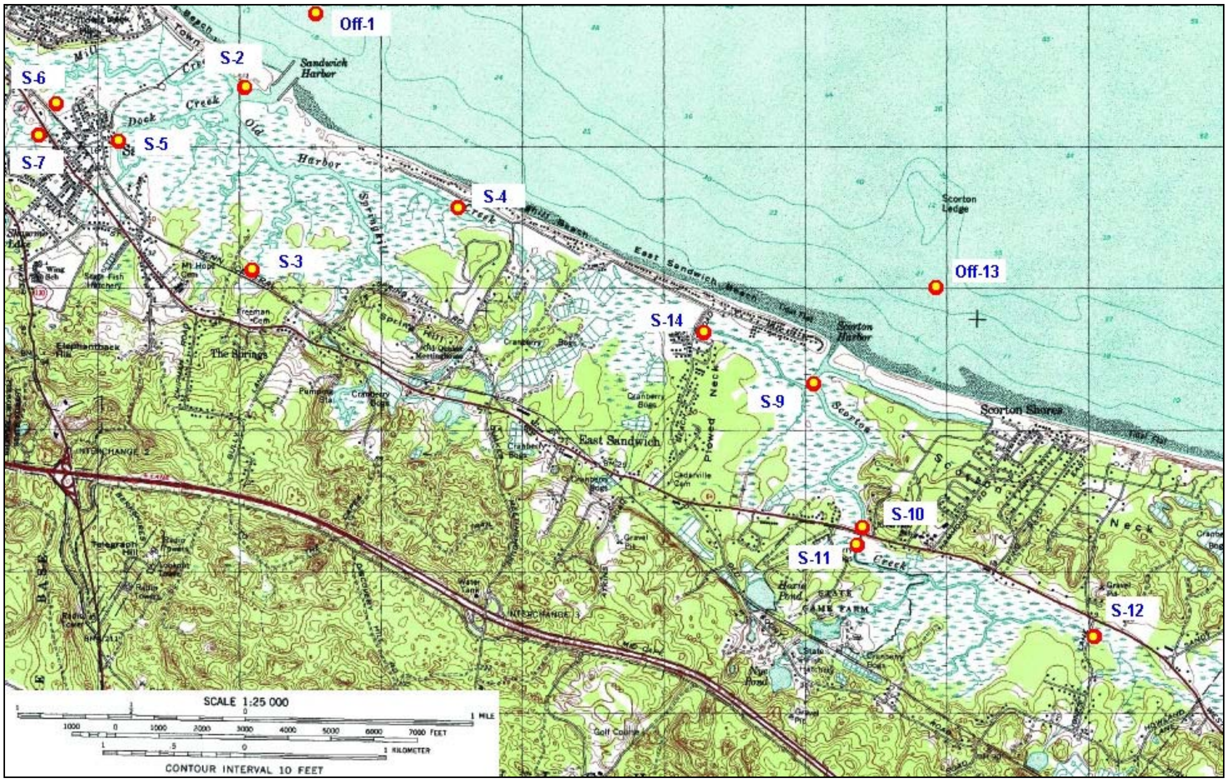


Figure V-3. Tide gage locations for the Scorton Creek Estuary of Sandwich.

To better quantify the changes to the tide from the inlet to the inside of the system, the standard tide datums were computed from the 43-day record. These datums are presented in Table V-1. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available; however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (NAVD) 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.

Frictional damping not only affects the range of the observed tide, it also causes a time lag in the arrival of high and low tide. Significant change in tide range is evident in Table V-1; however the peak of high tides does not change significantly, it is the bottom of the low tides that is truncated due to the depths of the marsh channels within the Scorton Creek system. The similar high tides is shown throughout the system in Figure V-5; showing that damping is not significant within system. Gage S-12 is shown in Figure V-5 to illustrate both attenuation and time lag. Table V-1 shows the mean tide range observed within Scorton Creek is approximately 7 feet while offshore within Cape Cod Bay it is approximately 9.5 feet.

A more thorough harmonic analysis of the tidal time series was also performed to produce tidal amplitude and phase of the major tidal constituents, and provide 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 each system.

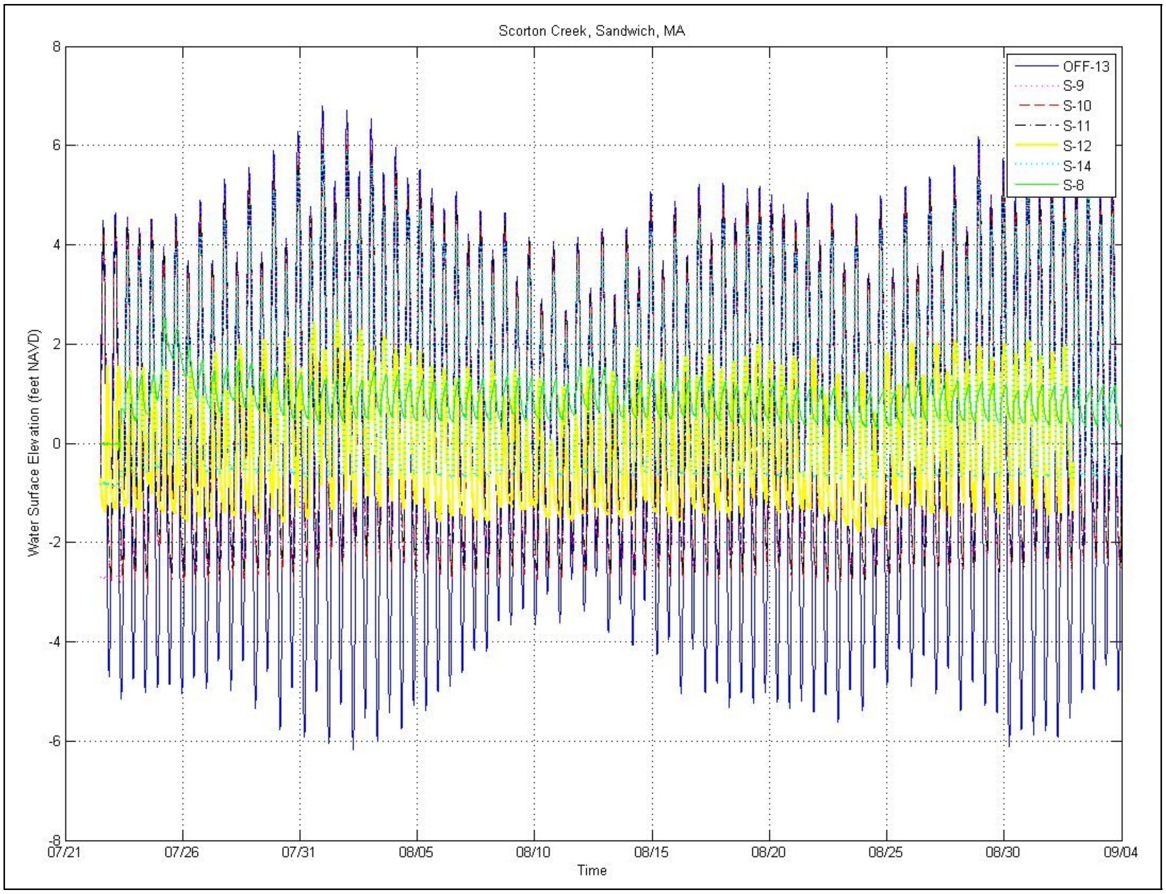


Figure V-4. Complete TDR records for gages deployed for the Scorton Creek system, Cape Cod Bay during late summer of 2008.

Table V-1. Tide datums computed from 43-day records collected offshore and in the Scorton Creek system in summer of 2008. Datum elevations are given relative to the NAVD vertical datum.

Tide Datum	Cape Cod Bay S-13 (010113) (feet)	Scorton Creek S-9 (013077) (feet)	Scorton Creek S-10 (9488) (feet)	Scorton Creek S-11 (010114) (feet)	Scorton Creek S-14 (013079) (feet)
Maximum Tide	6.79	6.62	6.33	6.25	6.57
MHHW	5.22	5.20	5.07	5.09	5.19
MHW	4.73	4.75	4.65	4.68	4.74
MTL	-0.03	1.15	1.13	1.18	2.40
MLW	-4.79	-2.45	-2.40	-2.32	0.05
MLLW	-5.07	-2.51	-2.49	-2.41	0.03
Minimum Tide	-6.17	-2.63	-2.66	-2.56	-0.03
Mean Range	9.52	7.19	7.05	7.00	4.79

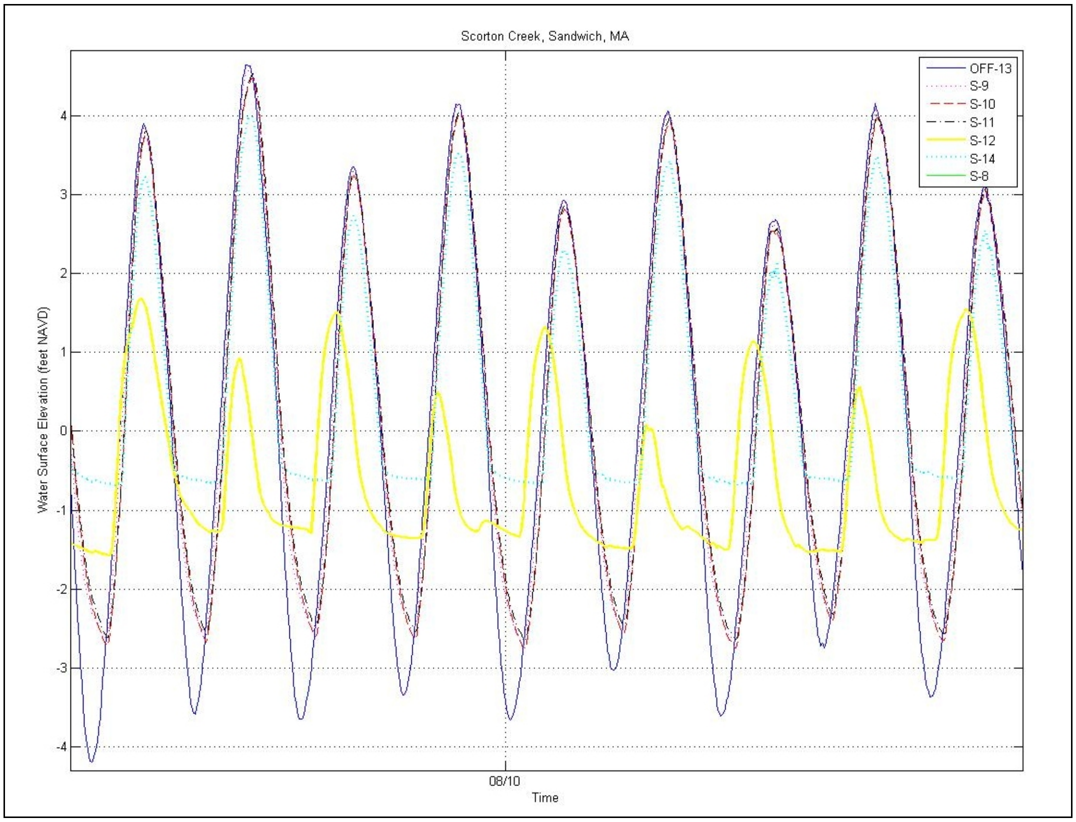


Figure V-5. Close-up of TDR record of tides recorded in Cape Cod Bay and Scorton Creek embayments connected to Cape Cod Bay.

A harmonic analysis was performed on the time series from each gage location. The results of this analysis are presented in Table V-2. 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. The observed astronomical tide is the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-6. Table V-2 presents the amplitudes of eight tidal constituents in the Scorton Creek estuary and Cape Cod Bay. The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to the tide signal at all of the gage deployment locations. The total range of the M_2 tide is twice the amplitude, e.g., 8.6 feet for the tide offshore in Cape Cod Bay. The K_1 and O_1 constituents represent diurnal tides that occur once daily associated with the sun and moon respectively.

Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, contribute significantly to the total tide signal, with amplitudes that are typically each about 10% of the total observed tide. 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 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 period for the M_6), results from frictional attenuation of the M_2 tide in shallow water.

Data in Table V-2 show how the constituents vary as the tide propagates into the upper reaches of the marsh system. Note the reduction in the M_2 amplitude between Cape Cod Bay and farther up into the marsh of Scorton Creek. The main M_2 and K_1 tidal constituents are smaller inside the marsh due to truncation of the lower portion of the tide. This truncation occurs because the channel bottoms in the system are high enough to limit the lower range of the tide. This effect is also apparent in Figure V-4. Frictional damping also is evident by the harmonic constituents, where drag on the channel sides and bottom causes some energy transfer from the M_2 to the higher order M_4 and M_6. This is seen by how the M_4 and M_6 increase in amplitude inside the systems, compared the constituents calculated for the Cape Cod Bay station.

Table V-2. Major tidal constituents determined for gage locations in the Scorton Creek Cape Cod Bay embayment of Sandwich, for the time period July through September 2008.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Cape Cod Bay, S-13	4.361	0.078	0.144	0.727	0.877	0.472	0.454	0.127
Scorton Creek, S-9	3.424	0.451	0.098	0.466	0.544	0.388	0.432	0.327
Scorton Creek, S-10	3.327	0.482	0.113	0.453	0.502	0.392	0.449	0.380
Scorton Creek, S-11	3.319	0.476	0.125	0.453	0.506	0.387	0.453	0.379
Scorton Creek, S-14	2.182	0.749	0.078	0.379	0.409	0.325	0.347	0.255

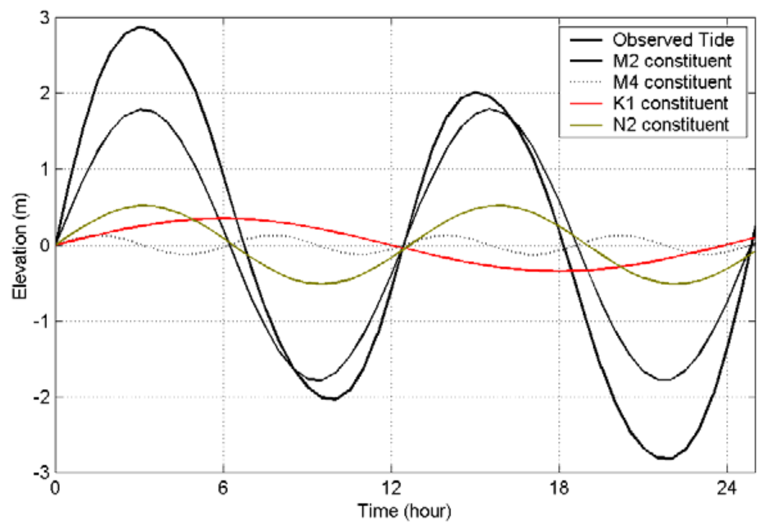


Figure V-6. Example of an observed astronomical tide as the sum of its primary constituents.

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. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow. The results of an analysis to determine the energy distribution (or variance) of the original water elevation time series for the Scorton Creek embayment system is presented in Table V-3 compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 23 constituents determined by the harmonic analysis). Subtracting the tidal signal from the original elevation time series resulted in 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. Figure V-7 shows the comparison of the measured tide from Scorton Creek, with the computed astronomical (predicted) tide resulting from the harmonic analysis, and the resulting non-tidal residual signal.

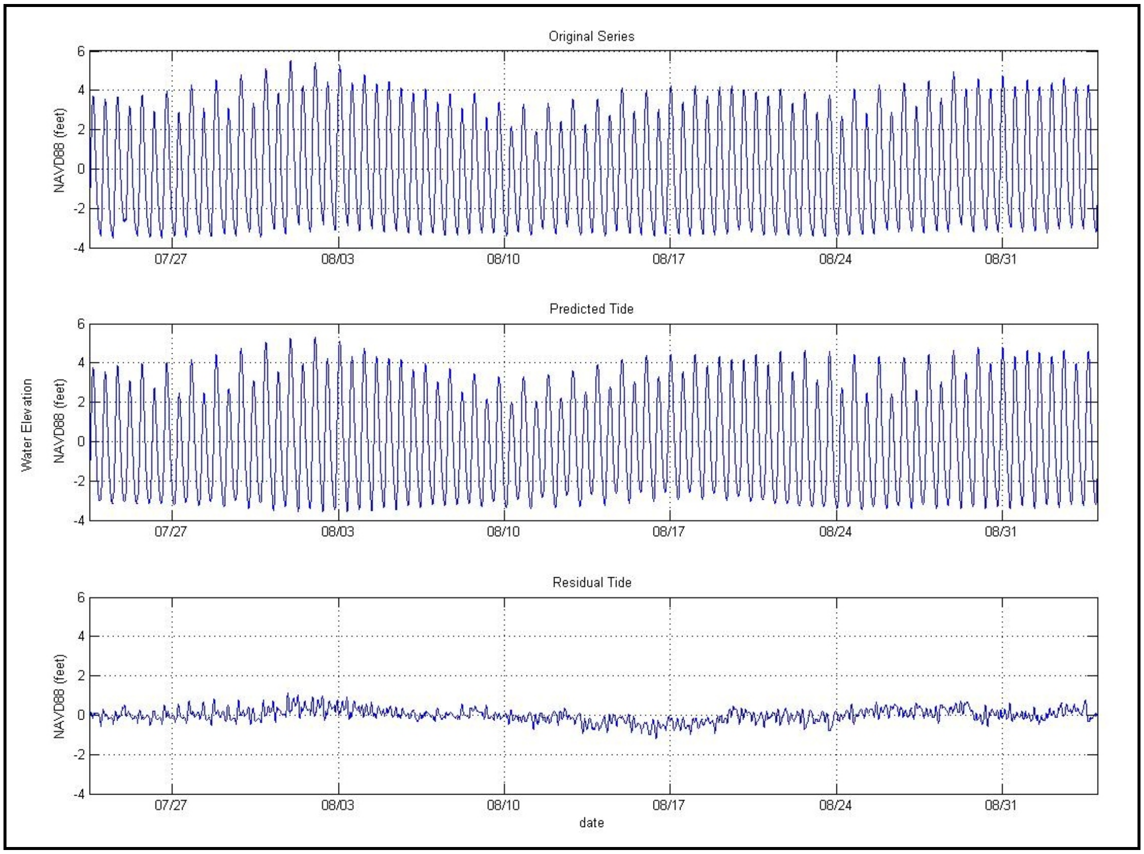


Figure V-7. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determine in the harmonic analysis of the Scorton Creek gage data S-10. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

Table V-3 shows that the variance of tidal energy was smallest in the data record from the offshore gage (S-13) located in Cape Cod Bay when compared to all the other deployed gages. This is to be expected since the tide range is the greatest at this location, compared to the other gage sites located within the Scorton Creek system. In general, the energy of the signal decreases with distance from the offshore gage, with the lowest energy found in upper regions of each system. The analysis also shows that tides are responsible for typically more than 90% of the water level changes in each system; wind effects in these data sets were negligible.

Table V-3. Percentages of Tidal versus Non-Tidal Energy for the Cape Cod Bay embayment of Scorton Creek in Sandwich.

TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Cape Cod Bay, S-13	0.062	99.4	0.6
Scorton Creek, S-9	0.080	98.8	1.2
Scorton Creek, S-10	0.103	98.4	1.6
Scorton Creek, S-11	0.108	98.3	1.7
Scorton Creek, S-14	0.096	97.1	2.9

V.3 HYDRODYNAMIC MODELING

For the Scorton Creek system, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing. The particular computed model code 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 and rivers. Applied Coastal staff members have utilized RMA-2 for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, Chatham embayments (Kelley, *et al*, 2001), Orleans Estuaries: Namskaket, Little Namskaket and Rock Harbor (Kelley, *et al*, 2007), Falmouth "finger" Ponds (Ramsey, *et al*, 2000), and Barnstable Harbor (Wood, *et al*, 1999).

V.3.1 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 are met.

V.3.2 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 2009 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of the system based on the tide gage data collected offshore East Sandwich Beach. Once the grid and boundary conditions were set, the model was calibrated to ensure that each computer model accurately represents the tidal dynamics of the real physical system. Various friction and eddy viscosity coefficients were adjusted, through several (20+) model calibration simulations to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid generation

The grid generation process was aided by the use of the SMS package. 2009 digital aerial orthophotos and recent bathymetry survey data were imported to SMS to facilitate the construction of finite element grids to represent each of the modeled estuaries. The aerial photographs were used to determine the land boundary of the system, as well as determine the surface coverage of salt marsh resources.

The completed grid mesh for the Scorton Creek system has a total of 4688 nodes that make up 1753 quadrilateral, triangular, and one dimensional elements. The maximum depth included in the main portion of the marsh model is -11.65 feet NAVD, in the main marsh channel at the downstream side of the Route 6 Bridge. Bathymetry data were interpolated to the developed finite element mesh of the system. The smaller marsh channels and marsh plain were represented through the use of one dimensional elements with additional off-channel storage used to represent the extensive marsh plains within Scorton Creek. The elevations of the marsh plain were based on elevations obtained from the Lidar survey of the marsh plain.

The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties of the modeled Scorton Creek Estuary. Areas of marsh were included through the use of off channel storage along one dimensional elements along the marsh channels because they represent a large portion of the total area of this system, and have a significant effect on system hydrodynamics. 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 in marsh creeks and channels was designed to provide a more detailed analysis in these regions of rapidly varying flow (Figure V-8). Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in harbor’s main basin and on the upper reaches of the modeled marsh plains. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

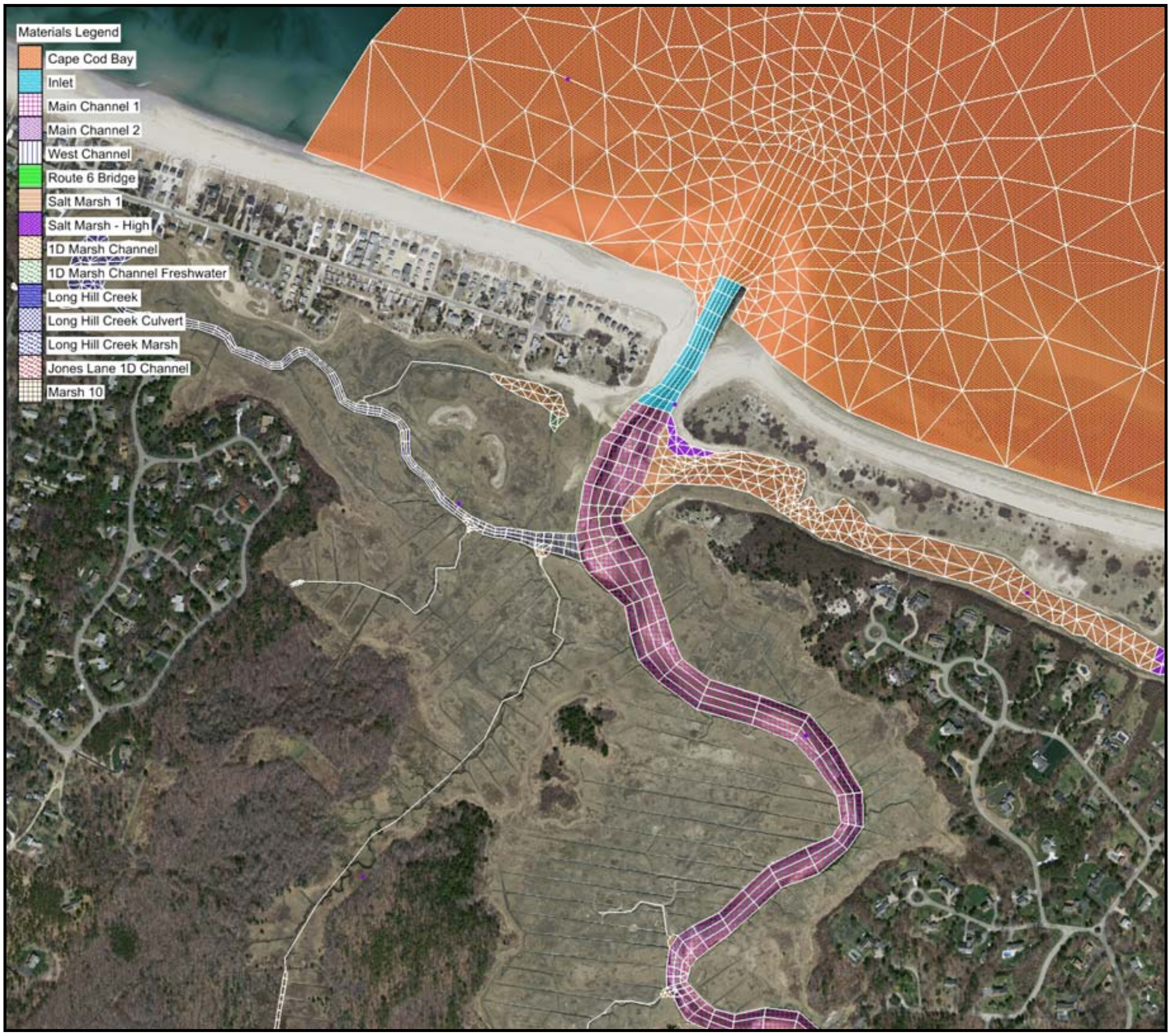


Figure V-8. Plot of hydrodynamic model grid mesh for Scorton Creek. Colors are used to designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.3.2.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2 model of Scorton Creek: 1) "slip" boundaries, 2) tidal elevation boundaries, and 3) flow 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. TDR measurement tide data collected offshore East Sandwich Beach provided the required data. The rise and fall of the tide in Cape Cod Bay is the primary driving force for estuarine circulation in the modeled system. Dynamic (time-varying) model simulations specified a new water surface elevation at the model open boundary every model time step (10 minutes); therefore, the model time step selected for the system corresponded to the time step of the tide data record. Flow boundaries were specified at the tide gate on Long Hill Creek and at Jones Lane where is crosses the marsh system. The flow boundaries were based on field measurements of the flow at those locations.

V.3.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the Scorton Creek model 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 required a close match between the modeled and measured tides. Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, an approximate 7-day period was modeled to calibrate the model based on dominant tidal constituents discussed previously in the data collection section. For Scorton Creek, the calibration period began July 27, 2008 at 12:00 (EDT). The 7-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for phase (time) lag and amplitude (height) damping of dominant tidal constituents

The calibrated and verified model was used to analyze existing detailed flow patterns and compute residence times. The flushing analysis is based on the 15 day period beginning July 27, 2008 at 12:00 (EDT). 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 15-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.3.2.3.1 Friction coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a characteristic 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 0.02 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 channel found in the lower portion of Scorton Creek, versus the meandering marsh creek channels of the upper marsh, 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.

Table V-4. Manning's Roughness and eddy viscosity coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-8.

System Embayment	Bottom Friction	eddy viscosity lb-sec/ft^2
Cape Cod Bay	0.027	50.0
Inlet	0.028	30.0
Main Channel 1	0.028	40.0
Main Channel 2	0.030	35.0
West Channel	0.028	40.0
Route 6 Bridge	0.032	25.0
Salt Marsh	0.031	75.0
Salt Marsh High	0.033	100.0
1D Marsh Channel	0.034	70.0
1D Marsh Channel FW	0.034	150.0
Long Hill Creek	0.026	30.0
Long Hill Creek Culvert	0.034	100.0
Long Hill Creek Marsh	0.033	100.0
Jones Lane 1D Channel	0.034	70.0
Marsh	0.034	150.0

V.3.2.3.2 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 30 and 150 lb-sec/ft^2. Higher values (up to 200 lb-sec/ft^2) were used on the marsh plain, and in shallow inlets, to ensure numerical stability.

V.3.2.3.3 Marsh porosity processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain included in the model. Cyclically wet/dry areas of the marsh will tend to store waters as the tide begins to ebb and then slowly release water as the water level drops within the creeks and channels. This store-and-release characteristic of these marsh regions was partially responsible for the distortion of the tidal signal, and the elongation of the ebb phase of the tide. On the flood phase, water rises within the channels and creeks initially until water surface elevation reaches the marsh plain, when at this point the water level remains nearly constant as water 'fans' out over the marsh surface. The rapid flooding of the marsh surface corresponds to a flattening out of the tide curve approaching high water.

Marsh porosity is a feature of the RMA-2 model that permits the modeling of hydrodynamics in marshes. This model feature essentially simulates the store-and-release capability of the marsh plain by allowing grid elements to transition gradually between wet and dry states. This technique allows RMA-2 to change the ability of an element to hold water, like squeezing a sponge. The marsh porosity feature of RMA-2 is typically utilized in estuarine systems where the marsh plain has a significant impact on the hydrodynamics of a system, such as the modeled Scorton Creek estuary which include marsh resources.

V.3.2.4 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model output for the measured data was achieved using the aforementioned values for friction and turbulent exchange. Figures V-9 through V-13 illustrate sections the 7-day simulation periods for the calibration model. Modeled (dashed line) and measured (solid line) tides are illustrated at each model location with a corresponding TDR. The ability of the model to represent the measured tides during the calibration period is demonstrated by the high degree of correlation (R^2) shown in Table V-5 for all stations.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in the system embayments. Four tidal constituents were selected for constituent comparison: the K_1, M_2, M_4 and M_6. Measured tidal constituent amplitudes are shown in Table V-5 for the calibration and verification simulations. The constituent amplitudes shown in this table differ from those in Table V-2 because constituents were computed for only the separate 7-day sub-sections of the 42-days represented in Table V-2. In Table V-6, error statistics are shown for the calibration.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The errors associated with tidal constituent amplitude for both the calibration and verification simulations were on the order of 0.00 to 0.30 ft, which is within the accuracy of the tide gages. Time lag errors for the main estuary reach were less than the time increment resolved by the model and tide data (10 minutes), indicating good agreement between the model and data.

V.4 FLUSHING CHARACTERISTICS

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through inlet of Scorton Creek, the primary mechanism controlling estuarine water quality within these systems is tidal exchange. For example, a rising tide offshore Scorton Creek creates a slope in water surface from the ocean into the upper reaches of the system. Consequently, water flows into (floods) the embayment. Similarly, each estuary drains to the ocean 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.

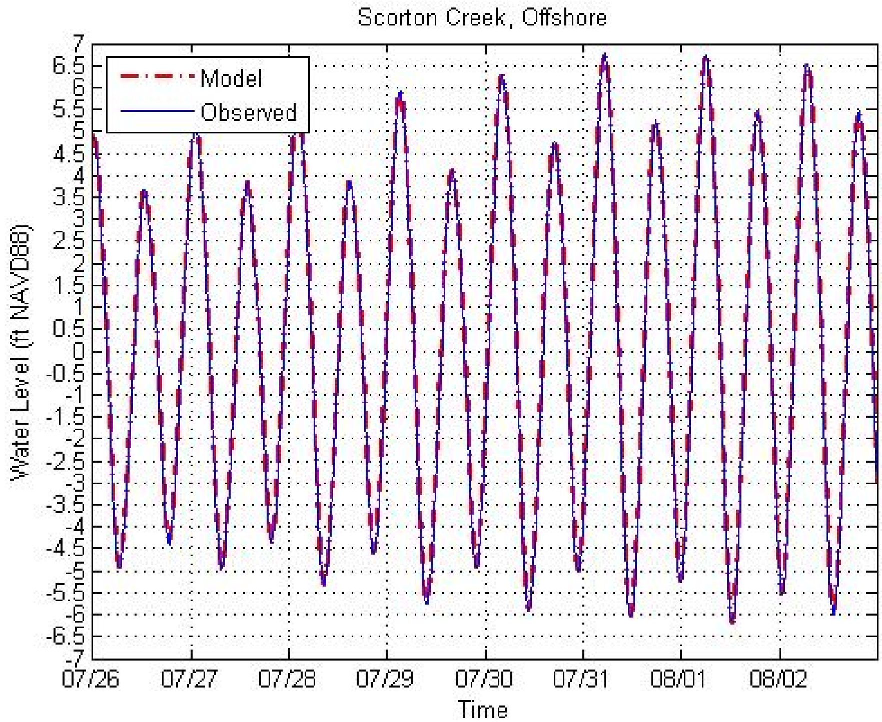


Figure V-9. Comparison of model output and measured tides for the TDR location offshore in Cape Cod Bay (S-13) for the final calibration model run.

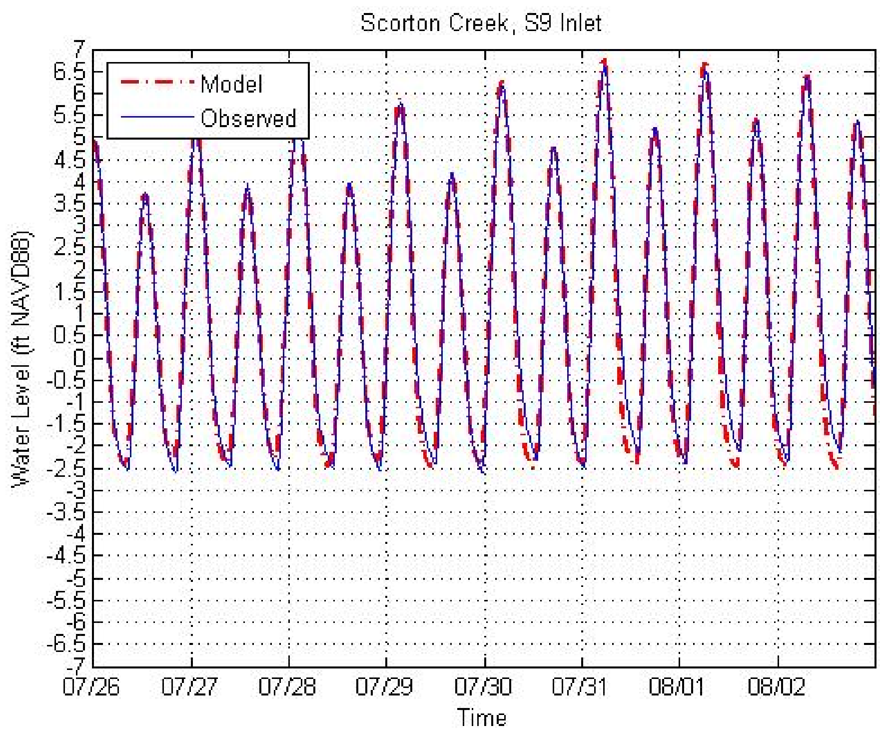


Figure V-10. Comparison of model output and measured tides for the TDR location in the inlet of Scorton Creek (S-9) for the final calibration model run.

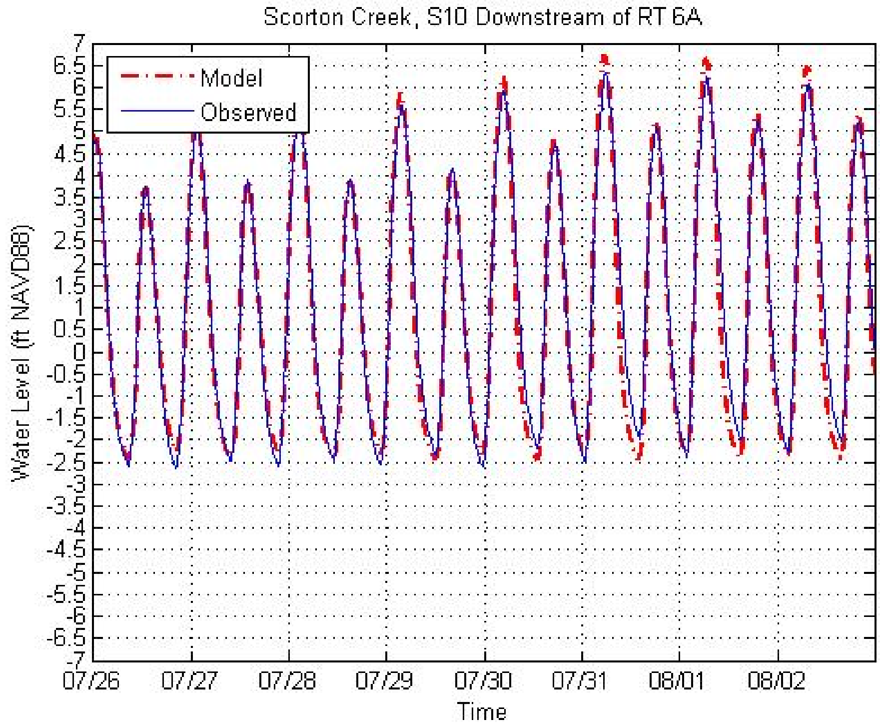


Figure V-11. Comparison of model output and measured tides for the TDR location downstream of the Route 6 Bridge (S-10) for the final calibration model run.

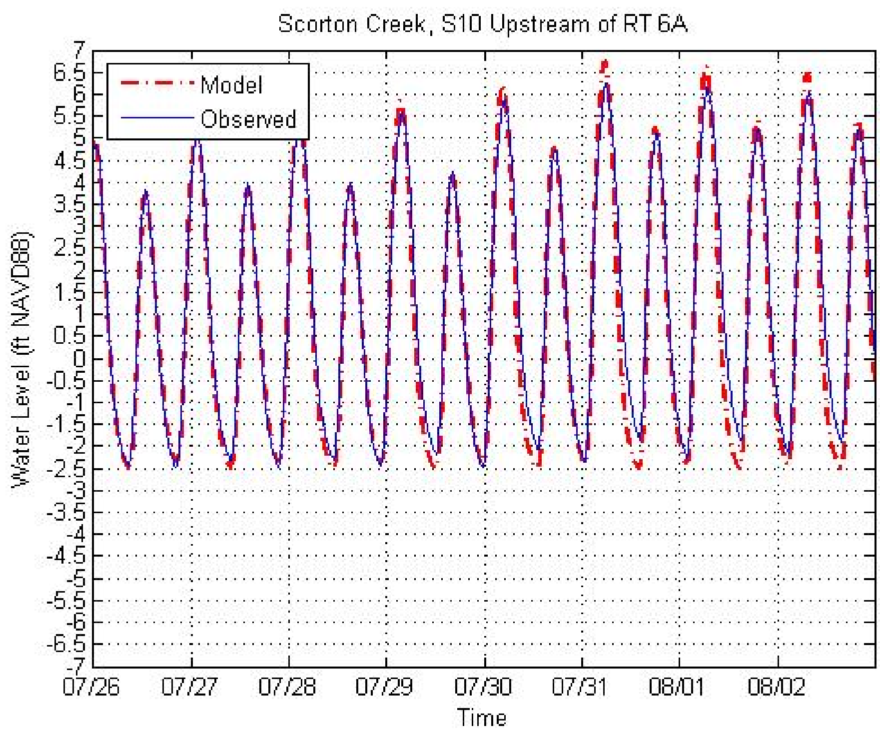


Figure V-12. Comparison of model output and measured tides for the TDR location upstream of the Route 6 Bridge (S-11) for the final calibration model run.

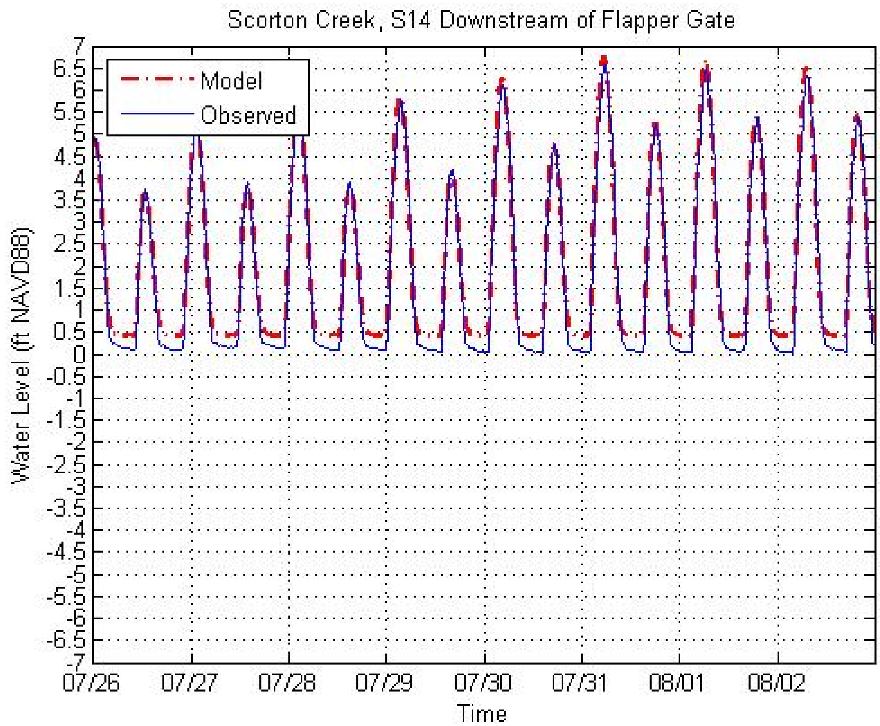


Figure V-13. Comparison of model output and measured tides for the TDR location downstream of the tide gate on Longhill Creek (S-14) for the final calibration model run.

Table V-5. Error statistics for the Scorton Creek hydrodynamic model, for model calibration.	
	Calibration
Location	R^2
Cape Cod Bay, S-13	1.00
Scorton Creek, S-9	0.98
Scorton Creek, S-10	0.96
Scorton Creek, S-11	0.95
Scorton Creek, S-14	0.97

Table V-6. Tidal constituents for measured water level data and calibrated model output for Scorton Creek during model calibration time period (calculated values based on 16 significant digits).					
Model calibration run					
Location	Constituent Amplitude (ft)				Phase (rad)
	M_2	M_4	M_6	K_1	ϕM_2
Cape Cod Bay*, S-13	5.05	0.10	0.17	0.84	-2.27
Scorton Creek, S-9	3.80	0.55	0.10	0.69	-2.16
Scorton Creek, S-10	3.72	0.80	0.17	0.72	-1.96
Scorton Creek, S-11	3.76	0.79	0.17	0.73	-1.96
Scorton Creek, S-14	2.32	0.91	0.12	0.56	-2.12
Measured tide during calibration period					
Location	Constituent Amplitude (ft)				Phase (rad)
	M_2	M_4	M_6	K_1	ϕM_2
Cape Cod Bay*, S-13	5.09	0.11	0.18	0.85	-2.36
Scorton Creek, S-9	3.71	0.57	0.13	0.69	-2.16
Scorton Creek, S-10	3.58	0.53	0.16	0.72	-2.01
Scorton Creek, S-11	3.54	0.50	0.18	0.72	-1.99
Scorton Creek, S-14	2.54	0.86	0.09	0.59	-2.16
Error					
Location	Error Amplitude (ft)				Phase error (min)
	M_2	M_4	M_6	K_1	ϕM_2
Cape Cod Bay*, S-13	0.04	0.00	0.01	0.01	-9.27
Scorton Creek, S-9	-0.09	0.02	0.04	0.00	1.79
Scorton Creek, S-10	-0.14	-0.27	-0.01	-0.01	-3.59
Scorton Creek, S-11	-0.22	-0.29	0.00	-0.01	-2.58
Scorton Creek, S-14	0.22	-0.05	-0.03	0.03	-3.18

*model open boundary

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 sub-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). To compute system residence time for a sub-embayment, the tidal prism of the sub-embayment replaces the total system tidal prism value in the above equation.

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 provide the only means of reducing the high nutrient levels.

The rate of pollutant/nutrient loading and the quality of water outside the estuary 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 modeling of Chapter VI provides this critical tool to evaluate the complex mechanisms governing estuarine water quality in Scorton Creek.

The mean volume and mean tidal prism is presented in Table V-7. Residence times were averaged for the tidal cycles comprising a representative 15 day period, and are listed in Table V-8 for Scorton Creek. The modeled time period used to compute the flushing rates was the same as the model calibration period, and included the transition from neap to spring tide conditions. Since the 15-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 sub-embayments.

Table V-7. Embayment mean volume and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Cape Cod Bay Embayments		
Scorton Creek	10,814,000	14,760,000

Table V-8. Computed System and Local residence times for Scorton Creek

Embayment	System Residence Time (days)
Scorton Creek	0.4

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 Scorton Creek 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 Embayments

Extensive field measurements and hydrodynamic modeling of the system were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated hydrodynamic model representing the transport of water within the Scorton Creek system. Files of node locations and node connectivity for the RMA-2 model grids 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 the 7 day period beginning July 26, 2008 1800 EST. This period corresponds to that used in the flushing analysis presented in Chapter V. 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 Embayments

Three primary nitrogen loads to sub-embayments 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 the Scorton Creek system, consisting of the background concentrations of total nitrogen in the waters entering from Cape Cod Bay. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayments

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 the area map presented 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 are the minimum required to provide a baseline for MEP analysis. Three years of data (collected between 2005 and 2007) were available for stations monitored by SMAST.

Table VI-1. Town of Sandwich water quality monitoring data, and modeled Nitrogen concentrations for the Scorton Creek 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	2005 mean	2006 mean	2007 mean	Mean	s.d. all data	N	model min	model max	model average
Scorton Creek	SC1	0.672	0.498	0.445	0.547	0.179	17	--	--	--
Scorton Creek	SC2	0.399	0.429	0.390	0.408	0.122	15	0.441	0.492	0.469
Scorton Creek	SC3	0.620	0.596	0.619	0.611	0.128	16	0.629	0.681	0.654
Scorton Creek	SC4	0.594	0.373	0.431	0.464	0.133	19	0.436	0.575	0.513
Scorton Creek	SC5	0.673	0.607	0.752	0.677	0.186	18	0.662	0.664	0.663
Scorton Creek	SC6	0.312	0.525	0.412	0.416	0.209	18	0.350	0.867	0.486
Scorton Creek	SC7	0.835	0.998	1.212	0.989	0.234	14	--	--	--
Scorton Creek	SC8	0.463	0.359	0.365	0.396	0.074	18	0.379	0.567	0.513
Scorton Creek	SC9	0.328	0.448	0.335	0.361	0.155	16	0.346	0.522	0.449
Scorton Creek	SC10	0.704	0.611	0.614	0.635	0.246	16	0.616	0.637	0.627
Scorton Creek	SC11	0.302	0.358	0.313	0.323	0.060	16	0.322	0.484	0.405
Scorton Creek	SC12	0.462	0.327	0.419	0.406	0.142	16	0.359	0.537	0.474
Scorton Creek - Inlet	SC13	0.300	0.294	0.338	0.310	0.039	16	0.318	0.500	0.391

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

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 Scorton Creek estuarine 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 simulate the fluid dynamics of Scorton Creek. 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 members have utilized RMA-2 for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, Chatham embayments (Kelley, *et al*, 2001), Orleans Estuaries: Namskaket, Little Namskaket and Rock Harbor (Kelley, *et al*, 2007), Falmouth "finger" Ponds (Ramsey, *et al*, 2000), and Barnstable Harbor (Wood, *et al*, 1999).

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 (based on the USGS watersheds), as well as the measured bottom sediment nitrogen fluxes. 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 Scorton Creek system.

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 the sub-embayments of the Scorton Creek system.

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 Scorton Creek also were 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 ground water flowing into the system from the watersheds. Scorton Creek has thirteen watersheds contributing to the groundwater flow, the combined flow rate into the system is 14.4 ft^3/sec (35,178 m^3/day).

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 7 tidal-day (174 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 Scorton Creek model.

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, and 4) the point source input developed from measurements of the discharge of the freshwater portion of Scorton Creek. Nitrogen loads from the marsh's watersheds were distributed along the length of the system. Loads from the watershed were applied at elements within the marsh creeks represented in the model. Benthic regeneration loads were distributed among another sub-set of grid cells in the marsh creeks.

The loadings used to model present conditions in the Scorton Creek system are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the creek channel surface area coverage (excluding the marsh plain) representing each separate core, 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. Benthic flux loads can be positive or negative. In Scorton Creek, the sum of all benthic flux in the system is negative, indicating that on average the bottom sediments of the creek channels and harbor basin are a net nitrogen sink.

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 the Cape Cod region offshore the entrance to the Creek was set at 0.318 mg/L, based on SMAST data collected in the summers of 2003 through 2005.

Table VI-2. Sub-embayment and surface water loads used for total nitrogen modeling of the Scorton Creek system, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions** for the listed sub-embayments.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Scorton Creek	32.403	0.395	-0.065
Surface Water Sources			
Long Hill Creek	4.584	--	--
Jones Lane	2.753	--	--

VI.2.4 Model Calibration

Calibration of the total nitrogen model of Scorton Creek 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 estuarine embayments of Cape Cod 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). The final values of *E* used in each sub-embayment of the modeled system 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 the 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 the Scorton Creek estuary system.

Embayment Division	E m^2/sec
Cape Cod Bay	25.0
Inlet	23.0
Main Channel 1	22.0
Main Channel 2	20.0
West Channel	20.0
Route 6 Bridge	20.0
Salt Marsh	12.0
Salt Marsh High	8.0
1D Marsh Channel	20.0
1D Marsh Channel FW	0.5
Long Hill Creek	2.0
Long Hill Creek Culvert	0.5
Long Hill Creek Marsh	4.0
Jones Lane 1D Channel	8.0
Marsh	1.0

Comparisons between model output and measured nitrogen concentrations are shown in plots presented in Figure VI-3. 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 water quality monitoring stations. The emphasis during calibration was to concentrate on representing the conditions measured at the data collection stations.

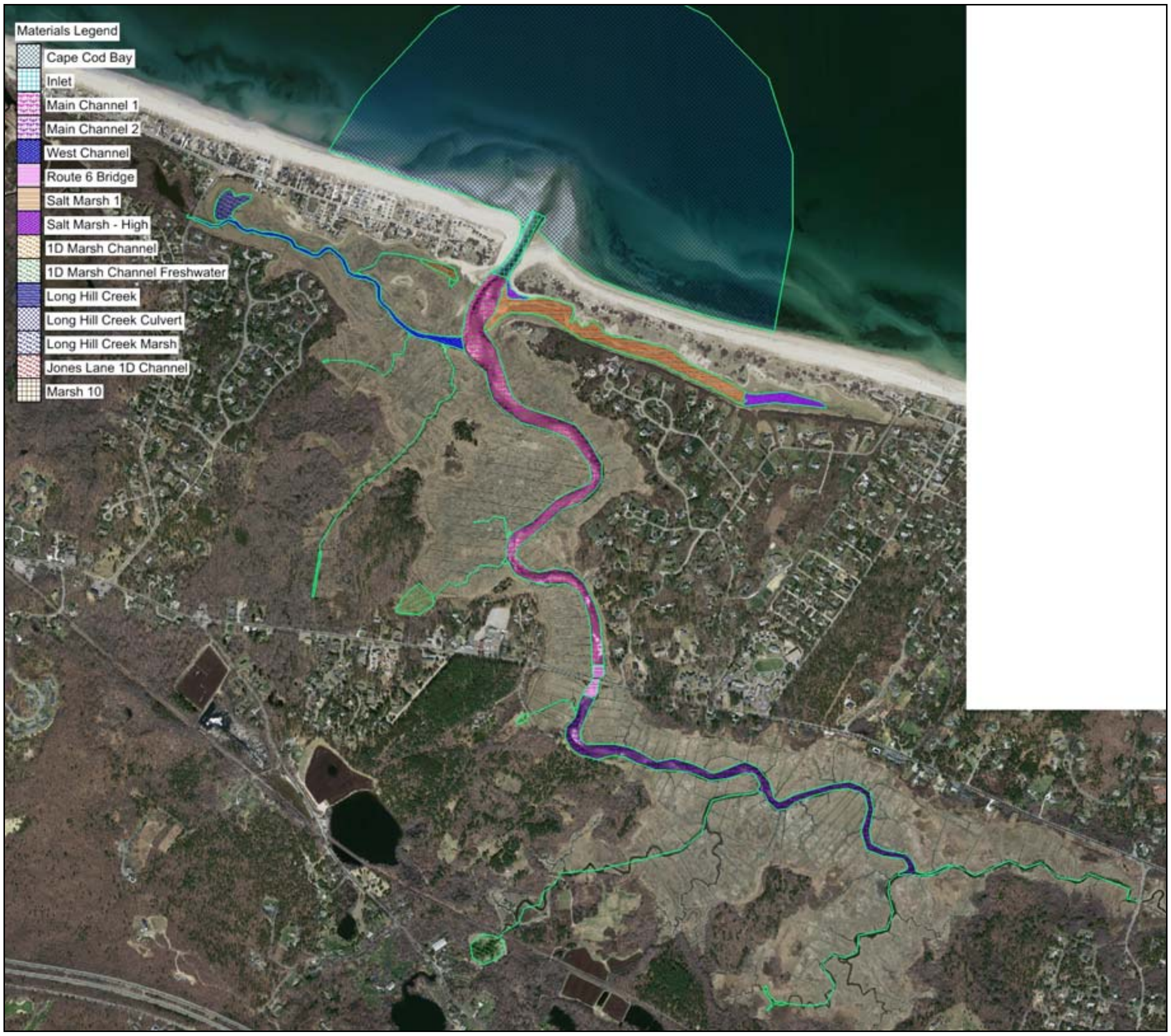


Figure VI-2. Map of Scorton Creek System water quality model longitudinal dispersion coefficients. Color patterns designate the different areas used to vary model dispersion coefficient values.

For model calibration, the maximum modeled TN was compared to mean measured TN data values, at each water-quality monitoring station. Usually, the mid-point between maximum modeled TN and average modeled TN is compared to mean measured TN data values, but the water quality samples for Scorton Creek were not carried out at the standard mid ebb tide as has been done on previous MEP evaluations. The water quality samples were collected at or before high tide by the Town of Sandwich. Thus the calibration target was set at the modeled minimum TN to reflect the different procedures used in collecting the monitoring data. Water quality monitoring stations SC1 and SC7 were not used during the calibration since both were outside the modeling domain.

Also presented in this figure are unity plot comparisons of measured data versus modeled target values for the system. The model provides a good representation for the Scorton Creek System, with rms error of 0.03 mg/L and an R^2 correlation coefficient of 0.95.

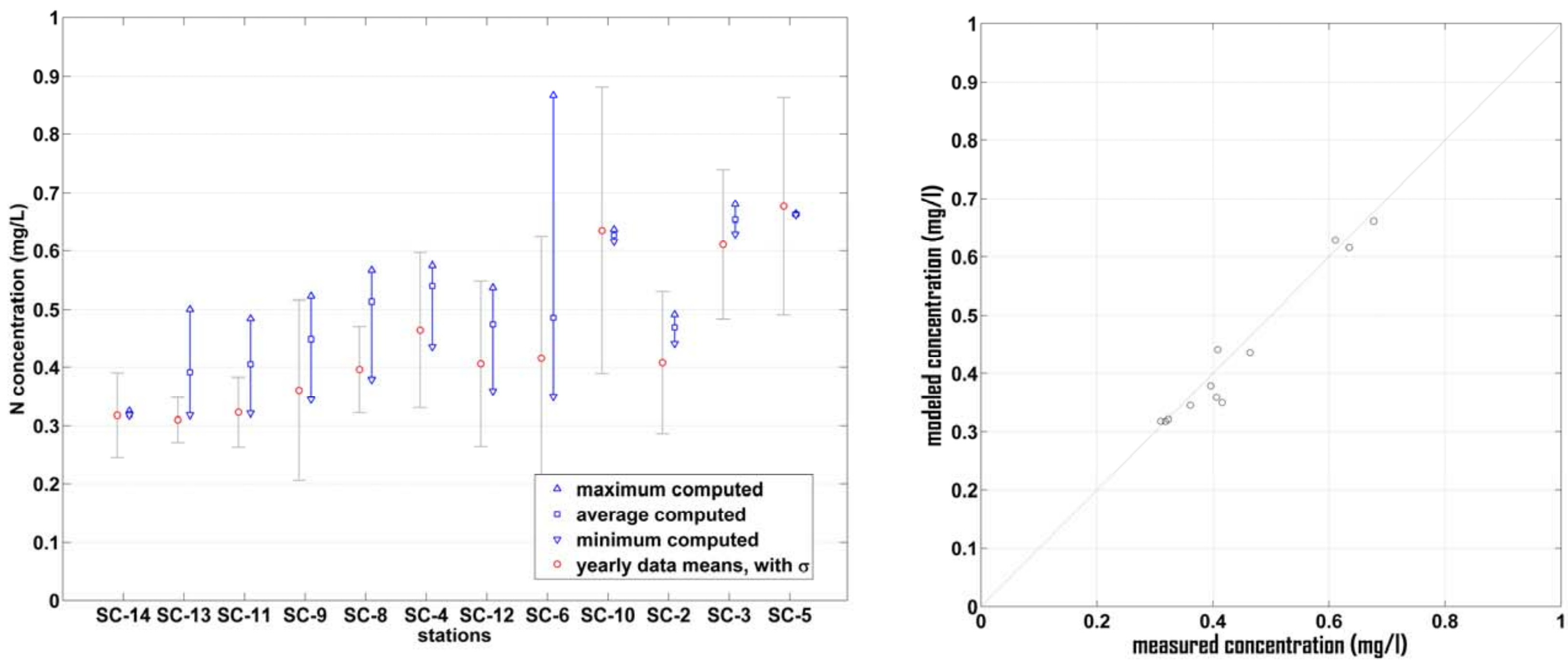


Figure VI-3. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Scorton Creek. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the 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.

A contour plot of calibrated model output is shown in Figure VI-4 for Scorton Creek System. 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.

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 Scorton Creek 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, at the freshwater stream discharges, and groundwater inputs. The open boundary salinity was set at 31.0 ppt. For groundwater inputs salinities were set at 0 ppt. Surface water stream flow rates for the streams were the same as those used for the total nitrogen model, as presented earlier in this section. The groundwater inputs used for the system was 14.4 ft^3/sec (35,178 m^3/day). Groundwater flows were distributed evenly in the model through the use of 1-D element input points positioned along each model's land boundary.

Comparisons of modeled and measured salinities are presented in Figure VI-5, with contour plots of model output shown in Figure VI-6. For salinity the calibration target is usually the mid-point between minimum modeled salinity and average modeled salinity which is compared to mean measured Salinity data values, but the water quality samples for Scorton Creek were not carried out at the standard mid ebb tide as has been done on previous MEP evaluations. The water quality samples were collect at or before high tide by the Town of Sandwich. Thus the calibration target was set at the modeled maximum salinity to reflect the different procedures used in collecting the monitoring data.

Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in Scorton Creek estuary system. The rms error of the models was 1.78 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 system.

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.

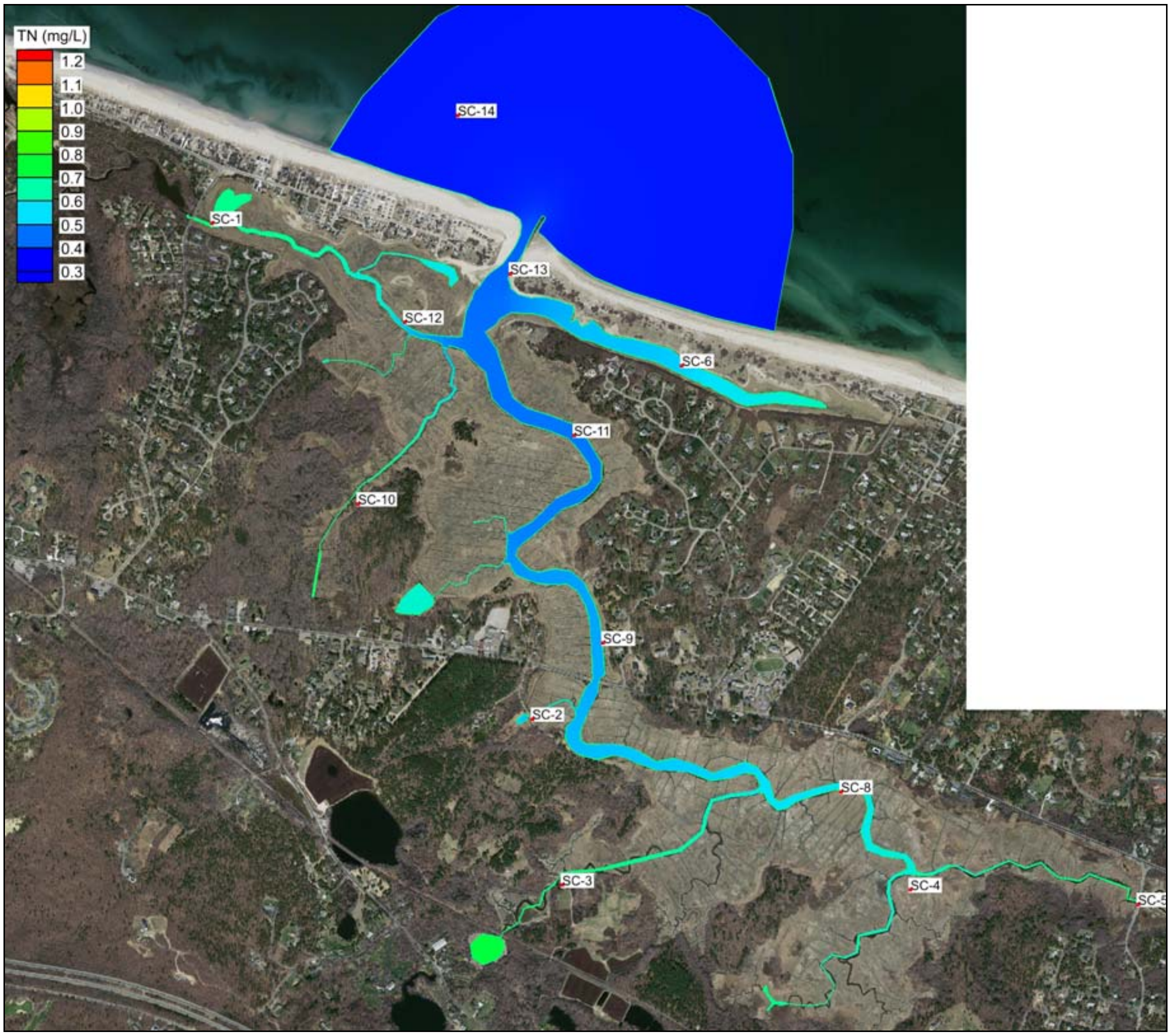


Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Scorton Creek system.

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others under build-out conditions. The build-out scenario indicates that there would be approximately a 24% increase in watershed nitrogen load to Scorton Creek as a result of potential future development, while of the surface water sources both have increases of different percentages. For the no load scenarios, a majority of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions.

For the build-out scenario, a breakdown of the total nitrogen load entering each sub-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 *vice versa*.

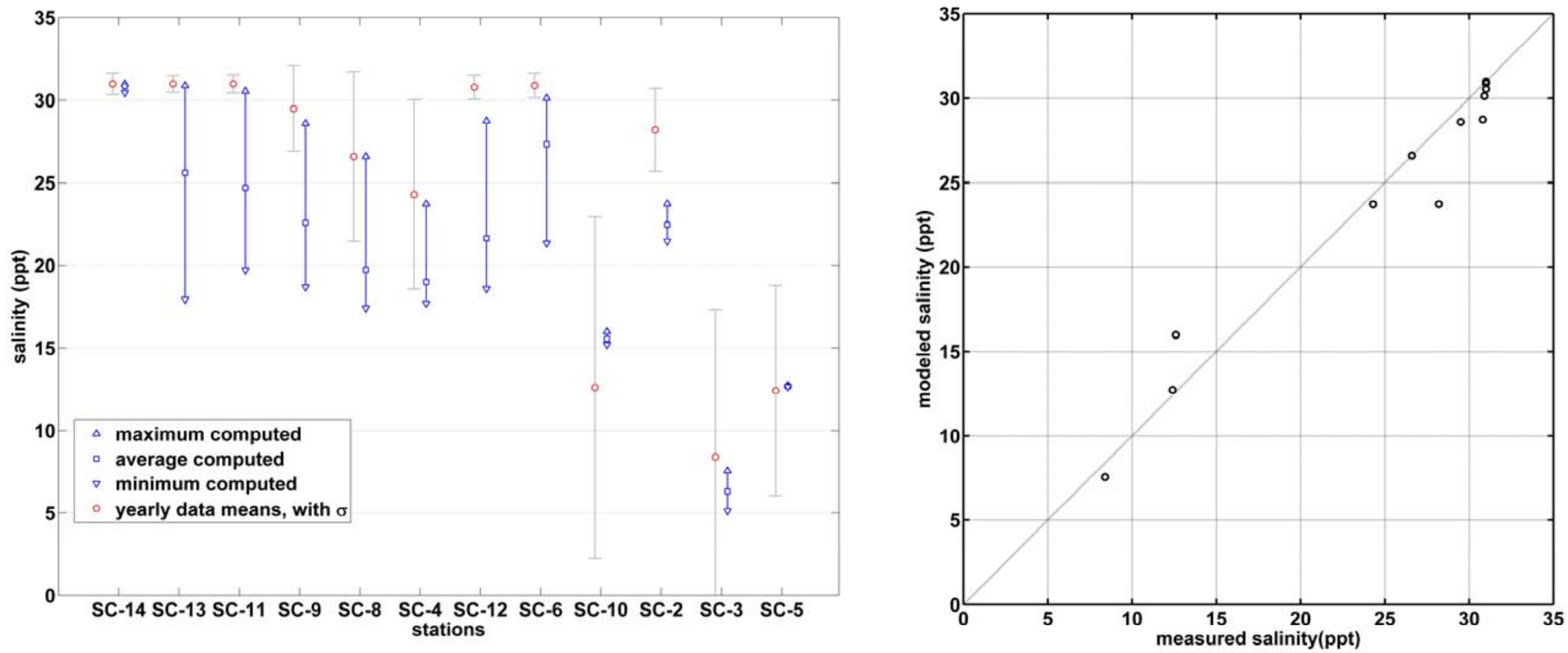


Figure VI-5. Comparison of measured and calibrated model output at stations in Scorton Creek. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle 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.

Figure VI-6. Contour plot of modeled salinity (ppt) in Scorton Creek.

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 Scorton Creek 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
Scorton Creek	32.403	40.079	+23.7%	3.885	-88.0%
Surface Water Sources					
Long Hill Creek	4.584	7.592	+65.6%	0.241	-94.7%
Jones Lane	2.753	3.321	+20.6%	0.162	-94.1%

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 Scorton Creek 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)
Scorton Creek	40.079	0.395	-0.004
Surface Water Sources			
Long Hill Creek	7.592	--	--
Jones Lane	3.321	--	--

Following development of the nitrogen loading estimates for the build-out scenario, the nutrient gradients within Scorton Creek were determined based on the incoming build-out load from the watershed and the offshore boundary condition (TN Concentration) from the water quality monitoring program to determine nitrogen concentrations across the system (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) remained nearly identical to the existing conditions modeling scenarios. Total N concentrations increased the most in the upper portions of the system, with the largest change at the Scorton Creek freshwater discharge and the least change occurring in the creek mouth near the system's inlet to Cape Cod Bay. In Table VI-6, the percent change *P* over background is calculated as:

$$P = (N_{scenario}-N_{present})/(N_{present}-N_{background})$$

where N is the nitrogen concentration at the indicated monitoring station for present conditions and the loading scenario (i.e., build-out in this case), and also in Cape Cod Bay (background).

Color contours of the TN gradient under build-out conditions are present in Figure VI-7. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-4, 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 Scorton Creek system (0.318 mg/L). The sentinel threshold station is in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Cape Cod Bay	SC-14	0.320	0.320	+0.2%
Scorton Creek - Inlet	SC-13	0.391	0.410	+5.9%
Scorton Creek	SC-11	0.405	0.425	+6.1%
Scorton Creek	SC-9	0.449	0.477	+8.8%
Scorton Creek	**SC-8**	**0.513**	**0.555**	**+13.1%**
Scorton Creek	SC-4	0.540	0.588	+15.2%
Scorton Creek	SC-12	0.474	0.560	+26.9%
Scorton Creek	SC-6	0.486	0.530	+14.1%
Scorton Creek	SC-10	0.627	0.710	+26.3%
Scorton Creek	SC-2	0.469	0.501	+10.2%
Scorton Creek	SC-3	0.654	0.716	+19.7%
Scorton Creek	SC-5	0.663	0.803	+44.2%

Figure VI-7. Contour plot of modeled total nitrogen concentrations (mg/L) in the Scorton Creek system, for projected build-out loading conditions.

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") scenarios 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 present conditions and build-out conditions, 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.

Following development of the nitrogen loading estimates for the build-out scenario, the nutrient gradients within Scorton Creek were determined based on the incoming build-out load from the watershed and the offshore boundary condition (TN Concentration) from the water quality monitoring program to determine nitrogen concentrations across the system. Again, total nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) 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 greatest reductions occurring in the upper portions of the system. Results for the system are shown pictorially in Figure VI-8.

Table VI-7. "**No anthropogenic loading**" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of the Scorton Creek 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)
Scorton Creek	3.885	0.395	-0.274
Surface Water Sources			
Long Hill Creek	0.241	--	--
Jones Lane	0.162	--	--

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 Scorton Creek system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). The sentinel threshold station is in bold print.

Sub-Embayment	monitoring station	present (mg/L)	no load (mg/L)	% change
Cape Cod Bay	SC-14	0.320	0.318	-0.5%
Scorton Creek - Inlet	SC-13	0.391	0.330	-15.8%
Scorton Creek	SC-11	0.405	0.335	-17.2%
Scorton Creek	SC-9	0.449	0.346	-22.8%
Scorton Creek	**SC-8**	**0.513**	**0.357**	**-30.4%**
Scorton Creek	SC-4	0.540	0.356	-34.1%
Scorton Creek	SC-12	0.474	0.296	-37.5%
Scorton Creek	SC-6	0.486	0.336	-30.8%
Scorton Creek	SC-10	0.627	0.353	-43.7%
Scorton Creek	SC-2	0.469	0.351	-25.1%
Scorton Creek	SC-3	0.654	0.429	-34.4%
Scorton Creek	SC-5	0.663	0.070	-89.4%

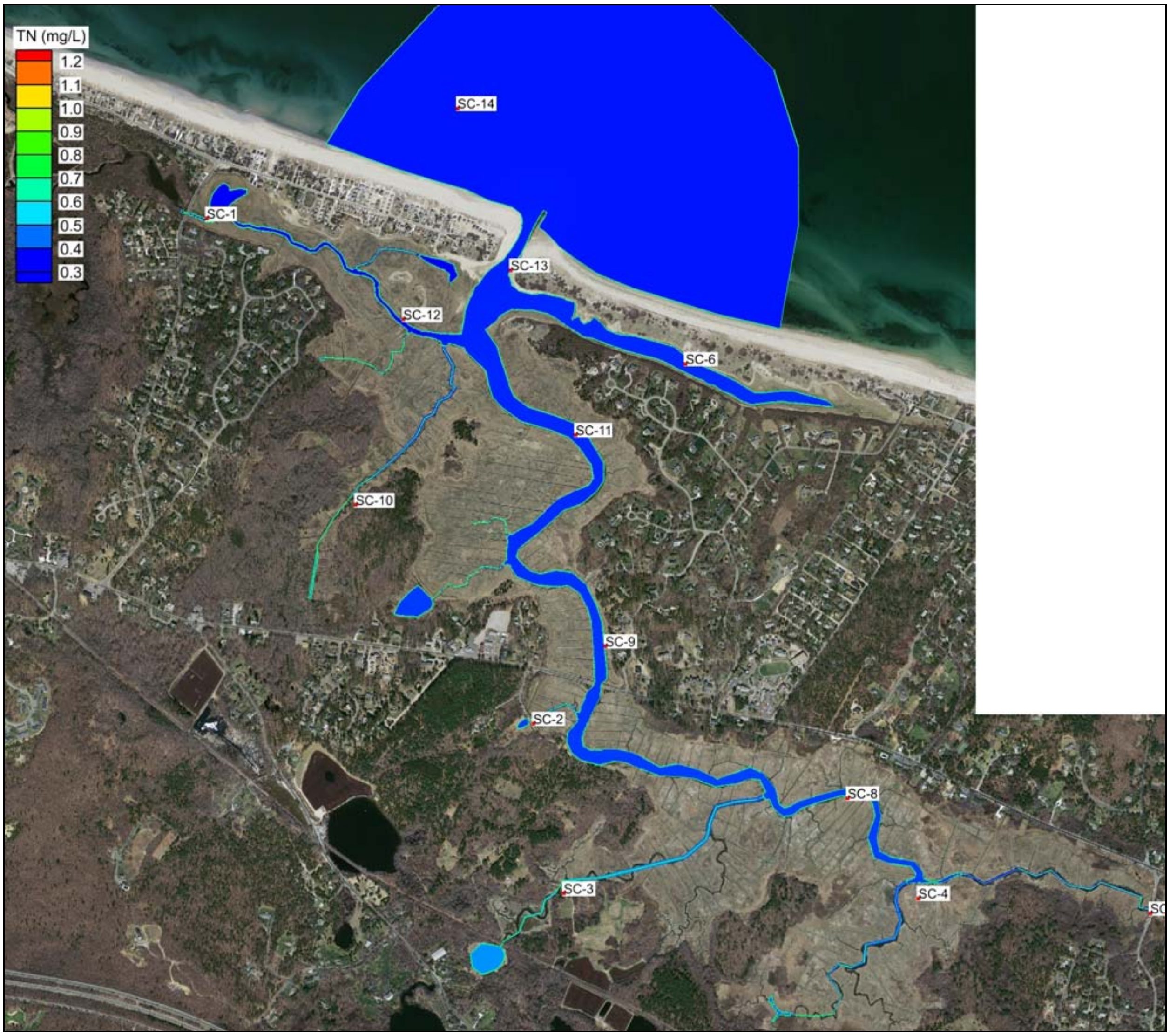


Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in Scorton Creek, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gaged 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 Scorton Creek Estuary in the Town of Sandwich, MA, our assessment is based upon data from the water quality monitoring database (2005, 2006, 2007) developed by the Town of Sandwich with technical support from the Coastal Systems Program (UMASS-SMAST), MassDEP surveys of eelgrass distribution as available (typically 1951, 1995, 2001), benthic animal communities (fall 2006), sediment characteristics (summer 2006), and dissolved oxygen records (summer 2006). 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 this system (Chapter VIII). It should be noted that nitrogen enrichment occurs through 2 primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or low rates of flushing due to restriction of tidal exchange with the low nitrogen waters of Cape Cod Bay. Scorton Creek has increasing nitrogen loading from the associated watersheds from shifting land-uses and has periodic restriction of tidal exchange. Fundamentally, restrictions of tidal exchange increase the sensitivity of an estuary to nitrogen inputs, however, this is more so the case in a classic embayment setting as opposed to systems like Scorton Creek that are functional salt marshes with higher assimilative capacities for nitrogen than open water embayments.

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 threshold determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll-*a* (Section VII.2), (2) as available from the MassDEP Eelgrass Mapping Program, 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 autonomous dissolved oxygen sensors in upper and lower Scorton Creek at locations that would be representative of the dissolved oxygen conditions at critical locations in the estuary, namely in the main channel of Scorton Creek upgradient of the inlet, a location in the tributary tidal creek that extends along the backside of the barrier beach towards Scorton Shores and in the central channel of the upper marsh above the Route 6A bridge (Figure VII-2). The dissolved oxygen moorings were deployed to record the frequency and duration of low oxygen conditions during the critical summer period. To the extent the information exists and is appropriate for the system being evaluated, 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. Unfortunately, MassDEP mapping of the eelgrass beds in the near shore waters of Sandwich Harbor was conducted (MassDEP Eelgrass Mapping Program, C. Costello), but surveying was not completed within Scorton Creek, as it has not historically supported eelgrass. Surveying completed by the SMAST-MEP Technical Team in the summer of 2006 also did not reveal the presence of any eelgrass, as would be expected in a tidal salt marsh system. As a result, temporal changes in eelgrass distribution could not provide a basis for evaluating recent increases (nitrogen loading) in nutrient enrichment of Scorton Creek. As a result, nutrient threshold determination was based strongly on results from the dissolved oxygen and chlorophyll mooring data as well as the benthic infaunal community characterization. Prior to evaluating nitrogen thresholds, analysis of inorganic N/P molar ratios within the watercolumn of the upper and lower tidal creeks of the Scorton Creek Estuary was conducted. The results support the contention that nitrogen is the nutrient to be managed in this estuary, as the ratio in lower and upper creeks are 9 and 5, respectively, clearly below the Redfield Ratio value (16) indicating that nitrogen additions will increase phytoplankton production in this estuary.

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 4 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters be able to maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Scorton Creek Estuary are currently 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 and that it is the designated water quality that is the target of TMDL's generated under the U.S. Clean Water Act. It is through the MEP and TMDL processes that site specific management targets are developed and under the Town's CWMP that management alternatives are designed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate estuaries 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 oxygen 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 bottom of the main tidal creeks within key regions of the Scorton Creek Salt Marsh (Figure VII-2). The dissolved oxygen sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployments. In addition periodic calibration samples were collected at the depth of each sensor 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 a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Scorton Creek Estuary were collected during the summer of 2006.

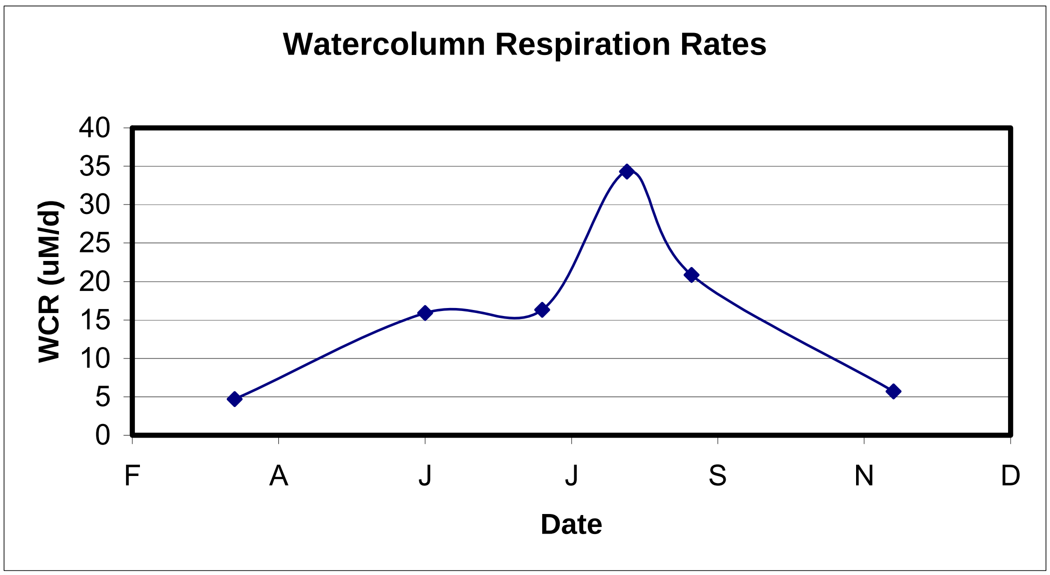


Figure VII-1. Example of typical average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System, Cape Cod (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter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

Similar to other estuaries in southeastern Massachusetts, the Scorton Creek Estuary evaluated in this assessment showed high frequency variation, related primarily to diurnal influences and to a lesser extent 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 at 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 37 day 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.

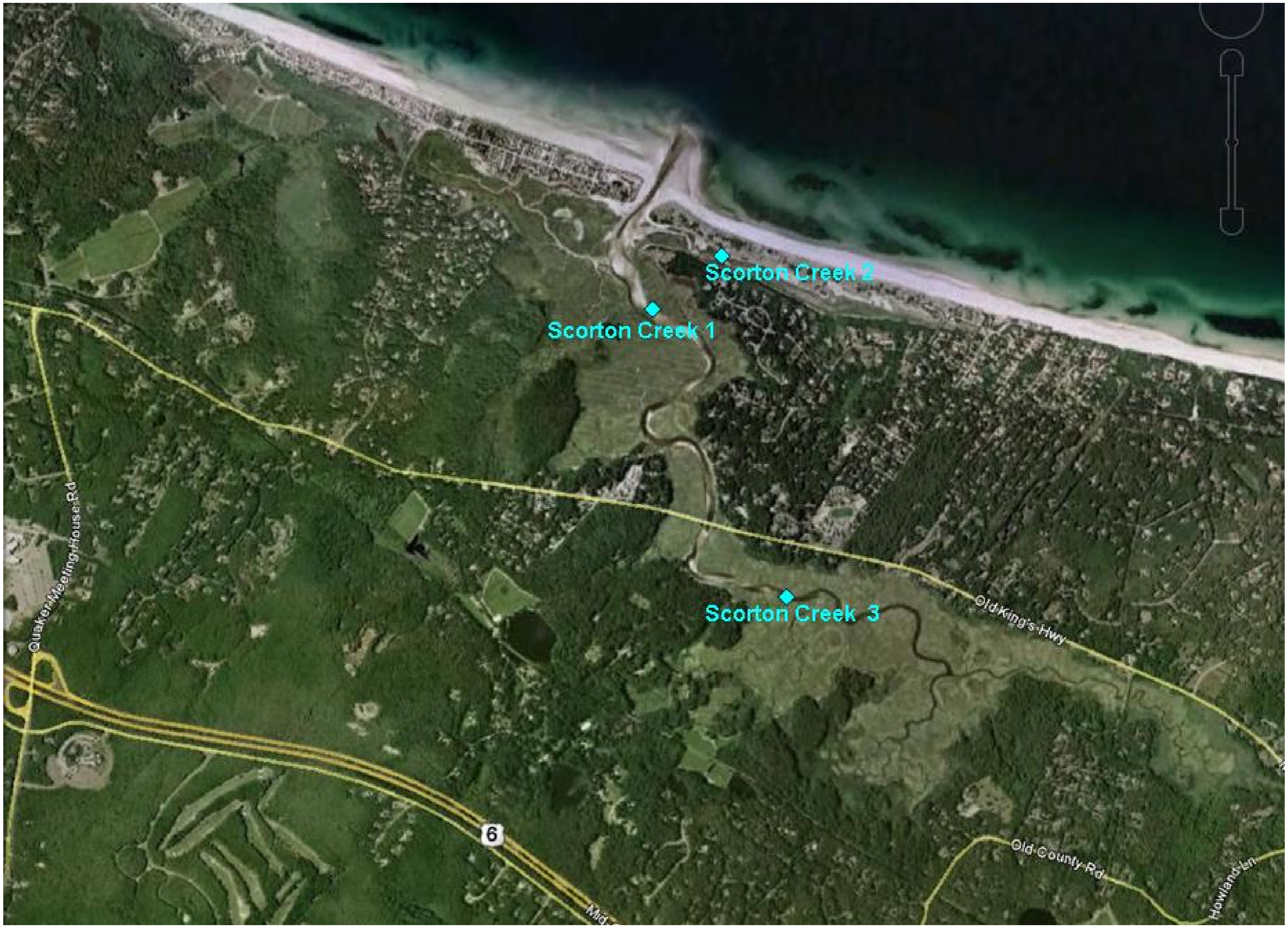


Figure VII-2. Aerial Photograph of the Scorton Creek system in the Town of Sandwich showing the location of the continuously recording Dissolved Oxygen / Chlorophyll-*a* sensors deployed during the Summer of 2006.

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 are consistent with the natural organic rich salt marsh creeks with high tidal flushing (Figures VII-3 through VII-8). The oxygen data is consistent with high organic matter inputs from the surrounding vegetated marsh and creek banks rather than from phytoplankton which show relatively low to moderate enrichment (chlorophyll-*a* levels generally <10 ug L^{-1}). The high velocity of tidal currents in the main channel and near complete draining of the creeks at low tide appears to reduce the settling of phytoplankton, hence sediment oxygen uptake.

The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in a particular location. Nitrogen enrichment also results in increased phytoplankton (or epibenthic algae) production, as evidenced by oxygen levels that rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). In Scorton Creek, oxygen levels only significantly exceeded atmospheric equilibrium occasionally, although oxygen depletion was commonly observed. The absence of elevated oxygen levels is consistent with the oxygen variations being the response to the naturally organic rich nature of tidal salt marsh creeks. In these systems the oxygen dynamic is driven by consumption within the tidal creeks, with re-oxygenation through phytoplankton production being limited. As the creeks drain nearly completely at low tide, the rise in oxygen levels is primarily through the entry of oxygen rich coastal waters on the flooding tide.

The dissolved oxygen records indicate that the upper (Scorton Creek 3 mooring) and lower regions of the central tidal salt marsh creek of the extensive Scorton Creek Salt Marsh frequently have oxygen declines to ~4 mg L^{-1}, while the tributary tidal creek along the backside of the barrier beach (Scorton Creek 2 mooring) shows periodic (but not prolonged) oxygen depletion to 2 mg L^{-1} (Table VII-1). Such oxygen depletion is typical of organic and nutrient rich temperate salt marsh creeks. Salt marshes are nutrient and organic matter enriched as part of their ecological design, which makes them such important nursery areas for adjacent offshore waters. However, a natural consequence of their organic rich sediments is periodic oxygen depletion within the tidal creeks, particularly during the summer. The observed level of oxygen depletion in the Scorton Creek salt marsh system is expected, as was the nearly identical pattern recorded by the MEP Technical Team in nearby Namskaket and Little Namskaket Creeks, both located on Cape Cod Bay in the Town of Orleans. Assessment of habitat quality must necessarily consider the natural function and tolerances of the specific estuarine ecosystems being evaluated. The specific results are as follows:

Scorton Creek DO/CHLA Moorings 1,2,3 (Figures VII-3 through VII-8):

The Scorton Creek Estuary is functionally a large tidal salt marsh with a central tidal creek. The upper reach has deeply incised narrow creeks surrounded by extensive emergent marsh vegetated with typical New England high and low marsh plants. The lower reach has broader creeks with sediments formed from marsh deposits and sand transported in by coastal processes. The mid and lower portions of the central creek have high tidal velocities as they drain water from the extensive upper marsh. As a result, the creek bottom sediments of the mid and lower reaches primarily consist of migrating fine and medium sands with some areas of coarse sand and gravel. The sides of the creeks consist of eroding salt marsh peat. In contrast, the upper creek areas support organic rich sediments with some fine sand mixed in. The tide range in adjacent Cape Cod Bay is large, ~10 ft (Chapter V), and the salt marsh areas are regularly flooded at high tide and the salt marsh creeks drain nearly completely with each ebb tide.

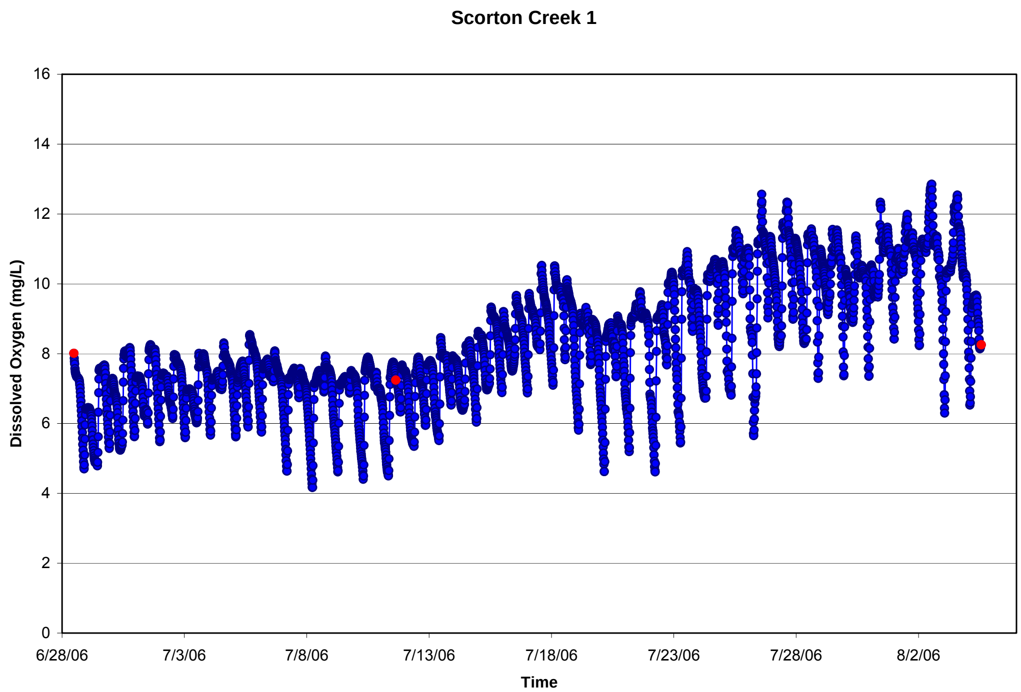


Figure VII-3. Bottom water record of dissolved oxygen at the Scorton Creek 1 station, Summer 2006. Calibration samples represented as red dots.

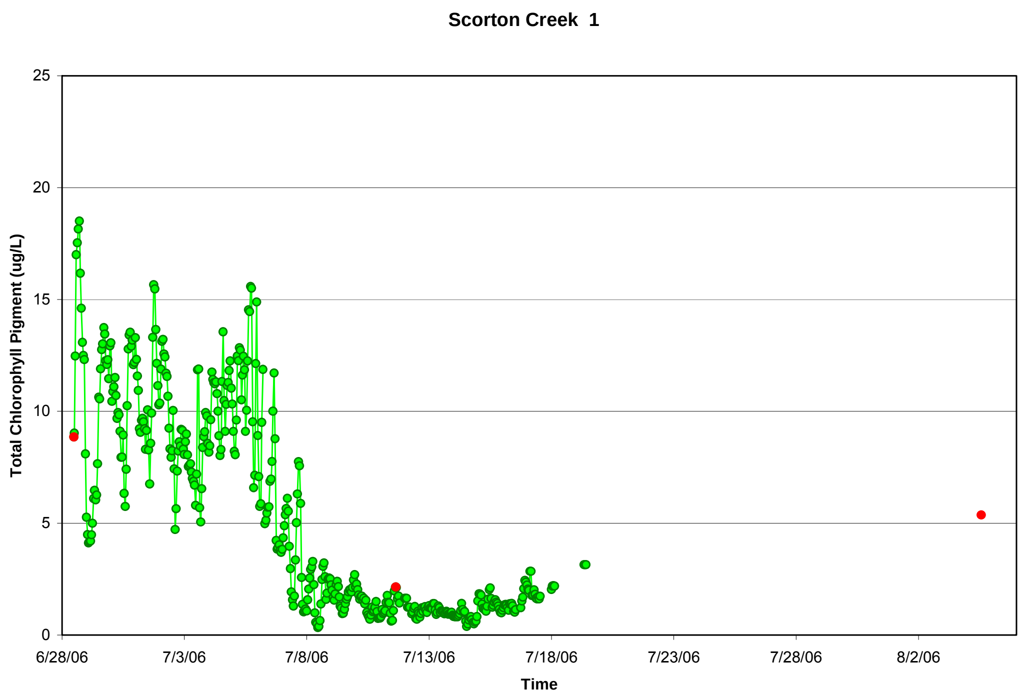


Figure VII-4. Bottom water record of Chlorophyll-*a* in the Scorton Creek 1 station, Summer 2006. Calibration samples represented as red dots.

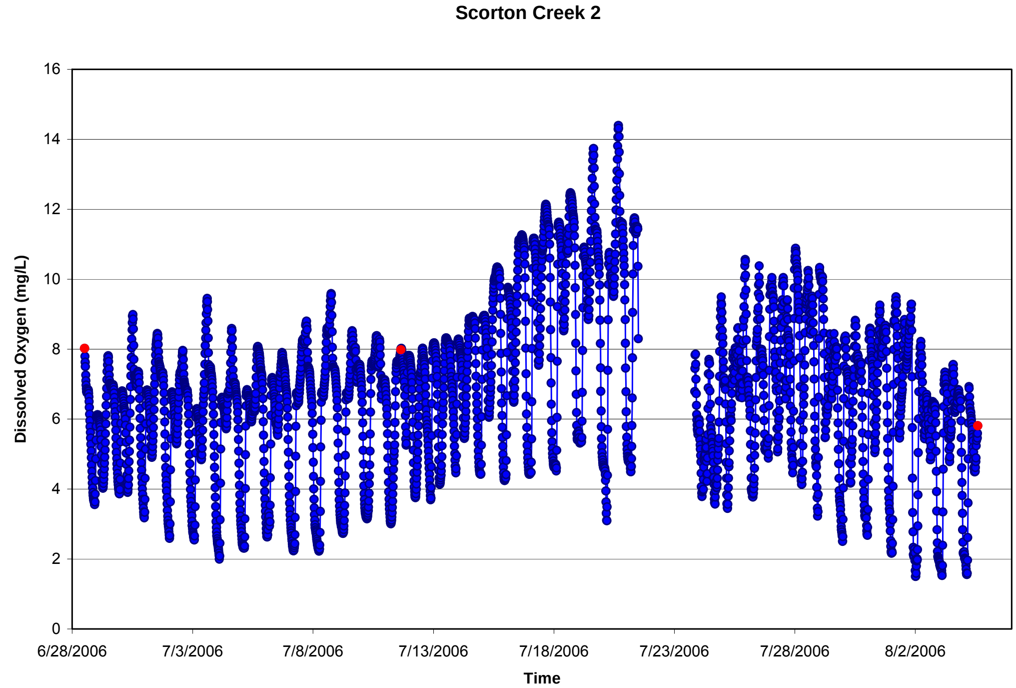


Figure VII-5. Bottom water record of dissolved oxygen at the Scorton Creek 2 station, Summer 2006. Calibration samples represented as red dots.

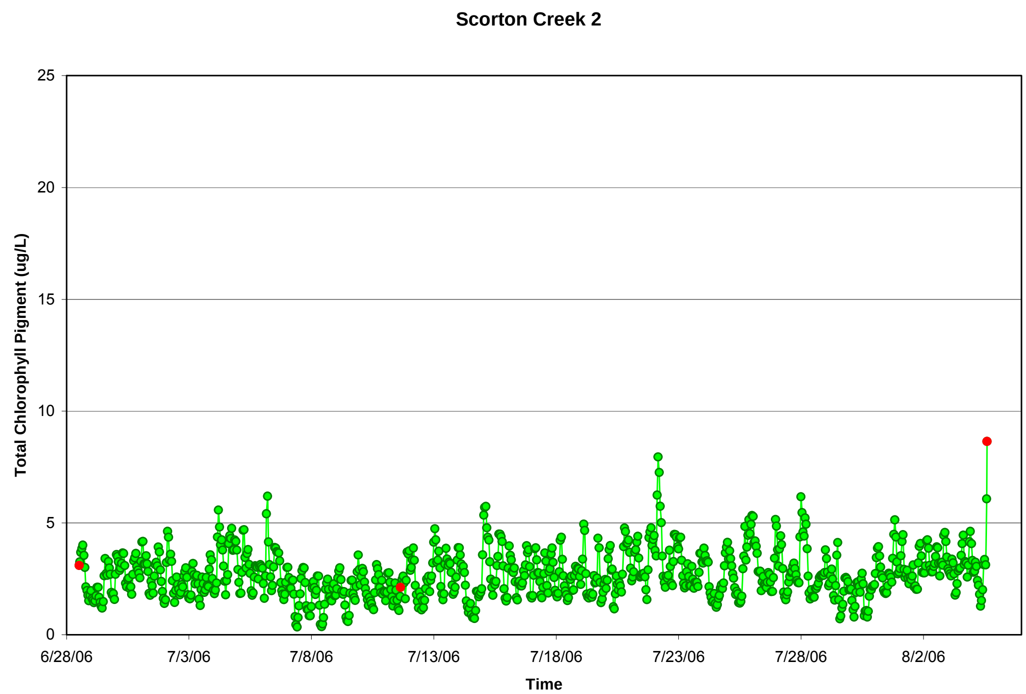


Figure VII-6. Bottom water record of Chlorophyll-*a* in the Scorton Creek 2 station, Summer 2006. Calibration samples represented as red dots.

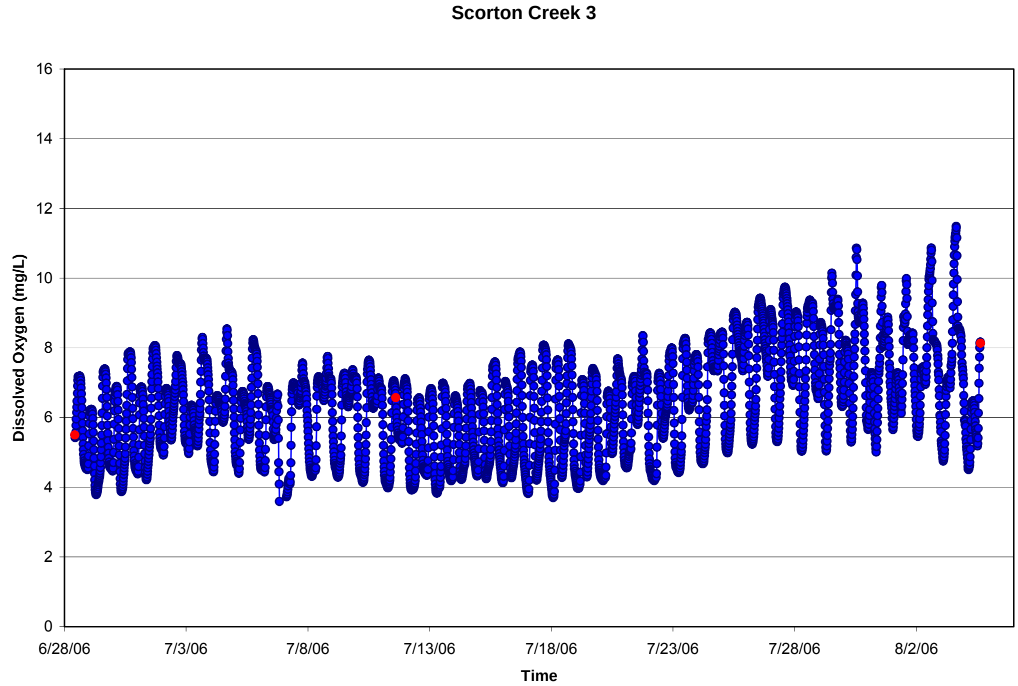


Figure VII-7. Bottom water record of dissolved oxygen at the Scorton Creek 3 station, Summer 2006. Calibration samples represented as red dots.

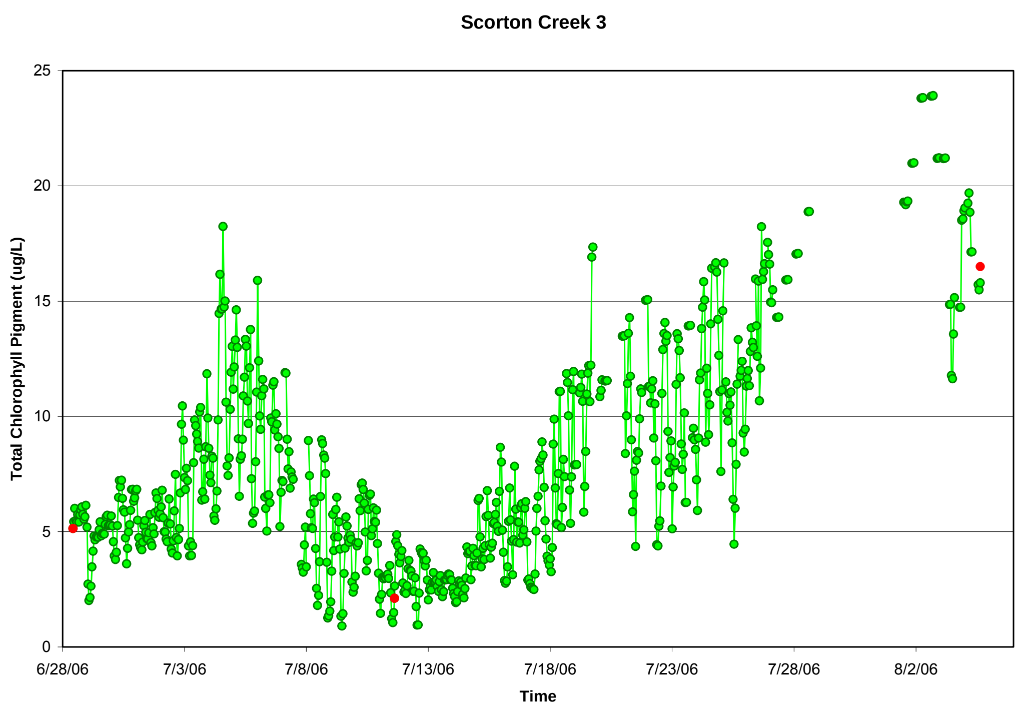


Figure VII-8. Bottom water record of Chlorophyll-*a* in the Scorton Creek 3 station, Summer 2006. Calibration samples represented as red dots.

Table VII-1. Days and percent of time during deployment of in situ sensors that bottom water oxygen was below various benchmark oxygen levels at each of the 3 mooring sites within the Scorton Creek Estuary. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Scorton Creek 1	**6/28/2006**	**8/4/2006**	**37.1**	**8%**	**2%**	**0%**	**0%**
			Mean	0.11	0.08	N/A	N/A
			Min	0.01	0.04	0.00	0.00
			Max	0.26	0.14	0.00	0.00
			S.D.	0.08	0.03	N/A	N/A
Scorton Creek 2	**6/28/2006**	**8/4/2006**	**37.07**	**33%**	**21%**	**12%**	**17%**
			Mean	0.21	0.17	0.15	0.14
			Min	0.02	0.01	0.04	0.01
			Max	0.46	0.31	0.28	0.24
			S.D.	0.10	0.08	0.08	0.07
Scorton Creek 3	**6/28/2006**	**8/4/2006**	**37.22**	**40%**	**22%**	**2%**	**0%**
			Mean	0.25	0.21	0.10	N/A
			Min	0.06	0.02	0.03	0.00
			Max	0.39	0.32	0.17	0.00
			S.D.	0.09	0.08	0.04	N/A

Table VII-2. Duration (days and % of deployment time) that chlorophyll-a levels exceed various benchmark levels at each of the 3 mooring sites within the Scorton Creek Estuary. "Mean" represents the average duration of each event over the benchmark level and "S.D." its standard deviation. Shortened deployment period for Scorton Creek 1 mooring due to sensor failure. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	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)
Scorton Creek 1	**6/28/2006**	**8/4/2006**	**19.0**	**100%**	**44%**	**21%**	**2%**	**0%**
Mean Chl Value = 5.3 ug/L			**Mean**	1.38	0.27	0.13	N/A	N/A
			Min	0.17	0.04	0.08	0.00	0.00
			Max	3.58	0.75	0.21	0.00	0.00
			S.D.	1.62	0.23	0.07	N/A	N/A
Scorton Creek 2	**6/28/2006**	**8/4/2006**	**37.25**	**100%**	**2%**	**0%**	**0%**	**0%**
Mean Chl Value = 2.7 ug/L			**Mean**	0.08	N/A	N/A	N/A	N/A
			Min	0.04	0.00	0.00	0.00	0.00
			Max	0.21	0.00	0.00	0.00	0.00
			S.D.	0.05	N/A	N/A	N/A	N/A
Scorton Creek 3	**6/28/2006**	**8/4/2006**	**29.04**	**100%**	**67%**	**29%**	**10%**	**2%**
Mean Chl Value = 7.9 ug/L			**Mean**	0.54	0.27	0.17	0.63	N/A
			Min	0.04	0.04	0.04	0.63	0.00
			Max	4.08	3.13	1.17	0.63	0.00
			S.D.	1.00	0.55	0.29	N/A	N/A

Moderate to large diurnal shifts in dissolved oxygen were measured at each of the 3 mooring sites, but in the upper and lower main creek moorings (1, 3) oxygen levels >4 mg L^{-1}, were maintained which is relatively high for salt marsh creeks. It is likely that the near draining of the creek water at low tide and the large tidal flows with low nutrient and high oxygen Cape Cod Bay water plays a significant role in the oxygen balance, as does the sandy sediments with only moderate levels of oxygen uptake (D. Schlezinger, personal communication). The tributary creek running behind the Scorton Shores barrier beach shows slightly greater oxygen excursions, but generally low chlorophyll-*a* levels (mean 2.7 ug L^{-1}). Overall, the oxygen conditions within the tidal creeks are reflective of temperate coastal salt marshes and do not indicate impairment to these naturally organic and nutrient rich ecosystems. The chlorophyll-*a* levels are consistent with this conclusion, in that the average levels over the deployments were low, ranging from 2.7 - 7.9 ug L^{-1}, with only the upper marsh and Scorton Shores creek showing enhancement (5.3 and 7.9 ug L-1, respectively) over offshore waters. The high tidal flushing does not allow a buildup of phytoplankton during the short residence time of phytoplankton within Scorton Creek, generally < 1 tidal cycle. The high velocities appear to reduce the sediment oxygen demand over other salt marshes, and generate sandy oxidized surface sediments over much of the mid and lower tidal reaches. However, it appears that transfers from the emergent marsh and from the upper to lower marsh are sufficient to create the observed periodic low oxygen levels. It should be noted that there were no prolonged (e.g. several day) hypoxic events in this system, as found in impaired open water basins. Instead, oxygen continuously cycled from atmospheric equilibration to ~ 4 or ~2 mg L^{-1} for sites 1 and 3 and 2, respectively. Depletion ranged from low (92% of record >6 mg L^{-1}) to moderate (12% of record <4 mg L^{-1}) for the lower main channel to moderate-high for the Scorton Shores Creek.

The relatively low chlorophyll-*a* concentrations at all the mooring locations show little enhancement over the offshore waters (chlorophyll in waters offshore from Scorton Creek are generally 2.5 ug/L), as the near complete exchange of tidal waters on each tide in the creeks does not allow for chlorophyll levels to build. The absence of prolonged (i.e. multi tidal cycle) oxygen depletion, typically found in nitrogen enriched embayments (due to stimulation of phytoplankton), supports the concept that tidal exchange and natural marsh processes are the primary controls on oxygen dynamics in this estuary. In fact, the daily average dissolved oxygen concentration varied inversely with the tidal amplitude suggesting that longer residence time and greater areal submergence of the marsh was responsible for the lowest oxygen observed oxygen levels. Further evidence for the dominance of marsh processes is the lack of linkage between the observed variations in chlorophyll and the extent of oxygen depletion. In embayments, oxygen minima are typically observed as a bloom declines (senesces), a pattern not seen at this site. Consistent with this latter observation is that the long-term average TN level at the mooring sites ranged from 0.40 mg N L^{-1} to 0.55 mg N L^{-1}. By comparison, in Namskaket Marsh (Town of Orleans), a salt marsh system similar to Scorton Creek, the tidally averaged TN level at the mooring site was 0.488 mg N L^{-1}. Like Scorton Creek, dissolved oxygen and chlorophyll levels in Namskaket Marsh exhibited characteristics consistent with healthy salt marsh environments. In traditional embayment systems on Cape Cod, these TN levels typically do not result in the level of oxygen depletion found in the salt marsh creeks of Scorton Creek or Namskaket and adjacent Little Namskaket Estuaries in the Town of Orleans. It is clear that the organic rich nature of salt marshes are a predominant control of oxygen levels. It should be noted that this level of TN is typical of salt marsh creeks, and does not indicate impairment in that type of environment. Overall, based upon the measured levels of oxygen and chlorophyll and comparisons to other unimpaired salt marshes tributary to Cape Cod Bay, it does not appear that the Scorton Creek Estuary is impaired based on these metrics.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data is key part of the MEP Approach. Surveys were conducted in 1995 and 2001 in the vicinity of the Sandwich Harbor System by the DEP Eelgrass Mapping Program to be integrated into the MEP effort. These surveys were essentially in the near shore waters of Cape Cod Bay, between the mouth of the Cape Cod Canal and the inlet to Sandwich Harbor but did not extend to the vicinity of Scorton Creek. The primary use of the data is to indicate (a) estuarine regions that have historically or presently support eelgrass habitat, and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1995 to 2001 (Figure VII-9); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

Eelgrass surveys were not undertaken for the tidal creek that constitutes the main channel of the Scorton Creek Salt Marsh by the MassDEP Eelgrass Mapping Program (C. Costello). In addition, the analysis of conditions from 1951 could not be performed due to the lack of adequate aerial photos. Tidal creeks like those in the Scorton Creek Salt Marsh do not generally support eelgrass habitat, particularly when the creeks nearly completely drain during each ebb tide, as is the case for Scorton Creek and its associated tributary tidal creeks. The absence of eelgrass, for at least the past 60-70 years, has also been documented for Namskaket and Little Namskaket Marshes, to the north also on Cape Cod Bay. The MEP Technical Team confirmed that eelgrass is not currently present in the tidal creeks to the Scorton Creek system while undertaking field surveys as part of the benthic regeneration and infauna studies and during the deployment and recovery of the instrument moorings (summer and fall 2006).

In contrast, eelgrass is present offshore to the west of the inlet to Sandwich Harbor. Based on the 2001 eelgrass survey conducted by the DEP Eelgrass Mapping Program. There was evidence of a potential small decline in the coverage of the offshore beds between 1995 and 2001 in that area. However, it is not possible at this time to determine if this represents an anthropogenically driven decline or natural variation at this site. Additional spatial and temporal sampling may be undertaken by the MassDEP Eelgrass Mapping Program to address the issue of the decline in eelgrass beds in that area as well as possibly determining if there is any eelgrass present in the near shore waters adjacent Scorton Creek.

Based on the salt marsh function of the Scorton Creek Estuary, historical absence of eelgrass in all of the other salt marsh systems on Cape Cod that exchange waters with Cape Cod Bay and the similarity of Scorton Creek to Namskaket Creek and Little Namskaket Creek that have not historically supported eelgrass habitat, the MEP Technical Team concludes that the Scorton Creek Estuary has not supported eelgrass for many decades, if not longer. Also, search of historic records has also not revealed evidence of eelgrass beds in Scorton Creek. The absence of eelgrass habitat is expected in this system given the periodic low dissolved oxygen levels and high water column nitrogen concentrations. Based upon all available information, it appears that the Scorton Creek Estuary is not structured to support eelgrass habitat. Therefore, threshold development for protection/restoration of this system will focus on infaunal habitat quality. This is typical for New England salt marshes, which are naturally organic and nutrient rich and generally contain little water in the creeks at low tide. This conclusion has been confirmed in a wide range of salt marsh dominated basins throughout southeastern Massachusetts by the MEP Technical Team.

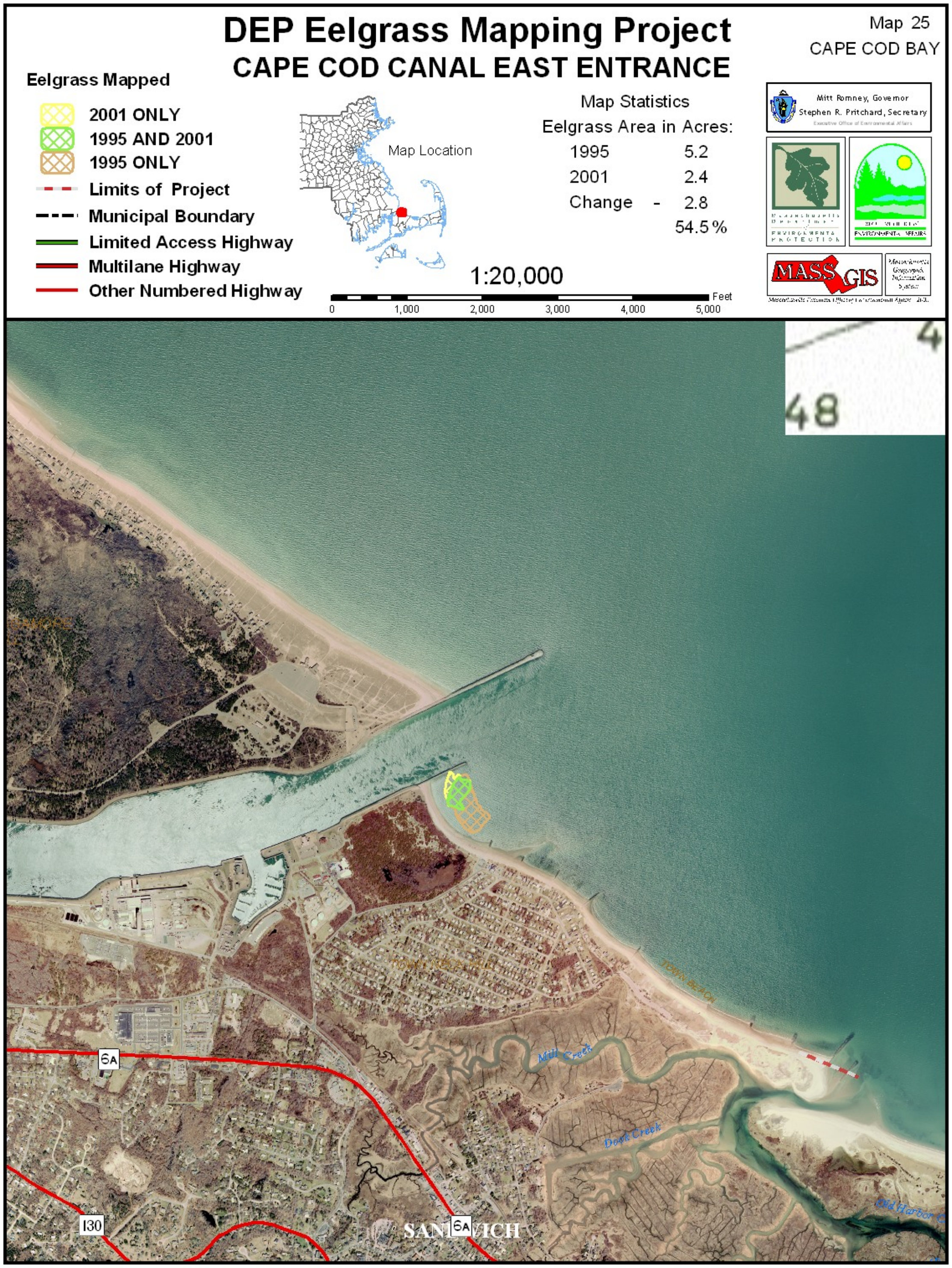


Figure VII-9. Eelgrass bed distribution offshore of Sandwich Harbor and the Cape Cod Canal. Beds delineated in 1995 are circumscribed by the brown outline with a composite of 1995 and 2001 outlined in green (map from the MassDEP Eelgrass Mapping Program). Surveying did not extend into the salt marsh tidal creeks, however, no eelgrass was observed in the Sandwich Harbor system and Scorton Creek Salt Marsh during SMAST-MEP surveying in 2006.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 6 locations within the major tidal creeks throughout the Scorton Creek Estuary (Figure VII-10), with replicate assays at each site. 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 (H') and Evenness (E) of the community. It should be noted that, although there are no eelgrass beds in the Scorton Creek Marsh system, this is almost certainly due to the system functioning as a large intertidal salt marsh. Scorton Creek like all the other large salt marshes tributary to Cape Cod Bay, does not appear to have historically supported eelgrass habitat and therefore the absence of eelgrass is "natural" and does not indicate impairment. As such, to the extent that the Scorton Creek Marsh can support healthy infaunal communities given specific nutrient conditions in the water column, 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).

SCORTON CREEK INFAUNAL CHARACTERISTICS:

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information (Table VII-3). 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.

Table VII-3. Benthic infaunal community data for the Scorton Creek Estuary, which is a major tidal salt marsh system tributary to Cape Cod Bay. Measured number of species and individuals, with 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 m^2. Stations refer to map in Figure VII-10.

	Sta. I.D.*	Total Species	Total Individuals	#Species Calc @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Scorton Creek Salt Marsh						
Upper Creek	**1**	**12**	**1026**	**8**	**2.13**	**0.61**
Mid-Lower Main Creek	**4,6,7**	**5**	**11**	**N/A**	**1.84**	**0.84**
Lower Tributary Creeks	**8,10**	**10**	**740**	**7**	**1.83**	**0.56**

* Station i.d.'s refer to Figure VII-10.

Figure VII-10. Aerial photograph of the Scorton Creek system showing location of benthic infaunal sampling stations (yellow symbols).

The infauna study of the Scorton Creek Estuary indicated that the tidal creeks of this salt marsh are presently supporting a salt marsh infaunal habitat typical of large unimpaired salt marsh systems on Cape Cod. Infauna communities within the tidal creek were indicative of the fine grained organic rich environment typical of salt marshes or of high velocity areas with sandy sediments and were consistent with the observed levels of oxygen depletion and water column TN. The communities within the upper reach had high numbers of individuals with moderate numbers of species and diversity and Evenness as also was found in the tributary creeks in the lower estuary, which also had some deep burrowers (e.g. *Clymenella*). In contrast, the mid and lower reach of the main channel was found to have high water velocities which have winnowed the fines from the sediments and created a medium to coarse sand with shifting sediments. The result is similar to tidal inlets or Chatham Harbor's shifting sands which do not support extensive benthic habitat, being naturally disturbed. The observed communities were typical of New England salt marsh creek bottom environments in summer. The soft-bottomed areas supported communities with organic enrichment tolerant species and were dominated by *Streblospio* and there were significant numbers of mollusks, crustaceans and polychaetes (SC-4, SC-8, SC-10). The sandy high velocity areas (SC-4, SC-6, SC-7) had unstable sediments and therefore did not have significant benthic animal habitat (Table VII-3). These latter areas, particularly SC-7, had high water quality, low chlorophyll-*a*, low TN (long term average, 0.30 mg N L^{-1}) and minimal oxygen depletion, consistent with physical disturbance, rather than nitrogen enrichment, structuring the environment.

Overall, the infauna survey indicated that most areas within the Scorton Creek system are supporting infauna habitat typical of organic rich New England salt marshes, hence high quality relative to this estuarine ecosystem type. The mid-lower main channel is structured by its high velocity and physical disturbance to the sediment. The lack of nitrogen related impairment is supported by the general absence of surface algal mats and macroalgae, with only a few patches of very sparse Ulva being observed. The near absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels within this system, 0.32-0.68 mg N L^{-1} (long-term average), but generally <0.5 mg N L^{-1}. By comparison, in Namskaket Marsh a similar salt marsh system also fed by low nutrient waters from Cape Cod Bay, macroalgal accumulations were also absent with similarly low total nitrogen levels of 0.42-0.66 mg N L^{-1} (tidally averaged). Infaunal habitat health status can further be confirmed when making an additional comparison to another similar marsh, Cockle Cove Creek (Chatham), which supports high quality habitats, both emergent marsh and creek bottom, at levels of 2 mg TN L^{-1}. Based upon all lines of evidence it appears that the Scorton Creek estuary is presently supporting high quality infaunal habitat and has not exceeded its threshold nitrogen level for assimilating additional nitrogen without impairment.

Other Resource Characteristics:

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish (Figure VII-11). As is the case with some systems on Cape Cod, all of the Scorton Creek Estuary is classified as prohibited for the taking of shellfish at any time during the year, i.e. Scorton Creek has reoccurring elevated numbers of indicator bacteria (fecal coliform). This is most likely due to bacterial inputs from wildlife and birds associated with the wetland and possibly from human activity (storm water or septic systems in the watershed). Despite the existing shellfish area classifications, the Scorton Creek Estuary is also classified as supportive of specific shellfish communities (Figure VII-12). The major shellfish species with potential habitat within the Scorton Creek Estuary are soft shell clams (*Mya*) and quahogs (*Mercenaria*) in the lower portion of the main tidal channel of the system up to the level of Route 6A as well as the tributary tidal creek that runs parallel to the backside of the barrier beach (Scorton Shores). Additionally, the area of hard substrate in the immediate vicinity of the inlet to the Scorton Creek system was classified as suitable for blue mussels.

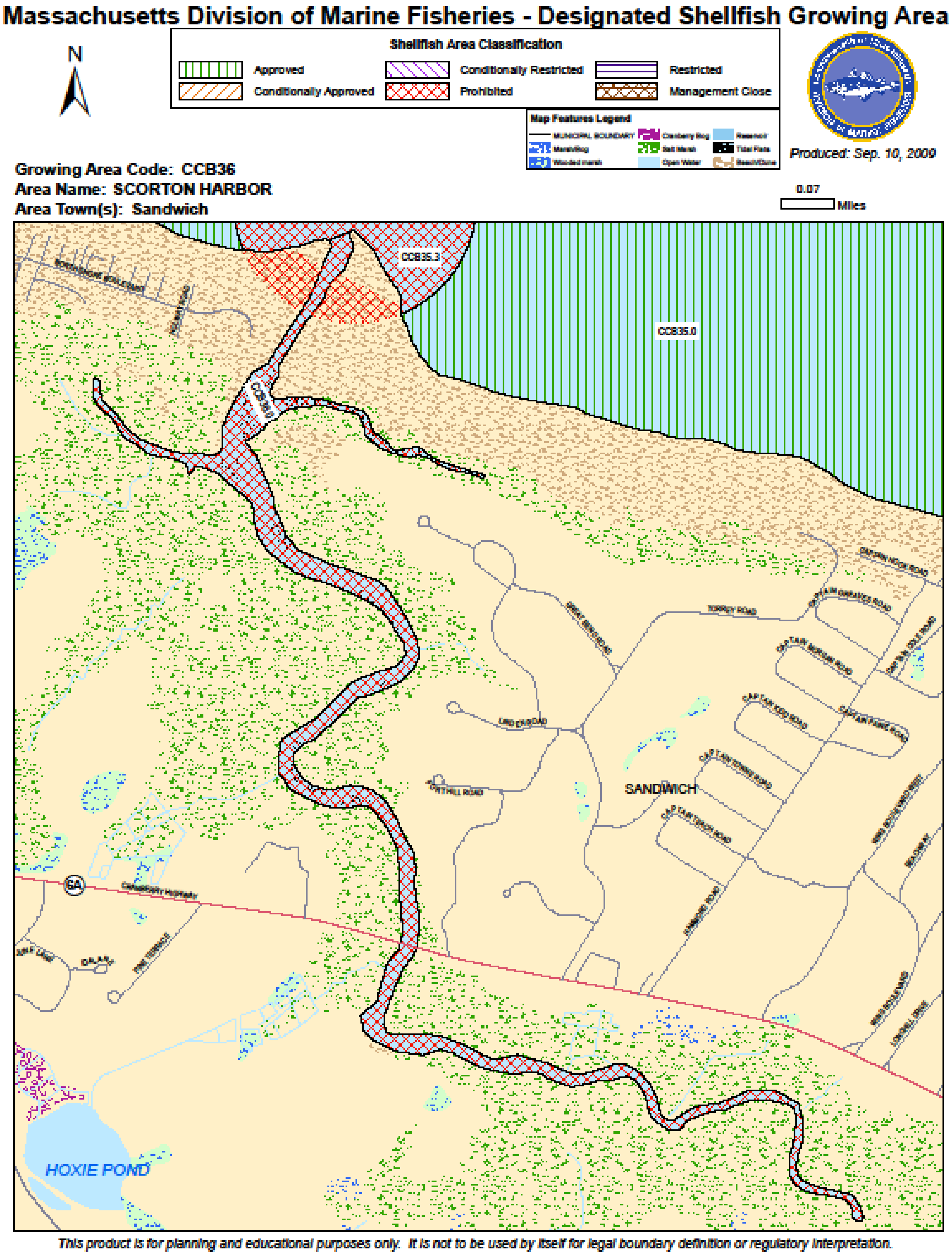


Figure VII-11. Location of shellfish growing areas in the Scorton Creek embayment system and the status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination from wildlife or human "activities", such as the location of marinas, septic tanks or stormwater discharges.

Figure VII-12. Location of shellfish suitability areas within the Scorton Creek Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence". The delineated areas generally coincide with creek bottoms dominated by fine and medium sand.

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 Scorton Creek estuarine system by the MEP 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 water quality monitoring baseline collected by the Town of Sandwich Water Quality Monitoring Program (2005-2007).

At present, the Scorton Creek Estuary is not showing nitrogen impairment of habitat throughout its tidal reaches. The upper reach appears to be a fully functional tidal salt marsh with deeply incised narrow creeks surrounded by extensive emergent marsh. This reach is typical of New England "pocket" marshes, with smaller tidal creeks and a marsh plain dominated by low marsh (*Spartina alterniflora*) and high marsh (*Spartina patens, Distichlis spicata*) plant communities with patches of fringing brackish marsh vegetation (*Juncus, Phragmites*). The lower reach of the marsh supports a larger central tidal creek within a large emergent vegetated marsh. The large central tidal creek has physically disturbed sediments due to the high tidal velocities (can be seen in the medium to coarse sands) and is also influenced by sand transport via nearshore coastal processes associated with adjacent Cape Cod Bay. These disturbances prevent establishment of stable infauna communities in the central basin, but as velocities are lower in the tributaries, infauna habitat is found there. Plant communities in the lower reach are similar to the upper reach except that there is less fringing brackish water species and the marsh grades to barrier beach/dune vegetation near the tidal inlet. The tide range in adjacent Cape Cod Bay is large, ~10 ft (Section V), and the salt marsh areas are regularly flooded at high tide and the salt marsh creeks drain nearly completely with each ebb tide.

While there is cyclical depletion of oxygen in the upper and lower marsh creeks, no prolonged hypoxia was observed. Chlorophyll levels are low in lower marsh creeks (<5 ug L-1) and low to moderate in the upper marsh creeks (5-9 ug L^{-1}). However, the near complete flushing of the system with each tidal cycle prevents blooms from occurring within the marsh. The lack of nitrogen related impairment is supported by the general absence of surface algal mats and macroalgae, with only a few patches of very sparse Ulva being observed. The near absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels (for a salt marsh) within this system, 0.32-0.68 mg N L^{-1} (long-term average), but generally <0.5 mg N L^{-1}.

All of the key indicators are consistent with the Scorton Creek Estuary, and particularly its tidal creeks, supporting high quality habitat, relative to its salt marsh structure and function (Section VII). Assessment of habitat quality must necessarily consider the natural function and tolerances of the specific estuarine ecosystem being evaluated. The lack of impairment of Scorton Creek is predictable from its moderate nitrogen levels and the greater tolerance of salt marshes to nitrogen loading than open water embayments.

Eelgrass: Eelgrass surveys were not undertaken for the tidal creek that constitutes the main channel of the Scorton Creek Salt Marsh by the MassDEP Eelgrass Mapping Program (C. Costello). In addition, the 1951 analysis could not be performed due to the lack of adequate aerial photos. However, tidal salt marsh creeks like those in the Scorton Creek Estuary do not generally support eelgrass habitat, particularly when the creeks nearly completely drain during each ebb tide, as is the case for Scorton Creek and its associated tributary tidal creeks. The absence of eelgrass, for at least the past 60-70 years, has been documented for Namskaket and Little Namskaket Marshes, to the east also on Cape Cod Bay. The MEP Technical Team confirmed that eelgrass is not currently present in the tidal creeks to the Scorton Creek system while undertaking field surveys as part of the benthic regeneration and infauna studies and during the deployment and recovery of the instrument moorings (summer and fall 2006).

Based on the salt marsh function of the Scorton Creek Estuary, historical absence of eelgrass in all of the other salt marsh systems on Cape Cod that exchange waters with Cape Cod Bay and the similarity of Scorton Creek to Namskaket Creek and Little Namskaket Creek, both of which have not historically supported eelgrass habitat, the MEP Technical Team concludes that the Scorton Creek Estuary has not supported eelgrass for many decades, if not longer. Also, search of historic records has not revealed evidence of eelgrass beds in Scorton Creek. The absence of eelgrass habitat is expected in this system given the periodic oxygen depletion and elevated water column nitrogen concentrations. Based upon all available information, it appears that the Scorton Creek Estuary is not structured to support eelgrass habitat. Therefore, threshold development for protection/restoration of this system should focus on infaunal habitat quality. This is typical for New England salt marshes, which are naturally organic and nutrient rich and generally contain little water in the creeks at low tide. This conclusion has been confirmed in a wide range of salt marsh dominated basins throughout southeastern Massachusetts by the MEP Technical Team.

The eelgrass data for the Scorton Creek Estuary are consistent with the results of the benthic infauna analysis and the water quality data for this system (see below).

Infaunal Animal Communities: The infauna animal communities within the Scorton Creek Estuary indicated that the tidal creeks of this salt marsh are presently supporting a salt marsh infaunal habitat typical of large, unimpaired salt marsh systems on Cape Cod. Infauna communities within the tidal creek were indicative of the fine grained organic rich environment typical of salt marshes or of high velocity areas with sandy sediments and were consistent with the observed levels of oxygen depletion and water column TN. The communities within the upper reach had high numbers of individuals with moderate numbers of species and diversity and Evenness as also found in the tributary creeks in the lower estuary, which also had some deep burrowers (e.g. *Clymenella*). In contrast, the mid and lower reach of the main channel was found to have high water velocities which have winnowed the fines from the sediments and created a medium to coarse sand with shifting sediments. The result is similar to tidal inlets or Chatham Harbor's shifting sands which do not support extensive benthic habitat, being naturally disturbed. The observed communities were typical of New England salt marsh creek bottom environments in summer. The soft-bottomed areas supported communities with organic enrichment tolerant species and were dominated by *Streblospio* and there were significant numbers of mollusks, crustaceans and polychaetes. The sandy high velocity areas had unstable sediments and therefore did not have significant benthic animal habitat (Table VII-3). Note that these latter areas had high water quality, low chlorophyll a, low TN (long term average, 0.30 mg N L^{-1}) and minimal oxygen depletion, consistent with physical disturbance, rather than nitrogen enrichment, structuring the environment.

Overall, the infauna survey indicated that most areas within the Scorton Creek system are supporting infauna habitat typical of organic rich New England salt marshes, hence high quality relative to this estuarine ecosystem type. The mid-lower main channel is structured by its high velocity and physical disturbance to the sediment. The lack of nitrogen related impairment is supported by the general absence of surface algal mats and macroalgae, with only a few patches of very sparse Ulva being observed. The near absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels within this system, 0.32-0.68 mg N L^{-1} (long-term average), but generally <0.5 mg N L^{-1}. Based upon all lines of evidence it appears that the Scorton Creek estuary is presently supporting high quality habitat and has not exceeded its threshold nitrogen level for assimilating additional nitrogen without impairment. Threshold development for this estuary will therefore focus on protection of resources and not exceeding the assimilative capacity of this system for nitrogen.

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout an embayment system, is to first identify a sentinel location within the estuary and second to determine the nitrogen concentration within the water column which will restore or maintain the desired habitat quality. The sentinel location is selected such that the restoration or protection 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.

Scorton Creek functions wholly as a large tidal salt marsh with a central creek and high rate of flushing, with the creeks generally empting at low tide. As a result, the Scorton Creek Estuary does not support eelgrass habitat, similar to other Cape Cod Bay salt marshes, like Namskaket and Little Namskaket Marshes (Orleans). MEP assessment of the other habitat metrics for Scorton Creek do not indicate nitrogen related habitat impairments (Table VIII-1). As a result, threshold development for the Scorton Creek Estuary targets its protection and focuses on sustaining infaunal habitat quality. The primary mechanism for nitrogen related infaunal habitat quality decline in salt marsh creeks of this type is through stimulation of macroalgal production and accumulation, which smothers benthic animals and at higher levels can negatively impact creek bank vegetation.

Determination of the critical nitrogen threshold for maintaining high quality habitat within the Scorton Creek Estuary is based primarily upon: 1) the structure and function of the system as a salt marsh, 2) macroalgal distribution, 3) current benthic community indicators and 4) nitrogen levels. Given the database it is possible to develop a site-specific threshold, which is a refinement upon general threshold analysis frequently employed.

The Scorton Creek Estuary is presently supportive of high quality salt marsh infaunal habitat throughout its tidal reach. While there is periodic summertime oxygen depletion of creek waters, the levels are consistent with unimpaired New England salt marsh systems. At present, significant (or even modest) macroalgal accumulations do not occur within this macro-tidal estuary at long term ebb tide concentrations of 0.677 mg N L^{-1} (headwaters) to 0.323 mg N L^{-1} (near tidal inlet). The inflowing water at the head of the marsh "freshwater" (average salinity of 1.3 ppt) has the highest TN level at 0.989 mg TN L^{-1}. By comparison, Namskaket Marsh supports a tidally averaged total nitrogen level of 0.662 mg N L-1 (headwaters) to 0.421 mg N L-1 (tidal inlet).

Table VIII-1. Summary of Nutrient Related Habitat Health within the Scorton Creek Salt Marsh on the Cape Cod Bay shore of the Town of Sandwich, MA, based upon assessment data presented in Chapter VII. The tidal reach of this estuary is a typical New England salt marsh with a large central tidal creek and as such it is nutrient and organic matter enriched. WQMP refers to the Water Quality Monitoring Program, 2005-2007.

Health Indicator	Scorton Creek Estuary		
	Upper Salt Marsh	**Lower Salt Marsh**	**Lower Tributary Creeks**
Dissolved Oxygen	H[1,2]	H[1,3]	H[1,4]
Chlorophyll	H[5]	H[6]	H[7]
Macroalgae	H[8]	H[8]	H[8]
Eelgrass	--[9]	--[9]	--[9]
Infaunal Animals	H[10]	--[11]	H[10]
Overall:	**H**	**H**	**H**

1 -- oxygen dynamics consistent with a naturally organic matter and nutrient rich New England salt marsh, oxygen depletions are typical of pristine salt marsh creeks on Cape Cod and elsewhere.
2-- Oxygen depletion to 4 mg L-1, but no prolonged (multi-tidal cycle) hypoxia.
3-- Minimal oxygen depletion, D.O. >6 mg L-1 for 92% of record.
4-- Cyclical depletions briefly to 4 and 2 mg L-1, no prolonged or even full tidal cycle depletion.
5-- low-moderate chlorophyll a, time-series average 7.9 ug/L, WQMP (Sta. SC-2,3,4,5,8) = 5-9 ug L^{-1}.
6 -- very low chlorophyll a levels time-series average 2.7 ug/L, WQMP ~3 ug L-1, Rt 6A to inlet, reflective of Cape Cod Bay waters.
7 -- low chlorophyll a levels time-series average 5.3 ug/L (mooring #1), WQMP 3.3 and 4.5 ug L-1, for Long Hill Creek and Scorton Shores Creek, respectively.
8 – *Ulva* and drift algae and surface algal mats very sparse to absent
9 – no evidence that this estuarine reach is supportive of eelgrass. Area is a salt marsh tidal creek, which drains at low tide.
10 – high numbers of individuals (>700), moderate numbers of species (10-12) and diversity (~2.0) and Evenness (~0.6) with some deep burrowers (e.g. *Clymenella*) in lower reach.
11–high water velocities have winnowed the fines from the sediments and created a medium to coarse sand with shifting sediments a natural physical disturbance similar to tidal inlets or Chatham Harbor's shifting sands which do not support extensive benthic habitat.

H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment; SD = Severe Degradation -- = not applicable to this estuarine reach

Since Scorton Creek Salt Marsh is presently below the level of nitrogen loading that would cause impairment to its infaunal habitats (i.e. below its nitrogen threshold level), a conservative estimate of the threshold was established. The threshold was based upon the site-specific data mentioned above and comparison to other similar systems on Cape Cod where detailed nitrogen threshold studies have been completed. This inter-estuarine comparison focused upon similar salt marshes which are presently experiencing higher nitrogen levels, with and without impairment.

The near complete absence of macroalgal accumulations is consistent with the relatively low total nitrogen levels within this system, 0.32-0.68 mg N L^{-1} (long-term average), but generally <0.5 mg N L^{-1}. By comparison, in Namskaket Marsh a similar salt marsh system also fed by low nutrient waters from Cape Cod Bay, macroalgal accumulations were also absent with similarly low total nitrogen levels of 0.42-0.66 mg N L^{-1} (tidally averaged). Infaunal habitat health status can further be confirmed when making an additional comparison to another similar marsh, Cockle Cove Creek (Chatham), which supports high quality habitats, both emergent marsh and creek bottom, at levels of 2 mg TN L^{-1}. Scorton Creek and Namskaket Creek both support benthic infauna habitat unimpaired by nitrogen enrichment and habitat typical of organic rich New England salt marshes, hence high quality relative to this estuarine ecosystem type. This is supported by the absence of macroalgal accumulations and algal mats within the creek bottoms, which can result if there is "excessive" external nitrogen loading. The absence of macroalgal accumulations in both systems is consistent with their relatively low total nitrogen levels (Scorton Creek, 0.32-0.65 mg TN L-1 vs. Namskaket Marsh, 0.42-0.66 mg TN L-1). These levels are significantly less than for a similar marsh, Cockle Cove Creek (Chatham), which supports high quality habitats, both emergent marsh and creek bottom, at levels of 2 mg TN L^{-1}. Based upon all lines of evidence it appears that the Scorton Creek Estuary is presently supporting high quality infaunal habitat and has not exceeded its threshold nitrogen level for assimilating additional nitrogen without impairment.

A detailed nitrogen threshold analysis of Cockle Cove Creek (Chatham) was conducted for a similarly configured salt marsh to Scorton Creek and Namskaket Marsh (SMAST 2006). In addition to having a similar structure, Cockle Cove Creek also supports similar benthic communities, sparse (to absent) macroalgal accumulations and similar tidal velocities within the central creek. In addition, the infaunal habitats within Namskaket and Cockle Cove Marsh are similar in composition and diversity (dominated by polychaetes and crustaceans, with some mollusks). Some of the key species (*Leptocheirus, Paranais*) were also observed in a study of another healthy salt marsh, Great Sippewisset Marsh on Cape Cod (Wiltse 1984).

Based on MEP analysis, it was ascertained that the major infaunal habitat issue related to nitrogen levels in salt marshes (e.g. Scorton Creek, Namskaket Marsh and Cockle Cove) was associated with the potential accumulation of macroalgae within the tidal creek bottom. Accumulations of macroalgae have been observed in a variety of nitrogen enriched salt marsh systems, for example Mashapaquit Marsh, West Falmouth Harbor and Aucoot Cove salt marsh (Town of Marion). These marshes differ from Namskaket and Cockle Cove Creek, in that they do not empty at low tide and they support broad creeks with low tidal velocities. These geomorphic differences allow these systems to grow and accumulate opportunistic drift algae (*Ulva*), which can then "smother" benthic communities. The MEP Technical Team assessed tidal velocities of these systems using a numerical hydrodynamic model. Cockle Cove Creek, Namskaket Creek and adjacent Little Namskaket Creek were found to have similar tidal velocities (1.1 ft sec^{-1}, 1.24 sec^{-1}, 1.13 ft sec^{-1}, respectively), but lower than the maximum velocities in the main channel of Scorton Creek (see Section V). None of these systems have macroalgal accumulations. In contrast, Mashapaquit and Aucoot Marshes had much lower tidal velocities (<0.5 ft sec^{-1}) and macroalgal accumulations have been observed. Algal accumulation did not appear to be controlled by nitrogen level alone, as Cockle Cove Creek had nitrogen levels 3-5 fold higher than the other systems, predominantly in plant available forms (nitrate, ammonium). The velocity data relates to the inability of drift algae to accumulate if there is no basin or low velocity areas to allow for settling. This is the case in Scorton Creek.

A principal component of the high tolerance of salt marsh systems to nitrogen inputs from groundwater and surface water inflows is that unlike embayments, creek waters cannot accumulate nutrients over multiple tidal cycles as in embayments. In addition, increasing the nitrogen concentration in the tidal waters that flood the marsh plain will have a negligible or possibly a stimulatory effect on marsh primary and likely secondary production (i.e. an enhancement of habitat). In addition, since the inflowing fresh waters flow down gradient through the marsh creek and out to Cape Cod Bay, the nitrogen levels will never exceed the inflowing freshwater nitrogen level.

For the Cockle Cove Creek system, it was determined that a highly conservative nitrogen threshold would yield a total nitrogen level of $\leq$2 mg N L^{-1} throughout the salt marsh, e.g. from headwaters to tidal inlet. As this system closely resembles the structure and hydrodynamics of Scorton Creek, this threshold level appears to be appropriate for Scorton Creek as well. It should be noted that the upper most marsh reach of Cockle Cove is currently exposed to 2-3 mg N L^{-1} and in Little Namskaket Marsh the range is 1.044 - 0.604 mg N L^{-1}., without discernible habitat impairment. Also, it is important to note that since the creek bottom sediments remove nitrate during transport, the TN concentration declines along the tidal reach. As such, the lower tidal reach has a significantly lower tidally averaged concentration compared to the headwaters. This can be seen in the existing TN gradient, where average TN in Scorton Creek headwaters are 0.66 mg N L^{-1} and 0.32 mg N L^{-1} near the inlet.

Putting all the assessment elements together, it appears that for Scorton, the critical values are a total nitrogen level of 2 mg N L^{-1} in the headwaters and a level of 1 mg N L^{-1} in the mid upper reach of the main channel (Station SC-8). This station is associated with the upper marsh infauna habitat and is not significantly diluted by direct freshwater inflows (like stations SC-3 or SC-7) and therefore (SC-8) was selected as the sentinel station for this system. The threshold (tidally averaged) total nitrogen level of 1 mg N L^{-1} was determined to be appropriate for the sentinel station (SC-8). It should be noted that the tidally averaged total nitrogen level at the middle marsh station in Cockle Cove Creek is currently 1.378 mg N L^{-1} and the tidal inlet station shows concentrations of 0.472 mg N L^{-1}, consistent with 1 mg N L^{-1} at the sentinel station in Scorton Creek. This threshold applies as long as the tidal creek maintains its present hydrodynamic characteristics (flushing and velocity). The nitrogen threshold for Scorton Creek is intentionally conservative based upon all available data from comparable systems. While not intuitive, the threshold in Scorton Creek is more restrictive of nitrogen loading than for Namskaket Marsh, since the station if further upgradient in the system than in Namskaket Marsh. This results from the fact that TN levels between the sentinel station in general are lower than at the sentinel station. However, it indicates that additional nitrogen may enter this system without impairment of its habitat quality throughout the estuary. The nitrogen loads associated with the threshold concentration at the sentinel location are discussed in Section VIII.3, below.

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 for infaunal habitats in the Scorton Creek Estuary. Contrary to most other estuarine systems evaluated as part MEP, the threshold concentration was set higher than present conditions, meaning that the system would be allowed to have a higher load than present to meet the threshold. The threshold level, at the sentinel station SC-

8, was set at 1.0 mg/L for the Scorton Creek system. It is important to note that load increases could be produced by increasing any or all sources of nitrogen to the system.

The nitrogen threshold developed in the previous section was used as a basis for determining whether or not the total nitrogen mass loading to Scorton Creek would require a reduction in order to maintain the existing habitat quality in the Scorton Creek Estuary. Total nitrogen thresholds derived in Section VIII.1 indicated that TN concentrations based on existing conditions and build-out conditions were well below the assimilative capacity of the system. Contrary to most other estuarine systems evaluated as part MEP but similar to conditions observed in Namskaket Marsh and Little Namskaket Marsh (Town of Orleans), the threshold concentration was set higher than present and build-out conditions, meaning that the system would be allowed to have a higher load than present and still meet the threshold. Therefore, watershed nitrogen loads were not lowered as loads are already well below the threshold. It is important to note that load increases could be produced by increasing any or all sources of nitrogen to the system, however, even at build-out, the loads to the Scorton Creek Estuary are below the systems assimilative capacity.

The nitrogen thresholds discussed in the previous section are the loads developed for the Present and Build-Out Loading Conditions as presented in Section VI. Based on the analysis of watershed loads, water quality parameters, and infaunal habitat, it was determined that the system would be allowed to have a higher load than present and still meet the threshold.

The septic loading is shown in Table VIII-2. The nitrogen septic loads have not changed from the Present Conditions loads developed in Section VI.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and build-out loading scenarios of the Scorton Creek 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 % change
Scorton Creek	23.532	115.373	+390.3%
Surface Water Sources			
Long Hill Creek	3.468	22.422	+546.4%
Jones Lane	2.049	9.586	+367.8%

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 septic loads depicted in Table VIII-2. The total nitrogen loads for Scorton Creek are presented in Table VIII-4. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water 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 Cape Cod Bay.

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 Scorton Creek 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
Scorton Creek	32.403	124.244	+283.4%
Surface Water Sources			
Long Hill Creek	4.584	23.537	+413.5%
Jones Lane	2.753	10.290	+273.7%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Scorton Creek 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)
Scorton Creek	124.244	0.395	0.614
Surface Water Sources			
Long Hill Creek	23.537	--	--
Jones Lane	10.290	--	--

Although the loading scenarios shown in Table VIII-5 provide one manner of achieving the selected threshold level for the sentinel site within the estuarine system, the specific example does not represent the only method for achieving this goal. However, the thresholds analysis provides general guidelines needed for the nitrogen management of this embayment.

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Scorton Creek. Sentinel threshold station is in bold print.					
Sub-Embayment	monitoring station	present (mg/L)	build out (mg/L)	threshold (mg/L)	Threshold % change
Cape Cod Bay	SC-14	0.320	0.320	0.325	+1.6%
Scorton Creek - Inlet	SC-13	0.391	0.410	0.594	+51.7%
Scorton Creek	SC-11	0.405	0.425	0.629	+55.3%
Scorton Creek	SC-9	0.449	0.477	0.775	+72.8%
Scorton Creek	**SC-8**	**0.513**	**0.555**	**1.006**	**+96.0%**
Scorton Creek	SC-4	0.540	0.588	1.117	+107.0%
Scorton Creek	SC-12	0.474	0.560	1.146	+141.9%
Scorton Creek	SC-6	0.486	0.530	0.971	+100.0%
Scorton Creek	SC-10	0.627	0.710	1.534	+144.7%
Scorton Creek	SC-2	0.469	0.501	0.844	+80.1%
Scorton Creek	SC-3	0.654	0.716	1.377	+110.6%
Scorton Creek	SC-5	0.663	0.803	2.452	+270.0%

IX. MANAGEMENT SCENARIO

IX.1 BACKGROUND

In 2012, the Town of Sandwich wastewater consultants, Wright-Pierce, created a Needs Assessment for the town's Comprehensive Water Resource Management Plan (CWRMP) (WP, 2012). As part of guidance from Massachusetts Department of Environmental Protection (MassDEP) community-based wastewater planning activities, a needs assessment identifies the wastewater needs in a town and is followed sequentially by: a) development and screening of alternatives to meet the identified needs and b) development of a comprehensive plan including details for all resulting preferred alternatives. The plan usually goes through state (MEPA) and county (DRI) regulatory review.

MEP assessments are usually a key for determining whether there are ecosystem and water quality problems or identified needs in target estuaries. If the MEP assessment indicates that a system is impaired, the MEP report will include a recommended nitrogen threshold concentration. If this threshold concentration is attained at one or more select "sentinel" stations, the system will be restored. These recommended nitrogen thresholds are typically adopted as Total Maximum Daily Loads (TMDLs) by MassDEP and EPA and, thus, are established as regulatory thresholds or restoration targets under the Clean Water Act. If an estuary is identified as impaired in a MEP report, MEP staff typically evaluates two options or scenarios to meet the N threshold: 1) review of inlet restrictions if collected MEP data indicates there is dampening of the tidal signal at the inlet and 2) development of one watershed nitrogen loading reduction strategy that attains the threshold concentration. This second scenario, which is called the threshold scenario, typically involves sequential reductions in wastewater loads within MEP subwatersheds until the threshold is just attained. This threshold scenario option is only one example to show that the threshold can be attained, but is not informed by either the cost or advisability of sewering. Other alternative options could also attain the threshold and often communities choose to evaluate a variety options to attain the threshold after a MEP report is completed.

Because the MEP report for the Sandwich Harbor estuary was not available at the time the Sandwich CWRMP was being prepared, Wright-Pierce incorporated a number of assumptions about Scorton Creek, including that it was impaired and required nitrogen reductions within its watershed to meet a prospective TMDL. Wright-Pierce proceeded with an analysis in the CWRMP that utilized the Cape Cod Commission watershed to Scorton Creek and a required 20% nitrogen reduction within the Scorton Creek watershed.

As indicated above, the MEP assessment has provided refinements that counter the assumptions in the CWRMP analysis. The MEP watersheds to Scorton Creek (see Figure III-2) provide a more refined watershed delineation that includes validation through the collection of streamflow data (see Section IV). In addition, and more importantly, the MEP analysis indicates that Scorton Creek is a healthy ecosystem and with excess capacity available for additional nitrogen loading. This capacity means that there should be no required reductions in nitrogen loading within the Scorton Creek watershed. It also means that additional nitrogen could be added to the watershed and the Scorton Creek ecosystem would remain healthy.

With the excess nitrogen loading capacity established, MEP staff were requested by the town and Wright-Pierce to complete a simple scenario that involved the following:

- 300,000 gpd effluent discharge at Parcels 28-41 and 28-58 (near the Industrial Park)

- 500,000 gpd effluent discharge at Parcel 28-102 (west of Quaker Meetinghouse Road near Oak Ridge School);
- 500,000 gpd effluent discharge at Parcel 34-002 (High School site)
- All effluent discharge at 8 mg/L TN
- All effluent imported into the Scorton Creek watershed

All potential effluent discharge sites are located within the Long Hill Creek subwatershed (Figure IX-1). As indicated in Sections III and IV, this subwatershed had a MEP gauge with data that indicated that only 40% of the watershed flow is discharged through the flapper valve into Scorton Creek. As currently conceptualized, the remaining watershed flow is discharged through the barrier beach due to regular ponding upstream of the flapper valve. For the purposes of this scenario, this relationship is maintained and 40% of the proposed additional effluent flow added to Long Hill Creek is discharged into Scorton Creek, while the remainder discharges through the barrier beach and into Cape Cod Bay.

IX.2 BUILD-OUT LOADING SCENARIO RESULTS

Based on the potential sewer options developed by the Town of Sandwich under their ongoing CWMP process, a scenario was developed to be modeled by the MEP. The scenario (1) as developed utilizes wastewater flows that were developed under Build-out conditions provided in Chapter 6, with an additional wastewater load entering the Long Hill Creek watershed. The load supplied by the Town calls for 1,300,000 gallons a day (MGD) at concentration of 8.0 mg/L to enter the watershed. Table IX-1 and Table IX-2 illustrate the overall change to septic and watershed loads resulting from this scenario. Under the build-out scenario shown in Chapter 6, the modeled concentration at the sentinel station is less than the threshold target (1.0 mg/L TN at SC-8). Additional nitrogen loading in the Long Hill Creek watershed does increase the nitrogen concentrations at the threshold station, but Scenario 1 still meets the threshold hold target (Tables IX-3 and IX-4).

Table IX-1. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling Build-out loading conditions and for Scenario 1. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	Build-out septic load (kg/day)	Build-out Scenario 1 septic load (kg/day)	septic load % change
Scorton Creek	31.208	31.208	0.0%
Surface Water Sources			
Long Hill Creek	6.477	14.142	118.4%
Jones Lane	2.616	2.616	0.0%

These scenario results do not account for potential subwatershed changes as a result of discharging the selected effluent volumes. The selected volumes of discharge at sites 2, 3, and 4 may have the potential to alter the groundwater flow paths near them enough to alter the subwatershed delineations. All of these sites are adjacent to subwatershed boundaries. Since site 1 is located near the center of its subwatershed, it is unlikely that the planned flow will alter the subwatershed delineation sufficiently to place the site in another subwatershed.

Groundwater modeling would be necessary to fully evaluate these changes and this is beyond the scope of the MEP.

Table IX-2. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of Build-out conditions and for Build-out Scenario 1. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	Build-out load (kg/day)	Build-out Scenario 1 load (kg/day)	% change
Scorton Creek	40.079	40.079	0.0%
Surface Water Sources			
Long Hill Creek	7.592	15.258	101.0%
Jones Lane	3.321	3.321	0.0%

Table IX-3. Sub-embayment loads used for total nitrogen modeling of the Scorton Creek system for build-out loading scenario with loading conditions for Build-out Scenario 1, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	scenario load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Scorton Creek	40.079	0.395	-0.004
Surface Water Sources			
Long Hill Creek	15.258	--	--
Jones Lane	3.321	--	--

Table IX-4. Comparison of model average total N concentrations from Present loading, Build-out loading, and the modeled Build-out Scenario1, with percent change, for the Scorton Creek. Sentinel threshold station is in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build out (mg/L)	Scenario 1 (mg/L)	Scenario % change
Cape Cod Bay	SC-14	0.320	0.320	0.320	+0.1%
Scorton Creek - Inlet	SC-13	0.391	0.410	0.416	+1.4%
Scorton Creek	SC-11	0.405	0.425	0.427	+0.6%
Scorton Creek	SC-9	0.449	0.477	0.479	+0.5%
Scorton Creek	**SC-8**	**0.513**	**0.555**	**0.557**	**+0.4%**
Scorton Creek	SC-4	0.540	0.588	0.590	+0.4%
Scorton Creek	SC-12	0.474	0.560	0.672	+20.1%
Scorton Creek	SC-6	0.486	0.530	0.531	+0.2%
Scorton Creek	SC-10	0.627	0.710	0.732	+3.1%
Scorton Creek	SC-2	0.469	0.501	0.504	+0.5%
Scorton Creek	SC-3	0.654	0.716	0.718	+0.3%
Scorton Creek	SC-5	0.663	0.803	0.803	+0.0%

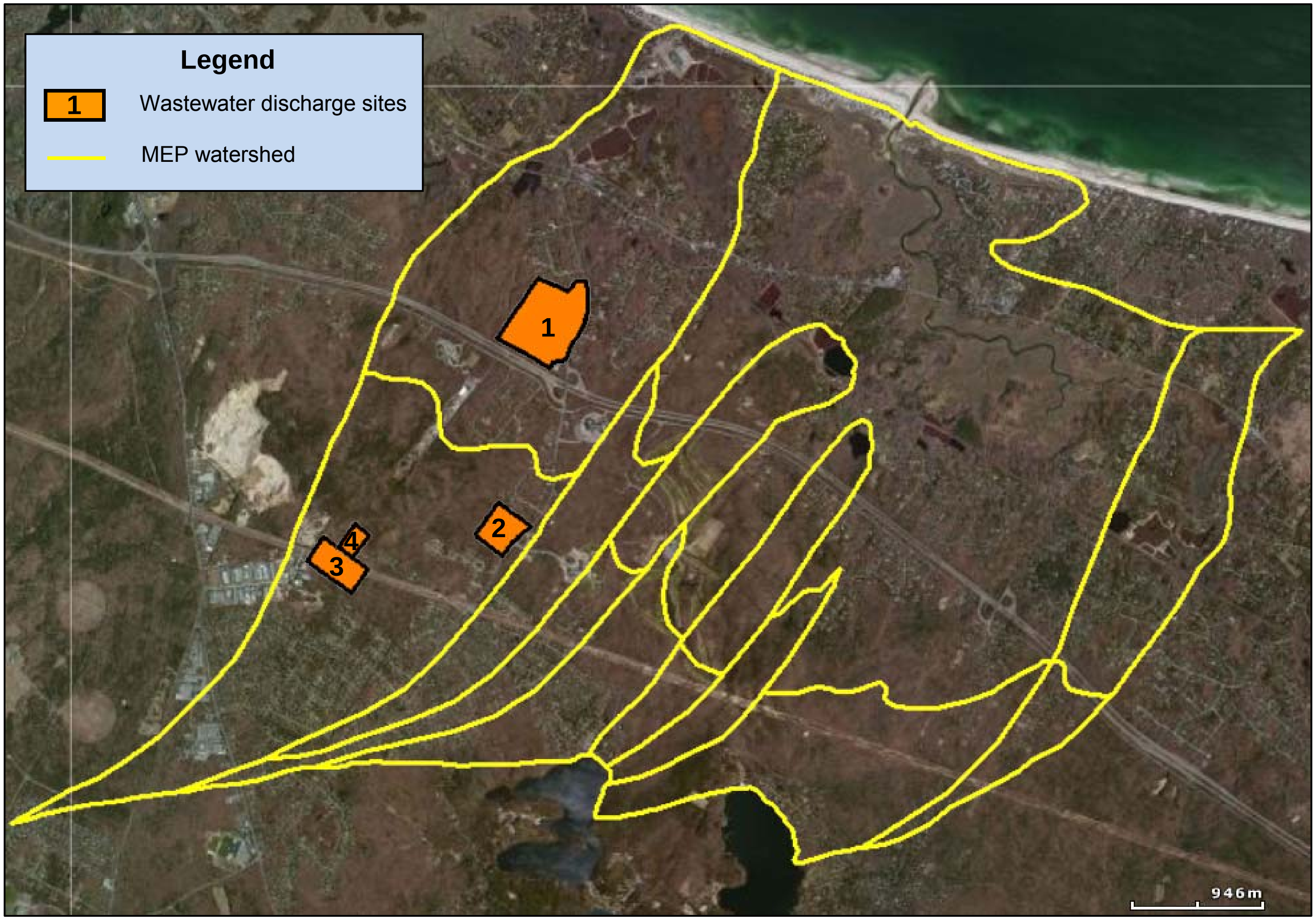


Figure IX-1. Scenario wastewater effluent discharge sites. Orange sites are effluent discharge sites selected in a scenario developed by Wright-Pierce and the town. All sites are within the Long Hill Creek subwatershed: site 1 (parcel 34-002; Sandwich High School) is located within Long Hill Creek LT10 (subwatershed #15), site 2 (parcel 28-102; west of Quaker Meetinghouse Road) is located within Long Hill Creek GT10 (subwatershed #14), and sites 3 and 4 (parcels 28-41 and 28-58, respectively; near the Industrial Park) are located within Long Hill Creek GT10 (subwatershed #14). Total effluent discharge for all sites is 1.3 million gallons per day with an effluent total nitrogen concentration of 8 mg/L.

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