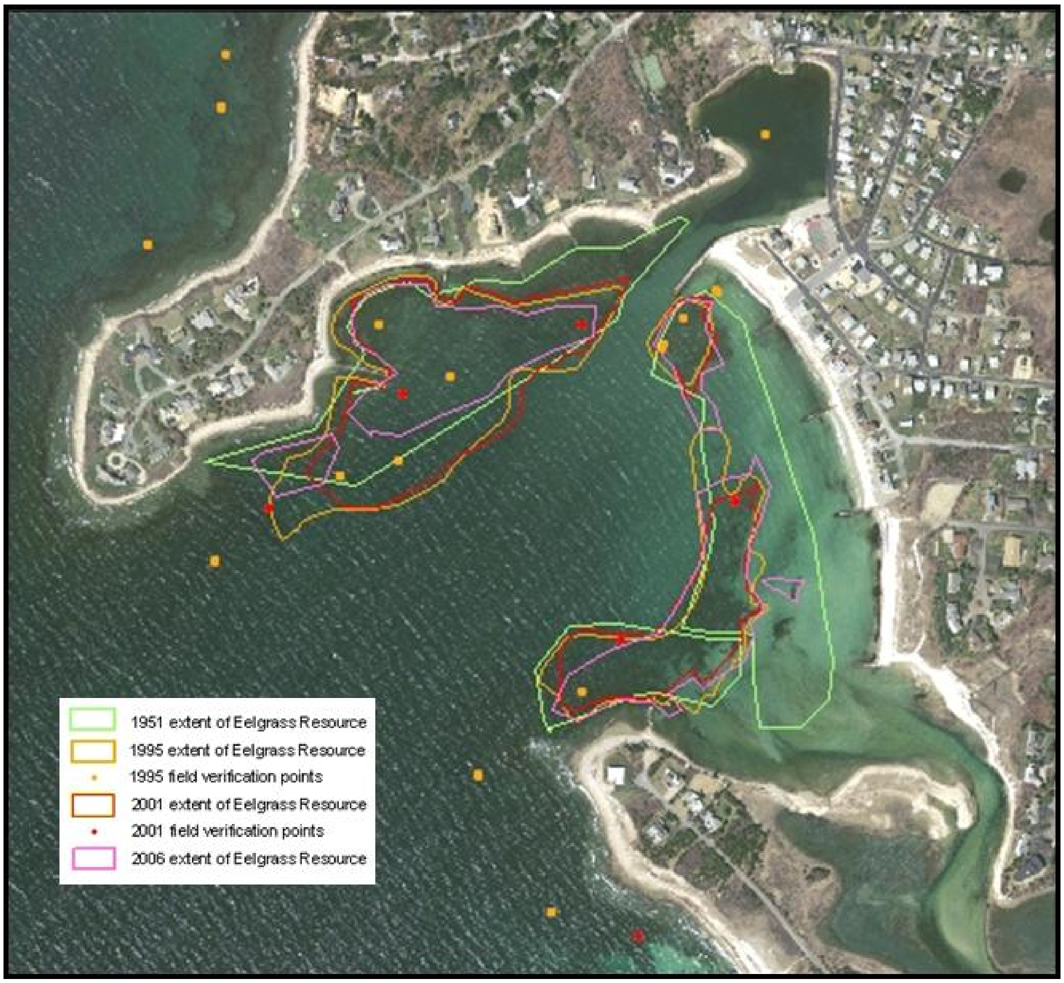
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Wild Harbor Embayment System Town of Falmouth, Massachusetts



University of Massachusetts Dartmouth
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Massachusetts Department of
Environmental Protection

FINAL REPORT – March 2013

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Massachusetts Estuaries Project

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Executive Summary

1. Background

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

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

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

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

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

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

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

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

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

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

The 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 Wild Harbor embayment system by using site-specific data collected by the MEP and water quality data from the Falmouth PondWatch Program (see Chapter 2) as well as the Coalition for Buzzards Bay (CBB) BayWatchers Program (assisted technically until 2008 by the University of Massachusetts-SMAST Coastal Systems Program). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Town of Falmouth Planning Department, and watershed boundaries delineated by USGS. This land-use data was used to determine watershed nitrogen loads within the Wild Harbor embayment system and the systems sub-embayments as appropriate (current and build-out loads are summarized in Table IV-3). Water quality within a sub-embayment is the integration of nitrogen loads with the site-specific estuarine circulation. Therefore, water quality modeling of this tidally influenced estuary included a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Once the hydrodynamics of the system was quantified, transport of nitrogen was evaluated from tidal current information developed by the numerical models.

A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the Wild Harbor 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 Buzzards Bay source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Wild Harbor embayment system was used to calibrate the water quality model, with validation using measured nitrogen concentrations (under existing loading conditions). The underlying hydrodynamic model was calibrated and validated independently using water elevations measured in time series throughout the embayments.

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

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

The 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 Wild Harbor 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 Wild Harbor 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 Wild Harbor embayment system in the Town of Falmouth. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. For Illustrative purposes, the MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of the embayment system. The concept was that since nitrogen loads associated with wastewater generally represent ~72% of the controllable watershed load to the Wild Harbor embayment system and are more manageable than other of the nitrogen sources, the ability to achieve needed reductions through this source is a good gauge of the feasibility for restoration of these systems.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout the Wild Harbor 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 Wild Harbor Estuary is beyond its ability to assimilate nitrogen without impairment and is showing a low to moderate level of nitrogen enrichment, with some moderate impairment of both eelgrass in the main basin and significant impairment of infaunal habitats in the inner Boat Basin (Table VIII-1), indicating that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system. In general, the habitat quality within the basins of this System is manifested by the temporal changes in eelgrass coverage and benthic community characteristics, which are consistent with the observed levels of nitrogen and organic matter enrichment and magnitude of oxygen depletion, as well as the sediment characteristics and general absence to only sparse macroalgal accumulations. The distribution and levels of habitat impairment within the Wild Harbor Embayment System is consistent with the low to moderate level of nitrogen enrichment. The Wild Harbor Embayment System presently shows a moderate impairment to eelgrass habitat within its outer basin, the main basin of Wild Harbor. The impairment is based upon the recent temporal trend in loss of eelgrass from the inner margin of the basin, at the inner Boat Basin boundary and loss at the deeper margin of the Nyes

Neck beds. Both the location and the temporal trend are consistent with nitrogen enrichment. However, as the rate of loss has been gradual and significant eelgrass resources still exist, that indicates that this estuarine basin is only just beyond its nitrogen threshold (i.e. the level of nitrogen a system can tolerate without impairment). The presence of stable dense eelgrass beds throughout the main basin of Wild Harbor and the generally high quality benthic animal habitat throughout the embayment system (except for the Boat Basin) also indicates a system just beyond its threshold.

It is significant that there is no evidence of eelgrass beds in the Wild Harbor River or in the Wild Harbor Boat Basin over the past 60 years. The Wild Harbor Boat Basin is an artificial embayment basin that is relatively deep and depositional, as evidenced by its organic enriched sediments. Eelgrass is not expected in the salt marsh dominated Wild Harbor River, which is naturally nutrient and organic matter enriched and becomes very shallow at low tide. The absence of evidence that the Wild Harbor Boat Basin and Wild Harbor River have ever supported eelgrass habitat, focuses management on benthic infaunal communities.

Overall, the infauna survey indicated that the main basin of Wild Harbor is supporting high quality benthic animal habitat with high diversity and Evenness, although the inner region is potentially moderately impaired. The infauna results were consistent with the eelgrass and water quality assessments. Similarly the Wild Harbor River is functioning as a non-nitrogen impaired salt marsh system with productive benthic communities typical of Cape Cod marsh creeks. In contrast, the enclosed Boat Basin presents a significantly impaired habitat for infaunal communities with low numbers of species and individuals as well as low diversity. The basin morphology tends to create a depositional environment, which increases the sensitivity to organic enrichment. The sediments are depositional, comprised of organic rich soft sulfidic mud. The sediment surface is overlain by an algal mat, which further reduces habitat quality. These sediment characteristics coupled with the periodic oxygen depletions to 3 mg L^{-1} and moderate chlorophyll levels are consistent with a benthic habitat significantly impaired by nitrogen enrichment. The present animal community is also indicative of a significantly impaired habitat, being dominated by stress tolerant species indicative of high levels of organic enrichment. About 50% of the total community consisted of a single species, *Capitella capitata*, generally found in embayments with high organic matter deposition and poor habitat quality. The levels of phytoplankton biomass, the levels of oxygen decline (3 mg L^{-1}) and the organic enrichment of the sediments are all clear evidence of nitrogen as the ultimate cause of the habitat impairment. Integration of all of the key metrics clearly indicates that the main basin of Wild Harbor is generally supporting high quality benthic animal habitat, while the Boat Basin is beyond its capacity to assimilate nitrogen loads without impairment (i.e. its just beyond its nitrogen threshold). Since Wild Harbor is also beyond its nitrogen threshold to support healthy eelgrass habitat, a slight reduction to enhance this habitat should also restore the moderate impairment of the benthic animal habitat within the inner region of this basin as well as within the Boat Basin.

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. The spatial pattern indicated that the magnitude of oxygen depletion, enhancement of chlorophyll a levels and total nitrogen concentrations increased from the offshore waters to the main basin of Wild Harbor and were highest within the inner Boat Basin.

Oxygen records obtained from both the moorings deployed in the Wild Harbor System show ecologically significant oxygen depletion in the inner Boat Basin, but oxygen levels in Wild

Harbor and Wild Harbor River were consistent with high quality habitat for these embayment and salt marsh creek sub-systems. The oxygen conditions in the inner versus outer embayment basins, are consistent with their basin structure, flushing, and nitrogen and chlorophyll a levels. Oxygen levels in the main basin of Wild Harbor generally showed only small to moderate depletion, with levels never dropping below 5 mg L^{-1} and almost always >5.5 mg L^{-1}. Outer basin oxygen conditions did exhibit daily excursions in oxygen levels, but the range of daily oxygen excursion was moderate and significantly less than that observed in the Wild Harbor Boat Basin, furthest from the low nutrient and high oxygen waters of Buzzards Bay water. The oxygen dynamics at both sites is a clear indication of nitrogen enrichment, but only the inner basin is showing declines stressful to benthic animal communities.

Although there was not an oxygen mooring in Wild Harbor River, the water quality monitoring program collected 129 oxygen reading at the mouth of the tidal river near the end of ebb tide. These data showed no reading below 3 mg L^{-1}, 2% <4 mg L^{-1}, and 87% >5 mg L^{-1}. Similarly, the long term average chlorophyll a (5.8 ug L^{-1}) is also in line with ebbing salt marsh waters. These moderate levels of oxygen depletion and chlorophyll a, coupled with the salt marsh dominance of the Wild Harbor River, support the contention that this tidal river is not impaired by nitrogen loading.

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 Falmouth Wild Harbor embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 72% of the controllable watershed nitrogen load to the embayment was from wastewater.

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

The threshold nitrogen levels for the Wild Harbor embayment system in Falmouth were determined as follows:

Wild Harbor Threshold Nitrogen Concentrations

- Following the MEP protocol, the restoration target for the Wild Harbor system should reflect both recent pre-degradation habitat quality and be reasonably achievable. Based upon the assessment data (Chapter VII), the Wild Harbor system is presently supportive of habitat in varying states of impairment, depending on the component sub-basins of the overall system but overall is only showing signs of moderate to low impairment.

- The Wild Harbor Embayment System presently shows a moderate impairment to eelgrass habitat within its outer basin, the main basin of Wild Harbor. The impairment is based upon the recent temporal trend in loss of eelgrass from the inner margin of the basin, at the inner Boat Basin boundary and loss at the deeper margin of the Nyes Neck beds. Both the location and the temporal trend are consistent with nitrogen enrichment. However, as the rate of loss has been gradual and significant eelgrass resources still exist, that indicates that this estuarine basin is only just beyond its nitrogen threshold (i.e. the level of nitrogen a system can tolerate without impairment).

- The presence of stable dense eelgrass beds throughout the main basin of Wild Harbor and the generally high quality benthic animal habitat throughout the embayment system (except for the Boat Basin) also indicates a system just beyond its threshold. The indication of impairment to eelgrass and infaunal animal habitat, to the extent that it was observed, is supported by the observed levels of oxygen depletion and clearly enhanced chlorophyll a levels in the inner Boat Basin waters. The spatial distribution of high quality and impaired habitats and the associated oxygen and chlorophyll a levels also parallels the gradient in watercolumn total nitrogen levels within this estuary.

- As eelgrass within the Wild Harbor Embayment System is a critical habitat structuring the productivity and resource quality of the entire system, and given that it is presently showing moderate impairment, restoration of this resource is the primary target for overall restoration of this system. Nutrient management planning for restoration of the eelgrass habitat at the boundary between the inner region of Wild Harbor and the Boat Basin should focus on reducing the level of nitrogen enrichment in embayment waters primarily through watershed nitrogen management within the Wild Harbor watershed. The loss of eelgrass within the main basin of Wild Harbor related to the beds along Nyes Neck is associated with a tidally averaged nitrogen (total nitrogen, TN) level of 0.447 mg N L^{-1} at the sentinel station (WH-1), while the high quality eelgrass habitat exist at lower TN levels, <0.35 mg N L^{-1}. These TN levels and habitat stability/decline are consistent with persistence and loss of eelgrass at similar depths in other estuaries in southeastern Massachusetts. .

- Based upon the present TN levels in the Wild Harbor System and comparison to other similar estuaries a threshold for tidally averaged TN at the sentinel station at the boundary of the inner and outer sub-embayment basins (WH-1) of 0.35 mg L^{-1} was selected. This threshold is similar to that for West Falmouth Harbor and Phinneys Harbor at similar depths, and is focused in part on restoring eelgrass where it had persisted until recently near the tidal inlet to the Boat Basin. In addition, lowering the level of nitrogen enrichment at the sentinel station will lower nitrogen levels within the Boat Basin and inner region of Wild Harbor with the parallel effect of improving impaired infaunal habitat.

It is important to note that the analysis of future nitrogen loading to the Wild Harbor estuarine system focuses upon additional shifts in land-use from forest/grasslands to residential and commercial development. However, the MEP analysis indicates that significant increases in nitrogen loading can occur under present land-uses, due to shifts in occupancy, shifts from seasonal to year-round usage and increasing use of fertilizers. 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 Wild Harbor estuarine system is that restoration will necessitate a reduction in the present (Falmouth 2009, Bourne 2008, Sandwich 2010) nitrogen inputs and management options to negate additional future nitrogen inputs.

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

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
Wild Harbor	0.323	2.827	7.499	--	10.326	1.033	-11.361	-0.002	0.34-0.52	0.35
Wild Harbor River	0.556	3.014	8.811	--	11.825	0.447	-0.423	11.848	0.41-0.60	--
Dam Pond Stream	0.178	0.455	1.052	--	1.507	--	--	1.507	--	--
Combined Total	**1.057**	**6.296**	**17.362**	**0.000**	**23.658**	**1.480**	**-11.784**	**13.353**	**0.34-0.60**	**0.35[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 1999 – 2009 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 is located at the Wild Harbor water quality station WH-1.

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Wild Harbor estuarine system in Falmouth, Massachusetts.

Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed reductions needed to achieve threshold load levels
Wild Harbor	10.326	4.552	1.033	-10.232	-4.647	-55.9%
Wild Harbor River	11.825	10.062	0.447	-0.359	10.150	-14.9%
Dam Pond Stream	1.507	1.507	--	--	1.507	0.0
Combined Total	**23.658**	**16.121**	**1.480**	**-10.591**	**7.010**	**-31.9%**

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

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

First and foremost we would like to recognize and applaud the commitment shown by the Town of Falmouth in carrying forward with the Massachusetts Estuaries Project as part of its watershed management planning and for their commitment to the restoration of all of their estuaries. Significant time and attention has been dedicated to this effort by Jerry Potamis and Amy Lowell, whose support has been instrumental to completion of these reports. Equally important has been technical support by the Town Planner, Brian Currie. We would also like to acknowledge the field support provided to the MEP by Brewer Fiddler's Cove Marina who gave us unrestricted use of the facility in support of critical field tasks. The MEP Technical Team would also like to thank Tony Williams of the Coalition for Buzzards Bay for providing the water quality monitoring baseline for this system. Without this baseline water quality data the present analysis would not have been possible. We also would like to recognize the Community Preservation Committee (CPC), the nutrient management committee and the CWMP review committee for the Town of Falmouth, in moving this MEP analysis forward to support estuarine management of one of the signature estuaries in the Town of Falmouth.

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I. INTRODUCTION

The Wild Harbor Estuarine System is located within the Town of Falmouth, on Cape Cod Massachusetts. This system is located on the eastern shore of Buzzards Bay between Megansett Harbor and West Falmouth Harbor. The Wild Harbor system is a complex estuary comprised of a large outer basin constrained by Nyes Neck to the north and Crow point to the south, a small inner basin and a significant salt marsh (Figure I-1). The small inner basin is tributary to the main outer basin of Wild Harbor along its northeastern shore. The inner basin is the main mooring area in the System with ~100 boat moorings and other boating activities, many associated with the Wild Harbor Yacht Club. The tidal wetland exchanges water through the southwestern shore of the outer basin of Wild Harbor near Crow Point and has developed within the estuarine portion of the Wild Harbor River. The Wild Harbor River contains most of the 110 acres of salt marsh contained within the Wild Harbor System. The unarmored inlet to Wild Harbor River is dynamic with rapidly changing sand shoals at its mouth.

The developed regions of the watershed to the Wild Harbor embayment system is distributed almost entirely within the Town of Falmouth with the exception that the uppermost portion of the watershed within the Massachusetts Military Reservation (MMR) falls within the Towns of Falmouth, Sandwich and Bourne. This upper watershed within the MMR (~1/4 of watershed) is mainly undeveloped and developed areas are on sewer, so it is not contributing a significant nitrogen load to the estuary. As a result, the primary stakeholder for the management and restoration of the Wild Harbor System is the Town of Falmouth.

The Wild Harbor System is one of the Town of Falmouth's moderately sized marine resources, however, it does not support an active marina as is found in nearby Fiddlers Cove. As such, boating activity is generally light during the summer months with the exception of small boat activity in the Wild Harbor boat basin associated with the Wild Harbor Yacht Club. At a time when many other coastal ponds and bays tributary to Buzzards Bay have been degraded by nutrients, water quality in Wild Harbor has generally remained moderately high due to its relatively small basins, tidal exchange with the high quality waters of Buzzards Bay and its relatively small watershed with significant undeveloped areas. However, portions of Wild Harbor associated with the Inner Basin have shown indications of nutrient enrichment. It should be noted that while Wild Harbor is presently beginning to have its marine resources damaged by nutrient enrichment, it was previously severely impacted by oil contamination.

In September 1969 the barge *Florida,* carrying no. 2 fuel oil went aground off West Falmouth spilling 180,000 gallons of fuel oil into Buzzards Bay. Rising winds drove the spill into Wild Harbor and its shores and marshes within the Wild Harbor River. Much of the oil settled along a narrow band in the Wild Harbor Marsh and Wild Harbor boat basin (inner basin). Oil reached the sediments of the embayment basins initially covering several square kilometers and 6.4 km of coastline. Subsequent storms and tidal currents expanded the contaminated sediment footprint several fold. Flooding tides carried the surface oil up onto the salt marsh areas where it entered the sediments and coated the marsh surface and plants. The initial impact on the harbor systems was the loss of benthic animals in contaminated areas and death of the salt marsh vegetation. After 4 years, the spill was still evident in invertebrates, fish and birds in the heavily oiled areas. The inner basin was still heavily contaminated 5 years post-spill and its animal populations reduced in abundance and dominated by opportunistic species. However, 20 years post-spill the majority of oil was "gone" and in the sub-tidal sediments the spilled oil was virtually all gone. Equally important, even in the salt marsh areas most heavily oiled in 1969, vegetation and animal populations appeared to have recovered (Teal 1993).

Fortunately, now more than 30 years later, the impact on the biological communities is not easily observed, although it is still possible to detect oil in the heavily oiled salt marsh areas at depth (Culbertson 2008). Now over 40 years later, while degraded oil is still present in some of the sediments of the Wild Harbor River salt marsh sediments, the Wild Harbor basins in general have fully recovered from the effects of the spill and no longer show signs of oil contamination. This recovery attests to the long-term impacts of oil contamination, particularly in salt marshes, and the resilience of estuaries to recover from stressors once the source of the stress has ceased. As such, signs of impairment in this system are not necessarily attributable to the oil spill any longer.

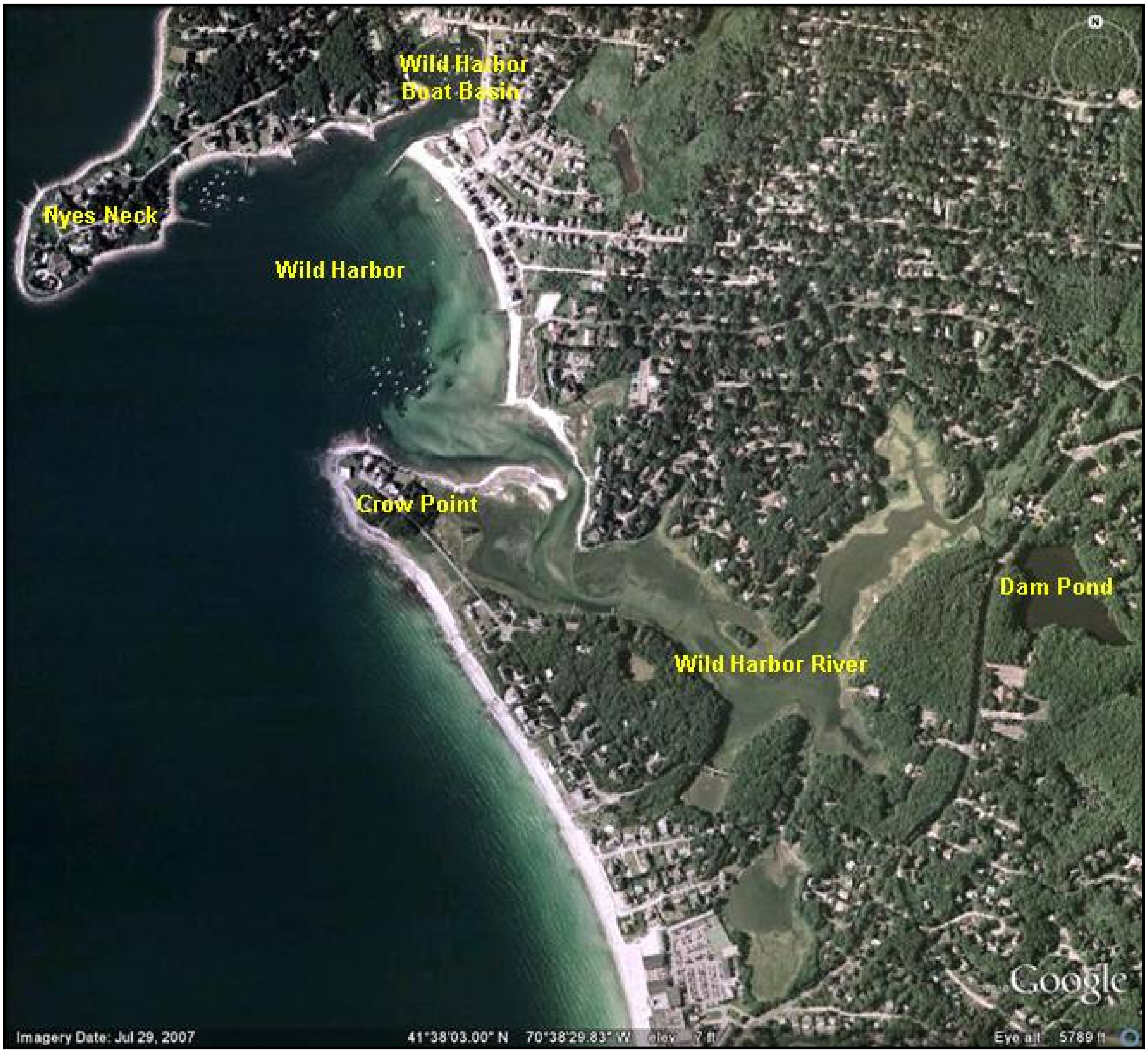

Figure I-1. Wild Harbor study region for the Massachusetts Estuaries Project assessment. Tidal waters enter the main basin, Wild Harbor, directly from Buzzards Bay with a portion of the tidal volume passing into the Wild Harbor River or into the Wild Harbor Boat Basin. Freshwaters primarily enter the headwaters of the Wild Harbor River from its watershed through a stream flowing from Dam Pond and to a lesser extent via direct groundwater discharge. Freshwater enters the other sub-embayments primarily through direct groundwater discharge.

The Wild Harbor System is a complex estuarine system composed of a large and small sub-embayment and a tidal river dominated by salt marsh. The habitat quality of Wild Harbor and its tributaries is linked to the level of tidal flushing through the inlets connecting Wild Harbor River and the boat basin to outer Wild Harbor and ultimately Buzzards Bay, which exhibits a moderate tide range of ~5 ft. It is the tide range in Buzzards Bay which is the primary driving force for tidal exchange (i.e. the volume of water flushed during a tidal cycle). BY comparison, the tide range off Stage Harbor Chatham is ~4.5 ft and in Wellfleet Harbor it is ~10 ft. In addition to tide range, the degree to which the inlets remain unobstructed also plays a critical role in the volume of water exchanged and the health of each estuarine component to the Wild Harbor System. This is particularly the case with the Wild Harbor River, as its inlet is not armored and sediment transport at the mouth is very dynamic and the area is prone to shoaling. Wild Harbor River acts as a mixing zone for terrestrial freshwater inflow and saline tidal flow from Buzzards Bay via outer Wild Harbor, however, the salinity characteristics of the system varies with the volume of freshwater inflow as well as the effectiveness of tidal exchange with outer Wild Harbor. Overall, the small freshwater contributing area and large tide range result in a relatively high average salinity (>27ppt) within Wild Harbor River.

Given the present hydrodynamic characteristics of the Wild Harbor embayment system, it appears that estuarine habitat quality is mostly dependent on the level of nutrient loading to embayment waters, rather than tidal restrictions. In Wild Harbor, specifically Wild Harbor River, in addition to nitrogen inputs, it is important to maintain tidal flows to the extent that they may be restricted by sediment deposition at the river's mouth to maintain habitat quality.

The watershed to the Wild Harbor System is somewhat geologically complex, being composed primarily of Falmouth Moraine and sand and gravel outwash glacial deposits. The watershed to the Wild Harbor River, is comprised primarily of bouldery glacial drift deposits of the Falmouth Moraine, while the sub-watershed soils to Wild Harbor and the Wild Harbor Boat Basin are mainly comprised of sand and gravel outwash from the Falmouth Moraine. These formations consist of material deposited during the retreat of the Cape Cod Lobe of the Laurentide Ice sheet. The material is highly permeable and as such, direct rainwater run-off is typically rather low for this type of coastal system. Therefore, most freshwater inflow to the estuary is via groundwater discharge, with some surface water discharge to the Wild Harbor River. Originally the Wild Harbor component basins 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. The main basin of Wild Harbor is formed from flooding of the depression bounded by Nyes Neck and Crows Point, while the Wild Harbor River is a small drown river valley estuary now dominated by salt marsh. In contrast to these "natural" basins, the Wild Harbor Boat Basin appears to have been created from the dredging of a small pocket salt marsh for a harbor which now has bulkheads along much of its shoreline. Since this was done about a century ago, this basin is presently functioning as coastal embayment and needs to be managed as such.

Similar to other embayments on Cape Cod, Wild Harbor is a mesotrophic (moderately nutrient impacted) shallow coastal estuarine system in the initial stages of nutrient impairment, as evidenced by a decline in eelgrass habitat. However, the main basin of Wild Harbor continues to support extensive eelgrass beds, but acreage has declined over the past 25 years. The presence of eelgrass is particularly important to the use of Wild Harbor as fish and shellfish habitat and in turn a source of larvae critical to coastal benthic animal and fish communities. The Wild Harbor System represents an important potential shellfish resource to the Town of Falmouth, primarily for quahogs, however, shellfishing is presently prohibited by the Massachusetts Division of Marine Fisheries year round for the entire Wild Harbor system as a result of the historic oil spill as well as bacterial contamination from watershed and wetland run-off. The shellfish closures and documented eelgrass loss within the main basin of Wild Harbor has raised concern in recent years over the health of estuarine resources within the Wild Harbor system. The Town of Falmouth has specifically targeted nutrient management within the watersheds to its estuarine systems as a way towards restoring and/or safe guarding the estuarine resources of the town. Wild Harbor is slated to be included in upcoming phases of the Town's Comprehensive Wastewater Management Planning effort.

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; but 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 shorelines. In particular, Wild Harbor, as well as other embayment systems on Cape Cod, are at risk of eutrophication from increasing nitrogen loads in discharging surface water and groundwater from land-use changes to its watershed. Given its structure, Wild Harbor currently exhibits a higher overall habitat health then most estuaries in the Town of Falmouth and much of Cape Cod.

Currently, the primary ecological threat to Wild Harbor marine resources is degradation resulting from nutrient enrichment. Loading of the critical eutrophying nutrient, nitrogen, to the embayment waters has increased over the past few decades with some additional increase possible unless nitrogen management is implemented. The nitrogen loading to Wild Harbor and other Falmouth embayments (Rands Harbor, Fiddlers Cove, Quissett Harbor, Oyster Pond, Little Pond, Great Pond, Green Pond, Bournes Pond), like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Town of Falmouth has been among the fastest growing towns in the Commonwealth over the past three decades and does not have centralized wastewater treatment throughout the entire Town. These unsewered areas contribute significantly to the nitrogen loading of the Wild Harbor system, both through transport in direct groundwater discharges to estuarine waters and through surface water flow to the estuarine reach of each system. As existing and probable increasing levels of nutrients impact Falmouth's coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

The Town of Falmouth, as the primary stakeholder to the Wild Harbor embayment systems, has been concerned over the resource quality of the Towns significant coastal resources, inclusive of the Wild Harbor system. In the mid-1980's the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. Nutrient limits were set for nitrogen in each of the Town's embayments. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts. To acquire baseline water quality data necessary for ecological management of Falmouth's coastal salt ponds and harbors, a citizen-based water quality monitoring program was initiated by the Town of Falmouth. Falmouth Pondwatch, was established to provide on-going nutrient related embayment health information in support of the By-law. The water quality monitoring program was based on a collaborative effort between scientists, citizens and representatives of the Town of Falmouth. As originally conceived, the monitoring program focused on data collection in three original ponds, Oyster Pond, Little Pond and Green Pond. By 1990, the scope of water quality data collection expanded to include two additional ponds, Great/Perch Pond and Bournes Pond. In 1992, the scope of data collection was once again expanded to include West Falmouth Harbor in order to evaluate the effects from a nutrient enriched wastewater plume generated by the Falmouth Wastewater Treatment Facility. Since 1997, technical aspects of the Falmouth PondWatch Program have been coordinated through the Coastal Systems Program at SMAST-UMassD. In addition, the Town of Falmouth has supported the Coalition for Buzzards Bay's Water Quality Monitoring Program which, through its association with the Coastal Systems Program at UMASS-SMAST, collected data on nitrogen related water quality within the Falmouth estuaries that exist adjacent Buzzards Bay. The collaborative CBB/SMAST water quality monitoring effort covered systems such as Wild Harbor and Megansett Harbor, Fiddlers Cove and Rands Harbor System beginning in 1992. The Coalition's BayWatcher Program has collected the principal baseline water quality data necessary for ecological management of Falmouth's embayments and harbors adjacent Buzzards Bay. The BayWatchers is a citizen-based water quality monitoring program run by the Coalition for Buzzards Bay (T. Williams, Project Coordinator) with technical and analytical assistance from the Coastal Systems Program at SMAST-UMD until 2008.

The common focus of the Coalition for Buzzards Bay BayWatcher Water Quality Monitoring Program effort has been to gather site-specific data on the current nitrogen related water quality throughout all the embayments tributary to Buzzards Bay and determine the relationship between observed water quality and habitat health. This multi-year effort was initiated in 1992, with significant support from the Buzzards Bay Project. The BayWatcher Water Quality Monitoring Program in Wild Harbor developed a water quality baseline for this system. Additionally, as remediation plans for various systems are implemented, the continued monitoring will help satisfy monitoring requirements by State regulatory agencies and provide quantitative information to the Town relative to the efficacy of remediation efforts. The MEP effort builds upon the water quality monitoring efforts and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Wild Harbor embayment system.

In conjunction with other Town efforts, the Town of Falmouth's Planning Office continues to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present MEP evaluation is part of that continuing effort. The estuarine specific watershed based nutrient loading model, the hydrodynamic models and the water quality models being developed under the MEP for Wild Harbor will be an additional set of tools the town can use to inform future nutrient management decisions. The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning and nitrogen management alternatives development needed by the Town of Falmouth. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Falmouth to develop and evaluate the most cost effective nitrogen management alternatives to restore the Town's valuable coastal resources currently being degraded by nitrogen overloading.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. The nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over this assimilative capacity, which begins to cause declines in their ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities. At its higher levels, enhanced loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human uses. However like nutrients, bacterial contamination is related to changes in land-use as 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 Falmouth) are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries.

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

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

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

In appropriate estuaries, TMDLs for bacterial contamination will also be conducted in concert with the nutrient effort (particularly if there is a 303d listing). However, the goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation. As part of the overall effort, the evaluation and modeling approach will be used to assess available options for meeting selected nitrogen goals, protective of embayment health.

The major Project goals are to:

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

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

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

The Linked Model has been applied for watershed nitrogen management in ca. 44 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options.

The Linked Watershed-Embayment Model when properly parameterized, calibrated and validated for a given embayment becomes a nitrogen management planning tool, which fully supports TMDL analysis. The Model suggests "solutions" for the protection or restoration of nutrient related water quality and allows testing of "what if" management scenarios to support evaluation of resulting water quality impact versus cost (i.e., "biggest ecological bang for the buck"). In addition, once a model is fully functional it can be "kept alive" and corrected for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

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

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

I.2 NITROGEN LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Wild Harbor embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer minerals (Weiskel and Howes 1992). Since even Cape Cod "rivers" are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidal reaches within the Wild Harbor and Wild Harbor River portion of the estuary follow this general pattern, where the primary nutrient of eutrophication in this system is nitrogen.

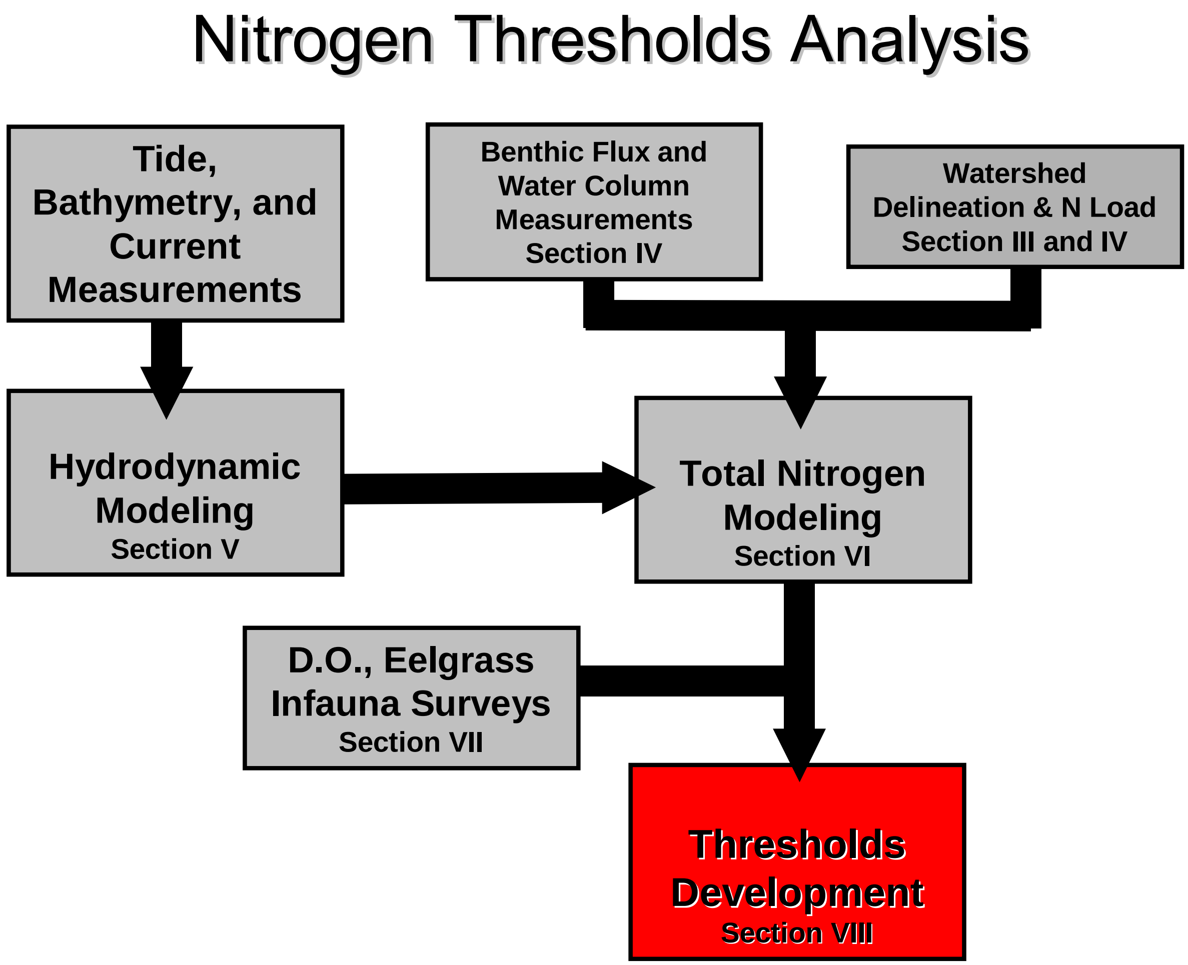

Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Section numbers refer to sections in this MEP report where the specified information is provided.

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

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

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

Even while the Wild Harbor estuary currently supports relatively healthy habitat, it does appear to be beyond its ability to assimilate additional nutrients without impacting ecological health. Nitrogen levels are elevated in the inner basin (Boat Basin) with a decline in benthic animal habitat. Similarly, eelgrass beds within the outer harbor basin have been stable for many decades, though areas that have persisted at the margins of the inner basin are declining. Together these habitat indicators indicate a system still supporting areas of high quality habitat, but just beyond its threshold (i.e. reaching its assimilative capacity). The result is that nitrogen management of this system is aimed at restoration of impaired habitat through the management of present and potential future increases in watershed nitrogen loading.

In general, nutrient over-fertilization is termed "eutrophication" and when the nutrient loading is primarily from human activities, it is considered "cultural eutrophication". Although the influence of human-induced changes has increased nitrogen loading to the system and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within a given embayment system 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" for water quality modeling of the Wild Harbor System; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within the system. Therefore, water quality modeling of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives. Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. The spread of pollutants may be analyzed from tidal current information developed by the numerical models.

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Wild Harbor System and the component basins of the system: the small boat basin associated with the Wild Harbor Yacht Club and the Wild Harbor River. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the systems. 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 in each estuarine receiving water.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic model for the system were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Virtually all nitrogen entering Falmouth's embayment systems is transported by freshwater, predominantly groundwater, either through direct discharge or after discharging to a stream flowing to estuarine waters. Concentrations of total nitrogen and salinity of Buzzards Bay / outer Wild Harbor source waters and within the boat basin and the Wild Harbor River were taken from the Coalition for Buzzards Bay BayWatchers Monitoring Program and from sampling of the Wild Harbor system by MEP staff during MEP related data collection activities. Measurements of nitrogen and salinity distributions throughout the estuarine waters of the system were used to calibrate and validate the water quality models (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 Wild Harbor Estuarine System for the Town of Falmouth. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Buzzards Bay (Section IV and VI respectively). 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 VI. 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 each embayment was performed that included a review of existing water quality information, temporal changes in eelgrass distribution, dissolved oxygen records and the results of a benthic infaunal animal analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration in a given estuarine basin. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for Wild Harbor. Finally, any additional analyses of the system relative to potential alterations of circulation and flushing (for instance as would be considered for the inlet of Wild Harbor River), is presented 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. In addition, the diverse avian fauna which feed upon infauna or fish communities are also affected and their numbers and diversity declines. This overall nutrient driven process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and ponds, it is not a necessarily a part of the natural evolution of a system.

In most marine and estuarine systems, such as the Wild Harbor System, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. As a result, there has been significant effort to develop tools for predicting how modification of watershed nitrogen loads and changes in tidal flushing quantitatively cause changes in the concentrations of water column nitrogen in the receiving estuary. Further development of these approaches generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2002).

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

In the case of Wild Harbor, numerous studies relating to environmental contamination and habitat health have been conducted over the past three decades that have informed the MEP habitat assessment and threshold development process. Directly supporting the present Massachusetts Estuaries Project effort to develop a nitrogen threshold for Wild Harbor were multiple historic investigations of the effects of the 1969 oil spill off West Falmouth which severely affected benthic animal communities in this estuary. These studies documented the impacts of fuel oil on benthic infaunal communities and established a critical point of reference for understanding the existing benthic infaunal community structure and the degree to which it is currently being effected by nutrients versus recalcitrant fuel oil. Some of the key studies of the Wild Harbor oil spill are summarized below.

The West Falmouth Oil Spill after 20 Years: Fate of Fuel Oil Compounds and Effects on Animals (Teal et.al., 1992; Teal 1993) – In 1989, researchers from Woods Hole Oceanographic Institution undertook sampling of sediments in Wild Harbor and Wild Harbor River in order to assess the degree to which that estuarine system may still be affected by oil that spilled from the barge Florida when it went aground off West Falmouth, MA. on September 29, 1969. Samples were taken for analysis of hydrocarbons in the sediments as well as infauna and were obtained from both sub-tidal and inter-tidal locations. After 20 years researchers found that little effect remains from the spilled No. 2 fuel oil. In the most heavily oiled area of the marsh there was still oil present at relatively high concentrations in marsh sediments to a depth of 15cm, however, the vast majority of oil that was spilled was no longer present. Based on the sampling that was completed, the oil spilled in the Wild Harbor system was virtually all gone from all the sub-tidal sampling sites and most of the inter-tidal marsh sampling sites. Based on the limited sediment sampling, the investigators concluded that 20-years post oil spill, less than 1% of the Wild Harbor Marsh area was still significantly affected by petroleum hydrocarbons. Furthermore, it was concluded that the Wild Harbor Marsh did not appear any different than any other healthy New England Salt Marsh, so long as the small remaining hydrocarbons remained undisturbed. Additionally, while the residual ecological effects were very small, investigators indicated that burrowing animals penetrating into contaminated sediments would be exposed to hydrocarbon levels that previously had significant biological effects. It was clearly stated that disturbance of the remaining affected marsh area could release enough oil to have observable local effects.

The West Falmouth oil spill after thirty years: The persistence of petroleum hydrocarbons in marsh sediments, Environmental Science and Technology (Reddy et.al., 2002) **–** Wild Harbor has been the site of intensive study resulting from the 1969 oil spill that occurred when the oil barge *Florida* went aground in West Falmouth. Between 650,000 and 700,000 liters of No. 2 fuel oil was released to Buzzards Bay and large quantities were pushed by the prevailing winds into the marshes of Wild Harbor, greatly affecting the plants and benthic habitat of that estuary. Sub-tidal and inter-tidal areas of Wild Harbor were heavily contaminated with oil. According to historical studies benthic animals such as worms, mollusks and crustaceans were significantly impacted by the spilled oil and salt marsh grasses covered by oil died within weeks of the spill. The spilled oil also entered the tidal Wild Harbor River where it was deposited in the still water areas of the marsh and became integrated into the marsh sediments. A sample site has been maintained in this area of the embayment system and has consistently been monitored over the decades since the spill to document long term-chronic effects of oil released to estuarine environments. Researchers concluded that fuel oil is still present in Wild Harbor river after 30-years. Residues of degraded No. 2 oil were detected in 11 of 18 sections of the 36-cm-long sediment core which was taken from the long term monitoring station in Wild Harbor River. The highest concentration of the of oil residue (specifically TPH) was observed in the 12cm to14cm section and the 14cm to 16cm section. Additionally, it was noted that the investigators were able to conclude that no degraded oil was detected in the top 6 cm (0-6 cm) and the lowest 8 cm (28-36 cm) of the core that was taken from Wild Harbor River. This is significant relative to understanding the degree to which benthic animal community structure may be more defined by the effects of nutrient enrichment rather than the acute or chronic effects of the 1969 oil release.

Wild Harbor Nutrient Related Water Quality Monitoring: The MEP analysis requires high quality water quality data in order to complete its assessment and modeling approach. The Coalition for Buzzards Bay's Water Quality Monitoring Program has been collecting data on nutrient related water quality throughout Buzzards Bay estuaries inclusive of Wild Harbor, outer Megansett Harbor, Fiddlers Cove and Rands Harbor for more than a decade. The Coalition's BayWatcher Program has collected the principal baseline water quality data necessary to support ecological management of each of Buzzards Bay's embayments and harbors. The BayWatchers is a citizen-based water quality monitoring program run by the Coalition for Buzzards Bay (T. Williams, Project Coordination) with technical and analytical assistance from the Coastal Systems Program at SMAST-UMD until 2008. The program has a USEPA and MassDEP approved Quality Assurance Project Plan (QAPP), which was operational over the entire period of 1999-2009 (data period for this MEP analysis).

The common focus of the Coalition for Buzzards Bay BayWatcher Water Quality Monitoring Program effort has been to gather site-specific data on the current nitrogen related water quality throughout all the embayments tributary to Buzzards Bay to support evaluations of observed water quality and habitat health. The BayWatcher Water Quality Monitoring Program in the Wild Harbor Embayment System developed a data set that elucidated the long-term water quality of this system (Figure II-1). The monitoring undertaken was a collaborative effort with CBB (Tony Williams) coordinating the field effort and chemical assays being completed by the SMAST Coastal Systems Analytical Facility. The Coastal Systems Analytical Facility is located in the School for Marine Science and Technology UMASS-Dartmouth, 706 S. Rodney French Blvd, New Bedford, MA, and the laboratory Points of Contact are Sara Sampieri 508-910-6325 (ssampieri@umassd.edu) or Mike Bartlett (mbartlett@umassd.edu). Use of the SMAST Analytical Facility ensured sufficient sensitivity and accuracy of the analytical protocols and that proper QA/QC procedures were followed to allow incorporation of the data into the MEP analysis. Baseline water quality data are a prerequisite to entry into the MEP. Implementation of the MEP's Linked Watershed-Embayment Approach necessarily incorporates the quantitative water column nitrogen data (1999-2009) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff.

Since the results of the long term Water Quality Monitoring Program (1999-2009) and initial habitat assessments suggest that portions of the Wild Harbor System are presently beyond their ability to assimilate nitrogen without impairment to key estuarine habitats, the Town of Falmouth undertook participation in the Massachusetts Estuaries Project to complete ecological assessment, nitrogen source identification and water quality modeling. The main purpose of the MEP effort is to quantitatively assess existing habitat quality of each of the Harbor basins and to develop nutrient thresholds to guide the Town's estuarine management planning relative to the Wild Harbor System.

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

- Designated Shellfish Growing Area – MassDMF (Figure II-2)

- Shellfish Suitability Areas – MassDMF (Figure II-3)
- Anadromous Fish Runs - MassDMF (Figure II-4)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-5)
- Mouth of Coastal Rivers – MassDEP Wetlands Program (Figure II-6)

The MEP effort builds upon earlier watershed delineation and land-use analyses, the hydrodynamic modeling, historical eelgrass surveys and water quality surveys discussed above. This information is integrated with MEP supported higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Wild Harbor Estuary. The MEP has incorporated appropriate and available data from pertinent previous studies to enhance the determination of nitrogen thresholds for the Wild Harbor System and to reduce costs to the Town of Falmouth.

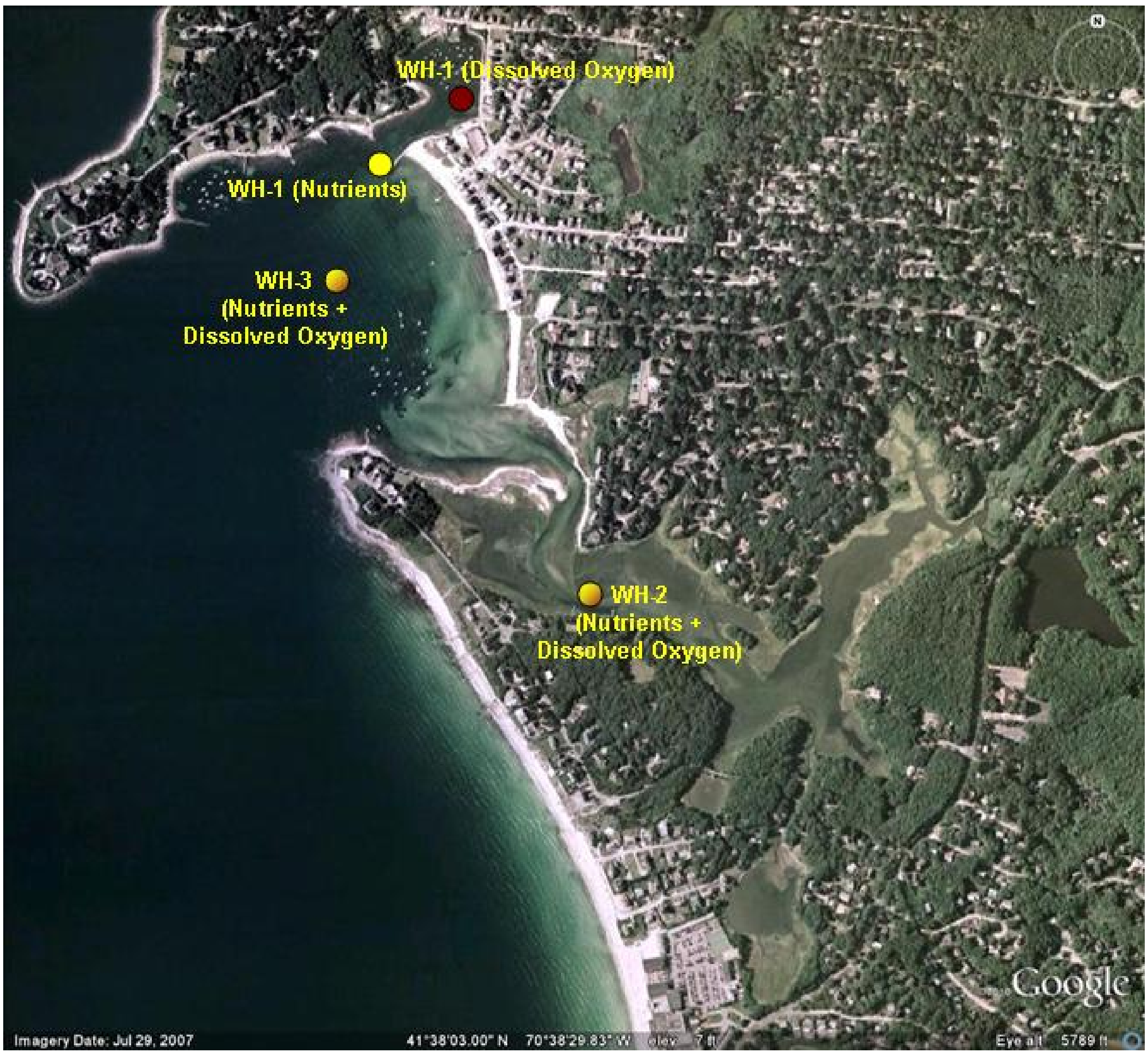

Figure II-1. Coalition for Buzzards Bay Water Quality Monitoring Program for Wild Harbor. Estuarine water quality monitoring stations sampled by the Coalition and analyzed by SMAST staff during summers 1999 to 2009.

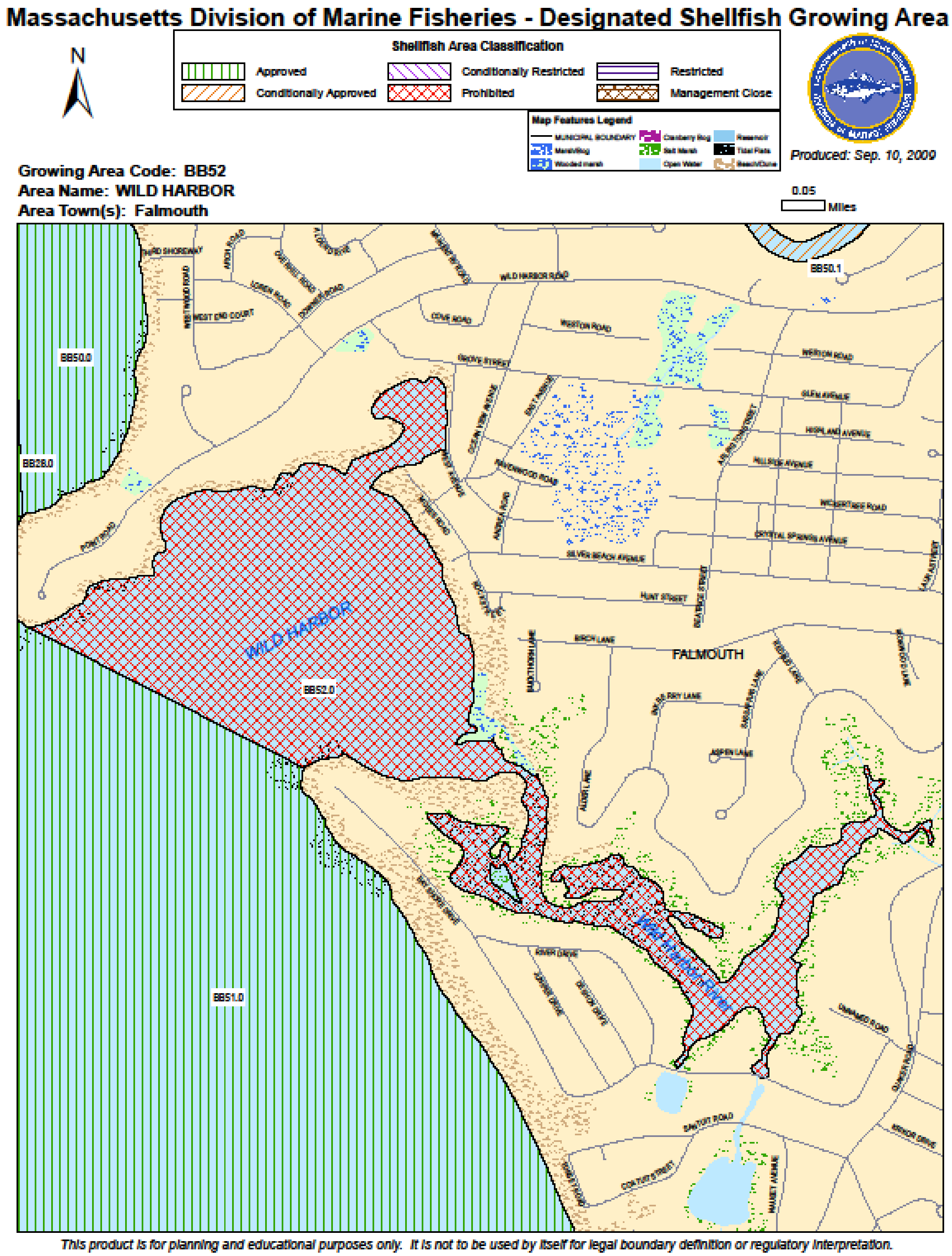

Figure II-2. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

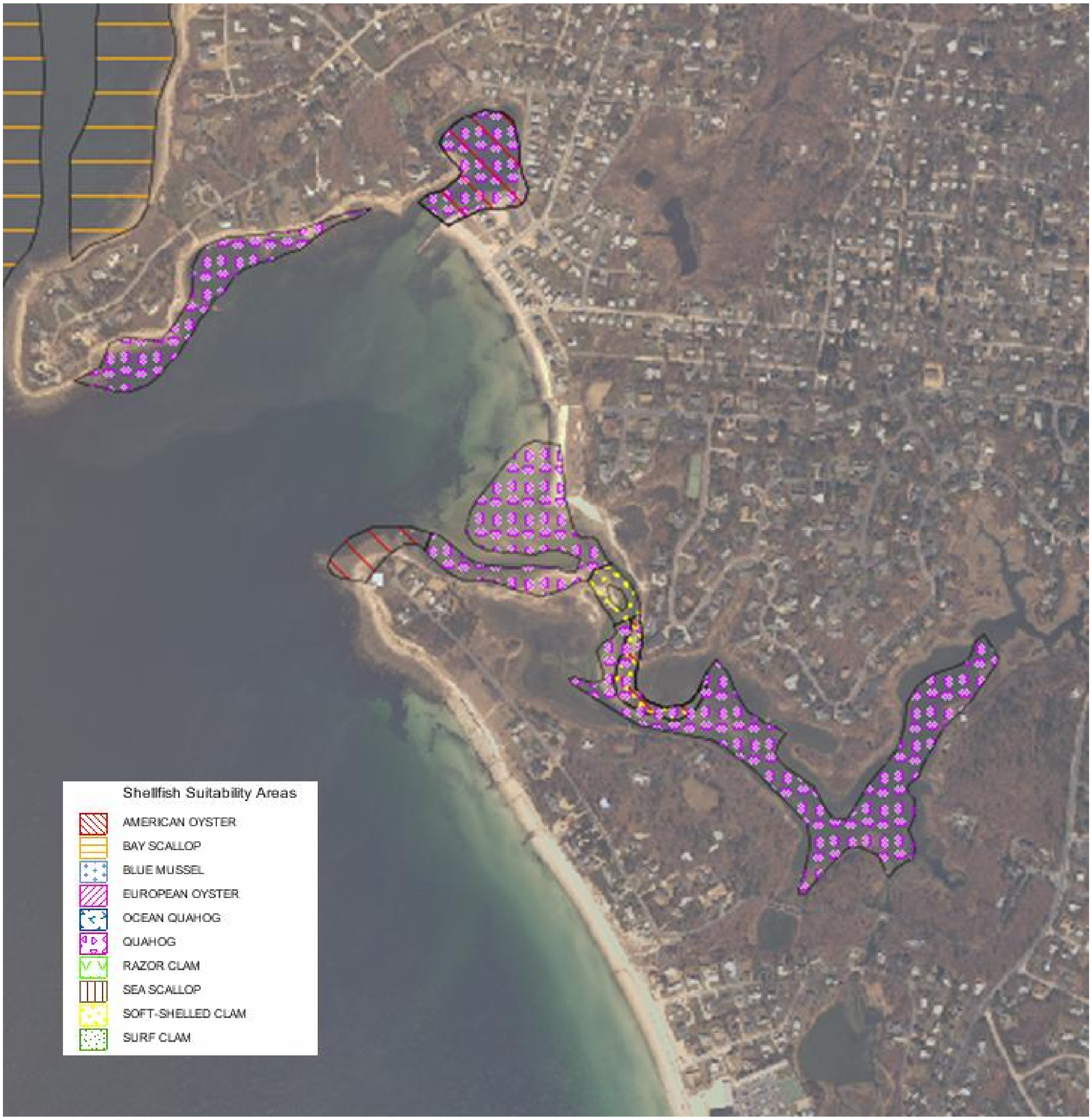

Figure II-3. Location of shellfish suitability areas within the Wild Harbor Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence".

Figure II-4. Anadromous fish runs within the Wild Harbor Estuary as determined by Mass Division of Marine Fisheries. The red diamonds show areas where fish were observed.

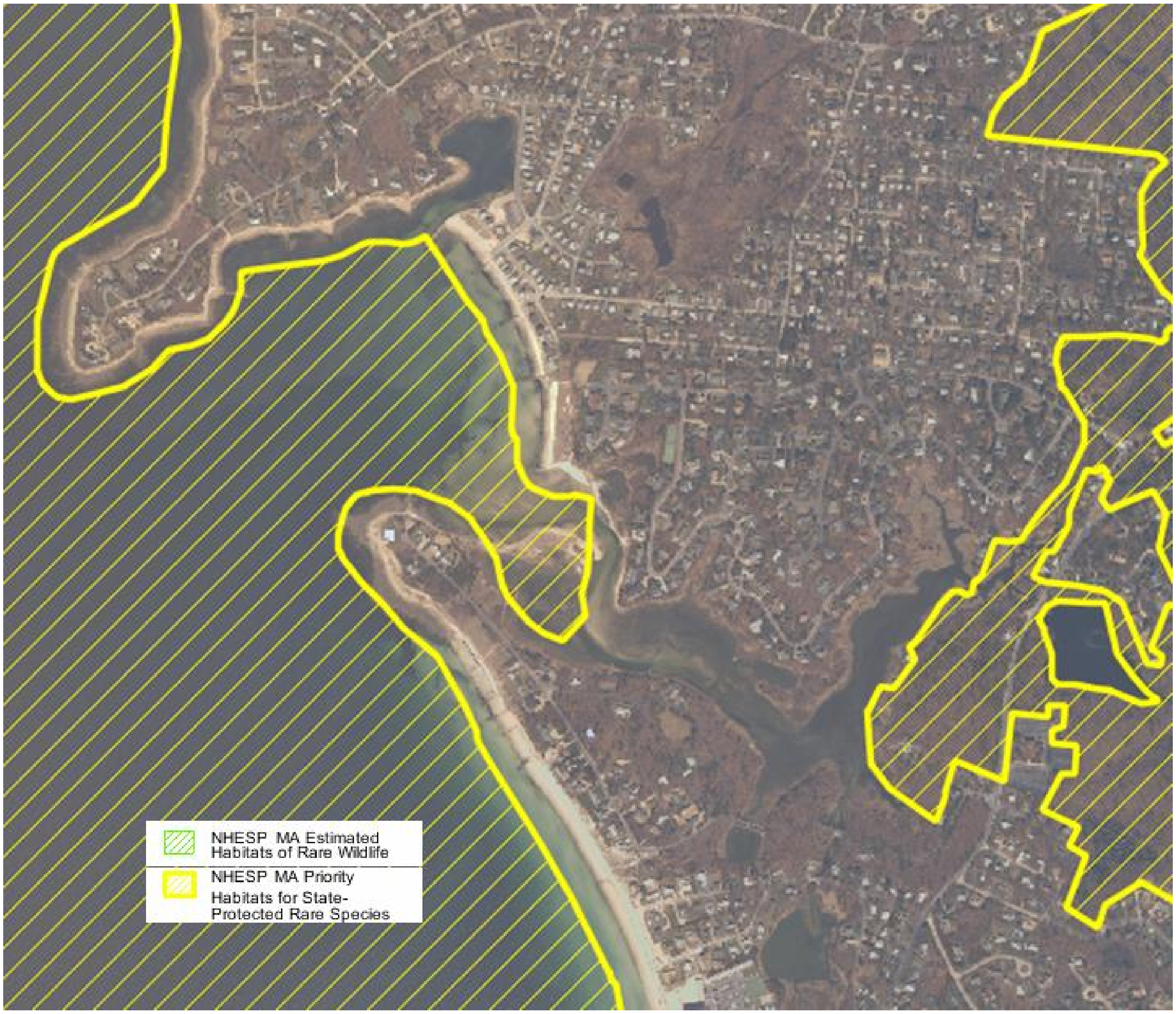

Figure II-5. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Wild Harbor Estuary as determined by - NHESP.

Figure II-6. Mouth of Coastal Rivers designation for Wild Harbor as determined by – MassDEP Wetlands 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 Wild Harbor embayment system. The Wild Harbor watershed is primarily located within the Town of Falmouth with its upper portions reaching into the Towns of Bourne and Sandwich and the Massachusetts Military Reservation.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Wild Harbor Embayment System under evaluation by the MEP Project Team. The Wild Harbor System is composed of a relatively open harbor with two (2) separate tributary basins: a tidal river dominated by salt marsh (Wild Harbor River) and a dredged inner embayment basin (Wild Harbor Boat Basin). The single surface water discharge enters the headwaters of the estuarine portion of the Wild Harbor River and drains a series of ponds and cranberry bogs that terminate approximately 1.5 km from the estuarine reach. Watershed modeling was undertaken to sub-divide the overall watershed to the Wild Harbor System into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These time of 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 Wild Harbor watershed is located along the western edge of the Sagamore flow cell. Model assumptions for calibration of the Wild Harbor Estuary included surface water discharges measured as part of the MEP stream flow program (2006 to 2007).

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water inputs and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to 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 Wild Harbor system and its various sub-watersheds, such as Dam and Wing Ponds, were delineated using the regional model of the Sagamore Lens flow cell (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Wild Harbor system and its sub-watersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 519 feet below NGVD 29 in the Sagamore Lens); since bedrock is approximately 150 feet below NGVD 29 in the Wild Harbor area the lowest model layer was inactive in this area of the model with variable thickness in the layer directly above. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location within the lens.

The glacial sediments that comprise the aquifer of the Sagamore Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The Wild Harbor system watershed is located in the Falmouth Moraine (also called the Buzzards Bay Moraine) in its southern region and in outwash deposits from these moraine deposits in its northern region. Both deposits are thought to be material deposited melting ice in a low energy depositional environment at the edge of a rapidly retreating ice lobe (Walter and Whealan, 2005). Modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that similar materials are permeable (*e.g.*, Masterson, *et al*., 1996) with lower hydraulic conductivity than the outwash plains that comprise most of the Cape. This distinction does not tend to impact groundwater flow direction and direct rainwater run-off is typically rather low as with most of the Cape. Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters in the groundwater models were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and stream flow data collected in 1989-1990 as well as 2003.

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

III.3 WILD HARBOR SYSTEM CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Wild Harbor embayment system, including Dam and Wing Ponds, the Wild Harbor River, and other sub-estuaries (Figure III-1) were determined by the United States Geological Survey (USGS). Model outputs of the watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (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 includes 10-yr time-of-travel boundaries. Overall, eight (8) sub-watershed areas, were delineated within the Wild Harbor 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 Wild Harbor System from the MEP delineated watershed is 13,969 m3/d. This flow includes corrections for outflow from Wing Pond, which straddles the southern boundary of the Wild Harbor System watershed.

The MEP watershed delineation is the second watershed delineation completed in recent years for the Wild Harbor 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 Wild Harbor system as a whole is 26% smaller than 1998 CCC delineation (1,941 acres vs. 2,634 acres, respectively). This significant difference is largely due to a change in flow paths just inland of the coast. The outer boundary of the MEP and CCC watersheds are coincident until approximately Quaker Road, then the MEP watershed lines continue east, while the CCC lines begin to bend toward the north. In addition, the CCC watershed did not include Wings Pond. The MEP watershed delineation also includes interior sub-watersheds to various components of the Wild Harbor system, such as ponds and streams 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).

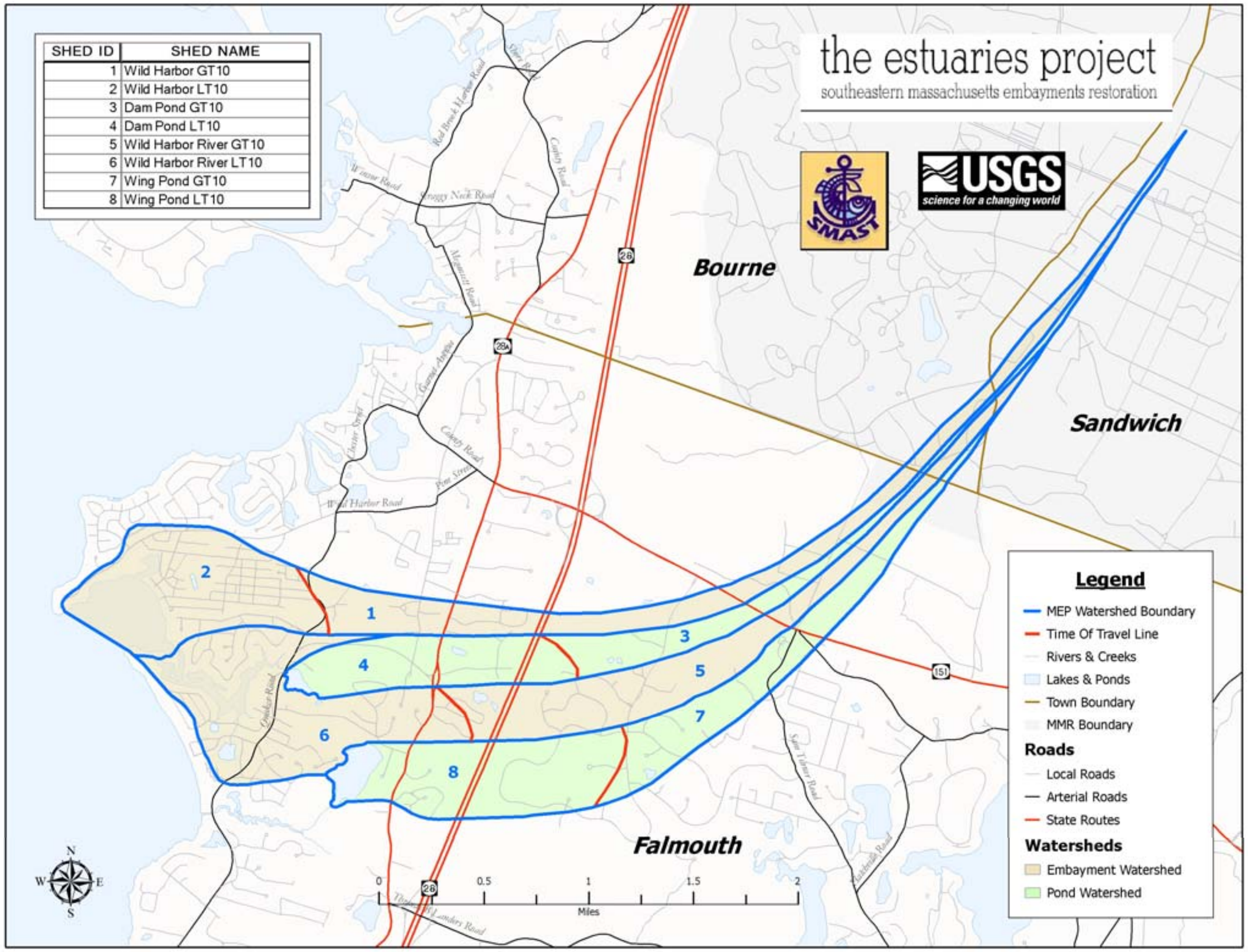

Figure III-1. Watershed delineation for the Wild Harbor estuary system. Sub-watershed delineations are based on USGS groundwater model output refined for pond and estuary shorelines and MEP streamflow measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes (see Section IV) and are designated with a "10" in the watershed names. Sub-watershed groups (*e.g.*, Wild Harbor River, Wild Harbor-Wild Harbor Boat Basin) were developed to match the functional estuarine sub-units in the water quality model (see Section VI).

Table III-1. Daily groundwater discharge from each of the sub-watersheds in the watershed to the Wild Harbor system estuary, as determined from the regional USGS groundwater model.					
Watershed	**#**	**Watershed Area (acres)**	**% contributing to Estuaries**	**Discharge**	
				m^3/day	**ft^3/day**
Wild Harbor GT10	1	314	100	2,408	85,021
Wild Harbor LT10	2	240	100	1,840	64,972
Dam Pond GT10	3	144	100	1,106	39,058
Dam Pond LT10	4	196	100	1,504	53,104
Wild Harbor River GT10	5	321	100	2,465	87,068
Wild Harbor River LT10	6	372	100	2,851	100,680
Wing Pond GT10	7	218	46	766	27,045
Wing Pond LT10	8	293	46	1,0307	36,371
TOTAL WILD HARBOR SYSTEM				13,969	493,318
Notes: 1) discharge volumes are based on 27.25 inches of annual recharge on watershed areas; 2) percentage of inflow from Wing Pond, which straddles the southern boundary of the overall Wild Harbor watershed boundary, 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 Wild Harbor system has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected; the model allows all this data to be organized and to be brought into congruence with adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model and the use of this model for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down gradient estuary. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Wild Harbor system (Section VI).

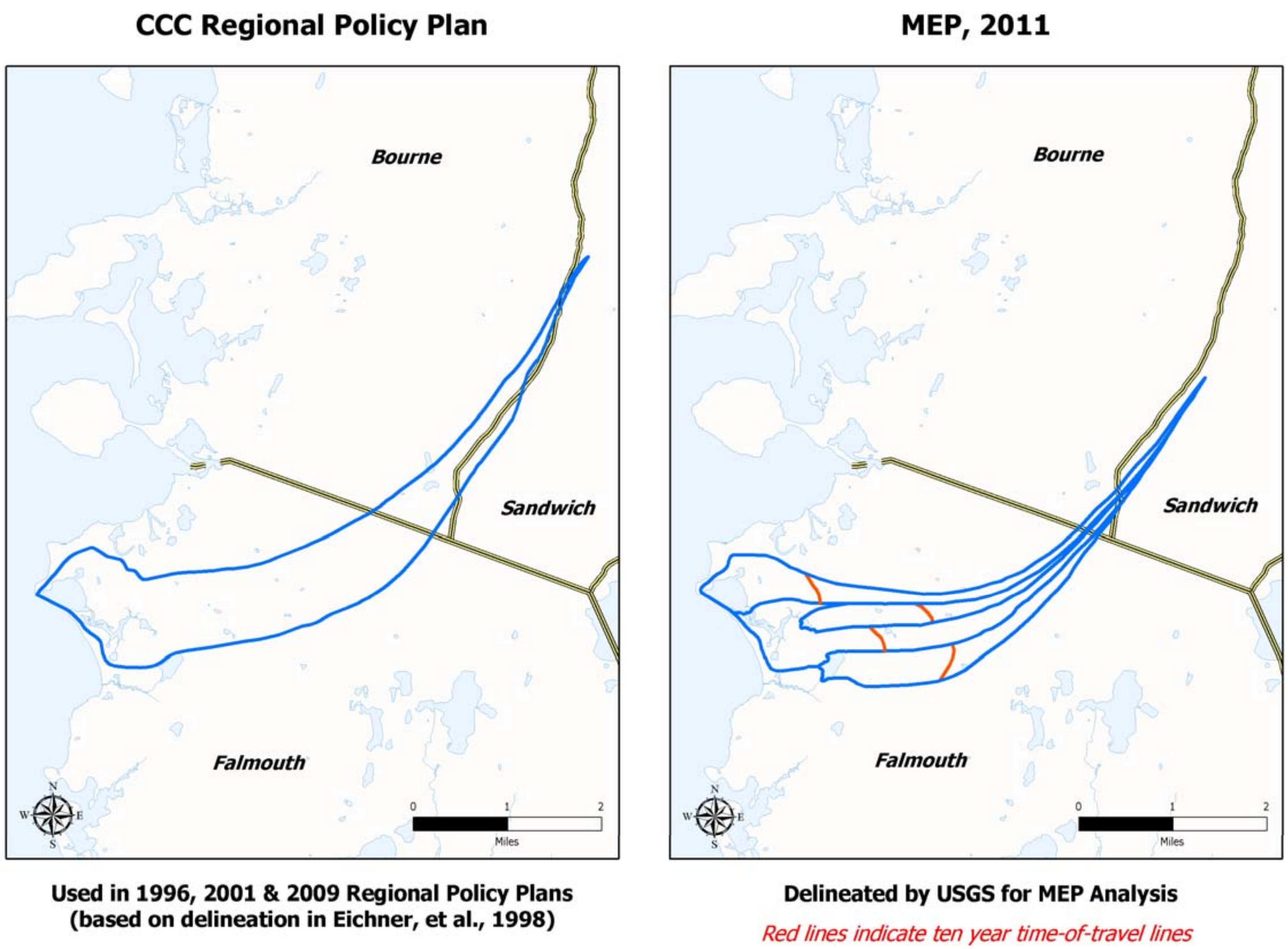

Figure III-2. Comparison of MEP Wild Harbor System watershed and sub-watershed delineations used in the current MEP assessment and the Cape Cod Commission watershed delineation (Eichner, *et al*., 1998), used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009). The MEP watershed area for the Wild Harbor system as a whole is 26% smaller than 1998 CCC delineation (1,941 acres vs. 2,634 acres, respectively), due primarily to an update in flow paths just inland of the coast.

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 Wild Harbor 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 Wild Harbor 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 Wild Harbor 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 Wild Harbor watershed, including watersheds to Dam and Wing Ponds and the Wild Harbor 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 collections points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. Ten-year time of travel sub-watersheds in the overall Wild Harbor watershed have been delineated for ponds, streams and the estuary itself. Simple review of less than and greater than 10-year time of travel watersheds indicates that 69% of the unattenuated nitrogen load from the whole watershed is within less than 10 year travel time to the estuary (Table IV-1). If this review is refined by looking at the stream flows, outflow out of the ponds, and adding in loads from precipitation on the estuary surface, the percentage that reaches the estuary within 10 years increases only slightly to 70%. The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuary (after accounting for natural attenuation, see below) and considering 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 Wild Harbor 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 Wild Harbor.

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
Wild Harbor GT10	1		989	989	0%
Wild Harbor LT10	2	2,779		2,779	100%
Dam Pond GT10	3		578	578	0%
Dam Pond LT10	4	726		726	100%
Wild Harbor River GT10	5		1,098	1,098	0%
Wild Harbor River LT10	6	2,450		2,450	100%
Wing Pond GT10	7		852	852	0%
Wing Pond LT10	8	1,776		1,776	100%
Wild Harbor Whole System		7,731	3,517	11,248	69%

Notes: if these loads are corrected to account for stream flows, outflow out of the ponds, and to add in loads from precipitation on the estuary surface, the percentage of watershed nitrogen load within 10-year time-of-travel to estuary increases to 70%

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data are used for some portion of the loads, while information developed from other detailed site-specific studies is applied to other portions of the watershed. The Linked Watershed-Embayment Management Modeling Approach (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 Wild Harbor System, the model used land-use data from the Towns of Falmouth, Bourne, and Sandwich 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 Wild Harbor watershed was determined based upon a site-specific study of stream flow and assumed and measured attenuation in the up-gradient freshwater ponds. Stream flow was characterized at the outlet immediately down gradient of Dam Pond. A sub-watershed to this stream discharge point allowed comparison between field collected data from the stream and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is generally estimated based on available information. Attenuation through the ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data are reliable enough to calculate a pond-specific nitrogen attenuation factor. Stream flow and associated surface water attenuation is included in the MEP's nitrogen attenuation and freshwater flow investigation, presented in Section IV.2.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. In the present effort, two freshwater ponds have delineated sub-watersheds within the Wild Harbor watershed: Dam Pond and Wing Pond. If smaller aquatic features that have not been included in this MEP analysis were providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly 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 Wild Harbor Estuary; 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 Wild Harbor includes portions of the Towns of Falmouth, Bourne, and Sandwich, Estuaries Project staff obtained digital parcel and tax assessor's data from the towns to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data for Falmouth are from 2009, while similar data from Bourne is from 2008 and Sandwich data are from 2010. The Bourne and Sandwich parcels are located within the Massachusetts Military Reservation (MMR) and the watershed area is enclosed in one parcel from each town. The land use databases contain traditional information regarding land use classifications (MassDOR, 2009) plus additional information developed by the towns and, in the case of development within the MMR, information developed by the Cape Cod Commission (CCC) GIS staff. The overall effort was completed with the assistance from GIS staff from the CCC.

Figure IV-1 shows the land uses within the Wild Harbor estuary watershed. Land uses in the study area are grouped into nine (9) land use categories: 1) residential, 2) commercial, 3) industrial, 4) mixed use, 5) agricultural (cranberry bogs), 6) undeveloped, 7) public service/government, including road rights-of-way, 8) open space, and 9) 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 represents tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, golf courses, open space, roads) and private groups like churches and colleges. It should be noted that there are some similar land uses that are classified in different categories; in this watershed, for example, the Golf Club of Cape Cod is classified by the town assessor as a commercial property (land use code 380), while the Ballymeade Country Club is classified as a multi-use property (land use code 038). Unclassified parcels are properties without any assessor land use classifications.

Residential land uses are the dominant land use type in the overall Wild Harbor watershed and occupy 37% of the watershed area (Figure IV-2). These land uses are also the dominant land use type in each of the sub-watershed groupings shown in Figure IV-2. Public service land uses occupy the second largest area within the whole watershed area (33%) and also within each of the sub-watershed groupings. Examples of these land uses are lands owned by town and state government (including open space and state-owned properties at the MMR), housing authorities, and churches. It is notable that land classified by the town assessor as undeveloped is 14% of the overall watershed area.

In all the sub-watershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type, ranging between 75% and 86% of all parcels in these sub-watersheds and 81% of all parcels in the Wild Harbor System watershed. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 96% to 100% of residential parcels in the individual sub-watershed groupings and 98% of the residential parcels throughout the Wild Harbor system watershed.

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

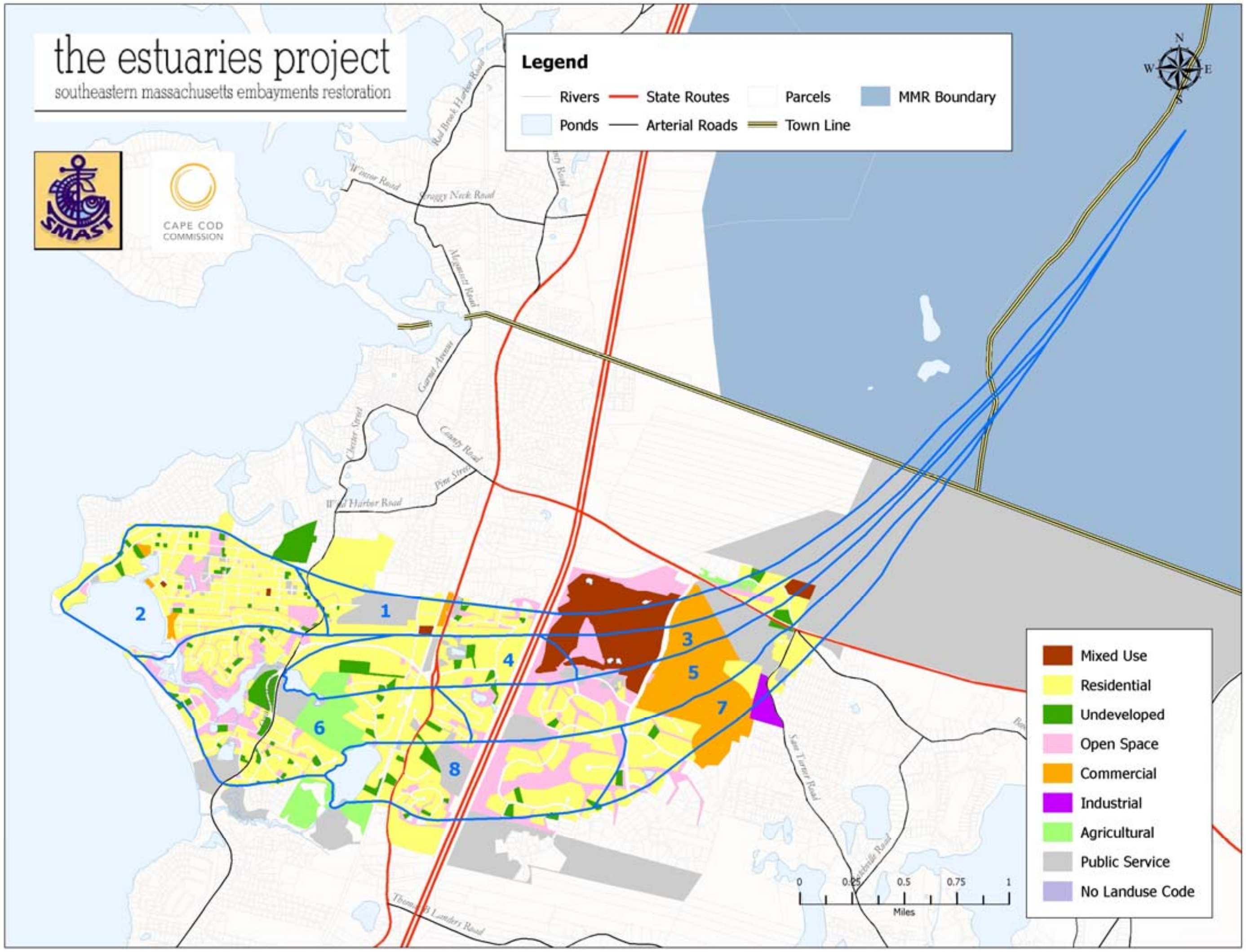

Figure IV-1. Land-use in the Wild Harbor system watershed and sub-watersheds. The total system watershed extends over portions of the Towns of Falmouth, Bourne, and Sandwich. Land use classifications are based on respective town assessor classifications and MADOR (2009) categories. Base assessor and parcel data for Falmouth are from the year 2009, while corresponding data from Bourne is from the year 2008 and Sandwich is from 2010. Bourne and Sandwich parcels are individual parcels with the Massachusetts Military Reservation (MMR).

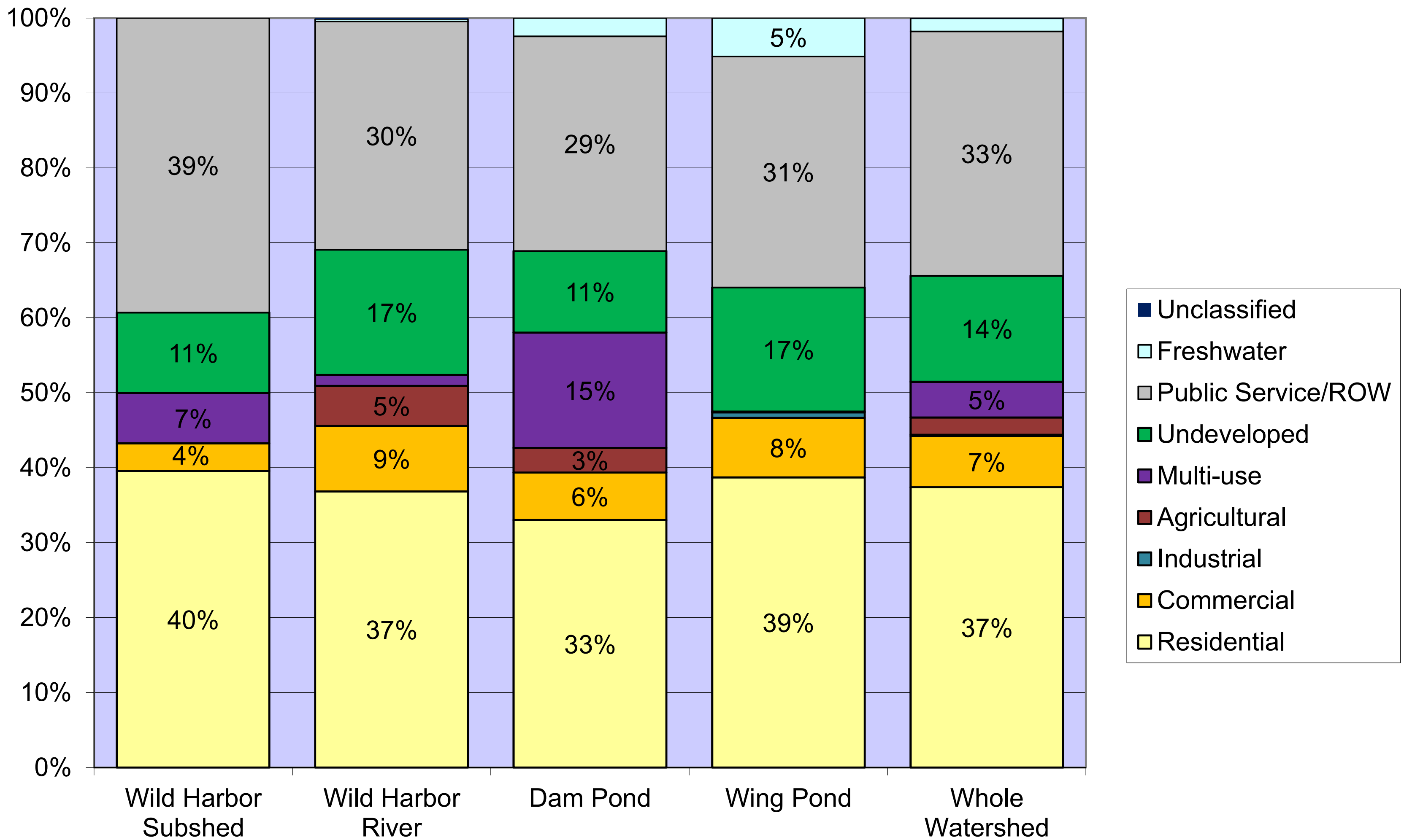

Figure IV-2. Distribution of land-uses by area within the Wild Harbor system watershed and four component sub-watersheds. Land use categories are generally based on town assessor's land use classification and groupings recommended by MADOR (2009). Unclassified parcels do not have an assigned land use code in the town assessor's databases. Only land-uses comprising 3% or more of the watershed area are shown.

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 are linked to assessor's parcel information using GIS techniques. The parcel specific water use data are 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 times N concentration) of 23.63, to convert water (per volume) to nitrogen load (N mass). This coefficient uses a per capita nitrogen load of 2.1 kg N person-yr^{-1} and is based upon direct measurements and corrects for changes in concentration that result from per capita shifts in water-use (*e.g.*, due to installing low plumbing fixtures or high versus low irrigation usage).

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

While census based population data have 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 are 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 the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to provide an independent validation of the average residential water use within the Wild Harbor watersheds, MEP staff reviewed US Census population values for the Town of Falmouth. Since Bourne and Sandwich occupy such a small portion of the watershed and all of the land area is within the MMR, 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 Falmouth is 2.36 people per housing unit with 69% year-round occupancy, while 2010 Census information indicates that the average occupancy dropped to 2.24 people per housing unit and the year-round occupancy also dropped to 64%. Average water use for single-family residences with municipal water accounts in the Wild Harbor MEP study area is 159 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the study area average is 143 gpd.

In order to provide a check on the water use, the Falmouth Census average occupancies were multiplied by the state Title 5 estimate of 55 gpd of wastewater per capita. The resulting flow estimates are 130 gpd of average estimated water use per residence based on 2000 Census occupancy and 123 gpd based on 2010 Census occupancy. Estimates of summer populations on Cape Cod derived from a number of approaches (*e.g.*, traffic counts, garbage generation, WWTF flows) suggest average population increases from two to three times year-round residential populations measured by the US Census. If it is assumed that the Falmouth population doubles for three months during the summer, the adjusted year-round occupancy would rise to 2.95 or 2.80 people per housing unit for the 2000 and 2010 Census occupancies, respectively. These occupancies multiplied by 55 gpd/person result in respective average flows of 162 gpd and 154 gpd or roughly equivalent to the measured average water use in the Wild Harbor watershed. 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 1,474 developed parcels in the Wild Harbor watershed. Developed parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs or town assessor valuations, and do not have a listed account in the water use databases. Of the 57 developed parcels without water use accounts, 52 (91%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and are assigned the Wild Harbor study area average water use of 159 gpd in the watershed nitrogen loading modules. Of the 5 remaining developed parcels, all 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 159 gpd as their water use in the watershed nitrogen loading model.

Wastewater Treatment Facilities and Alternative Septic Systems

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

MEP staff received a list of alternative, denitrifying septic systems in Falmouth and total nitrogen effluent monitoring data from the Barnstable County Department of Health and the Environment (personal communication, Brian Baumgaertel, 1/2011). From the BCDHE database, project staff identified one denitrifying septic system within the Wild Harbor watershed. This system had only one sampling run, so the wastewater nitrogen load from this site was not modified.

MEP staff also received information on parcels with existing or planned connections to the New Silver Beach Wastewater Treatment Facility (personal communication, Bob Shea, GIS Coordinator, 3/2011). The project plan has been to connect ~230 properties in the New Silver Beach area of North Falmouth to a new treatment facility, with discharge under the athletic fields at the elementary school (www.falmouthwastewaterprojects.org, (Figure IV-3). Project staff also received effluent flow information from Town Department of Public Works staff (personal communication, Amy Lowell, Assistant Wastewater Manager, 4/2011). This data showed that the WWTF began discharging effluent in August 2009 and town staff confirmed that approximately half of the planned connections occurred in the first six to nine months (personal communication, Jerry Potamis, Wastewater Superintendent, 4/2011). Given that the water quality data collected in the Wild Harbor occurred in 2006-2007 and the WWTF did not start operation until after this time, the WWTF collection and treatment is included in the buildout conditions, not the existing conditions assessment. According the MassDEP databases, no other GWDPs exist in the Wild Harbor system watershed.

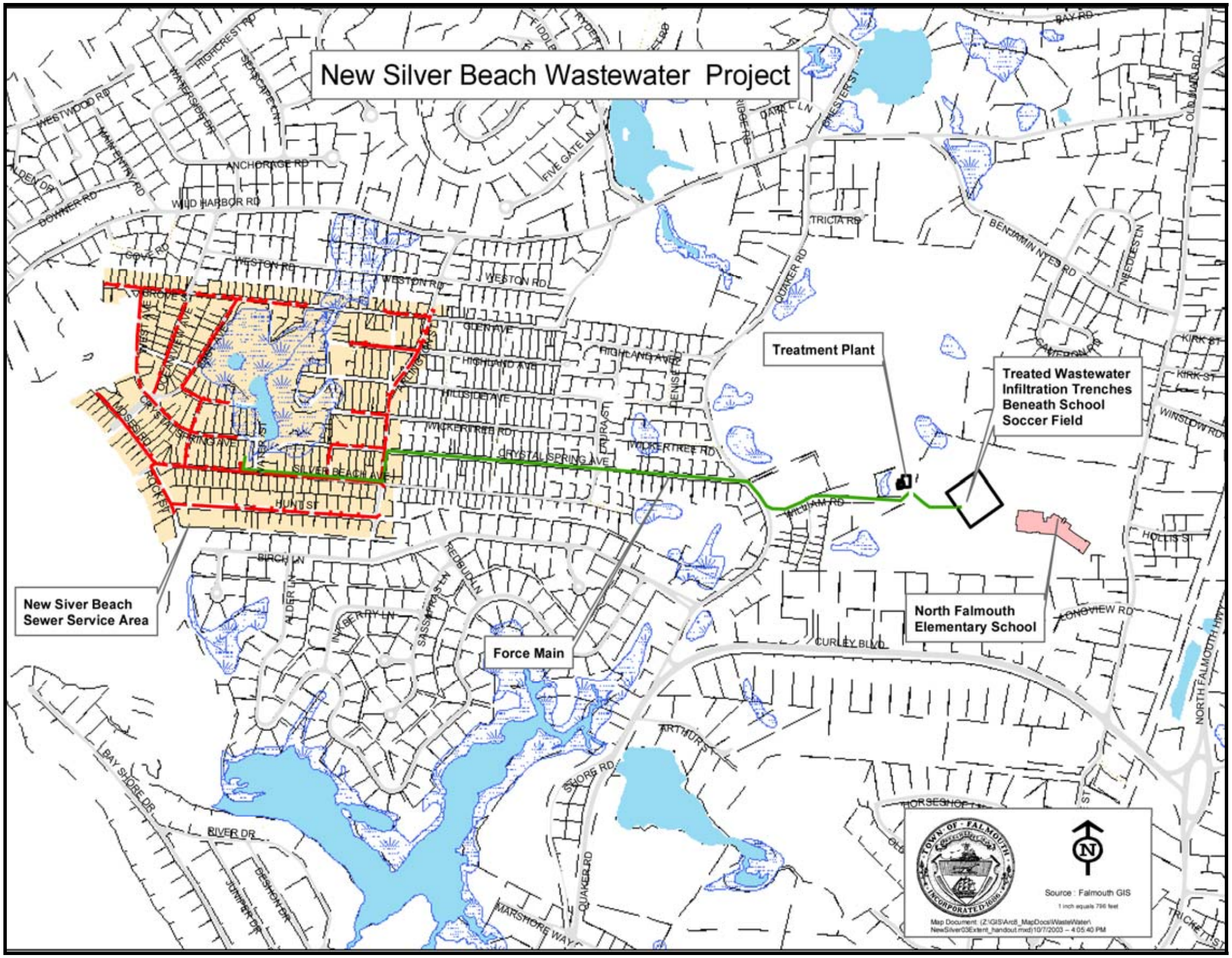

Figure IV-3. Parcels with existing or planned connections to the New Silver Beach Wastewater Treatment Facility. The project plan has been to connect ~230 properties in the New Silver Beach area of North Falmouth to a new treatment facility, with discharge under the athletic fields at the elementary school.

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 Wild Harbor system, MEP staff reviewed available regional information about residential lawn fertilizing practices and sought site-specific information for the following golf courses: Golf Club of Cape Cod and Ballymeade Country Club. An estimated nitrogen load is also included for the cranberry bogs in the watershed. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts, while MEP staff contacted the golf course superintendents in order to obtain course-specific fertilizer application rates.

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.

In order to obtain a site-specific estimate of nitrogen loading from the Golf Club of Cape Cod and Ballymeade Country Club, MEP staff contacted Charles Passios, Chief Operating Officer. When site-specific golf course fertilizer application rates are not available, MEP staff assign average application rates from courses that have provided this information. Mr. Passios approved the use of the averages for the two courses in the Wild Harbor watershed. Current MEP nitrogen application rate averages (all in pounds per 1,000 square feet per year) based on reporting from 19 courses are: greens, 3.6; tees, 3.3; fairways, 3.3, and roughs, 2.5.

As has been done in all MEP reviews, MEP staff reviewed the layout of both golf courses from aerial photographs, classified the various turf types, and, using GIS, assigned these areas to the appropriate sub-watersheds. The MEP average golf course nitrogen application rates were then applied to the respective turf areas, a standard MEP 20% leaching rate was applied, and annual load from each golf course to each sub-watershed was calculated.

Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture that has been conducted in southeastern Massachusetts (Howes and Teal, 1995). Based on this study, only the bog loses measurable nitrogen, the forested upland releases only very low amounts of nitrogen. For the watershed nitrogen loading analysis, the areas of active bog surface are based on a GIS coverage maintained by MassDEP for Water Management Act purposes. Cranberry bogs are located in two Wild Harbor sub-watersheds: Wild Harbor River and Dam Pond.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Wild Harbor 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 Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and the MassDEP Nitrogen Loading Computer Model Guidance Document (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the MEP nitrogen loading analysis for the Wild Harbor watershed are summarized in Table IV-2.

Road areas are based on MassHighway GIS information, which provides road width for various lengths of road segments. MEP staff utilized the GIS to sum these segments and their various widths by 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 Wild Harbor are shown in Figure IV-4.

The review of individual parcels straddling watershed boundaries includes corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. The Town of Falmouth provided GIS coverages of building footprints for the roof area calculations; MMR buildings, parking lots, and roads areas were digitized from aerial photos. 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 Wild Harbor 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 Wild Harbor 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:	159 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[4]	
Cranberry Bogs Nitrogen Leaching (kg/ha/yr)[1]	6.9	Commercial	
Total Nitrogen effluent concentration assigned to New Silver Beach (mg/l)[5]	10	Wastewater flow (gpd/1,000 ft2 of building):	180
		Building coverage:	15%
		Industrial	
Average Single Family Residence Building Size from watershed data (sq ft)	1,506	Wastewater flow (gpd/1,000 ft2 of building):	44
		Building coverage:	5%

Notes:
1) Howes and DeMoranville evaluation of cranberry bog export
2) Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.
3) Based on average flow of all single-family residences in the watershed
4) based on existing water use and water use for similarly classified properties throughout the Town of Falmouth
5) New Silver Beach WWTF began discharging effluent after the MEP water quality data collection in Wild Harbor, so its impact is included in the buildout assessment. Given that the WWTF only has approximately one year of treatment performance available, it is conservatively assigned a total nitrogen effluent concentration equivalent to its regulatory permit limit of 10 mg/l.

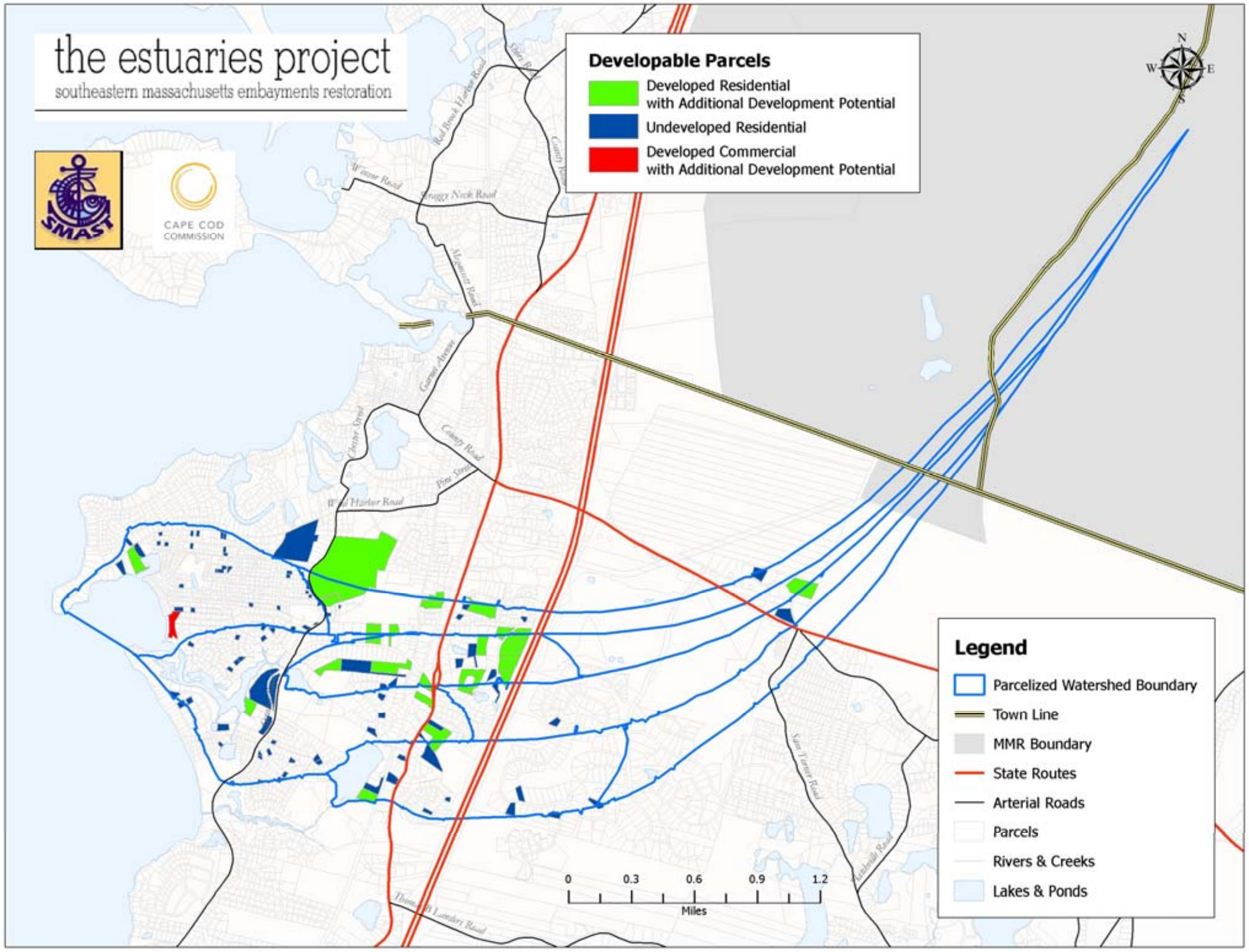

Figure IV-4. Parcels, Parcelized Watersheds, and Developable Parcels in the Wild Harbor watersheds. Parcels colored green and red are developed parcels (residential and commercial, respectively) with additional development potential based on current zoning, while parcel colored blue are undeveloped residential parcels classified as developable by the town assessor. The parcelized watersheds are drawn to minimize the division of properties for management purposes while achieving a match of area with the modeled watersheds of 2% or less. Developable parcels are based on town assessor classifications and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. All buildout results were reviewed with town staff (personal communication, Brian Currie, Town of Falmouth, 4/2011).

Following the assignment of all parcels, sub-watershed modules were generated for each of the eight (8) sub-watersheds in the Wild Harbor 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 are assigned to each sub-watershed. Individual sub-watershed information is then integrated to create the Wild Harbor Watershed Nitrogen Loading summary 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 watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Wild Harbor System watershed, 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 Wild Harbor study area, there are only two ponds with a delineated sub-watershed: Dam Pond and Wing Pond (sub-watersheds #4 and #8 respectively). Both are completely within the Wild Harbor River LT10 sub-watershed (#6), so all of the water from the two associated sub-watersheds and the accompanying attenuated nitrogen load is discharged to the salt marsh dominated tidal river.

Table IV-3. Wild Harbor 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 rate of 50% based on water quality monitoring from the Cape Cod Pond and Lake Stewards program or a modified factor if sufficient monitoring data are available. All nitrogen loads are kg N yr^{-1}.

		Wild Harbor N Loads by Input (kg/y):							% of Pond Outflow	Present N Loads			Buildout N Loads		
Watershed Name	Watershed ID#	Wastewater	Non-Golf Course Fertilizers	Golf Course Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Wild Harbor System		6,782	696	1,209	764	642	276	567		10,370		9,178	10,937		9,654
Wild Harbor GT10	1	501	46	278	113	-	52	70		989		989	1,060	-	1,060
Wild Harbor LT10	2	2,237	288	-	229	-	25	28		2,779		2,779	2,807	-	2,807
Wild Harbor Estuary Surface						377				377		377	377	-	377
Wild Harbor River Total		4,045	362	931	422	265	199	469		6,224		5,033	6,693	-	5,410
Wild Harbor River GT10	5	558	36	395	51	5	54	18		1,098		1,098	1,115	-	1,115
Wild Harbor River LT10	6	1,964	195	-	232	6	53	252		2,450		2,450	2,702	-	2,702
Wing Pond Total	WP	918	49	109	41	54	39	35	46%	1,210		605	1,245	-	622
Dam Pond Total	DP	606	83	427	98	38	53	164	100%	1,304		717	1,468	-	807
Wild Harbor River Estuary Surface						163				163		163	163	-	163
Dam Pond Total	DP	606	83	427	98	38	53	164		1,304	45%	717	1,468	45%	807
Dam Pond GT10	3	153	3	376	24	-	21	0		578		578	578	-	578
Dam Pond LT10	4	453	80	50	74	38	32	164		726		726	890	-	890
Wing Pond Total	WP	2,003	106	238	90	118	85	76		2,639	50%	1,320	2,715	50%	1,358
Wing Pond GT10	7	525	35	238	29	-	37	0		863		863	863	-	863
Wing Pond LT10	8	1,478	72	-	61	118	48	76		1,776		1,776	1,852	-	1,852

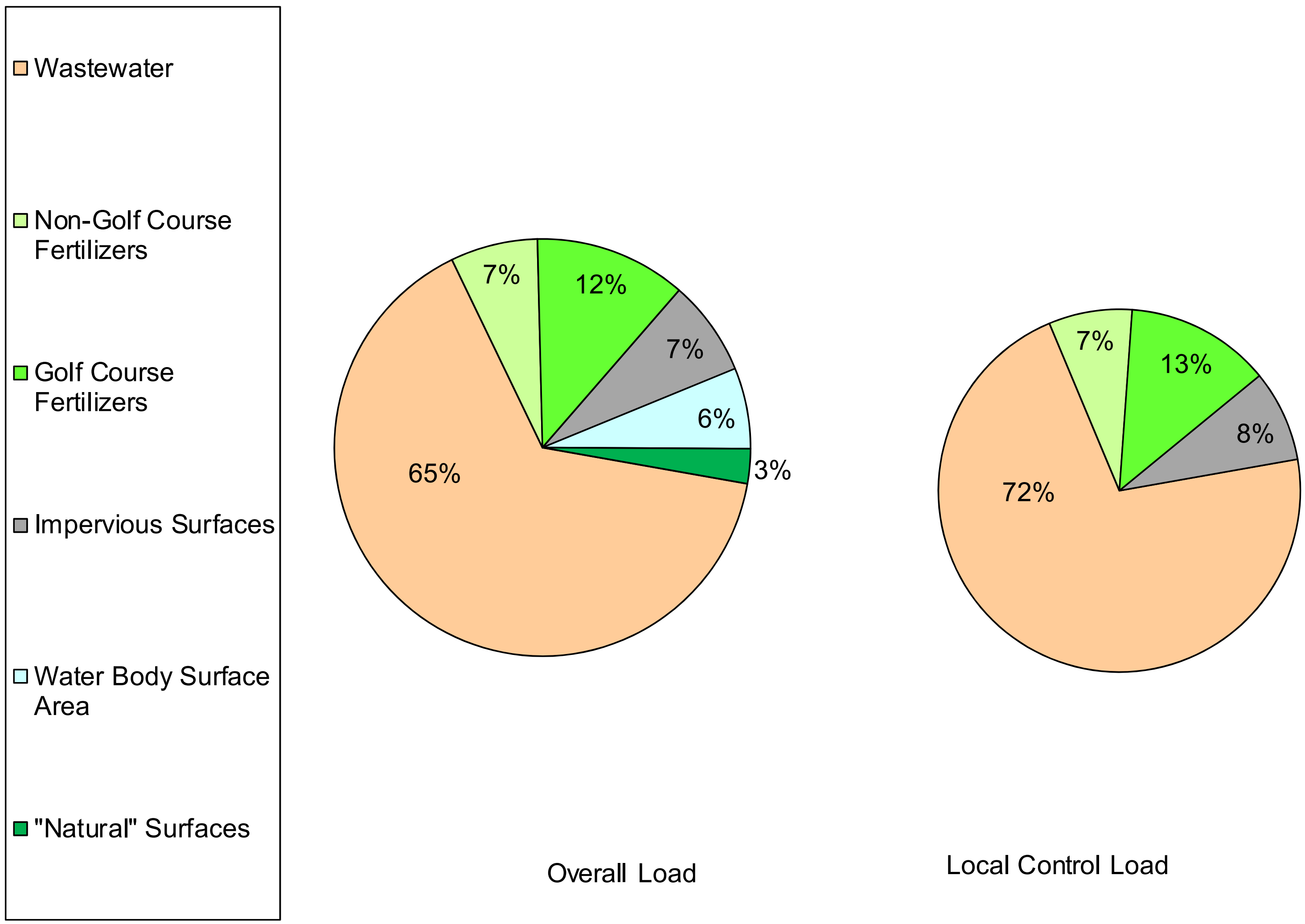

Figure IV-5. Land use-specific unattenuated nitrogen loads (by percent) to the whole Wild Harbor watershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control.

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/pond 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 and transformations, 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 through a stream outlet 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 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 the level of 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.

Many ponds on Cape Cod have been sampled through the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and the initiative of local volunteer pond sampling programs. The PALS Snapshots are regional volunteer pond one-time 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. Sampling protocols developed through the PALS program (Eichner *et al*., 2003) have been used for more extensive pond sampling programs in many communities on Cape Cod. Sampling under these protocols has included field collection of temperature and dissolved oxygen profiles and sampling has generally occurred at standardized depths that provide some evaluation of potential sediment nutrient regeneration. PALS water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. In some cases town programs have generated sufficient sampling data that modified MEP nitrogen attenuation rates can be reliably assigned to freshwater ponds.

Within the Wild Harbor watershed, there are two freshwater ponds with delineated watersheds: Dam Pond and Wing Pond. Neither pond has available pond-wide bathymetric data (Eichner *et al*., 2003) or sufficient water quality data collection outside of the MEP. Dam Pond has been sampled twice during the nine years of PALS Snapshots and Wing Pond has not been sampled through the PALS Program.

Although neither freshwater pond has sufficient in-pond sampling to assign an alternative MEP nitrogen attenuation rate, MEP staff did have a stream gauge located on the stream outlet of Dam Pond. Using the water quality and flow information collected from this gauge (see Section IV.2), MEP staff assigned a 45% attenuation rate to Dam Pond. This attenuation rate is a balance between the measured flow and load leaving the pond through the stream gauge and the likely discharge of a portion of the flow and load from the pond through its down gradient shoreline. Wing Pond is assigned the standard 50% nitrogen attenuation rate that has been determined to be a reasonably conservative attenuation rate for freshwater ponds in the MEP study area that are lacking sufficient pond-specific data.

Buildout

Part of the regular MEP watershed nitrogen load 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 density development 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 Falmouth assessors office 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. As an 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 Wild Harbor watersheds, MEP staff reviewed the results with Brian Currie, Falmouth Town Planner in April 2011. Suggested changes from town staff review were incorporated into the final buildout for Wild Harbor.

All the parcels with additional buildout potential within the Wild Harbor 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 unless the site is within the New Silver Beach WWTF collection area. As noted above, all connections to the New Silver Beach WWTF are included in the buildout scenario, given the timing of the plant start-up (August 2009) and the groundwater transport time. Existing properties within the collection area have a cumulative average water use of 12,621 gallons per day. This flow is assumed to be treated to 10 mg/l total nitrogen and removes 230 kg/yr of nitrogen from the Wild Harbor LT10 sub-watershed (#2). Cumulative unattenuated buildout loads from all sub-watersheds are indicated in a separate column in Table IV-3. As a result of the limited developable land and the nitrogen removals associated with the New Silver Beach WWTF, buildout additions within the Wild Harbor watersheds will increase the unattenuated system-wide nitrogen loading rate by only 6%.

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 Wild Harbor Embayment Systems 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 Wild Harbor watersheds). 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 Wild Harbor River portion of the Wild Harbor embayment system, a portion of the freshwater flow and transported nitrogen passes through a pond with a discrete discharge point (e.g. a culvert discharging into the head of the Wild Harbor River) prior to entering the broader estuary, thereby producing the opportunity for reductions in nutrient loading, primarily through nitrogen attenuation (Figure IV-6).

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.

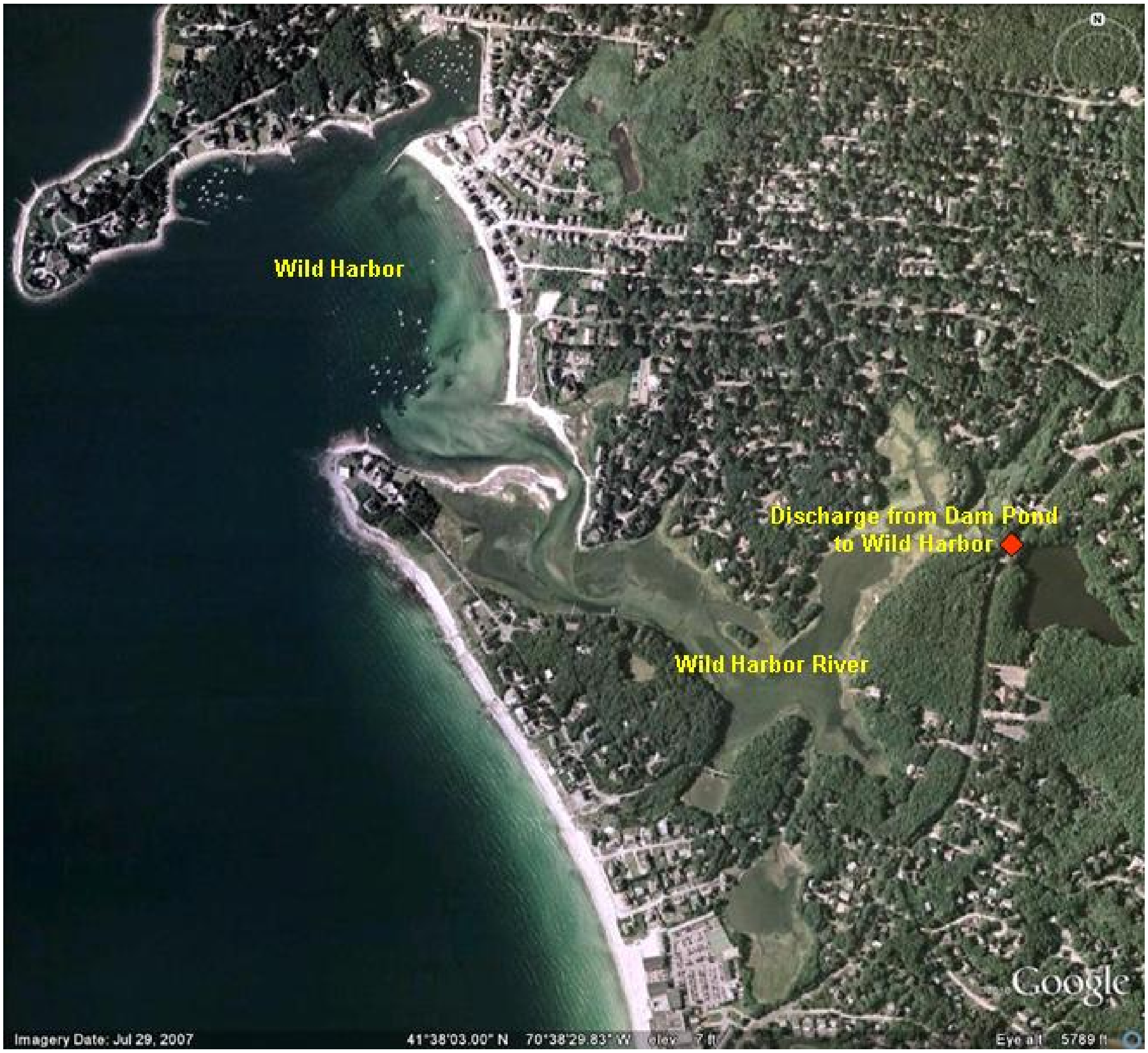

Figure IV-6. Location of the Stream gauge (red symbol) in the Wild Harbor River portion of the Wild Harbor Embayment System. Surface freshwater flow discharges from Dam Pond through a culvert to an upper tributary tidal creek within the Wild Harbor River salt marshes.

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 Wild Harbor 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 Wild Harbor watershed, 1) Dam Pond discharging to the tidally influenced Wild Harbor River. Wild Harbor River is a salt marsh dominated tidal river that discharges directly to the more open water basin that constitutes Wild Harbor.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater stream discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area up gradient from the various gauging sites. Flow and nitrogen load was determined at one gauge location immediately down gradient of Dam Pond for 16 months of record (Figures IV-7). During the study period, a velocity profile was completed at the gauge positioned in the small channel formed by the culvert discharging from Dam Pond. Velocity profiles were completed at the gauge 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).

Determination of flow at the gauge positioned at the culvert discharging from Dam Pond to Wild Harbor River was calculated and based on the measured values obtained for the cross sectional area of the creek as well as creek specific velocity. Freshwater discharge was represented by the summation of individual discharge calculations for each channel subsection for which a cross sectional area and velocity measurement were obtained. Velocity measurements across the entire channel cross section were not averaged and then applied to the total creek cross sectional area.

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

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

where by:

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

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

Periodic measurement of flows over the entire "stream" 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 gauge location. 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 Wild Harbor River and the down gradient Wild Harbor embayment and emanating from Dam Pond.

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 Wild Harbor River. Nitrogen discharge from the small creek capturing the freshwater discharge from Dam Pond was calculated using the paired daily discharge and daily nitrogen concentration data to determine the mass flux of nitrogen through the gauging site. For the creek gauge location, weekly water samples were collected (at low tide given the tidally influenced stage) in order to determine nutrient concentrations from which nutrient load was calculated when merged with the flow record. 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 gauging point currently reduces (percent attenuation) nitrogen loading to the overall embayment system.

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Dam Pond Discharge into Wild Harbor River

Dam Pond is a small freshwater pond and unlike many of the freshwater ponds on Cape Cod, this small pond has a direct surface discharge to the Wild Harbor River portion of the Wild Harbor Embayment System rather than draining solely to the aquifer along its down-gradient shore. This outflow through the culvert discharging to an un-named tidal creek, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation occurring within the pond. In addition, nitrogen attenuation also occurs within associated wetland areas, riparian zones and streambed associated with the salt marsh creek prior to entering the embayment basin of Wild Harbor. 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 Dam Pond up-gradient of the stream gauge site and the measured annual discharge of nitrogen to the salt marsh tidal creek associated with the head of Wild Harbor River, Figure IV-6.

At the gauge site (situated immediately down-gradient of the Dam Pond culvert passing under Quaker Road), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater creek discharging from Dam Pond and which carries the flows and associated nitrogen load to the head of the Wild Harbor River portion of the Wild Harbor embayment. As the small creek is tidally influenced, the gauge was located as far down gradient along the Creek reach such that freshwater flow dominated at low tide. Based on the stage record, the location of this specific gauge was clearly tidally influenced. This was further confirmed by salinity measurements conducted on the weekly water quality samples collected from the gauge site. Average salinity for all the water samples collected at low tide over the entire gauge deployment period was determined to be 3.2 ppt. As such, the overall flow record was adjusted to account for the slight salinity at the gauge location in order to quantify only the freshwater portion of the flow in the creek. As the salinity influence was slight and on many sample dates the salinity was less than 1.0 ppt, 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 May 25, 2006 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until November 9, 2007 for a total deployment of 18 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 gauge 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 the head of the Wild Harbor River portion of the Wild Harbor System. This measured attenuated mass discharge is reflective of the biological processes occurring primarily in Dam Pond as there was very little channel reach above the gauge site (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 the Creek receiving water from the Dam Pond culvert passing under Quaker Road 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 within 1% of the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September 20, 2006 and ending September 19, 2007 (low flow to low flow) was 1,979 m^3/day compared to the long term average flows determined by the USGS modeling effort (1,984 m^3/day which takes into consideration a small loss of volume via the down gradient shoreline).

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

Total nitrogen concentrations within the Creek outflow were moderate, 0.761 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 1.51 kg/day and a measured total annual TN load of 550 kg/yr. Nitrate, 0.32 mg N L^{-1}, accounted for less than half of the total nitrogen (42%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to Dam Pond was partially taken up by biological processes occurring in the pond prior to discharge. This is further supported by the fact that organic nitrogen (dissolved plus particulate organic nitrogen) constitutes virtually all of the remaining total nitrogen pool. The low concentration of inorganic nitrogen (0.34 mg N L^{-1}, 44% of TN) in the out flowing creek waters also suggests that plant production and biologic activity within the up gradient freshwater pond ecosystem can potentially be enhanced in Dam Pond. A more detailed biogeochemical investigation of Dam Pond would be warranted for this system in the event natural attenuation can be enhanced thereby decreasing the overall cost of nitrogen management in the Wild Harbor River sub-watershed.

From the measured nitrogen load discharged from Dam Pond to the to Wild Harbor River and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon lower total nitrogen load (550 kg yr^{-1}) discharged from the freshwater Creek compared to that added by the various land-uses to the associated watershed (1,304 kg yr^{-1}), the integrated attenuation in passage through Dam Pond prior to discharge to the estuary is 58% (i.e. 58% 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 capable of attenuating nitrogen. The directly measured nitrogen loads 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 the freshwater discharge from Dam Pond to the Wild Harbor River portion within the Wild Harbor Embayment System. The "Stream" data are from the MEP stream gauging site on the surface water discharge from Dam Pond. Watershed data are based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	**Dam Pond Discharge[(a)] to** Wild Harbor River	**Data Source**
Total Days of Record	365[(b)]	(1)
Flow Characteristics		
Stream Average Discharge (m3/day) **	1,979	(1)
Contributing Area Average Discharge (m3/day)	1,984	(2)
Discharge Stream 2006-07 vs. Long-term Discharge	0.25%	
Nitrogen Characteristics		
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.32	(1)
Stream Average Total N Concentration (mg N/L)	0.761	(1)
Nitrate + Nitrite as Percent of Total N (%)	42%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	1.51	(1)
TN Average Contributing UN-attenuated Load (kg/day)	2.75	(3)
Attenuation of Nitrogen in Pond/Stream (%)	45%	(4)

(a) Flow and N load to stream discharging from Dam Pond to Wild Harbor River includes apportionments of Pond contributing areas for ponds upgradient of Dam Pond.
(b) September 20, 2006 to September 19, 2007.
** Flow is an average of annual flow for 2006-2007

(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 Dam Pond Discharge to Wild Harbor River and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

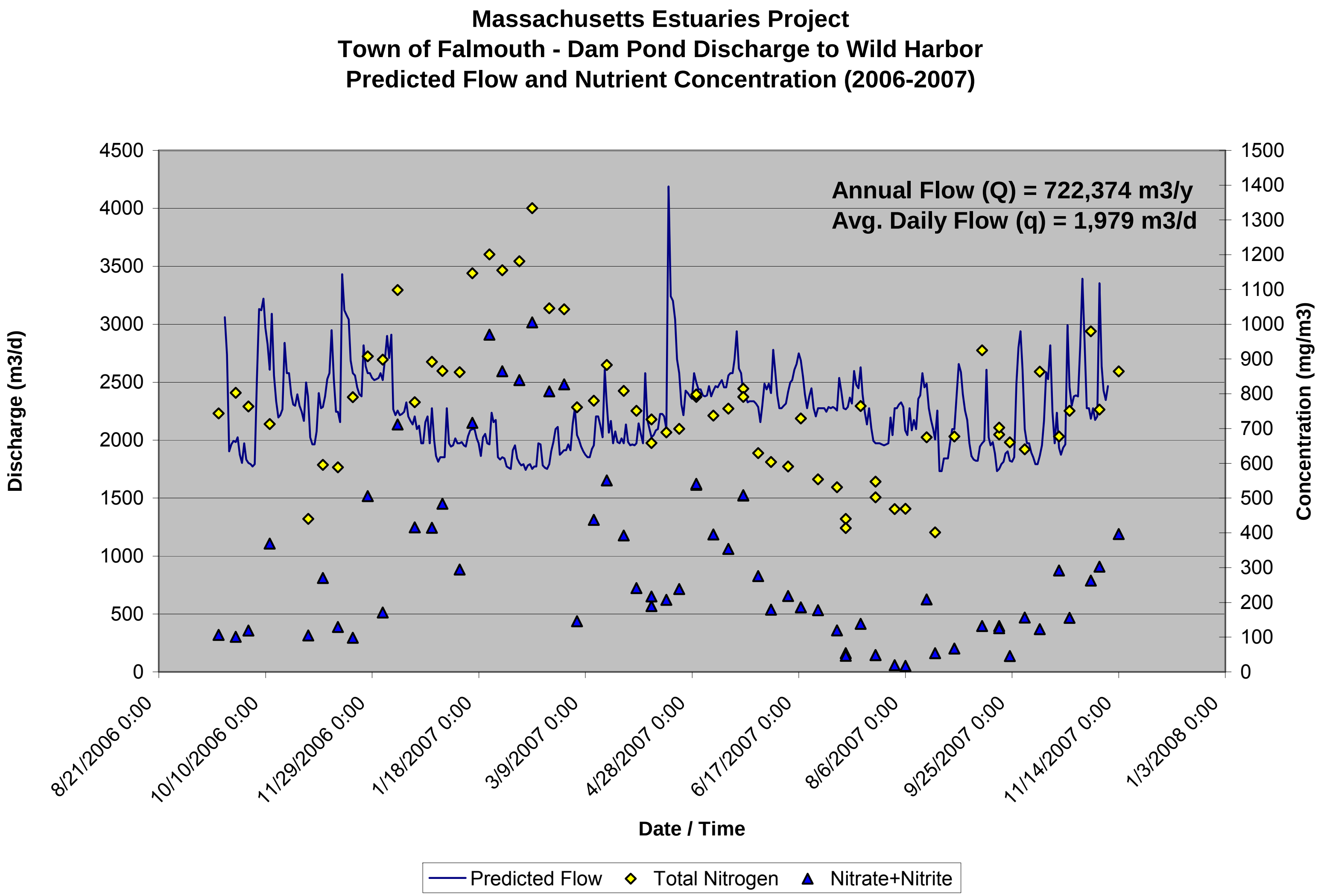

Figure IV-7. Dam Pond discharging directly into the head of the Wild Harbor River (solid red line), nitrate+nitrite (blue symbol) and total nitrogen (yellow symbol) concentrations for determination of annual volumetric discharge and nitrogen load (Table IV-4).

Table IV-5. Summary of annual volumetric discharge and nitrogen load from Dam Pond via surface water flow to the head of Wild Harbor River. Flows and loads based upon the data presented in Figures IV-7 and Table IV-4.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Wild Harbor River Dam Pond Disharge (MEP)	**September 20, 2006 to September 19, 2007**	**722,374**	**231**	**550**
Wild Harbor River Dam Pond Disharge (CCC)	Based on Watershed Area and Recharge	**724,014**	--	--

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

IV.3.1 Sediment-Water column Exchange of Nitrogen

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

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bio-available nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with deep depositional basins or salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer (e.g. Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh or Sesachacha Pond on the Island of Nantucket). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, for example in the margins of the main basin to Lewis Bay (Town of Barnstable, Cape Cod). In contrast, most embayments show low rates of nitrogen release throughout much of a basins area and, in regions of high deposition, typically support anoxic sediments with high release rates during summer months. The consequence of high deposition rates is that the basin sediments are unconsolidated, organic rich and sulfidic 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 Wild Harbor system. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

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

For the Wild Harbor Embayment System, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. A total of 16 cores were collected from 16 sites (Figure IV-8) in July-August 2006, focusing on obtaining an areal distribution that would be representative of nutrient fluxes throughout the system but also considering tributary “basins” such as the small boat basin as well as the salt marsh dominated Wild Harbor River. 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.

Wild Harbor System Benthic Nutrient Regeneration Cores

- WLD-1 1 core (Main Basin)
- WLD-2 1 core (Main Basin)
- WLD-3 1 core (Main Basin)
- WLD-4 1 core (Main Basin)
- WLD-5 1 core (Main Basin
- WLD-6 1 core (Main Basin)
- WLD-7 1 core (Boat Basin)
- WLD-8 1 core (Boat Basin)
- WLD-9 1 core (Wild Harbor River)
- WLD-10 1 core (Wild Harbor River)
- WLD-11 1 core (Wild Harbor River)
- WLD-12 1 core (Wild Harbor River)
- WLD-13 1 core (Wild Harbor River)
- WLD-14 1 core (Wild Harbor River)
- WLD-15 1 core (Wild Harbor River)
- WLD-16 1 core (Wild Harbor River Inlet)

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

Figure IV-8. Wild Harbor System locations (yellow symbols) of sediment sample collection for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-6.

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

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

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

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

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

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

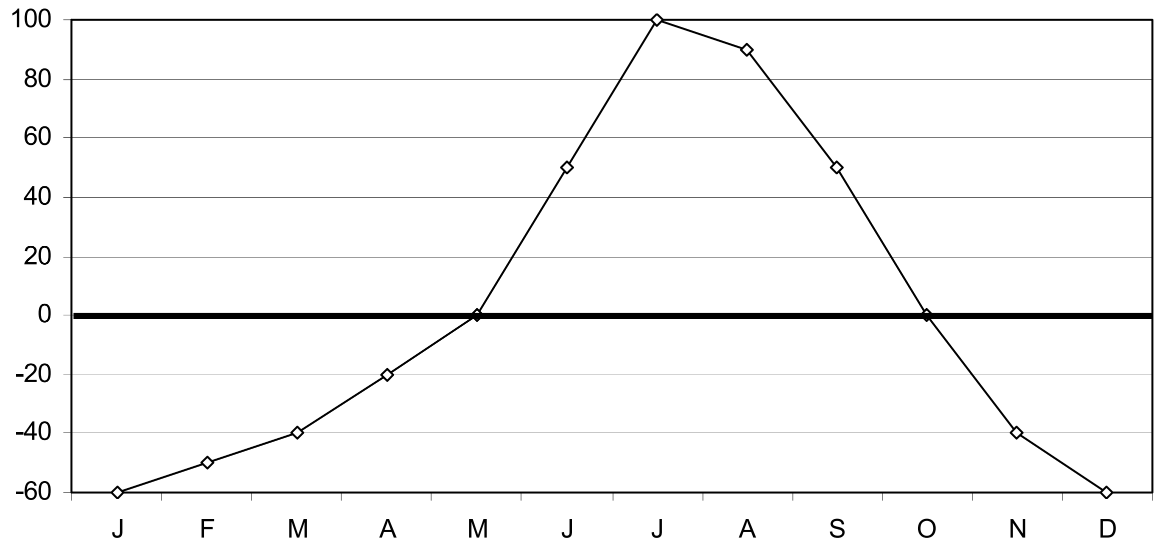

Figure IV-9. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

In order to determine the net nitrogen flux between water column and sediments, all of the above factors were taken into account. The net input or release of nitrogen within a specific embayment was determined based upon the measured total dissolved nitrogen uptake or release, and estimate of particulate nitrogen input.

Sediment Nitrogen Release by Standard Core Approach: Sediment sampling was conducted throughout the embayment basins of the Wild Harbor System. Sediment cores were collected from the outer main basin of Wild Harbor as well as the inner small boat basin associated and the salt marsh dominated Wild Harbor River. Generally, the distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content, as well as sediment type and an analysis of each site's tidal flow velocities. As expected flow velocities are generally low throughout the Wild Harbor System. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

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

Rates of net nitrogen release or uptake from the sediments within the Wild Harbor embayment system were comparable to other embayments of similar depth and configuration in southeastern Massachusetts. There was a clear pattern of sediment N flux. Sediments throughout both embayment basins showed net nitrogen uptake, the deep main basin having fluxes about twice the smaller Boat Basin of moderate depth, -36.7 and -17.1 mg N m^{-2} d^{-1}, respectively. The difference in sediment nitrogen flux reflects the configuration of the 2 basins and may also be correlated with the occurrence of dense eelgrass beds bordering the main basin, and the depositional nature of the smaller northern basin. The main basin sediments were well oxidized and generally fine sand with some organic matter, while the depositional Boat Basin sediments were organic enriched soft mud. In contrast sediment in the Wild Harbor River was generally organic mud with areas of salt marsh detrital deposits. These sediments are in balance with the overlying waters, with only a small net flux, 1.4 mg N m^{-2} d^{-1}.

The pattern of sediment nitrogen flux is similar to other similarly configured basins within southeastern Massachusetts with similar sediment characteristics. The large deep main basin of Wild Harbor is similar to the outer basin of nearby Quissett Harbor, which supports similarly oxidized sandy to sand/mud (mix) sediments and a net uptake of -12.2 mg N m^{-2} d^{-1}. Similarly, the sandy oxidized sediments at comparable depths within the main basins of the Nantucket Harbor Embayment System also show net nitrogen uptake in summer of -7.9 to -38.8 mg N m^{-2} d^{-1}; the outer basins of West Falmouth Harbor have rates of summer uptake of -11.8 mg N m^{-1} d^{-1}; and analogous basins within the Three Bays System show a similar pattern of net uptake in the outer basins ranging to lower uptake grading to net release in the innermost basins (Seapuit River to North Bay, -37.7 --> 57.7 mg N m^{-1} d^{-1}). The small inner Boat Basin with its organic enriched sediments also showed sediment nitrogen flux similar to other analogous systems, such as measured nitrogen flux rates in Farm Pond and Rushy Marsh, Barnstable, MA (-19.1 mg N m^{-2} d^{-1}), the depositional basins within the Parkers River Estuary, Seine Pond and Lewis Pond with net nitrogen uptake by their sediments of -16.9 and -11.8 mg L^{-1}, respectively, and the shallow enclosed tributary basins to the main basin of Edgartown Great Pond also showing comparable rates (-8.9 to -16.0, Wintucket, Turkeyland, Slough, Jobs Neck Coves). Finally, sediment-water column nitrogen exchange within the salt marsh dominated tidal river, Wild Harbor River, was also similar to other nearby Buzzards Bay tidal marsh basins. For example the creeks of the salt marsh dominated portions of the Back River (Bourne) to the north and the Slocums and Little River Estuaries (Dartmouth) across the Bay support similarly small net release rates of 6.5 mg N m^{-2} d^{-1} and 4.6-9.0 mg N m^{-2} d^{-1}, respectively. It appears that the sediment nitrogen release rates within the component basins to the Wild Harbor System are very comparable to analogous basins in other estuaries in the region, and specifically in Buzzards Bay. Net nitrogen flux rates for use in the water quality modeling effort for the component sub-basins of the Wild Harbor Embayment System (Chapter VI) are presented in Table IV-6.

There was a clear pattern of sediment N flux, with the more depositional and organic sediments of the inner basin supporting a moderate level of net nitrogen release, 32.0 mg N m^{-2} d^{-1}. Both the observed rates and their spatial distribution are similar to other estuarine basins in the region. A similarly configured estuary, Lagoon Pond (Martha's Vineyard) was found to have net nitrogen uptake in the basin formed behind the barrier beach (-2.3 mg N m^{-2} d^{-1}) and net release in the inner depositional basins of 8.4 and 31.8 mg N m^{-2} d^{-1}. The tributary embayment of Polpis Harbor (Nantucket Harbor) showed net nitrogen release (East Polpis, 14.6 mg N m^{-1} d^{-1}; West Polpis 65.9 mg N m^{-1} d^{-1}).

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

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			i.d. *
	Mean	**S.E.**	**N**	
Wild Harbor Embayment System				
Wild Harbor Main Basin	-36.7	9.7	7	1,2,3,4,5,6,16
Wild Harbor Boat Basin	-17.1	12.1	2	7,8
Wild Harbor River	1.4	13.1	6	9,10,11,12,13,15

* Station numbers refer to Figure IV-8.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This hydrodynamic study, funded by the Massachusetts Department of Ecological Restoration, was performed for Wild Harbor, located within the town of Falmouth, Massachusetts, on the eastern shoreline of Buzzards Bay. It is the receiving basin of groundwater flow from the densely developed Silver Beach neighborhood. A topographic map detail in Figure V-1 shows the general study area. Wild Harbor is moderately deep coastal embayment with a wide opening to the Bay that is bound by Nyes Neck to the NW and Crow Point to the SE. The average depth of the main Harbor basin bottom is -9.0 feet mean low water (MLW). An inner harbor area at Silver Beach and the Wild Harbor River (Figure V-2) are the two main attached sub-embayments of the harbor. The mean depth of the inner harbor is -4.3 feet MLW while the mean depth of the Wild Harbor River (excluding marsh plain) is only 0.0 feet MLW. The total surface coverage of the Wild Harbor system is approximately 140 acres, which includes 15 acres of salt marsh.

Circulation in the Harbor is dominated by tidal exchange with Buzzards Bay. From measurements made in the course of this study, the average tide range offshore from the Pond is 3.6 feet. As indicated by the lack of significant tide range attenuation at the upper portion of Wild Harbor River, tidal flushing is generally very efficient throughout the tidal reaches of the Harbor system. Two small salt ponds with outlets to Wild Harbor River have much less efficient tidal exchange. Potters Hole, a 3.7 acre pond at the southern-most reach of the system with an average bottom elevation of -0.5 feet MLW, is attached to Wild Harbor River through a tidal creek with two bridge crossings (Figure V-3). The mean tide range of Potters Hole is 2.6 feet, or a foot less than in the River. A second un-named pond (hence referred to as Noname Pond) with a small 2.5 foot concrete culvert under Bay Shore Drive (Figure V-4), is more severely restricted and has only a 1.1 foot mean tide range. Though the culvert and bridge abutments of these two small ponds do act to restrict tidal flow. With no historical data, it is not possible to assess how the present tide range of these two small ponds compares to the range that existed when the ponds were in their natural state. It is possible that these ponds were just as tidally restricted or even non-tidal prior to the construction of the road crossings.

The hydrodynamic study of the Wild Harbor system proceeded as two component efforts. In the first portion of the study, bathymetry and tide data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Wild Harbor was performed to determine the variation of embayment depths throughout the system. This survey addressed the previous lack of adequate bathymetry data for this area. In addition to the bathymetry survey, tides were recorded at six stations for 30 days. These tide data were necessary to run and calibrate the hydrodynamic model of the system.

A numerical hydrodynamic model of Wild Harbor and its attached sub-embayments was developed in the second portion of this study. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from Buzzards Bay were used to define the open boundary condition that drives the circulation of the model, and data measured within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

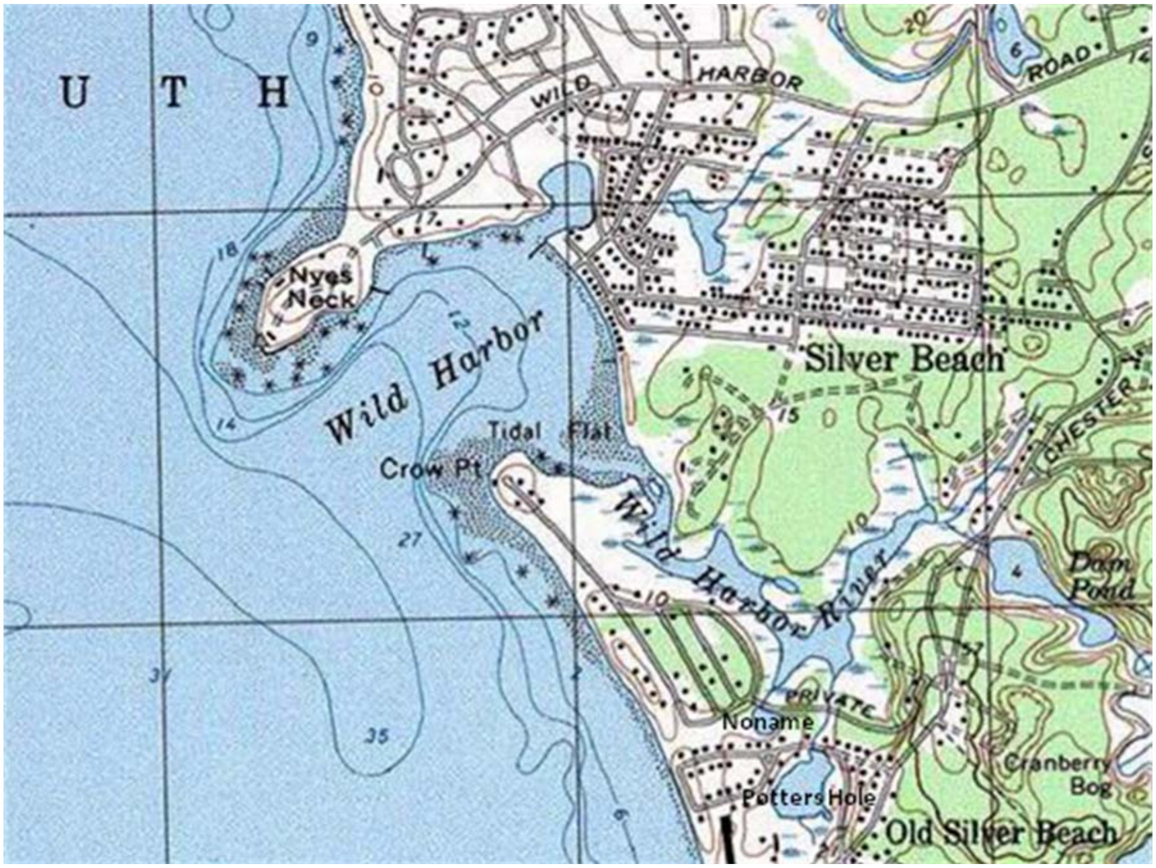

Figure V-1. Topographic map detail of Wild Harbor, including Wild Harbor River.

Figure V-2. View of marsh areas in Wild Harbor River.

Figure V-3. View of Santuit Road Bridge over Potters Hole Creek.

Figure V-4. View of Noname Pond culvert.

The calibrated hydrodynamic model of Wild Harbor is an integral piece of water quality model developed in the next chapter of this report. In addition to its use as the hydrodynamic basis for the TN and salinity models, the calibrated hydrodynamic model is a useful tool that can be used to investigate the tidal properties of the system.

V.2 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of Wild Harbor. Bathymetry data were collected throughout the system so that it could be accurately represented as a computer hydrodynamic model and flushing rates could be determined for the system. In addition to the bathymetry, tide data were also collected throughout the Harbor and River system (including the two attached small salt ponds and the inner harbor), in order to run the circulation model with real tides, and also to calibrate and verify its performance.

V.2.1 Bathymetry Data

A detailed bathymetric survey of Wild Harbor and Wild Harbor River was performed May 27 and June 14, 2005. A fathometer was used to take continuous soundings of the bottom as the survey vessel moved through the water. Positioning data were collected using a differential GPS. The actual survey paths followed by the survey craft are shown in Figure V-5. Soundings of Noname Pond and Potters Hole where made using a canoe and survey rod. Elevations along Potters Hole Creek and the Wild Harbor River marsh plain were measured using a survey level. The elevations were later confirmed using an RTK GPS system. The NOAA GEODAS data archive was used to as a source of bathymetry data for offshore areas in Buzzards bay not covered in the 2005 survey.

The resulting bathymetric surface created by interpolating the data to a finite element mesh is shown in Figure V-6. All soundings were tide corrected using tide data collected in the Ponds. The data were all rectified to the NGVD 29 vertical datum using benchmarks provided by the town. The benchmark elevations were confirmed using an RTK GPS system.

Results from the survey show that the deepest point in Wild Harbor is located at the opening to Buzzards Bay. The deepest depth measured in the course of the 2005 survey is -31.2 feet MLW. Generally, the average depth of the whole system is very shallow. Not including the marsh plain or any area offshore of the line between Nyes Neck and Crow Point, the mean depth of the system is -6.5 feet NGVD.

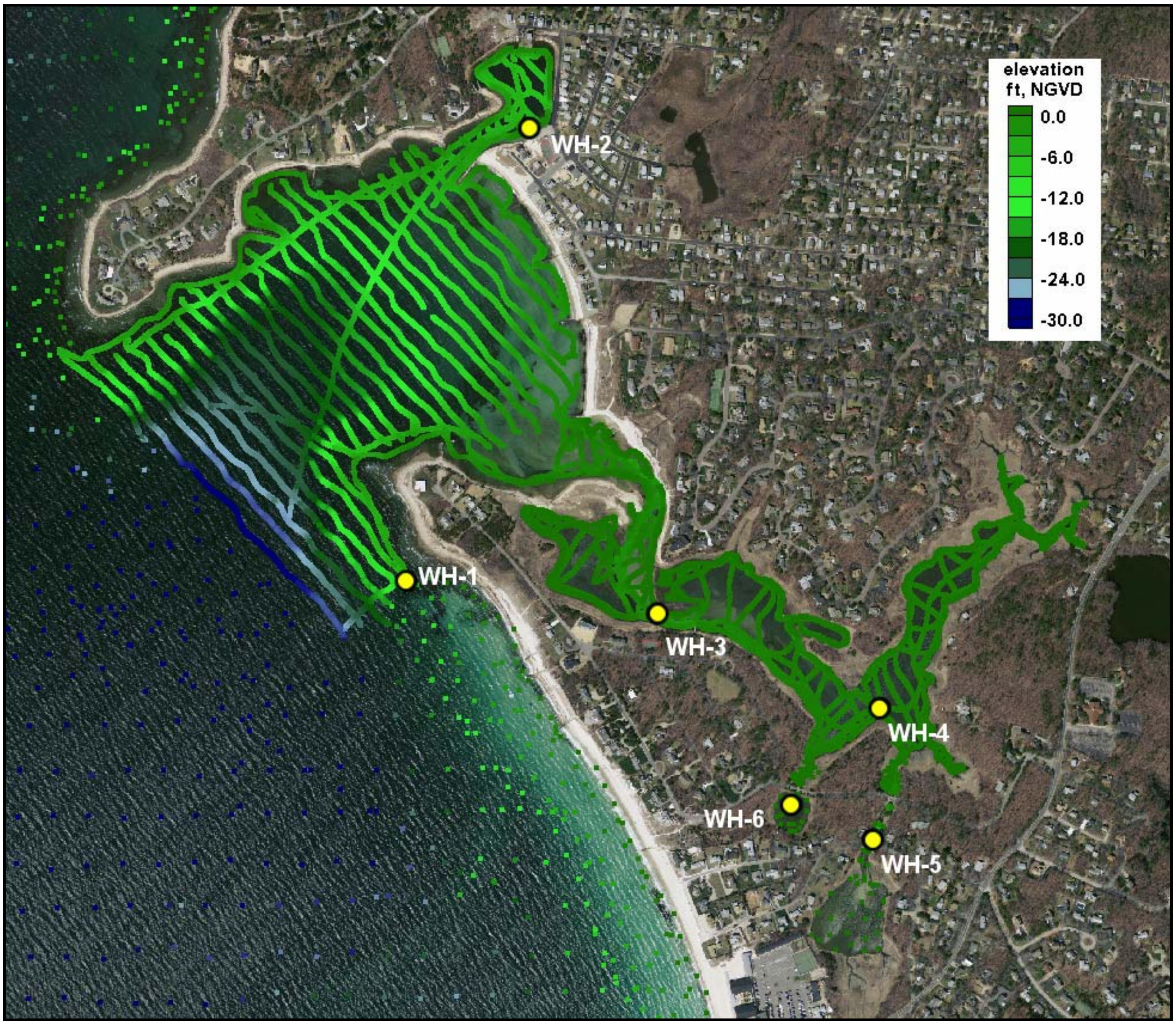

Figure V-5. Transects from the May and June 2005 bathymetry survey of Wild Harbor. Yellow markers show the locations of the tide recorders deployed for this study (WH-1 Buzzards Bay, WH-2 Inner Harbor, WH-3 Lower River, WH-4 Upper River, WH-5 Potters Hole, and WH-6 Noname Pond).

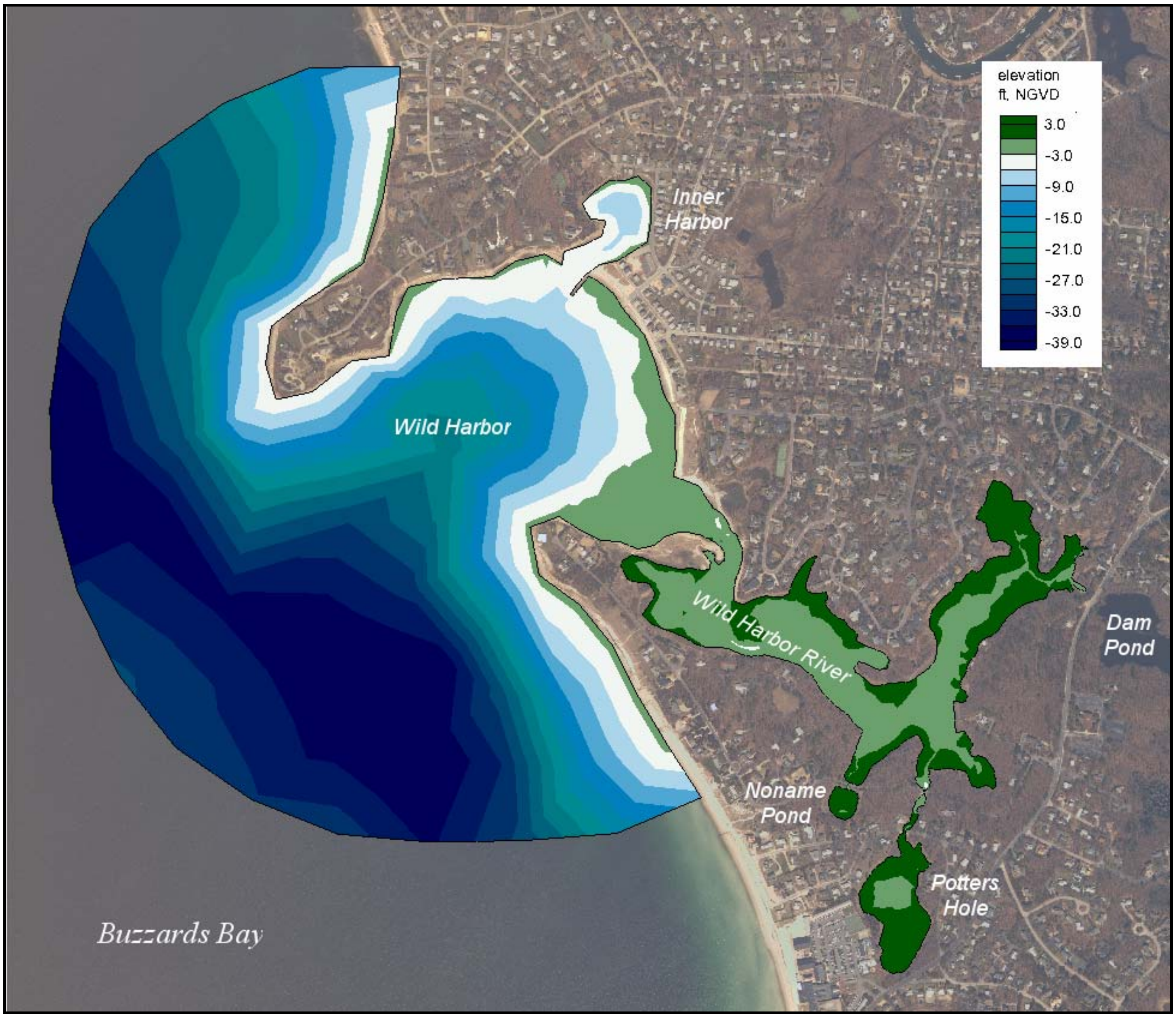

Figure V-6. Bathymetry data interpolated to the finite element mesh used with the RMA-2 hydrodynamic model. Contours represent the bottom elevation relative to mean low water (NGVD). The primary data source used to develop the grid mesh is the May/June 2005 survey of the system, with NOAA GEODAS data used for the offshore area.

V.2.2 Tide Data Collection and Analysis

Tide data records were collected concurrently at six gauging stations shown in Figure V-5, located in Buzzards Bay (WH-1), at the inner harbor (WH-2), at lower and upper portions of Wild Harbor River (WH-3 and WH-4), and in the two small salt ponds attached to Wild Harbor River (WH-5 and WH-6). The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 30-day period between May 17 and June 17, 2005. The elevation of each gauge was surveyed relative to the NGVD vertical datum. The Buzzards Bay tide record was used as the open boundary condition of the hydrodynamic model. Data from inside the system were used to calibrate the model.

Tide records longer than 29 days are necessary for a complete evaluation of tidal dynamics within the estuarine system. Although a one-month record likely does not include extreme high or low tides, it does provide an accurate basis for typical tidal conditions governed by both lunar and solar motion. For numerical modeling of hydrodynamics, the typical tide conditions associated with a one-month record are appropriate for driving tidal flows within the estuarine system.

Plots of the tide data from the three gauges are shown in Figure V-7 for the entire 30-day deployment. The spring-to-neap variation in tide range is discernable in these plots. The data record begins during a period of neap tides, where the minimum range is approximately 2 feet. A week later there is a period of spring tides, where the maximum range of 5.4 feet occurs on May 23. This is one day before the full moon of that month. Following this spring tide is a continuing cycle of neap and spring tides, though the transition is more muted than at the beginning of the month.

A visual comparison between tide elevations offshore and at the different stations in the system shows that the tide amplitude does not change much, even along the reach of the River. There is a great attenuation of the range in Potters Hole and Noname Pond. This loss of amplitude is described as tidal attenuation. In these two small salt ponds, the attenuation of the tide signal from Wild Harbor River is due to flow restrictions caused by the culvert at Noname Pond and the Bridge abutments and long creek channel at Potters Hole.

To better quantify the changes to the tide from the inlet to inside the system, the 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 were 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.

Frictional damping is evident in the reduction of the mean tide range in Potters Hole and Noname Pond, compared to the mean range offshore. The tide range in Noname Pond is the smallest in the system. It is only 1.1 feet, or 30% of the mean offshore range. Damping not only affects the range of the observed tide, it also causes a time lag in the time of high and low tide. Figure V-8 shows how the time of high in Noname Pond lags 40 minutes from the river. The time lag for low tide is larger. As Noname Pond slowly drains through its culvert, the time of low tide in the pond occurs almost two hours after low tide in the river.

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

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 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-9. The amplitudes and phase of 21 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of seven tidal constituents computed for the Wild Harbor station records. The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor

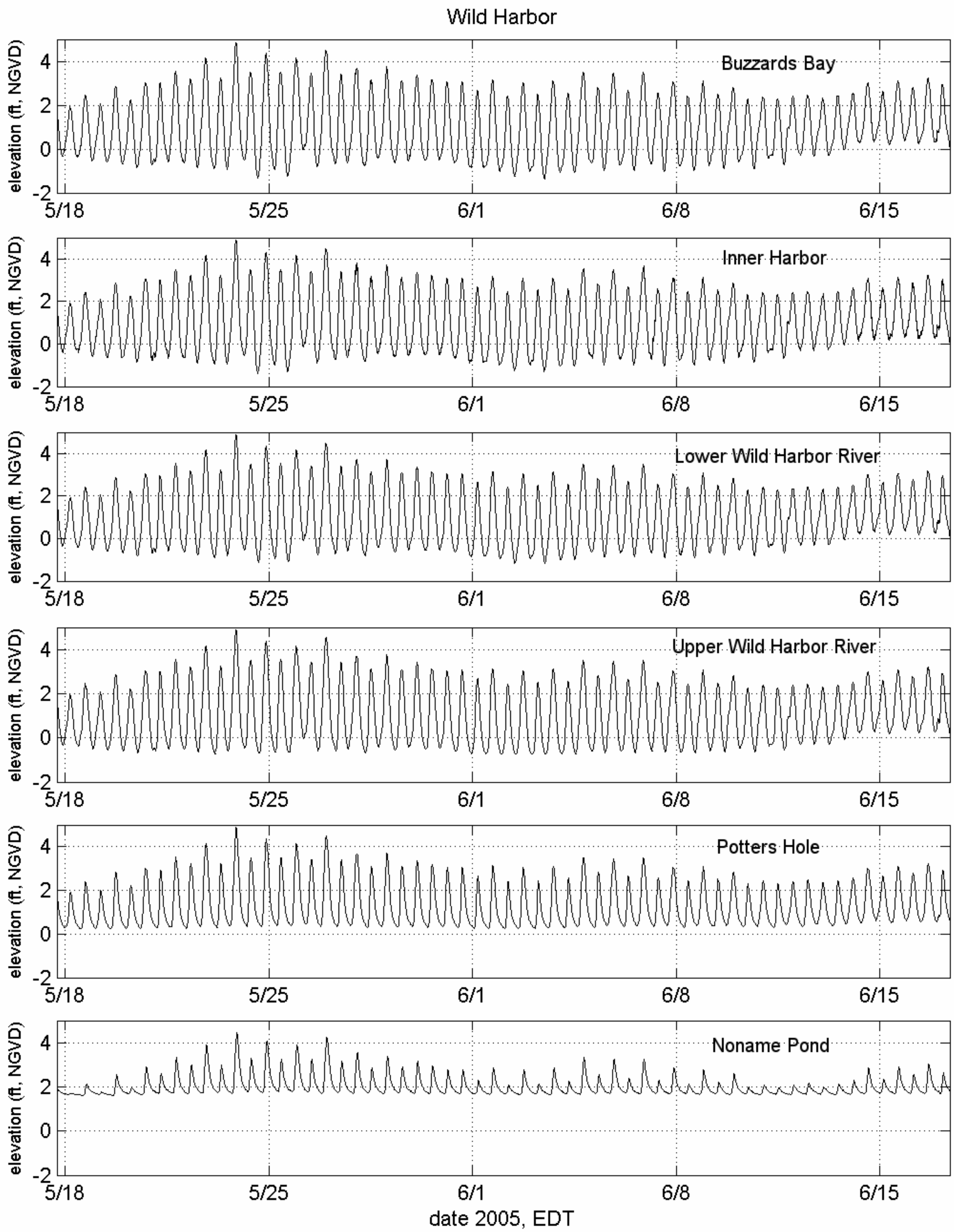

Figure V-7. Plots of observed tides for stations in Wild Harbor, for the 30-day period between May 17 and June 17, 2005. All water levels are referenced to the NGVD vertical datum.

Table V-1. Tide datums computed from 30-day records collected offshore and in the Wild Harbor system in May and June 2005. Datum elevations are given relative to NGVD vertical datum.						
Tide Datum	Buzzards Bay (feet)	Inner Harbor (feet)	Lower Wild Harbor River (feet)	Upper Wild Harbor River (feet)	Potters Hole (feet)	Noname Pond (feet)
Maximum Tide	4.9	4.9	4.9	4.9	4.9	4.5
MHHW	3.3	3.3	3.3	3.3	3.3	3.1
MHW	3.0	3.1	3.0	3.0	3.0	2.8
MTL	1.2	1.2	1.2	1.3	1.7	2.2
MLW	-0.6	-0.7	-0.6	-0.5	0.4	1.7
MLLW	-0.7	-0.8	-0.7	-0.5	0.3	1.7
Minimum Tide	-1.4	-1.4	-1.2	-0.7	0.2	1.6
Mean Range	3.6	3.8	3.6	3.5	2.6	1.1

to the signal with an offshore amplitude of 1.7 feet. The total range of the M_2 tide is twice the amplitude, or 3.4 feet.

The diurnal tides (once daily), K_1 and O_1, possess amplitudes of approximately 0.3 feet and 0.2 respectively. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, also contribute to the total tide signal, with amplitudes of 0.3 feet and 0.4 feet, respectively. 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.

Generally, it can be seen that as the total tide range is attenuated through the system there is a corresponding reduction in the amplitude of the individual tide constituents. One exception is the M_4 and M_6 amplitudes in Potters Hole, which are larger than for the offshore station. Again, this is due to energy transferring from the M_2 to these overtides due to frictional losses across the system.

Though there is little change in constituent amplitudes across the length of the main channel of the River, the phase change of the tide is easily seen from the results of the harmonic analysis. Table V-3 shows the delay of the M_2 at different points in the Wild Harbor system, relative to the timing of the M_2 constituent in Buzzards Bay, offshore the harbor entrance. Between the offshore and the upper river, there is only a 16 minute delay in the M_2. This is not a large delay, considering that the period of this constituent is more than 12 hours. A larger delay is seen in the analysis of the data from Potters Hole and Noname Pond. There is a 51 and 87 minute delay respectively in the phasing of the M_2 between the offshore and these embayments.

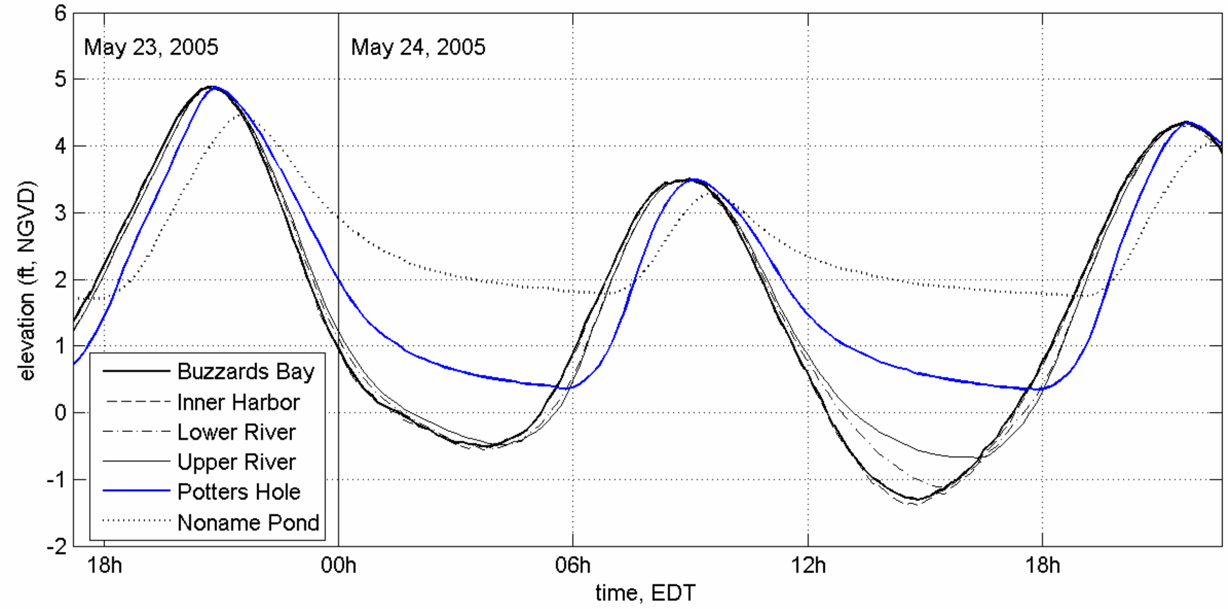

Figure V-8. Four-day tide plot showing tides measured in Buzzards Bay and at stations in the Wild Harbor system. Demonstrated in this plot is the frictional damping effect caused by the culvert connection between the inner reaches of Wild Harbor River and the Bay. The damping effects are seen as a reduction in tidal amplitude, as well as the lag in time of high and low tides from the offshore tide.

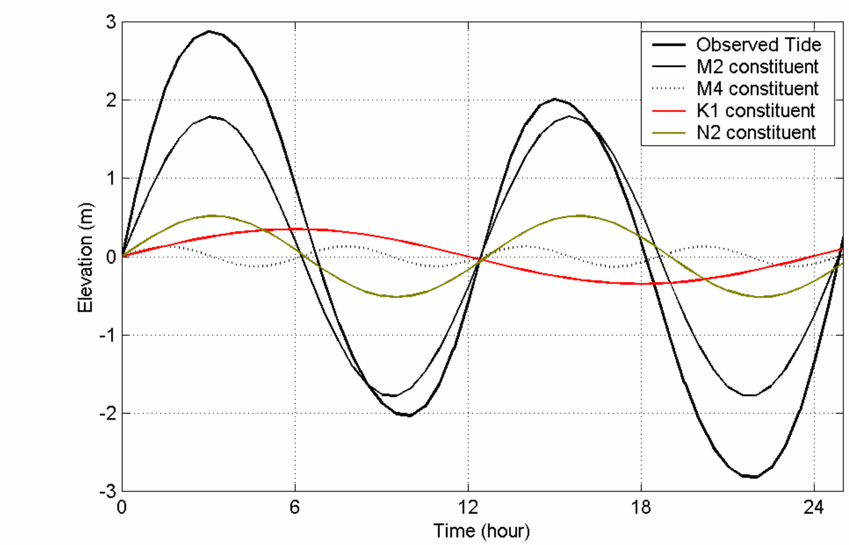

Figure V-9. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Tidal Constituents computed for tide stations in the Wild Harbor system and offshore in Buzzards Bay, May to June 2005.							
	Amplitude (feet)						
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82
Buzzards Bay	1.72	0.28	0.03	0.27	0.39	0.28	0.23
Inner Harbor	1.74	0.28	0.03	0.27	0.39	0.29	0.23
Lower River	1.70	0.27	0.05	0.26	0.39	0.29	0.23
Upper River	1.66	0.29	0.04	0.25	0.39	0.28	0.22
Potters Hole	1.17	0.39	0.09	0.17	0.30	0.23	0.20
Noname Pond	0.40	0.19	0.07	0.09	0.16	0.14	0.14

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 measured water elevation records for the gauge records in Wild Harbor compared to the energy content the astronomical tidal signal (re-created by summing the contributions from the 21 constituents determined by the harmonic analysis) is presented in Table V-3. 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-10 shows the comparison of the measured tide from Buzzards Bay, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-4 shows that the variance of tidal energy was largest in the offshore signal, as should be expected. The analysis also shows that tides are responsible for approximately 95% of the water level changes in Buzzards Bay and most of Wild Harbor. This indicates that the hydrodynamics of the system is influenced predominantly by astronomical tides.

The non-tidal residual is largest by percentage in Noname Pond, where the greatest tide attenuation occurs. Though the residual signal has an energy content that is more than 20% of the total water level fluctuation of the pond, the residual signal has the same absolute amplitude of the residual signal offshore. This means that the residual signal does not grow between the Bay and the Pond, which indicates that the residual signal in the pond results from water level fluctuations offshore in the Bay and not from some more localized phenomena such as wind.

Table V-3. M_2 tidal constituent phase delay (relative to Buzzards Bay) for gauge locations in the Wild Harbor system, determined from measured tide data.	
Station	Delay (minutes)
Inner Harbor	0.1
Lower Wild Harbor River	10.7
Upper Wild Harbor River	16.0
Potters Hole	50.8
Noname Pond	86.7

Table V-4. Percentages of Tidal versus Non-Tidal Energy for stations in the Wild Harbor system and Buzzards Bay, May to June 2005.			
TDR Location	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Buzzards Bay	1.77	94.8	5.2
Inner Harbor	1.69	95.0	5.0
Lower River	1.63	95.0	5.0
Upper River	1.55	94.9	5.1
Potters Hole	0.88	93.4	6.6
Noname Pond	0.19	88.6	21.4

V.3 HYDRODYNAMIC MODELING

For the modeling of the Wild Harbor system, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in the Pond. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2 for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, Chatham embayments (Kelley, *et al*, 2001), Falmouth "finger" Ponds (Howes *et al*, 2005), Three Bays (Kelley *et al*, 2003) and Barnstable Harbor (Wood, *et al*, 1999).

V.3.1 Model Theory

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

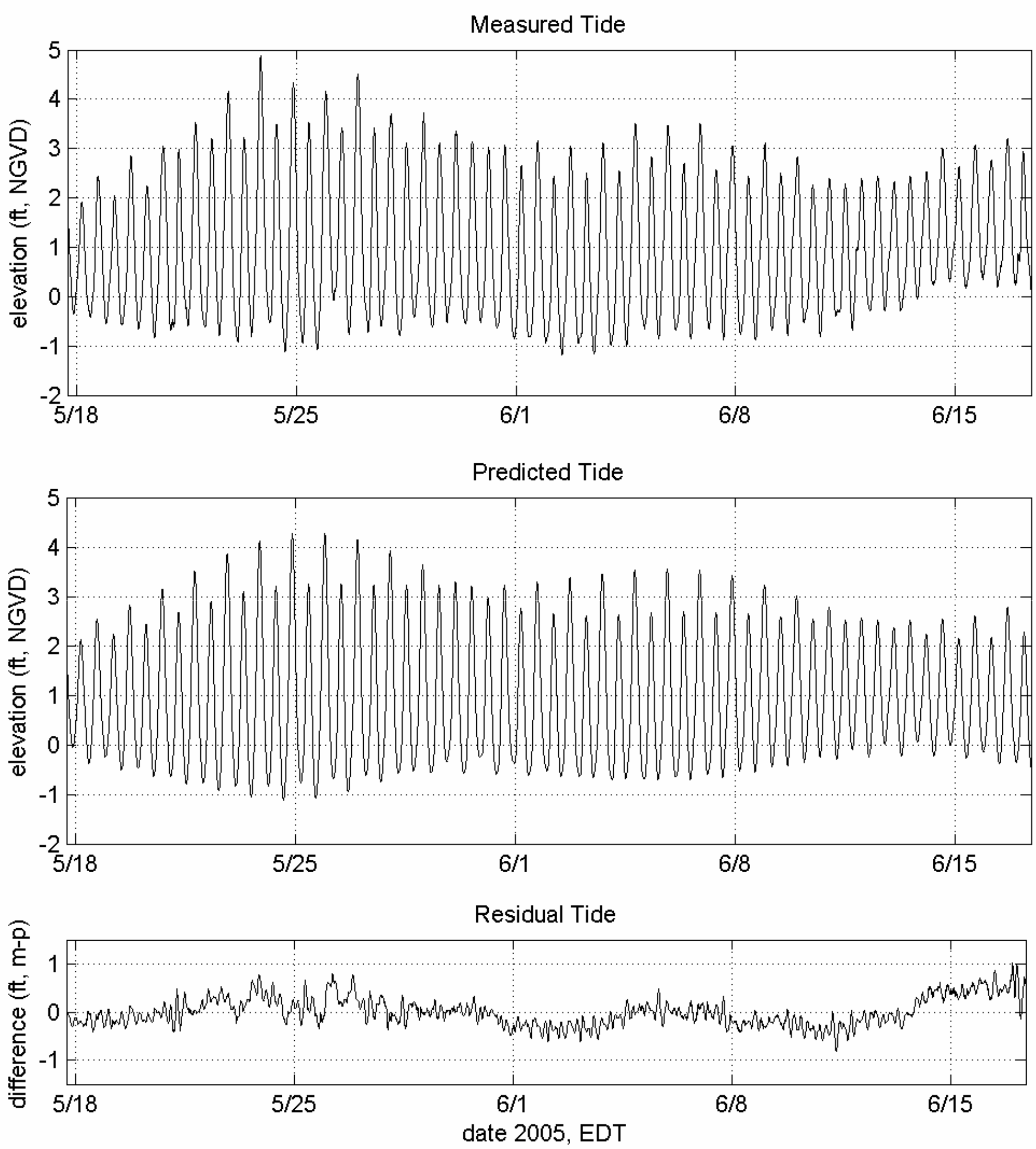

Figure V-10. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 21 individual tide constituents determine in the harmonic analysis of the Buzzards Bay gauge data, collected offshore Wild Harbor. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

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

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

V.3.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using 2005 and 2009 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of the Wild Harbor grid based on the tide gauge data collected offshore in Buzzards Bay. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid generation

The grid generation process was aided by the use of the SMS package. 2005 and 2009 digital aerial orthophotos and the 2005 bathymetry survey data were imported to SMS, and a finite element grid was generated to represent the estuary. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of salt marsh. The bathymetry data were interpolated to the developed finite element mesh of the system. The completed grid consists of 5694 nodes, which describe 2586 total 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -47ft (NGVD) along the open boundary of the grid in Buzzards Bay, and the typical modeled marsh plain elevation is 2.5 ft, based on RTK GPS measurements. The completed grid mesh of the Wild Harbor system is shown in Figure V-11.

The finite element grid for the system provides the detail necessary to evaluate accurately the variation in hydrodynamic properties of Wild Harbor. Areas of marsh were included in the model because they represent a significant portion of the total surface area of this system. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Grid resolution is generally governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution is employed where complex flow patterns are expected, generally near the inlet. Appropriate implementation of wider node spacing and larger elements reduces computer run time with no sacrifice of accuracy.

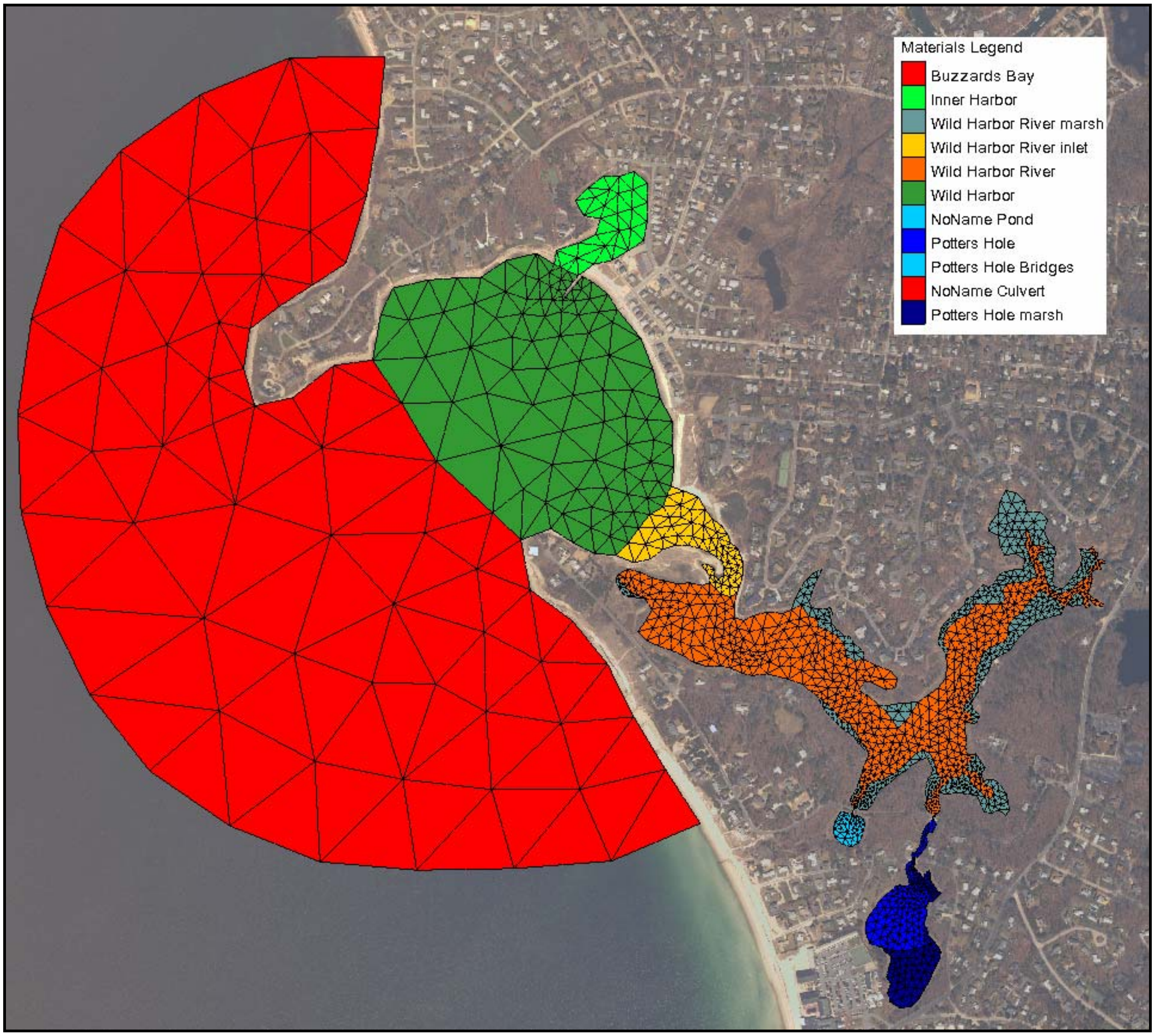

Figure V-11. Plot of hydrodynamic model grid mesh for Wild Harbor. Colors are used to designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.3.2.2 Boundary condition specification

Three types of boundary conditions were employed for the RMA-2 model of the Wild Harbor system: 1) "slip" boundaries, 2) tidal elevation boundaries, and 3) constant flow input boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified at the inlet from Buzzards Bay. TDR measurements provided the required data. The rise and fall of the tide in the Bay is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at the open boundary of the Wild Harbor grid every model time step. The model runs of Wild Harbor used a 10-minute time step, which the same as the 10-minute sampling rate of the measured tide data. Details concerning the constant flow input boundary conditions included in the hydro model are discussed in Chapter VI.

V.3.2.3 Calibration

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

Calibration of the hydrodynamic model required a close match between the modeled and measured tides from stations inside the system (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 14-day period (27 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section 2. The 14-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 14-day period beginning May 23, 2005 at 2000 EDT. This representative time period included one full cycle between spring and neap periods.

After the model was calibrated, an additional verification run was made in order test the model performance in a time period outside of the calibration period. The model verification was performed for the eight-day period beginning June 7, 2005 at 2130 EDT.

The calibrated model was used to analyze existing detailed flow patterns and compute residence times. The flushing analysis is based on the 14 day period beginning May 23, 2005, at 2000 EDT. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire 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.a Friction coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficients between 0.025 and 0.070 were specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels and marsh plains with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. For example, lower friction coefficients were specified for main basin of Wild Harbor, versus the marsh plain areas of the River, which provides greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based on 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 and eddy viscosity coefficients used in simulations of the Wild Harbor system. These embayment delineations correspond to the material type areas shown in Figure V-11.

System Embayment	bottom friction	eddy viscosity lb-sec/ft^2
Buzzards Bay	0.025	20.0
Inner Harbor	0.030	20.0
Wild Harbor River marsh	0.070	50.0
Wild Harbor River inlet	0.030	20.0
Wild Harbor River	0.030	30.0
Wild Harbor	0.025	20.0
Noname Pond	0.025	20.0
Potters Hole	0.035	20.0
Potters Hole bridges	0.040	150.0
Noname Pond culvert	0.090	50.0
Potters Hole Marsh	0.070	50.0

V.3.2.3.b Turbulent exchange coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 20 and 50 lb-sec/ft^2 (Table V-5). Higher values were used for the Potters Hole channel bridges.

V.3.2.3.c Marsh porosity processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain included in the model of the Wild Harbor 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.d Comparison of modeled tides and measured tide data

A best-fit of model output for the measured data was achieved using the aforementioned values for friction and turbulent exchange. Figures V-12 through V-17 illustrate sections the 14-day simulation periods for the calibration model. Modeled (solid line) and measured (dotted line) tides are illustrated at each model location with a corresponding TDR.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in the system embayments. Four tidal constituents were selected for constituent comparison: the K_1, M_2, M_4 and M_6. Measured tidal constituent amplitudes are shown in Table V-6 for the calibration and verification simulations. The constituent amplitudes shown in this table differ from those in Table V-2 because constituents were computed for only the separate 14-day sub-sections of the 30-days represented in Table V-2. In Tables V-6 and V-7, error statistics are shown for the calibration and verification.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The errors associated with tidal constituent amplitude for both the calibration and verification simulations were on the order of 0.01 ft, which is of the same order magnitude of the accuracy of the tide gages (0.032 ft). Time lag errors for the main estuary reach were less than the time increment resolved by the model and tide data (10 minutes), indicating good agreement between the model and data. The skill of the model calibration is also demonstrated by the high degree of correlation (R^2) and low RMS error shown in Table V-8 for all stations.

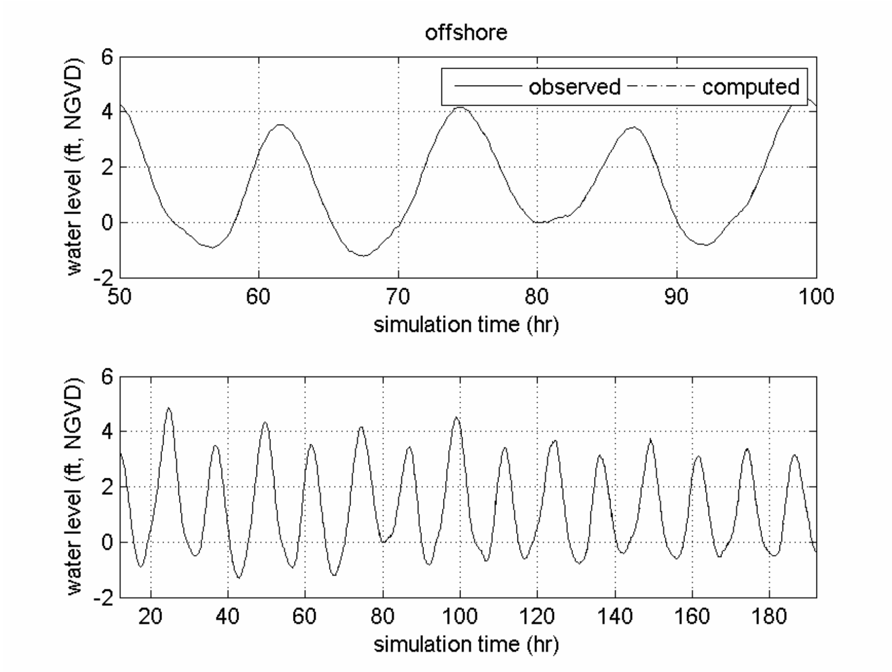

Figure V-12. Comparison of model output and measured tides for the TDR location offshore in Buzzards Bay for the final calibration model run (May 23, 2005 at 2000 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot.

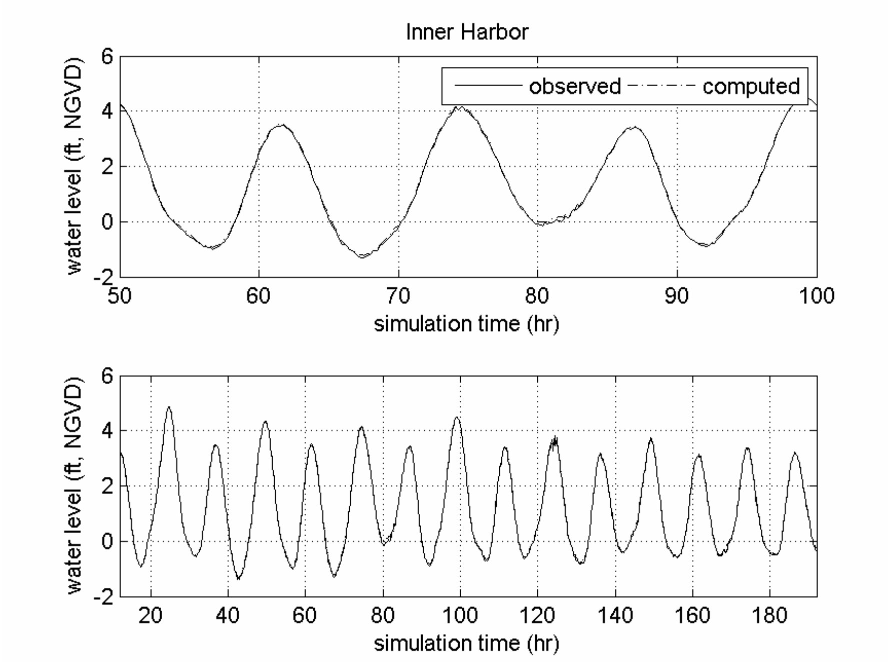

Figure V-13. Comparison of model output and measured tides for the TDR location in the inner harbor (WH-2) for the final calibration model run (May 23, 2005 at 2000 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot

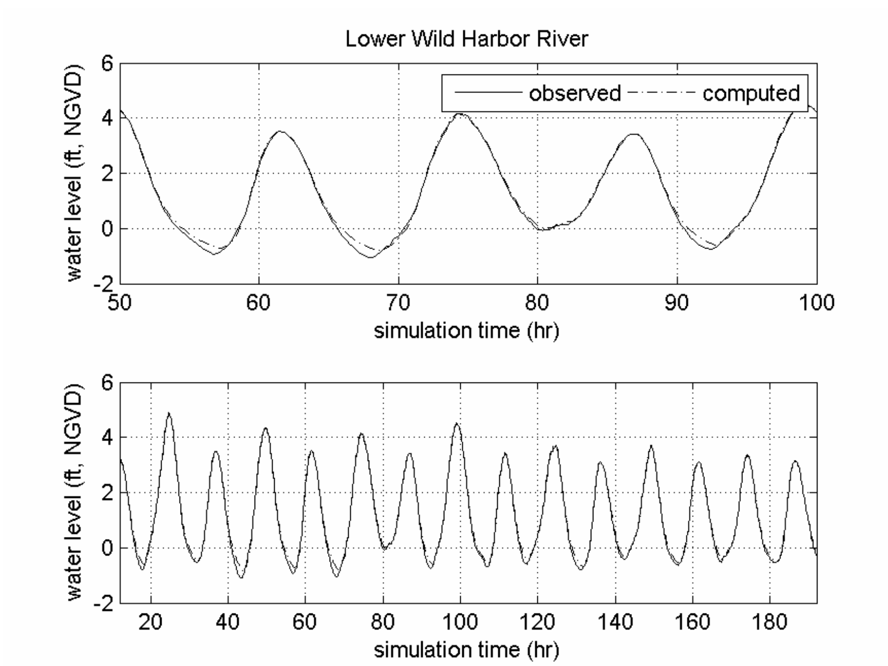

Figure V-14. Comparison of model output and measured tides for the TDR location at the lower portion of Wild Harbor River for the final calibration model run (May 23, 2005 at 2000 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot

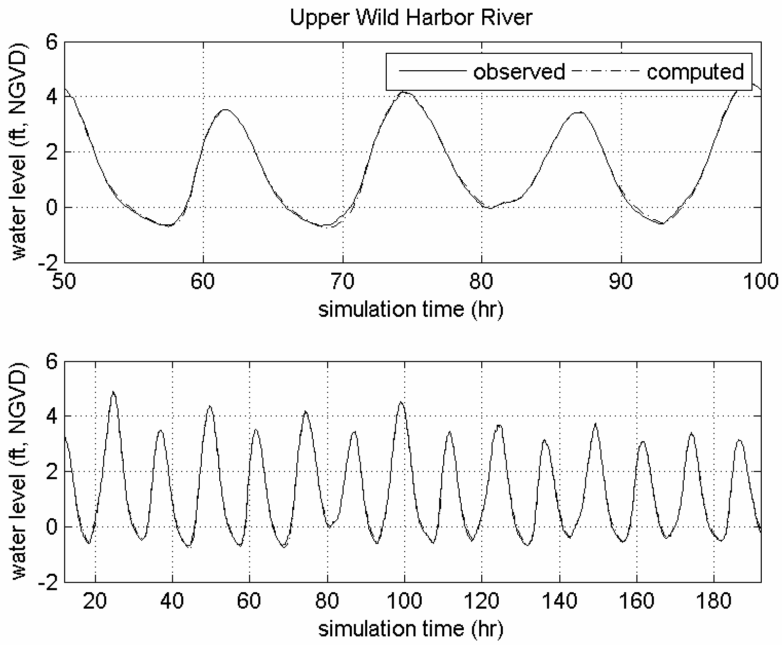

Figure V-15. Comparison of model output and measured tides for the TDR location at the upper portion of Wild Harbor River for the final calibration model run (May 23, 2005 at 2000 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot

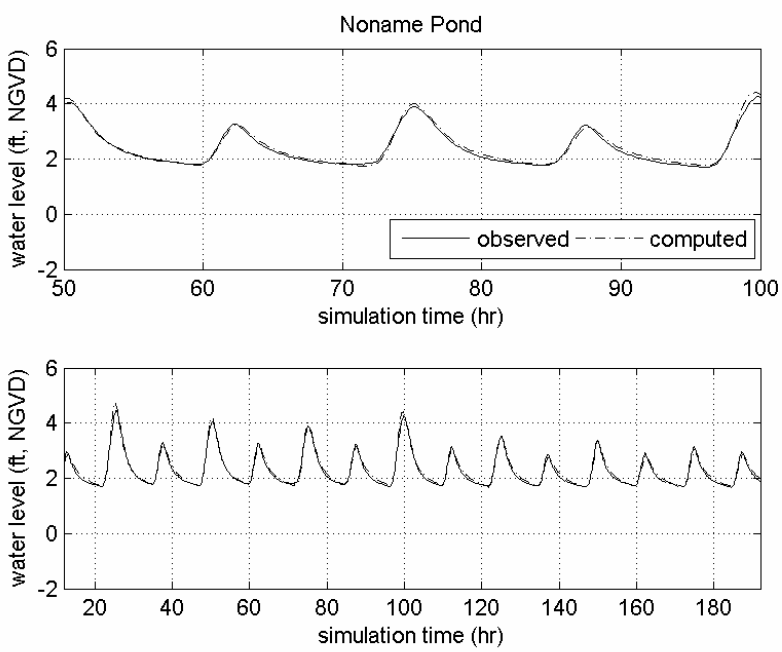

Figure V-16. Comparison of model output and measured tides for the TDR location in Noname Pond for the final calibration model run (May 23, 2005 at 2000 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot

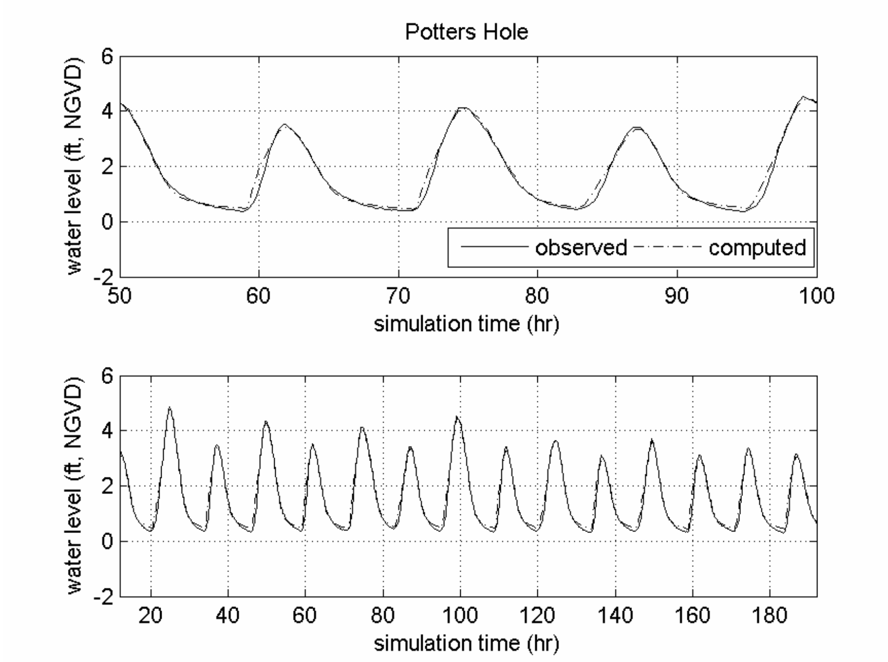

Figure V-17. Comparison of model output and measured tides for the TDR location in Potters Hole Creek for the final calibration model run (May 23, 2005 at 2000 EDT). The top plot is a 50-hour sub-section of the longer segment of the total modeled time period shown in the bottom plot

Table V-6. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for Wild Harbor, during modeled calibration time period.

Location	Constituent Amplitude (ft)				Phase (deg)	
Model calibration run						
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Buzzards Bay	1.90	0.28	0.02	0.36	20.4	47.4
Inner Harbor	1.90	0.28	0.02	0.36	20.4	47.4
Lower River	1.80	0.32	0.03	0.34	27.6	36.9
Upper River	1.78	0.34	0.04	0.35	29.9	35.7
Potters Hole	1.21	0.36	0.03	0.27	39.4	60.3
Noname Pond	0.47	0.19	0.06	0.19	66.4	86.9
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Buzzards Bay	1.95	0.31	0.02	0.34	20.8	49.4
Inner Harbor	1.96	0.31	0.02	0.34	20.9	49.0
Lower River	1.91	0.32	0.05	0.33	26.0	47.2
Upper River	1.85	0.36	0.04	0.33	28.6	42.6
Potters Hole	1.29	0.45	0.10	0.27	45.4	63.8
Noname Pond	0.50	0.23	0.09	0.17	62.7	90.8
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Buzzards Bay	0.05	0.03	0.00	-0.02	0.9	2.1
Inner Harbor	0.06	0.03	0.00	-0.02	1.0	1.6
Lower River	0.11	0.00	0.02	-0.01	-3.3	10.7
Upper River	0.07	0.02	0.00	-0.02	-2.8	7.0
Potters Hole	0.08	0.09	0.07	0.00	12.3	3.6
Noname Pond	0.03	0.04	0.03	-0.02	-7.6	4.0

V.3.4 Model Circulation Characteristics

The final calibrated model serves as a useful tool in investigating the circulation characteristics of the Wild Harbor 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. As an example, Figure V-18 shows color contours and vectors that indicate velocity during a single model time step, during a period of maximum flood currents at the inlet.

Table V-7. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for Wild Harbor, during modeled verification time period.

Model verification run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Buzzards Bay	1.35	0.26	0.02	0.17	-8.5	46.7
Inner Harbor	1.36	0.26	0.02	0.17	-8.5	46.7
Lower River	1.32	0.20	0.05	0.17	-2.6	41.1
Upper River	1.32	0.18	0.05	0.17	-0.8	38.8
Potters Hole	0.92	0.23	0.03	0.12	13.8	30.7
Noname Pond	0.24	0.10	0.04	0.04	61.8	72.8
Measured tide during verification period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Buzzards Bay	1.35	0.25	0.02	0.16	-8.5	46.9
Inner Harbor	1.36	0.25	0.02	0.16	-13.1	38.3
Lower River	1.34	0.22	0.04	0.17	-4.5	47.1
Upper River	1.33	0.19	0.04	0.16	-2.3	42.3
Potters Hole	0.98	0.27	0.06	0.14	16.4	18.6
Noname Pond	0.22	0.11	0.05	0.03	51.7	63.4
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Buzzards Bay	0.00	-0.01	0.00	-0.01	0.0	0.2
Inner Harbor	0.00	-0.01	0.00	-0.01	-9.6	-8.7
Lower River	0.02	0.02	-0.01	0.00	-3.9	6.2
Upper River	0.01	0.01	-0.01	-0.01	-3.1	3.6
Potters Hole	0.06	0.04	0.03	0.02	5.2	-12.5
Noname Pond	-0.02	0.01	0.01	-0.01	-21.1	-9.7

Table V-8. Error statistics for the Wild Harbor hydrodynamic model, for model calibration and verification.

	Calibration		Verification	
	R^2	RMS error	R^2	RMS error
Buzzards Bay	1.00	0.00	0.99	0.10
Inner Harbor	1.00	0.04	1.00	0.05
Lower River	0.99	0.12	0.99	0.09
Upper River	1.00	0.07	0.99	0.08
Potters Hole	0.97	0.17	0.96	0.15
Noname Pond	0.98	0.08	0.96	0.05

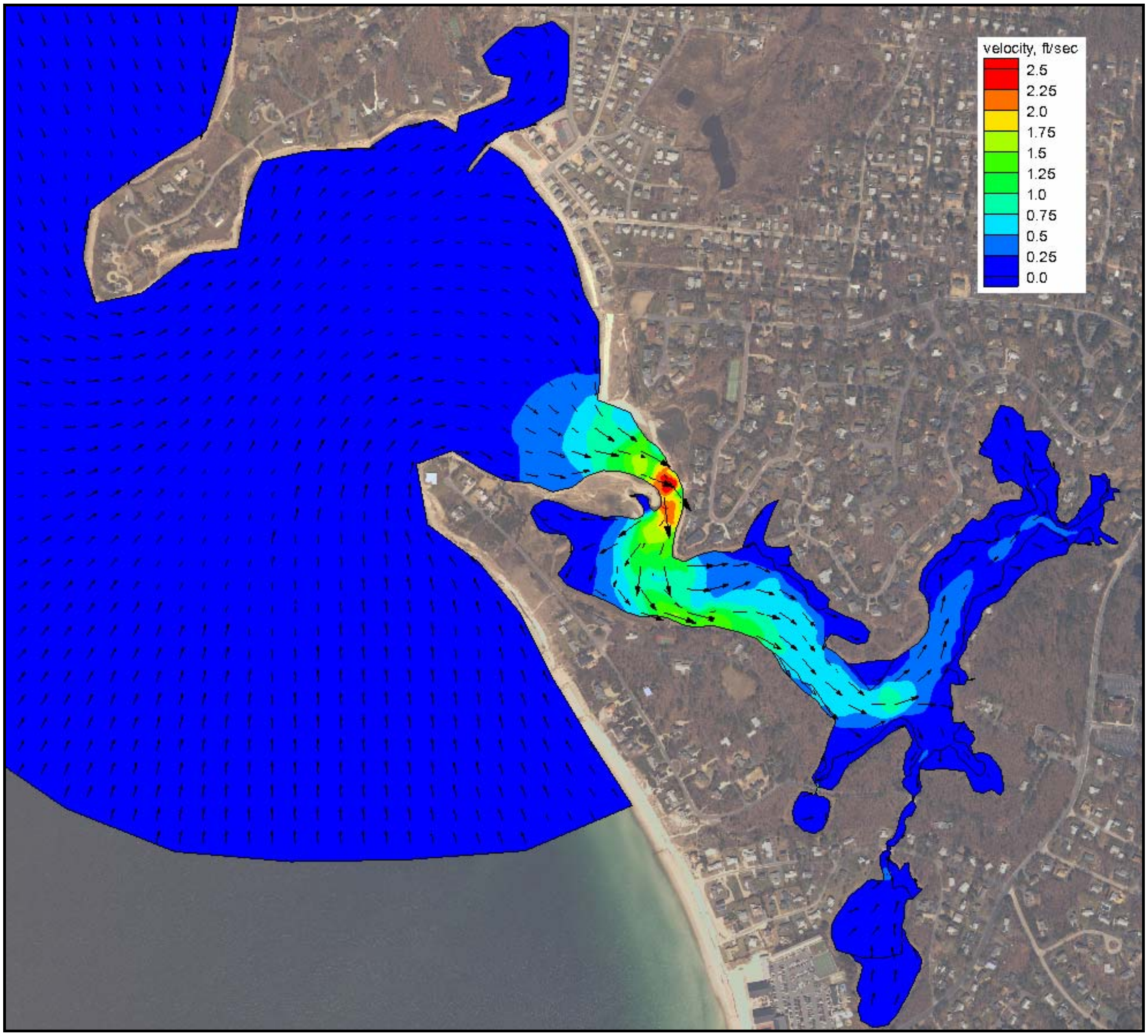

Figure V-18. Example of Wild Harbor hydrodynamic model output for a single time step during a flooding tide at the Wild Harbor River inlet. Color contours indicate velocity magnitude, and vectors indicate the direction of flow. Areas of marsh are also shown as the solid black lines within the model domain.

As another example, from the calibration model run of the Wild Harbor system, the total flow rate of water flowing through the inlet culvert can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-19. During spring tides, the maximum flood flow rates reach 2000 ft^3/sec at the Wild Harbor River inlet. Maximum ebb flow rates during spring tides are similar.

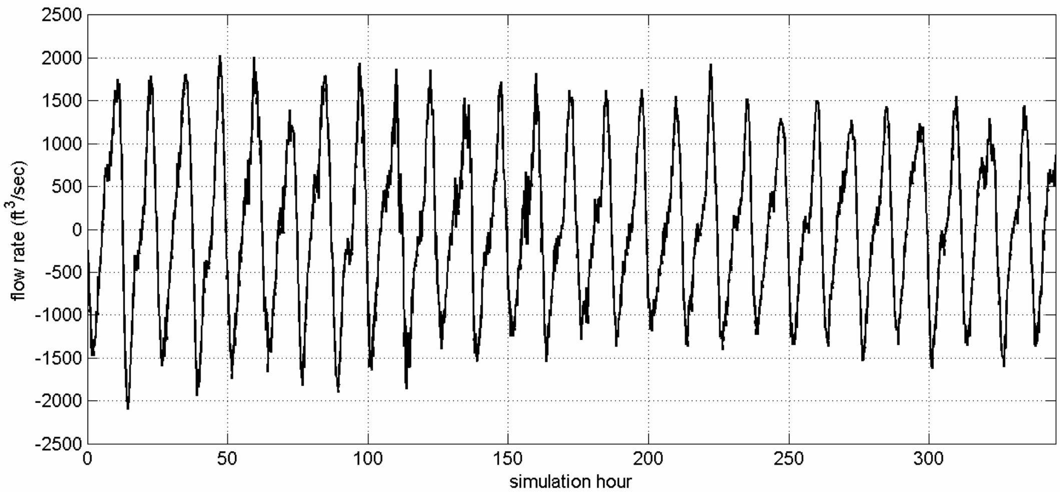

Figure V-19. Time variation of computed flow rates at the Wild Harbor River inlet. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Positive flow indicated flooding tide flows, while negative flow indicates ebbing tide flows.

Using the velocities computed in the model, an investigation of the flood or ebb dominance of different areas in the Wild Harbor system can be performed. Marsh systems are typically flood dominant, meaning that maximum flood tide velocities are greater than during the ebb portion of the tide. Flood dominance indicates a tendency to collect and trap sediment, which is required to maintain healthy marsh resources.

Flood or ebb dominance in channels of a tidal system can be determined by performing a harmonic analysis of tidal currents. A discussion of the method of relative phase determination is presented in Friedrichs and Aubrey (1988). For this method, the M_2 and M_4 tidal constituents of a tidal velocity time series are computed, similar to the tidal elevation constituents presented in Section V.3.2.

The relative phase difference is computed as the difference between two times the M_2 phase and the phase of the M_4, expressed as $\Phi=2M_2-M_4$. If Φ is between 270 and 90 degrees ($-90<\Phi<90$), then the channel is characterized as being flood dominant, and peak flood velocities will be greater than for peak ebb. Alternately, if Φ were between 90 and 270 degrees ($90<\Phi<270$), then the channel would be ebb dominant. If Φ is exactly 90 or 270 degrees, neither flood nor ebb dominance occurs. For Φ equal to exactly 0 or 180 degrees, maximum tidal distortion occurs and the velocity residuals of a channel are greatest. This relative phase relationship is presented graphically in Figure V-20.

Though this method of tidal constituent analysis provides similar results to a visual inspection of a velocity record (e.g., by comparing peak ebb and flood velocities), it allows a more exact characterization of the tidal processes. By this analysis technique, a channel can be characterized as being strongly, moderately, or weakly flood or ebb dominant.

Five points were selected for this velocity analysis: 1) entrance to the inner harbor, 2) the Wild Harbor River inlet, 3) the upper portion of Wild Harbor River, 4) Potters Hole Creek and 5) Noname Pond Culvert.

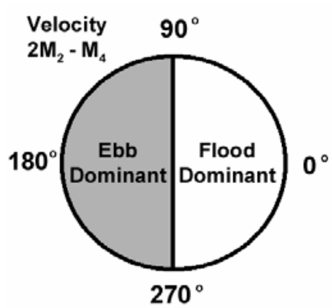

Figure V-20. Relative velocity phase relationship of M2 and M4 tidal velocity constituents and characteristic dominance, indicated on the unit circle. Relative phase is computed as the difference of two times the M2 phase and the M4 phase (2M2-M4). A relative phase of exactly 90 or 270 degrees indicates a symmetric tide, which is neither flood nor ebb dominant.

The results of this velocity analysis of the Wild Harbor model output show that, though the inlet area of the main basin is ebb dominant, the river (starting at the inlet) and its attached sub-embayments are indeed flood dominant, as is expected for a marsh. The computed values of $2M_2$-M_4 are presented in Table V-9.

Table V-9. Wild Harbor relative velocity phase differences of M_2 and M_4 tide constituents, determines using velocity records.

location	$2M_2$-M_4 relative phase (deg)	Characteristic dominance
Inner Harbor entrance	240.5	Moderate Ebb
Wild Harbor River Inlet	300.2	Moderate Flood
Upper Wild Harbor River	308.4	Moderate Flood
Potters Hole	325.9	Moderate Flood
Noname Pond	294.8	Moderate Flood

V.3.5 Flushing Characteristics

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

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

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

where T_{system} denotes the residence time for the system, V_{system} represents volume of the (entire) system at mean tide level, P equals the tidal prism (or volume entering the system through a single tidal cycle), and t_{cycle} the period of the tidal cycle, typically 12.42 hours (or 0.52 days). To compute system residence time for a sub-embayment, the tidal prism of the sub-embayment replaces the total system tidal prism value in the above equation.

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Potters Hole as an example, the **system residence time** is the average time required for water to migrate from Potters Hole, through the mid-reach of the Wild Harbor River, out through Wild Harbor, and into Buzzards Bay, where the **local residence time** is the average time required for water to migrate from Potters Hole to only Wild Harbor River (not all the way to the Bay). Local residence times for each sub-embayment are computed as:

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

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

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

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. It is impossible to evaluate an estuary's health based solely on flushing rates. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality is obtained from the calibrated hydrodynamic model in the following section of this report (Section VI) by extending the model to include pollutant/nutrient dispersion. The water quality model provides an additional valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Harbor system.

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were computed for the entire estuary, as well the four subdivisions of the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system.

Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet. Model divisions used to define the system sub-embayments include 1) the entire Wild Harbor system, 2) inner harbor basin, 3) Wild Harbor River, 4) Potters Hole and 5) Noname Pond. These system divisions follow the model material type areas designated in Figure V-11. Sub-embayment mean volumes and tide prisms are presented in Table V-10.

Table V-10. Embayment mean volumes and average tidal prism during simulation period for the Wild Harbor system.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Wild Harbor System	61,479,000	30,813,000
Wild Harbor inner harbor	1,521,000	1,010,000
Wild Harbor River	31,534,000	17,306,000
Potters Hole	360,000	499,000
Noname Pond	49,000	47,000

Residence times were averaged for the tidal cycles comprising a representative 14 day period (27 tide cycles), and are listed in Table V-11. The modeled time period used to compute the flushing rates started May 23, 2005 , similar to the model calibration period, and included the transition from neap to spring tide conditions. The RMA-2 model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume. Since the 14 day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

Table V-11. Computed System and Local residence times for embayments in the Wild Harbor system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Wild Harbor System	1.0	1.0
Wild Harbor inner harbor	31.5	0.8
Wild Harbor River	1.8	0.9
Potters Hole	63.8	0.4
Noname Pond	676.9	0.5

The computed flushing rates for the River system show that as a whole, the system flushes well. A flushing time of 1.0 days for the entire estuary shows that on average, water is resident in the system for only one day. System sub-embayments typically have local flushing times that are equal to or less than 1 day. Noname Pond has the shortest local flushing time, because this embayment has a small mean sub-embayment volume, relative to its tide prism. This indicates that even though the pond has a greatly restricted tide range compared to the River, with a residence time of 0.5 days, it flushes extremely well.

The generally low local residence times in all areas of the Wild Harbor system show that they would likely have good water quality if the system water with which it exchanges also has good water quality . For example, the water quality of Potters Hole would likely be good as long as the water quality of the River basin was also good. Actual water quality would still also depend upon the total nutrient load to each embayment.

For the smaller sub-embayments of the Wild Harbor system, computed system residence times are typically one to four orders of magnitude longer than their corresponding local residence time. System residence times provide a qualitative measure that helps to identify the relative sensitivity of different sub-embayments to nutrient loading.

Based on our knowledge of estuarine processes, we estimate that the combined errors associated with the method applied to compute residence times are within 10% to 15% of “true” residence times, for the Wild Harbor system. Possible errors in computed residence times can be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. In this study, the most significant errors associated with the bathymetry data result from the process of interpolating the data to the finite element mesh, which was the basis for all the flushing volumes used in the analysis. In addition, limited topographic measurements were available in some of the smaller sub-embayments of the system.

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

VI. WATER QUALITY MODELING

VI.1 DATA SOURCES FOR THE MODEL

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

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the embayments were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated hydrodynamic model representing the transport of water within the Wild Harbor system. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic model output used for the water quality model calibration was the 14 day (27 tide cycle) period beginning May 23, 2005 2000 EDT. This period overlaps with that used in the flushing analysis presented in Chapter V. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model 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 Embayments

Three primary nitrogen loads to sub-embayments are recognized in this modeling study: external loads from the watersheds, nitrogen load from direct rainfall on the embayment surface, and internal loads from the sediments. Additionally, there is a fourth load to the Wild Harbor system’s sub-embayments, consisting of the background concentrations of total nitrogen in the waters entering from Buzzards Bay. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in the area map presented in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data are the minimum required to provide a baseline for MEP analysis. 10 years of data (collected between 1999 and 2009) were available for stations in the harbor.

Table VI-1. Measured data and modeled nitrogen concentrations for the Wild Harbor estuarine system used in the model calibration plots of Figures VI-2 and VI-3. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of all measurements. Data represented in this table were collected in the summers of 1999 through 2009.

Sub-Embayment	Monitoring station	Data Mean	s.d. all data	N	model min	model max	model average
Wild Harbor inner harbor	WH-1	0.439	0.071	38	0.359	0.546	0.447
Wild Harbor River	WH-2	0.480	0.107	40	0.332	0.779	0.482
Buzzards Bay	CBB1	0.282	0.044	13	-	-	-

Wild Harbor inner harbor (WH-1) Annual TN means										
1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
0.462	0.378	0.452	0.396	0.520	0.392	0.394	0.519	0.434	0.490	0.427

Wild Harbor River (WH-2) Annual TN means										
1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
0.476	0.465	0.435	0.484	0.489	0.433	0.429	0.560	0.596	0.525	0.406

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 Wild Harbor estuarine system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of Wild Harbor. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. The MEP Technical Team has utilized this model in water quality studies of other embayment systems in southeastern Massachusetts, including Pleasant Bay (Howes *et al*., 2006); New Bedford Harbor (Howes *et al*., 2008) and Edgartown Great Pond, MA (Howes *et al*., 2008).

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

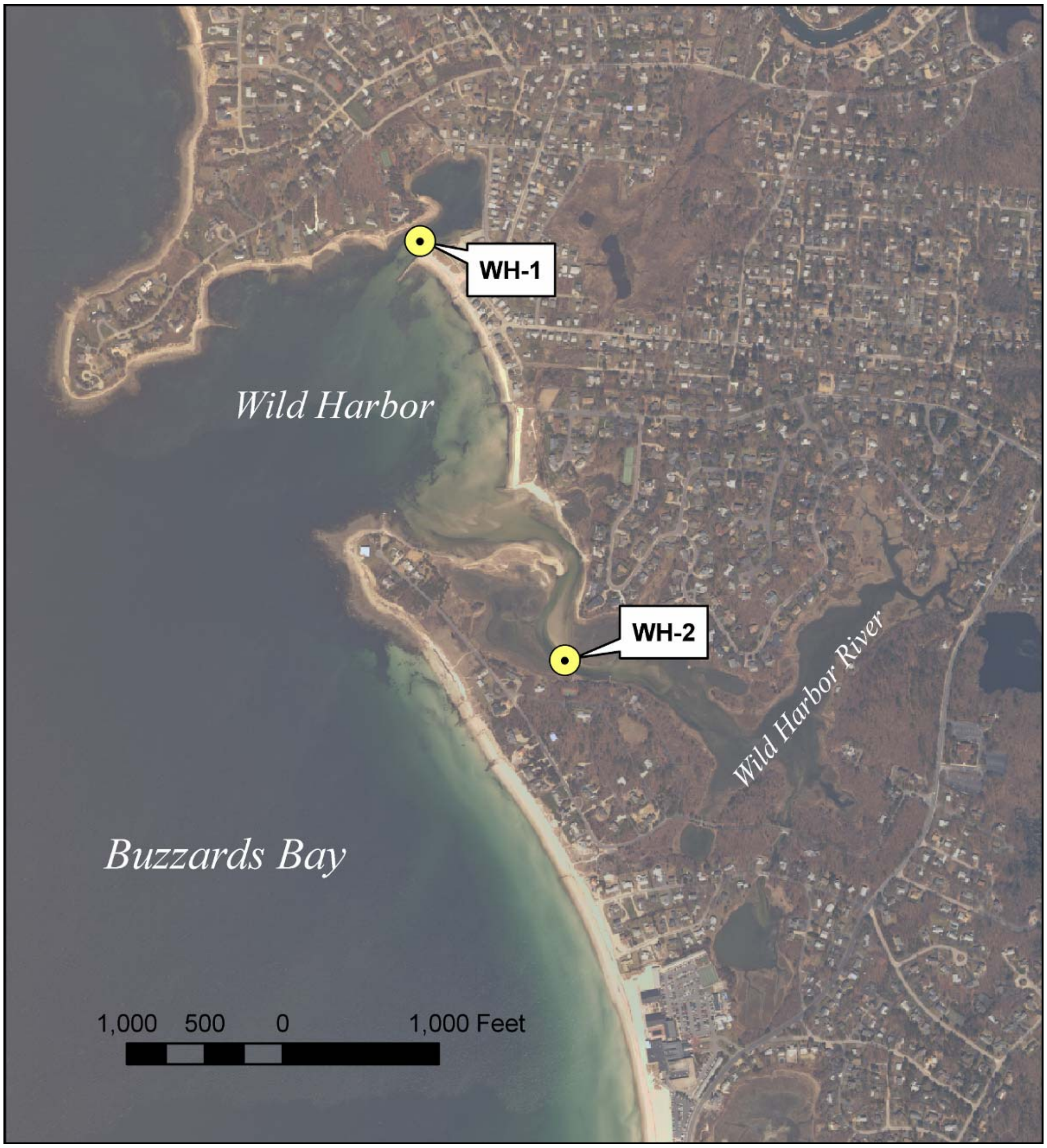

Figure VI-1. Estuarine water quality monitoring station locations in the Wild Harbor estuary system. Station labels correspond to those provided in Table VI-1.

VI.2.1 Model Formulation

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

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

where *c* in the water quality constituent concentration; *t* is time; *u* and *v* are the velocities in the *x* and *y* directions, respectively; D_x and D_y are the model dispersion coefficients in the *x* and *y* directions; and σ is the constituent source/sink term. Since the model utilizes input from the

RMA-2 model, a similar implicit solution technique is employed for the RMA-4 model.

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

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

VI.2.2 Water Quality Model Setup

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

For each 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 14 day (336 hour) period corresponding to the hydrodynamic model calibration. 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 Wild Harbor model.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, and 3) summer benthic regeneration. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed and direct atmospheric deposition loads for Wild Harbor River were evenly distributed at grid cells along the perimeter of the sub-embayment. Benthic regeneration loads were distributed among all the other, non-watershed loading elements of each material type described in Chapter V .

The loadings used to model present conditions in the Wild Harbor system are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. In all areas of Wild Harbor, including Wild Harbor River, the net benthic flux is negative which indicates a net uptake of nitrogen in the bottom sediments.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Buzzards Bay, offshore the harbor inlet, was set at 0.282 mg/L, based on SMAST data.

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Wild Harbor	10.326	1.033	-11.361
Wild Harbor River	11.825	0.447	-0.423
Dam Pond Stream	1.507	-	-
System Total	23.658	1.479	-11.784

VI.2.4 Model Calibration

Calibration of the total nitrogen model of Wild Harbor proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in Section V. Observed values of *E* in coast estuary areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled system are presented in Table VI-3. These values were used to develop the “best-fit” total nitrogen model calibration. For the case of TN modeling, “best fit” can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

Comparisons between calibrated model output and measured nitrogen concentrations are shown in plots presented in Figures VI-2 and VI-3. In these plots, means of the water column data and a range of two standard deviations of the annual means at each individual station are plotted against the modeled maximum, mean, and minimum concentrations output from the model at locations which corresponds to the MEP monitoring stations.

For model calibration, the average modeled TN was compared to mean measured TN data values, at both water-quality monitoring stations. The calibration target would fall near the modeled mean because the monitoring data are collected, as a rule, during mid ebb tide.

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

Embayment Division	E m^2/sec
Buzzards Bay	1.0
Inner Harbor	0.6
Wild Harbor River marsh	5.0
Wild Harbor River inlet	1.0
Wild Harbor River	1.0
Wild Harbor	0.5
Noname Pond	5.0
Potters Hole	5.0
Potters Hole Bridges	5.0
Potters Hole Marsh	5.0

Also presented in Figure VI-3 are unity plot comparisons of measured data verses modeled target values for each system. The computed R^2 correlation is 0.92 and the root mean squared (rms) error is less than 0.01 mg/L, both of which demonstrate an excellent fit between modeled and measured data for this system.

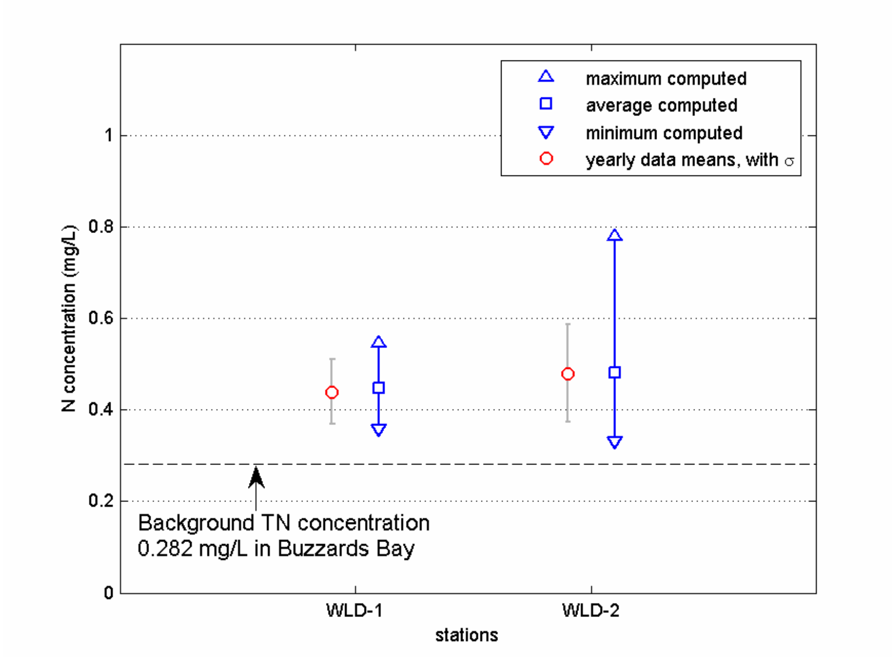

Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Wild Harbor system. Station labels correspond with the MEP IDs provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset

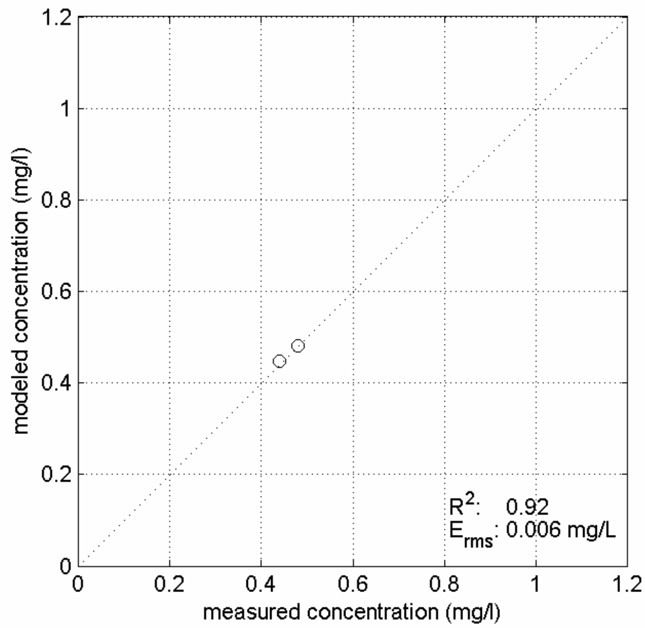

Figure VI-3. Model total nitrogen calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are 0.92and 0.006 mg/L respectively.

A contour plot of calibrated model output is shown in Figures VI-4. In this figure, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full 14-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 Wild Harbor system using salinity data collected at the same stations as the nitrogen data. For the salinity verification, none of the model dispersion coefficients were changed from the values used in the TN calibration. Comparisons of modeled and measured salinities are presented in Figures VI-5 and VI-6, with contour plots of model output shown in Figure VI-7. The R^2 correlation of the model and measurements is 0.96 and the rms error of the model is 0.4 ppt.

The only required inputs into the RMA4 salinity model of the system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, rain, surface water and groundwater inputs. The open boundary salinity was set at 31.6 ppt. Surface water and groundwater input salinities were set at 0 ppt. The only surface water input to the model is from Dam Pond Stream, which has a mean measured flow of 0.81 ft^3/sec (1,979 m^3/day). Groundwater inputs used for the model were 1.73 ft^3/sec (4,247 m^3/day) for the Wild Harbor basin watershed and 3.22 ft^3/sec (7,886 m^3/day) for the Wild Harbor River watershed. Average rainfall rates included in the simulation were 0.26 ft^3/sec (646 m^3/day) 0.11ft^3/sec (279 m^3/day) for Wild Harbor River. Groundwater and rainfall flows were distributed evenly in the model along elements positioned along the model's land boundary.

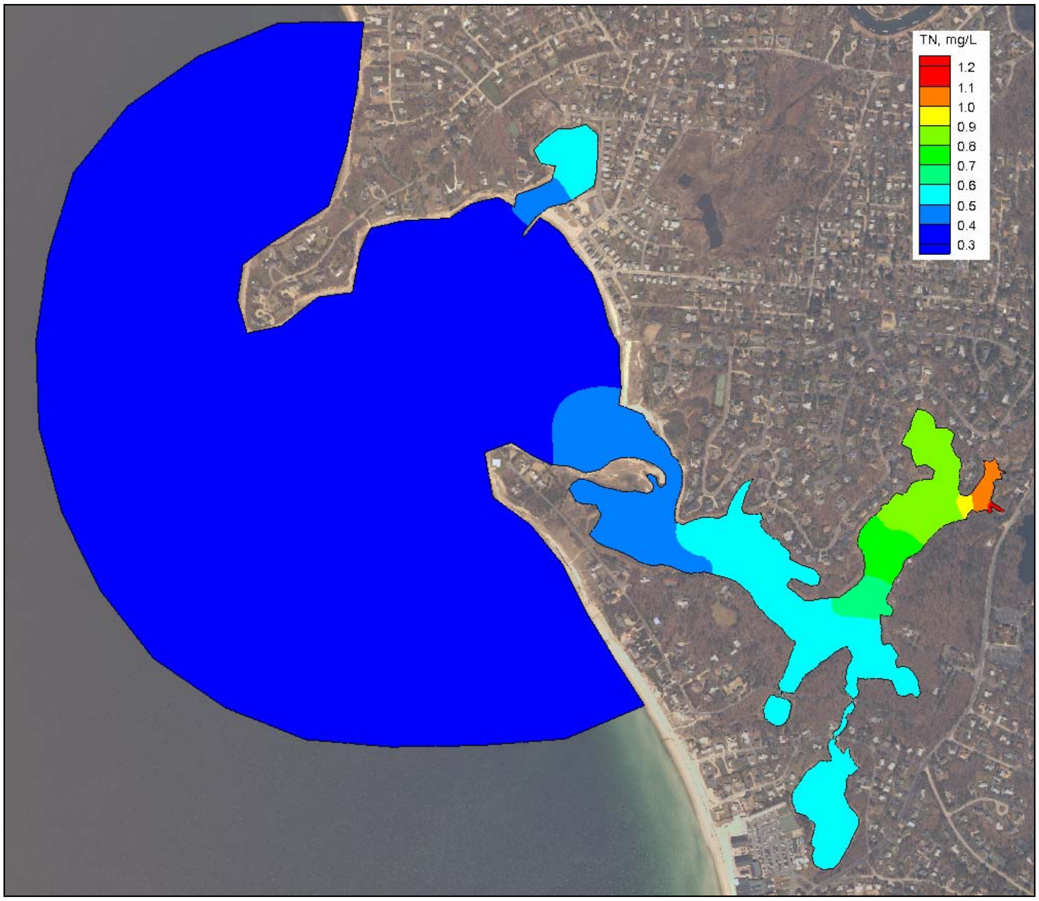

Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Wild Harbor system.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

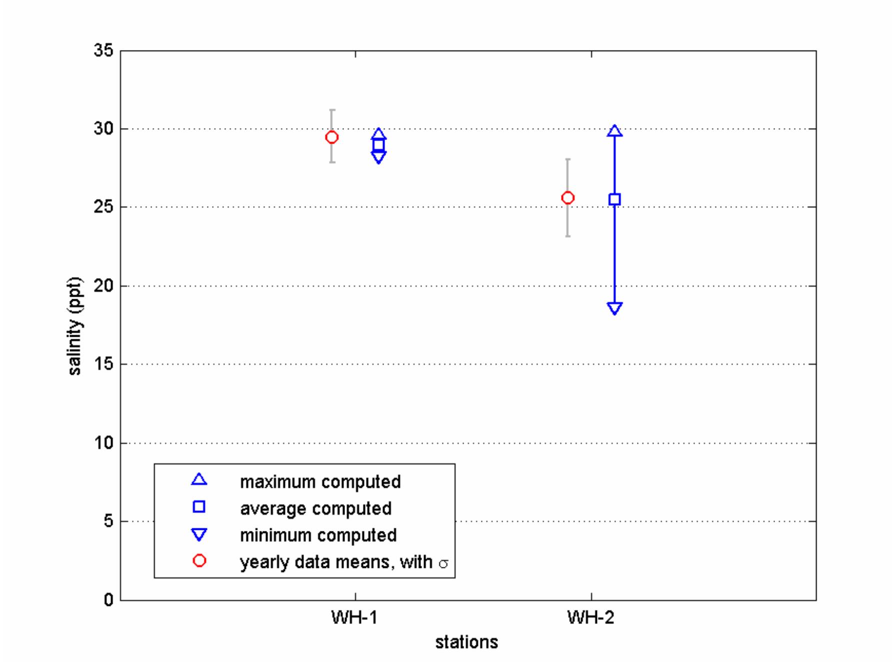

Figure VI-5. Comparison of measured and calibrated model output at stations in Wild Harbor. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with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

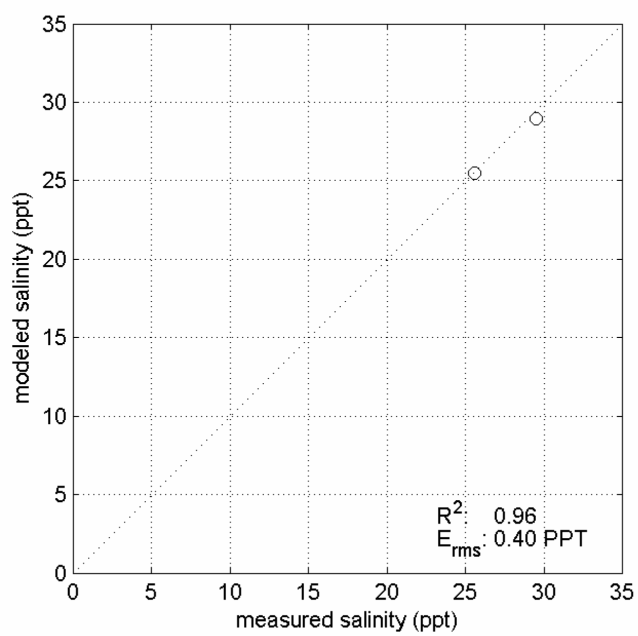

Figure VI-6. Model salinity target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) is 0.96 and RMS error for this model verification run is 0.40 ppt.

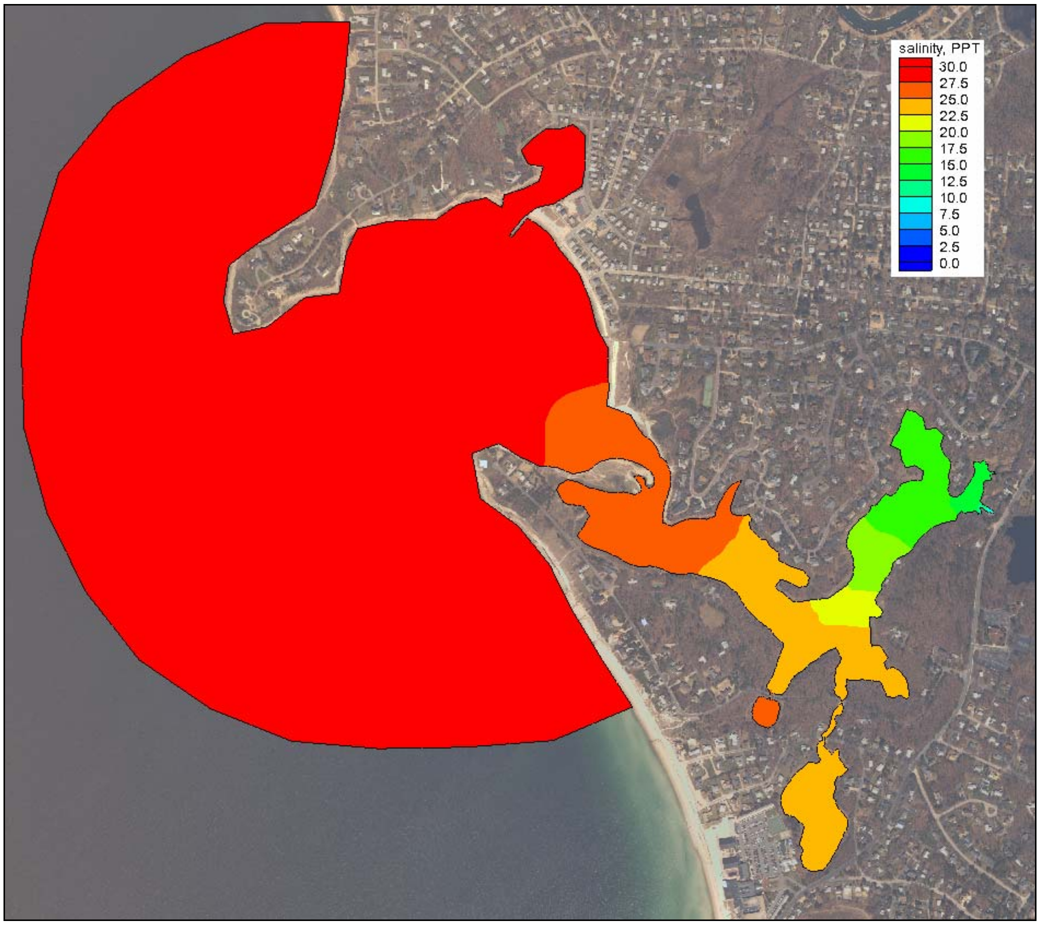

Figure VI-7. Contour Plot of average modeled salinity (ppt) in the Wild Harbor system.

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

sub-embayment	present load (kg/day)	Build-out (kg/day)	build-out % change	no load (kg/day)	no load % change
Wild Harbor	10.326	10.595	+2.6%	0.323	-96.9%
Wild Harbor River	11.825	12.156	+2.8%	0.556	-95.3%
Dam Pond Stream	1.507	2.230	+48.0%	0.178	-88.2%
System Total	23.658	24.981	+5.6%	1.058	-95.5%

VI.2.6.1 Build-Out

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. **Build-out** scenario sub-embayment and surface water loads used for total nitrogen modeling of the Wild Harbor system, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Wild Harbor	10.595	1.033	-11.556
Wild Harbor River	12.156	0.447	-0.434
Dam Pond Stream	2.230	-	-
System Total	24.981	1.479	-11.990

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of the system was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Buzzards Bay) remained identical to the existing conditions modeling scenarios. For build-out, the increase in modeled TN concentrations is greatest at the monitoring station in Wild Harbor River, where concentrations increase more than 10%. A contour plot showing average TN concentrations throughout the river system is presented in Figure VI-8 for the model of build-out loading.

Table VI-6. Comparison of model average total N concentrations from present loading and the **build-out scenario**, with percent change over background in Buzzards Bay (0.282 mg/L), for the Wild Harbor system.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	build-out (mg/L)	% change
Wild Harbor inner harbor	WH-1	0.447	0.455	+5.1%
Wild Harbor River	WH-2	0.482	0.503	+10.3%

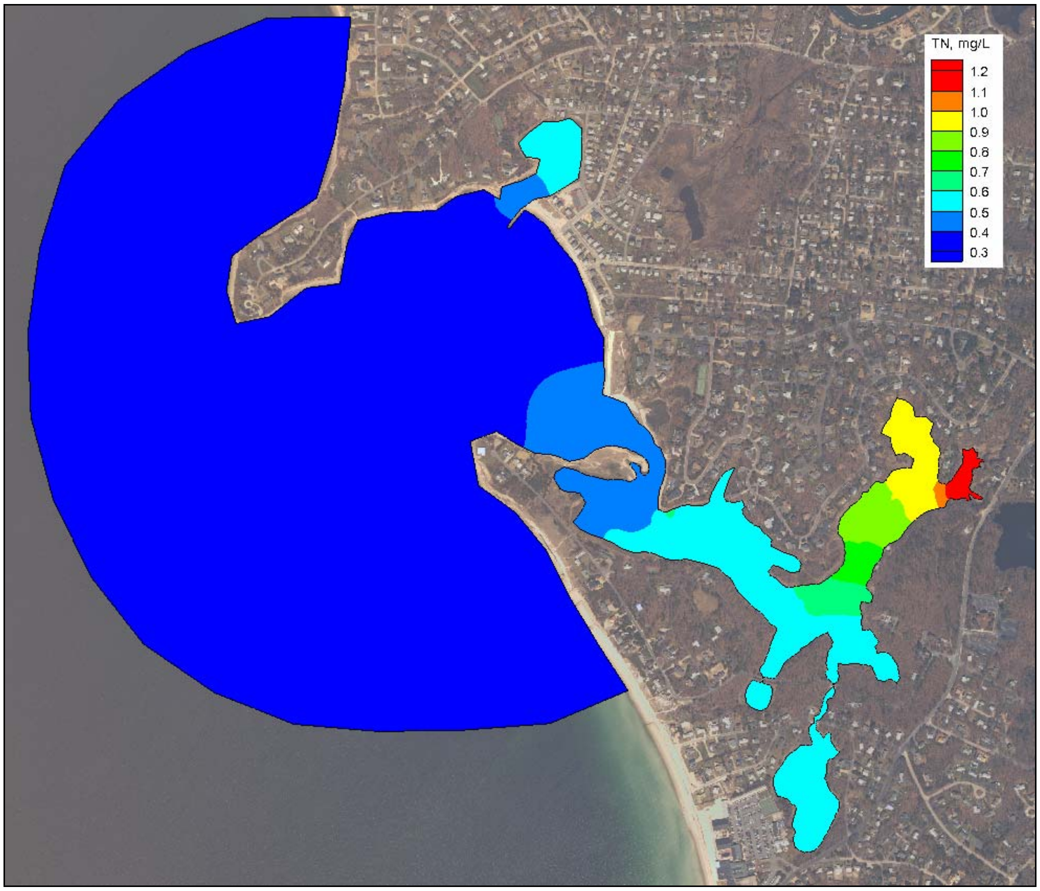

Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Wild Harbor system, for projected build-out scenario loading conditions.

VI.2.6.2 No Anthropogenic Load

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

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Wild Harbor	0.323	1.033	-7.835
Wild Harbor River	0.556	0.447	-0.229
Dam Pond Stream	0.178	-	-
System Total	1.058	1.479	-8.064

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations at each monitoring station. Again, total nitrogen concentrations in the receiving waters (i.e., Buzzards Bay) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was large, with all areas of the system experiencing reductions greater than 100%, compared to the background concentration of 0.282 in Buzzards Bay. TN concentrations drop below background in Buzzards Bay due to the projected large negative benthic flux of harbor sediments. A contour plot showing TN concentrations throughout the system is shown pictorially in Figure VI-9.

Table VI-8. Comparison of model average total N concentrations from present loading and the "**No anthropogenic loading**" ("no load"), with percent change over background in Buzzards Bay (0.282 mg/L), for the Wild Harbor system.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	no-load (mg/L)	% change
Wild Harbor inner harbor	WH-1	0.447	0.264	-111.1%
Wild Harbor River	WH-2	0.482	0.268	-107.2%

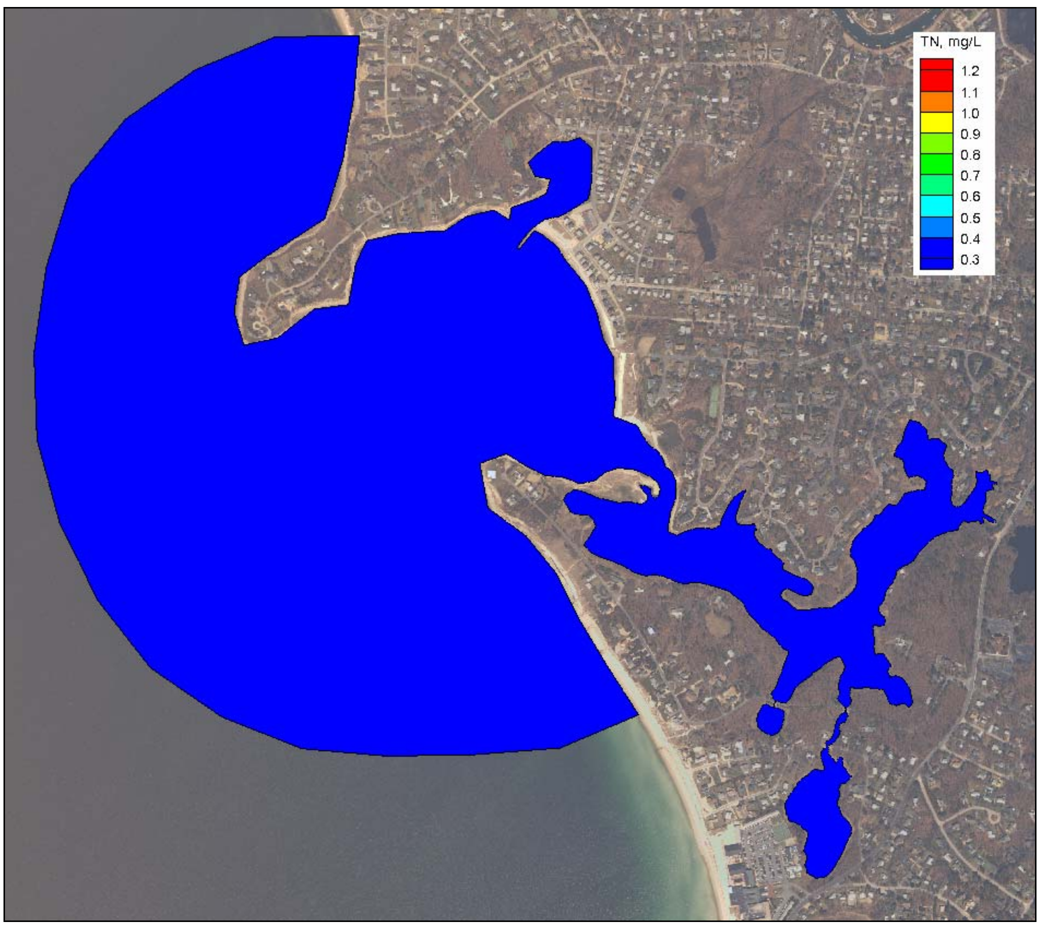

Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in Wild Harbor, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Wild Harbor embayment system the MEP assessment is based upon data from the water quality monitoring database (1999-2009) developed by the Coalition for Buzzards Bay, surveys of eelgrass distribution (1951, 1995, 2001, 2006), benthic animal communities (fall 2006), sediment characteristics (summer 2006), and dissolved oxygen records (summer 2006). These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for the overall system (Section VIII). It should be noted that nitrogen enrichment occurs through 2 primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or low rates of flushing due to restriction of tidal exchange with the low nitrogen waters of Buzzards Bay. Wild Harbor has enhanced nitrogen loading from its associated watersheds from shifting land-uses. Fundamentally, tidal exchange or circulation alters the sensitivity of an estuary or sub-estuary to nitrogen inputs.

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen threshold determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed autonomous dissolved oxygen sensors in Wild Harbor at locations that would be representative of the dissolved oxygen conditions at critical locations in each of the sub-systems, namely a mid-location in Wild Harbor River and a location within the Wild Harbor Yacht Club boat basin. There was also a dissolved oxygen mooring deployed in the main basin of Wild Harbor, (Figure VII-1). The dissolved oxygen moorings were deployed to record the frequency and duration of low oxygen conditions during the critical summer period. The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within the Wild Harbor system was conducted by the MassDEP Eelgrass Mapping Program (C. Costello) for comparison to historic records.

Figure VII-1. Aerial Photograph of the Wild Harbor System in the Town of Falmouth (with upper watershed extending into the Towns of Bourne and Sandwich) showing the location of the continuously recording Dissolved Oxygen / Chlorophyll-a sensors deployed during the Summer of 2006.

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 throughout the Wild Harbor System supports this contention that nitrogen is the nutrient to be managed as the N/P ratios in the main basin (1), inner Boat Basin (3) and Wild Harbor River (<6 at highest) are clearly below the Redfield Ratio value (16) indicating that nitrogen additions will increase phytoplankton production in this system. Within the Wild Harbor main basin, temporal changes in eelgrass distribution provide one basis for evaluating nitrogen enrichment, as there was identifiable eelgrass in 1951, 1995, 2001 and 2006. As a result, nutrient threshold determination for this basin 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 (such as the Wild Harbor River and the boat basin), benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H') and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 4 mg L-1. Massachusetts State Water Quality Classification indicates that SA (high quality) waters be able to maintain oxygen levels above 6 mg L-1. The tidal waters of the Wild Harbor 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 TMDLs 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-2). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the bottom of the embayment within key regions of the Wild Harbor system (Figure VII-1). The dissolved oxygen sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployments. In addition periodic calibration samples were collected at the depth of each sensor and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 25-30 days within the interval from July through mid-September. All of the mooring data from the Wild Harbor system were collected during the summer of 2006. These data are supplemented by the traditional "grab" sampling data from the water quality monitoring program, which generally collects <15 time points per summer, but is conducted over several years.

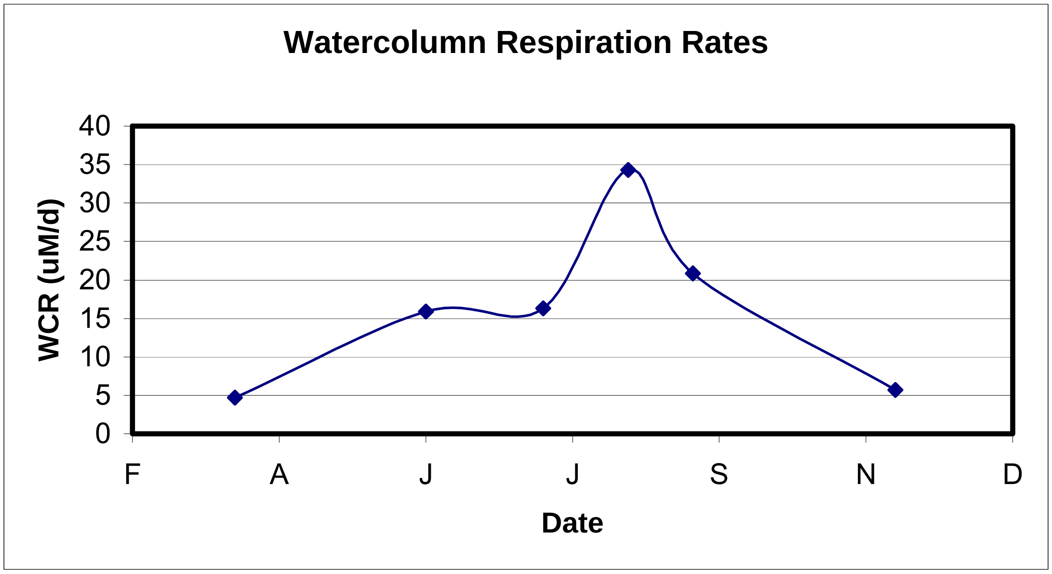

Figure VII-2. Example of typical average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System, Cape Cod (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

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

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 28 day and 40 day (Wild Harbor-outer and Wild Harbor-inner, respectively) deployment periods that these parameters were below/above various benchmark concentrations (Tables VII-1 and VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate slightly nutrient enriched waters within the outer portion of the Wild Harbor system (Wild Harbor-outer mooring location) and moderately nutrient enriched waters within the small inner basin (Wild Harbor-inner). These open-water embayment basins differ from the Wild Harbor River which functions as a tidal salt marsh. Salt marshes typically have a higher tolerance for nitrogen loading then open water basins, since they are naturally nutrient and organic matter enriched.

The dissolved oxygen data is described below and depicted in Figures VII-3 and VII-5). The oxygen data is consistent with organic matter enrichment of embayment systems (particularly in the Boat Basin), primarily from phytoplankton production 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 the Wild Harbor estuarine system.

The oxygen records obtained from both the moorings deployed in Wild Harbor show that the small Boat Basin (Wild Harbor-inner) has large enough daily oxygen excursions to indicate moderate nitrogen enrichment. More importantly, dissolved oxygen levels do occasionally drop below the 4 mg L^{-1} oxygen threshold that indicates dissolved oxygen stress. 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. A fundamental effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~8-12 mg L^{-1} at the Wild Harbor-inner mooring site). The evidence of oxygen levels above atmospheric equilibration indicates that this inner basin is nitrogen enriched as air equilibration is typically between 7 and 8 mg L^{-1}. Although there was not an oxygen mooring in Wild Harbor River, the water quality monitoring program collected 129 oxygen readings at the mouth of the tidal river near the end of ebb tide. These data showed no reading below 3 mg L^{-1}, 2% <4 mg L^{-1}, and 87% >5 mg L^{-1}. These moderate levels of oxygen depletion coupled with the salt marsh dominance of the Wild Harbor River, supports the contention that this tidal river is not impaired by nitrogen loading. It should be emphasized that salt marshes are naturally nutrient and organic matter rich, and frequently show significant levels of oxygen depletion on a diurnal basis. The embayment specific results of the dissolved oxygen/chlorophyll mooring program are as follows:

Wild Harbor-outer DO/CHLA Mooring (Figures VII-3 and VII-4):

Two moorings were deployed in the Wild Harbor system. One of the two instrument moorings was located in the main basin of Wild Harbor. This mooring was positioned centrally in the main outer basin and slightly closer to shore rather than the outer limit of the harbor before it becomes more representative of Buzzards Bay. This outer mooring is relatively well exposed to low nutrient waters entering Wild Harbor from Buzzards Bay. As stated above, instantaneous oxygen levels that drop below 4 mg L^{-1} are indicative of oxygen stress. Oxygen levels in Wild Harbor were not observed to decline to even close to this level, generally showing only small to moderate depletion, with levels never dropping below 5 mg L^{-1} and almost always >5.5 mg L^{-1} (Figure VII-3, Table VII-1). Overall the oxygen levels observed within the outer basin are not generally associated with habitat stress, particularly given that declines below 6 mg L^{-1} were generally of short duration. Outer basin oxygen conditions did exhibit daily excursions in oxygen levels, but the range of daily oxygen excursion was moderate and significantly less than that observed in the Wild Harbor Boat Basin, furthest from the low nutrient and high oxygen waters of Buzzards Bay water. The oxygen dynamics at both sites is a clear indication of nitrogen enrichment, but only the inner basin is showing declines stressful to benthic animal communities.

Oxygen levels occasionally exceeded 8 mg L^{-1} and mostly existed between 6 and 8 mg L^{-1}. The small daily oxygen excursion (~2.5 mg L^{-1}), with generally high oxygen levels is consistent with the low nutrient concentrations and relatively low phytoplankton biomass in this basin. Over the 28 day deployment chlorophyll levels were generally moderate to low without extensive blooms. Chlorophyll a regularly remained between 2-6 ug L^{-1} with a brief increase to 8-10 ug L^{-1}. The level of oxygen observed in this system is indicative of a generally healthy system, consistent with the low chlorophyll a levels and slight nitrogen enrichment (mooring chlorophyll-a average 3.8 ug L^{-1}). In this somewhat open portion of the Wild Harbor system, chlorophyll-a was less than the 5 ug L^{-1} benchmark 84 percent of the time (Table VII-2, Figure VII-4). Average chlorophyll levels over 10 ug L^{-1} have been used to indicate eutrophic conditions in embayments.

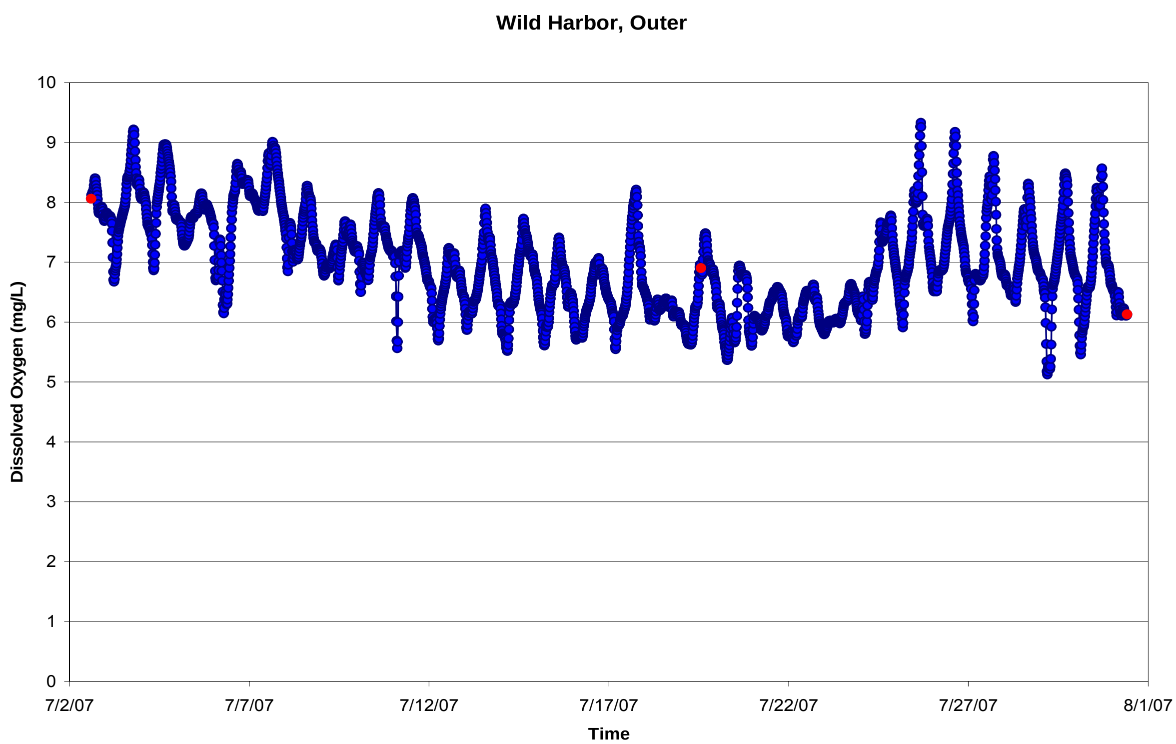

Figure VII-3. Bottom water record of dissolved oxygen in the Wild Harbor basin (Wild Harbor-outer station), Summer 2007. Calibration samples represented as red dots.

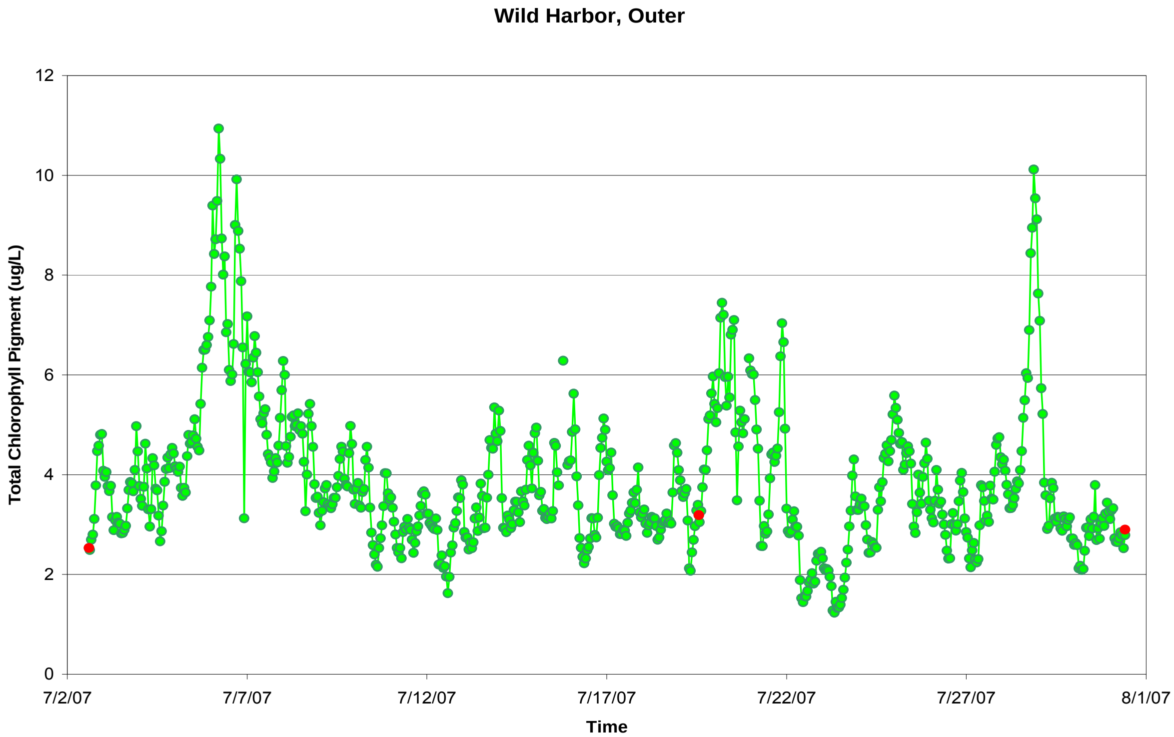

Figure VII-4. Bottom water record of Chlorophyll-*a* in the Wild Harbor basin (Wild Harbor-outer station), Summer 2007.Calibration samples represented as red dots.

Wild Harbor-inner DO/CHLA Mooring (Figures VII-5 and VII-6):

The second of the two instrument moorings was (Wild Harbor-inner) deployed in the Wild Harbor Boat Basin. This small basin is connected to the main basin of Wild Harbor (Wild Harbor-outer) by a small (~50 to 60 meter wide) armored inlet channel. The mooring was centrally located within the Boat Basin. Moderate daily excursions in oxygen levels were observed at this location (~4 mg L^{-1}), ranging from levels at and above air equilibration (regularly exceeding 10 mg L^{-1}). Equally important oxygen depletion to slightly hypoxic conditions where levels decline to 4 mg L^{-1} and also drop periodically to slightly below 4 mg L^{-1} were also documented (Figure VII-5, Table VII-1). Instantaneous oxygen levels that drop below 4 mg L^{-1} are indicative of oxygen stress to benthic animal communities. The effects of nitrogen loading and organic enrichment of the system are consistent with the phytoplankton blooms that were observed during the meter deployment period as well as the high rates of photosynthesis (carbon fixation) and the declines in oxygen after sunset stemming from respiration. It should be noted that the oxygen excursions are not as wide ranging in this portion of Wild Harbor as they are in estuarine systems that have significant habitat impairments, however, they are clearly wider ranging then the record collected from the Wild Harbor outer basin, indicating that there is habitat impairment in this small basin. However, oxygen levels dropped below 6 mg L^{-1} only 20 percent of the deployment period. Moreover, oxygen levels dropped below 4 mg L^{-1} only 1 percent of the deployment period, but did decline to 3 mg L^{-1}. Overall, the observed oxygen levels, although representing a stress, are consistent with only a moderate level of impairment, in part due to tidal exchange with the high quality waters of Buzzards Bay.

Over the 40 day deployment there does appear to be a clear 15 day period at the beginning of the deployment of moderately intense phytoplankton bloom where chlorophyll a increased to 20-25 ug L^{-1} and a few points during the bloom where chlorophyll-a concentrations peaked at just over 30 ug L^{-1}. The periodic low levels of oxygen observed in this system correspond to the period of high chlorophyll and is indicative of moderate habitat impairment (mooring chlorophyll-a average 9.4 ug L^{-1}). In the innermost portion of the main basin of the Wild Harbor system, chlorophyll-a exceeded the 10 ug L^{-1} benchmark 37 percent of the time and exceeded the 20 ug L^{-1} benchmark 4 percent of the time (Table VII-2, Figure VII-6). Average chlorophyll levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

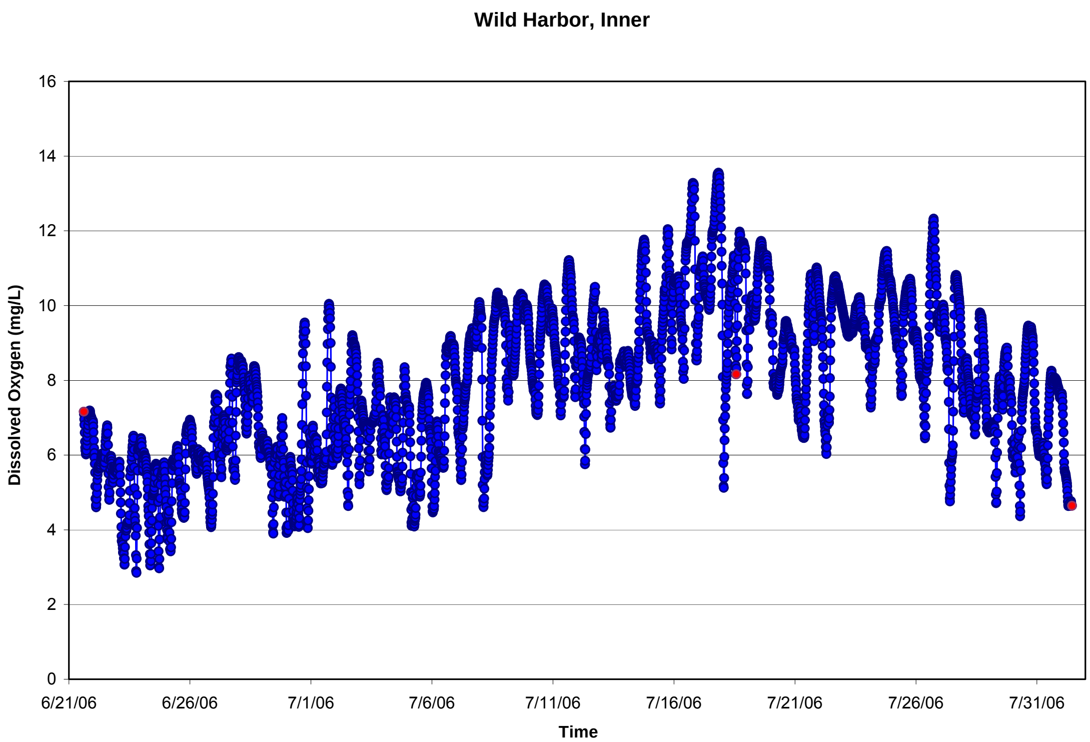

Figure VII-5. Bottom water record of dissolved oxygen at the Wild Harbor-inner station, Summer 2006. Calibration samples represented as red dots.

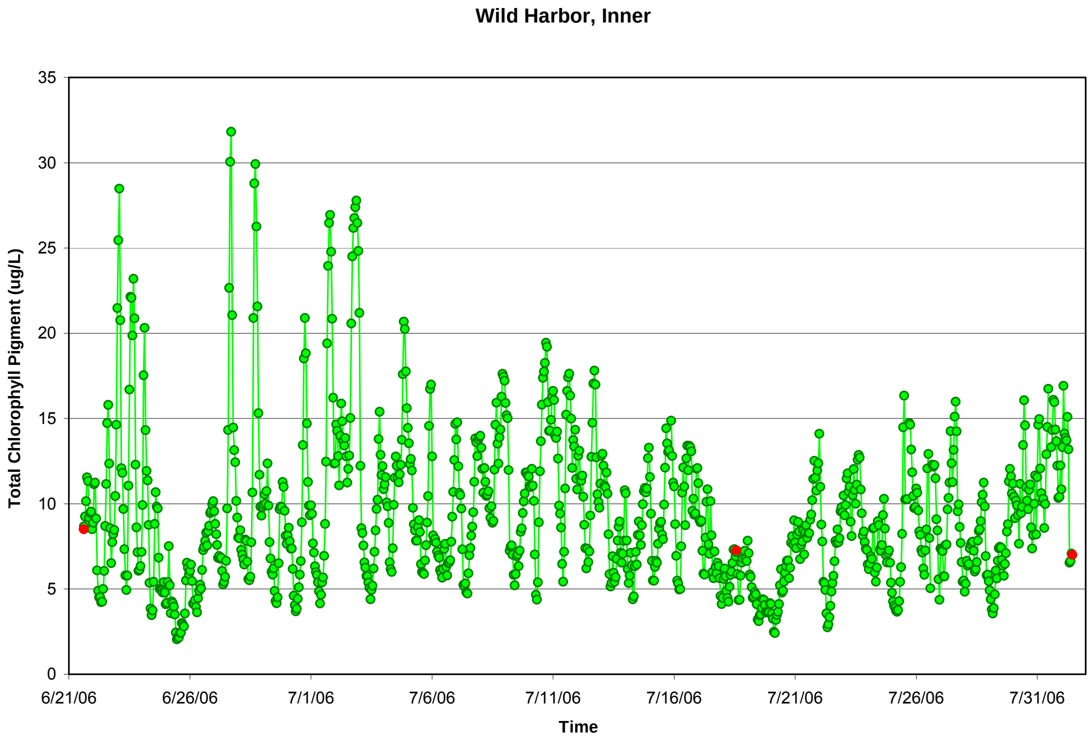

Figure VII-6. Bottom water record of Chlorophyll-*a* in the Wild Harbor-inner station, Summer 2006. Calibration samples represented as red dots.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Analysis of spatial and temporal trends in eelgrass distribution and review of historical data was conducted for the Wild Harbor Embayment System by the MassDEP Eelgrass Mapping Program as part of the MEP technical effort. Field surveys of the harbor were conducted in 1995, 2001 and 2006 by MassDEP, as part of this program. Additional observations during summer and fall 2006 were completed by the SMAST/MEP Technical Team during other MEP data collection efforts. Analysis of available aerial photography from 1951 was conducted to reconstruct the eelgrass distribution prior to the present level of development of the watershed. The primary use of the eelgrass data within the MEP approach is to indicate (a) if eelgrass once or currently colonizes a basin and (b) identify any large-scale system-wide shifts in distribution. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 to 2006 (Figure VII-7). These data were also compared with an eelgrass survey in the mid 1980's (Costa 1988), thus increasing the validity of the overall assessment of eelgrass trends in Wild Harbor. This temporal information can be used to determine the stability of the eelgrass community in many estuarine systems.

All of the available information on eelgrass relative to Wild Harbor indicates that the main basin (Wild Harbor) has and continues to support significant eelgrass beds. However, there is also clear evidence of eelgrass loss, primarily from the bed adjacent Nyes Neck. Eelgrass bed loss from this region of the harbor has been nearest the inlet to the Boat Basin (which receives nutrient enriched ebb tide waters) and from the deeper region of the bed. The loss has been continuous over the past 6 decades and the spatial and temporal pattern is consistent with nitrogen enrichment.

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 Wild Harbor 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)**
Wild Harbor, Inner	**6/21/2006**	**8/1/2006**	**40.8**	**20%**	**7%**	**1%**	**0.1%**
			Mean	0.19	0.12	0.06	0.02
			Min	0.02	0.02	0.02	0.01
			Max	1.33	0.38	0.18	0.02
			S.D.	0.26	0.10	0.05	0.01
Wild Harbor, Outer	**7/2/2007**	**7/31/2007**	**28.2**	**11%**	**0%**	**0%**	**0%**
			Mean	0.15	NA	NA	NA
			Min	0.02	0.00	0.00	0.00
			Max	0.36	0.00	0.00	0.00
			S.D.	0.10	NA	NA	NA

Table VII-2. Duration (days and % of deployment time) that chlorophyll a levels exceed various benchmark levels within the Wild Harbor 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)**
Wild Harbor, Inner	**6/21/2006**	**8/1/2006**	**40.8**	**88%**	**37%**	**9%**	**4%**	**1%**
Mean Chl Value = 9.4 ug/L			**Mean**	1.32	0.28	0.14	0.15	0.12
			Min	0.04	0.04	0.04	0.04	0.08
			Max	3.92	1.46	0.42	0.38	0.21
			S.D.	1.19	0.27	0.11	0.10	0.05
Wild Harbor, Outer	**7/2/2007**	**7/31/2007**	**28.7**	**16%**	**0.44%**	**0%**	**0%**	**0%**
Mean Chl Value = 3.8 ug/L			**Mean**	0.23	0.06	NA	NA	NA
			Min	0.04	0.04	0.00	0.00	0.00
			Max	1.21	0.08	0.00	0.00	0.00
			S.D.	0.32	0.03	NA	NA	NA

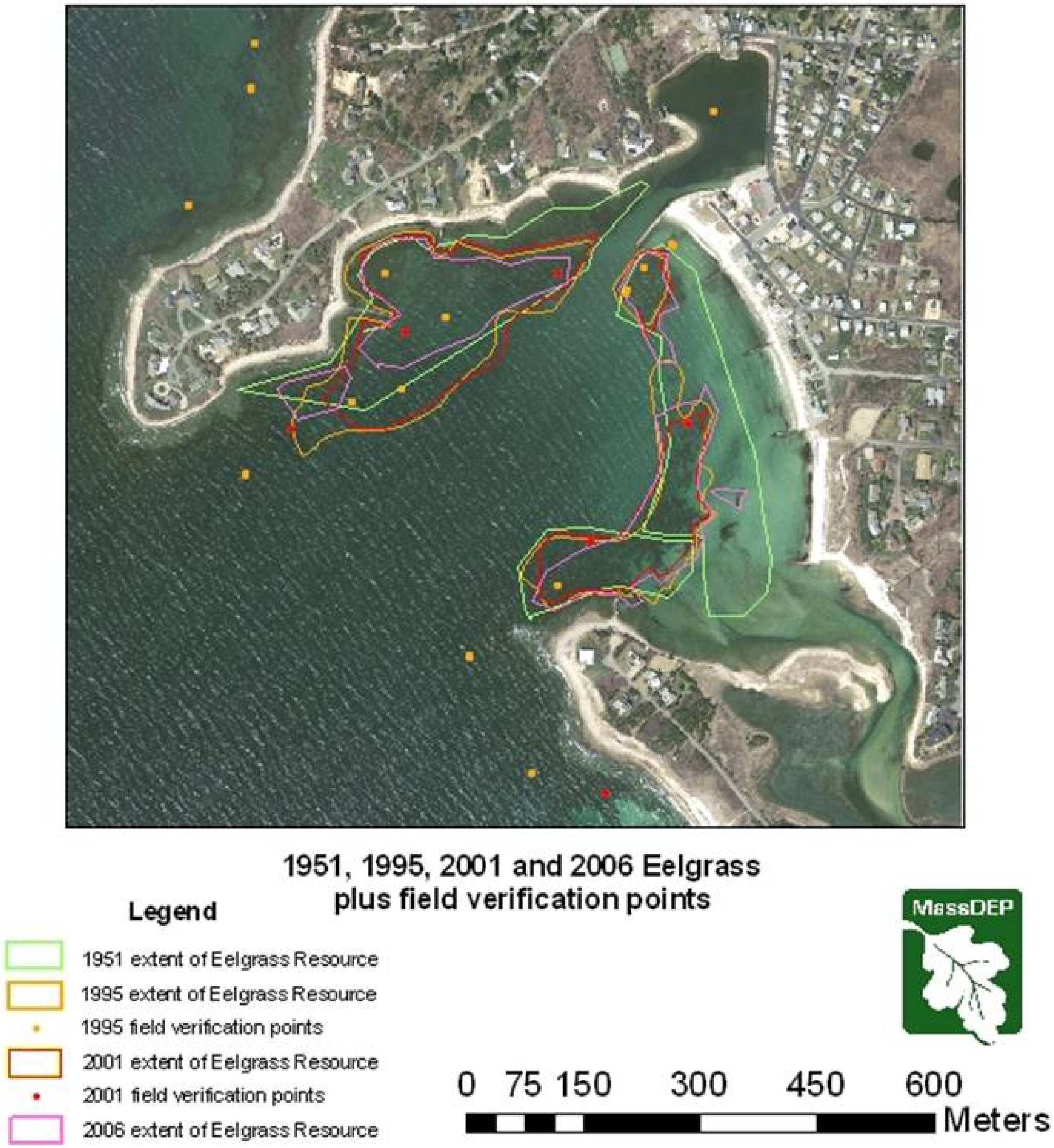

Figure VII-7. Eelgrass bed distribution throughout Wild Harbor. Beds delineated in 1951 are circumscribed by the green line, 1995 beds are circumscribed by the tan outline, 2001 beds are circumscribed by the brown line and 2006 beds are circumscribed by the pink line (map from the MassDEP Eelgrass Mapping Program). There is no evidence of eelgrass beds within the Wild Harbor Boat Basin or Wild Harbor River in any surveys over the past 60 years.

It is significant that eelgrass was not detected in the Wild Harbor River or in the Wild Harbor Boat Basin in the 1951 (or 1995, 2001, 2006) data. The Wild Harbor Boat Basin is an artificial embayment basin apparently in its present form as a result of dredging and bulkhead construction. This basin is relatively deep and is a depositional basin as evidenced by its organic enriched sediments. Eelgrass is not expected in the salt marsh dominated Wild Harbor River, which is naturally nutrient and organic matter enriched and becomes very shallow at low tide. The absence of evidence that the Wild Harbor Boat Basin and Wild Harbor River have ever supported eelgrass habitat, directs management to benthic infaunal communities (see below).

Other factors which influence eelgrass bed loss in embayments were also evaluated in the Wild Harbor system. Although the observed bed loss seems completely in-line with nitrogen enrichment, a brief listing of non-nitrogen related factors is presented here. Eelgrass bed loss does not seem to be directly related to mooring density, as the Wild Harbor System generally supports a low density of boat moorings in the areas where eelgrass loss has occurred and has moorings in areas where beds remain stable. Moorings associated with the eelgrass beds along Nyes Neck are concentrated in small area of the southeastern portion of the main basin as well as a small area of the northern portion of the main basin. Pier construction and boating pressure may be adding additional stress, but seem to be relatively minor factors in the overall system, with the exception of the boat basin (though that area does not show a history of eelgrass as far back as 1951). While in other systems, pressure on eelgrass from shellfishing activity can be significant, that would not be the case in Wild Harbor as shellfishing is prohibited throughout this system on a year round basis.

Based on the available data, it is possible to utilize the 1951 coverage data as an indication that eelgrass beds might be recovered if nitrogen management alternatives were implemented (Table VII-3). This determination is based upon the MASSDEP Mapping Program and would indicate that an area of eelgrass habitat within Wild Harbor of approximately 10 acres could be recovered. Note that restoration of this habitat will necessarily result in lower nitrogen levels in the boat basin, as well as the Wild Harbor River portion of the overall system (see Chapter VIII).

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

Table VII-3. Temporal changes in eelgrass coverage in the Wild Harbor Embayment System within the Town of Falmouth 1951 to 2006 (MassDEP, C. Costello).

EMBAYMENT	1951 (acres)	1995 (acres)	2001 (acres)	2006 (acres)	% Difference (1951 to 2006)
Wild Harbor	27.81	20.53	17.40	14.44	48%

VII.4 Benthic Infauna Analysis

Quantitative sediment sampling was conducted at 7 locations within the Wild Harbor Embayment System (Figure VII-8), with replicate assays at the majority of sampling sites. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. As mentioned previously, it does not appear that the Wild Harbor River or the Wild Harbor Boat Basin portions of this System have ever supported eelgrass. The Wild Harbor River is functioning as a salt marsh creek, which typically do not support eelgrass. The Wild Harbor Boat Basin is a dredged basin and presently has levels of oxygen depletion and chlorophyll a not generally supportive of eelgrass habitat. Therefore habitat quality in those specific areas will primarily focus on benthic infauna habitat. For the Boat Basin the oxygen, chlorophyll a levels and the depositional sulfidic sediments suggest a basin where management will focus on restoration rather than conservation. To the extent that the Boat Basin region of the Wild Harbor System can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded) and what nutrient concentrations would be supportive of healthy habitat. This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

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

Wild Harbor Infaunal Characteristics:

Overall, the infauna survey indicated that the main basin of Wild Harbor is supporting high quality benthic animal habitat with high diversity (23, 28 species; H'>3.4) and Evenness (>0.72), although the inner region is potentially moderately impaired. Similarly the Wild Harbor River is functioning as a non-nitrogen impaired salt marsh system with productive benthic communities typical of Cape Cod marsh creeks. In contrast, the enclosed Boat Basin presents a significantly impaired habitat for infaunal communities with low numbers of species (7) and individuals (123/sample), with low diversity (H'=1.7). The sediments are depositional, comprised of organic rich soft sulfidic mud. The sediment surface is overlain by an algal mat, which further reduces habitat quality. These sediment characteristics coupled with the periodic oxygen depletions to 3 mg L^{-1} and moderate chlorophyll levels are consistent with a benthic habitat significantly impaired by nitrogen enrichment (Table VII-4). The present animal community is also indicative of a significantly impaired habitat, being dominated by stress tolerant species indicative of high levels of organic enrichment. About 50% of the total community consisted of a single species, *Capitella capitata*, generally found in embayments with high organic matter deposition and poor habitat quality.

Figure VII-8. Aerial photograph of the Wild Harbor system showing location of benthic infaunal sampling stations (red symbols).

In contrast, high quality infaunal habitat is supportive of much higher numbers of individuals and species with high Diversity and Eveness. The main basin of Wild Harbor compares well with the highest quality benthic habitats on Cape Cod, such as in the outer basin of Quissett Harbor which also supported diverse communities of polychaetes, mollusks and crustaceans typically associated with high quality coastal environments. Throughout the Quissett Harbor Basin, infaunal animal communities colonizing sediments at depths less than 4.5 meters or throughout the Outer Basin are diverse ($\geq$22 species) and productive (>400 individuals per sample). The Outer Basin general ranked high based upon the key community indices, the Weiner Diversity Index (H') and Eveness, which had values greater than 3.4 and 0.77, respectively. Similarly, the outer stations within Lewis Bay in Barnstable currently support high quality benthic habitat as seen in the numbers of individuals (502 per sample), number of species (32), diversity (3.69) and Eveness (0.74).

Table VII-4. Benthic infaunal community data for the Wild Harbor embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2). Stations refer to map in Figure VII-17.

	Station	Total Species	Total Individuals	#Species Calc @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Wild Harbor						
Outer	**1**	**28**	**230**	**19**	**3.46**	**0.72**
Inner	**4**	**10**	**162**	**8**	**2.21**	**0.67**
River Mouth	**5**	**23**	**232**	**18**	**3.66**	**0.81**
Boat Basin						
Central	**7,8**	**7**	**123**	**7**	**1.71**	**0.65**
Wild Harbor River						
Creek	**10,12**	**17**	**543**	**10**	**2.62**	**0.64**
Station numbers refer to ID's on maps presented above.						

Other Resource Characteristics:

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available for use by the MEP Technical Team. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish (Figure VII-9). As is the case with some systems on Cape Cod, all of the enclosed waters of Wild Harbor are classified as prohibited for the taking of shellfish at any time of the year, indicating the system is impaired relative to the taking of shellfish. Despite the 1969 fuel oil spill which released ~180,000 gallons of No. 2 fuel oil a small portion of which remains buried in sediments of Wild Harbor River, the closure of Wild Harbor to shellfishing is due to overlying water quality which shows high fecal coliform bacteria levels which would be a result of both human activity (septic systems in the watershed) as well as natural fauna. Nevertheless, the Wild Harbor system has also been classified as supportive of specific shellfish communities (Figure VII-10). The major shellfish species with potential habitat within the Wild Harbor Estuary are mainly quahogs (*Mercenaria*) and soft shell clams (*Mya*). While habitat suitable to *Mercenaria* exists throughout the Wild Harbor River, the boat basin and the northern shore of Wild Harbor along Nyes Neck, suitable habitat for *Mya* only exists in a small portion of the Wild Harbor River in the vicinity of the inlet. Small portions of the Wild Harbor system where also identified as being suitable to the American Oyster. These areas are mainly the boat basin, a small area around Crows Point and a small area in the Wild Harbor River that is also suitable for soft shells clams. It should be noted that the observed pattern of shellfish growing area is consistent with the observed organic rich sediments within the Wild Harbor system. Moreover, improving benthic animal habitat quality should also expand the shellfish growing area within this system. This will not necessarily result in the opening of shellfishing beds as there will still exist the underlying concern over any remaining effects from petroleum hydrocarbons in the estuarine sediments, particularly within the Wild Harbor River.

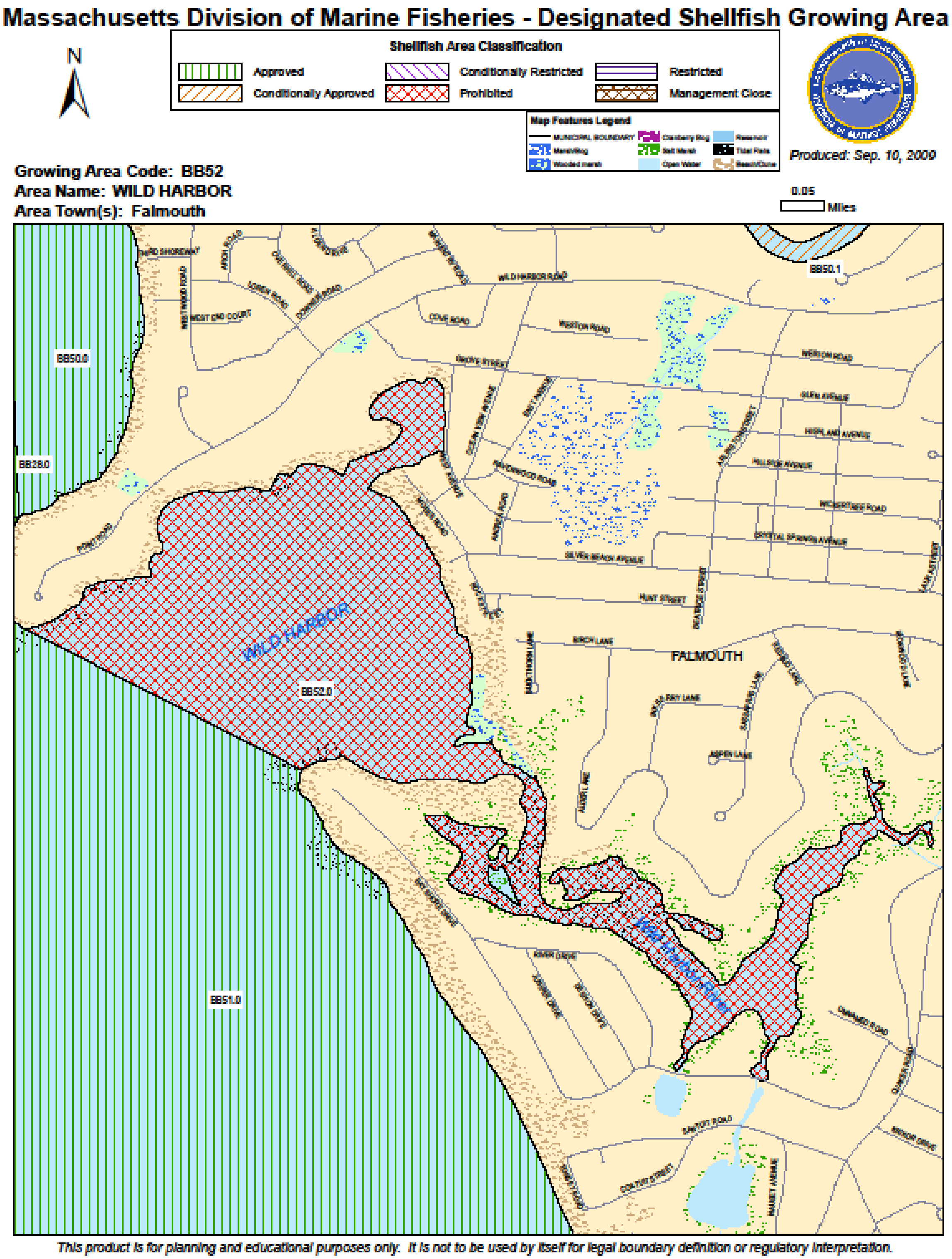

Figure VII-9. Location of shellfish growing areas in the Wild Harbor embayment system and the status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination (fecal coliform) or "activities", such as the location of marinas. For Wild Harbor, according to the MassDMF, closure is not due to 1969 fuel oil spill.

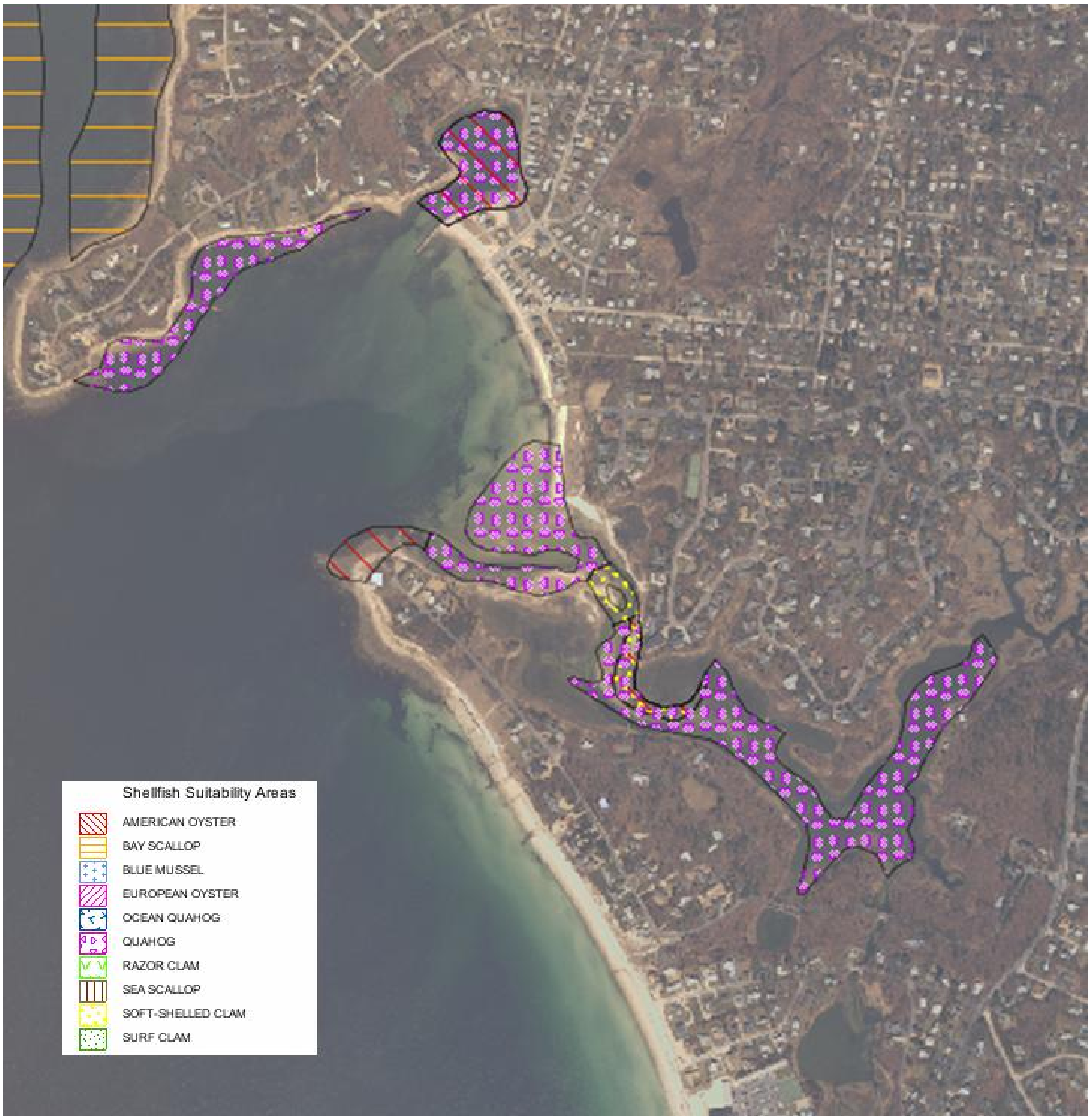

Figure VII-10. Location of shellfish suitability areas within the Wild Harbor Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence".

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll). Additional information on temporal changes within each sub-embayment of an estuary, its associated watershed nitrogen load and geomorphological considerations of basin depth, stratification and functional type further strengthen the analysis. These data were collected to support threshold development for the Wild Harbor Embayment System by the MEP and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the baseline Water Quality Monitoring Program conducted by the Coalition for Buzzards Bay's BayWatcher with analytical support from the Coastal Systems Analytical Facility at SMAST-UMass Dartmouth through 2008.

The Wild Harbor system is a complex estuary comprised of a large outer basin constrained by Nyes Neck to the north and Crow point to the south, a small inner basin and a significant salt marsh (Figure I-1). The small inner Boat Basin is tributary to the main outer basin of Wild Harbor along its northeastern shore. The tidal wetland, Wild Harbor River, exchanges water through the southwestern shore of the outer basin of Wild Harbor near Crow Point. The Wild Harbor River contains most of the 110 acres of salt marsh contained within the Wild Harbor System. Each of type of functional component to an estuary (salt marsh basin, embayment, tidal river, deep basin (sometimes drown kettles), shallow basin, etc.) has a different natural sensitivity to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of the specific basin and its ability to support eelgrass beds and infaunal communities. At present, the Wild Harbor Estuary is beyond its ability to assimilate nitrogen without impairment and is showing a low to moderate level of nitrogen enrichment, with some moderate impairment of both eelgrass in the main basin and significant impairment of infaunal habitats in the inner Boat Basin (Table VIII-1), indicating that nitrogen management of this system will be for restoration rather than for protection or maintenance of an unimpaired system.

The measured levels of oxygen depletion and enhanced chlorophyll a levels follows the spatial pattern of total nitrogen levels in this system (Chapter VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment. The spatial pattern indicated that the magnitude of oxygen depletion, enhancement of chlorophyll a levels and total nitrogen concentrations increased from the offshore waters to the main basin of Wild Harbor and were highest within the inner Boat Basin.

Oxygen records obtained from both the moorings deployed in the Wild Harbor System show ecologically significant oxygen depletion in the inner Boat Basin, but oxygen levels in Wild Harbor and Wild Harbor River were consistent with high quality habitat for these embayment and salt marsh creek sub-systems. The oxygen conditions in the inner versus outer embayment basins, are consistent with their basin structure, flushing, and nitrogen and chlorophyll a levels. The Boat Basin mooring measured the deep bottom waters of the basin, and was placed to measure oxygen levels associated with benthic animal habitat quality in this embayment basin. Moderate daily excursions in oxygen levels were observed at this location (~4 mg L^{-1}), with highs exceeding 10 mg $L^{-}1$ and periodic lows of 3 mg L^{-1} (Figure VII-3, Table VII-1). Instantaneous oxygen levels that drop below 4 mg L^{-1} are indicative of oxygen stress.

The oxygen levels are consistent with a system that is showing the effects of nitrogen enrichment and is periodically presenting stressful oxygen depletions to benthic communities.

Oxygen levels in the main basin of Wild Harbor generally showed only small to moderate depletion, with levels never dropping below 5 mg L^{-1} and almost always >5.5 mg L^{-1} (Figure VII-3, Table VII-1). Overall the oxygen levels observed within the outer basin are not generally associated with habitat stress, particularly given that declines below 6 mg L^{-1} were generally of short duration. Outer basin oxygen conditions did exhibit daily excursions in oxygen levels, but the range of daily oxygen excursion was moderate and significantly less than that observed in the Wild Harbor Boat Basin, furthest from the low nutrient and high oxygen waters of Buzzards Bay water. The oxygen dynamics at both sites is a clear indication of nitrogen enrichment, but only the inner basin is showing declines stressful to benthic animal communities.

The observed generally high oxygen levels in the main basin are consistent with the low nutrient concentrations and relatively low phytoplankton biomass in this basin. Over the 28 day deployment chlorophyll levels were generally moderate to low without extensive blooms and with levels typically between 2-6 ug L^{-1} (mooring chlorophyll-a average 3.8 ug L^{-1}). The level of oxygen observed in this system is indicative of a generally healthy system, consistent with the low chlorophyll a levels and slight nitrogen enrichment (Table VII-2, Figure VII-4).

Although there was not an oxygen mooring in Wild Harbor River, the water quality monitoring program collected 129 oxygen reading at the mouth of the tidal river near the end of ebb tide. These data showed no reading below 3 mg L^{-1}, 2% <4 mg L^{-1}, and 87% >5 mg L^{-1}. Similarly, the long term average chlorophyll a (5.8 ug L^{-1}) is also in line with ebbing salt marsh waters. These moderate levels of oxygen depletion and chlorophyll a, coupled with the salt marsh dominance of the Wild Harbor River, support the contention that this tidal river is not impaired by nitrogen loading. It should be emphasized that salt marshes are naturally nutrient and organic matter rich, and frequently show significant levels of oxygen depletion on a diurnal basis.

All of the available information on eelgrass relative to Wild Harbor indicates that the main basin (Wild Harbor) has and continues to support significant eelgrass beds. However, there is also clear evidence of eelgrass loss, primarily from the bed adjacent Nyes Neck. Eelgrass bed loss from this region of the harbor has been in the region nearest the inlet to the Boat Basin (which receives nutrient enriched ebb tide waters) and from the deeper region of the bed. The loss has been continuous over the past 6 decades and the spatial and temporal pattern is consistent with nitrogen enrichment.

It is significant that there is no evidence of eelgrass beds in the Wild Harbor River or in the Wild Harbor Boat Basin over the past 60 years. The Wild Harbor Boat Basin is an artificial embayment basin that is relatively deep and depositional, as evidenced by its organic enriched sediments. Eelgrass is not expected in the salt marsh dominated Wild Harbor River, which is naturally nutrient and organic matter enriched and becomes very shallow at low tide. The absence of evidence that the Wild Harbor Boat Basin and Wild Harbor River have ever supported eelgrass habitat, focuses management on benthic infaunal communities.

Overall, the infauna survey indicated that the main basin of Wild Harbor is supporting high quality benthic animal habitat with high diversity (23, 28 species; H'>3.4) and Evenness (>0.72), although the inner region is potentially moderately impaired. The infauna results were consistent with the eelgrass and water quality assessments. Similarly the Wild Harbor River is functioning as a non-nitrogen impaired salt marsh system with productive benthic communities typical of Cape Cod marsh creeks. In contrast, the enclosed Boat Basin presents a significantly impaired habitat for infaunal communities with low numbers of species (7) and individuals (123/sample), with low diversity (H'=1.7). The sediments are depositional, comprised of organic rich soft sulfidic mud. The sediment surface is overlain by an algal mat, which further reduces habitat quality. These sediment characteristics coupled with the periodic oxygen depletions to 3 mg L^{-1} and moderate chlorophyll levels are consistent with a benthic habitat significantly impaired by nitrogen enrichment (Table VII-5). The present animal community is also indicative of a significantly impaired habitat, being dominated by stress tolerant species indicative of high levels of organic enrichment. About 50% of the total community consisted of a single species, *Capitella capitata*, generally found in embayments with high organic matter deposition and poor habitat quality.

The inner Boat Basin supports significantly impaired benthic animal habitat. The basin morphology tends to create a depositional environment, which increases the sensitivity to organic enrichment, the levels of phytoplankton biomass, the levels of oxygen decline (3 mg L^{-1}) and the organic enrichment of the sediments are all clear evidence of nitrogen as the ultimate cause of the habitat impairment. Integration of all of the key metrics clearly indicates that the main basin of Wild Harbor is generally supporting high quality benthic animal habitat, while the Boat Basin is beyond its capacity to assimilate nitrogen loads without impairment (i.e. its just beyond its nitrogen threshold). Since Wild Harbor is also beyond its nitrogen threshold to support healthy eelgrass habitat, a slight reduction to enhance this habitat should also restore the moderate impairment of the benthic animal habitat within the inner region of this basin as well as within the Boat Basin.

In general, the habitat quality within the basins of this System is manifested by the temporal changes in eelgrass coverage and benthic community characteristics, which are consistent with the observed levels of nitrogen and organic matter enrichment and magnitude of oxygen depletion, as well as the sediment characteristics and general absence to only sparse drift macroalgal accumulations. The distribution and levels of habitat impairment within the Wild Harbor Embayment System is consistent with the low to moderate level of nitrogen enrichment. The recent losses of historically stable eelgrass habitat at the inner boundary of the main basin of Wild Harbor makes restoration of this resource the primary focus for nitrogen management, with the associated goal of restoring impaired benthic habitat within the inner Boat Basin. Determining the nitrogen target to restoring these habitats is the focus of the nitrogen management threshold analysis, below.

Table VIII-1. Summary of nutrient related habitat quality within the Wild Harbor System within the Town of Falmouth, MA, based upon assessments in Section VII.

Health Indicator	Wild Harbor Embayment System		
	Outer Basin	Inner Basin	River
Dissolved Oxygen	H[1]	MI[2]	H[3]
Chlorophyll	H[4]	MI[5]	H[6]
Macroalgae	H[7]	MI/SI[8]	H[7]
Eelgrass	MI[9]	--[10]	--[10]
Infaunal Animals	H/MI[11]	SI[12]	H[13]
Overall:	**MI[14]**	**MI/SI[15]**	H[16]

1- deep water mooring almost always >5.5 mg/L, >6 mg/L 89% of time, daily excursion ~2.5 mg/L
2- mooring oxygen <5mg/L 8%, <4mg/L 1% of time, periodically to 3mg/L, daily excursion ~2.5 mg/L WQMP: <5 mg/L 11%, < 6 mg/L 48%, 1% of samples <4 mg/L, none <3mg/L (N=86), daily excursions ~4 mg/L
3- Salt marshes naturally have periodic low D.O.; WQMP: <5 mg/L 13% of time, 3-4mg/L 2% of 129 samples, none <3mg/L.
4 - levels low for a coastal basin, averaging 3.8 ug L^{-1}, <5 ug L^{-1} 84% and <10 ug L^{-1} 100% of record;
5- levels moderate for a coastal basin, mooring average 9.4 ug L^{-1}, >10ug L^{-1} 37% of record; blooms 20-30 ug L^{-1}; WQMP long-term average 7.1 ug L^{-1}
6- Salt marsh dominated system, WQMP long-term average 5.8 ug L^{-1}
7- drift algae generally absent, some small patches, attached Codium in main basin only.
8- drift algae sparse or absent, but significant coverage by surface algal mat on sulfidic sediments.
9- basin supports high quality eelgrass habitat, but loss from beds in region between the Wild Harbor and Boat Basin and from the deeper edge of the beds along Nyes Neck. Temporal and spatial pattern of loss from the inner and deeper margins of the beds is typical of nitrogen enrichment effects.
10- no evidence of present or historic eelgrass habitat within this basin
11- Outer: high numbers of individuals, species (25), diversity (>3) and Evenness (>0.7), dominated by non-stress indicator species with crustaceans and mollusks, some deep burrowers; Deep central inner area, moderate numbers of individuals and species, dominated by non-stress indicator species.
12- low numbers of species, moderate number of individuals, dominated by organic enrichment tolerant species; low diversity and Evenness consistent with the organic rich sediments and periodic D.O. depletion to <4 mg/L.
13- Community typical of a functioning salt marsh. High numbers of individuals, moderate species (17), diversity (2.6) and Evenness (>0.64), naturally organic rich ecosystem.
14- Stable high quality eelgrass habitat with some loss at the margin to the Boat Basin and deeper margins; benthic infaunal animal communities are diverse and productive with non-stress indicator species, with the exception of the deep inner region. The level of eelgrass impairment is consistent with benthic habitat indicators, low chlorophyll and generally high D.O. Loss of marginal eelgrass coverage is indicative of nitrogen enrichment and rates a designation of "Moderate Impairment".
15- Significantly impaired benthic animal communities, organic enriched sulfidic sediments overlain with an algal mat. Moderate-high chlorophyll levels, consistent with periodic oxygen depletions to <4 mg/L.
16- All indicators support the contention that this basin is functioning as a non-nitrogen impaired salt marsh dominated tidal creek.

H = High quality habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severely Degraded; -- = not applicable to this estuarine reach
WQMP: Water Quality Monitoring Program

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 Wild Harbor 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 Wild Harbor Embayment System presently shows a moderate impairment to eelgrass habitat within its outer basin, the main basin of Wild Harbor. The impairment is based upon the recent temporal trend in loss of eelgrass from the inner margin of the basin, at the inner Boat Basin boundary and loss at the deeper margin of the Nyes Neck beds. Both the location and the temporal trend are consistent with nitrogen enrichment. However, as the rate of loss has been gradual and significant eelgrass resources still exist, that indicates that this estuarine basin is only just beyond its nitrogen threshold (i.e. the level of nitrogen a system can tolerate without impairment). The presence of stable dense eelgrass beds throughout the main basin of Wild Harbor and the generally high quality benthic animal habitat throughout the embayment system (except for the Boat Basin) also indicates a system just beyond its threshold. The indication of impairment to eelgrass and infaunal animal habitat, to the extent that it was observed, is supported by the observed levels of oxygen depletion and clearly enhanced chlorophyll a levels in the inner Boat Basin waters. The spatial distribution of high quality and impaired habitats and the associated oxygen and chlorophyll a levels also parallels the gradient in watercolumn total nitrogen levels within this estuary. The tidally averaged total nitrogen at the boundary between the inner Boat Basin and outer Wild Harbor basin, at the inner extent of the historic eelgrass beds, is 0.447 mg N L^{-1}. It should be noted that this inner extent of historic eelgrass was determined both from the 1951 (MassDEP, C. Costello) and the 1987 (Costa 1988) assessments. The relatively low levels of nitrogen within the central region of Wild Harbor, ~0.3 mg L^{-}1, are consistent with the general high quality of eelgrass and benthic animal habitat within this system. But the clear enrichment of the inner region (the boundary area losing eelgrass) and within the Boat Basin is consistent with habitat impairment documented for this estuary (Section VII). Restoring the impairments to eelgrass and benthic animal habitats is the focus of the nitrogen management threshold analysis (Section VIII.3).

As eelgrass within the Wild Harbor Embayment System is a critical habitat structuring the productivity and resource quality of the entire system, and given that it is presently showing moderate impairment, restoration of this resource is the primary target for overall restoration of this system. Nutrient management planning for restoration of the eelgrass habitat at the boundary between the inner region of Wild Harbor and the Boat Basin should focus on reducing the level of nitrogen enrichment in embayment waters primarily through watershed nitrogen management within the Wild Harbor watershed.

The loss of eelgrass within the main basin of Wild Harbor related to the beds along Nyes Neck is associated with a tidally averaged nitrogen (total nitrogen, TN) level of 0.447 mg N L^{-1} at the sentinel station (WH-1), while the high quality eelgrass habitat exist at lower TN levels, <0.35 mg N L^{-1}. These TN levels and habitat stability/decline are consistent with persistence and loss of eelgrass at similar depths in other estuaries in southeastern Massachusetts. The Nantucket Harbor Estuary tidally averaged levels in the lower reach of Head of the Harbor (0.340-0.353) were associated with recent loss of eelgrass coverage, while eelgrass was lost from West Falmouth Harbor when tidally averaged TN exceeded 0.35 mg L^{-1}. Similarly, within the deep basins of Quissett Harbor only slight eelgrass loss was observed at tidally averaged TN of 0.35 mg L^{-1} indicating that eelgrass in that system required slightly low TN levels of 0.34 mg L^{-1} for long-term stability. Based upon the present TN levels in the Wild Harbor System and comparison to other similar estuaries a threshold for tidally averaged TN at the sentinel station at the boundary of the inner and outer sub-embayment basins (WH-1) of 0.35 mg L^{-1} was selected.

This threshold is similar to that for West Falmouth Harbor and Phinneys Harbor at similar depths, and is focused in part on restoring eelgrass where it had persisted until recently near the tidal inlet to the Boat Basin. In addition, lowering the level of nitrogen enrichment at the sentinel station will lower nitrogen levels within the Boat Basin and inner region of Wild Harbor (Section VIII.3) with the parallel effect of improving impaired infaunal habitat. Therefore, the goal is to achieve the nitrogen target at the sentinel location to restore the historical eelgrass habitat within Wild Harbor basin, resulting also in the restoration of infaunal habitat throughout the embayment basins. The nitrogen loads associated with the threshold concentration at the sentinel location and secondary infaunal check stations are discussed in Section VIII.3, below.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Wild Harbor system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for Wild Harbor. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment. A comparison between present septic and total watershed loading and the loadings for the two modeled threshold scenarios is provided in Tables VIII-2 and VIII-3.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required more than 40% removal of septic load (associated with direct groundwater discharge to the embayment) for the entire system. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1. It should be noted that in this particular scenario the New Silver Beach sewer project will fulfill 8.5% of the nitrogen load reduction needed to restore the nitrogen impaired habitats within the Wild Harbor System.

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. For Example, removal of 77% of the septic load from the main Wild Harbor watershed results in a 56% reduction in total watershed nitrogen load. No load reduction was necessary for the Dam Pond watershed. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent 'worst-case' summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Buzzards Bay, as discussed in Section VI.2.6.1.

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

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.

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

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

sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Wild Harbor	7.499	1.725	-77.0%
Wild Harbor River	8.811	7.049	-20.0%
Dam Pond Stream	1.052	1.052	+0.0%
System Total	17.362	9.826	-42.5%

Table VIII-3. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Wild Harbor system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Wild Harbor	10.326	4.552	-55.9%
Wild Harbor River	11.825	10.062	-14.9%
Dam Pond Stream	1.507	1.507	+0.0%
System Total	23.658	16.121	-31.9%

Table VIII-4. Threshold sub-embayment loads used for total nitrogen modeling of the Wild Harbor system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Wild Harbor	4.552	1.033	-10.232
Wild Harbor River	10.062	0.447	-0.359
Dam Pond Stream	1.507	-	-
System Total	16.121	1.480	-10.591

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

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Wild Harbor inner harbor	**WH-1**	**0.447**	**0.353**	**-56.8%**
Wild Harbor River	WH-2	0.482	0.440	-21.2%

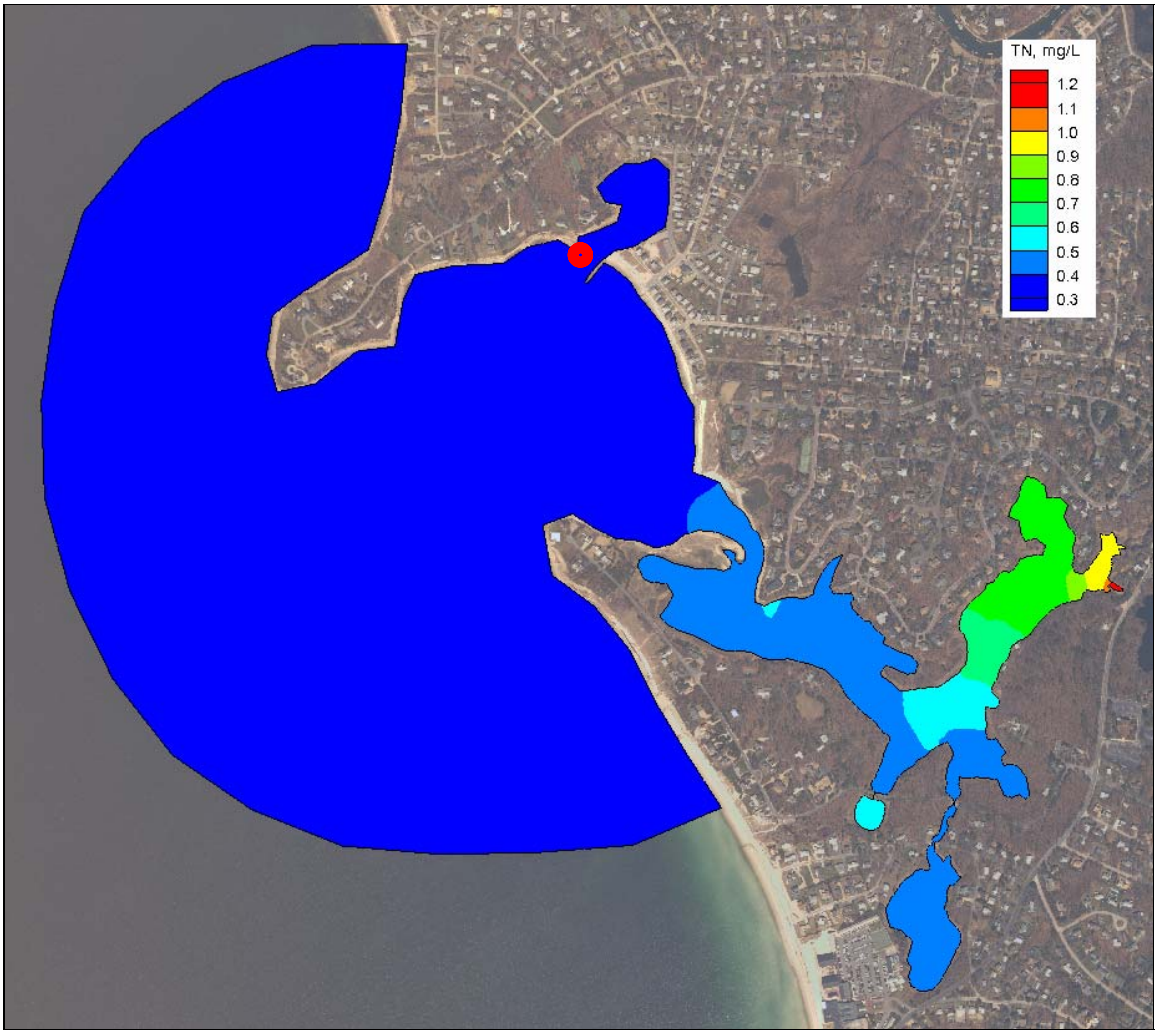

Figure VIII-1. Contour plot of tidally averaged modeled total nitrogen concentrations (mg/L) in the Wild Harbor system, for threshold. Red dot indicates sentinel station from water quality monitoring program, WH-1, at location of inner edge of historical eelgrass coverage.

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