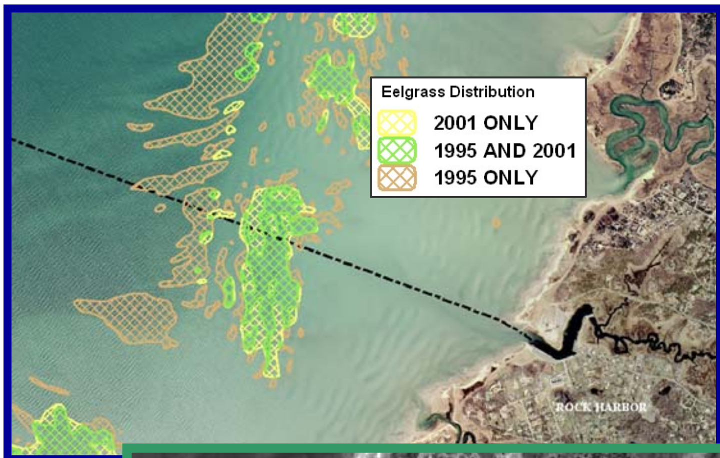
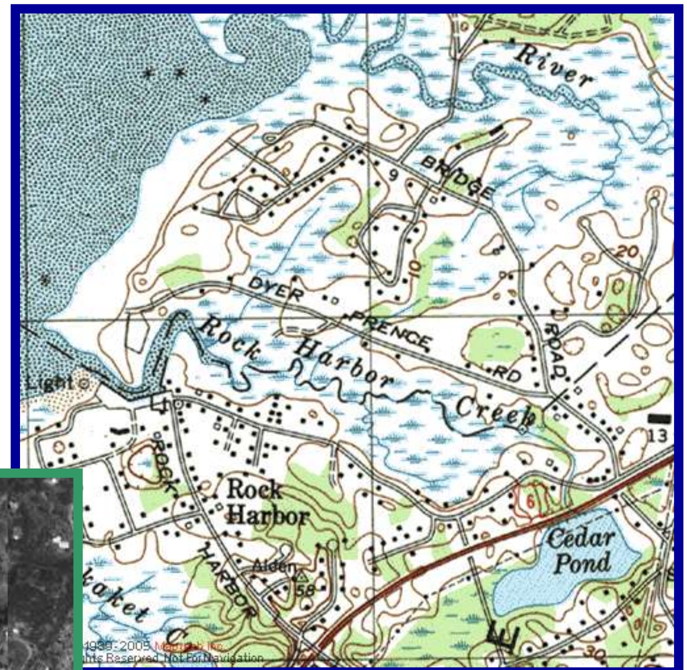
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Threshold for the Rock Harbor Embayment System, Orleans, Massachusetts





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Massachusetts Department of
Environmental Protection

FINAL REPORT – December 2008

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 Rock Harbor embayment system, a coastal embayment situated within the Town of Orleans, Massachusetts. Analyses of the Rock Harbor embayment system was performed to assist the Town with up-coming nitrogen management decisions associated with current and future wastewater planning efforts, as well as wetland restoration, anadromous fish runs, shell fishery, open-space, and harbor maintenance programs. As part of the MEP approach, habitat assessment was conducted on the embayment based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure.

Nitrogen loading thresholds for use as goals for watershed nitrogen management are the major product of the MEP effort. In this way, the MEP offers a science-based management approach to support the Town of Orleans 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 Rock Harbor embayment, (2) identification of all nitrogen sources (and their respective N loads) to embayment waters, (3) nitrogen threshold levels for maintaining Massachusetts Water Quality Standards within embayment waters, (4) analysis of watershed nitrogen loading reduction to achieve the N threshold concentrations in embayment waters, and (5) a functional calibrated and validated Linked Watershed-Embayment modeling tool that can be readily used for evaluation of nitrogen management alternatives (to be developed by the Town) for the restoration / protection of the Rock Harbor 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 Rock Harbor embayment system within the Town of Orleans is at risk of eutrophication (over enrichment) in its lower reaches due to enhanced nitrogen loads entering through groundwater and surface water from the increasingly developed watersheds to this complicated estuarine system that includes significant areas of salt marsh. 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 Towns that exist in the Rock Harbor watershed (including the Town of Eastham) have recognized the severity of the problem of eutrophication and the need for watershed nutrient management. Concern over declining resource quality of the estuarine systems of Orleans (inclusive of Rock Harbor) prompted the Town of Orleans to initiate the town-wide Orleans Water Quality Monitoring Program in 2001, which continues in a reduced form through present (2008). The 2001 Program was an expansion of a previous effort targeting Pleasant Bay, begun in 1997 by the Orleans Water Quality Task Force. The town-wide monitoring program is focused on restoring and protecting the estuarine habitats associated with the Town of Orleans and is being undertaken in concert with the DEP/SMAST Massachusetts Estuaries Project. This is a collaborative effort whereby the Town of Orleans provides the support, coordination and oversight of the program through its Planning Office and through its Wastewater Management Steering Committee and SMAST provides the technical and analytical aspects needed for the project through the MEP Technical Team.

The investigations undertaken prior to the Massachusetts Estuaries Project analysis summarized in this report provided significant information related directly to the implementation of the MEP Linked Management Modeling Approach and helped yield insight into the interpretation of the results. In addition, the Town of Orleans' comprehensive Water Quality Monitoring Program was of sufficient rigor to be used as the water quality baseline required for the MEP threshold analysis presented in this MEP Technical Report.

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 Rock Harbor embayment system by using site-specific data collected by the MEP and water quality data from the Orleans Water Quality Task Force Water Quality Monitoring Programs (see Chapter 2). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Planning Departments in each of the Towns represented in the Rock Harbor watershed, and watershed boundaries delineated by USGS. This land-use data was used to determine watershed nitrogen loads within the Rock Harbor embayment system and associated sub-embayments (current and build-out loads are summarized in Chapter IV). 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 Rock Harbor embayment system. Once the hydrodynamic properties of the estuarine system was 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 bio-available and total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis while nitrogen entering the coastal embayment was quantified by direct measurement of stream nutrient concentrations and freshwater flow, predominantly groundwater, in streams discharging directly to the embayment. Boundary nutrient concentrations in the Cape Cod Bay source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Rock Harbor 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 approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout and embayment system, is to first identify a sentinel location within the embayment and second to determine the nitrogen concentration within the water column which will restore that location to the desired habitat quality (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 Rock Harbor System, the restoration target should reflect both recent pre-degradation habitat quality and be reasonably achievable. The load reductions presented in the report represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation in this report of load reductions aims to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen reduction guidelines for nitrogen management of the Rock Harbor embayment system (a unique combination of salt marsh and open water in the form of an artificially created open water basin). Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. These scenarios should be developed in coordination with both the Towns in the Rock Harbor watershed in order to effectively examine the effect of load reductions on water column nutrient concentrations. The MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of each embayment system. The concept was that since septic system nitrogen loads generally represent 88%-92% of the controllable watershed load to the Rock Harbor embayment system and are more manageable than other of the nitrogen sources, the ability to achieve needed reductions through this source is a good gauge of the feasibility for restoration of these systems.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout Rock Harbor system based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. The Rock Harbor system is showing high habitat quality throughout its upper salt marsh reach (above WMO-17) and significant habitat impairment in its lower "embayment" reach (e.g. harbor portion, WMO-17 to inlet). The upper reach appears to be a fully functional tidal salt marsh with deeply incised narrow creeks surrounded by significant areas of emergent marsh. This reach is typical of New England "pocket" marshes, with smaller tidal creeks and a marsh plain dominated by low marsh and high marsh plant communities, along with patches of fringing brackish marsh vegetation.

In Contrast, the lower "embayment reach, comprised primarily of the harbor basin, functions as a small open water cove or harbor. This basin is depositional by structure, collecting both algal and salt marsh organic matter with accumulation of anoxic organic-rich fine sediments (sulfidic); it is highly tidal, with sufficient light penetration to allow periodic development of benthic algal mats; and its tidal inlet is influenced by sand transport via nearshore coastal processes associated with adjacent Cape Cod Bay. These features in combination with the observed levels of summer oxygen depletion (to 2 mg L^{-1}), indicate a significantly impaired habitat. This assessment is supported by the impoverished infaunal animal community which is dominated by small opportunistic stress indicator species common to disturbed or organic matter enriched basins.

Based upon all available information the present lack of eelgrass throughout the Rock Harbor System does not appear to be a response to watershed sourced nitrogen loading (e.g. changing watershed land-use). Instead, the absence of eelgrass habitat appears to result from the structure of the upper reach supportive of salt marsh and the lower reach being a maintained depositional basin. The absence of eelgrass within the harbor basin is likely the result of its configuration, in that it is a "relatively deep" depositional basin. In addition, in the lower reach, harbor activities also likely have limited the potential for colonization of this system. Most important relative to MEP nitrogen thresholds analysis, it does not appear that eelgrass beds have been present within the Rock Harbor System at any time over the past century, as indicated by MassDEP Eelgrass Mapping Program analysis and MEP Technical Team historical analysis. Therefore, nitrogen threshold development for protection/restoration of this estuarine system will necessarily focus on restoration of the impaired infaunal habitat within the harbor (embayment reach) and protection of the high quality infaunal habitat within the upper salt marsh reach

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 Rock Harbor embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 88%-92% of the controllable watershed nitrogen load to the embayment was from wastewater.

The Rock Harbor estuary is a composite of 2 different estuarine systems; an upper salt marsh reach and a lower embayment reach (the harbor). These systems have very different tolerances to nitrogen enrichment and are presently supporting habitats of very different quality, with high quality habitat in the salt marsh and significantly impaired habitat quality within the embayment (Table VIII-1). Since the embayment reach is clearly over its ability to assimilate its present nitrogen load without impairment, restoration will require nitrogen management. A correlative effect of nitrogen management will be that nitrogen loading to the upper salt marsh will also be reduced, although the upper salt marsh is well below its likely nitrogen loading threshold level to maintain its high quality habitats. Therefore the nitrogen threshold for the Rock Harbor System focusing on the habitat quality of the embayment reach will necessarily be protective of the upper salt marsh reach.

The threshold nitrogen levels for the Rock Harbor embayment system in the Towns of Orleans and Eastham were determined as follows:

Rock Harbor Threshold Nitrogen Concentrations

- As a result of the present significant impairment of the infaunal habitat within the embayment reach of the Rock Harbor Estuary and given that there is no evidence that this system has supported eelgrass over the past century, the threshold development necessarily focuses on the embayment reach. The threshold for restoring and maintaining high quality infaunal habitat within the embayment reach of Rock Harbor is 0.500 mg TN L^{-1} (tidally averaged) at the sentinel station located at the head of the harbor (upper region of harbor basin, Town of Orleans Water Quality Monitoring Program station WMO-17). .

- At present, the embayment reach of the Rock Harbor System has elevated total nitrogen levels (0.686 mg N L^{-1}, tidally averaged), with stressful levels of summer oxygen depletion (to 2 mg L^{-1}), sulfidic sediments and depleted infaunal communities dominated by stress indicator species. These observations strongly support the contention that this basin is significantly impaired through nitrogen enrichment. As this basin does not presently support high quality infaunal habitat, the nitrogen threshold analysis was based upon comparisons to a number of small embayments on Cape Cod.

- Based upon these observations, the MEP Technical Team concluded that an upper limit of 0.50 mg N L^{-1} tidally averaged TN would support healthy infaunal habitat in the lower embayment reach of the Rock Harbor System. Equally important, lowering nitrogen levels from the present 0.686 mg N L^{-1} to the threshold 0.500 mg N L^{-1} will lower nitrogen levels within the upper salt marsh (e.g. WMO-18, from 0.829 to 0.615 mg N L^{-1}), protective of those habitats. Therefore, it appears that achieving the nitrogen target at the sentinel location is restorative of infaunal habitat throughout the lower basin and protective of habitats within the upper salt marsh reach.

- The nitrogen concentration thresholds developed through the MEP analysis were used to determine the amount of total nitrogen mass loading reduction required for restoration of infaunal habitats in the Rock Harbor system. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for Rock Harbor. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented in the report represent only one of a suite of potential reduction approaches that need to be evaluated by the community. As discussed in Chapter 8, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required nearly 70% removal of septic load (associated with direct groundwater discharge to the embayment) for the entire system. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1

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

Table ES-1. Existing total and sub-embayment nitrogen loads to the estuarine waters of the Rock Harbor system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations. Loads to estuarine waters of the Rock Harbor system include both upper watershed regions contributing to the major surface water inputs.

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. [8] (mg/L)
ROCK HARBOR SYSTEM										
Rock Harbor	0.822	1.126	6.787	0.065	7.978	0.079	1.382	9.4393	0.66-1.10	--
Rock Harbor Creek	0.107	0.177	0.910	-	1.088	0.000	0.000	1.088	1.14	--
Rock Harbor System Total	0.929	1.303	7.698	0.065	9.066	0.079	1.382	10.527	**0.66-1.14**	**0.500**

[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 unattenuated 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 data collected between 2001 and 2006, ranges show the upper to lower regions (highest-lowest) of the indicated sub-embayment.
[8] benthic infauna threshold for sentinel site located at the head of the lower basin of the Harbor system.

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Acushnet River system.

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
ROCK HARBOR SYSTEM						
Upper Basin	7.978	2.633	0.079	1.207	3.919	-67.0%
Acushnet River (fresh water)	1.088	1.088	0.000	0.000	1.088	0.0%
Rock Harbor System Total	9.066	3.720	0.079	1.207	5.006	-59.0%

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

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The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the critical coastal resources of the Rock Harbor Embayment System and supported the application of the Linked Watershed-Embayment Model to Determine the Critical Nitrogen Loading Threshold for this system. Without these stewards and their efforts, this project would not have been possible.

The Wastewater Management Steering Committee and the Program Technical Team the Orleans Water Quality Monitoring Program was initiated in 2001 and continued through 2002 as a collaborative effort between the Coastal Systems Program at the School for Marine Science and Technology-UMass and the Town and Citizens of Orleans. The Program is coordinated by the Orleans Planning Department, George Meservey, and the Orleans Wastewater Management Steering Committee, Gussie McKusick.

First and foremost we would like to recognize and applaud the significant time and effort in data collection and discussion spent by members of the Town of Orleans Water Quality Monitoring Program and Water Quality Task Force. These individuals gave of their time to collect nutrient related water quality from this system, without which the present analysis would not have been possible. Of particular note are George Meservey of the Orleans Planning Department and Gussie McKusick of the Orleans Wastewater management Steering Committee, who were instrumental in instituting and initially coordinated the program and the recent Coordinator, Judy Scanlon. Similarly, many in the Town of Orleans helped in this effort since 2001, but particularly the members of the Wastewater Management Steering Committee, the Town Planning Office and members of the Board of Selectman.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by Tom Cambareri and Margo Fenn of the Cape Cod Commission; MaryJo Feurbach and Art Clark of the USEPA; and our MADEP colleagues: Andrew Gottlieb (now with the County Wastewater Collaborative), Arleen O'Donnell, Art Screpetis, Rick Dunn, Steve Halterman, and Russ Issacs. We are also thankful for the long hours in the field and laboratory spent by the technical staff, interns and students within the Coastal Systems Program at SMAST-UMD.

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I. INTRODUCTION

Rock Harbor is approximately a 50 hectare (124 acre) salt marsh on the northeastern coast of Cape Cod, with adjacent similar systems of Namskaket Marsh and Little Namskaket Marsh. These three salt marsh systems are located within the Towns of Orleans, Eastham and Brewster on Cape Cod Massachusetts. All three of these systems are situated on the north shore of Cape Cod and are tidal salt marshes receiving tidal flood water from Cape Cod Bay. Though all three systems are typical salt marshes, the Rock Harbor system has been significantly modified to create a harbor just inside the inlet. The harbor area encompasses the lower quarter of the main tidal creek and supports a small marina and docks for private and commercial fishing vessels. Both Namskaket Marsh and Little Namskaket Marsh have small, deeper water areas immediately inside their inlets to Cape Cod Bay. However, in both of these systems these regions are "natural" and do not retain significant water volume at low tide. Each of these three estuarine systems is predominantly tidal salt marsh with a central tidal creek and smaller tributary tidal creeks. Only Rock Harbor, exhibits "open water" embayment characteristics, associated with the dredged harbor basin. The watersheds contributing nitrogen to the waters of the Rock Harbor, Little Namskaket Marsh and Namskaket Marsh systems are distributed among the Towns of Orleans and Eastham, Orleans and Orleans and Brewster, respectively. Restoration of any degraded habitats within these estuarine salt marsh systems will depend upon the coordinated efforts of these municipalities and their citizens.

Overall, the Rock Harbor system is a typical New England salt marsh dominated by a central tidal creek and emergent marsh colonized by low marsh (*Spartina alterniflora*) and high marsh (*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 tide, 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 from its watershed. Observations by the USGS and the MEP Technical Team indicate a healthy functioning New England Salt Marsh.

Rock Harbor provides both wildlife habitat and a nursery to offshore fisheries, as well as serving as a storm buffer and nutrient sink for watershed derived nitrogen. In recognition of its natural beauty and critical value to regional environmental quality, Rock Harbor has been designated as an Area of Critical Environmental Concern and an Outstanding Resource Water by the Commonwealth of Massachusetts [Massachusetts General Law, chapter 21a, sections 2(7) and 40(e)].

Rock Harbor is a small estuary behind a barrier beach formed by coastal processes associated with nearshore Cape Cod Bay. This estuarine system 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. At present, streams are relatively small and discharge only a small portion of the aquifer recharge to the estuary. Streams are not well formed, with almost all freshwater entering through groundwater seepage either to the restricted wetlands at the head of the salt marsh, groundwater seepage to Cedar Pond or directly through marginal and creek bottom seepage.

Tidal exchange with Cape Cod Bay is through a single inlet through the barrier beach. 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 emergent salt marsh would be reduced. Any long term habitat management plan for the Rock Harbor System must recognize the importance of inlet dynamics and include options to maintain tidal exchange.

The primary ecological threat to Rock Harbor resources is degradation resulting from nutrient enrichment. 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 Nauset 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 Orleans has been among the fastest growing towns in the Commonwealth over the past two decades and does not have centralized wastewater treatment, although it is associated with the Tri-Town Septage Treatment Facility, located in the upper Namskaket Marsh watershed. As levels of nitrogen loading to coastal systems continue to increase, concern has grown in outer Cape Cod Towns over associated nutrient impacts.

Fortunately for the resource protection of Rock Harbor, its function as a tidal salt marsh makes it more tolerant of watershed nitrogen inputs than coastal embayments, like nearby Town Cove or Pleasant Bay. The greater sensitivity of embayments versus wetlands results from their lower tidal exchange rates, the fact that there is limited to no exposure of the sediments to the atmosphere at low tide (like the marsh plain), and the fact that these systems have evolved under much lower levels of productivity and organic matter loading than wetlands. For example, the organic carbon content of New England Salt Marsh vegetated sediments can frequently reach 20%, while embayment sediments are generally in the 1%-5% range. Yet another difference between system types is that oxygen depletion in the creeks of *pristine* wetlands can normally 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 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 levels about 7 times the natural background N input (75.6 g N m^{-2} each growing season).

Salt marsh creek bottoms and creek banks (such as those found in Rock Harbor Marsh, Little Namskaket Marsh and the Namskaket Marsh areas) 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 all three systems.

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 these New England salt marshes (like Little Namskaket and Namskaket Creek) 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 Rock Harbor System, relates to potential macroalgal issues.

The Towns of Orleans and Eastham are the primary stakeholders to the Rock Harbor Marsh Estuary, the focus of this report. The Town of Orleans was among the first communities to become concerned over the existing and potential future degradation of its estuaries. As such, the Town of Orleans (via the Planning Office) undertook a town-wide monitoring effort, inclusive of Rock Harbor. The comprehensive water quality monitoring program was developed to acquire the necessary background water quality monitoring data from all the embayment systems in the Town of Orleans 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 Orleans Water Quality Monitoring Program is consistent with that generated by the Town of Harwich and Chatham as well as the Pleasant Bay Alliance making the data for all the systems in this area cross comparable.

The common focus of the water quality monitoring efforts undertaken by the Towns of Orleans has been to gather site-specific data on the current nitrogen related water quality throughout the Rock Harbor, Namskaket Marsh and Little Namskaket Marsh systems as well as, Nauset Marsh and portions of Pleasant Bay as they exist within the Town of Orleans. 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 combined water quality data sets from the Orleans Water Quality Monitoring Program form a baseline from which to gauge long-term changes as watershed nitrogen management moves forward. This data has already proven to be of high quality and adequate for the development of management thresholds for each of the Town's coastal systems, as was demonstrated in the MEP nitrogen threshold analysis completed for the Pleasant Bay System (Howes et al. 2006). The Orleans Water Quality Monitoring Program efforts allowed the MEP to prioritize all of the Orleans systems for the next step in the restoration/protection and management process.

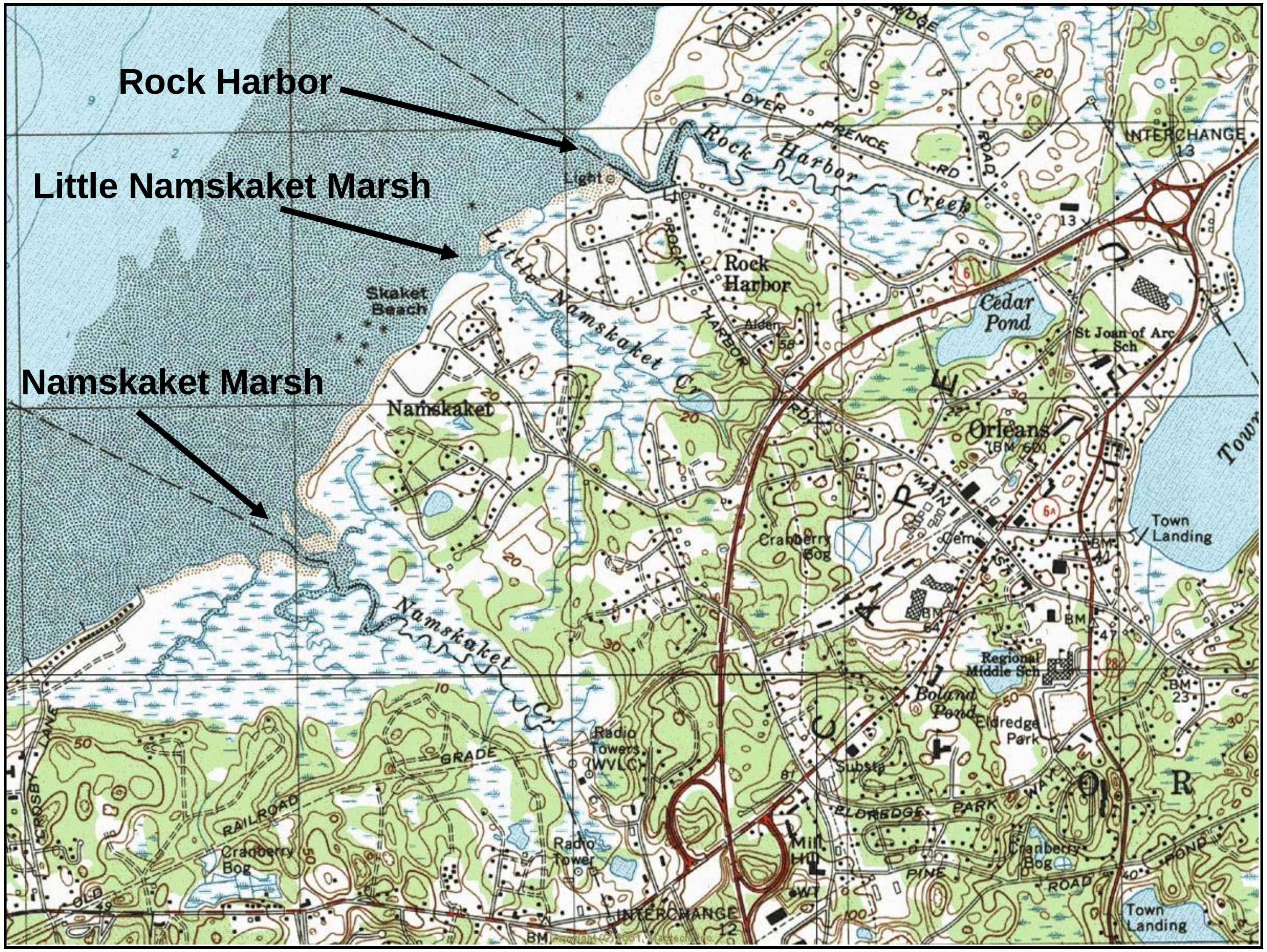


Figure I-1. Study region for the Massachusetts Estuaries Project nitrogen thresholds analysis for Rock Harbor, Little Namskaket Marsh and Namskaket Marsh. Tidal waters enter the system through one inlet in each of the three systems. Flood waters enter the systems from Cape Cod Bay. Freshwaters enters the head of each system from the watershed primarily through 1 surface water discharge point in each marsh, as well as direct groundwater discharge.

The MEP effort builds upon the efforts of the water quality monitoring program, and previous hydrologic and water quality analyses undertaken by the US Geological Survey (as in Namskaket Marsh), and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Rock Harbor, Namskaket and Little Namskaket marsh systems.

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 Orleans for restoration of the its impaired embayment 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 Orleans to develop and evaluate the most cost effective nitrogen management alternatives to restore those valuable coastal resources of Orleans that have been degraded by nitrogen overloading.

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

Municipalities are seeking guidance on the assessment of nitrogen sensitive embayments, as well as available options for meeting nitrogen goals and approaches for restoring impaired systems. Many of the communities have encountered problems with "first generation" watershed based approaches, which do not incorporate estuarine processes. The appropriate method must be quantitative and directly link watershed and embayment nitrogen conditions.

This "Linked" Modeling approach must also be readily calibrated, validated, and implemented to support planning. Although it may be technically complex to implement, results must be understandable to the regulatory community, town officials, and the general public.

The Massachusetts Estuaries Project represents the next generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (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 each of the 89 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment model available 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. 30 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 validated and unlike many approaches, accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics (Figure I-2). This methodology integrates a variety of field data and models, specifically:

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

I.2 SITE DESCRIPTION

The three Cape Cod Bay estuaries, associated with the Town of Orleans with Brewster and Eastham, Rock Harbor, Namskaket Creek and Little Namskaket Creek, are among the larger of the coastal salt marsh systems of Cape Cod. As such, these systems are comprised of ca. 500 acres of barrier beach, salt marsh, and tidal flat. These three systems are situated on the northern shore of Cape Cod and each exchanges tidal waters with Cape Cod Bay through a single tidal inlet. The inlets can be significantly affected by longshore sand transport (west to east), where shoaling can impede hydrodynamic exchange at the mouth of each system and, in the case of extreme events, close an existing inlet and open a new one. The existing inlets to the Namskaket Marsh and Little Namskaket Marsh systems are natural inlets and are not armored in any way. In contrast, the inlet to Rock Harbor is armored and a navigational channel is maintained, however, shoals are abundant in the vicinity of the inlet and depths vary significantly. Depths throughout these three systems vary due to their tidal salt marsh characteristics in combination with the large tidal range in Cape Cod Bay. At low tide large areas of these marshes become nearly dry tidal creeks with large exposed tidal flats with little to no water at the mouths of these systems.

Rock Harbor Marsh occupies a rectilinear indentation in the coast sheltered by barrier dunes and beach sands (Oldale et al. 1971). The present configuration of the Rock Harbor Salt Marsh System results from a combination of glacially dominated geologic processes including the deposition of glacial outwash deposits and tidal flooding of post-glacial valleys formed primarily by post-glacial surface water erosion of the outwash. This estuarine system 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. Tidal exchange with Cape Cod Bay is through a single inlet through the barrier beach. 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 driven 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 emergent salt marsh would be reduced. Any long term habitat management plan for the Rock Harbor System must recognize the importance of inlet dynamics and include options to maintain tidal exchange.

Similar to the larger adjacent Nauset marshes, Rock Harbor Marsh is a shallow coastal estuary dominated by salt marsh and tidal flats, as well as being located within a watershed that includes glacial outwash plain (Harwich Outwash Plain) and ice contact deposits (Nauset Height ice-contact deposits). 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 is highly permeable and varies 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. In the Rock Harbor Salt Marsh System almost all freshwater enters through groundwater seepage either to the restricted wetlands at the head of the salt marsh, groundwater seepage and surface water flow from Cedar Pond or directly through marginal and creek bottom seepage.

Rock Harbor 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. 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 Rock Harbor creek water is 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 moderately toward the tidal inlet. Organisms associated with these creeks have developed strategies for dealing with these large salinity variations.

The Rock Harbor Marsh Estuary is a typical New England salt marsh dominated by a central tidal creek and emergent marsh colonized by low marsh (*Spartina alterniflora*) and high marsh (*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 tide, 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 from its watershed.

Overall, Rock Harbor Marsh appears presently to be a healthy functioning New England salt marsh as noted by USGS studies (DeSimone et al. 1998) and more recently during the MEP Technical Team's field survey. Nitrogen levels within the tidal creeks do show evidence of impairing the resource. However, the primary ecological concern relative to loss of Rock Harbor Marsh resources is degradation resulting from nutrient enrichment (and possibly restriction of tidal flushing due to coastal processes). 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. The Town of Orleans has been among the fastest growing towns in the Commonwealth over the past two decades and does not have centralized wastewater treatment, although it is associated with the Tri-Town Septage Treatment Facility, located in the upper Namskaket Marsh watershed. As levels of nitrogen loading to coastal systems continue to increase, concern has grown in outer Cape Cod Towns over associated nutrient impacts. However, as noted above, salt marshes are relatively insensitive to degradation by 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 salt marsh system is partitioning of salt marsh and embayment (basins retaining significant tidal volume at low tide, i.e. Town Cove) receptors.

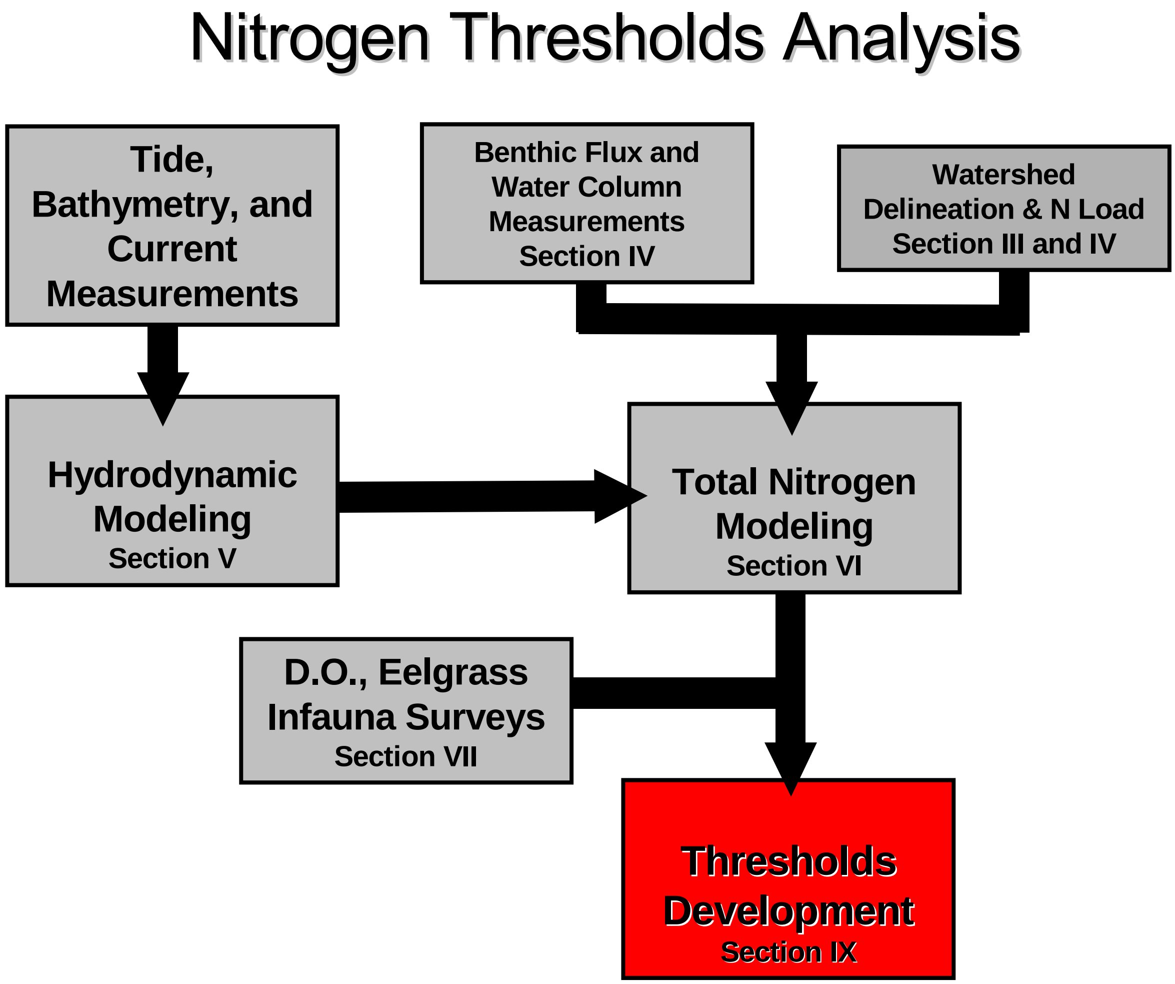


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

I.3 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 sandy glacial outwash aquifers, such as in the watersheds to the Rock Harbor, Namskaket and Little Namskaket Marsh embayment systems, 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 embayment system maintains a capacity to assimilate watershed nitrogen inputs without degradation. However, as loading increases a point is reached at which the capacity (termed assimilative capacity) is exceeded and nutrient related water quality degradation occurs. 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 we have integrated site-specific data on nitrogen levels and the gradient in N concentration throughout the marsh systems monitored by the Town of Orleans 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.4 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important “boundary conditions” for water quality modeling of the three Orleans 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 marshes 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 Monomoy 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 marsh systems themselves was taken from the water quality monitoring program run by the Towns of Orleans (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.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Rock Harbor Marsh Estuary in the Town of Orleans. 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 in developing a variety of alternative nitrogen management options for the marsh systems. Finally, analyses of the marshes was relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging options to improve nitrogen related water quality.

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: 1) excessive plankton and macrophyte growth (which leads to reduced water clarity), 2) organic matter enrichment of waters and sediments, with the concomitant resulting increased rates of oxygen consumption and periodic depletion of dissolved oxygen, (especially in bottom waters), and 3) 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 infaunal communities (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, all of which are dependent upon these highly productive estuarine systems as a habitat and food resource during migration or during different life cycle phases. This process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and pond, it is not a necessarily a part of the natural evolution of a system.

In most marine and estuarine systems, such as the Cape Cod Bay marshes of Namskaket, Little Namskaket and Rock Harbor, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen is controlled through managing inputs relative to tidal removal (via flushing), then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the approach 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. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using estuarine specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments and salt marshes of southeastern Massachusetts and Cape Cod, including the Namskaket Marsh, Little Namskaket Marsh and the Rock Harbor System (each being a combination of embayment in the lower portions and marsh in the upper reaches). 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 or unique features.

Given the structural, ecological, hydrologic and hydrodynamic similarities of the Namskaket Marsh, Little Namskaket Marsh and the Rock Harbor Systems, previous studies related to one system generally were found to provide useful information for the modeling and assessment of the other two. For example, nitrogen levels in flooding waters to Rock Harbor provides a boundary condition to Namskaket and Little Namskaket Marshes as they share common offshore waters. The comparability of the salt marsh regions of these three systems led the MEP Technical Team to evaluate the numerous studies completed over the past two decades as these were related to the nutrient related health of these estuaries. These investigations include both habitat assessments and studies relating to nitrogen loading, hydrodynamics and habitat health. While none of the previous studies provided a holistic view of even an individual system, they provide useful information to the present MEP effort.

The earlier efforts were generally survey studies to evaluate specific estuaries and their watersheds within the larger regional system, to assess the potential for specific watershed nitrogen sources to produce habitat declines within the receiving estuary, or focused studies of processes related to nitrogen cycling within the watershed and down gradient estuarine system. Given the need for diverse data sets to implement the Linked Management Approach, it was important to integrate all available information.

As part of the initial MEP process, the Orleans Wastewater Management Steering Committee and the Program Technical Team compiled more than 25 studies relating to the marine systems of Orleans. Of these 10 were selected as likely to contribute information or quantitative data to the Linked Management Approach. These studies were reviewed in 2001 by Project technical experts (all of whom are currently members of the MEP Technical Team) for (a) information or quantitative data to support the Linked Management Approach, (b) acceptability of results based upon quality assurance or comparability, and (c) data gaps seen in the integrated data set of the existing studies and present Program. Though these studies are related to other embayment systems, such as Nauset Marsh and Pleasant Bay, some information was potentially useful to the current MEP effort. In addition, there are a number of investigations, conducted by the USGS specifically related to nitrogen sources and inputs to Namskaket Marsh. A brief summary of the evaluations of the most pertinent studies is given below.

(1) The Coastal Impact of Groundwater Discharge: an assessment of anthropogenic nitrogen loading in Town Cove, Orleans, MA. Woods Hole Oceanographic Institution, November 1983.

<u>Project Summary</u>: A yearlong investigation of nitrogen transport and transformation within the watershed and estuarine waters of Town Cove was undertaken primarily to guide the Town of Orleans in deciding between a septage treatment and sewage treatment facility. A primary research goal was to determine the role of the watershed nitrogen load in the cultural eutrophication of the Cove and the roles of particulate deposition and remineralization in the productivity of the estuary. Measurements included groundwater nitrogen levels, system freshwater inflow, sediment nitrogen release, phytoplankton productivity and its limitation within the Cove and the potential impacts of increased nitrogen loading to Namskaket Marsh through surface disposal of treated septage effluent from a potential facility (now Tri-Town).

The conclusion of the project was that nitrogen and phosphorus are not limiting the phytoplankton and algal growth within Town Cove; that the Cove is serving as a depositional basin which enhances nitrogen release from the sediments during summer; that the central basin had periodic anoxia of bottom waters which resulted in the seasonal loss of benthic animal communities. Inorganic nutrient data could not be used “in a diagnostic fashion to estimate the degree of eutrophication in the water body”. This results from the rapid incorporation of inorganic nutrients into phytoplankton biomass.

(2) Groundwater Nutrient Concentrations Around Nauset Marsh. Final Report and Data Appendix A, National Park Service, Department of Interior, 1994.

Project Summary: The report describes the methods and results of the first phase of the study - a survey of shoreline groundwater nutrient concentrations (20 cm-40 cm below the water table along the high tide line) to be used for selection of study sites for hydrologic and denitrification measurements. Groundwater (<2ppt by refractometer) was collected for assay of inorganic nitrogen (ammonium, nitrate) and phosphorus (ortho-phosphate) concentrations at 14 shoreline locations around Nauset Marsh within the Cape Cod National Seashore. Samples were collected at 1 to 5 meter intervals along the high tide line. Nitrate levels within most sites were highly variable, frequently ranging over 100 fold, making comparisons to upland development problematic. No ground water flow information was provided.

This phase of the effort was "not intended nor should the data be interpreted as a systematic survey of ground water quality around the Nauset system". A future effort is referred to which would assess the importance of microbial denitrification within the sub-tidal and inter-tidal sediments of the receiving marine systems and the role of specific flow paths and sediment types. Five of the fourteen sites were recommended for this future work (1-Salt Pond Bay, 2-Salt Pond, 3-Weeset, 4-Shore Garden and 5-Eldredge, 4&5 are Orleans shores). The referred to denitrification studies are presented in #3, below. Groundwater nitrate levels were elevated in developed areas, similar to all other areas of Cape Cod. Nitrate levels typically reached 200 - 500 uM at each site with developed uplands.

(3) The Role of Sediment Denitrification in Reducing Groundwater-Derived Nitrate Inputs to Nauset Marsh Estuary, Cape Cod, Massachusetts. B.L. Nowicki, E. Requintina, D. Van Keuren, J. Portnoy, Estuaries 22:245-259.

Project Summary: Five sites within the Nauset Inlet System were assayed seasonally for sediment denitrification rates to gauge their potential impact on reducing groundwater transported nitrate inputs to estuarine waters. The five sites were selected in #2 above, to represent a range of nitrate levels and sediment types at the groundwater discharge sites.

There was no relationship between measured denitrification rates and associated groundwater nitrate concentrations. In fact, denitrification supported by groundwater nitrate was found to be small. Most denitrification was fueled by the coupled nitrification/denitrification within the sediment system. Denitrification was related with sediment organic matter content and varied seasonally with organic content and temperature.

The authors concluded that "denitrification did not contribute significantly to the direct loss of nitrate from incoming groundwater at Nauset marsh estuary. Therefore most groundwater nitrate reaches the estuary.

(4) Namskaket Marsh Studies Relating to Discharge of Treated Septage, by USGS and SMAST scientists (1998).

Project Summary: A large study has been on going within the upper watershed and salt marsh of Namskaket Creek. This investigation has resulted in numerous publications and technical reports. The focus of the study was to determine (a) the fate of nitrogen discharged to the aquifer from the Tri-Town Septage Treatment Facility, (b) predict potential effects on the Namskaket Marsh if the effluent nitrogen is discharged via groundwater, and (c) determine the timing and location of nitrogen discharge in groundwater.

Key findings related to the treated effluent plume from the Tri-Town Septage Treatment Facility located adjacent Namskaket Marsh stem from a number of USGS open file reports and journal articles. The basic results were:

- Assessment of the nitrogen mass to be discharged to the Cape Cod Bay Systems was quantified, but the discharge has still not reached the estuaries;
- the nitrogen plume is not being significantly attenuated during aquifer transport;
- the plume transport direction is sensitive to local recharge rates
- denitrification within the creek bottom sediments of Namskaket Marsh is large relative to the amount of nitrogen loading from the surrounding watershed and will increase if nitrate levels increase in entering freshwater
- groundwater does not enter the marsh through the marsh plain, but through a narrow seepage face at the upland/marsh boundary and through the creek bottoms.

In addition, the ability of the Namskaket Marsh to tolerate nitrogen inputs was evaluated. Nitrogen additions to both the vegetated marsh surface and creek bottoms were found to result in increased nitrogen loss through denitrification in the creek bottoms and increased production of the emergent marsh plants on the marsh plain.

These studies provide the information on the nitrogen tolerance of the salt marsh regions within the Cape Cod Bay Systems needed for the current nitrogen management program. These data will be incorporated into the present program.

Concern over declining resource quality of its estuarine systems led the Town of Orleans to initiate the town-wide Orleans Water Quality Monitoring Program in 2001, which continues in a reduced form through present (2007). The 2001 Program was an expansion of a previous effort targeting Pleasant Bay, begun in 1997 by the Orleans Water Quality Task Force. The town-wide monitoring program is focused on restoring and protecting the estuarine habitats associated with the Town of Orleans and is being undertaken in concert with the DEP/SMAST Massachusetts Estuaries Project. This is a collaborative effort whereby the Town of Orleans provides the support, coordination and oversight of the program through its Planning Office and through its Wastewater Management Steering Committee and SMAST provides the technical and analytical aspects needed for the project through the MEP Technical Team.

The Water Quality Monitoring Program uses primarily trained volunteers for the field data collection efforts. The sampling teams are equipped and trained prior to sampling and given refreshers as needed during the field season. The methods employed are directly comparable to other data collection efforts associated with SMAST, which involve nearly all of the embayments in southeastern Massachusetts, inclusive of Martha's Vineyard and the Island of Nantucket. Water samples are collected by trained volunteers and town staff for nutrient analysis by the Coastal Systems Analytical Facility at SMAST-UMass Dartmouth. As such, the data can be fully integrated into the MEP technical approach.

These prior investigations provided significant information related directly to the implementation of the MEP Linked Management Modeling Approach and helped yield insight into the interpretation of the results. In addition, the Town of Orleans' comprehensive Water Quality Monitoring Program was of sufficient rigor to be used as the water quality baseline required for the MEP threshold analysis presented in this MEP Technical Report.

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 to organize and analyze the available data use up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, surface water/groundwater interaction, groundwater travel time, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including Rock Harbor in the Towns of Orleans and Eastham. This estuary is situated along the northern edge of Cape Cod and is bounded by Cape Cod Bay.

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 Rock Harbor estuary system under evaluation by the Project Team. Further modeling of the estuary watersheds was undertaken to sub-divide the overall watersheds into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from "new" development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the overall Monomoy groundwater flow cell. Model assumptions for calibration were matched to surface water inputs and flows from current (2002 to 2004) stream gage information.

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 transport requires determination of the portion of the watershed that contributes directly to the stream and the portion of the groundwater system that discharges directly into the estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the Rock Harbor estuary system were delineated using a regional model of the Monomoy 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 of 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 Rock Harbor system and also to determine portions of recharged water that may flow through ponds and streams prior to discharging into coastal water bodies.

The Monomoy Flow Model grid consists of 164 rows, 220 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 (Walter and Whealan, 2005). 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 525 feet below NGVD 29). The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens. Since water elevations are less than +40 ft in the portion of the Monomoy Lens in which the Rock Harbor estuary resides, the three uppermost layers of the model are inactive. It should also be noted that the Rock Harbor watershed is located on the northernmost edge of the Monomoy model grid.

The glacial sediments that constitute the aquifer of the Monomoy Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence with sand and gravel sediments deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Rock Harbor estuary watershed extends over a number of geologic settings. The northern portion in Eastham contains sections of the Eastham Plains deposits, while the southern portion contains parts of the Harwich Outwash Plains (Oldale, *et al*., 1971). The Eastham Plains deposits generally consist of sand and gravel and slope toward Cape Cod Bay, which once contained a pro-glacial lake. These materials are younger than the Harwich Outwash Plains and are thought to indicate a glacial lobe positioned to the east of Eastham. The portions of the watershed closer to Cape Cod Bay are composed of Lake and Lake Bottom Deposits from the Cape Cod Bay pro-glacial lake and are generally composed of clays and silts mixed with some sand. On top of these materials are the marsh and swamp deposits that generally surround Rock Harbor and are generally composed of decaying plant material mixed with varying amounts of clay, silt, and sand. Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Town and water-level data.

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

III.3 ROCK HARBOR ESTUARY CONTRIBUTORY AREAS

Newly revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the Rock Harbor estuary system (Figure III-1). Model outputs of MEP watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, and (c) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineations for this watershed do not include 10 yr time of travel boundaries. Overall, three sub-watershed areas, including Cedar Pond, were delineated within the watershed to the Rock Harbor estuary.

Table III-1 provides the daily freshwater discharge volumes for each of the sub-watersheds as calculated by the groundwater model. The volumes presented in Table III-1 were used to assist in the salinity calibration of the tidal hydrodynamic models and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to measured surface water discharges. The total estimated freshwater inflow to the Rock Harbor estuary equals 4,734 m3/day, 96 percent of which is represented by groundwater input and the remainder is recharged on the surface of Cedar Pond.

The delineations completed by this revised MEP analysis are the second watershed delineation completed in recent years for this estuary. A pre-MEP delineation was completed by the Cape Cod Commission and was defined based on regional water table measurements collected from available wells over a number of years and normalized to average conditions (Eichner, *et al*., 1998). The initial analysis included data collected in Leab, et al. (1995). The Commission's delineation was incorporated into the CCC regulations through the Regional Policy Plan (CCC, 1996 & 2001). Figure III-2 compares the Commission delineation with the MEP delineation. The MEP watershed delineation changes the location of the groundwater divide between Cape Cod Bay and Town Cove and adds a refined watershed delineation for Cedar Pond.

Overall, the MEP contributing areas to the Rock Harbor estuary based upon the groundwater modeling effort is very similar in area to the previous delineations based upon available well data. The overall watershed area for the MEP delineation is 609 acres, while the watershed delineated by the CCC is 551 acres. However, the portions of the towns covered by these two delineations are different; a greater proportion of the MEP watershed is shifted closer to Cape Cod Bay. The MEP watershed to Rock Harbor is mostly within Orleans; 75% or 457 acres is in Orleans and 25% or 153 acres is in Eastham.

The evolution of the watershed delineations for the Rock Harbor estuary has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected. Moreover, the groundwater model allows all pertinent data to be organized and to be brought into congruence with data from 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 used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within specific 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.

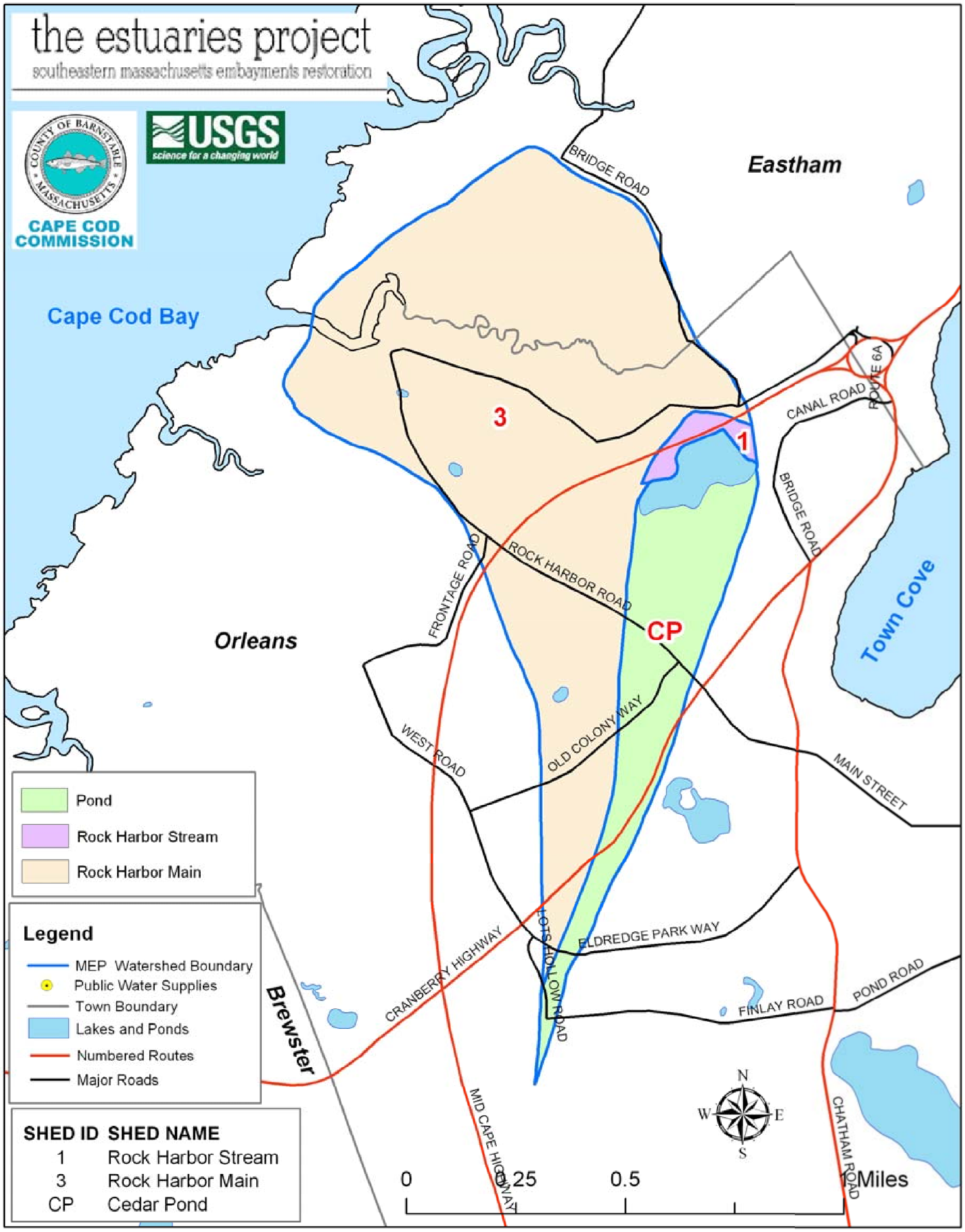


Figure III-1. Watershed delineation for the Rock Harbor estuary. Sub-watersheds to aquatic systems were selected based upon the functional estuarine sub-units in the water quality model (see section VI). Assigned watershed numbers are based on delineation of all Cape Cod Bay estuaries in Town of Orleans.

Table III-1. Daily groundwater discharge from each of the sub-watersheds to Cedar Pond and the Rock Harbor estuary, as determined from the USGS groundwater model.

Watershed	#	Watershed Area (acres)	% contributing to Harbor	Discharge	
				m^3/day	ft^3/day
Cedar Pond	CP	120	100	974	34,407
Rock Harbor Stream	1	10	100	76	2,694
Rock Harbor Main	3	480	100	3,683	130,068
TOTAL		609		4,734	167,169

Note: discharge volumes are based on 27.25 in of annual recharge over the watershed area; watershed area to Cedar Pond includes the surface of the pond and the discharge includes recharge on this surface area; rounding may result in slight discrepancies in totals.

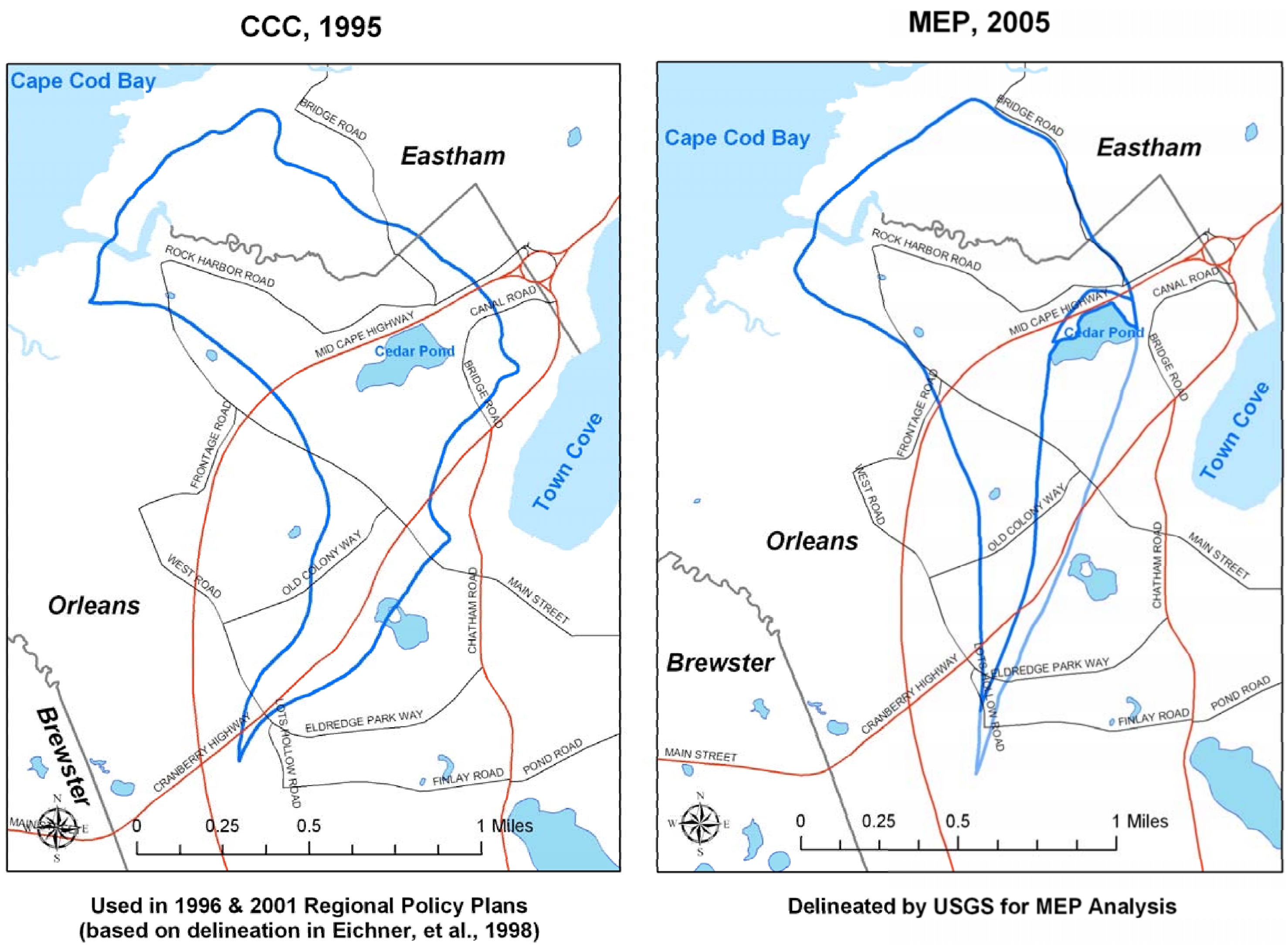


Figure III-2. Comparison of watershed and sub-watershed delineations used in the current analysis and the Cape Cod Commission delineation completed for the Coastal Embayment Project (Eichner, *et al*., 1998) and adopted into the Regional Policy Plan (CCC, 1996 & 2001).

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 Rock Harbor system. Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. This latter natural attenuation process results from biological processes that naturally occur within ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from sediments. Sediment nitrogen recycling results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters or in some cases a sink for nitrogen reaching the bottom. Failure to include the nitrogen balance of estuarine sediments generally leads to errors in predicting water quality, particularly in determination of summertime nitrogen load to embayment waters.

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP Technical Team members, CCC staff developed nitrogen-loading rates (Section IV.1) to the Rock Harbor estuary system (Section III). The Rock Harbor watershed was sub-divided to define contributing areas to Cedar Pond, the gauged tributary leaving Cedar Pond and the overall system. A total of three sub-watersheds were delineated for the Rock Harbor Estuarine System.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes, the time of groundwater travel provided by the USGS watershed model, and review of data at natural collections points, such as streams and ponds. Ten-year time of travel zones are a regular part of the watershed analysis, but for Rock Harbor all travel is within a ten-year travel time. MEP staff also reviewed land use development records for the age of developed properties in the watershed. Based on all these reviews, it was determined that Rock Harbor is currently in balance with its watershed load. 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 estuaries (after accounting for natural attenuation, see below).

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 studies is applied to other portions of the nitrogen load. The Linked Watershed-Embayment Management Model (Howes and Ramsey, 2001) uses a land-use Nitrogen Loading Sub-Model based upon sub-watershed-specific land uses and pre-determined nitrogen loading rates. For the Rock Harbor embayment system, the model used Town of Orleans and Town of Eastham land-use data transformed to nitrogen loads using both regional nitrogen loading factors and local watershed specific data (such as parcel by parcel water use or groundwater monitoring wells). 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 has not yet been included.

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) within the Rock Harbor System watershed was determined based upon a site-specific study of stream flow from the upper portion of the watershed (*i.e.*, Rock Harbor Stream) and within Cedar Pond. Sub-watersheds to these portions allowed comparisons between field collected data from the stream and pond and estimates from the nitrogen-loading sub-model. Stream flow and associated surface water attenuation is included in the MEP nitrogen attenuation calculation and freshwater flow investigation, presented in Section IV.2.

Natural attenuation during stream transport is a standard part of the data collection effort of the MEP. In the present effort, measurements were made of attenuation in the stream leading into the estuary portion of Rock Harbor. However, 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. Based upon these considerations, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the one sub-watershed that directly discharges groundwater to the estuary without flowing through one of the interim measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Rock Harbor Estuarine System. Moreover, measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Estuaries Project staff obtained digital parcel and tax assessors data from the Towns of Orleans and Eastham. Digital parcels and land use data are from 2005 for Orleans and 2002 for Eastham and were obtained from the respective town planning departments and Cape Cod Commission files. These land use databases contain traditional information regarding assessor land use classifications (MADOR, 2002) plus additional information developed by each of the towns. The parcel data and assessors' databases for all the towns were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows the land uses within the Rock Harbor Estuary watershed area. Land uses in the study area are grouped into seven land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) agricultural, 6) public service/government, including road rights-of-way, and 7) mixed use. These land use categories are aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2002). These categories are common to both towns in the watershed. “Public service” in the MADOR system is tax-exempt properties, including lands owned by government (e.g., wellfields, schools, golf courses, open space, roads) and private groups like churches and colleges.

In the Rock Harbor System watershed, the predominant land use based on area is residential, which accounts for 49% of the system watershed (Figure IV-2). Residential land uses also are the majority of the parcels in the system watershed; residential parcels are 62% of all the parcels. Most of the residential parcels are single-family residences (MADOR land use code 101); parcels classified by the town assessors as 101s are 79% of the total area of parcels classified as residential and 87% of the total number of residential parcels in the system watershed. Parcels classified as public service/government (MADOR land use codes in the 900s) are the second highest percentage (25%) of the area of the system watershed. Undeveloped areas account for 18% of the Rock Harbor Main sub-watershed, and 7% of the Cedar Pond sub-watershed. Most of the commercial properties are in the Cedar Pond sub-watershed where they are 43% of the area. Overall, undeveloped land uses account for 17% of the entire Rock Harbor watershed, while commercial properties account for approximately 10% of the system watershed area.

In order to estimate wastewater flows within the Rock Harbor study area, the Cape Cod Commission obtained parcel-by-parcel water use information from the Town of Orleans. Water supply in the Town of Eastham is generally supplied by on-site wells, therefore parcel by parcel water use is not available for these parcels. The water use data from Orleans includes information from the Orleans Water Department via the Planning Department. Orleans water data is twelve months of water use between 2002 and 2003. There is one wastewater treatment facility in the watershed: the Community of Jesus wastewater treatment facility. MEP staff linked water use information to the Orleans parcel and assessors data using GIS techniques. Average water use in the watershed was used for all developed Eastham properties. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2).

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}.

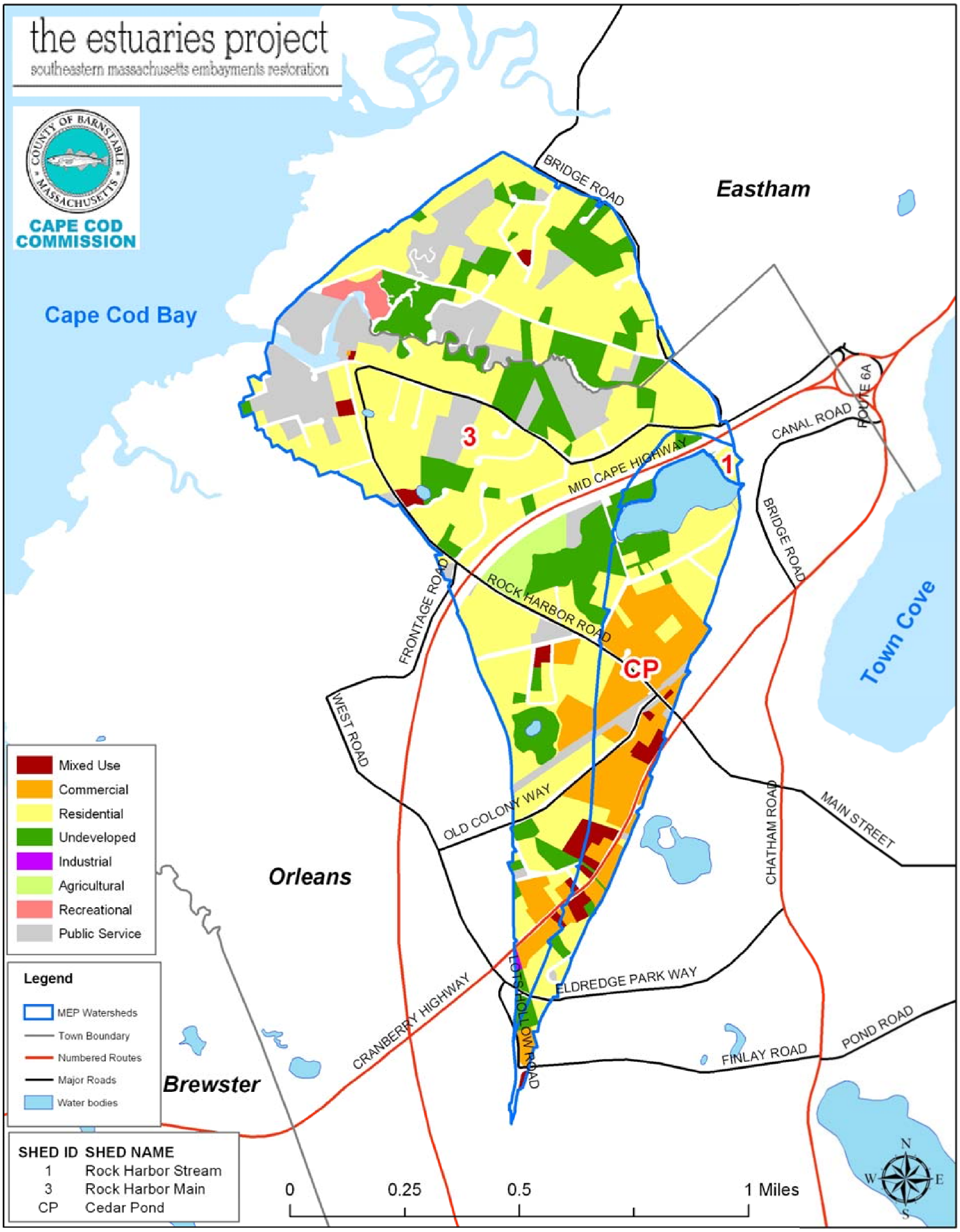


Figure IV-1. Land-use in the Rock Harbor estuary watershed. The watershed extends into portions of both the Town of Orleans and the Town of Eastham. Land use classifications are based on assessors' records provided by the towns.

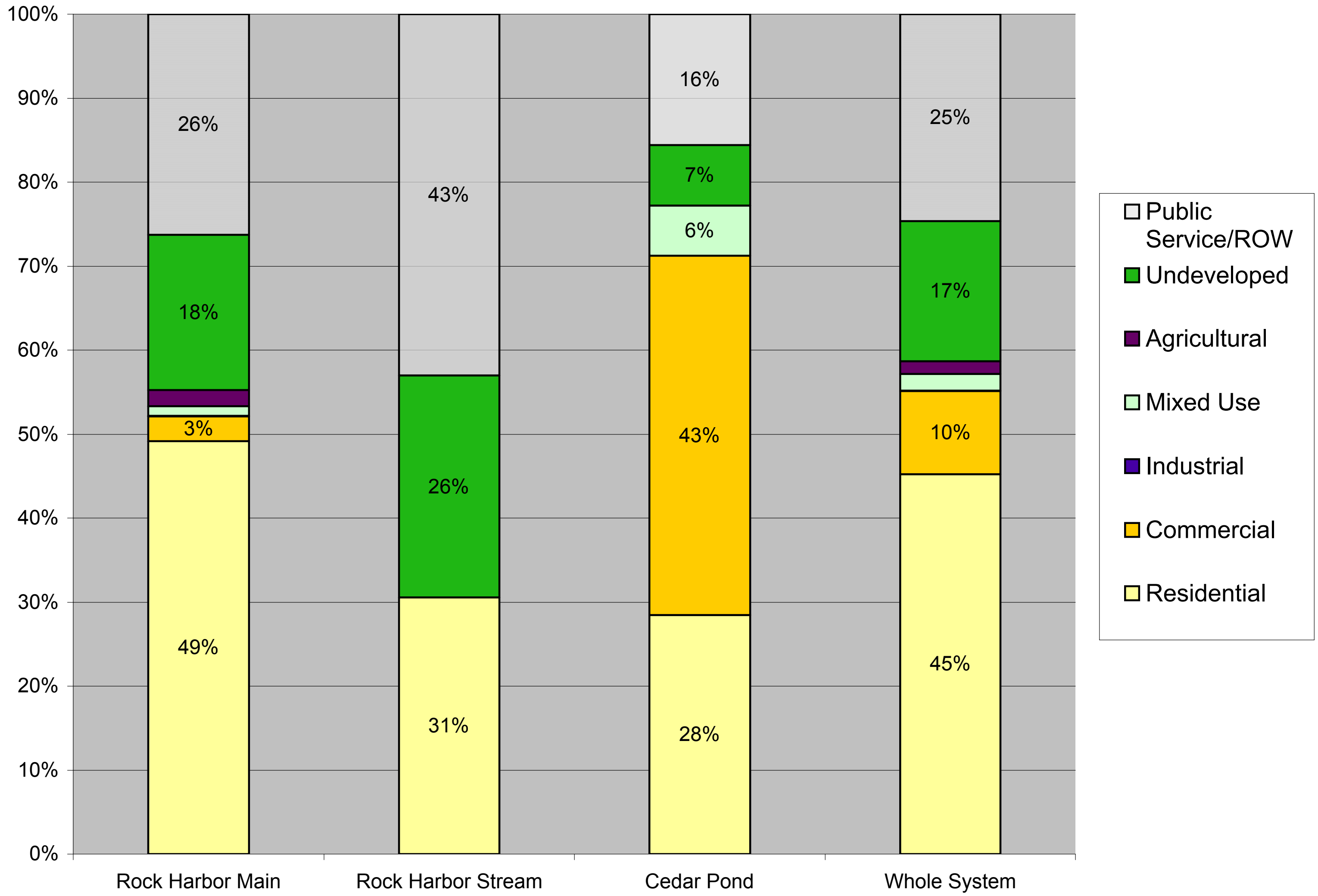


Figure IV-2. Distribution of land-uses within the major sub-watersheds and whole watershed to Rock Harbor. Only percentages greater than or equal to 3% are shown.

However, 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 assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down gradient in the aquifer.

All nitrogen losses within the septic system are incorporated into the MEP analysis. For example, information developed at the MASSDEP 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). Down gradient studies of septic system plumes indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

In its application of the water-use approach to septic system nitrogen loads, the MEP has ascertained for the Estuaries Project region that while the per capita septic load is well constrained by direct studies, the consumptive use and nitrogen concentration data are less certain. As a result, the MEP has derived a combined term the effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per 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). 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 cover large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that from the water use approach. This comparison matches some of the variability seen in census data itself. Census blocks, which are generally smaller areas of any given town, have shown up to a 13% difference in average occupancy form town-wide occupancy rates. These analyses provide additional support for the use of the water use approach in the MEP study region.

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy outwash aquifers; (b) has been validated in studies of the MEP Watershed "Module", where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Nitrogen Loading Coefficients (e.g., stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees in 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 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 Rock Harbor System watershed, MEP staff reviewed US Census population values for the Towns of Eastham and Orleans. 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 Eastham is 2.28 people per housing unit, while year-round occupancy of available housing units is 60%. Orleans had an average occupancy of 2.05 people per housing unit and a 61% year-round occupancy of available housing units. Average water use for the 182 single-family residences with municipal water accounts, which are all in Orleans, in the Rock Harbor watershed is 142 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the watershed average water use is 128 gpd. If this flow is then divided by 55 gpd, the average estimated occupancy in the watershed is 2.33 people per household. If the Census occupancy rates for the towns are weighted by their proportion of the watershed areas, the resulting occupancy is 2.11.

In most previously completed MEP studies, average water use based on population estimates and average water use based on measured flows have generally agreed fairly well. In cases where they have not, generally there have been clearly identified factors that help to explain the differences. One of the usual steps that MEP staff take to review apparent differences is to review more refined US Census information. In the Rock Harbor watershed, MEP staff determined that the Orleans portion of the watershed is mostly within US Census Tract 104, Block Group 1. Within this block group, which covers only a portion of the Town of Orleans, there were 865 people and 322 households during the 2000 US Census. These counts result in an average occupancy of 2.69 people per household. If this occupancy is multiplied by 55 gpd per person, the resulting average flow in this portion of Orleans is 148 gpd. Since this closely approximates the 142 gpd based on water use data in the watershed to Rock Harbor and Eastham has a higher town-wide occupancy rate than Orleans, MEP staff determined that the average water use for the watershed was reasonable and was an appropriate estimate for properties with wells and 68 additional residential dwellings estimated for the buildout scenario.

Water use information exists for 75% of the 354 developed parcels in the Rock Harbor watershed. The remaining 25% of the parcels (87 developed parcels) are assumed to utilize private wells for drinking water; 85% of these parcels are in Eastham. 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 87 parcels, 72 of them (70 of which are in Eastham) are classified as single-family residences (land use code 101) and another ten parcels are classified as other types of residential development [e.g. 109 (multiple houses on a single property)]. Four of the remaining five parcels are commercial properties (300s land use codes) and one is mixed use. MEP staff used current water use to develop a watershed-specific water use estimate for all uses that were assumed to utilize private wells.

Community of Jesus Wastewater Treatment Facility

As mentioned above, the Community of Jesus Wastewater Treatment Facility (CofJ-WWTF) is located within the Rock Harbor watershed, but two of its four effluent discharge fields are located within the Little Namskaket sub-watershed (Figure IV-3). The CofJ-WWTF began operating in June 2003 under a state Groundwater Discharge Permit that limits its nitrogen discharge to 10 mg/l or less (B. Dudley, MassDEP, personal communication, March 2007). MEP staff obtained effluent flow and nitrogen concentration data from MassDEP in order to assess potential impacts from the treatment facility. MEP staff also obtained facility design schematics from Cape Cod Commission files and spoke with the Orleans Health Agent regarding operational practices at the facility.

The CofJ-WWTF has four effluent fields: 1) North Anchor Drive, 2) South Anchor Drive, 3) Bay View Drive Parking Lot, and 4) Brother's Residence. According to the Orleans Health Agent, these fields are used equally (R. Canning, personal communication). Their total design capacity is 21,000 gpd. The two Anchor Drive fields are located within the Little Namskaket watershed, while the other two are located within the Rock Harbor watershed. Average flow at the facility based on monthly monitoring over 28 months (startup through 2005) is 5,538 gpd, while the average total nitrogen concentration in the effluent is 5.43 mg/l. Based on this information, the load to the Rock Harbor sub-watershed is 24 kg/yr.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized lawns, golf courses, and cranberry bogs, with lawns being the predominant source within this category. In order to add this source to the nitrogen loading model for the Rock Harbor system, MEP staff reviewed available information about residential lawn fertilizing practices and incorporated site-specific information to determine nitrogen loading from large tracks of turf in the watershed. Residential lawns are the only source of fertilizer load identified in the Rock Harbor estuary watershed.

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. The assessment accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn; these factors are generally 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 also be noted that professionally maintained lawns were found to have the higher rate of fertilizer application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

Although the above values are generally used in MEP modeling, the Orleans Wastewater Planning Committee conducted a 2003 survey of 340 homes throughout the Town of Orleans. The results of this survey indicated that the number of fertilizations per lawn in Orleans was similar to the number listed above, 1.76 versus 1.44, but there were a very high number of homes serviced by commercial lawn companies (over 1/3). These commercially-tended lawns were fertilized at higher rate than those developed above for standard homeowner-tended lawns. The overall results indicated potentially higher nitrogen loading per lawn in Orleans of 1.51 lb/lawn/yr (weighted average). MEP staff used this higher application rate throughout the watershed based on this site-specific information.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III).

Factors used in the MEP nitrogen loading analysis for the Rock Harbor watershed are summarized in Table IV-1.

Table IV-1. Primary Nitrogen Loading Factors used in the Rock Harbor MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Orleans data. *Data from Orleans lawn study.

Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing developed residential parcels wo/water accounts and buildout residential parcels:	142 gpd
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (sq ft)*	5,000		
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.51	Existing developed parcels w/water accounts:	Measured annual water use
Nitrogen leaching rate	20%	Buildout additions:	
Impervious Surfaces (based on ORL data):		Commercial wastewater	98 gpd/1,000 ft2 of building
Residences (sq ft):	2,056	Industrial wastewater	16 gpd/1,000 ft2 of building
Commercial building coverage for parcels:	13%		
Industrial building coverage for parcels:	10%		

IV.1.3 Calculating Nitrogen Loads

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

The review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (condominiums, Community of Jesus wastewater facility, etc.) was 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 Rock Harbor estuary. The assignment effort was undertaken to better define the sub-embayment loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Following the assignment of all parcels, sub-watershed modules were generated for each of the three sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. The individual sub-watershed modules were then integrated to create the Rock Harbor Watershed Nitrogen Loading module with summaries for each of the individual portions (or sub-embayments) of the larger system. The sub-embayments represent the functional 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 Rock Harbor System, the major types of nitrogen loads are: wastewater (e.g., septic systems), the Community of Jesus wastewater facility, fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-2). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-4 a-c). 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.

Cedar Pond Nitrogen Loads

Prior to the collection of stream flow information (see Section IV.2), Cedar Pond was identified as a freshwater pond. Cedar Pond is the most inland portion of the Rock Harbor estuary system and, as such, focuses a large portion of the watershed's groundwater discharge. Water quality data collected by volunteers coordinated through the Town of Orleans Marine and Fresh Water Quality Task Force (WQTF) and reviewed under the current Orleans Ponds Assessment (Eichner, 2007, in review) provides sufficient nitrogen data to develop a site-specific attenuation rate.

Cedar Pond is 15.1 acres with a volume of 113,371 cubic meters and is 4.6 m deep at its deepest point. The volume is based on bathymetric information collected by WQTF volunteers and interpreted by Cape Cod Commission staff. Between 2001 and 2004, volunteers collected over 26 water quality samples. These samples were analyzed by the SMAST and National Park Service laboratories and provided total nitrogen concentrations. Field data collected at the same time shows that the pond is impaired with average dissolved oxygen concentrations between June and September below state surface water standards for approximately 33% of the pond volume (Figure IV-5). In addition, 17% of the pond volume is on average anoxic (<1 ppm).

Because of the regular low oxygen conditions, nitrogen concentrations in samples collected below 1 m are impacted by nitrogen regenerated from the sediments. The average total nitrogen concentration of thirteen near-surface samples is 0.54 mg/l, while the average for paired samples from 2-3 meters is 2.17 mg/l. In relatively shallow ponds (<9 m) on the Cape, winds are generally sufficient to keep water columns well mixed. Because Cedar Pond is shallow by this standard, it would be expected that the water column should be well mixed. Average temperature readings generally confirm well-mixed conditions, so some portion of the high nitrogen concentrations in the deeper waters will mix into the upper, near-surface waters.

When calculating a nitrogen attenuation rate for Cedar Pond, MEP staff reviewed the available water quality data and the estimated nitrogen loads shown in Table IV-2. After comparing these results with data available from the stream gauge (Section IV.2), MEP staff conservatively estimated a nitrogen attenuation rate of 58% in Cedar Pond.

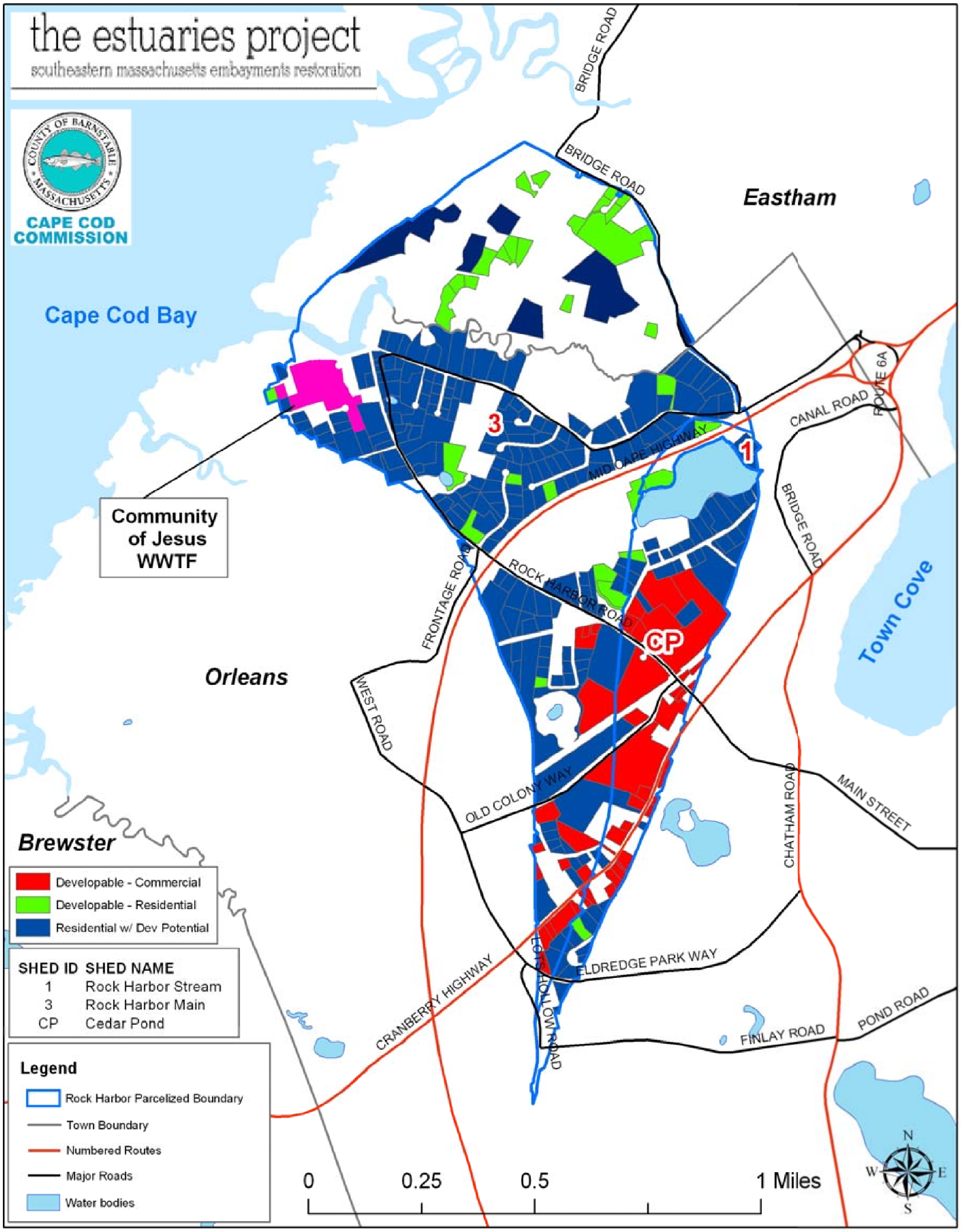
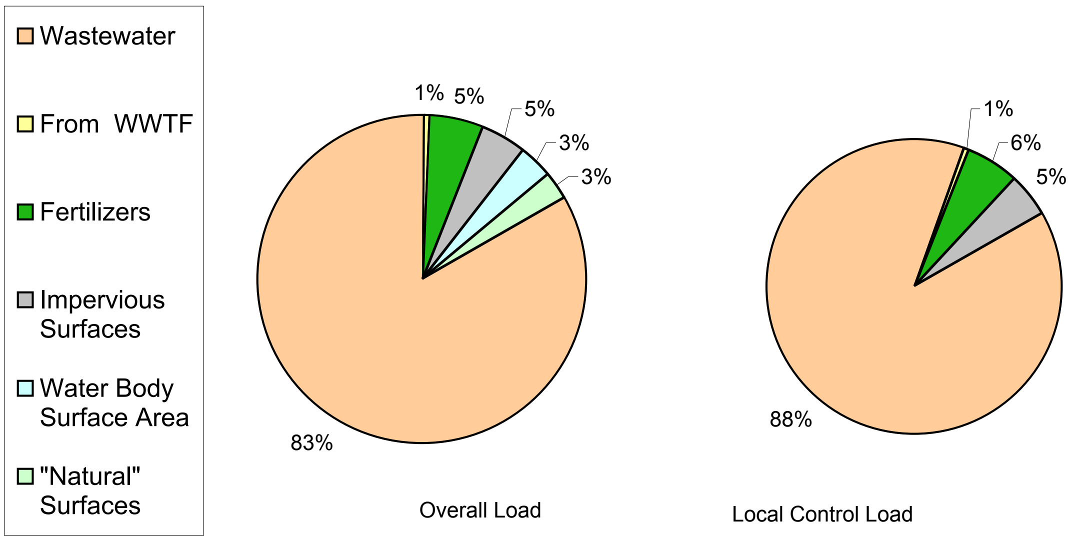


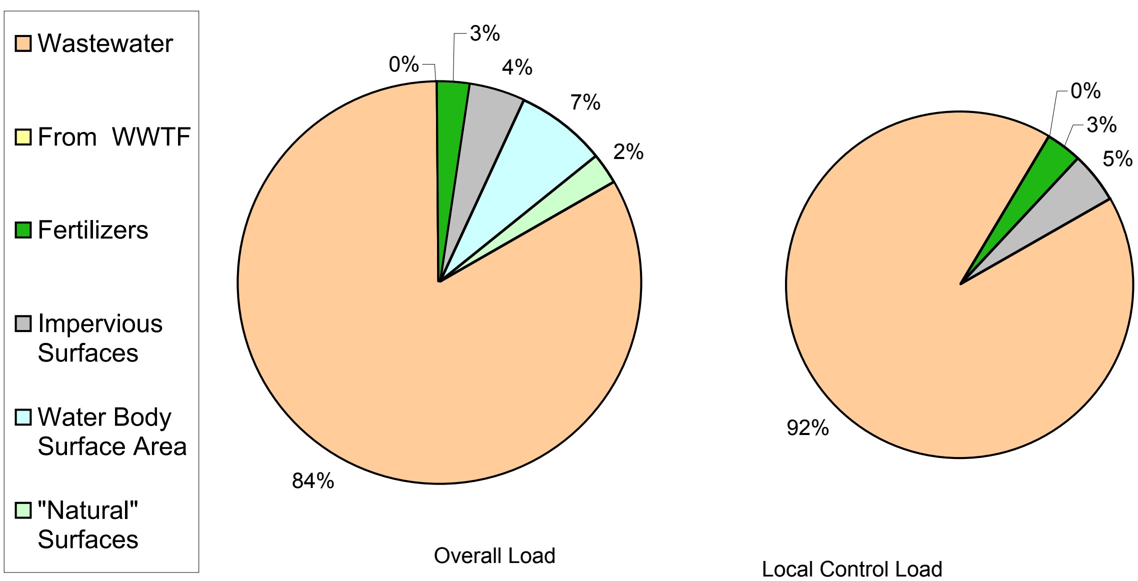
Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Rock Harbor watersheds.

Table IV-2. Rock Harbor Nitrogen Loads. Attenuation of nitrogen loads within the Rock Harbor system occurs as nitrogen moves through up-gradient streams and Cedar Pond during transport to the estuary. All values are kg N yr^{-1}. WWTF category includes partial load from the Community of Jesus Wastewater Treatment Facility.

		Rock Harbor N Loads by Input (kg/y):							% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	From WWTF	Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Rock Harbor System Total		3278	24	208	178	129	109	3973		3926		3299	7899		5325
Rock Harbor Main Total	3	2476	24	182	136	29	86	984		2933		2933	3916		3916
Rock Harbor Estuary surface deposition						29				29		29	29		29
Rock Harbor Stream Total	1+CP	802	0	27	42	70	24	2989		965	17%	337	3954	17%	1380
Rock Harbor Stream	1	1	0	1	4	0	2	23		8		8	31		31
Cedar Pond TOTAL		801	0	26	38	70	22	2966		957	58%	398	3923	58%	1632
Cedar Pond	CP	801	0	26	38		22	2966		886		886	3853		3853
Cedar Pond Estuary surface deposition						70				70		70	70		70



a. Rock Harbor System Overall



b. Rock Harbor Stream Total Subwatershed

Figure IV-4 (a-b). Land use-specific unattenuated nitrogen load (by percent) to the (a) overall Rock Harbor System watershed and (b) Rock Harbor Stream (including Cedar Pond) sub-watershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

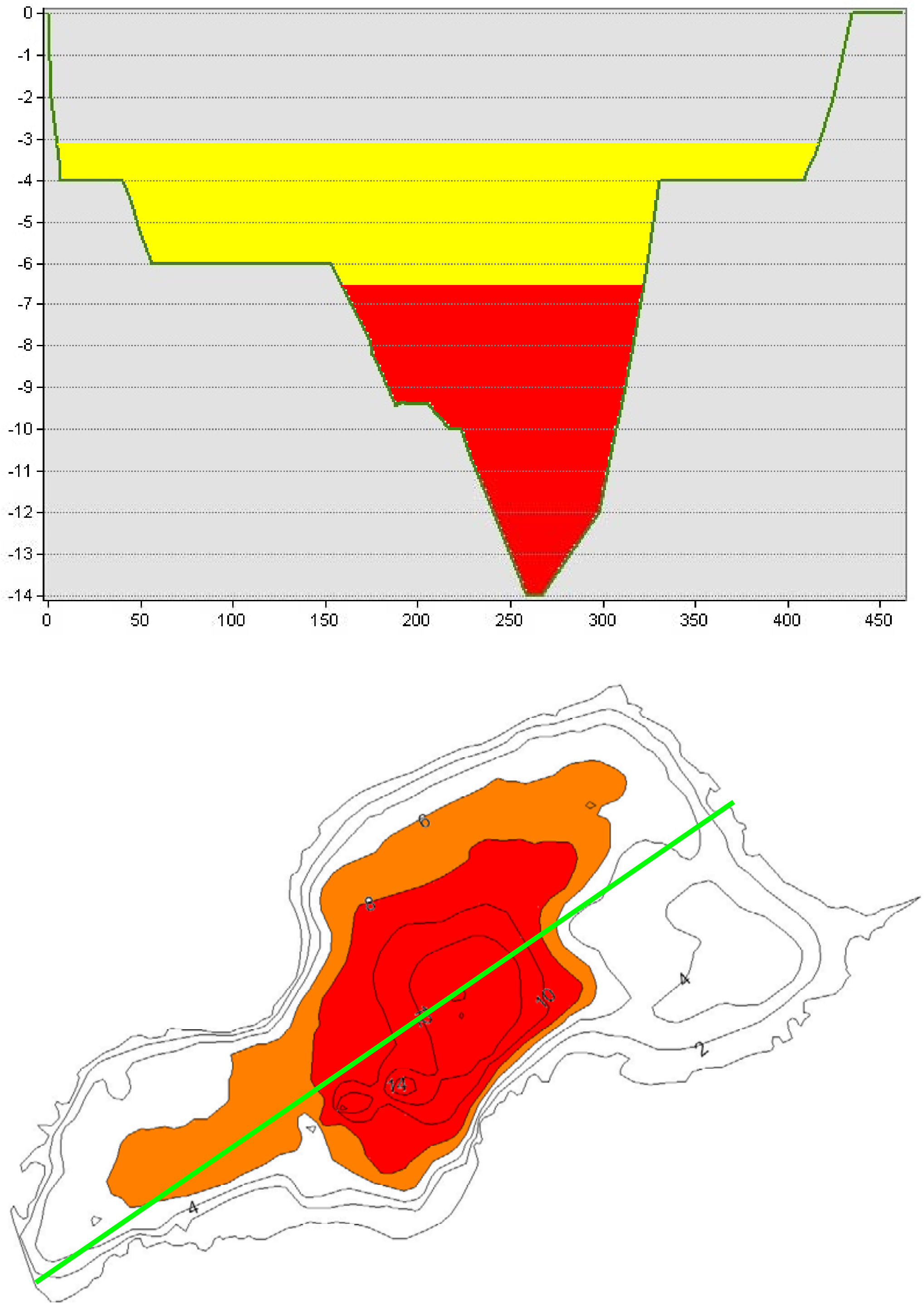


Figure IV-5. Cedar Pond Average Dissolved Oxygen and State Surface Water Standard. Depth profile shows portion of Cedar Pond that averages less than the state surface water 5-ppm standard for class SA warm water fisheries (shaded yellow); areas shaded red have average anoxic concentrations (<1 ppm). Averages are based on readings collected between June and September (18 profiles). Area on map is shaded orange since anoxic area only extends to 6.5 ft contour and only whole contours are shown. Green line on map shows depth profile track through the pond. Data and graphic from Eichner (2007, in review).

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watershed. For the Rock Harbor modeling, MEP staff consulted with respective town officials to determine the information that would be used in the assessment. Orleans buildout information was provided by the town Planning Department during the development of the Pleasant Bay MEP Technical Report; further communication with the Town Planner led to subsequent modification in the buildout parameters (G. Meservey, personal communication, February 2007). This change brought the buildout into congruence with buildout numbers being used in the current town Comprehensive Wastewater Management Plan process. Eastham buildout was completed with guidance from town officials and utilized standard MEP buildout parameters as developed by CCC staff.

A standard MEP buildout assessment is to evaluate town zoning to determine minimum lot sizes in each of the zoning districts, including overlay districts (*e.g.*, water resource protection districts). Larger lots are subdivided by the minimum lot size to determine the total number of new lots and existing developed properties are reviewed for additional development potential; for example, residential lots that are twice the minimum lot size, but have only one residence. In the Eastham buildout completed by MEP staff, parcels that are classified by the town assessor as developable residential (state class land use codes 130 and 131) but are less than the minimum lot size and are greater than 5,000 square feet are assigned one residence in the buildout; 5,000 square feet is a common minimum buildable lot size in Cape Cod town regulations. Properties classified by the Eastham assessor as “undevelopable” (e.g., codes 132, 392, and 442) were not assigned any development at buildout; this is different than the assumptions used in the Orleans buildout. Commercially developable properties were not subdivided in any of the towns; the area of each parcel and the factors in Table IV-2 were used to determine a wastewater flow for these properties. All the parcels included in the buildout assessment of the Rock Harbor watershed are shown in Figure IV-3. A nitrogen load for each additional parcel included in the buildout was developed using the factors in Table IV-1.

Overall, a cumulative nitrogen load for each additional residence or business is included in the unattenuated buildout indicated in a separate column in Table IV-2. The Community of Jesus Wastewater Treatment Facility is not assigned any additional load at buildout. Buildout additions within the overall Rock Harbor System watershed will increase the unattenuated loading rate by 101%.

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 Rock Harbor Marsh 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 region of the watersheds to these Town of Orleans systems). 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 case of the Orleans embayment system watersheds for Rock Harbor Marsh, a portion of the freshwater flow and transported nitrogen passes through a surface water system (Creek from Cedar Pond to Rock Harbor) prior to entering the estuarine system, producing the opportunity for 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. MEP conducted long-term measurements of natural attenuation relating to surface water discharge to the head of the Rock Harbor embayment system considered in this report in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). The additional site-specific study was conducted in the 1 major surface water flow system in the watershed to the head of the marsh 1) Creek discharging from Cedar Pond to the head of Rock Harbor salt marsh (Figure IV-6).

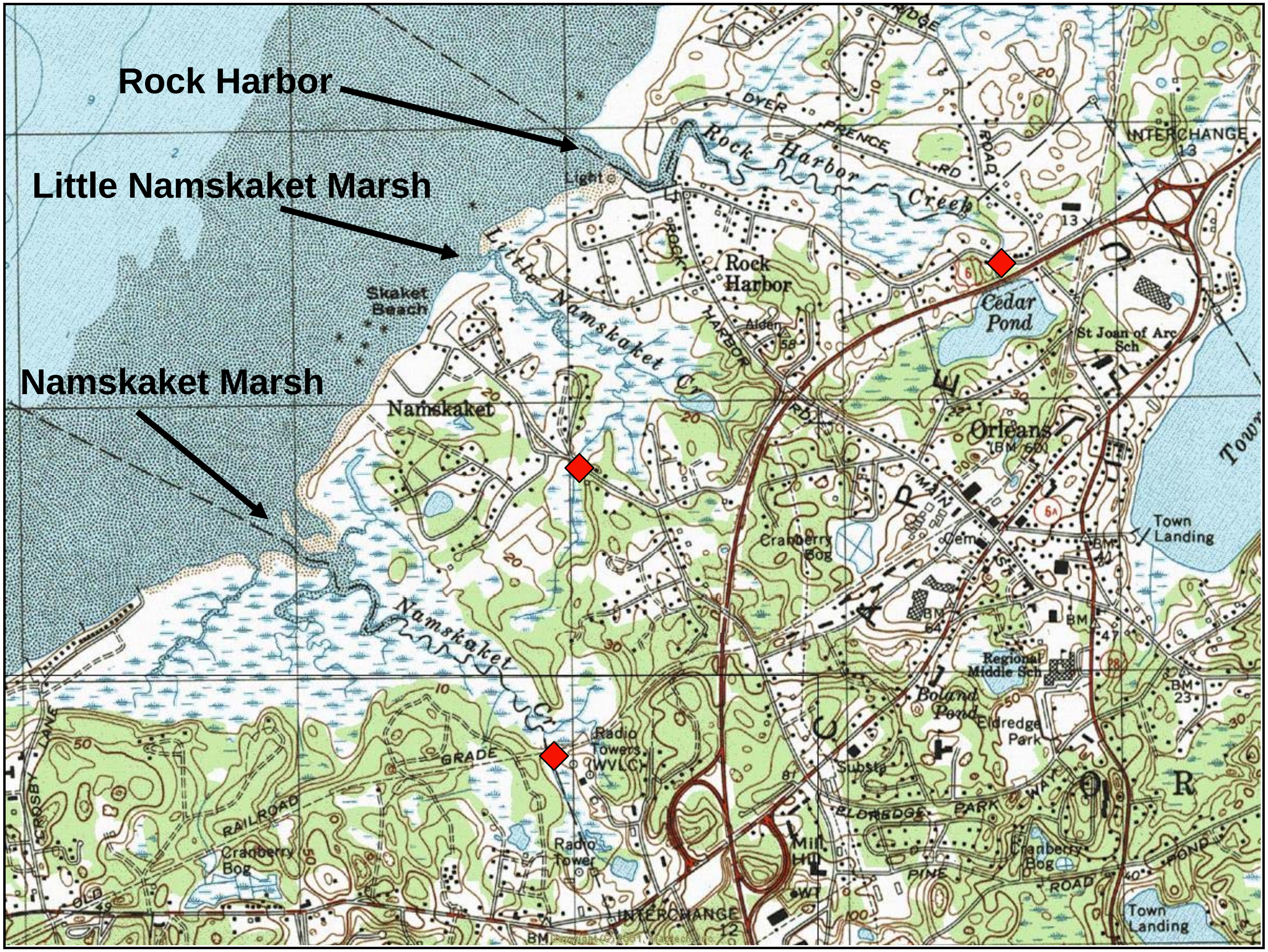


Figure IV-6. Location of Stream gages (red symbols) in the Rock Harbor, Little Namskaket Marsh and Namskaket Marsh embayment systems.

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 streams discharging to the estuaries provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area up-gradient from the various gauging sites. Flow and nitrogen load were measured at the gages in each freshwater stream site for between 16 and 24 months of record depending on the stream gauging location (Figures IV-7 and Table IV-4). During each study period, velocity profiles were completed on each creek every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

Determination of stream flow at each gage was calculated and based on the measured values obtained for stream cross sectional area and velocity. Stream discharge was represented by the summation of individual discharge calculations for each stream subsection for which a cross sectional area and velocity measurement were obtained. Velocity measurements across the entire stream cross section were not averaged and then applied to the total stream 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 river. These hourly stages values where 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. In the case of tidal influence on stream stage, the diurnal low tide stage value was extracted on a day-by-day basis in order to resolve the stage value indicative of strictly freshwater flow. The two low tide stage values for any given day were averaged and the average stage value for a given day was then entered into the stage – discharge relation in order to compute daily flow. A complete annual record of stream flow (365 days) was generated for the surface water discharges flowing into the Rock Harbor Marsh system.

The annual flow record for the surface water flow at the gage 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 head of the marsh system. Nitrogen discharge from the streams was calculated using the paired daily discharge and daily nitrogen concentration data to determine the mass flux of nitrogen through a specific gauging site. For the stream gage location, weekly water samples were collected (at low tide for a tidally influenced stage) in order to determine nutrient concentrations from which nutrient load was calculated. 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 stream flow 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 each pond currently reduces (percent attenuation) nitrogen loading to the embayment system.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge from Cedar Pond to head of Rock Harbor

Cedar Pond, located up-gradient of the stream gage deployed on the surface water flow into Rock Harbor, is a small freshwater pond and unlike many of the freshwater ponds on Cape Cod, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This freshwater outflow, stream into head of Rock Harbor, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and streambed. 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 stream above the gage site and the measured annual discharge of nitrogen to the tidal portion of the Rock Harbor marsh system, Figure IV-6.

At the Rock Harbor stream gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater portion of the Rock Harbor marsh estuarine system that carries the flows and associated nitrogen load to the mouth of Rock Harbor and ultimately, Cape Cod Bay. As portions of the Halls Creek system are tidally influenced, the gage was located above the saltwater reach such that freshwater flow could be measured without tidal influence (at low tide). To confirm that freshwater was being measured, salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average low tide salinity was determined to range between 6.9 ppt and 11.1 ppt depending on the season. Based on the salinity, a correction was made to the predicted daily flows obtained at the gage to account for the tidal influence. Considering the weekly salinity data, a boundary salinity obtained from a nearby offshore water quality monitoring station and the ability to correct gage data for salinity, the gage location was deemed acceptable for making freshwater flow measurements at low tide. Calibration of the gage was checked monthly. The gage on the Cedar Pond stream was installed on June 28, 2002 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 May 23, 2004 for a total deployment of 23 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 Cedar Pond stream site based upon these flow measurements and measured water levels at the gage site. 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 head of the Rock Harbor marsh system flowing into Cape Cod

Bay (Figure IV-7 and Table IV-3). 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 each gage site.

The annual freshwater flow record for the Cedar Pond stream 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 Cedar Pond Stream was 21% 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 1,271 m^3/day compared to the long term average flows determined by the USGS modeling effort (1,051 m^3/day).

The difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Creek flowing from Cedar Pond to Rock Harbor is in part due to a slight tidal influence confounding the freshwater flow record.
Due to site constraints, it was necessary to position the stream gage further down gradient in the stream flowing out of Cedar Pond and into the head of the Rock Harbor Marsh system. As such, even at low tide, the flow measured in the stream exhibited a slight salinity as mentioned above. The flow was corrected based on seasonal trends in salinity, however, it is possible that the flow in the stream is a slight over estimate of the freshwater flow discharging to Rock Harbor Marsh. The MEP Technical Team concurred that the over-estimate on flow and therefore load was conservative and used the measured flow and load in the water quality modeling.

Total nitrogen concentrations within the Cedar Pond outflow were low, 0.86 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 1.09 kg/day and a measured total annual TN load of 399 kg/yr. In the Cedar Pond stream outflow, nitrate was a small fraction of the total nitrogen load from Cedar Pond (13%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater pond and to the river was largely taken up by plants within the pond or stream ecosystems. The low concentration of inorganic nitrogen in the out-flowing stream waters suggests that plant production within the up-gradient ecosystems may be nitrogen limited. In addition, the low nitrate level may negate the possibility for additional uptake by the up-gradient Cedar Pond system or along the freshwater reach of the stream.

From the measured nitrogen load discharged by Cedar Pond stream to the head of Rock Harbor marsh 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 (399 kg yr^{-1}) discharged from the freshwater Cedar Pond stream compared to that added by the various land-uses to the associated watershed (957 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 58% (i.e. 58% of nitrogen input to watershed does not reach the estuary). This slightly higher level of attenuation compared to other streams evaluated under the MEP is expected given the hydraulic nature of the up-gradient pond which acts as a small kettle pond able to enhance attenuation via natural biological processes. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-3. Comparison of water flow and nitrogen discharges from streams (freshwater) discharging to the head of each Orleans marsh. 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	Rock Harbor Cedar Pond Discharge[(a)]	Namskaket Hurley Bog Discharge[(a)]	Little Namskaket Stream Discharge[(a)]	Data Source
Total Days of Record	365[(b)]	365[(b)]	365[(b)]	(1)
Flow Characteristics				
Stream Average Discharge (m3/day) **	1271	4050	203	(1)
Contributing Area Average Discharge (m3/day)	1051	4499	282	(2)
Discharge Stream relative to Long-term Discharge	-21%	10%	28%	
Nitrogen Characteristics				
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.116	0.352	0.54	(1)
Stream Average Total N Concentration (mg N/L)	0.86	0.837	1.013	(1)
Nitrate + Nitrite as Percent of Total N (%)	13%	42%	53%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	1.09	3.39	0.21	(1)
TN Average Contributing UN-attenuated Load (kg/day)	2.62	4.45	0.41	(3)
Attenuation of Nitrogen in Pond/Stream (%)	58%	24%	49%	(4)

(a) Flow and N load to streams discharging to Orleans salt marsh systems includes apportionments of Pond contributing areas.
(b) September, 2002 to August, 2003 (Rock Harbor), July 2002 to June 2003 (Little Namskaket), July 2002 to June 2004 (Namskaket)
** Flow is an average of annual flow for 2002-2003

(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 streams to Orleans Marshes;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates as applicable.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

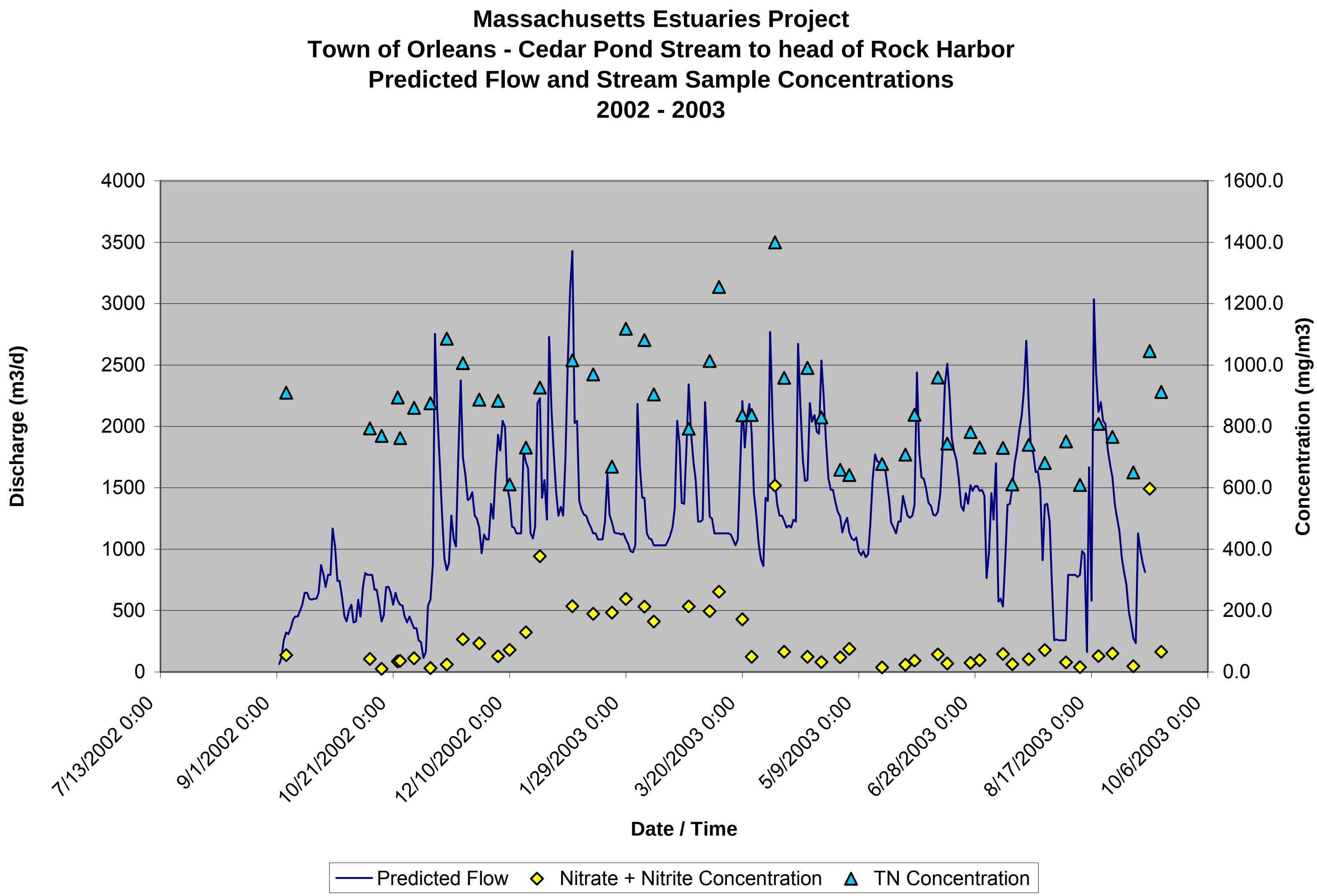


Figure IV-7. Cedar Pond stream discharge (solid blue line), nitrate+nitrite (yellow diamond) and total nitrogen (blue triangle) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Rock Harbor Marsh (Table IV-4).

Table IV-4. Summary of annual volumetric discharge and nitrogen load from the stream (freshwater) discharges into the Town of Orleans Cape Cod Bay salt marshes based upon the data presented in Figures IV-7 and Table IV-3.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Cedar Pond Stream (MEP)	**September 1, 2002 to August 31, 2003**	**463762**	**54**	**399**
Cedar Pond Stream (CCC)	Based on Watershed Area and Recharge	**385926**	--	--
Hurley Bog Stream (Freshwater) MEP	**July 1, 2002 to June 30, 2003** **July 1, 2003 to June 30, 2004** **Average**	**1669900** **1286518** **1478209**	**549** **492** **521**	**1175** **1302** **1239**
Hurley Bog Stream (Freshwater) CCC	Based on Watershed Area and Recharge	**1642008**	--	--
Stream to Little Namskaket (Freshwater) MEP	**July 1, 2002 to June 30, 2003**	**74033**	**40**	**75**
Stream to Little Namskaket (Freshwater) CCC	Based on Watershed Area and Recharge	**103091**	--	--

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 within the Rock Harbor Estuarine 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-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Rock Harbor salt marsh/embayment system predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the 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 bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the 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 bays.

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within 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 bioavailable 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. Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh). These latter salt marsh channels typify the upper region of the estuarine system under consideration in this threshold report, which is essentially tidal salt marsh with a dominant central creek, smaller tributary creeks and a salt marsh plain colonized mainly by low and high salt marsh plants. In contrast, the Rock Harbor "boat basin" area, which was created by dredging of the lower reach of the salt marsh tidal creek, functions primarily as a embayment. However, due to the basin configuration, its sediments receive enhanced deposition of organic matter from the overlying waters. The consequence of this deposition is that the basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

Failure to account for the nitrogen balance of the sediments from the tidal creeks and boat basin will result in significant errors in determination of the threshold nitrogen loading to the Rock Harbor 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-watercolumn nitrogen exchange

For the Rock Harbor System, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from 8 sites, 4 within the dredged harbor basin and 4 distributed along the tidal salt marsh creek **(Figure IV-8)** in July-August 2003. 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)** per incubation are as follows:

Rock Harbor Benthic Nutrient Regeneration Cores

- Station Rock-1 1 core (Main Basin)
- Station Rock-2/3 2 cores (Main Basin)
- Station Rock-4 1 core (Main Basin)
- Station Rock-5 1 core (Creek)
- Station Rock-6 1 core (Creek)
- Station Rock-7 1 core (Creek)
- Station Rock-8 1 core (Creek)

Figure IV-8. Rock Harbor embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

Sampling was distributed throughout the primary tidal reach of this system and the results for each site combined for calculating 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 (Harbormasters Office) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining 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. 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 organic levels within the sediment (oxic/anoxic) and the concentration of nitrate in the overlying water. Organic rich sediment systems with high overlying nitrate frequently show large net nitrogen uptake throughout the summer months, even though organic nitrogen is being mineralized and released to the overlying water as well. The rate of nitrate uptake, simply dominates the overall sediment nitrogen cycle.

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

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature **(Figure IV-9)**.

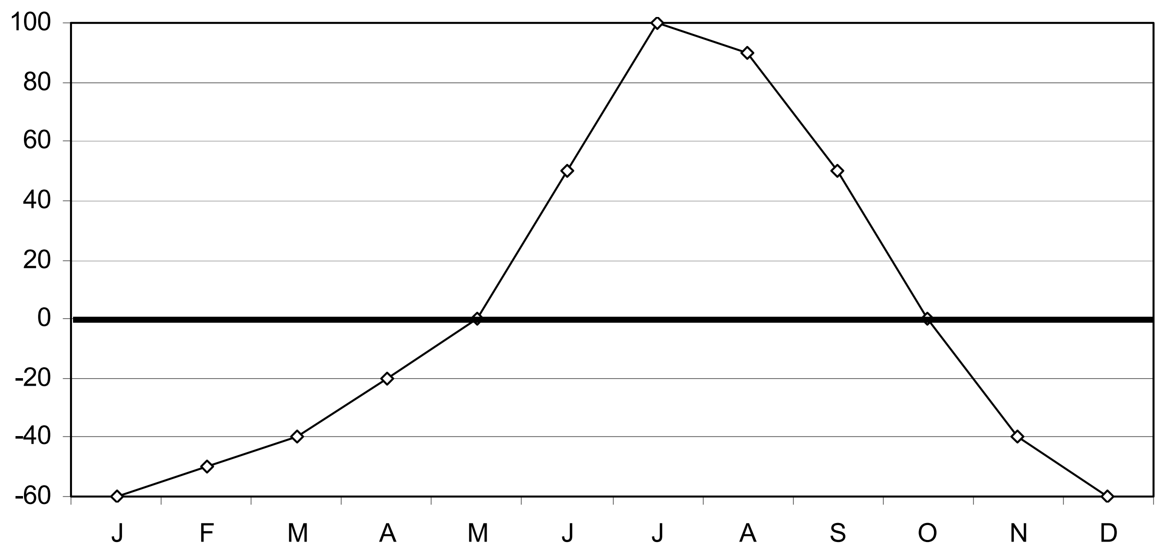


Figure IV-9. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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.

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

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site, 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 Rock Harbor Estuary were comparable to other similar embayments with similar configuration and flushing rates. In addition, the pattern of sediment N release was also similar to other systems, with the tidal salt marsh creek showing net nitrogen uptake and the tidal basin showing net nitrogen release. Sediment nitrogen release in the harbor basin was moderate/high, 80.8 mg N m^{-2} d^{-1} and is consistent with the depositional nature of the basin due to its configuration. However, the rate of nitrogen release was less than in heavily nitrogen loaded sub-embayments within the Pleasant Bay Estuary (~100 mg N m^{-1} d^{-1}). The salt marsh rates of uptake were also similar to other marsh systems with low watershed N loading. For example, net nitrogen uptake in the salt marsh creek (-20.6 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}). The Centerville River System also showed nitrogen loss from the sediments in the main depositional basin and net uptake in the salt marsh creeks (MEP Centerville River Final Nutrient Technical Report 2006, MEP Cockle Cove Technical Memorandum-Howes et al. 2006), a general pattern seen in a number of estuaries of similar structure within the MEP region.

Net nitrogen release rates for use in the water quality modeling effort for the component sub-basins of the Rock Harbor Estuarine System (Chapter VI) are presented in Table IV-5. There was a clear spatial pattern of sediment nitrogen flux, with net uptake of nitrogen by the salt marsh sediment of the tidal creek and net release by the sediments of the depositional harbor basin. The sediments within the Rock Harbor System showed nitrogen fluxes typical of similarly structured systems with low to moderate watershed nitrogen loading within the region 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-5. Rates of net nitrogen return from sediments to the overlying waters of the Rock Harbor Estuarine System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July -August rates.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			i.d. *
	Mean	S.E.	# sites	
Rock Harbor Estuary				
Upper Salt Marsh Reach	-20.6	7.1	4	ROCK 5 - 8
Lower Harbor Basin	80.8	53.9	4	ROCK 1 - 4

* Station numbers refer to Figures IV-7.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of a hydrodynamic model for the Rock Harbor 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 Rock Harbor 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 Rock Harbor system. A section of a topographic map in Figure V-1 shows the general study area. The Rock Harbor system has three main sub-divisions: 1) the main harbor basin,2) Rock Harbor marsh and 3) the marsh area separated from the main system by Dyer-Prence Road. The entire Harbor system has a surface coverage of 64 acres, including the attached harbor basin and marsh plain.

Circulation in the Rock Harbor system is dominated by tidal exchange with Cape Cod Bay. The Harbor is connected to the Bay through an inlet that empties onto a broad tidal flat, which restricts access to the harbor during portions of the tide. The lower portion of the system has been dredged to create a harbor, with jetties approximately 200 feet in length. The distance between the jetty tips is 170 feet.

Figure V-1. Topographic map detail of the Cape Cod Bay embayments of Orleans, Massachusetts, including Rock Harbor.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry, tide, and circulation velocity 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 Rock Harbor 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.

A numerical hydrodynamic model of the Rock Harbor 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 the Harbor 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 Rock Harbor 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 six locations, to run the circulation model with real tides, and also to calibrate and verify its performance.

V.2.1 Bathymetry Data Collection

Bathymetric data were collected in Orleans salt water embayments, as part of a larger town-wide effort, over the course of six separate surveys, occurring between October 30 and December 11, 2001. The surveys employed an Odem HydroTrac fathometer mounted on a 16 ft 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 Cape Cod Bay embayments were also surveyed, where depths allowed the passage of the survey boat. The actual survey paths are shown in Figure V-2, for surveys that were completed during this project. Collected bathymetry data will be tide-corrected, to account for the change in water depths as the tide level changes. 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 these systems using a level and stadia rod. The topography data were necessary to set the elevation of the marsh plain included in the hydrodynamic model.

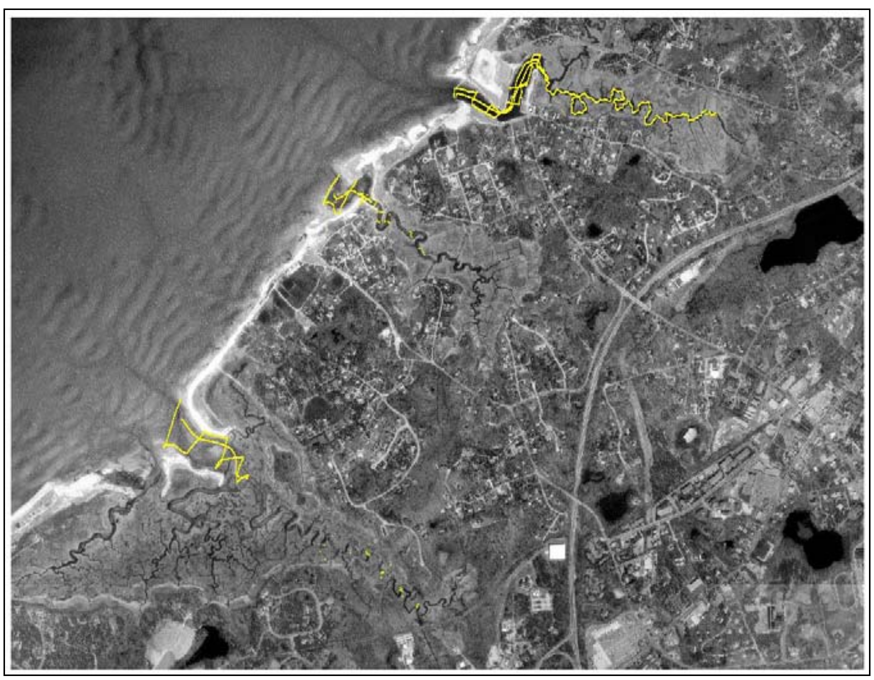


Figure V-2. Actual paths followed by recent (Fall, 2001) survey boat during bathymetry surveys of the Cape Cod Bay embayments of Orleans, MA. Fathometer paths are plotted on a composite 1994 aerial photograph of the area.

V.2.2 Tide Data Collection and Analysis

Tide data were collected for this study and in the fall of 2001. 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 (NGVD 29).

The instruments deployed in this study were typically 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 Orleans embayments. A 29-day tide record length is required to differentiate the several individual tidal constituents that make up the total observed tide. Tidal constituent determination is an ancillary analysis, which provides useful insight into the dynamics of an estuary. Typically, a shorter tide data record is used to run and calibrate hydrodynamic models, which is the primary purpose of collecting this data. A secondary purpose for the tide data, as mentioned previously, is to tide-correct bathymetry data, so that it can be referenced to some standard datum (e.g., NGVD 29).

The deployment locations for all the Orleans Cape Cod Bay embayment gages are shown in Figure V-3. Tides were measured offshore, at a location in the vicinity of the "target ship", and at stations inside each of the three Orleans Cape Cod Bay embayments systems, in Namskaket Creek, Little Namskaket Creek, and Rock Harbor. The total Cape Cod Bay deployment spanned early November to early January. For comparison, the complete tide data record is plotted in Figure V-4, from each gage deployed in the Orleans Cape Cod Bay embayments study.

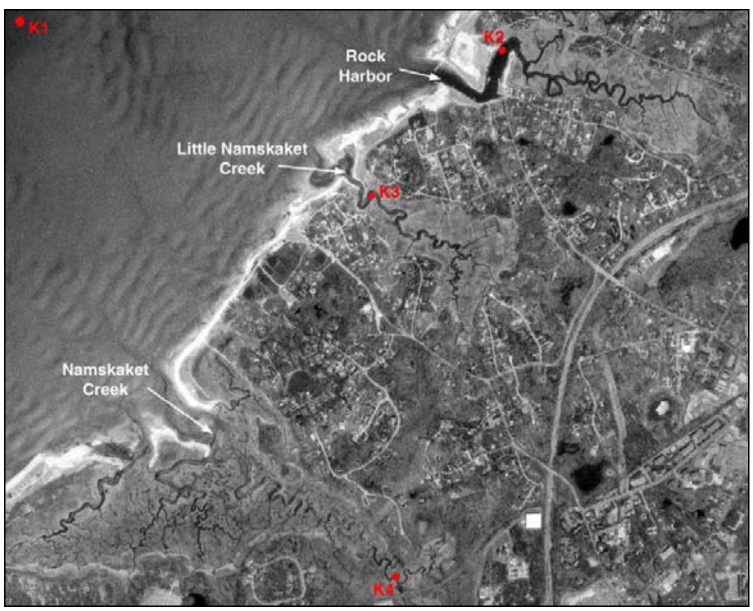


Figure V-3. Tide gage locations for the three Cape Cod Bay embayments of Orleans, at Rock Harbor (K2), Little Namskaket Creek (K3), and Namskaket Creek (K4).

An harmonic analysis was performed on the time series from each gauge location. The results of this analysis are presented in Table V-1. 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-5.

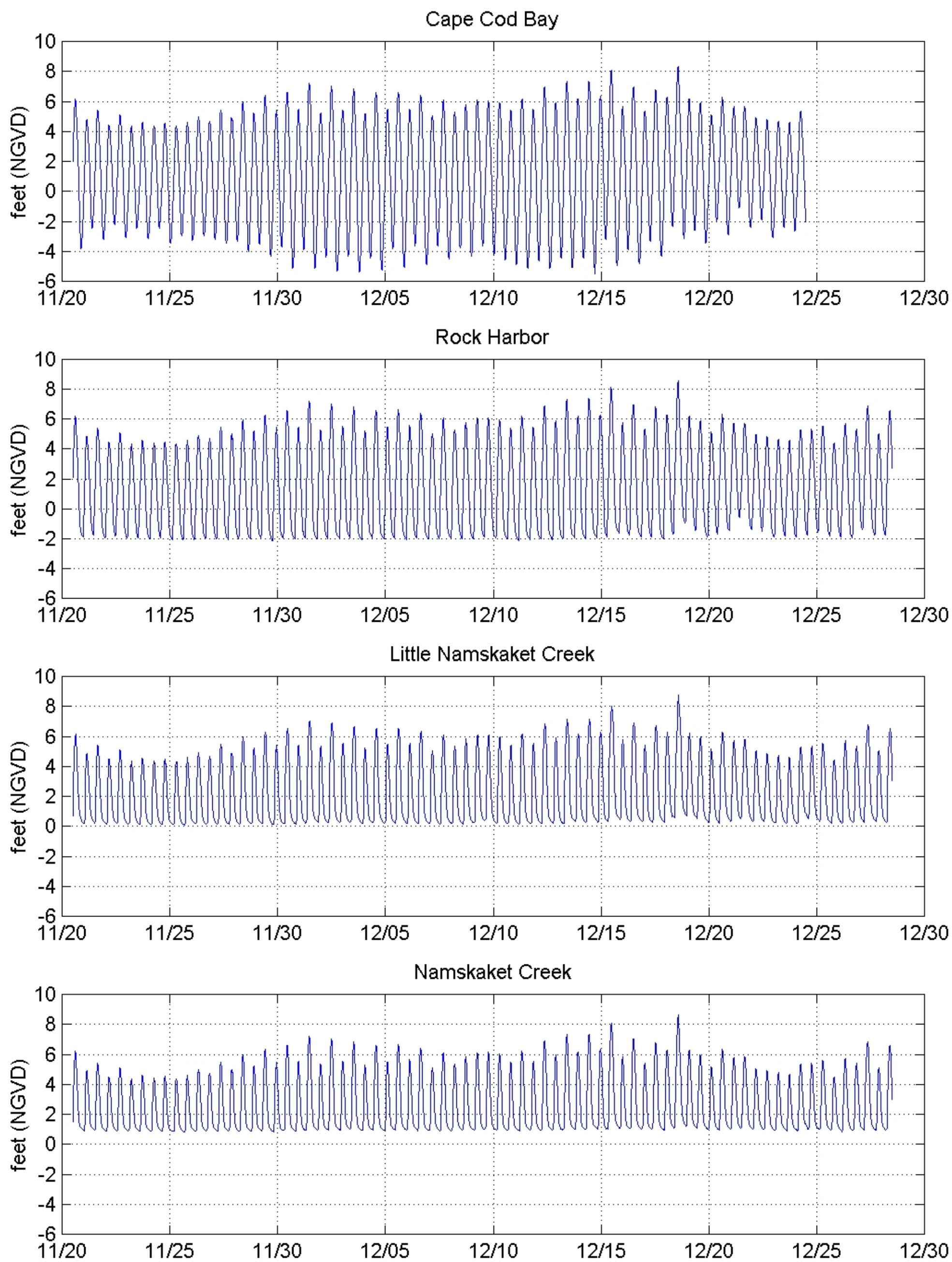


Figure V-4. Complete TDR records for gages deployed for the Cape Cod Bay embayments of Orleans during late 2001.

Table V-1. Major tidal constituents determined for gauge locations in the Cape Cod Bay embayments of Orleans, for the time period November 20 through December 24, 2001.

	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	4.58	0.09	0.22	0.69	0.84	0.64	0.41	0.19
Rock Harbor	3.82	0.50	0.48	0.48	0.58	0.50	0.34	0.24
Little Namskaket Creek	2.31	0.91	0.17	0.34	0.36	0.38	0.30	0.31
Namskaket Creek	2.67	0.92	0.18	0.37	0.42	0.44	0.33	0.33

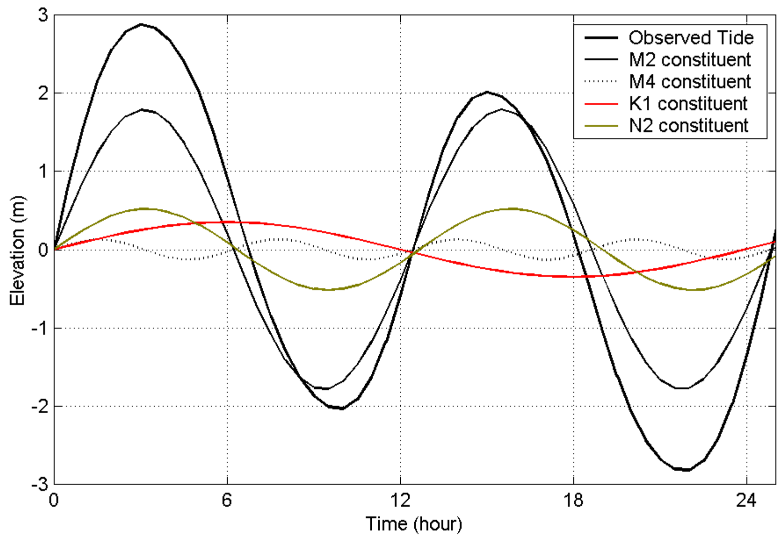


Figure V-5. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-1 presents the amplitudes of eight tidal constituents in the three Orleans Cape Cod Bay embayment systems. 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 gauge deployment locations. The total range of the M_2 tide is twice the amplitude, e.g., 4.6 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-1 show how the constituents vary as the tide propagates into the upper reaches of these marsh systems. Note the reduction in the M_2 amplitude between Cape Cod Bay and the main basin of Rock Harbor., and farther up into the marsh of Namskaket and Little Namskaket creeks. The main M_2 and K_1 tidal constituents are smaller inside the marshes due to truncation of the lower portion of the tide. This truncation occurs because the channel bottoms in each system are high enough to limit the lower range of the tide. This effect is also apparent in Figure V-6. 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.

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 embayment systems of Orleans is presented in Table V-2 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 Nauset Harbor, with the computed astronomical (predicted) tide resulting from the harmonic analysis, and the resulting non-tidal residual signal.

Table V-2 shows that the variance of tidal energy was largest in the data record from Cape Cod Bay for all the deployed gauges; as should be expected since the tide range is the greatest at this location, compared to the other sites. 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.

V.3 HYDRODYNAMIC MODELING

For the Rock Harbor 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), 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.

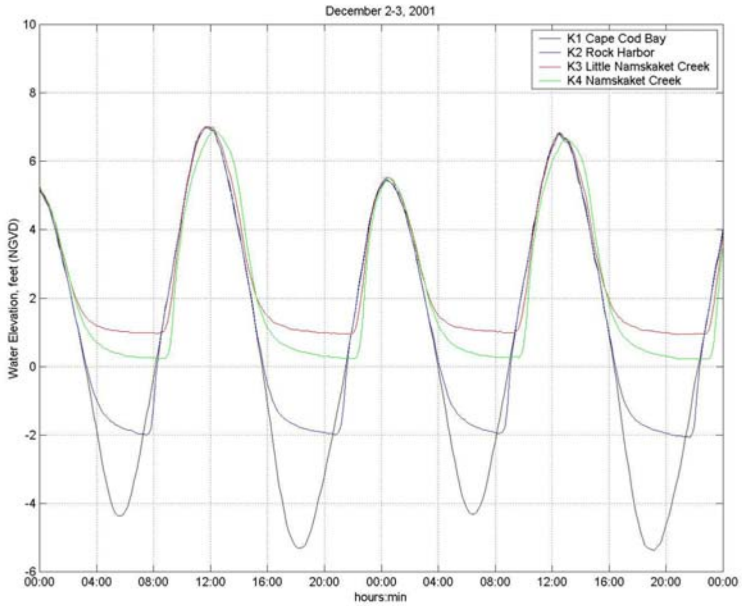


Figure V-6. Close-up of TDR record of tides recorded in Cape Cod Bay, and the three Orleans embayments connected to Cape Cod Bay. From this plot significant attenuation, is apparent caused by the influence of the extensive tidal flats that separate these systems from Cape Cod Bay at low tide. This effect of the marsh is seen by how ebbing water levels do not fall below a certain level, as if a weir or sill was present to prevent water from dropping below this level, while having very little effect on the amplitude and phase of the flooding portion of the tide.

Table V-2. Percentages of Tidal versus Non-Tidal Energy for the Cape Cod Bay embayments of Orleans.

TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Cape Cod Bay	10.74	98.5	1.5
Rock Harbor	7.56	97.5	2.5
Little Namskaket Creek	3.22	95.4	4.6
Namskaket Creek	4.18	94.6	5.4

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.

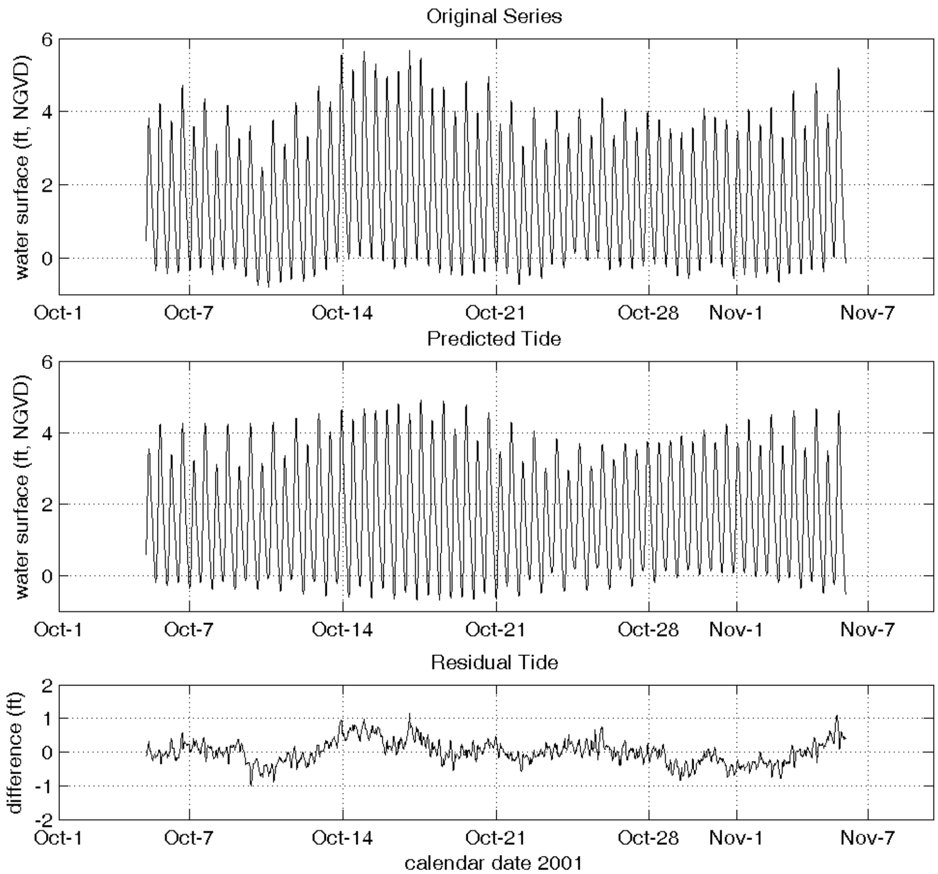


Figure V-7. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determine in the harmonic analysis of the Town Cove (sub-embayment of the Nauset Harbor system) gauge data. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

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

V.3.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each of the five separate finite element grid was generated using 1994 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 each system based on the tide gauge data collected offshore Skaket 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 for 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. 1994 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 Rock Harbor system is shown in Figure V-8. The Rock Harbor grid has a total of 5055 nodes that make up 1920 quadrilateral elements. The maximum depth included in the model is -13.1 feet NGVD, in the main harbor basin. Bathymetry data were interpolated to the developed finite element mesh of each system, as seen in Figure V-9. A typical marsh plain elevation of +5.0 ft NGVD was used, based on spot surveys of the marsh plain. This elevation also roughly corresponds to the mean high water (MHW) level in the marsh systems, which is typical.

The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties of the five modeled embayments. Areas of marsh were included in the model because they represent a large portion of the total area of this system, and have a significant effect on system hydrodynamics. Fine resolution was required to simulate channel constrictions that significantly impact the estuarine hydrodynamics, e.g. the Dyer Prence Road culvert, as well as marsh and tidal creeks found in the system. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the modeled estuaries.

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. 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.

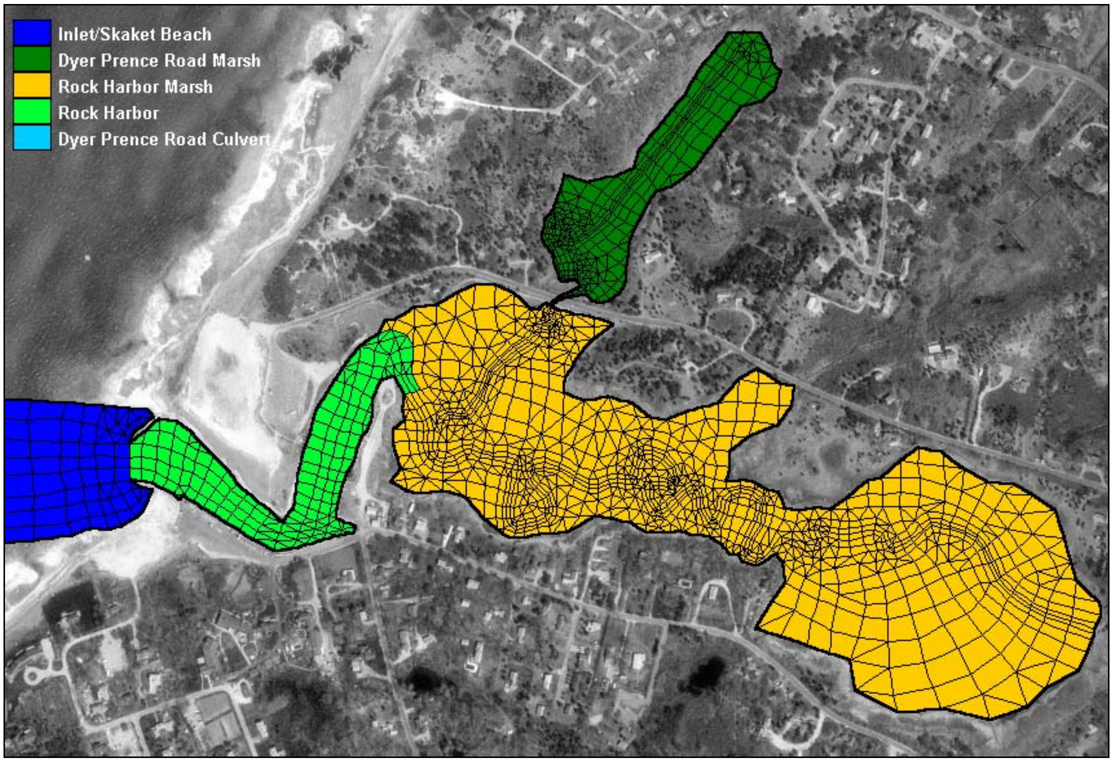


Figure V-8. Plot of numerical grid used for hydrodynamic modeling of the Rock Harbor system. Colored divisions indicate boundaries of different grid material types, as well as volumes used to compute flushing rates.

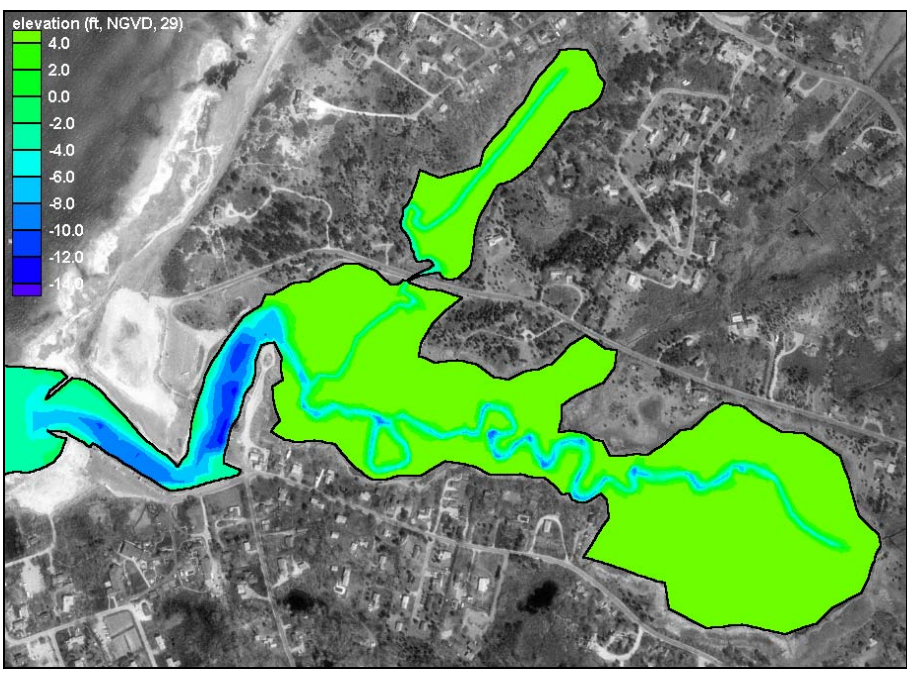


Figure V-9. Depth contour plot of the numerical grid for Rock Harbor showing 2-foot contour intervals relative to NGVD29.

V.3.2.2 Boundary condition specification

Two types of boundary conditions were employed for the RMA-2 model of Rock Harbor: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations.

A tidal boundary condition was specified at the inlet. TDR measurement tide data collected offshore Skaket 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 each system corresponded to the time step of the tide data record.

V.3.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the Rock Harbor 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 (typically 20+) 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 five-day period (10 tide cycles, or 5 lunar days) was modeled to calibrate the model based on dominant tidal constituents discussed previously in the data collection section (Section W). For Rock Harbor, the calibration period began November 30, 2001 at 1100 EST. The five-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

To provide average tidal forcing conditions for model verification and the flushing analysis, a separate time period was chosen that spanned the transition between spring and neap tide ranges (bi-weekly maximum and minimum tidal ranges, respectively). For The verification period for Rock Harbor began December 7, 2001 at 1600 EST.

The calibrated model was used to analyze existing detailed flow patterns and compute embayment residence times. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire seven-day simulation period used for each system. 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 Rock Harbor, 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-3.

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 80 and 100 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 four modeled Orleans embayments which include marsh resources.

The constituent calibration resulted in generally excellent agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.1 ft, which is of the same order of the accuracy of the tide gages (±0.12 ft). Time lag errors were typically less than the time increment resolved by the model (1/6 hours or 10 minutes), indicating good agreement between the model and data.

The hydrodynamic model's ability to predict propagation of the secondary non-linear constituents through the estuary is important for understanding the attenuation of the tidal signal and the impact this has on estuarine circulation. Of primary interest is the M_4 constituent, which can be used to determine the flood dominance (sediment trapping characteristics) of an estuarine system. Proper prediction of M_4 provides confidence in the model's accuracy, since this indicates that the model is capable of simulating the tidal wave form and size. Similar to the model predictions for M_2, comparison of the information from Table V-4 indicates that the modeled phase of the M_4 generally falls within one time step of the observed value.

Table V-3. Manning's Roughness 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
Rock Harbor Inlet (Skaket Beach)	0.150
Dyer Prence Road marsh plain	0.070
Rock Harbor Creek marsh plain	0.070
Lower Rock Harbor (Boat Basin)	0.030
Dyer Prence Road Culvert	0.100

V.3.2.3.4 Comparison of modeled tides and measured tide data

A best-fit of model predictions for the TDR deployments was achieved using the aforementioned values for friction and turbulent exchange. Figures V-10 and V-11 illustrate the five-day calibration simulation along with 50-hour sub-section, for the Rock Harbor model. Modeled (solid line) and measured (dotted line) tides are illustrated at each model location with a corresponding TDR.

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 modeled systems. Due to the duration of the model runs (approximately 5 days of simulated time), four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-4 for the calibration period differ from those shown previously in the data collection section of this report Table V-1 because constituents were computed for only the five-day section of the longer time periods represented in Table V-1. Table V-4 compares tidal constituent height and phase for modeled and measured tides at the TDR locations. The constituent phase shows the relative timing of each separate constituent at a particular location, and also the change (or phase lag) in timing of a single constituent at different locations in an estuary.

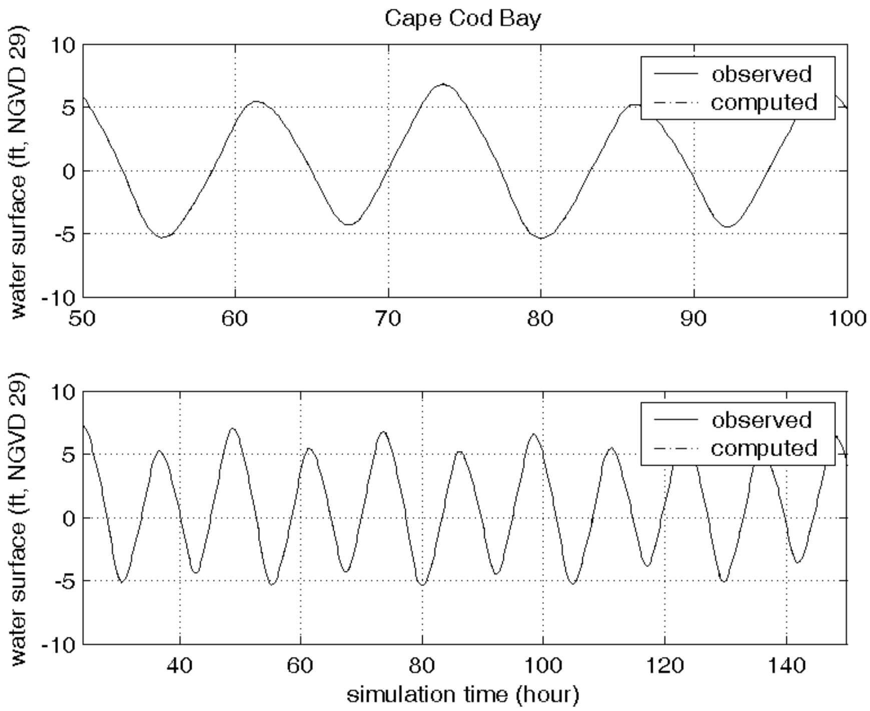


Figure V-10. Comparison of model output and measured tides for the TDR location offshore Skaket Beach (Cape Cod Bay), during model calibration time period. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

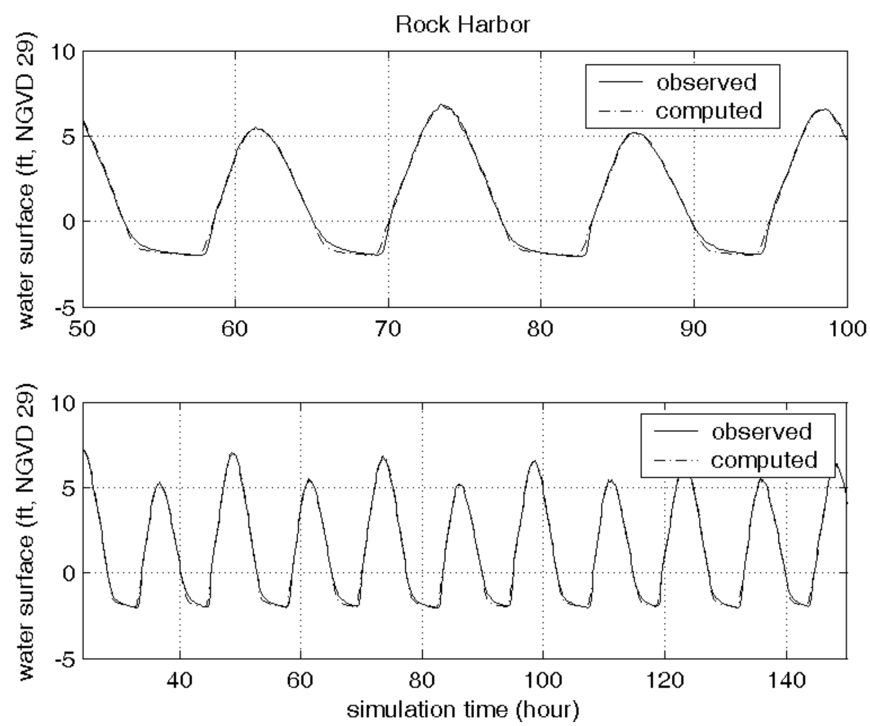


Figure V-11. Comparison of model output and measured tides for the TDR location in Rock Harbor, during model calibration time period. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

Table V-4. Tidal constituents for measured water level data and calibrated model output for Rock Harbor during model calibration time period.

Location	Constituent Amplitude (ft)				Phase (deg)	
Model calibration run						
Location	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Cape Cod Bay*[a]	5.14	0.12	0.28	0.98	10.6	185.3
Rock Harbor	4.11	0.60	0.17	0.75	13.2	13.1
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Cape Cod Bay*[a]	5.14	0.12	0.28	0.98	10.6	184.1
Rock Harbor	4.07	0.70	0.19	0.75	13.6	1.4
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Cape Cod Bay*[a]	0.00	0.00	0.00	0.00	0	-1
Rock Harbor	-0.04	0.10	0.02	0.00	1	-12

*model open boundary

V.3.2.3.4 Model Verification

The calibration procedure used in the development of the finite-element model of Rock Harbor required a match between measured and modeled tides. An additional model verification model run was performed to verify the model performance during time periods different from the calibration time period. In this fashion, the calibrated model is tested to ensure its accuracy when run for any time period outside the calibration period. The selected verification time period was chosen so that it did not overlap with the original calibration period. The results of the model verification runs are shown in Table V-5. The analysis of the verification model run in this table shows that the model performs with a similar degree of accuracy to the calibration run.

Table V-5. Tidal constituents for measured water level data and calibrated model output for Rock Harbor during model verification time period.

Location	Constituent Amplitude (ft)				Phase (deg)	
Model calibration run						
Location	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Cape Cod Bay*[a]	5.19	0.11	0.28	0.66	64.8	-33.7
Rock Harbor	4.22	0.58	0.20	0.54	67.6	116.1
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Cape Cod Bay*[a]	5.19	0.11	0.28	0.66	64.9	-37.8
Rock Harbor	4.17	0.69	0.22	0.52	67.7	105.4
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Cape Cod Bay*[a]	0.00	0.00	0.00	0.00	0	-4
Rock Harbor	-0.05	0.11	0.02	-0.02	0	-11

*model open boundary

V.4 FLUSHING CHARACTERISTICS

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through inlet of Rock Harbor, the primary mechanism controlling estuarine water quality within these systems is tidal exchange. For example, a rising tide offshore Rock Harbor 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.

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.

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. The **system residence time** of the Dyer-Prence Marsh is the average time required for water to migrate from the marsh, through the culvert under Dyer-Prence Road, and out through Rock Harbor inlet. Alternately, the **local residence time** is the average time required for water to migrate from the Dyer-Prence Marsh to just Rock Harbor (not all the way to the inlet). Local residence times for each sub-embayment are computed as:

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

where T_{local} denotes the residence time for the local sub-embayment, V_{local} represents the volume of the sub-embayment at mean tide level, P equals the tidal prism (or volume entering the local sub-embayment through a single tidal cycle), and t_{cycle} the period of the tidal cycle (again, 0.52 days).

Residence times are provided as a first order evaluation of estuarine water quality. Lower residence times generally correspond to higher water quality; however, residence times may be misleading depending upon pollutant/nutrient loading rates and the overall quality of the receiving waters. As a qualitative guide, **system residence times** are applicable for systems where the water quality within the entire estuary is degraded and higher quality waters 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 Rock Harbor.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were computed for the entire estuary, as well the two main sub-embayments within the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system. Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The mean volume and mean tidal prism of each modeled sub-embayment is presented in Table V-6.

Residence times were averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and are listed in Table V-7 for Rock Harbor. The modeled time period used to compute the flushing rates was the same as the model verification period, and included the transition from neap to spring tide conditions. Model divisions used to define the sub-embayments of each system include are shown in Figure V-8 of the previous section. The RMA-2 model calculated flow crossing specified grid lines for each sub-embayment, and this flow data was used to compute the mean tidal prism volume. Since the 7.25-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

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

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Cape Cod Bay Embayments		
Rock Harbor System	4,879,500	5,849,000
Prence Road Marsh	288,700	268,400

Table V-7. Computed System and Local residence times for embayment systems in Orleans, Massachusetts		
Embayment	System Residence Time (days)	Local Residence Time (days)
Rock Harbor System	0.4	0.4
Prence Road Marsh	9.4	0.6

The computed flushing rates for Rock Harbor show that the system generally flushes well, with flushing times less than a day for the whole system. Marsh systems like Rock Harbor are expected to flush very well, since in marshes, the mean volume is much smaller than mean tidal prism.

Based on our knowledge of estuarine processes, we estimate that the combined errors due to bathymetric inaccuracies represented in the model grid and the “strong littoral drift” assumption are within 10% to 15% of “true” residence times for all of the modeled Orleans embayments. Generally, possible errors in computed residence times could be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. In this study, the most significant errors associated with the bathymetry data result from the process of interpolating the data to the finite element meshes used to model each system, which provided the basis for all the flushing volumes used in the analysis. In addition, limited topographic measurements were available on the extensive marsh plain included in the

Minor errors may be also introduced in residence time calculations by simplifying assumptions. Flushing rate calculations assume that water exiting an estuary or sub-embayment does not return on the following tidal cycle. For regions where a strong littoral drift exists, this assumption is valid. However, water exiting a small sub-embayment on a relatively calm day may not completely mix with estuarine waters. In this case, the “strong littoral drift” assumption would lead to an under-prediction of residence time. Since littoral drift in Cape Cod Bay is typically is strong, the “strong littoral drift” assumption is completely appropriate, and would result in only minor errors in residence time calculations.

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 Rock Harbor 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 Rock Harbor 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 October 24, 2001 1830 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 Rock Harbor system's sub-embayments, 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. Typically, six years of data (collected between 2001 and 2006) were available for stations monitored by SMAST.

Table VI-1. Measured data and modeled nitrogen concentrations for the Rock Harbor estuarine system used in the model calibration plots of Figures VI-2 and VI-3. All concentrations are given in mg/L N. "Data mean" values are calculated as the average all samples. Data represented in this table were collected in the summers of 2001 through 2006, except the Cape Cod Bay station, which are from the 2003 through 2005 seasons.

Sub-Embayment	monitoring station	data mean	s.d. all data	N	model min	model max	model average	model target
Rock Harbor Mouth	WMO-15	0.854	0.219	34	0.357	0.677	0.447	-
Rock Harbor Basin	WMO-16	0.661	0.149	23	0.359	0.859	0.574	0.574
Rock Harbor Marsh	WMO-17	1.098	0.238	18	0.362	1.053	0.686	1.053
Rock Harbor Creek	WMO-18	1.142	0.436	24	0.660	1.062	0.829	1.062
Cape Cod Bay	WMO-14	0.357	0.059	15	0.357	0.357	0.357	0.357

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 Rock Harbor 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 Rock Harbor. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other Cape Cod embayments, including systems other systems in Falmouth (Howes *et al*., 2005); Mashpee, MA (Howes *et al*., 2004) and Chatham, MA (Howes *et al*., 2003).

The overall approach involves modeling total nitrogen as a non-conservative constituent, where bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. This modeling represents summertime conditions, when algal growth is at its maximum. Total nitrogen modeling is based upon various data collection efforts and analyses presented in previous sections of this report. Nitrogen loading information was derived from the Cape Cod Commission watershed loading analysis (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 Rock Harbor system.

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

VI.2.1 Model Formulation

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

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

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

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

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

Based on measured surface water flow rates from SMAST and groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include the latest estimates of flows from Cedar Pond (0.52 ft^3/sec).

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 Rock Harbor 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 Cedar Pond. Nitrogen loads from Harbor 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 Rock Harbor 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 Rock Harbor, the sum of all benthic flux in the system is positive, indicating that on average the bottom sediments of the creek channels and harbor basin are a net producer of nitrogen.

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 Rock Harbor was set at 0.357 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 Rock Harbor 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)
Rock Harbor	7.978	0.079	1.382
Cedar Pond	1.088	-	-
System Total	9.066	0.079	1.382

VI.2.4 Model Calibration

Calibration of the total nitrogen model of Rock Harbor 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 Rock Harbor estuary system.

Embayment Division	E m^2/sec
Open Boundary - Cape Cod Bay	5.0
Rock Harbor - main basin	0.1
marsh plain	5.0
Rock Harbor Creek	0.1
marsh channels	0.1

Comparisons between calibrated model output and measured nitrogen concentrations are shown in plots presented in Figures VI-2 and 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 SMAST monitoring stations.

For model calibration, the average modeled TN was compared to mean measured TN data value at station WMo-16. The calibration target would fall near the modeled mean TN because the monitoring data are collected, as a rule, during mid ebb tide. At stations WMO-17 and WMO-18, the model target was set at the maximum modeled TN concentration, because these stations are located in marsh creeks which are typically dry during the stage of the tide that the water samples are collected. Also presented in this figure are unity plot comparisons of measured data verses modeled target values for each system.

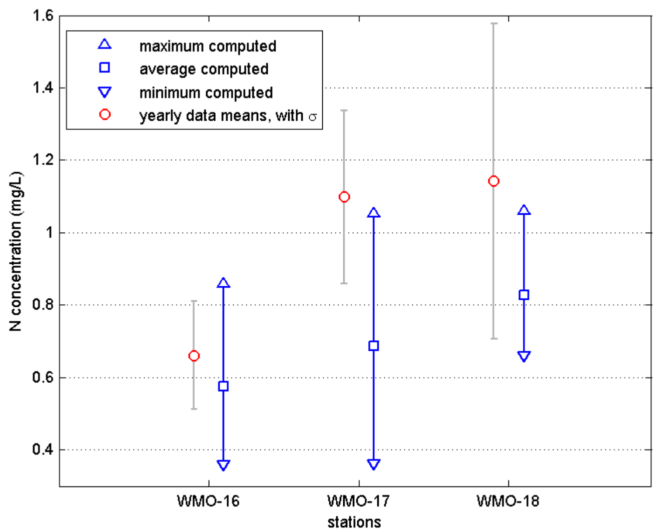


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Rock Harbor system. Station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation 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

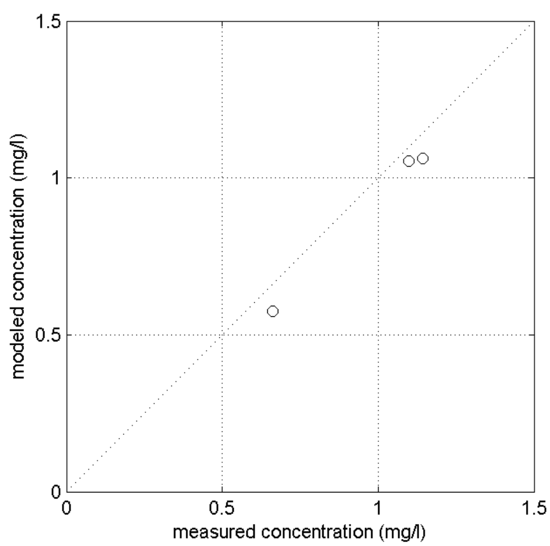


Figure VI-3. Model total nitrogen calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) are 0.89 and 0.07 mg/L, respectively.

A contour plot of calibrated model output is shown in Figures VI-4. In this figure, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full 7-tidal-day model simulation output period.

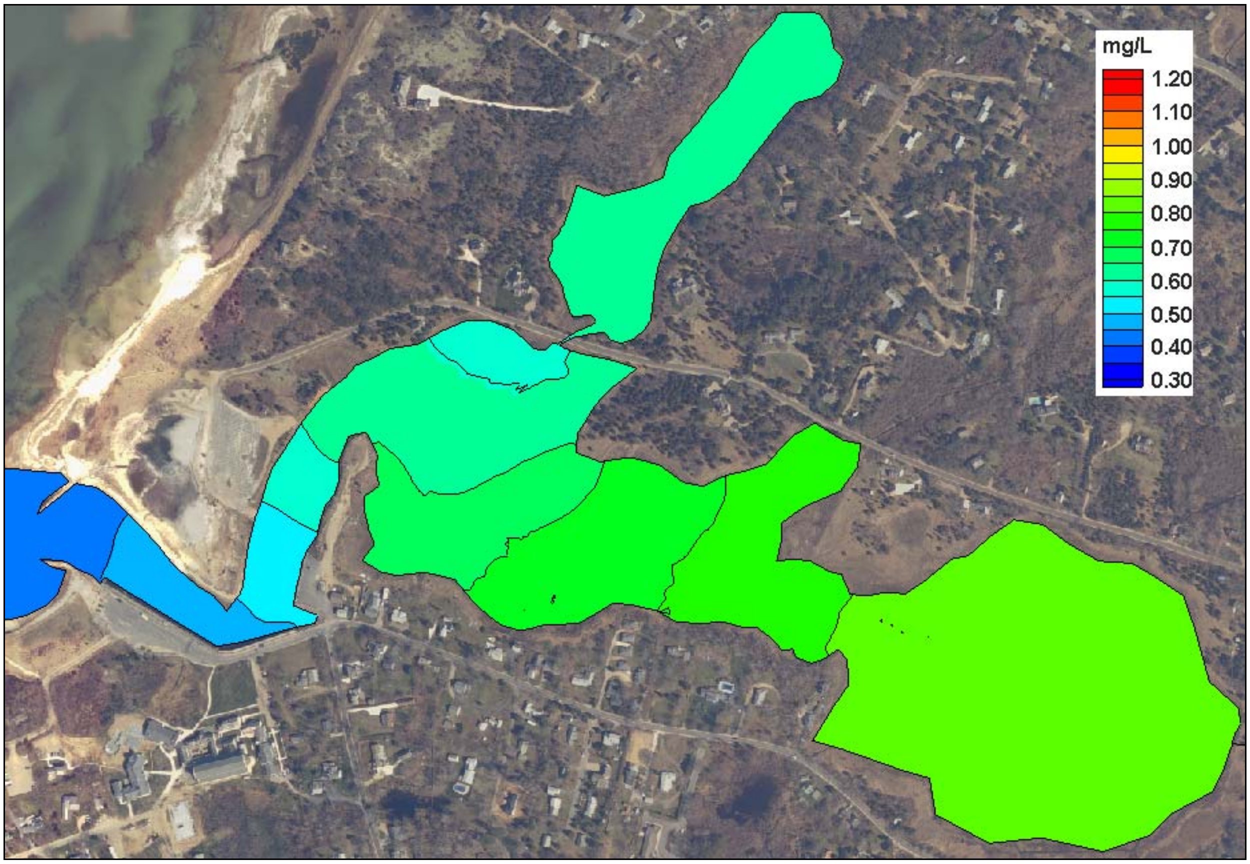


Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Rock Harbor system.

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 Rock Harbor 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 30.8 ppt. For surface water steams and 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 1.50 ft^3/sec (3,680 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 Figures VI-5 and VI-6, with contour plots of model output shown in Figure VI-7. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients throughout the Rock Harbor estuary system. The rms error of the model is less than 0.7 ppt, and correlation coefficient between the model and measured salinity data is 0.99. 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 Rock Harbor, 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.

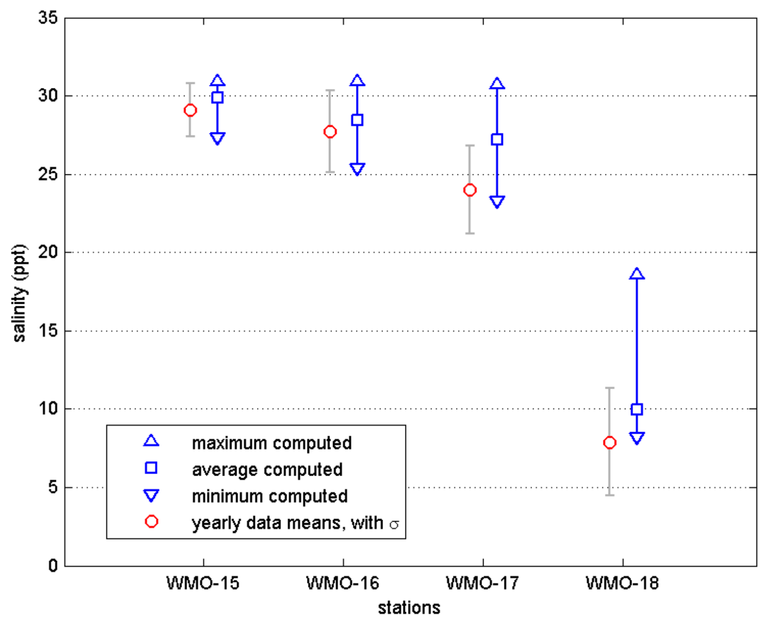


Figure VI-5. Comparison of measured and calibrated model output at stations in Rock Harbor. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with 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.

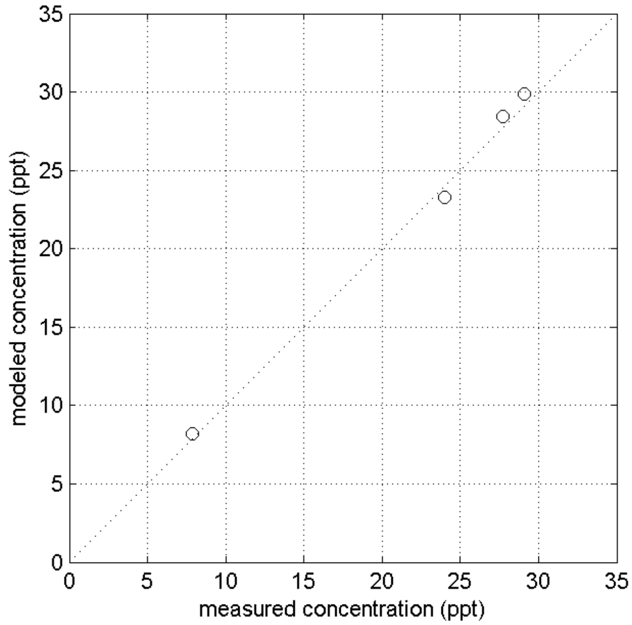


Figure VI-6. Model salinity target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) are 0.99 and 0.67 ppt, respectively.

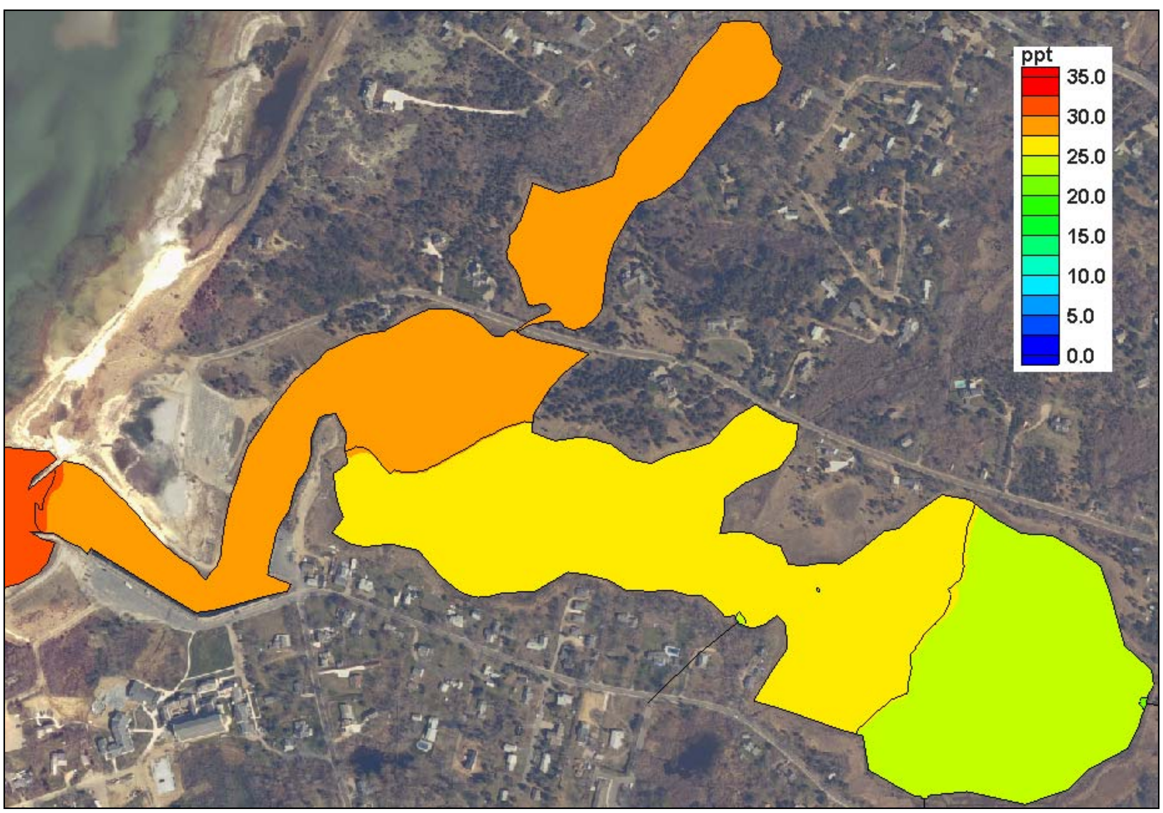


Figure VI-7. Contour Plot of modeled salinity (ppt) in the Rock Harbor system.

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 Rock Harbor 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
Rock Harbor	7.978	10.668	+33.7%	0.822	-89.7%
Cedar Pond	1.088	4.474	+311.3%	0.107	-90.2%
System Total	9.066	15.142	+67.0%	0.929	-89.8%

VI.2.6.1 Build-Out

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 *vise versa*.

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

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 Rock Harbor 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)
Rock Harbor	10.668	0.079	1.572
Cedar Pond	4.474	0.000	0.000
System Total	15.142	0.079	1.572

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model was run to determine nitrogen concentrations across the system (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Cape Cod Bay) remained 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 Rock Harbor Creek discharge (+100.5% at WMO-18), with the least change occurring in the Harbor mouth (15.9% at WMO-15) near the system's inlet to Cape Cod Bay.

Color contours of model output for the build-out scenario are present in Figure VI-8. 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 Rock Harbor system. The sentinel threshold station is in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Rock Harbor Mouth	WMO-15	0.447	0.518	15.9%
Rock Harbor Basin	WMO-16	0.574	0.752	30.8%
Rock Harbor Marsh	**WMO-17**	**0.686**	**0.963**	**40.4%**
Rock Harbor Creek	WMO-18	0.829	1.662	100.5%

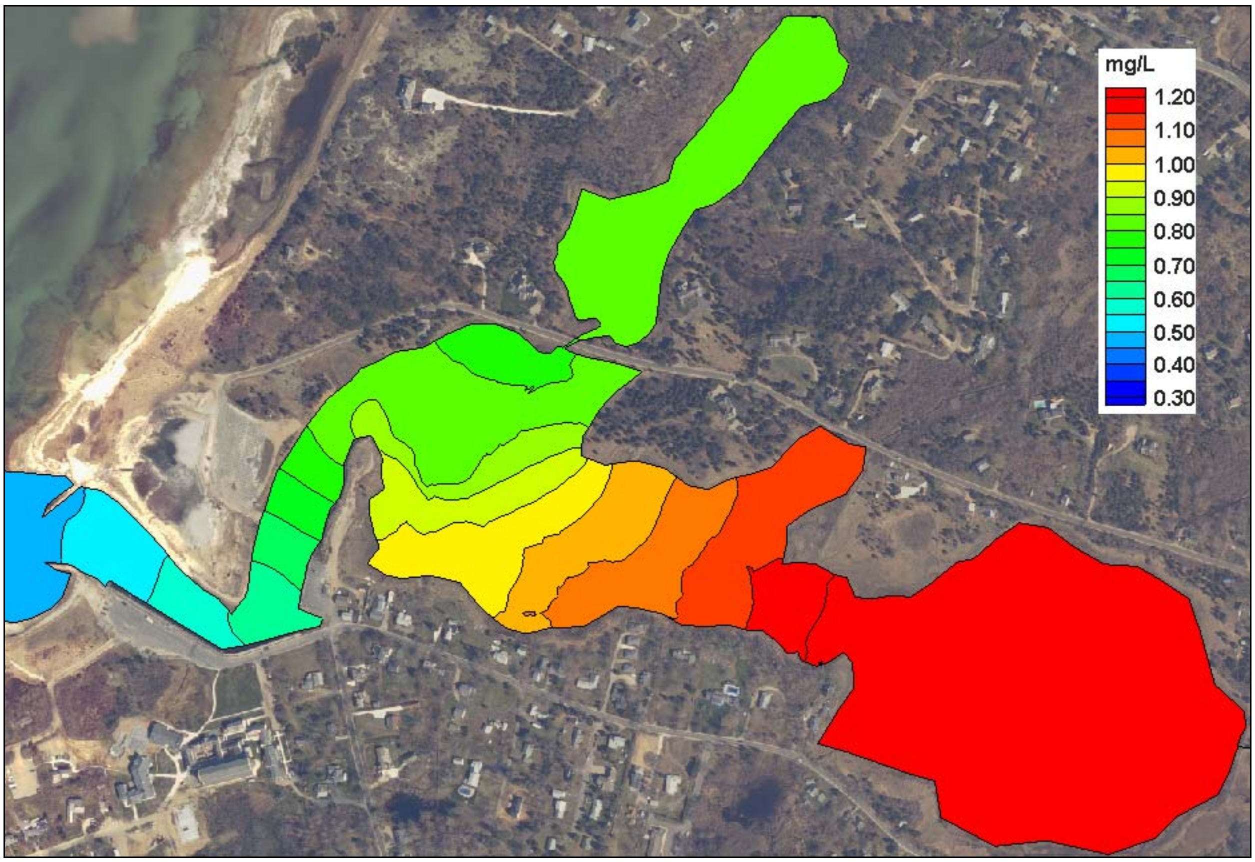


Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Rock Harbor system, for projected build-out loading conditions.

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 modeled present conditions and build-out scenario, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Table VI-7. “**No anthropogenic loading**” (“no load”) sub-embayment and surface water loads used for total nitrogen modeling of the Rock Harbor 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)
Rock Harbor	0.822	0.079	1.115
Cedar Pond	0.107	0.000	0.000
System Total	0.929	0.079	1.115

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run 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 reductions greater than 59% (at WMO-18) occurring the upper portions of the system. Results for the system are shown pictorially in Figure VI-9.

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 Rock Harbor 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
Rock Harbor Mouth	WMO-15	0.447	0.367	-17.8%
Rock Harbor Basin	WMO-16	0.574	0.380	-33.9%
Rock Harbor Marsh	**WMO-17**	**0.686**	**0.384**	**-44.0%**
Rock Harbor Creek	WMO-18	0.829	0.332	-59.9%

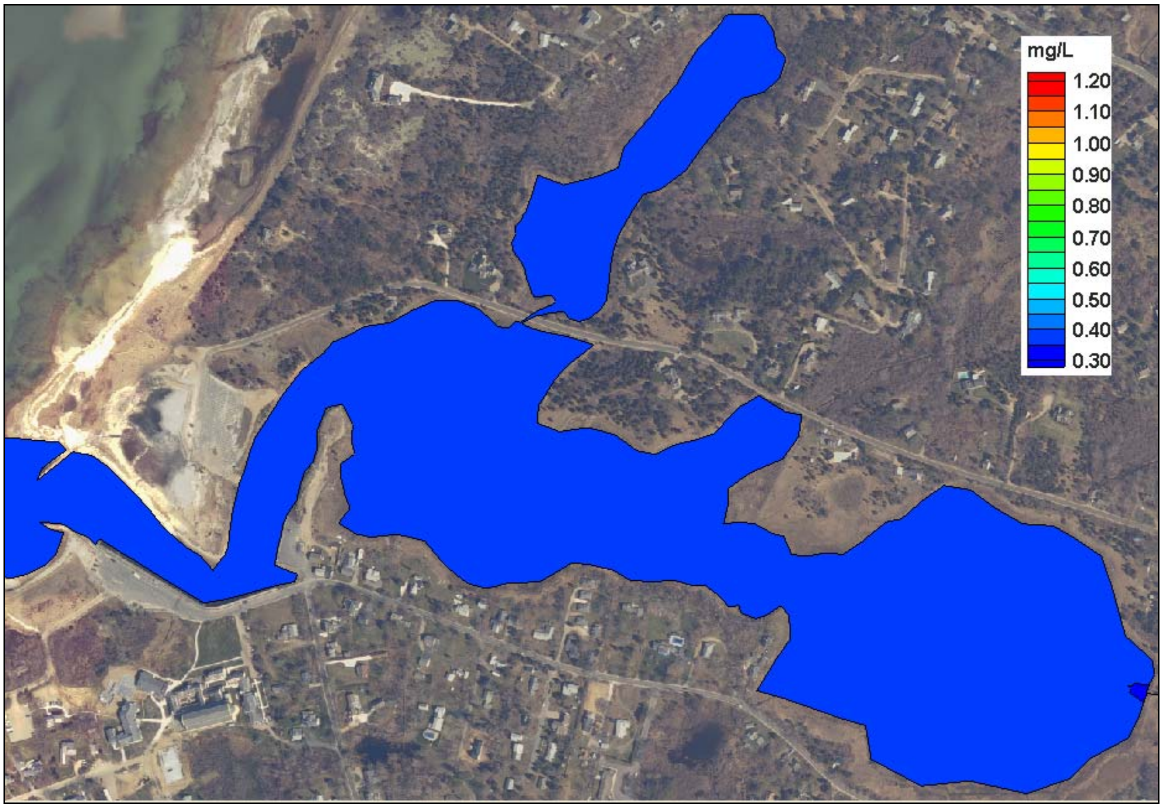


Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in Rock Harbor, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Rock Harbor embayment system in the Towns of Orleans and Eastham, MA, the MEP habitat assessment is based upon data from the water quality monitoring database developed by the Town of Orleans Water Quality Monitoring Program. Additionally, the assessment is comprised of data collection on eelgrass distribution, benthic animal communities and sediment characteristics, and dissolved oxygen records, all of which was conducted during the summer and fall of 2003. 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, support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within the upper portion of the main sub-embayment basin of the Rock Harbor system (the lower harbor basin) to record the frequency and duration of low oxygen conditions during the critical summer period. 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. Mapping of the eelgrass beds within the Rock Harbor system was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels.

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 3.8 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Rock Harbor, Little Namskaket Creek and Namskaket Creek systems 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. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (see Figure VII-1 for example). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, an autonomously recording oxygen sensor was moored 30 cm above the embayment bottom within the key region of the Rock Harbor (Figure VII-2) system. The sensor (YSI 6600) was first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition, periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during the deployment. The 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 Rock Harbor embayment system was collected during the summer of 2003.

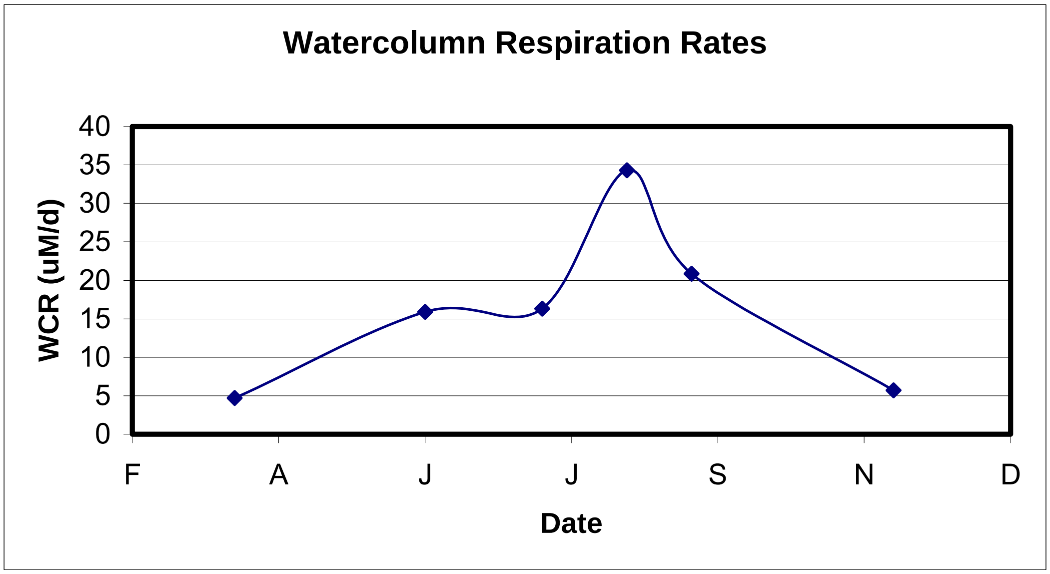


Figure VII-1. By example, average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Similar to other embayments in southeastern Massachusetts, the Rock Harbor System evaluated in this assessment showed high frequency variation, apparently related to diurnal and sometimes tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration 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 25-28 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. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate a highly nutrient and organic matter enriched basin and impaired benthic habitat at the mooring site within the Rock Harbor Estuary (Figures VII-3 and VII-4). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of these estuarine systems. 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. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). The clear evidence of oxygen levels above atmospheric equilibration indicates that the upper portion of the main basin to the Rock Harbor system is nutrient enriched.

The dissolved oxygen records indicate that the main sub-tidal basin of the Rock Harbor System (the harbor basin) is currently under seasonal oxygen stress, consistent with nitrogen and organic matter enrichment (Table VII-1). Tidally averaged total nitrogen levels at the mooring location were high (0.69 mg N L^{-1}) for a coastal sub-embayment (e.g. retains water at low tide). Other similarly configured sub-embayments on Cape Cod with similar TN levels also show significant oxygen depletion and impaired infaunal habitat quality (e.g. upper Bournes Pond or Great Pond, Falmouth). It should be noted that this level of TN is typical of upper tidal salt marsh creeks (e.g. Cockle Cove Creek, Chatham), and does not indicate impairment in that type of environment. The specific results are as follows:

Rock Harbor **(Figures VII-3 and VII-4):**

Rock harbor functions primarily as a tidal salt marsh within its upper estuarine reach and an embayment in its lower reach. 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. The lower harbor basin, functions as a tidal sub-embayment, retaining water at low tide and supporting plankton and fish communities throughout the tidal cycle. The oxygen dynamics of the lower basin are controlled by the level of organic matter enrichment through nitrogen loading from the watershed and from deposition of detritus from the extensive upper marshes. The structure of the harbor basin enhances organic matter deposition, which increases the level of oxygen depletion.

Relatively low levels of chlorophyll a were observed within the basin waters, although some enrichment was observed. The chlorophyll a concentrations result from the high flushing rate of harbor waters. Although there is a diurnal variation in the dissolved oxygen data, the primary signal is tidal. Many of the largest depletions in dissolved oxygen are coincident with low tide in the early morning, while the largest departures above air equilibration occurred when low tide occurred near sunset. The daily average dissolved oxygen concentration varied inversely with the tidal amplitude suggesting that longer residence time and greater areal submergence of the marsh may have been partially responsible for the lowest oxygen levels observed. Increased chlorophyll concentrations (8-12 ug L^{-1}) near the end of the deployment (August 23-28) appeared to have only a modest impact on dissolved oxygen depletion. Oxygen depletion in this system was frequently to ~3 mg L^{-1} or below. In open embayments oxygen depletion to <3 mg L^{-1} generally indicates habitat impairment, as is the case in the Rock Harbor lower basin. However, in salt marsh creeks the extent of oxygen depletion is consistent with the organically enriched nature of the ecosystem and common to even pristine salt marshes.

Figure VII-2. Aerial Photograph of the Rock Harbor system in Orleans showing the location of the Dissolved Oxygen mooring deployment conducted in the Summer of 2003.

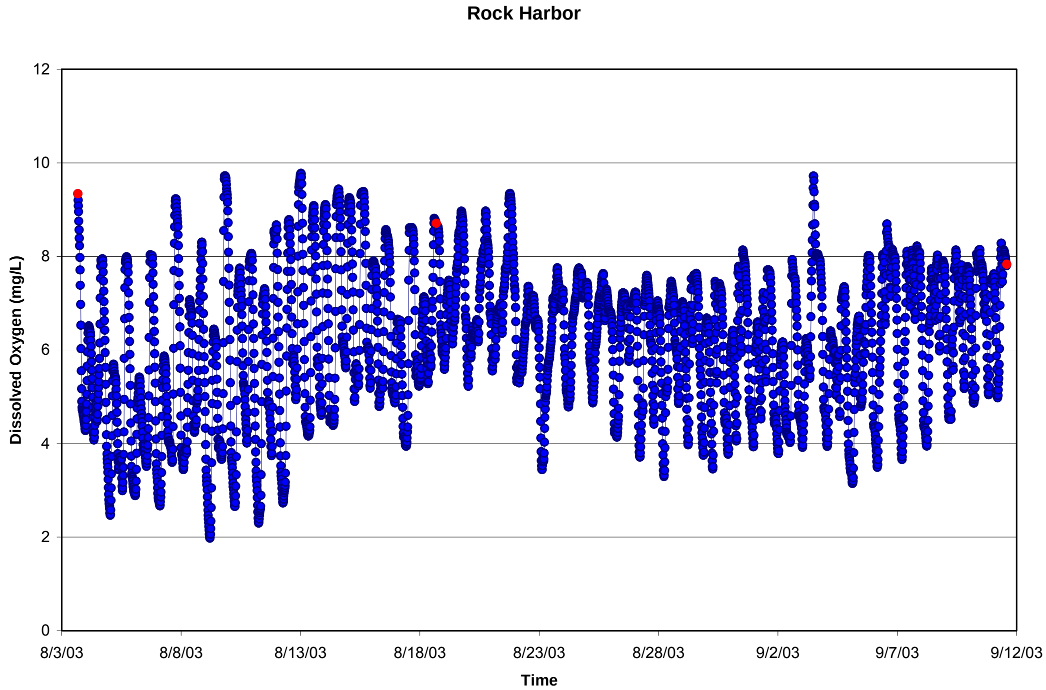


Figure VII-3. Bottom water record of dissolved oxygen in Rock Harbor, Summer 2003. Calibration samples represented as red dots

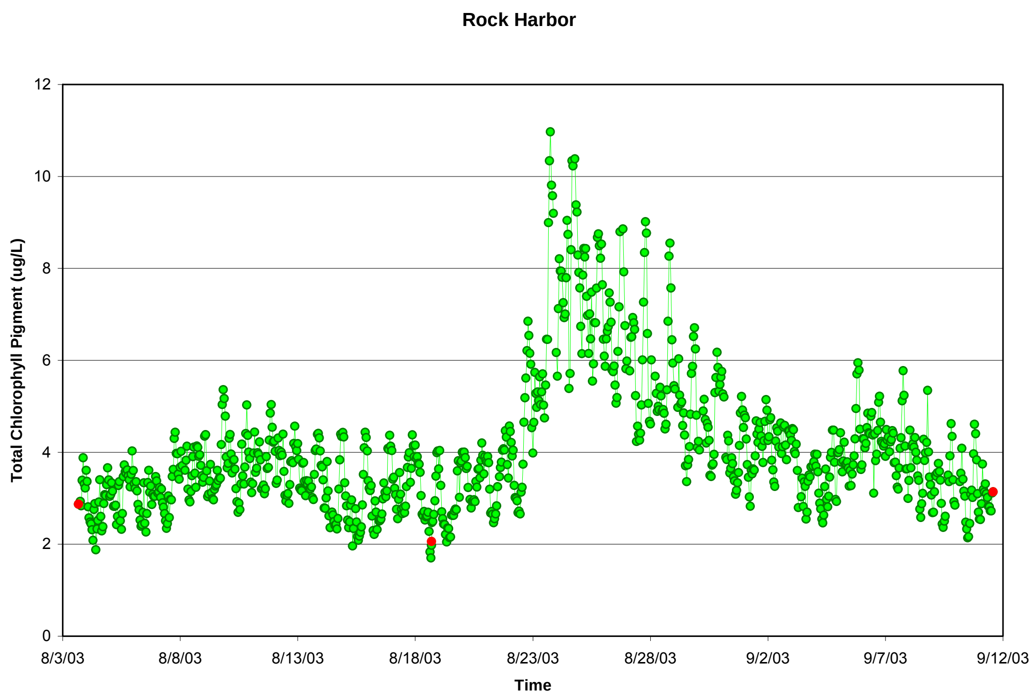


Figure VII-4. Bottom water record of Chlorophyll-*a* in Rock Harbor, Summer 2003. Calibration samples represented as red dots.

Table VII-1. Percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels.

Massachusetts Estuaries Project Town of Orleans: 2003	Dissolved Oxygen: Continuous Record, Summer 2003				
	Deployment Days	< 6 mg/L (% of days)	< 5 mg/L (% of days)	< 4 mg/L (% of days)	< 3 mg/L (% of days)
Rock Harbor	38.9	43%	25%	10%	2%
Namskaket	38.6	55%	34%	18%	16%
Little Namskaket	40.8	62%	43%	25%	9%

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

Embayment System	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)
Rock Harbor								
Rock Harbor	8/3/2003	9/11/2003	37.6	18%	1%	0%	0%	0%
		Mean		0.32	0.10	N/A	N/A	N/A
		S.D.		0.76	0.03	N/A	N/A	N/A
Namskaket	8/3/2003	9/11/2003	38.8	63%	18%	4%	2%	1%
		Mean		0.61	0.180	0.10	0.09	0.10
		S.D.		1.68	0.09	0.04	0.03	0.02
Little Namskaket	8/1/2003	9/11/2003	40.7	2%	0%	0%	0%	0%
		Mean		0.15	N/A	N/A	N/A	N/A
		S.D.		0.04	N/A	N/A	N/A	N/A

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data is key part of the MEP Approach. Surveys were conducted in the vicinity of the Rock Harbor System by the DEP Eelgrass Mapping Program as part of the MEP effort in 1995 and 2001. 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-5) and 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 Rock Harbor system by the MassDEP Eelgrass Mapping Program (C.Costello), as the Rock Harbor system is a tidal creek system and generally not supportive of eelgrass and the lower basin of the Harbor is an organic rich, depositional basin formed by dredging the lower portion of the tidal creek. In addition, the 1951 analysis could not be performed due to the lack of adequate aerial photos. However, there is no evidence of eelgrass previously colonizing this system. The MEP Technical Team did confirm the lack of eelgrass in the tidal creeks to the Rock Harbor system while undertaking field surveys as part of the benthic regeneration and infauna studies and during the deployment and recovery of the instrument moorings. Part of this effort included SMAST research divers conducting visual surveys throughout the harbor basin for the presence of eelgrass (and macroalgae).

In contrast, eelgrass is present offshore of the inlet to Rock Harbor as well as Little Namskaket Marsh and Namskaket Marsh. Based on the 2001 eelgrass survey conducted by the DEP Eelgrass Mapping Program offshore, there was evidence of a potential decline in the coverage of the offshore beds between 1995 and 2001 (Table VII-3). However, it is not possible at this time to determine if this represents an anthropogenic decline or natural variation at this site. Additional temporal sampling is planned to address this issue.

Based upon all available information, it appears that the Rock Harbor 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. While the lower basin of Rock Harbor has embayment characteristics (e.g. retains water at low tide, can support planktonic communities), it is an artificial basin created by dredging the lower reach of the main tidal creek. It is clearly a depositional basin that accumulates both watercolumn derived organic matter and marsh detritus. These characteristics are consistent with a basin not supportive of eelgrass habitat even at very low watershed nitrogen loads.

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

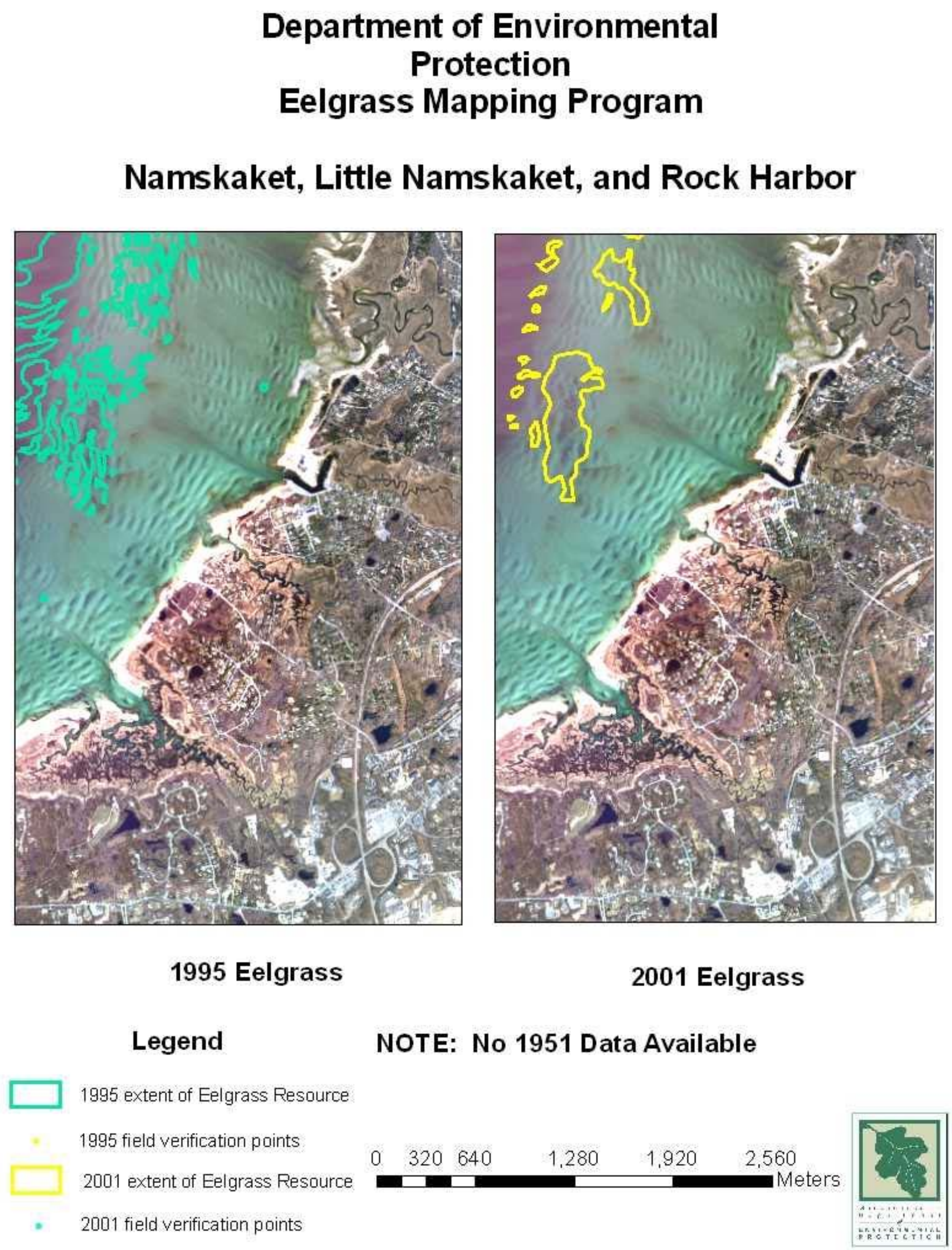


Figure VII-5. Eelgrass bed distribution immediately offshore the Rock Harbor, Little Namskaket Creek and Namskaket Creek systems. The 1995 coverage is depicted by the green outline and the 2001 coverage the yellow outline, which circumscribes the eelgrass beds as mapped by DEP Eelgrass Mapping Program. There is no evidence that these three systems have ever supported eelgrass habitat, as they are primarily tidal salt marshes.

Table VII-3. Changes in eelgrass coverage offshore from the Rock Harbor, Little Namskaket Creek and Namskaket Creek systems within the Town of Orleans over the past half century (C. Costello). The absence of eelgrass within these salt marsh systems is typical of tidal marshes throughout New England.

Temporal Change in Eelgrass Coverage				
Embayment	**1951 Acreage**	**1995 Acreage**	**2001 Acreage**	**% Loss 1995 - 2001**
Rock Harbor	NA[1]	0	0	NA[1]
Namskaket Creek	NA[1]	0	0	NA[1]
Little Namskaket Creek	NA[1]	0	0	NA[1]
Nearshore Cape Cod Bay	unavailable	**294.51**	**229.47**	**22%**
1 -- no historical evidence of eelgrass in this basin				

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 2 locations within the harbor basin in the which comprises the lower reach of the Rock Harbor Estuary (Figure VII-6). In some cases multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, although there are no eelgrass beds in the Rock Harbor system, this is almost entirely as a result of the structure of the system (Section VII.3). As such, to the extent that Rock Harbor 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).

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

The Infauna Study indicated that the sub-embayment basin of Rock Harbor is presently supporting a significantly impaired infaunal habitat. This basin, which was created by deepening the lower reach of the central tidal creek of the Rock Harbor salt marsh, is clearly depositional and receiving organic matter inputs from the watercolumn and up gradient marshes (marsh detritus was evident in the sediment samples). As a result, the system shows periodic oxygen depletion to <3 mg L^{-1} (Section VII.2) which is stressful to embayment infaunal animals. Sub-basins to Chesapeake Bay were determined to have stressful conditions if oxygen levels decline to 3.8 mg L^{-1}. Consistent with the level of oxygen depletion, tidally averaged total nitrogen (TN) levels at the mooring location were high for a coastal sub-embayment, 0.69 mg N L^{-1}. Other similarly configured sub-embayments on Cape Cod with similar TN levels also show significant oxygen depletion and impaired infaunal habitat quality (e.g. upper Bournes Pond or Great Pond, Falmouth). However, It should be noted that this level of TN is typical of upper tidal salt marsh creeks (e.g. Cockle Cove Creek, Chatham), and does not indicate impairment in that type of environment.

Infauna communities within the harbor basin of the Rock Harbor System were indicative of an environment significantly impaired by organic enrichment, consistent with the observed levels of oxygen depletion and watercolumn TN. The communities were low to moderate in numbers of individuals (64-121) and low in numbers of species (2-6), even for salt marsh dominated sub-embayments. These values do not compare well with even moderately impaired systems. For example, the infaunal community of the salt marsh pond of Mill Creek (Lewis Bay System) has 14 species, while the basin of the salt marsh system of Little River (Dartmouth) averaged 16 species over 5 sites distributed throughout the marsh basin and creeks. More significantly, the communities within the Rock Harbor basin were dominated by organic enrichment indicator species, like *Capitella capitata*, and *Streblospio benedicti*. While the configuration of the harbor basin and its location at the mouth of an extensive tidal salt marsh are important factors in the level of organic enrichment and habitat quality observed, nitrogen loads from the contributing watershed have likely exacerbated the stress. Therefore, recovery of the infaunal habitat within this basin and the role of the watershed N load were further investigated as part of the nitrogen thresholds analysis for this system. It should be noted that as there is no evidence that this system has ever supported eelgrass, its habitat quality is defined by its infaunal habitats.

Table VII-4. Benthic infaunal community data for the Rock Harbor embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.018 m2). Stations refer to map in Figure VII-6, (N) is the number of samples per site.

Location	Sta ID (N)	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Rock Harbor						
Lower Basin	Sta. 2/3 (2)	2	64	1	0.50	1.00
	Sta. 4 (2)	6	122	5	2.00	0.83

Figure VII-6. Aerial photograph of the Rock Harbor system showing location of benthic infaunal sampling stations (blue symbol).

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

Rock Harbor is an estuary dominated by salt marsh resources in its upper reach and embayment resources in its lower reach. The lower "harbor" reach consists of a tidal harbor basin maintained through dredging by the Towns of Eastham and Orleans. The present harbor basin is highly altered from its original state as the main tidal creek to the Rock Harbor marshes. This lower reach, "Rock Harbor", was a natural widening near the tidal inlet and has been used as a harbor since the colonial period. It became the focal point for major regional shipping activities in the early 1800's, with the construction of a town pier in 1814. In the 1900's the harbor continued to be altered for navigation and dockage. At present, bulkheads, piers and boat slips line most of the harbor reach. The inlet has hard structures on either shore, and the basin is periodically dredged for navigation. While the types of vessels have changed over the past century from quahoggers to recreational vessels, the harbor remains an important asset to the Towns of Orleans and Eastham. The purpose of this nitrogen threshold analysis is to support the resources of this marine basin and the significant habitats within the upper salt marshes. This area has been designated as an Area of Critical Environmental Concern and Outstanding Resource Water by the State of Massachusetts [Massachusetts General Law, chapter 21a, sections 2(7) and 40(e)], for planning purposes.

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 by the MEP to support threshold development for the Rock Harbor System 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 Orleans' Water Quality Monitoring Program. It should be noted that assessment of habitat quality must necessarily consider the natural function and tolerances of the specific estuarine ecosystem being evaluated.

At present, the Rock Harbor is showing high habitat quality throughout its upper salt marsh reach (above WMO-17) and significant habitat impairment in its lower "embayment" reach (e.g. harbor portion, WMO-17 to inlet). The upper reach appears to be a fully functional tidal salt marsh with deeply incised narrow creeks surrounded by significant areas of 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, along with patches of fringing brackish marsh vegetation (*Juncus, Phragmites*). 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. The upper tidal reach of the Rock Harbor system is comparable to the upper reaches of adjacent Little Namskaket and Namskaket Marshes in structure and present functioning. In Contrast, the lower "embayment reach, comprised primarily of the harbor basin, functions as a small open water cove or harbor. This basin is depositional by structure, collecting both algal and salt marsh organic matter with accumulation of anoxic organic-rich fine sediments (sulfidic); it is highly tidal, with sufficient light penetration to allow periodic development of benthic algal mats; and its tidal inlet is influenced by sand transport via nearshore coastal processes associated with adjacent Cape Cod Bay. These features in combination with the observed levels of summer oxygen depletion (to 2 mg L^{-1}), indicate a significantly impaired habitat (Table VIII-1). This assessment is supported by the impoverished infaunal animal community which is dominated by small opportunistic stress indicator species common to disturbed or organic matter enriched basins (*Capitella, Streblospio*).

Based upon all available information the present lack of eelgrass throughout the Rock Harbor System does not appear to be a response to watershed sourced nitrogen loading (e.g. changing watershed land-use). Instead, the absence of eelgrass habitat appears to result from the structure of the upper reach supportive of salt marsh and the lower reach being a maintained depositional basin. The absence of eelgrass in the salt marsh creeks 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 by the MEP Technical Team in a wide range of salt marsh dominated basins throughout southeastern Massachusetts, as well as for the adjacent Namskaket and Little Namskaket estuarine systems. The absence of eelgrass within the harbor basin is likely the result of its configuration, in that it is a "relatively deep" depositional basin. In addition, in the lower reach, harbor activities also likely have limited the potential for colonization of this system. Most important relative to MEP nitrogen thresholds analysis, it does not appear that eelgrass beds have been present within the Rock Harbor System at any time over the past century, as indicated by MassDEP Eelgrass Mapping Program analysis and MEP Technical Team historical analysis. Therefore, nitrogen threshold development for protection/restoration of this estuarine system will necessarily focus on restoration of the impaired infaunal habitat within the harbor (embayment reach) and protection of the high quality infaunal habitat within the upper salt marsh reach.

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

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

The Rock Harbor estuary is a composite of 2 different estuarine systems; an upper salt marsh reach and a lower embayment reach (the harbor). These systems have very different tolerances to nitrogen enrichment and are presently supporting habitats of very different quality, with high quality habitat in the salt marsh and significantly impaired habitat quality within the embayment (Table VIII-1). Since the embayment reach is clearly over its ability to assimilate its present nitrogen load without impairment, restoration will require nitrogen management. A correlative effect of nitrogen management will be that nitrogen loading to the upper salt marsh will also be reduced, although the upper salt marsh is well below its likely nitrogen loading threshold level to maintain its high quality habitats. Therefore the nitrogen threshold for the Rock Harbor System focusing on the habitat quality of the embayment reach will necessarily be protective of the upper salt marsh reach.

Table VIII-1. Summary of Nutrient Related Habitat Health within the Rock Harbor Estuary on the Cape Cod Bay shore of the Towns of Orleans and Eastham, MA, based upon assessment data presented in Chapter VII. The upper reach of this estuary is a typical New England salt marsh with a large central tidal creek, while the lower reach is an artificial "embayment", created from the lower portion of the central creek as a harbor.

Health Indicator	Rock Harbor Estuary	
	Upper Salt Marsh	**Lower Harbor Basin**
Dissolved Oxygen	--[1]	SI[2]
Chlorophyll	--[3]	H-MI[4]
Macroalgae	H[5]	SI[6]
Eelgrass	--[7]	--[8]
Infaunal Animals	--	SI-SD[9]
Overall:	**H[10]**	**SI**

1 -- oxygen mooring not placed in upper marsh creek as naturally organic matter and nutrient rich, oxygen depletions are typical of pristine salt marsh creeks.
2 – periodic oxygen depletions to <3 mg/L and frequently <4 mg/L.
3 -- tidal waters ebb nearly completely at low tide, chlorophyll levels reflect floodwaters.
4 – low to moderate chlorophyll a levels generally 2-8 ug/L, generally <5 ug/L
5 – *Ulva* and drift algae very sparse to absence
6 -- thick benthic algal mat throughout basin
7 – no evidence that this estuarine reach is supportive of eelgrass, as it is a salt marsh tidal creek, which drains at low tide..
8 – no evidence that this estuarine reach is supportive of eelgrass, this basin was created by dredging the main salt marsh tidal creek, and is structured as a depositional basin, with accumulations of marsh detritus.
9 – low numbers of individuals and species, dominated by stress indicator species (*Capitella*, *Streblospio*).
10 -- upper estuary salt marsh very similar in structure, communities and TN levels to high quality upper marsh reaches of adjacent Namskaket and Little Namskaket salt marsh systems

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

As a result of the present significant impairment of the infaunal habitat within the embayment reach of the Rock Harbor Estuary and given that there is no evidence that this system has supported eelgrass over the past century, the threshold development necessarily focuses on the embayment reach. The threshold for restoring and maintaining high quality infaunal habitat within the embayment reach of Rock Harbor is 0.500 mg TN L^{-1} (tidally averaged) at the sentinel station located at the head of the harbor (upper region of harbor basin, Town of Orleans Water Quality Monitoring Program station WMO-17). This determination is discussed below and the watershed nitrogen loads that will achieve this nitrogen threshold level are developed in the following Section VIII.3.

At present, the embayment reach of the Rock Harbor System has elevated total nitrogen levels (0.686 mg N L^{-1}, tidally averaged), with stressful levels of summer oxygen depletion (to 2 mg L^{-1}), sulfidic sediments and depleted infaunal communities dominated by stress indicator species. These observations strongly support the contention that this basin is significantly impaired through nitrogen enrichment. As this basin does not presently support high quality infaunal habitat, the nitrogen threshold analysis was based upon comparisons to a number of small embayments on Cape Cod. The 0.5 mg TN L^{-1} determined as the threshold compares well with:

- moderately impaired infaunal habitat at total nitrogen (TN) levels in the range of 0.535 mg N L^{-1}. within the Wareham River basin of the Wareham River Estuary;
- moderate impairment at tidally averaged TN levels of 0.526 mg N L^{-1} in Scudder Bay and at 0.543 mg TN L^{-1} in the middle reach of the Centerville River;
- Bournes Pond (Falmouth) and Popponesset Bay (Mashpee/Barnstable) where levels $\leq$0.5 mg N L^{-1} were found to be supportive of high quality infaunal habitat;
- Eel Pond in Bourne, where high quality infaunal habitat had a threshold level, 0.45 mg N L^{-1}, due to it being a "deep" depositional terminal basin (as opposed to the flow-through of the Rock Harbor basin;
- high quality infaunal animal habitat areas within the Wareham River System were at TN levels of 0.444-0.463 mg TN L^{-1}.

Based upon these observations, the MEP Technical Team concluded that an upper limit of 0.50 mg N L^{-1} tidally averaged TN would support healthy infaunal habitat in the lower embayment reach of the Rock Harbor System. Equally important, lowering nitrogen levels from the present 0.686 mg N L^{-1} to the threshold 0.500 mg N L^{-1} will lower nitrogen levels within the upper salt marsh (e.g. WMO-18, from 0.829 to 0.615 mg N L^{-1}), protective of those habitats. Therefore, it appears that achieving the nitrogen target at the sentinel location is restorative of infaunal habitat throughout the lower basin and protective of habitats within the upper salt marsh reach. 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 reduction required for restoration of eelgrass and infaunal habitats in the Rock Harbor system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for Rock Harbor. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment. A comparison between present septic and total watershed loading and the loadings for the two modeled threshold scenarios is provided in Tables VIII-2 and VIII-3.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required nearly 70% removal of septic load (associated with direct groundwater discharge to the embayment) for the entire system. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. For Example, removal of 79% of the septic load from the Rock Harbor watershed results in a 67% reduction in total watershed nitrogen load. No load reduction was necessary for the Cedar Pond watershed. 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, as discussed in Section VI.2.6.1.

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel station is shown in Table VIII-4. To achieve the threshold nitrogen concentrations at the sentinel station, reductions in TN concentrations of typically greater than 20% is required in the system, between the main harbor basin and the marsh.

The basis for the watershed nitrogen removal strategy utilized to achieve the embayment thresholds may have merit, since this example nitrogen remediation effort is focused on watersheds where groundwater is flowing directly into the estuary. For nutrient loads entering the systems through surface flow, natural attenuation in freshwater bodies (i.e., streams and ponds) can help by significantly reduce the load that finally reaches the estuary. Presently, this attenuation is occurring in Cedar Pond due to natural ecosystem processes and the extent of attenuation being determined by the mass of nitrogen which discharges to these systems. Future nitrogen management should take advantage of natural nitrogen attenuation, where possible, to ensure the most cost-effective nitrogen reduction strategies. However, "planned" use of natural systems has to be done carefully and with the full analysis to ensure that degradation of these systems will not occur. One clear finding of the MEP has been the need for analysis of the potential associated with restored wetlands or ecologically engineered ponds/wetlands to enhance nitrogen attenuation. Attenuation by ponds in agricultural systems has also been found to work in some cranberry bog systems, as well. Cranberry bogs, other freshwater wetland resources, and freshwater ponds provide opportunities for enhancing natural attenuation of their nitrogen loads. Restoration or enhancement of wetlands and ponds associated with the lower ends of rivers and/or streams discharging to estuaries are seen as providing a dual service of lowering infrastructure costs associated with wastewater management and increasing aquatic resources associated within the watershed and upper estuarine reaches.

Although the above modeling results 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-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Rock Harbor system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Rock Harbor	6.787	1.442	-78.8%
Cedar Pond	0.910	0.910	0.0%
System Total	7.698	2.352	-69.4%

Table VIII-3. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer, and the WWTF) used for modeling of present and threshold loading scenarios of the West Falmouth Harbor 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
Rock Harbor	7.978	2.633	-67.0%
Cedar Pond	1.088	1.088	0.0%
System Total	9.066	3.720	-59.0%

Table VIII-4. Threshold sub-embayment loads used for total nitrogen modeling of the Rock Harbor 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)
Rock Harbor	2.633	0.079	1.207
Cedar Pond	1.088	-	-
System Total	3.720	0.079	1.207

Table VIII-5. Comparison of model average total N concentrations from present loading and the threshold scenario, with percent change, for the Rock Harbor System. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). The threshold station is shown in bold print.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Rock Harbor Mouth	WMO-15	0.447	0.398	-11.1%
Rock Harbor Basin	WMO-16	0.574	0.454	-20.9%
Rock Harbor Marsh	**WMO-17**	**0.686**	**0.500**	**-27.1%**
Rock Harbor Creek	WMO-18	0.829	0.615	-25.8%

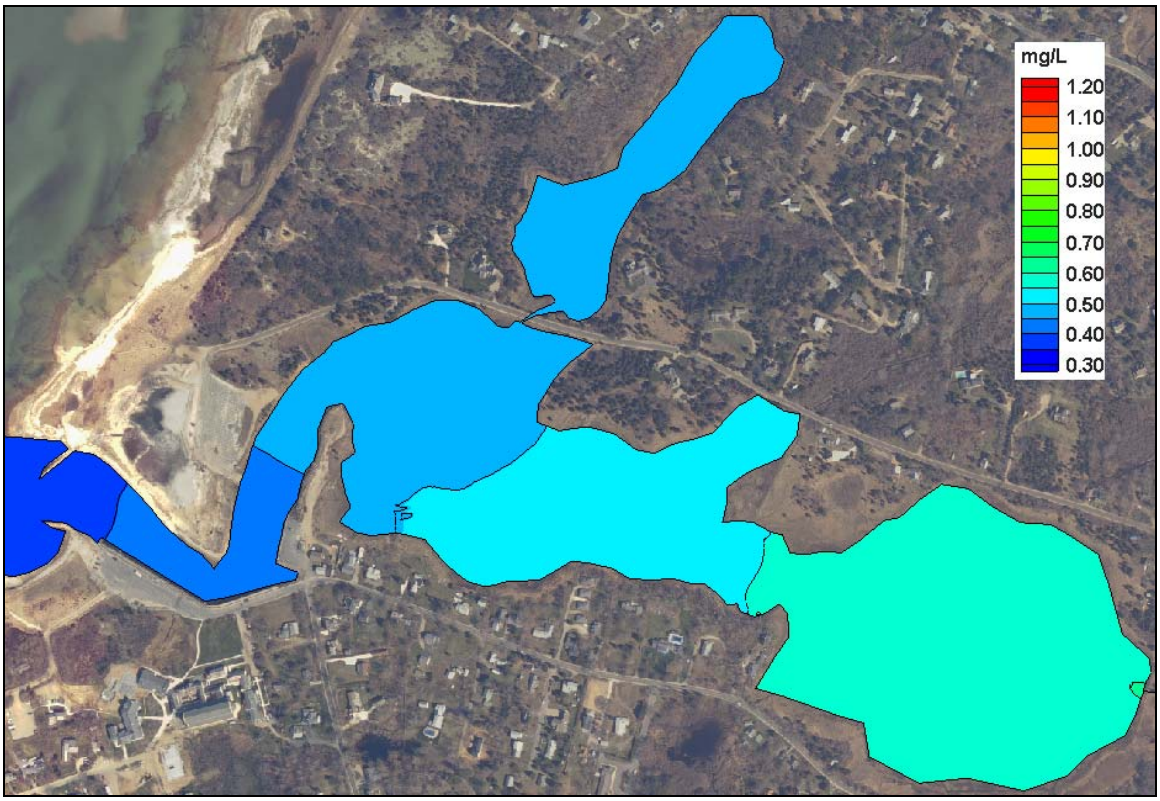


Figure VIII-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Rock Harbor system, for threshold conditions (0.50 mg/L at monitoring station WMO-17. The plot shows the entire area of Rock Harbor salt marsh that is flooded at high tide.

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