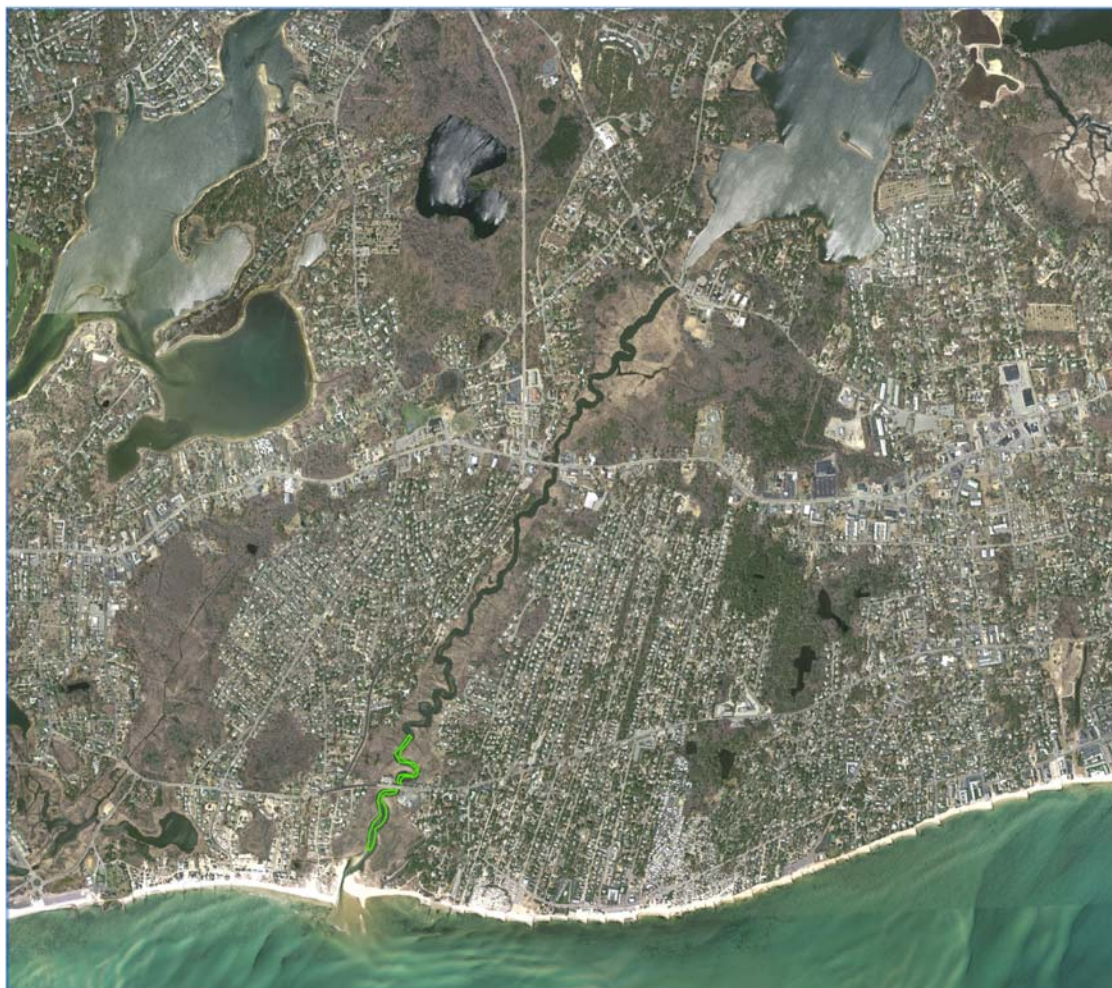# Massachusetts Estuaries Project

Linked Watershed-Embayment Approach to Determine Critical Nitrogen Loading Thresholds for the Swan Pond River Embayment System Town of Dennis, Massachusetts

University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

FINAL REPORT – October 2012

Massachusetts Estuaries Project

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Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Swan Pond River Embayment System, Dennis, Massachusetts

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 Swan Pond River embayment system, a coastal embayment primarily within the Town of Dennis, Massachusetts (very small portions of the overall watershed extend into the Towns of Harwich and Brewster). Analyses of the Swan Pond River embayment system was performed to assist the Town of Dennis 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 Dennis 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 Swan Pond River 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 Swan Pond River 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 Swan Pond River embayment system within the Town of Dennis 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 Dennis 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 Dennis has been working with the Town of Yarmouth that has also completed and implemented wastewater planning in other nearby regions not associated with the Swan Pond River system, specifically the Bass River embayment system which it shares with Dennis. In this manner, this analysis of the Swan Pond River system will be combined in the future with the results of the nutrient threshold analysis already completed of Bass River and the yet to be completed Sesuit Harbor MEP analysis to give the Town of Dennis the necessary results to plan out and implement a unified town-wide approach to nutrient management. The Town of Dennis with associated work groups have recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the 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 by the Buzzards Bay Project, the CCC models, and other relevant models. However, the Linked Model differs from other nitrogen management models in that it:

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

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

For a comprehensive description of the Linked Model, please refer to the *Full Report: Nitrogen Modeling to Support Watershed Management: Comparison of Approaches and Sensitivity Analysis*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm. A more basic discussion of the Linked Model is also provided in Appendix F of the *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm. The Linked Model suggests which management solutions will adequately protect or restore embayment water quality by enabling

towns to test specific management scenarios and weigh the resulting water quality impact against the cost of that approach. In addition to the management scenarios modeled for this report, the Linked Model can be used to evaluate additional management scenarios and may be updated to reflect future changes in land-use within an embayment watershed or changing embayment characteristics. In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries. Unlike many approaches, the Linked Model accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics and accommodates the spatial distribution of these processes. For an overview of several management scenarios that may be employed to restore embayment water quality, see *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm.

Application of MEP Approach: The Linked Model was applied to the Swan Pond River embayment system by using site-specific data collected by the MEP and water quality data from the Water Quality Monitoring Program initiated and conducted by the Dennis Water District and later transferred over to the Town of Dennis (2010), with technical guidance from the Coastal Systems Program at SMAST (see Chapter II). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Town of Dennis Planning Department, and watershed boundaries delineated by USGS. This land-use data was used to determine watershed nitrogen loads within the Swan Pond River embayment system and each of the systems sub-embayments as appropriate (current and build-out loads are summarized in Chapter IV). Water quality within a sub-embayment is the integration of nitrogen loads with the site-specific estuarine circulation. Therefore, water quality modeling of this tidally influenced estuary included a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Once the hydrodynamics of the system was quantified, transport of nitrogen was evaluated from tidal current information developed by the numerical models.

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

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

Threshold nitrogen levels for the embayment systems in this study were developed to restore or maintain SA waters or high habitat quality. High habitat quality was defined as supportive of eelgrass and infaunal communities. Dissolved oxygen and chlorophyll a were also considered in the assessment.

The nitrogen thresholds developed in Section VIII-2 were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Swan Pond River embayment 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 Swan Pond River system. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen reduction guidelines for nitrogen management of the Swan Pond River embayment system in the Town of Dennis. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. The MEP analysis has initially focused upon removal of nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of each embayment system. The concept was that since nitrogen loads associated with wastewater generally represent 75% - 79% of the controllable watershed load to the Swan Pond River 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 this system.

2. Problem Assessment (Current Conditions)

The Swan Pond River Embayment System is a complex estuary composed of two functional types of component basins: open water embayment (Swan Pond) and tidal river (Swan Pond River), with the upper reaches supporting significant salt marsh area. Each of these functional components has different natural sensitivities to nitrogen enrichment and organic matter loading. In addition, the extensive salt marsh introduces a level of natural organic enrichment which must be considered as part of the ecological assessment. Evaluation of eelgrass and infaunal habitat quality in the Swan Pond River system considered the natural structure of the system and the systems ability to support eelgrass beds and the types of benthic infaunal communities typically observed in a healthy estuarine system. A habitat assessment was conducted throughout the Swan Pond River embayment system based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements of dissolved oxygen and chlorophyll, and benthic community structure.

At present, the Swan Pond River is showing differences in nitrogen enrichment and habitat quality among its various component basins (Section VIII-1), with regions of clearly impaired habitat. The tidal waters of the Swan Pond River Embayment System are currently listed under the State Surface Water Classification as SA waters. The Swan Pond River Estuary is not presently meeting the water quality standards for SA waters. The result is

that as required by the Clean Water Act, TMDL processes and management actions must be developed and implemented for the restoration of resources within this estuary.

Overall, the system is showing some nitrogen-related habitat impairment within each of its component basins, however, there is a strong gradient. Swan Pond is a significantly impaired basin relative to benthic animal habitat, since it historically has not supported eelgrass. Nitrogen enrichment (through inputs and naturally low tidal exchange) has resulted in frequent large phytoplankton blooms, periodic hypoxia/anoxia, large macroalgal accumulations and a benthic community comprised of stress indicator species. The Swan Pond River is also nitrogen enriched, but has less nitrogen enrichment based primarily on its structure and high relative water turnover. While the lower reach currently supports only high quality to moderately impaired benthic habitat, its loss of historical eelgrass beds indicates that it has become a significantly impaired basin relative to eelgrass habitat. The upper tidal reach of the Swan Pond River, between Swan Pond and the lower River, is intermediate in habitat quality. The upper tidal reach (above Rt. 28) is moderately to significantly impaired based upon its benthic animal habitat, due primarily to organic and nitrogen rich waters ebbing from Swan Pond and natural enrichment processes associated with its extensive wetlands.

The frequency and duration of periodic hypoxia/anoxia in the Swan Pond system also indicates significant habitat impairment, consistent with nitrogen enrichment. The oxygen levels within Swan Pond indicate a coastal salt pond highly organic and nitrogen enriched with large daily oxygen excursions, large phytoplankton blooms and periodic hypoxia/anoxia. Bottom water oxygen frequently declined to <3 mg L^-, with periodic anoxia (<0.2 mg L^{-1}). These oxygen conditions are consistent with the parallel levels of organic and nitrogen enrichment as demonstrated by the regular algal blooms (phytoplankton, macroalgae) documented over the past 6 years and demonstrable large daily excursions in oxygen levels (10-15 mg L^{-1}). Consistent with the large diurnal oxygen excursions and frequent oxygen depletion in bottom waters were large phytoplankton blooms and macroalgal accumulations.

Since Swan Pond is the terminal basin of this estuary, it has the most enriched conditions within the estuary, with the Swan Pond River being intermediate in nutrient related water quality between Swan Pond and the high quality waters of Nantucket Sound. The tidal river has higher water quality than the terminal basin of Swan Pond, primarily as a result of its much higher water turnover and its more direct access to the high quality waters of Nantucket Sound which enter during flood tides. The upper reach of the River is primarily organic and nitrogen enriched by the ebbing waters of Swan Pond and direct watershed nitrogen loads, with some natural enrichment from the associated extensive tidal wetlands. Consistent with the observed levels of nitrogen and organic enrichment coupled with the distribution of oxygen depletion throughout the estuary, benthic animal habitat showed a wide range in quality. The upper terminal basin of Swan Pond is clearly significantly impaired while the Swan Pond River (lower portion of the system) presently shows a gradient in benthic animal habitat quality from low/intermediate levels in the upper reach to moderately high quality habitat in the lower reach. The gradient in benthic animal habitat parallels the key water quality parameters of oxygen and chlorophyll-a.

As documented by the MEP, the pattern of infaunal habitat quality throughout the Swan Pond River system is consistent with measured dissolved oxygen concentration, chlorophyll, nutrients and organic matter enrichment in this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland dominated such as Swan Pond or tidal embayment dominated, such as Swan Pond River. Based upon this analysis it is clear that the upper regions of the Swan Pond River Embayment System are significantly impaired by nitrogen and organic matter enrichment

while the lower basins are presently supporting high quality to moderately impaired benthic animal habitat.

3. Conclusions of the Analysis

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

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

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Town of Dennis Swan Pond River embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 75% - 79% of the controllable watershed nitrogen load to the embayment was from wastewater.

A major finding of the MEP clearly indicates that a single total nitrogen threshold can not be applied to Massachusetts' estuaries, based upon the results of over 50 estuarine analyses undertaken by the Massachusetts Estuaries Project, such as was completed in the Great, Green and Bournes Pond Systems, Popponesset Bay System, the Hamblin / Jehu Pond / Quashnet River analysis in eastern Waquoit Bay, the analysis of the nearby Lewis Bay and Rushy Marsh system, the analysis of Parkers River and Bass River shared by the Towns of Yarmouth and Dennis and the Pleasant Bay and Nantucket Sound embayments associated with the Town of Chatham (to mention a few). This is almost certainly going to be true for the other embayments within the MEP area, as well, inclusive of Swan Pond River River.

The threshold nitrogen levels for the Swan Pond River embayment system in Dennis were determined as follows:

Swan Pond River Threshold Nitrogen Concentrations

- Following the MEP protocol, the restoration target for the Swan Pond River system should reflect both recent pre-degradation habitat quality and be reasonably achievable. Based upon the assessment data (Chapter VII), the Swan Pond River system is presently supportive of habitat in varying states of impairment, depending on the component sub-basins of the overall system (e.g. Swan Pond or Swan Pond River).

- The primary habitat issue within the Swan Pond River Embayment System relates to the loss of eelgrass beds from the lower portion of the main channel that constitutes a portion of the overall system.   This loss of eelgrass classifies this area of the system as "significantly impaired". The impairments to both the infaunal habitat in Swan Pond and the loss of eelgrass habitat within the lower portion of the system are supported by the variety of other indicators, oxygen depletion, chlorophyll, and TN levels, which support

the conclusion that these impairments are the result of nitrogen enrichment, primarily from watershed nitrogen loading.

- The results of the water quality and infaunal data, coupled with the temporal trends in eelgrass coverage, clearly support the need to lower nitrogen levels within the Swan Pond River system in order to restore eelgrass habitat. The observed loss of eelgrass, presence of drift algae, oxygen and chlorophyll levels and benthic community structure suggests a system beyond the nitrogen threshold level that would be supportive of eelgrass. The high nitrogen levels within the Swan Pond River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates. This loading increase the nitrogen levels in the incoming tidal waters (0.3 mg L-1) by several fold (see Section VI). As there is no evidence of eelgrass coverage within Swan Pond or the upper reach of the Swan Pond River within the past 6 decades, they should not be considered for eelgrass restoration. In contrast, documented eelgrass within the lower Swan Pond River make restoration of this resource a primary target for overall restoration of the Swan Pond River System. Restoration of this habitat will require appropriate nitrogen management. Note that restoration of this eelgrass habitat will necessarily result in restoration of other resources throughout the system, particularly within the upper estuary.

- The approach for determining nitrogen loading rates that will support acceptable habitat quality throughout an embayment system is to first identify a sentinel location within the embayment and secondly, to determine the nitrogen concentration within the water column that will restore the 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. The sentinel station for the Swan Pond River Estuary is located at the long-term water quality monitoring station within middle of the lower reach of the River (SWP-2). This site was selected based upon its location at the upper most extent of the documented eelgrass coverage in this estuary.

- The observed loss of eelgrass, moderate oxygen and chlorophyll levels and benthic community structure within the lower reach of the Swan Pond River, indicate a system beyond the nitrogen threshold level that would be supportive of eelgrass, but relatively close to the level for supporting high quality infaunal habitat. The tidally averaged nitrogen levels for this lower reach were 0.651 mg N L^{-1} over the entire reach and 0.564 mg N L^{-1} in the region that historically supported eelgrass beds. These TN levels are slightly above the level typically supportive of infaunal communities (0.5-0.55 mg N L^{-1}), but well above levels supportive of eelgrass beds as found in a variety of MEP assessments of Cape Cod estuaries. Similarly, average total nitrogen levels within Swan Pond (1.056-1.207 mg N L^{-1}) are well above levels found in basins supportive of high quality benthic animal habitat.

- Based on all indicators and with the sentinel station located at the uppermost extent of the historical eelgrass coverage, the target nitrogen concentration (tidally averaged TN) for *restoration of eelgrass at the sentinel location within the lower reach of the Swan Pond River was determined to be 0.40 mg TN L^{-1}*. As there has not been eelgrass habitat within the Swan Pond River Estuary for over a decade, this primary threshold was based upon comparison to other local embayments of similar depths and structure that have been reviewed under MEP analysis.

- Although the nitrogen management target is restoration of eelgrass habitat, benthic infaunal habitat quality must also be supported as a secondary condition. At present, the regions with moderately to significantly impaired infaunal habitat within the Swan Pond River Embayment System have average tidal total nitrogen (TN) levels of 0.66 to 1.21 mg N L^{-1}. The observed impairments throughout this estuary are consistent with observations by the MEP Technical Team in other estuaries along Nantucket Sound (e.g. Perch Pond, Bournes Pond, Popponesset Bay) where levels <0.5 mg N L^{-1} were found to be supportive of healthy infaunal habitat and where moderately impaired habitat was found at ~0.6 mg N L^{-1}. Similarly, moderate impairment was also observed at TN levels (0.535-0.600 mg N L^{-1}) within the Wareham River Estuary, while the Centerville River system showed moderate impairment at tidally averaged TN levels of 0.526 mg N L-1 in Scudder Bay and at 0.543 mg TN L-1 in the deep middle reach of the Centerville River. Based upon these observations, the MEP Technical Team concluded that an upper limit of 0.50 mg N L-1 tidally averaged TN would support healthy infaunal habitat in the Swan Pond River, but given the shallow nature of Swan Pond and its significant salt marsh resources, _a tidally averaged TN of <0.55 mg N L-1 was appropriate for healthy infaunal communities_. It should be emphasized that these secondary criteria values were not used for setting nitrogen thresholds in this embayment system.

It is important to note that the analysis of future nitrogen loading to the Swan Pond River 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 Swan Pond River estuarine system is that restoration will necessitate a reduction in the present (Dennis and Brewster 2009, Harwich 2006) nitrogen inputs and management options to negate additional future nitrogen inputs.

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
Table ES-1. Existing total and sub-embayment nitrogen loads to the estuarine waters of Swan Pond River estuary system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations.										
SYSTEMS										
Swan Pond	0.863	5.065	13.058	--	18.123	1.885	-3.473	16.535	0.49-1.02	0.40
Swan Pond River - North	0.268	1.625	8.411	--	10.036	0.104	0.385	10.525	0.88-1.43	--
Swan Pond River - South	0.463	4.038	11.518	--	15.556	0.233	-1.346	14.276	1.04-1.55	--
System Total	**1.594**	**10.728**	**32.987**	**0.00**	**43.715**	**2.222**	**-4.434**	**41.336**	**0.49-1.55**	**0.400[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 2005 – 2010 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 Swan Pond River at water quality station SWP-2.

Table ES-2.	Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Swan Pond River estuary system, Town of Dennis, 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						
Swan Pond	18.123	5.063	1.885	-1.185	5.763	-72.1%
Swan Pond River - North	10.036	1.625	0.104	0.150	1.879	-83.8%
Swan Pond River - South	15.556	4.038	0.233	-0.586	3.518	-74.0%
System Total	**43.715**	**10.726**	**2.222**	**-1.621**	**11.160**	**-75.5%**

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

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I. INTRODUCTION

The Swan Pond River Embayment System is a complex estuary located within the Town of Dennis on Cape Cod, Massachusetts and exchanges tidal waters with Nantucket Sound to the south (Figure I-1). The Swan Pond River Estuary is comprised of a long meandering tidal river connecting a large kettle pond, Swan Pond, to the high quality marine waters of Nantucket Sound. Drowning of coastal river valleys to form tidal rivers and associated salt marshes has occurred gradually as a result of rising sea level following the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~18,000 years ago. The barrier beach around the inlet was formed from marine sands and gravel deposited by shoreline coastal processes, as sea level rose. The salt marsh which dominates both banks of the lower and upper portion of Swan Pond River lies along the narrow tidal channel (the "Swan River" portion of the system) that connects the Swan Pond basin to Nantucket Sound. The upper reach of the river and lower region of Swan Pond presently support significant wetland resources. Swan Pond is generally the same depth as Swan River, both being only 1.0 - 1.5 meters deep. The upper portion of the system is dominated by open water with significant wetlands, particularly in the region of the lower Pond/upper River. In fact the shoreline of nearly all of Swan Pond is bordered by a 50 feet to 100 feet of fringing salt marsh. The watershed to Swan Pond is primarily characterized as upland (with forest and single family residential development). The tidal river, Swan Pond River, is functionally divided into an upper and lower river, falling north and south of the Route 28 bridge, respectively. Overall, the Swan Pond River System presently supports some of the major salt marsh resources along this region of the Cape Cod coast.

The embayment is located within the sands and gravel outwash of the Harwich Outwash Plain Deposits overlying Pleistocene deposits, formed after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet. The material is highly permeable and varies in composition from well sorted medium sands to coarse pebble sands and gravels. The soils of the entire watershed are therefore composed in such a manner that direct rainwater run-off is typically rather low and most freshwater inflow to the estuary via groundwater discharge or groundwater fed surface water flow. The larger uppermost basin Swan Pond River is thought to have been formed from the melting of a block of ice trapped in the outwash deposits. Originally Swan Pond and Swan River were isolated from the sea, but as a result of rising sea level following the last glaciation approximately 18,000 years BP, they became estuarine systems ~6,000-8,000 years BP.

While the embayment basins of the Swan Pond River are distributed entirely within the Town of Dennis, its watershed includes portions of the Towns of Dennis, Harwich and Brewster. Almost all of the watershed recharge enters Swan Pond River through direct groundwater seepage around its shoreline. Only a small portion of the freshwater entry is via surface water discharge via the creek that flows along Hydaway Lane and discharges into the upper portion of Swan Pond (Hydaway Creek). The number of sub-embayments (Swan Pond River-lower, Swan Pond River-upper and Swan Pond) comprising the overall Swan Pond River System greatly increases the shoreline and decreases the travel time of groundwater from the sites of watershed nitrogen inputs to estuarine regions of discharge.

The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shores 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 shoreline. Like almost all embayment systems on Cape Cod, Swan Pond River is at risk of eutrophication

Figure I-1.   Major components of the Swan Pond River Embayment System assessed by the Massachusetts Estuaries Project. Tidal waters enter the main channel of the Swan Pond River system through a single inlet from Nantucket Sound. Freshwaters enter from the watershed primarily through direct groundwater discharge to the estuary and through one (1) surface water discharge (a small creek into Swan Pond).

from increasing nitrogen loads entering via groundwater from land-use shifts within its watershed. However, given its structure, Swan Pond River, is more susceptible to nitrogen enrichment than most estuaries in the region. This sensitivity to nitrogen inputs stems from the system's tidal hydrodynamics as structured and influenced by its morphology. The long meandering tidal river presents a high degree of natural friction to tidal flows resulting in an attenuation of the tide range that occurs in the upper River versus the range at the tidal inlet. In addition, the large basin of Swan Pond requires that a large volume of water be transported through the river on the flood and ebb tides to effect adequate flushing to reduce its nitrogen levels. The combined result of the friction in the tidal river and the large area of Swan Pond result in a tide range in Swan Pond of ~1 ft compared to ~3 ft at the inlet. The relatively low exchange of Swan Pond waters during each tidal cycle allows for a greater build up of nitrogen levels, making the Pond more sensitive than if the system had a shorter river and larger tide range in the pond. This is seen to a lesser extent in the nearby Parkers River System in Yarmouth, which presently has a reduced tide range in the large upper basin of Seine Pond and also shows clear signs of nutrient-related impairment, in part due to its inability to efficiently flush nutrients from the upper basin. However, in the Parkers River case, the reduced flushing in the large upper basin results primarily from a man-made physical restriction by the Route 28 Bridge. Swan Pond River, as a result of its unique hydrodynamics and relatively high watershed nitrogen inputs, is presently one of the most nitrogen enriched estuaries on Cape Cod and is showing nitrogen related water quality impairments of key estuarine habitats.

The Swan Pond River Embayment System is a complex (drown river valley) estuary exchanging tidal waters with the high quality waters of Nantucket Sound through a single inlet that is partially armored with a single jetty on its western shore (Figure I-1). At present the western jetty is "full" and sediment is periodically transported into the inlet channel, where a sand delta has formed. The tidal inlet is periodically dredged (most recently winter 2010) to remove accumulated sand and to maximize tidal flushing between the estuary and Sound. The Swan Pond River Estuary contains high salinity waters throughout its tidal reaches, generally 30 parts per thousand (ppt) near the inlet and 23 ppt in Swan Pond. The high salinities reflect the dominance of tidal flows, rather than freshwater inflows, in structuring this system. Sediment transport and deposition associated with coastal processes including coastal storms deposit sands within the tidal inlet resulting in periods of lower tidal flows and a small "freshening" of the upper estuary. Historically, this may have led to periodic, but significant, lowering of salinities compared to the present condition, where the inlet is regularly maintained by the Town of Dennis.

The primary ecological threat to the Swan Pond River system as a coastal resource is degradation resulting from nutrient enrichment. Like many enclosed estuarine systems on Cape Cod, it also has periodic bacterial contamination issues currently related to stormwater run-off from the watershed and likely animal sources primarily associated with waterfowl and the extensive tidal wetlands. Bacterial contamination causes periodic closures of shellfish harvest areas within the Swan Pond (SC36.1 and SC36.2 growing areas) sub-embayment, but this contamination does not result in habitat degradation (i.e. loss of animals and plants). In contrast, loading of the critical eutrophying nutrient, nitrogen, to the Swan Pond River System has impaired its animal and plant habitats and resulted in ecological changes and lost marine resources.

Nitrogen inputs to Swan Pond River have greatly increased over the past several decades with a further increase of ~10% possible given remaining developable land and present zoning (Section IV). The nitrogen loading to the Swan Pond River Estuary, like almost all embayments

in southeastern Massachusetts, results primarily from on-site disposal of residential (and some commercial) wastewater.

The Town of Dennis, like most of Cape Cod, has seen rapid growth over the past five decades and does not have a centralized wastewater treatment system or decentralized facilities that remove nitrogen.   As such, all of the developed areas in the Swan Pond River watershed are not connected to any municipal sewerage system and wastewater treatment and disposal is primarily through privately maintained on-site septic systems. As present and future increased levels of nutrients impact the coastal embayments in the Town of Dennis, water quality degradation will increase, with additional impairment and loss of environmental resources, as evidenced by the recent macroalgal blooms within Swan Pond.

As the primary stakeholder to the Swan Pond River System, the Town of Dennis (through the Dennis Water District) was among the first communities in southeastern Massachusetts to become concerned over degradation of its coastal waters.   Concern over declining habitat quality within its embayments led directly to the establishment of a comprehensive water quality monitoring program aimed at determining the degree to which waters of the Town's embayments may be impaired.   The Dennis Water District Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD, with analysis by the Coastal Systems Analytical Facility (Sara Sampieri, Laboratory Manager, ssampieri@umassd.edu, 508-910-6325). Over the past several years the Dennis Program has also operated in a coordinated manner with the Town of Yarmouth water quality monitoring program as a result of the shared embayment of Bass River. In addition to assessing the health of the estuaries in Dennis, the water quality monitoring program provides the required quantitative watercolumn nitrogen data (2005-2010) for validation of the MEP's Linked Watershed-Embayment Modeling Approach used in the present study. Entry into the MEP and TMDL compliance monitoring depends upon Town-supported water quality monitoring, as guided by SMAST.

The common focus of the Dennis Water Quality Monitoring effort has been to gather site-specific data on the current nitrogen-related water quality throughout the Swan Pond River System and determine its relationship to watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between watershed loading, tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program and adds several additional layers of high end data collection linking watershed characteristics to estuarine function.   The MEP approach includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each major sub-embayment. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater planning and nitrogen management alternatives development needed by the Town of Dennis in association with other watershed stakeholders, the Towns of Harwich and Brewster).   While the completion of this complex multi-step process of rigorous scientific investigation to support watershed-based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns to develop and evaluate the most cost effective nitrogen management alternatives to restore this valuable coastal resource which is currently being degraded by nitrogen overloading. It is important to note that the Swan Pond River System has been significantly altered by human activities over the past ~100 years or more (see Section I.2, below). As a result, the present nitrogen "overloading" appears to result partly from alterations to the geomorphology and

ecological systems. These alterations subsequently affect nitrogen loading and transport within the watershed and influence the degree to which nitrogen loads impact the estuary. Therefore, restoration of this system should focus on managing nitrogen through both management of nitrogen loading within the watershed and restoration/management of processes which serve to lessen the amount or impact of nitrogen entering the estuary, for example hydrodynamic solutions.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

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

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of Dennis) 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 reliable planning. Although it may be technically complex to implement, results must be understandable to the regulatory community, town officials, and the general public.

The Massachusetts Estuaries Project represents the next generation of watershed-based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MassDEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and other MEP partners 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 MEP approach

was selected after extensive review by the MassDEP and USEPA and associated scientists and engineers. It has subsequently been applied to more than 40 estuaries and reviewed by other state agencies, municipalities, non-profit environmental organizations, engineering firms, scientists and private citizens. Over the course of the extensive reviews, the MEP approach has proven to be robust and capable of yielding quantitative results to support management of a wide variety of estuaries.

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide MassDEP and municipalities in the region with technical guidance to support restoration and protection 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 1) the sources of the pollutant of concern (in this case, nitrogen) from both point and non-point sources, 2) the allowable load of the pollutant to meet the state water quality standards, and 3) the load allocation from all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an outline of an implementation plan.

For this project, the MassDEP recognizes that there are likely to be multiple ways to achieve the desired goals/allowable load, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, MassDEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly for shared watersheds) through the Comprehensive Wastewater Management Planning process.

The MEP nitrogen threshold analysis includes site-specific habitat assessments and watershed/embayment modeling approaches to develop and assess various nitrogen management alternatives for meeting selected nitrogen goals supportive of restoration/protection of embayment health.

The major MEP nitrogen management goals are to:

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

The core of the Massachusetts Estuaries Project analytical method is the Linked Watershed-Embayment Management Modeling Approach. This approach represents the "next generation" of nitrogen management strategies. It fully links watershed inputs with embayment circulation and nitrogen characteristics. The Linked Model builds on and refines well-accepted

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

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

The Linked Model has been applied for watershed nitrogen management of more than 40 embayments throughout southeastern Massachusetts as of the date of this report. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options. The MEP Technical Team, through SMAST-UMD, has conducted more than 200 scenarios to date.

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

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

- Watercolumn Monitoring - multi-year embayment nutrient sampling
- Hydrodynamics -
  - embayment bathymetry
  - site specific tidal record
  - current records (in complex systems only)
  - hydrodynamic model
- Watershed Nitrogen Loading
  - watershed delineation
  - stream flow (Q) and nitrogen load

- land-use analysis (GIS)
- watershed N model
- Embayment TMDL - Synthesis
  - linked Watershed-Embayment N Model
  - salinity surveys (for linked model validation)
  - rate of N recycling within embayment
  - D.O record
  - Macrophyte survey
  - Infaunal survey

Nitrogen Thresholds Analysis
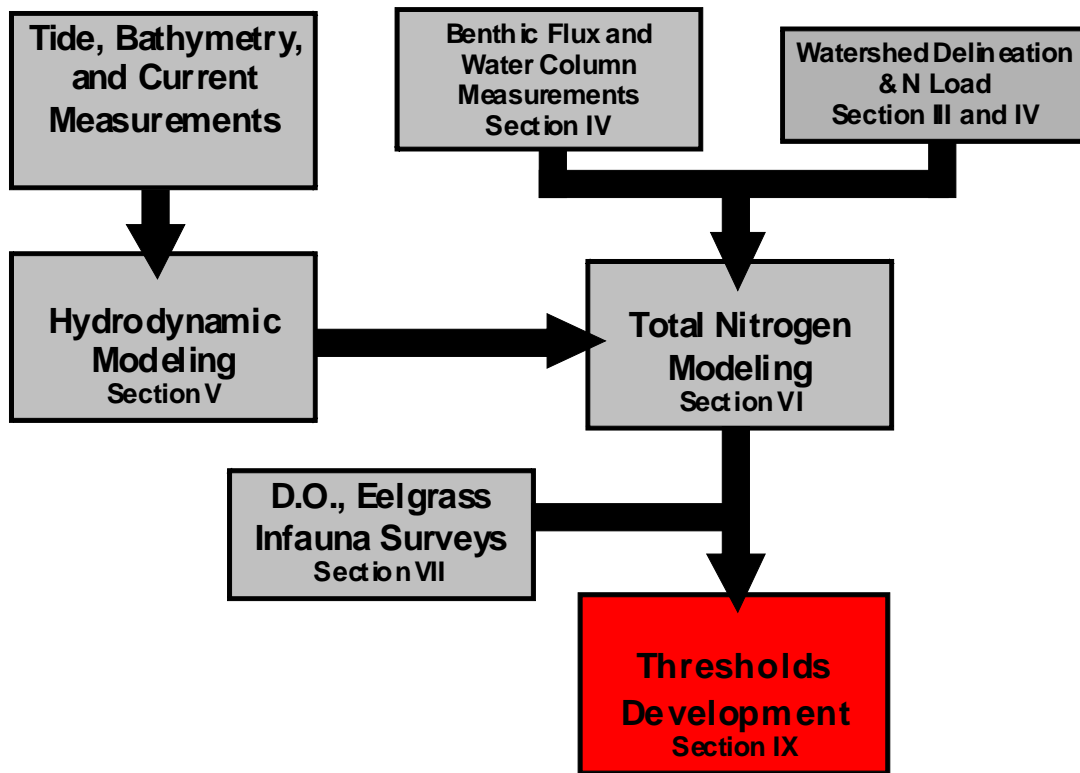
Figure I-2.   Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Note that the approach is not a single model, but a series of models linked by scientists and engineers who validate outputs and inputs to each model.

Within the overall Swan Pond River System is seen a diversity of estuarine habitats, including the main tidal channel portion of Swan Pond River which operates as tidal river with extensive wetlands and uppermost terminal basin of Swan Pond, the main embayment basin, that is a shallow kettle basin with fringing salt marsh along most of its shoreline. The Swan Pond River Estuary presently supports extensive wetlands throughout most of its tidal reach.

The Swan Pond River Estuary contains high salinity waters throughout its tidal reach, from ~30 parts per thousand (ppt) near the inlet to ~23 ppt in the upper basin of Swan Pond. The high salinities reflect the dominance of tidal flows, rather than freshwater inflows, in structuring these systems. Tidal forcing for this embayment system is generated from Nantucket Sound.

Nantucket Sound exhibits a moderate to low tide range, with a mean range of about 2.5 to 3.5 ft. Since the water elevation difference between Nantucket Sound and the Swan Pond River System 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 ~4.5 ft, while off Wellfleet Harbor is ~10 ft).

Tidal damping (reduction in tidal amplitude) through an embayment can range from negligible, indicating "well-flushed" conditions, to large attenuations caused by constricted channels and marsh plains or long meandering channels (causing high friction). These latter attenuations indicate a "restricted" system, where tidal flow and the associated flushing are inhibited. Tidal data in the Swan Pond system indicate that its tidal inlet is operating efficiently, because of periodic dredging to maintain navigation and flushing, and that the bridges over the Swan Pond River, including the Rt. 28 Bridge are also not presently restricting tidal flow. However, the long meandering tidal river has a naturally high friction to water flow and this causes a natural damping of tide range in the upper estuary (present tide range 0.9 ft).

I.2 NUTRIENT LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Swan Pond River System, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer minerals (Weiskel and Howes 1992). Since even Cape Cod "rivers" are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant-available nitrate, is readily transported through the generally, well-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). This export of watershed nitrogen is then paired with the coastal estuary ecosystems, which tend to

have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidal reaches within the Swan Pond River Estuary follow this general pattern, where the primary nutrient of eutrophication in this 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. This point can be termed the "nutrient threshold" and in estuarine management this threshold sets the target nutrient level for restoration or protection. Because nearshore coastal salt ponds and embayments are the primary recipients of nutrients carried via surface and groundwater transport from terrestrial sources, it is clear that activities within the watershed,

often miles from the water body itself, can have chronic and long lasting impacts on these fragile coastal environments.

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

Unfortunately, the upper reaches of the Swan Pond River System are beyond their ability to assimilate additional nutrients without impacting their ecological health. The MEP analysis clearly indicates that the system is presently impaired by nitrogen overloading and does not meet the Commonwealth's Water Quality Standards. Nitrogen levels are elevated throughout the System and eelgrass beds have not been observed within the Swan Pond River system as far back as 1995. The large macroalgae blooms in Swan Pond in 2006 and 2009 have provided dramatic evidence of habitat impairment and pushed nitrogen management into public forums. It is important to note that the present nitrogen enrichment of the Swan Pond River Embayment System results from the combination of increasing nitrogen inputs from its contributing watershed coupled with moderate to low flushing in the upper portions of the system. The MEP analysis evaluates both of these processes and any efficient management plan will likely include modifications of both nitrogen loading and, if possible, the tidal flushing rate.

Nitrogen related habitat impairment within the Swan Pond River Estuary shows a gradient of high to low moving from the upper basin of Swan Pond to the tidal inlet. The result is that nitrogen management of the three primary sub-embayments to the Swan Pond River system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed "eutrophication" and in certain instances can occur naturally over long periods of time. When the nutrient loading is rapid and primarily from human activities leading to changes in a coastal watershed, nutrient enrichment of coastal waters is termed "cultural eutrophication". Although the influence of human-induced changes has clearly increased nitrogen loading and contributed to the degradation in ecological health, it is sometimes possible that eutrophication could potentially occur without human influence and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a "pristine" system.

I.3 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important "boundary conditions" (e.g. watershed-derived and offshore nutrient inputs) for water quality modeling of the Swan Pond River System; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within each component of 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 the Swan Pond River System, including the tributary sub-embayment of Swan Pond. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for each of the systems. Once the hydrodynamic properties of each estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape groundwater model for sub-watershed areas designated by MEP. Almost all watershed sourced nitrogen entering the Swan Pond River System is transported groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Swan Pond River system were provided by the Dennis Water District Water Quality Monitoring Program (a coordinated effort between the District and the Coastal Systems Program at SMAST). Measurements of the salinity and nitrogen distributions throughout estuarine waters of the Swan Pond River System (2005-2010) were used to calibrate and validate the water quality model (under existing loading conditions).

I.4 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Swan Pond River System for the Towns of Dennis, Harwich and Brewster. 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 Towns of Dennis, Harwich and Brewster and Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Nantucket Sound (Section IV). Intrinsic to the calibration and validation of the linked-watershed embayment

modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section IV. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration/protection of the Swan Pond River estuary in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined system threshold for restoration or protection (Section VIII). This latter assessment represents only one of many solutions and is produced to assist the Towns in developing a variety of alternative nitrogen management options for this system. References are provided in Section IX.

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 dependant 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 ponds, it is not a necessarily a part of the natural evolution of a system.

In most marine and estuarine systems, such as the Swan Pond River Embayment System, the limiting nutrient, and thus the nutrient of primary management concern, is nitrogen. In large part, if nitrogen levels are controlled, either by controlling inputs or flushing, then habitat impairments resulting from eutrophication are 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 older 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 MEP 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 Swan Pond River 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.

Studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Swan Pond River System over the past decade along with site visits by MEP technical staff documenting macroalgal blooms that have occurred in Swan Pond. Available studies that were integrated into the present Massachusetts Estuaries Project effort to develop a nitrogen threshold for the Swan Pond River System are summarized below.

***Swan Pond River Stabilization Study: A Report Submitted to the Town of Dennis* –** The focus of this study which was completed in 1984 by D.G. Aubrey was on the inlet dynamics and flushing of the Swan Pond River system to inform the development of an inlet management plan. The study clearly stresses the importance of maintenance dredging in order to keep the inlet fully open for maximal circulation within the Swan Pond River system. The study indicated that if inlet maintenance was not performed on a regular basis the Swan Pond River system would revert to a "poorly flushed, nutrient-rich coastal pond with a drastically altered biological community." The study concluded that any long-term attempt at maintaining the viability of Swan Pond River as a saline, coastal estuary must include periodic maintenance dredging and that there were "no feasible methods to maintain Swan Pond River and Swan Pond as a coastal salt pond on a one-shot basis: periodic maintenance will always be required." The study evaluated several different methods for keeping the Swan Pond River a viable coastal system by enhancing flushing and tidal exchange. Several options were considered in the report: 1) the status quo (do nothing), 2) periodic maintenance dredging with no other remedial structural measures, 3) construction of larger jetties on the east and west sides of the inlet and 4) a combination of maintenance dredging and structural modification to the inlet mouth and interior basins.

***The Swan River Project: Report to Dennis Board of Health* –** The report was developed by T. A. Dumas in 1990 for the Town of Dennis. The purpose was to synthesize relevant information on Swan Pond and Swan River aimed principally at "solving" the bacterial contamination problem (and related shellfish harvest closure) in Swan Pond. The report identified waterfowl, stormwater and road runoff and septage/nitrates as being major contaminants to these estuarine basins. Actions by the Dennis Natural Resources Office, Shellfish Constable, Engineering Office and Department of Public Works were undertaken to remediate sources of contamination. These efforts included actions to reduce waterfowl (no feeding), upgrades of on-site wastewater disposal systems and ceasing to directly discharge stormwater to the estuary. Efforts also included Shellfish Sanitary Surveys for guiding analysis.

***Swan Pond Site Visit to Document 2006 Macroalgal Bloom* –** MEP Technical Director conducted a site visit to Swan Pond in response to an extensive but not uncommon macroalgal bloom that occurred in April 2006. The site visit was to document the extent of the bloom as it might inform the MEP nutrient threshold analysis of the overall Swan Pond River system.

The macroalgal bloom was dominated by a filamentous green algae, growing on the bottom of Swan Pond, most likely Cladophora. Cladophora and several other filamentous green macroalgae are common to the embayments of southeastern Massachusetts. These species are generally found in nitrogen-enriched basins, where eelgrass has declined. They can accumulate in the nitrogen-enriched shallow waters and can generate a relatively high biomass, during periods of the year when light penetrates to the bottom (such as in late winter and spring). As this algae grows, it can begin to lift-off the bottom either due to bubble formation during photosynthesis or due to turbulence from wind. Once at the surface, the macroalgae are blown into larger clumps and finally into the shallows where they can accumulate.

As observed in other estuarine systems analyzed by the MEP, the macroalgae accumulation observed in Swan Pond is a common sign of significant nitrogen enrichment of a shallow embayment (Figures II-1and II-2). Unfortunately, at the point that macroalgal blooms are visible, the embayment is well beyond its nitrogen loading tolerance and has become significantly impaired (as supported by MEP data collection efforts). While atmospheric deposition may play a minor role, nitrogen inputs from the local surrounding watershed typically dominate the nitrogen loading, and hence are the primary cause of the blooms. Note that the

nitrogen inputs can occur over time, with some nitrogen accumulating in the system as it cycles between the sediments and overlying waters. The major mechanisms for reversing the process and restoring a healthy estuarine system is generally to reduce the nitrogen inputs and/or to potentially increase the circulation or flushing of the waters. Swan Pond is particularly sensitive to nitrogen overloading as it is a relatively large basin, exchanging tidal waters with Nantucket Sound through the long-narrow channel of the Swan Pond River. The structure of the system tends to accumulate organic matter and nitrogen within the Pond, enhancing its eutrophication. The relative importance of watershed nitrogen management versus tidal flushing options to the restoration of Swan Pond is presently the focus of the SMAST/DEP Massachusetts Estuaries Project and this nutrient threshold report for Swan Pond River system.

Figure II-1.   View of upper (northern) Swan Pond April 12, 2006. Drift algae floating at surface in dense patches. Algae appears to have been produced within Swan Pond, associated with the bottom, and has lifted-off to the surface where it accumulates in the coves and nearshore due to wind action.

Swan Pond River Water Quality Monitoring Program - The Dennis Water District, while being actively engaged in the study and management of municipal infrastructure and natural resources, committed early on to gathering baseline water quality monitoring data in support of the MEP assessment and nitrogen threshold analysis of Dennis estuaries. For the Dennis Water District and the Town of Dennis, the focus of the Dennis Water Quality Monitoring Program was the gathering of site-specific data on the current nitrogen related water quality of the Town's estuaries to support assessment of present water quality and habitat health. Water quality monitoring of the Swan Pond River System as well as the Sesuit Harbor system on the north shore of Cape Cod was initiated and designed as a coordinated effort between the Town

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of Dennis Water District and The Coastal Systems Program at SMAST-UMD. The water quality monitoring program was initiated in 2005 and has continued uninterrupted through the summer of 2010 and is to be continued under a Town of Dennis imitative at least through 2012. During the first three years of baseline water quality data collection, six sampling events were undertaken each summer between June and early September, with a reduction to 4 events per summer in subsequent years.

Figure II-2.   Surface drift macroalgae forming large bloom in Swan Pond Dennis, early April 2006. The algae were a filamentous green marine species, most likely Cladophora.

The Dennis Water Quality Monitoring Program for Swan Pond River conducted system-wide synoptic sampling during each event, with sampling stations distributed throughout the open water basin of Swan Pond and the upper and lower reaches of the Swan Pond River (Figure II-3). Additionally, as remediation plans for this and the other town systems are implemented throughout the Town of Dennis, the continued monitoring is planned to provide quantitative information to the Town relative to the efficacy of remediation efforts and to support adaptive management planning.

The joint Town of Dennis / CSP Water Quality Monitoring Program provided the quantitative water column nitrogen data (2005-2010) required for the implementation of the MEP's Linked Watershed-Embayment Approach. The MEP effort also builds upon previous watershed delineation and land-use analyses, the previous embayment hydrodynamic analysis related to the establishment of the historic groin at the inlet and historical eelgrass surveys. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Swan Pond River Estuarine System. The MEP has incorporated appropriate data from previous studies available to the

MEP Technical Team to enhance the determination of nitrogen thresholds for the Swan Pond River System and to reduce the costs of this assessment to the Town of Dennis.
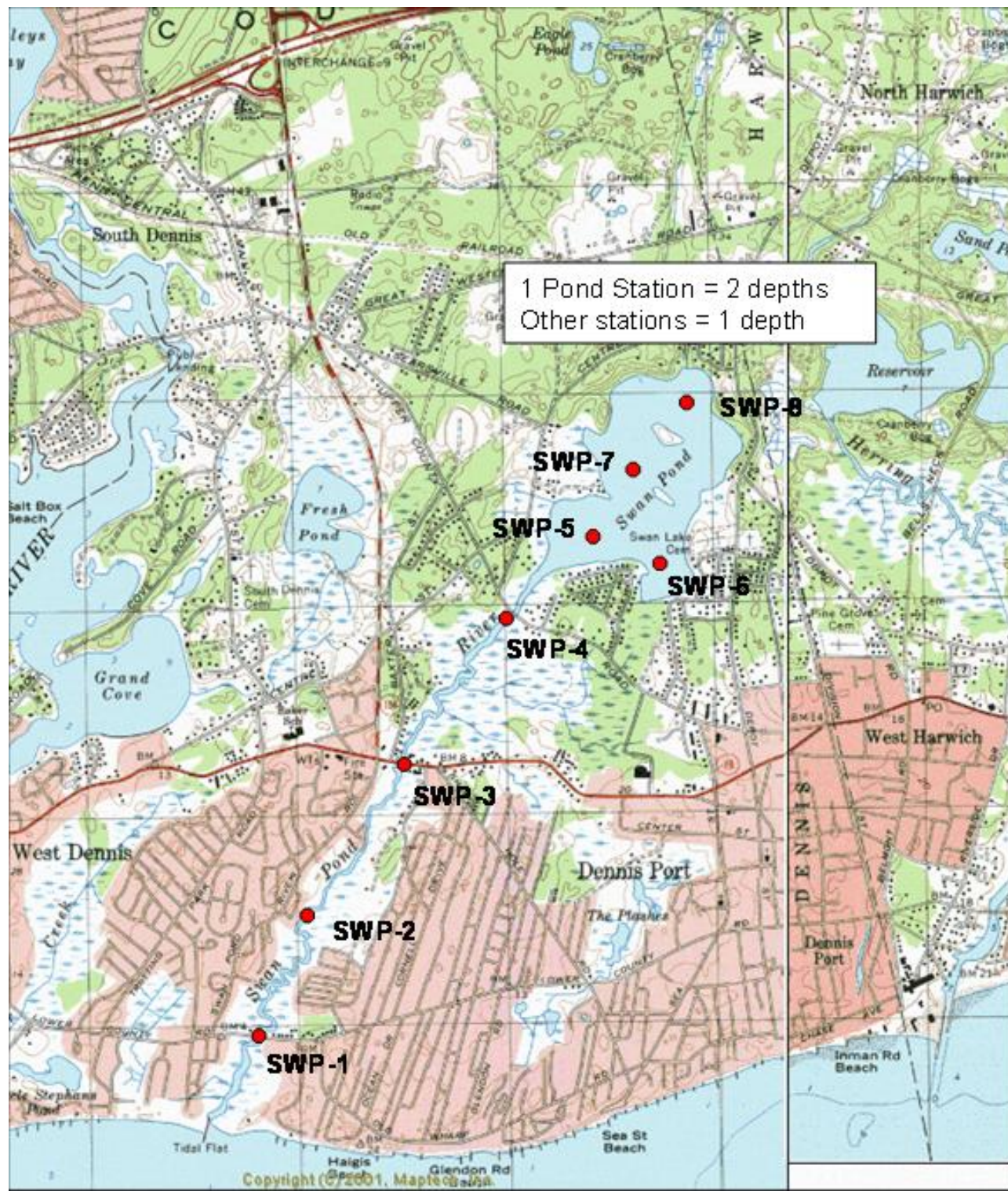
Figure II-3.   Town of Dennis Water Quality Monitoring Program. Estuarine water quality monitoring stations sampled by the Dennis Water District staff and volunteers. Surface and bottom sample collected from SWP-7.

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Regulatory Assessments of Swan Pond River Resources - The Swan Pond River Estuary contains a variety of natural resources of value to the citizens of the Town of Dennis 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-4 through II-8) for reference by those providing stewardship for this estuary. For the Swan Pond River Estuary these include:

- Mouth of River designation - MassDEP (Figure II-4)
- Designated Shellfish Growing Area – MassDMF (Figure II-5)
- Shellfish Suitability Areas - MassDMF (Figure II-6)
- Anadromous Fish Runs - MassDMF (Figure II-7)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-8)

Figure II-4.   Regulatory designation of the mouth of the Swan Pond River under the Massachusetts River Act (MassDEP). Designations recognize that the estuarine reach of rivers are part of the river system and therefore the mouth is at the discharge point to a bay or large embayment. Upland adjacent the "river front" inland of the mouth of the river has restrictions specific to the Act.

Massachusetts Division of Marine Fisheries - Designated Shellfish Growing Area
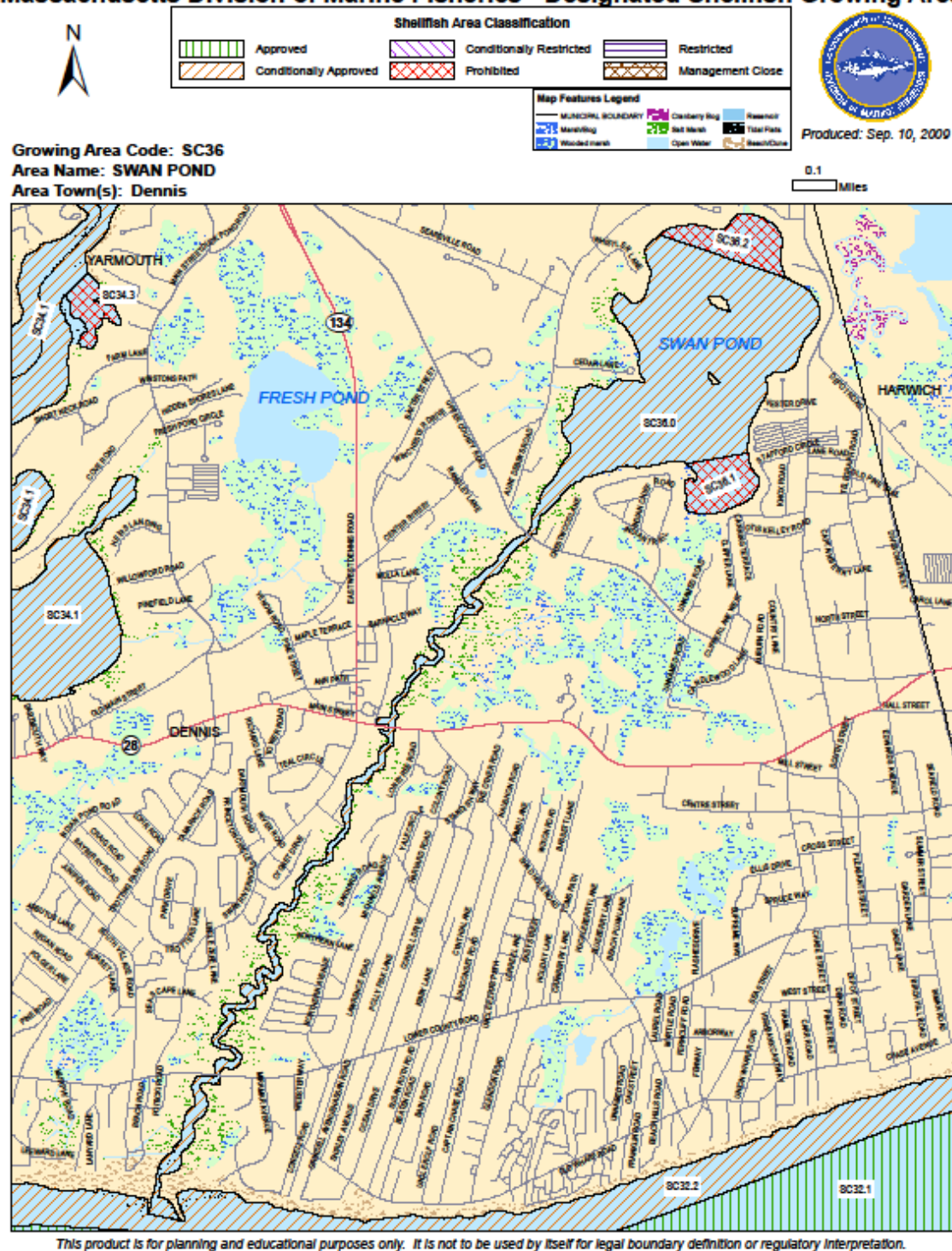
Figure II-5. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Massachusetts Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.
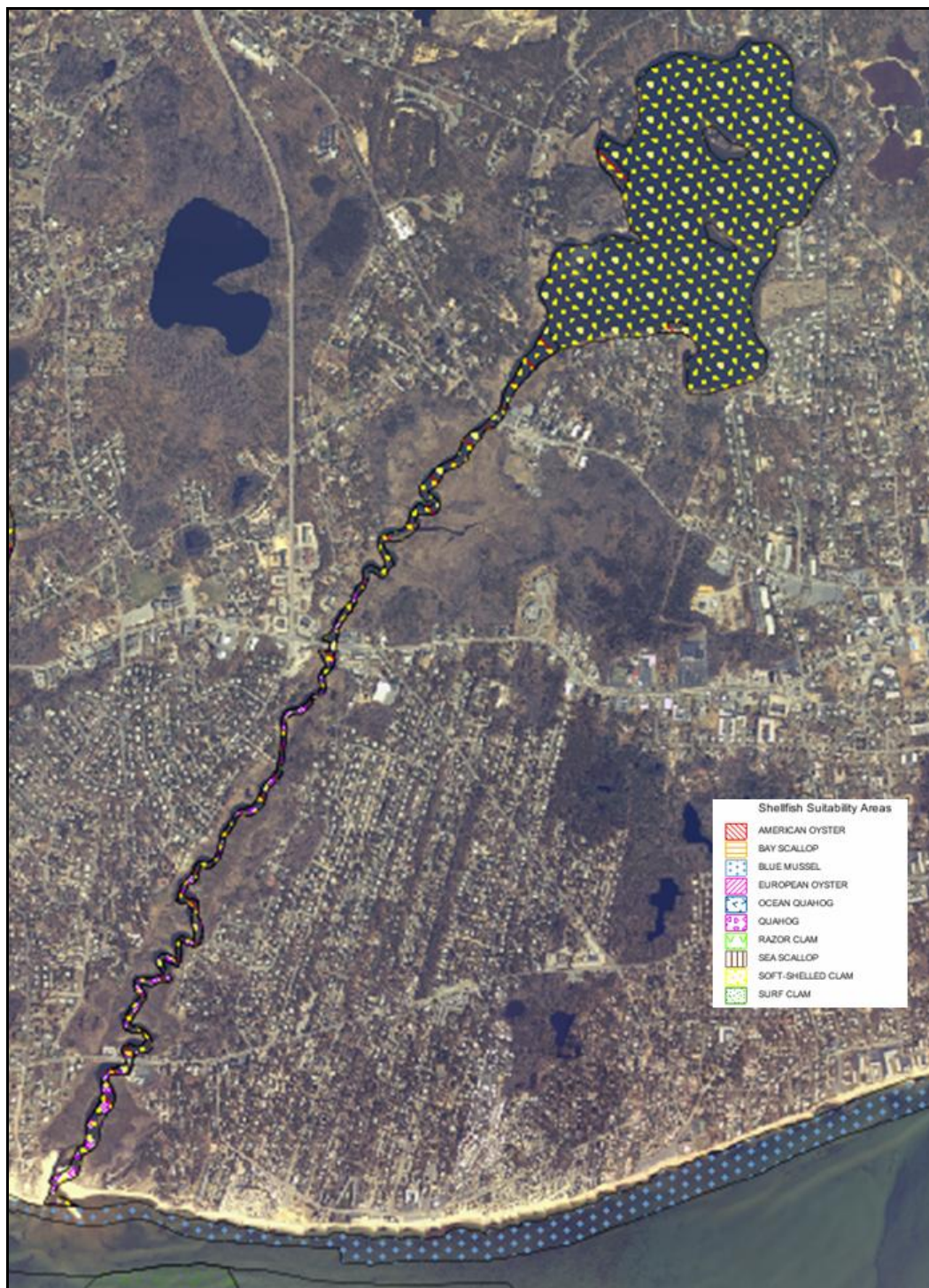
Figure II-6.   Location of shellfish suitability areas within the Swan Pond River Estuary as determined by Massachusetts Division of Marine Fisheries. Suitability does not necessarily mean "presence".   The major habitat is for soft shell clams (*Mya*) with some quahogs (*Mercenaria*) primarily within the tidal river.
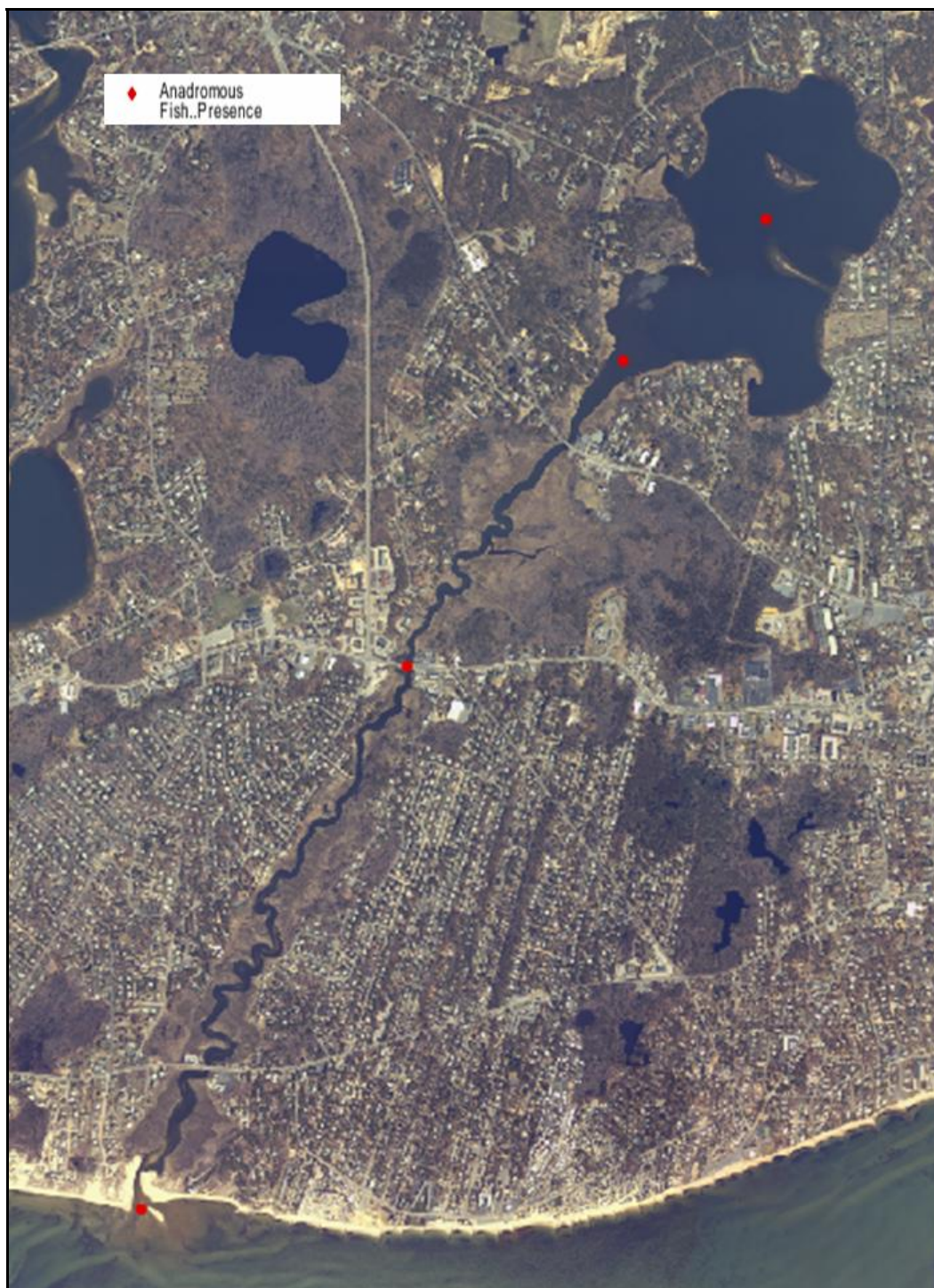
Figure II-7.   Anadromous fish runs within the Swan Pond River Estuary as determined by Massachusetts Division of Marine Fisheries. The red diamonds show areas where fish were observed. The uppermost sites are within Swan Pond located at the head of the Swan Pond River System.
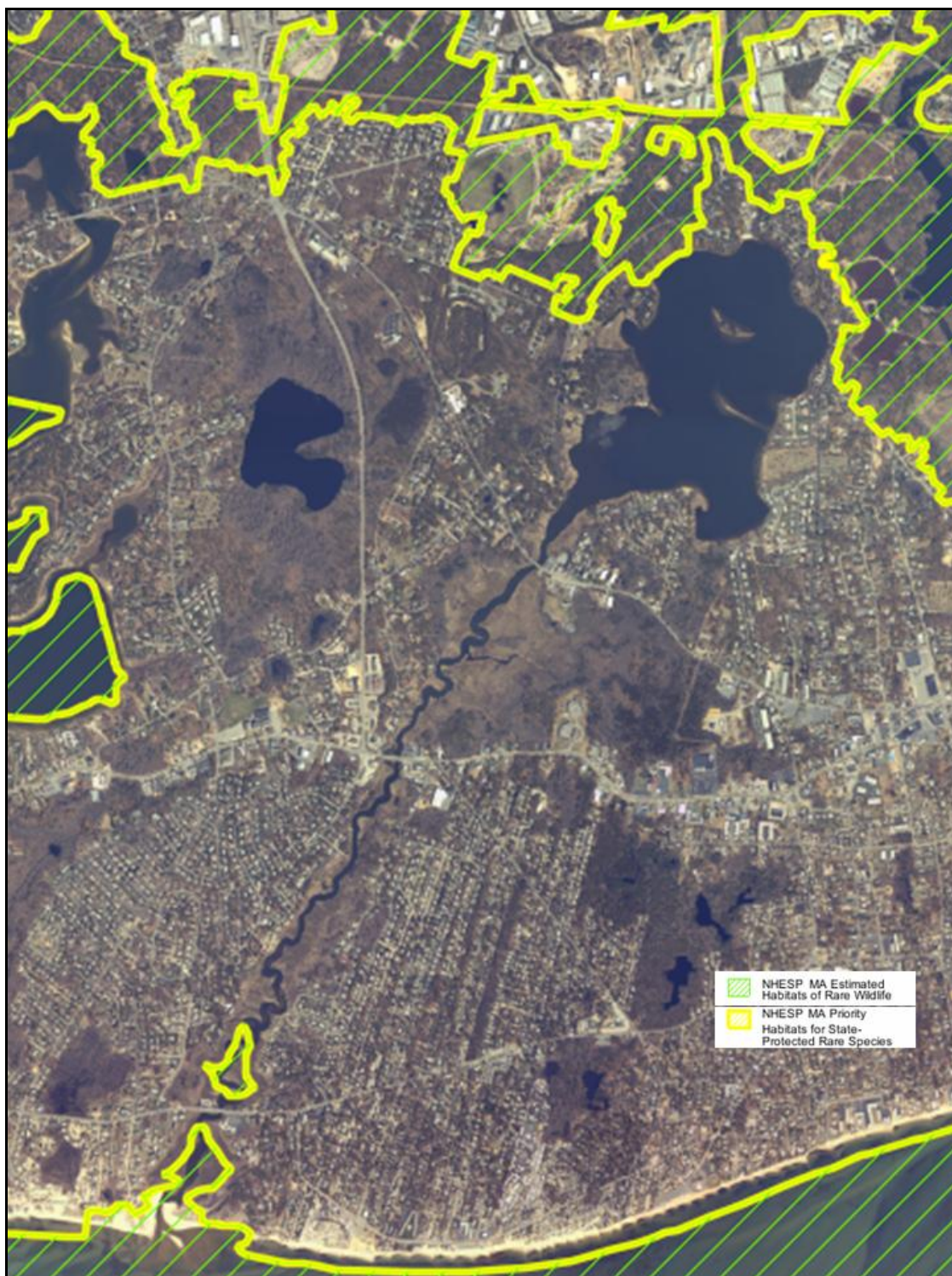
Figure II-8.   Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Swan Pond River Estuary as determined by the Massachusetts Natural Heritage and Endangered Species Program.

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 Swan Pond embayment system. The Swan Pond watershed is primarily located within the Town of Dennis with small portions within the Towns of Harwich and Brewster, Massachusetts.

In the present assessment, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Swan Pond embayment system under evaluation by the MEP Technical Team. The Swan Pond estuarine system is composed of a relatively shallow pond connected to Nantucket Sound by a nearly 3 km long tidal channel (Swan Pond River). Watershed modeling was undertaken to sub-divide the overall watershed to the Swan Pond system into functional sub-units based upon: (a) defining inputs from contributing areas to each major component of the embayment system (mostly bridge crossings across Swan Pond River), (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 gauge the potential mass of nitrogen from "new" development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in both the Sagamore and Monomoy flow cells on Cape Cod; the Swan Pond system is located near the southwestern edge of the Monomoy flow cell. Model assumptions for calibration of the Swan Pond Estuary included surface water discharges measured as part of the MEP stream flow program (2005 to 2006).

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 Swan Pond system and its various sub-watersheds, such as Eagle Pond and the northern and southern sections of Swan Pond River, were delineated using the regional model of the Monomoy Lens flow cell (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area 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 Swan Pond system and its sub-watersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The Monomoy Flow Model grid consists of 164 rows, 220 columns, and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below; the upper three layers of the Monomoy model are dry because of the Monomoy has a lower elevation than the overlapping Sagamore groundwater model that was constructed at the same time (Walter and Whealan, 2005). Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 525 feet below NGVD 29 in the Monomoy); since bedrock is 300 to 400 feet below NGVD 29 in most of the Swan Pond area, the two lowest model layers were active in this area of the model. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, were used to simulate the top of the water table, which varies in elevation depending on the location in both flow cells.

The glacial sediments that comprise the aquifer of the Monomoy Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Swan Pond system watershed is located in the Harwich Outwash Plain Deposits (Oldale, 1974). Modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that similar deposited materials are highly permeable (*e.g.*, Masterson, *et al.*, 1996). Given their high permeability, direct rainwater run-off is typically rather low for this type of watershed system. 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 and stream flow data collected in 1989-1990 as well as 2003 as appropriate for a given watershed.

The Monomoy Flow Cell groundwater model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000

with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. After accounting for the consumptive loss, water withdrawn from the modeled aquifer by public drinking water supply wells is evenly returned within residential areas designated as using on-site septic systems.

III.3 SWAN POND RIVER CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Swan Pond embayment system, including Eagle Pond, and the estuarine components of the upper (northern) and lower (southern) reaches of the Swan Pond River, and Swan 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 fresh pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (d) to more closely match the sub-estuary segmentation of the tidal hydrodynamic model and (e) to address streamflow measurements collected as part of the MEP. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineation also includes 10 yr time-of-travel boundaries. Overall, eight (8) sub-watershed areas, including a shared watershed with the Bass River estuary, were delineated within the Swan Pond study area.

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. The overall estimated freshwater flow into the Swan Pond system from the MEP delineated watershed is 16,972 m3/d. This flow includes inflow from the NW Dennis Wells subwatershed, which primarily discharges into Bass River (Howes, et al., 2010), but also discharges into the Swan Pond system.

The MEP watershed delineation is the second watershed delineation completed in recent years for the Swan 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).

The MEP watershed area for the Swan Pond system as a whole is the same area as the 1998 CCC delineation (2,379 acres vs. 2,380 acres, respectively). However, the areas that are included in the delineations are somewhat different. The northern portion of the MEP watershed is shifted to the west and includes outflow from the shared NW Dennis wells watershed. The MEP watershed also significantly excludes an area south of Route 28 and east of Swan Pond River that is included in the CCC delineation. In the MEP delineation, this area flows directly to Nantucket Sound. The MEP watershed delineation also includes interior sub-watersheds to various components of the Swan Pond system, such as ponds and public water supply wells that were not included in the CCC delineation. These refinements are another benefit of the update of the regional groundwater model (Walter and Whealan, 2005).

SHED ID	SHED NAME
1	Hydaway Lane Stream
2	Eagle Pond
3	Swan Pond GT10
4	Swan Pond LT10
5	Swan Pond River N GT10
6	Swan Pond River N LT10
7	Swan Pond River S
NW	NW Dennis Wells TOTAL

Legend
- Rivers & Creeks
- Lakes & Ponds
- Town Boundary

Roads
- Local Roads
- Arterial Roads
- State Routes

Watersheds
- Embayment Watershed
- Pond Watershed
- Well Watershed
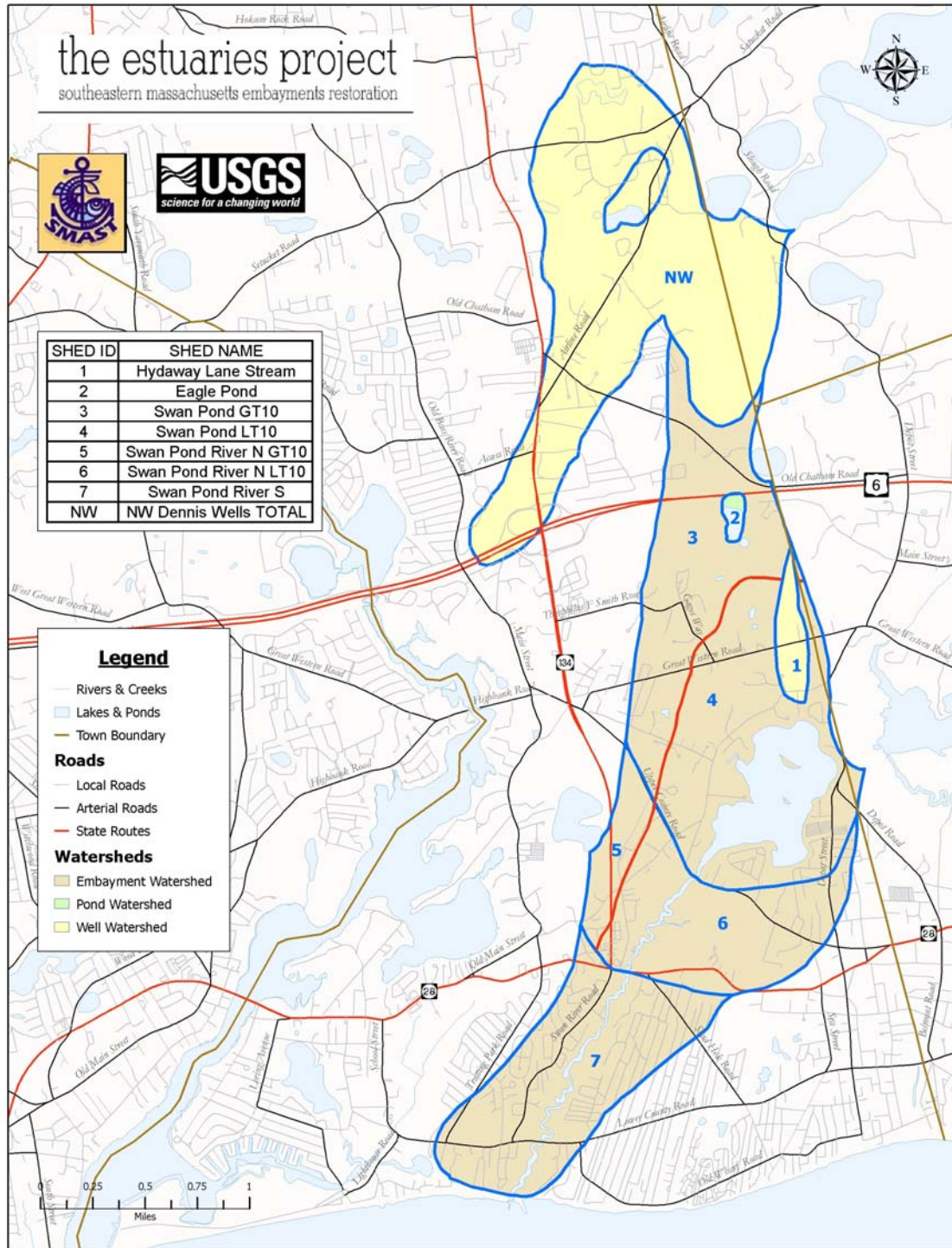
Figure III-1.   Watershed delineation for the Swan Pond estuary system. Subwatershed delineations (8) are based on USGS groundwater model output with refinements to better address pond and estuary shorelines and MEP stream gauge measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watershed groups (*e.g.*, Swan Pond River N) were selected based upon the functional estuarine sub-units in the water quality model (see Section VI). The watershed to NW Dennis Wells is shared with the Bass River system and is detailed in the Bass River MEP Report (Howes, *et al.*, 2010).

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

Watershed	#	Watershed Area (acres)	% contributing to Estuaries	Discharge	
				m³/day	ft³/day
Hydaway Lane Stream	1	50	100	382	13,473
Eagle Pond	2	13	100	96	3,403
Swan Pond GT10	3	449	100	3,444	121,622
Swan Pond LT10	4	469	100	3,600	127,134
Swan Pond River N GT10	5	88	100	675	23,823
Swan Pond River N LT10	6	367	100	2,815	99,403
Swan Pond River S	7	482	100	3,700	130,673
NW Dennis Wells	NW	967	30	2,261	79,843
TOTAL SWAN POND SYSTEM				16,972	599,374
Notes: 1) discharge volumes are based on 27.25 in of annual recharge within land areas; 2) percentage of inflow from NW Dennis Wells, which is shared with the Bass River estuary, is determined by length of downgradient watershed boundary, 3) these flows do not include precipitation on the surface of the estuary, 4) totals may not match due to rounding.					

The evolution of the watershed delineations for the Swan 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 strengthens the use of this model for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down gradient estuary. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Swan Pond system (Section V.1).

CCC Regional Policy Plan

MEP, 2011

Used in 1996, 2001 & 2009 Regional Policy Plans
(based on delineation in Eichner, et al., 1998)

Delineated by USGS for MEP Analysis

Red lines indicate ten year time-of-travel lines
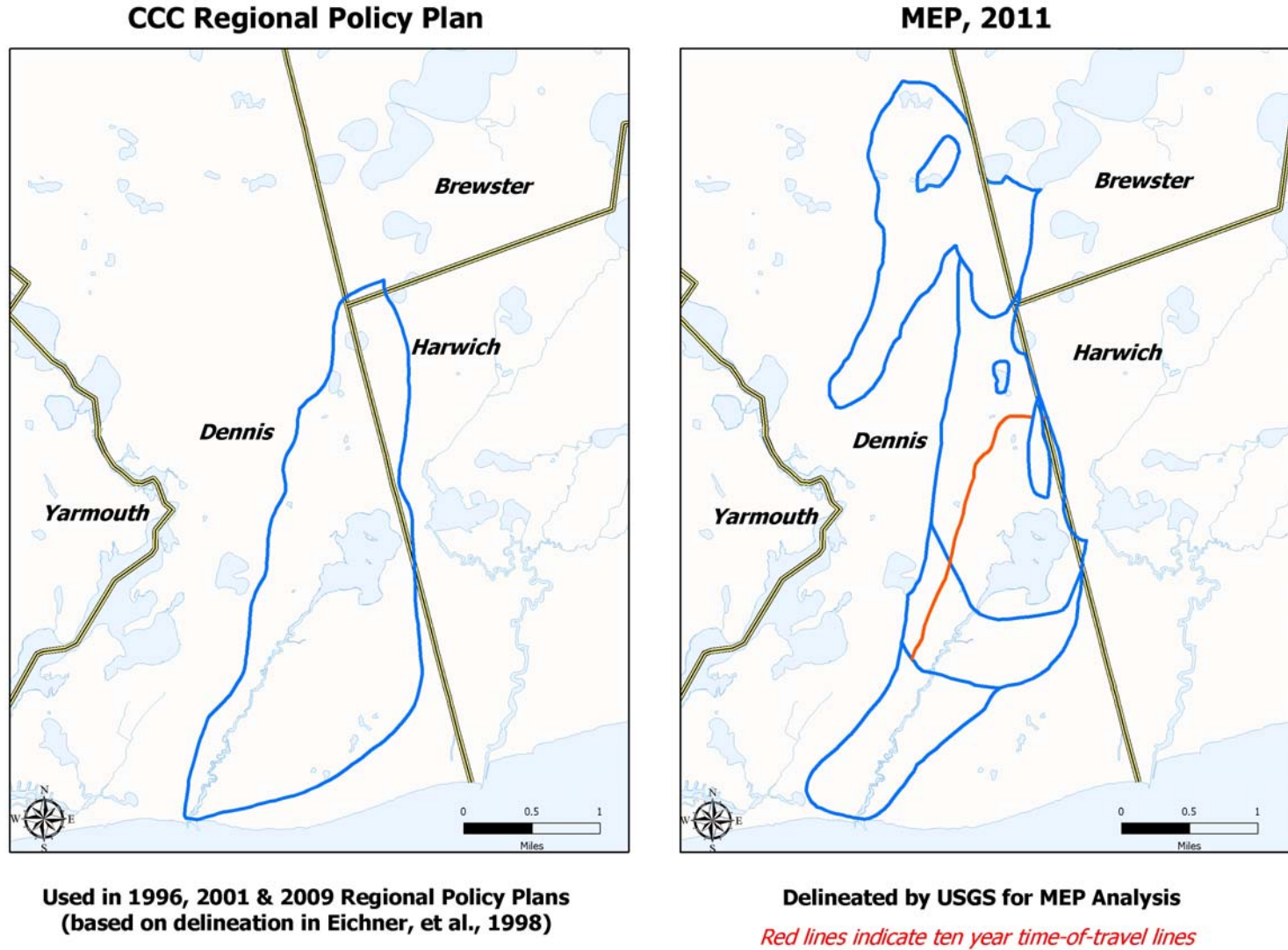
Figure III-2.   Comparison of MEP Swan Pond River watershed and sub-watershed delineations used in the current assessment and the Cape Cod Commission watershed delineation (Eichner, *et al.*, 1998), which has been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009). Note that while portions of the Towns of Brewster and Harwich are within the watershed, almost all of the watershed area falls within the Town of Dennis.

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 Swan 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. Permanent burial of nitrogen in the sediments is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters or in some cases a sink for nitrogen reaching the bottom. Failure to include the nitrogen balance of estuarine sediments and the watershed attenuation generally leads to errors in predicting water quality, particularly in determination of summertime nitrogen load to embayment waters.

In order to determine watershed nitrogen loading inputs to the Swan Pond estuary system, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of the estuary and its watersheds (Section III). The Swan Pond watershed was sub-divided to define contributing areas or sub-watersheds 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 eight (8) sub-watersheds were delineated in the overall Swan Pond River watershed, including watersheds to Eagle Pond and the northern and southern sections of Swan Pond River. 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 gauge 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. Only one ten-year time of travel sub-watershed in the overall Swan Pond watershed has been delineated: one to Swan Pond. This delineation means that the flow from the NW Dennis Wells

and Eagle Pond watersheds are beyond the 10-year time of travel line, but all other flow within the whole Swan Pond estuary watershed and its accompanying nitrogen load is within 10-years' time of travel. Simple review of less than and greater than watersheds indicates that 75% of the unattenuated nitrogen load from the whole watershed is within less than 10 year travel time to the estuary (Table IV-1). Comparison of the contributing flow to the NW Dennis Wells sub-watershed, which should balance the cumulative wellfield withdrawal rate in the USGS groundwater model, to the metered water use in the sub-watershed shows that only 3% of the pumped water is used within the sub-watershed. This finding indicates that the rest of the water is distributed to developed areas in other portions of Dennis, including both the Bass River and Swan Pond River watersheds. This also means that the majority of the nitrogen load from the NW Dennis Wells is likely distributed to less than 10 year time of travel subwatersheds since this is where most of the development is located. Discerning precisely how much more than 75% would require details about a number of factors about the wells and the drinking water distribution system, including determining the source well for drinking water received in different portions of town. In any case, these refinements would only serve to increase the percentage of nitrogen load that is within 10 years' time-of-travel to the estuary. The general conclusion is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuary (after accounting for natural attenuation, see below) and that the distinction between time of travel in the sub-watersheds is not important for modeling existing conditions. Overall and based on the review of all this information, it was determined that the Swan Pond estuary is currently in balance with its watershed load.

Table IV-1.   Percentage of unattenuated nitrogen loads in less than ten year time-of-travel sub-watersheds to Swan Pond.

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
Hyda Way Creek	1	121		121	100%
Eagle Pond	2		55	55	0%
Swan Pond GT10	3		2,290	2,290	0%
Swan Pond LT10	4	3,234		3,234	100%
Swan Pond River N GT10	5		573	573	0%
Swan Pond River N LT10	6	3,090		3,090	100%
Swan Pond River S	7	5,678		5,678	100%
NW Dennis Wells TOTAL	NW		1,025	1,025	0%
Swan Pond Whole System		12,122	3,943	16,066	75%
Notes: loads have been corrected to 1) include division of portions of nitrogen load from ponds and wellhead protection areas to down gradient sub-watersheds, and 2) exclude nitrogen loads that are discharged outside of the Swan Pond system watershed from ponds or wellhead protection areas on the system watershed boundaries. Loads exclude atmospheric loading on the estuary surface waters.					

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

sub-watershed specific land uses and pre-determined nitrogen loading rates based on regional analyses. For the Swan Pond estuary system, the model used land-use data from the Towns of Dennis, Harwich, and Brewster transformed into nitrogen loads using both regional nitrogen loading factors and local watershed-specific data (such as parcel-by-parcel water use and alternative septic system monitoring). Determination of the nitrogen loads required obtaining watershed specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the "potential" or unattenuated nitrogen load to each receiving embayment, since attenuation during transport is included at a later stage.

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) within the Swan Pond River watershed was determined based upon a site-specific study of streamflow and theoretical and measured attenuation in the up-gradient freshwater ponds. Streamflow was characterized at the stream near Hyda Way Lane (Hyda Way Creek). The land-use analysis within the sub-watershed to the pond allows comparison between field collected data from the stream monitoring and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is generally estimated based on the amount and quality of available information. Attenuation through freshwater ponds is conservatively assumed to equal 50% unless there is sufficient reliable available monitoring and pond physical data is reliable enough to calculate a pond-specific nitrogen attenuation factor. Streamflow and associated surface water attenuation is included in the MEP nitrogen attenuation and freshwater flow investigation, presented in Section IV.2.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. In the present effort, one freshwater pond (Eagle Pond) and one small stream (Hyda Way Creek) have delineated sub-watersheds within the overall Swan Pond River watershed. If smaller aquatic features that have not been included in this MEP analysis were providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly (<10%) overestimated given the distribution of nitrogen sources within the watershed.

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the sub-watersheds that directly discharge groundwater to the estuary without flowing through one of these interim pond and stream measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Swan 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 watershed to Swan Pond includes portions of the towns of Dennis, Harwich, and Brewster, Estuaries Project staff obtained digital parcel and tax assessor's data from each of the towns to serve as the land-use database for the watershed nitrogen loading model. Digital parcels and land use/assessors data for Dennis and Brewster are from 2009, while similar data from Harwich is from 2006. These land use databases contain traditional information regarding land use classifications (MassDOR, 2009) plus additional information developed by the towns.

This effort was completed with the assistance from GIS staff from the Cape Cod Commission (CCC).

Figure IV-1 shows the land uses within the Swan Pond estuary watershed. Land uses in the study area are grouped into eight (8) land use categories: 1) residential, 2) commercial, 3) industrial, 4) mixed use, 5) undeveloped (including residential open space), 6) public service/government, including road rights-of-way, 7) freshwater ponds, and 8) unclassified properties. These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2009). "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. Unclassified parcels are properties without any assessor land use classifications.

Public service land uses are the dominant land use type in the overall Swan Pond watershed and occupy 38% of the watershed area (Figure IV-2). Examples of these land uses are lands owned by town and state government (including open space and wellhead protection lands), housing authorities, and churches. Residential land uses occupy the second largest area with 34% of the watershed area. It is notable that land classified by the town assessor as undeveloped is 14% of the overall watershed area. The Swan Pond and Swan Pond River North sub-watersheds are where most of the public service lands are; parcel examples of public service lands in these sub-watersheds include the town landfill and transfer station and wetland areas between Upper County Road and Route 28.

In all the sub-watershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type, ranging between 36% and 90% of all parcels in these sub-watersheds and 79% of all parcels in the Swan Pond system watershed. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 59% to 100% of residential parcels in the individual sub-watershed groupings and 91% of the residential parcels throughout the Swan Pond system watershed.

In order to estimate wastewater flows within the Swan Pond study area, MEP staff also obtained parcel-by-parcel water use data from the Dennis Water District and the Town of Harwich. None of the parcels in the Brewster portion of the watershed have assigned water use accounts. Three years of water use was obtained from the Dennis Water District, while four years of water use was obtained from the Town of Harwich Water Department. The water use data was linked to the respective town parcel databases by the CCC 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.
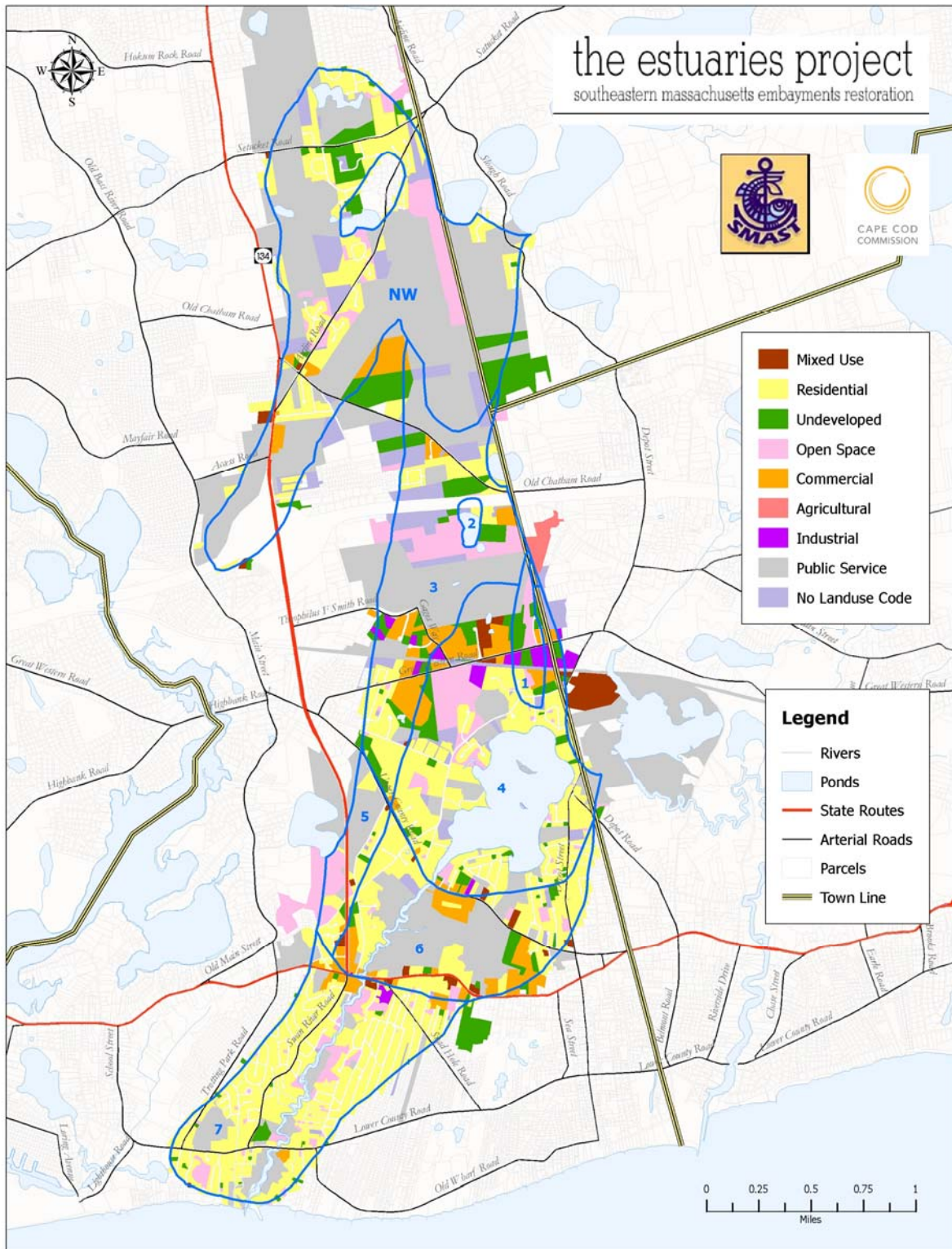
Figure IV-1.   Land-use in the Swan Pond system watershed and sub-watersheds.   The system watershed extends over portions of the Towns of Dennis, Harwich, and Brewster. Land use classifications are based on respective town assessor classifications and MADOR (2009) categories. Base assessor and parcel data for Dennis and Brewster are from the year 2009, while corresponding data from Harwich is from the year 2006.
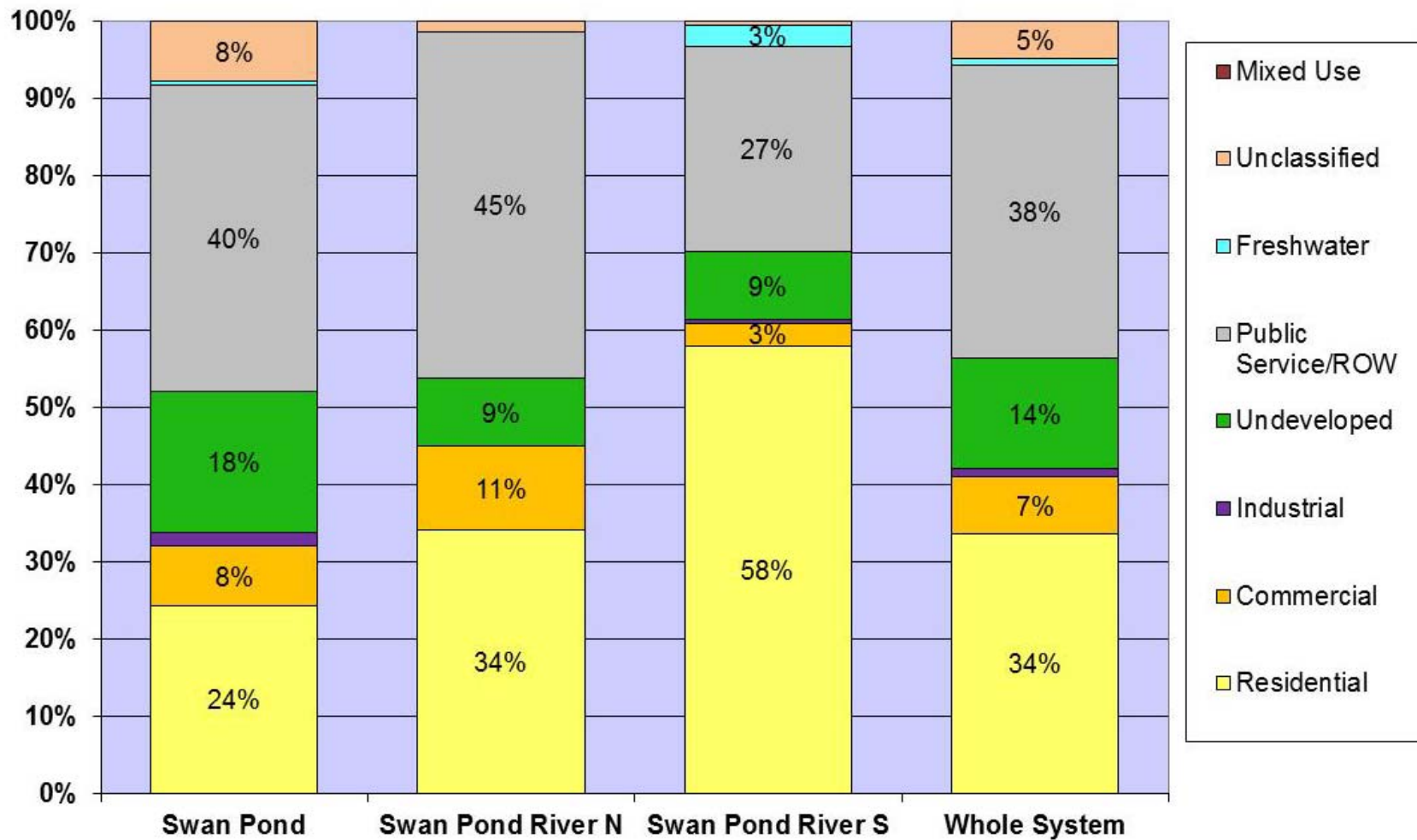
Figure IV-2.   Distribution of land-uses by area within the watershed to Swan Pond River and its three component sub-watersheds. Land use categories are generally based on town assessor's land use classification and groupings recommended by MADOR (2009). Unclassified parcels do not have an assigned land use code in the town assessor's databases. Only percentages greater than or equal to 3% are shown.
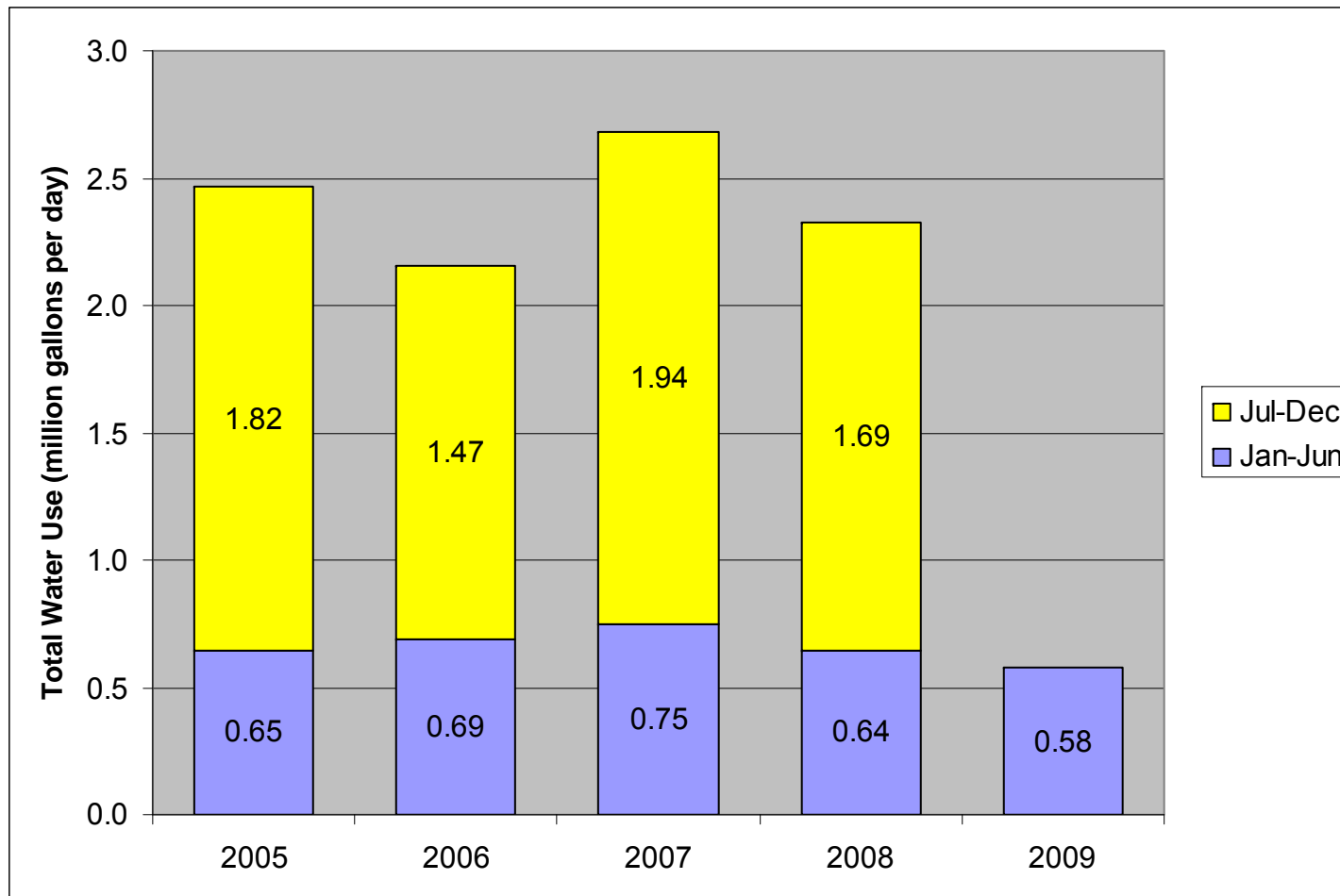
Figure IV-3.   Dennis Seasonal Water Uses (2005-2009). Water use was only available for the first portion of 2009 during the development of the watershed nitrogen loading model. In order to address this, MEP staff utilized three years of data based on July to June calendar years to calculate average water use for parcels. However, since the July to December 2006 period was exceptional low, this period was replaced with the comparable 2007 period when calculating the average parcel water use. Overall, substitution of this period results in total average flows for the July to December period that is approximately 2% higher than a simple average of 2005, 2007, 2008 total water use for this period.

While the water use for individual parcels in the Town of Harwich is a standard average across four years of data, the Town of Dennis water use for individual parcels is calculated slightly differently. MEP staff obtained water use from January 2005 through June 2009 from the Dennis Water District (Sheryl McMahon, Treasurer, 10/09). In order to try to keep the most up-to-date water use, MEP staff determined annual flows based on July to June years; so the most recent year in the data is July 2008 to June 2009. Upon review of these adjusted annual flows, the July to December 2006 cumulative flow was found to be exceptionally low (more than one standard deviation below the mean) (Figure IV-3). Staff then reviewed the July to December reporting periods to ensure that three years' worth of water data was used in the average water use for individual parcels and determined that the July to December 2007 data would be most appropriate. This data is cumulatively 2% higher than the mean of the 2005, 2007, and 2008 July to December monitoring period, but it provides a more recent set of flow than the next closest match (2005). As a result, the Town of Dennis individual parcel water uses are based on average water use from July 2008 to June 2009, July 2007 to June 2008, and a combined July to December 2007 and January to June 2007.

MEP staff also received alternative, denitrifying septic system total nitrogen effluent data from the Barnstable County Department of Health and the Environment (personal communication, Brian Baumgaertel, 12/10) and information on state Groundwater Discharge Permits granted by MassDEP (personal communication, Brian Dudley, 2/09). The total nitrogen monitoring data for denitrifying septic systems was used to develop wastewater nitrogen loads for sites in the BCDHE database. GWDPs are required under MassDEP regulations for wastewater treatment systems with design flows greater than 10,000 gallons per day. According the MassDEP databases, no GWDPs exist in the Swan Pond system watershed.

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 down gradient in the aquifer.

All nitrogen losses within the septic system are incorporated into the MEP analysis. For example, information developed at the MassDEP Alternative Septic System Test Center at the

Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa *et al.* 2001). Down gradient studies of septic system plumes 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 multiplied by 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 Coefficient, they were not used in its development. The MEP Technical Team has developed the septic system nitrogen load over many years, and the general agreement among the number of supporting studies has greatly enhanced the certainty of this critical watershed nitrogen loading term.

The independent validation of the water quality model (Section VI) and the reasonableness of the freshwater attenuation (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher loads that are generally used for septic systems in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%) needed to lower embayment nitrogen levels to a nitrogen target (e.g. nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to 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 Swan Pond watersheds, MEP staff reviewed US Census population values for the Town of Dennis. Since Brewster and Harwich occupy such a small portion of the watershed, they were not included in this validation analysis. 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 Dennis is 2.13 people per housing unit with 53% year-round occupancy, while 2010 Census information indicates that the average occupancy dropped to 2.05 people per housing unit and the year-round occupancy also dropped to 44%. Average water use for single-family residences with municipal water accounts in the Swan Pond MEP study area is 118 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the study area average is 107 gpd.

In order to provide a check on the water use, Yarmouth and Dennis 2000 Census average occupancies were averaged since they are approximately the same. Multiplying this occupancy by the state Title 5 estimate of 55 gpd of wastewater per capita results in an average estimated water use per residence of 117 gpd based on 2000 Census occupancy and 113 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 seasonal properties are occupied at twice the year-round occupancy for three months in Dennis, the estimated average town-wide water use would be 146 or 141 gpd for the 2010 and 2000 Census, respectively, while if the seasonal properties are occupied at three times the year-round occupancy for three months, the estimated average water use would be 169 or 176 gpd, respectively. Given that the average wastewater generation for the Swan Pond watershed closely approximates the flow estimates based on simple occupancy without any summer multipliers, it suggests that the Swan Pond system watershed

tends to be occupied by year-round residents. This analysis also suggests that the average water use is reasonably reflective of average wastewater estimates.

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

Water use information exists for 96% of the 2,026 developed parcels in the Swan Pond watershed. Parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs or town assessor valuations, and do not have a listed account in the water use databases. Of the 72 developed parcels without water use accounts, 51 (71%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and are assigned the Swan Pond study area average water use of 118 gpd in the watershed nitrogen loading modules. Of the 21 remaining developed parcels, 11 are other residential land uses. Given the preponderance of residential land uses among developed parcels without water use accounts, all of these parcels were assigned 118 gpd as their water use in the watershed nitrogen loading model.

Alternative Septic Systems

Project staff identified 29 alternative, denitrifying septic systems in the Swan Pond River watershed that have total nitrogen effluent data in the Barnstable County Department of Health and the Environment database (personal communication, Brian Baumgaertel, 12/10). These systems have between 3 and 41 total nitrogen effluent measurements. Among these denitrifying septic systems, average total nitrogen effluent concentrations ranged between 8 and 48 ppm. Project staff used these site-specific, average measured effluent total nitrogen concentrations and the average measured water use from the town records to calculate average annual loads from each of these sites. These loads were incorporated into the watershed nitrogen loading module for the Swan Pond.

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 Swan Pond system, MEP staff reviewed available regional information about residential lawn fertilizing practices. No golf courses or cranberry bogs are located within the Swan Pond system watershed.

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

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

Nitrogen Loading Input Factors: Dennis Landfill and other solid waste sites

MEP staff contacted MassDEP to obtain any nitrogen monitoring data for solid waste sites within the Swan Pond River System watershed. MassDEP has data on three sites within the watershed: 1) the Town of Dennis landfill, 2) S&J Exco, and 3) Robert Childs, Inc. (Mark Dakers, SERO, personal communication, 12/15/10). Development of nitrogen loads for each of these sites based on the available monitoring data is discussed in this section.

<u>Dennis Landfill</u>
A portion of the capped Town of Dennis landfill is primarily located with the Swan Pond GT10 sub-watershed (sub-watershed #3). Available data consists of contaminant concentrations from groundwater samples collected from 14 wells semi-annually between March 2005 and March 2010 (*e.g.*, 10 sampling runs).

The available Dennis landfill groundwater monitoring data includes nitrate-nitrogen concentrations, but does not include total nitrogen or ammonium-nitrogen data. MEP staff estimated the rest of the dissolved nitrogen concentration during each sampling run based on alkalinity concentrations and the relationship between alkalinity concentrations and ammonium-nitrogen concentrations from groundwater monitoring of the Town of Brewster landfill (Cambareri and Eichner, 1993). After calculation, the estimated ammonium-nitrogen concentrations are added to the measured nitrate-nitrogen concentrations to provide an estimate of dissolved inorganic nitrogen, which is also used as an estimate of total nitrogen.

After determining the estimated nitrogen concentrations in the 14 wells regularly sampled in the Dennis landfill monitoring network, MEP staff reviewed the location of the regularly monitored wells and the well construction details including the screen depths shown in the well logs. This review found that there are only a few wells that are screened at appropriate depths to accurately measure nitrogen export from the landfill. This review also found that the predominant groundwater flow direction from the landfill is toward the south-southeast and toward Swan Pond.

Given some of the uncertainties with the available data, MEP reviewed the alkalinity concentrations in the down gradient monitoring wells and selected those wells with average alkalinity concentrations greater than 100 mg/l CaCO3 and averaged the estimated nitrogen concentrations for the three wells that met this criterion. Based on this approach, the average total nitrogen concentration in groundwater from the Dennis Landfill is 5.64 ppm. Using this

concentration and the standard MEP recharge rate with the 18 acres of capped solid waste that exist within the watershed results in an annual nitrogen load of 290 kg from the Dennis landfill. This load is added to the sub-watershed #3 annual nitrogen load.

It is acknowledged that this approach for estimating a nitrogen load from the Dennis landfill includes a number of assumptions and is likely conservative. A detailed hydrogeologic assessment is beyond the scope of the MEP, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP 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, including, at a minimum, analysis of total nitrogen concentrations, would help to refine this assessment and future management options.

S&J EXCO Solid Waste Site
The S&J EXCO is a six (6) acre solid waste site located north of the intersection of Great Western Road and South Gages Way and is split by the 10-year time-of-travel line for the Swan Pond sub-watershed. MEP staff digitized the area of solid waste based on maps from MassDEP files and assigned 61% of the area to the Swan Pond GT10 sub-watershed (sub-watershed #3) and 39% to the Swan Pond LT10 sub-watershed (sub-watershed #4). Monitoring data available in MassDEP files is for two (2) sampling runs: June 2008 and May 2010. Available data consists of contaminant concentrations from groundwater samples collected from four (4) wells. Unlike the Dennis landfill data, the available S&J EXCO groundwater monitoring data includes both nitrate-nitrogen and Kjeldahl nitrogen concentrations, which can be added to determine total nitrogen.

The four monitoring wells at this site are located in a pattern to establish one up-gradient site and three down gradient sites. All wells have 15 ft screens that straddled the water table at the time of installation (December, 1986). The average total nitrogen concentration of the down gradient wells is 7.24 mg/l, while the up-gradient well has an average total nitrogen concentration based on two samples of 11.8 mg/l. Given the lack of definitive groundwater flow paths, it is unclear what is causing the elevated up-gradient concentration and its high concentration does not allow a definitive measure of the impact of the S&J EXCO site. However, a conservative estimate was derived from the average total nitrogen concentration of the down gradient wells, 7.24 mg N L^{-1}, and the standard MEP recharge rate over the 6 acres of solid waste on this site. This simple approach yields an annual nitrogen load of 122 kg from the S&J EXCO site. This load is divided between sub-watersheds #3 and #4 based on the area of solid waste within each sub-watershed.

It is acknowledged that this approach for estimating a nitrogen load from the S&J EXCO site includes a number of assumptions and is likely conservative. A detailed hydrogeologic assessment is beyond the scope of the MEP, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP 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, including, at a minimum, additional sampling for total nitrogen concentrations, would help to refine this assessment and future management options.

Robert Childs, Inc. Solid Waste Site
The Robert Childs, Inc. enterprise is a six (6) acre solid waste site located south of Great Western Road and almost entirely within the Swan Pond LT10 sub-watershed (sub-watershed #4). MEP staff digitized the area of solid waste based on maps from MassDEP files. Monitoring

data available in MassDEP files is for two (2) sampling runs: March 2010 and September 2010. Available data consists of contaminant concentrations from groundwater samples collected from four (4) wells. Unlike the Dennis landfill data, the available Robert Childs, Inc. groundwater monitoring data includes both nitrate-nitrogen and Kjeldahl nitrogen concentrations, which can be added to determine total nitrogen.

The four monitoring wells at this site are located in a pattern to establish one up-gradient site and three down gradient sites. Well logs detailing construction details were not available from MassDEP files. The average total nitrogen concentration of the down gradient wells is 2.33 mg/l, while the upgradient well had an average total nitrogen concentration based on two samples of 0.52 mg N L^{-1}. Given the lack of definitive groundwater flow paths and well construction details, it is unclear whether the well labeled up-gradient is in the same flow path as the wells installed as the down gradient wells. A conservative approach was again employed using the average total nitrogen concentration of the down gradient wells to estimate the nitrogen load from the Robert Childs, Inc. site. Based on the 2.33 mg N L^{-1} average concentration and the standard MEP recharge rate over the 6 acres of solid waste on this site an annual nitrogen load of 37 kg is added to the watershed from the Robert Childs, Inc. site. This load is added to the sub-watershed #4 load.

It is acknowledged that this approach for estimating a nitrogen load from the Robert Childs, Inc. site includes a number of assumptions and is likely conservative. A detailed hydrogeologic assessment is beyond the scope of the MEP, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP 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, including, at a minimum, additional sampling for total nitrogen concentrations, would help to refine this assessment and future management options.

Nitrogen loading from all three solid waste sites is ~2.5% of the present nitrogen loading to the estuary.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Swan 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 of road segments. MEP staff utilized the GIS to sum these segments and their various widths by sub-watershed. 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 sub-watershed. 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 sub-watershed and the sum of the area of the parcels within each sub-watershed. The resulting "parcelized" watersheds to Swan 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. Each of the towns provided GIS coverages of building footprints for the roof area calculations; Dennis footprints are from 2009, Brewster's footprints are from 2005, and Harwich footprints are from 2006. Individualized information for parcels with atypical nitrogen loading (solid waste sites, 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 Swan 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 Swan Pond MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Dennis, Harwich and Brewster-specific data.

Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Natural Area Recharge	0.072	**Water Use/Wastewater:**	
Direct Precipitation on Embayments and Ponds	1.09	Existing developed parcels wo/water accounts and buildout single-family residential parcels:	118 gpd[3]
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (sq ft)[2]	5,000	Existing developed parcels w/water accounts:	Measured annual water use
Residential Watershed Nitrogen Rate (lbs/lawn)[2]	1.08	Commercial and Industrial Buildings buildout additions[5]	
Town of Dennis Landfill Area[1]		Commercial	
Area of capped solid waste in watershed (acres)	18	Wastewater flow (gpd/1,000 ft2 of building):	127
Estimated TN concentration (mg/l)	5.64	Building coverage:	8%
Estimated Total Nitrogen Load (kg/yr)	290	Industrial	
Average Single Family Residence Building Size from watershed data (sq ft)	1,281	Wastewater flow (gpd/1,000 ft2 of building):	30
		Building coverage:	15%

Notes:
1) Analysis based on digitizing areas of solid waste and available monitoring data from MassDEP files
2) Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.
3) Based on average flow in all single-family residences in the watershed
4) Based on average flow in all residences that are not single-family residences in the watershed
5) based on existing water use and water use for similarly classified properties throughout the watershed
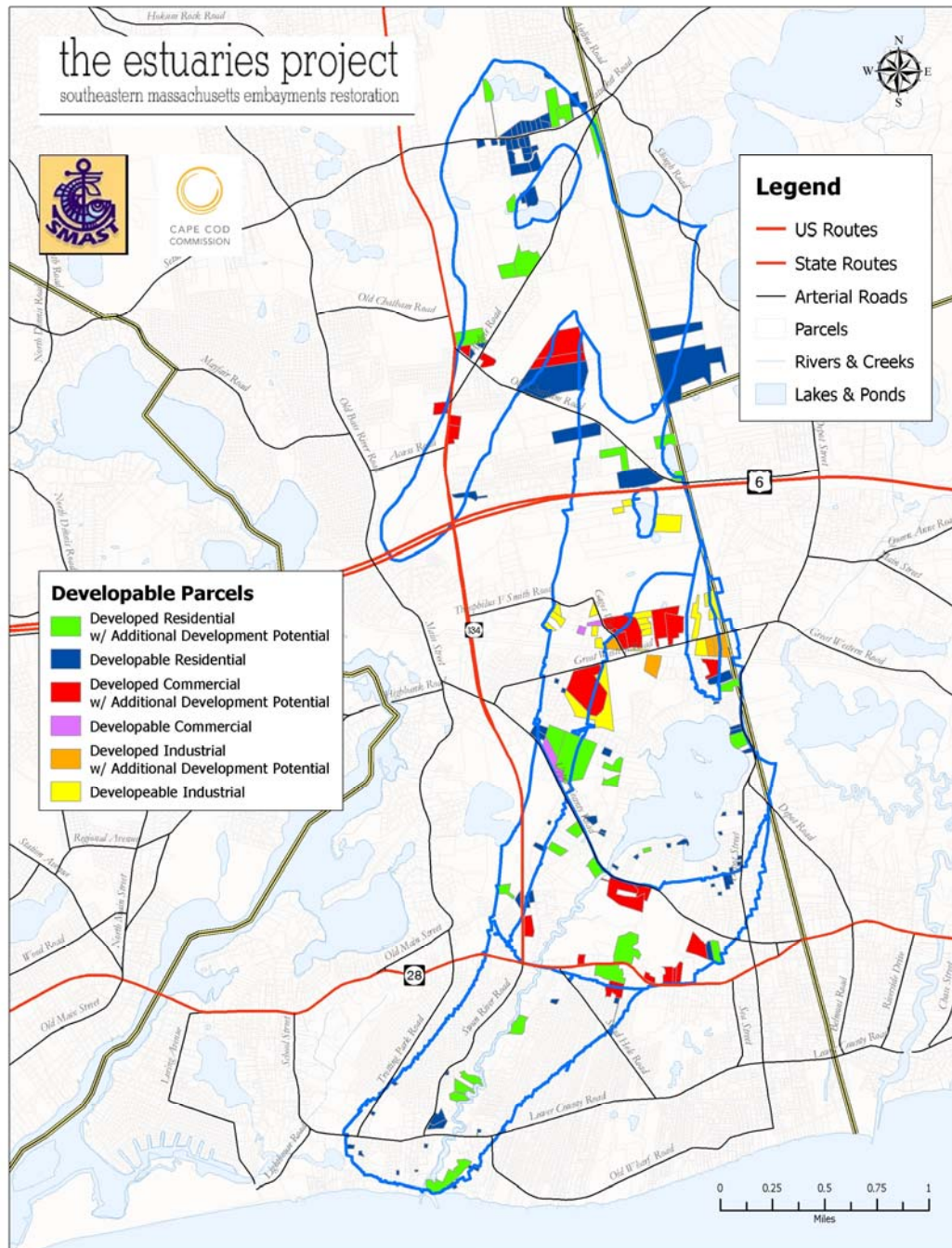
Figure IV-4.   Parcels, Parcelized Watersheds, and Developable Parcels within the Swan Pond River watershed. Parcels colored green, red, and orange are developed parcels (residential, commercial, and industrial, respectively) with additional development potential based on current zoning, while parcel colored blue, purple, and yellow are corresponding undeveloped 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 (Dan Fortier, Town of Dennis, personal communication, 12/10).

Following the assignment of all parcels, sub-watershed modules were generated for each of the eight (8) sub-watersheds in the Swan Pond study area. These sub-watershed 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 sub-watershed. Individual sub-watershed information is then integrated to create the Swan Pond Watershed Nitrogen Loading module with summaries for each of the individual eight sub-watersheds. The sub-watersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Swan Pond study area, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), fertilizers, impervious surfaces, direct atmospheric deposition to water surfaces, the solid waste sites, 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 down gradient sub-watershed, the length of shoreline on the down gradient side of the pond is used to apportion the pond-attenuated nitrogen load to respective down gradient watersheds. The apportionment is based on the percentage of discharging shoreline bordering each down gradient sub-watershed. In the Swan Pond study area, Eagle Pond is the only pond of sufficient size to have a delineated sub-watershed. Smaller ponds do not significantly affect the regional groundwater flow and the MEP does not typically delineate their contributing areas. Eagle Pond is completely within the Swan Pond GT10 sub-watershed, so all of the water from its watershed and the accompanying attenuated nitrogen load is discharged completely to sub-watershed #3.

Table IV-3. Swan Pond River Watershed Nitrogen Loads. Attenuated nitrogen loads shown below are based on measured and assigned attenuation factors assigned to up-gradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2), while pond attenuation factors are assigned a standard MEP nitrogen attenuation of 50% attenuation based on water quality monitoring from the Cape Cod Pond and Lake Stewards program. All nitrogen loads are kg N yr^{-1}.

Watershed Name	Watershed ID#	Swan Pond N Loads by Input (kg/y):							% of Pond Outflow	Present N Loads			Buildout N Loads		
		Wastewater	Landfill/ Solid Waste	Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Swan Pond System		12,127	450	1,062	1,979	875	323	2,266		16,816		16,707	19,082		18,942
Swan Pond		4,853	450	356	788	752	214	1,664		7,413		7,303	9,076		8,936
Hydaway Lane Stream	1	78		4	29	-	9	110		121		121	231	-	231
Eagle Pond	2	6		0	17	31	1	0	100%	55	50%	28	56	50%	28
Swan Pond GT10	3	1,499	364	59	290	-	77	407		2,290		2,290	2,696	-	2,696
Swan Pond LT10	4	2,535	85	179	362	-	73	476		3,234		3,234	3,710	-	3,710
NW Dennis Wells TOTAL	NW	733	-	113	92	33	55	671	30%	1,025		943	1,696	-	1,584
Swan Pond Estuary Surface						688				688		688	688		688
Swan Pond River N		3,070	-	146	375	38	71	505		3,701		3,701	4,206		4,206
Swan Pond River N GT10	5	430		28	102	-	13	18		573		573	591	-	591
Swan Pond River N LT10	6	2,640		119	273	-	58	486		3,090		3,090	3,576	-	3,576
Swan Pond River N Surface						38				38		38	38		38
Swan Pond River S		4,205	-	560	815	85	38	97		5,703		5,703	5,800		5,800
Swan Pond River S	7	4,205		560	815	-	38	97		5,618		5,618	5,715		5,715
Swan Pond River S Surface						85				85		85	85		85
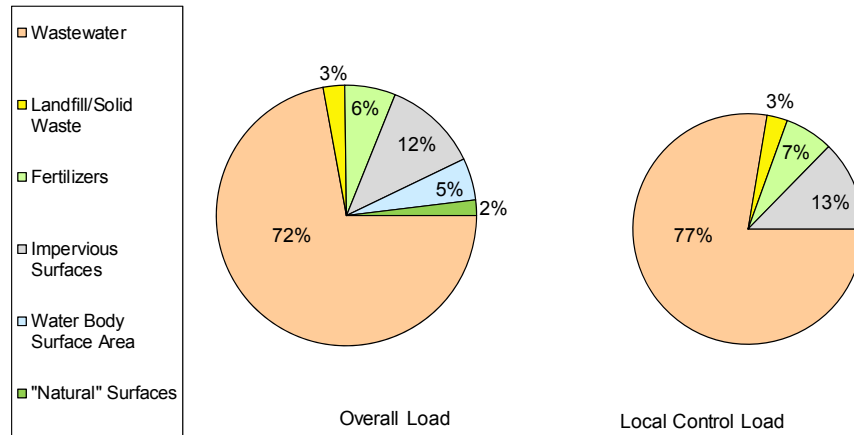
A. Whole System: Swan Pond River Estuary
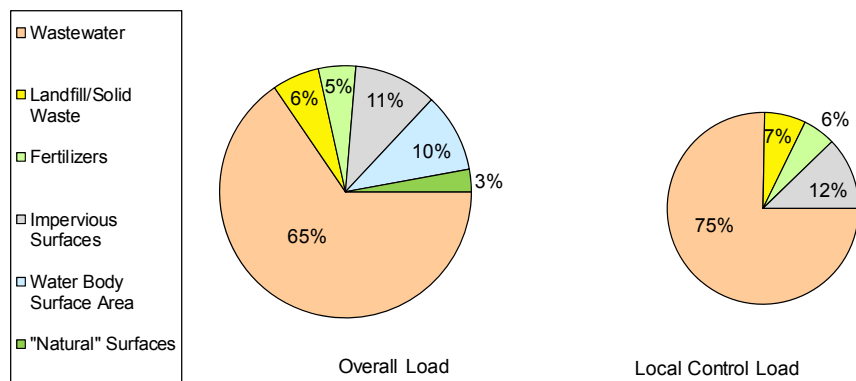
B. Swan Pond Subwatershed
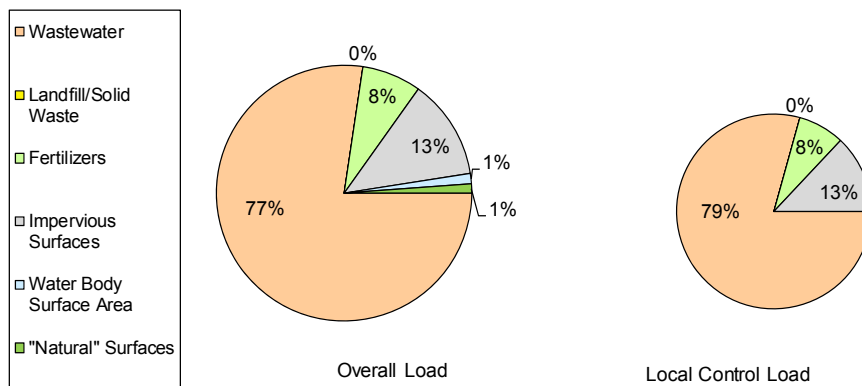
C. Swan Pond River (N & S)

Figure IV-5 .   Land use-specific unattenuated nitrogen loads (by percent) to the a) whole Swan Pond River watershed, b) Swan Pond sub-watershed, and c) combined Swan Pond River sub-watershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

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 up-gradient shoreline, then lake water flows back into the groundwater system along the down gradient 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 a 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 down gradient side of the pond and eventually discharge into the down gradient embayment or pass into a surface water feature such as a stream with associated stream outlet thus discharging N-load directly to the estuary. The nitrogen load summary in Table IV-3 includes both the unattenuated (nitrogen load to each sub-watershed) 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 the 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 gauge nitrogen attenuation.

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

Within the Swan Pond watershed, there is one freshwater pond with a delineated watershed: Eagle Pond. Eagle Pond does not have available pond-wide bathymetric data (Eichner, 2009). As such, a reasonable pond-specific nitrogen attenuation rate cannot be developed for this pond, even though Eagle Pond has been regularly sampled via the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and the locally-supported volunteer pond sampling program through the Town of Dennis Water Quality Advisory Committee.

The PALS Snapshots are regional volunteer pond sampling supported for the last nine years by SMAST and the Cape Cod Commission, with free laboratory services provided by the Coastal Systems Program Laboratory at SMAST. In addition, Dennis, Brewster, and Harwich have created local volunteer pond sampling programs that included regular sampling throughout multiple summers. All sampling runs in the Swan Pond watershed ponds have generally followed PALS protocols (Eichner *et al.*, 2003), which means that sampling 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.

Eagle Pond has been sampled since the original PALS Snapshot in 2001 and water quality data and samples have been collected during 42 sampling runs since then (Eichner, 2009). Unfortunately, given the lack of bathymetry, data is insufficient to assign a pond-specific nitrogen attenuation factor to Eagle Pond. The standard MEP freshwater pond 50% nitrogen attenuation rate is incorporated into the Swan Pond watershed nitrogen loading module of the linked watershed-estuary model for Eagle Pond.

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 and 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 sub-watershed 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 Dennis, Harwich, and Brewster 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 Swan Pond watersheds, MEP staff reviewed the results with town officials. MEP staff reviewed the initial Dennis buildout results with Dan Fortier, Town of Dennis Town Planner in December 2010. And the initial Brewster results were reviewed with Sue Leven, Town of Brewster Town Planner in June 2010. Suggested changes from all reviews were incorporated into the final buildout for the Swan Pond River watershed.

All the parcels with additional buildout potential within the Swan 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 are indicated in a separate column in Table IV-3. The watershed to Swan Pond River is presently nearing buildout under existing zoning, with the unattenuated nitrogen loading to the estuary at full buildout increasing by only 13%.

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

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

Embayment System being investigated under this nutrient threshold analysis was based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1).

If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such as the developed regions of the Swan Pond River watershed). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the watershed for the Swan Pond River embayment system, a portion of the freshwater flow and transported nitrogen passes through one main surface water systems (e.g. un-named creek discharging into the head of Swan Pond) prior to entering the estuary, producing the opportunity for reductions in nutrient loading, primarily through nitrogen attenuation.

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

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

watershed, 1) a creek running along Hyda Way Lane (which has been named Hyda Way Creek) and discharging to Swan Pond which is then connected to the main tidal channel of the Swan Pond River system (Figure IV-6).

Figure IV-6.   Location of the Stream gauge (red symbol) within the watershed to the Swan Pond River Embayment system. The site was monitored to determine freshwater flow and nitrogen load to the estuary. The gauging site on Hyda Way Creek was off Hyda Way Road.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load to the stream watershed, as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater stream discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen within the contributing area up gradient from the various gauging sites. Flow and nitrogen load were determined at the gauge location for 17 months of record (Figure IV-7). During the study period, a velocity profile was completed on the creek every month to two months. The summation of the products of creek subsection areas of the channel cross-section and the respective measured velocities represent the computation of instantaneous flow (Q) through the creek.

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

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

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

where by:

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

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

Periodic measurement of flows over the entire "stream" gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauge. Water level data obtained every 10-minutes was averaged to obtain hourly stages for the creek. These hourly stages values where then entered into the stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. A complete annual record of flow in the creek (365 days) was generated for the surface water discharge flowing into the head of Swan Pond / Swan Pond River embayment system from Hyda Way Creek.

The annual flow record for the surface water flow at the gauge was merged with the nutrient data set generated through the weekly water quality sampling performed at the gauge location to determine nitrogen loading rates to the head of the Swan Pond River System. Nitrogen discharge from Hyda Way Creek was calculated using the paired daily discharge and daily nitrogen concentration data to determine the mass flux of nitrogen through the specific gaging site. For the creek gauge location, weekly water samples were collected (at low tide for a tidally influenced stage) in order to determine nutrient concentrations from which nutrient load was calculated. In order to pair daily flows with daily nutrient concentrations, interpolation

between weekly nutrient data points was necessary. These data are expressed as nitrogen mass per unit time (kg/d) and can be summed in order to obtain weekly, monthly, or annual nutrient load to the embayment system as appropriate. Comparing these measured nitrogen loads based on flow in the creek and water quality sampling to predicted loads based on the land use analysis allowed for the determination of the degree to which natural biological processes within the watershed to the gauged creek currently reduces (percent attenuation) nitrogen loading to the overall embayment system.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Creek flowing into head of Swan Pond

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, the un-named creek which discharges into the upper portion of Swan Pond does not have an up-gradient pond from which that creek discharges. Rather, this small creek appears to be groundwater fed and emanates from a generally wooded area up-gradient of Centre Street. The outflow leaving the wooded areas up gradient of Centre Street travels through an upland environment just prior to discharging directly into the head of Swan Pond. The creek outflow from the wooded area up gradient of the gauge located ~100 meters down gradient of the Centre Street crossing of the un-named Creek may serve to contribute to the attenuation of nitrogen and also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within associated wetland areas, riparian zones and streambed associated with the Creek. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Creek above the gauge site and the measured annual discharge of nitrogen to the tidally influenced Swan Pond at the head of the overall Swan Pond River system, Figure IV-7.

At the Creek gauge site (situated down gradient of Centre Street and parallel to Hyda Way), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater creek portion of the Swan Pond River estuarine system that carries the flows and associated nitrogen load to the near shore waters of Cape Cod Bay. As Swan Pond is tidally influenced, the gauge was located as far down gradient along the Creek reach such that freshwater flow could be measured at low tide. To confirm that freshwater was being measured the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 0.1 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked approximately monthly each time the site was visited and a flow measurement obtained. The gauge on the Creek was installed on June 3, 2005 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until October 11, 2006 for a total deployment of 17 months.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the un-named Creek site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to Swan Pond and subsequently, the main tidal channel of the Swan Pond River system. This measured attenuated mass discharge is reflective of the biological

processes occurring in the stream channel and riparian zone contributing to nitrogen attenuation (Figure IV-7 and Table IV-4). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the gauge site.

The annual freshwater flow record for Hyda Way Creek as measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the Creek was 16% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September and ending in August (low flow to low flow) was 453 m^3/day compared to the long term average flows determined by the USGS modeling effort (382 m^3/day).

The difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Hyda Way Creek was considered to be insignificant given the relatively small flow and associated load. The similar long-term average flow based on watershed recharge rates and the MEP measured flow in Hyda Way Creek discharging to Swan Pond would indicate that the Creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within Hyda Way Creek outflow were low to moderate, 0.635 mg N L^{-1}, which when coupled with the low volumetric flows yielded a small average daily total nitrogen discharge to the estuary of 0.29 kg/day and a measured total annual TN load of 105 kg/yr. In the Creek (freshwater), nitrate (0.218 mg N L^{-1}) was significantly less than half of the total nitrogen pool (34%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging was largely taken up by plants within the stream ecosystem.   This is further supported when considering that dissolved and particulate organic nitrogen constituted 48% of the total nitrogen pool discharging from the Creek to Swan Pond. The low concentration of inorganic nitrogen (0.327 mg N L^{-1}) suggests that only small increases in nitrogen attenuation by enhancing biological removal associated with stream transport in this system are likely. Opportunities for enhancing nitrogen attenuation elsewhere in the overall watershed to Swan Pond River could be considered, but there appears to be only a small yield likely related to Hyda Way Creek.

From the measured nitrogen load discharged by the Hyda Way Creek to the Swan Pond River estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of creek watershed derived nitrogen presently occurring during transport to the estuary. Based upon lower total nitrogen load (105 kg yr^{-1}) discharged from the freshwater Creek compared to the nitrogen added by the various land-uses (121 kg yr^{-1}). Therefore, it appears that the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 13% (i.e. 13% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the hydrologic and biogeochemical characteristics of the up gradient pond(s) capable of attenuating nitrogen.   The directly measured nitrogen load from the Creek was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-4. Comparison of water flow and nitrogen discharges from Hyda Way Creek (freshwater) flowing along Hyda Way and discharging to Swan Pond and the estuarine reach of the Swan Pond River system. The Stream data is from the MEP stream gauging effort. Watershed freshwater discharge and nitrogen loading are based upon the MEP watershed modeling effort by USGS and the MEP watershed nitrogen model, respectively.

Stream Discharge Parameter	Un-named Creek Discharge[a] (Swan Pond)	Data Source
Total Days of Record	365[b]	(1)
Flow Characteristics		
Stream Average Discharge (m3/day) **	453	(1)
Contributing Area Average Discharge (m3/day)	382	(2)
Discharge Stream 2004-05 vs. Long-term Discharge	16%	
Nitrogen Characteristics		
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.218	(1)
Stream Average Total N Concentration (mg N/L)	0.635	(1)
Nitrate + Nitrite as Percent of Total N (%)	34%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	0.29	(1)
TN Average Contributing UN-attenuated Load (kg/day)	0.332	(3)
Attenuation of Nitrogen in Pond/Stream (%)	13%	(4)

(a) Flow and N load to stream discharging from Un-named Creek to Swan Pond includes apportionments of Pond contributing areas.
(b) September 1, 2005 to August 31, 2006.
 ** Flow is an average of annual flow for 2005-2006

(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages; the fractional flow path from each sub-watershed which contribute to the flow in the creek to Swan Pond; and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attenuation rates.
(4) Calculated based upon the measured TN discharge from the creek vs. the unattenuated watershed load.
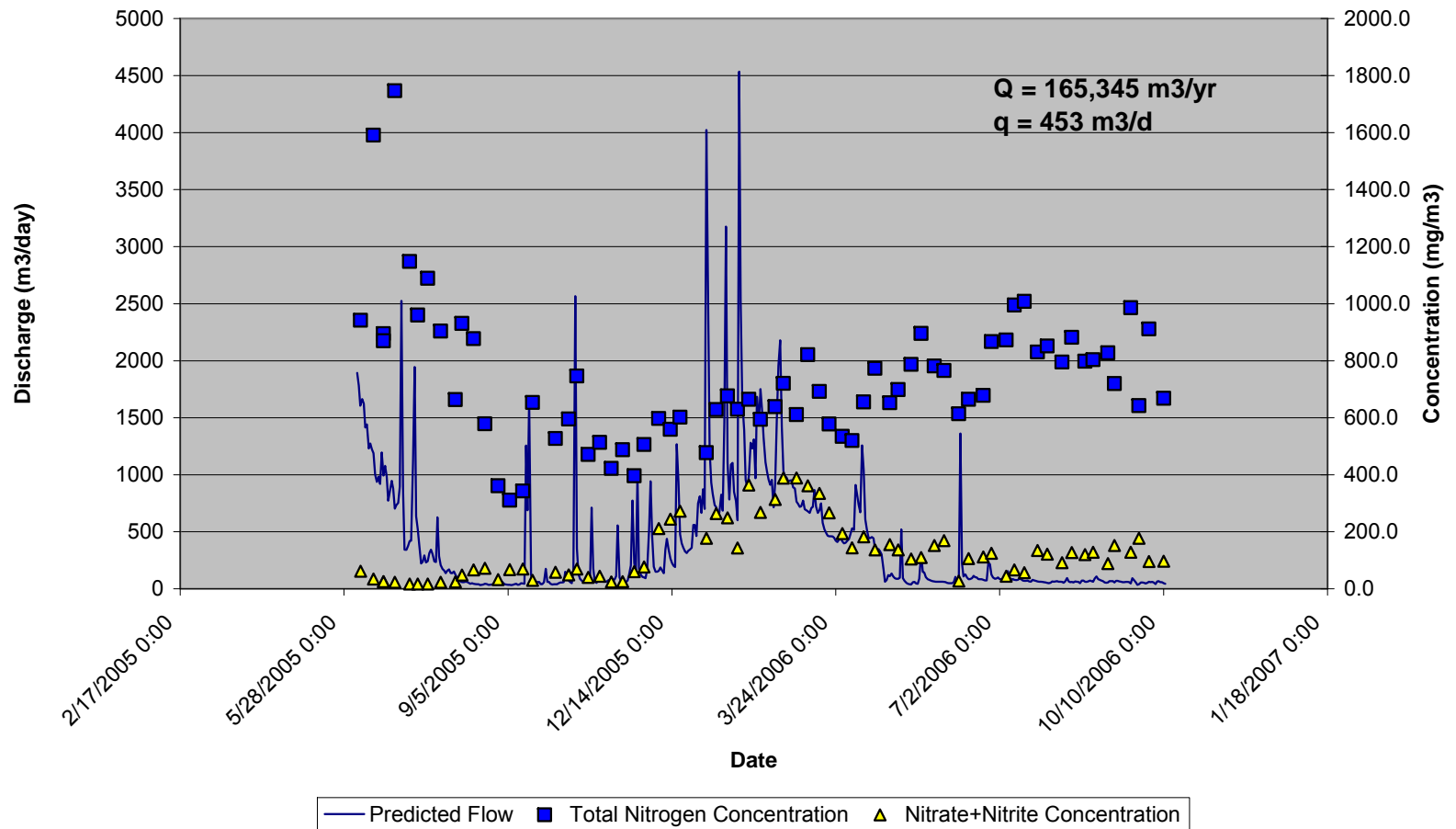
Figure IV-7.   Gauged Creek flowing along Hyda Way and discharging directly into the head of Swan Pond (solid blue line), nitrate+nitrite (yellow triangle) and total nitrogen (blue square) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Swan Pond River (Table IV-4).

Table IV-5.	Summary of annual volumetric discharge and nitrogen load from Hyda Way Creek discharge to Swan Pond (head of Swan Pond River estuarine system) based upon the data presented in Figure IV-X and Table IV-4.			
Embayment System	Period of Record	Discharge (m³/year)	Attenuated Load (kg/year)	
			Nox	TN
Swan Pond River (tidal channel) Un-named Creek to Swan Pond	September 1, 2005 to August 31, 2006	165,345	36	105
Swan Pond River (tidal channel) Un-named Creek to Swan Pond	Based on Watershed Area and Recharge	139,256	-	-

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

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Swan Pond River system predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the water column (once it entered) then predicting water column nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayment from the surrounding watersheds, it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton "particles". Most of these "particles" remain in the water column for sufficient time to be flushed out to a down gradient larger water body (like Nantucket 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 sediments. 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 associated nitrogen "load" become incorporated into the surficial sediments of the system.

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic

(oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with 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). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, such as found within much of the bordering region to the nearby Lewis Bay main basin. In contrast, regions of high deposition like Hyannis Inner Harbor, which is essentially a dredged boat basin, typically support anoxic sediments with elevated rates of nitrogen release during summer months. The consequence of this deposition is that these basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

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

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Swan Pond River Harbor embayment system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), a total of 17 sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from a total of 16 sites in the Swan Pond River system. Cores were collected from 9 sites within main tidal channel that constitutes Swan Pond River and 7 cores were collected from the main basin of Swan Pond up-gradient of Upper County Road (Figure IV-8). All the sediment cores for this system were collected in July-August 2005. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shore side field lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. . The sampling locations and numbers of cores collected are listed below. The spatial distribution of the stations is presented in Figure IV-8.
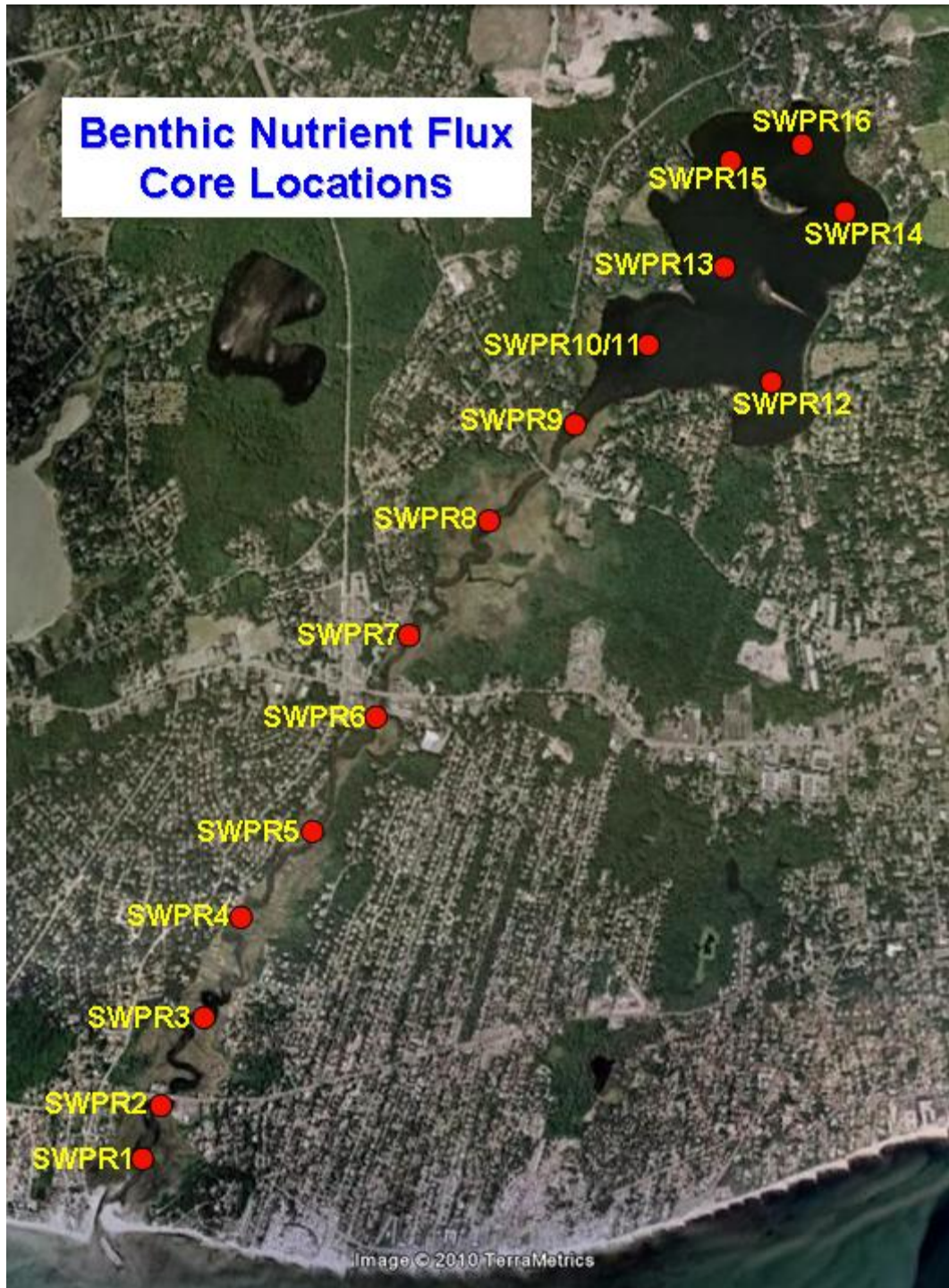
Figure IV-8.   Swan Pond River Embayment System sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers relate to station i.d.'s/numbers referenced in Table IV-6.

Swan Pond River Benthic Nutrient Regeneration Cores

- SWPR-1   1 core   (Lower River (south))
- SWPR-2   1 core   (Lower River (south))
- SWPR-3   1 core   (Lower River (south))
- SWPR-4   1 core   (Lower River (south))
- SWPR-5   1 core   (Lower River (south))
- SWPR-6   1 core   (Lower River (south))
- SWPR-7   1 core   (Upper River (north))
- SWPR-8   1 core   (Upper River (north))
- SWPR-9   1 core   (Upper River (north))
- SWPR-10/11   2 cores   (main basin Swan Pond)
- SWPR-12   1 core   (main basin Swan Pond)
- SWPR-13   1 core   (main basin Swan Pond)
- SWPR-14   1 core   (main basin Swan Pond)
- SWPR-15   1 core   (main basin Swan Pond)
- SWPR-16   1 core   (main basin Swan Pond)

Sampling was distributed throughout the upper open water basin of Swan Pond as well as the main tidal channel (Swan Pond River) connecting Swan Pond to Nantucket Sound. The benthic regeneration results for each site were combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-water column exchange follows the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1998) for nutrients and metabolism. Upon return to the field laboratory (Bass River Marina) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining samples frozen ($-20^{o}C$) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

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

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

Water column nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (water column and benthic), and uptake (e.g. photosynthesis). As stated above, during the warmer summer months the sediments of shallow embayments typically act as a net source of nitrogen to the overlying waters and help to stimulate eutrophication in organic rich systems. However, some sediments may be net sinks for nitrogen and some may be in "balance" (organic N particle settling = nitrogen release). Sediments may also take up dissolved nitrate directly from the water column and convert it to dinitrogen gas (termed "denitrification"), hence effectively removing it from the ecosystem. This process is typically a small component of sediment denitrification in embayment sediments,

since the water column nitrogen pool is typically dominated by organic forms of nitrogen, with very low nitrate concentrations. However, this process can be very effective in removing nitrogen loads in some systems, particularly in streams, ponds and salt marshes, where overlying waters support high nitrate levels.

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

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

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

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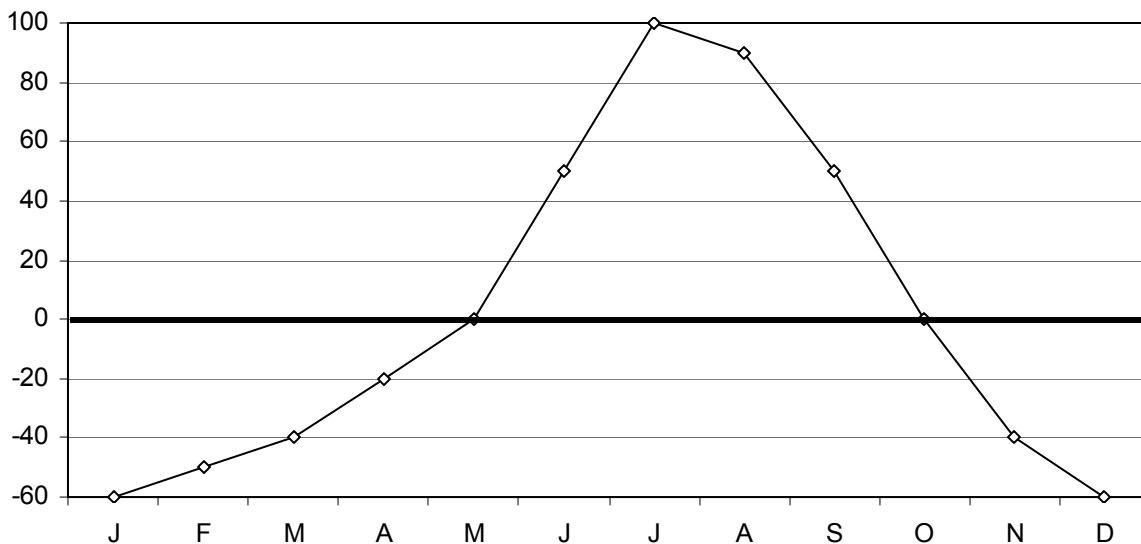
Figure IV-9.   Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

In order to determine the net nitrogen flux between water column and sediments, all of the above factors were taken into account. The net input or release of nitrogen within each of the three components of Swan Pond River were 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 component embayment basins (main basin of Swan Pond, upper and lower Swan Pond River) of the Swan Pond River System in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model. The distribution of cores in each harbor was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and sediment type and an analysis of each site's tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site, the average summer particulate carbon and nitrogen concentration within the overlying water and the tidal velocities from the hydrodynamic model (Chapter V). Depositional areas were also determined from an analysis of the sediment type. Two levels of settling were used.

If the sediments were organic rich and fine grained, and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated coarse-grained sediments and low organic content and high velocities, then half this settling rate was used. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. This approach has been previously validated in outer Cape Cod embayments (Town of Chatham embayments) by examining the relative fraction of the sediment carbon turnover (total sediment metabolism), which would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins are driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments. Additional, validation has been conducted on deep enclosed basins (with little freshwater inflow), where the fluxes can be determined by multiple methods. In this case the rate of sediment regeneration determined from incubations was comparable to that determined from whole system balance.

Net nitrogen release rates for use in the water quality modeling effort for the main basins of the Swan Pond River system (Chapter VI) are presented in Table IV-6. There was a clear spatial pattern of sediment nitrogen flux, with net uptake of nitrogen within the upper salt pond (Swan Pond) and within the lower Swan Pond River, and net nitrogen release within the upper Swan Pond River. The sediments within the Swan Pond River Estuary showed nitrogen fluxes typical of similarly structured systems within the region and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the level of nitrogen loading to this system and its relatively high flushing rate.

Table IV-6.	Rates of net nitrogen return from sediments to the overlying waters of the component basins of the Swan Pond River System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July -August rates.			
Location	**Sediment Nitrogen Flux (mg N m^{-2} d^{-1})**			**Station numbers/i.d. ***
	Mean	**S.E.**	**# sites**	
Swan Pond River Estuarine System				
Swan Pond Basin	-8.0	23.0	6	SWPR 10 - 16
Upper Swan Pond River	12.3	43.4	3	SWPR 7,8,9
Lower Swan Pond River	-10.3	15.3	6	SWPR 1 - 6
* Station numbers refer to Figure IV-11.				

Net nitrogen release or uptake from the sediments within the Swan Pond River Embayment System is comparable to other similar embayments with similar configuration and flushing rates in southeastern Massachusetts. In addition, the spatial pattern of sediment N release was also similar to other systems, with the upper embayment basin (Swan Pond) and lower tidal river showing a low rate of net nitrogen uptake and the middle basin, the upper reach of the tidal river, showing a low rate of net nitrogen release. For example a nearby estuary similarly configured to Swan Pond River, the Parker's River Estuary, also supports net nitrogen uptake in both of its basins: 1) large open water basin at the head of its tidal river (Seine Pond) has a net nitrogen uptake of -16.9 mg N m^{-2} d^{-1} and 2) the salt marsh basin near its inlet (Lewis

Pond) has a net nitrogen uptake of -11.8 mg N m^{-2} d^{-1}. Much like Upper Swan Pond River, the upper tidal river reach of Parker's River estuary also had a net nitrogen release (39.9 mg N m^{-2} d^{-1}) and lower high velocity channel (Lewis Pond Channel) has a net nitrogen uptake (-11.6 mg N m^{-2} d^{-1}) similar to the Lower Swan Pond River. The upper reach of the Swan Pond River was also similar to The River within Pleasant Bay, which has net nitrogen release rates of 12.0 - 34.2 mg N m^{-2} d^{-1}. In addition, the spatial pattern of sediment N release was also similar to other systems, with the salt marsh basins and creeks showing net nitrogen uptake, the embayment depositional basins with oxidized surficial sediments showing low rates of net nitrogen uptake and the depositional areas within the tidal river showing net nitrogen release. Swan Pond also has similar rate of net nitrogen uptake to other, similar terminal open water basins, such as the Eel Pond within the Phinneys Harbor System, Eel River within Three Bays, lower Muddy Creek and Frost Fish Creek, which have net nitrogen uptake rates of -8.6, -6.4, -16.0, -5.1 mg N m^{-2} d^{-1}, respectively. The net uptake in Swan Pond likely results from its depositional nature and organic enriched sediments. Also Mill Creek within Lewis Bay (-14.3 mg N m-2 d^{-1}) and above noted Lewis Pond in Parkers River have both similar rates of net nitrogen uptake and support significant salt marsh, as does Scudder Bay (-13.2 mg N m-2 d^{-1}) and lower Halls Creek (-11.1 mg N m-2 d^{-1}). Halls Creek supports sediments very similar to the lower reach of the Swan Pond River, with oxidized medium and fine sands, and a similar rate on net nitrogen uptake.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

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

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

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

A hydrodynamic study was performed for the Swan Pond River system, which is located in the Town of Dennis Port, Massachusetts on Cape Cod. A section of a topographic map in Figure V-1 shows the general study area. The Swan Pond River system is linear progressing from Nantucket Sound northeast to the head of the system in Swan Pond. The system is crossed by three bridges which do not cause any significant attenuation. The entire Swan Pond River system has a surface coverage of approximately 210 acres.

Circulation in the Swan Pond River system is dominated by tidal exchange with Nantucket Sound. The river is connected to Nantucket Sound through a narrow, shallow, partially structured inlet that often has a significant ebb shoal. The west side of the inlet is delineated with a rubble mound jetty which is approximately 680 feet-long. Over the length of the system, there is considerable attenuation of the tide range. Between the inlet and the head of the system, Swan Pond, the average tide range is reduced from 3.5 feet to 1.4 feet, a reduction of

2.1 feet or 60%. This reduction is caused by frictional losses along the 2.5 mile-long reach of the River, to the Pond at the head of the system.
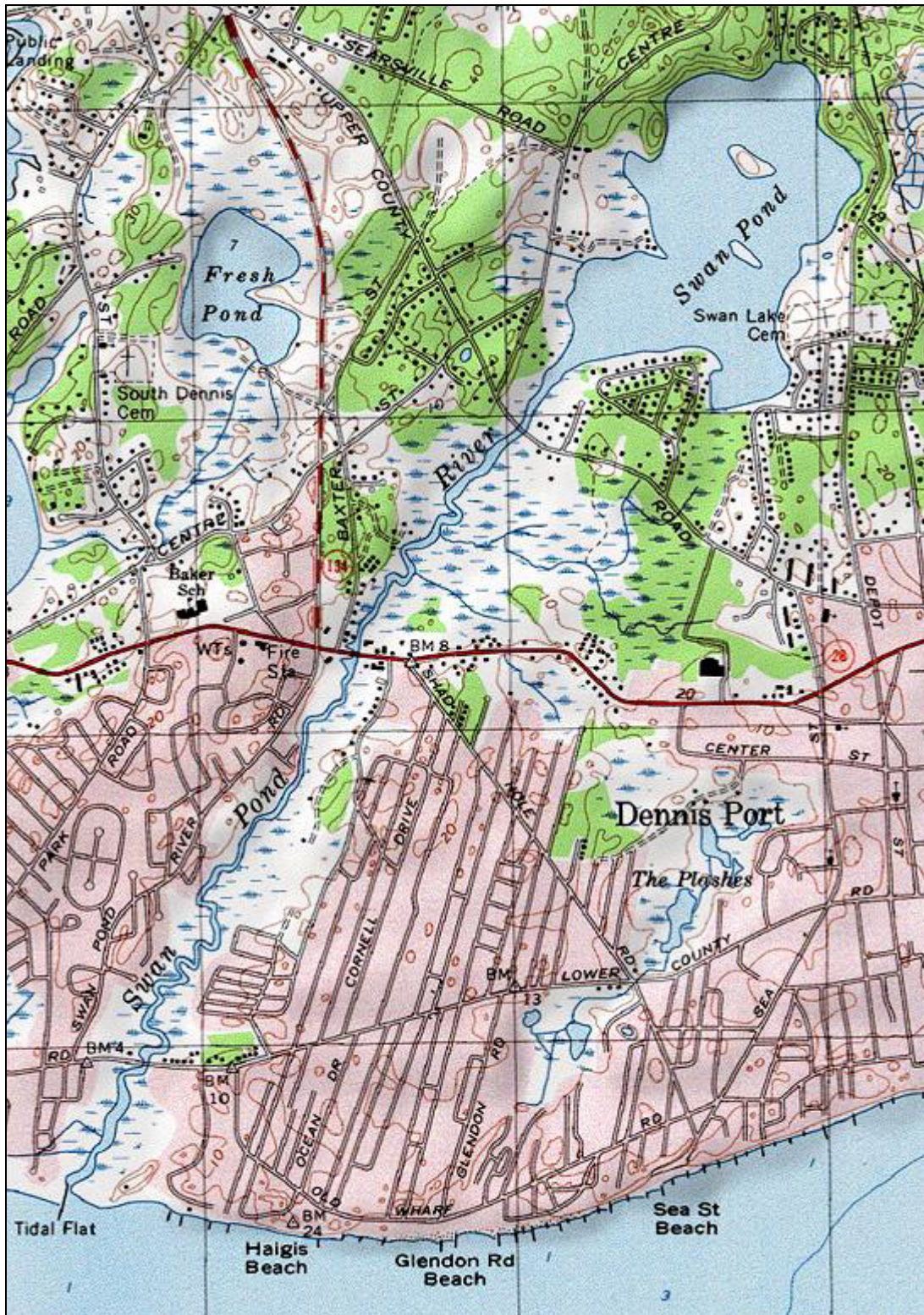
Figure V-1. Topographic map detail of the Swan Pond River, Dennis Port, Massachusetts.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry, tide, and circulation velocity data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Swan Pond River was performed to determine the present variation of embayment and channel depths throughout the system. In addition to bathymetry, tides were recorded at seven locations within the system for at least a complete lunar month (29.5 days). These tide data were necessary to run and calibrate the hydrodynamic model of the system. Finally, an Acoustic Doppler Current Profiler (ADCP) survey was completed during a single tide cycle to measure ebb and flood velocities across the inlet. The ADCP data were used to compute system flow rates and to provide an independent means of verifying the performance of the hydrodynamic model.

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

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

V.2 DATA COLLECTION AND ANALYSIS

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

V.2.1 Bathymetry Data Collection

Bathymetry data in Swan Pond River were collected during July 2005. The July 2005 survey employed a single-beam acoustic fathometer mounted to a small survey vessel. Positioning data were collected using a differential GPS. Survey paths and measured depths are shown in Figure V-2. The resulting bathymetric surface created by interpolating the data to a finite element mesh is shown in Figure V-3. All bathymetry was tide corrected, and referenced to the NGVD 29 vertical datum, using survey benchmarks located in the project area.
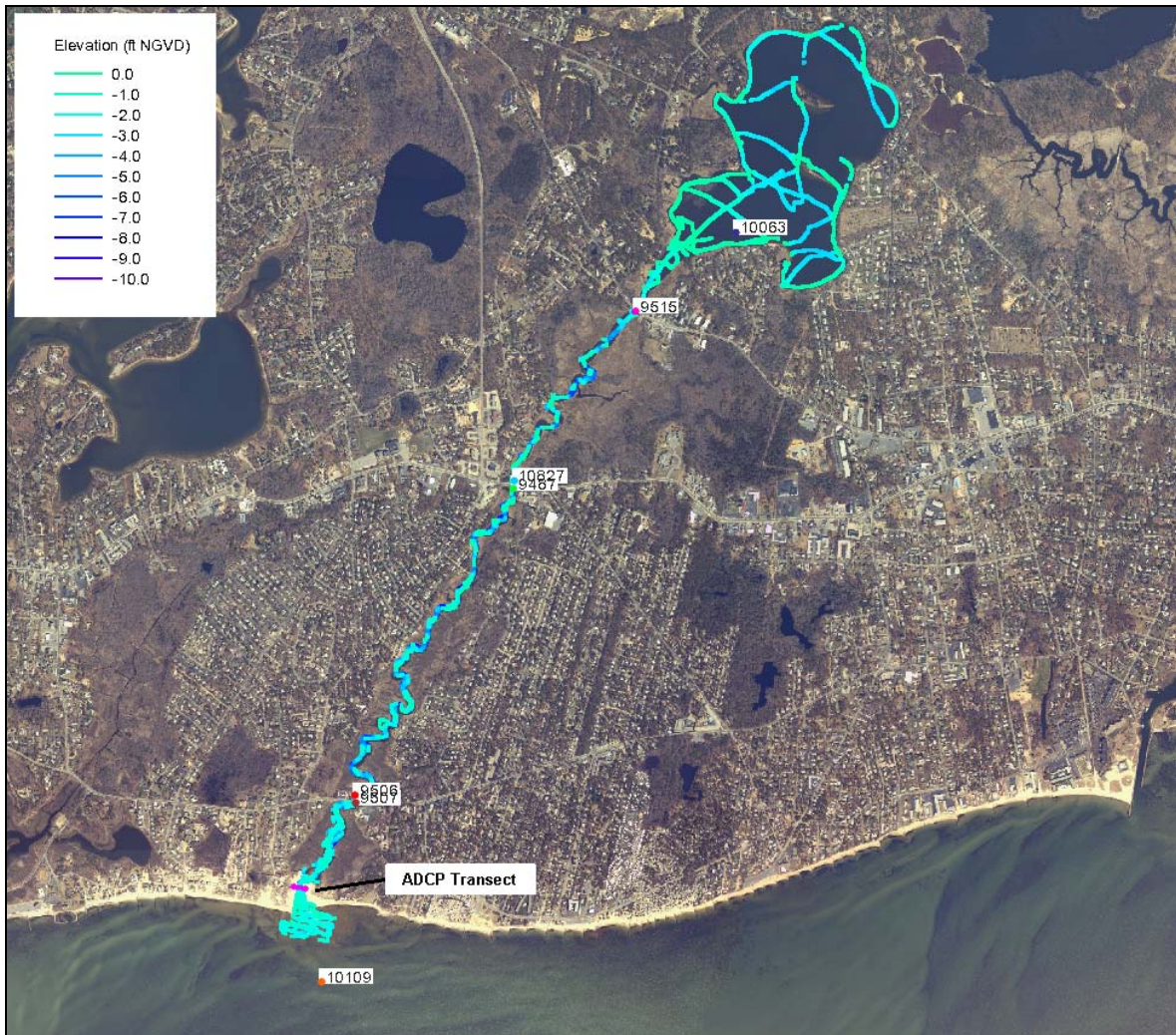
Figure V-2.   Transects (blue-shaded lines) from the bathymetry survey of Swan Pond River and markers (colored circles) show the locations of the tide recorders deployed for this study.

Results from the survey show that the overall river channel is shallow with a deeper thalweg at significant channel bends in the river. The channel bottom varies with changes in the channel width and orientation. At the head of the system, Swan Pond, the pond is shallow with an average depth of -1.8 feet, and a maximum depth of -3.8 feet.

V.2.2 Tide Data Collection and Analysis

Tide data records were collected at seven stations in the Swan Pond River System: 1) offshore the inlet (10109), 2) Lower County Road bridge-south (9507), 3) Lower County Road bridge-north (9506), 4) Route 28 bridge-south (9487), 5) Route 28 bridge-north (10827), 6) Upper County Road bridge-south (9515), and 7) Swan Pond at the head of the system (10063). The locations of the stations are shown in Figure V-2.   The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 30-day period beginning July 11, 2005. The elevation of each gauge was surveyed relative to NGVD 29.   Duplicate offshore gauges were deployed to ensure data recovery, since the offshore tide record is crucial for developing the open boundary condition of the hydrodynamic model of the Swan Pond River system. Data from the gauges inside the system were used to calibrate the model.
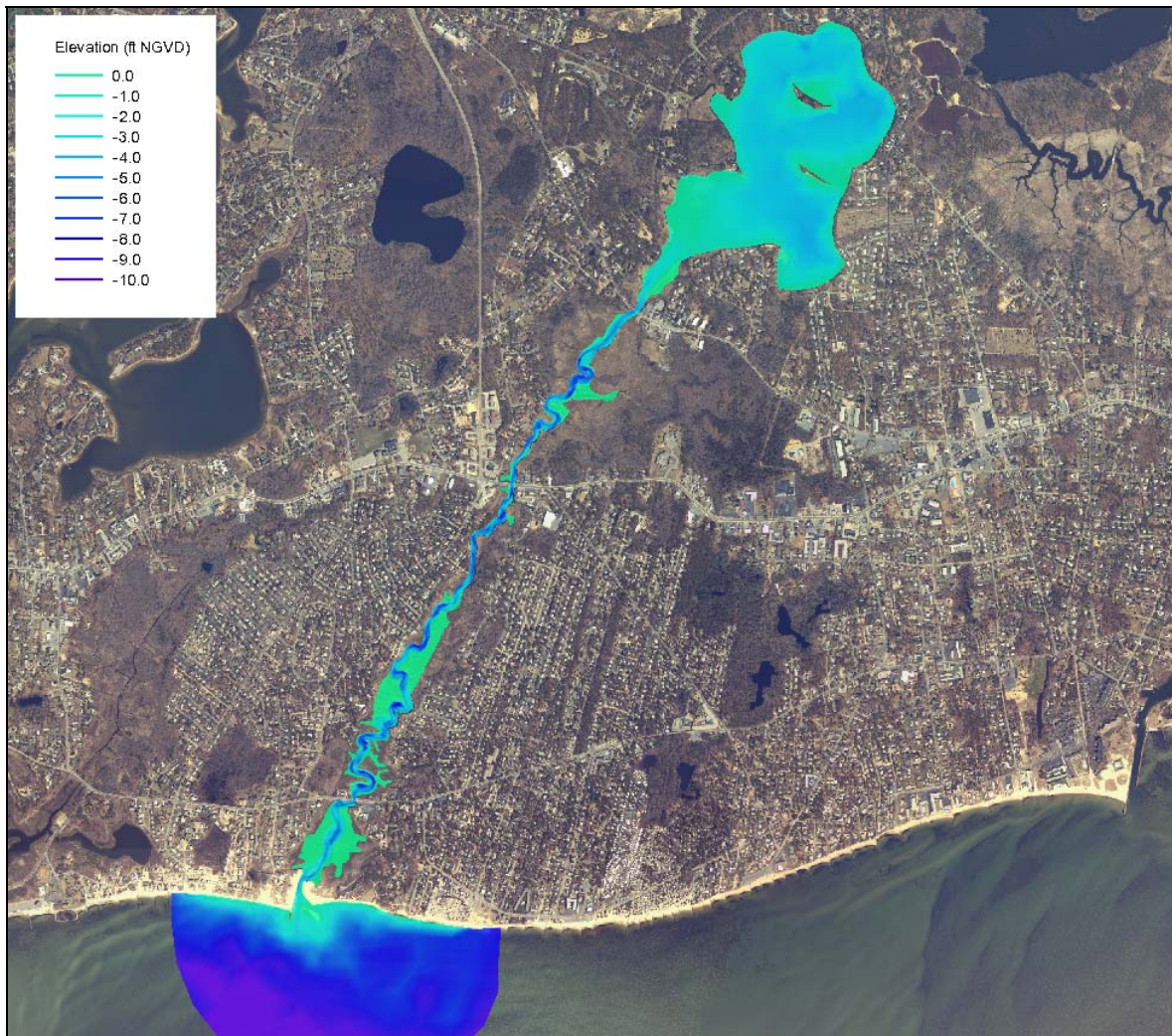
Figure V-3.   Plot of interpolated finite-element grid bathymetry of the Swan Pond River system, shown superimposed on 2005 aerial photos. Bathymetric contours are shown in color at one-foot intervals.

Plots of the tide data from seven representative gauges are shown in Figure V-4, for the entire 30-day deployment. The neap-to-spring variation in tide can be seen in these plots. From the plot of the data from offshore Swan Pond River inlet, the tide reaches its maximum spring tide range of approximately 5.8 feet around July 22. About seven days earlier the neap tide range is much smaller, as small as 2.9 feet.

A visual comparison in Figure V-5 between tide elevations at the stations in Swan Pond River shows that there is a reduction in the tide range as the tide propagates to the upper reaches of the system. The loss of amplitude with distance from the inlet is described as tidal attenuation. Frictional mechanisms dissipate tidal flow energy, resulting in a reduction of the height of the tide. Tide attenuation is accompanied by a time delay (or phase lag) in the time of high and low tide (relative to the offshore tide), which becomes more pronounced farther into an estuary.
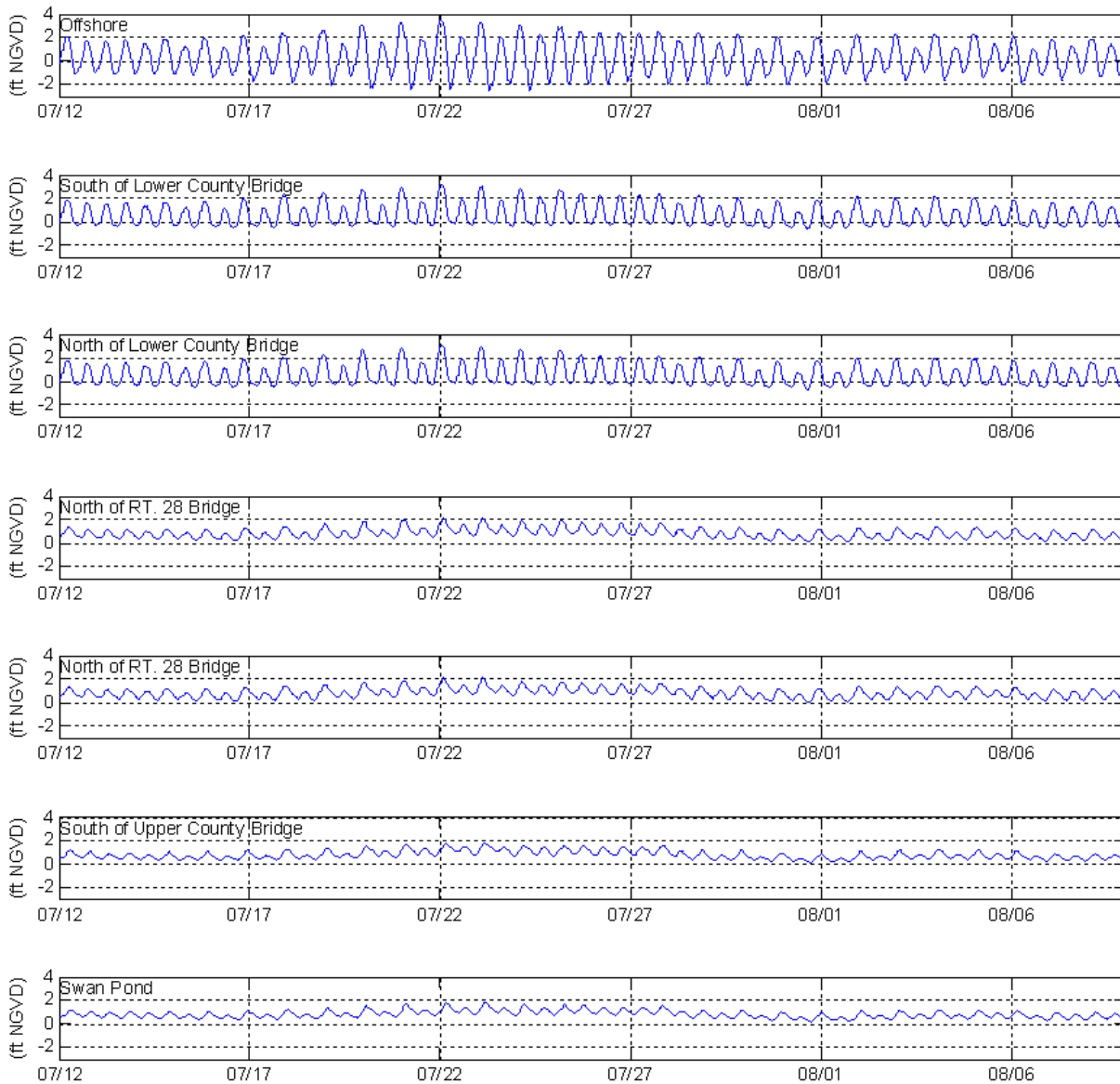
Figure V-4.   Plots of observed tides for the Swan Pond River system, for the 30-day period between July 12 and August 9, 2005. The top plot shows tides offshore Swan Pond River inlet, in Nantucket Sound. Tides recorded at downstream and upstream of Lower County Road, downstream and upstream of Route 28, downstream of Upper County Road, and in Swan Pond are also shown. All water levels are referenced to the National Geodetic Vertical Datum of 1929 (NGVD).
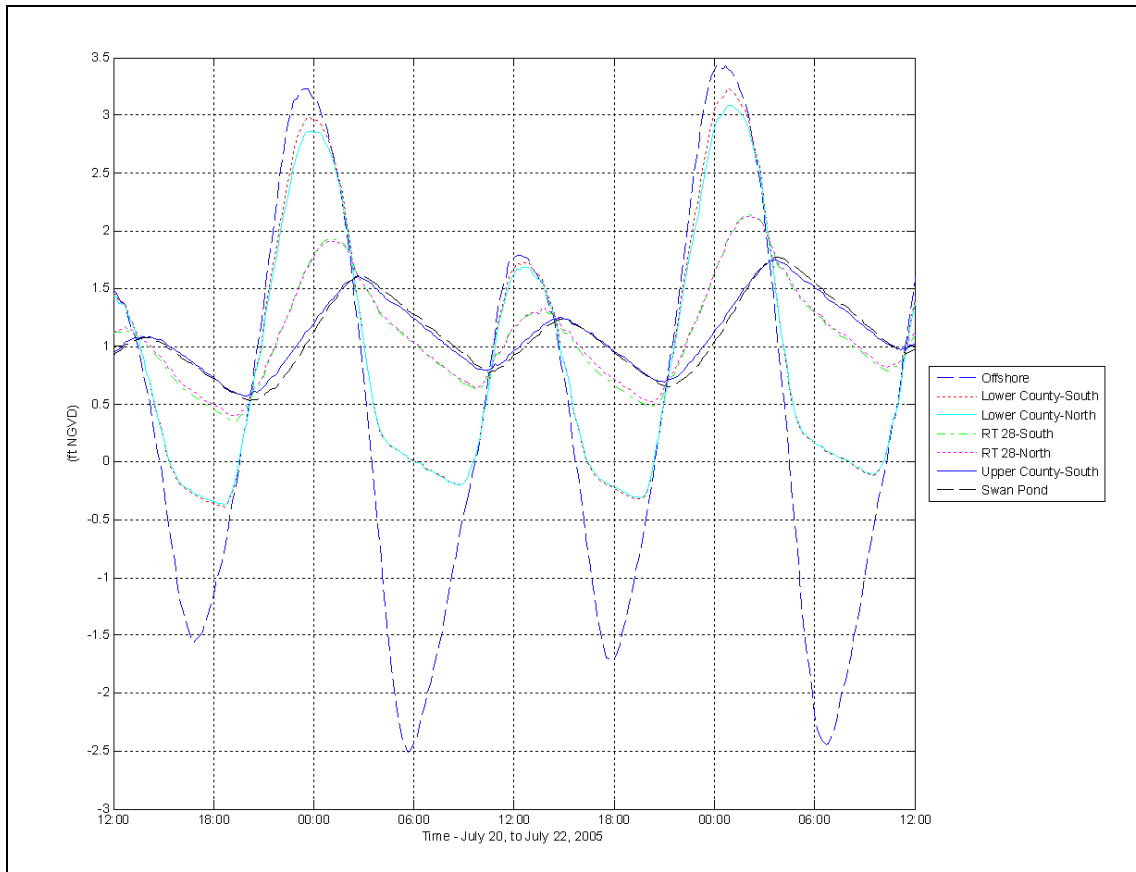
Figure V-5.   Plot showing two tide cycles at the stations in the Swan Pond River system plotted together. Demonstrated in this plot is the frictional damping effect caused by flow restrictions along the river's length. The damping effects are seen only as a lag in time of high and low tides from Nantucket Sound. The maximum time lag of low tide between the Sound and Swan Pond in this plot is 225 minutes (3.8 hours).

Standard tide datums were computed from the 30-day records. These datums are presented in Table V-1. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available; however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW.

As the tide propagates from Nantucket Sound to the upper reaches of the River attenuation of the tide occurs. This is observed as a reduction in the tide range and also as a delay in the time of high and low tide during each tide cycle. The mean tide range in Nantucket Sound is 3.5 feet. At the head of the system the mean tide range is reduced to 0.6 feet by frictional losses along the length of the River.

The tides in Nantucket Sound are semi-diurnal, meaning that there are typically two tide cycles in a day. There is usually a small variation in the level of the two daily tides. This variation can be seen in the differences between the MHHW and MHW, as well as the MLLW and MLW levels.

A more thorough harmonic analysis of the tidal time series was performed to produce tidal amplitude and phase of the major tidal constituents, and provide assessments of hydrodynamic 'efficiency' of the system in terms of tidal attenuation. This analysis also yielded a quantitative assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of the system.

Table V-1.	Tide datums computed from a 30-day period from the tide records collected in the Swan Pond River system. Datum elevations are given relative to NGVD 29.						
Tide Datum	Offshore	Lower County - South	Lower County - North	Route 28 - South	Route 28 - North	Upper County - South	Swan Pond
Maximum Tide	3.44	3.24	3.09	2.11	2.09	1.85	1.86
MHHW	2.41	2.25	2.16	1.47	1.46	1.26	1.31
MHW	1.94	1.84	1.76	1.25	1.25	1.12	1.16
MTL	0.18	0.74	0.70	0.80	0.81	0.81	0.85
MLW	-1.59	-0.35	-0.37	0.34	0.37	0.51	0.53
MLLW	-1.84	-0.40	-0.41	0.29	0.32	0.46	0.48
Minimum Tide	-2.61	-0.59	-0.61	0.00	0.03	0.13	0.16

A harmonic analysis was performed on the time series from each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The observed astronomical tide is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-6. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of eight tidal constituents in the Swan Pond River system.

The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to throughout the system. The total range of the M_2 tide is twice the amplitude, or 3.4 feet for the offshore gauge. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 has an amplitude of 0.12 feet near the system inlet, but is reduced progressive up to the head of the system. The M_6 has a very small amplitude in the system (less than 0.1 feet at all gauge stations).

For all the other included constituents, except for the fortnightly M_{sf}, amplitudes decrease with distance into the system. The other major tide constituents also show reductions progressing up to the head of the system. The diurnal tides (once daily), K_1 and O_1, possess amplitudes of approximately 0.4 feet. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, contribute significantly to the total tide signal, with amplitudes of 0.15 feet and 0.50 feet, respectively. 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, and has an amplitude of 0.1 ft at the base of the system and it increase to about 0.2 feet at the head of the system.
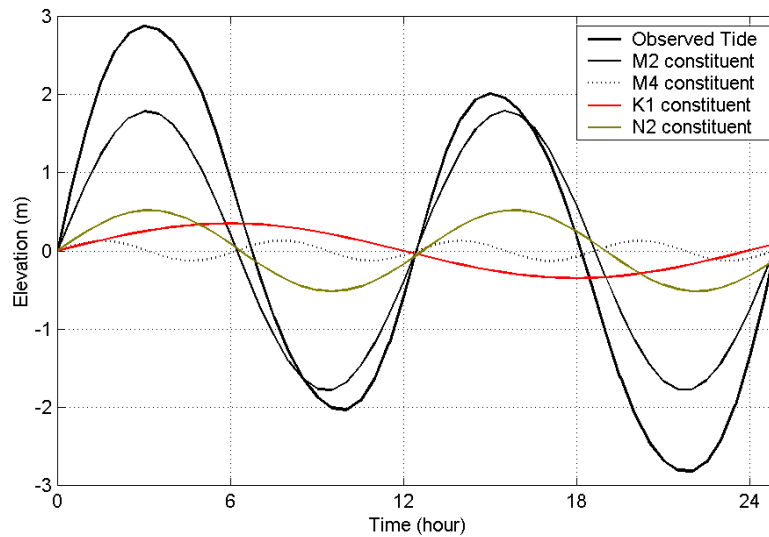
Figure V-6.   Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2.	Major tidal constituents determined for gauge locations in Swan Pond River, July 11 through August 9, 2005.							
	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Nantucket Sound (offshore)	1.69	0.12	0.07	0.15	0.50	0.40	0.36	0.12
Lower County - South	1.08	0.16	0.03	0.09	0.27	0.30	0.28	0.19
Lower County - North	1.05	0.15	0.03	0.09	0.26	0.29	0.27	0.19
Route 28 - South	0.40	0.06	0.01	0.03	0.09	0.16	0.16	0.21
Route 28 - North	0.39	0.06	0.01	0.03	0.09	0.16	0.16	0.22
Upper County - South	0.26	0.02	0.01	0.03	0.06	0.11	0.13	0.23
Swan Pond	0.27	0.03	0.01	0.02	0.06	0.12	0.13	0.22

There is a constant reduction in the constituent amplitudes across the length of the River, which is also exhibited in the phase change of the tide as seen from the results of the harmonic analysis. Table V-3 shows the delay of the M_2 at different points in the Swan Pond River system, relative to the timing of the M_2 constituent in Nantucket Sound, offshore the inlet. The analysis of the data from Swan Pond shows that there is a 190 minute delay between the inlet and the farthest reach of the system. The lag for each of the locations is illustrated in Figure V-5.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow. The results of an analysis to determine the energy distribution (or variance) of the original water elevation time series for the Swan Pond River system is presented in Table V-4 compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 23 constituents determined by the harmonic analysis). Subtracting the tidal signal from the original elevation time series resulted 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. Figure V-7 shows the comparison of the measured tide from the tide gauge south of the Lower County Road bridge and inside the inlet, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-3.	M_2 tidal constituent phase delay (relative to Nantucket Sound) for gauge locations in the Swan Pond River system, determined from measured tide data.
Station	**Delay (minutes)**
Lower County - South	20.0
Lower County - North	21.3
Route 28 - South	89.4
Route 28 - North	91.1
Upper County - South	171.9
Swan Pond	189.9

Table V-4 shows that the variance of tidal energy decreases for stations that are farther from the inlet. The analysis also shows that tides are responsible for approximately 99% of the water level variations at the base of the system and is slowly reduced to approximately 70% of the water level changes at the head within Swan Pond. The remaining variance was the result of atmospheric forcing, due to winds, barometric pressure gradients, or fresh water entering the system.

Table V-4.	Percentages of Tidal versus Non-Tidal Energy for stations in Swan Pond River, July to August 2005.		
TDR Location	**Total Variance (ft^2)**	**Tidal (%)**	**Non-tidal (%)**
Nantucket Sound (offshore)	1.77	98.8	1.2
Lower County - South	0.79	95.4	4.6
Lower County - North	0.75	95.2	4.8
Route 28 - South	0.17	81.9	18.1
Route 28 - North	0.16	82.6	17.4
Upper County - South	0.11	70.1	29.9
Swan Pond	0.11	72.9	27.1

It should be noted that the non-tidal energy is included within the data shown on the plots presented in Figure V-5 and V-6. For the observed data, the non-tidal energy is contained within the recorded water levels. The modeled water levels have the non-tidal energy included through the numerical representation of the estuarine system. The non-tidal from offshore in contained within the boundary condition, the remaining portions are represented within the system through friction, turbulence, groundwater enter the system, and barometric pressure. The plots do show slight variations in the water levels which is partial attributable to non-tidal forces acting on the system during the data collection period which were not represented within the numerical model due scale of the events and availability of data to represent it.
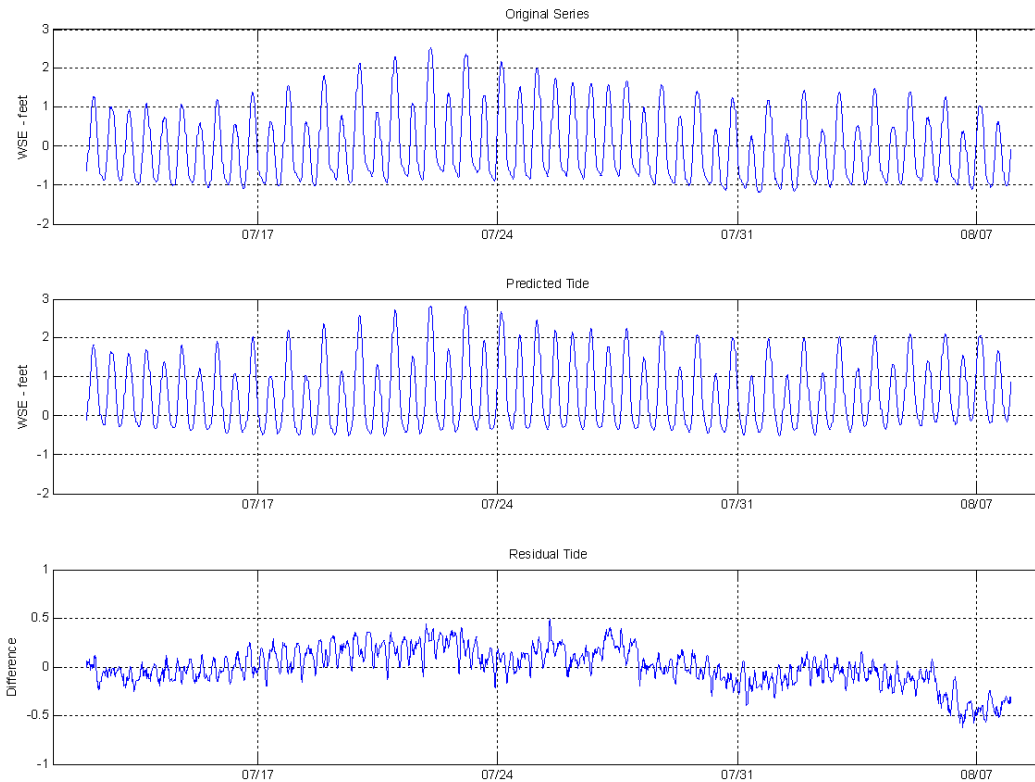
Figure V-7. Plot showing the comparison between the measured tide time series (top plot) inside the inlet, and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determined in the harmonic analysis of the Lower County-south gauge data. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

V.2.3 ADCP Data Analysis

Cross-channel current measurements were surveyed through a complete tidal cycle in the Swan Pond River system on July 26, 2005 to resolve spatial and temporal variations in tidal current patterns. The survey was designed to observe tidal flow across into and out of the system on regular intervals. The survey transect (indicated in Figure V-2) was located just inside the inlet. The data collected during this survey provided information that was necessary to model properly validate the hydrodynamic model of the Swan Pond River system.

The measurements were collected using an Acoustic Doppler Current Profiler (ADCP) mounted aboard a small survey vessel. The boat repeatedly navigated the pre-defined transect line across the inlet, approximately every 20 to 30 minutes, with the ADCP continuously collecting current profiles. This pattern was repeated for an approximate 12.5-hour duration to ensure measurements over the entire tidal cycle. The results of the data collection effort are high-resolution observations of the spatial and temporal variations in tidal current patterns throughout the survey area.

Measurements were obtained with a BroadBand 1200 kHz Acoustic Doppler Current Profiler (ADCP) manufactured by RD Instruments (RDI) of San Diego, CA. The ADCP was mounted to a specially constructed mast, which was rigidly attached to the rail of the survey

vessel. The ADCP was oriented to look downward into the water column, with the sensors located approximately 1 foot below the water surface. The mounting technique assured no flow disturbance due to vessel wake. The survey line was designed to measure as accurately as possible the volume flux through the inlet during a complete tidal cycle.

V.3 HYDRODYNAMIC MODELING

The focus of this study was the development of a numerical model capable of accurately simulating hydrodynamic circulation within the Swan Pond River system. Once calibrated, the model was used to calculate water volumes for selected sub-embayments as well as determine the volumes of water exchanged during each tidal cycle. These parameters are used to calculate system residence times, or flushing rates. The ultimate utility of the hydrodynamic model is to supply required input data for the water quality modeling effort described in Chapter VI.

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 four main steps required to implement RMA-2:
- Grid generation
- Boundary condition specification
- Calibration
- Verification

The extent of each finite element grid was generated using 2005 and 2009 color digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the inlet of the river system based on the tide gauge data collected offshore of Swan Pond River, in Nantucket Sound.

Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid generation

The grid generation process for the model was assisted through the use of the SMS package. The digital shoreline and bathymetry data were imported to SMS, and a finite element grid was generated to represent the estuary with 3135 elements and 8650 nodes. All regions in the system were represented by two-dimensional (depth-averaged) elements. The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties within the estuary. Fine resolution was required to simulate the channel constrictions that significantly impact the estuarine hydrodynamics. The completed grid is made up of quadrilateral and triangular two-dimensional elements. Reference water depths at each node of the model were interpreted from bathymetry data obtained in the recent field surveys and the recent NOAA LIDAR data along the coastline. The final interpolated grid bathymetry is shown in Figure V-8. The model computed water elevation and velocity at each node in the model domain.

Grid resolution is governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each region. Smaller cross channel node spacing in the meandering river channels was designed to provide a more detailed analysis in these regions of rapidly varying velocities and bathymetry. Widely spaced nodes were utilized in areas where velocity gradients were likely to be less acute; for example, on marsh plains and in broad, shoal sections in the model domain.

V.3.2.2 Boundary condition specification

Two types of boundary conditions were employed for the RMA-2 model of the Swan Pond River 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. Tidal boundary conditions were specified at the inlet from Nantucket Sound. TDR measurements from a gauge deployed offshore the inlet provided the required data.

The model was forced at the open boundary using water elevations measurements obtained in Nantucket Sound (described in section V.2.2). The measured time series consists of all physical processes affecting variations of water level: tides, winds, and other non-tidal oscillations of the sea surface. The rise and fall of the tide in Nantucket Sound is the primary driving force for estuarine circulation. Dynamic (time-varying) model simulations specified a new water surface elevation at the offshore boundaries every 10 minutes. The model specifies the water elevation at the offshore boundary, and uses the value to calculate water elevations at every nodal point within the system, adjusting each value according to solutions of the model equations. Changing water levels in Nantucket Sound produce variations in surface slopes within the estuary; these slopes drive water either into the system (if water is higher offshore) or out of the system (if water levels fall in the pond).

Figure V-8.   Plot of hydrodynamic model grid mesh for the Swan Pond River estuarine system. The model seaward boundary was specified with forcing functions consisting of water elevation measurements obtained in Nantucket Sound.

V.3.2.3 Calibration

After developing the finite element grid and specifying boundary conditions, the model was calibrated. Calibration ensured the model predicts accurately what was observed during the field measurement program. Numerous model simulations were required to calibrate the model, with each run varying specific parameters such as friction coefficients, turbulent exchange coefficients, fresh water inflow, and subtle modifications to the system bathymetry to achieve a best fit to the data.

Calibration of the hydrodynamic model required a close match between the modeled and measured tides in each of 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, a seven lunar-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 July 17, 2005 at 1900 EDT. This representative time period included the transition from neap to spring tide conditions, where the tide range and tidal currents are greatest, and model numerical stability is often most sensitive. To provide average tidal forcing conditions for model verification and the flushing analysis, a separate time period was chosen that spanned the transition between spring and neap tide ranges (bi-weekly maximum and minimum tidal ranges, respectively).

The calibrated model was used to analyze system 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 using 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 coefficient values of 0.025 were specified for all element material types. This values corresponds to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) (Henderson, 1966).

During calibration, friction coefficients were incrementally changed throughout the model domain. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within the estuary system. 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-5.

Table V-5.	Manning's Roughness coefficients used in simulations of modeled sub-embayments. These embayment delineations correspond to the material type areas shown in Figure V-9.
System Embayment	**Bottom Friction**
Nantucket Sound	0.031
Marsh Plain - lower	0.055
Inlet	0.036
Swan Pond River - upper	0.028
Swan Pond River - middle	0.032
Swan Pond River – lower	0.035
Swan Pond	0.028
Lower County Road Bridge	0.050
Route 28 Bridge	0.045
Upper County Road Bridge	0.034
Marsh Plain - upper	0.047
Marsh Plain - middle	0.052
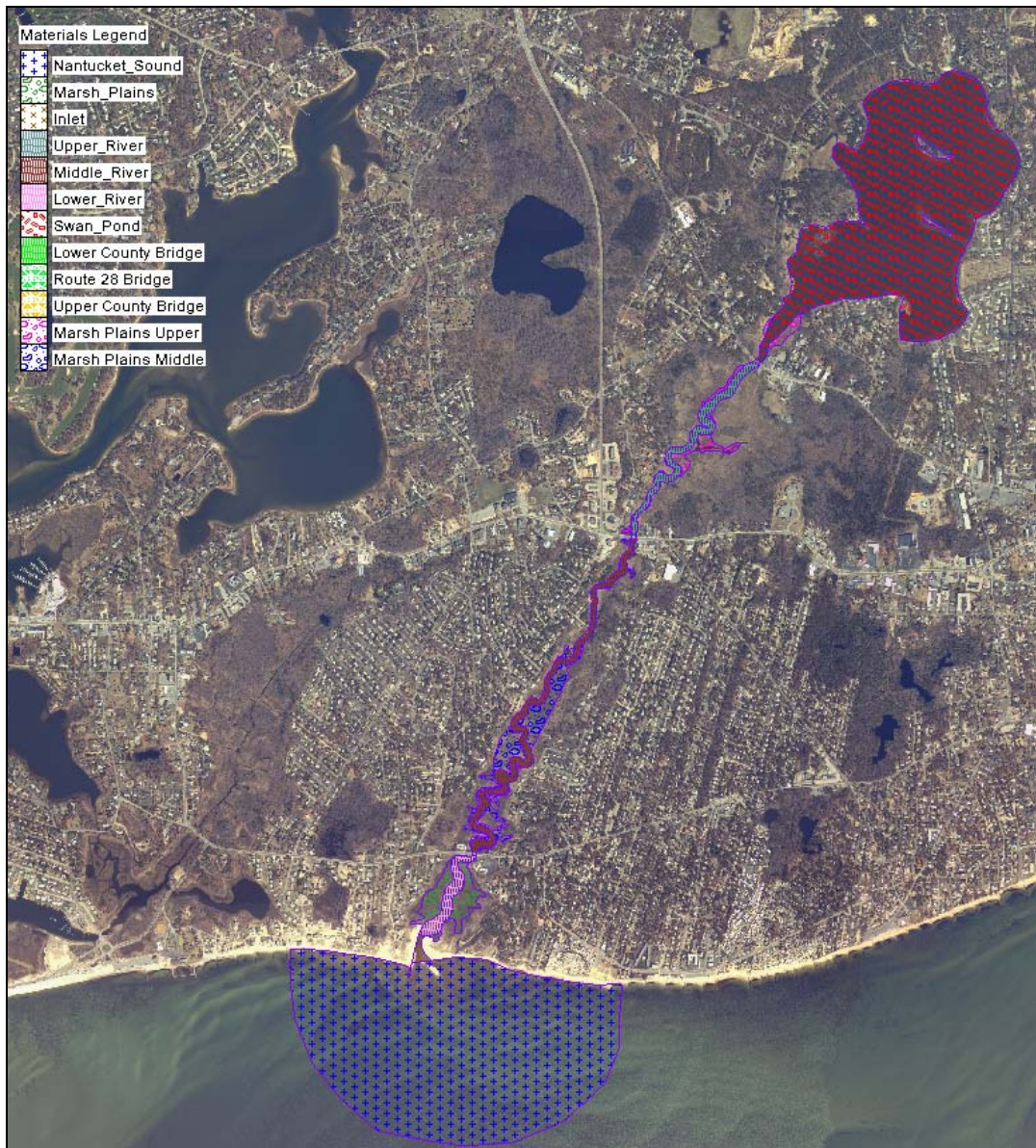
Figure V-9. Map of Swan Pond River hydrodynamic model friction coefficients. Color patterns designate the different areas used to vary model friction coefficient values.

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 swift, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). The model was mildly sensitive to turbulent exchange coefficients, with areas of marsh plain being most sensitive. In other regions where the flow gradients were not as strong, the model was much less sensitive to changes in the turbulent exchange coefficients. Typically, model turbulence coefficients (D) are set between 50 and 100 lb-sec/ft^2 (as listed in Table V-6). Higher values (up to 500 lb-sec/ft^2) are used on the marsh plain, to ensure solution stability.

Table V-6.	Turbulence exchange coefficients (D) used in simulations of modeled embayment system. These embayment delineations correspond to the material type areas shown in Figure V-9.
Embayment	D (lb-sec/ft^2)
Nantucket Sound	40.0
Marsh Plain - lower	150.0
Inlet	35.0
Swan Pond River - upper	25.0
Swan Pond River - middle	35.0
Swan Pond River – lower	35.0
Swan Pond	25.0
Lower County Road Bridge	35.0
Route 28 Bridge	35.0
Upper County Road Bridge	30.0
Marsh Plain - upper	150.0
Marsh Plain - middle	150.0

V.3.2.3.3 Marsh porosity processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain regions included in the model of the Swan Pond River system. Cyclically wet/dry areas of the marsh will tend to store waters as the tide begins to ebb and then slowly release water as the water level drops within the creeks and channels. This store-and-release characteristic of these marsh regions was partially responsible for the distortion of the tidal signal, and the elongation of the ebb phase of the tide. On the flood phase, water rises within the channels and creeks initially until water surface elevation reaches the marsh plain, when at this point the water level remains nearly constant as water 'fans' out over the marsh surface. The rapid flooding of the marsh surface corresponds to a flattening out of the tide curve approaching high water. Marsh porosity is a feature of the RMA-2 model that permits the modeling of hydrodynamics in marshes. This model feature essentially simulates the store-and-release capability of the marsh plain by allowing grid elements to transition gradually between wet and dry states.   This technique allows RMA-2 to change the ability of an element to hold water, like squeezing a sponge.

V.3.2.3.4 Comparison of modeled tides and measured tide data

Several calibration model runs were performed to determine how changes to various parameters (e.g. friction and turbulent exchange coefficients) affected the model results. These trial runs achieved excellent agreement between the model simulations and the field data. Comparison plots of modeled versus measured water levels at the six gauge locations within Swan Pond River and the offshore gauge are presented in Figures V-10 through V-16.
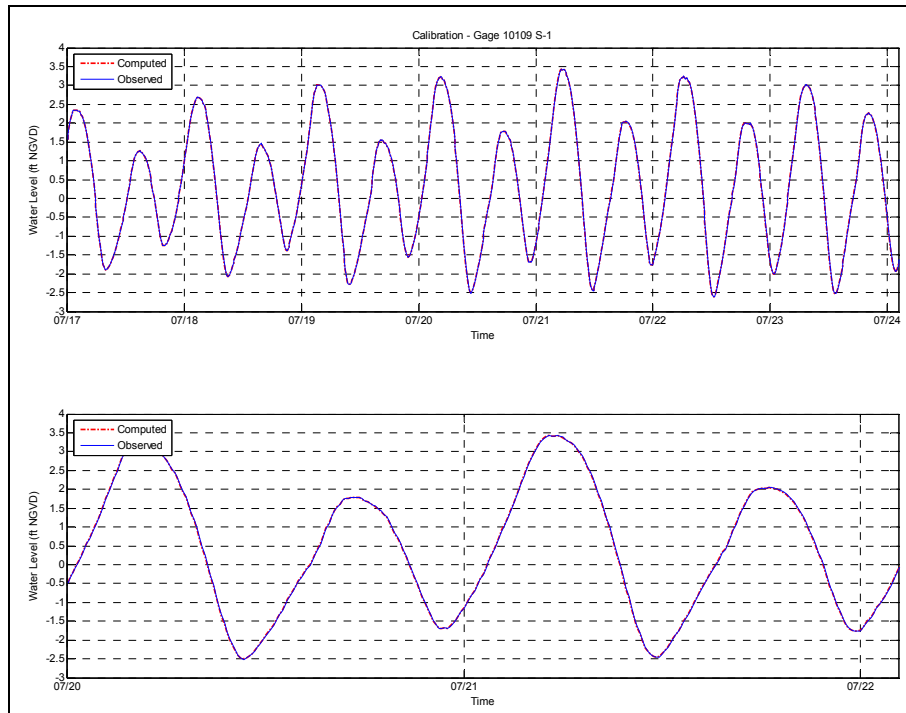
Figure V-10.   Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the offshore gauging station. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.
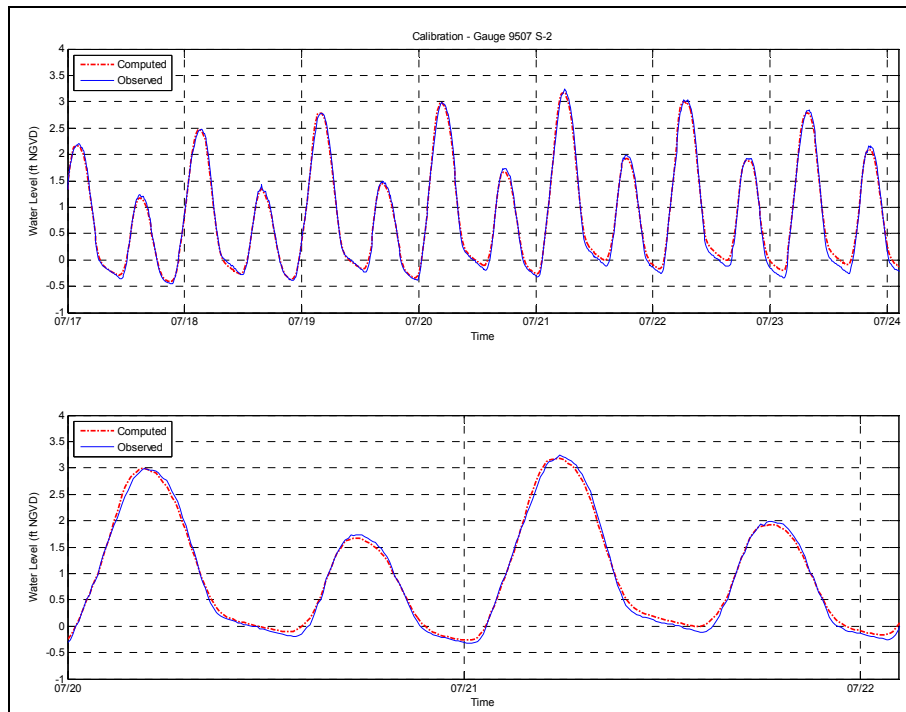
Figure V-11.   Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the gauging station south of Lower County Road. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.
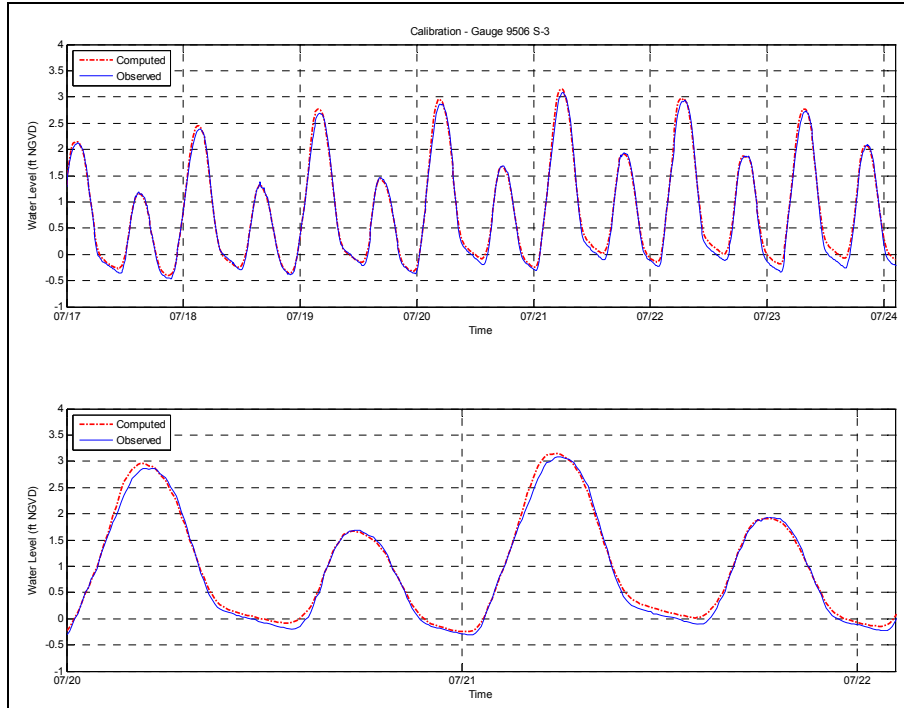
Figure V-12.   Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the gauging station north of Lower County Road. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.
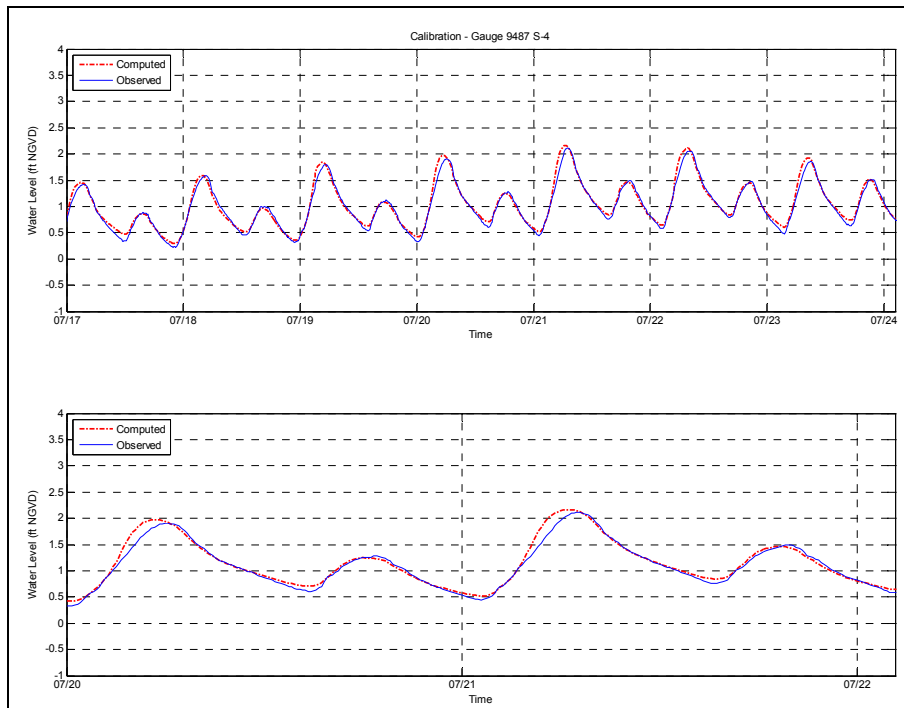
Figure V-13.   Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the gauging station south of Route 28. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.
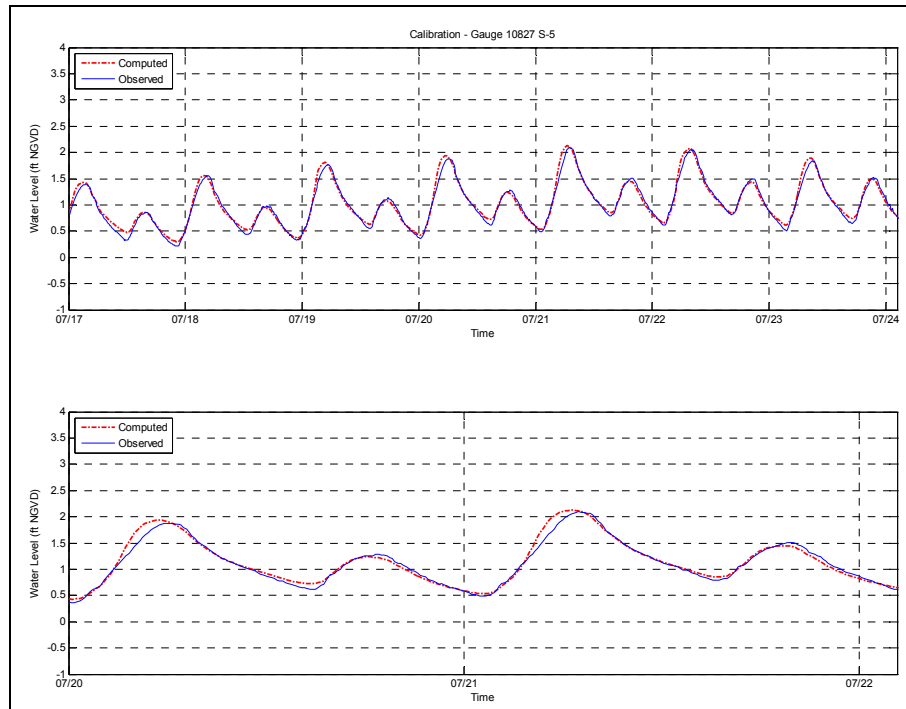
Figure V-14. Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the gauging station north of Route 28. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.
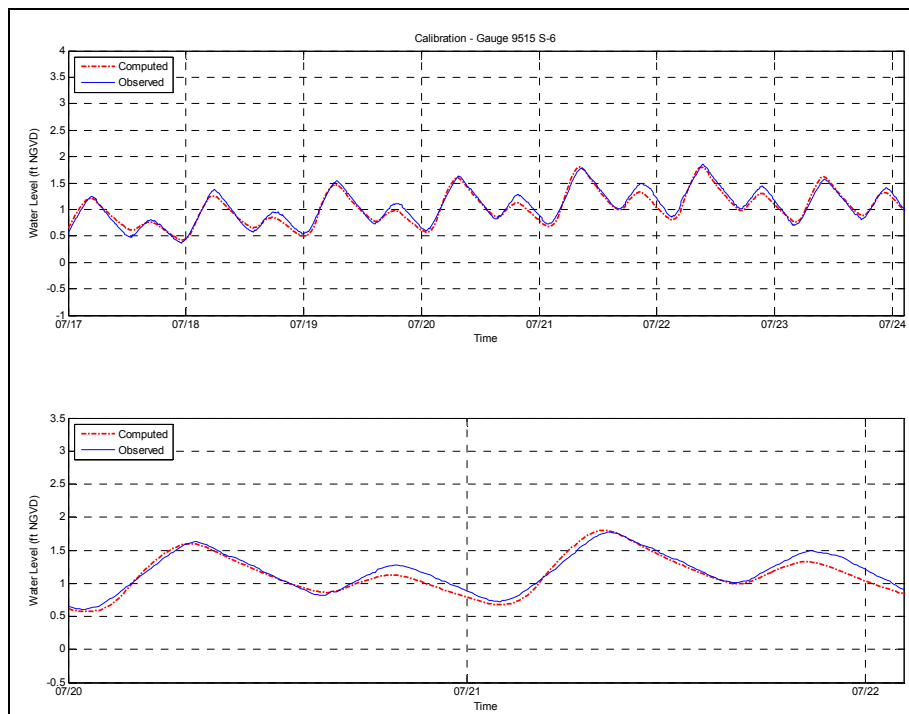
Figure V-15. Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the gauging station south of Upper County Road. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.
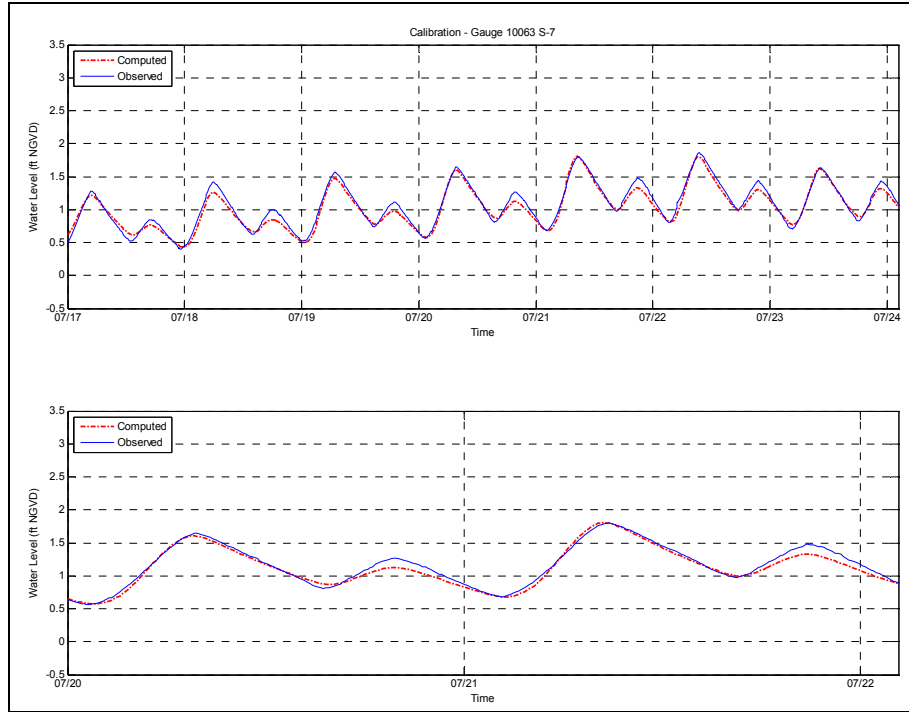
Figure V-16.   Comparison of water surface variations simulated by the model (dashed line) to those measured within the system (solid line) for the calibration time period, for the Swan Pond gauging station. The bottom plot is a 48-hour sub-section of the total modeled time period, shown in the top plot.

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 (principle lunar semidiurnal constituent) was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled system. 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-7 for the calibration period differ from those in Table V-2 because constituents were computed for only the seven-day section of the 30-days represented in Table V-2. Table V-7 compares tidal constituent amplitude (height) and relative phase (time) for modeled and measured tides at the TDR locations. The constituent phase shows the relative timing of each separate constituent at a particular location, and also the change (or phase lag) in timing of a single constituent at different locations in an estuary.

Table V-7. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for the Swan Pond River system, during modeled calibration time period.

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (rad)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound	2.103	0.180	0.114	0.653	1.27	-1.75
Lower County South	1.213	0.214	0.042	0.472	1.41	2.72
Lower County North	1.194	0.210	0.041	0.466	1.42	2.73
Route 28 - South	0.439	0.103	0.012	0.268	1.91	3.02
Route 28 - North	0.424	0.097	0.012	0.262	1.94	3.04
Upper County South	0.263	0.035	0.003	0.211	2.70	-2.23
Swan Pond	0.261	0.040	0.003	0.208	2.94	-1.78
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (rad)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound	2.111	0.186	0.120	0.659	1.19	-1.98
Lower County South	1.267	0.235	0.052	0.483	1.35	2.70
Lower County North	1.227	0.223	0.050	0.465	1.36	2.73
Route 28 - South	0.455	0.085	0.008	0.252	1.95	3.11
Route 28 - North	0.442	0.081	0.008	0.241	1.97	3.12
Upper County South	0.295	0.027	0.010	0.164	2.72	-2.14
Swan Pond	0.306	0.039	0.007	0.187	2.86	-1.91
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound	0.008	0.006	0.006	0.005	-9.5	-13.9
Lower County South	0.054	0.021	0.009	0.011	-7.8	-1.7
Lower County North	0.033	0.013	0.009	-0.001	-7.1	0.1
Route 28 - South	0.016	-0.018	-0.004	-0.015	4.5	5.3
Route 28 - North	0.018	-0.016	-0.004	-0.021	3.4	4.6
Upper County South	0.032	-0.007	0.007	-0.047	2.2	5.0
Swan Pond	0.045	-0.002	0.004	-0.021	-10.3	-8.0

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, which is better than the order of accuracy of the tide gauges ($\pm$0.12 ft). Time lag errors were typically less than the time increment resolved by the model (1/6 hours or 10 minutes), indicating good agreement between the model and data.

V.3.2.4 ADCP verification of the Swan Pond River system

An additional model verification check was possible by using collected ADCP velocity data to verify the performance of the model in representing the system dynamics. Computed flow

rates from the model were compared to flow rates determined using the measured velocity data. The ADCP data survey efforts are described in Section V.2.3. For the model ADCP verification, the hydrodynamic model was run over a time period that included the ADCP survey on July 26, 2005.

The verification model period was performed for an approximate eight-day period, beginning 0400 hours EDT July 24, 2005 and ending 0400 EDT August 1, 2005. This time period included a 24-hour model spin-up period, and a tide cycle period used to compare to the ADCP data. Model flow rates were computed in RMA-2 at a continuity line (channel cross-section) that correspond to the actual ADCP transect followed in the survey across the inlet to Swan Pond River.

A comparison of the measured and modeled volume flow rates in the across the inlet to Swan Pond River is shown in Figure V-17. The top plot shows the flow comparison, and the lower plot shows the time series of tide elevation for the same period. Each ADCP point (blue triangles shown on the plots) is a summation of flow measured along the ADCP transect. The 'bumps' and 'skips' of the flow rate curve (more evident in the model output) can be attributed to the effects of winds (i.e., atmospheric effects) on the water surface and friction across the seabed periodically retarding or accelerating the flow through the inlets, and inside the system channels. If water surface elevations changed smoothly as a sinusoid, the volume flow rate would also appear as a smooth curve. However, since the rate at which water surface elevations change does not vary smoothly, the flow rate curve is expected to show short-period fluctuations.

Data comparisons at all five ADCP transect show exceptionally good agreement with the model predictions. The calibrated model accurately describes the discharge magnitude at both lines. For both transects the R^2 correlation coefficients between data and model results are equal or greater than 0.97. The RMS error computed from each transect is less than 42.2 ft3/sec, which is 9.3% of the maximum measured discharge rate. Correlation statistics between the modeled and measured flows for each ADCP transect are presented in Table V-8.
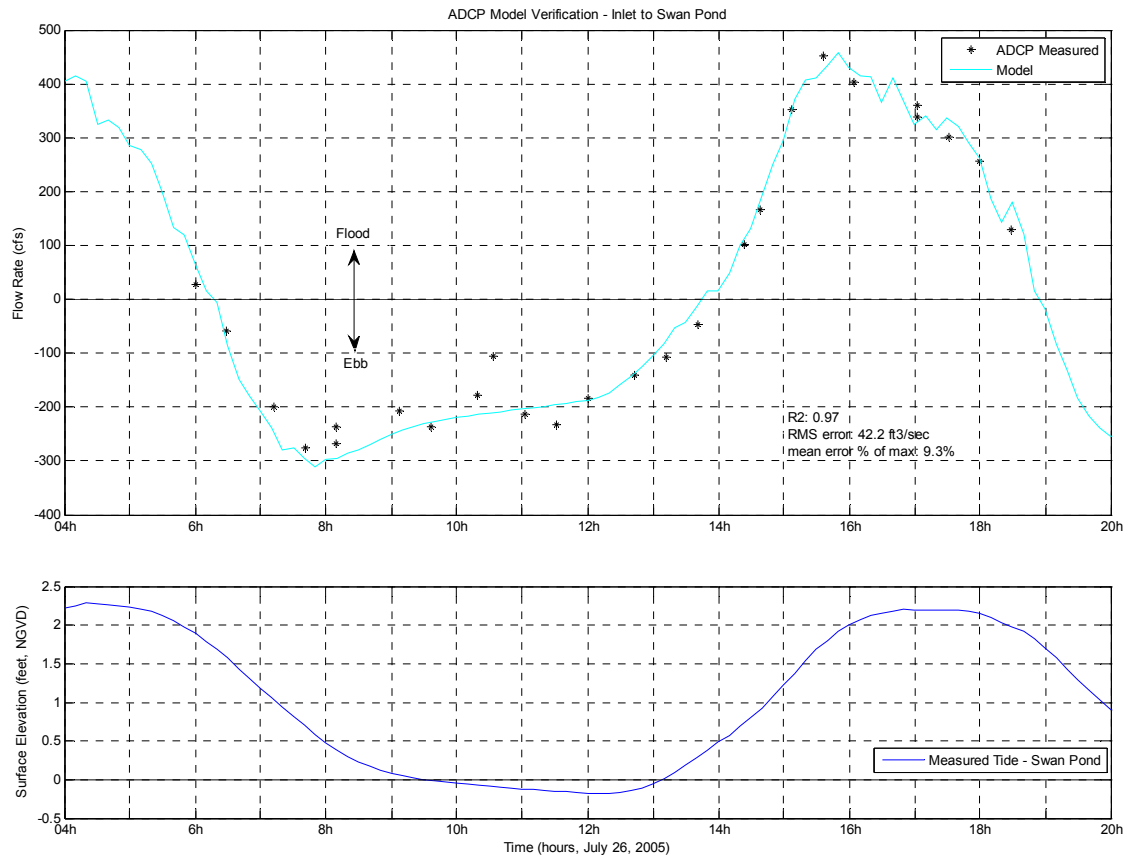
Figure V-17. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the inlet to Swan Pond River, over a tidal cycle on July 26, 2005. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation within the inlet. (R^2 =0.97, E_{RMS}=42.2 ft^3/sec).

Table V-8.	Correlation statistics between modeled and measured total flow rates at the ADCP transects used in the model verification of the Swan Pond River model.			
Transect	R^2 correlation	RMS error (ft^3/sec)	Max Error (ft^3/sec)	Min Error (ft^3/sec)
Inlet	0.97	42.2	106.1	1.5

V.3.2.6 Model Circulation Characteristics

The final calibrated model serves as a useful tool in investigating the circulation characteristics of the Swan Pond River system. Using model inputs of bathymetry and tide data, current velocities and flow rates can be determined at any point in 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.

From the model run of the system, maximum ebb velocities in the inlet channel are slightly larger than velocities during maximum flood. However, across the rest of the system in flood velocities are slightly larger than the ebb velocities. Maximum depth-averaged flood velocities in the model are approximately 1.6 feet/sec at the narrows near the Lower County Bridge, while

maximum ebb velocities are about 1.0 feet/sec. Close-up views of model output are presented in Figure V-18 and V-19, which show contours of velocity magnitude along with velocity vectors that indicate flow direction, each for a single model time-step, at the portion of the tide where maximum ebb velocities occur (in Figure V-18), and for maximum flood velocities in Figure V-19.

In addition to depth-averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs at the two system inlets is seen in the plot of flow rates in Figure V-20. Maximum flow rates are roughly equal during flood and ebb tides. At the inlet, the modeled maximum flow rate during spring tides is 9,100 ft^3/sec
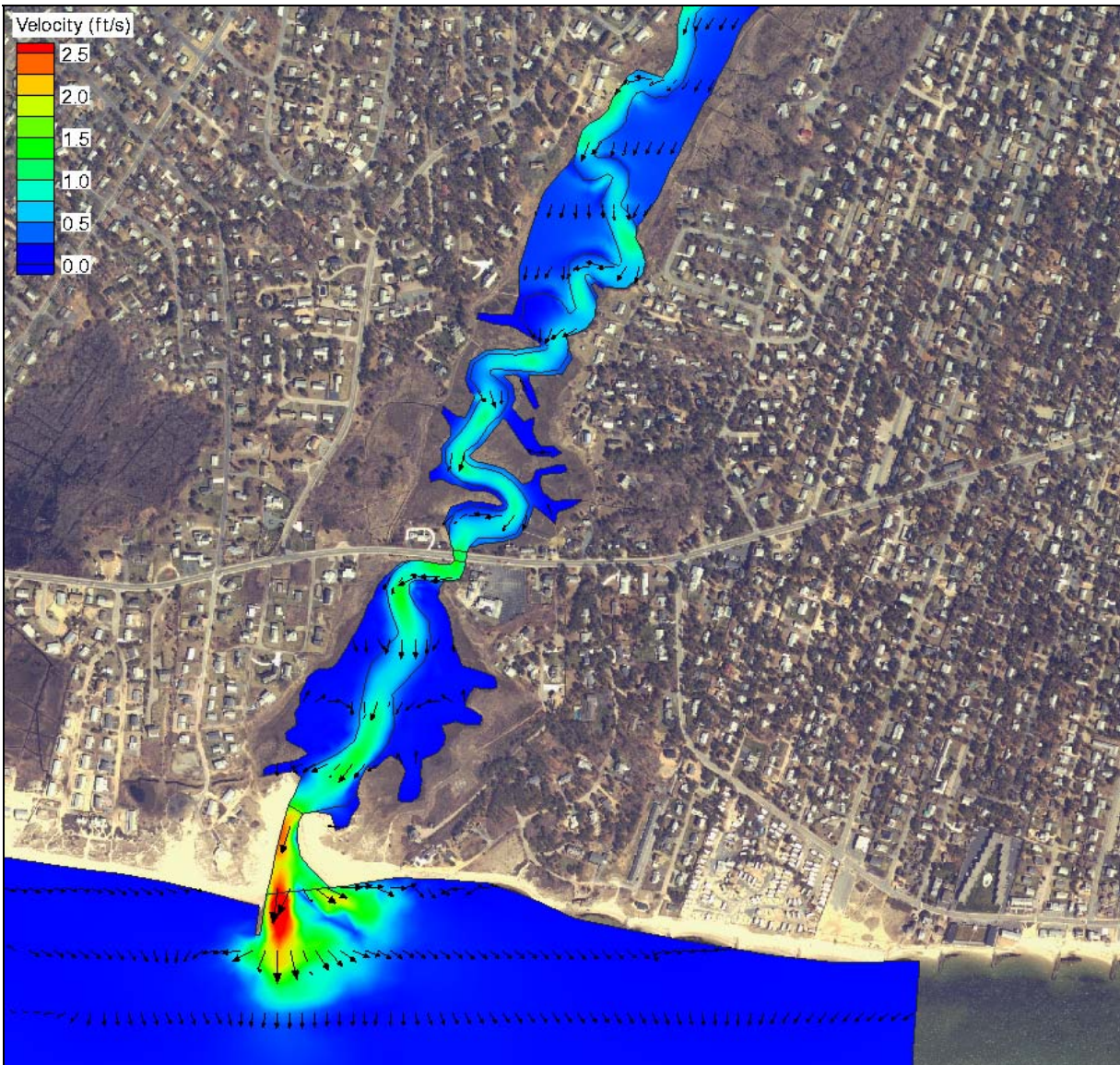
Figure V-18.   Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.
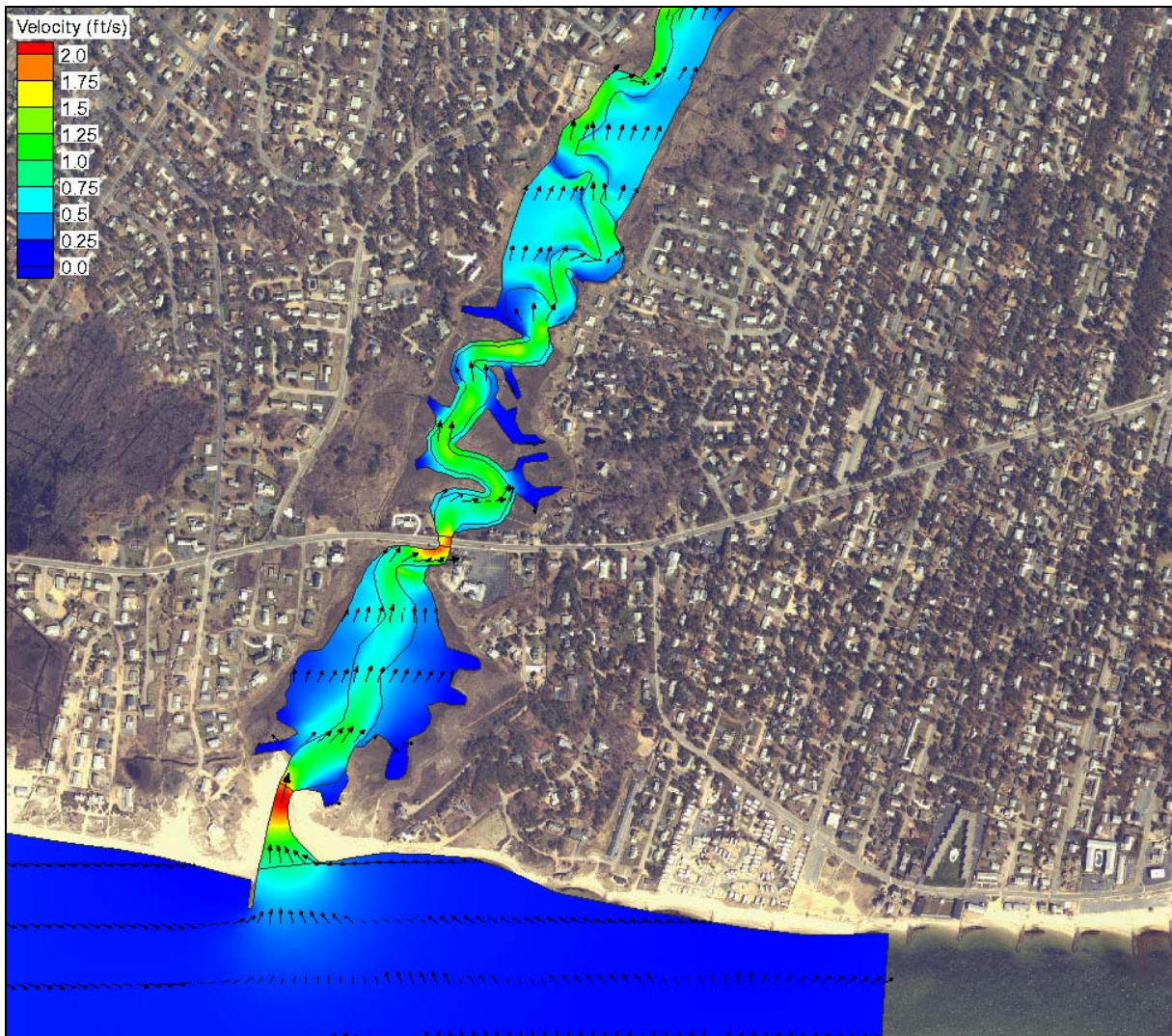
Figure V-19.   Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle.   Color contours indicate velocity magnitude, and vectors indicate the direction of flow.
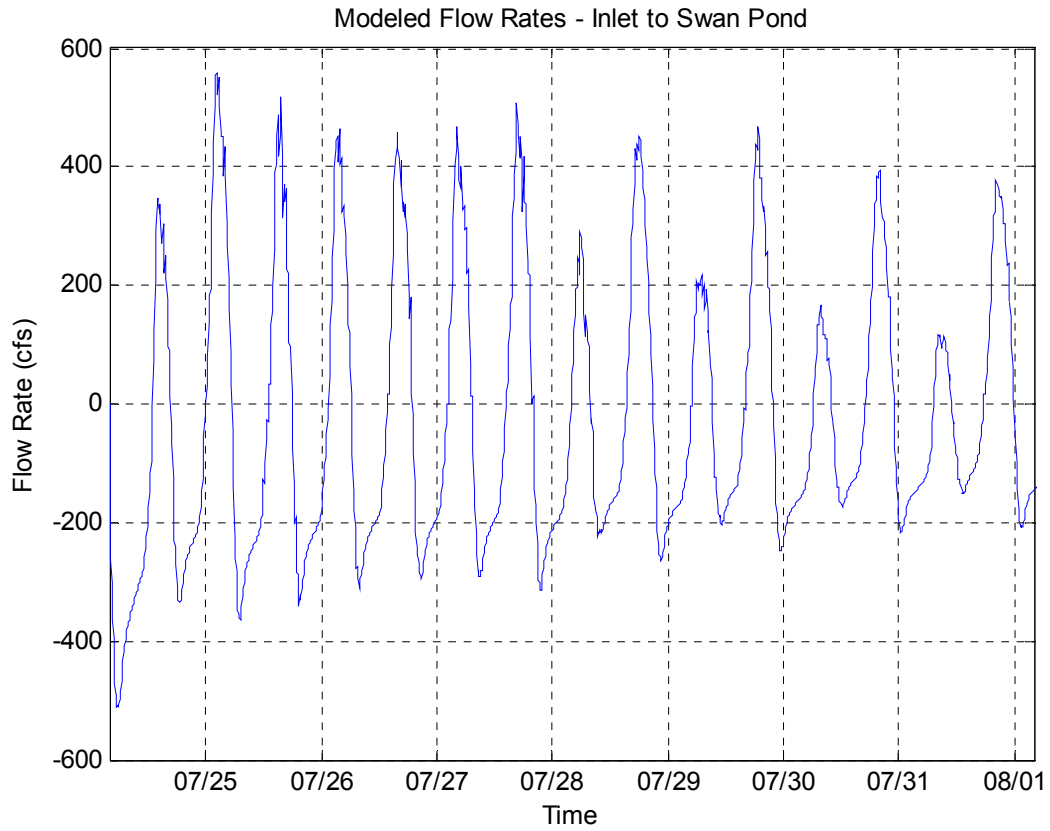
Figure V-20.   Time variation of computed flow rates at the inlet to Swan Pond River. Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

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 Swan Pond River System. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayment

Extensive field measurements and hydrodynamic modeling of the embayment were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated model output representing the transport of water within the system embayment. Files of node locations and node connectivity for the RMA-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 an 11-tidal cycle period in July 2005. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model 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 Swan Pond River System, consisting of the background concentrations of total nitrogen in the waters entering from Nantucket Sound. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Six years of data (collected between 2005 and 2010) were available for stations monitored by SMAST in the Swan Pond River System.

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 Swan Pond River 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 Swan Pond River System. Like RMA-2 numerical code, RMA-4 is a two-dimensional depth averaged finite element model capable of simulating time-dependent

Table VI-1.	Town of Dennis water quality monitoring data, and modeled Nitrogen concentrations for the Swan Pond River System used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.												
Sub-Embayment	Monitoring station	2005 mean	2006 mean	2007 mean	2008 mean	2009 mean	2010 mean	mean	s.d. all data	N	model min	model max	model average
Swan Pond River	SWP-1	0.494	0.640	0.540	0.490	0.449	0.712	0.556	0.134	38	0.315	1.050	0.465
Swan Pond River	SWP -2	0.581	0.825	0.607	0.567	0.688	0.773	0.673	0.188	30	0.325	1.175	0.662
Swan Pond River	SWP -3	0.803	1.015	0.781	0.851	0.875	0.841	0.862	0.282	30	0.335	1.206	0.827
Lower Swan Pond	SWP -5	0.963	1.135	1.133	0.954	0.979	0.971	1.036	0.279	29	0.570	1.216	1.056
Lower Swan Pond	SWP -6	0.933	0.971	1.433	0.879	1.268	1.181	1.123	0.331	26	1.090	1.196	1.143
Upper Swan Pond	SWP -7	1.039	1.141	1.412	1.141	1.260	1.178	1.197	0.395	59	1.112	1.223	1.168
Upper Swan Pond	SWP -8	1.098	1.113	1.199	1.547	1.063	1.163	1.159	0.336	28	1.146	1.262	1.207

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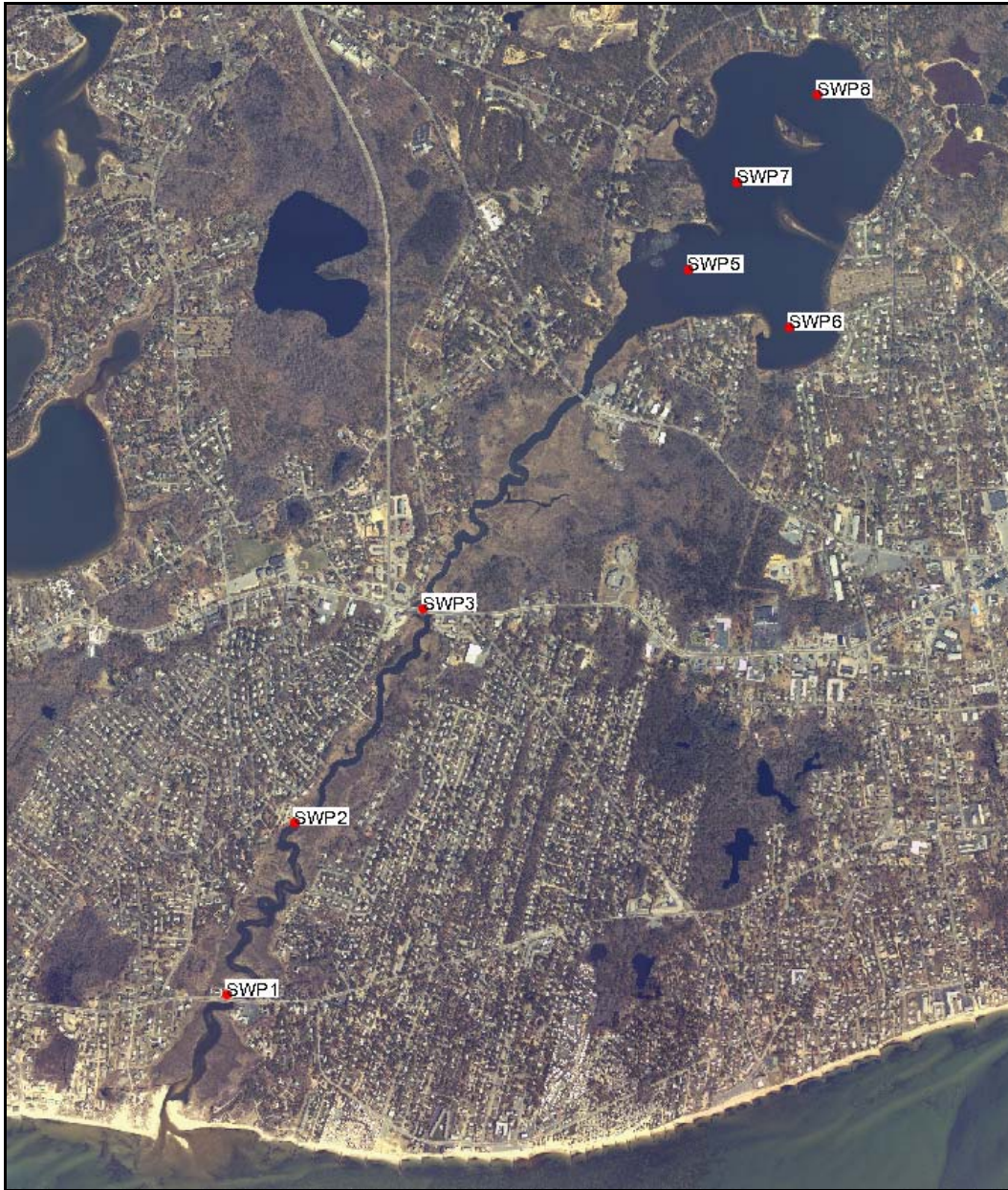
Figure VI-1. Estuarine water quality monitoring station locations in the Swan Pond River System. Station labels correspond to those provided in Table VI-1.

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 Swan Pond River 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 Swan Pond River System was used for the water quality constituent modeling portion of this study.

Based on groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include ground water flowing into the system from the watersheds. Swan Pond has three watersheds contributing to the groundwater flow, the combined flow rate into the system is 2.92 ft³/sec (7,140 m³/day), upper Swan Pond River watersheds have a groundwater flow rate of 1.42 ft³/sec (3,489 m³/day), and lower Swan Pond River watershed has a groundwater flow rate

of 1.51 ft^3/sec (3,700 m^3/day).

For the model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (28 day) spin-up period. At the end of the spin-up period, the model was run for an additional 5 tidal-day (125 hour) period. Model results were recorded only after the initial spin-up period. The time step used for the water quality computations was 10 minutes, which corresponds to the time step of the hydrodynamics input for the Swan Pond River 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 each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed direct atmospheric deposition load for Swan Pond was evenly distributed at grid cells that formed the perimeter of the embayment. Benthic regeneration load was distributed among another sub-set of grid cells which are in the interior portion of each basin.

The loadings used to model present conditions in Swan Pond River 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 each sub-embayment (excluding marsh coverage, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, the benthic flux is generally low. With two northern portion of Swan Pond River being positive and the two regions having a negative benthic flux (Swan Pond and lower Swan Pond River).

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 Nantucket Sound was set at 0.305 mg/L, based on SMAST data from the Nantucket Sound. Swan Pond River Inlet is adjacent to Bass River with a similar boundary condition. All boundary conditions considered for the modeling are within 0.42 standard deviations of Swan Pond River boundary of 0.305 mg/L. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Nantucket Sound.

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 Swan Pond River (pond and marsh) 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.

Table VI-2. Sub-embayment loads used for total nitrogen modeling of the Swan Pond River System, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions.**			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Swan Pond	18.123	1.885	-3.473
Swan Pond River - North	10.036	0.104	0.385
Swan Pond River - South	15.556	0.226	-1.346

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Swan Pond River System.	
Embayment Division	E m^2/sec
Nantucket Sound	2.6
Marsh Plain - lower	1.6
Inlet	2.3
Swan Pond River - upper	2.2
Swan Pond River - middle	2.4
Swan Pond River – lower	2.4
Swan Pond	1.4
Lower County Road Bridge	2.3
Route 28 Bridge	2.3
Upper County Road Bridge	1.5
Marsh Plain - upper	1.4
Marsh Plain - middle	1.4
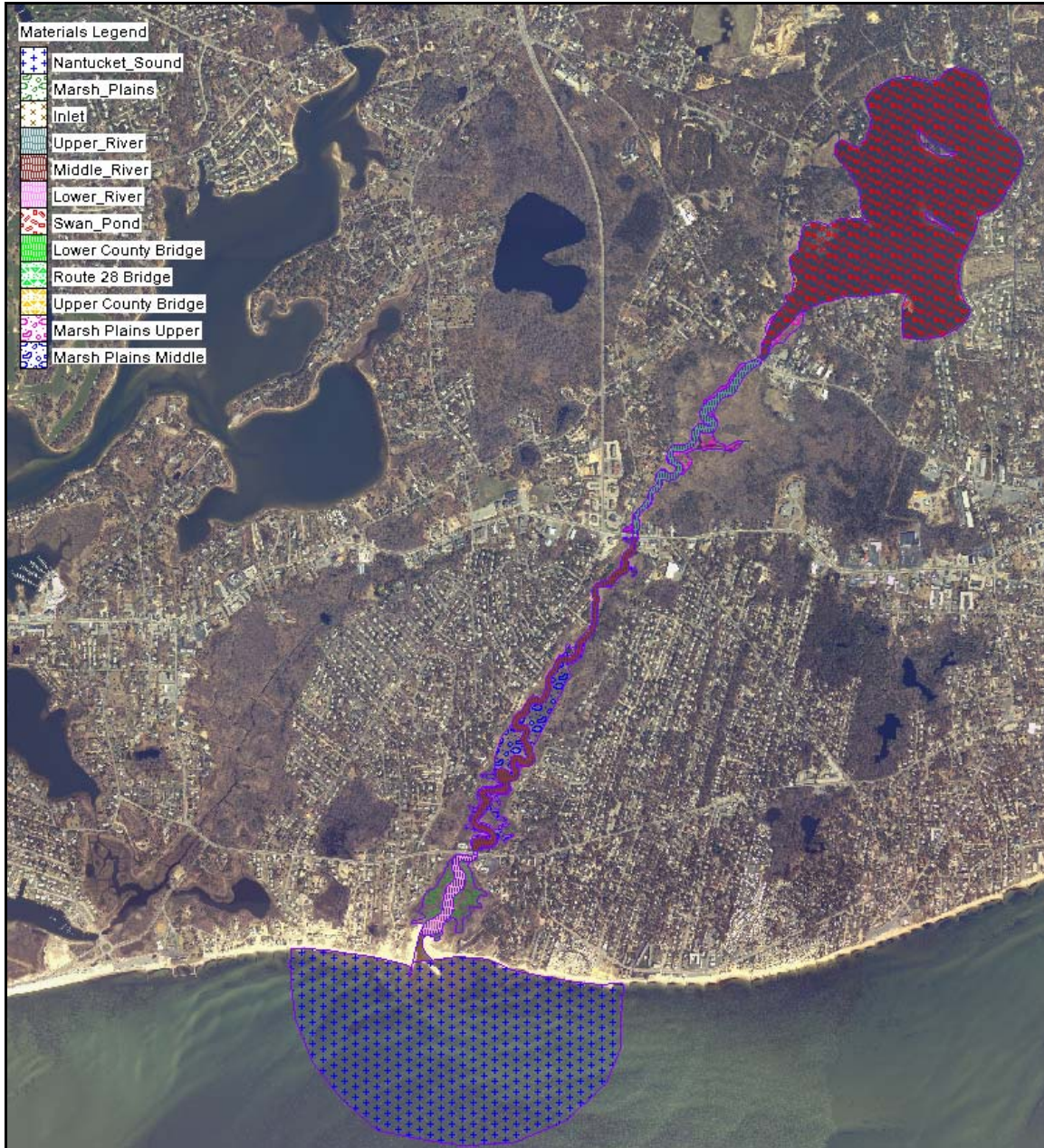
Figure VI-2.   Map of Swan Pond River water quality model longitudinal dispersion coefficients. Color patterns designate the different areas used to vary model dispersion coefficient values.

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.

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.
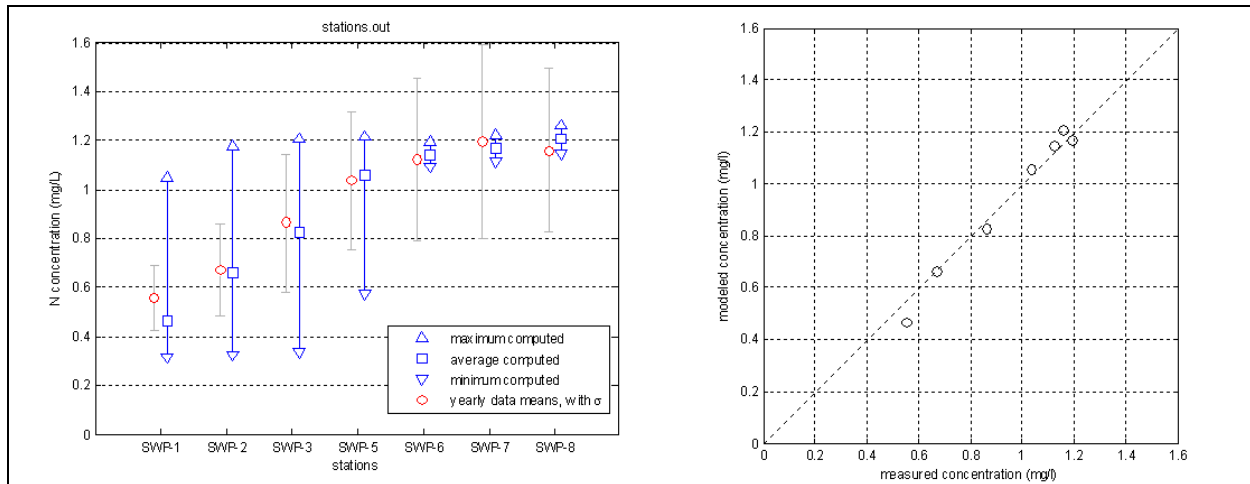
Figure VI-3.   Comparison of measured total nitrogen concentrations and calibrated model output at stations in Swan Pond River System. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed 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 Swan Pond River System, with rms error of 0.04 mg/L and an R^2 correlation coefficient of 0.96.

A contour plot of calibrated model output is shown in Figure VI-4 for Swan Pond River 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 Swan Pond River 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 30.9 ppt. For groundwater inputs salinities were set at 0 ppt. The total groundwater input used for the model was 6.01 ft^3/sec (14,711 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 Swan Pond River System. The approximate location of the sentinel threshold station for Swan Pond River System (SWP-2) is shown.

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 Swan Pond River System. The rms error of the models was 1.37 ppt, and correlation coefficient was 0.72. The salinity data from 2005 to 2007 was noticeably fresher than from the period 2007 to 2010. On average the salinity is 1.2 ppt higher in the later period, which confirms the trends shown within the model. If the salinity data from 2007 to 2010 was utilized for the model comparison, the rms error of the models would be 0.51 ppt, with a correlation coefficient of 0.95. The salinity differences between the 2005-2007 and 2007-2010 periods likely involves a variety of factors, i.e. metrological differences between the period, correlation between rainfall events and sampling, shoaling within the inlet, etc. The entire dataset was used to verify the model. 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.
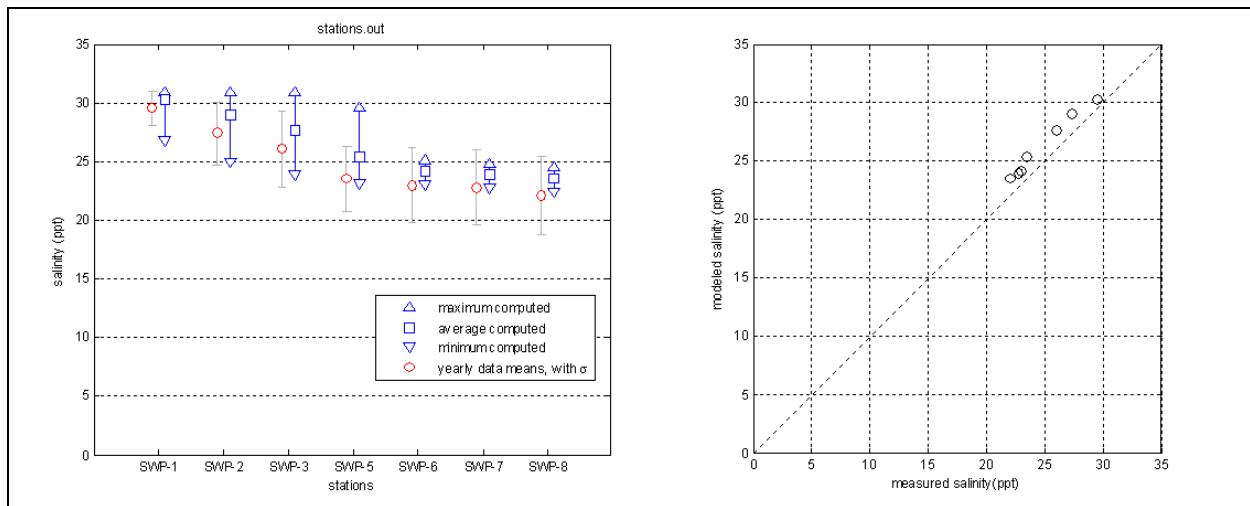
Figure VI-5.   Comparison of measured and calibrated model output at stations in Swan Pond River System. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (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.
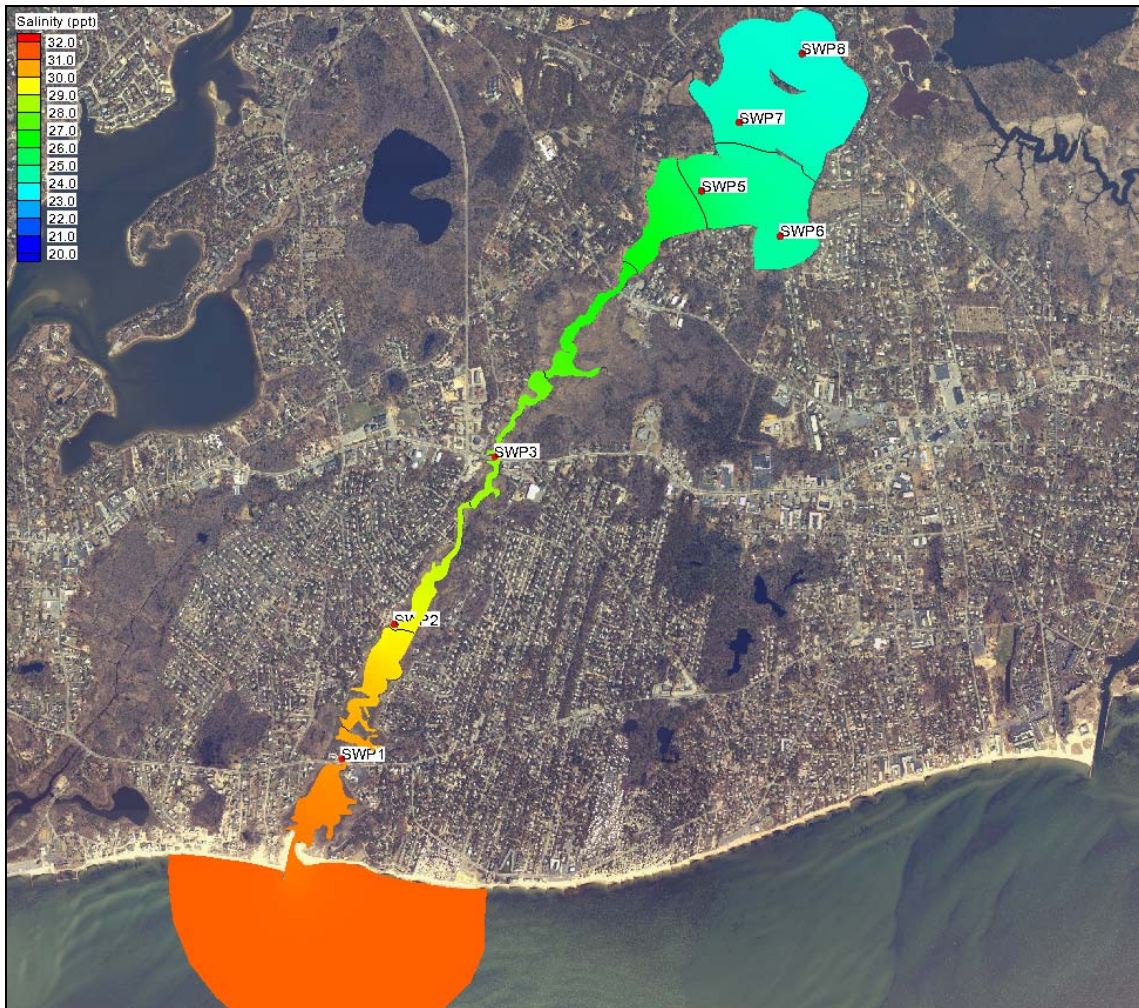
Figure VI-6.   Contour plots of modeled salinity (ppt) in Swan Pond River System.

Table VI-4.   Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Swan Pond River 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
Swan Pond	18.123	22.600	+24.7%	0.863	-95.2%
Swan P R N	10.036	11.419	+13.8%	0.268	-97.3%
Swan P R S	15.556	15.822	+1.7%	0.463	-97.0%

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 Swan Pond River as a result of potential future development. Specific watershed areas would experience large load increases, for example the loads to Swan Pond would increase 25% 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 significantly lower than existing conditions by over 96% overall.

For the build-out scenario, a breakdown of the total nitrogen load entering the Swan Pond River 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:

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

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 Swan Pond River 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)
Swan Pond	22.600	1.885	-3.845
Swan Pond River - North	11.419	0.104	0.426
Swan Pond River - South	15.822	0.233	-1.473

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Swan Pond River System was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The stations in Swan Pond River show steady increase in nitrogen from the inlet to the head of the system. 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 Swan Pond River System. Sentinel threshold station is in bold print.			
Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Lower River	SWP-1	0.465	0.484	+4.0%
Lower River	**SWP-2**	**0.661**	**0.712**	**+7.6%**
Upper Swan River	SWP-3	0.826	0.907	+9.8%
Swan Pond	SWP-5	1.055	1.182	+12.1%
Swan Pond	SWP-6	1.143	1.291	+13.0%
Swan Pond	SWP-7	1.168	1.324	+13.4%
Swan Pond	SWP-8	1.210	1.378	+13.8%

Figure VI-7.   Contour plots of modeled total nitrogen concentrations (mg/L) in Swan Pond River System, for projected build-out loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Swan Pond River System (SWP-2) is shown.

VI.2.6.2 No Anthropogenic Load

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

Table VI-7.	"No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of Swan Pond River 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)
Swan Pond	0.863	1.885	-0.558
Swan Pond River - North	0.268	0.104	0.085
Swan Pond River - South	0.463	0.233	-0.380

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., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions ranging from 32% inside the inlet to Swan Pond with greater than 67% reduction in total nitrogen. 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 Swan Pond River System. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold station is in bold print.			
Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Lower River	SWP-1	0.465	0.314	-32.5%
Lower River	**SWP-2**	**0.661**	**0.329**	**-50.3%**
Upper Swan River	SWP-3	0.826	0.343	-58.4%
Swan Pond	SWP-5	1.055	0.366	-65.3%
Swan Pond	SWP-6	1.143	0.377	-67.0%
Swan Pond	SWP-7	1.168	0.381	-67.4%
Swan Pond	SWP-8	1.210	0.388	-68.0%
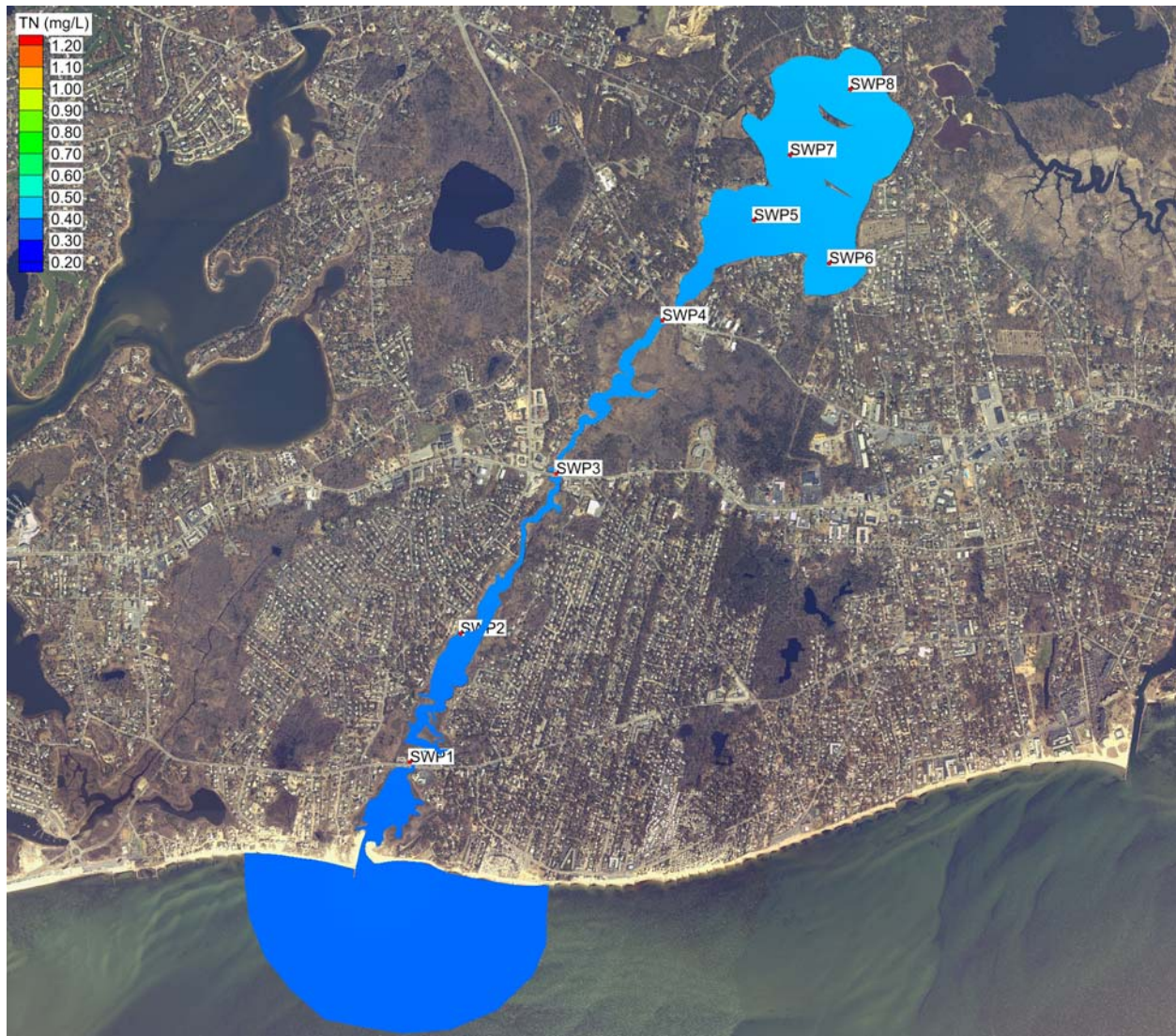
Figure VI-8.   Contour plots of modeled total nitrogen concentrations (mg/L) in Swan Pond River System, for no anthropogenic loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Swan Pond River System (SWP-2) is shown.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Swan Pond River Embayment System in the Town of Dennis, MA, our assessment is based upon data from the water quality monitoring database developed by the Dennis Water District and surveys of eelgrass distribution, benthic animal communities and sediment characteristics, and time-series measurements of dissolved oxygen and chlorophyll-a conducted during the summer and fall of 2005. These data form the basis of an assessment of this system's present nutrient-related habitat quality, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for this system (Chapter VIII). It should be noted that nitrogen enrichment occurs through two primary mechanisms, 1) high rates of nitrogen entering from the surrounding watershed and/or 2) low rates of flushing due to "restricted" tidal exchange with the low nitrogen waters of Nantucket Sound. The Swan Pond River Estuary has enhanced nitrogen loading from its watershed from shifting land-uses and Swan Pond has a low tidal exchange due to the long meandering channel of the Swan Pond River. Fundamentally, natural or man-made 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 that 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 concentrations (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 an autonomous dissolved oxygen sensor in the central region of Swan Pond such that it would be representative of the dissolved oxygen conditions within this shallow salt pond. The site was selected within the central region of the main basin, such that it was removed from the influence of inflowing waters from the Swan Pond River (Nantucket Sound) and near a long-term water quality monitoring station. The dissolved oxygen mooring was deployed to record the frequency and duration of low oxygen conditions and also supporting information on phytoplankton biomass (chlorophyll a), 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 Swan Pond River Embayment 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. Analysis of inorganic N/P molar ratios within the watercolumn of the major basins comprising the Swan Pond River Estuary support the contention that nitrogen is the nutrient to be managed, as the ratio in Swan Pond (~1) and Swan Pond River (~3) are far below the Redfield Ratio value (16) indicating that nitrogen additions will increase phytoplankton production in this system. Within the Swan Pond River System, temporal changes in eelgrass distribution provide evidence of nitrogen impacts, as eelgrass beds were identified in the 1951 MassDEP Survey, but not in the 1995, 2001, or 2006/7 surveys. As a result, nutrient threshold determination was based on results from the dissolved oxygen and chlorophyll mooring data, the eelgrass distribution as well as the benthic infaunal community characterization.

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 4 mg L-1. Massachusetts State Water Quality Classification indicates that SA (high quality) waters be able to maintain oxygen levels above 6 mg L-1 (314 CMR 4). The tidal waters of the Swan Pond River 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 concentrations (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, an autonomously recording oxygen sensor was moored 30 cm above the embayment bottom within the central region of Swan Pond, the main open water basin within the Swan Pond River System (Figure VII-2). The dissolved oxygen sensor (YSI 6600) was first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during 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 Swan Pond system was collected during the summer of 2005. These data are supplemented by the traditional "grab" sampling data from the estuary-wide water quality monitoring program, which collected samples on ~30 dates during summers, 2005-2010.
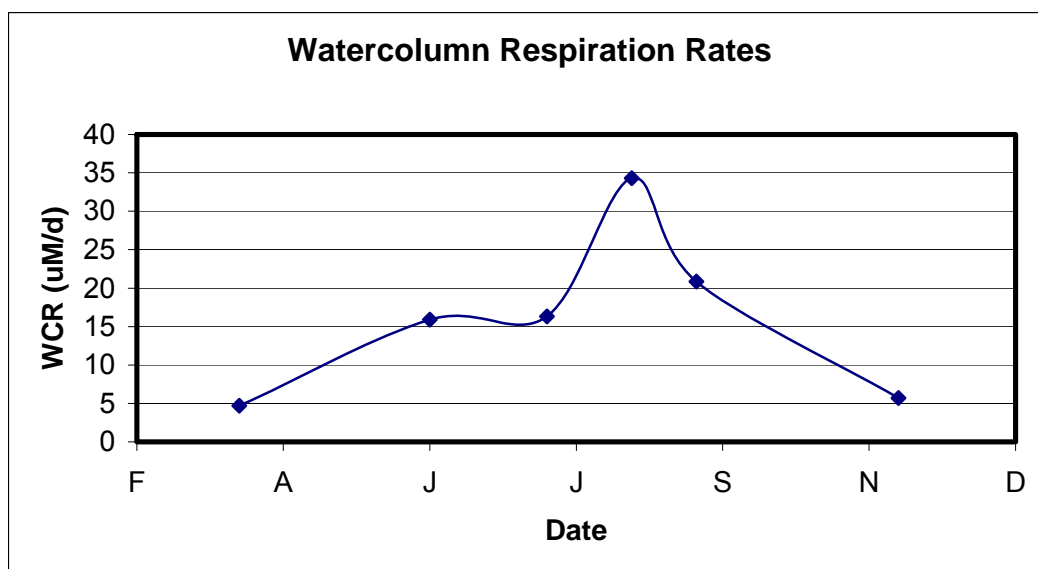
Figure VII-1.   Example of typical average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System, Cape Cod (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Similar to other embayments in southeastern Massachusetts, the Swan Pond River System 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 in Swan Pond, 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 26 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both

the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.
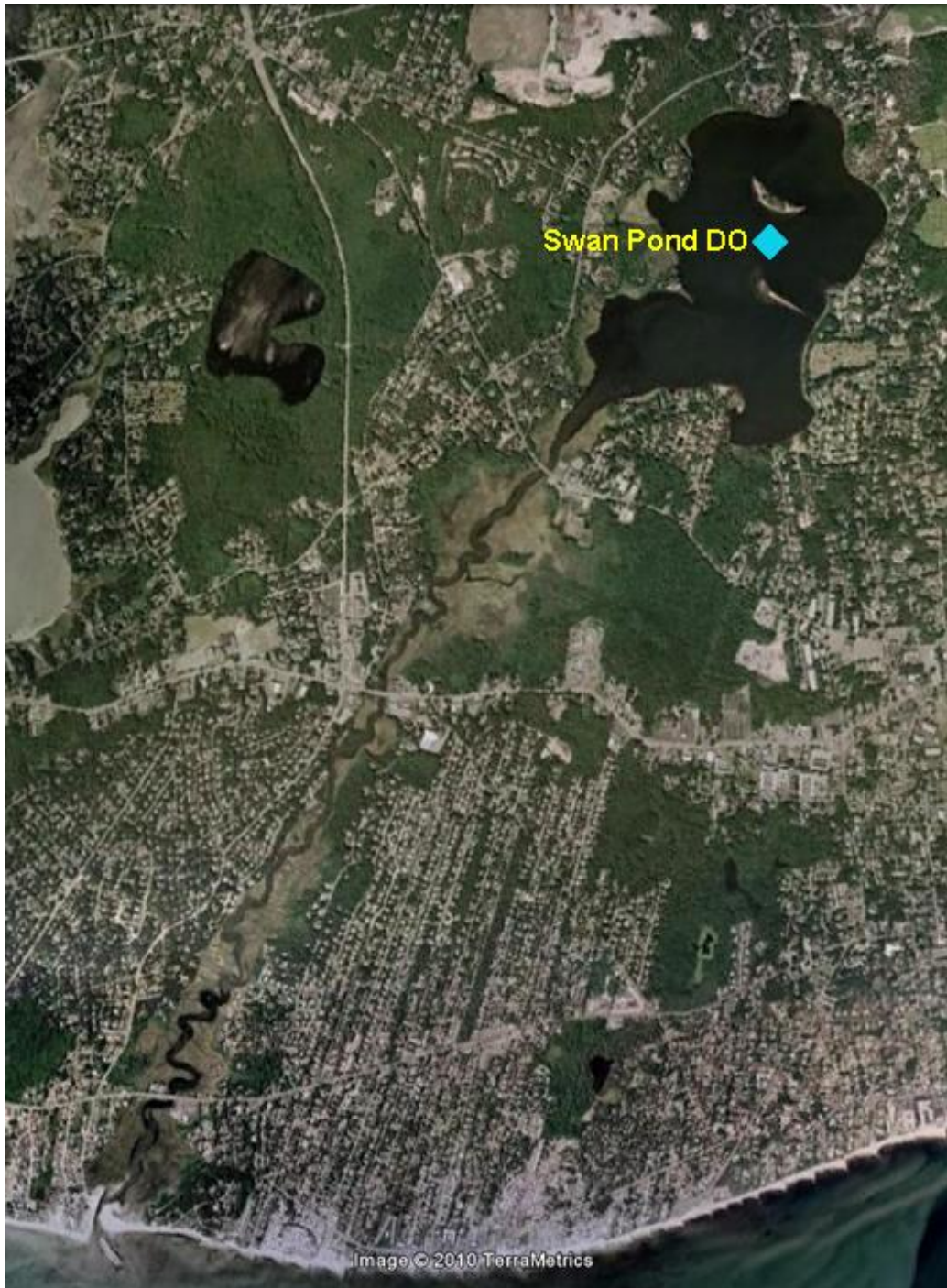
Figure VII-2.   Aerial Photograph of the Swan Pond River system in the Town of Dennis showing the location of the continuously recording Dissolved Oxygen / Chlorophyll-a sensors deployed during the Summer of 2005. Swan Pond is the terminal basin in this estuary and therefore has the highest level of nitrogen enrichment.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate significantly nutrient enriched waters within the upper basin (Swan Pond) of the Swan Pond River system (Figures VII-3). Since Swan Pond is the terminal basin of this estuary, these measured levels represent the most enriched conditions, as the Swan Pond River is intermediate in nutrient related water quality conditions between Swan Pond and the high quality waters of Nantucket Sound. Within the basin of Swan Pond the oxygen data is consistent and accompanied by significant organic matter enrichment, primarily from phytoplankton production (and accumulated macroalgal biomass) as seen from the parallel measurements of chlorophyll a. 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 of this estuarine system.

The oxygen record for Swan Pond shows that this coastal salt pond is highly nitrogen enriched with large daily oxygen excursions and large phytoplankton blooms. The oxygen excursions were extreme, frequently 10 mg L^{-1} and periodically 15 mg L^{-1}, some of the highest observed by the MEP. 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 (generally ~7-8 mg L^{-1}) in these shallow basins. The clear evidence of oxygen levels above atmospheric equilibration also supports the contention that Swan Pond is nitrogen and organic matter enriched, primarily through in situ production by phytoplankton (and macroalgae). The measured extent of dissolved oxygen depletion indicates that Swan Pond River is presently experiencing significant oxygen stress during summer. This level of stress is typically observed in southeastern Massachusetts estuaries with a high degree of habitat impairment, particularly benthic animal habitat. The embayment specific results are as follows:

Swan Pond DO/CHLA Mooring (Figures VII-3 and VII-4):

The Swan Pond instrument mooring was located within the central basin of Swan Pond, the terminal basin at the northern end of the Swan Pond River System, well away from the tidal inlet connecting the estuary to the waters of Nantucket Sound. The mooring was centrally located in Swan Pond. Large daily excursions in oxygen levels were observed at this location, ranging from levels more than twice air equilibration to hypoxic conditions. Bottomwater oxygen frequently declined to <3 mg L^{-}, with periodic anoxia (<0.2 mg L^{-1}) Figure VII-3, Table VII-1. The Swan Pond basin is highly nitrogen enriched with parallel levels of organic enrichment as demonstrated by the regular algal blooms (phytoplankton, macroalgae) that have been documented over the past 6 years, consistent with the large daily excursions in oxygen levels (10-15 mg L^{-1}). Diurnal oxygen excursions result from high rates of photosynthesis (carbon fixation) increasing oxygen levels above air equilibration and high rates of dark respiration lowering levels to below equilibration.

Day time oxygen levels regularly exceeded 15 mg L^{-1} and periodically exceeded 20 mg L^{-1}. These high oxygen levels are likely the result of the combined effects of photosynthesis by the large phytoplankton biomass and macroalgae and relatively quiescent waters (Figure VII-3). Over the 30 day deployment there appear to be multiple moderately intense phytoplankton blooms where chlorophyll a increased to 10-15 ug L^{-1} and one extremely intense bloom event where chlorophyll-a concentrations exceeded 100 ug L^{-1}. The very high bloom value was confirmed independently by a calibration sample taken during the bloom (Figure VII-4). Average

chlorophyll a levels during the deployment were high, 15.3 ug L-1, with and even higher multi-year average observed by the Dennis water quality monitoring program, 29 ug L^{-1}.
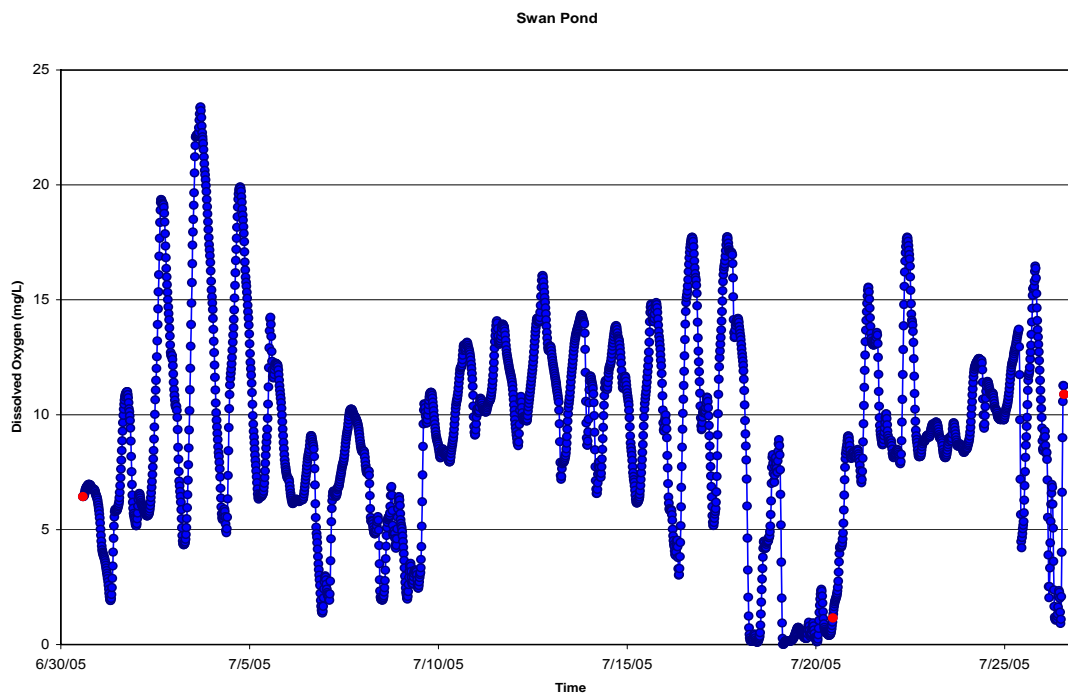
Swan Pond

Figure VII-3. Bottom water record of dissolved oxygen at the Swan Pond station, Summer 2005. Calibration samples represented as red dots.
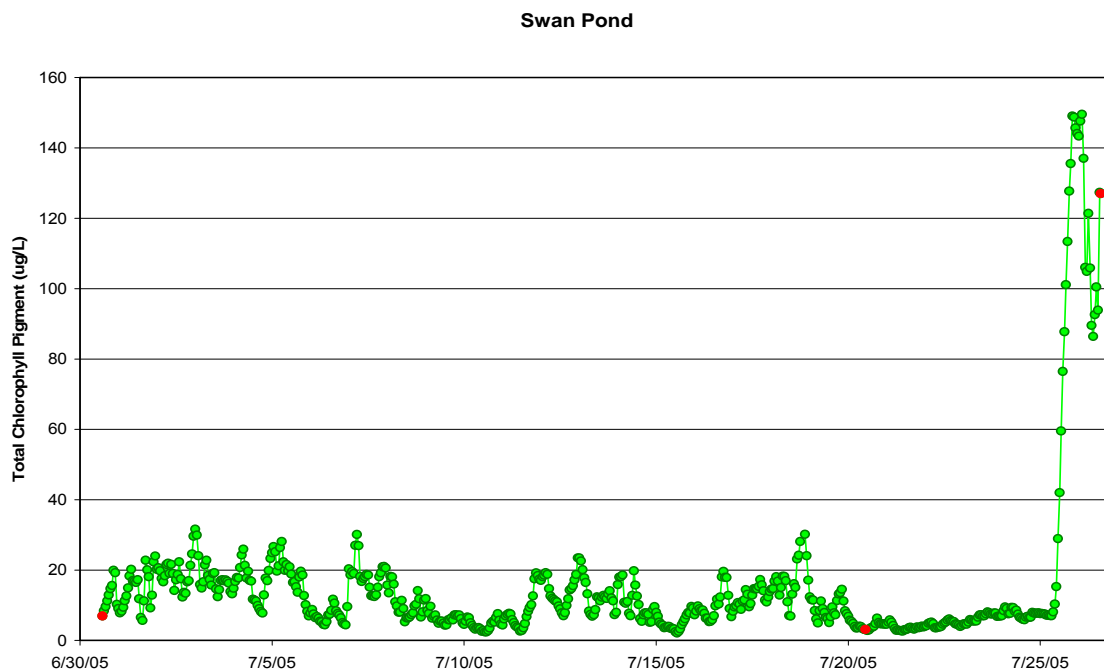
Swan Pond

Figure VII-4. Bottom water record of Chlorophyll-*a* in the Swan Pond station, Summer 2005. Calibration samples represented as red dots. Note that the large bloom in late July was confirmed by the field calibration sample.

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 Swan 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)
Swan Pond Dennis	6/30/2005	7/26/2005	26.0	23%	18%	14%	11%
			Mean	0.38	0.33	0.36	0.30
			Min	0.08	0.02	0.02	0.03
			Max	1.68	1.65	1.55	1.51
			S.D.	0.39	0.42	0.45	0.44

Table VII-2.   Duration (days and % of *in situ* sensors deployment time) that chlorophyll a levels exceed various benchmark levels within the Swan Pond 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)
Swan Pond Dennis	6/30/2005	7/26/2005	26.0	83%	47%	30%	12%	7%
Mean Chl Value = 15.3 ug/L			Mean	1.35	0.53	0.32	0.17	0.24
			Min	0.08	0.04	0.08	0.04	0.04
			Max	5.75	2.75	1.21	1.17	1.17
			S.D.	1.77	0.62	0.26	0.25	0.41

Consistent with the large diurnal oxygen excursions and large phytoplankton blooms and macroalgal accumulations, oxygen depletion was frequent in bottom waters. The frequency and duration of periodic hypoxia/anoxia in this system indicates significant habitat impairment, and is consistent with nitrogen enrichment. In the Swan Pond portion of the overall system, chlorophyll a exceeded the 10 ug L^{-1} benchmark 47% of the time and was over 20 ug L^{-1} 12% 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, which is well below the summer conditions within Swan Pond.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data are a key part of the MEP Approach. Surveys were conducted in the Swan Pond River Estuary, particularly within the main tidal channels (Swan Pond River up to Route 28) by the MassDEP Eelgrass Mapping Program (C. Costello). The most recent survey was conducted in 2006/07 as part of the MEP program following two earlier field surveys by MassDEP in 1995 and 2001. MassDEP also analyzed aerial photographs from 1951 and was able to delimit the eelgrass beds that existed prior to the significant development of the watershed that has occurred over the past 60 years. The 1951 data were validated through discussion with the Town of Dennis, including individuals with long-term, on-site knowledge of this system, particularly Swan Pond and the main tidal channel. The 2001 map was field validated by the MassDEP Eelgrass Mapping Program. The primary use of the data is to indicate (a) estuarine regions that have historically or presently support eelgrass habitat, and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 and to 2006/07 (Figure VII-5); the overall period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community, as well as its distribution.

At present there are no eelgrass beds within the Swan Pond River Estuary as determined from the 1995, 2001 and 2006/07 surveys. The eelgrass survey conducted by the MEP Technical Team in 2005 was associated with the benthic regeneration and benthic animal surveys and during the deployment and recovery of the instrument moorings. The MEP Technical Team confirmed both the lack of eelgrass beds in Swan Pond and the main channel of the Swan Pond River system from Nantucket Sound into Swan Pond. However, divers did observe a few eelgrass patches in the lowest reach of the River nearest the tidal inlet. This suggests that if water quality conditions are improved, that re-establishment of historic eelgrass beds should be rapid and proceed without the need for transplants. The 1951 assessment indicated beds within this same lower region of the Swan Pond River system. The 1951 analysis was based upon high quality aerial photos. To validate the eelgrass distribution of 1951, the Dennis Natural Resources Department was asked about historic eelgrass beds within the Swan Pond River Estuary (without access to the MassDEP map). First-hand accounts indicated that eelgrass was not present in Swan Pond, but beds had been observed in patches within the lower reach of the Swan Pond River. These first-hand accounts dated back to the 1940's and 1950's and matched with the independent assessment by MassDEP (Figure VII-5). In contrast to most of the Swan Pond River Estuary, eelgrass beds are present offshore of the inlet as well as offshore of adjacent Bass River and nearby Lewis Bay and Parkers River estuaries within the Town of Yarmouth.
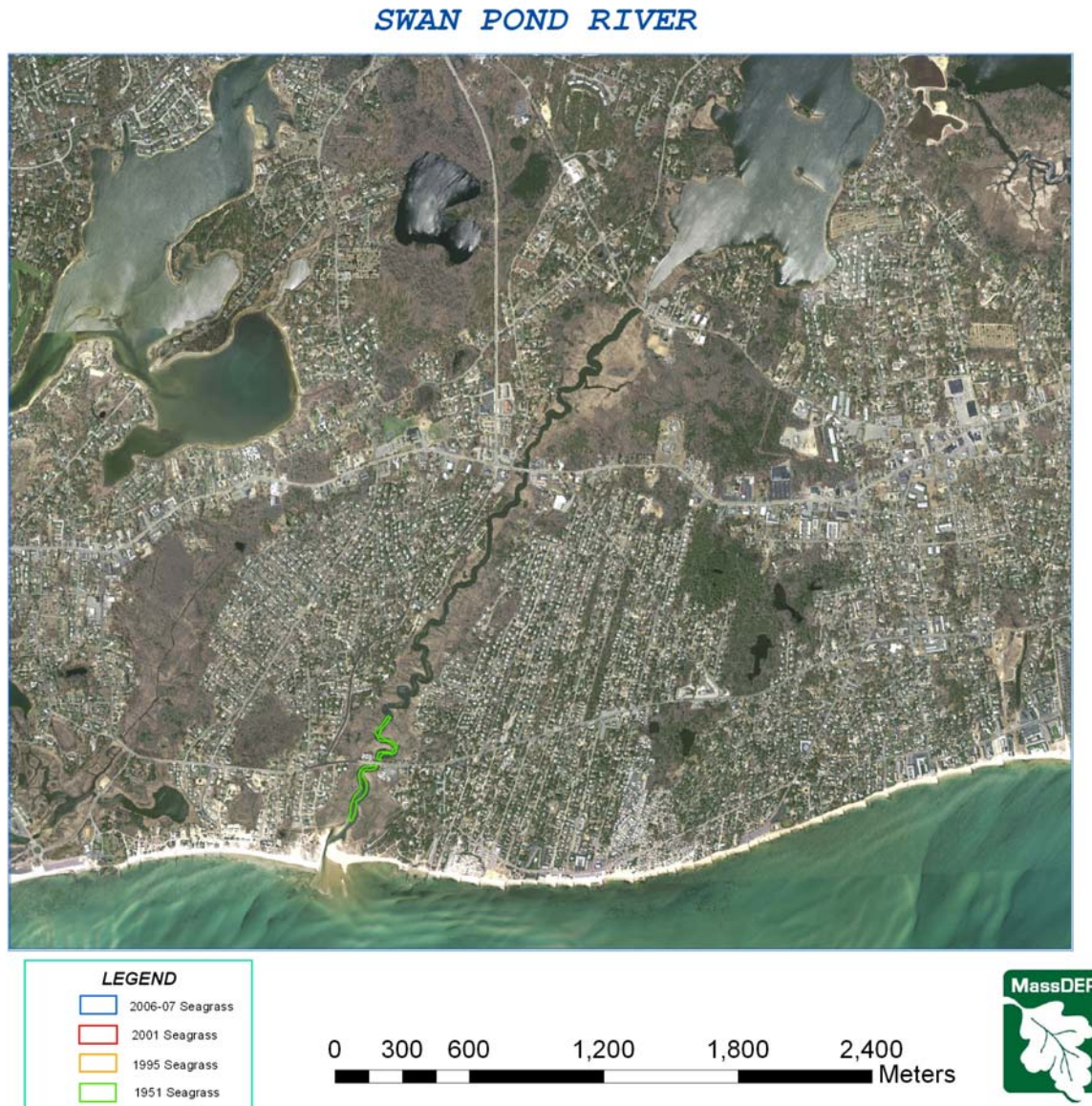
SWAN POND RIVER

Figure VII-5.   Eelgrass bed distribution within the Swan Pond River System. 1951 beds delineated from aerial photography are circumscribed by the green outline. There were no beds identified in 1995, 2001 or 2006/07. (Map from the MassDEP Eelgrass Mapping Program).

Overall, the historical distribution of eelgrass within the Swan Pond River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. Shallow salt marsh basins, like the upper portion of the tidal river above Route 28 (with extensive associated wetlands) and Swan Pond typically do not support eelgrass beds. Additionally, the lower reach of the Swan Pond River with its access to nearby Nantucket Sound flood tidal waters is capable of supporting water quality (and clarity) sufficient to sustain eelgrass beds, although its present lack of beds is consistent with its present nitrogen and organically enriched condition. Lowering the present nitrogen loads, hence enrichment of tidal waters within the lower River, will improve water clarity

and oxygen levels, and with sufficient loading reduction will allow the re-establishment of the historic eelgrass beds.

The present nitrogen loading to the Swan Pond River System is sufficient to raise significantly the nitrogen level in tidal waters from ~0.30 mg L^{-1} (flood waters) to 0.66 mg L-1 in the region of the historic eelgrass beds within the lower Swan Pond River. Therefore, the current absence of eelgrass within this system is expected given the high nitrogen levels and high chlorophyll levels in all basins, and in the areas historically supportive of eelgrass beds. Typically eelgrass beds exist at much lower nitrogen levels (0.35 - 0.45 mg N L^{-1}) than presently found in this system (0.47 - 1.21 mg N L^{-1}). The high nitrogen levels within the Swan Pond River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates, which increases the nitrogen levels in the incoming tidal waters (0.3 mg L^{-1}) by several fold (see Section VI).

Based upon the MassDEP Eelgrass Mapping Program surveys, it appears that about 2-3 acres of eelgrass beds could be restored in this estuary through nitrogen management. This is based upon the acreage of eelgrass bed lost since 1951 (Table VII-3). Nitrogen management to restore this resource would necessarily lower the nitrogen enrichment of the up-gradient estuarine waters, resulting in lowered organic enrichment, higher oxygen and improved benthic animal habitats.

Table VII-3.   Change in eelgrass coverage within the Swan Pond River Embayment System, Town of Dennis, as determined by the MassDEP Eelgrass Mapping Program (C. Costello).

EMBAYMENT	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Swan Pond River	2.47	0.00	0.00	100%

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at eight (8) locations within the Swan Pond River Embayment System (Figure VII-6), 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 dissolved oxygen) to highly stressed (high organic matter loading-low dissolved oxygen). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that given the loss of eelgrass from 1951 to 2006, the Swan Pond River system is considered impaired by nutrient overloading. Similarly, based upon the oxygen and chlorophyll a levels and macroalgal accumulations in Swan Pond and the associated very high watercolumn total nitrogen (>1.0 mg N L^{-1}), it is clear that the upper

Figure VII-6.   Aerial photograph of the Swan Pond River System showing location of benthic infaunal sampling stations (red symbol).

estuary is highly impaired by nitrogen enrichment. To the extent that the system can support eelgrass in the lower portion and healthy infaunal communities in areas that have never supported eelgrass habitat, the benthic infauna analysis provides another determinant as to the level of present impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic (E) can range from 0-1 (one being most even), while the diversity index (H') 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 H'>3) and evenness (E~0.7). The converse is also true, with poorest habitat quality found where diversity is <1 and evenness is <0.5.

Overall, the Infauna Survey indicated that the basins comprising the Swan Pond River system are presently supporting impaired benthic infaunal habitat (Table VII-4). The upper terminal basin of Swan Pond is clearly significantly impaired, with low diversity (H'=1.6) and Evenness (E=0.56). While the benthic animal community has a very high number of individuals (>1000 per sample), it is comprised of a few species (8). More importantly, the species dominating the community are indicative of stress, with about 50% of the total community consisting of a well known organic enrichment indicator, *Capitella capitata*. This species is typical of embayments with high organic matter deposition and poor habitat quality. The high numbers of individuals are due to the unstable environment in which small rapidly growing/short lived species proliferate. Based upon all habitat indices, Swan Pond basin is presently supporting significantly impaired to severely degraded habitat. This could be predicted from the very high phytoplankton biomass, with large blooms and accumulations of macroalgae, resulting in periodic hypoxic/anoxic levels of oxygen depletion of bottom waters.

Table VII-4. Benthic infaunal community data for the Swan Pond River embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2). Stations refer to map in Figure VII-7, (N) is the number of samples per site.

Location	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)	Stations SWPR-
Swan Pond River Embayment System						
Swan Pond	8	1333	5	1.62	0.56	10,12,14,15
Swan Pond River Upper	12	729	8	1.83	0.52	8
Swan Pond River Lower	25	346	17	3.31	0.72	1,3,5
* Station i.d.'s refer to site map above.						

The Swan Pond River presently shows a gradient in benthic animal habitat quality from low/intermediate levels in the upper reach to moderately high quality habitat in the lower reach. The gradient in benthic animal habitat parallels the key water quality parameters of oxygen and chlorophyll a. The upper reach of the Swan Pond River supports a benthic animal community with a high number of individuals (>700), low-moderate number of species (12), but is not dominated by stress indicator species. However, the community is dominated by species common estuaries with organic enrichment and has low diversity (H'=1.8) and Evenness

(E=0.52). The low Diversity, Evenness, and number of species are indicative of a moderate to high level of impairment. Additionally, this basin has significant associated salt marsh, which may be contributing to the benthic habitat quality. However, the low quality ebb tidal waters from Swan Pond appear to be a major factor in the present habitat quality of this reach of the River and all the benthic metrics clearly show impairment of benthic habitat.

In contrast to the upper basins, the lower reach of the Swan Pond River is presently supporting high quality to moderately impaired benthic animal habitat. The lower reach has generally small depletions of oxygen and moderate to high chlorophyll a levels, primarily for waters ebbing from the upper estuary. However, during the flood tide period, this reach contains water approaching the high quality of Nantucket Sound. Consistent with the water quality, the lower reach of the Swan Pond River supports a benthic animal community with a moderate to high number of individuals (>300), moderate to high number of species (25), but with some stress indicator species (~20%). The community has a high diversity (H'=3.3) and Evenness (E=0.72) is composed of polychaetes, crustaceans and mollusks, with some deep burrowers. Integrating all of the benthic animal information, indicates a generally high quality habitat but with some slight impairment, due to the stress indicator species (tubificids). However, restoration of benthic animal habitat in Swan Pond will clearly improve water quality during ebb tides in the Swan Pond River, also restoring benthic animal habitat in these downgradient basins as well.

The benthic animal communities within the basins of the Swan Pond River were compared to high quality environments, such as the Outer Basin of Quissett Harbor, which provides additional confirmation of impaired habitat. 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 Evenness (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 (tubificids, capitellids).

In contrast, the Swan Pond River System is supporting a similar gradient of impaired benthic animal habitat as the adjacent Bass River System and other nearby estuaries of similar structure, such as the Parkers River. The uppermost basin of the Parkers River (Seine Pond) also supports poor benthic habitat throughout the basin with few species and individuals (i.e. low secondary production), with an average of 6 species and 48 individuals per sample (similar to Dinah and Kelleys Ponds in Bass River and Swan Pond). Seine Pond also has very low diversity (1.25) and Evenness <1 (0.65) and is dominated by stress indicator species associated with organic matter enrichment. The lower reach of Parker's River , closer to the high quality Nantucket Sound waters, currently supports higher species numbers (27 species), more in line with a high quality benthic animal habitat. However, the Diversity (2.94) and Evenness (0.61) indices suggest a moderate level of impairment in this lower reach, much like the lower Swan Pond River reach.

Overall, the pattern of infaunal habitat quality throughout the Swan Pond River System is consistent with measured dissolved oxygen concentration, chlorophyll, nutrients and organic matter enrichment in this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland influenced such as the upper reach of the Swan Pond River or tidal embayment with wetland influence (Swan Pond). Based upon this analysis it is clear that the upper regions of the Swan

Pond River Embayment System are significantly impaired by nitrogen and organic matter enrichment while the lower basin (lower reach of River) is presently supporting high quality to moderately impaired benthic animal habitat. The proximate cause of impairment is organic matter enrichment and oxygen depletion, stemming ultimately from nitrogen enrichment. Total nitrogen levels within the upper basin (Swan Pond) presently exceed 1.0 mg N L-1, a level associated with significant impairment of benthic animal habitat in southeastern Massachusetts estuaries.

Other Benthic Resource Characteristics:

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish. As is the case with some embayments on Cape Cod, all of the enclosed waters of Swan Pond River are classified as conditionally approved for shellfishing during specific times of the year with certain areas of the upper Swan Pond prohibited for the taking of shellfish during any season of the year (Figure VII-7). However, regulatory closure of the Swan Pond River basins due to bacterial contamination has been improving in recent years through the efforts of citizen stewards and Town of Dennis officials, the Board of Health and Department of Public Works. Significant efforts were undertaken to implement Title 5 regulations and to reduce direct surface water runoff to the estuary. In addition, a by-law against feeding waterfowl reduced their numbers (hence waste inputs) relative to Swan Pond. In December 2008, MassDMF was able to reclassify a large portion of Swan Pond, ~144 acres, from "Prohibited" to "Conditionally Approved", a change which reopened significant shellfish beds to harvest from December through April 30 each year. The change was possible through an active program to find and eliminate or mitigate sources of contamination. The steps taken by the citizens of Dennis and Town officials, boards and departments have been credited with reducing bacterial contamination resulting in the opening of this important shellfishing area (Hickey, 2008).

The Swan Pond River system has been classified as supportive of specific shellfish communities (Figure VII-8). The major shellfish species with potential habitat within the Farm Pond Estuary are soft shell clams (*Mya*) throughout the system and quahogs (*Mercenaria*) extending essentially up the main tidal channel from the inlet to approximately the level of the Route 28 bridge crossing. Improving water quality to restore eelgrass in the lower portion of the system as well as benthic animal habitat quality throughout should also expand the shellfish growing area within this system. Historically, Swan Pond and Swan River contained significant soft-shelled clam and American Oyster resources and limited quahogs (Hickey 2008).
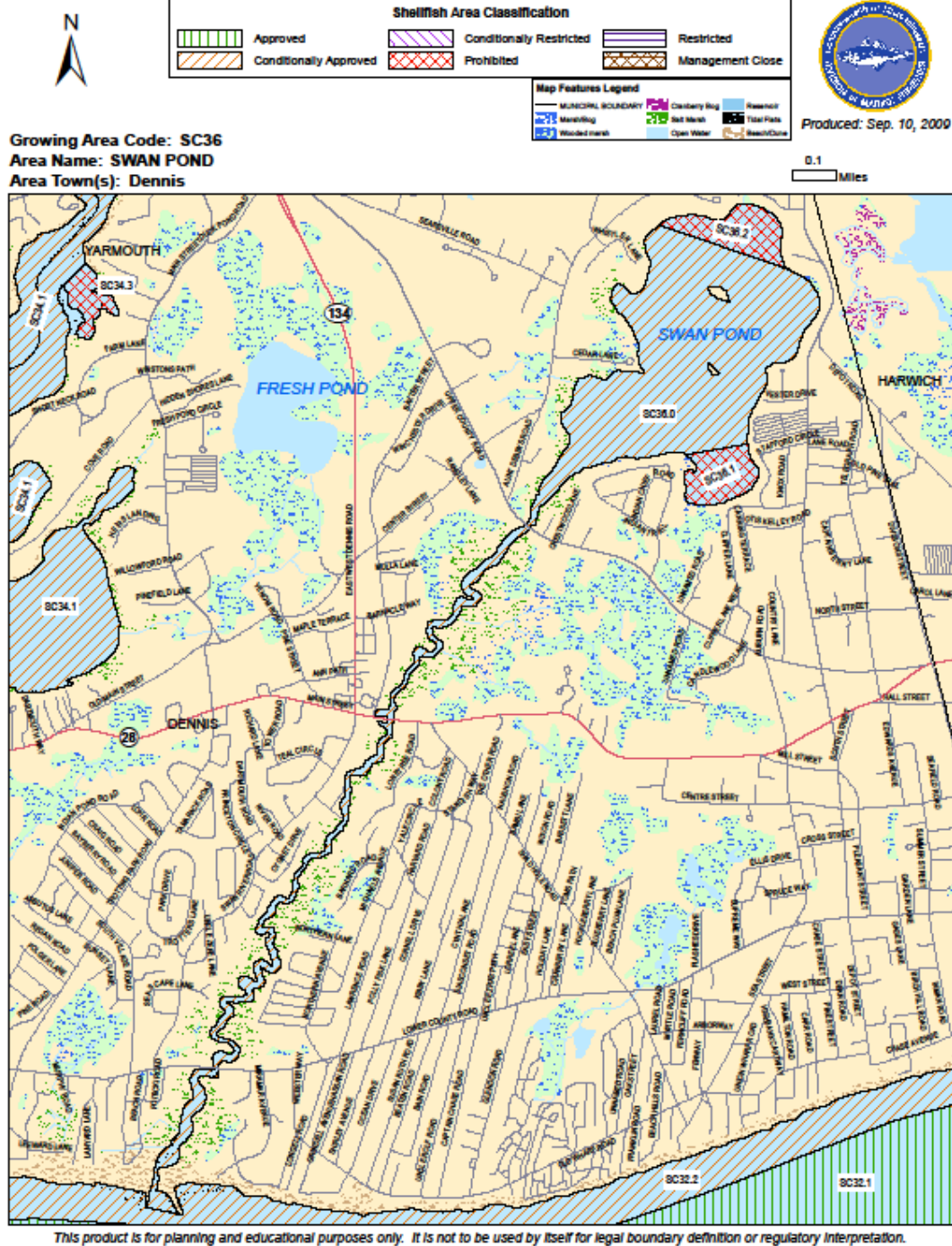
Figure VII-7. Location of shellfish growing areas in the Swan Pond River system and the status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.
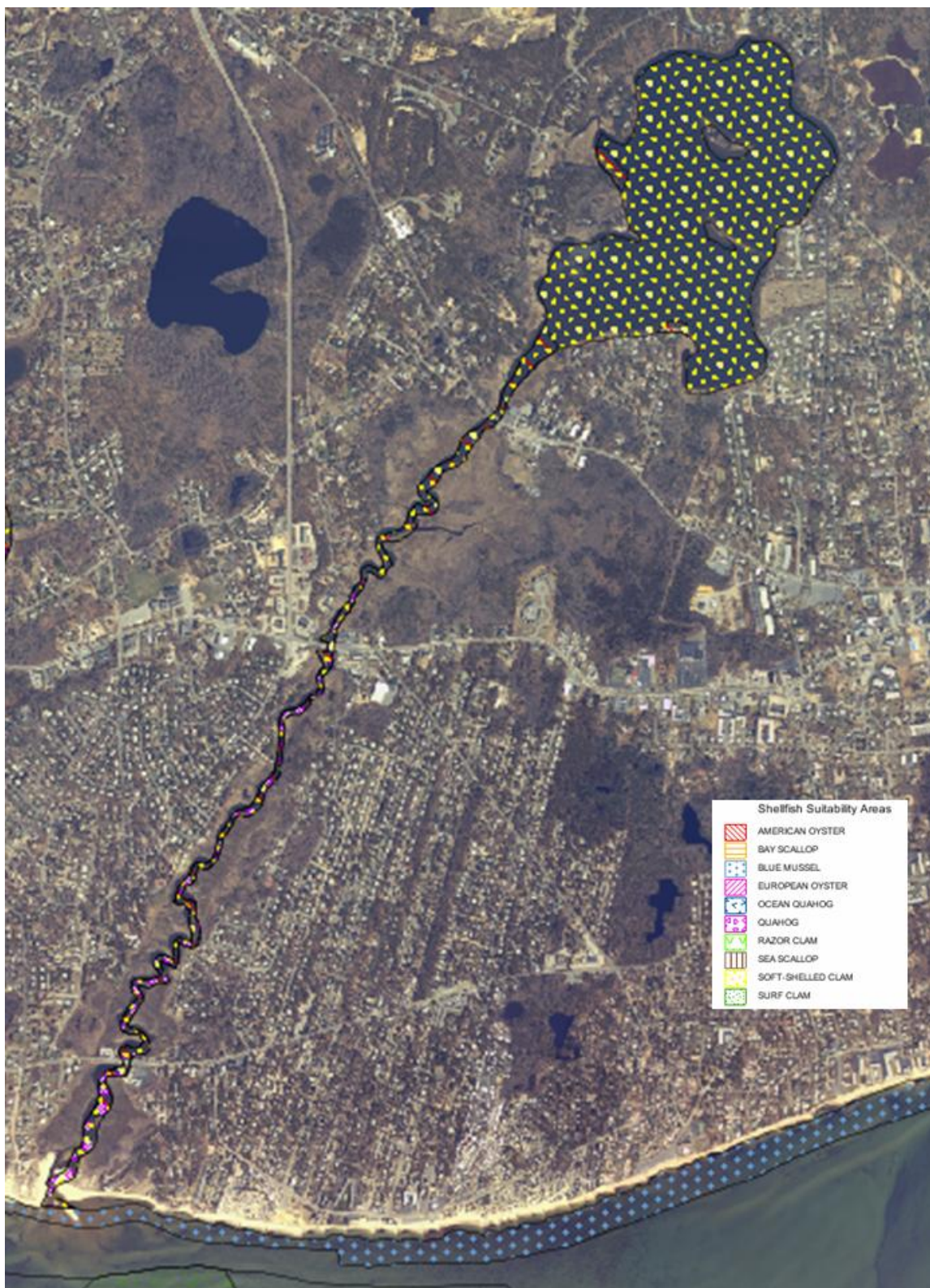
Figure VII-8. Location of shellfish suitability areas within the Swan Pond River Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence".

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 and its associated watershed nitrogen load further strengthen the analysis. These data were collected to support threshold development for the Swan Pond River 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 Dennis Water Quality Monitoring Program conducted by the Dennis Water District with technical and analytical support from the Coastal Systems Program at SMAST-UMass Dartmouth.

The Swan Pond River Embayment System is a complex estuary composed of two functional types of component basins: open water embayment (Swan Pond) and tidal river (Swan Pond River), with the upper reaches supporting significant salt marsh area. Each of these functional components has different natural sensitivities to nitrogen enrichment and organic matter loading. In addition, the extensive salt marsh introduces a level of natural organic enrichment. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of each system and the ability to support eelgrass beds and the types of benthic infaunal communities that they support. At present, the Swan Pond River is showing differences in nitrogen enrichment and habitat quality among its various component basins (Table VIII-1), with regions of clearly impaired habitat. The tidal waters of the Swan Pond River Embayment System are currently listed under the State Surface Water Classification as SA waters. The Swan Pond River Estuary is not presently meeting the water quality standards for SA waters. The result is that as required by the Clean Water Act, TMDL processes and management actions must be developed and implemented for the restoration of resources within this estuary.

Overall, the system is showing some nitrogen-related habitat impairment within each of its component basins, however, there is a strong gradient. Swan Pond is a significantly impaired basin relative to benthic animal habitat, since it historically has not supported eelgrass. Nitrogen enrichment (through inputs and naturally low tidal exchange) has resulted in frequent large phytoplankton blooms, periodic hypoxia/anoxia, large macroalgal accumulations and a benthic community comprised of stress indicator species (e.g. 50% of the community is *Capitella capitata*). These stress indicator species are found in highly disturbed organic enriched basins, they are small opportunistic organisms, growing quickly and reproducing before environmental conditions become lethal. The Swan Pond River is also nitrogen enriched, but has less nitrogen enrichment based primarily on its structure and high relative water turnover. While the lower reach currently supports only high quality to moderately impaired benthic habitat, its loss of historical eelgrass beds indicates that it has become a significantly impaired basin relative to eelgrass habitat. The upper tidal reach of the Swan Pond River is intermediate in habitat quality between Swan Pond and the lower River. The upper tidal reach (above Rt. 28) is moderately to significantly impaired based upon its benthic animal habitat, due primarily to organic and nitrogen rich waters ebbing from Swan Pond and natural enrichment processes associated with its extensive wetlands. The result is high phytoplankton biomass with some oxygen depletion and organic enriched sediments and an animal community with very low diversity. Even as a salt marsh influenced "basin", the benthic habitat though

moderately productive is clearly impaired as indicated by the high chlorophyll levels and some of the infaunal indicators. The upper River has not historically supported eelgrass habitat. Fortunately, the high tidal velocities in the tidal river prevent significant accumulation of macroalgae, in both the upper and lower reaches, preventing the level of nitrogen enrichment impacts that occur in Swan Pond. Overall, however, the regions of significant and moderate habitat impairment comprise >90% of the estuarine area of the Swan Pond River Embayment System.

The frequency and duration of periodic hypoxia/anoxia in the Swan Pond system also indicates significant habitat impairment, consistent with nitrogen enrichment. The oxygen levels within Swan Pond indicate a coastal salt pond highly organic and nitrogen enriched with large daily oxygen excursions, large phytoplankton blooms and periodic hypoxia/anoxia. Bottomwater oxygen frequently declined to <3 mg L^-, with periodic anoxia (<0.2 mg L^{-1}). These oxygen conditions are consistent with the parallel levels of organic and nitrogen enrichment as demonstrated by the regular algal blooms (phytoplankton, macroalgae) documented over the past 6 years, the Swan Pond basin also demonstrated has large daily excursions in oxygen levels (10-15 mg L^{-1}). Diurnal oxygen excursions result from high rates of photosynthesis (carbon fixation) increasing oxygen levels above air equilibration and high rates of dark respiration lowering levels to below equilibration and are evidence of a system "out-of-balance". Swan Pond is presently experiencing significant oxygen stress during summer.

Consistent with the large diurnal oxygen excursions and frequent oxygen depletion in bottom waters were large phytoplankton blooms and macroalgal accumulations. In the Swan Pond portion of the overall system, chlorophyll a exceeded the 10 ug L^{-1} benchmark 47% of the time and was over 20 ug L^{-1} 12% of the time-series record. The MEP documented multiple moderately intense phytoplankton blooms where chlorophyll a increased to 10-15 ug L^{-1} and one extremely intense bloom event where chlorophyll-a concentrations exceeded 100 ug L^{-1} in a single 30 day period. Average chlorophyll a levels during this 30 day record were high, 15.3 ug L-1, with an even higher in multi-year average, 29 ug L^{-1}, observed by the Dennis water quality monitoring program. Average chlorophyll levels over 10 ug L^{-1} have been used to indicate eutrophic conditions in embayments. The chlorophyll a concentrations in Swan Pond are consistent with the measured extent of dissolved oxygen depletion and these are consistent with its level of nitrogen enrichment and algal biomass. This level of stress is typically observed in southeastern Massachusetts estuaries with a high degree of habitat impairment, particularly benthic animal habitat.

Since Swan Pond is the terminal basin of this estuary, it has the most enriched conditions within the estuary, with the Swan Pond River being intermediate in nutrient related water quality between Swan Pond and the high quality waters of Nantucket Sound. The tidal river has higher water quality than the terminal basin of Swan Pond, primarily as a result of its much higher water turnover and its more direct access to the high quality waters of Nantucket Sound which enter during flood tides. The upper reach of the River is primarily organic and nitrogen enriched by the ebbing waters of Swan Pond and direct watershed nitrogen loads, with some natural enrichment from the associated extensive tidal wetlands. However, this basin is presently impaired based upon its high chlorophyll a levels (multi-year: ~20 ug L^{-1}) and periodic hypoxia/anoxia. The lower reach of the Swan Pond River supports the highest quality habitat within this estuary, with some oxygen depletions and moderate to high chlorophyll a levels (~10 ug L^{-1}) entering from up-gradient on the ebbing tides. However, the twice daily tidal flushing with adjacent Nantucket Sound waters results in there being only a moderate level of nitrogen enrichment, but enough to result in the loss of historic eelgrass beds from this basin.

Consistent with the observed levels of nitrogen and organic enrichment coupled with the distribution of oxygen depletion throughout the estuary, benthic animal habitat showed a wide range in quality. The upper terminal basin of Swan Pond is clearly significantly impaired, with low diversity (H'=1.6) and Evenness (E=0.56). While the benthic animal community has a very high number of individuals (>1000 per sample), there are few species (8). More importantly, the species dominating the community are indicative of stress, with about 50% of the total community consisting of a well-known organic enrichment indicator species, *Capitella capitata*. This species is typical of embayments with high organic matter deposition and poor habitat quality. The high numbers of individuals are due to the unstable environment in which small rapidly growing/short lived species proliferate. Based upon all habitat indices, Swan Pond basin is presently supporting significantly impaired to severely degraded habitat and this is consistent with the very high phytoplankton biomass, with large blooms and accumulations of macroalgae, and periodic hypoxic/anoxic levels of oxygen depletion of bottom waters.

Swan Pond River presently shows a gradient in benthic animal habitat quality from low/intermediate levels in the upper reach to moderately high quality habitat in the lower reach. The gradient in benthic animal habitat parallels the key water quality parameters of oxygen and chlorophyll a. The upper reach of the Swan Pond River supports a benthic animal community with a high number of individuals (>500), low-moderate number of species (12), but is not dominated by stress indicator species. However, the community is dominated by species common to basins with organic enrichment and has low diversity (H'=1.8) and Evenness (E=0.52). The low diversity, Evenness and number of species are indicative of a moderate to high level of impairment, this basin has significant associated salt marsh, which may be contributing to the observed benthic habitat quality. However, the low quality ebb tidal waters from Swan Pond appear to be a major factor in the present habitat quality of this reach of the River and all of the benthic metrics clearly show impairment of benthic habitat.

In contrast to the upper basins, the lower reach of the Swan Pond River is presently supporting high quality to moderately impaired benthic animal habitat. The lower reach has generally small depletions of oxygen and moderate to high chlorophyll a levels, primarily for waters ebbing from the upper estuary. However, during the flood tide period, this reach contains water approaching the high quality of Nantucket Sound. Consistent with the water quality, the lower reach of the Swan Pond River supports a benthic animal community with a moderate to high number of individuals (>300), moderate to high number of species (25), but with some stress indicator species (~20%). The community has a high diversity (H'=3.3) and Evenness (E=0.72) is composed of polychaetes, crustaceans and mollusks, with some deep burrowers. Integrating all of the benthic animal information, indicates a generally high quality habitat but with some slight impairment, due to the stress indicator species (tubificids). However, achieving the N-threshold that would be restorative of benthic animal habitat in Swan Pond will clearly improve water quality during ebb tides in the Swan Pond River, also restoring benthic animal habitat in these down gradient basins as well.

Overall, the pattern of infaunal habitat quality throughout the Swan Pond River System is consistent with measured dissolved oxygen concentration, chlorophyll, nutrients and organic matter enrichment in this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland influenced such as the upper reach of the Swan Pond River or tidal embayment with wetland influence like Swan Pond. Based upon this analysis it is clear that the upper regions of the Swan Pond River Embayment System are significantly impaired by nitrogen and organic matter enrichment while the lower basin (lower reach of River) is presently supporting high quality to moderately impaired benthic animal habitat. The proximate cause of impairment is organic

matter enrichment and oxygen depletion, stemming ultimately from nitrogen enrichment. Total nitrogen levels within the upper basin (Swan Pond) presently exceed 1.0 mg N L-1, a level associated with significant impairment of benthic animal habitat in southeastern Massachusetts estuaries.

The absence of eelgrass throughout the Swan Pond River Estuary is consistent with the observed nitrogen and the chlorophyll levels and functional basin types comprising this estuary. The lower Swan Pond River supported eelgrass beds in 1951 under lower nitrogen loading conditions. This eelgrass was lost sometime prior to 1995.

The historical distribution of eelgrass and its present absence within the Swan Pond Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. Shallow salt marsh influenced basins, particularly when situated as terminal salt ponds at the end of a long tidal river, like Swan Pond typically do not typically support eelgrass beds. These basins are configured to have elevated nitrogen levels due to their focus of groundwater and surface water inflows and their lower flushing rates, compared to lower reaches of their estuary. In the case of Swan Pond, the friction caused by the length and meandering of the tidal river reduces the tidal prism by about two-thirds, compared to a similar basin with a more direct connection with Nantucket Sound. As a result, Swan Pond has likely been nutrient enriched with poor water clarity for many decades. The upper River, though supporting greater water turnover, is also enriched in nitrogen and organic matter, as seen in the high phytoplankton biomass in its waters and organic enrichment of its sediments. This enrichment stems from the combination of natural enrichment from its associated wetlands, inputs from its watershed, and significantly from inputs of the highly enriched waters ebbing from Swan Pond. As a result, the present and historic absence of eelgrass beds from this basin is also expected. Given the structure of the basins and the absence of eelgrass beds historically and at present, nitrogen management to promote eelgrass beds in these basins is not indicated.

In contrast, the lower reach of the Swan Pond River with its direct inflow of low nutrient high quality water from Nantucket Sound on each flood tide and its high water turnover would be expected to have sufficient water clarity and oxygen levels to support eelgrass beds at the lower levels of watershed nitrogen loading in 1951. However, given the sensitivity of eelgrass to declining light penetration resulting from nutrient enrichment and secondary effects of organic enrichment and oxygen depletion, the current absence of eelgrass within this system is expected given the high nitrogen levels and high chlorophyll levels measured in all basins (>10 ug L-1 chlorophyll a). Total nitrogen levels in the lower River (0.47 - 0.83 mg N L-1) while lower than the upper River (0.83 - 1.06 mg N L-1) and Swan Pond (1.6 -1.21 mg N L-1) are higher than typically associated with eelgrass beds in southeastern Massachusetts (0.35 - 0.45 mg N L-1).

The high nitrogen levels within the Swan Pond River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates. This loading increase the nitrogen levels in the incoming tidal waters (0.3 mg L-1) by several fold (see Section VI). As there is no evidence of eelgrass coverage within Swan Pond or the upper reach of the Swan Pond River within the past 6 decades, they should not be considered for eelgrass restoration. In contrast, documented eelgrass within the lower Swan Pond River make restoration of this resource a primary target for overall restoration of the Swan Pond River System. Restoration of this habitat will require appropriate nitrogen management. Note that restoration of this eelgrass habitat will necessarily result in restoration of other resources throughout the system, particularly within the upper estuary. Swan Pond and the upper Swan Pond River are the

discharge basins for much of the watershed nitrogen load to this estuary. The lower reach of the Swan Pond River is the channel through which the nitrogen and organic matter enriched waters from the upper estuary is flushed out on ebbing tides. Nitrogen management focused on lowering nitrogen levels within the lower River will require lowering of nitrogen levels throughout the upper estuary. Therefore lowering nitrogen loading to improve of infaunal habitats within these upper basins will also result in improving eelgrass habitat in the lower River.

Based upon the above analysis (Table VIII-1), eelgrass habitat was selected as the primary nitrogen management goal for the lower reach of the Swan Pond River and infaunal habitat quality the management target for Swan Pond and the upper reach of the Swan Pond River. These goals are the focus of the MEP threshold analysis presented in Section VIII.3.

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

The approach for determining nitrogen loading rates that will support acceptable habitat quality throughout an embayment system is to first identify a sentinel location within the embayment and secondly, to determine the nitrogen concentration within the water column that will restore the 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 (Section VIII.2), the Linked Watershed-Embayment Model is used to sequentially adjust nitrogen loads until the targeted nitrogen concentration is achieved (Section VIII.3).

Determination of the critical nitrogen threshold for maintaining high quality habitat within the Swan Pond River Embayment System is based primarily upon the nutrient and oxygen levels, temporal trends in eelgrass distribution and current benthic community indicators. Given the information on a variety of key habitat characteristics, it is possible to develop a site-specific threshold, which is a refinement upon more generalized threshold analyses frequently employed.

The Swan Pond River Embayment System presently supports a range of infaunal habitat quality. A gradient in nutrient related habitat degradation was observed from the most inland reach of the overall system (Swan Pond) to the higher quality habitat near the tidal inlet. While the upper tidal river is partially naturally nutrient and organic matter enriched (due to extensive salt marsh), the existing benthic communities and high chlorophyll-a level still suggest a moderate level of impairment for this portion of the overall Swan Pond System. However, the primary habitat issues within the Swan Pond River Embayment System relate to the loss of the eelgrass beds from the lower Swan Pond River as well as the highly degraded benthic animal habitat in the upper estuary, in particular in Swan Pond. The loss of eelgrass classifies the lower Swan Pond River as "significantly impaired", although this estuarine basin presently supports high quality to moderately impaired infaunal communities. The impairments to both the infaunal habitat and the eelgrass habitat within the component basins of the Swan Pond River Embayment System are supported by the variety of other indicators including oxygen depletion, chlorophyll, and TN levels, all of which support the conclusion that these impairments are the result of nitrogen enrichment, primarily from watershed nitrogen loading (Table VIII-1).

Table VIII-1. Summary of nutrient related habitat quality within the Swan Pond River Estuary within the Town of Dennis, MA, based upon assessments in Section VII. Swan Pond is a coastal salt pond connected to Nantucket Sound by the Swan Pond River, both of these estuarine components support significant salt marsh resources. DWQMP is the Dennis Water Quality Monitoring Program.

Health Indicator	Swan Pond River Embayment System		
	Swan Pond River		Swan Pond
	Upper River	Lower River	
Dissolved Oxygen	MI-SI[1]	MI[2]	SI[3]
Chlorophyll	SI[4]	MI-SI[5]	SD[6]
Macroalgae	H-MI[7]	H-MI[7]	SD[8]
Eelgrass	--[9]	SI[10]	--[9]
Infaunal Animals	MI-SI[11]	H-MI[12]	SI-SD[13]
Overall:	MI-SI[14]	MI[15]	SI-SD[16]

1- primarily a tidal river with extensive salt marsh which adds a natural component to its organic enrichment, low D.O. primarily related to low oxygen ebb waters from Swan Pond, DWQMP: DO min. all stations (2005-10) = <0.2 mg/L, <2 mg/L 4%, <4 mg/L ~15% of 30 dates

2 - a tidal river with fringing salt marsh, moderate-low oxygen declines primarily related to up-gradient low oxygen ebb waters, DWQMP (2005-10): D.O.<4 mg/L 11%, <5 mg/L 40% of 30 dates

3 - oxygen depletions frequent: D.O.<4 mg/L 18%, <3 mg/L 11% of record, periodically <1 mg/l; DWQMP: D.O. min. all stations (2005-10) = <0.2 mg/L, central basin <2 mg/L 7%, <4 mg/L 20% of 30 dates

4 - very high summer chlorophyll a levels, averaging 20 ug/L (DWQMP, 2002-2008)

5 - moderate to high chlorophyll a levels, DWQMP average 2005-10 = 10-15 ug/L

6 - eutrophic chlorophyll a levels, average = 15.3 ug/L, bloom >100 ug/L; DWQMP: mean 2005-2010 = 29 ug/L

7 - generally sparse, but with patches of drift algae, *Ulva*, with some filamentous species.

8 - dense patches of drift algae, *Ulva*, with some filamentous species, small patches of SAV, Ruppia (which is common to salt marsh ponds) consistent with the salt marsh influences in this system.

9- no evidence this basin is supportive of eelgrass.

10- MassDEP (C. Costello) indicates that eelgrass beds lost from this reach of tidal river between 1951-1995.

11- high numbers of individuals, low-moderate number of species (12), low numbers of stress indicator species, dominated by organic enrichment species; low diversity (1.8) and Evenness (0.52).

12- moderate-high numbers of individuals and species (25), stress indicator species (~20% of population), high diversity (3.3) and Evenness (0.7), community of polychaetes, crustaceans & mollusks, some deep burrowers.

13- high numbers of individuals (>1000), low number of species (8) dominated by a few opportunistic species, the stress indicator species *Capitella* ~50% of population, very low diversity (1.6) and Evenness (0.56).

14- Moderate to Significant Impairment , primarily due to sustained high chlorophyll levels & periodic D.O. depletion. Dominated by outflows of low D.O., high organic matter waters from Swan Pond, with some natural enrichment from associated wetlands.

15-Significant Impairment based upon loss of eelgrass from system, 1951-1995. Infauna habitat with some impairment as stress indicator species (~20% of population) and moderate-high numbers of individuals, moderate-high number of species (25), but with high diversity (3.3) and Evenness (0.7) and community composed of polychaetes, crustaceans and mollusks, some deep burrowers.

16-Significant Impairment to Severely Degraded: low numbers of species & high numbers of individuals, low diversity (H') & Evenness (E), dominated by stress indicator species (*Capitella*) and organic enrichment tolerant species, oxygen stress with periodic anoxia, very high phytoplankton biomass, large macroalgal accumulations.

DWQMP: Dennis Water Quality Monitoring Program (2005-2010)
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation;   -- = not applicable to this estuarine reach

The habitat assessment data are also internally consistent. Overall, the oxygen and chlorophyll data for the Swan Pond River Estuary clearly indicate a system supporting sub-tidal habitats impaired by nitrogen, ranging from highly stressed (Swan Pond) to moderately stressed (Lower River). These observations are consistent with the high levels of total nitrogen (TN) throughout the estuary. The gradient in impairment follows the gradient in nitrogen enrichment, where Swan Pond has very high tidally averaged TN levels (1.06-1.21 mg L^{-1}) declining to the Lower River nearest the tidal inlet (0.47-0.66 mg L^{-1}). While the lower River supports lowest nitrogen levels within the system, the levels are still quite high and indicate a basin incapable of supporting eelgrass beds and with a moderate level of impairment to benthic animal habitat (see Sections VII-3 & VII-4).

The observed loss of eelgrass, moderate oxygen and chlorophyll levels and benthic community structure within the lower reach of the Swan Pond River, indicate a system beyond the nitrogen threshold level that would be supportive of eelgrass, but relatively close to the level for supporting high quality infaunal habitat. The tidally averaged nitrogen levels for this lower reach were 0.651 mg N L^{-1} over the entire reach and 0.564 mg N L^{-1} in the region that historically supported eelgrass beds. These TN levels are slightly above the level typically supportive of infaunal communities (0.5-0.55 mg N L^{-1}), but well above levels supportive of eelgrass beds as found in a variety of MEP assessments of Cape Cod estuaries. Similarly, average total nitrogen levels within Swan Pond (1.056-1.207 mg N L^{-1}) are well above levels found in basins supportive of high quality benthic animal habitat. Parallel measurements made in Swan Pond of oxygen depletion and very high chlorophyll a levels stemming from frequent phytoplankton blooms, as well as accumulations of drift macroalgae, are all consistent with a basin significantly impaired by nitrogen enrichment. It is clear that a significant reduction in nitrogen loading or increase in tidal flushing (or both) will be required for restoration of Swan Pond, the main embayment basin in the Swan Pond River Estuary.

The results of the water quality and infaunal surveys, coupled with the temporal trends in eelgrass coverage, clearly supports the need to lower nitrogen levels within the lower reach of the Swan Pond River to restore eelgrass habitat. Based on all indicators, the lowering of nitrogen levels will also be necessary to restore infaunal habitat within Swan Pond and to a lesser extent within the upper reach of the River. It is likely that restoration of the impaired infaunal habitats within Swan Pond and the Swan Pond River will be achieved with the restoration of eelgrass habitat within the lower reach of the River.

The eelgrass and water quality information supports the conclusion that eelgrass beds within the lower reach of the Swan Pond River should be the primary target for restoration of the Swan Pond River Embayment System and that restoration requires appropriate nitrogen management. From the historical analysis, it appears that while only a modest acreage of eelgrass can be restored (all in the lower reach), achieving its restoration will be coupled with restoration of large areas of severely degraded benthic animal habitat within the upper estuary (above Rt. 28), more than 150 acres in Swan Pond alone as well as reduced oxygen depletion that presently cause periodic fish kills. Therefore, the sentinel station for the Swan Pond River Estuary is located at the long-term water quality monitoring station within middle of the lower reach of the River (SWP-2). This site was selected based upon its location at the upper most extent of the documented eelgrass coverage in this estuary (Figure VII-5).

With the sentinel station located at the uppermost extent of the historical eelgrass coverage, the target nitrogen concentration (tidally averaged TN) for restoration of eelgrass at the sentinel location within the lower reach of the Swan Pond River was determined to be 0.40 mg TN L^{-1}. As there has not been eelgrass habitat within the Swan Pond River Estuary for over

a decade, this threshold was based upon comparison to other local embayments of similar depths and structure that have been reviewed under MEP analysis. The historic Swan Pond River eelgrass habitat appears to have been patchy and like other similar basins, found mainly within the areas of more stable sediments.

The threshold for eelgrass restoration in Swan Pond River is similar to those selected by the MEP for nearby systems like the Bournes Pond Estuary, where eelgrass has historically been confined to the lower estuarine basin (main open water stem of the channel) at TN levels of 0.42 mg TN L^{-1}, although at a shallower depth than the channel of lower Swan Pond River. Similarly other MEP observations found that the lower reach of the Green Pond Estuary, supports a sparse (slowly declining) eelgrass "bed" at tidally averaged TN levels of 0.41 mg TN L^{-1}, while the region near the inlet to Waquoit Bay eelgrass patches persist at 0.395 mg N L^{-1}. Given the depth of the lower Swan Pond River a lower threshold than Bournes Pond and about the same as the patches in Waquoit Bay is appropriate. It should be noted that this threshold targets eelgrass habitat throughout the lower reach of the River, not just the shallow fringing areas.

Although the nitrogen management target is restoration of eelgrass habitat (and associated water clarity, shellfish and fisheries resources), benthic infaunal habitat quality must also be supported as a secondary condition. Benthic animals are more tolerant of nutrient and organic matter enrichment than eelgrass, which requires clear waters and high oxygen levels. At present, the regions with moderately to significantly impaired infaunal habitat within the Swan Pond River Embayment System have average tidal total nitrogen (TN) levels of 0.66 to 1.21 mg N L^{-1}. The observed impairments throughout this estuary are consistent with observations by the MEP Technical Team in other estuaries along Nantucket Sound (e.g. Perch Pond, Bournes Pond, Popponesset Bay) where levels <0.5 mg N L^{-1} were found to be supportive of healthy infaunal habitat and where moderately impaired habitat was found at ~0.6 mg N L^{-1}. Similarly, moderate impairment was also observed at TN levels (0.535-0.600 mg N L^{-1}) within the Wareham River Estuary, while the Centerville River system showed moderate impairment at tidally averaged TN levels of 0.526 mg N L-1 in Scudder Bay and at 0.543 mg TN L-1 in the deep middle reach of the Centerville River.

Based upon these observations, the MEP Technical Team concluded that an upper limit of 0.50 mg N L-1 tidally averaged TN would support healthy infaunal habitat in the Swan Pond River, but given the shallow nature of Swan Pond and its significant salt marsh resources, a tidally averaged TN of <0.55 mg N L-1 was appropriate. This higher threshold in Swan Pond is similar those selected for Lewis Pond in Parker's River and the upper reach of the Mashpee River and is only slightly higher than the non-wetland influenced basins of the upper Bass River. Since the goal is restoration throughout Swan Pond, the benthic animal restoration TN level targets the pond-wide tidally averaged TN level (average long-term monitoring stations, SWP-5,6,7,8).

It should be emphasized that these secondary criteria values were not used for setting nitrogen thresholds in this embayment system. These values merely provide a check on the acceptability of conditions within the tributary basins at the point that the threshold level is attained at the sentinel station. The results of the Linked Watershed-Embayment modeling are used to ascertain that when the nitrogen threshold is attained, TN levels in these regions are within the acceptable range. The goal is to achieve the nitrogen target at the sentinel location and restore eelgrass habitat within the lower reach of the Swan Pond River and restore infaunal habitat throughout the Swan Pond River Embayment System.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were in turn used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Swan Pond River System. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for the Swan Pond River System (SWP-2 is located approximately at the upper limit of historic eel grass within the system). A secondary threshold (check) was set at the head of the system to restore benthic animal habitat, the secondary threshold averages the measured concentrations at stations SWP-5, SWP-6, SWP-7, and SWP-8. 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. Community discussions should review this option and consider evaluation of other alternatives. The presentation below 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 required removal of 100% of the septic nitrogen load from all the watersheds within the Swan Pond River basin (100% of the septic nitrogen load removed from watersheds 1, 2, 3, 4, 5, 6, 7, and 8). 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 Swan Pond River 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
Swan Pond	13.058	0.000	-100.0%
Swan Pond River - North	8.411	0.000	-100.0%
Swan Pond River - South	11.518	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 all septic loads from Swan Pond River watersheds results in the total nitrogen loads presented in Table VIII-4. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent 'worst-case' summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Nantucket Sound.
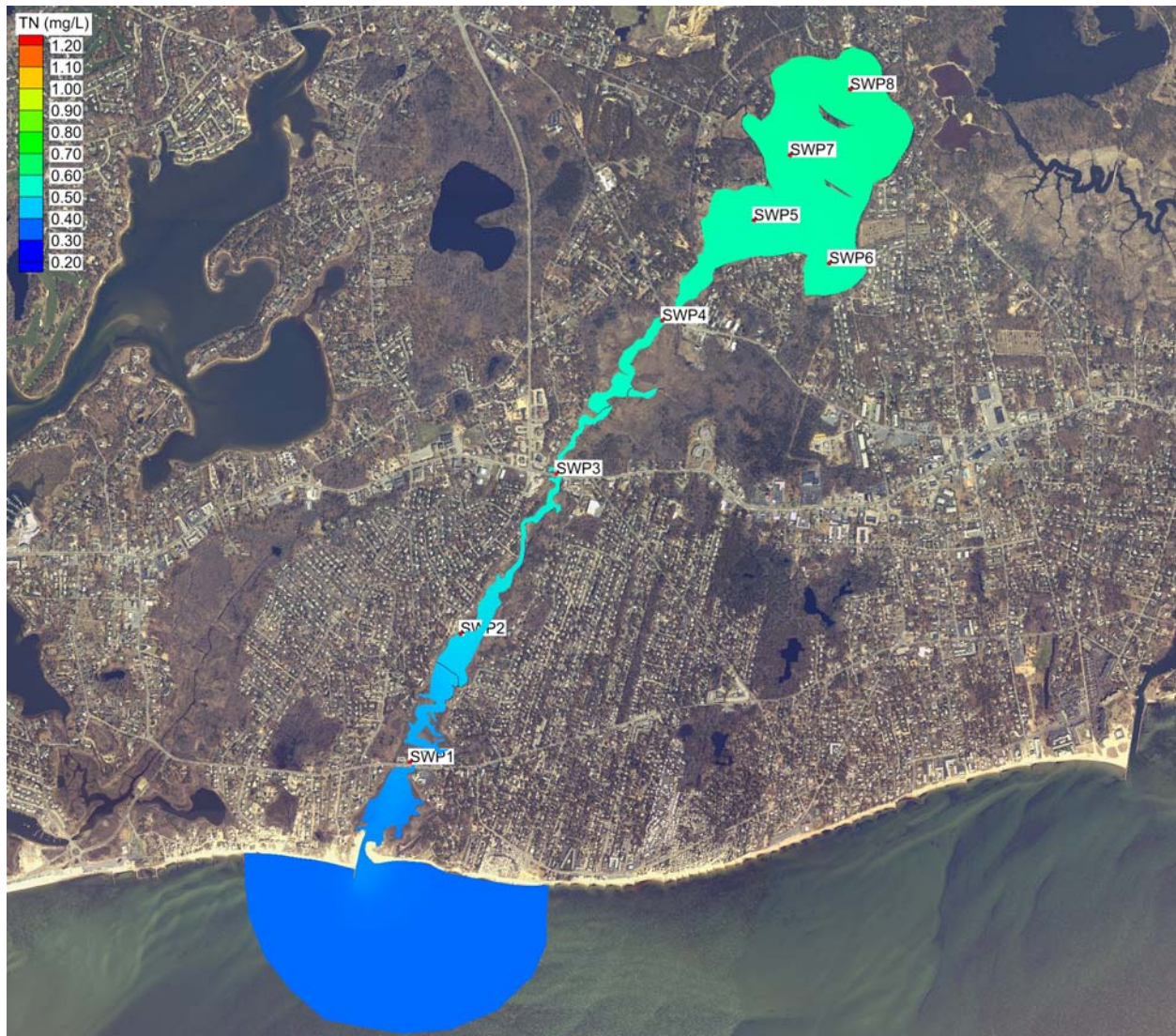
Figure VIII-1.   Contour plot of modeled average total nitrogen concentrations (mg/L) in Swan Pond River system, for threshold conditions (0.40 mg/L at water quality monitoring station SWP-2, with a secondary threshold of 0.55 mg/L averaged from the water quality monitoring stations in Swan Pond). The approximate location of the sentinel threshold station for Swan Pond River (SWP-2) is shown.

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 Swan Pond River system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.		
sub-embayment	present total load (kg/day)	threshold load (kg/day)	threshold % change
Swan Pond	18.123	5.063	-72.1%
Swan Pond River - North	10.036	1.625	-83.8%
Swan Pond River - South	15.556	4.038	-74.0%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Swan Pond River system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	threshold load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Swan Pond	5.063	1.885	-1.185
Swan Pond River - North	1.625	0.104	0.150
Swan Pond River - South	4.038	0.226	-0.586

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel stations is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station within the lower reach of the Swan Pond River, a reduction in TN concentration of approximately 40% was required at the long-term monitoring station SWP-2 (Sentinel Station).

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Swan Pond River system. Sentinel threshold station, SWP-2, to restore eelgrass habitat within the lower tidal river is in bold print. The nitrogen level to restore benthic animal habitat in Swan Pond is <0.55 mg TN L-1, as the average of SWP-5, 6, 7, 8.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Lower River	SWP-1	0.465	0.346	-25.6%
Lower River	**SWP-2**	**0.661**	**0.398**	**-39.8%**
Upper Swan River	SWP-3	0.826	0.445	-46.1%
Swan Pond	SWP-5	1.055	0.515	-51.2%
Swan Pond	SWP-6	1.143	0.544	-52.4%
Swan Pond	SWP-7	1.168	0.553	-52.6%
Swan Pond	SWP-8	1.210	0.569	-53.0%

The basis for the watershed nitrogen removal strategy utilized to achieve the embayment thresholds may have merit, since this example nitrogen remediation effort is focused on watersheds where groundwater is flowing directly into the estuary. For nutrient loads entering the systems through surface flow, natural attenuation in freshwater bodies (i.e., streams and ponds) can significantly reduce the load that finally reaches the estuary. Presently, this attenuation is occurring due to natural ecosystem processes and the extent of attenuation being determined by the mass of nitrogen which discharges to these systems. The watershed nitrogen reaching these systems is currently "unplanned", resulting primarily from the widely distributed non-point nitrogen sources (e.g. septic systems, lawns, etc.). Future nitrogen management should take advantage of natural nitrogen attenuation, where possible, to ensure the most cost-effective nitrogen reduction strategies. However, "planned" use of natural systems for their nitrogen attenuation has to be done carefully and with the full analysis to ensure that degradation of these systems will not occur. One clear finding of the MEP has been the need for analysis of the potential to enhance nitrogen attenuation in restored wetlands or

ecologically engineered ponds/wetlands. Attenuation by ponds in agricultural systems has also been found to work in some cranberry bog systems, as well. Cranberry bogs, other freshwater wetland resources, and freshwater ponds provide opportunities for enhancing natural attenuation of their nitrogen loads.   Restoration or enhancement of wetlands and ponds associated with the lower ends of rivers and/or streams discharging to estuaries are seen as providing a dual service of lowering infrastructure costs associated with wastewater nitrogen management and increasing the watershed and upper estuarine benefits that these types of aquatic resources provide.

Although the above modeling results provide one manner of achieving the selected threshold level for the sentinel site within the estuarine system, the specific example does not represent the only method for achieving this goal. However, the thresholds analysis provides general guidelines needed for the nitrogen management of this embayment.

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