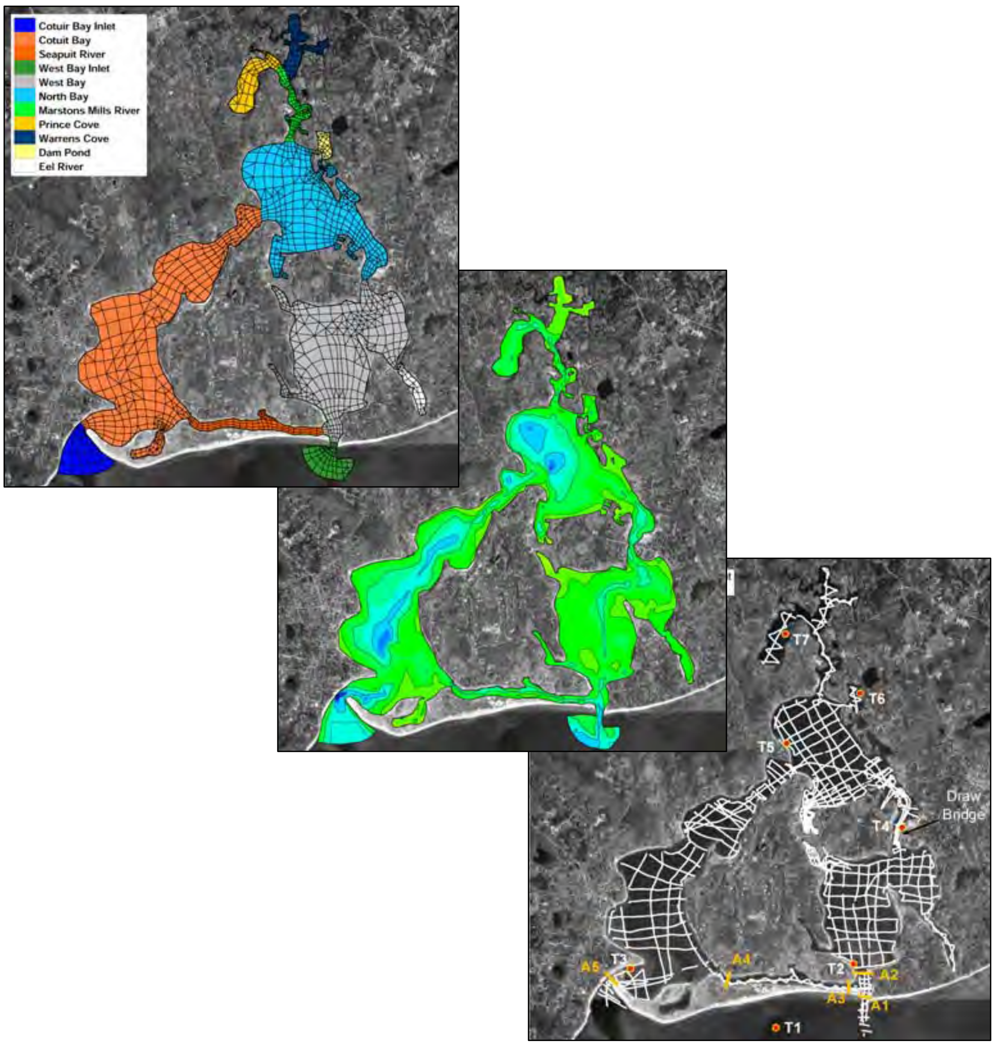
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Three Bays, Barnstable, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of Environmental Protection

FINAL REPORT –APRIL 2006

Massachusetts Estuaries Project

LINKED WATERSHED-EMBAYMENT MODEL TO DETERMINE CRITICAL NITROGEN LOADING THRESHOLDS FOR THREE BAYS IN THE TOWN OF BARNSTABLE, MA

FINAL REPORT – APRIL 2006

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Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Rushy Marsh Barnstable, Massachusetts

Executive Summary

1. Background

This report presents the results generated from the implementation of the Massachusetts Estuaries Project's Linked Watershed-Embayment Approach to the Three Bays embayment system, a coastal embayment within the Town of Barnstable, Massachusetts. Analyses of the Three Bays embayment system was performed to assist the Town with up-coming nitrogen management decisions associated with the Towns' current and future wastewater planning efforts, as well as wetland restoration, anadromous fish runs, shell fishery, open-space, and harbor maintenance programs. As part of the MEP approach, habitat assessment was conducted on the embayment based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. Nitrogen loading thresholds for use as goals for watershed nitrogen management are the major product of the MEP effort. In this way, the MEP offers a science-based management approach to support the Town of Barnstable 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 Three Bays 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 Three Bays 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 Three Bays embayment system within the Town of Barnstable 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 Barnstable 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 it plans to rapidly implement. The Town of Barnstable has also completed and implemented wastewater planning in other regions of the Town not associated with the Three Bays embayment system. The Town has nutrient management activities related to their tidal embayments, which have been associated with the MEP effort in the Centerville River/Harbor and the Lewis Bay embayment systems. The Town of Barnstable and work groups have recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Town. 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 Three Bays embayment system by using site-specific data collected by the MEP and water quality data from the Water Quality Monitoring Program conducted by Three Bays Preservation in partnership with the Town of Barnstable, with technical guidance from the Coastal Systems Program at SMAST (see Chapter 2). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Town of Barnstable Planning Department, and watershed boundaries delineated by USGS. This land-use data was used to determine watershed nitrogen loads within the Three Bays embayment system and each of the systems sub-embayments as appropriate (current and build-out loads are summarized in Table IV-5). 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 Three Bays 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 Vineyard Sound source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Three Bays 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 Rushy Marsh 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 Three Bays 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 Three Bays embayment system in the Town of Barnstable. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. The MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of each embayment system. The concept was that since septic system nitrogen loads generally represent 85% - 90% of the controllable watershed load to the Three Bays 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 Three Bays system based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. At present, the Three Bays system is showing significantly impaired to severely degraded habitat quality in the Prince Cove and Warrens Cove sub-embayments as well as the upper portion of North Bay. The lower portion of North Bay as well as Eel Pond are showing indications of moderate bordering on significant impairment while Cotuit Bay and West Bay are both showing signs of moderate impairment. All of the habitat indicators are consistent with this evaluation of the whole of system (Chapter VII).

The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). The clear evidence of oxygen levels above atmospheric equilibration indicates that the Three Bays system is eutrophic.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality within the estuary. The major sub-embayments to the Three Bays system (Cotuit Bay, West Bay, North Bay and Prince Cove) are currently under seasonal oxygen stress, consistent with nitrogen enrichment (Chapter VII). That the cause is nitrogen enrichment is supported by parallel observations of chlorophyll a (Table VII-2). Oxygen conditions and chlorophyll a levels generally improved with decreasing distance to the tidal inlet, although all basins showed oxygen depletions to <4 mg L^{-1}. There was also a clear gradient in chlorophyll a, with highest levels in the uppermost reaches and lowest levels near the tidal inlet to Nantucket Sound. The results of the summer oxygen and chlorophyll a studies are consistent with the absence of eelgrass throughout the Three Bays System and the near absence of animal communities throughout the upper basins where oxygen depletions routinely dropped below 3 mg/L.

Currently, there are no remaining eelgrass beds within the Three Bays System. However, it appears that all of the major sub-embayments had water quality conditions capable of supporting eelgrass (except in the deeper channels and basin depths) in 1951. However, eelgrass appears to have been restricted to the shallows (North and Cotuit Bays) or to Prince Cove and West Bay basins. If the issue in 1951 was nitrogen enrichment, the pattern of the beds would have been very different, with more eelgrass in lower Cotuit Bay and West Bay and much less in Prince Cove and North Bay (except in the very shallows). Instead, it is likely that disturbance related to activities in North and Cotuit Bays associated with training during WWII played a role in the North and Cotuit Bay pattern of beds in the 1951 assessment. Whatever the cause, it is clear that in the recent past, the Three Bays system was capable of supporting eelgrass within each of its major sub-embayments. It also appears that the recent losses (post 1951) are associated with nitrogen enrichment, as in virtually every other embayment in southeastern Massachusetts. The absence of eelgrass in each basin and the fact that they supported eelgrass in the recent past classifies each basins eelgrass habitat as “significantly impaired” (Table VIII-1).

The current absence of eelgrass in each of the major sub-embayments of the Three Bays System is consistent with the observed oxygen depletions in each basin and the high chlorophyll levels in the upper regions. The greater depths in the Three Bays Estuary also makes oxygen depletions more likely than in shallow basins with the same nitrogen levels. This results from the fact that deeper systems are more likely to periodically stratify. The central deep basins in North Bay and Prince Cove are particularly sensitive to eelgrass loss as it takes less intense phytoplankton blooms to reduce light penetration to the bottom, and thereby prevent eelgrass growth. In addition, the basins are sensitive to periodic oxygen depletion. At this time, it is not clear if these regions have historically (100 years) supported eelgrass. However, eelgrass beds fringing these basins are well documented. As regards the lack of eelgrass within the lowermost portion of Cotuit Bay and the Seapuit River, it is likely associated with the documented highly dynamic coastal processes in this area. The level of natural disturbance in this region is very high (sand transport, overwash, etc). Physical stability is important to the ability of eelgrass beds to form and persist.

The Infauna Study indicated that most of the upper areas of the Three Bays system are presently significantly impaired to severely degraded by nitrogen enrichment (Prince Cove, Warrens Cove and portions of North Bay), while the lower basins of Cotuit Bay and West Bay are moderately impaired (Table VII-4). Prince Cove, Warrens Cove and 2 of 3 sites in North Bay are virtually devoid of infaunal animal communities. The central region of North Bay currently supports a transitional community dominated by amphipods, indicative of organic matter enrichment. In contrast, Cotuit and West Bays generally have ~500-2000 individuals per grab and 16-26 species. While there are stress indicator species (generally *Capitella* or *Streblospio*) in numbers at these locations there are also other species indicative of a healthy environment and overall high diversity. Overall, the pattern of infaunal community quality is consistent with the pattern of oxygen depletion and chlorophyll a during summer and the absence of eelgrass. All sites showed some level of degradation, either in number of individuals, diversity or the presence of stress indicator species.

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 Barnstable Three Bays embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 85% - 90% 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 Great, Green and Bournes Pond Systems, Popponesset Bay System, the Hamblin / Jehu Pond / Quashnet River analysis in eastern Waquoit Bay, the analysis of the adjacent Rushy Marsh system and the Pleasant Bay and Nantucket Sound embayments associated with the Town of Chatham. This is almost certainly going to be true for the other embayments within the MEP area, as well.

The threshold nitrogen levels for the Three Bays embayment system in Barnstable were determined as follows:

Three Bays Threshold Nitrogen Concentrations

- Following the MEP protocol, the restoration target for the Three Bays system should reflect both recent pre-degradation habitat quality and be reasonably achievable. Based upon the assessment data (Chapter VII), eelgrass bed restoration within Cotuit Bay and West Bay, with restoration of marginal beds in North Bay and Prince Cove is supportable. In addition, in the central basins of North Bay and Prince Cove, where eelgrass habitat has not been documented, as well as in Warrens Cove, restoration of infaunal habitat is necessary. Achieving these habitat quality targets will also result in mitigation of the present macroalgal accumulation problem in Warrens Cove. To achieve these habitat restoration targets, for the Three Bays system a single sentinel location was selected with secondary criteria that must be achieved at other locations. The secondary criteria serve only as checks to make sure that the targets are achieved when the nitrogen threshold at the sentinel station has been reached.
- The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.38 mg TN L^{-1} at the sentinel location and 0.40 mg TN L^{-1} within the marginal regions (shallows) of North Bay. This secondary level to check restoration of marginal beds in North Bay (O.40 mg TN L^{-1}) is consistent with the analysis of restoration of fringing eelgrass beds in nearby Great Pond, and analysis where eelgrass

beds in deep waters could not be supported at a tidally averaged TN of 0.412 mg TN L^{-1} at depths of 2 m. Similarly prior MEP analysis in Bournes Pond indicated that tidally averaged TN levels of 0.42 mg TN L^{-1} excluded beds from all but the shallowest water. The MEP Technical Team cannot specify the exact extent of marginal beds to be restored in the upper deep basins. At tidally averaged TN levels of 0.42 mg TN L^{-1} the eelgrass habitat would be restricted to very shallow waters, while at 0.40 mg TN L^{-1} the eelgrass habitat should reach to 1-2 meters depth, based upon the data from nearby systems. In addition, the persistence of eelgrass beds through 1995-2001 in the shallow waters of south Windmill Cove, but in a stable physical setting, were at nitrogen levels (tidally averaged TN ~0.40 mg L-1).

- Since infaunal animal habitat is also a critical resource to the Three Bays System, the secondary metric for a successful restoration (after eelgrass) will be to restore the significantly impaired/severely degraded habitats in the Prince Cove/Warrens Cove and North Bay basins. In the upper more muddy basins of other nearby systems, healthy infaunal habitat is associated with nitrogen levels of TN <0.5 mg TN L^{-1}. This was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 were found supportive of high infaunal habitat quality in this system. In the Three Bays System, present healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay) However, the impaired areas are at nitrogen levels of TN >0.5 mg TN L^{-1} (North Bay) and are severely degraded at nitrogen levels of TN >0.6 mg TN L^{-1}. This is consistent with the findings discussed above from other systems and fully supports a secondary nitrogen criteria for the upper muddy basins of 0.5 mg TN L^{-1}.

It is important to note that the analysis of future nitrogen loading to the Three Bays 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 useage and increasing use of fertilizers (presently less than half of the parcels use lawn fertilizers). Therefore, watershed-estuarine nitrogen management must include management approaches to prevent increased nitrogen loading from both shifts in land-uses (new sources) and from loading increases of current land-uses. The overarching conclusion of the MEP analysis of the Three Bays estuarine system is that restoration will necessitate a reduction in the present (2004) 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 Three Bays system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations. Loads to estuarine waters of the Three Bays system include both upper watershed regions contributing to the major surface water inputs (Marstons Mills River and Little River).

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)
THREE BAYS SYSTEM										
Cotuit Bay [a]	2.447	5.515	20.225	--	25.740[b]	5.786	-54.443	-22.917	0.39-0.44	--
West Bay	1.170	3.578	15.490	--	19.068	4.233	3.815	27.117	0.38-0.48	--
Seapuit River	0.452	0.847	2.921	0.016	3.767	0.452	-5.418	-1.199	0.32	--
North Bay	1.970	4.468	24.978	--	29.447	3.953	67.522	100.922	0.50-0.52	--
Prince Cove [a]	3.964	10.337	24.836	0.092	35.173[c]	1.230	0.512	36.914	0.60-0.70	--
Warren Cove	1.945	5.052	6.975	--	12.027	--	8.830	20.857	0.64	--
Prince Cove Channel	0.515	0.770	4.767	--	5.537	--	2.345	7.882	0.64	--
Three Bays System Total	12.463	30.567	100.192	0.108	130.759	15.655	23.162	169.576	**0.32-0.70**	0.38

[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. Warren Cove and Prince Cove Channel atmospheric loads are included with the Prince Cove Load.
[6] composed of natural background, fertilizer, runoff, septic system atmospheric deposition and benthic flux loadings
[7] average of 1999 – 2004 data, ranges show the upper to lower regions (highest-lowest) of a sub-embayment.
[8] Eel grass threshold for sentinel site located at “The Narrows” between North Bay and Cotuit Bay (0.38 mg/L TN), and infaunal target for Prince Cove of 0.50 mg/L TN.
[a] Include loads from surface water sources (i.e., Marstons Mills River to Prince Cove and Little River to Cotuit Bay).
[b] Sum of Cotuit Bay watershed and Little River input.
[c] Sum of Marstons Mills River outflow from Mill Pond + Lower Marstons Mill South (Cresent below Mill Pond, WS#18) + Prince Cove (groundwater watershed).

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

Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed reductions needed to achieve threshold load levels
THREE BAYS SYSTEM						
Cotuit Bay	25.740	22.335	5.786	-45.788	-17.666	-13.2%
West Bay	19.068	15.970	4.233	3.469	23.673	-16.2%
Seapuit River	3.767	3.767	0.452	-5.371	-1.152	0.0%
North Bay	29.447	4.468	3.953	45.202	53.624	-84.8%
Prince Cove	35.173	17.890	1.230	0.323	19.444	-49.1%
Warren Cove	12.027	5.052	--	6.225	11.277	-58.0%
Prince Cove Channel	5.537	0.770	--	1.541	2.311	-86.1%
Three Bays System Total	130.759	70.254	15.655	5.602	91.511	-46.3%

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

ACKNOWLEDGMENTS

The Massachusetts Estuaries Project Technical Team would like to acknowledge the contributions of the many individuals who have worked tirelessly for the restoration and protection of the critical coastal resources of the Three Bays 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 significant time and effort in data collection and discussion spent by members of the Three Bays Water Quality Monitoring Program and SMAST "volunteers". These individuals gave of their time to collect nutrient samples from this system, without which the present analysis would not have been possible. Of particular note are Lindsey Counsell of Three Bays Preservation, Coordinator of the Monitoring Program, and John Jacobsen of the Town of Barnstable, Coordinator of the Town-wide Monitoring Program. Similarly, many in the Town of Barnstable helped in this effort, the Barnstable Department of Public Works and the Barnstable Shellfish Department, as well as the Barnstable Nutrient Management Committee. The technical team would like to specifically acknowledge the efforts of Mark Ells for helping to move this effort forward and Three Bays Preservation for their energy and perseverance in the stewardship of one of the regions key coastal resources.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by Tom Camberari and Margo Fenn of the Cape Cod Commission; Art Clark and Nora Conlon of the USEPA; and our MADEP colleagues: Andrew Gottlieb, Arleen O'Donnell, Art Screpetis, Rick Dunn, Steve Halterman, Russ Issac, Glenn Haas, and Ron Lyberger. We are also thankful for the long hours in the field and laboratory spent by the many technical staff (Jennifer Antosca, Michael Bartlett, and David White), interns and students within the Coastal Systems Program at SMAST-UMD.

Support for this project was provided by the Town of Barnstable, Barnstable County, MADEP, and the USEPA, with in-kind support from Three Bays Preservation Inc.

SUGGESTED CITATION

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2005). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Three Bays, Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

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I. INTRODUCTION

The Three Bays Embayment System is a complex estuary located primarily within the Town of Barnstable on Cape Cod, Massachusetts with a southern shore bounded by water from Nantucket Sound (Figure I -1). The Bay's watershed is distributed primarily among the Towns of Barnstable and Sandwich, with a small region adjacent Santuit Pond (<1% of the watershed) within the Town of Mashpee and comprised primarily "protected" forest land. The Town of Sandwich has jurisdiction over land and associated land uses in the uppermost portions of the overall watershed to The Three Bays system. Specifically, portions of the Three Bays watershed that exist within the Town of Sandwich are generally situated above the Spectacle Pond, Lawrence Pond and Triangle Pond system and within the contributing area to the upper and mid reaches of the Marstons Mills River. However, the majority of the watershed falls within the Town of Barnstable, which includes the watershed contributing direct groundwater discharge to the estuary and contributing to the lower Marstons Mills River and to the Little River. Although land-uses closest to an embayment generally have greater impact than those in the upper portions of the watershed, which are subject to nitrogen attenuation during transport through natural aquatic systems (e.g. ponds, rivers, wetlands etc.) prior to discharge to the embayment, effective restoration of the Three Bays System, will require both the Towns of Barnstable and Sandwich to be active in nutrient management and restoration discussions and planning.

The large number of sub-embayments to the Three Bays System greatly increases the shoreline and decreases the travel time of groundwater (and its pollutants) from the watershed recharge areas to bay regions of discharge. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, the Three Bays system and its sub-embayments along the Barnstable shores are at risk of eutrophication (over enrichment) from high nitrogen loads in the groundwater and runoff from their watersheds.

The Three Bays Embayment System is a complex estuary, with multiple inlets and sub-embayments (Cotuit Bay, West Bay, North Bay, Prince's/Warrens Cove). The estuary receives tidal waters from Nantucket Sound into its two large lower basins, Cotuit Bay to the west of Osterville Grand Island, and West Bay to the east of Grand Island. Floodwaters from Nantucket Sound enter the two large lower basins of the Three Bays system through 2 tidal inlets and flow through the Seapuit River (Figure I-1). Both Cotuit Bay and West Bay exchange tidal waters with upgradient North Bay through "natural" channels. A third tidal passage apparently once existed through the salt marsh between Little Island and Grand Island, but this no longer exists, due to the causeway supporting the roadway from the mainland to Grand Island. Further upgradient of North Bay are two smaller sub-embayments (Prince's Cove and Warren's Cove). These smaller sub-embayments (including Tim's Cove adjacent to Cotuit Bay, Eel River adjacent to West Bay and Seapit River connecting Cotuit Bay to West Bay) constitute important components of the Town's natural and cultural resources.

Figure I-1. Study region for the Massachusetts Estuaries Project analysis of the Three Bays Embayment System. Tidal waters enter the Bay through two inlets from Nantucket Sound. Freshwaters enter from the watershed primarily through 2 surface water discharges (Marstons Mills River and Little River) and direct groundwater discharge.

The present the Three Bays system results from tidal flooding of drowned river valleys formed primarily by the Marstons Mills River discharging to the Princes Cove/Warrens Cove sub-embayment upgradient of North Bay as well as incorporated ancient kettle ponds. Little River may also have contributed slightly to the formation, discharging to the head of Cotuit Bay, although the upper reach of Little River is primarily man-made for herring production. Drowning of the river valleys occurred gradually as a result of rising sea level following the last glaciation approximately 18,000 years BP. It appears that the Three Bays system has had multiple inlet positions as an estuary. Coastal processes, including the formation of a barrier spit (beach and dune deposits) have altered the positions of the tidal inlet(s) to the Three Bays system, affected tidal exchange and are responsible for enclosing Cotuit Bay and West Bay from Nantucket Sound as sea level rose. The Bay is presently separated from Nantucket Sound by a barrier spit (Dead Neck), which grew from the southeastern shore, and is a very dynamic geomorphic feature. The Bay presently exchanges tidal water with Nantucket Sound through two maintained inlets. The smaller man-made eastern inlet has been stabilized with riprap where as the larger western inlet flowing into Cotuit Bay remains unarmored and is maintained by dredging.

The primary ecological threat to the Three Bays embayment system as a coastal resource is degradation resulting from nutrient enrichment. Although the watershed and the Bay have some organic contamination and bacterial contamination issues, these do not appear to be having large system-wide impacts (Howes et al. MEP Bacterial Tech Report 2004). Bacterial contamination causes closures of shellfish harvest areas regularly within the Prince’s Cove and Warren’s Cove sub-embayments as well as portions of North Bay. In contrast, loading of the critical eutrophying nutrient, nitrogen, to the Three Bays System has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to the Bay, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater.

The Town of Barnstable has been among the fastest growing towns in the Commonwealth over the past two decades and does have a centralized wastewater treatment system located in Hyannis, however, the vast majority of the Three Bays watershed is not connected to any municipal sewerage system, but rather, rely on privately maintained septic systems for treatment and disposal of wastewater. As existing and probable increasing levels of nutrients impact Barnstable’s coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

As the primary stakeholder to the Three Bays System, the Town of Barnstable was among the first communities to become concerned over perceived degradation of Bay waters. The concern over declining habitat quality followed significant on-going efforts to preserve open space within the Marstons Mills River and Little River sub-watersheds. This local concern also led to the conduct of several studies (see Chapter II) of nitrogen loading to the system (Cape Cod Commission 1998) and the formation of a citizens organization, Three Bays Preservation Inc., to provide local stewardship of the Three Bays system and to assist in advancing restoration of the System within the Town. One of the initial projects of Three Bays Preservation was to establish, in 1999, a nitrogen related water quality monitoring program throughout the Three Bays system to support restoration efforts. The Three Bays Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD and over the past several years have been incorporated into Barnstable’s Town-wide embayment monitoring program. This effort provides the quantitative watercolumn nitrogen data (1999-2004) required for the implementation of the MEP’s Linked Watershed-Embayment Approach used in the present study.

Since the initial results of the Water Quality Monitoring Program and the land-use studies indicated that parts of the Three Bays system were presently impaired by land-derived nitrogen inputs, the Town of Barnstable and Three Bays Preservation undertook additional site-specific data collection to support MEP’s ecological assessment and modeling project. The effort was part of the Town’s Wastewater Facilities Planning effort and was aimed at restoration of the resources within the Three Bays system. Under the direction of the Town of Barnstable DPW, the Three Bays System was included in the second tier of the Massachusetts Estuaries Project (rank #15).

The common focus of the Barnstable effort has been to gather site-specific data on the current nitrogen related water quality throughout the Three Bays System and determine its relationship to watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program, and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each major sub-embayment. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater planning and nitrogen management alternatives development needed by the Town of Barnstable. 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, most notably from members of the local non-governmental organization (NGO) Three Bays Preservation. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Barnstable to develop and evaluate the most cost effective nitrogen management alternatives to restore this valuable coastal resource which is currently being degraded by nitrogen overloading. It is important to note that the Three Bays System has been significantly altered by human activities over the past ~400 years (see Section I.2, below). As a result, the present nitrogen "overloading" appears to result partly from alterations to the geomorphology and ecological systems. These alterations subsequently affect nitrogen loading within the watershed and influence the degree to which nitrogen loads impact the estuary. Therefore, restoration of this system should focus on managing nitrogen through both management of nitrogen loading within the watershed and restoration/management of processes which serve to lessen the amount or impact of nitrogen entering the estuary.

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 and the food chain which they support. At higher levels, nitrogen 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 frequently related to changes in land-use as watersheds become more developed. The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities.

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

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

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

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region’s coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the DEP and municipalities 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 outline of an implementation plan. For this project, the DEP recognizes that there are likely to be multiple ways to achieve the desired goals, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, DEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly) through the Comprehensive Wastewater Management Planning process.

In appropriate estuaries, bacterial technical reports will be developed in support of a Cape Cod wide TMDL for bacterial contamination. As possible, these analyses of bacterial contamination will be conducted in concert with the nutrient effort (particularly if there is a 303d listing), as was the case for the Prince’s Cove sub-embayment to the Three Bays system. The MEP (through SMAST) has already completed the Technical Analysis and Report to support the inclusion of this system in the Cape Cod wide bacterial TMDL that the MASSDEP is in the process of producing. The goal of the bacterial program is to provide information to guide targeted sampling for specific source identification and remediation.

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

The major MEP nitrogen management goals are to:

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

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

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

The Linked Model has been applied for watershed nitrogen management in approximately 15 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 facilitates the evaluation of nitrogen management alternatives relative to meeting water quality targets within a specific embayment. The Linked Watershed-Embayment Model also enables Towns to evaluate improvements in water quality relative to the associated cost. In addition, once a model is fully functional it can be "kept alive" and updated for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

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

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

I.2 SITE DESCRIPTION

The Three Bays embayment system exchanges tidal water with Nantucket Sound through two inlets at the east and west ends of a barrier beach referred to as Dead Neck. The eastern most inlet connecting Nantucket Sound to West Bay was opened by dredging in the early 1900's and is armored on both the Oyster Harbors Beach side as well as the Wianno Beach side. For the MEP analysis, the Three Bays estuarine system has been partitioned into five general sub-embayment groups: the 1) Cotuit Bay, 2) West Bay, 3) North Bay, 4) Prince's Cove and 5) Warren's Cove (see Figure I-1). The estuarine reach of the Marstons Mills River was considered as part of the Prince's Cove / Warren's Cove sub-embayment system flowing into the head of North Bay in the modeling and thresholds analysis.

Within the Three Bays System, the tidal portion of Prince's Cove and Warren's Cove sub-embayment system (Marstons Mills River) including the upper portion of North Bay show the greatest diversity of estuarine habitats, with most of the System's salt marsh area, shallow tidal flats and large salinity fluctuations being present in this area. In contrast, Cotuit Bay and West Bay show more typical embayment characteristics dominated by open water areas, small fringing salt marshes, relatively stable salinity gradients and relatively large basin volumes relative to tidal prism. Although the upper two sub-embayment systems up-gradient of North Bay and the open water portions of Cotuit Bay and West Bay exhibit different hydrologic characteristics (river dominated versus tidally dominated), the tidal forcing for these systems is generated from Nantucket Sound. Nantucket Sound, adjacent Oyster Harbors Beach (Dead Neck), exhibits a moderate to low tide range, with a mean range of about 2.5 ft. Since the water elevation difference between Nantucket Sound and the Three Bays system is the primary

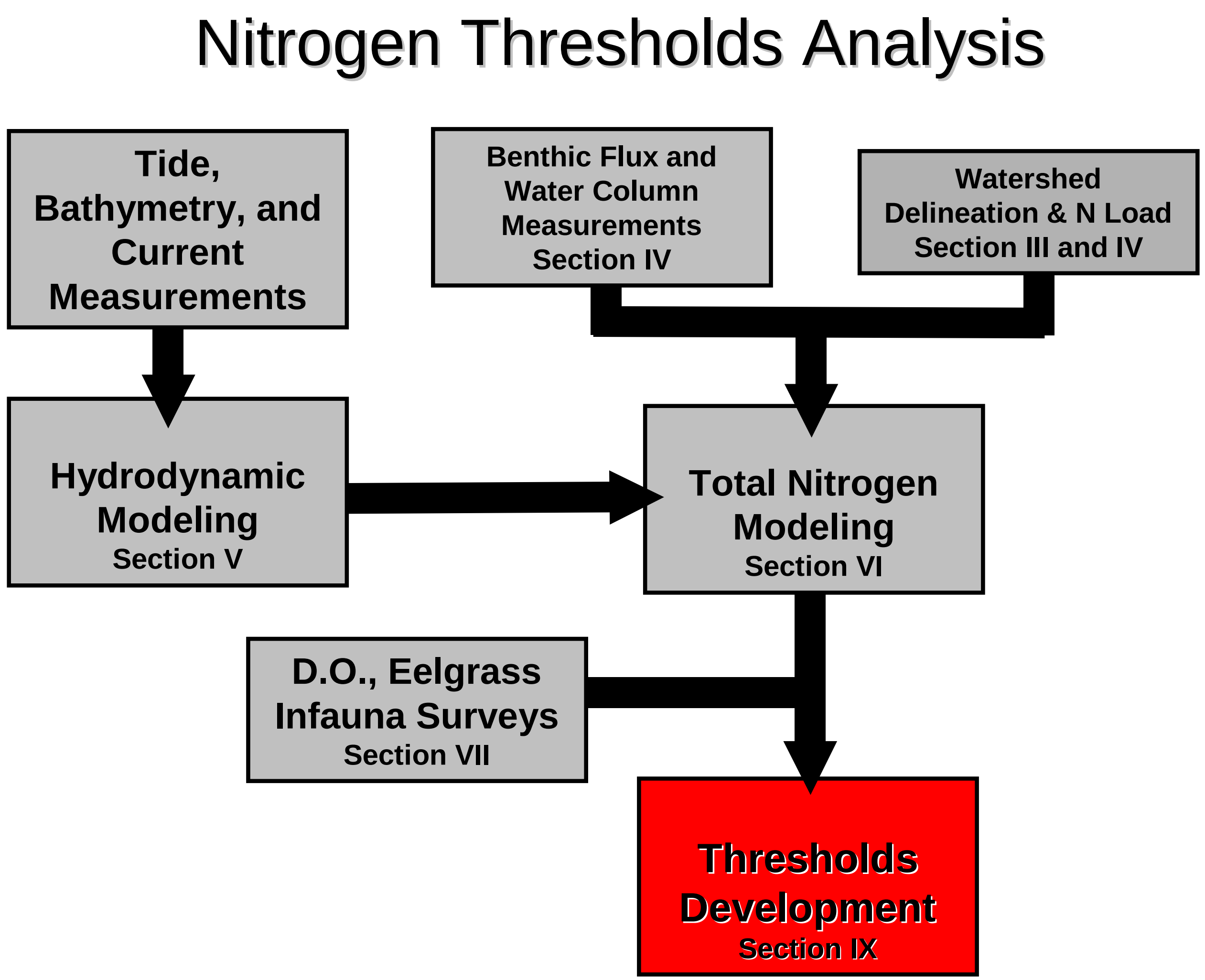


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, Wellfleet Harbor is ~10 ft).

Tidal damping (reduction in tidal amplitude) through an embayment can range from negligible, indicating “well-flushed” conditions, or show tidal attenuation caused by constricted channels and marsh plains, indicating a “restrictive” system, where tidal flow and the associated flushing are inhibited. Tidal data indicate only minimal tidal damping through the two inlets into the Three Bays system. It appears that both the tidal inlets are operating efficiently, possibly due to the active inlet maintenance program. Similarly, within the Three Bays System, the tide propagates to the sub-embayments with negligible attenuation, consistent with generally well-flushed conditions throughout.

History of Change

As management alternatives are being developed and evaluated, it is important to note that the Three Bays System is naturally a relatively dynamic system. Equally important is the recognition that it has been significantly altered by man’s activities over the past ~300 years and particularly over the past century.

Management of coastal systems requires not only an understanding of both present conditions, but also of the history of physical and environmental alteration. In addition, within degraded or partially degraded systems, an evaluation of the system’s “maximum level of sustainable environmental health” is also needed. It is clear that there has been significant alteration of the Three Bays hydrologic and biological systems over the past several centuries since the early days of the mills along Goodspeeds River. What follows is a brief description of the Three Bays system focusing on major upland or embayment alterations relating to present system health (Howes and Hampson 2000).

While the nutrient related health of the Three Bays System as it exists today is very much linked to changes wrought by human activities, it is the physical structure of the system laid down by the retreat of the Laurentide Ice Sheet that still controls much of the Bays’ tolerance to nutrient inputs. The physical structure, shape and depth of a coastal embayment plays a major role in its susceptibility to ecological impacts from nutrient loading. Physical structure (geomorphology), which includes embayment bathymetry, inlet configuration and saltwater reaches, when coupled with the tidal range of the adjacent open waters, determines the system’s rate of flushing. System flushing rate is generally the primary factor for removing nutrients from active cycling within coastal bays and harbors like the Three Bays system. As a result maximizing system flushing is one of the standard approaches for controlling the nutrient related health of coastal embayments.

As the Cape Cod Bay and Buzzards Bay Lobes of the Ice Sheet retreated, the sandy outwash plain that now holds the Three Bays watershed was formed. This sandy outwash produced the highly permeable soils found throughout upper Cape Cod. It is the permeability of the soils which has resulted in the importance of groundwater flow as a major pathway for nutrient transfers from sub-watersheds to adjacent coastal waters in this region. The presence of both groundwater and surface water pathways for input of nutrients into the present estuary has significant impact on its response to changing nutrient loadings with the surrounding watershed from changing land-uses.

As sea level rose and flooded the present basins of the Three Bays system, salt marshes began to form and an estuarine ecosystem began to function. At present it is not clear to what extent the basins of the embayment were formed from flooding kettle ponds versus merely flooding erosional valleys. However, given the shape and depth of the basins compared to other non-kettle systems on the southern shore of the upper Cape, it seems likely that kettle ponds with freshwater stream inflows and outflow were incorporated. With further sea-level rise the present marine beach deposits of Dead Neck, Sampson's Island, Bluff Point, northern tip of Grand Island and near the bridge to Little Island began to develop. The result was a complex estuarine system with a single inlet to Nantucket Sound through Cotuit Bay and major freshwater inputs through the Marstons Mills River and to a lesser extent, Little River.

Based upon studies from other regions of Cape Cod, it is likely that Native Americans utilized the resources of the Three Bays System for several thousand years before ceding the region to Captain Miles Standish in 1648. Native Americans likely used both the upland and estuarine resources. The marine food sources of the system would provide both shellfish (scallops, oysters and quahogs) and fish, particularly herring. According to James Otis, the name Mystic was the Native American term for small streams and ponds, particularly where herring and trout abounded. The largest lake within the watershed is still called Mystic Lake reputedly from this early term.

In 1653 when Roger Goodspeed, the first European to settle within the Three Bay watershed, settled by the Marstons Mills River (for awhile named Goodspeeds River), the Three Bays system was different from the present system in both its circulation and water quality. The upland was largely forested with some open lands, the Marstons Mills River was free flowing (no dams) and had more extensive freshwater marshes within its lower reaches and the embayment was connected to Nantucket Sound via a single inlet. While this single inlet almost certainly reduced the tidal exchange with high quality Nantucket Sound waters, the much lower terrestrial input of nutrients suggests a high quality estuarine system. However, it is also likely that, similar to today, within the region of the estuarine reach of the Marstons Mills River and associated salt marshes the sediments and bay waters were among the most nutrient and organic matter rich within the Three Bays System. However, the aquatic and upland components of this System began to change rapidly. By 1689 a fulling mill was constructed on the Marstons Mills River. In 1704-5 the dam was constructed thus altering the pathway of surface water transport and associated nutrients to the estuary. Town records indicating the leasing of herring rights and the requirement that all mills maintain fishways is testament to the magnitude of the herring population supported by this system. An active herring run within the Marstons Mills River continues to this day.

During the 1800's utilization of the estuary and its watershed continued to increase. Regions of the watershed were cleared for agricultural land and the Grist Mill at Marstons Mills continued operations past 1842. Within the watershed were changes to the freshwater systems which attenuate nitrogen during transport to bay waters. Most notable were the modification of riparian zones either through channelization, restriction, or filling of freshwater wetlands and, in some cases, transformation to cranberry agriculture. Most of the alterations reduced the nutrient buffering capacity of these systems, magnifying the nitrogen loading to the bay which greatly increased in the next century. Land clearing was accelerated by the development of salt works on the shores of the Bay which used fire to fuel evaporation for salt production. This activity peaked in 1812 and then declined.

Direct use of marine resources focused on oyster production, where oysters were initially pickled and shipped in barrels to market. In these earlier centuries, as today, oysters were cultured on the Bays' bottom. One of the first growers, Captain George Fisher who was granted a large section of Cotuit Bay shipped oysters to widespread U.S. markets. With the demand for oysters, the natural beds surrounding Grand Island became depleted and spat were imported from Long Island for grow out. However, at least for awhile, seed could be collected at the mouth of nearby Popponesset Bay on deployed scallop shells to supply the grow out needs. During this period, scallops were harvested within the Three Bays system in quantity and even at the turn of the century scallops represented a major economic resource. This record of substantial scallop harvest indicates that eelgrass beds were likely prevalent throughout the Bays. This suggests that the water was clearer (greater transparency due to less phytoplankton), hence less nutrient loading from the watershed was occurring. During this period the population was still small, for example there were only 36 homes in Santuit, Little River and Cotuit combined. Throughout the 1800's the residents relied heavily on coastal resources as salt making, oyster production, fishing, farming, ship-building and coastal trading.

By far the greatest changes to the Three Bays watershed and estuary have occurred during the last 100 years. The most obvious change has been the dramatic shift in land-use to residential housing during the last half of the 1900's. With this shift and the advent of fertilized lawns, has come a dramatic increase in the amount of nitrogen, which enters the Three Bays system. The previous large shifts in land-use, primarily from forest to agriculture did not have the same resultant increase in nitrogen loading, as the historic population was <10% of today. The present year-round population per square mile is greater than the entire town population of 50 years ago (total population based on 2000 census for Towns of Mashpee, Sandwich, and Barnstable are 12,946, 20,136 and 47,821 respectively). Unfortunately previous reductions in the capacity of the freshwater systems to attenuate nitrogen prior to its entry into the Three Bays system has accelerated the rate of nitrogen impairment as land-use changed. While this may be a partial cause of the present estuarine decline, it may also represent a potential opportunity for restoration of bay systems.

It is the recent increase in nitrogen load which is responsible for the observed declines in estuarine habitat quality throughout most of the Three Bays Estuary. In addition to this multi-decadal shift in watershed nitrogen loading, there were likely pulses of nutrients to the system during the 1940's associated with the military training areas within the Bays. The associated barracks, warehouses and storage tanks installed during WWII would result in a “new” source of nitrogen loading and the paving of the beach from Baxters Neck to Point Isabella (western shore of North Bay) may have also increased bacterial contamination in the adjacent waters.

In the early 1900's there was another major change to the Three Bays Estuarine System. Until this time, tidal exchange with Nantucket Sound was restricted to a single inlet to Cotuit Bay. However, a second inlet was opened which likely increased the flushing of West Bay, which previously had exchanged estuarine waters via the Seapuit River and through North Bay. Regardless of the extent to which this second inlet increased the flushing out of nutrient rich estuarine waters, it will have helped to buffer the Bays against the coming nutrient increases in the latter part of the century. Recent efforts to maintain the Bays for navigation may have also helped to maintain tidal exchanges, but the extent that this may have helped lessen the effects of increased watershed loadings has not been determined. Dredging of the Narrows from Prince Cove to North Bay in 1957 and the inlet to Cotuit Bay (most recently in the late 1990's) are two of the more notable examples of recent efforts.

Unfortunately while nutrient related decline in environmental health of the Three Bays System will be reduced by maximizing tidal exchange with the high quality waters of Nantucket Sound, the growing watershed nutrient loading and the structure of the system will require watershed management to restore the estuarine habitats within the Three Bays system to meet the high level of quality designated by the State Water Quality Standards. Watershed management will likely involve reduction of nitrogen inputs from various sources and possibly the removal of large loads (e.g. wastewater) from the watershed. Watershed management targeted at embayment restoration will usher in a new phase in the ever changing Three Bays System to the benefit of both present and future generations.

I.3 NUTRIENT LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Three Bays 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 Three Bays Estuary follow this general pattern, where the primary nutrient of eutrophication in these systems is nitrogen.

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

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

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

Unfortunately, almost all of the estuarine reaches within the Three Bays System are near or beyond their ability to assimilate additional nutrients without impacting their ecological health. Nitrogen levels are elevated throughout the Systems and eelgrass beds have not been observed within the Three Bays system for over a decade, although some plants were observed within the shallows of the upper estuary until 1995. Nitrogen related habitat impairment within the Three Bays Estuary shows a gradient of high to low moving from the inland reaches to the tidal inlet. The result is that nitrogen management of the primary sub-embayments to the Three Bays system is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed "eutrophication" and in certain instances can occur naturally over long periods of time. When the nutrient loading is rapid and primarily from human activities leading to changes in a coastal watershed, nutrient enrichment of coastal waters is termed "cultural eutrophication". Although the influence of human-induced changes has increased nitrogen loading to the systems and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within the Three Bays sub-embayments 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.4 WATER QUALITY MODELING

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

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all nitrogen entering the Three Bays System is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Three Bays system were taken from the Three Bays Water Quality Monitoring Program (a coordinated effort between the Town of Barnstable, Three Bays Preservation and the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the Systems (1999-2004) were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Three Bays System for the Town of Barnstable. 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 Nantucket Sound (Section IV). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section IV. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of the Bay in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined Bay threshold for restoration. 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 this system. Finally, analyses of the Three Bays System were undertaken relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging options to improve nitrogen related water quality. The results of the nitrogen modeling for each scenario have been 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 shellfisherman and to the sport-fishery and offshore finfishery, which are dependant upon these highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. This process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and pond, it is not a necessarily a part of the natural evolution of a system.

In most marine and estuarine systems, such as the Three Bays system, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the approach generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2002).

These tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including the Three Bays System. As the MEP approach requires substantial amounts of site specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality or unique features.

Numerous studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Three Bays System over the past 10 years. In the late 1980’s and early 1990’s local concern over the health of the sub-embayments to the Three Bays system, particularly in the upper reaches, focused upon closures of shellfish beds. Field measurements by the Town of Barnstable in the mid/late 1990’s indicted that the greater issue of habitat degradation from nitrogen enrichment was occurring particularly in the region of Prince Cove. This concern about nutrient related habitat declines resulted in a nitrogen loading and flushing analysis by the Cape Cod Commission under the Cape Cod Coastal Embayment Project (Eichner et al. 1998). In that study the major sub-watersheds to the Three Bays system were delineated based upon available water table measurements, a land-use nitrogen loading model was implemented to determine nitrogen inputs to bay waters and a box model was used to evaluate flushing rates of the estuary's sub-basins. The box model incorporated both August and October salinity data and the results of a dye study in Prince Cove. Although the approach yields only approximations, the results clearly supported the concept that nitrogen inputs to the Three Bays system were impacting the water quality system-wide, with greatest degradation in the upper regions of Prince Cove and Warrens Cove. Further, the landuse analysis indicated that on-site septic disposal of wastewater was the major single source of watershed nitrogen to bay waters. While the overall results of this study have held true, the box model is insufficient to simulate changes in nitrogen within the estuary under different management alternatives. In addition, as quantitative surveys of embayment nitrogen levels were not yet available, the model could not be validated. In addition, as the landuse model did not account for nitrogen attenuation by the surface freshwater ecosystems within the watershed (no data available), it over estimated the role of nitrogen sources in upper (inland most) sub-watersheds compared to the direct groundwater watersheds to the estuary. While base data from this earlier study was incorporated by the MEP, direct use of the modeling results was problematic. Since the landuse model was based upon the 1996 watershed delineations from well data, rather than the MEP's USGS West Cape Model (see Chapter III), the contributing areas are slightly different. Due to the difference in watershed areas and the MEP's update and refinements to the watershed nitrogen loading model (e.g. to incorporate attenuation and new nitrogen source information), the results from the MEP are different and supercede those of this earlier study.

At this same time (1996) a group of private citizens formed a non-profit NGO, Three Bays Preservation, to address environmental problems within the Three Bays System. Three Bays Preservation initially focused upon dredging the tidal inlet between Cotuit Bay and Nantucket Sound, with a small amount of dredging in West Bay. As part of this process a 2-D hydrodynamic model was used to evaluate the effects of dredging on water circulation and flushing of the Three Bays system. The model is similar to that used by MEP, but it appears that the model did not calibrate and therefore the effort was revisited by MEP. In addition, as the earlier model focused on the effects of dredging on circulation, the model needed to be redeveloped with appropriate grid spacing to support the MEP water quality modeling effort. After permitting, the dredging effort was carried out with the Town of Barnstable in the winters of 1998-99 (187,000 cu yd), 1999-00 (24,700 cu yd), 2001 (15,000 cu yd) and 2002 (9,000 cu yd). The dredged sand was used to nourish Dead Neck for barrier beach stabilization.

Despite these dredging activities, it was clear that inlet maintenance was not sufficient to redress the nitrogen related declines observed throughout large portions of the Three Bays System. In 1999 Three Bays Preservation established a nitrogen related water quality monitoring program throughout the Three Bays system to support restoration efforts. The Three Bays Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD and over the past several years the program has been incorporated into Barnstable's Town-wide embayment monitoring program. To date water column sampling has been conducted throughout the system on 6-8 dates per summer, June-August. Initial results of the Three Bays Water Quality Monitoring Program (Howes and Hampson 2000) indicated that:

(a) "some areas within the Estuary are presently showing nutrient related water quality declines and there is a wide variation in habitat quality within the Three Bays System. In general the quality of habitat in the Three Bays system shifts from high quality near the inlets to Nantucket Sound to poor quality (eutrophic) in the inner reaches. Prince's Cove, and the region of the mouth of the Marstons Mill River through the narrows to North Bay are showing poor nutrient related environmental health. However, the loss of eel grass beds from most of the Three Bays system proper indicates that the system has undergone nutrient loading related shifts";

(b) "the effects of watershed nutrient inputs can be seen in diminished water transparency, increased chlorophyll and nitrogen levels in the upper system and North Bay.

(c) "the Three Bays Estuary appears to be nitrogen limited, i.e. additions of nitrogen will increase algal production";

(d) "the organic matter within the Three Bays Estuary appears to be produced by phytoplankton supported by inputs of watershed nitrogen and recycled nitrogen within the Bays, as opposed to entering the system in surface water flows";

This effort provides the quantitative watercolumn nitrogen data (1999-2004) required for the implementation of the MEP's Linked Watershed-Embayment Approach used in the present study. In addition, there have been a variety of smaller studies relating to bacterial contamination (see MEP Bacterial Technical Report for Princes Cove 2005) and macroalgal management conducted by Three Bays Preservation.

The MEP has incorporated all appropriate data from all previous studies to enhance the determination of nitrogen thresholds for the Three Bays System and to reduce costs to the Town of Barnstable.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). 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 to the USGS to organize and analyze the available data utilize up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation, surface water/groundwater interaction, groundwater travel time, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Three Bays embayment system located in the Town of Barnstable, Massachusetts.

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 Three Bays system under evaluation by the Project Team. The Three Bays estuarine system is a complex estuary, with two tidal inlets, large central islands, and a significant intrusion into the Sagamore groundwater lens. Further watershed modeling was undertaken to sub-divide the overall watershed to the Three Bays 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 generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the Three Bays watershed. Model assumptions for calibration were matched to surface water inputs and flows from current (2002 to 2003) stream gage information.

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by the 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 the stream and the portion of the groundwater system that discharges directly into the estuary as groundwater seepage.

III.2 MODEL DESCRIPTION

Contributing areas to the Three Bays system and local freshwater bodies were delineated using a regional model of the Sagamore Lens (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 main basins of the Three Bays embayment system and also to determine portions of recharged water that may flow through freshwater ponds and streams prior to discharging into the tidal waters of the Three Bays system.

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. Layers 1-7 are stacked above NGVD 29 and layers 8 to 20 extend below. Layer 18 has a thickness of 40 feet and layer 19 extends to 240 feet below sea level. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens. Since the Three Bays watershed extends to the top portion of the Sagamore Lens, most of the uppermost layers of the groundwater model are active in its delineation

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 sediments generally show a fining downward with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Three Bays watershed is situated in the midst of the very-coarse grained Mashpee Pitted Plain deposits (Masterson, et al., 1996). Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water level and streamflow data collected in May 2002.

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

III.3 THREE BAYS CONTRIBUTORY AREAS

Newly revised watershed and sub-watershed boundaries to the Three Bays embayment system were determined by the United States Geological Survey (USGS) (Figure III-1). Model outputs of MEP watershed boundaries were “smoothed” to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, and (c) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. Overall, 59 sub-watershed areas, including thirteen freshwater ponds and six public water supply wellfields, were delineated within the watershed to the Three Bay embayment system.

Table III-1 provides the daily freshwater discharge volumes for each of the subwatersheds as calculated by the groundwater model and these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models in order to determine hydrologic turnover in the lakes/ponds, as well as for comparison to measured surface water discharges. The MEP delineation includes 10 yr time of travel boundaries. The overall estimated freshwater inflow to the estuarine waters of the Three Bays system from the MEP watershed is 95,608 m3/d.

The delineations completed by the MEP are the second watershed delineation completed in recent years for the Three Bays estuary. Figure III-2 compares the delineation completed under the current effort with the delineation completed by the Cape Cod Commission in 1995 as part of the Cape Cod Embayment Project (Eichner, et al., 1998). The delineation completed in 1995 was defined based on regional water table measurements collected from available wells over a number of years and normalized to average conditions; delineations based on this previous effort were incorporated into the Commission's regulations through the Regional Policy Plan (CCC, 1996 & 2001).

Overall, the MEP contributing area to the Three Bays system based upon the groundwater modeling effort is similar to the previous delineation based upon available well data, the MEP area being only 2.5% or 324 acres smaller. However, the specific land areas enclosed by the 2 delineation efforts are slightly different. The primary spatial difference is a slight thinning of the portion of the watershed close to the Cape Cod Bay/Vineyard Sound groundwater divide, a slightly more northern location for the top of the Sagamore Lens, and a more eastern location for the western boundary of the system watershed. The change in the western boundary is largely due to a better consideration of Santuit Pond and its relationship to Popponesset Bay than was possible in the earlier effort (see Figure III-2). There was insufficient data to support the development of the subwatersheds to the individual great fresh ponds in the earlier delineation effort.

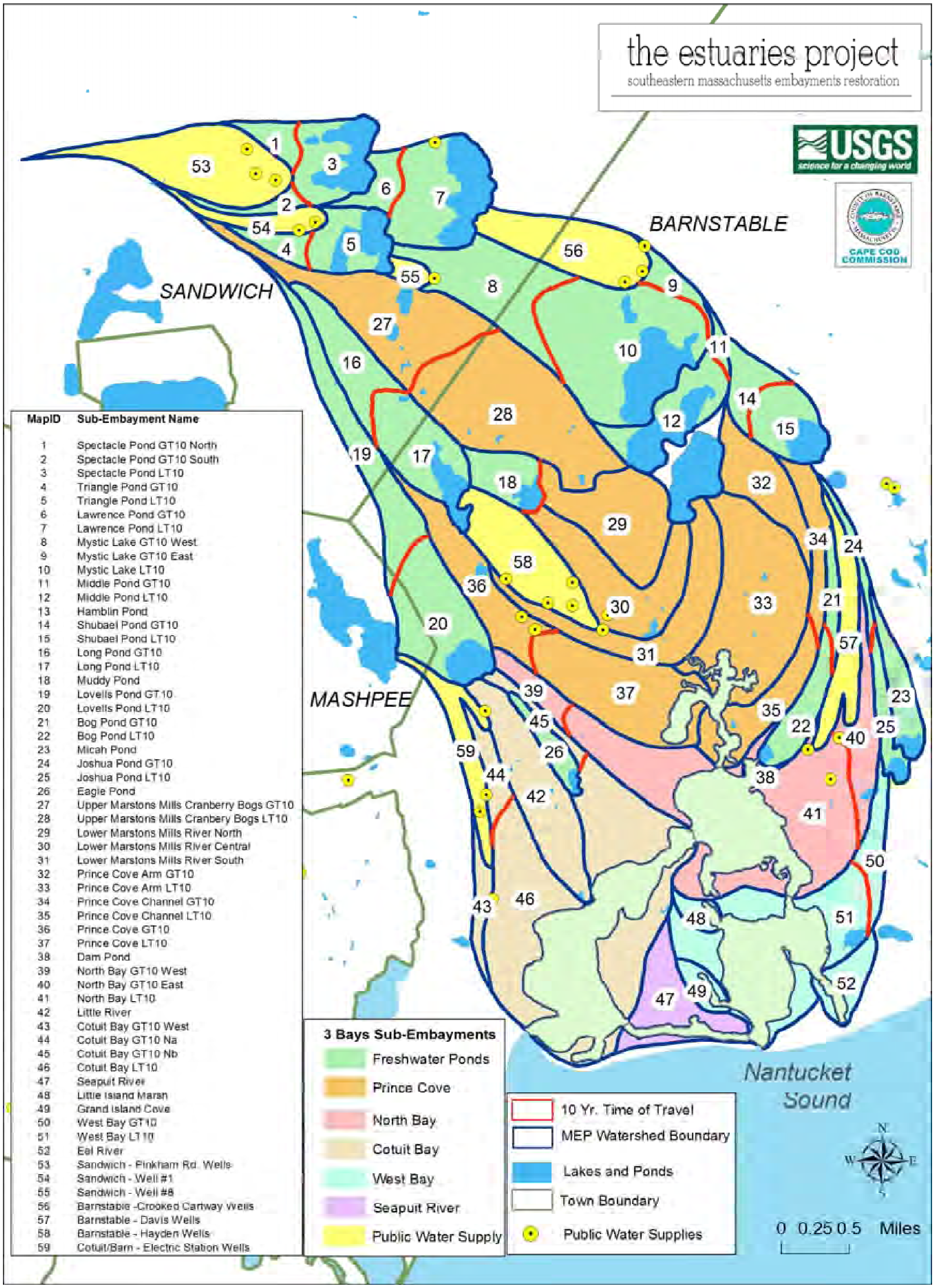


Figure III-1. Watershed and sub-watershed delineations for the Three Bays estuary system. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI).

Table III-1. Daily groundwater discharge to each of the sub-embayments in the Three Bays system, as determined from the USGS groundwater model.

Watershed	Watershed #	Discharge	
		m^3/day	ft^3/day
Spectacle Pond GT10N	1	463	16357
Spectacle Pond GT10S	2	329	11632
Spectacle Pond LT10	3	1572	55508
Triangle Pond GT10	4	423	14924
Triangle Pond LT10	5	1305	46087
Lawrence Pond GT10	6	601	21234
Lawrence Pond LT10	7	2491	87948
Mystic Lake GT10W	8	2540	89690
Mystic Lake GT10E	9	469	16545
Mystic Lake LT10	10	4795	169309
Middle Pond GT10	11	190	6726
Middle Pond LT10	12	1722	60794
Hamblin Pond	13	1235	43606
Shubael Pond GT10	14	699	24699
Shubael Pond LT10	15	1275	45025
Long Pond GT10	16	1378	48651
Long Pond LT10	17	1992	70320
Muddy Pond	18	961	33930
Lovells Pond GT10	19	2065	72910
Lovells Pond LT10	20	2506	88485
Bog Pond GT10	21	387	13671
Bog Pond LT10	22	965	34065
Micah Pond	23	897	31682
Joshua Pond GT10	24	345	12171
Joshua Pond LT10	25	613	21657
Eagle Pond	26	528	18632
Upper MM River GT10	27	3314	117010
Upper MM River LT10	28	4735	167195
Lower MM River N	29	2403	84843
Lower MM River Mid	30	2415	85272
Lower MM River S	31	2258	79727
Prince Cove Arm GT10	32	1496	52834
Prince Cove Arm LT10	33	4236	149562
Prince Cove Channel GT10	34	919	32441
Prince Cove Channel LT10	35	1395	49265
Prince Cove GT10	36	1869	65989
Prince Cove LT10	37	2926	103319
Dam Pond	38	183	6466
North Bay GT10 W	39	548	19347
North Bay GT10 E	40	1205	42562
North Bay LT10	41	5332	188282

Table III-1 (continued). Daily groundwater discharge to each of the sub-embayments in the Three Bay system, as determined from the USGS groundwater model.			
Watershed	Watershed #	Discharge	
		m³/day	ft³/day
Little River	42	2487	87805
Cotuit Bay GT10W	43	495	17485
Cotuit Bay GT10Na	44	550	19438
Cotuit Bay GT10Nb	45	354	12488
Cotuit Bay LT10	46	6720	237270
Seapuit River	47	1593	56233
Little Island Marsh	48	290	10232
Grand Island Cove	49	369	13037
West Bay GT10	50	738	26048
West Bay LT10	51	2679	94601
Eel River	52	675	23843
Sandwich - Pinkham Rd. Wells	53	2879	101651
Sandwich - Well #1	54	670	23657
Sandwich - Well #8	55	238	8418
Barnstable - Crooked Cartway Wells	56	2156	76125
Barnstable - Davis Wells	57	1383	48831
Barnstable - Hayden Wells	58	2225	78565
Cotuit/Barnstable - Electric Station Wells	59	1128	39818
Note: Discharge rates are based on 27.25 inches per year of recharge (Walter and Whealan, 2005)			

The evolution of the watershed delineations for the Three Bays 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 brought into congruence with data from adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading as errors in loading depend upon the land-uses that are included/excluded within 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.

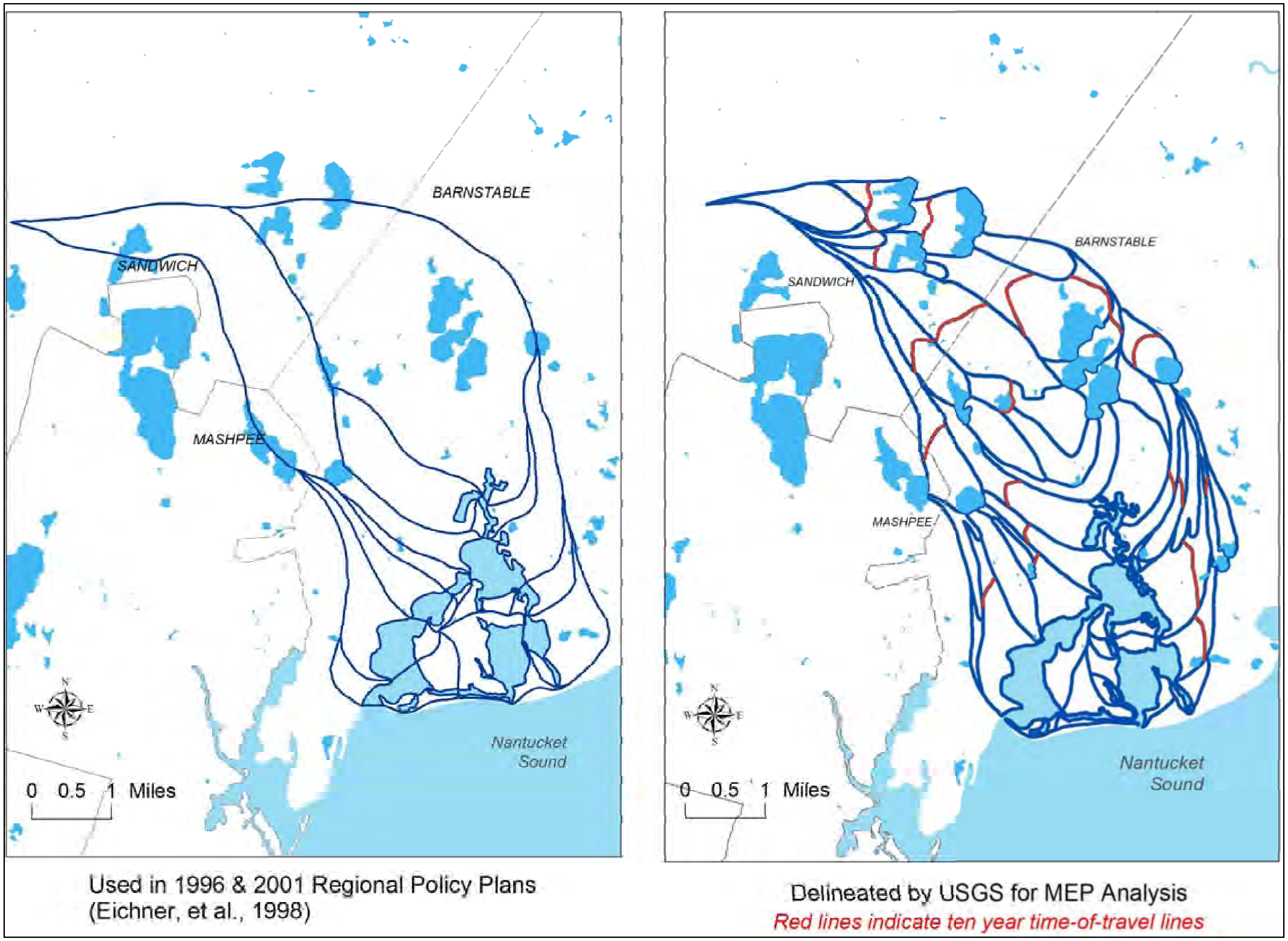


Figure III-2. Comparison of previous CCC (left) and MEP (right) Three Bay watershed and subwatershed delineations. The MEP watershed area is only 2.5% or 324 acres smaller than the 1996 delineation, but encloses slightly different land areas. The difference is primarily in region close to the Cape Cod Bay/Vineyard Sound groundwater divide and stems from a better consideration of Santuit Pond and its relationship to Popponesset Bay than was possible in the earlier effort.

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 Three Bays system. Determination of watershed nitrogen inputs to these embayment systems requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or aquifer, (b) confirmation that a groundwater transported load has reached the embayment at the time of analysis, and (c) quantification of nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. This latter natural attenuation process results from biological processes that naturally occur within ecosystems. Failure to account for attenuation of nitrogen during transport results in an over-estimate of nitrogen inputs to an estuary and an underestimate of the sensitivity of a system to new inputs (or removals). In addition to the nitrogen transport from land to sea, the amount of direct atmospheric deposition on each embayment surface must be determined as well as the amount of nitrogen recycling within the embayment, specifically nitrogen regeneration from sediments. Sediment nitrogen recycling results primarily from the settling and decay of phytoplankton and macroalgae (and eelgrass when present). During decay, organic nitrogen is transformed to inorganic forms, which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if it is not included in determination of summertime nitrogen load.

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) to the Three Bays embayment system (Section III). The Three Bays watershed was sub-divided to define contributing areas for each of the major inland freshwater systems and each major sub-embayment to the Three Bays system. The overall watershed to the system was further sub-divided into regions greater and less than 10 year groundwater travel time from the receiving estuary yielding a total of 59 sub-watersheds in all. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds for Three Bays and to each sub-embayment.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes and the time of groundwater travel provided by the USGS watershed model. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel (LT10) and greater than 10 year time of travel (GT10) watersheds (Table IV-1), previous nitrogen loading assessments (Eichner, et al., 1998), land use development records, and water quality modeling, it was determined that Three Bays is currently in balance with its watershed load. At present, the bulk (75%) of the watershed nitrogen load is less than 10 years travel time to the estuarine waters of Three Bays. Therefore, the distinction of less than 10 year and greater than 10 year time of travel regions within a subwatershed (Figure III-1) was eliminated and the number of subwatersheds was reduced to 37 (Figure IV-1). Potential errors resulting from long

Table IV-1. Percentage of unattenuated nitrogen loads in less than 10 time of travel subwatersheds to Three Bays. Note that not all of the "outflow" from these ponds is to the Three Bays Estuary. The load to the estuary is presented below in Table IV-5.

WATERSHED		LT10	GT10	TOTAL	%LT10
Name	#	kg/yr	kg/yr	kg/yr	
Spectacle Pond	1	662	395	1056	63%
Triangle Pond	2	629	315	944	67%
Lawrence Pond	3	727	26	753	96%
Mystic Lake	4	3429	2460	5890	58%
Middle Pond	5	857	67	924	93%
Hamblin Pond	6	693	0	693	100%
Shubael Pond	7	1520	711	2231	68%
Long Pond	8	2270	1396	3666	62%
Muddy Pond	9	525	0	525	100%
Lovells Pond	10	1734	1752	3486	50%
Bog Pond	11	1097	294	1391	79%
Micah Pond	12	869	0	869	100%
Joshua Pond	13	763	411	1174	65%
Eagle Pond	14	567	0	567	100%
Upper Marstons Mills River	15	4583	2819	7402	62%
Lower MM River N	16	2162	0	2162	100%
Lower MM River Mid	17	1929	0	1929	100%
Lower MM River S	18	1157	0	1157	100%
Prince Cove Arm	19	2521	1490	4011	63%
Prince Cove Channel	20	1327	506	1833	72%
Prince Cove	21	3204	1033	4236	76%
Dam Pond	22	118	0	118	100%
North Bay	23	5821	2404	8225	71%
Little River	24	1617	0	1617	100%
Cotuit Bay	25	4796	1304	6100	79%
Seapuit River	26	1375	0	1375	100%
Little Island Marsh	27	338	0	338	100%
Grand Island Cove	28	204	0	204	100%
West Bay	29	3619	1719	5338	68%
Eel River	30	915	0	915	100%
Sandwich - Pinkham Rd. Wells	31	1500	0	1500	100%
Sandwich - Well #1	32	590	0	590	100%
Sandwich - Well #8	33	19	0	19	100%
Barnstable - Crooked Cartway Wells	34	1012	0	1012	100%
Barnstable - Davis Wells	35	1479	0	1479	100%
Barnstable - Hayden Wells	36	1231	0	1231	100%
Cotuit/Barnstable - Electric Station Wells	37	711	0	711	100%
TOTAL		58569	19102	77671	75%

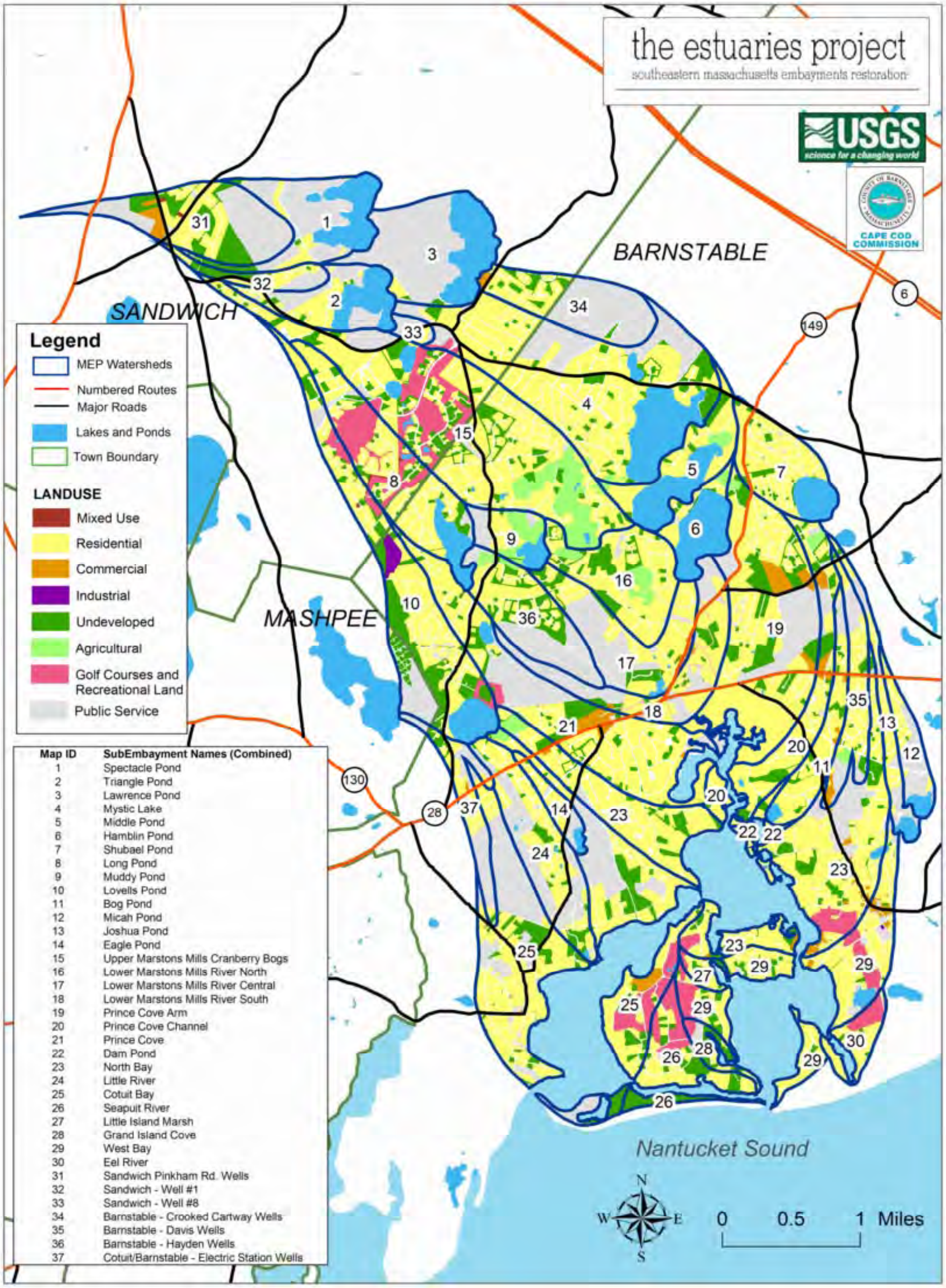


Figure IV-1. Land-use coverage in the watershed to the Three Bays Embayment System. The watershed encompasses portions of the Towns of Barnstable, Sandwich, and Mashpee and land use classifications are based on assessors' records provided by each of the towns.

travel times appear to be relatively small in the Three Bays System, since development in the watershed nearest the estuary (lower Marstons Mill River and groundwatersheds to the bays) indicate that 82% of the nitrogen sources are within 10 years travel times. Longer travel times in the watersheds to the fresh ponds does not create a proportional error, as these sources are generally significantly attenuated (i.e. nitrogen is lost) during transport. In addition, the watersheds nearest the embayment have been relatively stable over recent years. The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries (after accounting for natural attenuation, see below).

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes & Ramsey, 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land-uses and pre-determined nitrogen loading rates. For the Three Bays embayment system, the model used Barnstable, Sandwich, and Mashpee-specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local watershed-specific data (such as parcel by parcel water use). Note that the small portion of the watershed in Mashpee (<1%) is predominantly open space adjacent Santuit Pond. 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” nitrogen load to each receiving embayment, since attenuation during transport has not yet been included.

Natural attenuation within the Three Bays watershed of nitrogen during transport from land-to-sea (Section IV.2) was determined based upon site-specific studies within the freshwater portion of the Marstons Mills River and Little River and through the freshwater ponds within the watershed. Attenuation during transport through each of the major fresh ponds was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Attenuation during transport through each of the major fresh ponds was assumed to equal 50% based on available monitoring of selected Cape Cod lakes. Available historic data collected from individual fresh ponds in the Three Bays watershed confirmed the appropriateness of this general assumption. Attenuation factors based upon site-specific studies were developed to three ponds (Mystic, Middle, and Hamblin) based on an extensive dataset collected during 2004 (Eichner, et al., in preparation). Internal nitrogen recycling was also determined throughout the tidal reaches of the Three Bays embayment; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Estuaries Project staff obtained digital parcel and tax assessors data from the Towns of Barnstable, Sandwich, and Mashpee. Digital parcels and land use data are from 2004 for Barnstable, 2000 for Sandwich, and 2001 for Mashpee and were obtained from the Town of Barnstable GIS Unit, the Town of Sandwich Planning Department, and the Town of Mashpee Planning Department, respectively. These land use databases contain traditional information regarding land use classifications (MADOR, 2002) plus additional information developed by each of the towns; different information is available depending on the town (e.g., Mashpee has developed information about impervious surfaces (building area, driveways, and parking area) on individual lots). The parcel coverages and assessors' database were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows the land uses within the Three Bays study area. Land use in the Three Bays study area can be apportioned into one of eight land use types: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) agricultural, 6) mixed use, 7) golf course and recreational land, and 8) public service/government, including road rights-of-way. “Public service” is the land classification assigned by the Massachusetts Department of Revenue to tax exempt properties, including lands owned by government (e.g., wellfields, schools, open space, roads) and private groups like churches and colleges. Massachusetts Assessors land uses classifications (MADOR, 2002), which are common to each town in the watershed, are aggregated into these eight land use categories.

In the Three Bays watershed, the predominant land use based on area is residential, which accounts for slightly less than half (46%) of the watershed area; public service (government owned lands including open space, roads, and rights-of-way) is the second highest percentage of the watershed (28%). In addition, 76% of the parcels in the system watershed are classified as single family residences (MADOR land use code 101) and single family residences account for 88% of the residential land area. In addition, residential land uses are the predominant land use in all the major Three Bays subwatersheds with a range of 39% to 63% of the subwatershed areas. Public service land uses are the second highest percentage in all of the major subwatersheds except for the Seapuit River subwatershed where undeveloped land uses are the second highest. Overall, undeveloped land uses account for 11% of the whole Three Bays watershed and are shown in Figure IV-2. Commercial properties account for only about 1% of the Three Bays watershed area.

In order to estimate wastewater flows within the Three Bays watershed, MEP staff also obtained parcel by parcel water use information from the Cotuit Water District and Centerville, Osterville, Marstons Mills (COMM) Water District from the Town of Barnstable GIS Unit, the Sandwich Water District, and the Mashpee Water District. The COMM water use data is three years (2001-2003) of biannual readings, while the Cotuit water use data is only for one year (October 2002 through October 2003). Sandwich water use data is two and a half years (January 2002 through June 2004), while Mashpee water use data is three years (1997 through 1999). Water use information was linked to the parcel and assessors data using GIS techniques. Water use for each parcel was converted to an annual volume for purposes of the nitrogen loading calculations. Two wastewater treatment facilities (WWTFs) currently exist in the Three Bays watershed: one at the Cotuit Landing shopping area (21,600 gallon per day design capacity) in the Prince Cove subwatershed and another at the Barnstable Horace Mann Charter School (32,000 gallon per day design capacity) in the Joshua Pond subwatershed.

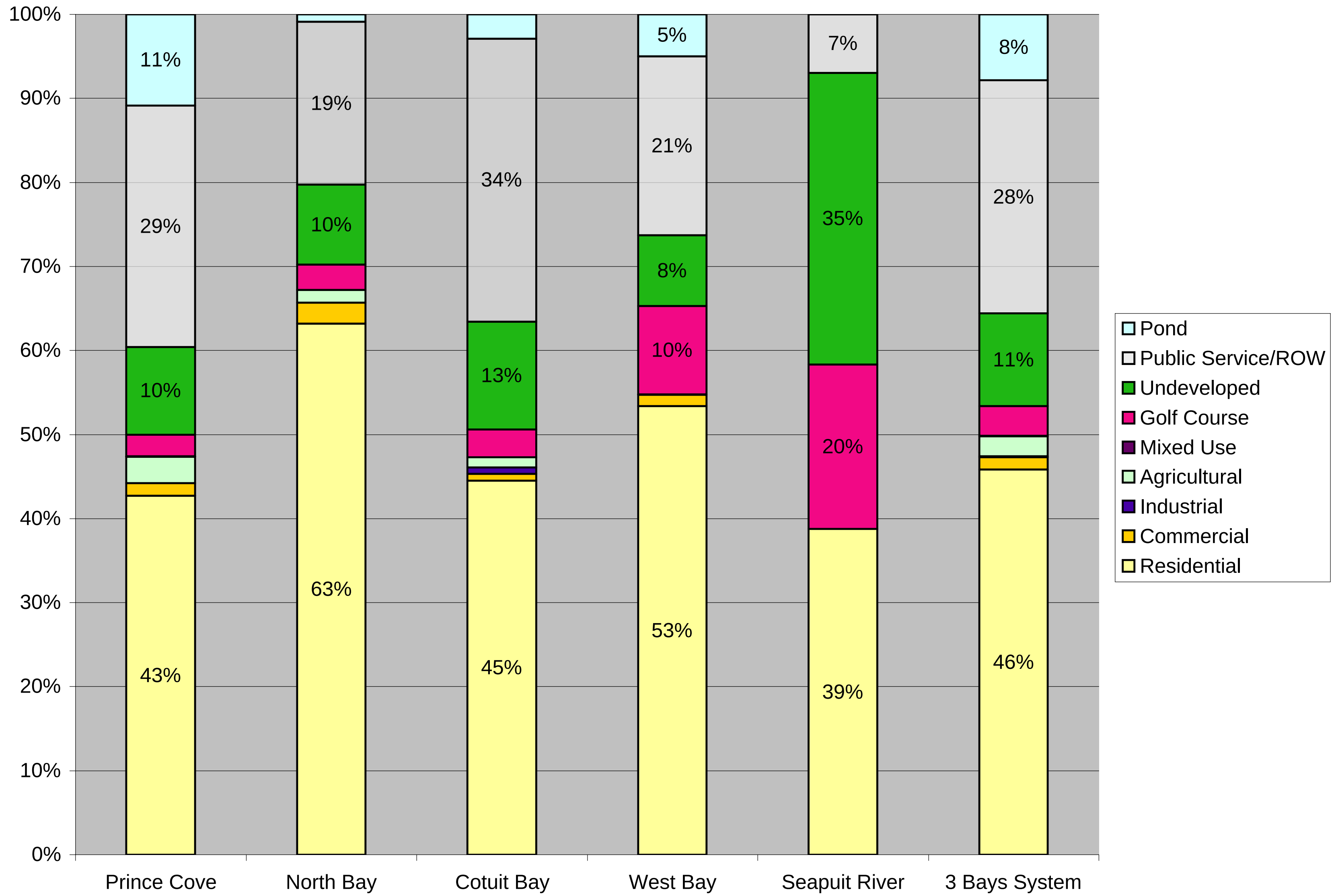


Figure IV-2. Distribution of land-uses within the major subwatersheds and whole watershed to Three Bays. Only percentages greater than or equal to 5% are shown.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

Except for the small WWTFs, all wastewater within the Three Bays watershed is returned to the aquifer through individual on-site septic systems. Wastewater-based nitrogen loading from the individual parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system or WWTF.

Similar to many other watershed nitrogen loading analyses, 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 directly measured septic system and per capita loads determined 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 in many of the watersheds throughout southeastern Massachusetts, census data yields accurate estimates of total population only in specific watersheds (see below). To correct for this uncertainty, the MEP employs a water-use approach. The water-use approach (Weiskel and Howes 1991) is applied on a parcel-by-parcel basis within a watershed, where usually an average of multiple years annual water meter data is linked to assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down-gradient in the aquifer. All losses within the septic system are incorporated. For example, information developed at the DEP 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). Aquifer studies indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

In its application of the water-use approach to septic system nitrogen loads, the MEP has ascertained for the Estuaries Project region that while the per capita septic load is well constrained by direct studies, the consumptive use and nitrogen concentration data are less certain. As a result, the MEP has derived a combined term the effective N Loading Coefficient (consumptive use times N concentration) of 23.63, to convert water (per cubic meter) to nitrogen load (N grams). This term 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, etc.).

The resulting nitrogen loads, based upon the above 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. For example, 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). The selected “effective N loading coefficient” also agrees with available watershed nitrogen loading analyses conducted on other Cape Cod estuaries. Aside from the concurrence observed between modeled and observed nitrogen concentrations in the estuary analyses completed under the MEP, analyses of other estuaries completed using this effective septic system nitrogen loading coefficient, the modeled loads also match observed concentrations in streams in the MEP region. Modeled and measured nitrogen loads were determined for a small sub-watershed to West Falmouth Harbor (Smith and Howes 2006, in review) where a small stream drained the aquifer from a residential neighborhood. In this effort, the measured nitrogen discharge from the aquifer was within 5% of the modeled N load. A second 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 measured and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year and under the ecological situation (Samimy and Howes unpublished data).

While census based population data has limitations in the highly seasonal MEP region, part of the regular MEP analysis is to compare expected water used based on average residential occupancy to measured average water uses. This is performed as a quality assurance check to increase certainty in the final results. This comparison has shown that the larger the watershed the better the match between average water use and occupancy. For example, in the cases of the combined Great Pond, Green Pond and Bournes Pond watershed in the Town of Falmouth and the Popponesset Bay/Eastern Waquoit Bay watershed, which cover large areas and have significant year-round populations, the septic nitrogen loading based upon the census data is within 5% of that from the water use approach. This comparison matches some of the variability seen in census data itself, census blocks, which are generally smaller areas of the towns 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 outwash aquifers; (b) has been validated in studies of the MEP Watershed “Module”, where there is been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP coefficients for stormwater, lawn fertilization, etc; (c) the MEP septic nitrogen loading coefficient agrees in specific studies of consumptive water use and N 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 worked out 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, it is also conservative in watersheds dominated by residential land-uses. Sensitivity analysis by MEP Technical Team showed that higher septic nitrogen loading factors (up to 33% larger), resulted in only slight changes in the required nitrogen removal (estimated at 1% to 5% lower)), to lower embayment nitrogen levels to a nitrogen target (e.g. nitrogen threshold, cf. Section VIII.

For the purposes of the Three Bays nitrogen loading modeling, performance data for the two WWTFs in the watershed were obtained from a review of WWTF effluent monitoring data associated with the DEP permit. Measurement and reporting of average monthly flows and effluent discharge N concentrations are required of both the Barnstable Horace Mann Charter School and Cotuit Landing WWTF. Averages of effluent flow and total nitrogen concentrations were incorporated into the model based on monitoring data collected between 2002 and 2004 (Table IV-2).

Table IV-2. Wastewater Treatment Facilities in the Three Bays Watershed

System Name	Average Effluent Characteristics		
Facility Name	Flow (gallons per day)	Total Nitrogen Concentration (mg/liter)	Annual Nitrogen Load (kg N/yr)
Barnstable Horace Mann Charter School	3,907[a]	4.64[b]	25
Cotuit Landing	5,206 [c]	4.69 [d]	34

Notes: [a] average flow (April 2002-September 2004); [b] average discharge concentration (April 2002-September 2004); [c] average flow (September 2002-December 2004); [d] average discharge concentration (September 2002-December 2004); all data from DEP records (personal communication, B. Dudley SERO)

Water use information exists for 85% of the 9,153 parcels in the Three Bays watershed. Only 1,411 parcels appear to utilize private wells for drinking water. These are properties that are classified with land use codes that should be developed (e.g., 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have a listed account in the water use databases. Of the 1,411 parcels, almost all, (92%, 1,295) are classified as single family residences (land use code 101), 12 are classified as commercial land uses (codes between 300 to 389), one is a plant nursery, and the remainder is other residential land uses (i.e., multiple houses on one lot, two family residences, condominiums, etc).

Water use and wastewater flows from the parcels on private wells were extrapolated from the other parcels in the watershed. To develop nitrogen loads from these developed parcels in the nitrogen loading modeling. MEP staff developed water use estimates based on average water uses of similar land uses in the Three Bays watershed (Table IV-3). For the residential land uses, the average water use of single family residences (227 gallons per day) was assigned to all residential parcels assumed to have private wells, as well as all new residences estimated through the buildout analysis. Estimates for commercial properties were treated differently. For most of the commercial properties, the area of the buildings occupying a given site and the percentage of the property occupied by the buildings are available. This information was combined with water use information to develop a flow per 1,000 square feet of building for existing properties (18 gpd/ 1,000 ft2) and an average building coverage (28% of the lot) for use in buildout projections. It is recognized that a variety of commercial land uses and a wide range of water use (e.g., small offices with one or two employees to large water users, like restaurants) are contained in the average, but the developed average has been selected as the most appropriate estimate for these land uses. Commercial and Industrial parcels are relatively minor nitrogen contributors to the Three Bays watershed, accounting for <3% of the total water use. The ranges in Table IV-3 are very similar to those observed in the MEP analysis of water use in other watersheds.

Table IV-3. Average Water Use in Three Bays Watershed based upon multi-year water meter data.

Land Use	State Class Codes	# of Parcels in Study Area	Water Use (gallons per day)	
			Study Area Average	Study Area Range
Residential	101	5,668	227	0 to 3,203
Commercial	300 to 389	103	379	0 to 2,910
Industrial	400 to 439	2	49	--

Note: All data for analysis supplied by towns or water districts. Only one industrial property has water use within the study area

In order to provide an independent validation of the residential water use average within the study areas, MEP staff reviewed US Census population values. In non-seasonal watersheds, estimates of census data can provide a good estimate of total population for comparison to total population estimates derived from water meter data. 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 each person generates 55 gpd of wastewater. Average occupancy within the Town of Barnstable during the 2000 US Census was 2.44 people per household, 2.46 in Mashpee, and 2.30 in Sandwich; using a weighted average based on the percent area of each town within the watershed, the Three Bays watershed 2000 Census occupancy is 2.41. If 2.41 were multiplied by 55 gpd, 132 gpd would be calculated as the average residential wastewater flow in the Three Bays watershed. However, the measured average total residential water use in the Three Bays watershed is 227 gpd, which indicates either a higher per capita water use, a higher per capita consumptive use (water used that does not go to septic disposal), or a higher annualized population (occupancy) than indicated by the census. It is almost certain that the latter explanation is working in this watershed, namely that the census data is underestimating (by ~1/3) the extent of population, which resides annually or seasonally in this highly seasonal watershed (see below).

In most previously completed MEP studies, average population and average water use have generally agreed fairly well, although the Oyster Pond analysis also did not show good agreement. As a result of the relatively poor agreement between census and water use population estimates in the Three Bays watershed, MEP staff reviewed more refined US Census information and water use information for each parcel within the watershed. Besides reviewing data on town and state levels, the US Census also develops information for smaller areas (i.e., tracts and blocks). Average occupancy in the Census tracts within the Three Bays watershed range between 2.18 and 3.05, which would produce a wastewater generation rate of 120 to 168 gpd per residence, if 55 gpd is assumed for each person. The top end of this range is still lower than the measured water use in the Three Bays watershed, but this range does suggest a higher potential population in the watershed than a review of town-wide data and supports the need for the MEP parcel-by-parcel water use approach, rather than use of census data alone. The overall review of Census data suggests that the Three Bay watershed is an area where water use should be increasing.

MEP staff then reviewed the average water uses measured in the subwatersheds of the Three Bay system. While the overall average for single family residences (SFRs) is 227 gpd, averages in the subwatersheds varied widely with a range between 78.5 and 1,072 gpd. The range for water uses in watersheds with 100 SFRs or greater is 152 to 374 gpd. The standard deviation among all the watershed averages is 161 gpd; the 132 gpd population estimated average fits within one standard deviation of the 227 gpd measured water use mean.

The purpose of the water use approach is to provide direct measurements, which relate to wastewater generation, rather than to make approximations using census data. The water use approach was developed for application in watersheds where accurate annualized population estimates were unavailable or could not be accurately determined. Three Bays is such a watershed, in that it has a large highly seasonal population with likely higher “real” occupancy rates than found in census figures. Given all the above analysis and the absence of information suggesting errors in the water use data, MEP staff decided to continue to use the Three Bays watershed-specific average water use for residential parcels without water use and for the 1,530 additional residential parcels included in the buildout analysis.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized lawns and golf courses, with lawns being the predominant source within this category. In order to add this source to the nitrogen loading model for the Three Bays system, MEP staff reviewed available information about residential lawn fertilizing practices and, for the Three Bays watershed, attempted to contact managers of large tracts of turf, such as golf courses, to incorporate site-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 was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. The assessment accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20% resulted in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn for use in the 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 were found to have the higher rate of fertilization application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

There are four golf courses in the Three Bays watershed: Wianno Golf Club, Oyster Harbors Club, Ridge Club, and Holly Ridge. Golf courses usually have different fertilizer application rates for different turf areas, usually higher annual application rates for tees and greens (~3-4 pounds per 1,000 square feet) and lower rates for fairways and roughs (~2-3.5 pounds per 1,000 square feet). Fertilizer application rates were available from Wianno Golf Club and Oyster Harbors Club; the application rates at the other two courses were assumed to be an average of application rates from the two watershed courses within application information and four other Cape Cod golf courses outside the watershed.

Fertilizer application information was also requested for ball fields and cemeteries within the Three Bays watershed, as well as the Marstons Mills Airport. Contacts for this information stated that none of these had active fertilizer applications. Only residential fertilizer applications are included in the nitrogen loading for the Three Bays system.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). 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 (Howes and Teal, 1995). Only the bog loses measurable nitrogen, the forested upland areas release only very low amounts. For the land-use N loading analysis, the areas of active bog surface are based on 85% of the total property area with cranberry bog land use codes. Factors used in the nitrogen loading analysis for the Three Bays watershed are summarized in Table IV-4.

IV.1.3 Calculating Nitrogen Loads

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

This review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (condominiums, WWTFs, etc.) were also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure, has a negligible effect on the total nitrogen loading to the Three Bays estuary. The assignment effort was undertaken to better define the sub-embayment loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Table IV-4. Primary Nitrogen Loading Factors used in the Three Bays MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Barnstable, Sandwich, and Mashpee watershed-specific data for Three Bays. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001. **Commercial assumptions also utilized for existing developed properties without water use.

Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing residential developed parcels wo/water accounts	227 gpd
Wastewater Coefficient	23.63		
Fertilizers:			
Average Residential Lawn Size (ft^2)*	5,000		
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08	Existing developed parcels w/water accounts	Measured annual water use
Cranberry Bogs nitrogen application (lbs/ac)	31	Buildout Parcels Assumptions:	
Cranberry Bogs nitrogen attenuation	34%	Residential parcels:	227 gpd
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined from site-specific information		Commercial and industrial parcels**	18 gpd/1,000 ft2 of building
		Commercial and industrial building coverage**	28%

Following the assignment of all parcels to individual subwatersheds in the Three Bays watershed, spreadsheets were generated for each of the 59 sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. As mentioned above, these results were then condensed to 37 subwatersheds based upon the time of travel analysis (<10 yr vs. > 10 yr) discussed above.

The results from the 37 individual sub-watershed assessments in the Three Bays study area were then integrated to generate nitrogen loading tables relating to each of the individual estuaries and their major components: Prince Cove, Cotuit Bay, West Bay, North Bay, and the Seapuit River. The sub-embayments represent the functional embayment units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated sub-embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Three Bays system, the major types of nitrogen loads are: wastewater, fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-5). 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 (Figures IV-4 a-f for Three Bays). The Three Bays annual watershed nitrogen input is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model. Natural attenuation within Marstons Mills River and Little River is also directly measured (Section IV.2) and compared to the attenuated annual watershed nitrogen load from the land-use sub-model

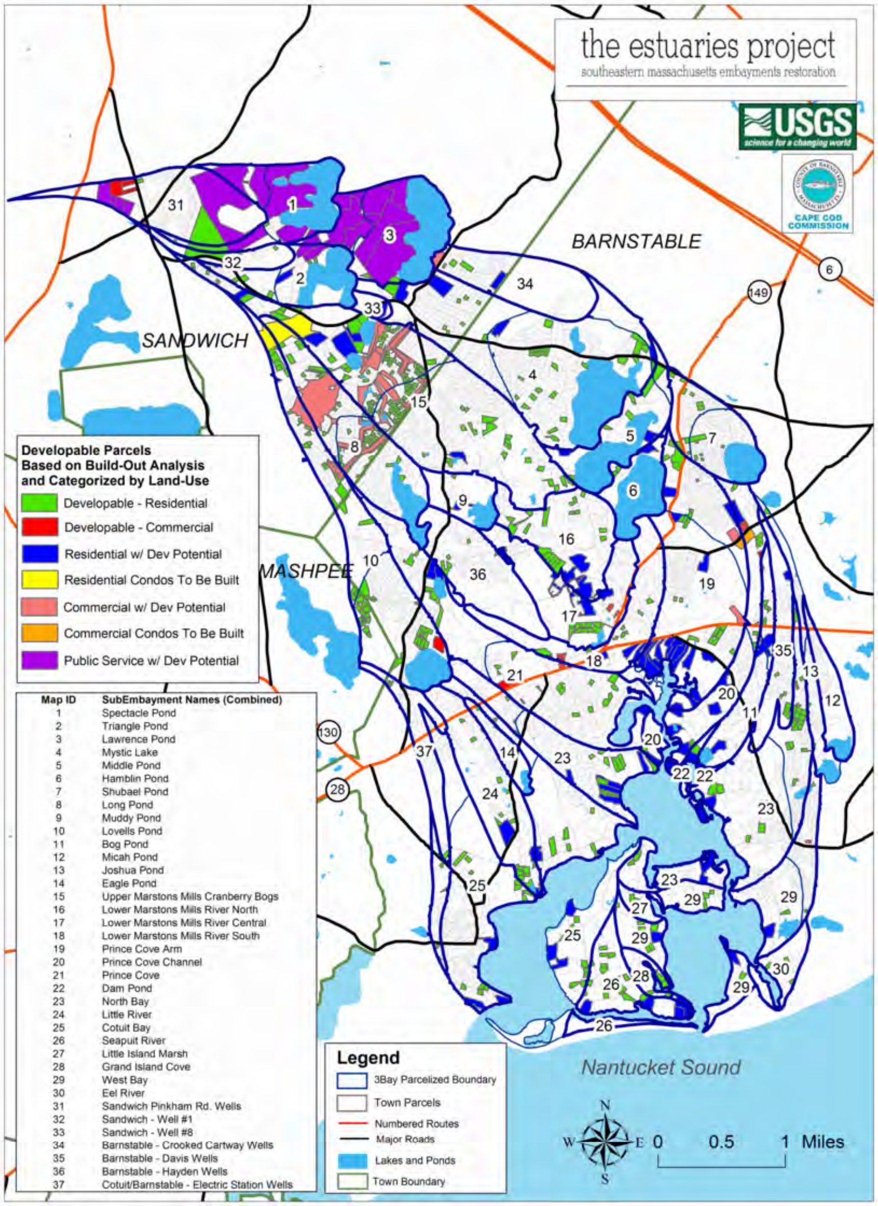
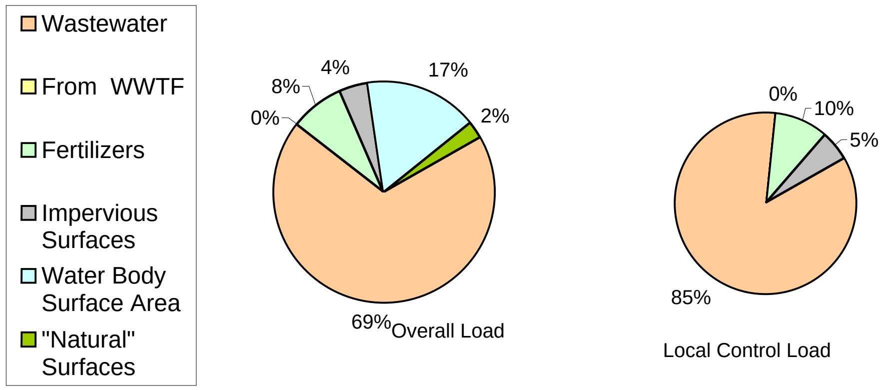


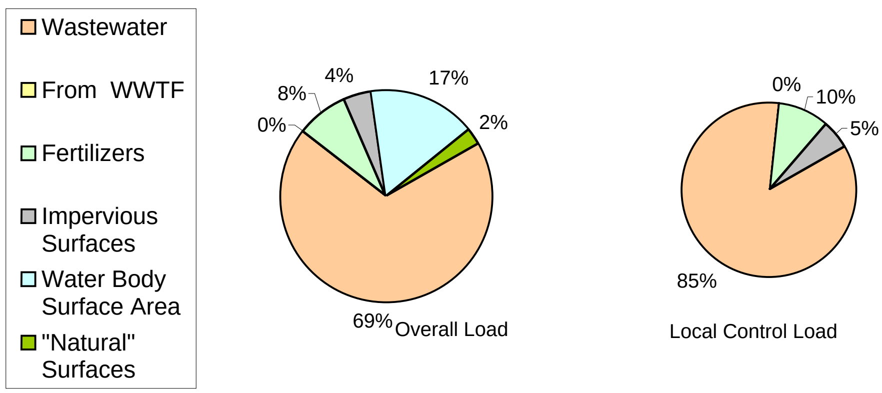
Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the sub-watersheds to the Three Bays Estuary.

Table IV-5. Nitrogen Loads to the tidal waters of the Three Bays Estuary. Attenuation of the Three Bays system nitrogen loads occurs as nitrogen moves through up-gradient ponds, the Marstons Mills River, and Little River during transport to the estuary.

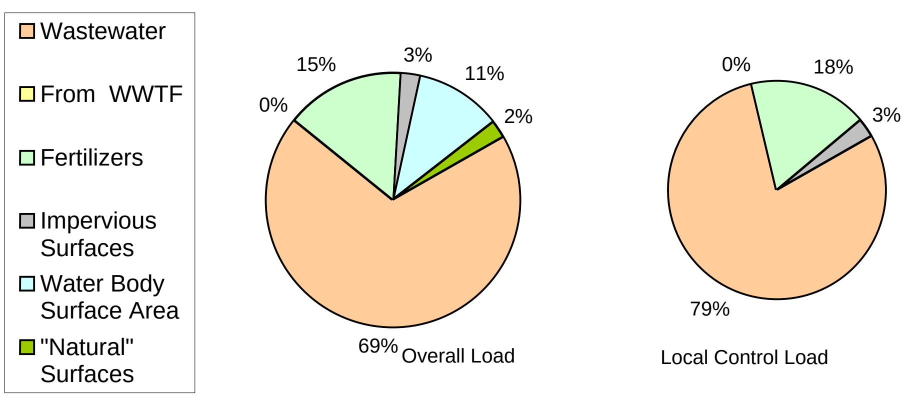
		Three Bay N Loads by Input:							% of Pond Outflow	Present N Loads			Buildout N Loads		
Watershed Name	Watershed ID#	Wastewater	From WWTF	Fertilizers	Impervious Surfaces	Water Body Surface	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Three Bay Estuary		53584	39	7920	2828	8555	1641	10372		74567		54657	84939		61296
Cotuit Bay Estuary	42,43,44,45, 46,59 + EP	9739	0	1086	589	2366	352	1915		14132		11653	16047		12929
Cotuit Bay Estuary surface deposition						2112				2112		2112	2112		2112
Little River Total	42 + LVP	2370	0	222	155	93	91	642		2932	30%	1592	3575	30%	1888
Seapuit River	47	1066	0	235	39	165	35	270		1540		1540	1810		1810
Seapuit River Estuary surface deposition						165				165		165	165		165
West Bay Estuary	48,49,50,51 + JP, MIP	5832	6	913	260	1623	97	372		8730		8505	9102		8873
West Bay Estuary surface deposition						1545				1545		1545	1545		1545
North Bay Estuary	21,22,38,39, 40,41,57 + LVP	9898	0	994	477	1527	221	1210		13117		12191	14327		13259
North Bay Estuary surface deposition						1443				1443		1443	1443		1443
Prince Cove Estuary	27,28,29,30, 31,32,33,34, 35,36,37 + TP,MDP,MP,HP	27050	34	4692	1463	2874	935	6605		37048		20768	43652		24425
Prince Cove Estuary surface deposition						449				449		449	449		449
Prince Cove Channel	34,35 + SP	1882	0	111	118	40	58	392		2208		2021	2600		2396
Prince Cove Arm	32,33 + HP	5666	0	458	323	842	221	1793		7510		4390	9303		5581
Prince Cove Subwatershed	36,37 + LP	4514	34	498	227	120	125	806		5519		4877	6324		5547
Lower MM River South	31,58 + LP,HP	4902	0	782	296	653	192	1121		6825	30%	2662	7946	30%	3023
Mill Pond	Lower MM River N & C	10086	0	2842	500	771	339	2493		14537	30%	6369	17030	30%	7431
Lower MM River Central	30 + MDP	2192	0	238	122	248	79	376		2880		2114	3256		2359
Lower MM River N (streamgauge)	27,28,29 + TP,MDP,MP,HP	7893	0	2604	378	523	260	2118		11657	30%	6984	13775	30%	8256



a. Three Bays System Overall

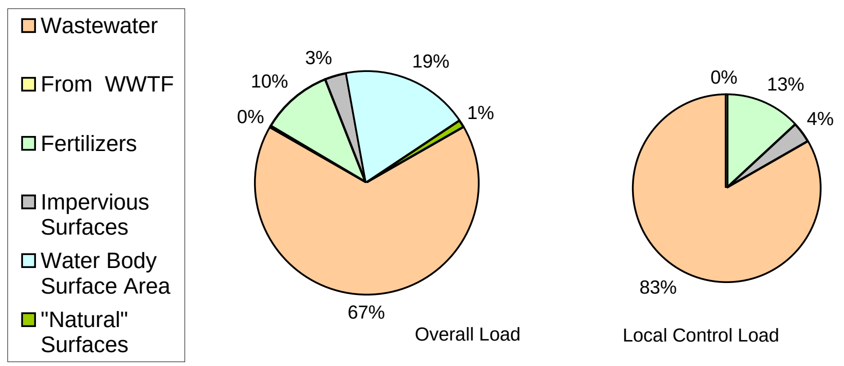


b. Cotuit Bay

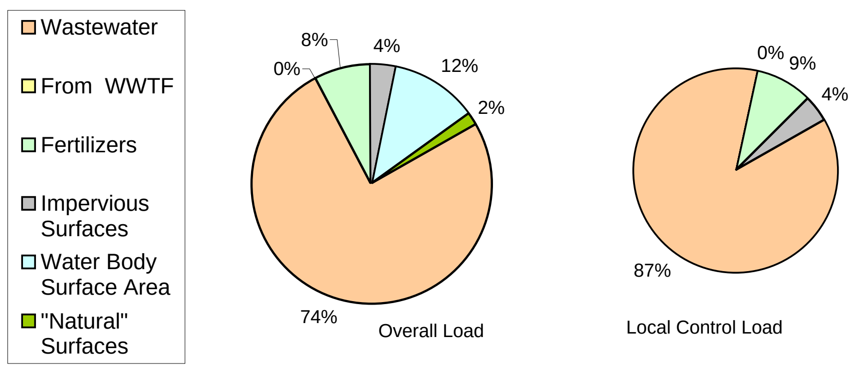


c. Seapuit River

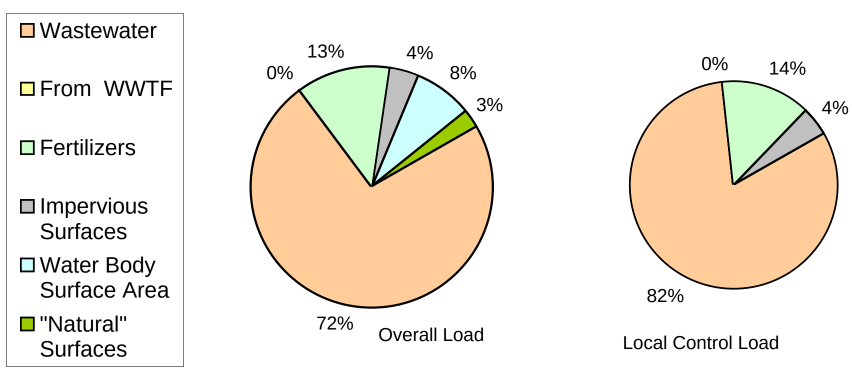
Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) to the (a) overall Three Bays System watershed, (b) Cotuit Bay subwatershed, and (c) Seapuit River subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.



d. West Bay



e. North Bay



f. Prince Cove

Figure IV-4 (d-f). Land use-specific unattenuated nitrogen load (by percent) to the (d) West Bay subwatershed, (e) North Bay subwatershed, and (f) Prince Cove subwatershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

Since groundwater outflow from a pond can enter more than one downgradient subwatershed, the length of shoreline on the downgradient side of the pond was used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment was based on the percentage of discharging shoreline bordering each downgradient subwatershed. So for example, Lawrence Pond has a downgradient shoreline of 7,020 feet; 54% of that shoreline discharges out of the Three Bays watershed, 26% discharges to the Crooked Cartway wellfield subwatershed (watershed 34 in Figure IV-1), and 19% discharges to the Mystic Lake subwatershed (watershed 34 in Figure IV-1). The nitrogen load discharging from Lawrence Pond is divided among these subwatershed based on these percentages of the downgradient shoreline.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call “windows on the aquifer.” Groundwater typically flows into the pond along the upgradient shoreline, then lake water flows back into the groundwater system along the downgradient shoreline. Occasionally a Cape Cod pond will have a stream outlet or herring run. Since the nitrogen loads flow into the pond with the groundwater, the relatively more productive ecosystems in the ponds incorporate some to the nitrogen, retain some of it in the sediments, and change it among its various oxidized and reduced forms. As result of these interactions, some of the nitrogen is removed from the watershed system, mostly through burial in the sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining, reduced loads flow back into the groundwater system along the downgradient side of the pond or through a stream outlet and eventually discharge into the downgradient embayment. The nitrogen load summary in Table IV-5 includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads reaching the estuary.

Pond nitrogen attenuation in freshwater ponds is generally set at 50% in MEP analyses based upon investigations within the MEP region. 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. However, if sufficient monitoring information is available for a specific fresh pond, an alternative attenuation rate is incorporated into the watershed nitrogen load modeling. In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond is reviewed. Available bathymetric information is reviewed relative to measured pond temperature profiles to determine whether an epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) exists in each pond. Of the ponds in the Three Bays study area, bathymetric information is available for all the ponds with delineated watersheds except Bog Pond, Muddy Pond, and Triangle Pond. Of the ponds with bathymetric information, Hamblin, Mystic, Shubael, and Spectacle are deep enough to develop strong temperature stratification and a separate epilimnion. Generally, if a strong epilimnion develops, the epilimnion is the active area of nitrogen uptake and transport to the shallow sediments and to the isolated hypolimnion. During stratification lower thermal layers are impacted by sediment regeneration of nitrogen.

In MEP analyses, freshwater recharging the downgradient shoreline, the available nitrogen concentrations and vertical structure of individual ponds are reviewed to determine the amount of nitrogen passing through. Analysis of the vertical structure (stratification) is made to determine whether the entire volume of the pond or the epilimnion is used in the hydraulic turnover calculations. Turnover time is how long it takes the recharge from the upgradient watershed to completely exchange the water in the pond or, in the case of a thermally stratified pond, just the epilimnion. The total mass of nitrogen in the pond or epilimnion is adjusted using the pond turnover time to determine the annual nitrogen load returned to the aquifer through the downgradient shoreline. This mass is then compared to the nitrogen load coming from the pond watershed to determine the nitrogen attenuation factor for the pond. However, the effort is undertaken as validation that overall a 50% attenuation factor is reasonable (i.e. the calculated values are $\geq$50%). In almost all cases that the MEP has examined to date, monitoring data has supported the use of a 50% attenuation as a conservative estimate.

The standard attenuation assumption for the other ponds in the Three Bays watershed was checked through the use of pond water quality information collected from the annual Cape Cod Pond and Lake Stewardship (PALS) water quality snapshot. The PALS Snapshot is a collaborative Cape Cod Commission/SMAST Program that allows trained, citizen volunteers of each of the 15 Cape Cod towns to collect pond samples in August and September using a standard protocol. Snapshot samples have been collected every year between 2001 and 2005. The standard protocol for the Snapshot includes field collection of dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at various depths depending on the total depth of the pond. PALS Snapshot data is available in the Three Bays watershed for the following ponds: Bog, Eagle, Hamblin, Joshua, Lawrence, Long, Lovells, Micah, Mill, Muddy, Mystic, Shubael, Spectacle, and Triangle. Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH.

In the Three Bays watershed there are 3 ponds with sufficient data to support a pond-specific attenuation factor: Mystic Lake, Middle Pond, and Hamblin Pond. These ponds form the Indian Ponds Complex and have been the subject of detailed nutrient related water quality monitoring between May and November 2004 and long-term monitoring (1999-2004).. The detailed 2004 water quality sampling effort was by Indian Ponds Association (IPA) volunteers and the longer term data from Three Bays Preservation. All samples were analyzed by the SMAST Coastal Systems Analytical Facility Laboratory with support from the University of Massachusetts-Dartmouth. Interpretation of the results is part of a current project that is being completed by the Cape Cod Commission for the Town of Barnstable and the IPA (Eichner, et al, in preparation).

The Indian Ponds analysis allowed the comparison of nitrogen, phosphorus, and water budgets for the three ponds. Water budget development began with the USGS watersheds (Figure III-1) and volume estimates based on Massachusetts Division of Fisheries and Wildlife bathymetric maps. Using the USGS recharge rate (27.25 inches per year), residence times were developed for each of the ponds. These residence times were checked against observed phosphorus concentrations and estimates of phosphorus loading. This analysis suggested that pumping of the public water supply wells (watersheds 31, 32, 33, and 34 on Figure IV-1) upgradient of the ponds was removing recharge and thereby altering the pond water budgets (e.g. lowering turnover). Nitrogen loads from the MEP analysis were then compared with measured total nitrogen concentrations and total mass within the ponds. Average total mass in the ponds over the entire sampling season were then compared with the watershed loading estimates and corrected to account for residence times. Based on this analysis, the nitrogen attenuation rates for the Indian Ponds are: 87%, Mystic Lake; 40%, Middle Pond; and 52%, Hamblin Pond.

Table IV-6 summarizes the pond attenuation estimates calculated from land-use modeled nitrogen inflow loads and nitrogen loads recharged to the downgradient aquifer or to outflow streams from each pond based on pond characteristics and measured nitrogen levels. Nitrogen attenuation within these ponds averaged 73% (20% s.d.), with a range of 40% to 97%.

However, a caveat to these attenuation estimates, except for the Indian Ponds, is that they are based upon nitrogen outflow loads from water column samples collected during summer and are not necessarily representative of the annual nitrogen loads transferred downgradient. Sampling at least comparable to that available for Mystic Lake, Middle Pond, and Hamblin Pond would be recommended for use of an attenuation rate different than 50%.

Table IV-6. Nitrogen attenuation by Freshwater Ponds in the Three Bays watershed based upon 2001 through 2004 Cape Cod Pond and Lakes Stewardship (PALS) program sampling and 2004 IPA/Barnstable sampling of Mystic Lake, Middle Pond, and Hamblin Pond. *Site specific nitrogen attenuation by these systems. Overall, estimates support the use of a 50% pond N attenuation rate within the Three Bays watershed for the MEP Linked N Model approach.

Pond	PALS ID	Area acres	Maximum Depth m	Overall turnover time Yrs	N Load Attenuation %
Bog Pond	BA-802	7.2	0.8	0.03	59%
Eagle Pond	BA-815	8.5	4.2	0.51	86%
Hamblin Pond*	BA-668	115.4	17.6	1.00	52%
Joshua Pond	BA-807	14.7	7.1	0.74	97%
Lawrence Pond	SA-431	133.8	8.2	2.40	91%
Long Pond	BA-675	54.8	5.5	0.48	81%
Lovells Pond	BA-759	55.5	10.3	0.78	46%
Micah Pond	BA-797	16.0	10.5	0.97	95%
Middle Pond*	BA-640	104.6	9.3	0.56	40%
Muddy Pond	BA-694	24.6	4	0.71	65%
Mystic Lake*	BA-584	148.4	13.5	1.10	87%
Shubael Pond	BA-664	55.1	11.2	1.92	91%
Spectacle Pond	SA-409	97.1	12.2	1.20	48%
Triangle Pond	SA-504	83.1	9.1	2.35	85%
				Mean	73%
				std dev	20%

Data sources: all areas from CCC GIS; Max Depth from MADFW or Cape Cod PALS monitoring; Volume for turnover time calculations from MADFW bathymetric maps (www.mass.gov/dfwele/dfw/dfw_pond.htm) and estimates based on max depth (max depth estimated for Bog, Muddy, and Triangle) ; TN concentrations for attenuation calculation from PALS monitoring and IPA study (Eichner, et al., in preparation)

Buildout

Part of the regular MEP watershed nitrogen loading modeling is to prepare a buildout assessment of potential development within the study area watershed. For the Three Bays modeling, MEP staff consulted with town planners to determine the parameters that would be used in the assessment and, in the case of Sandwich in the Three Bays watershed, reviewed the development potential of each property in the watershed. The standard buildout assessment is to evaluate town zoning to determine minimum lot sizes in each of the zoning districts, including overlay districts (*e.g*., water resource protection districts). Larger lots are subdivided by the minimum lot size to determine the total number of new lots. Staff also review developed properties with additional development potential; for example, residential lots that are twice the minimum lot size, but only have one residence. Parcels that are classified as developable residential (state class land use codes 130 and 131) but are less than the minimum lot size and are greater than 5,000 square feet are assigned an additional residence in the buildout; 5,000 square feet is a minimum lot size in some Cape Cod town zoning regulations. Commercial properties are not subdivided; the area of each parcel and the factors in Table IV-4 were used to determine a wastewater flow for these properties. The Sandwich Town Planner also requested that the Three Bays buildout include additional development potential for selected public service properties (state class land use codes in the 900s). All the parcels included in the buildout assessment of the Three Bays watershed are shown in Figure IV-3. A nitrogen load for each additional parcel included in the buildout and these was determined for the existing development using the factors presented in Table IV-4 and discussed above. A summary of total potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-5.

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 or sewering analysis) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of each sub-embayment of the overall Three Bays embayment system under study were based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1). If all of the nitrogen applied or discharged within a watershed reaches an embayment the watershed land-use loading rate represents the nitrogen load to the receiving waters. This condition exists in watersheds where nitrogen transport is through groundwater in sandy outwash aquifers with direct discharge to estuarine waters. 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 on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and removal (e.g. burial 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 which 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. Typical of other large embayment systems in the MEP region, most of the freshwater flow and transported nitrogen entering Three Bays, first passes through a surface water system, and frequently multiple systems, producing the opportunity for significant nitrogen attenuation.

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving 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 2001). Similarly, in 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 discharge from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Similarly, the small tidal basin of Frost Fish Creek in the Town of Chatham showed ~20% nitrogen attenuation or watershed nitrogen load prior to discharge to Ryders Cove. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives, and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements were undertaken as part of the MEP Approach. MEP conducted multiple studies on natural attenuation relating to sub-embayments of the Three Bays system in addition to the natural attenuation measures by fresh kettle ponds, addressed above. These additional site-specific studies were conducted in each of the 2 major surface water flow systems (i.e. the Marstons Mills River discharging to the tidal portion of the Prince Cove/Warren's Cove sub-embayment and Little River discharging to Cotuit Bay).

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 Marstons Mills River (immediately upgradient of Mill Pond just above Route 28) and Little River (immediately upgradient of Old Post Road) provide a direct integrated measure of all of the processes presently attenuating nitrogen in the sub-watersheds upgradient from the gauging sites. In the Marstons Mills River, a separate study was conducted where paired flow and nutrient measurements were made at the gauge site and at the outflows from Mill Pond. These data were used to determine the level of nitrogen attenuation resulting from passage through Mill Pond itself (based upon the N load entering the pond from the river and the watershed load discharging to the Pond through groundwater inflows).

The upper watershed regions to the Rivers account for more than half of the entire watershed area to the Three Bays System. Flow and nitrogen load were measured at the Marstons Mills site for 22 months of record and at the Little River site for 23 months of record (Figure IV-5). During the study period, velocity profiles were completed on each river every month to two months, with an effort to capture the range of flows experienced at each site. 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 gauges. A complete annual record of stream flow (365 days) was generated for both the Marstons Mills River and the Little River. The annual flow records for both rivers were merged with the nutrient data sets generated through the weekly water quality sampling to determine nitrogen loading rates to the tidally influenced portion of the Marstons Mills River and to Cotuit

Bay in the case of the Little River. Comparing these measured nitrogen loads based on stream flow and water quality sampling to predicted loads based on the land use analysis allowed for the determination of the degree to which natural biological processes within the watersheds to each river currently reduce (percent attenuation) nitrogen loading to the Three Bays embayment system.

Figure IV-5. Location of Stream gauges (red diamonds) on the Marstons Mills River discharging to Warren's Cove and Little River discharging to Cotuit Bay in the Three Bays Embayment System. The red triangle represents the site of periodic measurements coupled to the gauge site to determine attenuation by the associated terminal fresh pond, Mill Pond.

IV.2.2 Surface Water Discharge and Attenuation of Watershed Nitrogen: Marstons Mills River to Warren's Cove/Prince's Cove (head of North Bay)

The Mystic Lake – Middle Pond – Hamblin Pond group is one of the largest pond complexes on Cape Cod and the complex is referred to as the "Indian Ponds". Unlike many freshwater ponds, this pond system has stream outflow through down gradient cranberry bogs rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, the Marstons Mills River, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and stream bed associated with the Marstons Mills River and the terminal man-made pond, Mill Pond. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Marstons Mills River above the gauge site and the measured annual discharge of nitrogen at the gauge location. Attenuation by Mill Pond was assessed by measurements of nitrogen load entering and exiting the Pond in surface water flows and the nitrogen entering from the Mill Pond sub-watershed determined from the land use model, Figure IV-6. Note that <20% of the Marstons Mills River watershed load is through groundwater discharge to Mill Pond.

At the Marstons Mills River gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater portion of the Marstons Mills River (immediately upgradient of Mill Pond) that carries the flows and associated nitrogen load to the Bay. Calibration of the gauge was checked monthly. The gauge on the Marstons Mills River was installed on February 18, 2002 and was set to operate continuously for 22 months such that two summer seasons would be captured in the flow record. Due to an instrument upgrade in July of 2002, stage data collection was extended until December 31, 2003. The 12 month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured approximately monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Marstons Mills River site based upon these 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. These measurements (Figure IV-6 and Table IV-7) coupled with input/output measurements from Mill Pond allowed for the determination of both total volumetric discharge and nitrogen mass discharge to the estuarine portion of the Marstons Mills River. In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the Marstons Mills River (see below), determined from measured stage and the stage – discharge relation developed by the MEP, was compared to the flows determined by the USGS modeling effort (Table IV-8). The measured freshwater discharge from the Marstons Mills River was in good agreement with the long-term average flow from the groundwater model. The gauge on the Marstons Mills River showed an annual average flow 1.16 times the long term average. This is good agreement, especially considering that the rainfall during the deployment period was higher than the long-term average. Intra-annual variation of Marstons Mills flows stems not just from normal stream hydrologic cycles, but from hydrologic management associated with the upgradient cranberry bog system located at the head waters of the Marstons Mills River. Releases of water from the cranberry bog system affect the measured flows observed at the Marstons Mills River gauge. The measured freshwater flows and associated attenuated total nitrogen loads as obtained using the MEP gauge on the Marstons Mills River (with a slight adjustment for measured freshwater inflow directly from the Mill Pond watershed) were used in the water quality model discussed in Section VI.

Total nitrogen concentrations within the Marstons Mills River outflow to Mill Pond were relatively high, 0.799 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 12,850 g/day (12.85 kg/d) and a measured total annual TN load of 4,690 kg/yr. In the Marstons Mills River, nitrate was the predominant form of nitrogen (60%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was not completely taken up by plants within the pond or stream ecosystems. The high concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited.

From the measured nitrogen load discharged by the Marstons Mills River to Mill Pond and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the Bay. Based upon lower nitrogen load (12.85 kg N d^{-1}, 4,690 kg yr^{-1}) discharged from the freshwater Marstons Mills River and the nitrogen mass entering from the associated watershed (31.94 kg N d^{-1}, 11,657 kg yr^{-1}) the integrated measure of nitrogen attenuation by the pond/river ecosystem is 60%. Paired measures of water flow and nitrogen load into and out of Mill Pond based using the land use nitrogen load to the gauge site plus that the load directly to Mill Pond indicate an integrated watershed nitrogen attenuation (i.e. nitrogen loading to the estuarine reach of the Marstons Mill River (e.g. below Mill Pond) of ~64%. The added attenuation stems from nitrogen removal within Mill Pond itself. This nitrogen removal by Mill Pond was smaller than the 50% typical of kettle ponds as a result of the shallow nature of the pond (low hydraulic residence time) and that it is a “flow through” pond, i.e. it is dominated by surface water inflows and outflows. The estimate of nitrogen removal by Mill Pond based upon paired measurements in August and September is consistent with the 30% removal typical of rivers and streams. Additional inflow/outflow data is being collected to further refine the Mill Pond specific attenuation rate for assessment of management alternatives (see Table IV-7). The directly measured nitrogen loads from the rivers were used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-7. Comparison of water flow and nitrogen discharges from Marstons Mills River to Mill Pond and Little River discharging to Cotuit Bay. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS. Note that the total nitrogen load exiting Mill Pond is only ~10% higher than the loading at the gauge site, due to attenuation of River transported nitrogen and nitrogen entering from the Mill Pond sub-watershed.

Stream Discharge Parameter	Stream Discharge to Little River[(a)]	Stream Discharge to Marstons Mills River[(a)]	Data Source
Total Days of Record	365[(b)]	365[(b)]	(1)
Flow Characteristics			
Stream Average Discharge (m3/day) 2002-2003	3483	16091	(1)
Contributing Area Average Discharge (m3/day) Long Term	4211	13922	(2)
Discharge Stream 2002-03 vs. Long-term Discharge	83%	116%	(5)
Contributing Area, with 40% reduction in Lovell's Pond recharge (m3/d)	3521	--	(7)
Discharge Stream 2002-03 vs. Long-term Discharge without LP recharge	99%	--	
Nitrogen Characteristics			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.856	0.481	(1)
Stream Average Total N Concentration (mg N/L)	1.137	0.799	(1)
Nitrate + Nitrite as Percent of Total N (%)	75%	60%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	3.96	12.85	(1)
TN Average Contributing Area Attenuated Load (kg/day)	4.36	19.13	(2)
TN Average Contributing UN-attenuated Load (kg/day)	8.03	31.94	(3)
Attenuation of Nitrogen in Pond/Stream (%)	51%	60%	(4)
MEP "Best Estimate" of Nitrogen Load in Mill Pond outflow (kg/d)	--	14.52	(6)

(a) Flow and N load to stream discharging to Little River includes Lovell Pond contributing area.
(a) Flow and N load to stream discharging to Marstons Mills RIver includes "Indian Ponds" contributing area.
(b) October 11, 2002 to October 10, 2003.
(1) MEP gage site data
(2) Calculated from MEP watershed delineations to Long Pond and Mares Pond for flow to Little Pond;
the fractional flow path from each sub-watershed which contribute to the flow in the stream to Little Pond;
and the annual recharge rate. Calculated nitrogen exiting Mill Pond using MEP standard attenuation factors.
(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.
(5) Little River measured flows 17% lower than long term average due to hydrologic diversions from Lovell's Pond to nearby cranberry bogs
(6) Nitrogen Load exiting Mill Pond based upon paired measurements of inflow and outflow freshwater and loads, and modeled watershed loading
(7) Modeled discharge in Little River if 40% of the upper watershed flow (above Lovell's Pond) is diverted through management practices.

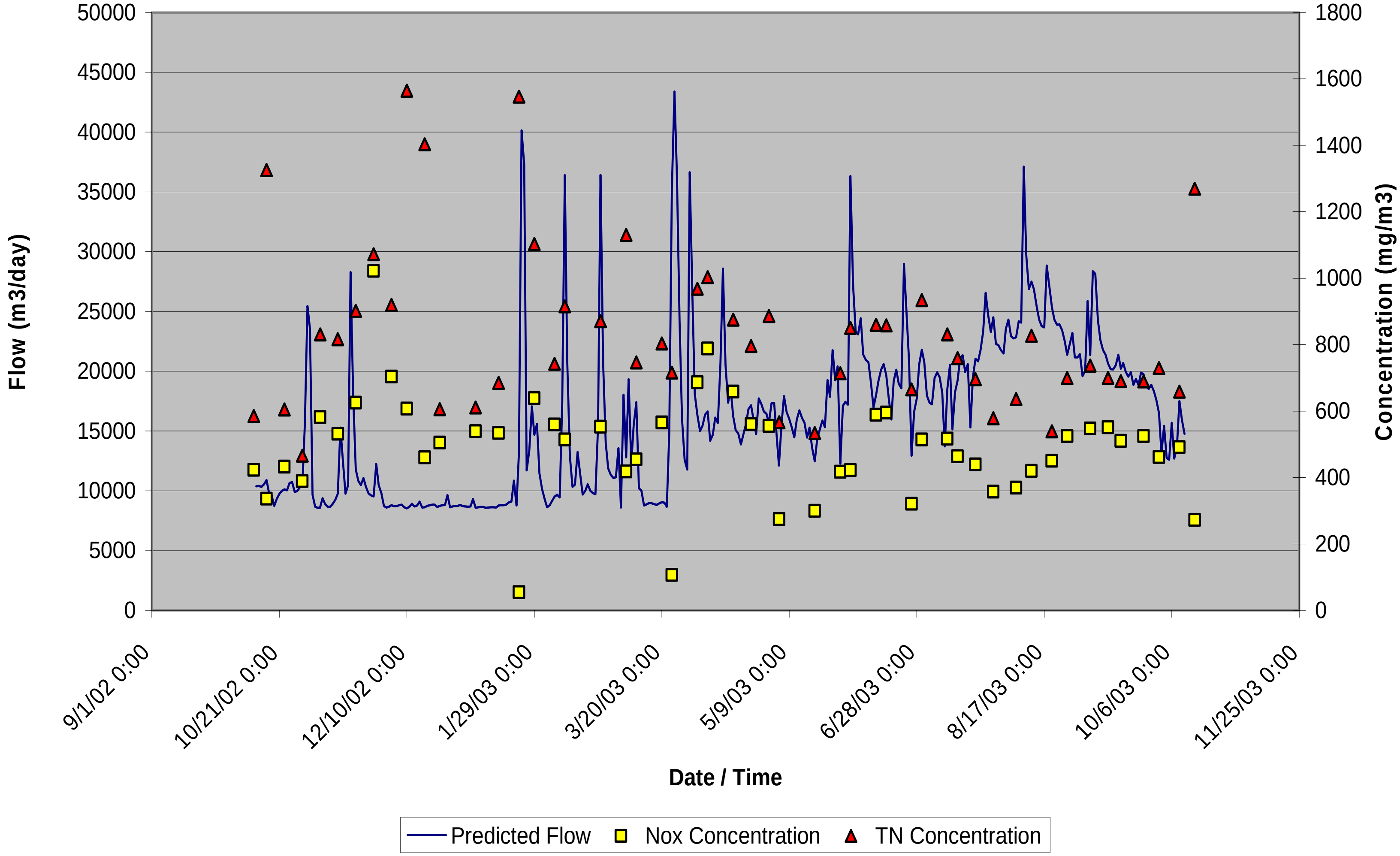


Figure IV-6. Marstons Mills River annual discharge developed from a stream gauge maintained above Mill Pond immediately upgradient of Route 28, Town of Barnstable, October 2002 to October 2003. Nutrient samples (Nox – Nitrate+Nitrite) were collected weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-7).

IV.2.3 Freshwater Discharge and Attenuation of Watershed Nitrogen: Little River to Cotuit Bay

Lovell's Pond is one of the larger ponds within the study area and unlike many of the freshwater ponds, the Lovell's Pond and adjoining cranberry bogs has stream outflow to the Little River, rather than discharging solely to the aquifer on the down-gradient shore. The stream, Little River, has highly manipulated flows into and out of Lovell's Pond. The hydrologic management of this system is such that outflow from Santuit Pond to Lovell's Pond and ultimately to Little River has been significantly altered from year-to-year as part of both cranberry agriculture and as part of fisheries management programs. As in the Marstons Mills River (see IV.2.2 above) the stream outflow from Lovell's Pond may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. Nitrogen attenuation also occurs within the wetlands and stream-bed associated with the Little River. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Little River above the gauge site and the measured annual discharge of nitrogen.

At the Little River gauge site (Figure IV-5), a continuously recording vented calibrated water level gauge was installed to yield the level of water for the determination of freshwater flow. Calibration of the gauge was checked monthly. The gauge on the Little River was installed on January 14, 2002 and was set to operate continuously for 23 months such that two summer seasons would be captured in the flow record. Due to the desire to have simultaneous measurement of river discharge from the Mashpee and Santuit Rivers, stage data collection was extended until December 5, 2003 (to match the Marstons Mills River). The 12 month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Little River site based upon these 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. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to the western shore of Cotuit Bay (Figure IV-7 and Table IV-7). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The annual freshwater flow record for the Little River (see below), determined from measured stage and the stage – discharge relation developed by the MEP, was compared to the flows determined by the USGS modeling effort (Table IV-8). The measured freshwater discharge from the Little River was approximately 17 percent lower than the long-term average modeled flows. The lower values are attributable to the fact that the functional watershed to Little River is currently smaller than historically. Water management of historic flows related to Santuit Pond appears to have reduced the effective recharge area. An estimate of the amount of reduction in recharge area necessary to account for the observed difference in flows is only ~40% of the uppermost watershed (Table IV-7). The reduction in outflow from Lovell's Pond is supported by observations of the upper most reach of the Little River at its headwaters (Lovell's Pond) going dry during summer. Site reconnaissance identified that Lovell's Pond is connected to the upper most reach of the Little River by a century old ceramic pipe that is greatly compromised along its length prior to discharging to a man made ditch that ultimately becomes the Little River. More importantly, channels transporting water from Santuit Pond have been recently managed for the protection of fisheries to not discharge water associated with cranberry agriculture or eutrophic Santuit Pond waters to Lovell's Pond. However, as the nitrogen load discharging to Cotuit Bay through the Little River is a direct measurement of the integrated upgradient watershed it represents the current nitrogen loading to the estuary and was used in the water quality modeling for nitrogen threshold determination.

Total nitrogen concentrations within the Little River outflow were relatively high, 1.14 mg N L^{-1}. Average daily total nitrogen discharge from the Little River to the estuary was 3,960 g/day (3.96 kg/d) with a measured total annual TN load of 1,446 kg/yr. As in the Marstons Mills River, nitrate was the predominant form of nitrogen (75%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was not completely taken up by plants within the pond or stream ecosystems. The high concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited. Nitrate in the Little River is a higher fraction of the total nitrogen pool than in the Marstons Mills River (75% vs. 60%) suggesting that there is less attenuation of inorganic nitrogen in the Little River system. This is partially attributable to the fact that Little River flow passes through fewer ponds and wetlands while also having a shorter river reach prior to discharging to Cotuit Bay.

From the measured nitrogen load discharged by the Little River to the estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to Cotuit Bay. Based upon the lower measured nitrogen load (3.96 kg N d^{-1}, 1,440 kg N yr^{-1}) discharged from the Little River and nitrogen mass entering from the associated watershed (8.03 kg N d^{-1}, 2,932 kg yr^{-1}), the integrated measure of nitrogen attenuation by the pond/river ecosystem is 51%. This is consistent with the land-use model which yielded and integrated nitrogen attenuation of 46%, since pond and stream attenuation in the watershed model use conservative attenuation factors (Table IV-6). However, while the directly measured Little River nitrogen load should accurately represent the sub-watershed loading to Cotuit Bay, the issues relating to hydrologic manipulation of upper watershed recharge to Lovell's Pond may result in an overestimate of the unattenuated loading value, hence a slight overestimate of the integrated attenuation rate (i.e. actual current rate may be <51%). Directly measured nitrogen loads from the rivers were used in the Linked Watershed-Embayment Modeling of water quality (Chapter VI).

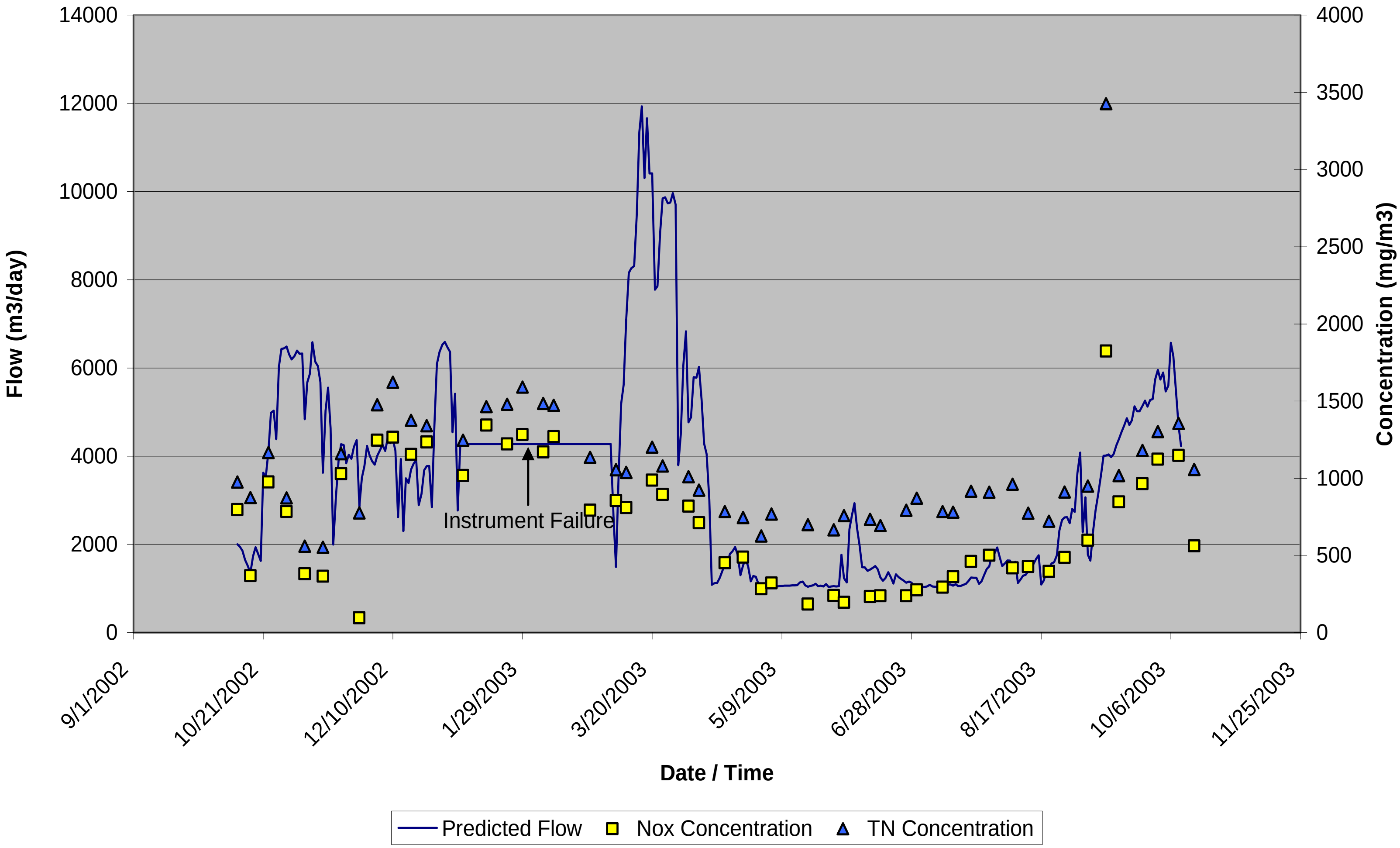


Figure IV-7. Little River annual discharge developed from a stream gauge maintained in the outflow from Lovell's Pond discharging to Cotuit Bay, Town of Barnstable, October 2002 to October 2003. Nutrient samples (Nox – Nitrate+Nitrite) were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-7).

Table IV-8. Summary of Flow and Nutrient loads from both the Marstons Mills River discharging to tidally influenced Warren's Cove (head of North Bay) and the Little River discharging to Cotuit Bay

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Marstons Mills River Marstons Mills River (CCC)	October 11, 2002 to October 10, 2003 Based on Watershed Area and Recharge	5873174 5081596	2824	4690
Little River to Cotuit Little River (CCC)	October 11, 2002 to October 10, 2003 Based on Watershed Area and Recharge	1271311 1536977	1088	1446

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

The overall objective of the Benthic Nutrient Flux Task was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters within each major basin area within the Three Bays System. The mass exchange of nitrogen between watercolumn and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

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

In general the fraction of the phytoplankton population which enters the surficial sediments of a shallow embayment: (1) increases with decreased hydrodynamic flushing, (2) increases in low velocity settings, (3) increases within small enclosed basins (e.g. Prince’s Cove, Warren’s Cove, etc). To some extent, the settling characteristics can be evaluated by observation of the grain-size and organic content of sediments within an estuary.

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content, that bioavailable nitrogen is returned to the embayment watercolumn 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 we have investigated, 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. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for Determining Sediment-Watercolumn Nitrogen Exchange

For the Three Bays System, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected in the Three Bays system from 20 sites (Figure IV-8) in August 2002. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample. As part of a separate research investigation, the rate of oxygen uptake was also determined and measurements were made of sediment bulk density, organic nitrogen, and carbon content. These measurements were made by the Coastal Systems Program at SMAST-UMD.

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 a small boat. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. The number of core samples from each site (see Figure IV- 8) per incubation were as follows:

Three Bays Embayment System

- Station 1 – 1 core (Prince’s Cove)
- Station 2 – 1 core (Prince’s Cove)
- Station 3 – 1 core (Prince’s Cove)
- Station 4 – 1 core (Warren’s Cove)
- Station 5 – 1 core (Marstons Mills River – tidal reach)
- Station 6 – 1 core (North Bay)
- Station 7 – 1 core (North Bay)
- Station 8 – 1 core (North Bay)
- Station 9 – 1 core (North Bay)
- Station 10 – 1 core (West Bay)
- Station 11 – 1 core (West Bay)
- Station 12 – 1 core (West Bay)
- Station 13 – 1 core (West Bay)
- Station 14 – 1 core (Seapuit River)
- Station 15 – 1 core (Cotuit Bay)
- Station 16 – 1 core (Cotuit Bay))
- Station 17/18 – 2 cores (Cotuit Bay)
- Station 19 – 1 core (Cotuit Bay)
- Station 20 – 1 core (Eel River)

Sampling was distributed throughout the embayment system to capture general spatial heterogeneity and the results for each site combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

Sediment-watercolumn exchange follow the methods of Jorgensen (1977), Klump and Martens (1983), and Howes *et al.* (1995) for nutrients and metabolism. Upon return to the field laboratory (Wianno Yacht Club located on the shores of North Bay, private residence on the shore of Rushy Marsh) the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and ortho-phosphate (Murphy and Reilly 1962) assays were conducted within 24 hours and the remaining sample frozen (-20^{o}C) for assay of nitrate + nitrite (Cd reduction: Lachat Autoanalysis), and DON (D'Elia *et al.* 1977). Rates were determined from linear regression of analyte concentrations through time.

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

Figure IV-8. Three Bays System locations (red diamonds) of sediment sample collection for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-9.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

Watercolumn nitrogen levels are the balance of inputs from direct sources (land, rain etc), losses (denitrification, burial), regeneration (watercolumn 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 watercolumn 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 watercolumn 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 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, which relate primarily to sediment and watercolumn oxygen status. However, for the modeling of nitrogen within an embayment it is the relative balance of nitrogen input from watercolumn to sediment versus regeneration which is critical. Similarly, it is the net balance of nitrogen fluxes between water column and sediments during the modeling period that must be quantified. For example, a net input to the sediments represents an effective lowering of the nitrogen loading to down-gradient systems and net output from the sediments represents an additional load.

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

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

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

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 watercolumn 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 ammonium release, measured nitrate uptake or release, and estimate of particulate nitrogen input. Dissolved organic nitrogen fluxes were not used in this analysis, since they were highly variable and generally showed a net balance within the bounds of the method.

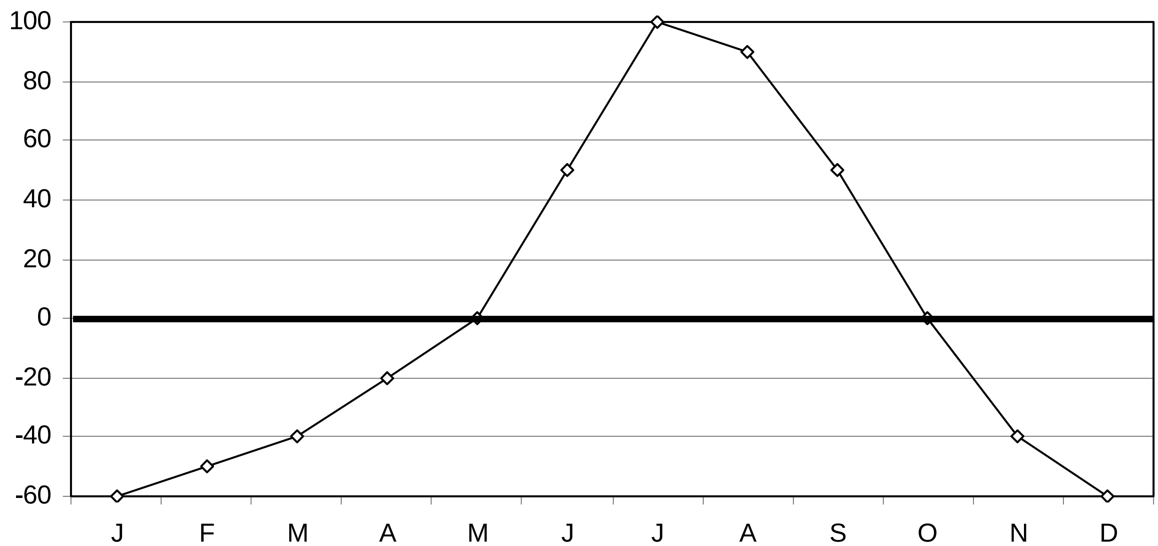


Figure IV-9. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment sampling was conducted within each of the sub-embayments of the Three Bays System in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-8). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and bulk density and an analysis of each site's tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site and the average summer particulate carbon and nitrogen concentration within the overlying water. Two levels of settling were used. If the sediments were organic rich and a fine grained and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated a coarse grained sediments and low organic content and high velocities, then half this settling rate was used. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. This approach was 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.

Net nitrogen release or uptake from the sediments within the Three Bays embayment for use in the water quality modeling effort (Chapter VI) are presented in Table IV-9. Net nitrogen release from the sediments of the Three Bays sub-embayments shows significant spatial variation, but is typical of other embayment within the MEP region. The general pattern is for higher release from the more heavily nitrogen loaded basins, typically in the inner reaches of the estuary, and lower release or uptake in the basins closer to the tidal inlets. This overall pattern reflects the particle distribution within Three Bays due to phytoplankton production and deposition. This was also the pattern within adjacent Popponesset Bay, which has a similar pattern of loading and multiple large sub-embayments. The high rates of nitrogen release in Warrens Cove reflect both its eutrophic status and its function as a salt marsh basin. Lowering the nitrogen inputs to the inner basins will result in lower net nitrogen release rates over relatively short time scales.

Table IV-9. Rates of net nitrogen return from sediments to the overlying waters of the Three Bays 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 August rates.

		Sediment Nitrogen Release		
Sub-Embayment	**Station**	**Mean mg N m^{-2} d^{-1}**	**std. dev. mg N m^{-2} d^{-1}**	N
Prince Cove	1-3	10.3	13.3	3
Warrens Cove	4	108.8	19.2	1
Channel to North Bay	5	24.6	5.1	1
North Bay	6-9	51.4	8.6	4
West Bay	10-13	4.5	8.2	4
Eel River	20	-6.4	11.4	1
Cotuit Bay	15-19	-29.1	3.3	5
Seapuit River	14	-37.7	5.4	1

Higher nitrogen net fluxes from sediments of the upper more nitrogen enriched basins also may result from differences in sediment nitrogen cycling.. There is an indication that the very reducing (anoxic) nature of the Princes Cove and Warrens Cove sediments may be increasing the percentage of nitrogen which is released from the sediments versus the amount of nitrogen being lost to denitrification via the pathway of mineralization→ nitrification → denitrification. The coupled nitrification-denitrification step in the pathway is significantly influenced by the availability of oxygen within the surficial sediments for nitrifying bacteria. That the anoxic/sulfidic nature of the sediment of these basins may be affecting enhancement of nitrogen release is supported by comparisons of measured release with estimates of total nitrogen regeneration (i.e. maximum potentially releasable).. Using this rough approximation, a greater proportion of the potential release rates of nitrogen is achieved in the upper basins than from the other sites. Note that this approach yields general patterns and cannot be used to determine accurate nitrogen removal rates. Lowering nitrogen loading to these upper systems should improve sediment oxidation and improve nitrogen removal rates by these sediments, although quantifying this enhancement is highly site specific. However, based upon this information a linear model for the lowering of nitrogen release with lowered watershed nitrogen loading is conservative.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

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

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

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

A hydrodynamic study was performed for the Three Bays system, which is located on the south shore of Cape Cod. A section of a topographic map in Figure V-1 shows the general study area. The three Bays system has four major subdivisions, Cotuit Bay, West Bay, North Bay, and Prince Cove. Cotuit Bay and West Bay both have there own outlets to Nantucket sound, and are connected though the Seapuit River and also via North Bay. Osterville Grand Island and Little Island lie within the Three Bays system. Dead Neck is the barrier island that describes the southern boundary of this estuarine system, and protects Grand Island from the open waters of Nantucket Sound.

Figure V-1. Topographic map detail of the Three Bays System, in Barnstable, Massachusetts.

The entire Three Bays system has a surface coverage of 1251 acres, including several small sub-embayments attached to the system's main sub-embayments. Cotuit Bay is the largest sub-embayment of the Three Bays system, covering 469 acres. The average depth of the whole embayment is 6.2 ft. West Bay has an area coverage of 343 acres and an average depth of 5.3 ft. North Bay has an area coverage of 309 acres, and an average depth of 5.3 ft. Prince Cove together with Warren Cove and the Marstons Mills River are the northernmost reaches of the Three Bays system, with a 93-acre coverage. The Marstons Mills River is the largest surface source of fresh water into the estuary.

Circulation in the Three Bays system is dominated by tidal exchange with Nantucket Sound. There is negligible attenuation of the tide range throughout the system, even into its uppermost reaches in Prince Cove. This indicates that there is little loss of tidal energy through the system, either due to bottom friction in shallow areas or from channel restrictions, e.g., at the system inlets and the Little Island draw bridge.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry, tide, and circulation velocity data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Three Bays was performed to determine the variation of embayment and channel depths throughout the system. This survey addressed the previous lack of available bathymetry data for this system. In addition to the survey, tides were recorded at seven locations within Three Bays for 44 days. These tide data were necessary to run and calibrate the hydrodynamic model of the system. Finally, an Acoustic Doppler Current Profiler (ADCP) survey was completed during a single tide cycle to measure ebb and flood velocities across five channel transects. The ADCP data were used to compute system flow rates and to provide an independent means of verifying the performance of the hydrodynamic model.

A numerical hydrodynamic model of the Three Bays system was developed in he 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 offshore Dead Neck were used to define the open boundary conditions that drive the circulation of the model at the two system inlets, and data from the five TDR stations within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

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

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE ESTUARINE SYSTEM

A general understanding of the hydrodynamic controls and coastal processes influencing estuarine dynamics provides the initial framework for the hydrodynamic analysis. In addition, both natural and anthropogenic changes to the estuarine system can guide the evaluation of effective alternatives to enhance tidal circulation and improve water quality.

The southern coast of Cape Cod between the Popponesset Bay and West Bay entrances can be considered a moderately dynamic region, where natural wave and tidal forces continue to reshape the shoreline. Due to the protection afforded by the islands of Marthas Vineyard and Nantucket, the south shore of Cape Cod is protected from the influence of long period open ocean wave conditions. Similar to many portions of the Massachusetts coast, the available sediment supply influences the migration and/or stability of tidal inlets. Tidal inlets can become overwhelmed by the gradual wave-driven migration of a barrier beach separating the estuaries from the ocean. In addition to these natural coastal processes, man-made structures often can influence the stability of a shoreline/tidal inlet system.

V.2.1 Natural Coastal Processes

For the Three Bays estuarine system, the process of barrier spit elongation continues to have a significant influence on tidal exchange. Over the past 80 years, the Sampsons Island barrier elongation has influenced tidal exchange through the Cotuit Bay entrance, where the

West Bay inlet has gradually become more vital to tidal exchange within the Three Bays system. Figure V-2 illustrates the gradual narrowing of the Cotuit Bay entrance since 1938. This narrowing has caused migration of the navigation channel to the west. By 2003, some coastal engineering structures along the west side of the inlet became undermined and collapsed into the channel. As the Sampsons Island spit continues to elongate, the Cotuit Bay entrance becomes narrower and the inlet becomes less efficient for tidal exchange.

Figure V-2. Historical shoreline positions in the vicinity of the Cotuit Bay entrance between 1938 and 2004.

Figure V-3 shows the historic shoreline change fronting Rushy Marsh and the Three Bays region between 1938 and 2005. Much of the accretion along the barrier beach separating Rushy Marsh from Nantucket Sound is a result of the Popponesset spit remnants joining the barrier beach fronting Rushy Marsh. After this spit welded onto the existing shoreline, the net west-to-east directed littoral drift has "straightened" the shoreline in this region, where slight erosion has been observed along the beach fronting the southwest end of the pond and significant accretion has been observed along the beach fronting the remainder of the pond. To the northeast of Rush Marsh Pond, the shoreline has experienced erosion over the past 60+ years, likely resulting from the westerly migration of Sampsons Island.

Although littoral drift generally moves from west-to-east along the south shore of Cape Cod, due primarily to the prevailing southwest winds, a sediment transport reversal occurs along the shoreline of Dead Neck and Sampsons Islands. This local reversal is due to a slightly different shoreline orientation relative to the incident wave climate. As shown in Figure V-3, the east-to-west littoral drift generally causes erosion along the eastern portion of Dead Neck, with a corresponding accretion/spit elongation at the west end of Sampsons Island. The observed erosion rates along the eastern portion of Dead Neck are moderated by ongoing beach nourishment efforts (described below).

Figure V-3. Observed shoreline change from 1938 to 2001/2005 for the shoreline area in the vicinity of Rushy Marsh Pond and Three Bays in Barnstable.

V.2.2 Anthropogenic Changes Influencing Rushy Marsh Pond

Manmade coastal structures along the shoreline immediately west of Cotuit Bay entrance consist primarily of groins along this updrift shoreline. Based on site observations, most of these structures are not effective barriers to natural littoral drift; therefore, beach compatible material continues to supply the beach systems along the shoreline of Cotuit Bay. The volume of material transported along this shoreline stretch is relatively small, due primarily to the quiescent wave conditions within the protected waters of Nantucket Sound. The conclusion that the longshore sediment transport rate is relatively low is further supported by the stable shoreline northeast of Rushy Marsh Pond and the small maintenance dredging volumes required to maintain the entrance to Cotuit Bay (which receives littoral sediments from both the east and the west).

The Three Bays estuarine system was significantly modified during the 1920s by the construction of the West Bay cut. Prior to this time, only the Cotuit Bay entrance connected the Three Bays system to Nantucket Sound. Figure V-4 illustrates the condition of the system immediately before development of West Bay inlet. Creation of this structured inlet altered the tidal exchange within the Three Bays system, significantly reducing the volume of water flowing through the Cotuit Bay entrance.

Figure V-4. Bathymetry map of the Three Bays estuarine system in 1897.

Over the past 20 years, significant efforts have been made by homeowners on Grand Island, as well as Three Bays Preservation, Inc. (a non-profit group concerned with the overall environmental conditions of Three Bays), to maintain the integrity of the Dead Neck barrier beach system. This maintenance of the barrier system has been in the form of beach nourishment as described below.

In order to enhance the storm protection capability of the eastern end of Dead Neck, two major beach nourishments have been completed on the island, adjacent to West Bay inlet. This segment of the Dead Neck shoreline has historically been the most erosive area of the island, due to it proximity to West Bay inlet. The inlet, with its jetties, effectively interrupts littoral transport from updrift beaches. This has been the case since the opening of the inlet in the first half of the 20th century. Net transport along the Dead Neck shoreline is directed toward the Cotuit Bay entrance from the West Bay entrance. Therefore, as it is cut off from the natural source of sediment to the east, the east end of the island continues to erode (apparent as shoreline retreat and lowering of the island), further inhibiting its ability to adequately serve as storm protection for the area directly landward of the island (in the Seapuit River) and in West Bay.

For the first nourishment in 1985, 120,000 cubic yards of sand were placed along the section of beach starting at West Bay inlet and extending 2,400 feet westward (Wood, et al., 1996). Beach compatible material dredged from the West Bay inlet entrance channel was the source of sand used for this project. The design template of the nourishment had a berm elevation of +12.0 ft NGVD and a width of 100 ft. On average, the fill template required 50 to 60 cubic feet of sand per foot length of beach and had a stated design life of 10 years.

The performance of the 1985 nourishment was monitored on a semi-annual basis up to 1993. In 1993 approximately 7% of the nourishment volume remained in the template area. The average volume loss rate for the 7.5-year monitoring period was 14,880 cubic yards per year. Erosion rates along the template were higher than average at the completion of the nourishment as the beach fill profile equilibrated. Toward the end of the monitoring period, erosion rates were again accelerated due to a series of severe storms which impacted this shoreline during this time, including Hurricane Bob (August, 1991), the “no-name” northeast storm of October, 1991 and the Blizzard of 1993.

The second nourishment project commenced in the first half of 1999 and was completed in winter 2000 (Woods Hole Group, 2001). Figures V-5 and V-6 show the condition of the Dead Neck barrier beach in 1999 (immediately preceding nourishment) and 2000 (immediately following nourishment), respectively. The fill template for this nourishment had an elevation of +13 MLW, and a berm crest width of 150 feet. The sand used for the fill was available from channel maintenance dredging at Cotuit and West Bay Inlets. For this nourishment, 187,300 cubic yards were initially placed along a 2,000 foot-length of the Dead Neck shoreline, starting at the West Bay inlet. This resulted in an average fill volume of 95 cubic yards per foot of shoreline. In 2000, the fill was supplemented with an additional 25,100 cubic yards of sand, placed over the easternmost 1,000 feet of the island. The total volume of sand for the 1999-2000 nourishment project was therefore 212,400 cubic yards.

Since the completion of the nourishment in the first quarter of 2000, the movement of the Dead Neck shoreline has been monitored annually through the use of Differential GPS (DGPS) shoreline surveys and cross-shore profile measurements. These two data sets were used to determine shoreline change rates and volume loss in the 1999-2000 nourishment area, as well as shoreline change rates for the entire seaward shoreline of the island. In Figure V-7, measured shoreline change rates are indicated by color bars along the Dead Neck shoreline. For the whole seaward facing shoreline the maximum erosion rate was computed to be -23.4 feet per year (ft/yr). This maximum rate occurred about 1100 feet west of the West Bay Inlet. The average change rate over the 2,000 foot length of the 1999 nourishment was -17.0 feet during this period. At the western end of Dead Neck, the westernmost 1,000 feet of shoreline from Cotuit Inlet was accretional, with an average change rate of +4.9 ft/yr. Based on volume calculations, 36% of the fill volume has been lost from the nourishment area over the last five years. At the time of the September 2004 survey, there were 137,000 yd^3 of sand remaining. Over the approximate five-year period since the bulk of the 1999 nourishment was completed, the average rate of volume loss has been -13,700 yd^3/yr.

Figure V-5. The eastern portion of Dead Neck in 1999, showing at least two locations where the beach had significant storm overwash areas.

Figure V-6. Dead Neck Beach immediately following the beach nourishment in 2000, where the beach width had been increased significantly to prevent breaching of the barrier.

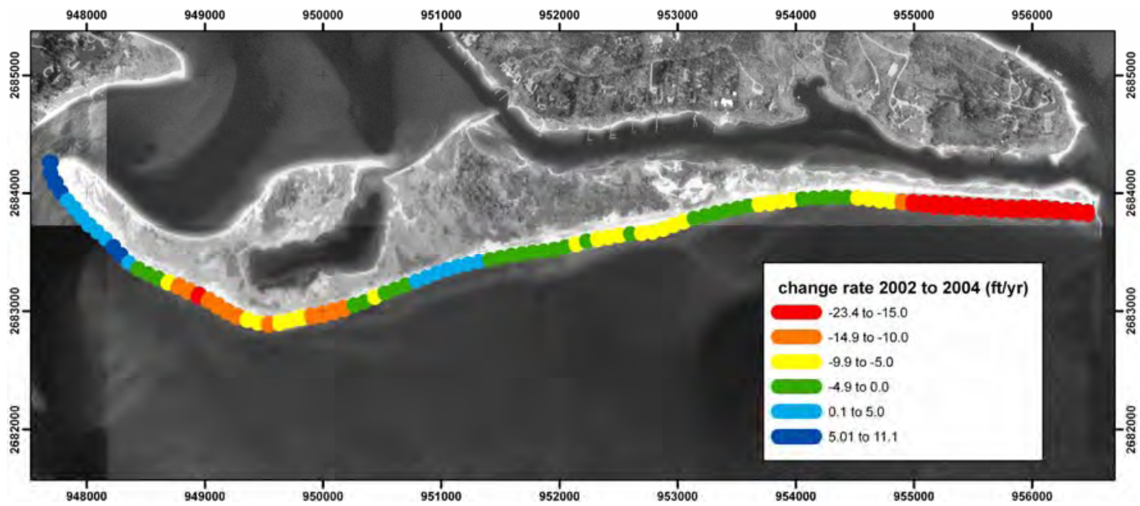


Figure V-7. Results of shoreline change analysis using the 2002 and 2004 GPS shorelines. Color bars indicate a range of shoreline change computed along Dead Neck. Negative rates indicate erosion, and are represented by the colors green, yellow, orange and red. Areas of accreting shoreline are indicated by light and dark blue.

V.3 DATA COLLECTION AND ANALYSIS

The field data collection portion of this study was performed to characterize the physical properties of the Three Bays estuary. Bathymetry were collected throughout the system so that it could be accurately represented as a computer hydrodynamic model, and so that flushing rates could be determined for the system sub-embayments. In addition to the bathymetry, tide data were also collected at seven locations, to run the circulation model with real tides, and also to calibrate and verify its performance.

V.3.1 Bathymetry Data Collection

Bathymetry data in Three Bays were collected during October 2002. Supplemental bathymetry were also available from a February 2002 survey of North and South Coves in Grand Island. The October 2002 survey employed a bottom tracking Acoustic Doppler Current Profiler (ADCP) mounted on a 12 ft motor skiff. Positioning data were collected using a differential GPS. The survey design included gridded transects at roughly 400 ft spacings in the main embayments, and finer spacings at the inlets. Survey paths are shown in Figure V-8. The resulting bathymetric surface created by interpolating the data to a finite element mesh is shown in Figure V-9. All bathymetry was tide corrected, and referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29), using survey benchmarks located in the project area.

Results from the survey show that the deepest point in the Three Bays system is located at Cotuit Inlet, and is –23.6 ft NAVD. The greatest depths (approximately 15 ft) of Cotuit Bay are located in its southern portion, near the village of Cotuit. The greatest depths of West Bay are in the dredged navigation channel (approximately 10 ft) from the West Bay inlet to the entrance to North Bay. The deepest depths of the entire Three Bays system (apart from the inlets) are located in North Bay (approximately 17 ft) near its entrance to Cotuit Bay.

V.3.2 Tide Data Collection and Analysis

Tide data records were collected at seven stations in the Three Bays estuary: 1) offshore Dead Neck, 2) West Bay (Grand Island), 3) Cotuit Bay (Bluff Point), 4) North Bay (Oyster Harbors Marina), 5) North Bay (Point Isabella), 6) Dam Pond, and 7) Prince Cove. An eighth tide gauge had been deployed in Cotuit Bay of off Handy Point, but data were not recovered from this location due to failure of the gauge. The locations of the stations are shown in Figure V-8. The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 44-day period between October 2, 2002 and November 15, 2002. The elevation of each gauge was surveyed relative to NGVD 29. Two gauges were deployed together offshore Dead Neck by SCUBA divers using a screw anchor. Duplicate offshore gauges were deployed to ensure data recovery, since the offshore tide record is crucial for developing the open boundary condition of the hydrodynamic model of the Three Bays system. Data from the other six locations were used to calibrate the model.

Plots of the tide data from three representative gauges are shown in Figure V-10, for the entire 44-day deployment. The spring-to-neap variation in tide can be seen in these plots. From the plot of the data from offshore Dead Neck, the tide reaches its maximum spring tide range of approximately 4.0 feet around October 7, and about seven days later the neap tide range is much smaller, as small as 1.5 feet. The second spring tide should occur around October 21, but the tide range is not clearly larger than either seven days before or after this date. The largest spring tide range is expected to occur at the time of the new moon, which occurred October 6 and again on November 4. The muted spring tide of October 21 occurred during the full moon. The causes of this odd feature of the tide in this are discussed from the results of the harmonic analysis later in this section.

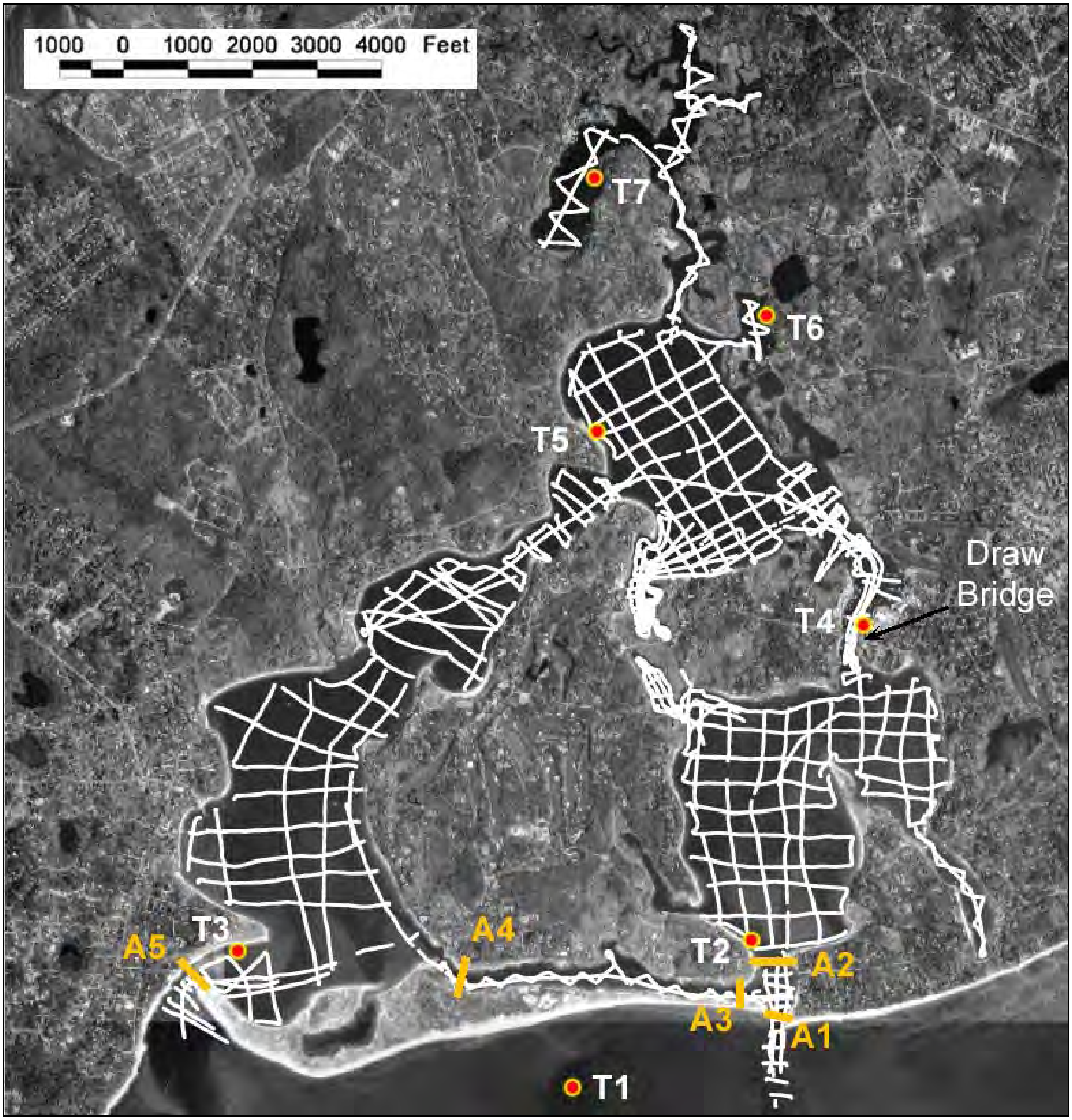


Figure V-8. Transects from the bathymetry survey of the Three Bays system. Yellow markers show the locations of the three tide recorders deployed for this study.

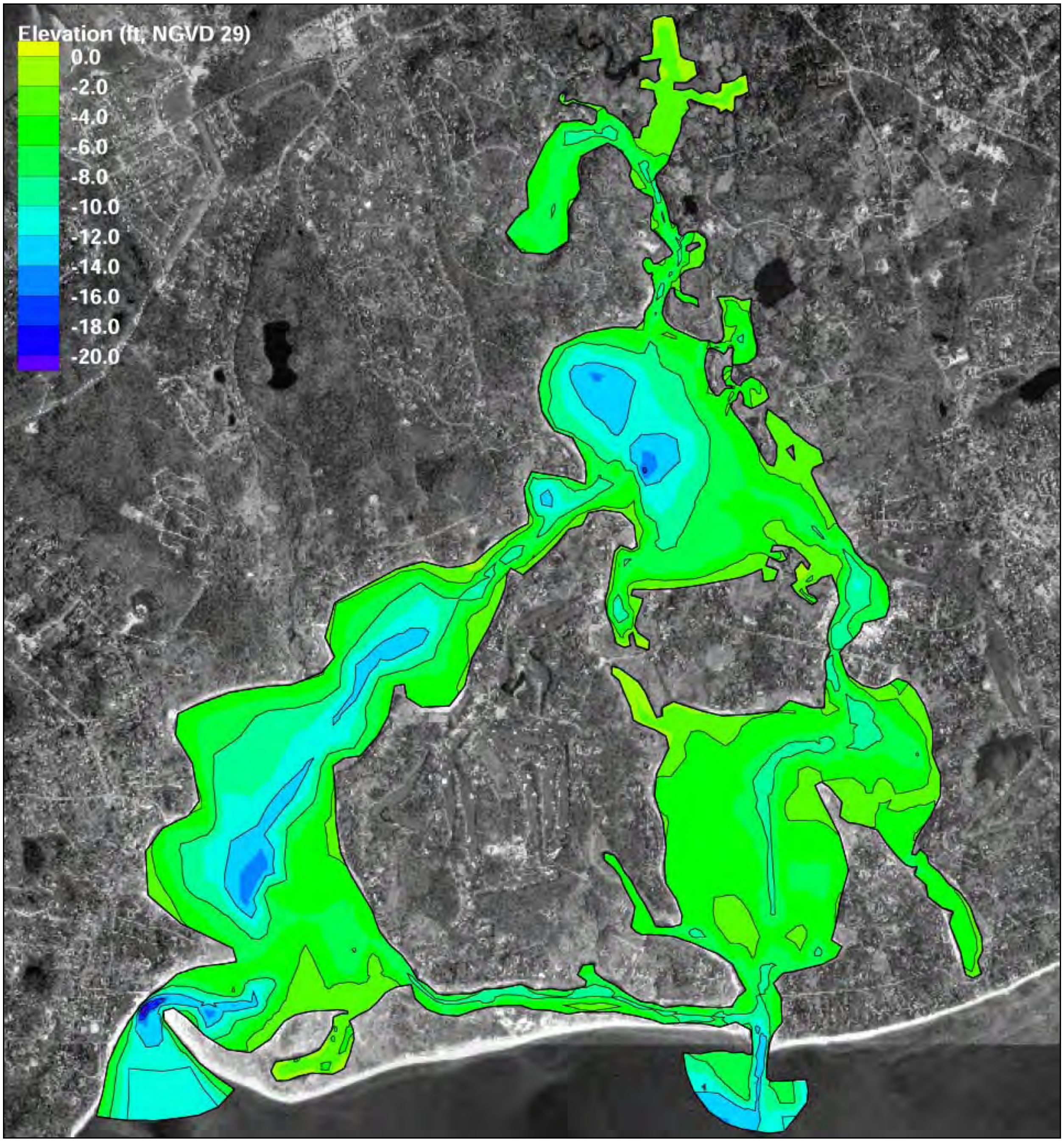


Figure V-9. Plot of interpolated finite-element grid bathymetry of the Three Bays system, shown superimposed on 1994 aerial photos of the system locale. Bathymetric contours are shown in color at two-foot intervals, and also as lines at four-foot intervals.

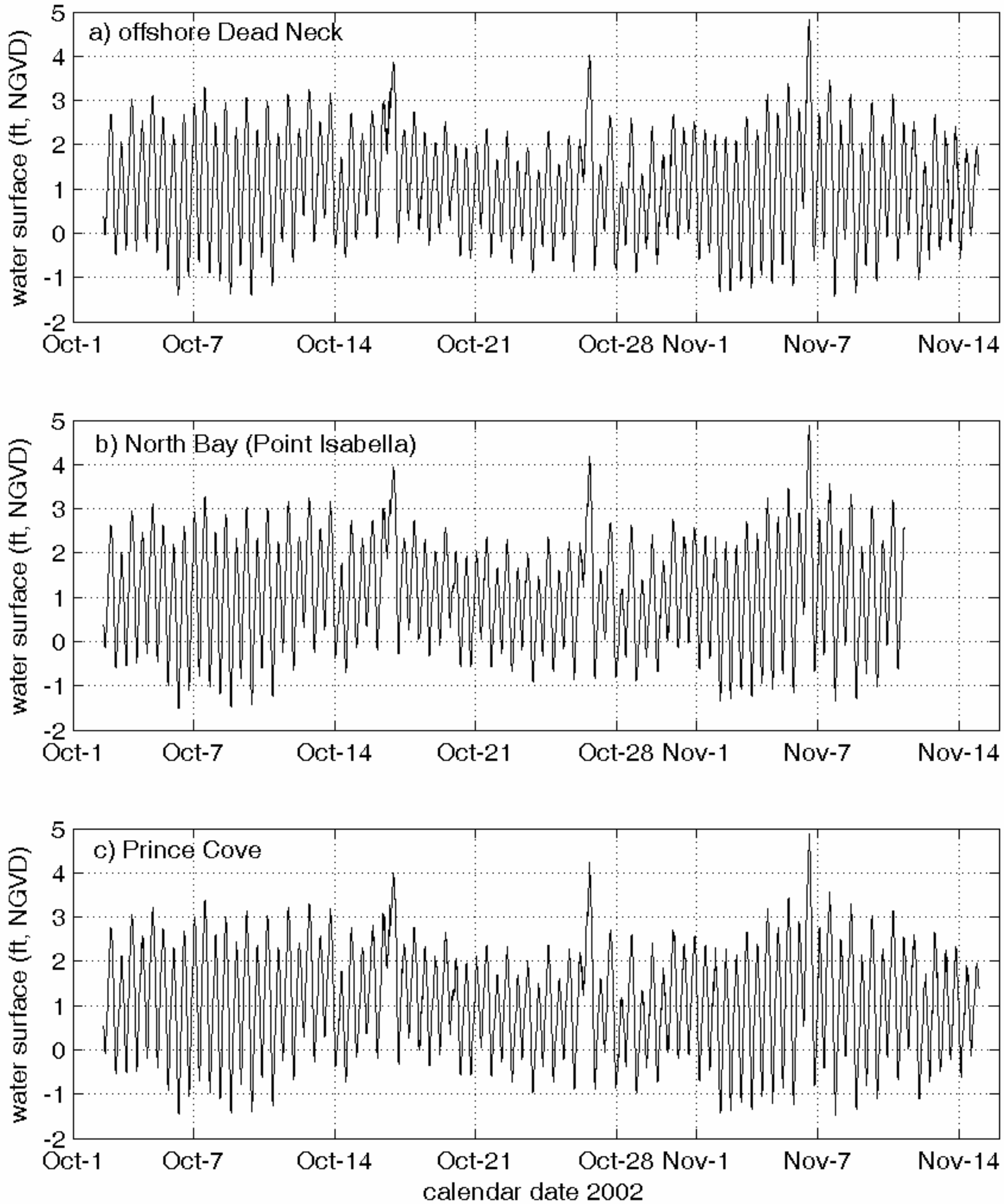


Figure V-10. Plots of observed tides for the Three Bays system, for the 44-day period between October 2 and November 15, 2002. The top plot shows tides offshore Dead Neck, in Nantucket Sound. The middle plot shows tides recorded in North Bay at Point Isabella, and the bottom plot shows tides recorded at Prince Cove, in the upper reaches of the Three Bays system. All water levels are referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

Also seen in this record are three distinct storm events, with peak storm surge levels occurring October 16, October 26, and November 6. Though the water level at peak surge is not substantially higher than the apparent normal peak spring tide levels, these surges stand out because of the relatively small tide range in this area of Nantucket Sound.

A visual comparison in Figure V-11 between tide elevations at the three stations shows that there negligible reduction in the tide range in the upper reaches of the Three Bays system. The loss of amplitude with distance from the inlet is described as tidal attenuation. Frictional mechanisms dissipate tidal flow energy, resulting in a reduction of the height of the tide. Tide attenuation is accompanied by a time delay (or phase lag) in the time of high and low tide (relative to the offshore tide), which becomes more pronounced farther into an estuary. The tide lag greatest in Prince Cove, as seen in Figure V-10, where low tide in this sub-embayment occurs approximately 50 minutes after low tide in Nantucket Sound.

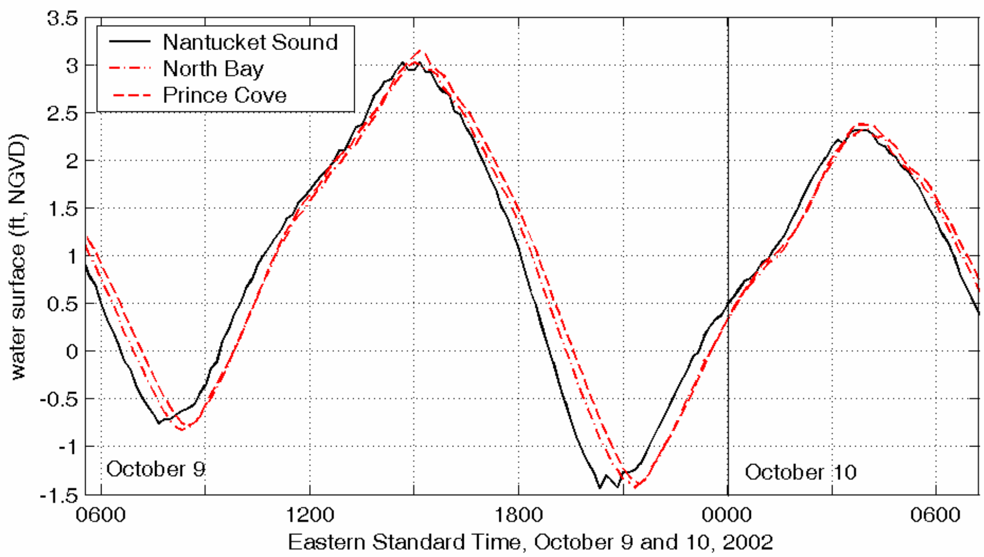


Figure V-11. Plot showing two tide cycles tides at three stations in the Three Bays system plotted together. Demonstrated in this plot is the minor frictional damping effect caused by flow restrictions at the inlets. The damping effects are seen only as a lag in time of high and low tides from Nantucket Sound. The time lag of low tide between the Sound and Prince Cove in this plot is 50 Minutes.

Standard tide datums were computed from the 44-day records. These datums are presented in Table V-1. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available, however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW. The lack of tide attenuation through the Three Bays estuary is apparent by how there is essentially no change in the elevation of each of the datums, from Nantucket Sound to Prince Cove. This is true for even the maximum and minimum tide levels.

Table V-1. Tide datums computed from 44-day records collected offshore Dead Neck and in Cotuit Bay, West Bay, and Price Cove. Datum elevations are given relative to NGVD 29.

Tide Datum	Offshore	Cotuit Bay	West Bay	Prince Cove
Maximum Tide	4.8	4.8	4.7	4.9
MHHW	2.8	2.8	2.8	2.9
MHW	2.4	2.4	2.4	2.4
MTL	1.0	1.0	1.0	1.0
MLW	-0.4	-0.4	-0.4	-0.5
MLLW	-0.7	-0.7	-0.7	-0.8
Minimum Tide	-1.4	-1.5	-1.4	-1.6

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

A more thorough harmonic analysis of the tidal time series was performed to produce tidal amplitude and phase of the major tidal constituents, and provide assessments of hydrodynamic 'efficiency' of the system in terms of tidal attenuation. This analysis also yielded a quantitative assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of 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 is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-12. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of eight tidal constituents in the Three Bays system.

The M_2, or the familiar twice-a-day lunar semi-diurnal tide, is the strongest contributor to the signal with an amplitude of 1.2 ft throughout the system. The total range of the M_2 tide is twice the amplitude, or 2.4 ft. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 has an amplitude of 0.2 feet in all sub-embayments of the system. The M_6 has a very small amplitude in the system (less than 0.1 feet). There is no change in the M_2 or its harmonics through the estuary, which is a further indication that friction losses in the system are minimal, and that Three Bays flushes very efficiently, even to its farthest reaches at Prince Cove.

The other major tide constituents also show little variation across the system. The diurnal tides (once daily), K_1 and O_1, possess equal amplitudes of approximately 0.4 feet. Other semi-diurnal tides, the S_2 (12.00 hour period) and N_2 (12.66-hour period) tides, contribute significantly to the total tide signal, with amplitudes of 0.2 feet and 0.4 feet, respectively. The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon, and has an amplitude less than 0.1 ft.

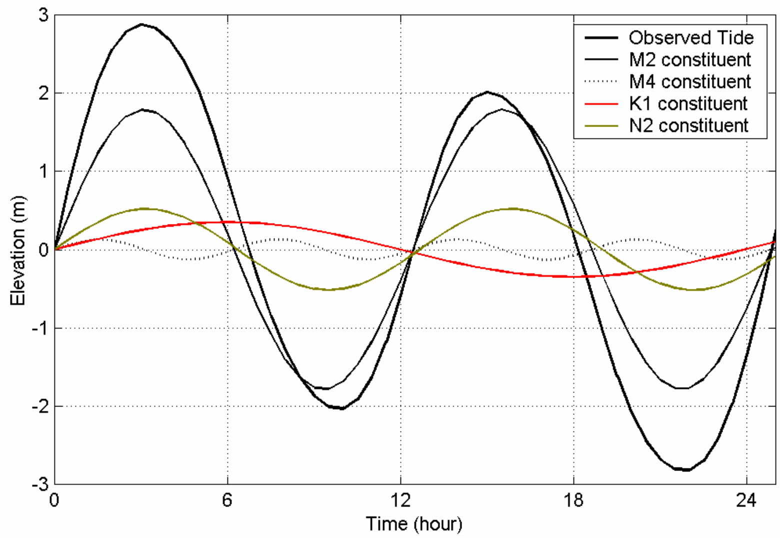


Figure V-12. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Major tidal constituents determined for gauge locations in Three Bays, October 2 through November 14, 2002.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Nantucket Sound	1.20	0.17	0.07	0.15	0.41	0.35	0.35	0.06
Cotuit Bay	1.19	0.15	0.06	0.14	0.42	0.35	0.35	0.05
West Bay	1.20	0.15	0.07	0.15	0.41	0.35	0.35	0.06
Oyster Harbor Marina	1.20	0.14	0.08	0.15	0.41	0.35	0.35	0.05
North Bay	1.19	0.14	0.09	0.15	0.41	0.35	0.35	0.09
Dam Pond	1.17	0.14	0.08	0.14	0.40	0.35	0.35	0.05
Prince Cove	1.20	0.14	0.09	0.15	0.41	0.35	0.35	0.05

Though there is no change in constituent amplitudes, 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 Three Bays system, relative to the timing of the M_2 constituent in Nantucket Sound, offshore Dead Neck. The greatest delay is at the Dam Pond TDR station, which also showed the largest reduction of the M_2 amplitude (Table V-2). Compared to other locations instrumented in this study, Dam Pond shows the greatest tidal attenuation.

Table V-3. M_2 tidal constituent phase delay (relative to Nantucket Sound) for gauge locations in the Three Bay system, determine from measured tide data.

Station	Delay (minutes)
Cotuit Bay (Bluff Point)	7.1
West Bay	10.1
Oyster Harbor Marina (Draw Bridge)	15.3
North Bay (Point Isabella)	12.0
Dam Pond	23.1
Prince Cove	20.5

Results of the harmonic analysis provide the reason why the transition from spring to neap tide ranges is not as apparent, as it is at other areas (e.g., Cape Cod Bay), as discussed earlier. The cause of the mute transition between spring and neap tide ranges is the relatively large amplitudes of the N_2 (larger lunar elliptic semidiurnal constituent) and O_1 (lunar diurnal constituent) constituents. From the analysis of other tide records from around southeastern Massachusetts, the N_2 has a typical amplitude that is less than 10% of the total tide, and the O_1 is typically less than 7% of the total tide amplitude. At Three Bays however, the N_2 and O_1 have much larger amplitudes relative to the total tide, at 15% and 13%, respectively. These constituents are slightly out of phase with the M_2 and K_1 (normally the greater contributors to the total tide amplitude), and therefore add and subtract from the total observed tide signal in cycles that are different (longer) than the 7 lunar day transition from spring to neap tides. In other areas (again, like Cape Cod Bay), the N_2 and O_1 represent a smaller percentage of the total observed tide, so their effect on the observed tide would be smaller.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large compared to tidal flow. The results of an analysis to determine the energy distribution (or variance) of the original water elevation time series for the Three Bays system is presented in Table V-4 compared to the energy content of the astronomical tidal signal (re-created by summing the contributions from the 23 constituents determined by the harmonic analysis). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. Figure V-13 shows the comparison of the measured tide from Nantucket Sound, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-4. Percentages of Tidal versus Non-Tidal Energy for Three Bays embayments, August to September 2001.

TDR LOCATION	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Nantucket Sound (offshore)	1.048	86.9	13.1
Cotuit Bay (Bluff Point)	1.031	85.5	14.5
West Bay	1.035	86.6	13.4
Oyster Harbor Marina (North Bay)	1.042	86.3	13.7
North Bay (Point Isabella)	1.070	85.7	14.3
Dam Pond	1.001	85.8	14.2
Prince Cove	1.042	85.9	14.1

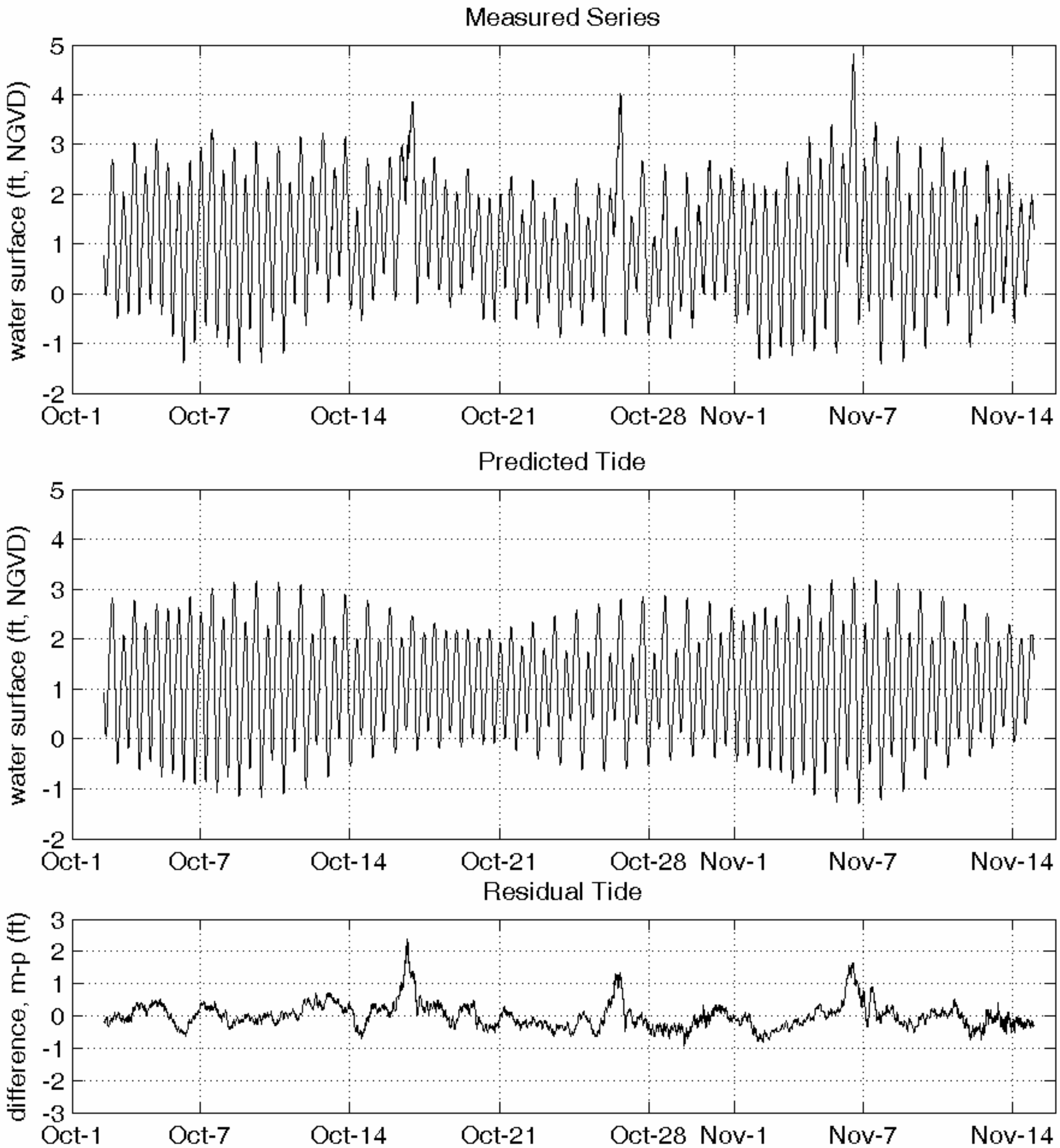


Figure V-13. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determine in the harmonic analysis of the Nantucket Sound (offshore Dead Neck) gauge data. The residual tide shown in the bottom plot is computed as the difference between the measured and predicted time series (r=m-p).

Table V-4 shows that the variance of tidal energy was essentially equal in all parts of the system; as should be expected given the minimal tidal attenuation through the system. The analysis also shows that tides are responsible for approximately 86% of the water level changes in Three Bays system. The remaining 14% was the result of atmospheric forcing, due to winds, or barometric pressure gradients. The largest tide residuals occurred at the three dates discussed earlier, October 16, October 26 and November 6. These are storm-induced surges caused by low pressure fronts moving through the area at those times, as indicated in regional meteorological data records.

V.3.3 ADCP Data Analysis

Cross-channel current measurements were surveyed through a complete tidal cycle in the Three Bays system on October 24, 2002 to resolve spatial and temporal variations in tidal current patterns. The survey was designed to observe tidal flow across five transects in the system at hourly intervals. These transects (indicated in Figure V-8) were located at the two system inlets, in the Seapuit River, as well as the entrance to West Bay, north of the eastern end of the Seapuit River. The data collected during this survey provided information that was necessary to model properly the hydrodynamics of the Three Bays system.

Figures V-15 through V-23 show color contours of the current measurements observed during the flood and ebb tides at each of the three transects. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. For example, at West Bay inlet, positive along-channel flow is to the north, and positive cross-channel flow is moving to east. In Figure V-14, the lower left panel shows depth-averaged currents across the channel projected onto a 1994 aerial photograph of the inlet. The lower right panel of each figure indicates the stage of the tide that the survey transect was taken by a vertical line through the water elevation curve.

At the inlets, maximum measured currents in the water column were between 2.5 and 3.0 ft/sec (1.5 and 1.8 knots), with the larger velocities occurring at West Bay inlet. Flow rates (computed using the ADCP velocity data) through both inlets were roughly the same during the measured tide cycle. Maximum flood flows in the morning of the October 24 were 3000 ft^3/sec at West Bay inlet, and 3200 ft^3/sec at Cotuit Bay inlet. In the afternoon, maximum ebb flows were 3500 ft^3/sec and 4700 ft^3/sec at West Bay inlet and Cotuit Bay inlet, respectively.

The ADCP data from the two Seapuit River transects were critical for the determination of the direction and magnitude of tidal flows within the river. The data show that as the tide floods into Cotuit Bay and West Bay inlets, the river flows from east to west. During the ebb stage of the tide, the flow direction in the river reverses, flowing to the east. Further analysis (Section 3) shows that the flow through the Seapuit River is driven by a slight difference in the timing of the stage of the tide between West Bay inlet and Cotuit Bay inlet.

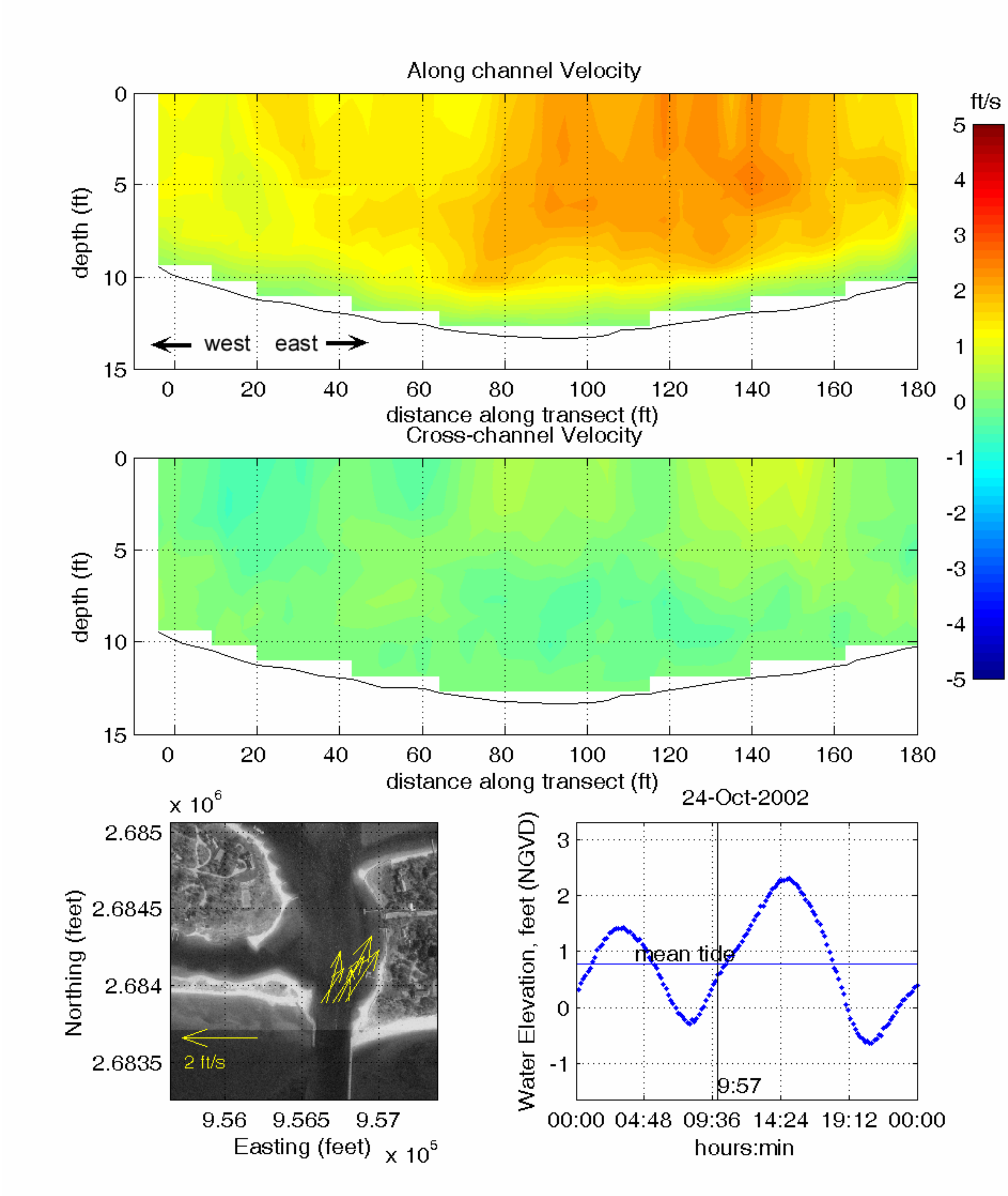


Figure V-14. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across West Bay inlet measured at 9:57 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

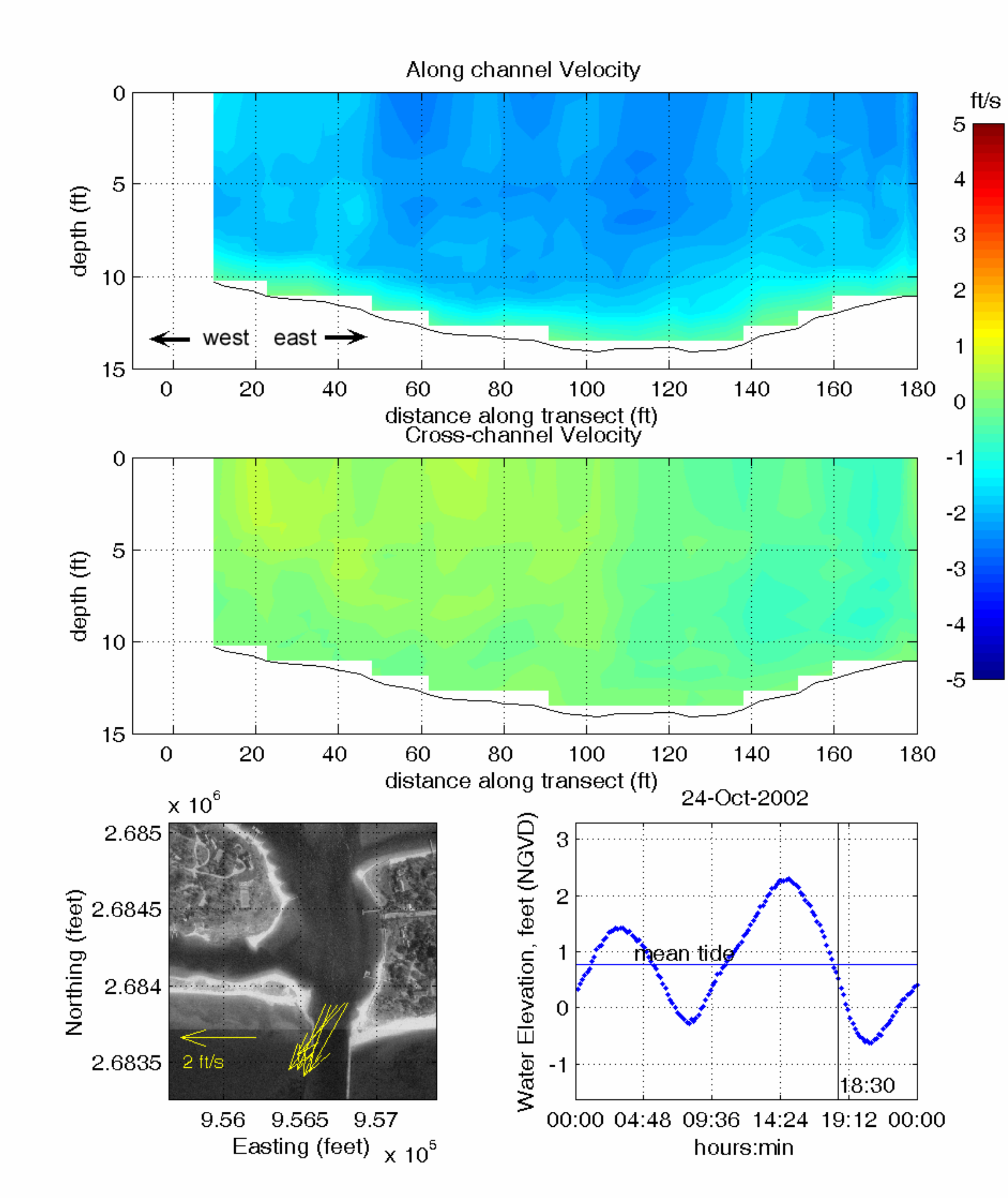


Figure V-15. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across West Bay inlet measured at 18:30 on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

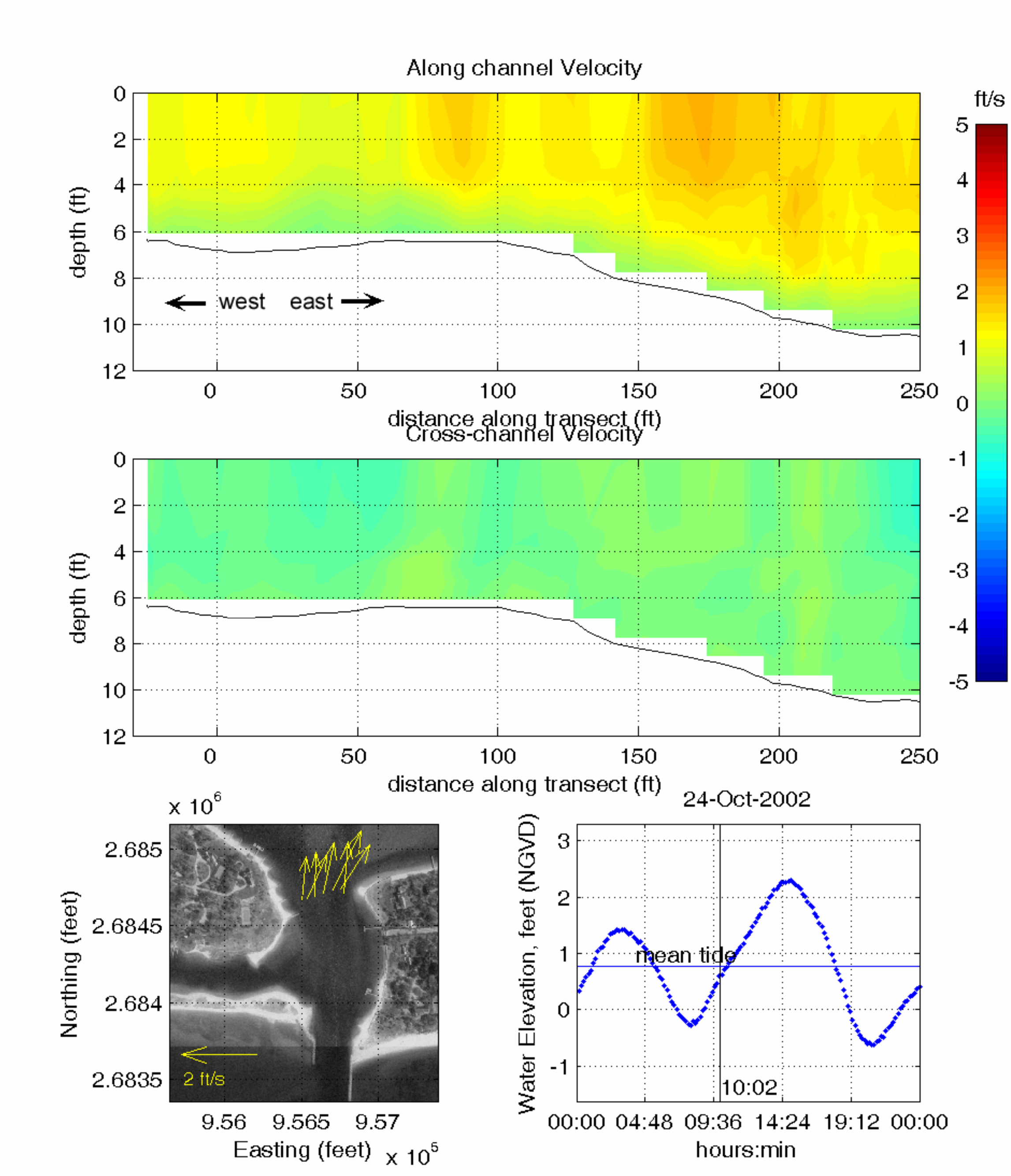


Figure V-16. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the entrance to West Bay, measured at 10:02 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

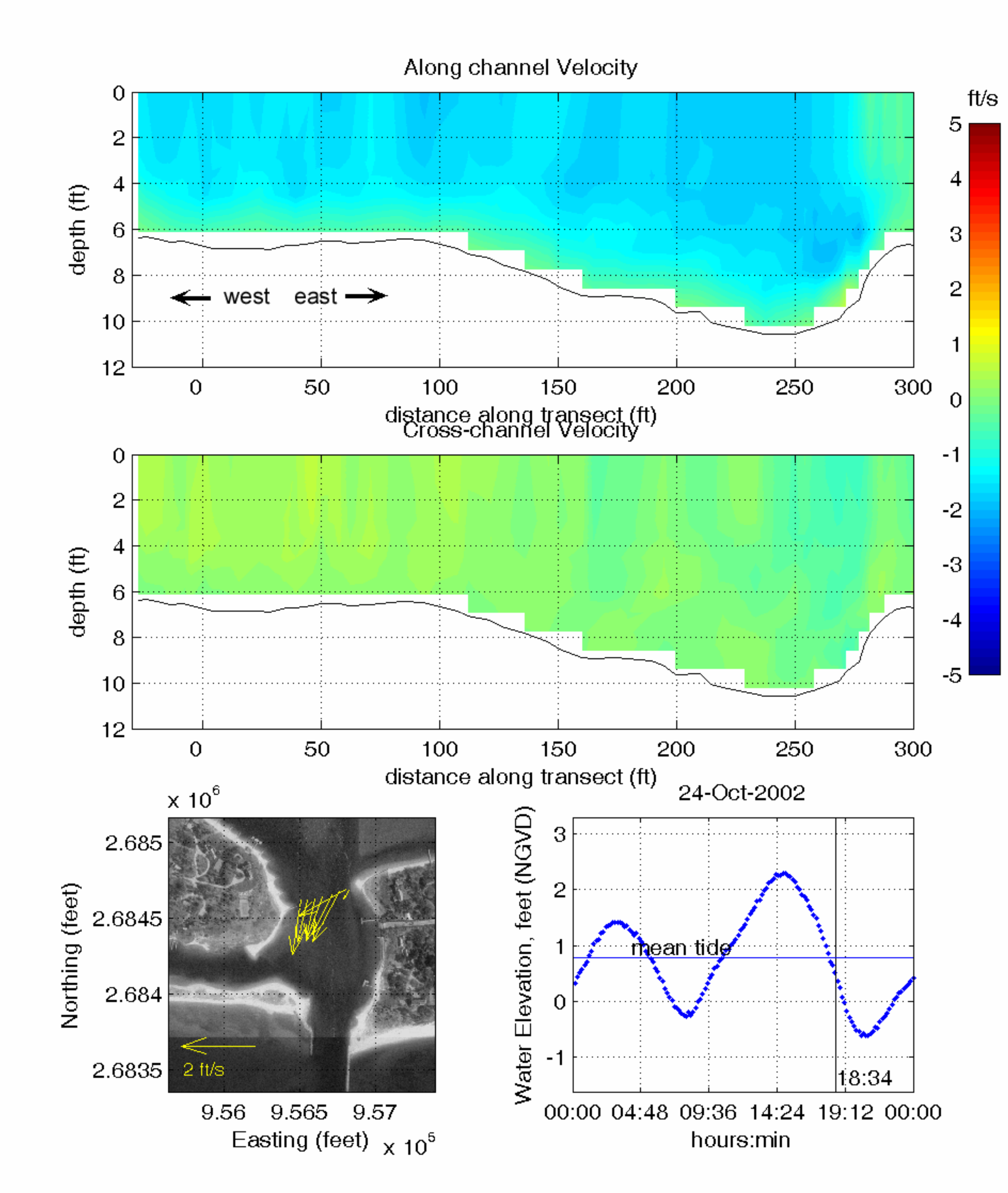


Figure V-17. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the entrance to West Bay, measured at 18:34 on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

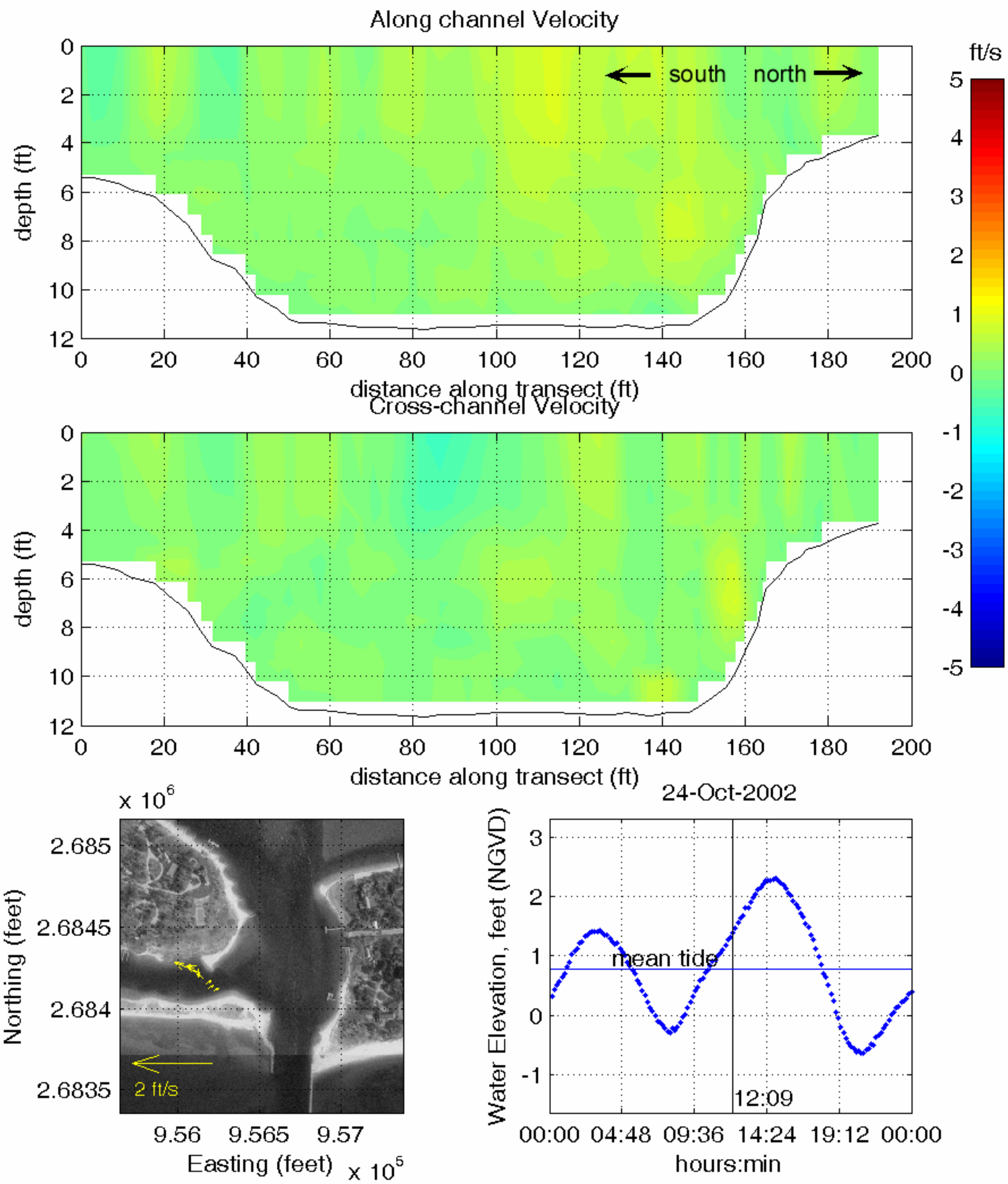


Figure V-18. Color contour plots of along-channel and cross-channel velocity components for transect line run north-to-south across the eastern end of the Seapuit River, measured at 12:09 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

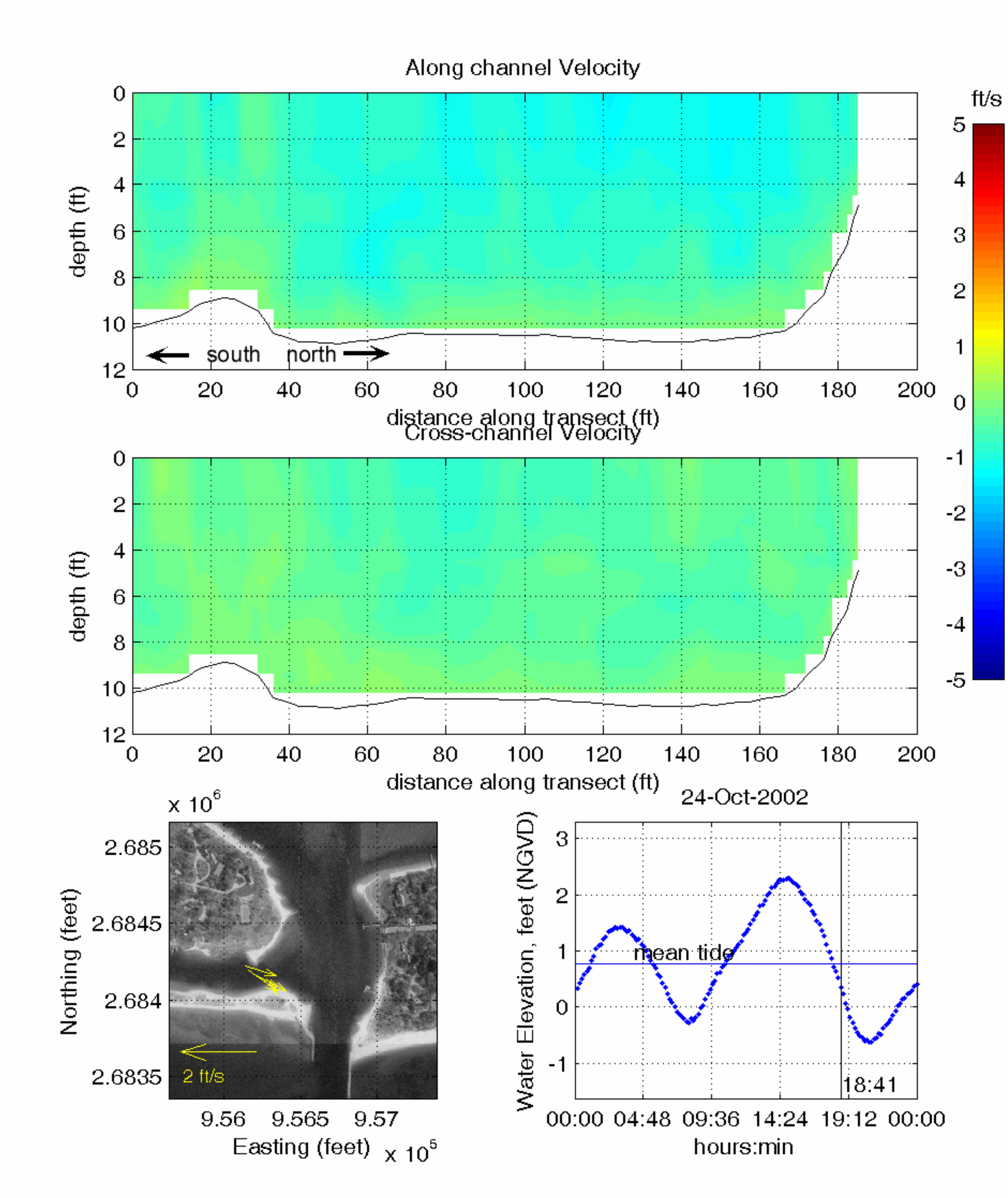


Figure V-19. Color contour plots of along-channel and cross-channel velocity components for transect line run east-to-west across the eastern end of the Seapuit River, measured at 18:41on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into Cotuit Bay from the West Bay inlet, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

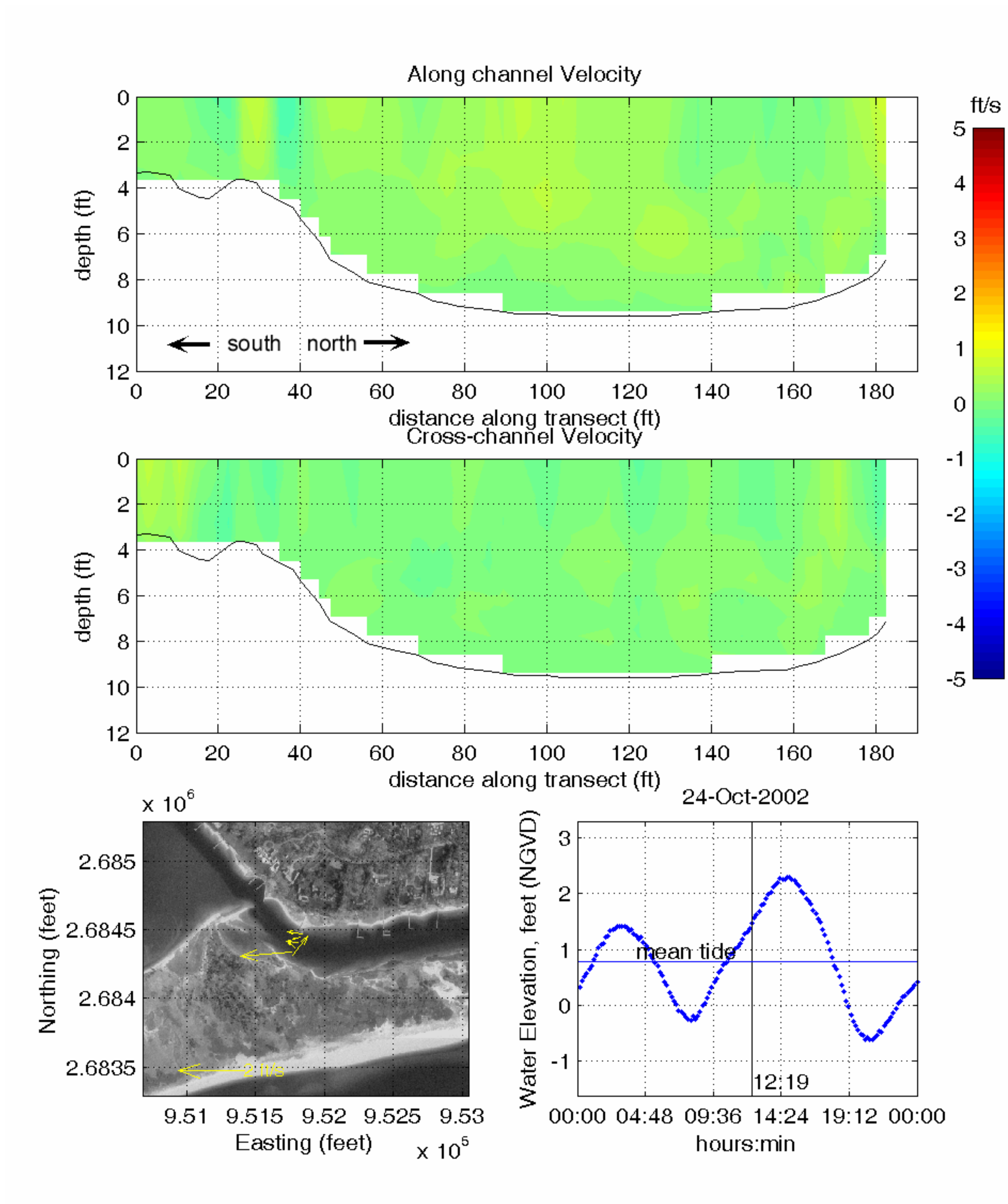


Figure V-20. Color contour plots of along-channel and cross-channel velocity components for transect line run north-to-south across the western end of the Seapuit River, measured at 12:19 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into Cotuit Bay from the West Bay inlet, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

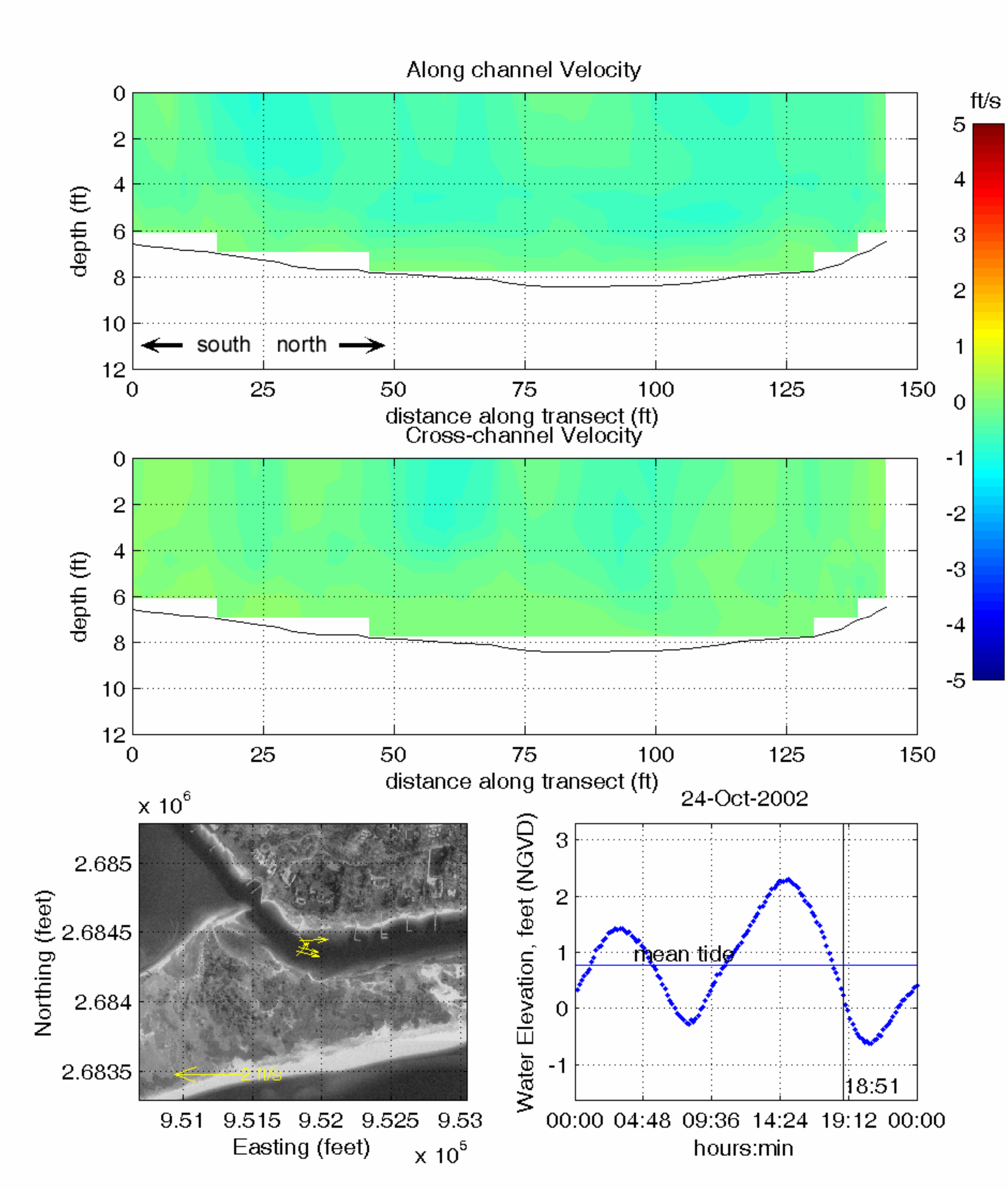


Figure V-21. Color contour plots of along-channel and cross-channel velocity components for transect line run north-to-south across the western end of the Seapuit River, measured at 18:51 on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into Cotuit Bay from the West Bay inlet, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

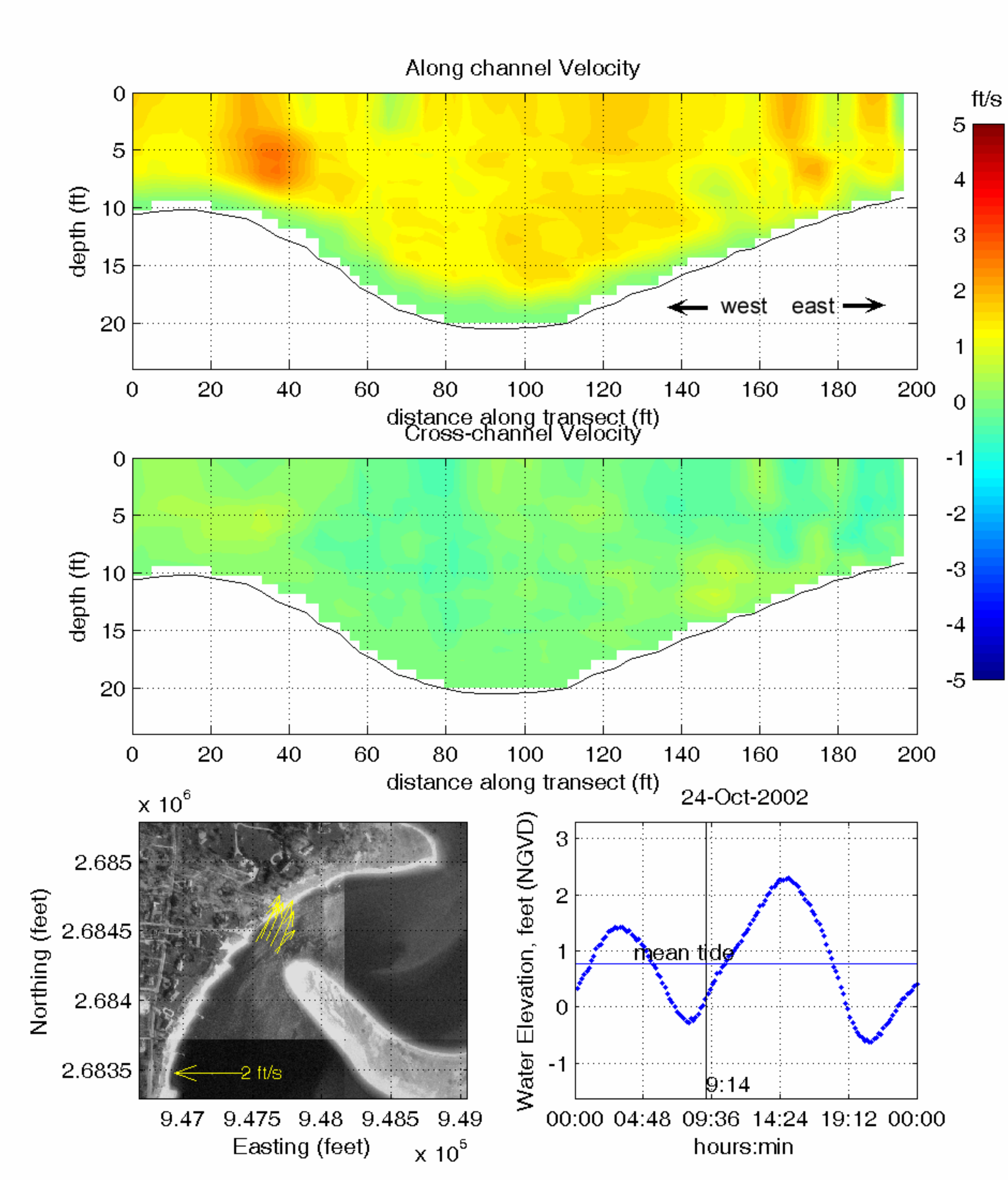


Figure V-22. Color contour plots of along-channel and cross-channel velocity components for transect line run west-to-east across Cotuit Bay Inlet, measured at 9:14 on October 24, 2001 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

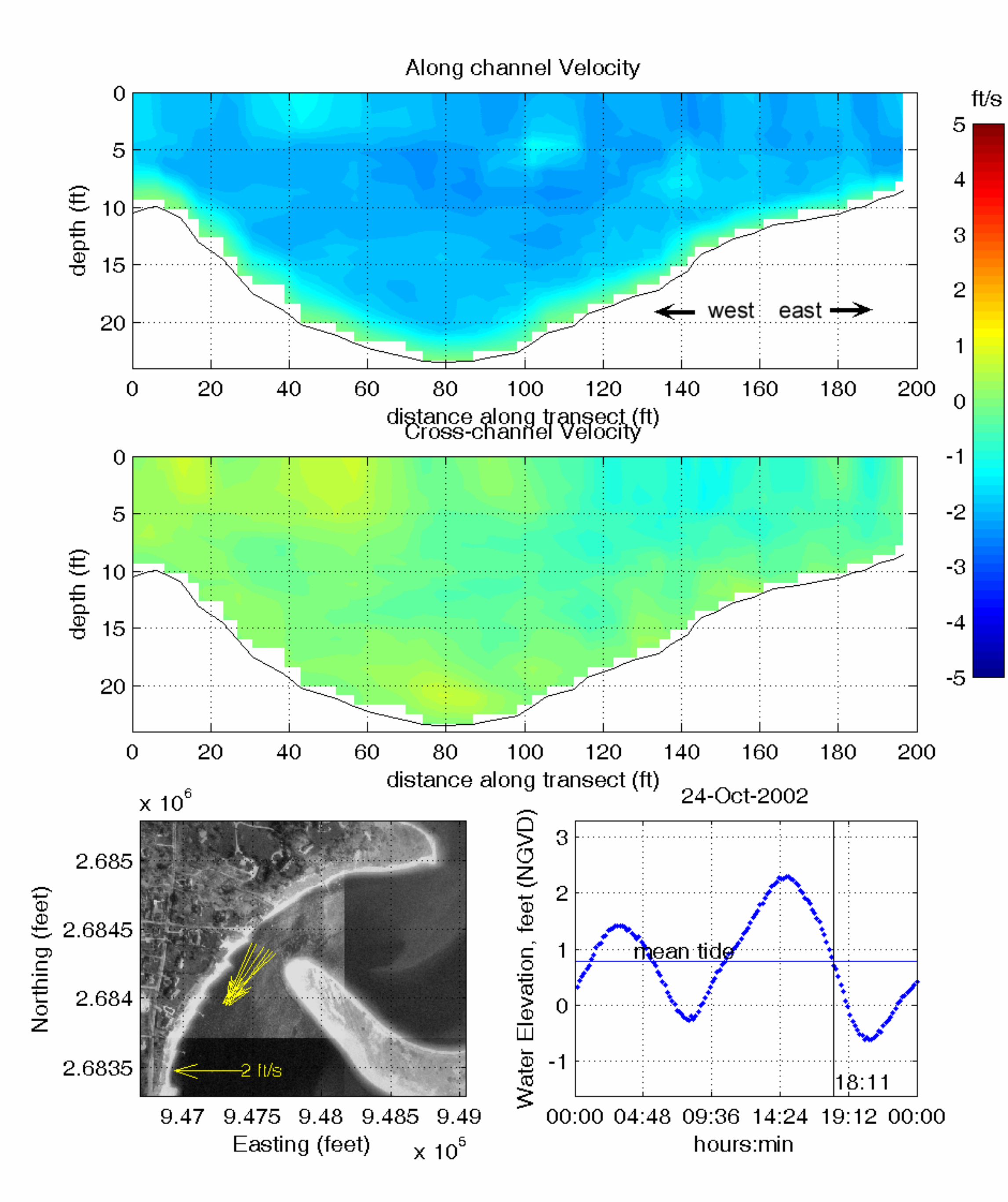


Figure V-23. Color contour plots of along-channel and cross-channel velocity components for transect line run west-to-east across Cotuit Bay Inlet, measured at 18:11 on October 24, 2001 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, while positive cross-channel velocities (middle panel) are oriented 90° clockwise of positive along-channel. Lower left plot shows scaled velocity vectors projected onto a 1994 aerial photo of the survey area. A tide plot for the survey day is also given.

V.4 HYDRODYNAMIC MODELING

For the modeling of the Three Bays system, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in these systems. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2 for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, Chatham embayments (Kelley, *et al*, 2001), Falmouth "finger" Ponds (Ramsey, *et al*, 2000), and Barnstable Harbor (Wood, *et al*, 1999).

V.4.1 Model Theory

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

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

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

V.4.2 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 1994 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrances of the Three Bays system (i.e., Cotuit Bay Inlet and West Bay Inlet) based on the tide gauge data collected offshore Dead Neck, in Nantucket Sound. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several (10) model calibration simulations for each system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.4.2.1 Grid generation

The grid generation process was aided by the use of the SMS package. 1994 digital aerial orthophotos and recent bathymetry survey data were imported to SMS, and a finite element grid was created to represent the estuary. The aerial photographs were used to determine the land boundary of the system. Bathymetry data were interpolated to the developed finite element mesh of the system. The completed grid consists of 5,190 nodes, which describe 1766 total 2-dimensional (depth averaged) quadratic elements, and covers 1337 acres. The maximum nodal depth is –22.9 ft (NGVD 29), in the throat of Cotuit Inlet. The completed grid mesh of the Three Bays system is shown in Figure V-24, and grid bathymetry was shown previously in Figure V-9.

The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties throughout the Three Bays embayments. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in tidal creeks and channels was designed to provide a more detailed analysis in these regions of rapidly varying flow (e.g., the Cotuit Inlet, West Bay Inlet, and the Marstons Mills River). Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in the main bodies of Cotuit Bay, West Bay, North Bay, and Prince Cove. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

V.4.2.2 Boundary condition specification

Two types of boundary conditions were employed for the RMA-2 model of the Three Bays system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Tidal boundary conditions were specified at both inlets from Nantucket Sound, Cotuit Bay Inlet and West Bay Inlet. TDR measurements from a gauge deployed offshore Dead Neck provided the required data.

The rise and fall of the tide in Nantucket Sound is the primary driving force for estuarine circulation in this system. Dynamic (time-varying) model simulations specified a new water surface elevation at Cotuit Inlet and West Bay Inlet every model time step of 10 minutes, which corresponds to the time step of the TDR data measurements. A time lag of 3 minutes was added to the tidal boundary condition at Cotuit inlet (lagging relative to West Bay Inlet) to account for the propagation of the tide from east to west. Over the 1.7-mile distance between the two inlets, the slight difference in tidal phase subtly affects the flow dynamics within the Three Bays system. Results from the RMA-2 model of the system show that the phase difference drives the flow through the Seapuit River. The 3 minute lag was determined based on the typical difference in time of high and low tides between Hyannisport and Cotuit, as publish in tide tables by NOAA. By this method, the tide propagates an average 2800 ft/min. Alternately, a similar result can be computed using the shallow wave equation $C = \sqrt{gh}$, where C is the speed of the tidal wave computed from the gravitational constant g, and the typical water depth *h*. This method results in a phase delay of 5 minutes. Model results discussed later in this section show that the 3-minute delay induces the correct magnitude and flow direction in the Seapuit River.

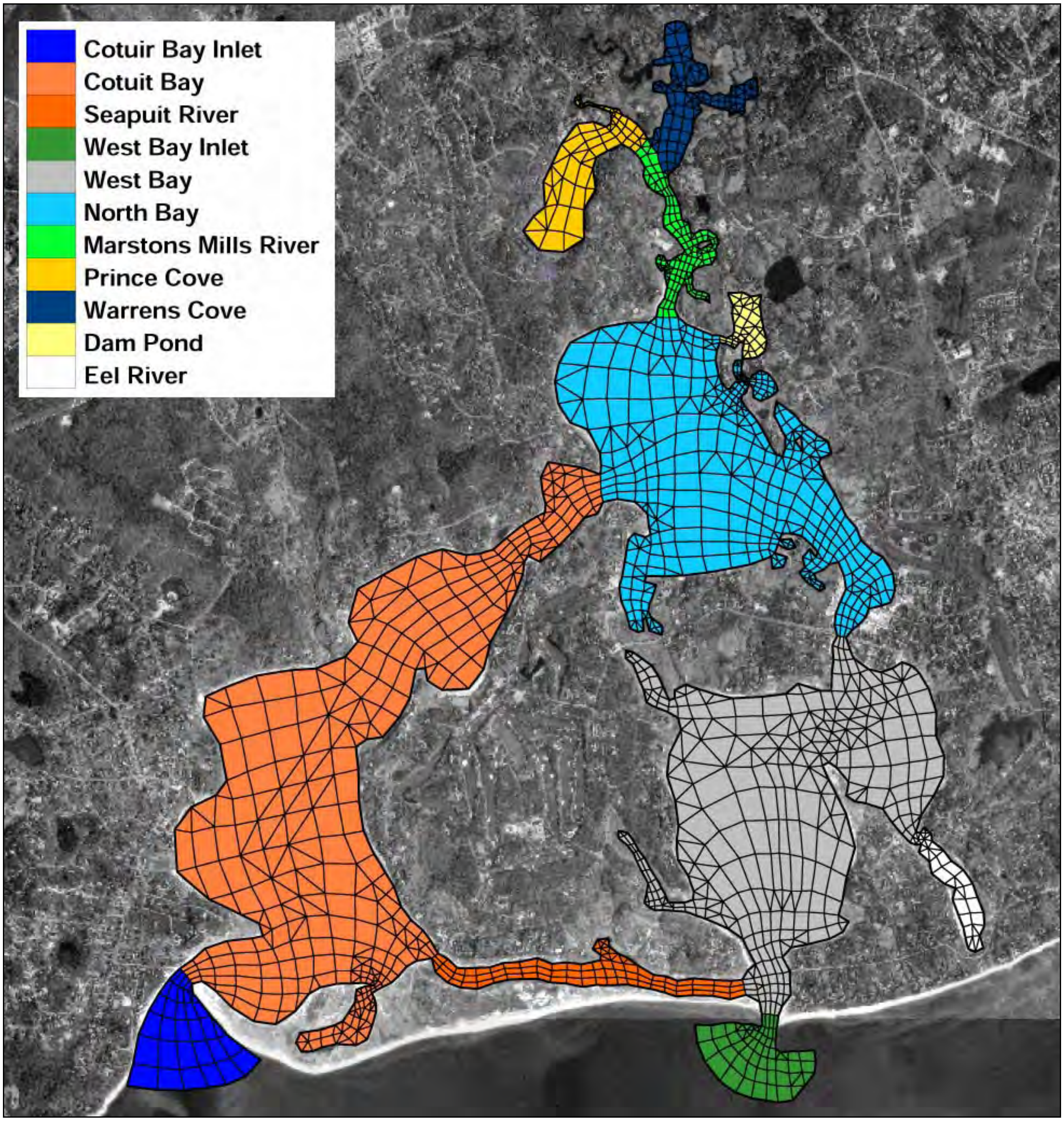


Figure V-24. Plot of hydrodynamic model grid mesh for the Three Bays system of Barnstable, MA. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.4.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Three Bays 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 required (typically 20+) for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the hydrodynamic model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, a five lunar-day period (10 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section 3.2. The five-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents.

The calibration was performed for a five-day period beginning October 8, 2002 1430 EST. This representative time period included the spring tide range of conditions, where the tide range and tidal currents are greatest, and model numerical stability is often most sensitive. To provide average tidal forcing conditions for model verification and the flushing analysis, a separate time period was chosen that spanned the transition between spring and neap tide ranges (bi-weekly maximum and minimum tidal ranges, respectively). For model verification and the flushing analysis, the 7 lunar-day period (14 tide cycles, or 7.25 solar days) beginning October 11, 2002 0000 EST was used.

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

V.4.2.3.1 Friction coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficient values of 0.03 were specified for all element material types. This values corresponds to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) (Henderson, 1966).

During calibration, friction coefficients were incrementally changed throughout the model domain. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within the estuary system. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-5.

Table V-5. Manning's Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-24.	
System Embayment	Bottom Friction
Cotuit Bay Inlet	0.030
Cotuit Bay	0.030
Seapuit River	0.025
West Bay Inlet	0.030
West Bay	0.030
North Bay	0.030
Marstons Mills River	0.035
Prince Cove	0.035
Warrens Cove	0.035
Dam Pond	0.030
Eel River	0.030

V.4.2.3.2 Turbulent exchange coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). Typically, model turbulence coefficients were set between 80 and 200 lb-sec/ft^2. In most cases, the Three Bays system was relatively insensitive to turbulent exchange coefficients. The exception was at the inlets, where higher exchange coefficient values (200 lb-sec/ft^2) were used to ensure numerical stability in these areas characterized by strong turbulent flows and large velocity magnitudes.

V.4.2.3.3 Wetting and Drying

Modeled hydrodynamics were complicated by wetting/drying cycles in shallow flats included in the model of the Three Bays system. A method was employed to simulate the periodic inundation and drying of tidal flats in the system. Nodal wetting and drying is a feature of RMA-2 that allows grid elements to be removed and re-inserted during the course of the model run. The wetting and drying feature has two keys benefits for the simulation, 1) it enhances the stability of the model by eliminating nodes that have bottom elevations that are higher than the water surface elevation at that time, and 2) it reduces total model run time because node elimination can reduce the size of the computational grid significantly during periods of a model run. Wetting and drying is employed for estuarine systems with relatively shallow borders and/or tidal flats.

V.4.2.3.4 Comparison of modeled tides and measured tide data

A best-fit of model predictions for the TDR deployment was achieved using the aforementioned values for friction and turbulent exchange. Figures V-25 through and V-31 illustrate the five-day calibration simulation along with a 50-hour sub-section. 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

(principle lunar semidiurnal constituent) was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6 for the calibration period differ from those in Table V-2 because constituents were computed for only the five-day section of the 38-days represented in Table W. Table V-6 compares tidal constituent amplitude (height) and relative phase (time) for modeled and measured tides at the TDR locations. The constituent phase shows the relative timing of each separate constituent at a particular location, and also the change (or phase lag) in timing of a single constituent at different locations in an estuary.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.01 ft, which is better than the order of accuracy of the tide gauges (±0.12 ft). Time lag errors were typically less than the time increment resolved by the model (1/6 hours or 10 minutes), indicating good agreement between the model and data.

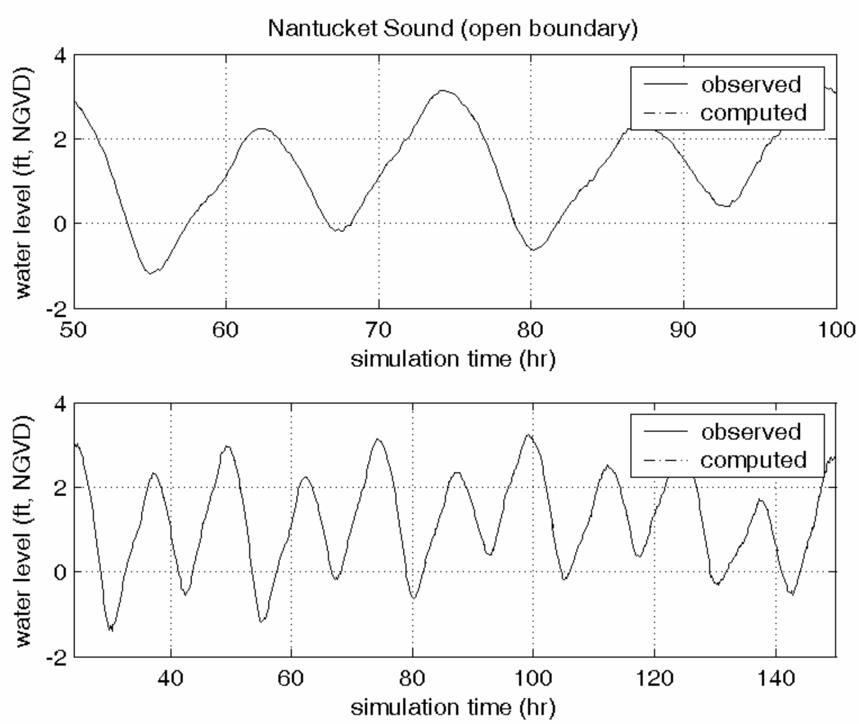


Figure V-25. Comparison of model output and measured tides for the TDR location offshore Dead Neck, in Nantucket Sound. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

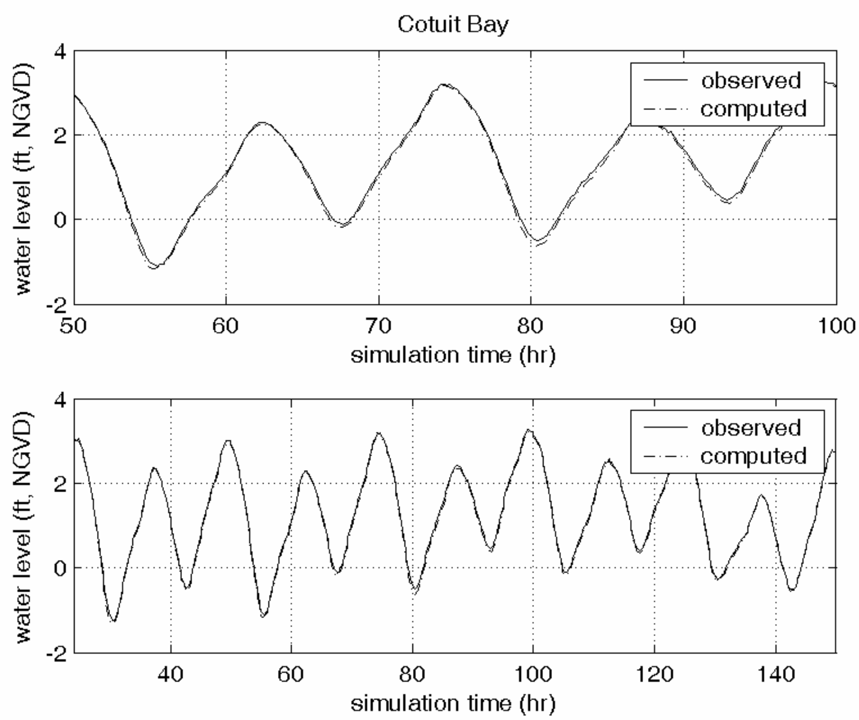


Figure V-26. Comparison of model output and measured tides for the TDR location in lower Cotuit Bay. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

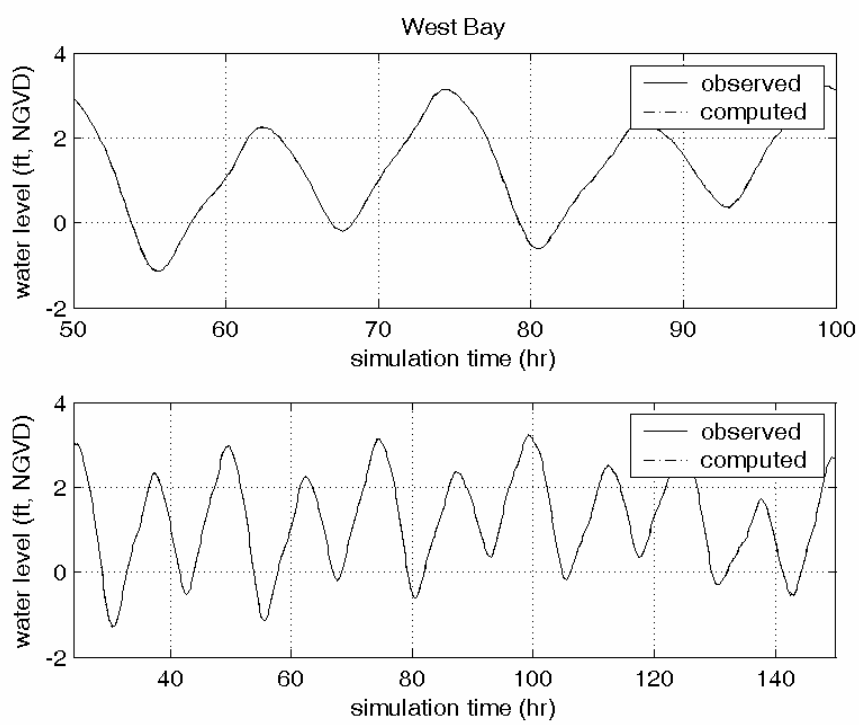


Figure V-27. Comparison of model output and measured tides for the TDR location in West Bay. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

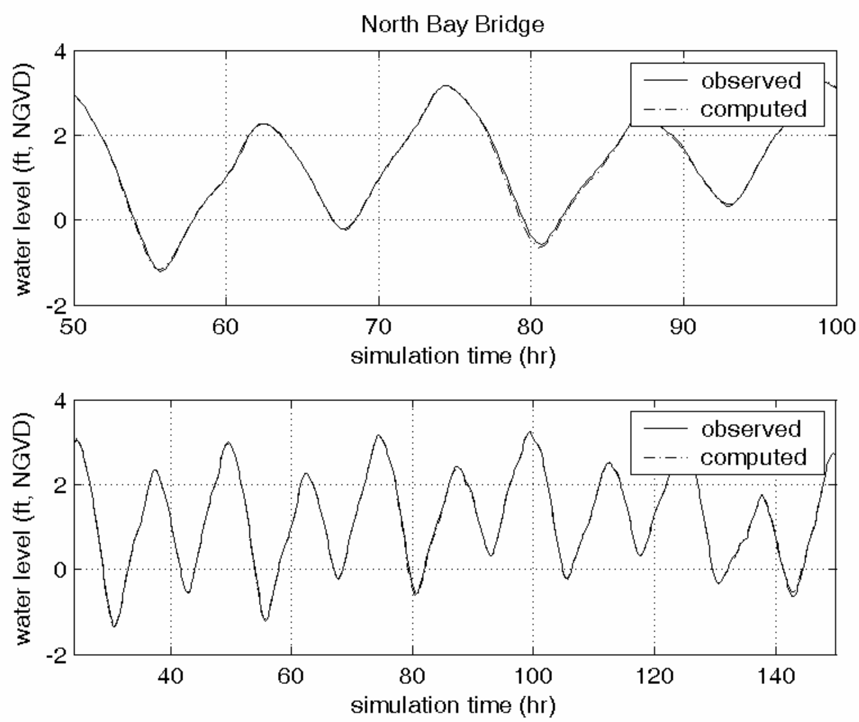


Figure V-28. Comparison of model output and measured tides for the TDR location at Oyster Harbors Marina, near the Little Island draw bridge. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

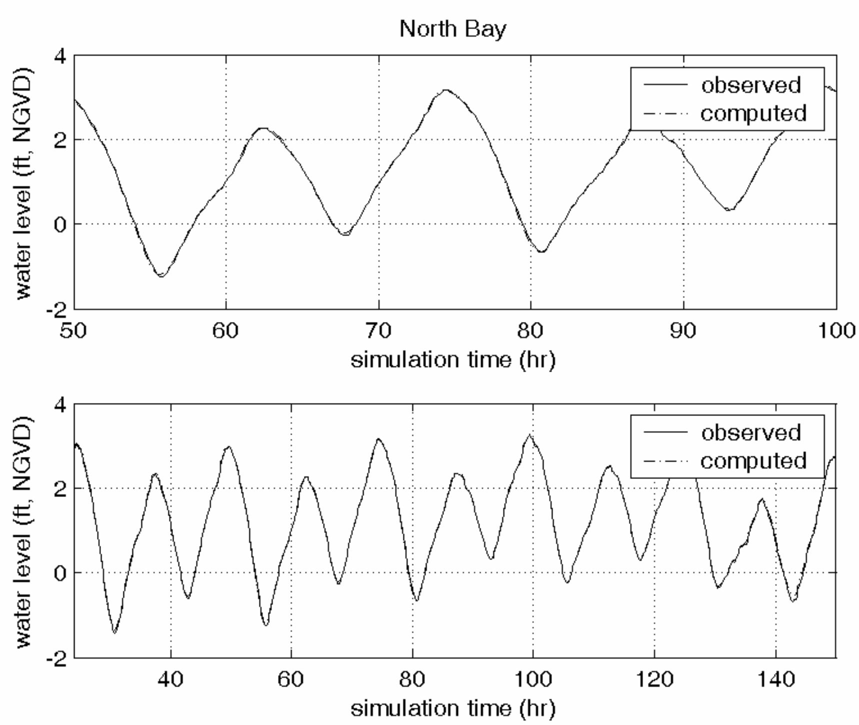


Figure V-29. Comparison of model output and measured tides for the TDR location in North Bay, off Point Isabelle. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

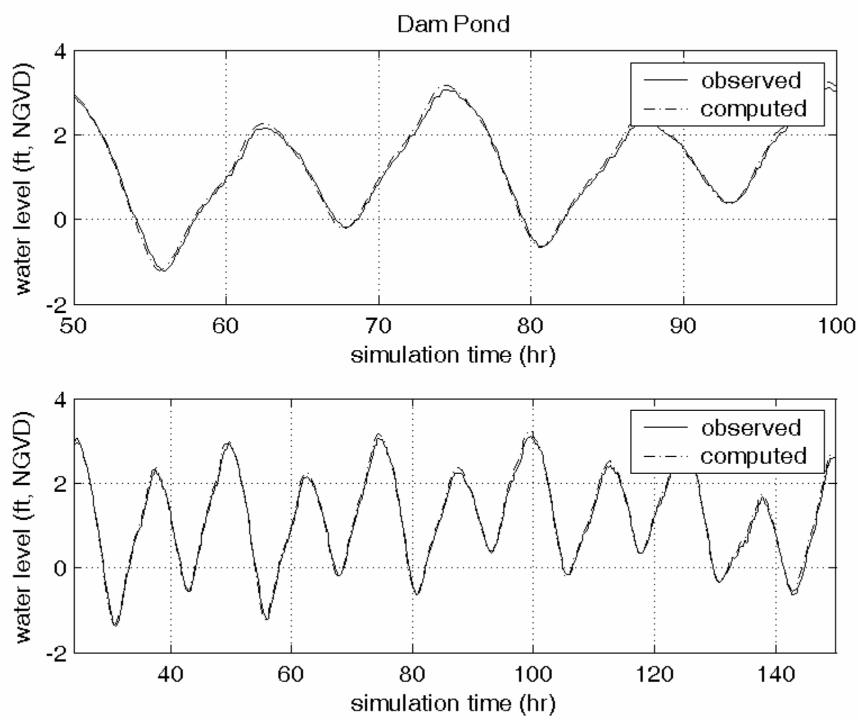


Figure V-30. Comparison of model output and measured tides for the TDR location in Dam Pond. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

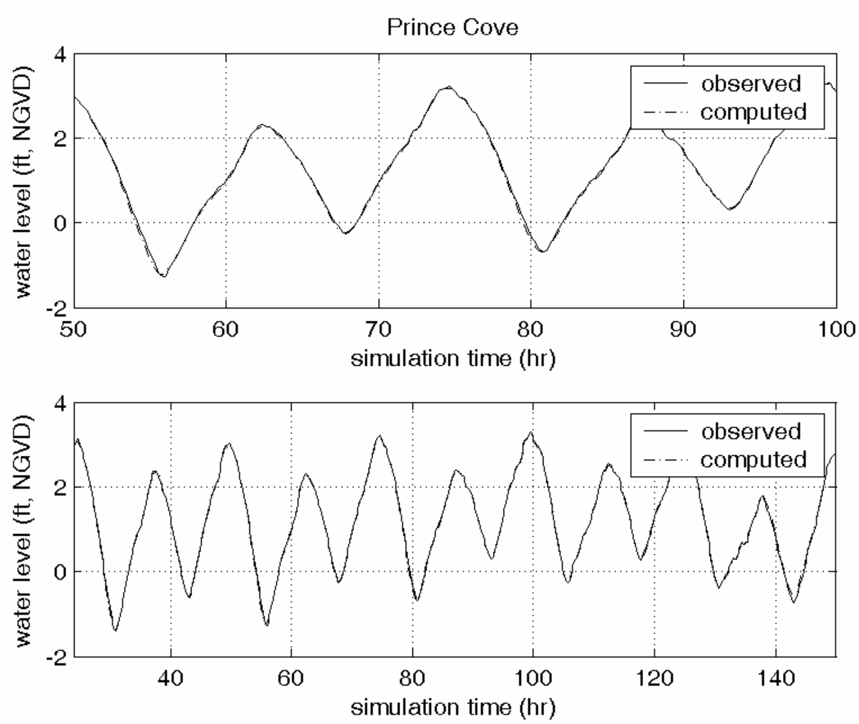


Figure V-31. Comparison of model output and measured tides for the TDR location in Prince Cove. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

V.4.2.4 Model Verification

The calibration procedure used in the development of the Three Bays finite-element model required a match between measured and modeled tides. An additional model verification run was performed to verify the model performance during time periods different from the calibration time period. In this fashion, the calibrated model is tested to ensure its accuracy when run for any time period outside the calibration period. The results of the model verification runs are shown in Table V-7. The analysis of the verification model runs in this table shows that the model performs with a similar excellent degree of accuracy to the calibration runs.

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

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound*	1.39	0.20	0.09	0.41	26.0	152.8
Cotuit Bay	1.39	0.20	0.09	0.42	30.2	162.2
West Bay	1.39	0.18	0.10	0.42	31.2	165.7
North Bay Bridge	1.39	0.18	0.11	0.42	33.0	172.2
North Bay	1.40	0.17	0.11	0.42	33.4	174.0
Dam Pond	1.40	0.17	0.11	0.42	33.8	175.0
Prince Cove	1.40	0.17	0.12	0.42	35.5	180.5
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound*	1.39	0.20	0.09	0.41	26.0	152.7
Cotuit Bay	1.37	0.19	0.10	0.41	30.1	164.3
West Bay	1.38	0.18	0.10	0.41	31.0	167.4
North Bay Bridge	1.39	0.17	0.11	0.40	33.5	176.6
North Bay	1.38	0.18	0.10	0.41	33.8	178.8
Dam Pond	1.35	0.17	0.12	0.41	37.5	185.5
Prince Cove	1.39	0.16	0.13	0.41	35.9	185.4
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound*	0.00	0.00	0.00	0.00	0	0
Cotuit Bay	0.02	0.01	-0.01	0.01	0	0
West Bay	0.01	0.00	0.00	0.01	0	2
North Bay Bridge	0.00	0.01	0.00	0.02	1	5
North Bay	0.02	-0.01	0.01	0.01	1	5
Dam Pond	0.05	0.00	-0.01	0.01	8	11
Prince Cove	0.01	0.01	-0.01	0.01	1	5

*model open boundary

Table V-7. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for the Three-Bays system, 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
Nantucket Sound*	1.13	0.17	0.06	0.40	175.8	93.2
Cotuit Bay	1.13	0.16	0.07	0.40	179.7	102.2
West Bay	1.13	0.16	0.07	0.40	180.5	105.0
North Bay Bridge	1.13	0.15	0.07	0.40	182.1	110.0
North Bay	1.14	0.16	0.08	0.40	182.4	111.3
Dam Pond	1.14	0.15	0.08	0.40	182.7	112.1
Prince Cove	1.14	0.15	0.08	0.40	184.4	116.7
Measured tide during verification period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound*	1.13	0.17	0.07	0.40	176.1	92.1
Cotuit Bay	1.12	0.16	0.07	0.40	179.5	102.2
West Bay	1.13	0.16	0.08	0.39	184.5	119.8
North Bay Bridge	1.13	0.16	0.08	0.39	182.3	112.2
North Bay	1.13	0.16	0.07	0.39	180.4	105.7
Dam Pond	1.09	0.16	0.09	0.40	187.8	124.9
Prince Cove	1.13	0.16	0.09	0.40	184.5	119.8
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound*	0.00	0.00	-0.01	0.00	1	-1
Cotuit Bay	0.01	0.00	0.00	0.00	-1	0
West Bay	0.00	0.00	-0.01	0.01	0	1
North Bay Bridge	0.00	-0.01	-0.01	0.01	1	2
North Bay	0.01	0.00	0.01	0.01	1	3
Dam Pond	0.05	-0.01	-0.01	0.00	10	13
Prince Cove	0.01	-0.01	-0.01	0.00	0	3

*model open boundary

V.4.2.5 ADCP verification of the Three Bays system

An additional model verification check was possible by using collected ADCP velocity data to verify the performance of the Three Bays system model. Computed flow rates from the model were compared to flow rates determined using the measured velocity data. The ADCP data survey efforts are described in Section 2. For the model ADCP verification, the Three Bays model was run for the period covered during the ADCP survey on October 24, 2002. Model flow rates were computed in RMA-2 at continuity lines (channel cross-sections) that correspond to the actual ADCP transects followed in each survey (i.e., across each inlet, at the entrance to West Bay, and at the east and west ends of the Seapuit River).

Comparisons of the measured and modeled volume flow rates in the Three Bays system are shown in Figures V-32 through V-36. For each figure, the top plot shows the flow comparison, and the lower plot shows the time series of tide elevation for the same period. Each ADCP point (blue triangles shown on the plots) is a summation of flow measured along the ADCP transect. The 'bumps' and 'skips' of the flow rate curve (more evident in the model output) can be attributed to the effects of winds (i.e., atmospheric effects) on the water surface and friction across the seabed periodically retarding or accelerating the flow through the inlets, and inside the system channels. If water surface elevations changed smoothly as a sinusoid, the volume flow rate would also appear as a smooth curve. However, since the rate at which water surface elevations change does not vary smoothly, the flow rate curve is expected to show short-period fluctuations.

Data comparisons at all five ADCP transect show exceptionally good agreement with the model predictions. The calibrated model accurately describes the discharge magnitude at each line, even in the Seapuit River where flow rates are an order of magnitude less than at each inlet. For all transects the R^2 correlation coefficients between data and model results are between 0.99 and 0.83. The lowest correlation is at the West Seapuit transect (R^2 value of 0.83) which is still good considering the low volume flow rates, which are more difficult to measure at this transect. Correlation statistics between the modeled and measured flows for each ADCP transect are presented in Table V-8.

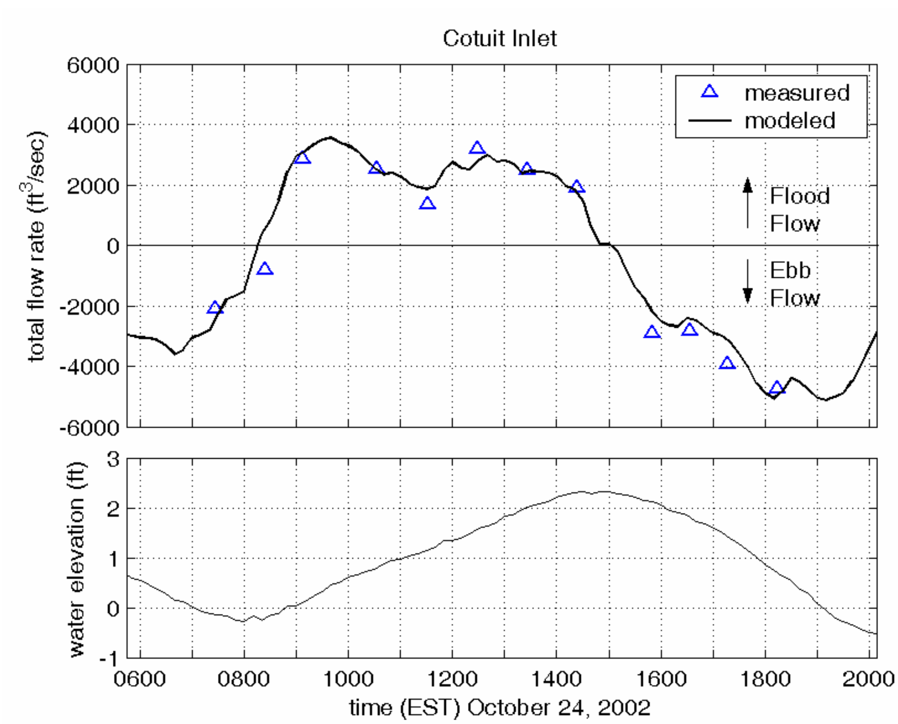


Figure V-32. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the Cotuit Bay Inlet over a tidal cycle on October 24, 2001. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation offshore Dead Neck. (R^2 =0.96).

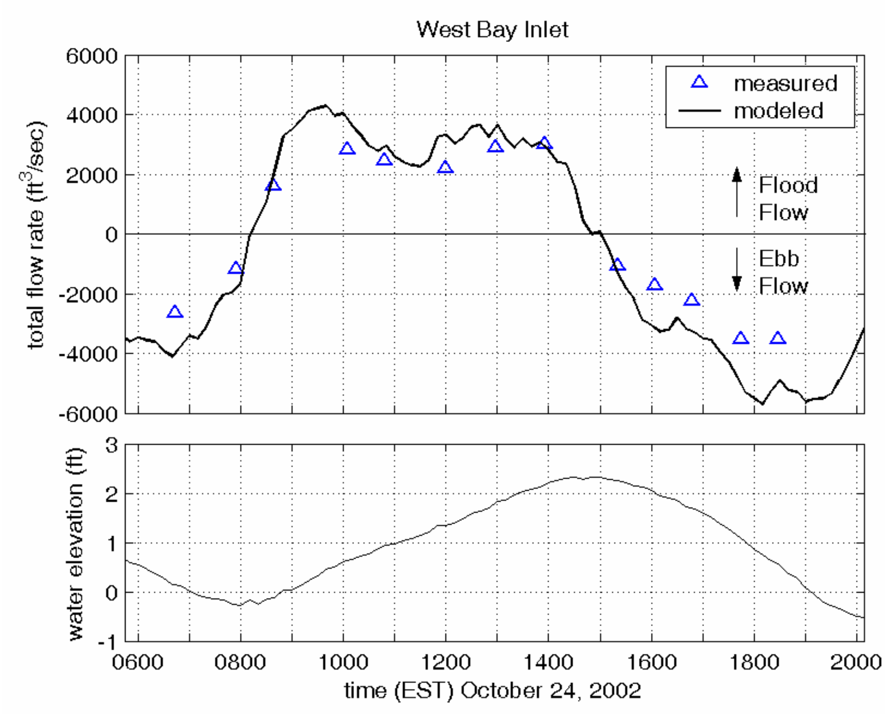


Figure V-33. Comparison of measured volume flow rates versus modeled flow rates (top plot) through West Bay Inlet over a tidal cycle on October 24, 2001. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation offshore Dead Neck. (R^2=0.84).

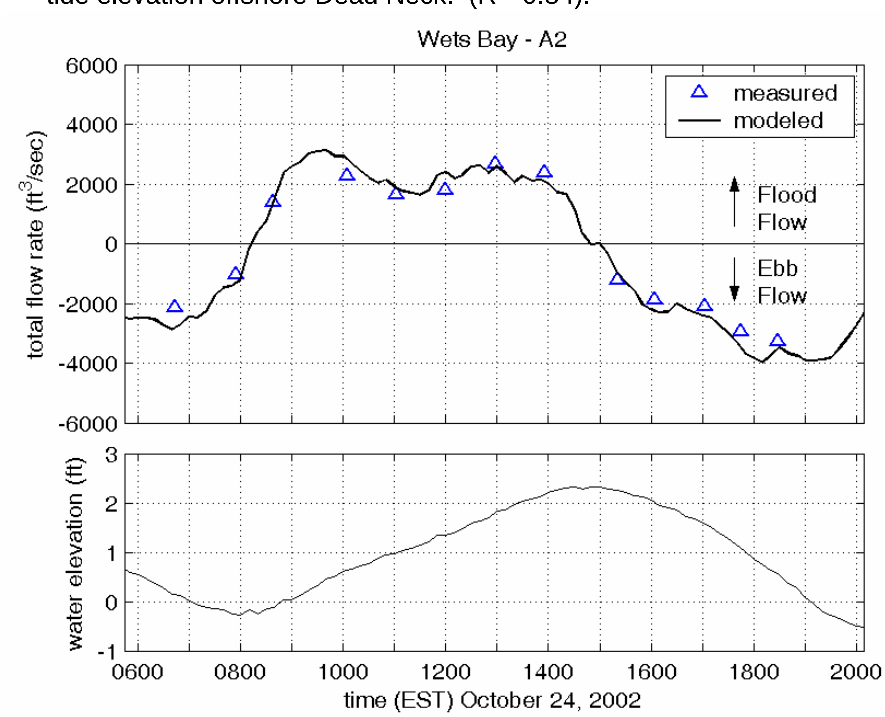


Figure V-34. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the entrance to West Bay, at transect A2, over a tidal cycle on October 24, 2001. Flood flows into the bay are positive (+), and ebb flows out of the bay are negative (-). The bottom plot shows the tide elevation offshore Dead Neck. (R^2=0.97).

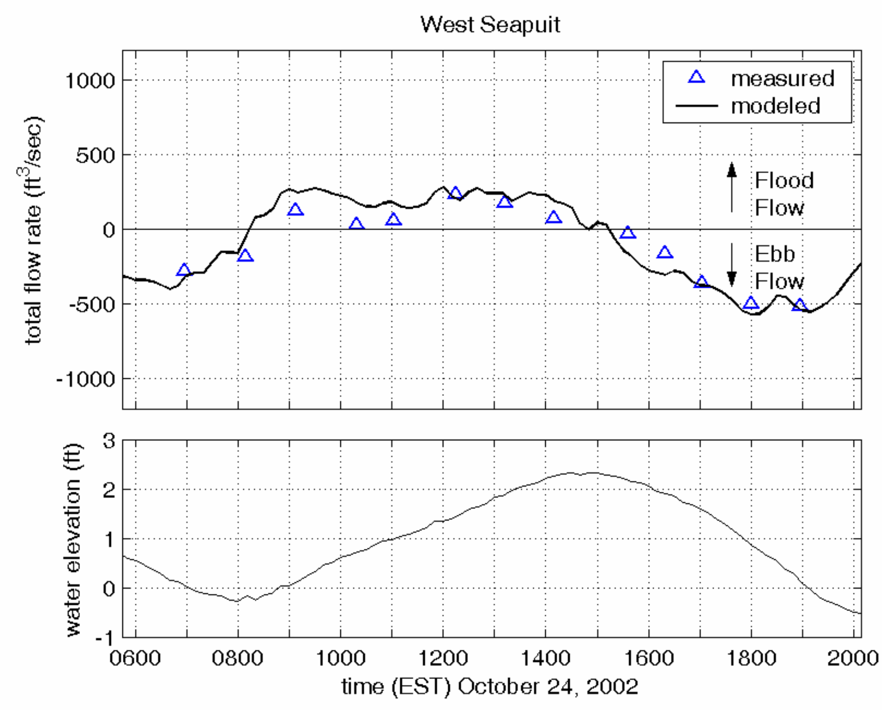


Figure V-35. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the west entrance to the Seapuit River over a tidal cycle on October 24, 2001. Flood flows are positive (+), and ebb flows are negative (-). The bottom plot shows the tide elevation offshore Dead Neck. (R^2=0.83).

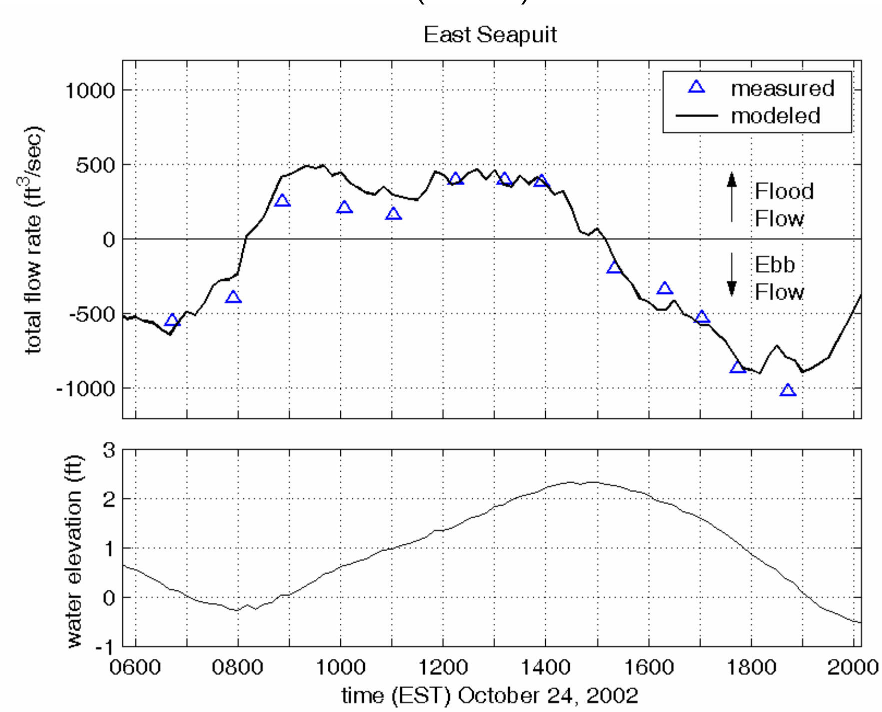


Figure V-36. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the east entrance to the Seapuit River over a tidal cycle on October 24, 2001. Flood flows are positive (+), and ebb flows are negative (-). The bottom plot shows the tide elevation offshore Dead Neck. (R^2=0.94).

Table V-8. Correlation statistics between modeled and measured total flow rates at the five ADCP transects.				
Transect	R^2 correlation	RMS error (ft^3/sec)	Max Error (ft^3/sec)	Min Error (ft^3/sec)
Cotuit Bay Inlet	0.96	595	1613	20
West Bay Inlet	0.84	1007	1784	203
West Bay – A2	0.97	369	740	65
West Seapuit	0.83	99	156	9
East Seapuit	0.94	114	212	2

V.4.2.6 Model Circulation Characteristics

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

From the model run of the Three Bays system, maximum ebb velocities in the inlet channels are slightly larger than velocities during maximum flood. Maximum depth-averaged flood velocities in the model are approximately 1.5 feet/sec at West Bay Inlet and 1.1 ft sec at Cotuit Bay inlet, while maximum ebb velocities are about 2.1 feet/sec at West Bay inlet and 1.7 ft/sec at Cotuit Bay Inlet. At both the Little Island Bridge and the Cotuit Bay entrance to North Bay, typical peak flood and ebb velocities are 0.8 ft/sec and 1.1 ft/sec, respectively. Close-up views of model output are presented in Figure V-37 and V-38, which show contours of velocity magnitude along with velocity vectors that indicate flow direction, each for a single model time-step, at the portion of the tide where maximum ebb velocities occur (in Figure V-37), and for maximum flood velocities in Figure V-38.

In addition to depth-averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. For the flushing analysis in the next section, flow rates were computed across 12 separate transects in the Three Bays system. The variation of flow as the tide floods and ebbs at the two system inlets is seen in the plot of flow rates in Figure V-39. Maximum flow rates occur during ebbing tides in this system. During spring tides, the maximum flood flow rates reach 4500 ft^3/sec at West Bay Inlet. Maximum ebb flow rates during spring tides are slightly greater at West Bay Inlet, about 6800 ft^3/sec. Minimum flood flows at West Bay Inlet during neap tides are 3300 ft^3/sec, and minimum ebb flows during neap tides are approximately 4300 ft^3/sec. The flow magnitudes through Cotuit Inlet are typically 10% less than the flows through West Bay Inlet.

The model is useful to demonstrate some of the unique hydrodynamic traits of the Three Bays system. The Seapuit River, for example, is a rare feature among estuaries in general. It is connected at both ends to separate embayments, which in turn have their own inlets to the ocean. It would be difficult to define the upstream and downstream portions of this tidal river based only on its geographical characteristics. However, ADCP measurements, and model results show that there is a definite "downstream" flow direction. Flow through the Seapuit River is driven by the time difference (lag) of the tide stage between Cotuit Inlet and West Bay Inlet. When the tide is flooding, the river will flow from east to west (the "upstream" direction), and *vice versa* for an ebbing tide. Both model and measurements show that peak flows at the west end of the river are less than peak flows at the east end. At first, this result may seem problematic, as a violation of mass conservation. The expectation is that for a river without additional tributary input, flow across any cross-stream transect should be the same. However, for the Seapuit River, a large portion of the flow into the river during a flooding tide does not exit the other end because it stays within the river basin, resulting in an increase in water elevation. Integrating the flow curves over a half tide cycle (e.g., from slack low to slack high tide) and subtracting the total flow though the west transect from the total flow trough the east transect results in the total tide prism between the two transects (i.e., the tide prism of the Seapuit River). This can be expressed as

$$V_{prism} = \int \left[\left(\frac{dV}{dt} \right)_{east} - \left(\frac{dV}{dt} \right)_{west} \right] dt$$

where V_{prism} is the prism volume, and *dV/dt* is the volume flow rate across the indicated transect. During the ADCP survey time period, the tide prism is computed to be about 3.6×10^6 cubic feet.

Another feature of the Three Bays system model is a persistent tidal eddy (or gyre) in Cotuit Bay, which is set-up during flooding tides. The eddy can be seen in model output shown in Figure V-40, to the north of Bluff Point in Cotuit. The eddy has a faint counter-clock-wise rotation, with velocity magnitudes that are less than 0.2 ft/sec.

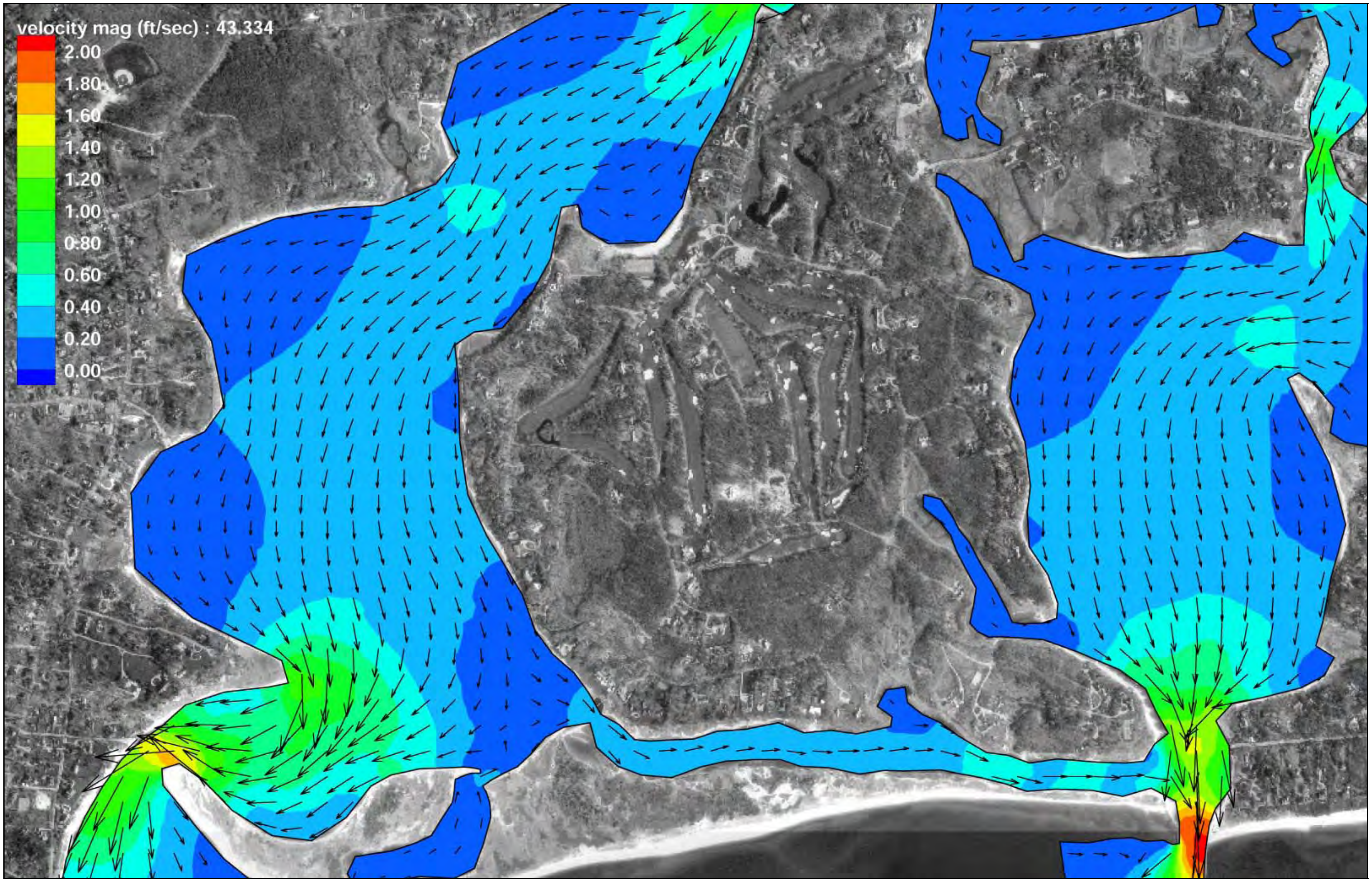


Figure V-37. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

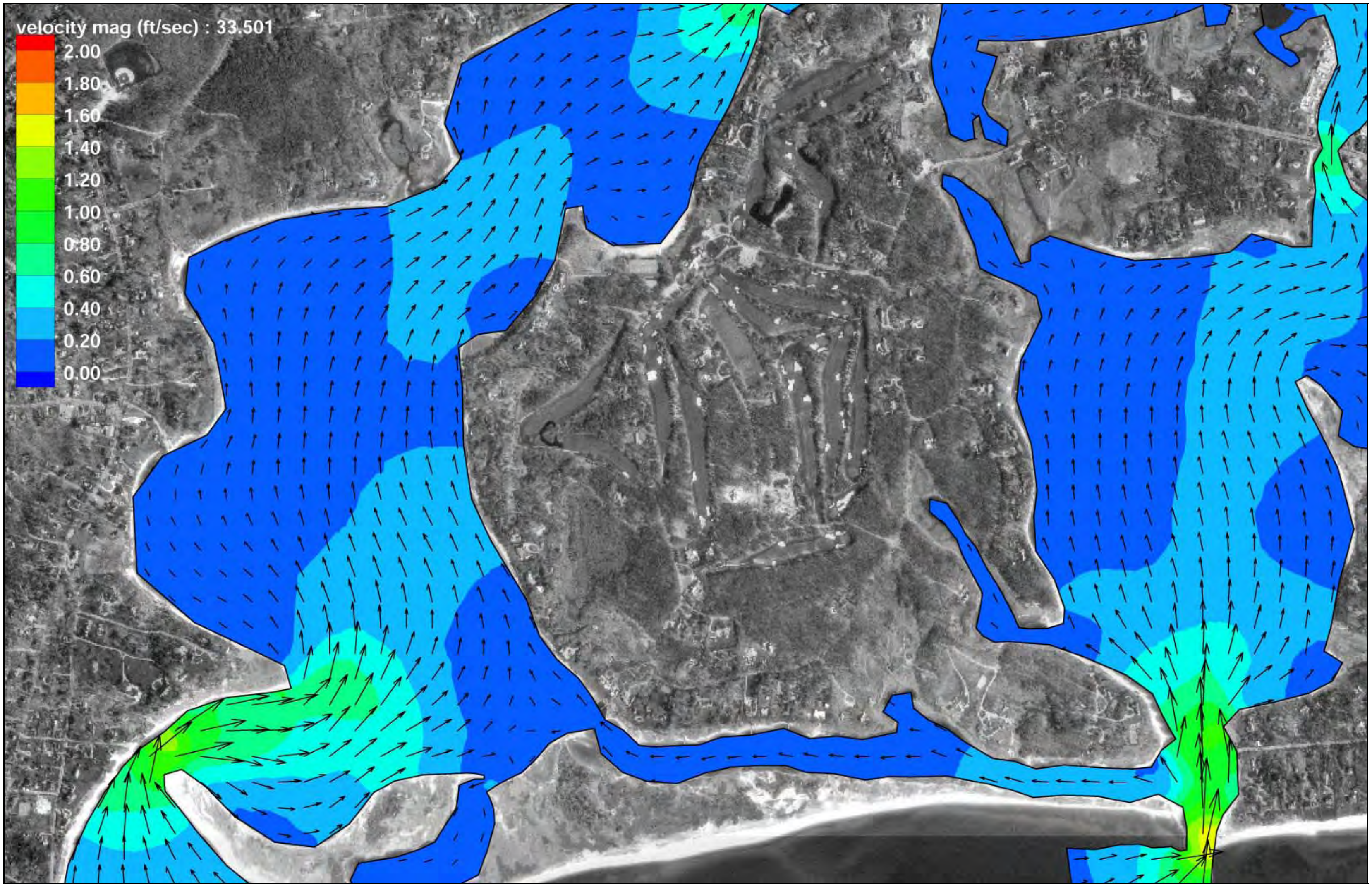


Figure V-38. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

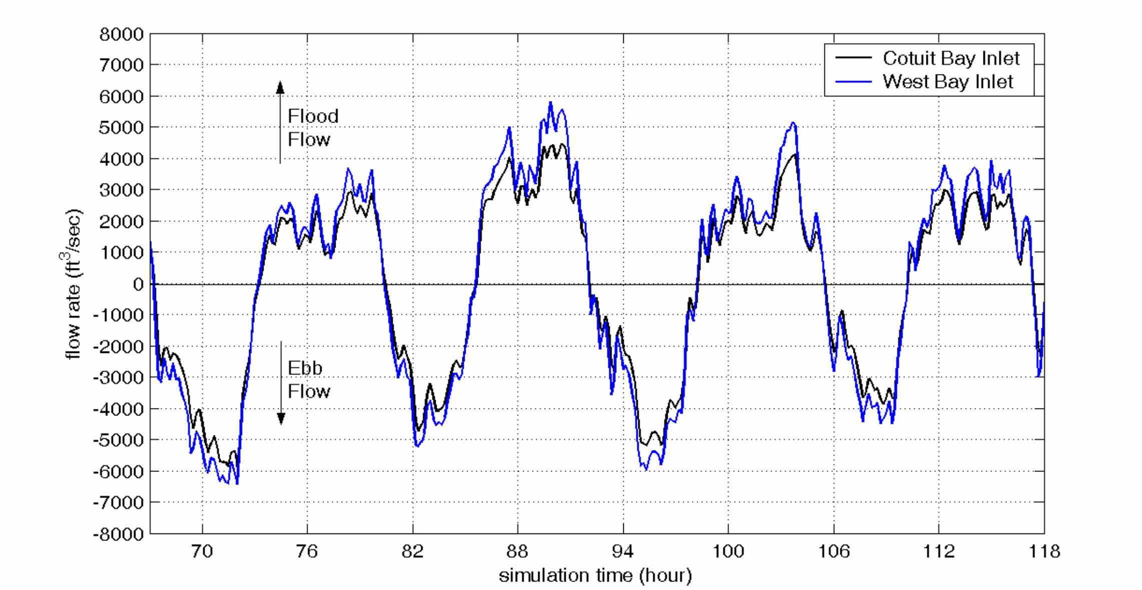


Figure V-39. Time variation of computed flow rates for the two inlets of the Three Bays system. . Plotted time period represents four tide cycles (12.42 h cycle). Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

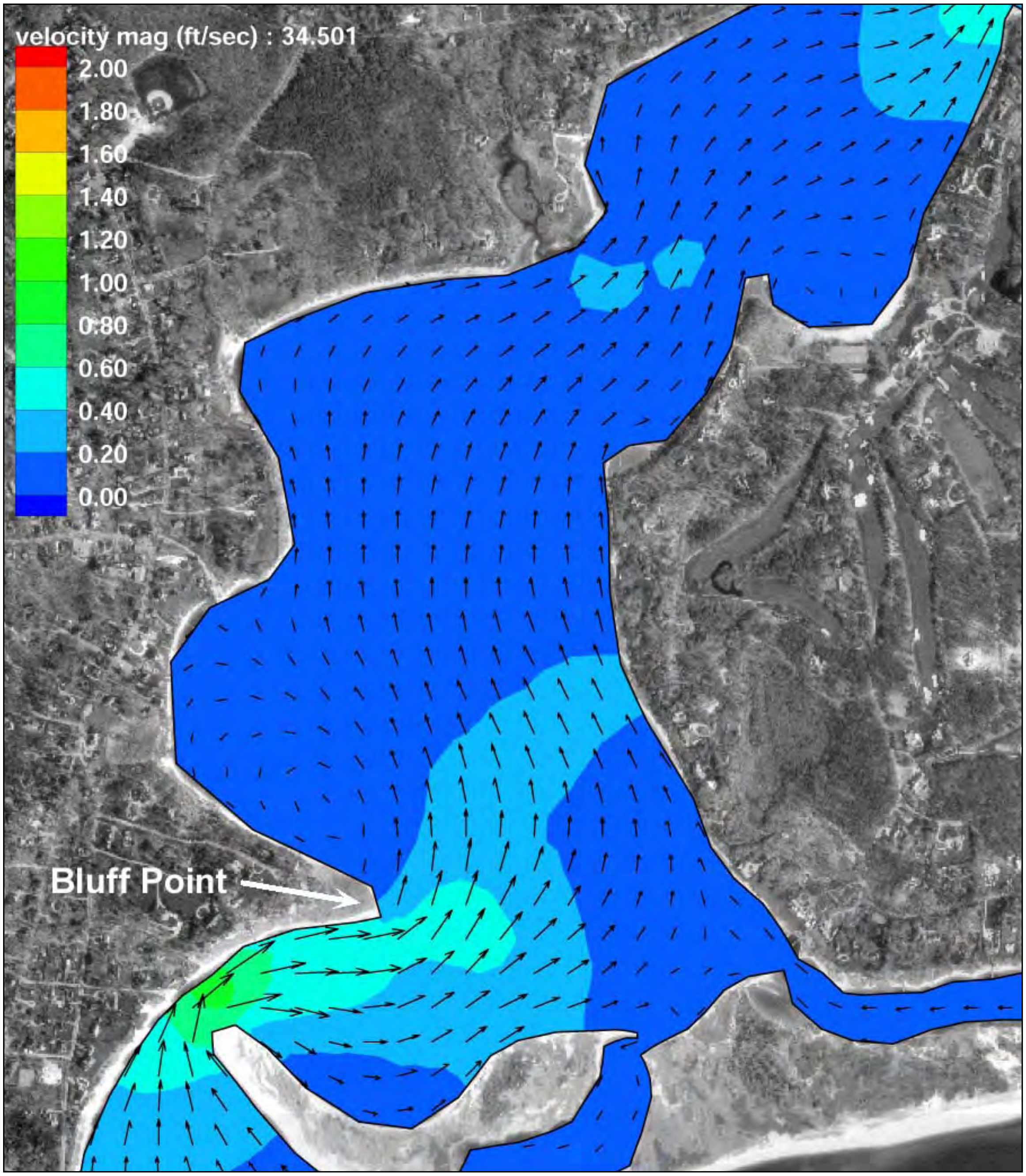


Figure V-40. Close-up of Cotuit Bay, showing output from the Three Bays hydrodynamic model at a single time step, where a recirculation eddy (or gyre) has set up on the north side of Bluff Point.

V.5 FLUSHING CHARACTERISTICS

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through each inlet, the primary mechanism controlling estuarine water quality within the modeled Three Bays system is tidal exchange. A rising tide offshore in Nantucket Sound creates a slope in water surface from the ocean into the modeled systems. Consequently, water flows into (floods) the system. Similarly, the estuary drains into the open waters of Nantucket Sound 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 Three Bays 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 Dam Pond as an example, the **system residence time** is the average time required for water to migrate from Dam Pond, through North Bay, out through West Bay or Cotuit Bay, and into Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Dam Pond to just North Bay (not all the way to the Sound). 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 Three Bays system this approach is applicable, since it assumes the main system has relatively lower quality water relative to Nantucket Sound.

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 will be obtained from the calibrated hydrodynamic model by extending the model to include pollutant/nutrient dispersion. The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Three Bays 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 six sub-embayments within the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system.

Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet. Model divisions used to define the system sub-embayments include 1) the entire Three Bays system, 2) North Bay, including the Marstons Mills River, Prince Cove and Warrens Cove, 3) the Marstons Mills River with Prince Cove and Warrens Cove, 4) Prince Cove, 5) Warrens Cove, 6) Dam Pond, and 7) Eel River. These system divisions follow the model material type areas designated in Figure V-24. Sub-embayment mean volumes and tide prisms are presented in Table V-9.

Residence times were averaged for the tidal cycles comprising a representative 7 lunar day period (14 tide cycles), and are listed in Table V-10. The modeled time period used to compute the flushing rates was that same as the model verification 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 7 lunar 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-9. Embayment mean volumes and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Three Bays System	429,117,000	140,570,000
North Bay	139,666,000	45,824,000
Marstons Mills River	25,236,000	10,834,000
Prince Cove	13,007,000	4,553,000
Warrens Cove	5,047,000	3,614,000
Dam Pond	2,798,000	1,200,000
Eel River	4,035,000	1,702,000

Table V-10. Computed System and Local residence times for embayments in the Three Bays system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Three Bays System	1.6	1.6
North Bay	4.8	1.6
Marstons Mills River	20.5	1.2
Prince Cove	48.8	1.5
Warrens Cove	61.4	0.7
Dam Pond	185.1	1.2
Eel River	130.5	1.2

The computed flushing rates for the Three Bays system show that as a whole, the system flushes well. A flushing time of 1.6 days for the entire estuary shows that on average, water is resident in the system less than two days. All system sub-embayments have local flushing times that are equal to or less than 2 days. Warrens Cove has the shortest local flushing time, because of its small mean sub-embayment volume, relative to its tide prism.

The low local residence times in all areas of the Three Bays 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 Eel River would likely be good as long as the water quality of West Bay 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 Three Bays system, computed system residence times are typically one or two 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 Three Bays 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 Nantucket Sound typically is strong because of the effects of the local winds and tidal induced mixing within Nantucket Sound, 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 Three Bays 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 Three Bays 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 output for the water quality model calibration was a the 7 lunar-day period (14 tide cycles, or 7.25 solar days) beginning October 11, 2002 0000 EST. This period corresponds to that used in the flushing analysis presented in Chapter V. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model had reached a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

Three primary nitrogen loads to sub-embayments are recognized in this modeling study: external loads from the watersheds, nitrogen load from direct rainfall on the embayment surface, and internal loads from the sediments. Additionally, there is a fourth load to the Three Bays system’s sub-embayments, consisting of the background concentrations of total nitrogen in the waters entering from Nantucket Sound. This load is represented as a constant concentration along the two seaward boundaries of the model grid (at Cotuit Bay and West Bay).

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in the area map presented in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data are the minimum required to provide a baseline for MEP analysis. Typically, six years of data (collected between 1999 and 2004) were available for stations monitored by SMAST and the Three Bays Alliance in the Three Bays system.

Table VI-1. Measured data and modeled Nitrogen concentrations for the Three Bays 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 the separate yearly means. Data represented in this table were collected in the summers of 1999 through 2004, except the Vineyard sound station, which covers a longer time period.

Sub-Embayment	monitoring station	data mean	s.d. all data	N	model min	model max	model average
Mill Pond (fresh water)	TB1	1.022	0.246	36	-	-	-
Prince Cove - south	TB2	0.699	0.192	38	0.685	0.699	0.695
Prince Cove - north	TB3	0.602	0.131	37	0.612	0.666	0.639
Warrens Cove	TB4	0.642	0.151	36	0.561	0.642	0.595
North Bay - north	TB5	0.498	0.135	105	0.504	0.531	0.518
North Bay - south	TB6	0.515	0.129	36	0.483	0.517	0.500
North Windmill Cove	TB7	0.511	0.120	103	0.498	0.523	0.511
West Bay - north	TB8	0.383	0.117	34	0.327	0.418	0.363
West Bay - west	TB9	0.376	0.078	38	0.299	0.362	0.327
Eel River	TB10	0.481	0.125	34	0.468	0.500	0.486
Seapuit River	TB11	0.322	0.068	67	0.287	0.305	0.295
Cotuit Bay - north	TB12	0.438	0.076	64	0.364	0.484	0.414
Cotuit Bay - south	TB13	0.389	0.077	75	0.298	0.350	0.321
South Windmill Cove	TB15	0.431	0.090	27	0.369	0.467	0.402
Mellon Cove	TB16	0.411	0.094	24	0.369	0.417	0.392
Dam Pond	TB17	0.508	0.073	5	0.513	0.531	0.523
Vineyard Sound	NS	0.280	0.065	196	-	-	0.280

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

The overall approach involves modeling total nitrogen as a non-conservative constituent, where bottom sediments act as a source or sink of nitrogen, based on local biochemical characteristics. This modeling represents summertime conditions, when algal growth is at its maximum. Total nitrogen modeling is based upon various data collection efforts and analyses presented in previous sections of this report. Nitrogen loading information was derived from the Cape Cod Commission watershed loading analysis (based on the USGS watersheds), as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model of the Three Bays system.

Figure VI-1. Estuarine water quality monitoring station locations in the Three Bays estuary system. Station labels correspond to those provided in Table VI-1. Sentinel station for threshold development depicted with red symbol.

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 Three Bays 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 Three Bays also were used for the water quality constituent modeling portion of this study.

Based on measured surface water flow rates from SMAST and groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include the latest estimates of flows from the Marstons Mills River (to Prince Cove)) and Little River (to Cotuit Bay). The Marstons Mills River has a mean measured flow rate of 6.6 ft^3/sec (16,100 m^3/day), which is 6.9% of the volume of the average tide prism of Prince Cove. Little River has average flows of 1.1 ft^3/sec (3,500 m^3/day, which represents only 0.1% of the Cotuit Bay tide prism.

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 5 tidal-day (125 hour) period. Model results were recorded only after the initial spin-up period. The time step used for the water quality computations was 10 minutes, which corresponds to the time step of the hydrodynamics input for the Three Bays model.

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, 3) summer benthic regeneration, 4) point source inputs developed from measurements of the freshwater portions of the Marstons Mills River and Little River. 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 Cotuit Bay were evenly distributed at grid cells that formed the perimeter of the sub-embayment. Benthic regeneration loads were distributed among another sub-set of grid cells which are in the interior portion of each basin.

The loadings used to model present conditions in the Three Bays estuary 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. For present conditions, some sub-embayments (e.g., North Bay) have more than twice the loading rate from benthic regeneration as from watershed loads. For other sub-embayments the benthic flux is relatively low or negative (Cotuit Bay and Seapuit River) indicating 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 the Nantucket Sound region offshore Three Bays was set at 0.280 mg/L, based on SMAST data from Vineyard Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Vineyard Sound.

Table VI-2. Sub-embayment and surface water loads used for total nitrogen modeling of the Three Bays system, with sub-watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions** for the listed sub-embayments. *Warrens Cove and Prince Cove Channel direct atmospheric deposition is included in the Price Cove load.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Cotuit Bay	21.778	5.786	-54.443
West Bay	19.068	4.233	3.815
Seapuit River	3.767	0.452	-5.418
North Bay	29.447	3.953	67.522
Prince Cove	13.362	1.230	0.512
Marstons Mills R. South (below Mill Pond)	7.293	-	-
Warren Cove	12.027	*	8.830
Prince Cove Channel	5.537	*	2.345
Surface Water Sources			
Marstons Mills River	14.518	-	-
Little River	3.962	-	-

VI.2.4 Model Calibration

Calibration of the total nitrogen model of Three Bays proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data.

Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in Section V. Observed values of *E* (Fischer, *et al.*, 1979) vary between order 10 and order 1000 m^2/sec for riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents (from tides or atmospheric forcing). Generally, the relatively quiescent estuarine embayments of the south shore of Cape Cod require values of *E* that are lower compared to the riverine estuary systems evaluated by Fischer, *et al.*, (1979). Observed values of *E* in these calmer areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled system are presented in Table VI-3. These values were used to develop the "best-fit" total nitrogen model calibration. For the case of TN modeling, "best fit" can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

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

Embayment Division	E m^2/sec
Cotuit Bay Inlet	5.0
Cotuit Bay	10.0
Seapuit River	5.0
West Bay Inlet	5.0
West Bay	10.0
South Windmill Cove	0.8
Mellon Cove	1.0
North Bay	10.0
Prince Cove - north	5.0
Prince Cove - south	1.0
Warrens Cove	5.0
Dam Pond	10.0
Eel River	0.5
Little River	1.0
Marstons Mills River	5.0

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

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

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for each system. Computed root mean squared (rms) error is less than 0.03 mg/L, which demonstrates the exceptional fit between modeled and measured data for this system.

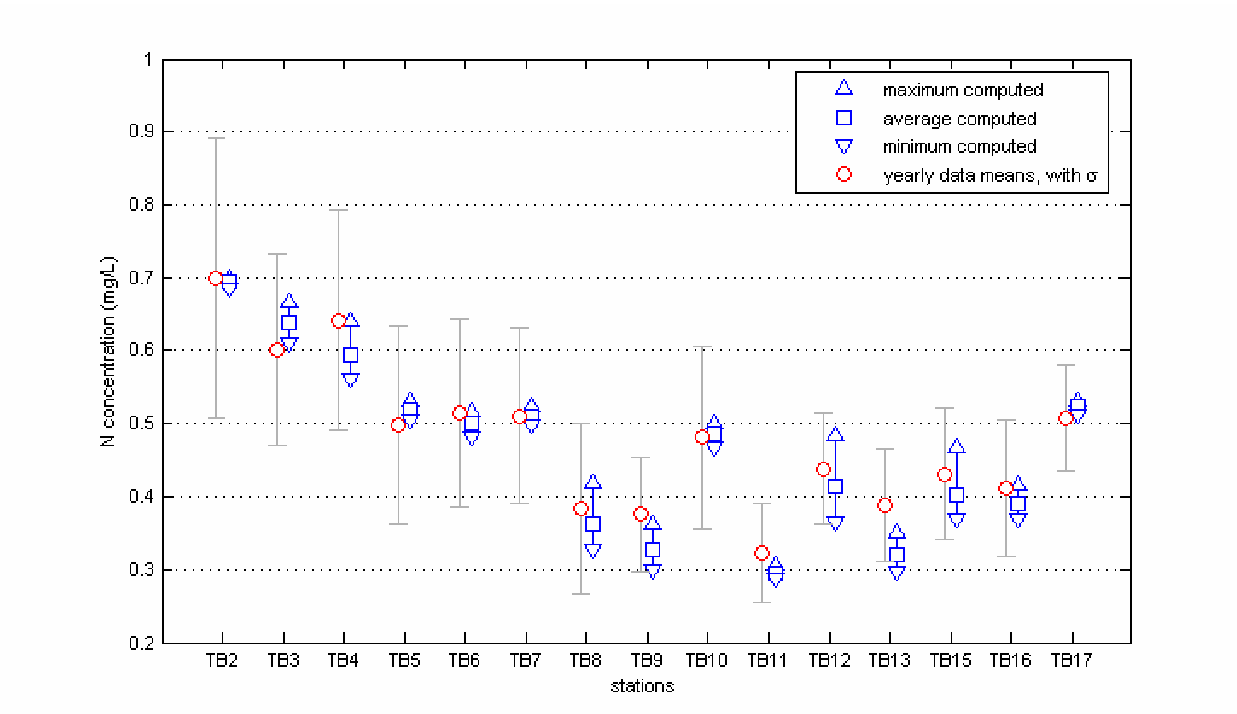


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Three Bays system. Station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation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

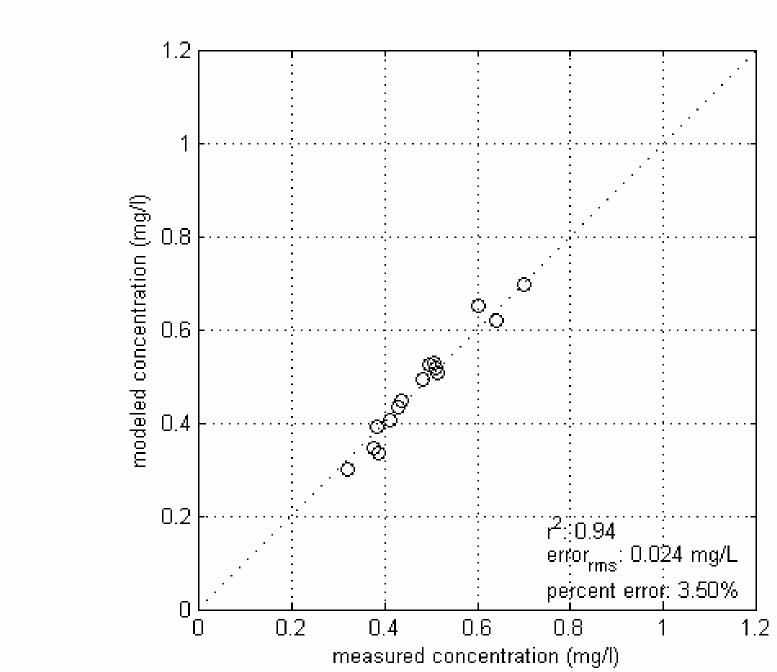


Figure VI-3. Model total nitrogen calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are also presented.

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 5-tidal-day model simulation output period.

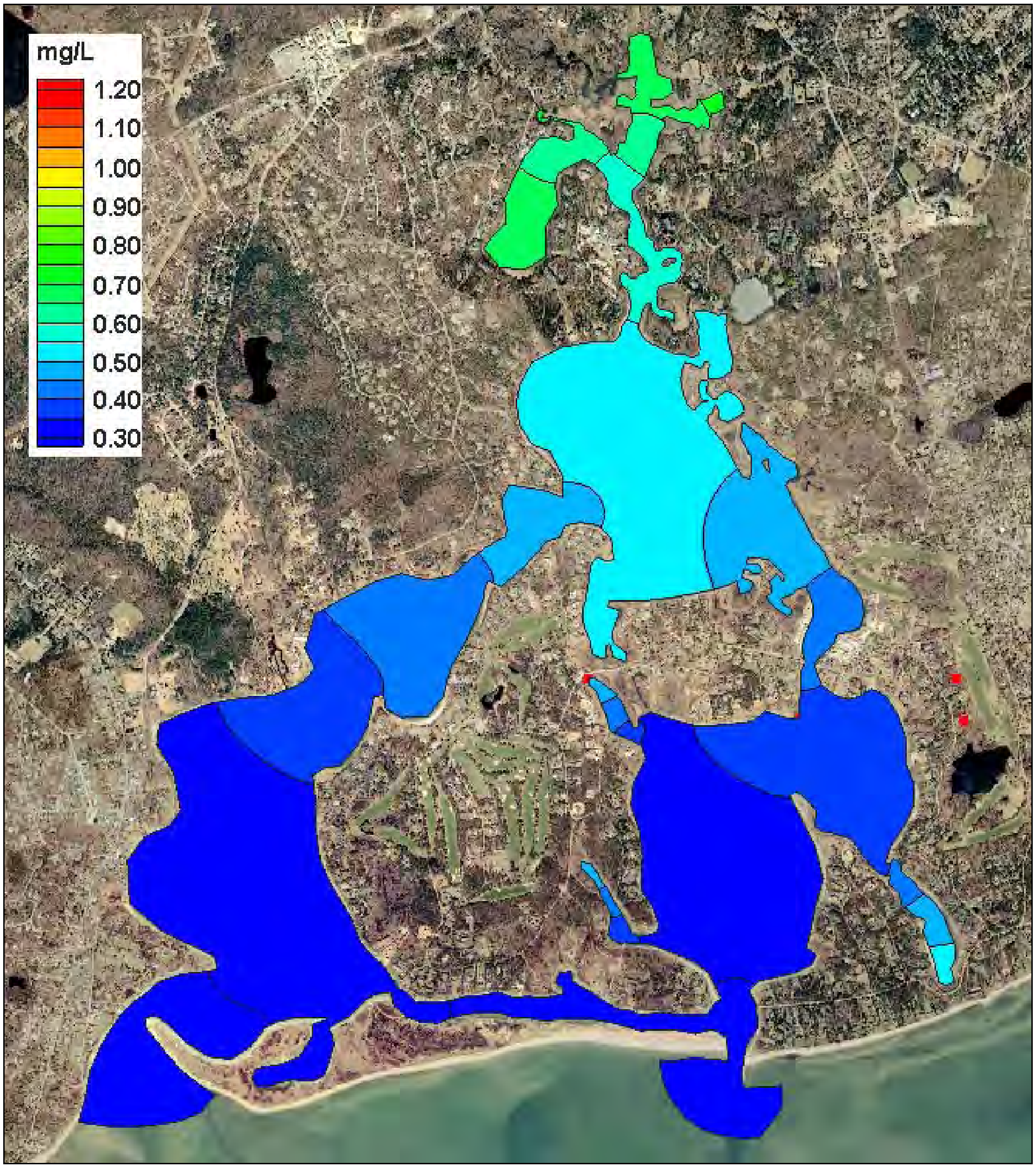


Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Three Bays system.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Three Bays system using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, at the freshwater stream discharges, and groundwater inputs. The open boundary salinity was set at 29.6 ppt. For surface water steams and groundwater inputs salinities were set at 0 ppt. Surface water stream flow rates for the streams were the same as those used for the total nitrogen model, as presented earlier in this section. Groundwater inputs used for each model were 16.01 ft^3/sec (39,100 m^3/day) for Prince Cove, 5.40 ft^3/sec (13,200 m^3/day) for North Bay, 3.22 ft^3/sec (7,900 m^3/day) for West Bay, 6.57 ft^3/sec (16,100 m^3/day) for Cotuit Bay, and 0.77 ft^3/sec (1,900 m^3/day) for the Seapuit River. Groundwater flows were distributed evenly in the model through the use of several 1-D element input points positioned along each model's land boundary.

Comparisons of modeled and measured salinities are presented in Figures VI-5 and VI-6, with contour plots of model output shown in Figure VI-7. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients throughout the Three Bays estuary system. The rms error of the three models is less than 0.8 ppt, and correlation coefficient between the model and measured salinity data is 0.78. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical system.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

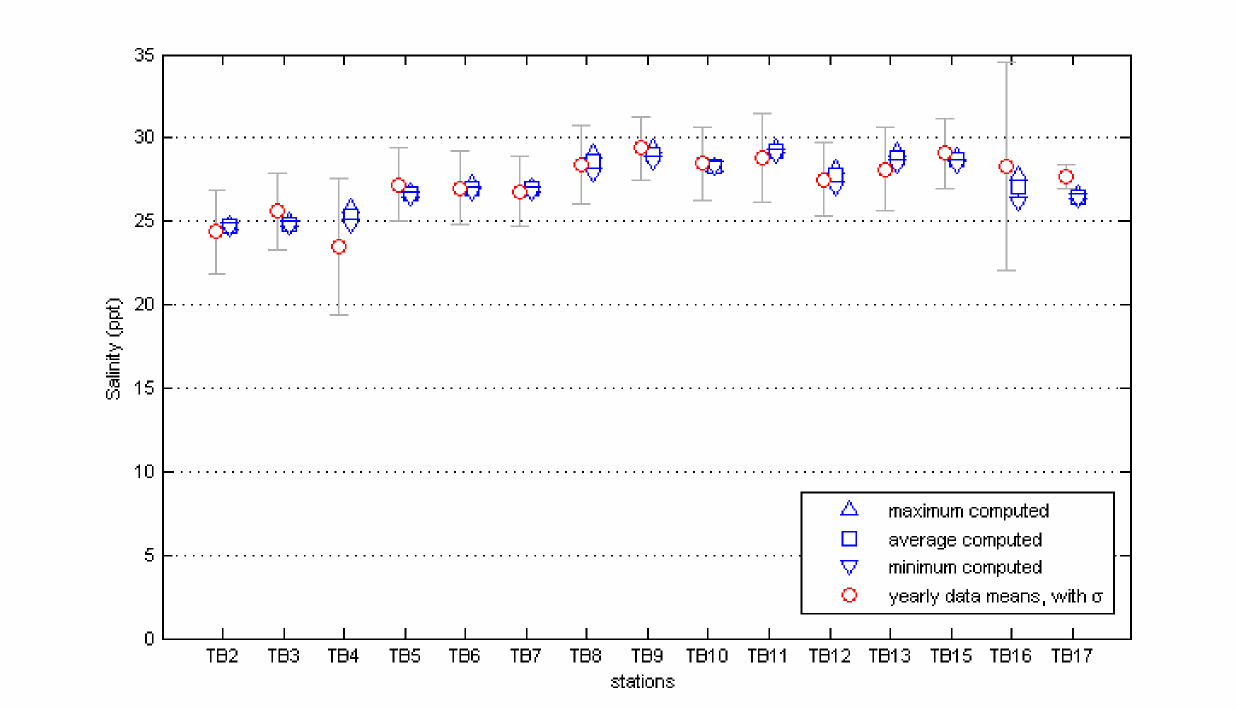


Figure VI-5. Comparison of measured and calibrated model output at stations in Three Bays. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the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

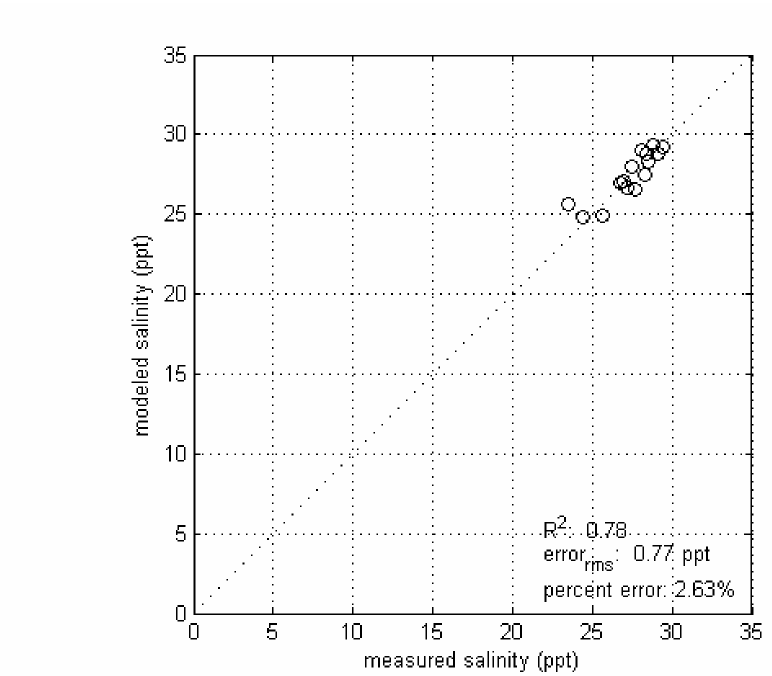


Figure VI-6. Model salinity target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

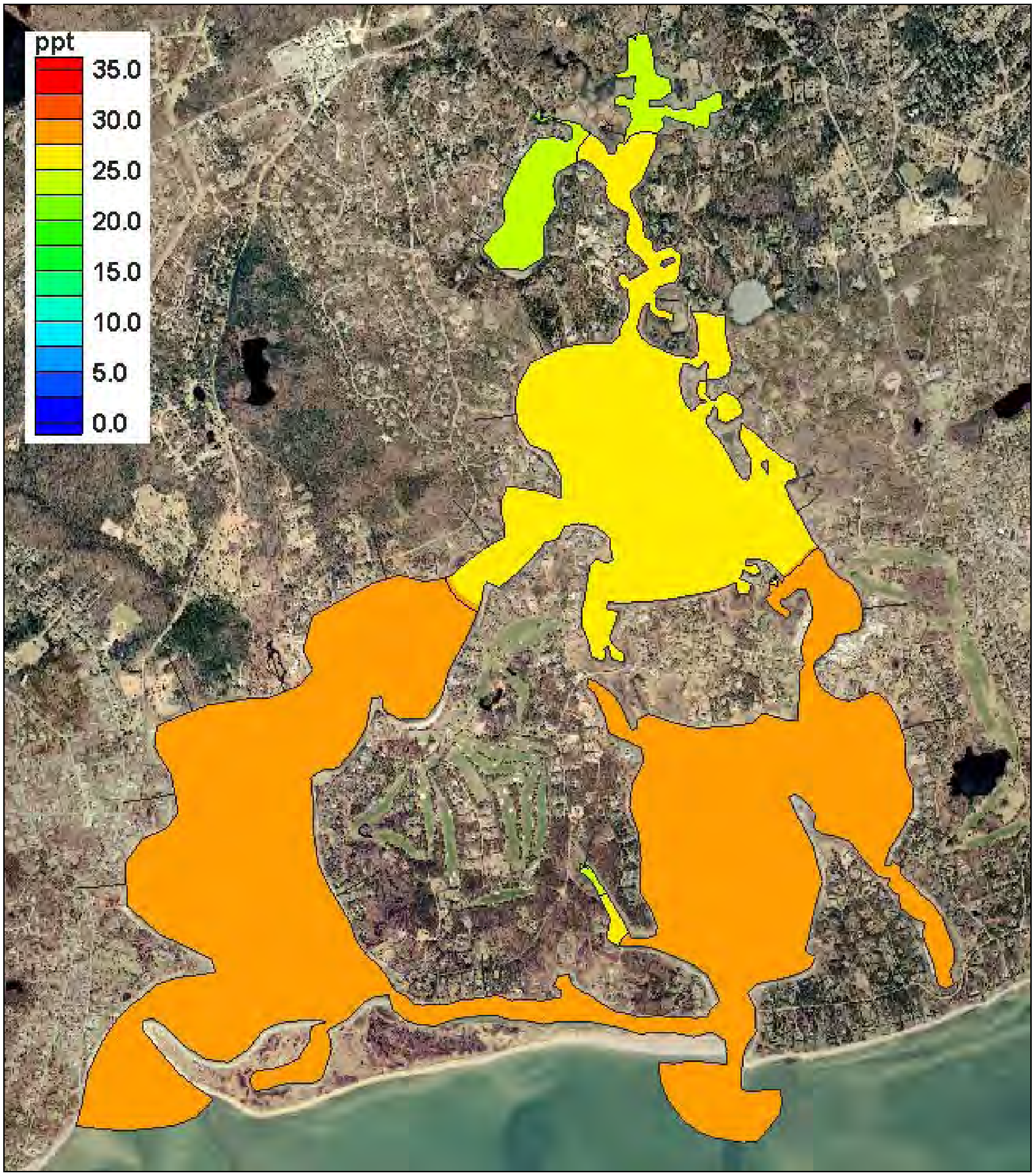


Figure VI-7. Contour Plot of modeled salinity (ppt) in the Three Bays system.

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 Three Bays 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
Cotuit Bay	21.778	24.463	12.3%	1.975	-90.9%
West Bay	19.068	20.077	5.3%	1.170	-93.9%
Seapuit River	3.767	4.507	19.6%	0.452	-88.0%
North Bay	29.447	32.370	9.9%	1.970	-93.3%
Prince Cove	13.362	15.197	13.7%	1.137	-91.5%
Marstons Mill R. South (below Mill Pond)	7.293	8.282	13.6%	1.186	-83.7%
Warrens Cove	12.027	15.290	27.1%	1.945	-83.8%
Prince Cove Channel	5.537	6.564	18.6%	0.515	-90.7%
Surface Water Sources					
Marstons Mills River	14.518	17.008	17.2%	1.641	-88.7%
Little River	3.962	4.830	21.9%	0.471	-88.1%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be less than a 6% increase in watershed nitrogen load West Bay as a result of potential future development. Other watershed areas would experience much greater load increases, for example the loads to the Little River would increase 22% from the present day loading levels. A maximum increase in watershed loading resulting from future development would occur in the Warrens Cove watershed, where the increase would be 3.263 kg/day, or 27% more than present conditions. For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions by over 85%.

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Three Bays 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)
Cotuit Bay	24.463	5.786	-57.241
West Bay	20.077	4.233	3.821
Seapuit River	4.507	0.452	-5.562
North Bay	32.370	3.953	72.223
Prince Cove	15.197	1.230	0.559
Marstons Mill R. South (below Mill Pond)	8.282	-	-
Warrens Cove	15.290	-	9.491
Prince Cove Channel	6.564	-	2.526
Surface Water Sources			
Marstons Mills River	17.008	-	-
Little River	4.830	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of each system were run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. Total N concentrations increased the most in the upper portions of the system, with the largest change at a station in Prince Cove (+7.6% at TB2), with the least change occurring in West Bay (+1.3% at TB9) closer to the inlet to Nantucket Sound. Color contours of model output for the build-out scenario are present in Figure VI-8. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-4, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Three Bays system.				
Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Prince Cove - south	TB2	0.695	0.748	+7.6%
Prince Cove - north	TB3	0.639	0.684	+7.1%
Warrens Cove	TB4	0.595	0.634	+6.6%
North Bay - north	TB5	0.518	0.545	+5.2%
North Bay - south	TB6	0.500	0.525	+4.8%
North Windmill Cove	TB7	0.511	0.536	+4.9%
West Bay - north	TB8	0.363	0.371	+2.2%
West Bay - west	TB9	0.327	0.331	+1.3%
Eel River	TB10	0.486	0.500	+2.9%
Seapuit River	TB11	0.295	0.297	+0.9%
Cotuit Bay - north	TB12	0.414	0.430	+3.8%
Cotuit Bay - south	TB13	0.321	0.327	+1.7%
South Windmill Cove	TB15	0.402	0.412	+2.3%
Mellon Cove	TB16	0.392	0.398	+1.7%
Dam Pond	TB17	0.523	0.551	+5.3%

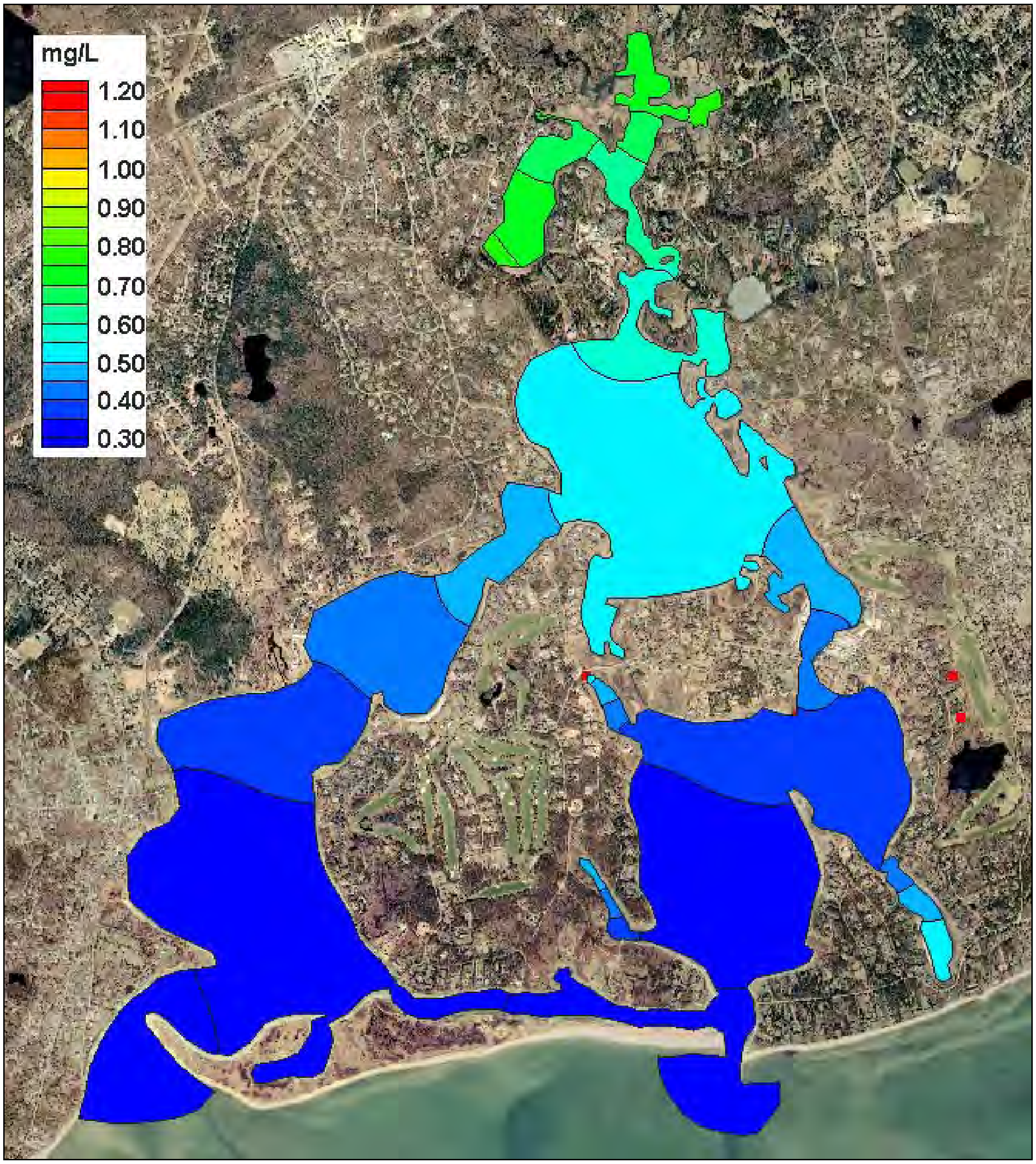


Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Three Bays system, for projected build-out loading conditions.

VI.2.6.2 No Anthropogenic Load

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

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of the Three Bays 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)
Cotuit Bay	1.975	5.786	-35.901
West Bay	1.170	4.233	3.360
Seapuit River	0.452	0.452	-4.412
North Bay	1.970	3.953	36.481
Prince Cove	1.137	1.230	0.224
Marstons Mill R. South (below Mill Pond)	1.186	-	-
Warrens Cove	1.945	-	4.409
Prince Cove Channel	0.515	-	1.125
Surface Water Sources			
Marstons Mills River	1.641	-	-
Little River	0.471	-	-

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions greater than 45% (at TB2) occurring the upper portions of the system. Results for each system are shown pictorially in Figure VI-9.

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for the Three Bays system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	no load (mg/L)	% change
Prince Cove - south	TB2	0.695	0.379	-45.4%
Prince Cove - north	TB3	0.639	0.370	-42.1%
Warrens Cove	TB4	0.595	0.365	-38.7%
North Bay - north	TB5	0.518	0.350	-32.4%
North Bay - south	TB6	0.500	0.346	-30.8%
North Windmill Cove	TB7	0.511	0.349	-31.6%
West Bay - north	TB8	0.363	0.305	-15.9%
West Bay - west	TB9	0.327	0.294	-10.0%
Eel River	TB10	0.486	0.336	-30.9%
Seapuit River	TB11	0.295	0.279	-5.4%
Cotuit Bay - north	TB12	0.414	0.315	-24.0%
Cotuit Bay - south	TB13	0.321	0.286	-10.9%
South Windmill Cove	TB15	0.402	0.315	-21.6%
Mellon Cove	TB16	0.392	0.310	-20.9%
Dam Pond	TB17	0.523	0.351	-32.9%

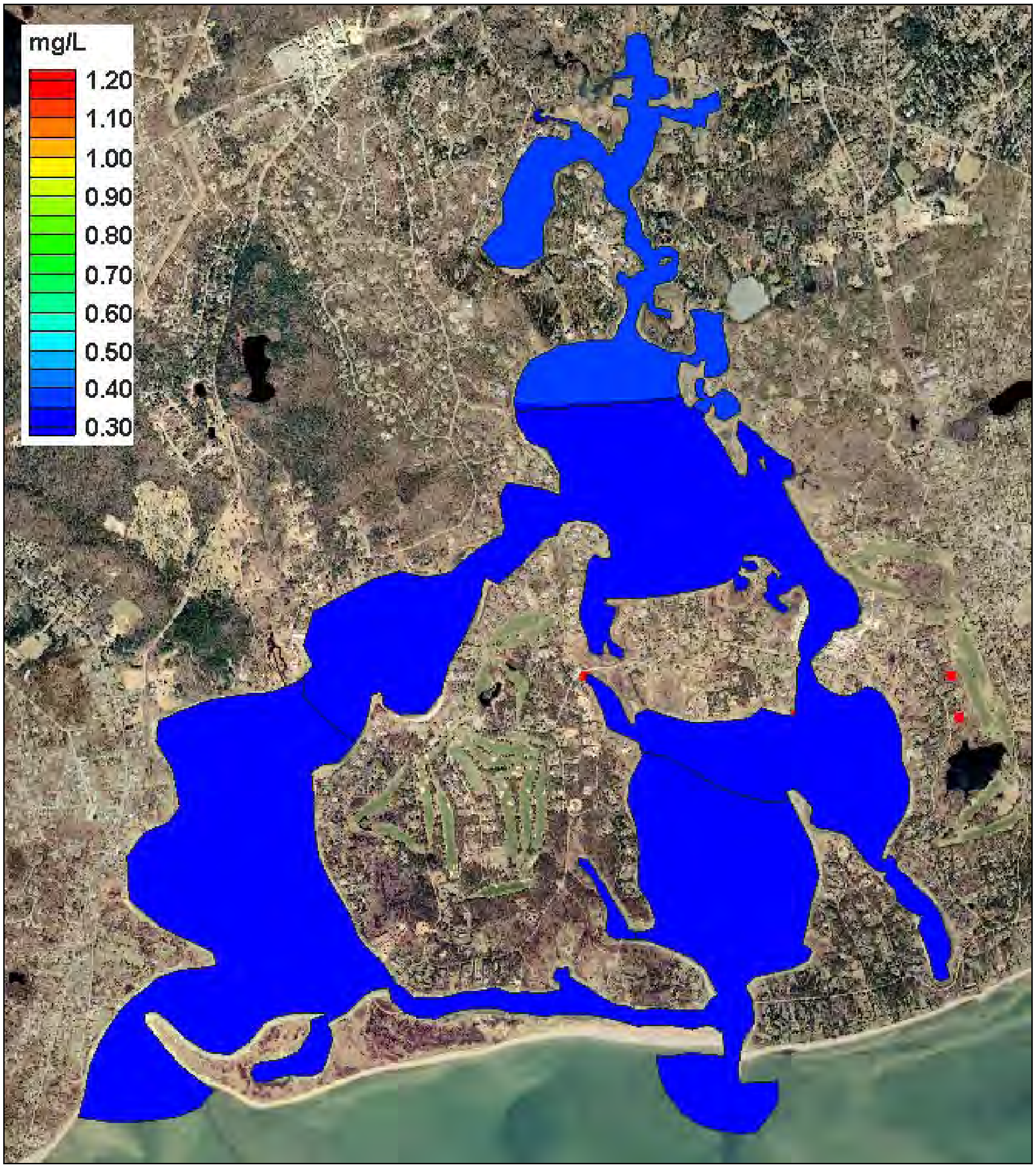


Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in Three Bays, for no anthropogenic loading conditions.

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 Three Bays embayment systems, the MEP assessment is based upon data from the water quality monitoring database and MEP surveys of eelgrass distribution, benthic animal communities and sediment characteristics, and dissolved oxygen records conducted during the summers of 2000-2002. 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 these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

There are a variety of indicators that can be used in concert with water quality monitoring data for evaluating the ecological health of 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 thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within the upper portion of the Three Bays system (Princes Cove and North Bay, 2002), as well as closer to the inlets to the Three Bays system (Cotuit bay and West Bay, 2002), to record the frequency and duration of low oxygen conditions during the critical summer period. This work was in association with Three Bays Preservation and the Town of Barnstable.

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 Three Bays System was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to nutrient related 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. Within the Three Bays System, temporal changes in eelgrass distribution provides a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing-new inlet) in nutrient enrichment.

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

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

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Three Bays System (Figure VII-2). The sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Three Bays embayment system was collected during the summers of 2000, 2001 and 2002.

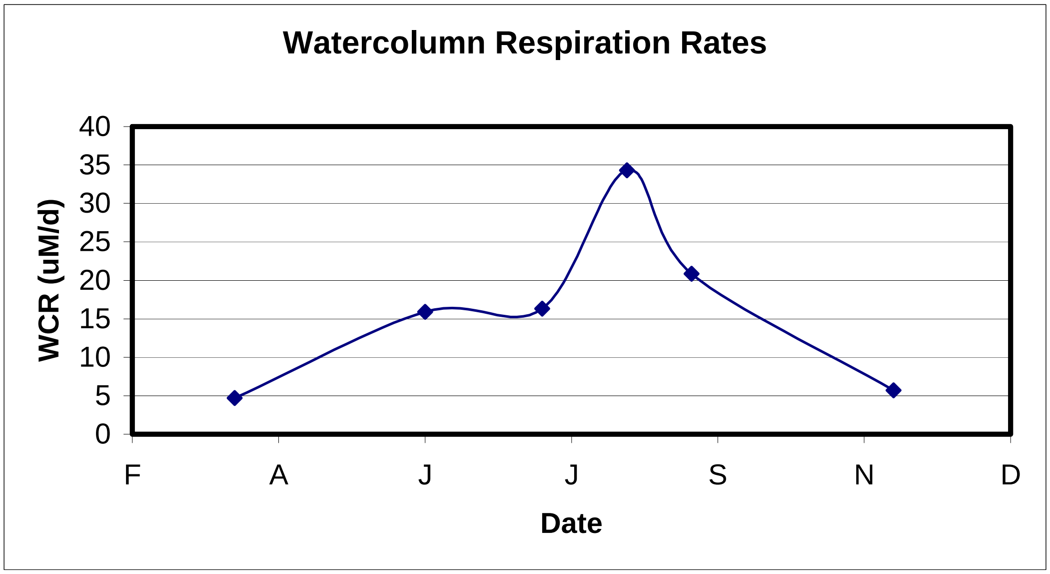


Figure VII-1. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

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

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 25-30 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality at all mooring sites within the estuary (Figures VII-3 through VII-12, Three Bays system). The oxygen data throughout the estuary is consistent with elevated organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of these estuarine systems. The oxygen records further indicate that the upper tidal reaches of each estuary have the largest daily oxygen excursion, with daily excursions in excess of 6 mg L-1 common. This further supports the assessment of a high degree of nutrient enrichment..

The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production,

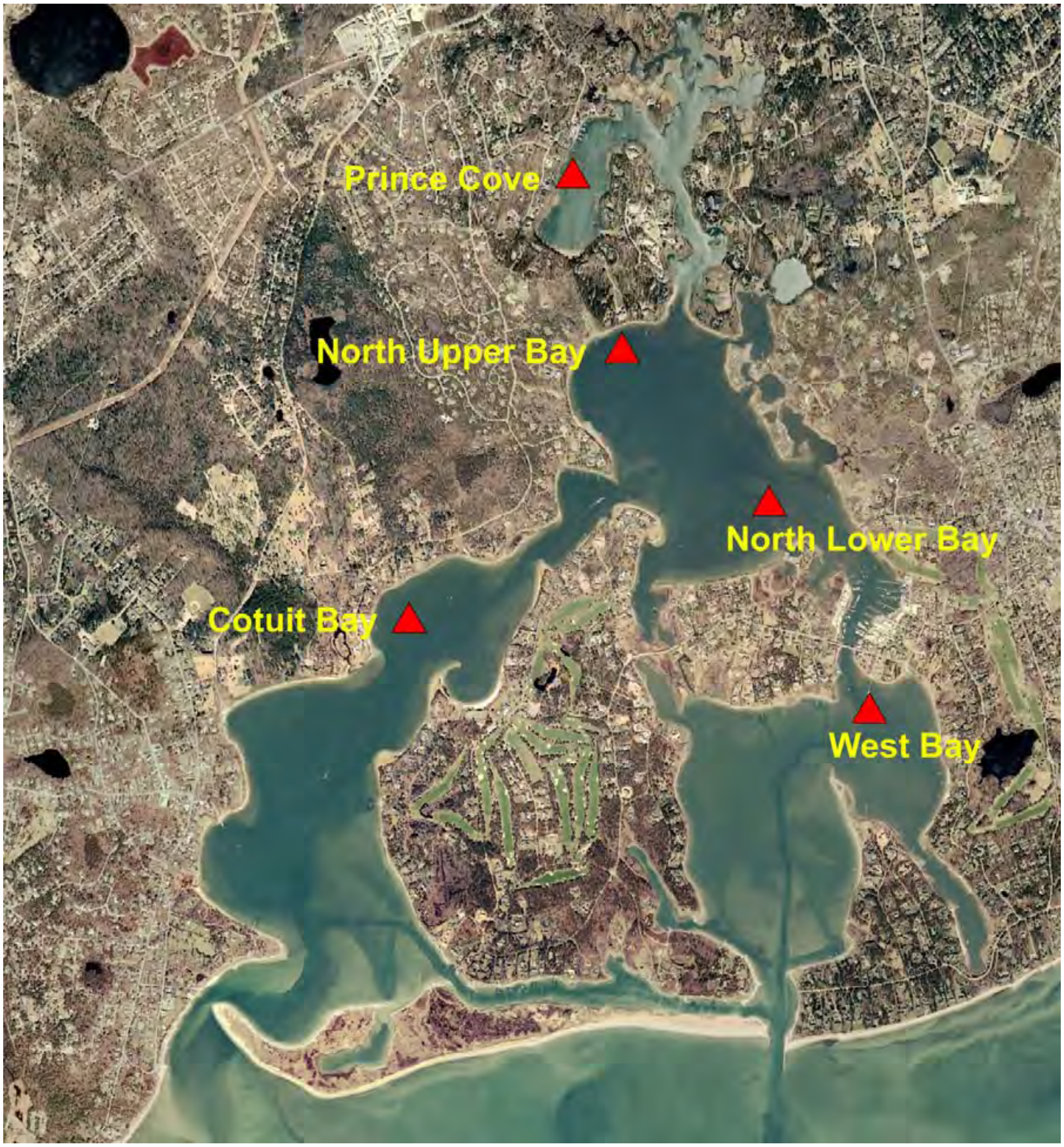


Figure VII-2. Aerial Photograph of the 3 Bays embayment system in the Towns of Mashpee and Barnstable showing locations of Dissolved Oxygen mooring deployments conducted in the Summer of 2002.

oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). This was also periodically within the upper basins, further supporting the contention that the upper basins are currently eutrophic. The mooring data also shows a gradient of impairment with high levels in the upper sub-embayments (Prince Cove, Warrens Cove, North Bay) and better conditions in the lower basins (Cotuit Bay and West Bay). However, there was clear oxygen depletion at all mooring sites, which indicates that additional nitrogen loading will cause further habitat decline at all sites

The dissolved oxygen records indicate that the major subembayments to the Three Bays system (Cotuit Bay, West Bay, North Bay and Prince Cove) are currently under seasonal oxygen stress, consistent with nitrogen enrichment (Table VII-1). That the cause is nitrogen enrichment is supported by parallel observations of chlorophyll a (Table VII-2). Oxygen conditions and chlorophyll a levels generally improved with decreasing distance to the tidal inlet, although all basins showed oxygen depletions to <4 mg L^{-1}. There was also a clear gradient in chlorophyll a, with highest levels in the uppermost reaches and lowest levels near the tidal inlet to Vineyard Sound. The results of the summer oxygen and chlorophyll a studies are consistent with the absence of eelgrass throughout the Three Bays System and the near absence of animal communities throughout the upper basins where oxygen depletions routinely dropped below 3 mg/L (see below).

Table VII-1. Bottom water dissolved oxygen levels within the principal sub-embayments to the Three Bays Estuary. Percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels during summer deployments, 2000-02.

Sub-Embayment	Dissolved Oxygen: Continuous Record, Summer 2000- 2002				
	Deployment Days	**< 6 mg/L (% of days)**	**< 5 mg/L (% of days)**	**< 4 mg/L (% of days)**	**< 3 mg/L (% of days)**
Princes Cove	14.3	60%	39%	27%	14%
North Bay Upper	31.1	66%	44%	24%	11%
North Bay Lower	27	46%	9%	1%	0%
Cotuit Bay	10.0	73%	25%	1%	0%
West Bay	25.8	49%	24%	19%	9%

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

Sub-Embayment	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)	Mean Chl a Level (ug/L)
Princes Cove	8/9/2000	8/23/2000	14.3	**100%**	**93%**	**84%**	**63%**	**39%**	
		Mean		14.3	1.33	1.00	0.39	0.20	23.0
		S.D.		N/A	1.62	1.04	0.59	0.17	
North Bay									
Upper	8/19/2001	9/19/2001	31.1	**95%**	**68%**	**30%**	**10%**	**2%**	
		Mean		2.26	1.01	0.42	0.19	0.11	12.7
		S.D.		6.08	2.54	0.33	0.12	0.10	
Lower	8/16/2001	9/12/2001	27	**89%**	**24%**	**4%**	**0%**	**0%**	**8.3**
		Mean		0.89	0.20	0.13	N/A	N/A	
		S.D.		2.03	0.22	0.07	N/A	N/A	
Cotuit Bay	7/13/2002	8/10/2002	28.1	**81%**	**32%**	**8%**	**2%**	**0%**	
		Mean		1.26	0.25	0.13	0.08	0.04	8.8
		S.D.		3.08	0.29	0.15	0.05	N/A	
West Bay	7/15/2002	8/10/2002	25.8	**72%**	**14%**	**0%**	**0%**	**0%**	
		Mean		0.81	0.25	0.08	N/A	N/A	6.8
		S.D.		1.34	0.19	N/A	N/A	N/A	

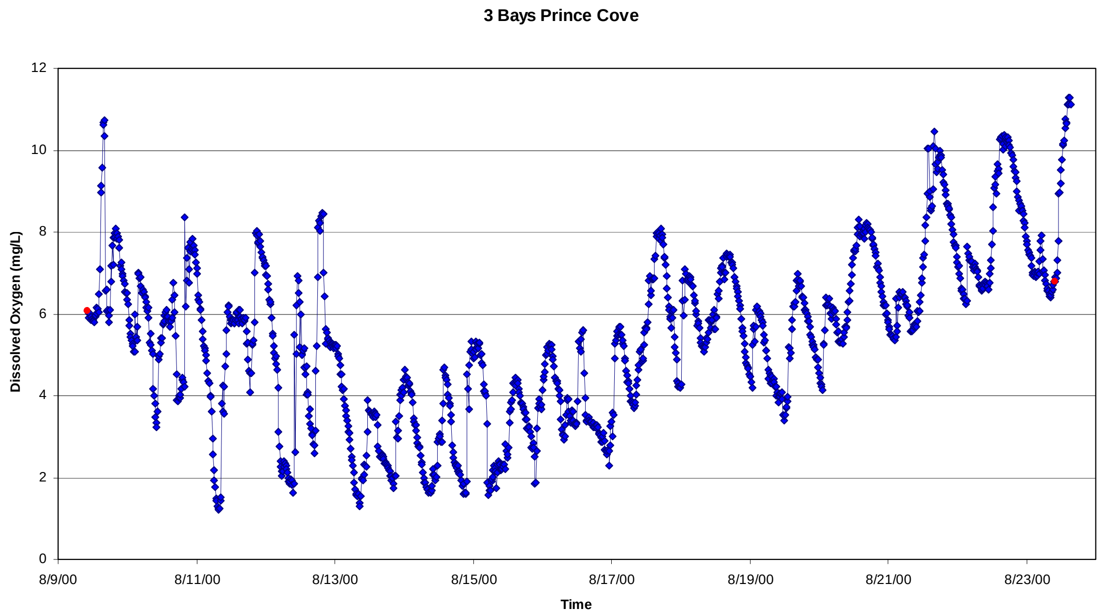


Figure VII-3. Bottom water record of dissolved oxygen at the Three Bays Prince Cove station, Summer 2000. Calibration samples represented as red dots.

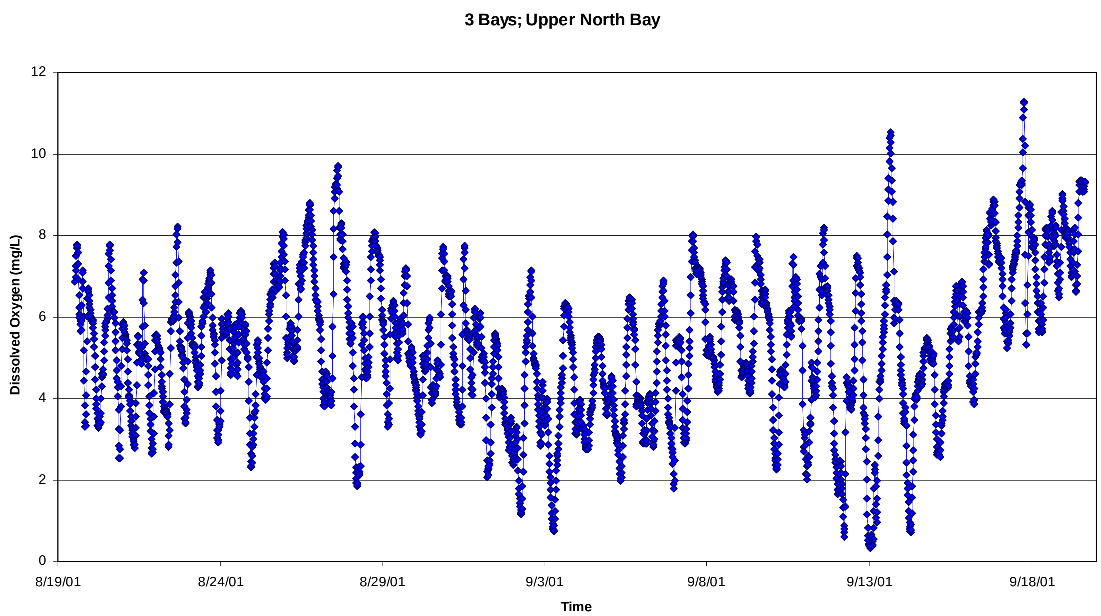


Figure VII-4. Bottom water record of dissolved oxygen in the Three Bays Upper North Bay station, Summer 2001. Calibration samples represented as red dots.

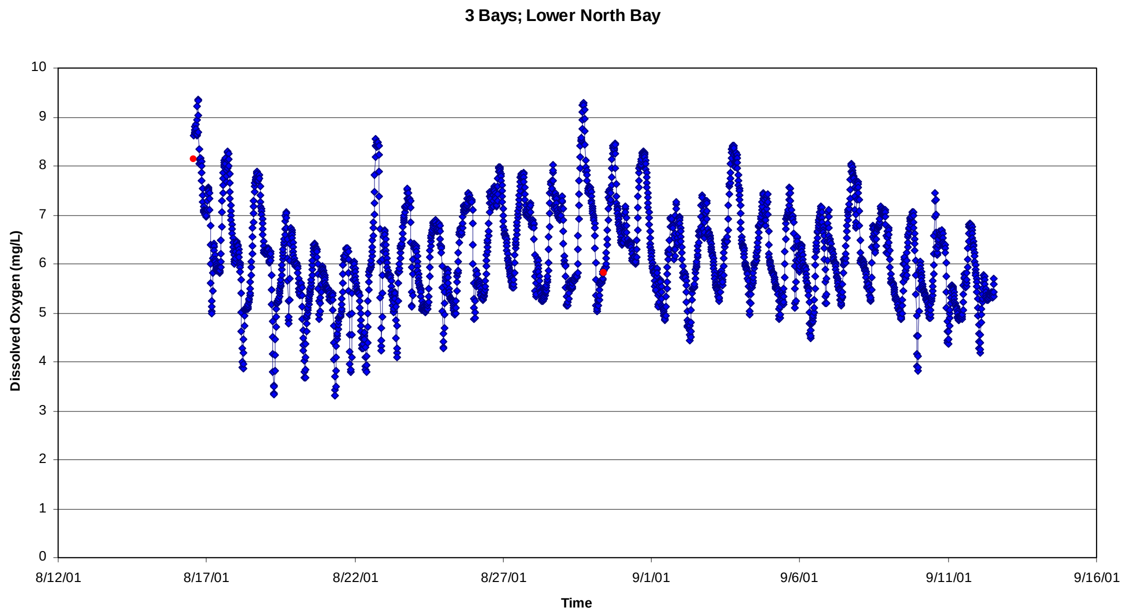


Figure VII-5. Bottom water record of dissolved oxygen in the Three Bays Lower North Bay station, Summer 2001. Calibration samples represented as red dots.

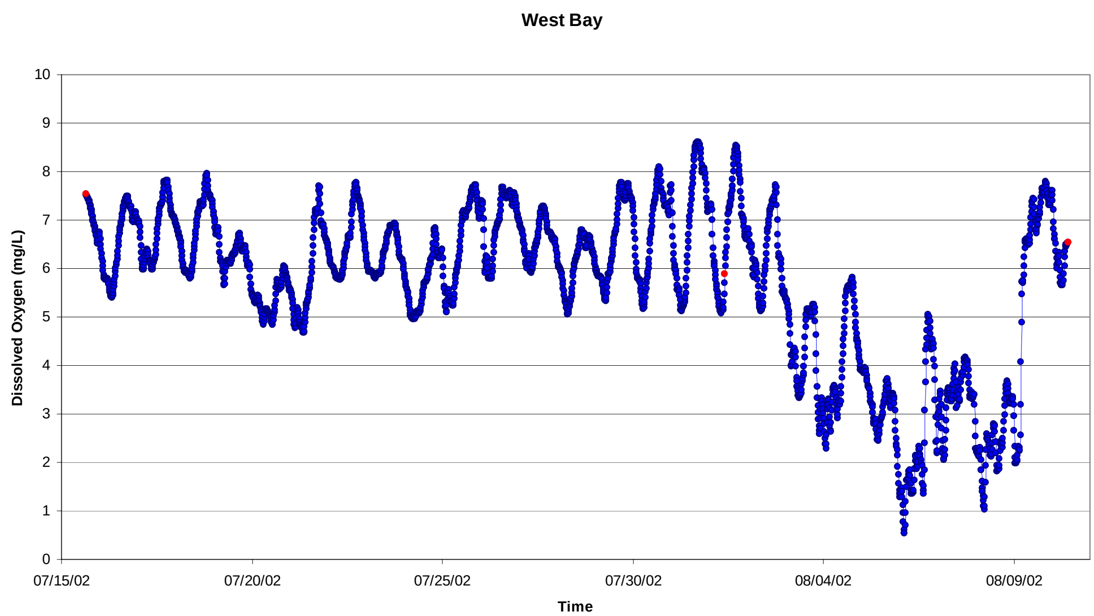


Figure VII-6. Bottom water record of dissolved oxygen in the Three Bays West Bay station, Summer 2002. Calibration samples represented as red dots.

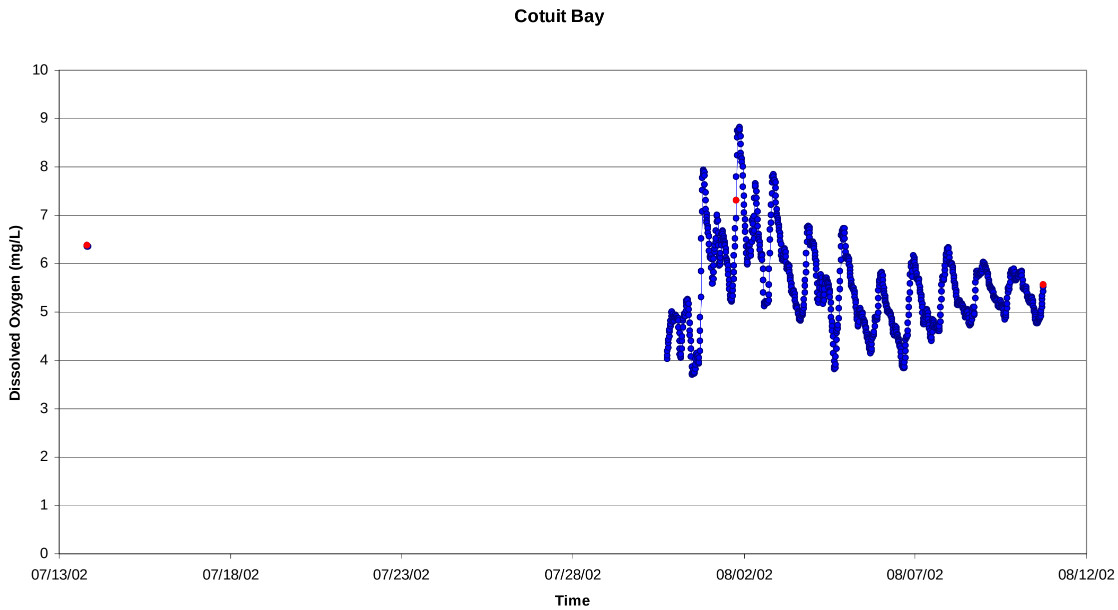
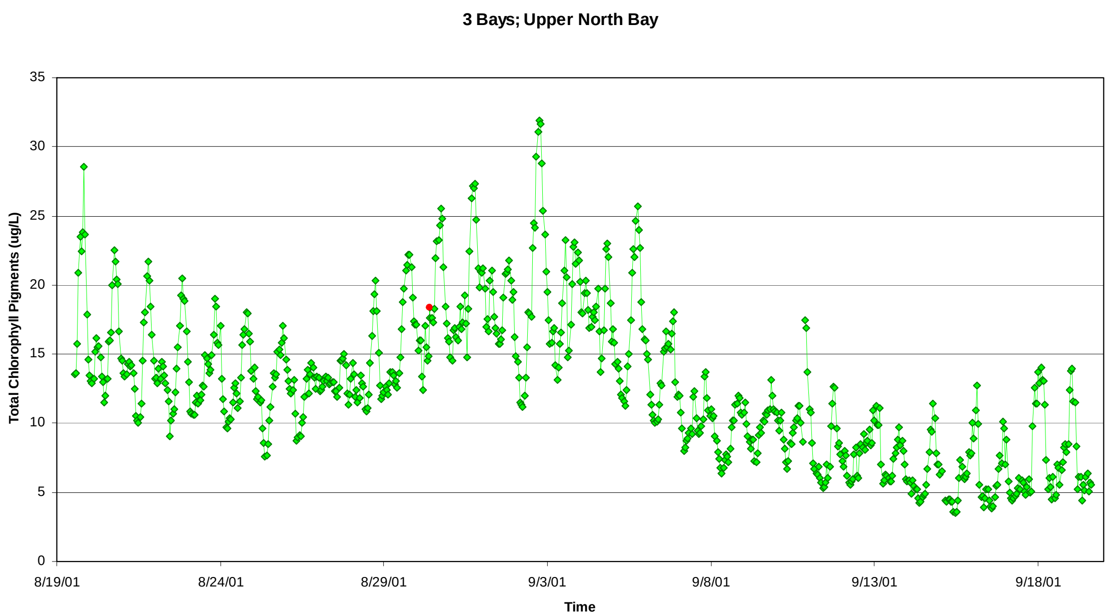


Figure VII-7. Bottom water record of dissolved oxygen in the Three Bays Cotuit Bay station, Summer.



2002 Calibration samples represented as red dots

Figure VII-8. Bottom water record of Chlorophyll-*a* at the Three Bays Prince Cove station, Summer 2000. Calibration samples represented as red dots.

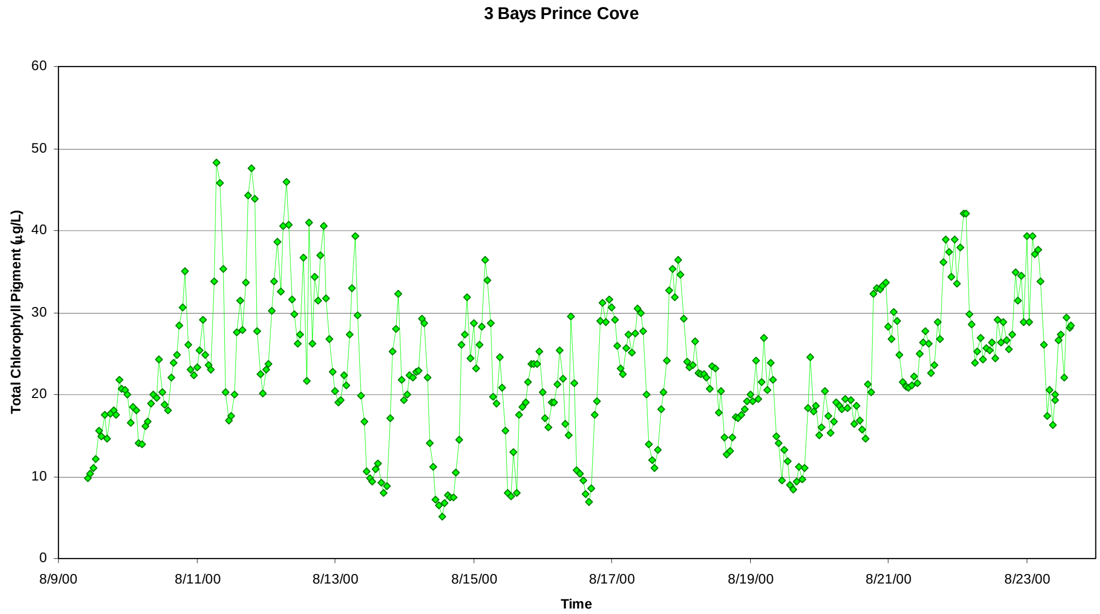


Figure VII-9. Bottom water record of Chlorophyll-*a* in the Three Bays Upper North Bay station, Summer 2001. Calibration samples represented as red dots.

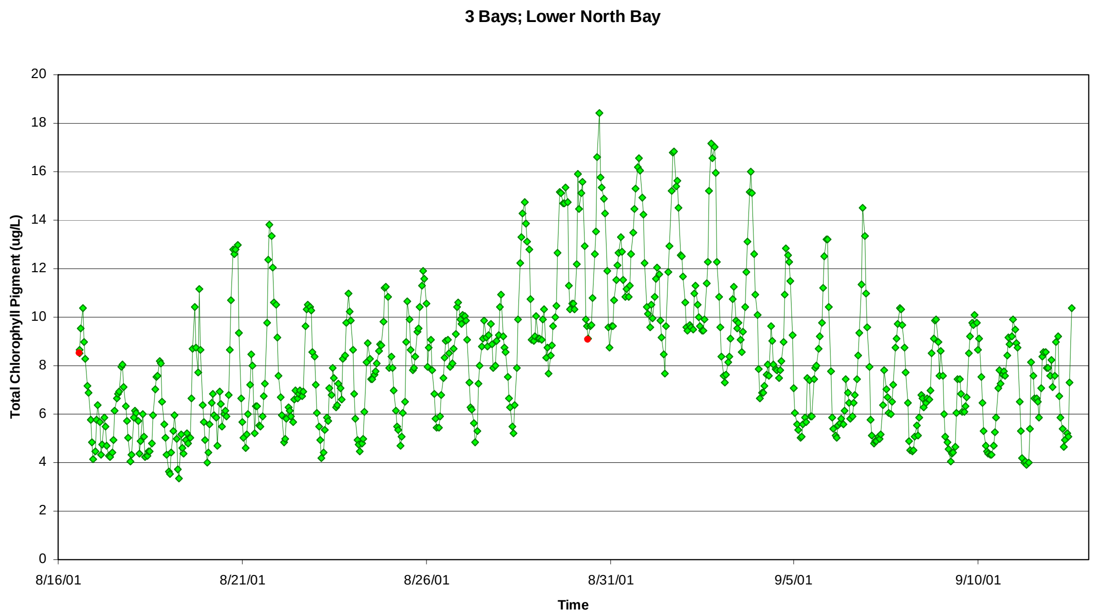


Figure VII-10. Bottom water record of Chlorophyll-*a* in the Three Bays Lower North Bay station, Summer 2001. Calibration samples represented as red dots.

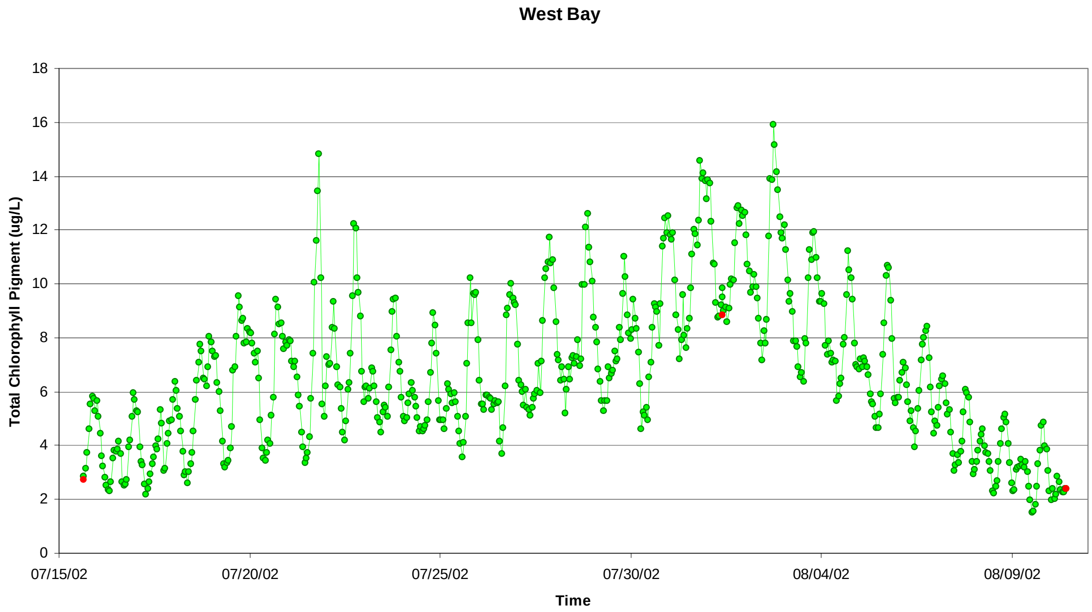


Figure VII-11. Bottom water record of Chlorophyll-*a* in the Three Bays West Bay station, Summer 2002. Calibration samples represented as red dots

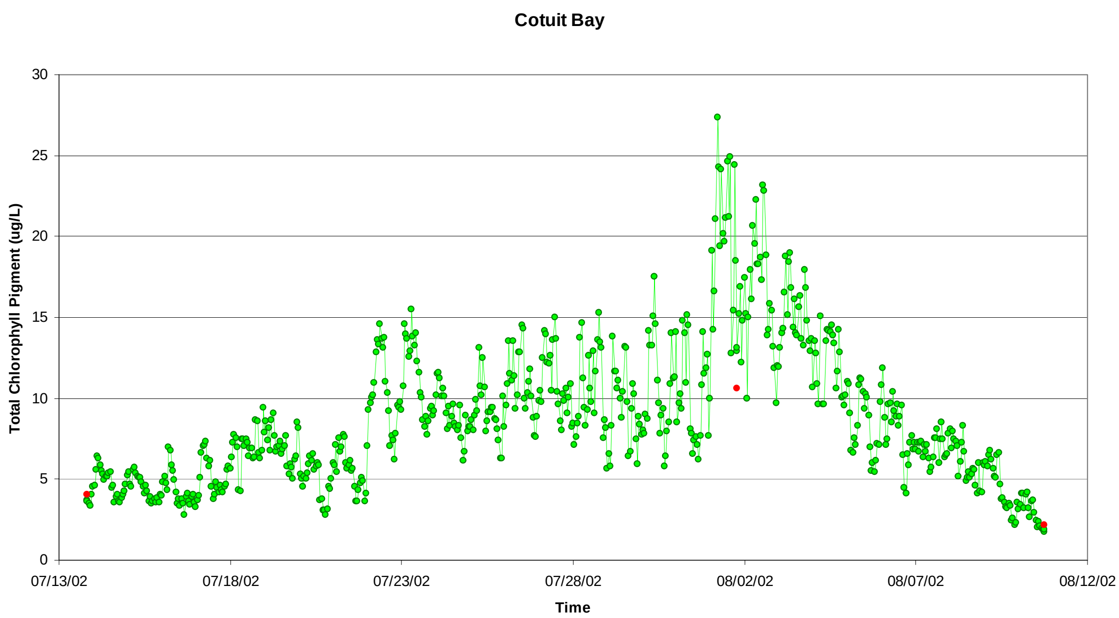


Figure VII-12. Bottom water record of Chlorophyll-*a* in the Three Bays Cotuit Bay station, Summer 2002. Calibration samples represented as red dots.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted for the Three Bays System by the DEP Eelgrass Mapping Program as part of the MEP Technical Team. Surveys were conducted in 1995 and 2001, as part of this program. Additional analysis of available aerial photos from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. The 1951 data were only anecdotally validated, while the 1995 and 2001 maps were field validated. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred or are presently underway. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figure VII-13 and VII-14); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

At present, eelgrass beds are not present within the Three Bays System. In addition, to the DEP mapping, this has been confirmed by the multiple MEP staff conducting the infaunal and sediment sampling and the mooring studies and by the Town of Barnstable Shellfish Department. The final remaining sparse patches in the shallows of Prince Cove have not been seen since 2003. The current lack of eelgrass beds is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within this system. However, it appears that all of the major subembayments had water quality conditions capable of supporting eelgrass (except in the deeper channels and basin depths) in 1951. However, eelgrass appears to have been restricted to the shallows (North and Cotuit Bays) or to Prince Cove and West Bay basins. If the issue in 1951 was nitrogen enrichment, the pattern of the beds would have been very different, with more eelgrass in lower Cotuit Bay and West Bay and much less in Prince Cove and North Bay (except in the very shallows). Instead, it is likely that another factor may have been acting on the distribution, possibly related to the disturbance related to activities in North and Cotuit Bays associated with training during WWII. The landing craft training area in North Bay and the deep central basin of North Bay support distribution observed. However, whatever the cause, it is clear that Three Bays was once capable of supporting eelgrass within each of its major subembayments. It also appears that the recent losses (post 1951) are associated with nitrogen enrichment, as in virtually every other embayment in southeast Massachusetts.

The present absence of eelgrass in each of the major subembayments of the Three Bays System is consistent with the observed oxygen depletions in each basin and the high chlorophyll levels in the upper regions. The ability of the central deep basins in North Bay and Prince Cove are particularly sensitive to eelgrass loss as it takes less intense phytoplankton blooms to reduce light penetration to the bottom, and thereby prevent eelgrass growth. At this time, it is not clear if these regions have historically (100 years) supported eelgrass. However, eelgrass beds fringing these basins are well documented.

Other factors which influence eelgrass bed loss in embayments may also be at play in the Three Bays System, though the recent loss seems completely in-line with nitrogen enrichment. However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss does not seem to be directly related to mooring density, as boat mooring areas are relatively well constrained. Similarly, pier construction and boating pressure may be adding additional stress in nutrient enriched areas, but do not seem to be the overarching factor. It is not possible at this time to determine the potential effect of shellfishing on eelgrass bed distribution, although it should be noted that the Bay has been an important shellfish area for 100's of years during which eelgrass was prevalent.

As for the lack of eelgrass within the lowermost portion of Cotuit Bay, it is likely that it is associated with the highly dynamic coastal processes in this area. This is particularly true in the nearshore to the spit, where there have been several storm overwash events and the need for beach nourishment to maintain the spit. Also, this region needs periodic maintenance dredging of the tidal inlet, without which the tidal flushing of the Bay would be reduced, magnifying the eutrophic conditions of the entire system.

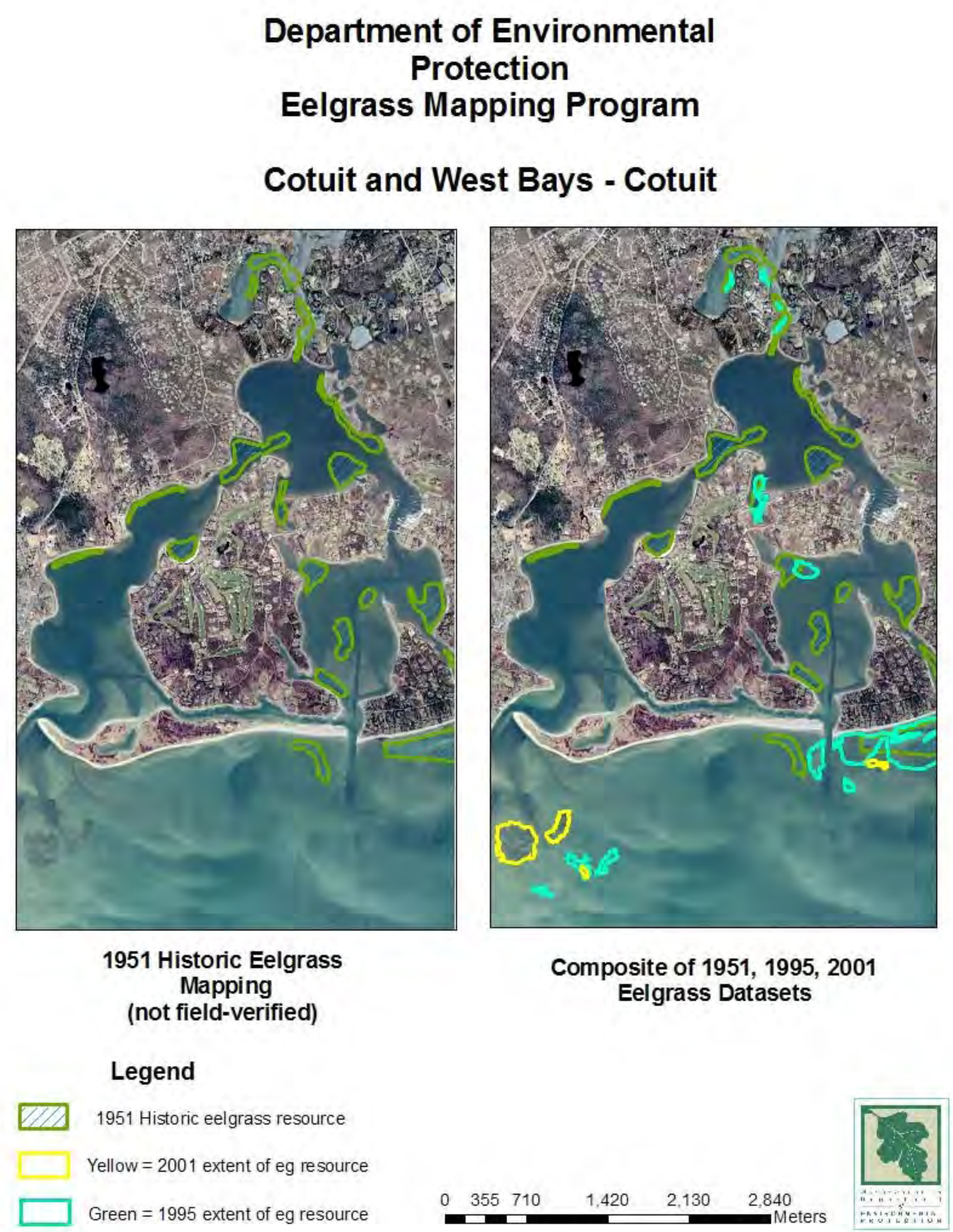


Figure VII-13. Eelgrass bed distribution within the Three Bays System. The 1951 coverage is depicted by the green thatched outline inside of which circumscribes the eelgrass beds. The green (1995) and yellow (2001) areas were mapped by DEP. All data was provided by the DEP Eelgrass Mapping Program.

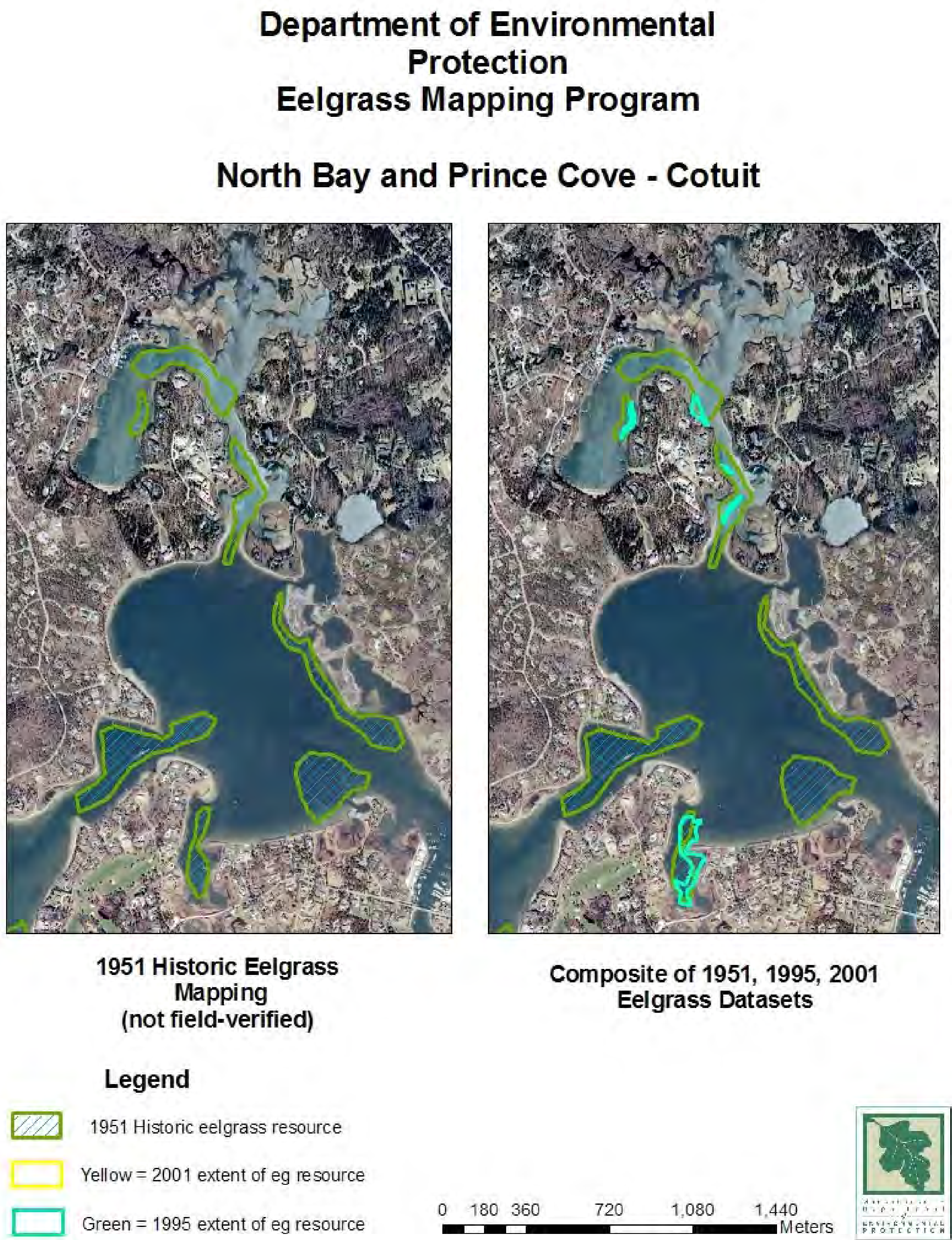


Figure VII-14. Eelgrass bed distribution within North Bay and Prince Cove/Warrens Cove subembayments to the Three Bays System. The 1951 coverage is depicted by the green thatched outline inside of which circumscribes the eelgrass beds. The green (1995) and yellow (2001) areas were mapped by DEP. All data was provided by the DEP Eelgrass Mapping Program.

It is not possible to determine a general idea of short- and long-term rates of change in eelgrass coverage from the mapping data, since there is only limited temporal data and limited presence in recent surveys.. However, it is possible to utilize the 1951 coverage data as an indication that a minimum eelgrass bed area that might be recovered on the order of 130 acres, if nitrogen management alternatives were implemented (Table VII-3). Even more area is likely, if the “natural” pattern of marginal beds can be restored, that were not observed in the 1951 mapping. Even a restoration of fringing beds in North and Cotuit Bays and restoration of West Bay would result in several times the 1951 acreage. Note that restoration of this habitat will necessarily result in restoration of other resources throughout the Three Bays System and in the region of Princes Cove. While it is unlikely that Warrens Cove will support eelgrass after restoration, given its salt marsh basin habitat, the present macroalgal problem would cease,(due to the nitrogen reductions required to restore the eelgrass.

The relative pattern of these data is consistent with the results of the oxygen and chlorophyll a patterns described in the previous section and the benthic infauna analysis, below.

Table VII-3. Changes in eelgrass coverage in the Three Bays system within the Town of Barnstable over the past half century (C. Costello).

EMBAYMENT	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Three Bays	**130.36**	**11.19**	**0.00**	**100%**
There is presently no eelgrass in the Three Bays embayment system.				

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 11 locations throughout the Three Bays System (Figure VII-15). In some cases multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, portions of the Three Bays System are clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records , 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. However, the number of species and individuals must also be taken into account as a high diversity can be achieved in a population decimated by organic loading. Also, as stated above the specific species must also be examined as a large number of stress indicator species (e.g. Capitellids) would be indicative of a stressful environment, even if the number of individuals and species is relatively high.

The Infauna Study indicated that most of the upper areas of the Three Bays system are presently significantly impaired to severely degraded by nitrogen enrichment (Prince Cove, Warrens Cove and portions of North Bay), while the lower basins of Cotuit Bay and West Bay are moderately impaired (Table VII-4).

Prince Cove, Warrens Cove and 2 of 3 sites in North Bay are virtually devoid of infaunal animal communities. They support populations of <50 individuals per 0.0625 m^2, which is more than an order of magnitude less than in a healthy environment. Three of five locations had 11 or less individuals. The central region of North Bay currently supports a transitional community dominated by amphipods, indicative of organic matter enrichment. In contrast, Cotuit and West Bays generally have ~500-2000 individuals per grab and 16-26 species. While there are stress (generally *Capitella* or *Streblospio*) indicator species in numbers at these locations there are also other species indicative of a healthy environment and overall high diversity. Overall, the pattern of infaunal community quality is consistent with the pattern of oxygen depletion and chlorophyll a during summer and the absence of eelgrass. The Three Bays System is clearly significantly impaired to severely degraded in its uppermost basins with higher quality benthic habitat in the lower Cotuit and West Bays. However, all sites showed some level of degradation, either in number of individuals, diversity or the presence of stress indicator species. Lowering nitrogen inputs to this system should allow a relatively rapid recovery of communities in the 2 large lower basins, with higher levels of nitrogen management required to restore benthic habitat to North Bay and Prince Cove and Warrens Cove. These upper basins currently support little to no viable benthic habitat.

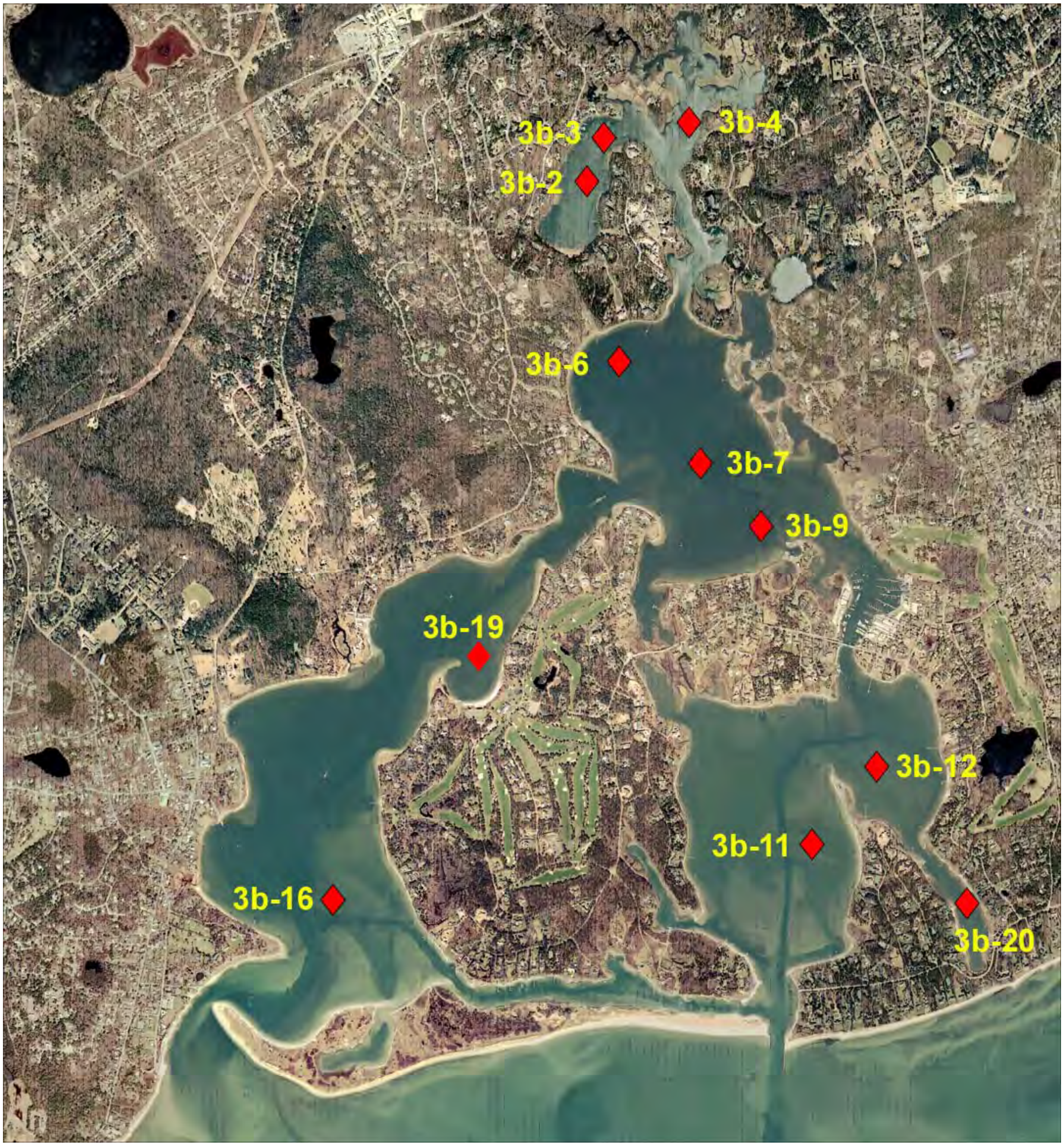


Figure VII-15. Aerial photograph of the Three Bays embayment system showing location of benthic infaunal sampling stations (red symbol).

Table VII-4. Benthic infaunal community data for the Three Bays 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). Values are averages of grab samples a-c.

Location	ID	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)	Infaunal Indicators
Prince Cove							
Upper	2 a,b,c	4.7	43	NA	0.90	0.43	SI
Lower	3 a,b,c	9.3	50	NA	2.49	0.82	SI
Warrens Cove							
Mid	4 a,b,c	4.7	7	NA	2.01	0.86	SI
North Bay							
Upper	6 a,b,c	4.7	11	NA	1.90	0.85	SI
Mid Basin	7 a,b,c	14.3	821	7.5	1.35	0.36	MI
Lower	9 a,b,c	3.0	7	NA	1.91	0.92	SI
Cotuit Bay							
Upper Basin	19 a,b,c	16.3	535	12.6	2.99	0.75	H/MI
Lower Basin	16 a,b,c	16.0	233	14.1	3.26	0.82	H
West Bay							
Upper	12 a,b,c	17.3	501	13.5	3.39	0.82	H/MI
Mid/Lower	11 a,b,c	26.3	1895	10.5	2.02	0.42	H/MI
Eel Pond							
Upper	20 a,b,c	11.0	466	8.9	2.28	0.67	MI

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

VIII.1 ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were collected to support threshold development for the Three Bays System by the MEP Technical Team and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels determined from the baseline Three Bays Water Quality Monitoring Program collected by Three Bays Preservation and the Town of Barnstable. At present, Three Bays, is showing a gradient of significantly impaired (upper basins) to moderately impaired (Cotuit Bay, West Bay) habitat quality. The lower basins show moderate impairment based upon all 3 parameters (eelgrass, infauna, D.O.), in spite of their proximity to the tidal inlet and the high quality waters of Nantucket Sound. All of the habitat indicators show consistent patterns of habitat health in each of the major sub-embayments and that habitat impairments are consistent with nitrogen enrichment (Chapter VII).

Eelgrass: The Three Bays Estuary is relatively deep compared to others along the south shore of Cape Cod from Falmouth to Barnstable (Chapter V). Water depths are well within the range for eelgrass growth in Massachusetts, under suitable conditions of light penetration. However, the need for more transparent waters requires lower nitrogen loads to these deep basins, compared to shallower basins, to support this habitat.

Currently, there are no remaining eelgrass beds within the Three Bays System. However, it appears that all of the major sub-embayments had water quality conditions capable of supporting eelgrass (except in the deeper channels and basin depths) in 1951. However, eelgrass appears to have been restricted to the shallows (North and Cotuit Bays) or to Prince Cove and West Bay basins. If the issue in 1951 was nitrogen enrichment, the pattern of the beds would have been very different, with more eelgrass in lower Cotuit Bay and West Bay and much less in Prince Cove and North Bay (except in the very shallows). Instead, it is likely that disturbance related to activities in North and Cotuit Bays associated with training during WWII played a role in the North and Cotuit Bay pattern of beds in the 1951 assessment. Whatever the cause, it is clear that in the recent past, the Three Bays system was capable of supporting eelgrass within each of its major sub-embayments. It also appears that the recent losses (post 1951) are associated with nitrogen enrichment, as in virtually every other embayment in southeastern Massachusetts. The absence of eelgrass in each basin and the fact that they supported eelgrass in the recent past classifies each basins eelgrass habitat as “significantly impaired” (Table VIII-1).

The current absence of eelgrass in each of the major sub-embayments of the Three Bays System is consistent with the observed oxygen depletions in each basin and the high chlorophyll levels in the upper regions. The greater depths in the Three Bays Estuary also makes oxygen depletions more likely than in shallow basins with the same nitrogen levels. This results from the fact that deeper systems are more likely to periodically stratify. The central deep basins in North Bay and Prince Cove are particularly sensitive to eelgrass loss as it takes less intense phytoplankton blooms to reduce light penetration to the bottom, and thereby prevent eelgrass growth. In addition, the basins are sensitive to periodic oxygen depletion. At this time, it is not clear if these regions have historically (100 years) supported eelgrass. However, eelgrass beds fringing these basins are well documented. As regards the lack of eelgrass within the lowermost portion of Cotuit Bay and the Seapuit River, it is likely associated with the documented highly dynamic coastal processes in this area. The level of natural disturbance in this region is very high (sand transport, overwash, etc). Physical stability is important to the ability of eelgrass beds to form and persist. Nitrogen levels in lower Cotuit Bay would presently support eelgrass habitat (tidally averaged TN 0.32 mg L-1) as they are much lower than those in many other eelgrass beds in nearby systems that are physically stable. In addition, the persistence of eelgrass beds through 1995-2001 in the shallow waters of south Windmill Cove, but in a stable physical setting, were at much higher nitrogen levels (tidally averaged TN 0.40 mg L-1).

Table VIII-1. Summary of Nutrient Related Habitat Health within the Three Bays Estuary on the south shore of Barnstable , MA., based upon assessment data presented in Chapter VII.

Health Indicator	Three Bays Estuary						
	Prince Cove	Warrens Cove	North Bay		Cotuit Bay	West Bay	Eel Pond
			Upper	Lower			
Dissolved Oxygen	SI/SD[1]	SI/SD	SI/SD[1]	MI/SI[2]	MI/SI[2]	SI	H/MI[3]
Chlorophyll	SI	SI	MI/SI	MI/SI	MI	H/MI	--
Macroalgae	MI	SD[4]	--	--	MI[6]	MI	SI
Eelgrass	SI	SI	SI	SI	SI	SI	SI
Infaunal Animals	SD[7]	SD[8]	SD[8]	MI/SI	H/MI	H/MI	MI
Overall:	**SI/SD**	**SD**	**SI/SD**	**MI/SI**	**MI**	**MI**	**MI/SI**

1 – periodic oxygen depletions to <2 mg/L and frequently <4 mg/L.
2 – infrequent oxygen depletions to 3-4 mg/L, periodic 4-5 mg/L., generally >5 mg/L.
3 – generally >5 mg/L., grab sample data only.
4 – macroalgal floating accumulations during summer
5 – moderate macroalgal accumulations on bottom.
6 – low to moderate patches on bottom only.
7 – modest numbers of individuals dominated by stress indicator species.
8 – absence of infaunal community (<15 individuals/grab).
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation
-- = not applicable to this estuarine reach

Nitrogen management of this system is likely to restore at least an area equivalent to the 130 acres observed in 1951 (Table VII-3). Even more area is likely, if the “natural” pattern of marginal beds can be restored (not observed in the 1951 mapping). Even a restoration of fringing beds in North and Cotuit Bays and restoration of West Bay would result in several times the 1951 acreage. Note that restoration of this habitat will necessarily result in restoration of other resources throughout the Three Bays System and in the region of Princes Cove. While it is unlikely that Warrens Cove will support eelgrass after restoration, given its salt marsh basin habitat, the present macroalgal problem would cease due to the nitrogen reductions required to restore the eelgrass. Eelgrass recovery following nitrogen management would likely follow the pattern of beds first being re-established in the marginal areas in the lower basins and move to the deeper regions and the margins of the upper sub-embayments.

Water Quality: The dissolved oxygen (DO) records indicate that the major sub-embayments to the Three Bays system (Cotuit Bay, West Bay, North Bay and Prince Cove) are currently under seasonal oxygen stress, consistent with nitrogen enrichment (Table VII-1). That the cause is nitrogen enrichment is supported by parallel observations of chlorophyll a (Table VII-2). Oxygen conditions and chlorophyll a levels generally improved with decreasing distance to the tidal inlet, although all basins showed oxygen depletions to <4 mg L^{-1}. There was also a clear gradient in chlorophyll a, with highest levels in the uppermost reaches and lowest levels near the tidal inlet to Nantucket Sound. The results of the summer oxygen and chlorophyll a studies are consistent with the absence of eelgrass throughout the Three Bays System and the near absence of animal communities throughout the upper basins where oxygen depletions routinely dropped below 3 mg/L (see below). These observations are consistent with a classification of the upper basins (North Bay and Prince Cove and Warrens Cove) as generally “Significantly Impaired” and the lower basins (Cotuit Bay, West Bay) as “Moderately Impaired”.

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 3.8 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Three Bays System are currently listed as SA under the State Classification.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality at all MEP DO mooring sites within the estuary (Figures VII-3 through VII-12). The oxygen data throughout the estuary is consistent with elevated organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of these estuarine systems. The oxygen records further indicate that the upper tidal reaches of each estuary have the largest daily oxygen excursion, with daily excursions in excess of 6 mg L-1 common. This further supports the assessment of a high degree of nutrient enrichment.

The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). This was periodically observed within the upper basins, further supporting the contention that the upper basins are currently eutrophic. The oxygen and chlorophyll data also shows a gradient of impairment with high levels of impairment in the upper sub-embayments (Prince Cove, Warrens Cove, North Bay) and better conditions in the lower basins (Cotuit Bay and West Bay). However, there was clear oxygen depletion at all mooring sites, which indicates that additional nitrogen loading will cause further habitat decline at all sites

Infaunal Communities: The Infauna Study indicated that most of the upper areas of the Three Bays system are presently significantly impaired to severely degraded by nitrogen enrichment

(Prince Cove, Warrens Cove and portions of North Bay), while the lower basins of Cotuit Bay and West Bay are moderately impaired (Table VII-4).

Prince Cove, Warrens Cove and 2 of 3 sites in North Bay are virtually devoid of infaunal animal communities. They support populations of <50 individuals per 0.0625 m^2, which is more than an order of magnitude less than in a healthy environment. Three of five locations had 11 or less individuals. The central region of North Bay currently supports a transitional community dominated by amphipods, indicative of organic matter enrichment. In contrast, Cotuit and West Bays generally have ~500-2000 individuals per grab and 16-26 species. While there are stress indicator species (generally *Capitella* or *Streblospio*) in numbers at these locations there are also other species indicative of a healthy environment and overall high diversity. Overall, the pattern of infaunal community quality is consistent with the pattern of oxygen depletion and chlorophyll a during summer and the absence of eelgrass. All sites showed some level of degradation, either in number of individuals, diversity or the presence of stress indicator species. Lowering nitrogen inputs to this system should allow a relatively rapid recovery of communities in the Cotuit and West Bays, with higher levels of nitrogen management required to restore benthic habitat to North Bay and Prince Cove and Warrens Cove. These upper basins currently support little to no viable benthic habitat. The infaunal community based classification for each sub-embayment is fully supported by the water quality and eelgrass data discussed in the text above.

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

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

For the Three Bays System, the restoration target should reflect both recent pre-degradation habitat quality and be reasonably achievable. Based upon the assessment data (Chapter VII), eelgrass bed restoration within Cotuit Bay and West Bay, with restoration of marginal beds in North Bay and Prince Cove is supportable. In addition, in the central basins of North Bay and Prince Cove, where eelgrass habitat has not been documented, as well as in Warrens Cove, restoration of infaunal habitat is necessary. Achieving these habitat quality targets will also result in mitigation of the present macroalgal accumulation problem in Warrens Cove.

To achieve these habitat restoration targets, for the Three Bays Estuary a single sentinel location was selected with secondary criteria that must be achieved at other locations. The secondary criteria serve only as checks to make sure that the targets are achieved when the nitrogen threshold at the sentinel station has been reached. The appropriate sentinel location within Three Bays was determined to be in the upper region of the narrows between North Bay and Cotuit Bay (at the entrance to the Narrows). This location was selected because (1) it is relatively deep (reflecting the larger Three Bays basins) and it supported a major eelgrass bed in the 1951 survey; (2) achieving the threshold nitrogen level at this location will result in high quality habitat conditions throughout Cotuit and West Bays; (3) restoration of nitrogen concentrations at this location should result in conditions similar to 1951 within Prince and Warrens Coves and North Bay; (4) nitrogen levels restorative of eelgrass beds at the deeper sentinel location should provide for marginal beds in the shallows of Prince Cove and North Bay and (5) achieving the threshold nitrogen level at the sentinel location will require removal of sufficient nitrogen related stress as to restore infaunal animal habitat in the adjacent deeper waters of Prince Cove and North Bay.

The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.38 mg TN L^{-1} at the sentinel location and 0.40 mg TN L^{-1} within the marginal regions (shallows) of North Bay. This secondary level to check restoration of marginal beds in North Bay (O.40 mg TN L^{-1}) is consistent with the analysis of restoration of fringing eelgrass beds in nearby Great Pond, and analysis where eelgrass beds in deep waters could not be supported at a tidally averaged TN of 0.412 mg TN L^{-1} at depths of 2 m. Similarly prior MEP analysis in Bournes Pond indicated that tidally averaged TN levels of 0.42 mg TN L^{-1} excluded beds from all but the shallowest water. The MEP Technical Team cannot specify the exact extent of marginal beds to be restored in the upper deep basins. At tidally averaged TN levels of 0.42 mg TN L^{-1} the eelgrass habitat would be restricted to very shallow waters, while at 0.40 mg TN L^{-1} the eelgrass habitat should reach to 1-2 meters depth, based upon the data from nearby systems. In addition, the persistence of eelgrass beds through 1995-2001 in the shallow waters of south Windmill Cove, but in a stable physical setting, were at nitrogen levels (tidally averaged TN ~0.40 mg L-1).

The target nitrogen concentration for restoration of eelgrass at the sentinel location system was determined to be 0.38 mg TN L^{-1}. It was not possible to make this determination based upon an analysis of the relationship of measured nitrogen levels to existing eelgrass beds in Three Bays, as all beds have been lost. Instead, the value stems from (1) the analysis of Stage Harbor, Chatham which also exchanges tidal water with Nantucket Sound and for which a MEP target has already been set), (2) analysis of nitrogen levels within the eelgrass bed in adjacent Waquoit Bay, near the inlet (measured TN of 0.395 mg N L^{-1}, tidally corrected <0.38 mg N L^{-1}), and (3) a similar analysis in West Falmouth Harbor. The sentinel station under present loading conditions supports a measured nitrogen level at mid-ebb tide of 0.438-0.498 mg TN L^{-1} and a tidally corrected average concentration of 0.485 mg TN L^{-1},

Since infaunal animal habitat is also a critical resource to the Three Bays System, the secondary metric for a successful restoration (after eelgrass) will be to restore the significantly impaired/severely degraded habitats in the Prince Cove/Warrens Cove and North Bay basins. In the upper more muddy basins of other nearby systems, healthy infaunal habitat is associated with nitrogen levels of TN <0.5 mg TN L^{-1}. This was found for Popponesset Bay where based upon the infaunal analysis coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 were found supportive of high infaunal habitat quality in this system.

In the Three Bays System, present healthy infaunal areas are found at nitrogen levels of TN <0.42 mg TN L^{-1} (Cotuit Bay and West Bay) However, the impaired areas are at nitrogen levels of TN >0.5 mg TN L^{-1} (North Bay) and are severely degraded at nitrogen levels of TN >0.6 mg TN L^{-1}. This is consistent with the findings discussed above from other systems and fully supports a secondary nitrogen criteria for the upper muddy basins of 0.5 mg TN L^{-1}.

It must be stressed that the nitrogen threshold for the Three Bays System is at the sentinel location. The secondary criteria should be met when the threshold is met at the sentinel station and also serve as a “check”. The nitrogen loads associated with the threshold concentration at the sentinel location are discussed in Section VIII.3, below.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Three Bays system. The load reductions associated with the nitrogen thresholds developed in the previous section only represent one of many different ways to reduce load from the watershed in order to meet the threshold. 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 one example from a suite of potential reduction approaches that need to be evaluated by the communities in the Three Bays system watershed. The purpose of the scenario presented is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

To develop the scenario, 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 Three Bays.

As shown in Table VIII-2 for the specific load reduction scenario (reductions in septic effluent discharges only), the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required 100% removal of septic load (associated with direct groundwater discharge to the embayment) for five of the eight total lower sub-watersheds of the system. In addition, a portion of the septic load entering the headwaters of the system from the Marstons Mills River also must be removed to meet the threshold nitrogen concentrations. For the load reduction scenario evaluated, the Marstons Mills River watershed required removal of approximately 25% of the septic load. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figures VIII-1 and VIII-2.

Tables VIII-3 and VIII-4 provide additional loading information associated with this thresholds scenario. 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 100% of the septic load from the Warrens Cove sub-watershed results in an 58% reduction in total nitrogen load. For the Marstons Mills River, septic load reduction of 25% resulted in total attenuated watershed load reduction of 17%. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent ‘worst-case’ summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Nantucket Sound.

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-5. To achieve the threshold nitrogen concentrations at the sentinel station, a reduction in TN concentration of typically greater than 20% is required for the upper portions of the system, in North Bay and Prince Cove.

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

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

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading in one possible load reduction scenario for the Three Bays system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Cotuit Bay	17.022	13.618	-20.0%
West Bay	15.490	12.392	-20.0%
Seapuit River	2.921	2.921	0.0%
North Bay	24.978	0.000	-100.0%
Prince Cove	11.192	0.000	-100.0%
Warrens Cove	6.975	0.000	-100.0%
Prince Cove Channel	4.767	0.000	-100.0%
Marstons Mills R. South (below Mill Pond)	3.573	0.000	-100.0%
Surface Water Sources			
Marstons Mills River	10.071	7.553	-25.0%
Little River	3.203	3.203	0.0%

Table VIII-3. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading in one possible load reduction scenario for the Three Bays 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
Cotuit Bay	21.778	18.374	-15.6%
West Bay	19.068	15.970	-16.2%
Seapuit River	3.767	3.767	0.0%
North Bay	29.447	4.468	-84.8%
Prince Cove	13.362	2.170	-83.8%
Warrens Cove	12.027	5.052	-58.0%
Prince Cove Channel	5.537	0.770	-86.1%
Marstons Mills R. South (below Mill Pond)	7.293	3.721	-49.0%
Surface Water Sources			
Marstons Mills River	14.518	12.000	-17.3%
Little River	3.962	3.962	0.0%

Table VIII-4. Threshold sub-embayment and surface water loads used for total nitrogen modeling of the Three Bays system under one possible scenario, with sub-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)
Cotuit Bay	18.374	5.786	-45.788
West Bay	15.970	4.233	3.469
Seapuit River	3.767	0.452	-5.371
North Bay	4.468	3.953	45.202
Prince Cove	2.170	1.230	0.323
Warrens Cove	5.052	-	6.225
Prince Cove Channel	0.770	-	1.541
Marstons Mills Crescent	3.721	-	-
Surface Water Sources			
Marstons Mills River	12.000	-	-
Little River	3.962	-	-

Table VIII-5. Comparison of model average total N concentrations from present loading and the threshold scenario (reduction in septic effluent discharge only), with percent change, for the Three Bays system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Prince Cove - south	TB2	0.695	0.460	-33.8%
Prince Cove - north	TB3	0.639	0.446	-30.1%
Warrens Cove	TB4	0.595	0.433	-27.2%
North Bay - north	TB5	0.518	0.400	-22.9%
North Bay - south	TB6	0.500	0.392	-21.7%
North Windmill Cove	TB7	0.511	0.396	-22.5%
West Bay - north	TB8	0.363	0.326	-10.1%
West Bay - west	TB9	0.327	0.307	-6.1%
Eel River	TB10	0.486	0.427	-12.2%
Seapuit River	TB11	0.295	0.287	-2.6%
Cotuit Bay - north	TB12	0.414	0.346	-16.5%
Cotuit Bay - south	TB13	0.321	0.298	-7.1%
South Windmill Cove	TB15	0.402	0.364	-9.5%
Mellon Cove	TB16	0.392	0.367	-6.2%
Dam Pond	TB17	0.523	0.402	-23.3%

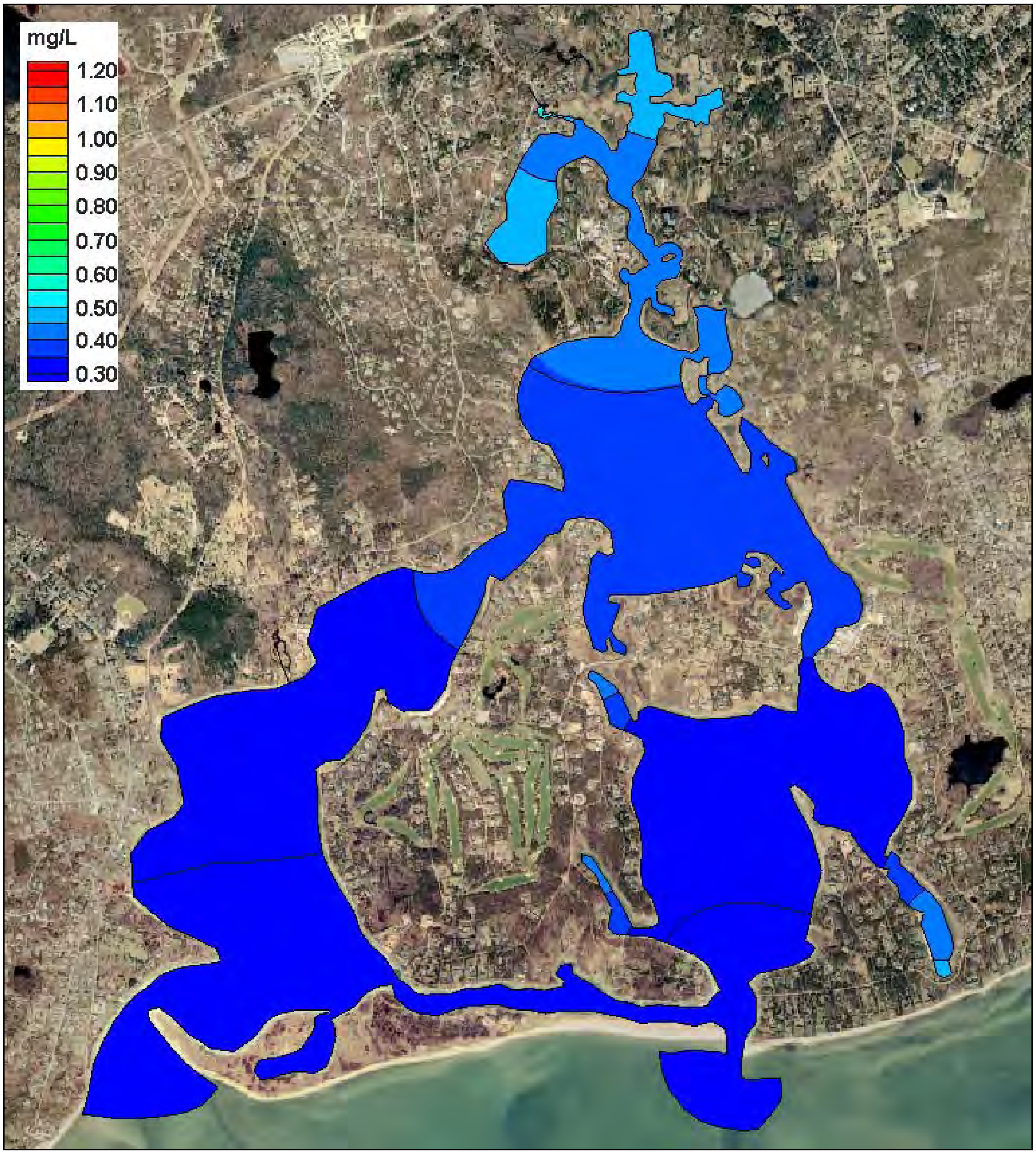


Figure VIII-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Three Bays system, for threshold conditions (0.38 mg/L at the narrows between North Bay and Cotuit Bay).

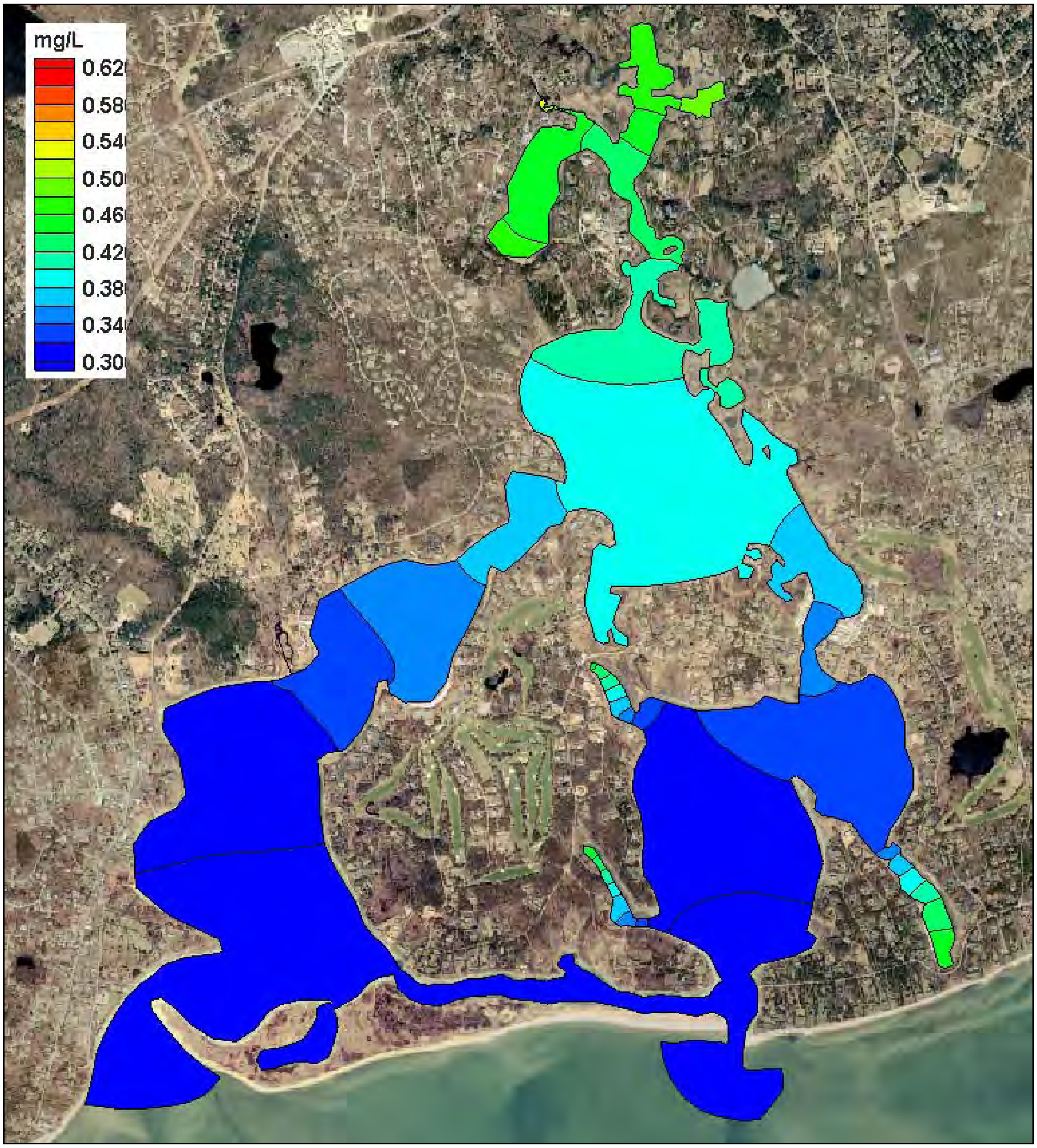


Figure VIII-2. Same results as for Figure VIII-1, but shown with finer contour increments for emphasis. Contour plot of modeled total nitrogen concentrations (mg/L) in the Three Bays system, for threshold conditions (0.38 mg/L at the narrows between North Bay and Cotuit Bay).

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

IX.1 DREDGING OF COTUIT BAY INLET

An investigation of water quality impacts resulting from a proposed widening of Cotuit Bay inlet was performed using the existing calibrated hydrodynamic and total nitrogen model of the Three Bays system. For this dredging scenario, the inlet to Cotuit Bay is widened by 300 feet, and to a depth of 8 feet NGVD. This would be achieved by removing the western tip of Sampsons Island (attached to Dead Neck), which is an accreting sand spit, supplied with sand from the updrift (eastern) portion of the island. Widening the inlet would alleviate erosional pressure on the western shore of the inlet, while returning the inlet to conditions that existed in the 1960's. It was also considered possible that the proposed dredging would benefit tidal exchange between Three Bays and Nantucket Sound, and therefore improve water quality in the system.

To quantitatively assess the water quality impacts resulting from dredging the inlet, the Three Bays hydrodynamic model was modified to include the improvements at the inlet and then re-run with the same offshore tidal boundary conditions as was used for the model runs of present conditions. A comparison of hydrodynamic model output for present and post-dredge model output is presented in Table IX-1. The resulting changes to the Three Bays system hydrodynamics due to dredging are very small. As an example, the tide prism of the entire Three Bays system increases only by 0.1%, while its mean volume is essentially unchanged. The resulting changes to computed flushing rates are similarly small.

Table IX-1. Comparison of modeled hydrologic conditions in the Three Bays system for present conditions and the Cotuit Bay Inlet dredging scenario. Computed residence times are shown to three decimal places in order to show the change resulting from the proposed dredging at the inlet.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)	Local Residence Time (days)	System Residence Time (days)
PRESENT				
Three Bays System	429,117,000	140,570,000	1.580	1.580
North Bay	139,666,000	45,824,000	1.577	4.846
Marstons Mills River	25,236,000	10,834,000	1.205	20.497
Prince Cove	13,007,000	4,553,000	1.478	48.774
Warrens Cove	5,047,000	3,614,000	0.723	61.447
Dam Pond	2,798,000	1,200,000	1.207	185.057
Eel River	4,035,000	1,702,000	1.227	130.475
DREDGED COTUIT INLET				
Three Bays System	429,149,000	140,747,000	1.578	1.578
North Bay	139,655,000	45,887,000	1.575	4.840
Marstons Mills River	25,236,000	10,850,000	1.204	20.469
Prince Cove	13,007,000	4,560,000	1.476	48.703
Warrens Cove	5,047,000	3,619,000	0.722	61.366
Dam Pond	2,797,000	1,202,000	1.204	184.763
Eel River	4,034,000	1,706,000	1.224	130.179

The small changes in total system tidal volume flux, resulting from widening the inlet, are not unexpected considering that the analysis of tidal energy distribution in the Three Bays system (section V.2.1) showed that there is very little tidal attenuation between the two inlets to Nantucket Sound and even in the most distant reaches of the system, in Prince Cove. The small degree of tidal attenuation indicates that the system presently flushes efficiently, and therefore increasing the size of either inlet could not significantly increase tidal exchange with Nantucket Sound.

Although the total system-wide tidal volume does not change by a significant amount, there is a quantifiable change in how tidal flows are distributed within the Three Bays system. The distribution of tidal fluxes computed in the model at the two outlet channels from North Bay to West Bay (that the Little Island draw bridge) and Cotuit Bay (at the Narrows) are presented in Table IX-2. For present conditions, 63% of North Bay's total tidal exchange is through the Narrows to Cotuit Bay, for both ebb and flood tides. The remaining 36% is via the channel to West Bay. For the dredging scenario, tidal exchange between North Bay and Cotuit Bay increases further, to about 67% for both the ebb and flood portions of the tide.

A change in flow distribution can also be seen at the inlets to the Three Bays system. As shown in Table IX-3 for present conditions, the total tide exchange between the Three Bays system and Nantucket Sound is evenly split between the two inlets. Dredging of the Cotuit Bay entrance increases the flow through Cotuit inlet to approximately 55% of the system total. Because the total flow in and out of the system remains constant, the flow though West Bay inlet must decrease to 45%. Additional changes occur in the Seapuit River. Based upon the model results, dredging of Cotuit Bay inlet would cause the flow through the Seapuit River to be reduced by approximately 35%, which also means that the maximum velocities in the river channel would be reduced. Model results indicate that maximum channel velocities would be reduced be approximately 25% in the dredged inlet scenario.

Table IX-2. Comparison of the distribution of tidal flows ebbing from and flowing to North Bay (to Cotuit Bay and West Bay) for present conditions and for the Cotuit Bay inlet dredging scenario. Percentages are based on the total hydraulic flux entering or exiting North Bay .

Flow Pathway	Present		Dredged Cotuit Bay Inlet	
	Flood	Ebb	Flood	Ebb
North Bay via Cotuit Bay	62.5%	65.2%	65.9%	68.1%
North Bay via West Bay	37.5%	34.8%	34.1%	31.9%

Table IX-3. Comparison of the distribution of tidal flows ebbing from and flowing to the Three Bays system (via Cotuit Bay inlet and West Bay inlet) for present conditions and for the Cotuit Bay inlet dredging scenario. Percentages are based on the total hydraulic flux entering or exiting the entire Three Bays system .

Flow Pathway	Present		Dredged Cotuit Bay Inlet	
	Flood	Ebb	Flood	Ebb
Cotuit Bay Inlet	49.9%	51.3%	55.41%	54.65%
West Bay Inlet	50.1%	48.7%	44.59%	45.35%

Changes in the Three Bays system resulting from widening Cotuit Bay were further quantified by modeling TN using the modified hydrodynamics. The dredging scenario was modeled using the nitrogen loading distribution and model parameters determined previously for present conditions (Table VI-2). In Figure IX-1, a contour plot is presented that shows TN changes between the dredging scenario and present conditions.

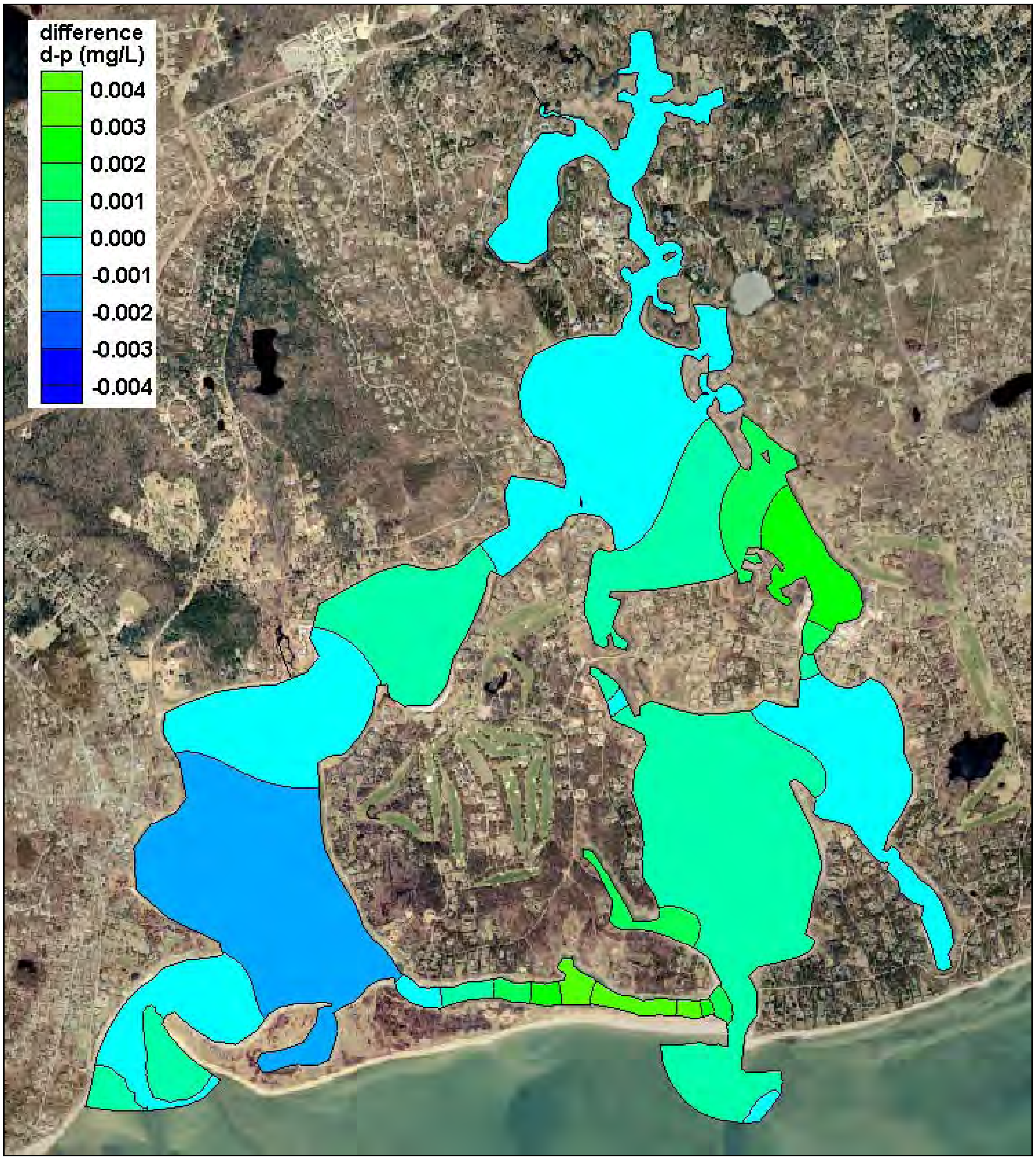


Figure IX-1. Contour plot of total nitrogen concentration change between present hydrodynamic conditions and the dredging scenario where Cotuit Bay inlet is widened to by approximately 300 ft to a depth of 8 ft NGVD. The difference is computed as dredged minus present (d-p) concentrations. Therefore, minus values indicate nitrogen concentration reductions associated with Cotuit Inlet dredging.

Similar to the hydrodynamic model results, changes to nitrogen concentrations throughout the Three Bays system are relatively small, with a maximum range of +0.004 to -0.002 mg/L, as a result of Cotuit Inlet dredging. Pre- and post-dredge TN concentrations at each of the water quality monitoring stations are shown in Table IX-4. The largest increase in modeled TN occurs in the eastern end of the Seapuit River, near West Bay Inlet; however, this slight increase would not cause any type of ecological shift for this region. The greatest decrease in TN occurs in the southeastern extent of Cotuit Bay, specifically in Treasure Cove and near the western mouth of the Seapuit River. Generally, water quality improvements are seen in the main basin of Cotuit Bay, as well as in North Bay, Prince Cove, and Warrens Cove. Small increases in average modeled TN concentrations are seen in West Bay, the Seapuit River, and the southeastern portion of North Bay. None of the changes are large enough to substantially impact water quality, either in a positive or negative way. However, it should be noted that dredging of the Cotuit Bay Inlet would return the system to similar conditions to the 1950s (the basis for the eelgrass restoration target). During this time period, Cotuit Inlet was the dominant inlet to the Three Bays estuarine system. Due to the larger overall volume and depth of Cotuit Bay relative to West Bay, it is beneficial from a water quality perspective to have the dominant inlet be the Cotuit Bay entrance. Based upon information from the Town of Barnstable, navigation and safety also remain concerns at the existing Cotuit Bay Inlet. These factors, along with the quantifiable improvements to Cotuit Bay water quality, may prove to be viable reasons for moving forward with Cotuit Inlet dredging.

Table IX-4. Comparison of model average total N concentrations from present loading and the threshold scenario, with percent change, for the Three Bays system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Prince Cove - south	TB2	0.695	0.695	0.0%
Prince Cove - north	TB3	0.639	0.638	-0.2%
Warrens Cove	TB4	0.595	0.594	-0.2%
North Bay - north	TB5	0.518	0.518	-0.2%
North Bay - south	TB6	0.500	0.501	+0.2%
North Windmill Cove	TB7	0.511	0.511	0.0%
West Bay - north	TB8	0.363	0.363	0.0%
West Bay - west	TB9	0.327	0.328	+0.3%
Eel River	TB10	0.486	0.485	-0.2%
Seapuit River	TB11	0.295	0.298	+1.0%
Cotuit Bay - north	TB12	0.414	0.415	+0.2%
Cotuit Bay - south	TB13	0.321	0.320	-0.3%
South Windmill Cove	TB15	0.402	0.402	0.0%
Mellon Cove	TB16	0.392	0.393	+0.3%
Dam Pond	TB17	0.523	0.523	-0.2%

X. REFERENCES

Aubrey, D.A. and A.G. Gaines, 1982. Recent evolution of an active barrier beach complex: Popponesset Beach, Cape Cod, Massachusetts. Woods Hole Oceanographic Institution Technical Report, WHOI-83-3.

Aubrey, D.A. and M.R. Goud, 1983. Coastal sediment transport: Popponesset Beach, Massachusetts. Woods Hole Oceanographic Institution Technical Report, WHOI-83-26.

Aubrey Consulting, Inc., 1994. Hydrodynamic and Water Quality Modeling Study of the Popponesset Bay Estuary, MA. Technical Report. 37 pp.

Brawley, J.W., G. Collins, J.N. Kremer, C.-H. Sham, and I. Valiela. 2000. A time-dependent model of nitrogen loading to estuaries form coastal watersheds. Journal of Environmental Quality 29:1448-1461.

Brigham Young University, 1998. “User’s Manual, Surfacewater Modeling System.”

Cambareri, T.C. and E.M. Eichner, 1998. Watershed Delineation and Ground Water Discharge to a Coastal Embayment. *Ground Water*. 36(4): 626-634.

Cape Cod Commission, 1998. “Cape Cod Coastal Embayment Project.” Barnstable, MA.

Cape Cod Commission Water Resources Office, 1991. Technical Bulletin 91-001, Nitrogen Loading.

Cape Cod Commission Water Resources Office, 1998. Cape Cod Coastal Embayment Project Interim Final Report.

Cape Cod Commission, 1998. Cape Cod Coastal Embayment Project: A Nitrogen Loading Analysis of Popponesset Bay. Cape Cod Commission Technical Report.

Costa, J.E., B.L. Howes, I. Valiela and A.E. Giblin. 1992. Monitoring nitrogen and indicators of nitrogen loading to support management action in Buzzards Bay. In: McKenzie et al. (eds.) Ecological Indicators, Chapt. 6, pp. 497-529.

Costa, J.E., G. Heufelder, S. Foss, N.P. Millham, B.L. Howes. 2002. Nitrogen Removal Efficiences of Three Alternative Septic System Technologies and a Conventional Septic System. Environment Cape Cod 5(1): 15-24.

DeSimone, L.A. and B.L. Howes. 1996. Denitrification and nitrogen transport in a coastal aquifer receiving wastewater discharge. *Environmental Science and Technology* 30:1152-1162.

Dyer, K.R., 1997. Estuaries, A Physical Introduction, 2nd Edition, John Wiley & Sons, NY, 195 pp.

Robert I. Fultz Associates and Coast and Harbors Institute 2002

Eichner, E.M. and T.C. Cambareri, 1992. Technical Bulletin 91-001: Nitrogen Loading. Cape Cod Commission, Water Resources Office, Barnstable, MA. Available at:

http://www.capecodcommission.org/regulatory/NitrogenLoadTechbulletin.pdf

Eichner, E.M., T.C. Cambareri, G. Belfit, D. McCaffery, S. Michaud, and B. Smith, 2003. Cape Cod Pond and Lake Atlas. Cape Cod Commission. Barnstable, MA.

Eichner, E.M., T.C. Cambareri, K. Livingston, C. Lawrence, B. Smith, and G. Prahm, 1998. Cape Coastal Embayment Project: Interim Final Report. Cape Cod Commission, Barnstable, MA.

Fischer, H. B., List, J. E., Koh, R. C. Y., Imberger, J., and Brooks, N. H. (1979). *Mixing in inland and coastal waters*. Academic. San Diego.

FitzGerald, D.M., 1993. "Origin and Stability of Tidal Inlets in Massachusetts." In: Coastal and Estuarine Studies: Formation and Evolution of Multiple Tidal Inlets, Volume 29, Symposium on Hydrodynamics and Sediment Dynamics of Tidal Inlets (D. G. Aubrey and G.S. Geise, eds.). American Geophysical Union, Washington, D.C. pp. 1-61.

Geise, G.S., 1988. "Cyclical Behavior of the Tidal Inlet at Nauset Beach, Massachusetts: Application to Coastal Resource Management." In: Lecture Notes on Coastal and Estuarine Studies, Volume 29, Symposium on Hydrodynamics and Sediment Dynamics of Tidal Inlets (D. Aubrey and L. Weishar, eds.), Springer-Verlag, NY, pp. 269-283.

Gaines and Fultz 2005

Harbaugh, A.W. and McDonald, M.G., 1996. User's Documentation for MODFLOW-96, an update to the U.S. Geological Survey Modular Finite-Difference Ground-Water Flow Model: U.S. Geological Survey Open-File Report 96-485, 56p.

Henderson, F. M., 1966. *Open Channel Flow*. Macmillan Publishing Company, New York. pp. 96-101.

Howes, B.L., R.I. Samimy and B. Dudley, 2003. Massachusetts Estuaries Project, Site-Specific Nitrogen Thresholds for Southeastern Massachusetts Embayments: Critical Indicators Interim Report

Howes, B.L., J.S. Ramsey and S.W. Kelley, 2000. Nitrogen modeling to support watershed management: comparison of approaches and sensitivity analysis. Final Report to MA Department of Environmental Protection and USEPA, 94 pp. Published by MADEP.

Howes B.L., J. Ramsey, E.M. Eichner, R.I. Samimy, S. W. Kelley, D.R. Schlezinger (2005). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Oyster Pond System, Falmouth, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes, B.L. and D.R. Schlezinger, 1997. Nutrient Related Water Quality within the Popponesset Bay System. Final Report to the Town of Mashpee, pp. 29.

Howes, B.L. and D.R. Schlezinger, 1998. Nutrient Related Water Quality within the Popponesset Bay System. Environment Cape Cod. 2(5):1-21.

Jorgensen, B.B. 1977. The sulfur cycle of a coastal marine sediment (Limfjorden, Denmark). Limnology Oceanography, 22:814-832.

King, Ian P., 1990. "Program Documentation - RMA2 - A Two Dimensional Finite Element Model for Flow in Estuaries and Streams." Resource Management Associates, Lafayette, CA.

Klump, J. and C. Martens. 1983. Benthic nitrogen regeneration. In: Nitrogen in the Marine Environment, (Carpenter & Capone, eds.). Academic Press.

K-V Associates Inc., 1991. A cumulative impact assessment plan to reduce and control sources of contamination in the Mashpee and Santuit/Shoestring Bay River Estuaries. Final Report to the Town of Mashpee.

Lindeburg, Michael R., 1992. *Civil Engineering Reference Manual, Sixth Edition*. Professional Publications, Inc., Belmont, CA.

Massachusetts Department of Environmental Protection, 1999. DEP Nitrogen Loading Computer Model Guidance. Bureau of Resource Protection. Boston, MA. Available at: http://www.state.ma.us/dep/brp/dws/techtool.htm

Massachusetts Department of Revenue. November, 2002. Property Type Classification Codes.

Millham, N.P. and B.L. Howes, 1994a. Freshwater flow into a coastal embayment: groundwater and surface water inputs. Limnology and Oceanography 39: 1928-1944.

Millham, N.P. and B.L. Howes, 1994b. Patterns of groundwater discharge to a shallow coastal embayment. Marine Ecology Progress Series 112:155-167.

Norton, W.R., I.P. King and G.T. Orlob, 1973. "A Finite Element Model for Lower Granite Reservoir", prepared for the Walla Walla District, U.S. Army Corps of Engineers, Walla Walla, WA.

Pollock, D.W., 1994. User's Guide to MODPATH/MODPATH_PLOT, version 3 – A particle tracking post-processing package for MODFLOW, the U.S. Geological Survey modular three dimensional finite-difference ground-water-flow-model: U.S. Geological Survey Open-File Report 94-464, [variously paged].

Ramsey, J.S., B.L. Howes, S.W. Kelley, and F. Li, 2000. "Water Quality Analysis and Implications of Future Nitrogen Loading Management for Great, Green, and Bournes Ponds, Falmouth, Massachusetts." Environment Cape Cod, Volume 3, Number 1. Barnstable County, Barnstable, MA. pp. 1-20.

Ramsey, John S., Jon D. Wood, and Sean W. Kelley, 1999. "Two Dimensional Hydrodynamic Modeling of Great, Green, and Bournes Ponds, Falmouth, MA." Applied Coastal Research and Engineering, Inc. report prepared for the Town of Falmouth and Horsley & Witten, Inc. 41 pp.

Ramsey, J.S., B.L. Howes, N.P. Millham, and D. Bourne. 1995. Hydrodynamic and water quality study of West Falmouth Harbor, Falmouth MA. Aubrey Consulting Inc. Technical Report for Town of Falmouth, pp. 81.

Ryther, J.H., and W.M. Dunstan. 1971. Nitrogen, phosphorous and eutrophication in the coastal marine environment. Science, 171:1008-1012.

Scheiner, D. 1976. Determination of ammonia and Kjeldahl nitrogen by indophenol method. Water Resources 10: 31-36.

Smith, R.L., B.L. Howes and J.H. Duff. 1991. Denitrification in nitrate-contaminated groundwater: occurrence in steep vertical geochemical gradients. Geochimica Cosmochimica Acta 55:1815-1825.

Smith, K.N. and B.L. Howes, 2006. Attenuation of watershed nitrogen by a small New England salt marsh. Manuscript in review.

Taylor, C.D. and B.L. Howes, 1994. Effect of sampling frequency on measurements of seasonal primary production and oxygen status in near-shore coastal ecosystems. Marine Ecology Progress Series 108: 193-203.

USGS web site for groundwater data for Massachusetts and Rhode Island: http://ma.water.usgs.gov/ground_water/ground-water_data.htm

Van de Kreeke, J., 1988. “Chapter 3: Dispersion in Shallow Estuaries.” In: Hydrodynamics of Estuaries, Volume I, Estuarine Physics, (B.J. Kjerfve, ed.). CRC Press, Inc. pp. 27-39.

Weiskel, P.K. and B.L. Howes, 1991. Quantifying Dissolved Nitrogen Flux Through a Coastal Watershed. Water Resources Research, Volume 27, Number 11, Pages 2929-2939.

Weiskel, P.K. and B.L. Howes, 1992. Differential Transport of Sewage-Derived Nitrogen and Phosphorous through a Coastal Watershed. Environmental Science and Technology, Volume 26. Pages 352-360

Wood, J.D., J.S. Ramsey, and S. W. Kelley, 1999. “Two-Dimensional Hydrodynamic Modeling of Barnstable Harbor and Great Marsh, Barnstable, MA.” Applied Coastal Research and Engineering, Inc. report prepared for the Town of Barnstable. 28 pp.

Zimmerman, J.T.F., 1988. “Chapter 6: Estuarine Residence Times.” In: Hydrodynamics of Estuaries, Volume I, Estuarine Physics, (B.J. Kjerfve, ed.). CRC Press, Inc. pp. 75-84.