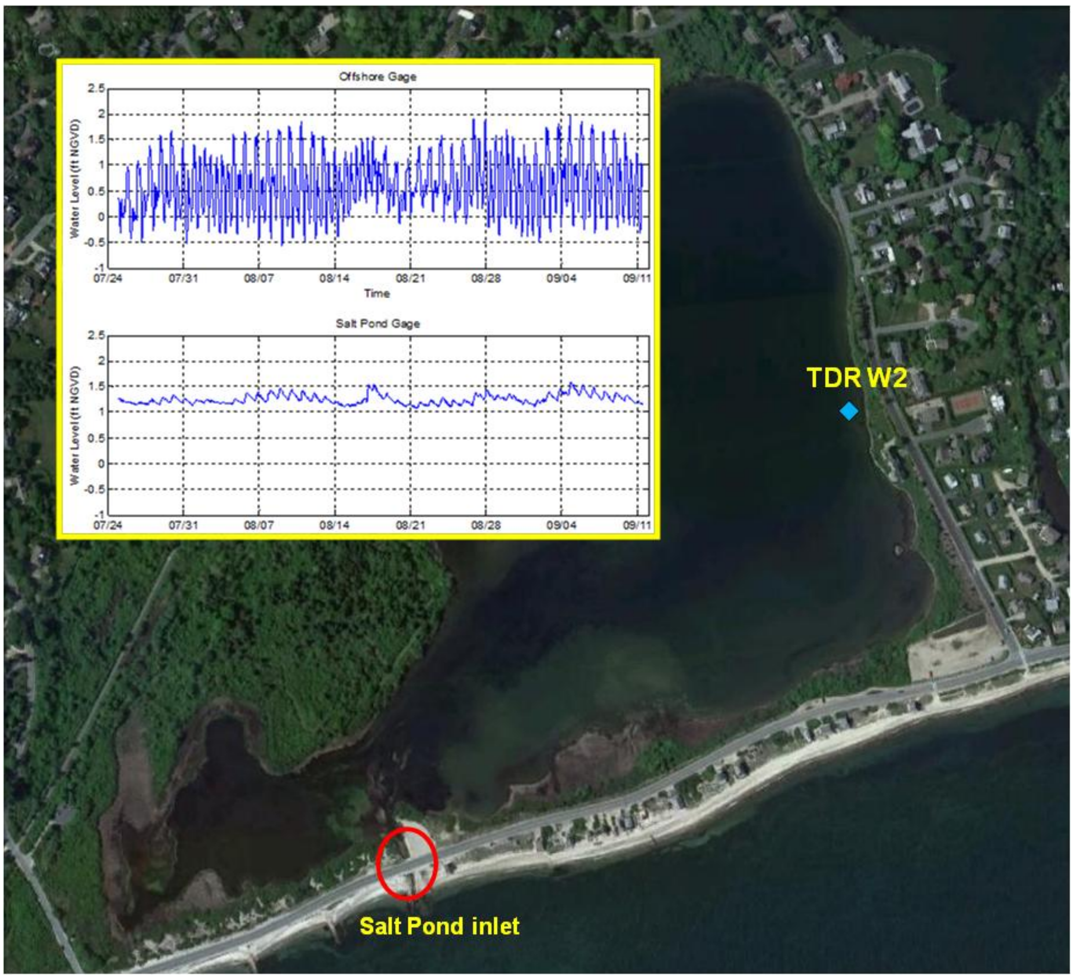
Massachusetts Estuaries Project

Linked Watershed-Embayment Approach to Determine Critical Nitrogen Loading Thresholds for the Salt Pond Embayment System Town of Falmouth, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of Environmental Protection

FINAL REPORT – May 2014

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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 Salt Pond embayment system, a coastal embayment within the Town of Falmouth, Massachusetts. Analyses of the Salt Pond embayment system was performed to assist the Town of Falmouth with up-coming nitrogen management decisions associated with the current and future wastewater planning efforts of the Town, as well as wetland restoration, anadromous fish runs, shell fishery, open-space, and harbor maintenance programs. As part of the MEP approach, habitat assessment was conducted on the embayment based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. Nitrogen loading thresholds for use as goals for watershed nitrogen management are the major product of the MEP effort. In this way, the MEP offers a science-based management approach to support the Town of Falmouth 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 Salt Pond embayment, (2) identification of all nitrogen sources (and their respective N loads) to embayment waters, (3) nitrogen threshold levels for maintaining Massachusetts Water Quality Standards within embayment waters, (4) analysis of watershed nitrogen loading reduction to achieve the N threshold concentrations in embayment waters, and (5) a functional calibrated and validated Linked Watershed-Embayment modeling tool that can be readily used for evaluation of nitrogen management alternatives (to be developed by the Town) for the restoration of the Salt Pond 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 Salt Pond embayment system within the Town of Falmouth is at risk of eutrophication (over enrichment) from enhanced nitrogen loads entering through groundwater from the increasingly developed watershed to this coastal system. Eutrophication is a process that occurs naturally and gradually over a period of tens or hundreds of years. However, human-related (anthropogenic) sources of nitrogen may be introduced into ecosystems at an accelerated rate that cannot be easily absorbed, resulting in a phenomenon known as cultural eutrophication. In both marine and freshwater systems, cultural eutrophication results in degraded water quality, adverse impacts to ecosystems, and limits on the use of water resources.

The Town of Falmouth has recognized the severity of the problem of eutrophication and the need for watershed nutrient management and is currently developing a Comprehensive Wastewater Management Plan which the Town plans to implement upon its completion. The Town of Falmouth has been working with the Town of Mashpee that has also completed and implemented wastewater planning in other nearby regions not associated with the Salt Pond system, specifically the Waquoit Bay embayment system. In this manner, this analysis of the Salt Pond system is yielding results which can be utilized by the Town of Falmouth along with MEP results developed for the other estuaries of the town (specifically, Rands Harbor, Fiddlers Cove, Wild Harbor, West Falmouth Harbor, Falmouth Inner Harbor, Little Pond, Quissett Harbor, Oyster Pond, Great Pond, Green Pond, Bournes Pond, Eel Pond/Childs River and Waquoit Bay) in order to give the Town of Falmouth the necessary results to plan out and implement a unified town-wide approach to nutrient management. The Town of Falmouth with associated working groups has recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Towns. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns' nutrient management groups to predict the impacts on water quality from a variety of proposed management scenarios.

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

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

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

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

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

The greatest assets of the Linked Model Approach 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 Salt Pond embayment system by using site-specific data collected by the MEP and water quality data from the volunteer efforts of scientists and graduate researchers within the Coastal Systems Program-SMAST. CSP staff and students undertook the collection of the necessary minimum three years baseline data in order to support entry of Salt Pond into the MEP. These "research volunteers" at CSP-SMAST initiated data collection in summer 2006 and created a 7 year baseline of summer water quality for the pond (2006-2012). Evaluation of upland nitrogen loading was conducted by the MEP, data were provided by the Town of Falmouth Planning Department, and watershed boundaries delineated by USGS. These land-use data were used to determine watershed nitrogen loads within the Salt Pond embayment system (current and build-out loads are summarized in Table IV-3). Water quality within an 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 Salt Pond embayment system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates. Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic model 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. Boundary nutrient concentrations in Vineyard Sound source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Salt Pond embayment system were 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 embayment.

MEP Nitrogen Thresholds Analysis: The threshold nitrogen level for an embayment represents the average water column concentration of nitrogen that will support the habitat quality being sought. The water column nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition).

The water column nitrogen concentration is modified by the extent of sediment regeneration. Threshold nitrogen levels for the embayment systems in this study were developed to restore or maintain SA waters or high habitat quality. High habitat quality was defined as supportive of eelgrass and infaunal communities. Dissolved oxygen and chlorophyll-*a* were also considered in the assessment.

The nitrogen thresholds developed in Section VIII-2 were used to determine the amount of total nitrogen mass loading reduction required for restoration of infaunal habitats (no documented historical eelgrass but for one observation of very limited eelgrass by the inlet in 2007, see Section VII for detail) in the Salt Pond 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 station chosen for the Salt Pond system. It is important to note that load reductions can be produced by reduction of any or all sources, enhancing flushing of the system or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented in Section VIII represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen reduction guidelines for nitrogen management of the Salt Pond embayment system in the Town of Falmouth. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. For Illustrative purposes, 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 the embayment system. The concept was that since nitrogen loads associated with wastewater generally represent 76% of the controllable watershed load to the Salt Pond 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 the system.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout the Salt Pond embayment system based upon available water quality monitoring data, time-series water column oxygen measurements of dissolved oxygen and chlorophyll, and benthic community structure (changes in eelgrass distribution could not be used as a metric due to lack of historical eelgrass presence). Salt Pond is currently functioning as a typical coastal embayment with restricted tidal exchange with the waters of Vineyard Sound. Each of type of functional component to an estuary (salt marsh basin, embayment, tidal river, deep basin {sometimes drown kettles}, shallow basin, etc.) has a different natural sensitivity to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of the specific basin and its ability to support eelgrass beds and infaunal communities. At present, the Salt Pond Estuary is beyond its ability to assimilate nitrogen without further impairment. The system is showing a high level of nitrogen enrichment, with no eelgrass habitat and moderate to significantly impaired benthic animal habitats (depending on location in the pond), regions of periodic hypoxia and phytoplankton blooms and a stratified deep basin with prolonged anoxia (Table VIII-1), these findings indicate that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system.

The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Chapter VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment and restriction of tidal flows. The spatial pattern indicated that the magnitude of oxygen depletion, enhancement of chlorophyll-*a* levels and total nitrogen concentrations were affected by the watercolumn stratification stemming from the basin geomorphology and reduced tidal action such that waters and sediments below 3 meters depth are subjected to prolonged anoxia and potential infaunal habitat is only in the shallow margins of the main basin and in the region of the tidal channel.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate moderate to high nutrient enriched waters within the margins of the main basin and tidal channel region of Salt Pond, respectively. The oxygen data is consistent with organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll-*a* and in the tidal channel, which also has patchy accumulations of macroalgae. The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of the Salt Pond Estuary.

Salt Pond is functionally a basin with very limited fringing wetland habitat along its shoreline to the main basin. The sediments are currently soft muds rich in organic matter in the center deep portion of the pond, which has had periods of prolonged anoxia since at least the 1970's. The primary factor in the oxygen depletion of the deep waters is the strong salinity stratification of the deep basin, which restricts ventilation with the atmosphere. It is the shallower (<3m) margins of the basin that support some sandy areas and compacted muds with oxidized surface layers. All of the available information related to eelgrass within the Salt Pond Estuary, including the 2007 survey, indicate that no eelgrass is present within Salt Pond and eelgrass beds have not been observed historically, although small sparse patches of *Ruppia* occur in some shoreline areas. The absence of eelgrass beds is expected in this system given the high chlorophyll-*a* (averages 14-15 ug L^{-1}) and periodic low dissolved oxygen levels and high water column nitrogen concentrations. Given the absence of eelgrass at present and the lack of evidence of prior eelgrass habitat within this system, management should focus on benthic animal habitat, primarily within the marginal areas.

Overall, the infauna survey indicated that the shallow margin (<3 m) around the deep kettle "hole" is supportive of the moderate quality infaunal habitat within Salt Pond, showing areas that are moderately and significantly impaired. The sediments within the deep basin (>3 m) are overlain by anoxic bottom water and are devoid of benthic animals. Intermediate to these 2 regions, the tidal channel is depositional, with significant oxygen declines and significantly impaired benthic habitat.

Classification of habitat quality necessarily included the structure of the estuarine basin, specifically that it is fully representative of a tidal embayment, as opposed to a tidal river or salt marsh basin and if a basin is structurally impaired or impaired by nitrogen enrichment.. Integration of all of the metrics clearly indicates that the shallow areas of Salt Pond are generally supporting benthic animal habitat that is moderately or significantly impaired. The proximate cause of impairment is organic matter enrichment and oxygen depletion, stemming ultimately from nitrogen enrichment. Total nitrogen levels within the estuary at present are >0.90 mg TN L^{-1}, a level generally found associated with a significant level of impairment of benthic animal habitat in southeastern Massachusetts estuaries. The lack of historical eelgrass beds in Salt Pond and the present impairment to benthic animal habitat from nitrogen enrichment makes restoration of infauna habitat resource the primary focus for nitrogen management.

3. Conclusions of the 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 integration of the watershed nitrogen load, the nitrogen concentration in the inflowing tidal waters (boundary condition) and dilution and flushing via tidal flows. The water column nitrogen concentration is modified by the extent of sediment regeneration and by direct atmospheric deposition.

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

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Town of Falmouth Salt Pond embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 76% of the controllable watershed nitrogen load to the embayment was from wastewater (septic and WWTF).

A major finding of the MEP clearly indicates that a single total nitrogen threshold can not be applied to Massachusetts' estuaries, based upon the results of the Fiddlers Cove, Rands Harbor, Wild Harbor, Little Pond, Falmouth Inner Harbor, Great, Green and Bournes Pond Systems, Popponesset Bay System, and the nearby Eel Pond and Hamblin / Jehu Pond / Quashnet River analysis in eastern Waquoit Bay, among many other systems analyzed by the MEP. This is almost certainly going to be true for the other embayments within the MEP area, as well, inclusive of Salt Pond.

The threshold nitrogen levels for the Salt Pond embayment system in Falmouth were determined as follows:

Salt Pond Threshold Nitrogen Concentrations

- Following the MEP protocol, the restoration target for the Salt Pond Embayment system should reflect both recent pre-degradation habitat quality and be reasonably achievable. The approach for determining nitrogen loading rates, which will maintain acceptable habitat quality throughout and embayment system, is to identify a sentinel location within the embayment or sub-embayment (as necessary) and second, to determine the nitrogen concentration within the water column which will restore that location to the desired habitat quality. Within the Salt Pond Estuary the most appropriate sentinel "station" was to use the average of the 3 long-term monitoring stations (<3 m) in Figure VI-1. This average approach has been used in other open single basin estuaries throughout the MEP region. The average was selected because of the heterogeneity in the benthic animal habitat in this stratified basin and the need to meet acceptable quality conditions throughout the basin.

- Following the MEP protocol, since eelgrass has not been documented in Salt Pond, restoration of infaunal habitat is the restoration goal. Infaunal animal habitat is a critical resource to the Salt Pond Estuary and estuaries in general. Since there are no unimpaired infaunal animal habitat areas remaining in the Salt Pond system, comparisons to the soft bottom basins of other nearby estuarine systems were relied upon for setting the nitrogen threshold for healthy infaunal habitat at a nitrogen level of TN <0.5 mg TN L^{-1}. This level was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 were found to be supportive of high infaunal habitat quality in this system. Similarly, in the Three Bays System, healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay), with impairment in areas where nitrogen levels of TN >0.5 mg TN L^{-1} (North Bay), and severe degradation at nitrogen levels of TN >0.6 mg TN L^{-1}. Present TN levels within the Salt Pond mixed layer during summer are ~0.90 mg TN L^{-1}, consistent with the observed lack of eelgrass beds and impaired benthic animal habitat.

It is important to note that the analysis of future nitrogen loading to the Salt Pond 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. 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 Salt Pond system is that given the relatively low watershed nitrogen load to Salt Pond, it will be difficult to lower TN levels by ~0.4 mg L^{-1} to meet the threshold. The nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations were not attainable even with 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the systems watershed. The limited circulation within the system prevents the threshold goals from being achieved. This is consistent with the MEP measurements of significantly restricted tidal flows between Salt Pond and Vineyard Sound. This has been found in other estuaries with similar restrictions (e.g. Rushy Marsh Pond, Farm Pond). In such cases a reduction of the tidal restriction is needed to lower the level of nitrogen enrichment and restore the impaired habitats. This will likely be the case for Salt Pond, as well.

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

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
SYSTEMS										
Salt Pond	0.241	1.263	3.488	--	4.751	0.789	1.439	6.979	0.35-1.41	--
System Total	**0.241**	**1.263**	**3.488**	**--**	**4.751**	**0.789**	**1.439**	**6.979**	**0.35-1.41**	**0.50[8]**

[1] assumes entire watershed is forested (i.e., no anthropogenic sources)
[2] composed of non-wastewater loads, e.g. fertilizer and runoff and natural surfaces and atmospheric deposition to lakes
[3] existing wastewater treatment facility discharges to groundwater
[4] composed of combined natural background, fertilizer, runoff, and septic system loadings
[5] atmospheric deposition to embayment surface only
[6] composed of natural background, fertilizer, runoff, septic system atmospheric deposition and benthic flux loadings
[7] average of 2006 – 2012 data, ranges show the upper to lower regions (highest-lowest) of an sub-embayment. individual yearly means and standard deviations in Table VI-1.
[8] Threshold for sentinel site located in Salt Pond was determined to be the average of the 3 long-term monitoring stations.

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for Salt Pond systems, Town of Falmouth, Massachusetts.

Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed reductions needed to achieve threshold load levels
SYSTEMS						
Salt Pond	4.751	--	0.789	--	--	--
System Total	**4.751**	--	**0.789**	--	--	--

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

ACKNOWLEDGMENTS

The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the critical coastal resources of the Town of Falmouth and drove for the completion of the Linked Watershed-Embayment Modeling Approach to Determine Critical Nitrogen Loading Thresholds for the Salt Pond Embayment System. Without these stewards and their efforts, this project would not have been possible.

First and foremost we would like to recognize and applaud the commitment shown by the Town of Falmouth in carrying forward with the Massachusetts Estuaries Project as part of its watershed management planning and for their commitment to the restoration of all of the estuaries of the Town. Significant time and attention has been dedicated to this effort by Jerry Potamis and Amy Lowell, whose support has been instrumental to completion of these reports. Equally important has been the technical support provided by the Town Planner, Brian Currie. We also would like to recognize the Nutrient Management Committee and the CWMP review committee for the Town of Falmouth, in moving this MEP analysis forward to support estuarine management of all of Falmouth's estuaries, including Salt Pond. We would also like to acknowledge the field support provided to the MEP by the Town of Falmouth Marine Department who gave us unrestricted use of the municipal marine facility to complete critical field tasks. The MEP Technical Team would also like to acknowledge the Coastal Systems Program Staff and graduate students for making available their research data related to the nutrient related water quality of this system. Without this baseline water quality data the present analysis would not have been possible.

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

Support for this project was provided by the Town of Falmouth via the Community Preservation Committee and the MassDEP.

PROPER CITATION

Howes B., R.I. Samimy, T. Ruthven, E.M. Eichner, J. S. Ramsey, D. Schlezinger (2013). Massachusetts Estuaries Project Linked Watershed-Embayment Approach to Determine Critical Nitrogen Loading Thresholds for the Salt Pond Embayment System, Town of Falmouth, MA, Massachusetts Department of Environmental Protection. Boston, MA.

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I. INTRODUCTION

The Salt Pond embayment system is located within the Town of Falmouth, on Cape Cod Massachusetts. The system has a southern shore bounded by water from Vineyard Sound (Figure I-1). The open waters and watershed of this salt pond system are fully within the Town of Falmouth. The present configuration of the Salt Pond embayment results from tidal flooding of a coastal kettle pond composed of a main deep basin (~ 5m) which shoals towards the barrier beach and inlet. Salt Pond is separated from the ocean by a barrier beach and does not have any small creeks discharging into the pond. Tidal flooding of the kettle pond is through a single inlet and is possible as a result of rising sea level since the end of the last glacial period. It appears that Salt Pond has had managed inlet channels since before 1880, and may have originally been a fresh pond (to slightly brackish depending on the amount of overwash from storms in any given year) artificially opened by local residents.

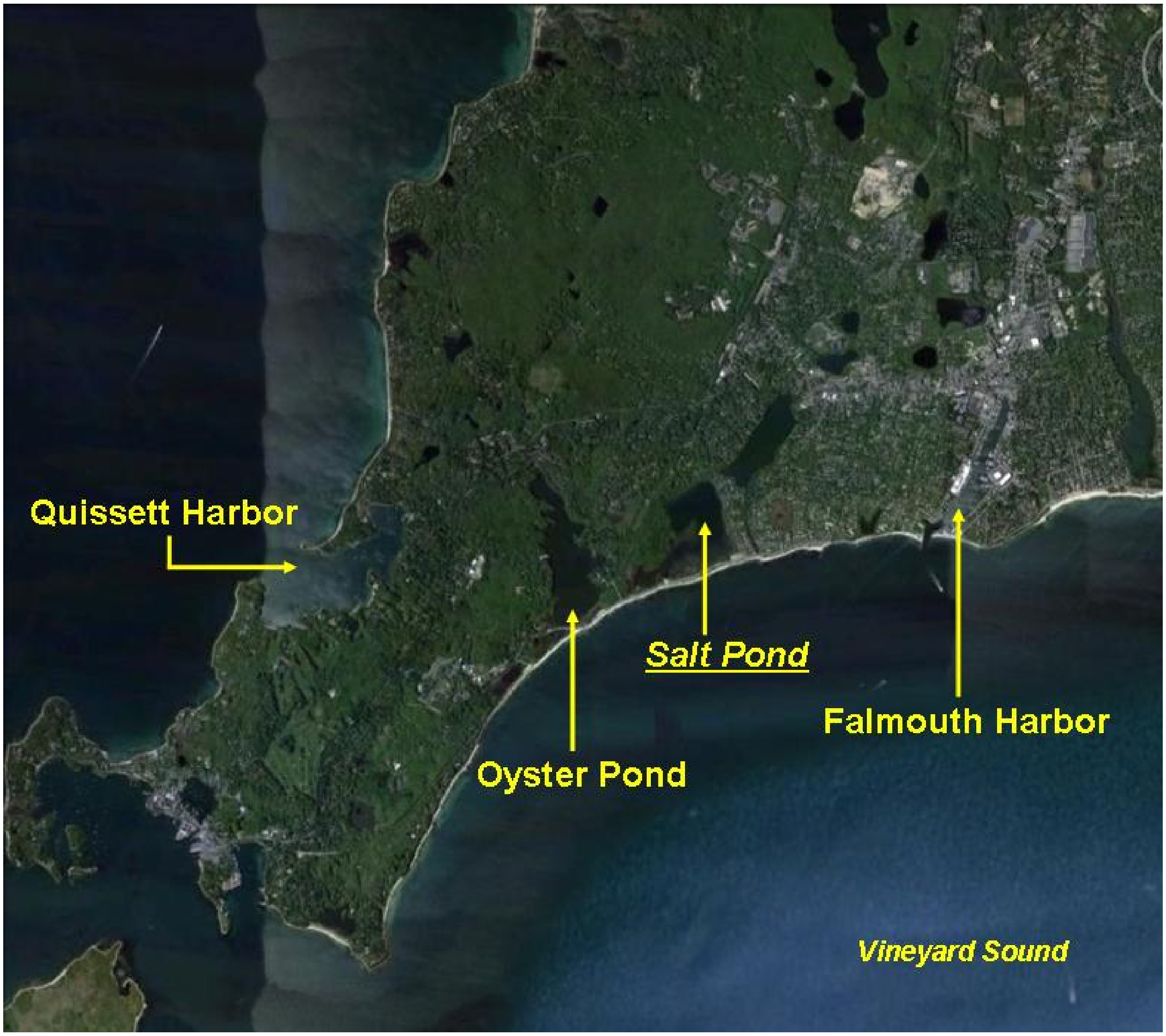

Figure I-1. Study region proximal to the Salt Pond embayment system for the Massachusetts Estuaries Project nutrient analysis.

As is typical with other Falmouth embayments (Great, Green, Bournes and Oyster Pond) Salt Pond is separated from Vineyard Sound by a barrier beach. The Salt Pond embayment exchanges tidal water directly with Vineyard Sound. Tidal flow is through a small tidal channel that passes under Surf Drive through a culvert. The beach and the opening to Salt Pond are very dynamic geomorphic features due to the influence of littoral transport processes. As a result of periodic sedimentation, generally associated with storms, the tidal channel and culvert must be maintained to sustain the small areas of salt marsh located landward of the barrier beach as well as the brackish nature of Salt Pond. Occlusion of the tidal channel has periodically resulted in reduced circulation and tidal flooding of the pond with associated decreases in water quality. The need for periodic maintenance of tidal inlets to sustain tidal flows is typical of the flood dominated embayments on Cape Cod. For example, Bournes Pond (Falmouth) became very restricted and finally completely isolated from Vineyard Sound waters in the late 1970's/early 1980's and was re-opened with a fixed inlet in mid 1980's. Similarly, protection of the natural resources of Salt Pond requires maintenance of the tidal flows. The tidal flushing of Salt Pond and effects on pond salinities and nutrient concentrations will be discussed further in Sections V and VI.

Similar to the Great, Green, Bournes, Little and Oyster Pond embayment systems, Salt Pond is a shallow coastal salt pond. Among these, Salt Pond, Oyster Pond and Perch Pond (part of Great Pond System) and Quissett Harbor to the north, are kettle basins, which tend to be deeper (5 m vs. 2 m) and more bowl shaped than the eroded drowned river valley basins of the other estuaries. All are located within a glacial outwash plain, the Mashpee Pitted Plain, consisting of material deposited after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~18,000 years ago. The outwash material is highly permeable and varies in composition from well-sorted medium sands to coarse pebble sands and gravels (Millham and Howes, 1994). As such, direct rainwater run-off to the adjacent estuaries is generally small with most freshwater inflow via groundwater discharge or in some cases groundwater fed surface water flows (e.g. stream to the head of Little Pond, Coonamessett River to Great Pond, Backus River to Green Pond, Mosquito Creek to Oyster Pond etc.) (Figure I-2). Salt Pond currently functions as a brackish great salt pond receiving freshwater primarily via groundwater discharge and with restricted tidal inflows of saline Vineyard Sound waters. The salinity characteristics of Salt Pond varies with the volume of freshwater inflow combined with the volume of tidal exchange with Vineyard Sound. Based on summer water quality data collected by the Coastal Systems Program, annual salinity averages range between a minimum of 15.4 ppt and a maximum of 27.7 ppt with an average salinity in the pond across three stations of 21 ppt.

In addition to shifting Salt Pond from a fresh to a salt water basin, a secondary ecologically significant consequence of the opening of Salt Pond has been a semi-permanent salt stratification of surface and bottom waters. The colder salty waters entering from Vineyard Sound, due to their greater density, tend to "fall" into the deep (~5 m) main basin (27 ppt), with more brackish waters (20-21 ppt) formed by mixing of salt water with entering freshwater comprising the surface mixed layer. The resulting salinity stratification, observed over much of the summer period, reduces exchange between surface and bottom waters and results in hypoxic to anoxic conditions and high nitrogen levels in bottom waters. This stratification is, in part, a natural phenomenon resulting from the basin geomorphology and low rate of tidal exchange. However, it does restrict the available habitat for infaunal animals and other marine functional groups to waters and sediments associated with the aerobic mixed surface layer. Adjacent, Oyster Pond, has similar geomorphology and has had similar salinity and oxygen stratification (Emery, 1969, revised 1997), until management actions were undertaken.

Figure I-2. Salt Pond embayment system for the Massachusetts Estuaries Project assessment and threshold analysis. Tidal waters enter the salt pond through one inlet to Vineyard Sound. Freshwaters enter from the watershed primarily through direct groundwater discharge, precipitation and stormwater from roads. The limited fringing wetland area results primarily from the limited tide range (0.25 ft) due to the restricted inlet.

Salt Pond, along with the other salt pond embayments along the south coast of Falmouth, constitutes an important component of the Town's natural and cultural resources. In addition, the pond's location in a heavily residential area of Falmouth greatly increases the potential contaminant inputs associated with development. As there are no significant streams discharging to the estuary, groundwater is the dominant pathway for freshwater and nitrogen transport to Salt Pond.

As a stratified kettle basin with restricted tidal exchange, the Salt Pond Estuary is relatively sensitive to the effects of nutrient enrichment from watershed based sources. In general, the nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, Salt Pond as well as the Great, Green, Bournes, Little and Oyster Pond embayment systems along the southern shore of Falmouth are at risk of eutrophication from high nitrogen loads in the groundwater and runoff from their watersheds.

The primary ecological threat to Salt Pond resources is degradation resulting from nutrient enrichment, coupled with its present water column stratification. Loading of the critical eutrophying nutrient, nitrogen, to the embayment waters has been greatly increased over the past several decades as population has increased, with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other Falmouth estuaries, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Town of Falmouth has been among the fastest growing towns in the Commonwealth over the past three decades and does not have centralized wastewater treatment throughout the entire Town. As existing and probable increasing levels of nutrients impact Falmouth's coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

The Town of Falmouth (via the Planning Office) was one of the first communities in Massachusetts to become concerned over perceived degradation of its embayment systems due to nutrient overloading. In the mid-1980's the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. Nutrient limits were set for nitrogen in each of the Town's embayments. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts. To acquire baseline water quality data necessary for ecological management of Falmouth's coastal salt ponds and harbors, a citizen-based water quality monitoring program was initiated by the Town of Falmouth. Falmouth PondWatch was established to provide on-going nutrient related embayment health information in support of the By-law. The water quality monitoring program was based on a collaborative effort between scientists, citizens and representatives of the Town of Falmouth. As originally conceived, the monitoring program focused on data collection in three original ponds, Oyster Pond, Little Pond and Green Pond. By 1990, the scope of water quality data collection expanded to include two additional ponds, Great/Perch Pond and Bournes Pond. In 1992, the scope of data collection was once again expanded to include West Falmouth Harbor in order to evaluate the effects from a nutrient enriched wastewater plume generated by the Falmouth Wastewater Treatment Facility. Since 1997, technical aspects of the Falmouth PondWatch Program have been coordinated through the Coastal Systems Program at SMAST-UMassD. However, the PondWatch Program never expanded to include all the Falmouth embayments such as Salt Pond, Falmouth Inner Harbor or Waquoit Bay and Eel Pond. These specific systems are being monitored outside of PondWatch by other programs but still coordinated through the Coastal Systems Program at SMAST.

Nevertheless, the Falmouth PondWatch Program, as the water quality monitoring effort came to be known, continues to play an active role in the collection of baseline water quality data to this day, though it has evolved beyond its original mandate of providing basic environmental data relative to the Coastal Pond Overlay Bylaw (Nutrient Bylaw). The PondWatch Program brings together, as requested by Town boards, ecological information relative to specific water quality issues. Additionally, as remediation plans for various systems are implemented, the continued monitoring satisfies demands by State regulatory agencies and provides quantitative information to the Town relative to the efficacy of remediation efforts. Lastly, the PondWatch Program has grown into being a repository of environmental data on Falmouth's coastal ponds. The database includes basic water quality monitoring data in addition to special project data on watershed nutrient loading and watershed delineation, circulation characteristics of the ponds, wetland delineations and plant and animal distributions. It should also be noted that the Town of Falmouth Planning Office continues to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present evaluation is part of that continuing effort.

Faced with the lack of necessary water quality baseline data for Salt Pond to conduct the Massachusetts Estuaries Project's assessment, the scientists and graduate researchers within the Coastal Systems Program-SMAST undertook the collection of the necessary minimum three years baseline data needed for entry of Salt Pond into the MEP. These "research volunteers" at CSP-SMAST initiated data collection in summer 2006 and created a 7 year baseline of summer water quality for the Pond (2006-2012). The water quality data collected in Salt Pond was collected using the same protocols developed and implemented for the collection of water quality data by the PondWatch Program as well as the Coalition for Buzzards Bay BayWatcher Program monitoring the other estuaries in the Town of Falmouth. In this manner all water quality data collected from all the embayments in the Town of Falmouth would be consistent and cross comparable.

Unfortunately, monitoring has documented that most regions within the Town's coastal ponds, including Salt Pond, are currently showing water quality declines and are beyond the limits set by the By-law. Based on the wealth of information obtained over the many years of study of these coastal ponds, in addition to the nutrient analyses undertaken as a precursor to the Massachusetts Estuaries Project, the Salt Pond embayment system was included in the Massachusetts Estuaries Project to receive state-of-the-art analysis and modeling.

The common focus of the Falmouth Pond Watch Program effort as well as water quality monitoring undertaken by the Coalition for Buzzards Bay and voluntarily by the Coastal Systems Program has been to gather site-specific data on the current nitrogen related water quality throughout all of the coastal embayments of Falmouth and 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 water quality in Salt Pond. The Coastal Systems Program voluntary water quality monitoring in Salt Pond over the past 7 years developed a data set that elucidated the long-term trend of declining water quality and its relation to watershed based nutrient loading. The MEP effort builds upon the water quality monitoring program undertaken in Salt Pond and includes higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Salt Pond embayment system.

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 Falmouth. 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 SMAST/MassDEP Massachusetts Estuaries Project, the results stem directly from the efforts of a 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 Falmouth to develop and evaluate the most cost effective nitrogen management alternatives to restore these valuable coastal resource which are currently being 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. 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 declining 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 use. Similar to nutrients, bacterial contamination is related to changes in land-use as watershed become more developed. Regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to culture, economy, and tax base of Massachusetts's coastal communities.

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of Falmouth) 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 newest generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MA DEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts.

The Massachusetts Estuaries Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the DEP with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system will serve as the basis for the development of Total Maximum Daily Loads (TMDLs). Development of TMDLs is required pursuant to Section 303(d) of the Federal Clean Water Act. TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an implementation plan. That plan must identify, among other things, the required activities to achieve the allowable load to meet the allowable loading target, the time line for those activities to take place, and reasonable assurances that the actions will be taken.

The major Project goals are to:

- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of 70 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment's model "alive" to address future regulatory needs.

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

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

The Linked Model has been applied for watershed nitrogen management in ca. 55 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating the effectiveness of various watershed nitrogen management options on improving the associated estuarine habitats.

It is this ability to use a calibrated and validated Linked Watershed-Embayment Model for a specific estuary as a nitrogen management planning tool, which makes it invaluable for TMDL analysis and compliance. 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.

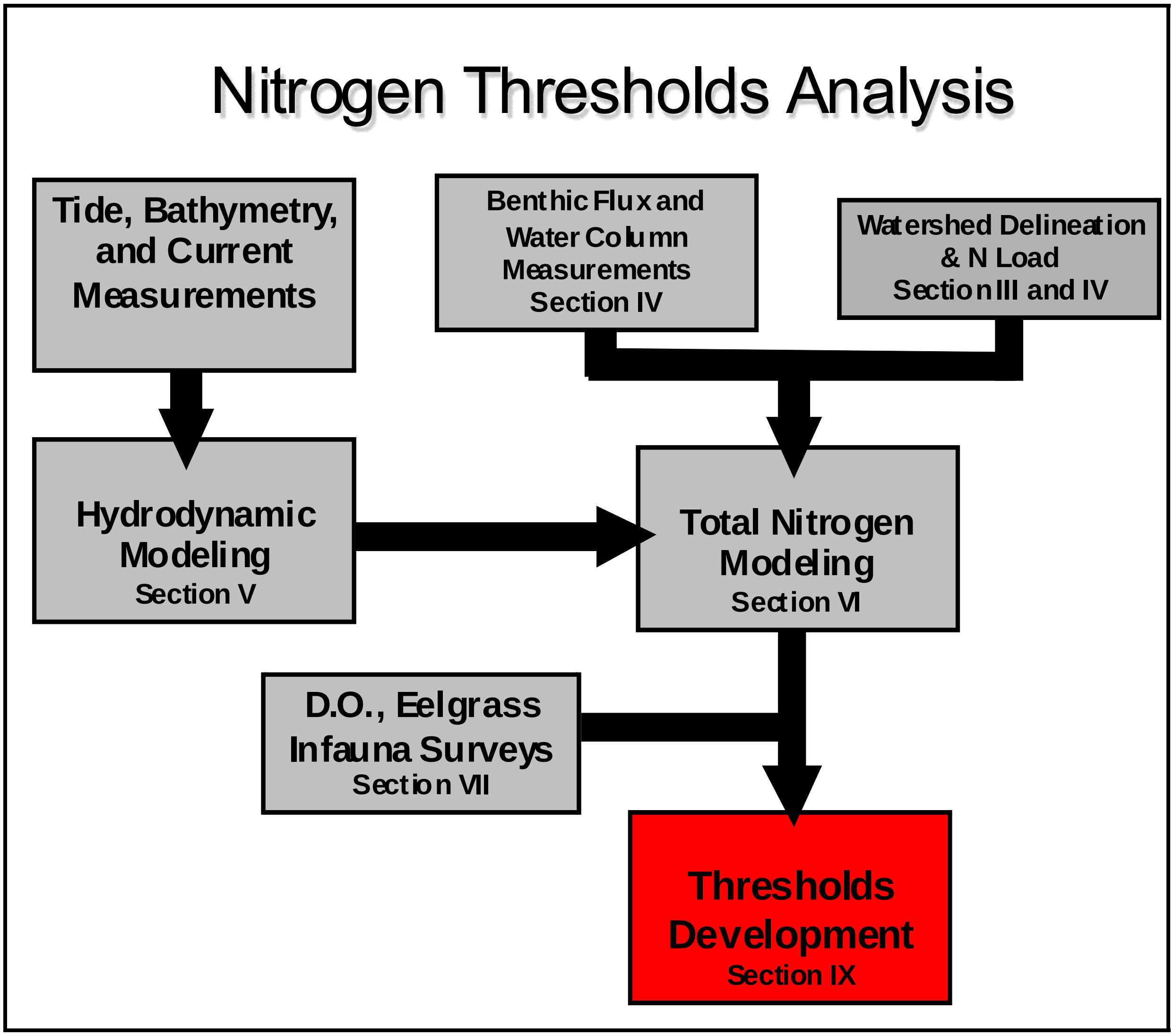

Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Section numbers refer to sections in this MEP report where the specified information is provided.

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-3). 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 coastal salt ponds of Falmouth including Salt Pond are generally oriented north-south, and open to varying degrees to Vineyard Sound via armored inlets. These inlets are affected significantly by longshore sand transport (west to east) as is the case with Salt Pond, where shoaling can impede hydrodynamic exchange within the inlet channel. As depicted in Figure I-3 Salt Pond has only a small shallow channel to Vineyard Sound which restricts tidal flows, in part due to a small flood tidal delta. The dimensions of the armored channel, elevation of the channel base, and distance from the inlet entrance to the pond, all contribute to the present reduction in flows into the pond.

Salt Pond, unlike other salt ponds in Falmouth (e.g. Great/Perch Pond, Green Pond, Bournes Pond and Little Pond) is a kettle pond that likely alternated between being a coastal salt pond (varying salinity from fresh to brackish depending on extent of overwash from storms) and an estuary (depending on connectivity to Vineyard Sound and extent of tidal influence) in the geologic past. Currently, it functions as a salt pond with restricted introduction of saltwater from Vineyard Sound. Salt Pond is a shallow mesotrophic (moderately nutrient impacted) to eutrophic (nutrient-rich) coastal pond on the southern coast of Falmouth. The shores of the pond can be characterized as slightly sloping to a height above the pond of less than 5 meters on both the northern and the southeastern side of the pond with gravel and few boulders as well as limited salt marsh habitat along the south and southwestern sides of the pond. The slightly sloping shores of the pond are generally backed by relatively flat outwash plain composed of sands and gravels deposited by glacial meltwater streams. The north end of Salt Pond is a kettle left by the melting of a residual mass of glacial ice within the Buzzards Bay moraine. The southward extension of the pond consists of a shallow basin that has shoaled over time with sediments transported into the pond with each incoming tide.

Although the salt pond embayment systems of Falmouth bounding Vineyard Sound exhibit slightly different hydrologic characteristics (river dominated versus tidally dominated), the tidal forcing for these systems is generated from Vineyard Sound. Vineyard Sound, adjacent the barrier beach separating the Salt Pond embayment system from the ocean, exhibits a moderate to low tide range, with a mean range of about 0.5 m offshore of the tidal inlet. Since the water elevation difference between Vineyard Sound and Salt Pond is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~1.5 m, Wellfleet Harbor is ~3 m). Tidal influence on Salt Pond by Vineyard Sound is further limited by restriction by the tidal inlet and associated sedimentation of the tidal channel, further diminishing the flow of Vineyard Sound water into the system. It appears that there are 2 potential pathways for improving Salt Pond habitat quality: (1) through enhancing tidal exchange and flushing and (2) managing nitrogen inputs. The current estuarine hydrodynamics and watershed nitrogen loads and subsequent impacts on pond resources are the focus of the present MEP analysis described in this report.

Since Salt Pond is comprised of a single main basin, nitrogen loading to this estuary was determined for the estuary as a whole as depicted in Figure I-2. Based upon the watershed and Pond being fully within the Town of Falmouth, it appears that nitrogen management for Pond restoration depends upon the efforts by Town Departments and citizens. As management alternatives are being developed and evaluated, it is important to note that Salt Pond is presently brackish and only very weakly tidal. Its basin configuration results in pulses of salt water entering the deep basin (~5m) during extreme high tides. As a result the deep basins stratify and become hypoxic-anoxic below 3-4 meters depth. This physical characteristic of the pond and resultant anoxic bottom waters and sediments in the affected areas, was also observed in adjacent Oyster Pond when it had a similar salinity regime. Prior to the installation of the inlet salinity control structure in Oyster Pond, it also supported low oxygen waters over pond regions covered by waters 3 meters in depth (compared to 4 meters in depth currently). Given its proximity to Vineyard Sound and the low elevation of the barrier beach, it is not possible to prevent the entry of salt water during hurricanes, when the barrier beach is over-washed. Therefore, it is not realistic to attempt to convert Salt Pond to fully freshwater pond. Similarly, it will be difficult to target the deep waters for “restoration”, as they naturally become anoxic if stratification exists and evidence suggests that they were periodically anoxic over the past ~100 years. However, these conditions appear to have persisted under highly restricted tidal flows and improvements may be possible if sufficient tidal flows can be implemented to render the water column isohaline, thus removing the dominant source of stratification. It should be noted that increasing the oxygenated water column from 3 to 4 meters will result in a significant increase in benthic infauna habitat, since the pond waters and sediments <4 meters depth account for most of the Pond resource (see Section VII).

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 watershed to the Salt Pond embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (Weiskel and Howes 1992). Since even Cape Cod “rivers” are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidally influenced areas within Salt Pond follow this general pattern, where the primary nutrient of eutrophication in the system is nitrogen.

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

Each embayment system maintains a capacity to assimilate watershed nitrogen inputs without degradation. However, as loading increases a point is reached at which the capacity (termed assimilative capacity) is exceeded and nutrient related water quality degradation occurs. 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, and the Falmouth Coastal Overlay Bylaw). While each approach may be different, they all focus on changes in nitrogen loading from watershed to embayment, and aim at projecting the level of increase in nitrogen concentration within the receiving waters. Each approach depends upon estimates of circulation within the embayment; however, few directly link the watershed and hydrodynamic models, and virtually none include internal recycling of nitrogen (as was done in the present effort). However, determination of the “allowable N concentration increase” or “threshold nitrogen concentration” used in previous studies had a significant uncertainty due to the need for direct linkage of watershed and embayment models and site-specific data. In the present effort we have integrated site-specific data on nitrogen and phosphorous levels monitored by the Coastal Systems Program (UMASS-SMAST) staff and graduate students with site-specific habitat quality data (phytoplankton blooms, benthic animals) to “tune” general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Unfortunately, the Salt Pond system is currently beyond its ability to assimilate additional nutrients without impacting ecological health. Nitrogen levels are elevated throughout the system and infaunal animal communities are diminished and lack diversity in many areas. The result is that nitrogen management of the Salt Pond system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed “eutrophication” and when the nutrient loading is primarily from human activities, “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to the system and contributed to the degradation in ecological health, it is also possible that some of the eutrophication within Salt Pond may occur without man’s influence (primarily due to basin structure and inlet dynamics) and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a “pristine” system.

I.4 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into Salt Pond. A two-dimensional depth-averaged hydrodynamic model was applied to the mixed layer, based upon the tidal currents and water elevations was employed for the system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates within the mixed layer of the pond. Exchanges between the deep basin hypolimnetic waters and the mixed layer were evaluated by an examination of rates of exchange via eddy diffusion.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model (Sagamore Lens) for sub-watershed areas designated by MEP. Almost all nitrogen entering Falmouth's salt ponds is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Vineyard Sound source waters was taken from the Falmouth PondWatcher Monitoring Program (supported by the Town of Falmouth and the Coastal Systems Program at SMAST-UMD). Nutrient related water quality throughout the Salt Pond system was collected voluntarily by staff and students from the Coastal Systems Program. Measurements of nitrogen and salinity distributions throughout the system 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 Salt Pond system for the Town of Falmouth. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Town of Falmouth Planning Department supplied data and water-use data. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling in Section VI. Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data from within the embayment and offshore waters (conducted by municipalities). These data and its relevance to the water quality modeling of present conditions are discussed in Section VI. Predictions based upon the water quality model to evaluate nitrogen levels at watershed build-out, and with removal of anthropogenic nitrogen sources are also presented in Section VI. In addition, an ecological assessment of the embayment was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration of this salt pond. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the Salt Pond system.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient related habitat quality. Effects include excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, organic matter enrichment of waters and sediments with the concomitant increased rates of oxygen consumption and periodic depletion of dissolved oxygen, especially in bottom waters, and the limitation of the growth of desirable species such as eelgrass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic 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, which are dependent upon these highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. 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 Salt Pond System, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the 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 embayment specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including the Salt Pond System. As the MEP approach requires substantial amounts of site specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality or unique features.

A limited number of studies have been conducted in the Salt Pond system but did not specifically relate to nitrogen loading, hydrodynamics and habitat health. As such, there is generally little historical data collected from Salt Pond to serve as a starting point from which to build upon by the Massachusetts Estuaries Project. Prior to the MEP, there was no comprehensive baseline of nutrient related water quality in Salt Pond and as such Coastal Systems Scientists voluntarily undertook water quality monitoring in Salt Pond in order to generate the necessary water quality data base for calibration and validation of the MEP hydrodynamic and water quality model. The following is a brief summary of some the historical research undertaken in Salt Pond as well as a description of the water quality monitoring undertaken to date:

Research Studies: Salt Pond has long been used as a natural laboratory by scientists and visiting scientists associated with the research institutions in Woods Hole. The reason is the result of the high nitrogen loading and poor tidal exchange and circulation, which results in a relatively stable stratified surface layer overlying anoxic bottom waters in the confined deep basin. The result is a redox stratified marine basin which mimics the Black Sea and other anoxic basins. As such, research on redox processes and oxic/anoxic interfaces, and microbial cycling can be investigated in a natural system. The result has been a number of research papers focused on these areas:

- ***Population Dynamics of Marine Magnetotactic Bacteria in a Meromictic Salt Pond described by qPCR. Simmons, S. L., Bazylinski, D. A. and Edwards, K. J. (2007), Environmental Microbiology, 9: 2162–2174.*** In the summer of 2003 a study was undertaken by scientists from Woods Hole Oceanographic Institute (WHOI) to quantitatively describe the distribution of species of magnetite- and griegite- producing magnetotactic bacteria (MTB) in a natural system, specifically Salt Pond.

- ***Magnetic Properties of Marine Magnetotactic Bacteria in a Seasonally Stratified Coastal Pond (Salt Pond, MA, USA). Moskowitz, B. M., Bazylinski, D. A., Egli, R., Frankel, R. B. and Edwards, K. J. (2008), Geophysical Journal International, 174: 75–92.***

- ***Dimethylsulfide in a stratified coastal pond. Wakeham, S.G., B.L. Howes and J.W.H. Dacey. Nature 310:770-772, 1984.***

- ***Biogeochemistry of dimethylsulfide in a seasonally stratified coastal salt pond. Wakeham, S.G., B.L. Howes, J.W.H. Dacey, R.P. Schwarzenbach and J. Zeyer***.
- ***Geochimica Cosmochimica Acta 51:1675-1684. 1987.***

- ***Primary production of protein. II. Algal protein metabolism and its relationship to the composition of particulate organic matter in a well mixed euphotic system. Lohrenz, S.E., C.D. Taylor and B.L. Howes. Marine Ecology Progress Series 40:175-183, 1987.***

- ***Occurrence and distribution of diverse populations of magnetic protists in a chemically-stratified coastal salt pond. Bazylinski, D.A., D.R. Schlezinger, B.L. Howes and R.B. Frankel. Chemical Geology. 169:319-328, 2000.***

Ancillary data provided in these papers was evaluated relative to the MEP modeling and assessment effort, particularly useful was information on stratification and water column chemical structure.

BUMP Study - Assessment of Groundwater nutrient inputs to Salt Pond, Falmouth, MA: Concentration, nitrogen load, storm effects and snow dumping (2005) completed by R. Robertson and K. Bentley. This effort was a student project at Boston University. The concept was to determine nitrogen loading from a land-use model and confirm the loading from 24 sites around the shore of Salt Pond. While a good educational exercise, the project used a very different watershed delineation than available to the MEP, which invalidates the estimated load. Equally important, the ground water sampling technique used was inadequate to determine the nitrogen profile of discharging groundwater, as has been observed previously (Howes et al., 2004). No water quality or hydrodynamic information was collected.

Salt Pond Water Quality Monitoring Program - The MEP analysis requires high quality water quality data in order to complete its assessment and modeling approach. Water quality monitoring in the Town of Falmouth's estuaries has been a multi-faceted program generally based on technical support from the Coastal System Program-SMAST, providing consistency of results. Water quality monitoring across all of the embayments in the Town of Falmouth has been completed over the years through four main groups depending on the embayment system: 1) Coalition for Buzzards Bay for Buzzards Bay systems, 2) the Falmouth PondWatch for south coast estuaries 3) the Coastal Systems Program-SMAST for south shore systems not monitored by PondWatch (Salt Pond and Falmouth Inner Harbor) and 4) Water Quality Monitoring Collaborative (Mashpee Wampanoag Tribe, Town of Mashpee, CSP-SMAST) for Waquoit Bay and Eel Pond.

The Coalition for Buzzards Bay's Water Quality Monitoring Program has been collecting data on nutrient related water quality throughout Buzzards Bay estuaries for more than a decade, inclusive of Quissett Harbor, Wild Harbor, outer Megansett Harbor, Fiddlers Cove and Rands Harbor in the Town of Falmouth. The Coalition's BayWatcher Program has collected the principal baseline water quality data to support ecological management of each of Falmouth's Buzzards Bay embayments. The BayWatchers is a citizen-based water quality monitoring program coordinated by T. Williams with technical and analytical assistance from the Coastal Systems Program at SMAST-UMD through 2008. The program has a USEPA and MassDEP approved Quality Assurance Project Plan (QAPP). However, the program generally samples from shore or docks.

The Town of Falmouth implemented a water quality monitoring program to collect baseline water quality data in specific south shore systems in the town in collaboration with researchers now at the Coastal Systems Program at SMAST. The Town of Falmouth has long recognized the potential threat of nutrient over-enrichment of its coastal salt ponds and embayments. In the mid-1980s the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts. The water quality monitoring program, Falmouth PondWatch, was established to provide on-going nutrient related embayment health information in support of the By-law. The first three Ponds to undergo water quality monitoring in the Town of Falmouth where Oyster Pond, Little Pond and Green Pond. These approaches were primarily initiated for planning as development within coastal watersheds progressed. The Town of Falmouth Planning Department has continued to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present study is part of that continuing effort. Unfortunately, monitoring has documented that most regions within the Town's coastal ponds are currently showing water quality declines and are beyond the limits set by the By-law.

Over time the PondWatch Water Quality Monitoring Program expanded to also collect water quality data from Great Pond, Bourne's Pond and West Falmouth Harbor. Because of these efforts, all PondWatch estuaries have completed Massachusetts Estuaries Project assessments and also have USEPA accepted TMDL's to support the Town of Falmouth's on-going restoration efforts. However, PondWatch did not include monitoring of Salt Pond or Falmouth Inner Harbor, Eel Pond or Waquoit Bay. Faced with the lack of necessary water quality baseline data for Salt Pond and Falmouth Inner Harbor to conduct the Massachusetts Estuaries Project's assessment, the scientists and graduate researchers within the Coastal Systems Program-SMAST undertook the collection of the necessary minimum three years baseline data in order to support entry of Salt Pond and Falmouth Inner Harbor (MEP analysis completed June 2011) into the MEP. These "research volunteers" at CSP-SMAST initiated data collection in summer 2006 and created a 7 year baseline of summer water quality for Salt Pond (2006-2012). The water quality data collected in Salt Pond was collected using the same protocols developed and implemented for the collection of water quality data by the other 3 groups monitoring the other estuaries in the Town of Falmouth. Note that both PondWatch and the other south shore programs collect samples from the middle of estuarine basin from boats rather than shoreside to better represent water quality conditions within the estuaries.

The common focus of all 4 water quality monitoring efforts has been to gather site-specific data on the current nitrogen related water quality throughout all the embayments of the Town of Falmouth to support evaluations of observed water quality and habitat health. The CSP-SMAST effort in the Salt Pond Embayment System developed the only water quality baseline for this system (Figure II-1). Samples were analyzed at the SMAST Coastal Systems Analytical Facility. The Coastal Systems Analytical Facility is located in the School for Marine Science and Technology UMASS-Dartmouth, 706 S. Rodney French Blvd, New Bedford, MA, and the laboratory Points of Contact are Sara Sampieri 508-910-6325 (ssampieri@umassd.edu) or Mike Bartlett (mbartlett@umassd.edu). Use of the SMAST Analytical Facility ensured sufficient sensitivity and accuracy of the analytical protocols and that proper QA/QC procedures were followed to allow incorporation of the data into the MEP analysis. The baseline water quality data were a prerequisite to entry into the MEP. Implementation of the MEP's Linked Watershed-Embayment Approach necessarily incorporates the quantitative water column nitrogen data (2006-2012) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff.

Regulatory Assessments of Salt Pond Resources - The Salt Pond Estuary contains a variety of natural resources of value to the citizens of the Town of Falmouth as well as to the Commonwealth. As such, over the years surveys have been conducted to support protection and management of these resources. The MEP gathers the available information on these resources as part of its assessment, and presents them here (Figures II-2 through II-5) for reference by those providing stewardship for this estuary. For the Salt Pond Estuary these include:

- Mouth of River designation - MassDEP (not applicable)
- Designated Shellfish Growing Area – MassDMF (Figure II-2)
- Shellfish Suitability Areas - MassDMF (Figure II-3)
- Anadromous Fish Runs - MassDMF (Figure II-4)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-5)

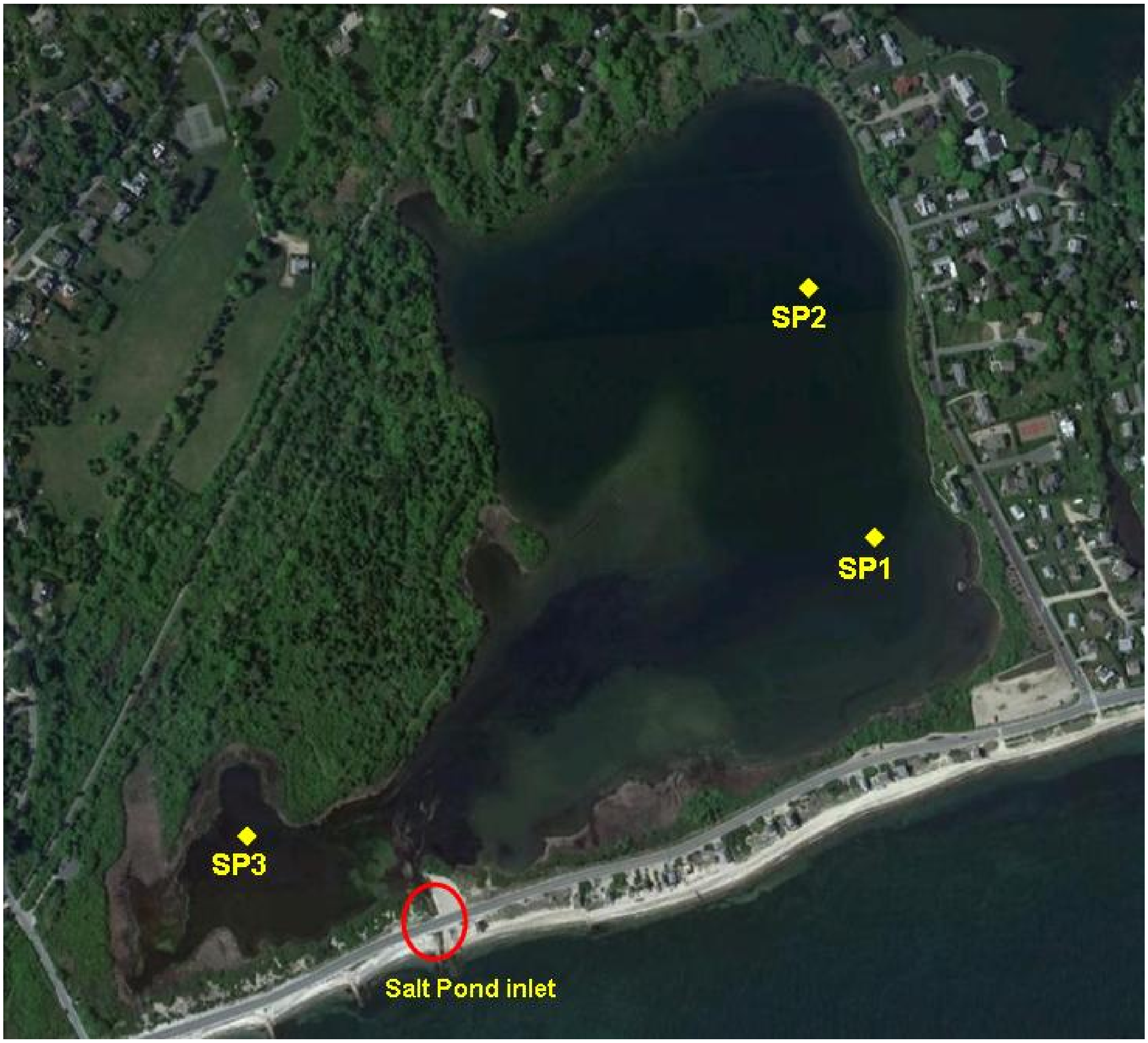

Figure II-1. Coastal Systems Program-SMAST Water Quality Monitoring Program for Salt Pond. Estuarine water quality monitoring stations sampled by the CSP and analyzed at the CSP-SMAST analytical facility during summers 2006 to 2012.

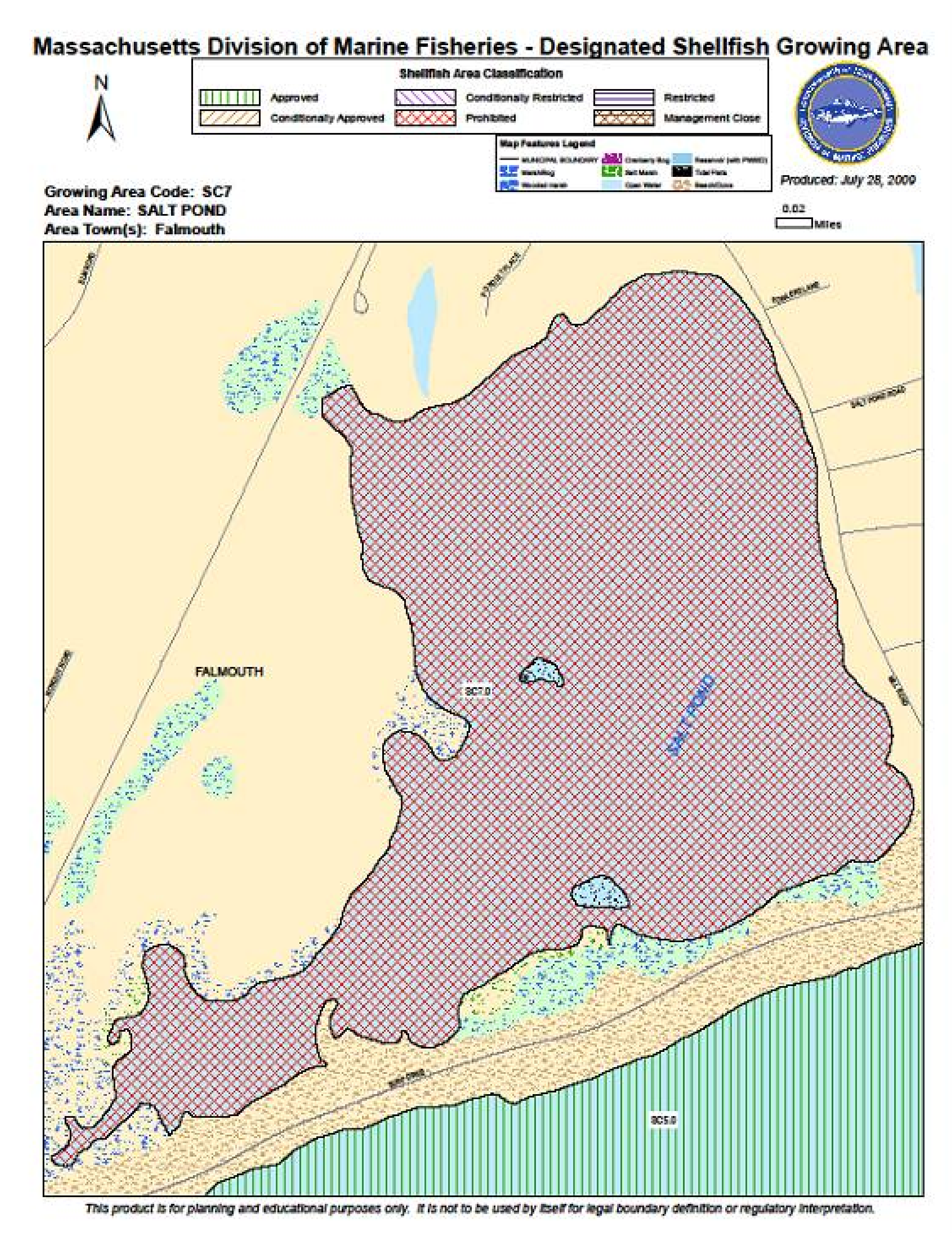

Figure II-2. Location of designated shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Shellfishing in Salt Pond is presently prohibited due to bacterial contamination, likely resulting from wildlife and birds, although some stormwater inflows may periodically occur. Increasing tidal flushing for nitrogen management will likely improve bacteria levels, as well.

Figure II-3. Location of shellfish suitability areas within the Salt Pond Estuary as determined by Mass Division of Marine Fisheries. Suitability indicates habitat that could support a specific shellfish species and does not necessarily mean that the shellfish is present or growing at a specific location.

Figure II-4 Anadromous fish run within the Salt Pond Estuary as determined by Mass Division of Marine Fisheries. The red diamond shows area where fish were observed.

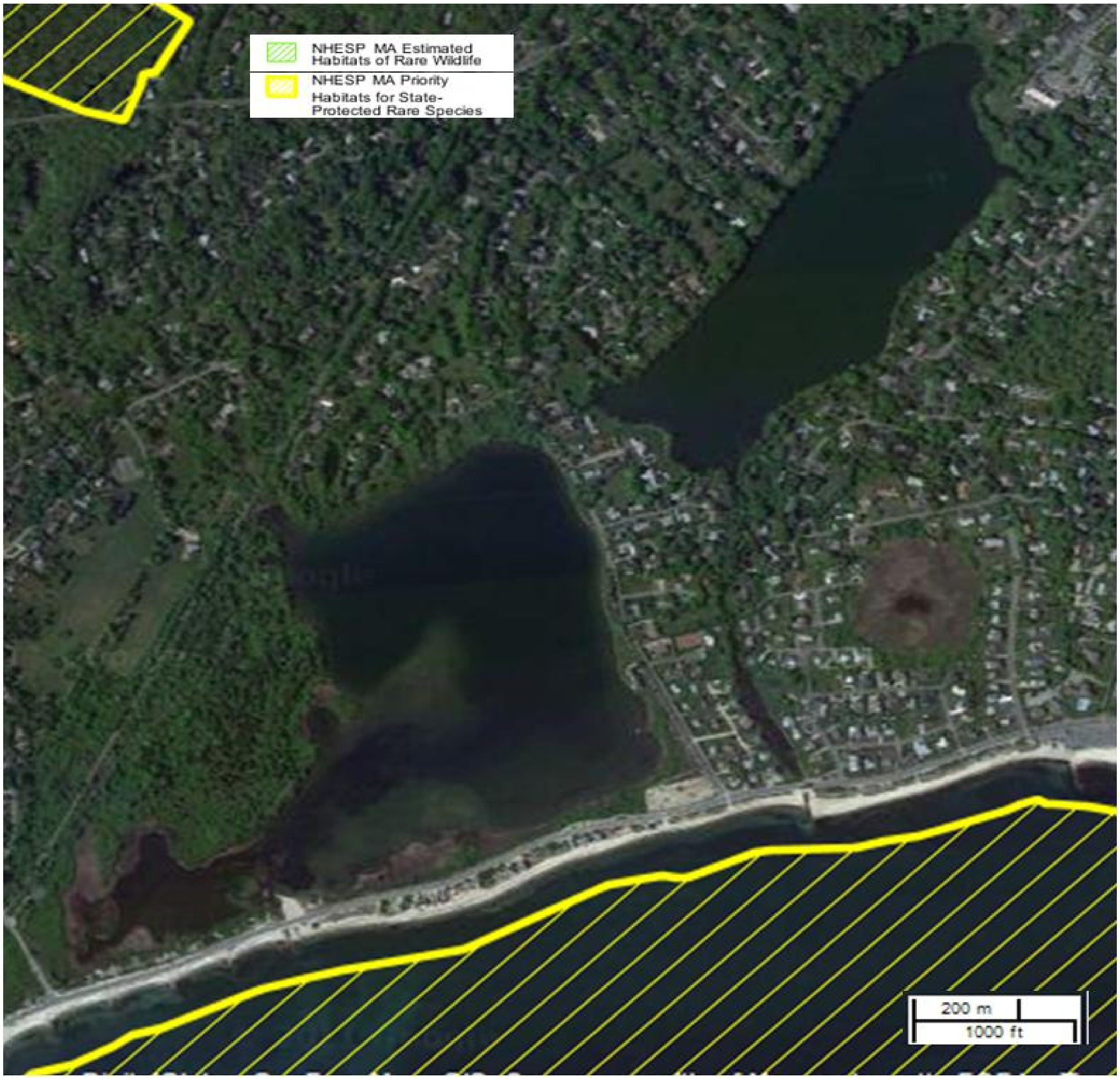

Figure II-5. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Salt Pond Estuary as determined by - NHESP.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation. These questions include surface water/groundwater interactions, groundwater travel times, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Salt Pond estuary system. Salt Pond and its watershed are located entirely within the Town of Falmouth.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watersheds or contributing areas to the Salt Pond estuary system under evaluation by the Project Team. The Salt Pond estuarine system has a single main central basin exchanging tidal waters through a single armored inlet to Vineyard Sound. Watershed modeling was undertaken to sub-divide the overall watershed to the Salt Pond system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These travel distributions within each sub-watershed are used as a procedural check to gage the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical groundwater model employed is also being used to evaluate the contributing areas to public water supply wells in the regional Sagamore flow cell; the Salt Pond watershed is located along the western edge of the Sagamore flow cell.

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water inputs and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to a stream and the portion of the groundwater system that discharges directly into an estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the Salt Pond system and its various sub-watersheds, such as Long Pond, were delineated using the regional model of the Sagamore Lens flow cell (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer.

The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Salt Pond system and its sub-watersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams (that may remove nitrogen) prior to discharging into the coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 519 feet below NGVD 29 in the Sagamore Lens); since bedrock is approximately 200 feet below NGVD 29 in most of the Salt Pond area the lowest model layer was inactive in this area of the model with variable thickness in the layer directly above. 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 within the lens.

The glacial sediments that comprise the aquifer of the Sagamore Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The Salt Pond system watershed is located in the Buzzards Bay Moraine, which is thought to have material deposited in place by melting ice in a low energy depositional environment at the edge of a rapidly retreating Buzzards Bay Lobe of the continental ice sheet, and the Falmouth Ice-Contact Deposits, which have material that was deposited at the edge of stationary Buzzards Bay Lobe (Walter and Whealan, 2005). Modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that these materials tend to have lower hydraulic conductivity than the outwash plains that comprise most of the Cape (*e.g.*, Masterson, *et al*., 1996), but this distinction does not tend to impact groundwater flow direction. Direct rainwater run-off is typically rather low in these materials, which is similar to most of the Cape. 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 in the groundwater models were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns. There were no specific streams discharging to the Salt Pond system therefore no flow data were collected for this analysis.

The Sagamore Lens groundwater model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss (Walter and Whealan, 2005). This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the consumptive loss, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within residential areas designated as using on-site septic systems.

III.3 SALT POND ESTUARY CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Salt Pond embayment system, including Long Pond and Grews Pond (Figure III-1), were determined by the United States Geological Survey (USGS). Model outputs of the 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, (c) to include water table data in the lower regions of the watersheds near the coast (as available), and (d) to more closely match the sub-estuary 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 delineation includes 10-yr time-of-travel boundaries. Overall, 5 sub-watershed areas, were delineated within the Salt Pond study area, including watersheds to Long Pond and Grews Pond.

Since the Salt Pond watershed includes Long Pond, which is the primary drinking water supply for the Town of Falmouth, calculations of freshwater discharge to Salt Pond need to correct for the drinking water withdrawals. MEP staff received 2000-2011 water withdrawal volumes for Long Pond from the Town of Falmouth (personal communication, Jerry Potamis and Mary Beth Wiser, September, 2012). This information shows that withdrawal volumes from Long Pond ranged between 7,112 and 11,168 m3/d with an 11 year average of 9,562 m3/d. These withdrawals averaged 59% of total town drinking water withdrawals during the same period. Long Pond is both the primary drinking water supply for the Town of Falmouth and a kettle hole pond. As a drinking water supply, watershed flows are removed and distributed throughout the town, including transfer to development within the pond watershed. The portion of the Long Pond watershed flow that is not removed for drinking water supply is available to flow back into the groundwater system and be transferred through the pond shoreline to down-gradient subwatersheds. Land use development within the watershed is adding some portion of the removed Long Pond supply back into the watershed flow. This circular flow pathway is unique among the Salt Pond watersheds because only Long Pond’s watershed flow is used in this way and, thus, this is the only subwatershed where this calculation is included.

Table III-1 provides the daily freshwater discharge volumes for various sub-watersheds as calculated from the groundwater model; these volumes were used in the salinity calibration of the tidal hydrodynamic model and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to the directly measured surface water discharges. This flow includes corrections for outflow from the watershed for Grews Pond, which straddles the northern boundary of the Salt Pond watershed and public drinking water supply withdrawal from the Long Pond watershed. The overall Long Pond watershed has freshwater inflow of 11,367 m3/d, which includes inflow from a portion of Mares Pond, which straddles the Long Pond and Great Pond Estuary watershed boundary. After correcting for the average withdrawal for drinking water supply (9,562 m3/d) and return flow based on water use within the Salt Pond watershed, 1,944 m3/d discharges through the downgradient shoreline of Long Pond (Table III-2). Based on the portion of downgradient flow from Long Pond captured by Grews Pond (13%), only 258 m3/d of freshwater discharges to Salt Pond from the Long Pond watershed. The overall estimated freshwater flow into the Salt Pond system from the MEP delineated watershed is 2,999 m3/d.

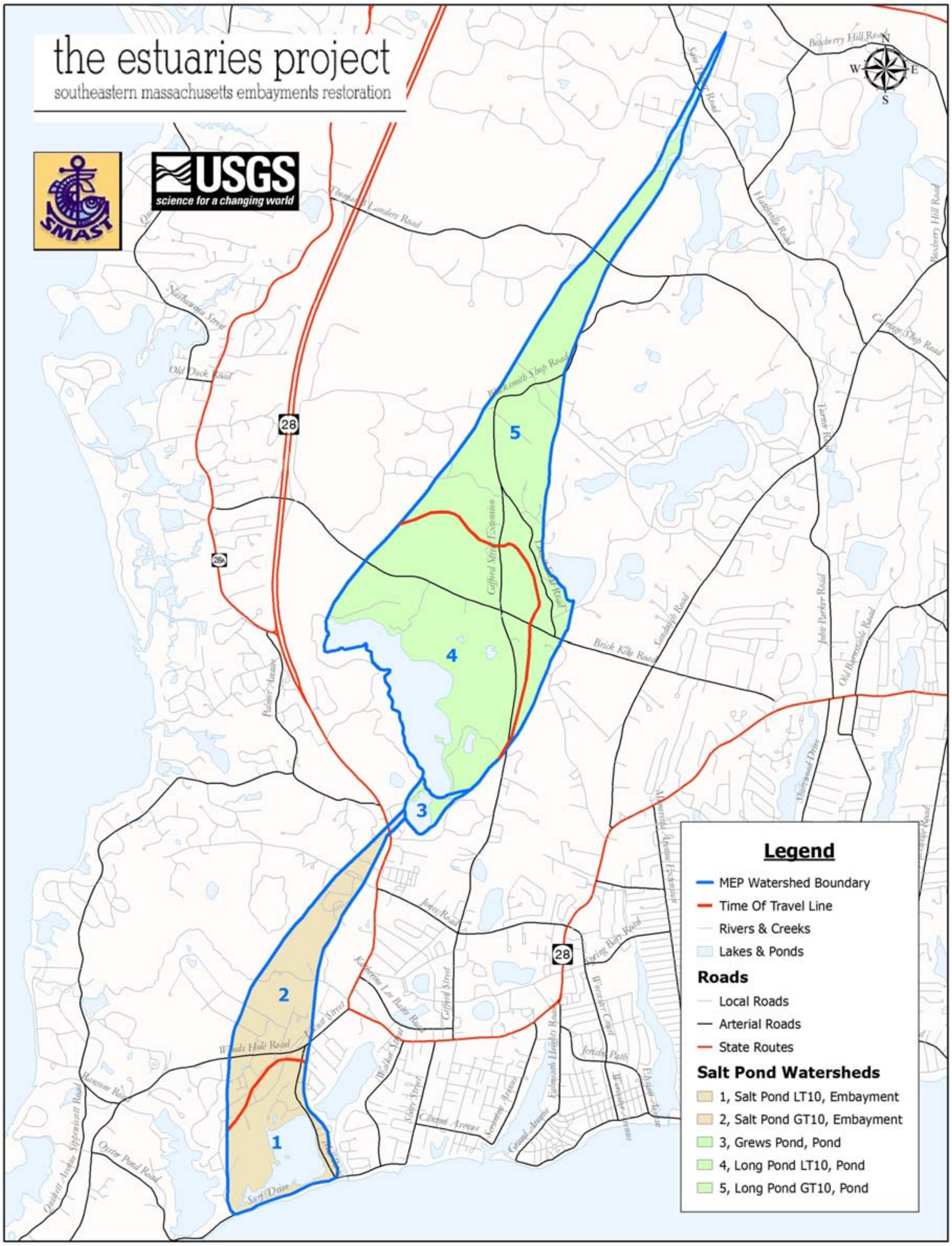

Figure III-1. Watershed delineation for the Salt Pond Estuary. Subwatershed delineations are based on USGS groundwater model output with modifications to better address pond and estuary shorelines. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above).

Table III-1. Daily groundwater discharge from each of the sub-watersheds in the watershed to the Salt Pond Estuary, as determined from the regional USGS groundwater model.

Watershed	#	Total Watershed Area		% contributing to Estuaries	Discharge
		Acres	hectares		m^3/day
Salt Pond LT10	1	139	56	100%	1,066
Salt Pond GT10	2	239	97	100%	1,837
Grews Pond	3	28	11	20%	44
Long Pond LT10	4	690	279	3%	24
Long Pond GT10	5	518	210	3%	18
Mares Pond	6	541	219	1%	10
SALT POND SYSTEM TOTAL					2,999

Notes:
1) discharge volumes are based on 27.25 inches of annual recharge on watershed areas;
2) Grews Pond, Long Pond, and Mares Pond flows include corrections for portion of downgradient flow reaching Salt Pond,
3) Long Pond and Mares Pond flows include corrections for average withdrawal of municipal drinking water from Long Pond,
4) these flows do not include precipitation on the surface of the estuary, and
5) totals may not match due to rounding.

Table III-2. Long Pond (Falmouth) Drinking Water Withdrawals (2000-2011)

year	Long Pond pumped	Total Town-wide pumped	Long Pond % of total town pumped	Long Pond pumped
	MG/yr	MG/yr	%	m3/d
2000	938	1,462	64%	9,727
2001	981	1,603	61%	10,173
2002	910	1,695	54%	9,437
2003	872	1,589	55%	9,043
2004	923	1,652	56%	9,571
2005	1,077	1,696	63%	11,168
2006	686	1,478	46%	7,112
2007	898	1,656	54%	9,315
2008	830	1,518	55%	8,607
2009	800	1,334	60%	8,295
2010	1,013	1,529	66%	10,505
2011	900	1,578	57%	9,338
Average*	922	1,587	59%	9,562

Notes:
1) All data supplied by Town of Falmouth (personal communication, Jerry Potamis and Mary Beth Wiser, September, 2012)
2) Averages are have been corrected for statistical outliers (outside 2x stdev range)

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

After accounting for the contributing portions of the watershed to Long Pond and Grews Pond, the MEP watershed area for the entire Salt Pond system as a whole is 40% larger than 1998 CCC delineation (424 acres vs. 302 acres, respectively). This significant difference is mostly due to a change in the location of the groundwater divide between Vineyard Sound and Buzzards Bay. The northern edge of the MEP watershed is this divide and is located to the north of the CCC divide. This more northern location allows flowpaths to include Grews Pond and Long Pond. In addition to these pond watersheds, the MEP watershed delineation also includes the 10 year time-of-travel sub-watersheds to Salt Pond, which were not included in the CCC delineation. These refinements are another benefit of the update of the regional groundwater model (Walter and Whealan, 2005)

The evolution of the watershed delineation for the Salt Pond system has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model and the use of this model for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down gradient estuary. The MEP watershed delineations were used to develop the watershed nitrogen loads to each portion of the watershed and ultimately to the estuarine waters of the Salt Pond system (Section V.1).

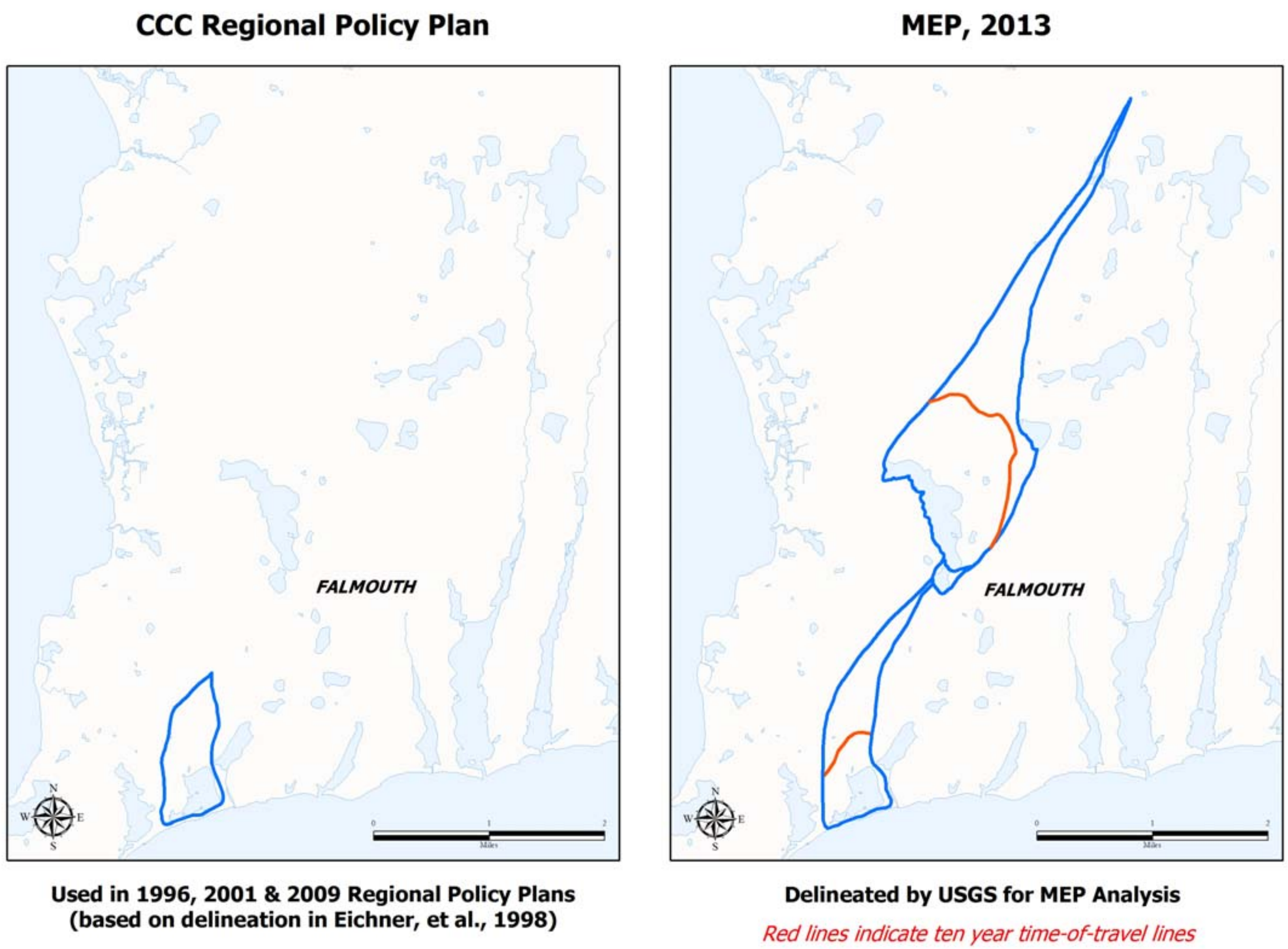

Figure III-2. Comparison of MEP Salt Pond watersheds and sub-watershed delineations and the Cape Cod Commission watershed delineations (Eichner, *et al*., 1998). CCC delineations have been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009). After accounting for the portions of the watershed to Long Pond and Grews Pond that contribute to Salt Pond, the MEP watershed area for the whole Salt Pond system as a whole is 40% larger than 1998 CCC delineation (424 acres vs. 302 acres, respectively). Note the sub-watershed area shown is the total watershed to Long Pond, but only a very small portion of the discharge from this sub-watershed reaches Salt Pond.

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 Salt Pond estuary 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 prior to reaching the estuary. This latter natural attenuation process results from biological processes that naturally occur within these ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from 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 in the sediments is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters or in some cases a sink for nitrogen reaching the bottom. Failure to include the nitrogen balance of estuarine sediments and nitrogen attenuation during transport through the watershed generally leads to errors in predicting water quality, particularly as associated with summertime nitrogen load to embayment waters.

In order to determine watershed nitrogen loading inputs to the Salt Pond Estuary, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of the estuary and its sub-watersheds (Section III). The Salt Pond watershed was sub-divided to define contributing areas or subwatersheds to each of the major inland freshwater systems and to each major portion of the estuary. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches estuary waters in less than 10 years or greater than 10 years. A total of 5 sub-watershed areas, were delineated within the greater Salt Pond Estuary Watershed. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each portion of the estuary (see Chapter III).

The initial task in the MEP land use analysis is to gage whether or not nitrogen discharges to the watershed have reached the estuary. This involves a temporal review of land use changes, the time of groundwater travel provided by the USGS watershed model, and review of data at natural collection points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. Ten-year time of travel subwatersheds in the Salt Pond watersheds have been delineated for the ponds and the estuary. Review of less than and greater than 10 years of travel watersheds indicates that 46% of the unattenuated nitrogen load from the Salt Pond watersheds is within less than 10 year travel time to the estuary (Table IV-1). Further review of the land use information in the greater than 10 year portion of the Salt Pond watershed indicates that the average year built for single residences, which are the predominant parcel type in the watershed, is 1963. This indicates that most of the development within the GT10 subwatershed has been contributing nitrogen load to Salt Pond for conservatively more than 30 years. The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuary and that the distinction between time-of-travel in the subwatersheds is not important for modeling existing conditions. Overall and based on the review of all this information, it was determined that the nitrogen conditions within the Salt Pond Estuary are currently in balance with its watershed load.

Table IV-1. Percentage of attenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Salt Pond.

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
Salt Pond LT10	1	634		634	100%
Salt Pond GT10	2		1,087	1,087	0%
Grews Pond TOTAL	GP		13	13	0%
Salt Pond Estuary Surface		288		288	100%
Salt Pond System TOTAL		**922**	**1,100**	**2,201**	**46%**

Notes: 1) these loads are corrected to account for only portions of Long Pond and Grews Pond watersheds that discharge within the Salt Pond watershed, 2) sums may not add due to rounding, 3) review of year-built information for single family residences (the predominant parcel type in the Salt Pond GT10 subwatershed) indicates that the average residence has existed for 50 years; even with the groundwater flow lag, nitrogen loads from these parcels would be reflected in Salt Pond water quality data during the 2000s; for this reason, the water quality and the watershed nitrogen load are in balance.

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed site-specific studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes and Ramsey, 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land uses and pre-determined nitrogen loading rates based on regional analyses. For the Salt Pond Estuary, the models used land-use data from the Town of Falmouth transformed into nitrogen loads using both regional nitrogen loading factors and local watershed-specific data (such as parcel-by-parcel water use). Determination of the nitrogen loads required obtaining watershed-specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the “potential” or unattenuated nitrogen load to each receiving embayment, since attenuation during transport is included at a later stage.

Natural attenuation of nitrogen during transport from a watershed to an estuary is generally determined based on site-specific studies of streamflow and pond characteristics. In the Salt Pond watershed there are no significant streams that could be gaged and nitrogen attenuation in the upgradient freshwater ponds was based on a conservative assessment from Falmouth and other Cape Cod ponds. The attenuated loads from these ponds was further adjusted based on the portion of the downgradient discharge that remains within the Salt Pond watershed and, in the case, of Long Pond, the removal of water for municipal drinking water. Attenuation in ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data is sufficiently reliable to calculate a pond-specific nitrogen attenuation factor.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. In the present effort, two freshwater ponds have delineated subwatersheds within the Salt Pond watershed: Long Pond and Grews Pond. 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 (~5%) overestimated given the distribution of nitrogen sources within the watershed.

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary. Internal nitrogen recycling was also determined throughout the tidal reaches of the Salt Pond Estuarine System; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Since the watersheds to Salt Pond are completely within the Town of Falmouth, Estuaries Project staff obtained digital parcel and tax assessor's data from the town to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data for Falmouth are from 2009. The land use database contains traditional information regarding land use classifications (MassDOR, 2012) plus additional information developed by the town. The overall effort was completed with assistance from GIS staff from the Cape Cod Commission.

Figure IV-1 shows the land uses within the Salt Pond estuary watersheds. Land uses in the study area are grouped into seven (7) land use categories: 1) residential, 2) commercial, 3) industrial, 4) open space, 5) undeveloped, 6) public service/government, including road rights-of-way, and 7) freshwater pond surfaces. These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2012). "Public service" in the MADOR system is tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, golf courses, open space, roads) and private groups like churches and colleges.

Public service land uses are the dominant land use type in the overall watershed, as well as the two groundwater subwatersheds to Salt Pond (LT10 and GT10). Public service lands are 49% and 55% of these two groundwater subwatersheds and 53% of the overall system watershed. Examples of these land uses in the Salt Pond GT10 subwatershed are lands owned by the Three Hundred Committee and the town, with Salt Pond Sanctuaries overseeing most of the land directly bordering Salt Pond. Residential land uses occupy the second largest area within the two groundwater subwatershed (46% and 38%, respectively) and 23% of the overall watershed area. It is notable that land classified by the town assessor as undeveloped is 3% of the overall watershed area.

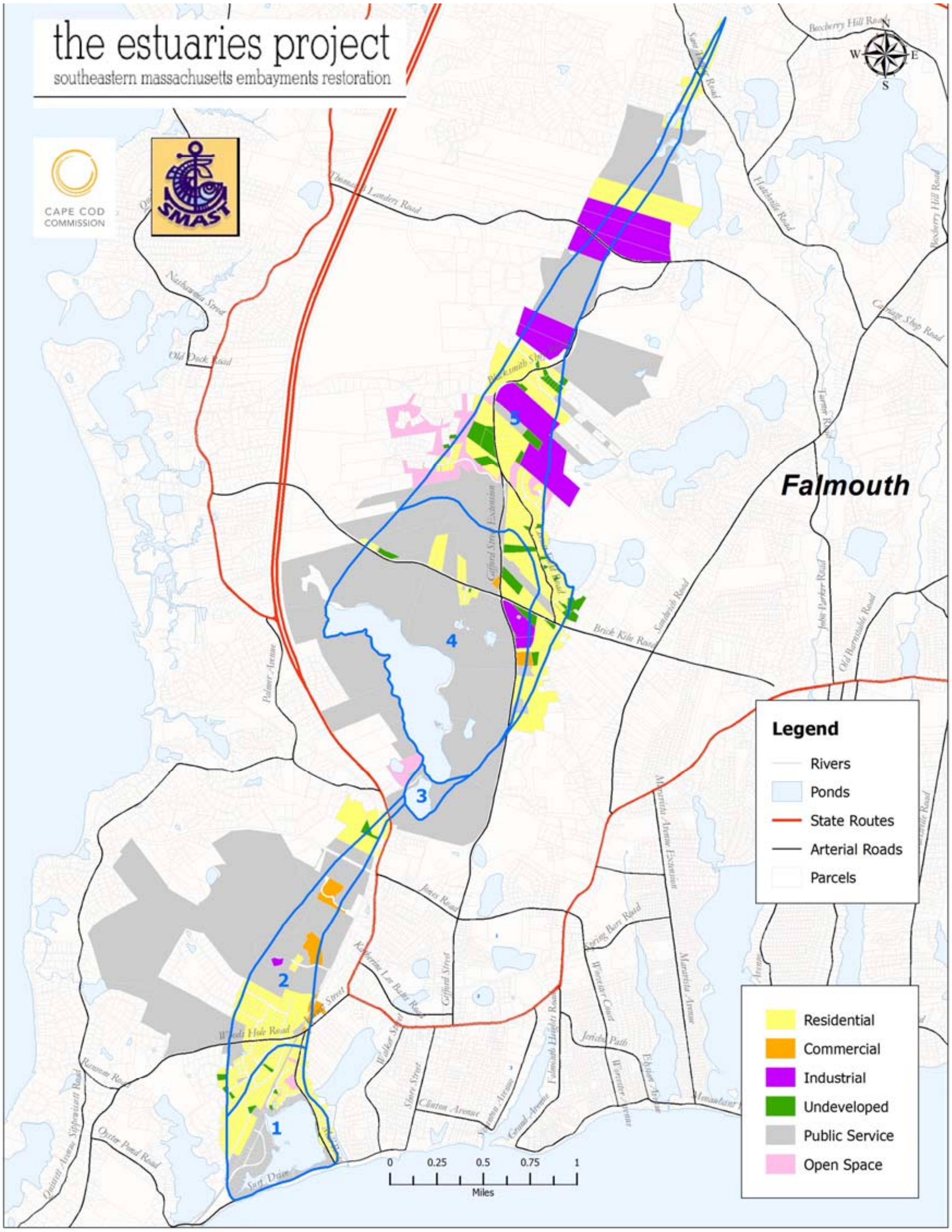

Figure IV-1. Land-use in the Salt Pond Estuary watershed and subwatersheds. The system watersheds are all within the boundaries of the Town of Falmouth. Only a small portion of the recharge and load from the upper watersheds (3 & 4) contribute to estuary loading. Land use classifications are based on town assessor classifications and MADOR (2012) categories. Base assessor and parcel data for Falmouth are from the year 2009.

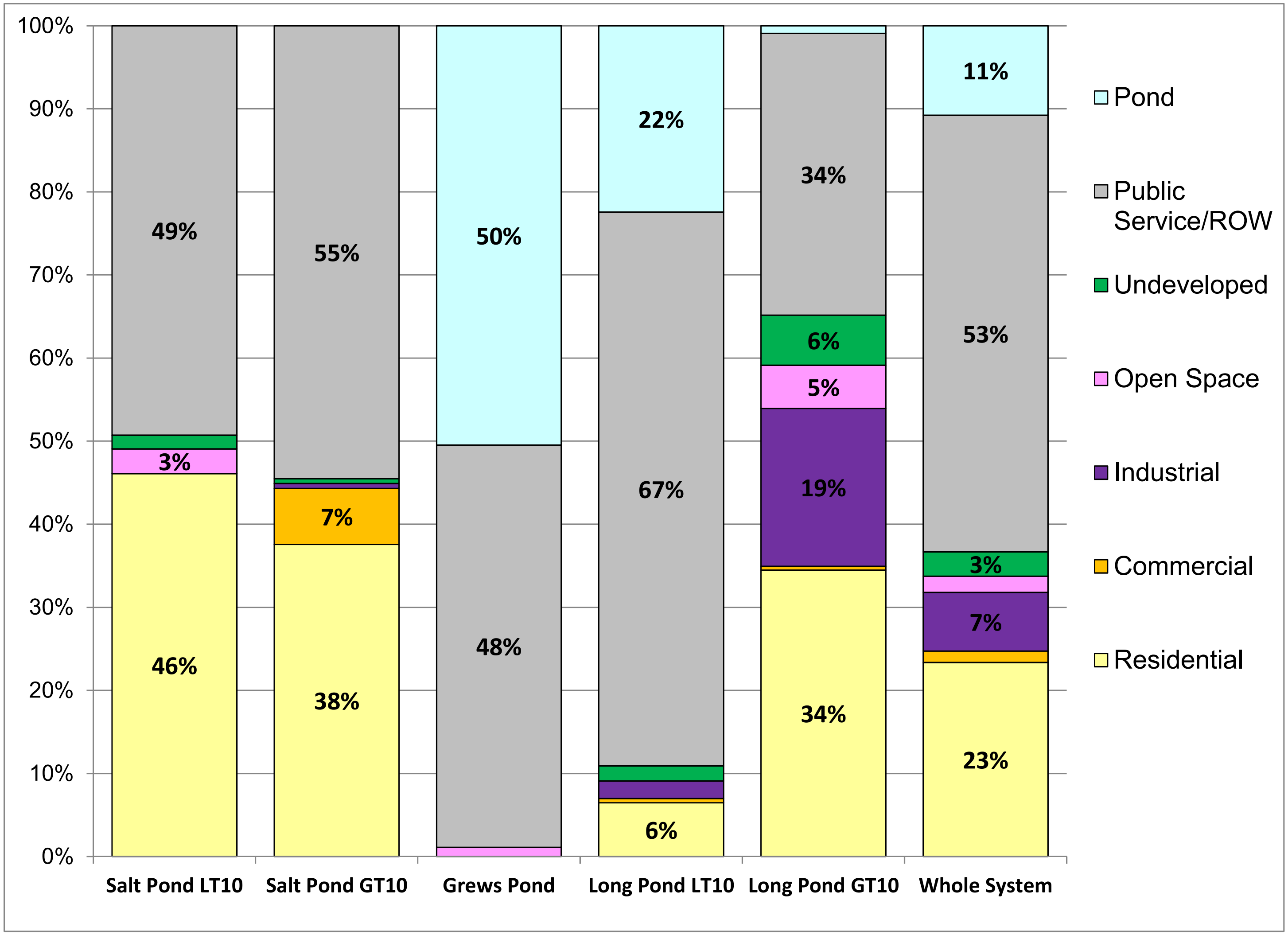

Figure IV-2. Distribution of land-use types by area within the Salt Pond Estuary subwatersheds and whole system watershed. Land use categories are generally based on town assessor's land use classification and groupings recommended by MADOR (2012). Only percentages greater than or equal to 3% are labeled.

It is also useful to partition the land-use areas per parcel type (Figure IV-2) by number of parcels comprising each type. In all the subwatershed groupings shown in Figure IV-2, except for Grews Pond, residential parcels predominate by number, ranging between 48% in the Long Pond LT10 subwatershed to 80% in the Salt Pond GT10 subwatershed. In the Grews Pond subwatershed, public service parcels are 75% of the total parcel count. Residential parcels are also the dominant parcel type in the overall Salt Pond watershed with 67% of the parcels. Public service parcels are the next highest percentages in all the subwatersheds except for the Long Pong GT10 subwatershed where undeveloped parcels are the second highest percentage after residential parcels. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 84% to 97% of residential parcels in the individual subwatershed groupings. Single-family residences are 94% of the residential parcels throughout the Salt Pond system watershed.

In order to estimate wastewater flows within the Salt Pond sub-watersheds, MEP staff also obtained parcel-by-parcel water use data from the Town of Falmouth (personal communication, Bob Shea, GIS Coordinator, 11/10). Three years of water use (fiscal years 2008, 2009 and 2010 was obtained from the town. The water use data was linked to the respective town parcel databases by the town GIS Department staff. Measured water use is used to estimate wastewater-based nitrogen loading from the individual parcels; average water use for each parcel is used for parcels with multiple years of data. The final wastewater nitrogen load for each parcel is based upon the measured water-use, wastewater nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2). All parcels are assumed to use on-site septic systems unless additional information is available.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

The Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a *per capita* nitrogen load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon a number of studies and additional information that directly measured septic system and per capita loads on Cape Cod or in similar geologic settings (Nelson *et al*. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter *et al*., 1990, Brawley *et al*., 2000, Howes and Ramsey 2000, Costa *et al*. 2001). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}.

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

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

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

The nitrogen loads developed using this approach have been validated in a number of long and short term field studies where integrated measurements of nitrogen discharge from watersheds could be directly measured. Weiskel and Howes (1991, 1992) conducted a detailed watershed/stream tube study that monitored septic systems, leaching fields and the transport of the nitrogen in groundwater to adjacent Buttermilk Bay. This monitoring resulted in estimated annual per capita nitrogen loads of 2.17 kg (as published) to 2.04 kg (if new attenuation information is included). Further, modeled and measured nitrogen loads were determined for a small sub-watershed to Mashapaquit Creek in West Falmouth Harbor (Smith and Howes, manuscript in review) where measured nitrogen discharge from the aquifer was within 5% of the modeled N load. Another evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The modeled and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year in this type of ecological situation (Samimy and Howes, unpublished data).

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

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy soils and outwash aquifers; (b) has been validated in studies of the MEP Watershed “Module”, where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Nitrogen Loading Coefficients (*e.g.*, stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees with specific studies of consumptive water use and nitrogen attenuation between the septic tank and the discharge site; and (d) the watershed module provides estimates of nitrogen attenuation by freshwater systems that are consistent with a variety of ecological studies. It should be noted that while points b-d support the use of the MEP Septic N Loading 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 in other MEP assessments add additional weight to the nitrogen loading coefficients used in MEP analyses. While the MEP septic system nitrogen load is a reasonable estimate, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher loads that are generally used for septic systems in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%)) needed to lower embayment nitrogen levels to a nitrogen target (*e.g.,* nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (*i.e*., attenuated loads).

In order to provide an independent validation of the average residential water use within the Salt Pond watershed, MEP staff reviewed US Census population values for the Town of Falmouth. 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 Falmouth is 2.36 people per housing unit with 69% year-round occupancy, while 2010 Census information indicates that the average occupancy dropped to 2.24 people per housing unit and the year-round occupancy also dropped to 64%. Average water use for single-family residences with municipal water accounts in the combined Salt Pond MEP study area is 140 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the study area average is 126 gpd.

In order to provide a check on the water use, the Falmouth Census average occupancies were multiplied by the state Title 5 estimate of 55 gpd of wastewater per capita results, The resulting flow estimates are 130 gpd of average estimated water use per residence based on 2000 Census occupancy and 123 gpd based on 2010 Census occupancy. Estimates of summer populations on Cape Cod derived from a number of approaches (*e.g.*, traffic counts, garbage generation, WWTF flows) suggest average population increases from two to three times year-round residential populations measured by the US Census. If it is assumed that the Falmouth population doubles for three months during the summer, the adjusted year-round occupancy would rise to 2.95 or 2.80 people per housing unit for the 2000 and 2010 Census occupancies, respectively. These occupancies multiplied by 55 gpd/person result in respective average flows of 162 gpd and 154 gpd. If population is assumed to triple, the respective average flows are 194 gpd and 185 gpd. This analysis suggests that seasonal adjustments of population in these watersheds are relatively minor. The above analysis also suggests that water use, on average, is a reasonable estimate of wastewater generation within the study area.

Water use information exists for 97% of the 446 developed parcels in the overall Salt Pond watersheds. Developed parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs or town assessor valuations, and do not have a listed account in the water use databases. Of the 13 developed parcels without water use accounts, 12 (92%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and are assigned the Salt Pond study area average water use of 140 gpd in the watershed nitrogen loading modules. Given the preponderance of residential land uses among developed parcels without water use accounts, all parcels without water use in the Salt Pond study area were assigned the Salt Pond study area average water single family residence use of 140 gpd as their water use in the watershed nitrogen loading model.

Wastewater Treatment Facilities and Alternative Septic Systems

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

MEP staff received a list of alternative, denitrifying septic system in Falmouth and total nitrogen effluent monitoring data from the Barnstable County Department of Health and the Environment (personal communication, Brian Baumgaertel, 1/11). Based on this database, no alternative, denitrifying septic systems were identified within the Salt Pond watershed.

MEP staff also contacted MassDEP to review whether any Groundwater Discharge Permits (GWDPs) are on file for any sites in the Salt Pond study area. A GWDP is required under MassDEP regulations for wastewater treatment systems with design flows greater than 10,000 gallons per day. According to the MassDEP databases, no GWDPs exist in the Salt Pond watershed.

According to town GIS coverages, there are 30 parcels that are connected to the municipal sewer system within the Salt Pond watershed. Since the municipal wastewater treatment facility treats and discharges collected wastewater within the West Falmouth Harbor watershed, wastewater flows from these parcels are assumed to be completely removed from the Salt Pond watershed.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of watershed nitrogen loading to estuaries is usually fertilized areas: lawns, golf courses, and cranberry bogs. Residential lawns are usually the predominant source within this category. In order to add this source to the nitrogen loading model for the Salt Pond systems, MEP staff reviewed available regional information about residential lawn fertilizing practices. No golf courses or cranberry bogs exist within the 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, prior to the start of the estuaries project 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 nitrogen leaching rate of 20% results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn; these factors are used in the MEP watershed nitrogen loading calculations. It should also be noted that a recent technical review of lawn fertilizer leaching in settings similar to those on Cape Cod confirmed that the 20% leaching rate is appropriate (HWG, 2009). It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. It should be noted that professionally maintained lawns in the three town survey were found to have the higher rate of fertilizer application and hence higher estimated annual contribution to groundwater of 3 lb/lawn/yr. Residential lawns are the only source of turf fertilizers in the Salt Pond watershed.

Nitrogen Loading Input Factors: Town of Falmouth Landfill

MEP staff contacted MassDEP staff to obtain any nitrogen monitoring data for solid waste sites within the Salt Pond system watershed. MassDEP databases identified only one solid waste site within the watershed: the Town of Falmouth landfill. Project staff obtained monitoring well construction details, boring logs, well locations and 2001-2010 sample results from MassDEP files (Mark Dakers, SERO, personal communication, March/April 2011). Development of the nitrogen load for the Town of Falmouth landfill is based on the available monitoring and spatial data that is discussed in this section.

The Falmouth Landfill is located partially within the Long Pond GT10 subwatershed (Figure IV-3). According to MassDEP files, it is unlined, was opened in 1955, accepted municipal solid waste, was closed in 1998, and capped in 2000. A series of monitoring wells have been installed around the landfill, along perceived flow paths between the landfill and Long Pond, and along the upgradient edge of Long Pond (see Figure IV-3). MEP staff reviewed the screen lengths and depths for the wells, as well as key water quality constituents from their sampling (*e.g.*, specific conductivity, dissolved oxygen, alkalinity, and the two nitrogen analytes that were usually reported: nitrate-nitrogen and total Kjeldahl nitrogen). Based on this review, staff determined that the key wells for assessing the impact of the landfill are WC-6S and 558D. These two wells have a combined average total nitrogen concentration (nitrate-nitrogen plus total Kjeldahl nitrogen) of 21 mg/L based on 2001-2010 sampling. Based on historic aerials and the information from MassDEP files, project staff determined that there is approximately 162,200 square meters (40 acres) of solid waste at the landfill. For the purposes of estimating the nitrogen load from the site, project staff conservatively assumed that the whole parcel was within the Long Pond watershed, that the average TN concentration for the two most impacted wells are representative of the landfill impact, and that the standard MEP recharge is passing through the site. Using this approach, the site has an annual nitrogen load of 2,365 kg. This load was added to the Long Pond GT10 subwatershed nitrogen load.

It is acknowledged that this approach for estimating a nitrogen load from the Falmouth landfill includes a number of assumptions. It is appropriate based on the available data. A detailed assessment of all the available data is beyond the scope of the MEP, staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP.

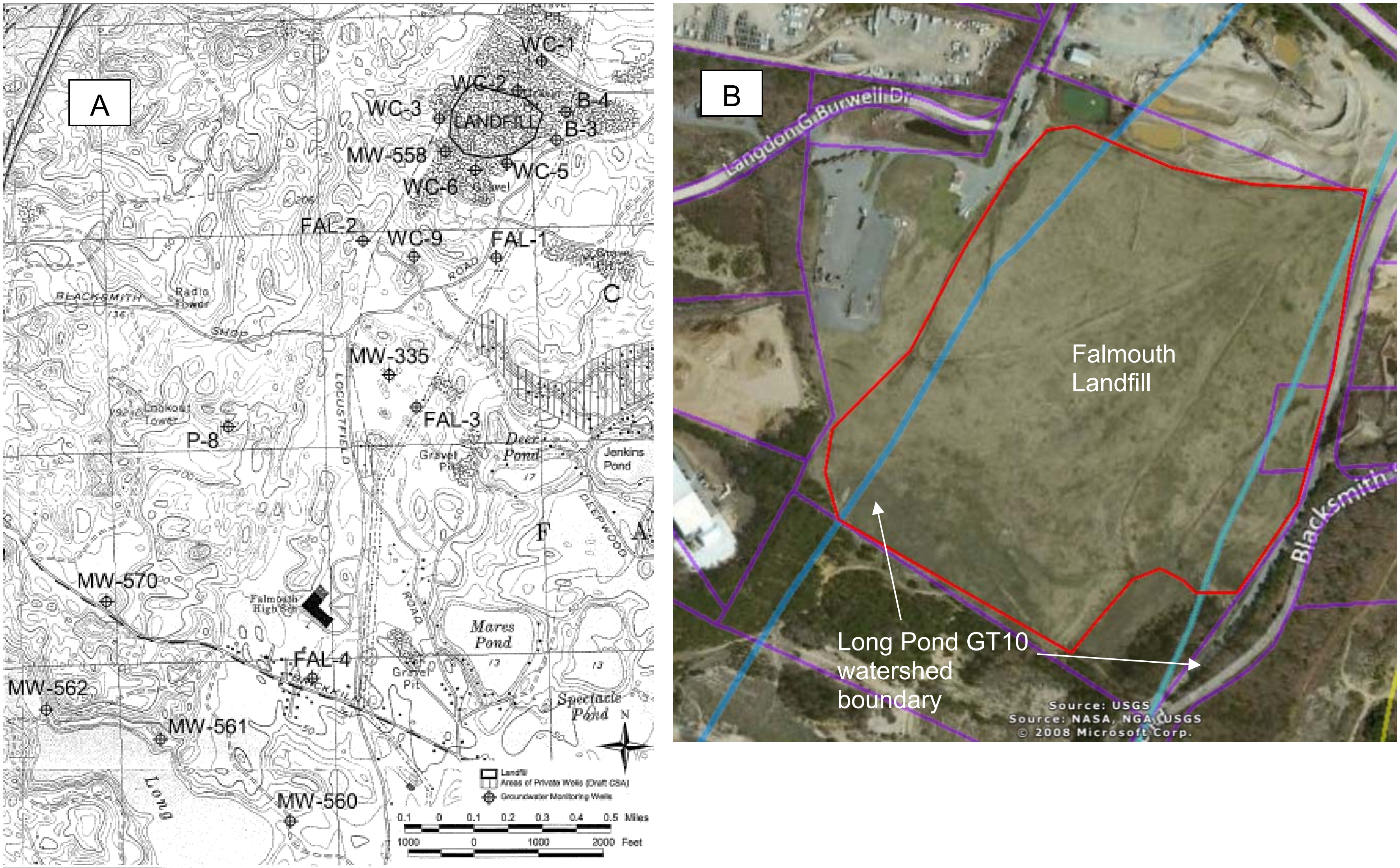

Figure IV-3. Falmouth Landfill Monitoring Wells and Aerial of Landfill Property. A shows the monitoring well (MW) network around the Falmouth Landfill and between the landfill and Long Pond. B shows aerial of the landfill site (captured between March and October 2011) with the estimated solid waste extent (outlined in red) and the Long Pond GT10 subwatershed boundaries (blue lines).

to include conservatism in nitrogen loading estimates when uncertainty exists in the data. A more refined evaluation and assessment of the established monitoring well network would help to refine this assessment and future management options.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Salt Pond assessment are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the CCC's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and MassDEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the MEP nitrogen loading analysis for the Salt Pond watersheds are summarized in Table IV-2.

Road areas are based on MassHighway GIS information, which provides road width for various lengths of road segments. MEP staff utilized the GIS to sum these segments and their various widths by subwatershed. Project staff also checked this information against parcel-based rights-of-way.

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information is linked to the parcel coverages, parcels are assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel is located within a respective subwatershed. Following the assigning of boundary parcels, all large parcels are examined individually and are split (as appropriate) in order to obtain less than a 2% difference between the total land area of each subwatershed and the sum of the area of the parcels within each subwatershed. The resulting "parcelized" watersheds to Salt Pond are shown in Figure IV-4.

The review of individual parcels straddling watershed boundaries includes corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. The Town of Falmouth provided GIS coverages of building footprints for the roof area calculations. All developed parcels are assumed to have 825 square foot driveways. Individualized information for parcels with atypical nitrogen loading (golf courses, denitrifying septic systems) is also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the Salt Pond estuary. The assignment effort is undertaken to better define sub-estuary loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

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

<table>
<tr><th>Nitrogen Concentrations:</th><th>mg/l</th><th>Recharge Rates:</th><th>in/yr</th></tr>
<tr><td>Road Run-off</td><td>1.5</td><td>Impervious Surfaces</td><td>40</td></tr>
<tr><td>Roof Run-off</td><td>0.75</td><td>Natural and Lawn Areas</td><td>27.25</td></tr>
<tr><td>Natural Area Recharge</td><td>0.072</td><td colspan="2">Water Use/Wastewater:</td></tr>
<tr><td>Direct Precipitation on Embayments and Ponds</td><td>1.09</td><td rowspan="3">Existing developed parcels wo/water accounts and buildout single-family residential parcels:</td><td rowspan="3">140 gpd[2]</td></tr>
<tr><td>Wastewater Coefficient</td><td>23.63</td></tr>
<tr><td colspan="2">Fertilizers:</td></tr>
<tr><td>Average Residential Lawn Size (sq ft)[1]</td><td>5,000</td><td>Existing developed parcels w/water accounts:</td><td>Measured annual water use</td></tr>
<tr><td rowspan="2">Residential Watershed Nitrogen Rate (lbs/lawn)[1]</td><td rowspan="2">1.08</td><td colspan="2">Commercial and Industrial Buildings buildout additions[3]</td></tr>
<tr><td colspan="2">Commercial</td></tr>
<tr><td colspan="2" rowspan="3">Golf course fertilizer applications based on MEP averages derived from 19 other courses</td><td>Wastewater flow (gpd/1,000 ft2 of building):</td><td>180</td></tr>
<tr><td>Building coverage:</td><td>15%</td></tr>
<tr><td>Industrial</td><td></td></tr>
<tr><td rowspan="2">Average Single Family Residence Building Size from watershed data (sq ft)[4]</td><td rowspan="2">1,586</td><td>Wastewater flow (gpd/1,000 ft2 of building):</td><td>44</td></tr>
<tr><td>Building coverage:</td><td>5%</td></tr>
</table>

Notes:
1) Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.
2) Based on average flow of all single-family residences in the watershed
3) based on existing water use and water use for similarly classified properties throughout the Town of Falmouth
4) Used for buildout calculations; building footprints for each parcel were used for the existing conditions nitrogen loads

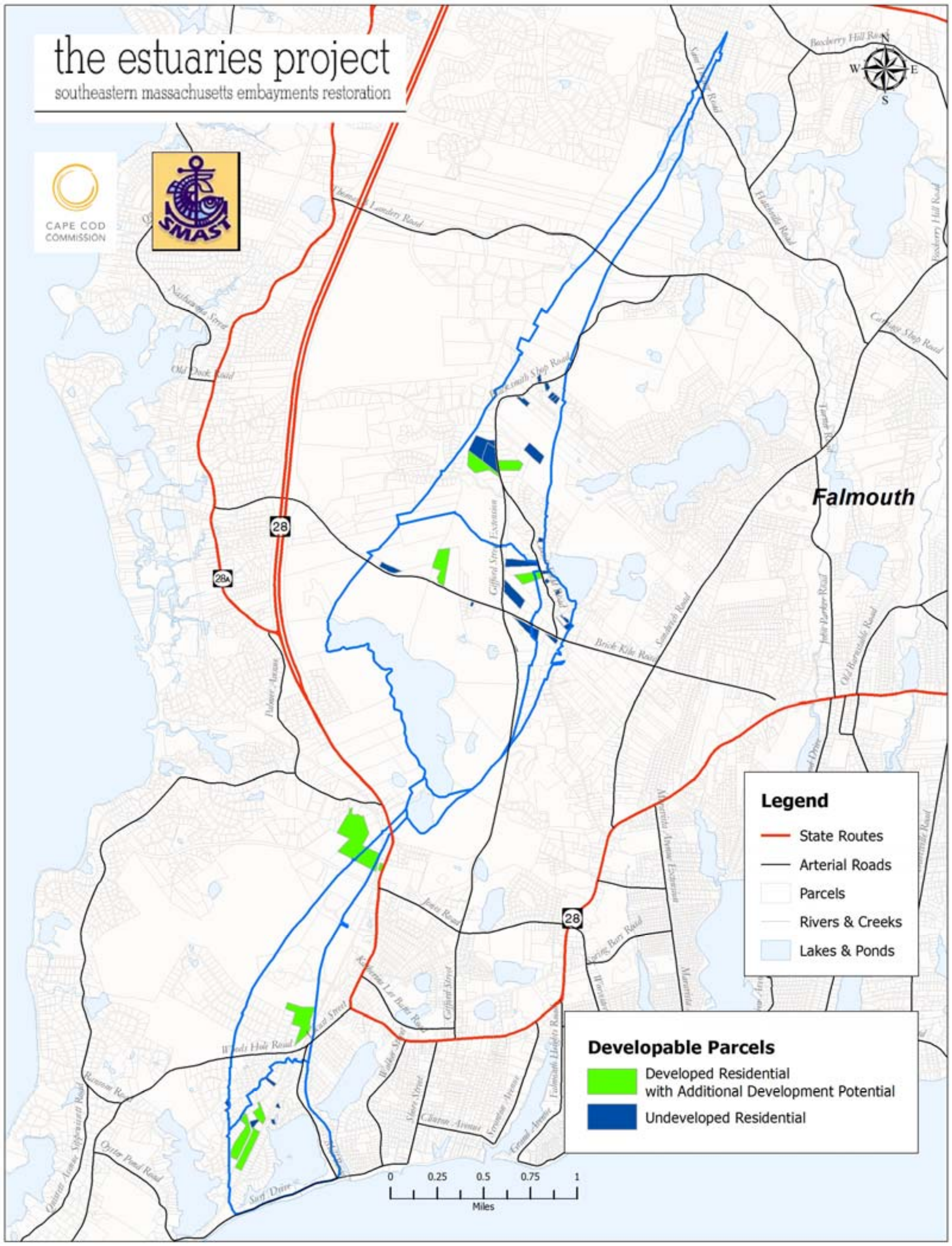

Figure IV-4. Parcels, Parcelized Watersheds, and Developable Parcels in the Salt Pond watersheds. Parcels colored green are developed residential parcels with additional development potential based on current zoning, while parcel colored blue are undeveloped residential parcels classified as developable by the town assessor. The parcelized watersheds are drawn to minimize the division of properties for management purposes while achieving a match of area with the modeled watersheds of 2% or less. Developable parcels are based on town assessor classifications and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. All buildout results were reviewed with town staff and modified based on comments (personal communication, Brian Currie, Town of Falmouth, 4/13).

Following the assignment of all parcels to appropriate subwatersheds, subwatershed modules were generated to determine nitrogen loads for each of the 5 sub-watersheds contributing to the Salt Pond Estuary. These subwatershed modules summarize, among other things: water use, parcel area, parcel frequency by land use category, private wells, and road area. All relevant nitrogen loading data is assigned to each subwatershed. Individual sub-watershed information is then integrated to create the Salt Pond Watershed Nitrogen Loading summary module with summaries for each of the individual 5 subwatersheds and the total for the estuary system. The subwatersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to help focus development of nitrogen management alternatives. Within the Salt Pond study area, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), turf fertilizers, impervious surfaces, direct atmospheric deposition to water surfaces, the solid waste site (landfill), and recharge within natural areas (Table IV-3). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-5). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model.

One of these attenuation adjustments occurs in the freshwater ponds. Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond is used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds unless modified by measured streamflow. The apportionment is based on the percentage of discharging shoreline bordering each downgradient sub-watershed. In the Salt Pond study area, there are two ponds with delineated subwatersheds: Long Pond and Grews Pond. Both of these have groundwater inputs and outputs, but their outputs need to account for the water and nitrogen withdrawn by the use of Long Pond as a municipal drinking water supply source.

Table IV-3. Salt Pond Watershed Nitrogen Loads, with partitioning by subwatershed. Attenuated nitrogen loads shown below are based on assigned attenuation factors assigned to upgradient freshwater ponds. Pond attenuation factors were assigned a standard MEP nitrogen attenuation rate of 50% based on water quality monitoring from the Cape Cod Pond and Lake Stewards (PALS) program. Long Pond nitrogen load is corrected for municipal drinking water withdrawal; 17% of the watershed nitrogen load remains after the drinking water withdrawal. The remaining nitrogen load for Long Pond is assigned to downgradient resources, like Grews Pond, based on the portion of the downgradient shoreline that is captured by the downgradient subwatersheds. All nitrogen loads are kg N yr^{-1}.

		Salt Pond N Loads by Input (kg/y):							% of Pond Outflow	Present N Loads			Buildout N Loads		
Watershed Name	Watershed ID#	Wastewater	Landfill	Turf Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Salt Pond System		1,273	11	118	280	304	62	128		2,048		2,021	2,176		2,148
Salt Pond LT10	1	423	-	57	134	-	20	84		634	-	634	719	-	719
Salt Pond GT10	2	843	-	60	143	-	40	42		1,087	-	1,087	1,129	-	1,129
Grews Pond TOTAL	GP	7	11	1	3	17	2	2	20%	39	-	13	41	-	14
Salt Pond Estuary Surface						288				288	-	288	288	-	288
Grews Pond TOTAL	GP	34	54	3	13	83	8	10		195	50%	65	205	50%	67
Grews Pond	3	-	-	-	4	64	3	0		71	-	71	71	-	71
Long Pond TOTAL	LP	34	54	3	9	19	5	10	13%	124	-	59	134	-	64
remaining after pumped out									17%						
Long Pond TOTAL	LP	1,505	2,365	113	396	841	241	444		5,461	50%	2,605	5,905	50%	2,814
Long Pond LT10	4	338	-	14	117	695	103	138		1,267	-	1,267	1,404	-	1,404
Long Pond GT10	5	898	2,365	83	235	22	91	253		3,694	-	3,694	3,947	-	3,947
Mares Pond (GGB)	MP	268	-	16	44	125	47	54	50%	500	50%	250	554	50%	277

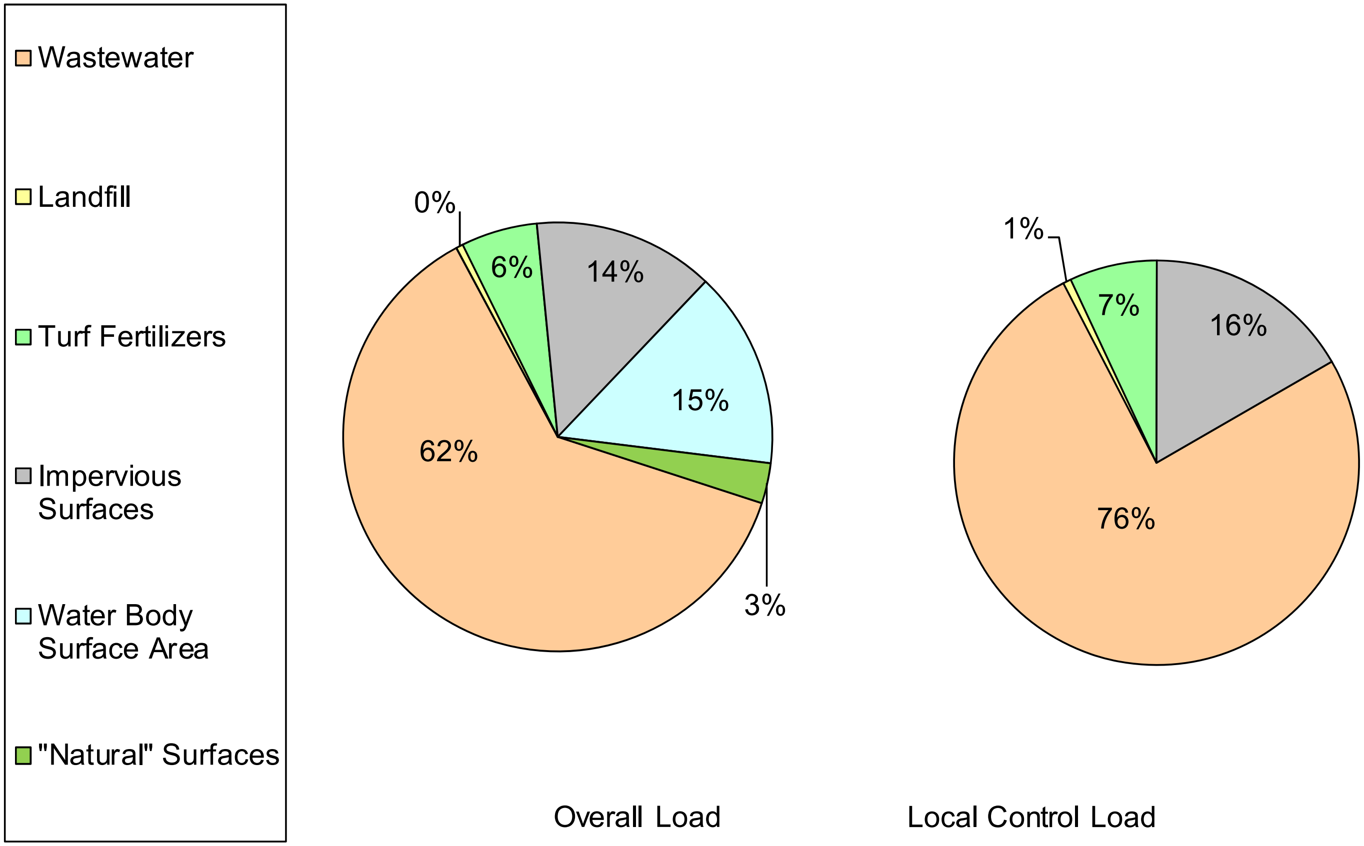

Figure IV-5. Land use-specific unattenuated nitrogen loads (by percent) to the whole Salt Pond watershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control.

Freshwater Pond Nitrogen Loads

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

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

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

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

Many ponds on Cape Cod have been sampled through the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and the initiative of local volunteer pond sampling programs. The PALS Snapshots are regional volunteer one-time pond sampling that happens each year and was initiated and is supported for the last twelve years by SMAST and the Cape Cod Commission, with free laboratory services provided by the Coastal Systems Program Laboratory at SMAST. Sampling protocols developed through the PALS program (Eichner *et al*., 2003) have been used for more extensive pond sampling programs in many communities on Cape Cod. Sampling under these protocols has included field collection of temperature and dissolved oxygen profiles and sampling has generally occurred at standardized depths that provide some evaluation of potential sediment nutrient regeneration. PALS water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. In some cases, enhanced town programs have generated sufficient sampling data that refined pond-specific nitrogen attenuation rates.

There are three freshwater ponds with delineated watersheds that contribute to the Salt Pond watershed: Long Pond, Grews Pond, and Mares Pond. Available bathymetry was found for Grews Pond and Mares Pond (Eichner *et al*., 2003), but Long Pond bathymetry does not appear to be available. However, none of the ponds has sufficient water quality data collection outside of the MEP to provide a basis for an alternative nitrogen attenuation rate. Review of PALS data shows that none of the ponds were ever sampled during the twelve years of PALS Snapshots and Town of Falmouth staff review of available water quality data found no nitrogen data of Long Pond (personal communication, Jerry Potamis, September 2012). Since these ponds have insufficient pond-specific data, they were assigned the standard MEP pond 50% nitrogen attenuation rate. More refined evaluation of these ponds would offer the opportunity to refine this attenuation rate and allow for better targeting of watershed nitrogen management options. As noted in Table IV-3 and Table III-1, only small portions of the ponds discharge into the main subwatersheds to Salt Pond, so only comparatively small nitrogen loads are added to the Salt Pond Estuary from the ponds within its watershed.

Buildout

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

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

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

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

Following the completion of the initial buildout assessment for the Salt Pond watersheds, MEP staff reviewed and modified the initial results with Brian Currie, Falmouth Town Planner in April 2013. Suggested changes from town staff review were incorporated into the final buildout for Salt Pond.

All the parcels with additional buildout potential within the Salt Pond watershed are shown in Figure IV-4. Each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for wastewater and impervious surfaces. Residential additions also include lawn fertilizer nitrogen additions. All wastewater loads are assumed to come from standard on-site septic systems. Cumulative unattenuated buildout loads from all subwatersheds are indicated in a separate column in Table IV-3. Buildout additions within the Salt Pond subwatersheds will increase the unattenuated system-wide nitrogen loading rate by 6%.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out or sewering analysis) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the watershed of the Salt Pond System were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment, the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport is exclusively through groundwater in sandy outwash aquifers as a result of the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. In the watershed to Salt Pond, unlike most watersheds in southeastern Massachusetts, nitrogen does not pass through a surface water ecosystem on its path to the adjacent embayment. Nitrogen transport through surface water systems, unlike aerobic aquifers, supports the conditions for nitrogen retention and denitrification. As there were no significant streams within the Salt Pond watershed, the watershed loading approach considered that nitrogen reaching the water table was transported without attenuation through the groundwater system until discharge either to Long Pond or Grews Pond or directly to the Salt Pond Estuary.

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

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

IV.3.1 Sediment-Water column Exchange of Nitrogen

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

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within enclosed tributary basins, particularly if they are deeper than the adjacent embayment. To some extent, the settling characteristics can be evaluated by observation of the grain-size and organic content of sediments within an estuary.

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bio-available nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with deep depositional basins (e.g. Salt Pond, Oyster Pond, Seine Pond, Swan Pond) 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 or Sesachacha Pond on the Island of Nantucket). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, for example in the margins of the main basin to Lewis Bay (Town of Barnstable, Cape Cod). In contrast, most embayments show low rates of nitrogen release throughout much of a basin's area and, in regions of high deposition, typically support anoxic sediments with high release rates during summer months. The consequence of high deposition rates is that the basin sediments are unconsolidated, organic rich and sulfidic in nature (MEP field observations).

Failure to account for site-specific nitrogen balance of sediments and its spatial variation from the tidal creeks and embayment basins will result in significant errors in determination of the threshold nitrogen loading to the Salt Pond system. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for Determining Sediment-Water column Nitrogen Exchange

For the Salt Pond Estuary, 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. A total of 12 cores were collected from 11 sites (Figure IV-6) in July-August 2007, focusing on obtaining a spatial distribution that would be representative of nutrient fluxes throughout the system. Duplicate cores were taken at one site. 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 established at the Town of Falmouth Harbormaster facility in the adjacent Falmouth Inner Harbor system. 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 sampling locations and numbers of cores collected are listed below. The spatial distribution of the stations is presented in Figure IV-6.

Salt Pond System Benthic Nutrient Regeneration Cores

- SP-1 1 core (Inlet Basin)
- SP-2 1 core (Inlet Basin)
- SP-3 1 core (Main Basin-Shallow)
- SP-4 1 core (Main Basin-Shallow)
- SP-5 1 core (Main Basin-Shallow)
- SP-6 1 core (Main Basin-Shallow)
- SP-7 1 core (Main Basin-Deep)
- SP-8 1 core (Main Basin-Shallow)
- SP-9/10 2 cores (Main Basin-Shallow)
- SP-11 1 core (Main Basin-Shallow)
- SP-12 1 core (Main Basin-Deep)

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

Figure IV-6. Salt Pond Estuary. Locations (yellow symbols) of sediment sample collection for determination of nitrogen regeneration rates and/or infaunal community analysis. Numbers are for reference to station listing above. Not all stations were sampled for both parameters.

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 (Town of Falmouth Harbormaster Facility, Falmouth Inner Harbor), the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and orthophosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining samples frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

Chemical analyses were performed by the Coastal Systems Analytical Facility at the School for Marine Science and Technology (SMAST) at the University of Massachusetts in New Bedford, MA [*508-910-6325*]. The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

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

In order to model the nitrogen distribution within an embayment it is important to be able to account for the net nitrogen flux from the sediments within each part of each system. This requires that an estimate of the particulate input and nitrate uptake be obtained for comparison to the rate of nitrogen release. Only sediments with a net release of nitrogen contribute a true additional nitrogen load to the overlying waters, while those with a net input to the sediments serve as an "in embayment" attenuation mechanism for nitrogen. Sediments within deep depositional basins or those in systems with low flushing rates tend to show a low or negative net release of 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-7).

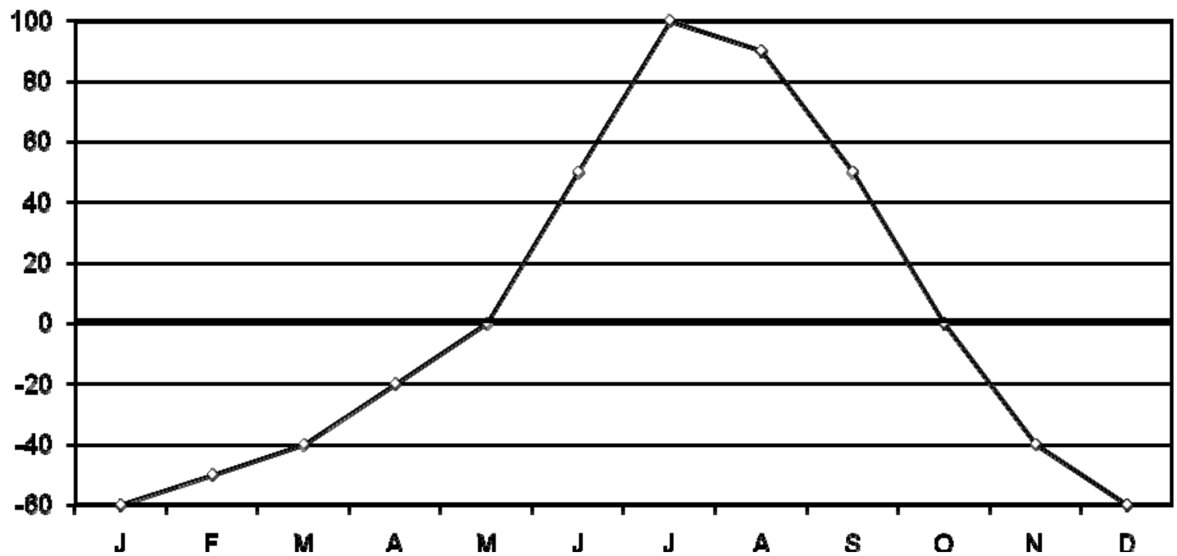

Figure IV-7. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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 Nitrogen Release by Standard Core Approach: Sediment sampling was conducted throughout the Salt Pond system to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-6). Generally, the distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content, as well as sediment type and an analysis of each site's tidal flow velocities. As expected flow velocities are generally low throughout the Salt Pond system with the exception of the area in the immediate vicinity of the small inlet (culvert under Surf Drive on the southwestern end of the pond) that connects Salt Pond to Vineyard Sound. 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 at 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). Salt Pond is operating as a marine (20 ppt) kettle pond with low tidal exchange rates and relatively deep water compared to other estuaries in southeastern Massachusetts. The low tidal exchange results in very low velocities and the deep water increases the time required to settle particles to the bottom. The average water depth in Salt Pond is ~3m, compared to <1 meter in other estuaries along Falmouth's south coast. Settling rates of one half to two thirds of the rates in other large, but shallower, basins were used. Adjusting the measured sediment releases is essential in order to not over-estimate the sediment nitrogen source and to account for those sediment areas that are net nitrogen sinks for the aquatic system. This approach was validated in outer Cape Cod embayments (Town of Chatham) by examining the relative fraction of the sediment carbon turnover (total sediment metabolism) that 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 other 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.

Rates of net nitrogen release or uptake from the sediments within the Salt Pond embayment system were comparable to other embayments of similar depth and configuration in southeastern Massachusetts. There was a clear pattern of sediment N flux. The main basin and small basin near the tidal inlet appear to be depositional with depths in the central region of the main basin in excess of 5 meters (see bathymetric map, Section V). Consistent with the morphology of the basins, the main basin sediments consist mainly of anoxic unconsolidated sulfidic muds, with the shallow margin sediments ranging from consolidated fine sand behind the barrier beach to organic rich anoxic muds with a thin algal mat and oxidized surface layer in the landward margin to the deep basin.

The deep waters of Salt Pond stratify and show periodic hypoxia/anoxia during summer. Nitrogen transformations in these deep sediments was determined not to have impact on surface water nitrogen levels during stratification. Vertical mixing in Salt Pond during summer is predominantly driven by wind. Mixing by tides, so important to many estuarine systems, is relatively unimportant due to the restricted nature of the Salt Pond exchange with Vineyard Sound. Nitrogen settling below the mixed surface layer of Salt Pond enters sediments overlain by unmixed bottom waters. Nitrogen release from these sediments must exit the sediment surface through diffusion, as there are typically no animals to enhance release through their activities (bioturbation). Once in the bottom waters, the released nutrients move upward (and horizontally) by diffusion. Evidence of this diffusion cell in bottom waters can be seen in dissolved sulfide profiles (Wakeham et al. 1984). The result is nutrients build up in bottom waters during summer, with release in fall/winter storm driven mixing. The effect of this "capping" of about 20% of the bottom sediments of the pond, is that nutrient release in these regions does not support a mixed layer nitrogen balance during summer. Therefore, for the modeling of summer water quality (Section VI) the measured sediment nutrient release was distributed over the >80% of the pond bottom that is overlain by a mixed aerobic watercolumn. The rates in the Salt Pond system (-13.0 to 11.8 mg N m^{-2} d^{-1}) were similar to other enclosed depositional basins, for example in the Town of Falmouth, Great Pond and Perch Pond (-16.4 and –20.2 mg N m^{-2} d^{-1}). Rates also agreed well with other highly restricted basins like Swan Pond (-8.0 m^{-2} d^{-1}), Seine Pond (-16.9 m^{-2} d^{-1}), Prince Cove (10.3 m^{-2} d^{-1}), Trapps Pond (9.2 m^{-2} d^{-1}), Town Cove (-18.8 to 64.4 m^{-2} d^{-1}). Similarly, in the nearby Vineyard Sound estuary (Popponessett Bay), sediment regeneration ranged from -17 to 85 mg N m^{-2} d^{-1}.

The benthic regeneration rate from the sediment incubations was also consistent with changes in the surface waters of Salt Pond during the summer of 2007. The physical structure of Salt Pond provides for a whole system approach for determining sediment net nutrient release. Since Salt Pond has a low tidal exchange relative to its daily inputs, it is possible to approximate the sediment nutrient release from changes in the mixed layer nitrogen mass. Over short periods (June 21 – July 31), Salt Pond's nitrogen inputs from its watershed and atmosphere and its outputs to Vineyard Sound were assumed to be constant. During this period losses or gains of nitrogen from the mixed layer can be attributed to net release from the sediments or net deposition. This approach provides an estimate of nitrogen release for the whole of Salt Pond for comparison to the sediment core measurements. Based upon average total nitrogen information from the SMAST water quality monitoring project, for June through July collected in 2007, there appears to be an increase in nitrogen in pond waters (Figure IV-8), which equates to a build-up in the mixed layer of 0.010 mg L^{-1} d^{-1}. This compares well with the measurements from the shallow margin sediments that are in contact with the mixed layer, which yielded a net projected increase in the mixed layer TN of 0.006 mg L^{-1} d^{-1}. From these results it appears that the 2 approaches are in good agreement. Given the uncertainties in using the whole system flux calculations in a tidal basin, the core incubation measurements were the primary focus for the water quality modeling (Section VI).

Overall, the sediment nitrogen flux within Salt Pond appears to be in balance with the overlying waters and the nitrogen flux rates are consistent with the level of nitrogen loading to this system, the basin morphology and tidal exchange. Net nitrogen flux rates for use in the water quality modeling effort for the Salt Pond Embayment System (Chapter VI) are presented in Table IV-4.

Table IV-4. Rates of net nitrogen return from sediments to the overlying waters of the Salt Pond Estuary. 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})			SP- # I.D. *
	Mean	S.E.	N	
Salt Pond Embayment System				
Inlet Basin	**-13.0**	**1.0**	**2**	**1, 2**
Shallow Main Basin	**10.2**	**9.2**	**8**	**3,4,5,6,8,9,10,11**
Main Basin	**11.8**	**40.1**	**2**	**7, 12**

* Station numbers refer to Figure IV-5.

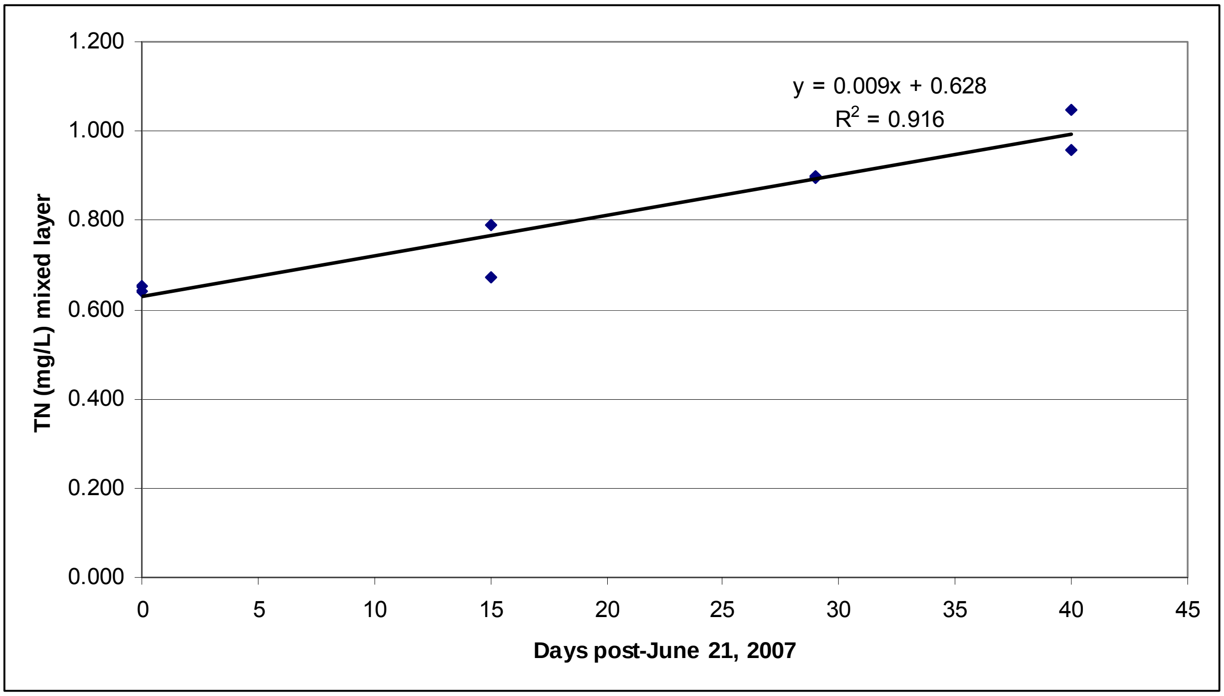

Figure IV-8. Change in average TN concentration (mg/L) in mixed surface waters of Salt Pond during summer 2007.

V. HYDRODYNAMIC MODELING

V.1. INTRODUCTION

This section summarizes the field data collection efforts and the development of a hydrodynamic model for the Salt Pond estuary system (Figure V-1). For this system, the final calibrated model offers an understanding of water movement through the estuary, and provides the first step towards evaluating water quality, as well as a tool for later determining nitrogen loading "thresholds". Tidal flushing information is utilized as the basis for a quantitative evaluation of water quality. Nutrient loading data combined with measured environmental parameters within the Salt Pond area 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, as well as determining the likely positive impacts of various alternatives for improving overall estuarine health, enabling the bordering residents to understand how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

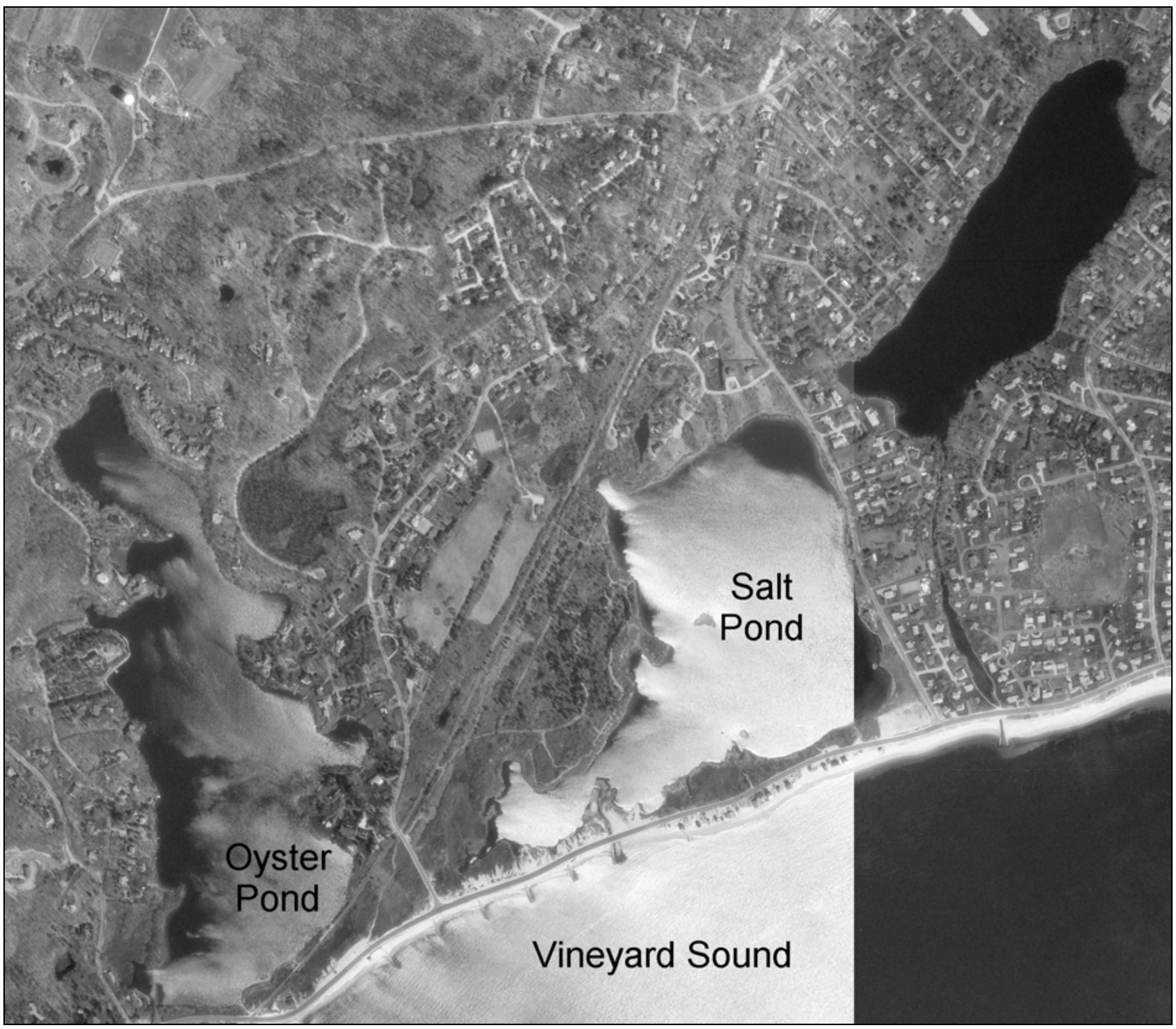

Figure V-1. Aerial photograph of the Salt Pond.

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.

Estuarine water quality is dependent upon nutrient and pollutant loading and the processes that help flush nutrients and pollutants from the estuary (e.g., tides and biological processes). Relatively low nutrient and pollutant loading and efficient tidal flushing are indicators of high water quality. The ability of an estuary to flush nutrients and pollutants is proportional to the volume of water exchanged with a high quality water body (i.e. Vineyard Sound). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For the Salt Pond system, the most important parameters are the tide range (affected by the singular culvert passing under Surf Drive) along with the shape, length and depth of the estuary.

Shallow coastal embayments 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 of the surrounding area 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.

The Salt Pond system (Figure V-1) located entirely within the Town of Falmouth is a tidally influenced embayment, with a south facing inlet to Vineyard Sound on the southern shoreline of Falmouth, MA. Since the water elevation difference between Vineyard Sound and the Pond is the primary driving force for tidal exchange, the local tide range limits the volume of water flushed during a tidal cycle and must be carefully considered.

Circulation in the Salt Pond estuarine system was simulated using the RMA-2 numerical hydrodynamic model. The aerial photograph in Figure V-1 shows the general study area. Salt Pond has only one main embayment that extends northward from the coastline. The pond covers approximately 62 acres. The system is shallow, with a mean depth of 2.0 feet. The deepest section of Salt Pond is in the northeast corner of the pond, with a mean depth of 16.0 feet.

Circulation in the system is dominated by tidal exchange with Vineyard Sound. From measurements made during the course of the hydrodynamic analysis of the system, the average tide range in Vineyard Sound is approximately 2.0 feet. The flow restrictions caused by inlet channel, produces major reductions in the tide range from Vineyard Sound. The reductions are on the order of 1.6-1.8 feet. Even with the significant reduction in tidal amplitude, the pond exhibits a small but clear tidal signal.

This hydrodynamic modeling effort proceeded in two steps. Step one involved collection of bathymetry and tide data in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the analysis. The bathymetry survey of Salt Pond was performed to determine the variation of embayment and channel depths throughout the system. This survey addressed the previous lack of adequate bathymetry data for this area. In addition to the bathymetry survey, tides were recorded at two locations within the system for 29 days. These tide data were necessary to run and calibrate the hydrodynamic model of the system.

A numerical hydrodynamic model of the system was developed in the second step of the overall hydrodynamic analysis of the system. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from a gage located offshore of the entrance to Salt Pond was used to define the open boundary conditions that drives the exchange of water through the entrance of Salt Pond. Data from the tide gage within the pond was 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 Salt Pond was used to compute the flushing rate for the system. Though water quality in an embayment cannot be directly inferred by use of the computed flushing rate alone, it can serve as a useful indicator of an embayments flushing performance relative to other systems. 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. FIELD DATA COLLECTION AND ANALYSIS

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

Bathymetry data was collected throughout the pond since the historical bathymetry data lacked accuracy and/or detail necessary for evaluation of tidal hydrodynamics. Tidal elevation measurements were used for both forcing conditions and to evaluate tidal attenuation through the inlet into the system. Figure V-2 shows the location of the tide gages.

V.2.1 Bathymetry

Bathymetry, or depth, of Salt Pond was measured during field a survey in August 2003. The survey was completed using a small vessel equipped with a precision fathometer interfaced to a differential GPS receiver. The fathometer has a depth resolution of approximately 0.1 foot and the differential GPS provides x-y position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder and GPS were logged to a data recorder. Within the inlet, standard rod and level survey methods where used to determine elevations due to the shallow water depths.

GPS positions and echo sounder measurements were merged to produce data sets consisting of water depth as a function of x-y horizontal position (in Massachusetts Mainland State Plane, 1983). The data were combined with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NGVD 1929 vertical datum (NGVD29). The resulting xyz files were input to mapping software to calculate depth contours for the system shown in

Figure V-3. The 1942 NOAA bathymetry dataset was used to supplement the bathymetry data collected in 2003. The bathymetry offshore of Salt Pond is stable and does not experience significant shoaling, sand wave migration, or other physical changes that would make the dataset inappropriate for use in the offshore regions of the numerical model which reside beyond the areas of focus as part of this MEP study.

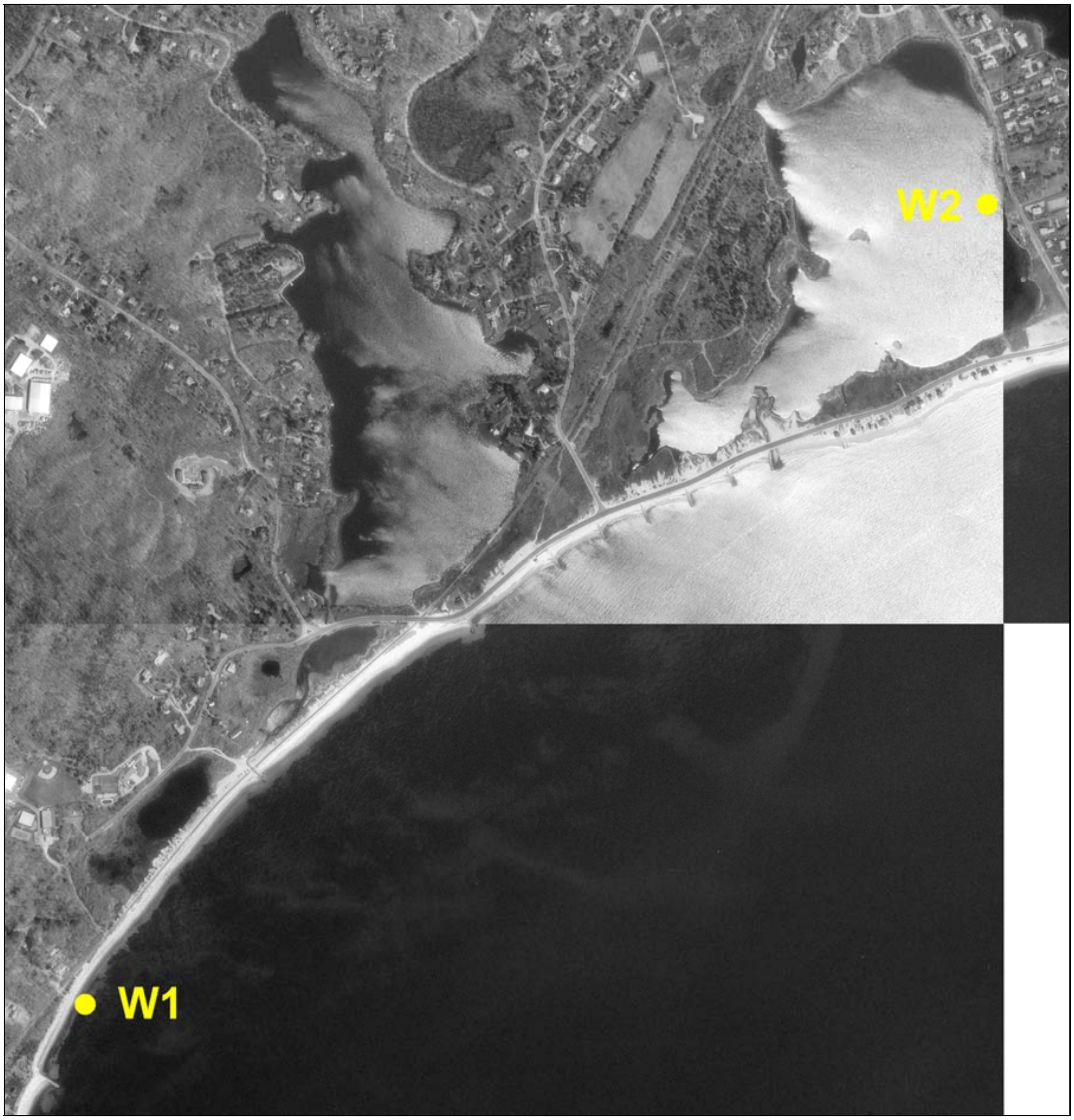

Figure V-2. Salt Pond with tide gage locations labeled as W1 and W2.

V.2.2 Water Elevation Measurements and Analysis

Changes in water surface elevation were measured using internal recording tide gages. These tide gages were installed on fixed platforms (such as pier pilings or screw anchors) to record changes in water pressure over time. Variations in the water surface can be due to tides, wind set-up, or other low frequency oscillations of the sea surface. The tide gages were installed in 2 locations in Salt Pond (Figure V-2) on July 25, 2003 and recovered on

September 11, 2003. Data records span at least 29 days to yield an adequate time period for resolving the primary tidal constituents.

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

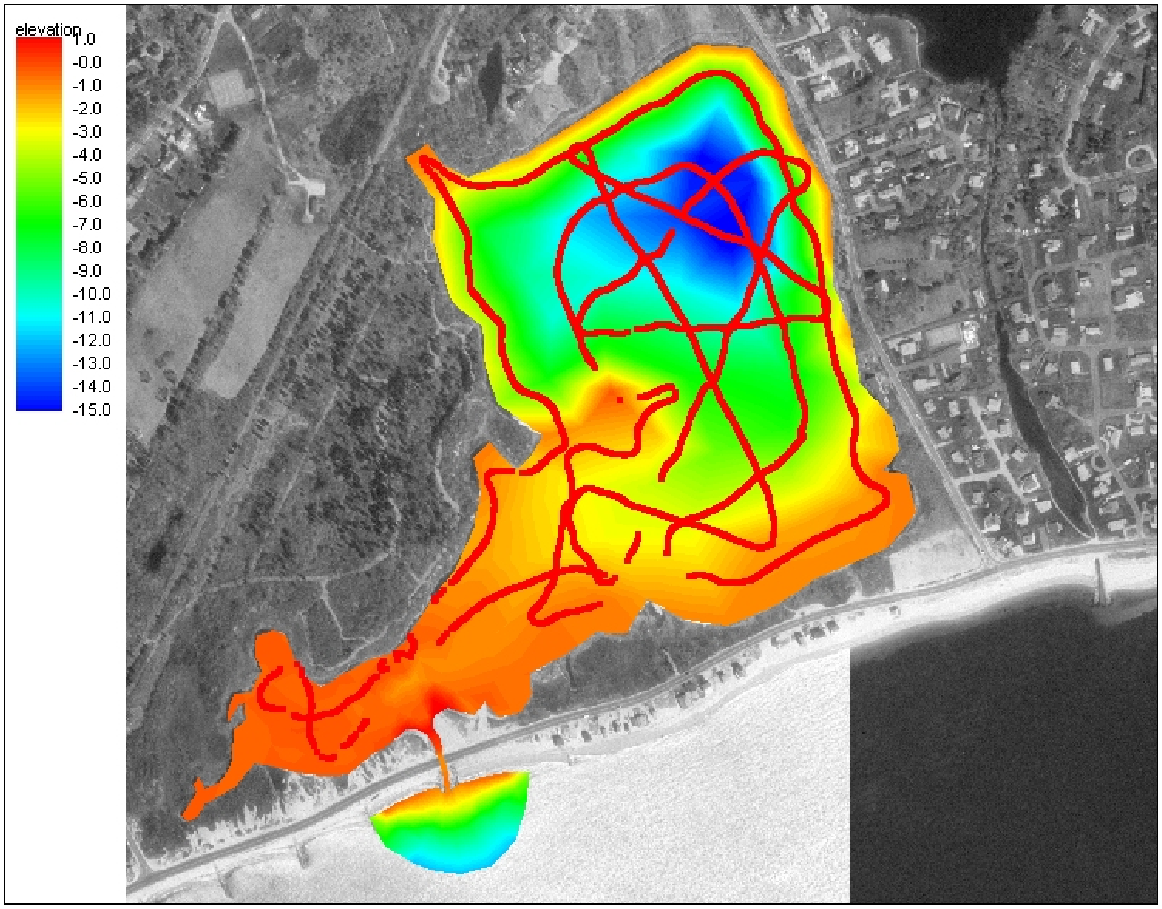

Figure V-3. Depth contour plots of the numerical grid for the Salt Pond at 0.25-foot contour intervals relative to NGVD29.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric readings were obtained from the NOAA buoy in Buzzards Bay (site BUZM3), interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in water pressure above the instrument. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (for example, feet of water above the tide gage). Several of the sensors were surveyed into local benchmarks to provide vertical rectification of the water level; these survey values were used to adjust the water surface to a known vertical datum. The result from each gage is a time series representing the variations in water surface elevation relative to NGVD29. Figure V-4 present the water levels at each gage location.

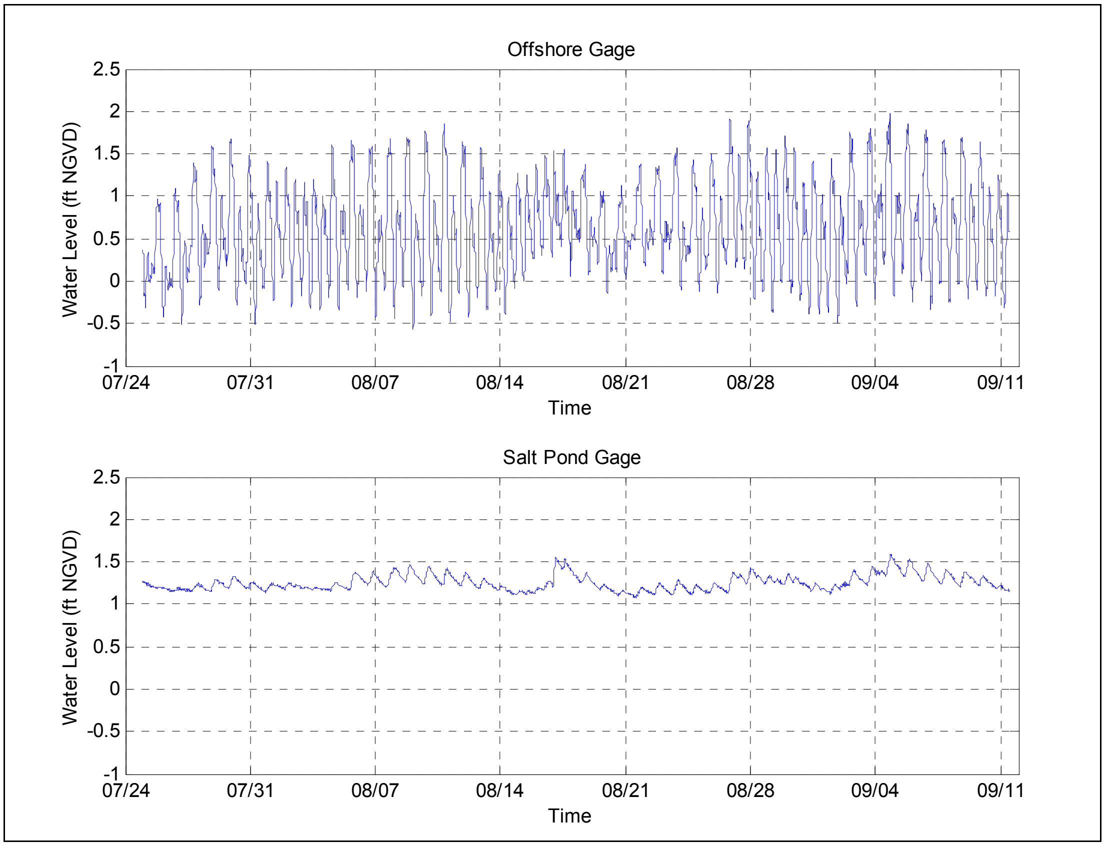

Figure V-4. Tidal elevation observations for offshore Salt Pond (location W1), and within Salt Pond (location W2).

Figure V-4 shows the tidal elevation for the period July 25 through September 11, 2003 at offshore gage and in Salt Pond. The offshore curve has a predominant 12.42-hour variation around the lunar semi-diurnal (twice-a-day), or M_2, tidal constituent. Modulation of the lunar and solar tides, results in the spring-neap fortnightly cycle, typically evidence by a gradual increase and decrease in tide range. The neap (or minimum) tide range was approximately 1.2 feet, occurring August 21. The spring (maximum) tide range was approximately 2.3 feet, and occurred on August 9. The Salt Pond curve shows the significant influence the inlet has upon the tidal signal within the pond. The flow restrictions reduce the tidal signal to a few tenths of a foot.

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

Harmonic analyses were performed on the time series for each gage location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-1 presents the amplitudes of the eight largest tidal constituents. The M_2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 0.51 feet at the offshore gage. The range of the M_2 tide is twice the amplitude, or

1.02 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.25 feet. The N_2 (12.66-hour period) semi-diurnal tide, also contributes significantly to the total tide signal with an amplitude of 0.19 feet. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 is approximately 30% of the amplitude of the M_2 in the offshore gage (about 0.17 feet). The M_6 amplitude is relatively small throughout the system (less than 0.06 feet). The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon. The observed astronomical tide is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency.

Table V-1. Tidal Constituents, Salt Pond July-September 2003

	AMPLITUDE (feet)							
Period (hours)	M2 12.42	M4 6.21	M6 4.14	S2 12.00	N2 12.66	K1 23.93	O1 25.82	Msf 354.61
Offshore	0.511	0.172	0.06	0.113	0.187	0.245	0.235	0.043
Salt Pond	0.021	0.001	0.002	0.004	0.005	0.033	0.030	0.026

Table V-1 also shows how the constituents vary as the tide propagates through the inlet into the pond. The most significant reduction is in the M_2 amplitude between the Vineyard Sound (offshore) gage and Salt Pond. Usually, a portion of the energy lost from the M_2 tide is transferred to higher harmonics, and is observed as an increase in the amplitude of the M_4 and M_6 constituents over the length of the estuary. However, in the Salt Pond system M_2, M_4 and M_6 are all clearly smaller than the amplitudes due to the extreme damping from inlet channel geometry restricting flow and a creating frictional drag through inlet

Table V-2 presents the phase delay of the M_2 tide through the inlet to gage in Salt Pond. Phase delay is another indication of tidal damping resulting from the inlet, and results with a later high tide in Salt Pond (Figure V-5). The greater the frictional effects, the longer the delay between locations. The delay in Salt Pond is 171 minutes. In general, the delays increase with increasing distance from the offshore gage. However in this case the inlet is producing significant damping and hence large phase delay.

Table V-2. M_2 Tidal Attenuation, July-September 2003 (Delay in minutes relative to Vineyard Sound)

Location	Delay (minutes)
Offshore (Vineyard Sound)	--
Salt Pond	171.2

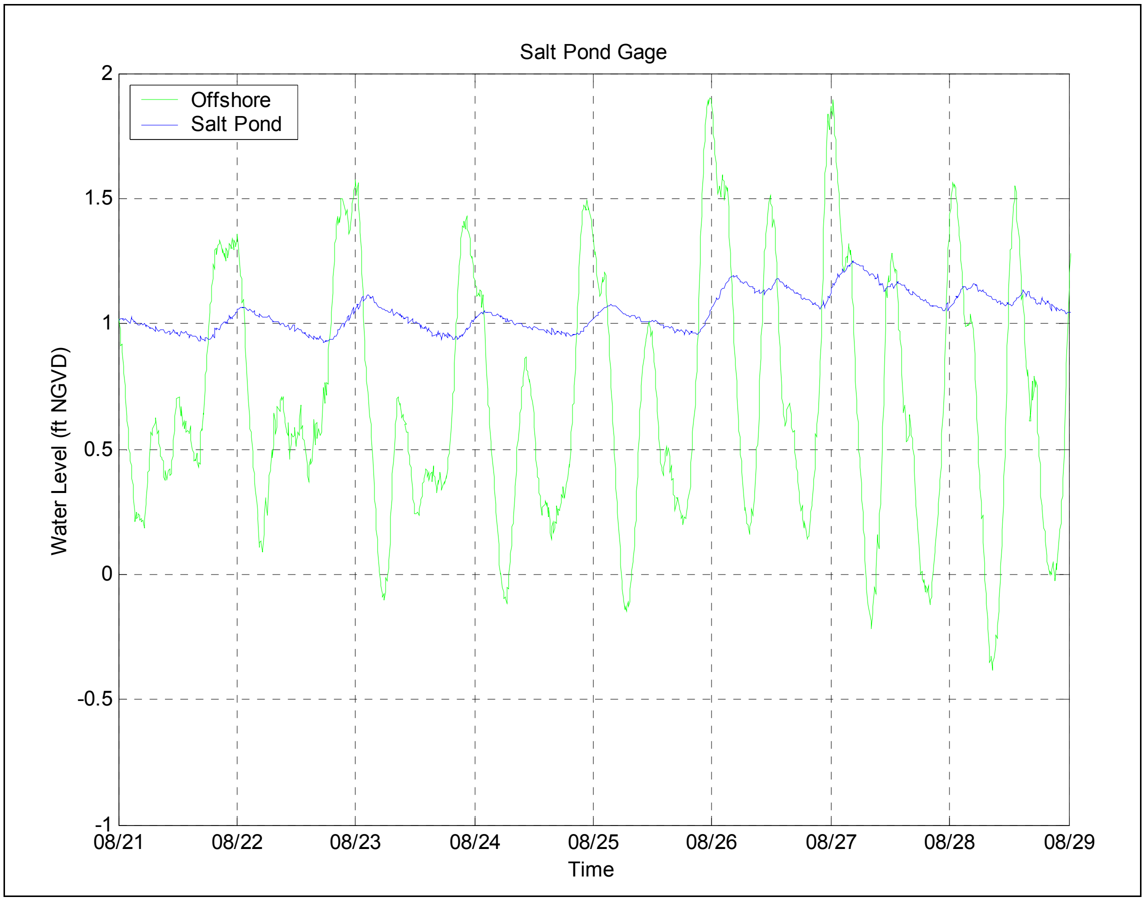

Figure V-5. Comparison of water surface elevation observations for Vineyard Sound (offshore), and in Salt Pond. Damping effects are seen as a decrease in the tidal amplitude, as well as a lag in the time of high and low tides from Vineyard Sound.

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. Vineyard Sound is a relatively shallow semi-enclosed basin, therefore the water surface responds readily to wind-forcing. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large. This analysis calculated the energy (or variance) of the original water elevation time series, and compared these energy values to that of the purely tidal signal (re-created by summing the contributions from the 23 known harmonic constituents). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. The results of this analysis for are presented in Table V-3.

Table V-3. Percentages of Tidal versus Non-Tidal Energy, Salt Pond, 2003				
	Total Variance ($ft^2 \cdot sec$)	Total (%)	Tidal (%)	Non-tidal (%)
Offshore	0.282	100	91.8	7.9
Salt Pond	0.009	100	23.9	76.1

The variability analysis shows that a majority of the change in water surface elevation in Vineyard Sound was due to tidal processes. However, in Salt Pond more than three-quarters of the energy was the result of non-tidal processes. The significant increase in non-tidal energy is due to tidal damping and frictional losses through the inlet. Local effects of wind blowing across the pond surface will also increase the energy of non-tidal processes. The analysis results indicate that tidal signal entering Salt Pond is significantly impacted by the secondary and physical processes, however the pond remains tidal. Figure V-5 illustrates that the water surface within the pond fluctuates daily as a result of the tide in Vineyard Sound, therefore even with the reduced tidal signal, tidal processes have a significant contribution to the hydrodynamics that govern the movement of water in and through the pond.

V.3. HYDRODYNAMIC MODELING

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

V.3.1 Model Theory

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

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

The time dependence of the governing equations is incorporated within the solution technique needed to solve the set of simultaneous equations. This technique is implicit; therefore, unconditionally stable. Once the equations are solved, corrections to the initial estimate of velocity and water elevation are employed, and the equations are re-solved until the convergence criteria 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 finite element grid was generated using the shorelines from 1994 digital aerial photographs from the MassGIS online orthophoto database. More recent aerial photographs are available and were examined as part of the project. However, the horizontal extents of Salt Pond remain stable and unchanged across the catalogue of aerial photographs that are available. The 1994 photographs were selected due contrast between the land/water boundary. These photographs provide a clear boundary which added the process of defining the numerical model extents.

A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of Salt Pond based on the tide gage data collected near the entrance to Salt Pond in Vineyard Sound. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid Generation

The grid generation process was aided by the use of the SMS package. A 1994 digital aerial orthophoto and the bathymetry survey data were imported to SMS, and a finite element grid was generated to represent the embayment and inlet. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of adjoining salt marshes. The bathymetry data was interpolated to the developed finite element mesh of the system. The completed grid consists of 1003 nodes, which describes 312 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -12.0 ft (NGVD 29), along the offshore boundary to Vineyard Sound. In the model grid, a typical marsh plain elevation of +0.2 ft (NGVD 29) was used, based on spot surveys of the marsh. The completed grid mesh of Salt Pond is shown in Figure V-6.

The finite element grid for each system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties in Salt Pond. Areas of marsh were included in the model because they represent a key portion of the estuary along the northern extents of Salt Pond, and may have an effect on hydrodynamics within the system. Fine grid resolution was required to simulate the channel constrictions and the inlet structures which have a significant impact the estuarine hydrodynamics. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary.

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

V.3.2.2 Boundary Condition Specification

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

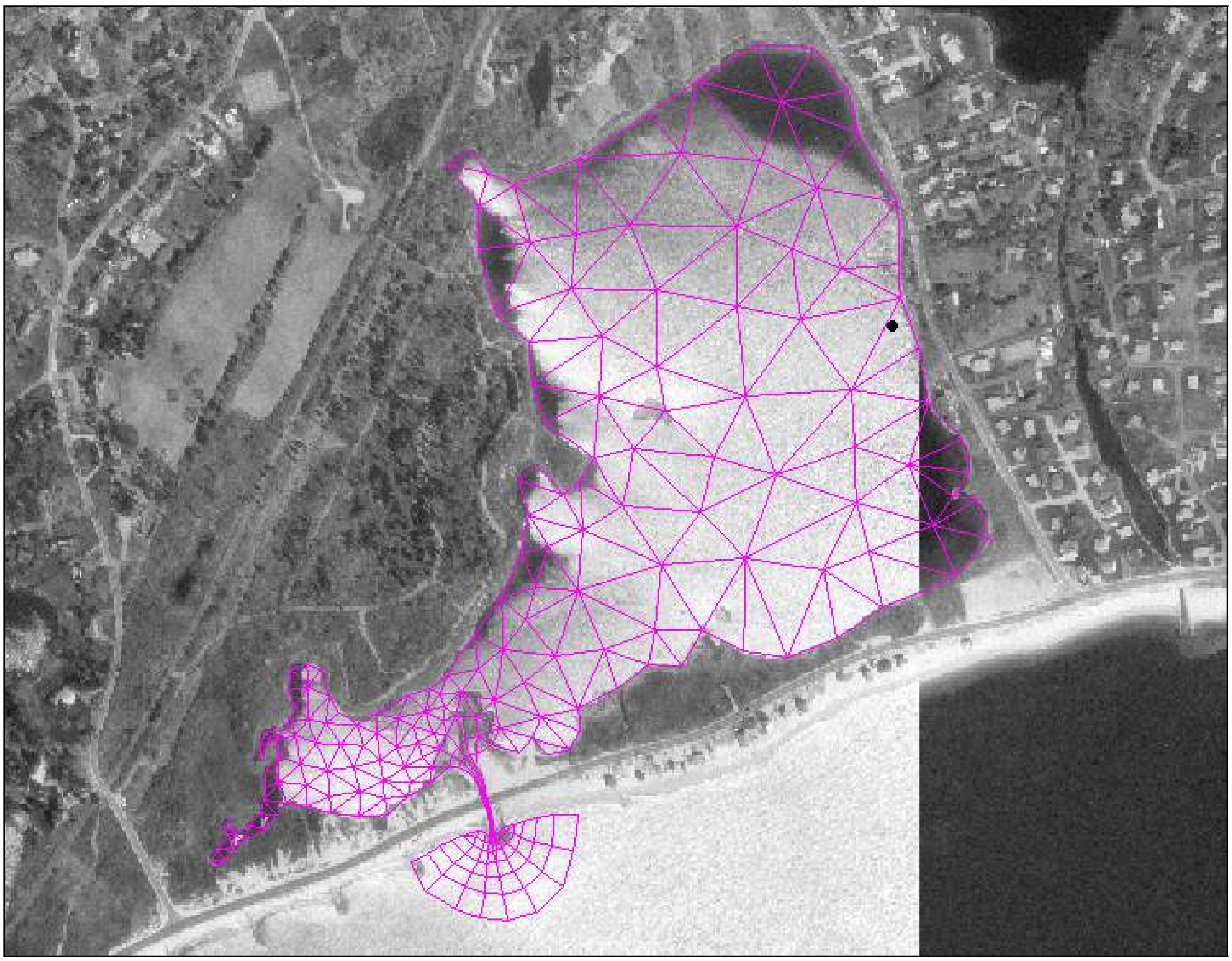

Figure V-6. Plot of hydrodynamic model grid mesh for Salt Pond.

V.3.2.3 Calibration

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

Calibration of the hydrodynamic model requires a close match between the modeled and measured tides in the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, an approximate seven-day period (14 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.3.2. The seven-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 time (phase) lag and height damping of dominant tidal constituents

The calibration was performed for a seven-day period beginning August 5, 2003 at 1800 EDT. This representative time period included the spring tide range of conditions, where the tide range and tidal currents are greatest.

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

V.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 measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficients between 0.020 and 0.080 were specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels and marsh plains with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy/silty bottom found in Salt Pond, versus the rock lined channel in the inlet, which provides greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-4. The extents of each material type are shown in Figure V-7.

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 50 and 80 lb-sec/ft^2.

Table V-4. Manning's Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-7.

System Embayment	Bottom Friction
Offshore	0.025
Culvert	0.039
Inlet Channel	0.035
Salt Pond	0.025
Marsh	0.035

Figure V-7. Hydrodynamic model grid material properties. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.3.2.3.3 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model predictions for the first TDR deployment was achieved using the aforementioned values for friction and turbulent exchange. Figure V-8 illustrates the seven-day calibration simulation for Salt Pond. Modeled (solid line) and measured (dotted line) tides are illustrate the tide record.

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, 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-5 for the calibration period differ from those in Table V-2 because constituents were computed for only the seven-day section of the 26-days represented in Table V-2. Table V-5 compares tidal constituent height and phase for modeled and measured tides at the TDR locations.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.01 ft in Salt Pond, which is of the same order of the accuracy of the tide gages (0.032 ft). Time lag errors were typically less than the time increment resolved by the model (0.10 hours or 10 minutes), indicating good agreement between the model and data.

Table V-5. Tidal constituents for measured water level data and calibrated model output for Salt Pond.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (rad)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Salt Pond	0.034	0.003	0.003	0.069	1.1	0.9
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (rad)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Salt Pond	0.025	0.002	0.003	0.067	1.0	0.7
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Salt Pond	0.00	-0.01	0.00	0.00	-5.6	-16.6

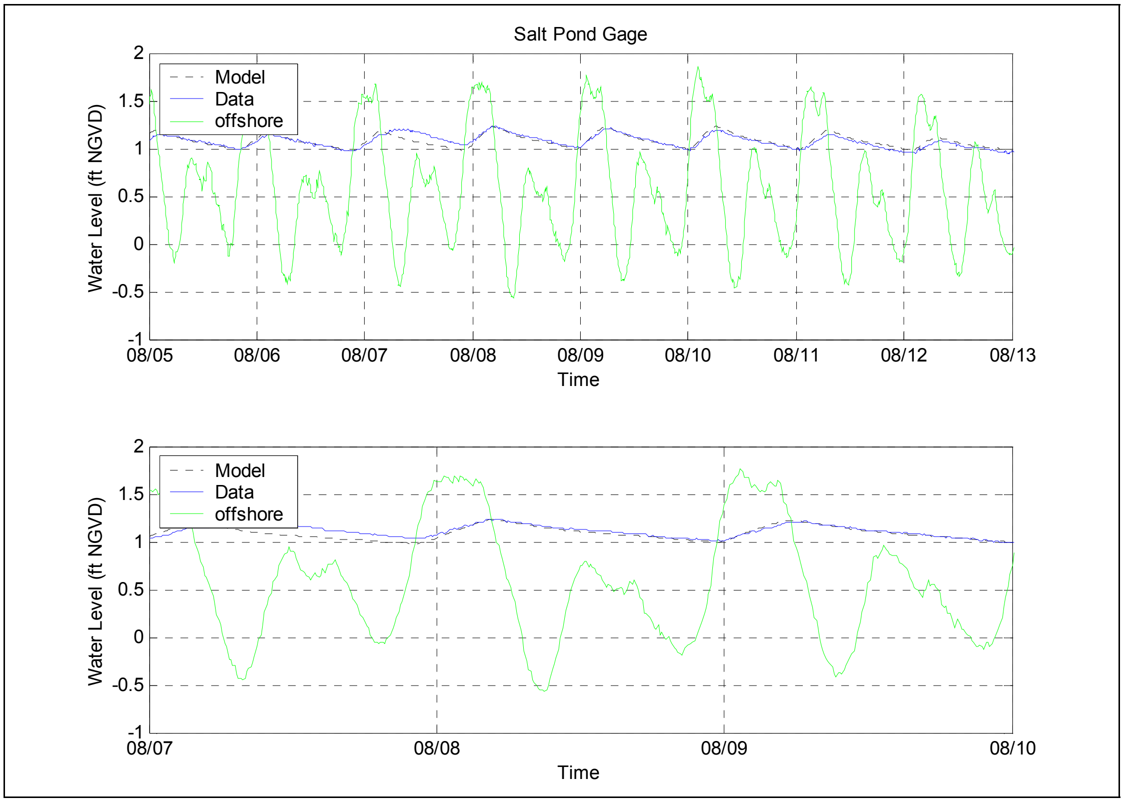

Figure V-8. Comparison of model output and measured tides for the TDR gage in Salt Pond.

The hydrodynamic model was then verified over a separate seven-day period, beginning August 21, 2003 at 2100 EDT, to ensure the coefficients established during calibration would accurately represent the estuary process beyond the calibration period. The verification period represented the variation from neap tide range of conditions toward spring tide conditions. The verification resulted in excellent agreement between modeled and measured tides. Figure V-9 show the comparison of modeled and measured tides, while Table V-6 shows the constituent agreement.

Table V-6. Tidal constituents for measured water level data and verified model output for Salt Pond.

Model verification run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Salt Pond	0.029	0.004	0.002	0.044	-0.5	2.9
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Salt Pond	0.025	0.003	0.002	0.041	-0.4	3.0
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Salt Pond	0.00	0.00	0.00	0.00	16.7	4.8

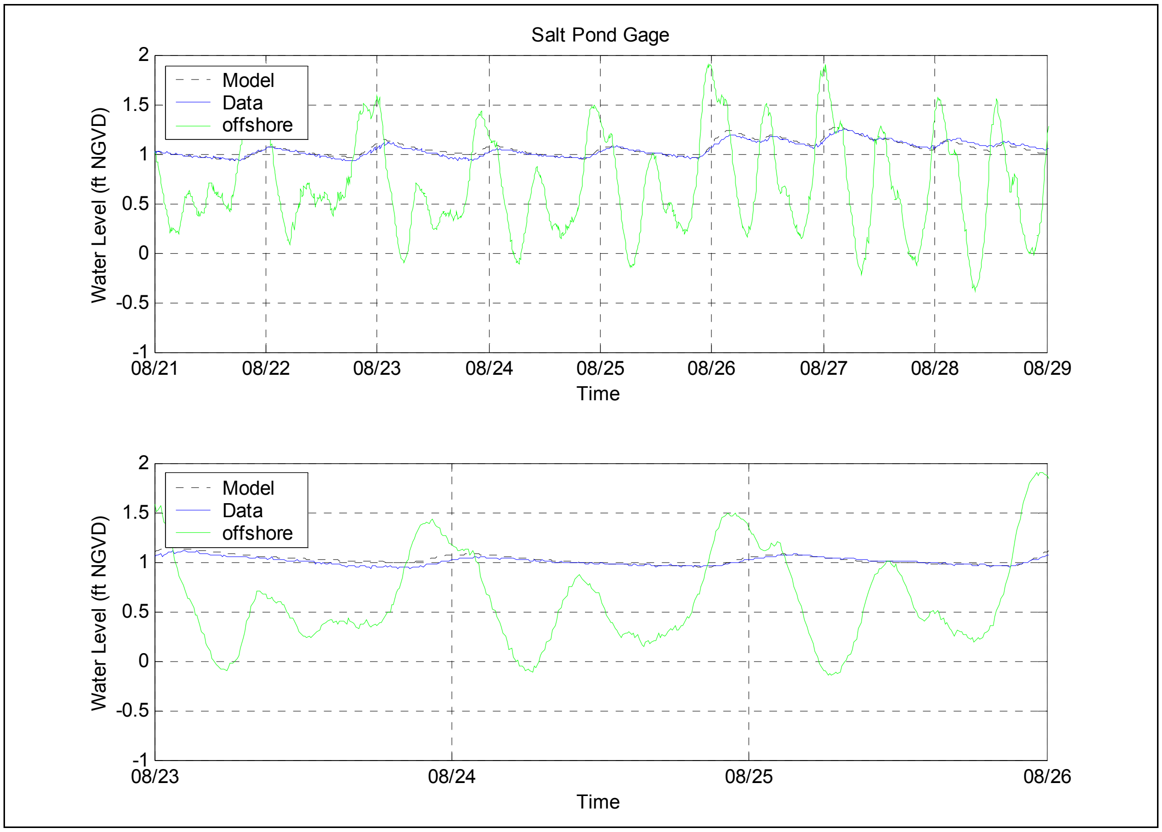

Figure V-9. Comparison of model output and measured tides for the TDR gage in Salt Pond.

V.3.2.4 Model Circulation Characteristics

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

Examining the results from the model run of Salt Pond shows flood velocities in the channels are slightly larger than velocities during maximum ebb. The maximum velocities occur in the inlet with Vineyard Sound, the maximum depth-averaged flood velocities in the model are approximately 3.0 feet/sec, while maximum ebb velocities are about 2.3 feet/sec. A close-up of the model output is presented in Figures V-10 and V-11, which shows contours of velocity magnitude, along with velocity vectors which indicate the direction of flow, for a single model time-step.

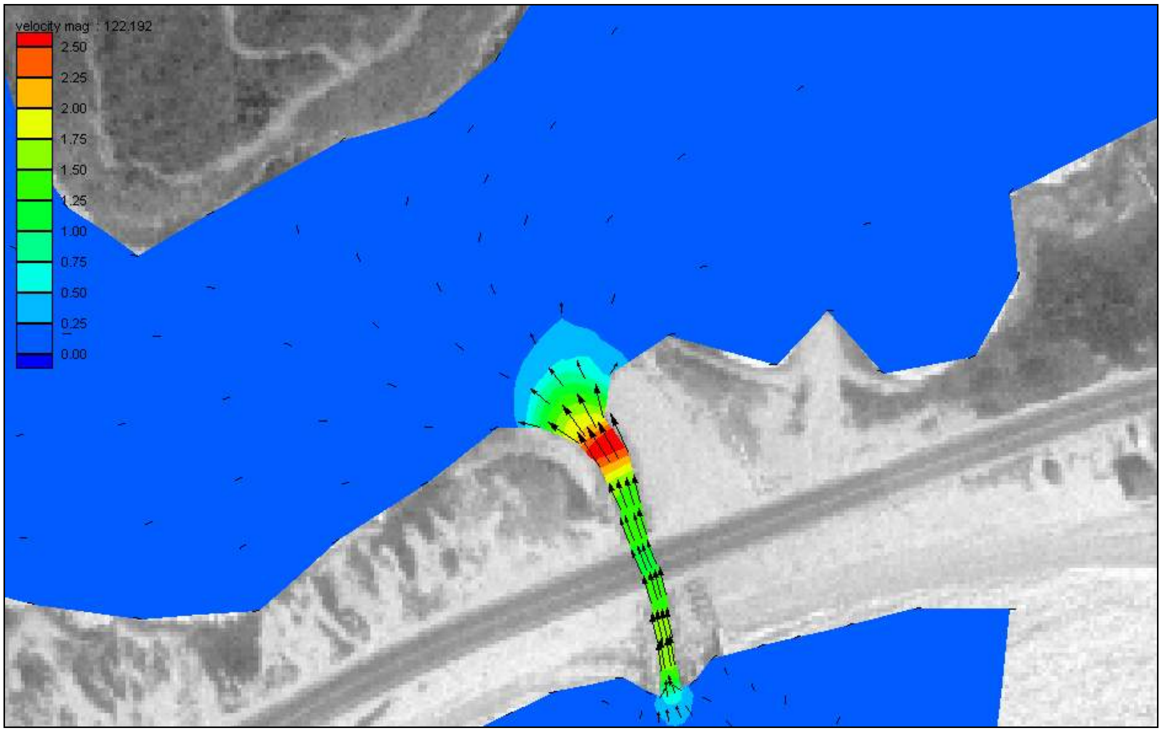

Figure V-10. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

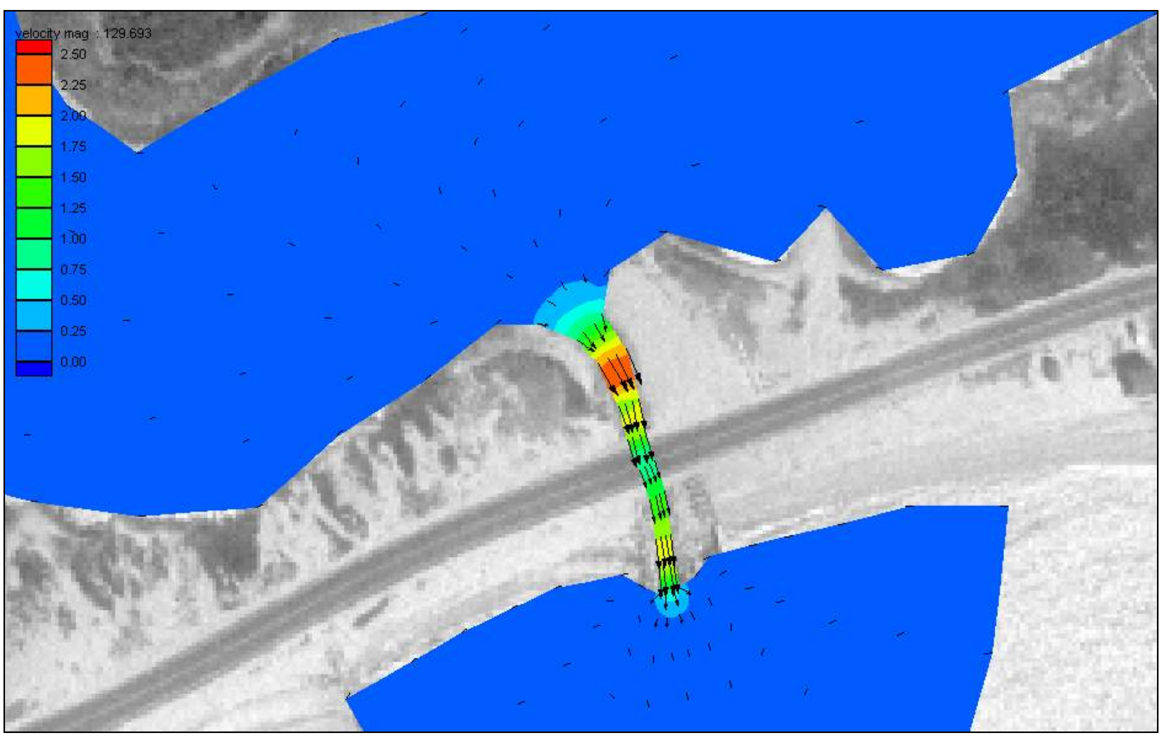

Figure V-11. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

In addition to depth averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. For the flushing analysis in the next section, flow rates were computed across the inlet. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-12. Maximum flow rates occur during flood tides in this system, an indication that this estuary system is flood dominant, and likely a sediment sink (a system that accumulates sediment). During spring tides, the maximum flood flow rates reach 32 ft^3/sec through the inlet. Maximum ebb flow rates are less, approximately 12 ft^3/sec.

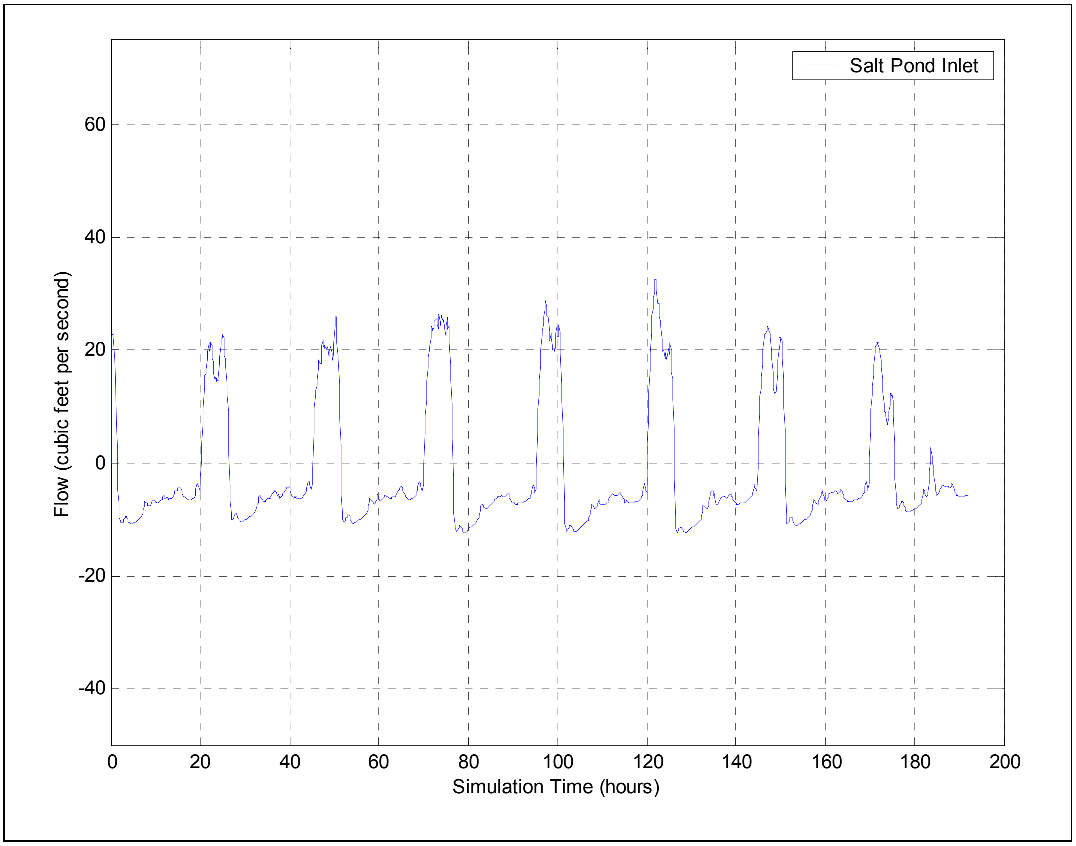

Figure V-12. Time variation of computed flow rate for inlet to Salt Pond. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Plotted time period represents three tide cycles (12.42 h cycle). Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

V.4. FLUSHING CHARACTERISTICS

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through the inlet, the primary mechanism controlling estuarine water quality within the modeled Salt Pond system is tidal exchange. A rising tide offshore in Vineyard Sound creates a slope in water surface from the ocean into the modeled systems. Consequently, water flows into (floods) the system. Similarly, the estuary drains into the open waters of Vineyard Sound on an ebbing tide. This exchange of water between each system and the ocean is defined as tidal flushing. The calibrated hydrodynamic model is a tool to evaluate quantitatively tidal flushing of each system, and was used to compute flushing rates (residence times) and tidal circulation patterns.

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

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

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

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

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 model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the system.

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

The residence time was averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and is listed in Table V-7. The modeled time period used to compute the flushing rate was the modeled calibration period, and included the transition from spring to neap tide conditions. The model calculated flow crossing specified grid lines along the inlet to compute the tidal prism volume. Since the 7.25-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system embayment.

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

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Salt Pond	3,127,050	429,946

Table V-7. Computed residence times for Salt Pond system.

Embayment	System Residence Time (days)
Salt Pond	3.7

The computed flushing rate for Salt Pond shows that the system takes approximately 3.7 days for the volume of the pond to be exchanged. This suggests that the system has marginal tidal flushing. This method assumes all the water in the system is exchanged. In reality, the water in the southwest end of the system is most likely to be exchanged, meaning that water quality continues to decrease in the upper portion of the system.

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

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

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

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayment

Extensive field measurements and hydrodynamic modeling of the embayment were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated model output representing the transport of water and nitrogen within the system embayment. Files of node locations and node connectivity for the RMA-2 model grid were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 22-tidal cycle period in August 2003. Each modeled scenario (e.g., present conditions, build-out) required the model to be run for a 28-day spin-up period, to allow the model to reach a dynamic "steady state", and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayment

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

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Six years of data (collected between 2006 and 2012) were available for stations monitored by SMAST in the Salt Pond System. The MEP study approach relies on a multi-year water quality data collection effort combined with an 'average' hydrodynamic period to represent summer conditions in the estuary of interest, Salt Pond. Thus, the analysis approach is not dependent on a specific time period, rather multi-year averaging of data to take into account variations in tide, temperature, rainfall, loading, groundwater, etc. that may have short-term influences on the data or even result in inter-annual differences over the data collection period. Due to this approach, the differences in data collection periods are not significant. The only circumstance where discrepancies in time periods might present a concern would be where significant physical changes in systems physical characteristics (i.e. geometry, etc.) have occurred between the data collection periods.

Table VI-1. Town of Falmouth water quality monitoring data, and modeled Nitrogen concentrations for the Salt Pond System used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.

Sub-Embayment	Monitoring station	2006 mean	2007 mean	2008 mean	2009 mean	2010 mean	2011 mean	2012 mean	mean	s.d. all data	N	model min	model max	model average
Salt Pond	STP1	0.756	0.788	1.167	0.805	1.157	1.383	0.953	1.080	0.561	26	0.946	0.964	0.958
Salt Pond	STP2	0.874	0.865	1.086	0.721	1.174	1.407	0.951	0.986	0.287	87	0.954	0.965	0.962
Salt Pond	STP3	0.764	0.845	1.129	0.805	1.269	1.364	0.913	0.900	0.204	66	0.675	0.992	0.826
Salt Pond	Inlet	--	0.351	0.393	0.371	0.625	0.484	0.451	0.406	0.123	26	0.280	0.532	0.379

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 Salt Pond 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 the Salt Pond System. 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 numerous water quality studies of other embayments.

Figure VI-1. Estuarine water quality monitoring station locations in the Salt Pond System. Station labels correspond to those provided in Table VI-1.

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 SMAST and 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 system.

VI.2.1 Model Formulation

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

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

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

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

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

Based on groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include ground water flowing into the system from the watersheds. Salt Pond has a single watershed contributing to the groundwater flow, the combined flow rate into the system is 1.23 ft^3/sec (2,999 m^3/day).

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

VI.2.3 Boundary Condition Specification

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

The loadings used to model present conditions in the Salt Pond 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 surface area coverage computed for the embayment (excluding marsh coverage, when present), resulting in a total flux for the embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, the benthic flux is positive within Salt Pond.

In addition to mass loading boundary conditions set within the model domain, a concentration along the model open boundary was specified. The model uses the specified concentration at the open boundary during the flooding tide periods of the model simulations. TN concentration of the incoming water is set at the value designated for the open boundary. The boundary concentration in Vineyard Sound was set at 0.28 mg/L, based on SMAST data from Vineyard Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Vineyard Sound.

Table VI-2. Embayment loads used for total nitrogen modeling of the Salt Pond System, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions.**

embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Salt Pond	4.751	0.789	1.439

VI.2.4 Model Calibration

Calibration of the total nitrogen model proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in Figure VI-2. 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 areas of Salt Pond 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 systems are presented in Table VI-3. These values were used to develop the "best-fit" total nitrogen model calibration. For the case of TN modeling, "best fit" can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

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

Comparisons between model output and measured nitrogen concentrations are shown in plots presented in Figure VI-3. In these plots, means of the water column data and a range of two standard deviations of the annual means at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the SMAST monitoring stations.

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Salt Pond System.	
Embayment Division	E m^2/sec
Vineyard Sound	4.0
Inlet	3.0
Culvert	3.0
Salt Pond	0.5
Marsh	0.5

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

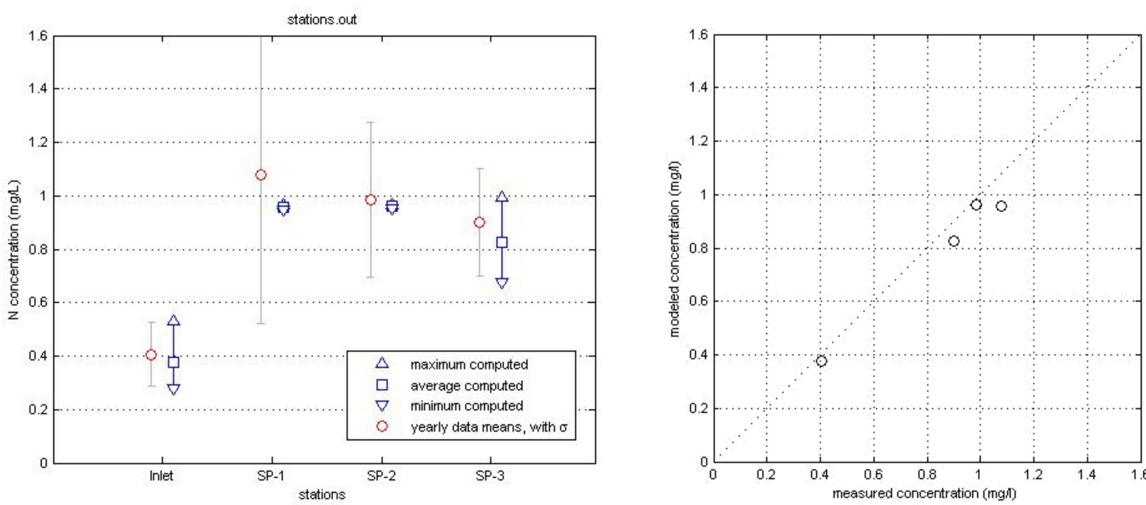

Figure VI-3. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Salt Pond System. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for the system. The model fit is exceptional for the Salt Pond System, with rms error of 0.07 mg/L and an R^2 correlation coefficient of 0.99.

A contour plot of calibrated model output is shown in Figure VI-4 for Salt Pond System. In the figure, color contours indicate nitrogen concentrations throughout the model domain. The output in the figure show average total nitrogen concentrations, computed using the full 5-tidal-day model simulation output period.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Salt Pond System using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, and groundwater inputs. The open boundary salinity was set at 31.4 ppt. For groundwater inputs salinities were set at 0 ppt. The total groundwater input used for the model was 1.23 ft^3/sec (2,999 m^3/day) distributed amongst the watersheds. Groundwater flows were distributed evenly within each watershed through grid cells that formed the perimeter along each watershed's land boundary.

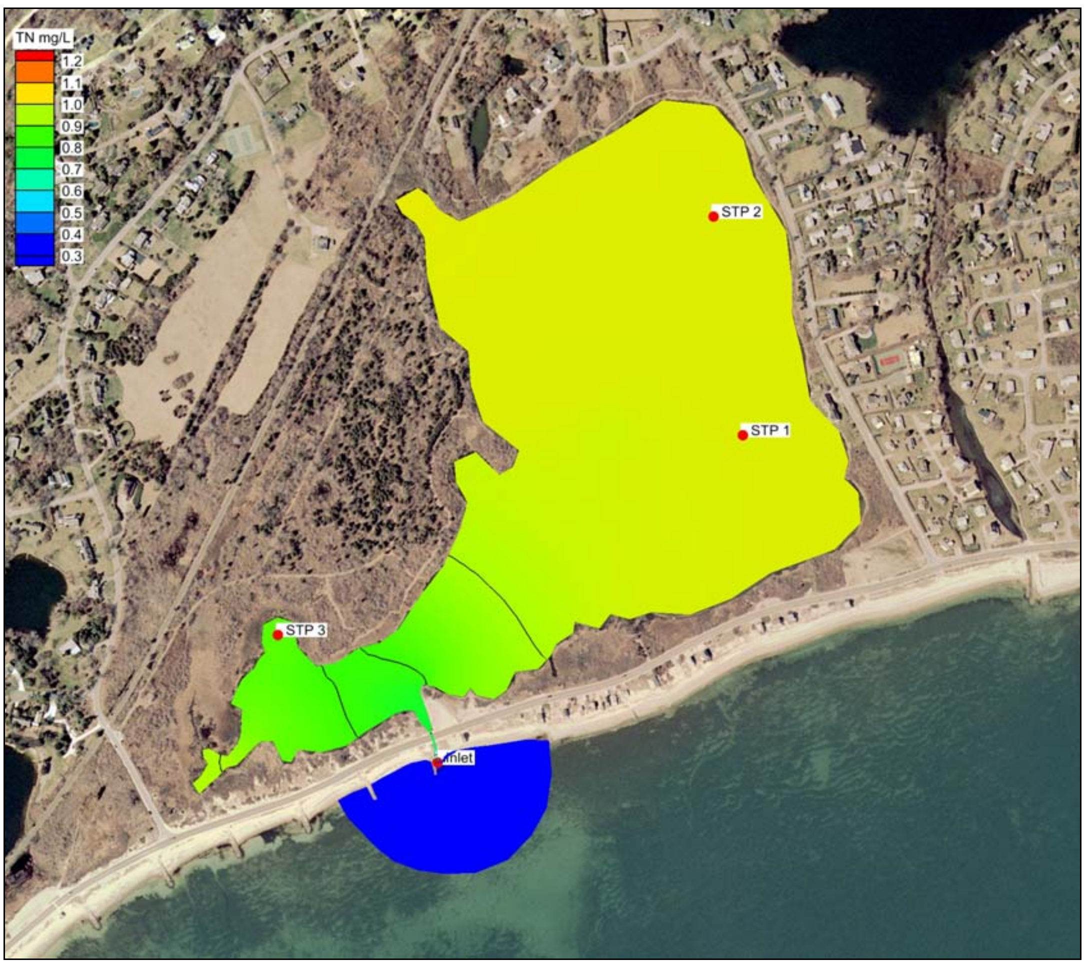

Figure VI-4. Contour plots of average total nitrogen concentrations from results of the present conditions loading scenario, for Salt Pond System. The three water quality stations within the pond are averaged together as the sentinel threshold station for Salt Pond System.

Comparisons of modeled and measured salinities are presented in Figure VI-5, with contour plots of model output shown in Figure VI-6. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in Salt Pond System. The rms error of the models was 1.7 ppt, and correlation coefficient was 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 systems.

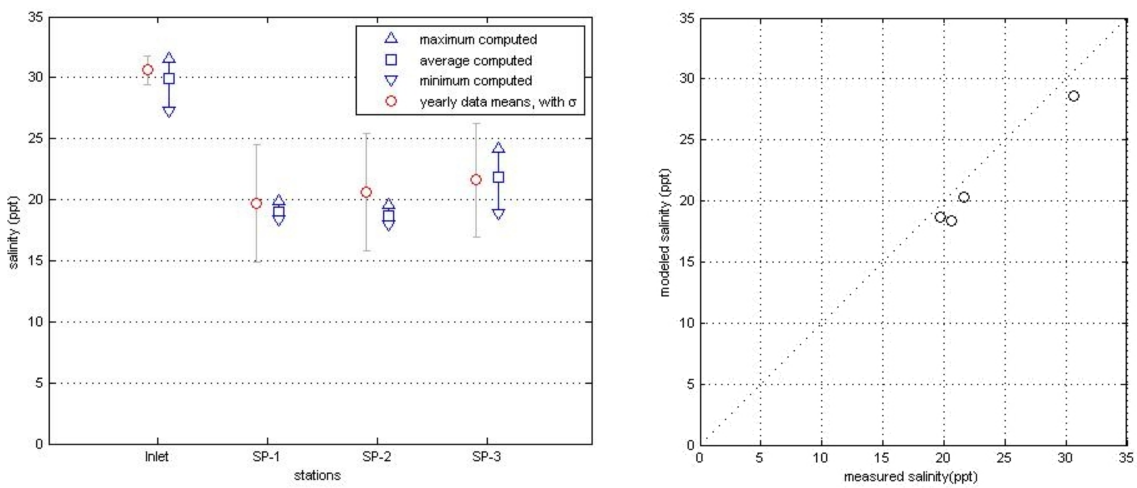

Figure VI-5. Comparison of measured and calibrated model output at stations in Salt Pond System. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Salt Pond 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
Salt Pond	4.751	5.099	+7.3%	0.241	-94.9%

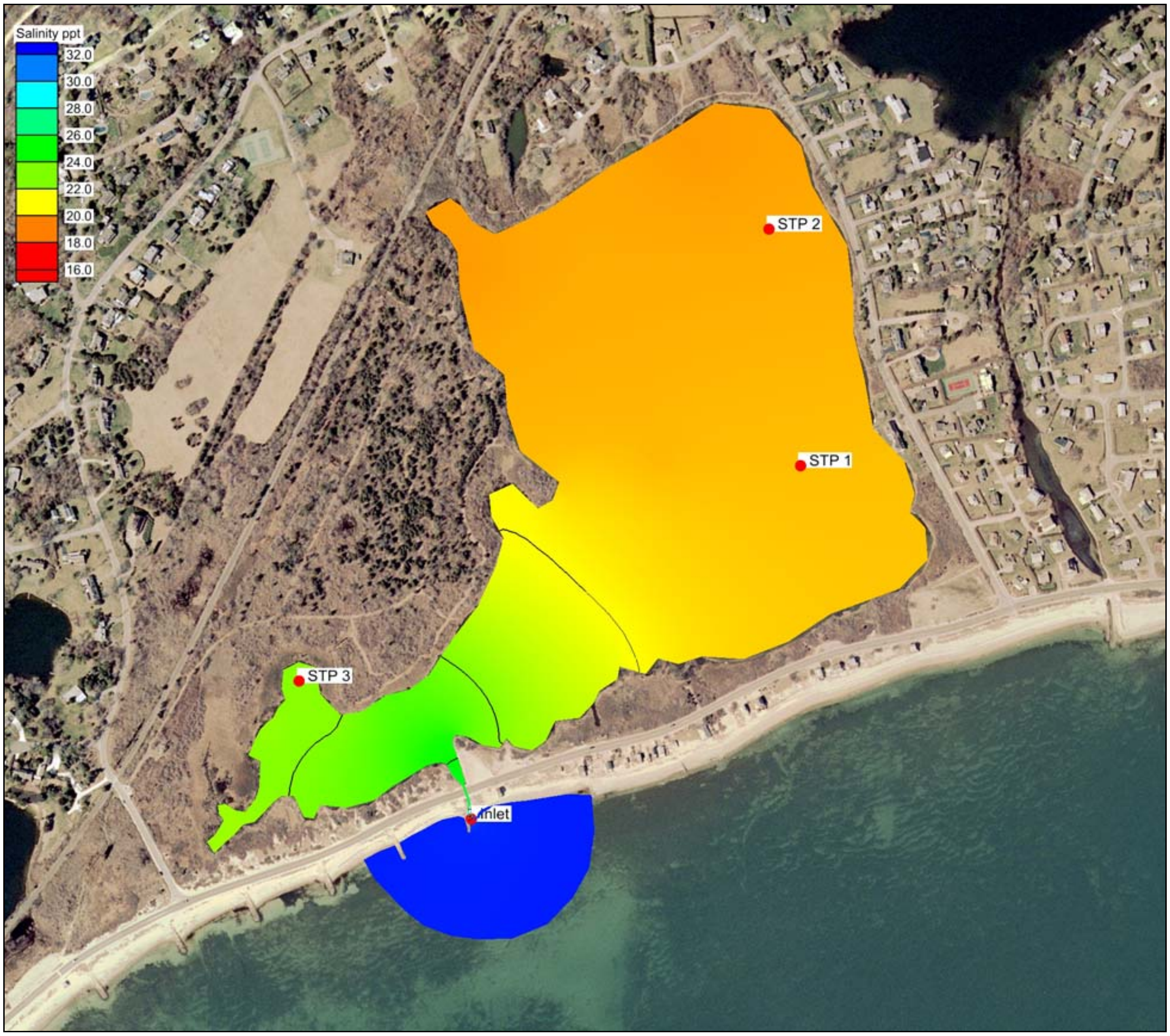

Figure VI-6. Contour plots of modeled salinity (ppt) in Salt Pond System.

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be a increase in watershed nitrogen load to the Salt Pond as a result of potential future development. Specific watershed areas would experience large load increases, for example the loads to Salt Pond would increase 7% from the present day loading levels. For the no load scenarios, a majority of the load entering the watershed is removed; therefore, the load is lower than existing conditions by 94% overall.

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Salt Pond 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)
Salt Pond	5.099	0.789	1.516

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Salt Pond System was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. The stations in Salt Pond show small increase in nitrogen across the water quality stations within the pond. Color contours of model output for the build-out scenario are present in Figure VI-7. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-4, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Salt Pond System. Sentinel threshold station is the average of the three stations within the pond.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Inlet		0.383	0.386	+0.8%
Salt Pond	SP-1	0.927	0.941	+1.5%
Salt Pond	SP-2	0.924	0.938	+1.5%
Salt Pond	SP-3	0.908	0.930	+2.4%

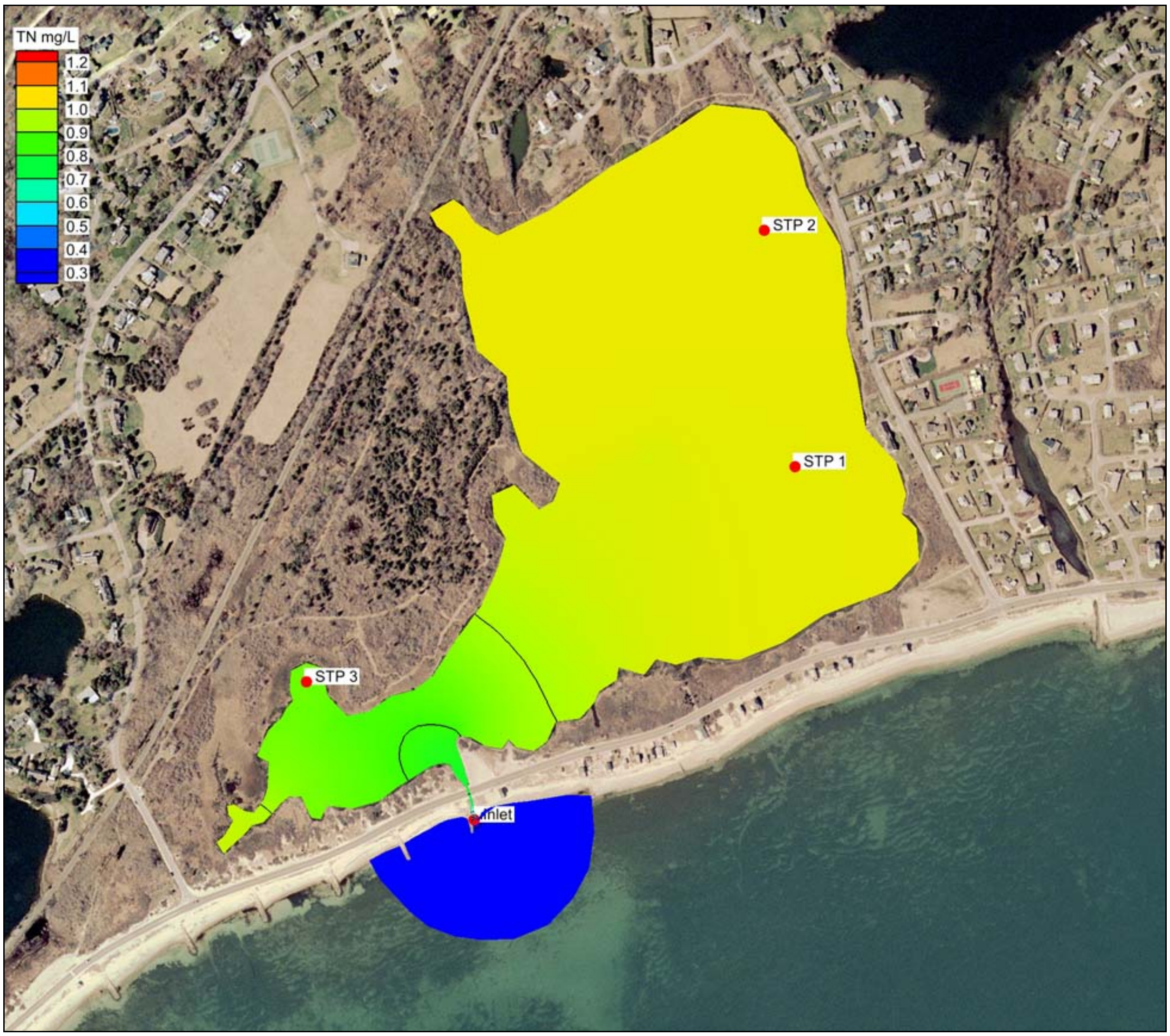

Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Salt Pond System, for projected build-out loading conditions, and bathymetry. The three water quality stations within the pond are averaged together as the sentinel threshold station for Salt Pond System.

VI.2.6.2 No Anthropogenic Load

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

Table VI-7. “No anthropogenic loading” (“no load”) sub-embayment and surface water loads used for total nitrogen modeling of Salt Pond 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)
Salt Pond	0.241	0.789	0.463

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from “no load” was noteworthy as shown in Table VI-8, with reductions ranging from 23% to 38% within the pond. Results for each system are shown pictorially in Figure VI-8.

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 Salt Pond System. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold station is the average of the three stations within the pond.

Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Inlet		0.383	0.332	-13.2%
Salt Pond	SP-1	0.927	0.702	-24.2%
Salt Pond	SP-2	0.924	0.705	-23.7%
Salt Pond	SP-3	0.908	0.563	-38.0%

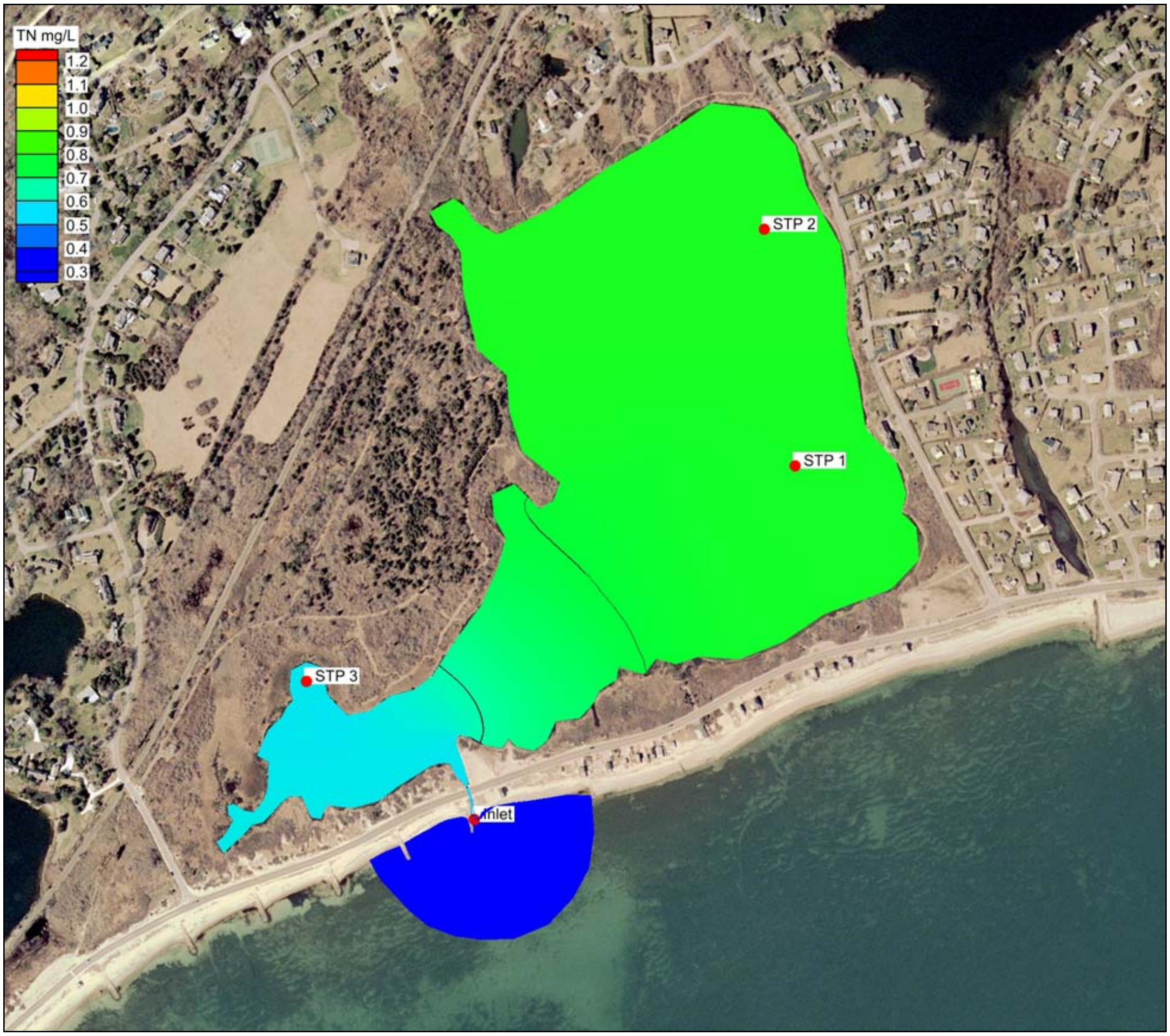

Figure VI-8. Contour plots of modeled total nitrogen concentrations (mg/L) in Salt Pond System, for no anthropogenic loading conditions, and bathymetry. The three water quality stations within the pond are averaged together as the sentinel threshold station for Salt Pond System.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gaged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Salt Pond embayment system in the Town of Falmouth, MA, our assessment is based upon data from the water quality monitoring database developed by the Coastal Systems Program – SMAST (2006-2012), surveys of eelgrass distribution (MEP typically relies on 1951, 1995, 2001, 2006), benthic animal communities (fall 2007), sediment characteristics (summer 2007), and dissolved oxygen time-series records (summer 2007). These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for both of these systems (Section VIII). It should be noted that nitrogen enrichment occurs through 2 primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or low rates of flushing due to restriction of tidal exchange with the low nitrogen waters of Vineyard Sound. Salt Pond has increasing nitrogen loading from the associated watersheds from shifting land-uses and has a significant restriction to tidal flows due to the small tidal inlet and channel configuration. Fundamentally, restrictions of tidal exchange increase the sensitivity of an estuary to nitrogen inputs.

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen threshold determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll-a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed autonomous dissolved oxygen sensors in Salt Pond at locations that would be representative of the dissolved oxygen conditions at critical locations in the system. Sensors (2) were deployed to capture oxygen conditions within an upper location in Salt Pond (main basin), furthest removed from the influence of inflowing waters from Buzzards Bay, and within the channel to the inlet from Vineyard Sound. The dissolved oxygen moorings were deployed 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 Salt Pond system was conducted for comparison to historic records (MassDEP 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. It appears that Salt Pond does not presently support eelgrass habitat and there is no historic record of eelgrass beds in the past 6 decades. Therefore, development of the MEP nitrogen threshold for this system must focus on other parameters of habitat quality, primarily benthic animal habitat.

Analysis of inorganic N/P molar ratios within the water column of Salt Pond suggest that nitrogen with phosphorus are the nutrients to be managed, as the ratio in Salt Pond (16) and the inlet channel (14) approximate the Redfield Ratio value (16) indicating that nitrogen and phosphorus additions will increase phytoplankton production in this basin (just offshore of inlet, N/P<5). Conversely, reducing nitrogen levels should reduce phytoplankton growth. Within the Salt Pond system, since temporal changes in eelgrass distribution could not provide a basis for evaluating nutrient related habitat quality, nutrient threshold determination was based on results from the dissolved oxygen and chlorophyll mooring and water quality monitoring data, macroalgae surveys, and the benthic animal community characterization.

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 stress-organic enrichment 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: 1) the Wild Harbor oil spill (Hampson, 1978), 2) benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution-Sanders, 1958 and 1960), 3) infaunal studies completed in New Bedford Inner and Outer Harbors (SMAST-unpublished data), and 4) the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H’) and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 4 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters be able to maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Salt Pond embayment are currently listed under this Classification as SA. It should be noted that the Classification system represents the water quality that the embayment should support, not the existing level of water quality and that it is the designated water quality that is the target of TMDL's generated under the U.S. Clean Water Act. It is through the MEP and TMDL processes that site specific management targets are developed and under the Town's CWMP that management alternatives are designed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the bottom of the embayment within key regions of the Salt Pond system (Figure VII-2). The dissolved oxygen sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployments. In addition periodic calibration samples were collected at the depth of each sensor and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 25-30 days within the interval from July through mid-September. All of the mooring data from the Salt Pond Estuary were collected during the summer of 2007.

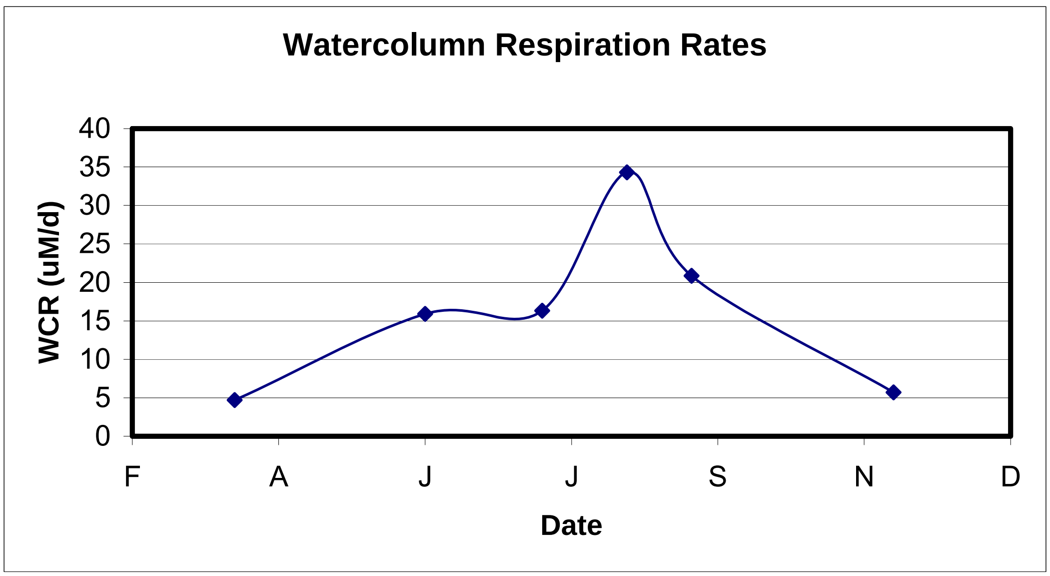

Figure VII-1. Example of typical average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System, Cape Cod (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter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

Similar to other embayments in southeastern Massachusetts, the Salt Pond Estuary 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 28 and 29 day deployment periods 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.

Figure VII-2. Aerial Photograph of the Salt Pond Estuary in the Town of Falmouth showing the location of the continuously recording Dissolved Oxygen / Chlorophyll-a sensors deployed during the Summer of 2007.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate moderate (North DO1) to high (South DO2) nutrient enriched waters within the upper basin and tidal channel region of Salt Pond, respectively. The dissolved oxygen data is further described below and depicted in Figures VII-3 and VII-5. The oxygen data is consistent with organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll-*a* and in the tidal channel, additionally accumulating macroalgae. The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of the Salt Pond Estuary.

The oxygen record for the upper mixed layer of the main basin of Salt Pond shows levels of oxygen depletion and daily oxygen excursions and chlorophyll-*a* levels indicative of nitrogen enrichment. Oxygen records from the Pond coupled with the multi-year monitoring by SMAST-Coastal Systems Program scientists indicates that oxygen levels are frequently lower than atmospheric equilibration but only infrequently decline to <4 mg L^{-1} (Figure VII-3). However, within the tidal channel oxygen depletion is significant (Figure VII-5), with frequent depletions to <4 mg L^{-1}. The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. 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 innermost and tidal channel reaches of this system are nitrogen enriched. Measured dissolved oxygen depletion indicates that portions of Salt Pond present oxygen stress to benthic animal communities. The prolonged stratification of the deep "hole" in the kettle basin (>5 m) has resulted in long-term hypoxia/anoxia of deep bottom waters and the loss of benthic animal communities. The upper mixed layer remains aerobic and it is the focus of the embayment specific results which follow:

Salt Pond North (DO1) DO/CHLA Mooring (Figures VII-3 and VII-4):

Two moorings were deployed in the mixed upper layer of the Salt Pond system. One of the two instrument moorings was located in the upper basin of Salt Pond at the farther end of the basin away from the inlet connecting the estuary to lower nutrient water from Vineyard Sound (Figure VII-2). The second instrument was placed in the lower portion of this system in the depositional tidal channel (discussed below). The mooring in the upper portion of the system was located in the upper northwest corner of Salt Pond. Moderate daily excursions in oxygen levels were observed at this location, however, hypoxic conditions where levels decline to 4 mg L^{-1} or less (Figure VII-3, Table VII-1) were never attained at this mooring location. Instantaneous oxygen levels that drop below 4 mg L^{-1} are indicative of oxygen stress.

Oxygen levels regularly exceeded 6 mg L^{-1} and ranged from between 8 and 10 mg L^{-1} in the later part or the deployment period. These moderately high oxygen levels are primarily the result of photosynthesis by high phytoplankton biomass and relatively quiescent waters. Over the 28 day deployment there appear to be multiple moderate phytoplankton blooms where chlorophyll-*a* increased to 10-14 ug L^{-1} and a few periods of bloom activity where chlorophyll-*a* concentrations peaked at just over 16 ug L^{-1} (Figure VII-4). The very infrequent low levels of oxygen observed in this system is indicative of moderate habitat impairment which is also consistent with the relatively reduced chlorophyll-*a* levels, also indicative of slight nitrogen enrichment (average chlorophyll-*a* by mooring, 9.1 ug L^{-1}; water quality monitoring program, 11.9-13.5 ug L^{-1}). In this northern portion of the main basin of the Salt Pond system, chlorophyll-a exceeded the 10 ug L^{-1} benchmark 25 percent of the time-series and the 15 ug L^{-1} level only 2 percent of the record (Table VII-2, Figure VII-4). Average chlorophyll levels over 10 ug L^{-1} have been used to indicate eutrophic conditions in embayments.

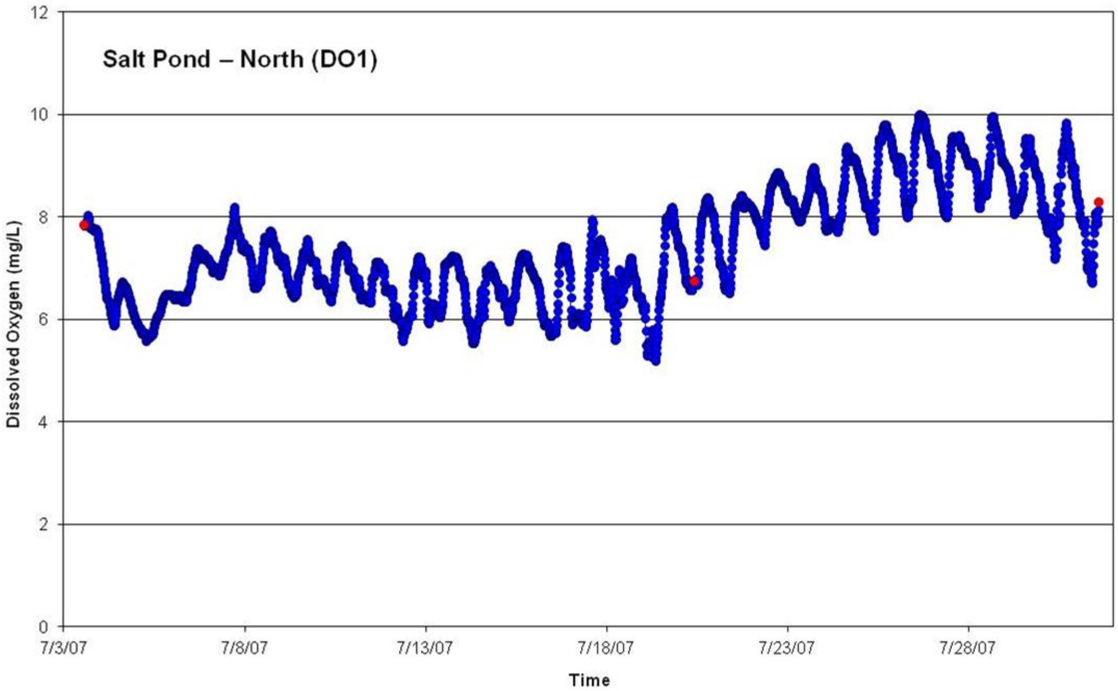

Figure VII-3. Bottom water record of dissolved oxygen at the Salt Pond North station, Summer 2007. Calibration samples represented as red dots.

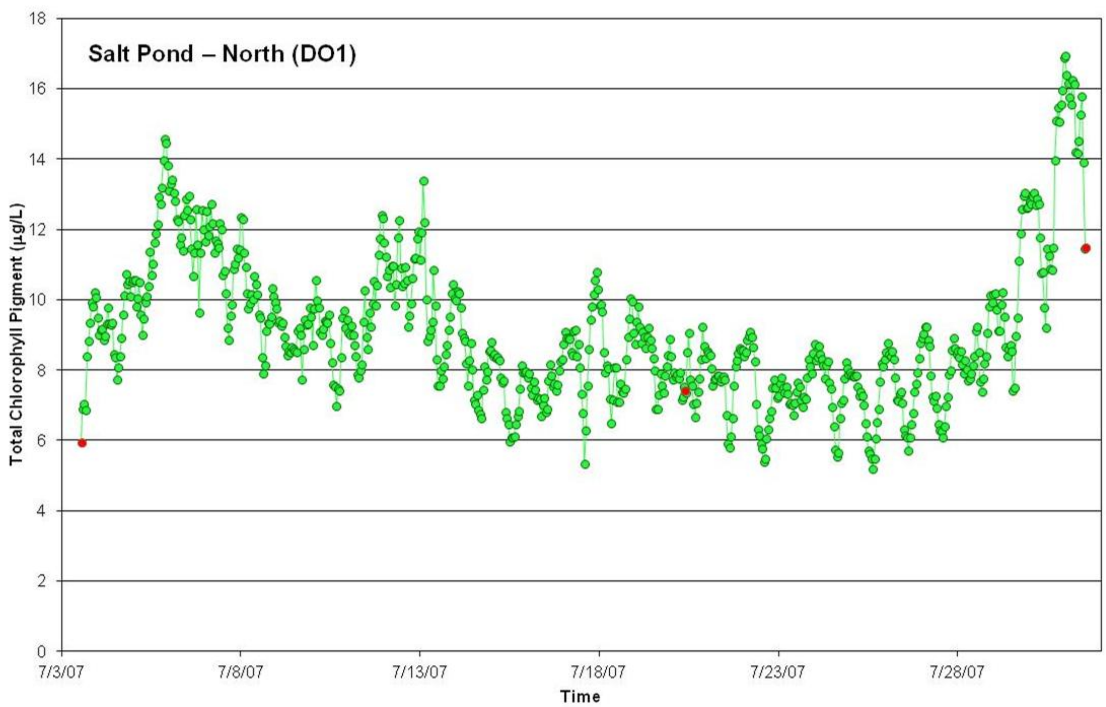

Figure VII-4. Bottom water record of Chlorophyll-*a* in the Salt Pond North station, Summer 2007. Calibration samples represented as red dots.

Salt Pond South (DO2) DO/CHLA Mooring (Figures VII-5 and VII-6):

The second of the two instrument moorings (Salt Pond South DO2) deployed in the Salt Pond system was located approximately in the lower portion of the system, slightly up-gradient of the inlet connecting the estuary to lower nutrient water from Vineyard Sound (Figure VII-2). The mooring was centrally located in a channel of deeper water that connects the lower part of the pond to the main basin. This channel was clearly depositional, with soft organic-rich sulfidic muds with patchy accumulations of drift macroalgae. As a result, oxygen conditions at this location were notably different compared to the DO record collected from the northern portion of the main basin, both in the range of oxygen excursion and the extent of oxygen depletion. Oxygen concentrations ranged from ca. 12 mg L^{-1}, well above air equilibration (~8 mg L^{-1}), to slightly hypoxic conditions where levels decline to 2-3 mg L^{-1} (Figure VII-5, Table VII-1). Instantaneous oxygen levels that drop below 4 mg L^{-1} are indicative of oxygen stress to many animals. The organic enrichment of the system is demonstrated by the moderate-high chlorophyll-*a* levels, organic rich sediments and accumulation of drift algae observed during the deployment period. A key feature of this organic enrichment is the high rate of oxygen production from photosynthesis (carbon fixation) during daylight and the rapid depletion after sunset stemming from respiration.

Oxygen levels regularly exceeded 8 mg L^{-1} and even 10 mg L^{-1}. These high oxygen levels are likely the result of the effects of photosynthesis by the phytoplankton and to a lesser extent macroalgae, documented in the vicinity of the DO mooring. Over the 29 day deployment there appears to be a noticeable increase in phytoplankton biomass in the pond where chlorophyll-*a* gradually increased to over 10 ug L^{-1} and then reached levels between 15-20 ug L^{-1} towards the end of the deployment period for nearly a ten day period. The low levels of oxygen and large daily excursions observed in this system are indicative of moderate to significant habitat impairment within the depositional tidal channel. Phytoplankton levels gradually increased over the deployment period, chlorophyll-*a* exceeded the 10 ug L^{-1} benchmark 31 percent of the time (Table VII-2, Figure VII-6). The mooring average chlorophyll-*a* level was only 8.6 ug L^{-1}, but reached 17 ug L-1 by the end of the deployment. The SMAST water quality monitoring program showed similar values averaging 14.3 ug L^{-1} over the 7 summers of monitoring (2006-2012). Average chlorophyll levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

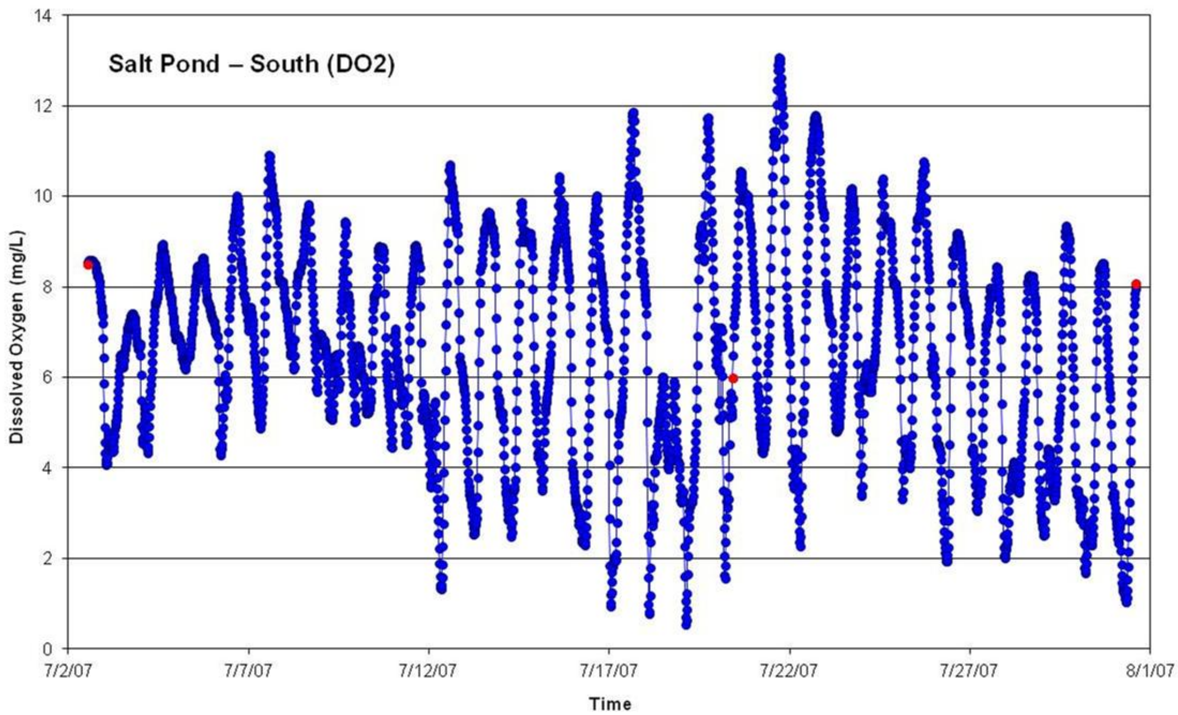

Figure VII-5. Bottom water record of dissolved oxygen at the Salt Pond South station, Summer 2007. Calibration samples represented as red dots.

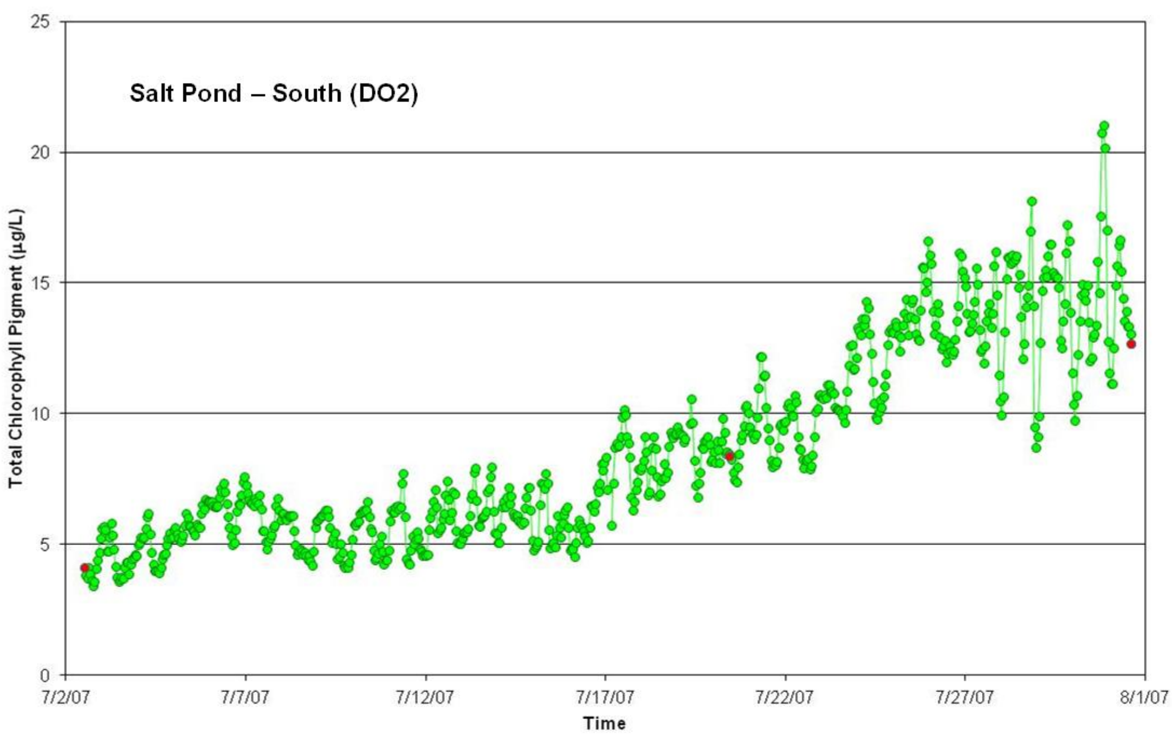

Figure VII-6. Bottom water record of Chlorophyll-*a* at the Salt Pond South station, Summer 2007. Calibration samples represented as red dots.

Table VII-1. Days and percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels within the Salt Pond embayment system. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Salt Pond North (DO1)	**7/2/2007**	**7/31/2007**	**28.0**	**8%**	**0%**	**0%**	**0%**
			Mean	0.18	NA	NA	NA
			Min	0.03	0.00	0.00	0.00
			Max	0.57	0.00	0.00	0.00
			S.D.	0.18	NA	NA	NA
Salt Pond South (DO2)	**7/2/2007**	**7/31/2007**	**29.0**	**40%**	**14%**	**17%**	**8%**
			Mean	0.35	0.28	0.21	0.13
			Min	0.04	0.03	0.02	0.04
			Max	0.94	0.58	0.53	0.38
			S.D.	0.22	0.17	0.15	0.09

Table VII-2. Duration (days and % of deployment time) that chlorophyll a levels exceed various benchmark levels within the Fiddlers Cove 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.

Mooring Location	Start Date	End Date	Total Deployment (Days)	>5 ug/L Duration (Days)	>10 ug/L Duration (Days)	>15 ug/L Duration (Days)	>20 ug/L Duration (Days)	>25 ug/L Duration (Days)
Salt Pond North (DO1)	**7/2/2007**	**7/31/2007**	**29.07**	**97%**	**25%**	**2%**	**0%**	**0%**
Mean Chl Value = 9.1 ug/L			Mean	14.04	0.30	0.31	NA	NA
			Min	0.04	0.04	0.08	0.00	0.00
			Max	28.04	1.46	0.54	0.00	0.00
			S.D.	19.80	0.37	0.32	NA	NA
Salt Pond South (DO2)	**7/2/2007**	**7/31/2007**	**29.07**	**87%**	**31%**	**7%**	**0%**	**0%**
Mean Chl Value = 8.6 ug/L			Mean	1.48	0.76	0.15	0.13	NA
			Min	0.04	0.04	0.04	0.13	0.00
			Max	15.33	3.42	0.42	0.13	0.00
			S.D.	3.64	0.97	0.11	NA	NA

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted for the Salt Pond Estuary by the DEP Eelgrass Mapping Program and by other members of the MEP Technical Team. Field survey data was collected in 2007, in concert with the MEP benthic recycling and infaunal animal sampling. Additional analysis of available aerial photos from 1951 was completed to determine eelgrass distribution under conditions of lower watershed nitrogen loading. According to the MassDEP eelgrass mapping program, the 1951 aerial photography was of marginal quality and not interpretable in a definitive manner, thus of limited value. Eelgrass presence determined in 2007 was by direct observation by MEP divers. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of the data sets can provide a view of temporal trends in eelgrass distribution from 1951 to 2007; the period in which watershed nitrogen loading increased (although present level is only moderate to low). This temporal information can be used to determine the stability of the eelgrass community.

At present, no eelgrass is present within Salt Pond, although small sparse patches of *Ruppia* occur in some shoreline areas. Salt Pond is functionally a basin with very limited fringing wetland habitat along its shoreline to the main basin. The sediments are currently soft muds rich in organic matter in the center deep portion of the pond, which has had periods of prolonged anoxia since at least the 1970's. The primary factor in the oxygen depletion of the deep waters is the strong salinity stratification of the deep basin, which restricts ventilation with the atmosphere. This stratification presently results from the depth of the basin and the highly restricted tidal exchange by the present inlet system. It is the shallower (<3m) margins of the basin that support some sandy areas and compacted muds with oxidized surface layers. The absence of eelgrass beds is expected in this system given the high chlorophyll-*a* and low dissolved oxygen levels and high water column nitrogen concentrations. Given the absence of eelgrass at present and the lack of evidence of prior eelgrass habitat within this system, management should focus on benthic animal habitat, primarily within the marginal areas.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 9 locations within the Salt Pond Embayment System (Figure VII-7), with replicate assays at each site. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or organic-enriched/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 while the Salt Pond Estuary does not support eelgrass beds and is showing signs of moderate (margins of the pond) and significant impairment (tidal channel) based on DO and chlorophyll-*a* levels and macroalgal distribution, it is still possible to improve habitat conditions to be more supportive of healthy benthic communities. To the extent that this system can support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded) and what nutrient concentrations would be supportive of healthy habitat. This assessment is also important for the establishment of site-specific nitrogen thresholds (Section VIII).

Figure VII-7. Aerial photograph of the Salt system showing location of benthic infaunal sampling stations (yellow symbols).

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.

Salt Pond Infaunal Characteristics:

Overall, the infauna survey indicated that the shallow margin (<3 m) around the deep kettle "hole" is supportive of the highest quality infaunal habitat within Salt Pond, showing areas that are moderately or significantly impaired. The sediments within the deep basin (>3 m) are overlain by anoxic bottom water and are devoid of benthic animals. Intermediate to these 2 regions, the tidal channel is depositional, with significant oxygen declines and significantly impaired benthic habitat. (Table VII-3).

The prolonged anoxia of the deep basin (>3 m), stems largely from the geomorphology of the basin, its deep water and the highly restricted tidal flushing of this system. The result is strong salinity stratification of the water column, which prevents down mixing of oxygenated surface waters. The result is the prolonged anoxia and lack of benthic animal habitat below ~3 meters depth. It should be noted that recovery of this habitat for benthic animals cannot occur until the physical stratification is significantly diminished. In contrast, the shallow area around the deep basin has aerobic bottom waters, relatively consolidated sediments and oxidized surface sediments. The result is high numbers of animals (~500 per sample), moderate-high numbers of species (17) with moderate diversity and Evenness. The community does have moderate numbers of organic enrichment-stress indicator species (Tubificids, Capitellids) comprising ~1/3 of the community. All of the community metrics, indicator species, numbers of individuals, species and indices, document a benthic animal habitat moderately-significantly impaired by organic enrichment, hence ultimately nitrogen enrichment. Uncharacteristically, the tidal channel to the inlet is currently supporting lower quality habitat than sediments at the same depth in the shallow margins of the main basin. The tidal channel is clearly supporting significantly impaired benthic animal habitat. This is documented in the low species numbers (5), low numbers of individuals (~100 per sample) and low diversity (1.5). Equally diagnostic, the community is dominated by organic enrichment-stress indicator species, which comprise more than 2/3 of the community. It appears that the "stress" is organic enrichment due to organic matter and macroalgal deposition within the channel. The result is periodic oxygen depletion (Figure VII-5) and sulfidic sediments. These environmental features are fully consistent with the community observations and the animal habitat designations described above. Note that communities with low numbers of stress indicator species and total species numbers of 20-25 are generally associated with high quality estuarine benthic habitats.

The benthic animal communities were compared to high quality environments, such as the Outer Basin of nearby Quissett Harbor, as a benchmark. The Outer Basin of Quissett Harbor supports benthic animal communities with $\geq$28 species, >400 individuals with high diversity (H' $\geq$3.7) and Evenness (E $\geq$0.77). Similarly, outer stations within Lewis Bay in Barnstable currently support similarly high quality benthic habitat as seen in the numbers of individuals (502 per sample), number of species (32), diversity (3.69) and Eveness (0.74). Equally important these communities are not consistent with nutrient enrichment being composed of a variety of polychaete, crustacean and mollusk species, as opposed to stress tolerant small opportunistic oligochaete worms.

Classification of habitat quality necessarily included the structure of the estuarine basin, specifically that it is fully representative of a tidal embayment, as opposed to a tidal river or salt marsh basin and if a basin is structurally impaired or impaired by nitrogen enrichment. Integration of all of the metrics clearly indicates that the shallow areas of Salt Pond are generally supporting benthic animal habitat that is moderately or significantly impaired. The proximate cause of impairment is organic matter enrichment and oxygen depletion, stemming ultimately from nitrogen enrichment. Total nitrogen levels within the estuary at present are >0.90 mg TN L^{-1}, a level generally found associated with a significant level of impairment of benthic animal habitat in southeastern Massachusetts estuaries.

Table VII-3. Benthic infaunal community data for the Salt Pond embayment system (Figure VII-7). Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2).

	Depth Range (m)	Total Species	Total Individuals	#Species Calc @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Salt Pond						
Shallow Margin	**0.5 - 2.0**	**17**	**492**	**11**	**2.51**	**0.63**
Deep Basin	**3.1 - 5.25**	**1**	**1**	**N/A**	**0.26**	**0.17**
Channel	**1.5**	**5**	**86**	**2**	**1.48**	**0.72**

Other Resource Characteristics:

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and as available to the MEP Technical Team. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish (Figure VII-8). As is the case with some systems on Cape Cod, all of the enclosed waters of Salt Pond are classified as prohibited year round for the taking of shellfish, indicating the system is impaired relative to the taking of shellfish. This could be due to bacterial concerns which would be a result of both human activity (septic systems in the watershed) as well as natural fauna. Nevertheless, the Salt Pond system has also been classified as supportive of specific shellfish communities (Figure VII-9). The major shellfish species with potential habitat within the Salt Pond Estuary are soft shell clams (*Mya*) and the American oyster. Theoretically suitable habitat for *Mya* is essentially along the shallow waters at the lower edge of the main basin of Salt Pond relatively close to the inlet whereas American Oyster habitat is theoretically more favorable in the shallow waters in the middle portion of the main basin.

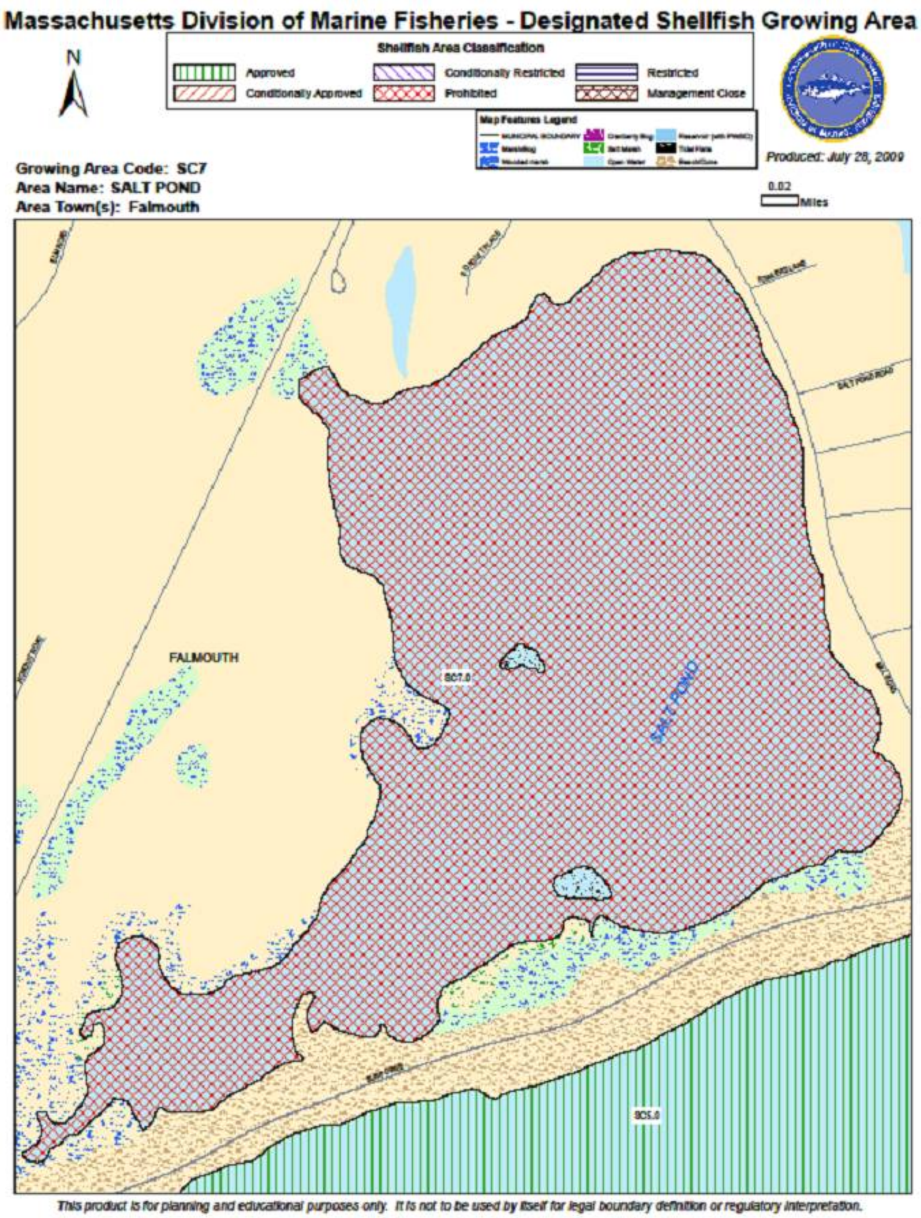

Figure VII-8. Location of shellfish growing areas in the Salt Pond embayment system and the status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

Figure VII-9. Location of shellfish suitability areas within the Salt Pond Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence".

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll). Additional information on temporal changes within each sub-embayment of an estuary, its associated watershed nitrogen load and geomorphological considerations of basin depth, stratification and functional type further strengthen the analysis. These data were collected to support threshold development for the Salt Pond Embayment System by the MEP and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the baseline Water Quality Monitoring Program conducted by the Coastal Systems Program with assays by the Coastal Systems Analytical Facility at SMAST-UMass Dartmouth.

The Salt Pond Embayment System is a simple estuary created by the drowning of a coastal kettle on the southern coast of Falmouth Cape Cod adjacent Vineyard Sound by rising sea levels several thousand years ago. The Pond consists of a single main kettle basin with a deep central region (>5 meters) and a tidal channel to the inlet. The tidal inlet is restricted and as a result of the small tide range, there is little fringing salt marsh. Salt Pond is currently functioning as a typical coastal embayment with restricted tidal exchange with the waters of Vineyard Sound. Each of type of functional component to an estuary (salt marsh basin, embayment, tidal river, deep basin {sometimes drown kettles}, shallow basin, etc.) has a different natural sensitivity to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of the specific basin and its ability to support eelgrass beds and infaunal communities. At present, the Salt Pond Estuary is beyond its ability to assimilate nitrogen without further impairment. The system is showing a high level of nitrogen enrichment, with no eelgrass habitat and moderate to significantly impaired benthic animal habitats (depending on location in the pond), regions of periodic hypoxia and phytoplankton blooms and a stratified deep basin with prolonged anoxia (Table VIII-1), these findings indicate that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system. In should be noted that nitrogen management includes both source reduction and in the case of a tidally restricted embayment, enhanced tidal flushing.

The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Chapter VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment and restriction of tidal flows. The spatial pattern indicated that the magnitude of oxygen depletion, enhancement of chlorophyll-*a* levels and total nitrogen concentrations were affected by the watercolumn stratification stemming from the basin geomorphology and reduced tidal action such that waters and sediments below 3 meters depth are subjected to prolonged anoxia and potential infaunal habitat is only in the shallow margins of the main basin and in the region of the tidal channel. Given the reduced tidal action, these latter regions show only a slight gradient, with Salt Pond operating more as an almost closed pond system.

Table VIII-1. Summary of nutrient related habitat quality within the Salt Pond Estuary within the Town of Falmouth, MA, based upon assessments in Section VII. The single main basin of this tidally restricted embayment consists of mixed layer (<3m), deep basin (>3 m) and channel to inlet. WQMP indicates SMAST Water Quality Monitoring Project.

Health Indicator	Salt Pond Embayment System		
	Channel to Inlet	**Main Basin Shallow [a]**	**Main Basin Deep [b]**
Dissolved Oxygen	SI[1]	H/MI[2]	SD[3]
Chlorophyll	MI[4]	MI[5]	--[6]
Macroalgae	MI/SI[7]	MI[8]	--[9]
Eelgrass	--[10]	--[10]	--[10]
Infaunal Animals	SI[11]	MI/SI[12]	SD[13]
Overall:	**SI[14]**	**MI[15]**	**SI[16]**

a – "shallow" is all waters and sediments <3 meters depth, large area surrounding deep "hole", sediments overlain by aerobic bottom water;
b – "deep" is watercolumn and sediment at >3 meters depth, limited to the relatively small area of the deep "hole" (5+ m), sediments overlain by anoxic bottom water, since stratification results from the present tidal dynamics and basin geomorphology, the very poor habitat is a natural condition;
1 – channel to inlet South: oxygen depletion frequently <6 mg/L, 40% of record and <4 mg/L 17% of record in 1 m deep channel; WQMP 53% of samples <6 mg/L, 17% <5 mg/L, 6%<4 mg/L. Channel is depositional, highly organic enriched sediments and macroalgae.
2 – Mixed layer. Main basin North: oxygen depletion frequently >6 mg/L, 92% and >5 mg/L 100% of record; WQMP 88% of samples >6 mg/L only 4% <4 mg/L over monitoring period.
3 – summer stratification and extended hypoxia/anoxia below 3 meters;
4 – moderate-high summer chlorophyll levels generally ~10 ug/L, averaging 9 ug/L at mooring site, average level in summer WQMP (2006-2012) was 14.3 ug/L, station SP-3.
5 – moderate-high summer chlorophyll levels generally ~10 ug/L, averaging 9 ug/L at mooring site, mean summer WQMP (2006-2012), 11.9-13.5 ug/L, stations SP-1,2.
6 – high bacterial chlorophyll levels associated with the oxycline, >40 ug/L common, but not directly related to nitrogen levels.
7 -- patches of drift macroalgae accumulating over organic rich, soft sulfidic muds
8 -- generally sparse drift algae and thin algal mat, some *Ruppia*.
9 -- macroalgae and algal mat absent, due to prolonged anoxia of bottom waters
10 – no evidence that this basin historically supported eelgrass habitat;
11 -- low-moderate numbers of individuals, low # species, low diversity, dominated by organic enrichment-stress tolerant opportunistic species, (Tubificids, *Capitella,* >65% of organisms)
12 -- high numbers of individuals, moderate number of species, moderate diversity and Evenness, some areas with significant numbers of organic enrichment and stress tolerant opportunistic species, (Tubificids, *Capitella* accounting for ~1/3 of total organisms) other areas higher quality.
13 -- benthic animals absent due to prolonged anoxia in bottom waters, due to structural stratification.
14 -- Significant Impairment , primarily due to periodic D.O. depletion and significantly impaired animal communities dominated by stress indicator species, patches of macroalgal accumulation with moderate-high summer chlorophyll
15 -- Moderate Impairment, primarily due to regions with 1/3 community as organic enrichment species, algal mat in some areas and moderate-high summer chlorophyll.
16 -- Degraded habitat, resulting from structural stratification and prolonged anoxia. Benthic animal community absent.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation; -- = not applicable to this estuarine reach

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll-*a* levels indicate moderate to high nutrient enriched waters within the margins of the main basin and tidal channel region of Salt Pond, respectively. The oxygen data is consistent with organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll-*a* and in the tidal channel, which also has patchy accumulations of macroalgae. The measured levels of oxygen depletion and enhanced chlorophyll-*a* levels follows the spatial pattern of total nitrogen levels in this system (Section VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of the Salt Pond Estuary.

The shallow areas (<3m depth) of Salt Pond generally maintain aerobic bottomwaters, and are currently supporting impaired benthic animal habitat. In contrast, the stratified deep waters (>3 M) of the deep kettle hole are anoxic and devoid of benthic animals. The tidal channel region was clearly depositional, with soft organic-rich sulfidic muds with patchy accumulations of drift macroalgae. As a result, oxygen conditions at this location were notably different compared to the DO record collected from the shallow margins of the main basin, both in the range of oxygen excursion and the extent of oxygen depletion. The shallow margins (<3 m) to the main basin have a large fetch which allows wind driven mixing of the surface waters and maintains bottom waters generally >5 mg L^{-1}. This contrasts with the semi-enclosed channel region which shows wide excursions in daily oxygen levels from ~1 mg L^{-1} to ~12 mg L^{-1}, hypoxic conditions where levels decline to 2-3 mg L^{-1} were common. Instantaneous oxygen levels that drop below 4 mg L^{-1} are indicative of oxygen stress to many animals. The organic enrichment of the system is demonstrated by the moderate-high chlorophyll-*a* levels, organic rich sediments and accumulation of drift algae, observed during the deployment period. The large daily oxygen excursion is indicative of organic enrichment, where high rates of oxygen production from photosynthesis (carbon fixation) during daylight raise oxygen levels faster than ventilation to the atmosphere and rapid oxygen uptake after sunset stemming from respiration removes oxygen faster than it can be replaced by mixing from the atmosphere.

Salt Pond also consistently supports multiple moderate phytoplankton blooms where chlorophyll-a increases to >15 ug L^{-1}, with summer averages of 14-15 15 ug L^{-1} observed in the SMAST water quality monitoring program (2006-2012). Average chlorophyll levels over 10 ug L^{-1} have been used to indicate eutrophic conditions in embayments.

Salt Pond is functionally a basin with very limited fringing wetland habitat along its shoreline to the main basin. The sediments are currently soft muds rich in organic matter in the center deep portion of the pond, which has had periods of prolonged anoxia since at least the 1970's. The primary factor in the oxygen depletion of the deep waters is the strong salinity stratification of the deep basin, which restricts ventilation with the atmosphere. This stratification presently results from the depth of the basin and the highly restricted tidal exchange by the present inlet system. It is the shallower (<3m) margins of the basin that support some sandy areas and compacted muds with oxidized surface layers. All of the available information related to eelgrass within the Salt Pond Estuary, including the 2007 survey, indicate that no eelgrass is present within Salt Pond and eelgrass beds have not been observed historically, although small sparse patches of *Ruppia* occur in some shoreline areas. The absence of eelgrass beds is expected in this system given the high chlorophyll-*a* (averages 14-15 ug L^{-1}) and periodic low dissolved oxygen levels and high water column nitrogen concentrations. Given the absence of eelgrass at present and the lack of evidence of prior eelgrass habitat within this system, management should focus on benthic animal habitat, primarily within the marginal areas.

Overall, the infauna survey indicated that the shallow margin (<3 m) around the deep kettle "hole" is supportive of the moderate quality infaunal habitat within Salt Pond, showing areas that are moderately and significantly impaired. The sediments within the deep basin (>3

m) are overlain by anoxic bottom water and are devoid of benthic animals. Intermediate to these 2 regions, the tidal channel is depositional, with significant oxygen declines and significantly impaired benthic habitat. (Table VII-3).

The prolonged anoxia of the deep basin (>3 m) stems largely from the geomorphology of the basin, its deep water and the highly restricted tidal flushing of this system. The result is strong salinity stratification of the water column, which prevents down mixing of oxygenated surface waters. The result is the prolonged anoxia and lack of benthic animal habitat below ~3 meters depth. It should be noted that recovery of this habitat for benthic animals cannot occur until the physical stratification is significantly diminished. In contrast, the shallow area around the deep basin has aerobic bottom waters, relatively consolidated sediments and oxidized surface sediments. The result is high numbers of animals (~500 per sample), moderate-high numbers of species (17) with moderate diversity and Evenness. The community does have moderate numbers of organic enrichment-stress indicator species (Tubificids, Capitellids) comprising ~1/3 of the community. All of the community metrics, indicator species, numbers individuals and species and indices, document a benthic animal habitat moderately to significantly impaired by organic enrichment, hence ultimately nitrogen enrichment. The tidal channel to the inlet is currently supporting lower quality habitat than sediments at the same depth in the shallow margins of the main basin. The tidal channel is clearly supporting significantly impaired benthic animal habitat. This is documented in the low species numbers (5), low numbers of individuals (~100 per sample) and low diversity (1.5). Equally diagnostic, the community is dominated by organic enrichment-stress indicator species, which comprise more than 2/3 of the community. It appears that the "stress" is organic enrichment due to organic matter and macroalgal deposition within the channel. The result is periodic oxygen depletion (Figure VII-5) and sulfidic sediments. These environmental features are fully consistent with the community observations and the animal habitat designations described above. Note that communities with low numbers of stress indicator species and total species numbers of 20-25 are generally associated with high quality estuarine benthic habitats.

Classification of habitat quality necessarily included the structure of the estuarine basin, specifically that it is fully representative of a tidal embayment, as opposed to a tidal river or salt marsh basin and if a basin is structurally impaired or impaired by nitrogen enrichment.. Integration of all of the metrics clearly indicates that the shallow areas of Salt Pond are generally supporting benthic animal habitat that is moderately or significantly impaired. The proximate cause of impairment is organic matter enrichment and oxygen depletion, stemming ultimately from nitrogen enrichment. Total nitrogen levels within the estuary at present are >0.90 mg TN L^{-1}, a level generally found associated with a significant level of impairment of benthic animal habitat in southeastern Massachusetts estuaries.

The lack of historical eelgrass beds in Salt Pond and the present impairment to benthic animal habitat from nitrogen enrichment makes restoration of infauna habitat resource the primary focus for nitrogen management Determining the nitrogen target to restoring these habitats is the focus of the nitrogen management threshold analysis, below.

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

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

Within the Salt Pond Estuary the most appropriate sentinel "station" was to use the average of the 3 long-term monitoring stations (<3 m) in Figure VI-1. This average approach has been used in other open single basin estuaries throughout the MEP region. The average was selected because of the heterogeneity in the benthic animal habitat in this stratified basin and the need to meet acceptable quality conditions throughout the basin.

Following the MEP protocol, since eelgrass has not been documented in Salt Pond, restoration of infaunal habitat is the restoration goal. Infaunal animal habitat is a critical resource to the Salt Pond Estuary and estuaries in general. Since there are no unimpaired infaunal animal habitat areas remaining in the Salt Pond system, comparisons to the soft bottom basins of other nearby estuarine systems were relied upon for setting the nitrogen threshold for healthy infaunal habitat at a nitrogen level of TN <0.5 mg TN L^{-1}. This level was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 were found to be supportive of high infaunal habitat quality in this system. Similarly, in the Three Bays System, healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay), with impairment in areas where nitrogen levels of TN >0.5 mg TN L^{-1} (North Bay), and severe degradation at nitrogen levels of TN >0.6 mg TN L^{-1}. Present TN levels within the Salt Pond mixed layer during summer are ~0.90 mg TN L^{-1}, consistent with the observed lack of eelgrass beds and impaired benthic animal habitat.

Given the relatively low watershed nitrogen load to Salt Pond, it will be difficult to lower TN levels by ~0.4 mg L^{-1} to meet the threshold. This is consistent with the MEP measurements of significantly restricted tidal flows between Salt Pond and Vineyard Sound. This has been found in other estuaries with similar restrictions (e.g. Rushy Marsh Pond, Farm Pond). In such cases a reduction of the tidal restriction is needed to lower the level of nitrogen enrichment and restore the impaired habitats. This will likely be the case for Salt Pond, as well.

The response of Salt Pond nitrogen levels to reductions in watershed nitrogen inputs to achieve the TN threshold are developed in the next section (VIII.3).

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 Salt Pond System. Tidally averaged total nitrogen thresholds derived in Section VIII.2 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 Salt Pond. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

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

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Salt Pond 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
Salt Pond	3.488	0.000	-100.0%

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. Removal of 100% of the septic load from the watershed of Salt Pond results in an 73% reduction in total nitrogen load. Table VIII-4 shows the breakdown of threshold sub-embayment loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent 'worst-case' summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Vineyard Sound.

Table VIII-3. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Salt Pond 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
Salt Pond	4.571	1.277	-73.1%

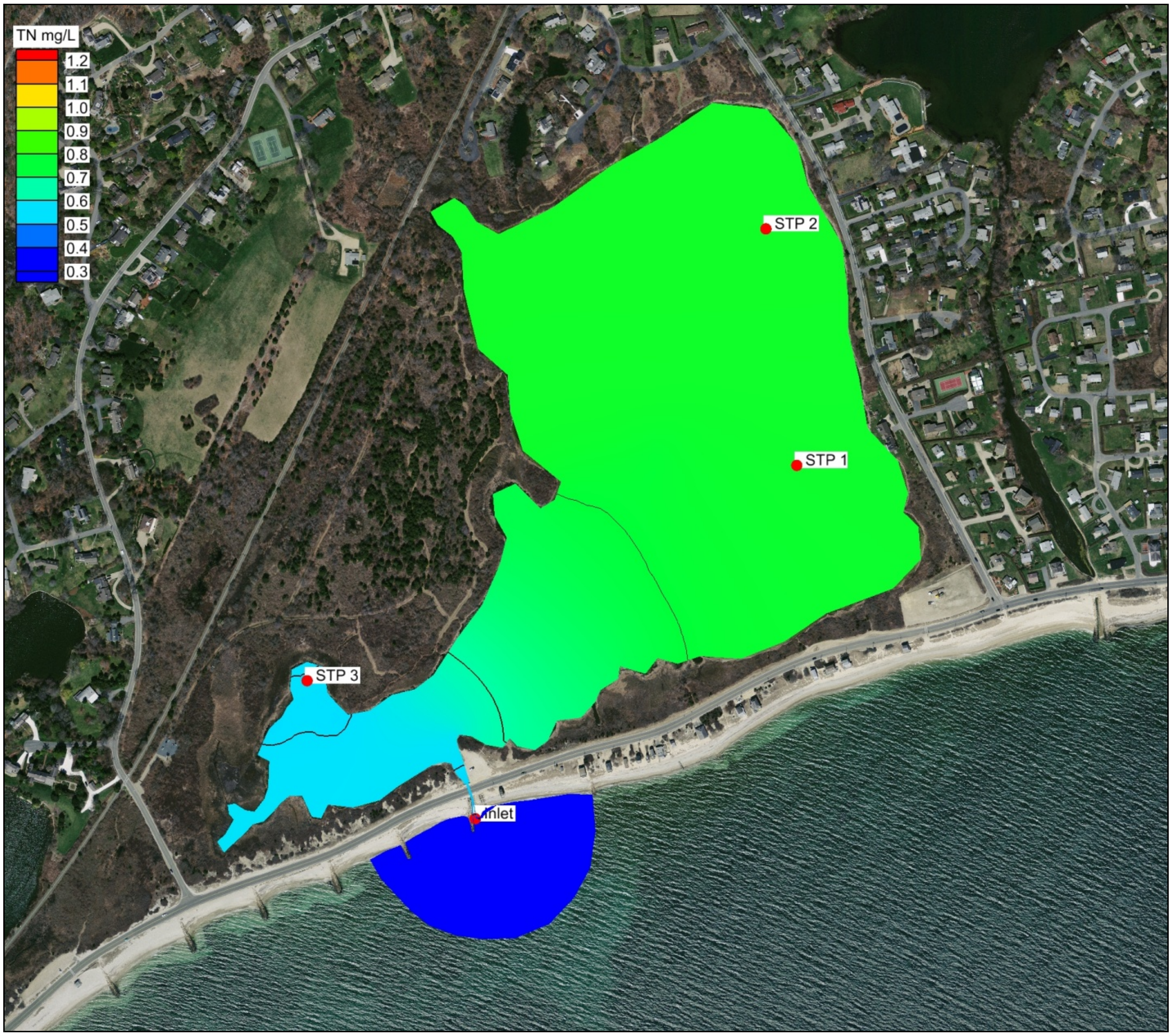

Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in Salt Pond System with 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the systems watershed. The removal of septic loads does not bring the total nitrogen concentrations within the system below the threshold of 0.5 mg/L. The three water quality stations within the pond are averaged together as the sentinel threshold station for Salt Pond System.

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Salt Pond 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)

Salt Pond	1.277	0.789	0.668

Comparison of model results between existing loading conditions and the selected loading scenario attempting to achieve the target TN concentrations at the sentinel station is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station, a different structural approach to increase circulation in the system is required for Salt Pond.

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Salt Pond system. Sentinel threshold station is the average of the three stations within the pond. The TN threshold is 0.5 mg L^{-1}.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Inlet		0.383	0.345	-9.9%
Salt Pond	SP-1	0.927	0.759	-18.2%
Salt Pond	SP-2	0.924	0.760	-17.8%
Salt Pond	SP-3	0.908	0.649	-28.5%

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

IX.1 WATER QUALITY IMPROVEMENTS TO SALT POND BY WIDENING THE INLET AND CULVERT UNDER PRESENT LOADING CONDITIONS

As a courtesy to the Town of Falmouth, a scenario was completed to assess potential water quality improvements resulting from increased tidal exchange in the Salt Pond estuary. Water quality improvements may be possible by improving tidal exchange in an estuary. Salt Pond has limited tidal exchange and could benefit from flushing improvements that cannot entirely be realized through the removal of septic loading. Tidal attenuation through the existing culvert is very high, where the average tide range in Salt Pond is less than 7% of the offshore range. Attenuation of the tide range in this system is primarily caused by an undersized inlet and culvert, sedimentation within the inlet and development of shallow flood and ebb shoals. In contrast, for Falmouth Harbor tide attenuation is near zero and tidal attenuation in Little Pond is approximately 57%, compared to the range offshore in Vineyard Sound. Widening the inlet and culvert offers the potential for increase tidal flushing to remove the excess nitrogen from the system, however, this analysis is only examining the impacts upon water quality. To fully assess inlet widening, a more detailed analysis of inlet stability, maintenance requirements, and potential environmental impacts would be required. The current inlet and culvert suffer from shoaling and sedimentation which obstructs the exchange of water between the Vineyard Sound and the pond. Sedimentation and shoaling will likely remain significant design and maintenance factor with wider inlets in the same location. A detailed analysis and study will be required to address design and sustainability issues associated with widening the inlet; the issues are beyond the scope of the MEP analysis.

The existing concrete box culvert under Surf Drive is 6-feet in width with an open bottom. The outer inlet varies in width due to the deterioration of the rubble mound jetties that bound the inlet. One possible solution to improve water quality within Salt Pond would be to increase the size of the inlet and culvert with the goal of improving tidal exchange between the sound and the pond. The Town has a narrow right away at the site of the current inlet, widening the inlet and culvert width to 12 feet, would maximize the inlet and culvert width while allowing room for rubble mound jetty structures to be reconstructed with 1:1.5 side slopes to protect the inlet while remaining within the existing right away. To quantitatively assess inlet improvements, model simulations were executed to simulate Salt Pond hydrodynamics and water quality with a wider 12.0-ft wide inlet and culvert.

Hydrodynamic model results for existing and improved inlet conditions are presented in Figure IX-1. In the top plot, tide attenuation is apparent by the lack of a tidal signature in the water surface elevations in the pond. The bottom plot shows a decrease in the mean water level in the pond, but the tidal signature in the pond is still heavily attenuated due to the significant shoaling within the pond and inlet. To significantly reduce tidal attenuation within the pond, dredging appears to be required through the outer inlet, culvert, and the flood shoal within Salt Pond. The flood shoals appears to be a significant factor in attenuation.

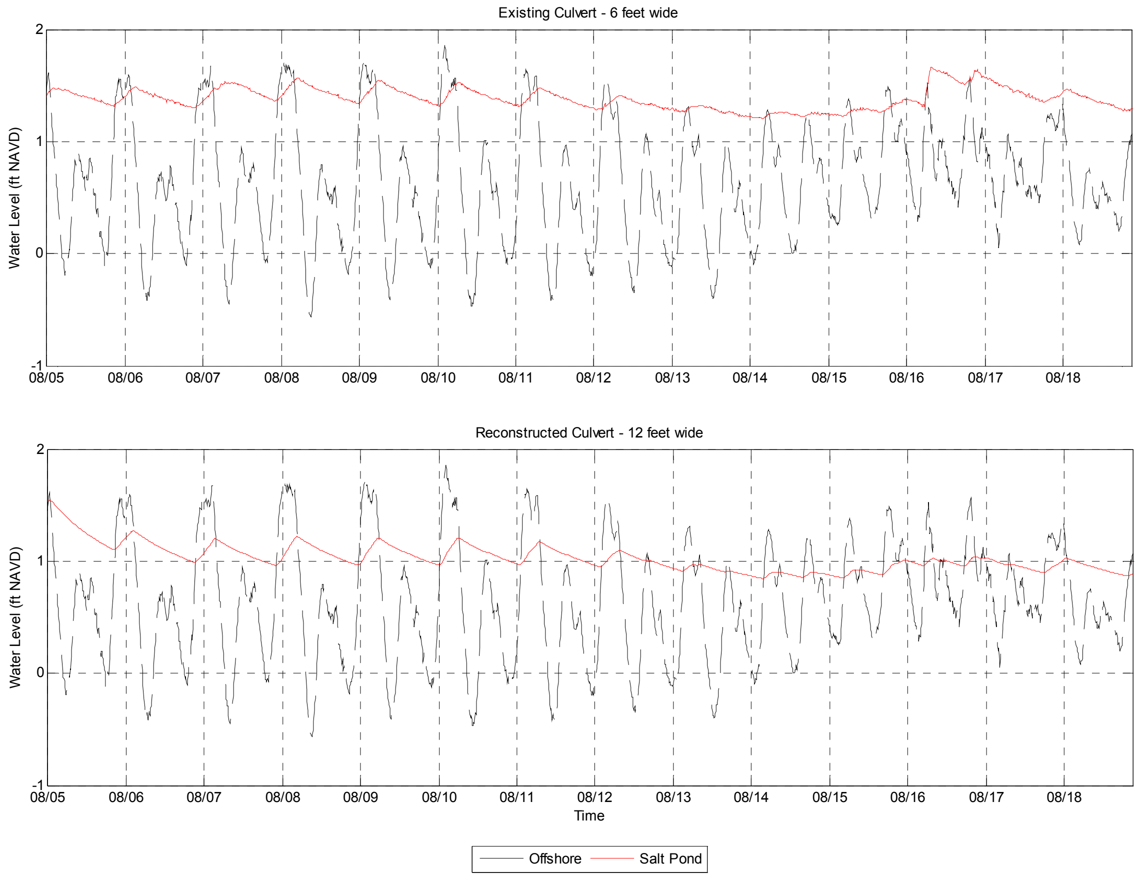

Figure IX-1. Plots showing a comparison of typical tides for modeled existing conditions (top plot) and the proposed reconstructed 12 ft-wide inlet (bottom plot).

A water quality model run was performed using the hydrodynamic model output of the proposed 12-foot wide inlet and culvert. The water quality simulation represents present loading conditions with the reconstructed inlet and the results are presented in Table IX-1. The TN concentrations are reduced with the widened inlet (i.e., up to an 10% reduction at WQ Station SP-3), however the reductions are not large enough to meet the threshold limits set for Salt Pond (Tidally averaged total nitrogen of 0.50 mg/L as derived in Section VIII.2).

Table IX-1. Comparison of model average total N concentrations from present loading and the widened 12 ft-wide inlet scenario with present loading, with percent change. Sentinel threshold station is the average of the three stations within the pond.

Sub-Embayment	monitoring station	present (6 ft) (mg/L)	Widened (12 ft) present (mg/L)	% change
Inlet		0.383	0.383	+0.2%
Salt Pond	SP-1	0.927	0.868	-6.3%
Salt Pond	SP-2	0.924	0.868	-6.1%
Salt Pond	SP-3	0.908	0.816	-10.1%

As an added scenario, the widened 12-foot inlet was rerun using the loading conditions developed for the Threshold Analysis as described in Section 8. The water quality simulation represents nitrogen load reductions within the system that call for 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the systems watershed with the widened inlet and culvert. The results are presented in Table IX-2. The TN concentrations are further reduced from the previous scenario with the widening and present loading conditions. The reductions in loading due to the full septic removal reduced average concentrations approximately 26%, but the reductions associated with the septic removal and inlet widening are not large enough to meet the threshold limits set for Salt Pond.

Table IX-2. Comparison of model average total N concentrations from present loading and the widened 12 ft-wide inlet scenario with threshold loading, with percent change. Sentinel threshold station is the average of the three stations within the pond.

Sub-Embayment	monitoring station	present (6 ft) (mg/L)	Widened (12 ft) threshold (mg/L)	% change
Inlet		0.383	0.345	-10.0%
Salt Pond	SP-1	0.927	0.714	-23.0%
Salt Pond	SP-2	0.924	0.716	-22.5%
Salt Pond	SP-3	0.908	0.588	-35.3%

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