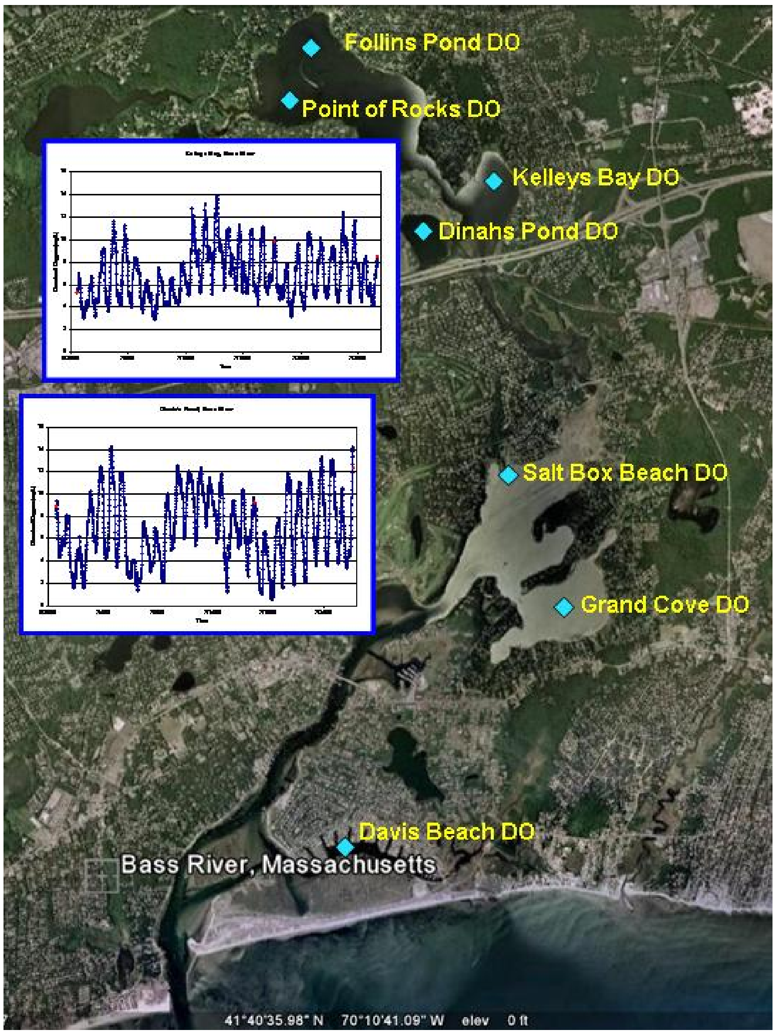
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Bass River Embayment System Towns of Yarmouth and Dennis, Massachusetts



University of Massachusetts Dartmouth
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Massachusetts Department of
Environmental Protection

FINAL REPORT – April 2011

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Executive Summary

1. Background

This report presents the results generated from the implementation of the Massachusetts Estuaries Project's Linked Watershed-Embayment Approach to the Bass River embayment system, a coastal embayment entirely within the Towns of Yarmouth and Dennis, Massachusetts. Analyses of the Bass River embayment system was performed to assist the Towns of Yarmouth and Dennis with up-coming nitrogen management decisions associated with the current and future wastewater planning efforts of the Towns, as well as wetland restoration, management of anadromous fish runs and shell fisheries, and open-space management 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 resource planning in the Towns of Yarmouth and inform the decision-making process. The primary products of this effort are: (1) a current quantitative assessment of the nutrient related health of the Bass River embayment, (2) identification of all nitrogen sources (and their respective N loads) to embayment waters, (3) nitrogen threshold levels for maintaining Massachusetts Water Quality Standards within embayment waters, (4) analysis of watershed nitrogen loading reduction to achieve the N threshold concentrations in embayment waters, and (5) a functional calibrated and validated Linked Watershed-Embayment modeling tool that can be readily used for evaluation of nitrogen management alternatives (to be developed by the Towns) for the restoration of the Bass River embayment system.

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

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

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

The Towns of Yarmouth and Dennis have recognized the severity of the problem of eutrophication and the need for watershed nutrient management and are currently engaged in wastewater management at a variety of levels. Moreover, the Town of Yarmouth is working collaboratively with the Town of Dennis relative to the MEP nutrient threshold analysis of the Bass River system. For the Town of Yarmouth, this analysis of the Bass River system will be considered relative to the recently completed nutrient threshold analysis of Parkers River and Lewis Bay in order to plan out and implement a unified town-wide approach to nutrient management for Yarmouth. Similarly, the MEP analysis of Bass River will be considered relative to the ongoing MEP analysis of Swan Pond River and Sesuit Harbor such that a unified nutrient management approach can be developed in the Town of Dennis. The Towns of Yarmouth and Dennis, along with associated working groups, have recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets is required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Towns in the study region. 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.mass.gov/dep/water/resources/coastalr.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.mass.gov/dep/water/resources/coastalr.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.mass.gov/dep/water/resources/coastalr.htm.

Application of MEP Approach: The Linked Model was applied to the Bass River embayment system by using site-specific data collected by the MEP and water quality data from the Town of Yarmouth and Town of Dennis Water Quality Monitoring Programs. The water quality monitoring program was conducted with technical guidance from the Coastal Systems Program at SMAST (see Section II). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Planning Departments in the Towns of Yarmouth and Dennis, watershed boundaries were delineated by USGS and the Cape Cod Commission rendered additional insights and GIS support. The watershed delineations and the land-use data was used to determine watershed nitrogen loads within the Bass River embayment system and each of the systems sub-embayments as appropriate (current and build-out loads are summarized in Section IV). Water quality within a sub-embayment is the integration of nitrogen loads with the site-specific estuarine circulation. Therefore, water quality modeling of this tidally influenced estuary included a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Once the hydrodynamics of the system was quantified, transport of nitrogen was evaluated from tidal current information developed by the numerical models.

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

MEP Nitrogen Thresholds Analysis: The threshold nitrogen level for an embayment represents the average water column concentration of nitrogen that will support the habitat quality being sought. The water column nitrogen level is ultimately controlled by the watershed nitrogen load and the nitrogen concentration in the inflowing tidal waters (boundary condition). The water column nitrogen concentration is modified by the extent of sediment regeneration. Threshold nitrogen levels for the embayment systems in this study were developed to restore or maintain SA waters or high habitat quality. High habitat quality was defined as supportive of eelgrass and infaunal communities. Dissolved oxygen and chlorophyll a were also considered in the assessment.

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

The Massachusetts Estuaries Project's thresholds analysis, as presented in this technical report, provides the site-specific nitrogen reduction guidelines for nitrogen management of the Bass River embayment system in the Towns of Yarmouth and Dennis. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment, however, within this report an additional hydrodynamic analysis was completed to examine the potential affect of widening the railroad bridge that passes over the main channel of Bass River. Using the calibrated hydrodynamic model of Bass River, an analysis was performed to evaluate flushing improvements that would be possible if culverts were added to the railroad bridge crossing of the Bass River. A series of culvert options were investigated, including three where a single 20-foot-wide culvert was placed 30 feet west of the existing bridge span. Three additional modeled options included a second 20-foot-wide culvert placed 20 feet east of the bridge span. Each modeled scenario was run with the inlet in its present location and existing dimensions.

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 nitrogen loads associated with wastewater generally represent 80% of the controllable watershed load to the whole Bass River embayment system and are more manageable than other of the nitrogen sources, the ability to achieve needed reductions through this source is a good gauge of the feasibility for restoration of these systems.

2. Problem Assessment (Current Conditions)

A habitat assessment was conducted throughout the Bass River embayment system based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements of dissolved oxygen and chlorophyll, and benthic community structure. At present, the Bass River system is showing differences in nitrogen enrichment and habitat quality among its various component basins (Table VIII-1).

Overall, the system is showing some nitrogen related habitat impairment within each of its semi-enclosed component basins, however, there is a strong habitat quality gradient. Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay and Grand Cove are presently supporting significantly impaired infauna habitat. The Bass River is also nitrogen enriched, but has less nitrogen enrichment based primarily on its structure and high water turnover. While the mid and lower reaches of the Bass River system currently supports high quality benthic habitat, the loss of historical eelgrass coverage in those regions of the system indicates that these areas have become significantly impaired relative to eelgrass habitat. Finally, Weir Creek is a small shallow tidal basin with extensive wetlands in its upper reaches and as such has not historically supported eelgrass. Weir Pond has been deepened for navigation and currently functions as a wetland influenced basin with natural organic matter inputs and periodic low oxygen. As such, it is currently supporting moderately to highly productive diverse infaunal communities. However, based upon the high chlorophyll levels and some of the infaunal indicators, it may be showing some modest impairment of benthic habitat. Overall, the regions of significant and moderate habitat impairment (eelgrass or benthic infaunal) comprise >90% of the estuarine area of the Bass River Embayment System.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels within Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay and Grand Cove indicate high levels of nutrient enrichment and impaired habitat quality. The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of these estuarine basins. The upper reaches of the Bass River Estuary generally show significant oxygen depletions and phytoplankton blooms and some basins have macroalgae accumulations. Kelleys Bay and Dinahs Pond also had frequent large daily excursions in oxygen levels ranging from levels in excess of air equilibration to periods of oxygen depletion to < 4 mg L^{-1}. High measured chlorophyll-a levels were found in Kelleys Bay with moderate levels in Dinah Pond. Given the high measured chlorophyll concentrations and the documented presence of macroalgae in Kelleys Bay, as well as large oxygen excursions, it appears that Kelleys Bay is presently over-enriched with nitrogen. Similarly, given the moderate chlorophyll a levels but similarly large oxygen excursions and epiphyte growth and eelgrass in Dinah Pond, the same designation of nutrient over enrichment is set for Dinah Pond.

The major semi-enclosed basin in the mid/lower reaches of the Bass River is Grand Cove. Being in the mid reach of the estuary provides only a slightly better habitat quality for this system. Grand Cove also shows large daily oxygen excursions resulting mainly from oxygen uptake associated with the diurnal cycle as well as tidal influence. Consistent with the oxygen data, chlorophyll a was moderately elevated averaging ~7.6 ug L^{-1}. However the basin supports macroalgal accumulations, further evidence of nitrogen enrichment of this basin. These observations are consistent with the loss of historic eelgrass beds and impaired benthic habitats and the tidally averaged summertime TN level >0.5 mg N L^{-1} (0.52 mg N L^{-1}).

The mid and lower reaches of the Bass River support moderate levels of oxygen depletion (seldom dropping to 4 mg L^{-1}), lower daily excursions and chlorophyll levels generally 4 - 10 ug L^{-1}. These reaches generally do not show macroalgal accumulations and support high quality benthic habitat. The strong horizontal gradient in water quality results mainly from the high nitrogen waters entering from the upper estuary on the ebb tide and the low nitrogen waters entering from the Sound on the Flood tide. Tidally averaged TN levels within the River range from 0.52 - 0.39 mg N L^{-1}, and lower directly in the tidal inlet (0.34 mg N L^{-1}).

Overall, the pattern of high nitrogen, resulting in high phytoplankton biomass and periodic low oxygen depletion was found throughout the upper reaches and in Grand Cove grading to high water quality in the mid and lower reaches of the Bass River. Management of nitrogen levels through reductions in watershed nitrogen inputs or increased tidal flushing are required for restoration of eelgrass and infaunal habitats within the Bass River Embayment System.

The Bass River reaches (mid and lower) extending from the tidal inlet to Rt. 6, were found to presently support high quality benthic habitat. These reaches presently support some of the highest quality benthic animal habitat assessed by the MEP on Cape Cod. These sites also tended to have low to moderate levels of oxygen depletion and chlorophyll a blooms and were generally not accumulating drift macroalgae. In contrast, the enclosed sub-basins of the upper portion of the Bass River system are presently supporting impaired benthic animal habitat. Mill Pond, Follins Pond and Grand Cove are generally dominated by organic enrichment indicators, consistent with high chlorophyll levels, moderate to significant oxygen depletion and accumulations of macroalgae. Dinah Pond and Kelleys Bay showed slightly more impairment of benthic infauna habitat. Overall, the pattern of infaunal habitat quality throughout the Bass River system is consistent with measured dissolved oxygen concentration, chlorophyll, nutrients and organic matter enrichment in this system.

The loss of historic eelgrass beds throughout the mid and lower basins of the Bass River Estuary is consistent with the observed nitrogen and the chlorophyll levels and functional basin types comprising this estuary. The Bass River basins below Rt. 6 supported eelgrass beds in 1951 under lower nitrogen loading conditions.

The historical distribution of eelgrass and its present absence within the Bass River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. The semi-enclosed basins of the upper estuary (Mill Pond, Follins Pond, Kelleys Bay) has likely been nutrient enriched with poor water clarity for many decades. While the presence of eelgrass within Dinah Pond was somewhat surprising, the heavy epiphyte growth over each of the plants is similar to that observed in other Cape Cod estuaries at similar shallow depths and elevated nitrogen levels (e.g. Little Pond, Falmouth). In contrast, the lower tidal reaches of the Bass River with lower nitrogen inputs might be expected to have sufficient water clarity and oxygen levels to support eelgrass beds. However, given the sensitivity of eelgrass to declining light penetration resulting from nutrient enrichment and secondary effects of organic enrichment and oxygen depletion, the current absence of eelgrass within this system is expected given the water depths, nitrogen levels and chlorophyll levels.

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 Bass River system in the Towns of Yarmouth and Dennis comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 80% of the controllable watershed nitrogen load to the overall embayment was from wastewater.

A major finding of the MEP clearly indicates that a single general 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 and the analysis of the nearby Parkers River and Lewis Bay systems. This is almost certainly going to continue to be true for the other embayments within the MEP area, as well, inclusive of Bass River.

The threshold nitrogen levels for the Bass River embayment system in Yarmouth and Dennis were determined as follows:

Bass River Threshold Nitrogen Concentrations

- Following the MEP protocol, the restoration target for the Bass River system should reflect both recent pre-degradation habitat quality and be reasonably achievable. Based upon the assessment data (Section VII), the Bass River system is presently supportive of habitat in varying states of impairment, depending on the component sub-basins of the overall system (e.g. upper portions like Follins Pond, Kelleys Bay and Dinah Pond compared to lower sections of the system such as Grand Cove and Weir Creek.

- The primary habitat issue within the Bass River Embayment System relates to the general loss of eelgrass beds from the lower and middle portions of the system as well as degraded infaunal habitat in the upper portions of the system. The Bass River Embayment System presently supports nitrogen related habitat impairment throughout the tidal reach. A gradient in nutrient related habitat degradation was observed from the most inland reaches of the overall system (Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay) to the higher quality habitat within the Bass River and near the tidal inlet. The loss of eelgrass classifies the mid and lower reaches (and Grand Cove) as "significantly impaired", although the River reaches presently support high quality infaunal communities. The impairments to both the infaunal habitat and the eelgrass habitat within the component basins of the Bass River Embayment System are supported by the variety of other indicators including oxygen depletion, chlorophyll, and TN levels, all of which support the conclusion that these impairments are the result of nitrogen enrichment, primarily from watershed nitrogen loading

- The eelgrass and water quality information supports the conclusion that eelgrass beds within the lower and middle reaches of the Bass River system should be the primary target for restoration of the Bass River Embayment System and that restoration requires a reduction in nitrogen enrichment through appropriate watershed nitrogen management and/or increased tidal exchange. Infaunal habitat quality is the management target for the upper basins, primarily Follins Pond (Mill Pond is brackish).

- The oxygen and chlorophyll data for the Bass River Estuary clearly indicate a system supporting sub-tidal habitats impaired by nitrogen, ranging from highly stressed (Mill Pond, Follins Pond, Kelleys Bay) to moderately stressed (Dinah Pond and possibly Weir Creek). These observations are consistent with the high levels of total nitrogen (TN) throughout the estuary. The gradient in impairment follows the gradient in nitrogen enrichment, where the upper ponds have high ebb tide TN levels ($\geq$0.70 mg N L^{-1}) declining to the Lower River (0.39 mg N L-1) to the tidal inlet (0.34 mg N L^{-1}). While the lower River supports lowest nitrogen levels within the system, the levels are still higher than can support eelgrass beds in deep basins (see Sections VII-3 & VII-4)..

- The observed loss of eelgrass, moderate oxygen and chlorophyll levels and benthic community structure within the mid and lower Bass River reaches, suggests a system beyond the nitrogen threshold level that would be supportive of eelgrass, but currently supporting high quality infaunal habitat. The average nitrogen levels for these regions were 0.39 - 0.50 mg N L^{-1}, the uppermost reach of the Bass River and Grand Cove appear to be above the level supportive of infaunal communities at 0.52 - 0.61 mg N L^{-1}) and well above levels supportive of eelgrass beds as found in a variety of MEP assessments of Cape Cod estuaries. Similarly, the total nitrogen levels at mid-ebb tide within the basins above Rt. 6 (0.61-95 mg N L^{-1}) are well above levels found in basins supportive of high quality benthic animal habitat. These upper basins have significant oxygen excursions and depletions, high chlorophyll a levels as well as accumulations of drift macroalgae (in places), consistent with basins significantly impaired by nitrogen enrichment. The sentinel station for the Bass River Estuary is located at the long-term water quality monitoring stations within the mid reach of the River (BR-6 & BR-7). These sites were selected based upon its location at the upper most extent of the documented eelgrass coverage in this estuary (see Figure VII-6). The concept is to restore the fringing eelgrass beds along the River channel at BR-6 and extensive beds at BR-7 and below.

- A single sentinel station was selected at the long term monitoring station, BR-7, for the re-establishment of the expansive beds at this location and in the region between this station and the tidal inlet, as well as the fringing beds within the river channel between BR-7 and BR-6. This determination is directly linked to analysis of the historical eelgrass coverage. The target nitrogen concentration (tidally averaged TN) for restoration of eelgrass at the sentinel location was determined to be 0.42 mg TN L^{-1}, with a secondary check to lower the River channel TN level to ~0.45 mg N L^{-1}. As there has not been significant eelgrass habitat within the Bass River Estuary for over a decade, this threshold was based upon comparison to other local embayments of similar depths and structure under MEP analysis

For restoration of the Bass River Embayment System, the primary nitrogen threshold at the sentinel station will need to be achieved. At the point that the threshold level is attained at the sentinel station, water column nutrient concentrations will also be at a level that will be supportive of healthy infaunal communities. The results of the Linked Watershed-Embayment modeling are used to ascertain that when the nitrogen threshold is attained, TN levels in the regions associated with the secondary criteria of healthy infauna are also within an acceptable range. The goal is to achieve the nitrogen target at the sentinel location and restore healthy eelgrass habitat throughout lower and middle region of the Bass River system as well as infaunal habitat throughout the embayment but particularly in the upper reaches.

It is important to note that the analysis of future nitrogen loading to the Bass River estuarine system focuses upon additional shifts in land-use from forest/grasslands to residential and commercial development. However, the MEP analysis indicates that significant increases in nitrogen loading can occur under present land-uses, due to shifts in occupancy, shifts from seasonal to year-round usage and increasing use of fertilizers. Therefore, watershed-estuarine nitrogen management must include management approaches to prevent increased nitrogen loading from both shifts in land-uses (new sources) and from loading increases of current land-uses. The overarching conclusion of the MEP analysis of the Bass River estuarine system is that restoration will necessitate a reduction in the present (2009) 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 Bass River estuary system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations.

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
BASS RIVER SYSTEM										
Run Pond[9]	0.132	1.370	7.014	0.00	8.384	0.222	--	8.606	--	--
Bass River - Lower	0.556	6.906	29.858	0.00	36.764	2.995	-11.699	28.060	0.51-0.31	--
School Street Marsh	0.485	2.386	9.496	0.00	11.882	0.247	4.371	16.500	--	--
Bass River - Middle	1.784	13.162	54.512	0.00	67.674	3.841	29.285	100.800	0.73-0.34	0.420
Grand Cove	0.323	1.134	6.159	0.00	7.293	1.071	17.911	26.275	0.55-0.49	--
Dinah's Pond	0.126	0.778	3.559	0.00	4.337	0.310	-2.016	2.631	0.72-0.66	--
Kelleys Bay	0.627	3.718	16.408	0.00	20.126	0.778	28.157	49.061	0.75-0.59	--
Follins Pond	1.367	7.036	27.085	0.00	34.121	2.658	39.596	76.375	0.77-0.72	0.520
Mill Pond	1.019	7.822	19.416	0.00	27.238	0.833	1.609	29.680	0.96-0.94	--
Bass River System Total	**6.419**	**44.312**	**173.507**	**0.00**	**217.819**	**12.955**	**107.214**	**337.988**	**0.61-0.21**	**0.420[8]**

1 assumes entire watershed is forested (i.e., no anthropogenic sources)
2 composed of non-wastewater loads, e.g. fertilizer and runoff and natural surfaces and atmospheric deposition to lakes
3 existing wastewater treatment facility discharges to groundwater
4 composed of combined natural background, fertilizer, runoff, and septic system loadings
5 atmospheric deposition to embayment surface only
6 composed of natural background, fertilizer, runoff, septic system atmospheric deposition and benthic flux loadings
7 average of 2003 – 2009 data, ranges show the upper to lower regions (highest-lowest) of an sub-embayment. Individual yearly means and standard deviations in Table VI-1.
8 Threshold for sentinel site located in Bass River at water quality station BR-7 and secondary sites BR-2 and BR-3.
9 The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Bass River estuary system, Towns of Yarmouth and Dennis, MA.

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
BASS RIVER SYSTEM						
Run Pond[5]	8.384	8.384	0.222	0.000	8.606	0.00%
Bass River - Lower	36.764	36.764	2.995	-9.796	29.963	0.00%
School Street Marsh	11.882	11.882	0.247	3.610	15.739	0.00%
Bass River - Middle	67.674	29.833	3.841	24.042	57.716	-55.92%
Grand Cove	7.293	7.293	1.071	13.699	22.063	0.00%
Dinah's Pond	4.337	0.778	0.310	-1.120	-0.032	-82.06%
Kelleys Bay	20.126	3.860	0.778	17.337	21.975	-80.82%
Follins Pond	34.121	7.858	2.658	19.540	30.056	-76.97%
Mill Pond	27.238	7.847	0.833	0.607	9.287	-71.19%
Bass River System Total	**217.819**	**114.499**	**12.955**	**67.919**	**195.373**	**-47.43%**

(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.
(5) The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.

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I. INTRODUCTION

The Bass River Embayment System is a complex estuary located within the Towns of Yarmouth and Dennis on Cape Cod, Massachusetts and which exchanges tidal waters with Nantucket Sound to the south (Figure I-1). The Bass River Estuary is comprised of a tidal river connecting a series of large kettle ponds (Mill Pond, Follins Pond, Dinahs Pond), to Nantucket Sound and with a small lagoonal tributary basin behind the barrier beach, which had filled with salt marsh and now has been partially filled and developed. The barrier beach was formed by growth of a spit from east to west, consisting of marine sands and gravel deposited by coastal processes during post-glacial sea-level rise. The groundwater defined watershed to the lower portion of the Bass River system is situated in sand and gravel deposits whereas the upper groundwater watershed is characterized by bouldery glacial drift overlying outwash sand and gravel of the Falmouth Moraine.

The late Wisconsinan Laurentide ice sheet reached its maximum extent and southernmost position about 20,000 years before present (BP), as indicated by the presence of terminal moraines on Martha's Vineyard and Nantucket and the southern limit of abundant gravel on the sea floor of Nantucket Sound and Vineyard Sound (Schlee and Pratt, 1970; Oldale, 1992; Uchupi et al., 1996). The lobate ice front was comprised of the Buzzards Bay lobe that deposited the moraine along the western part of Martha's Vineyard, the Cape Cod Bay lobe that deposited the moraines across eastern Martha's Vineyard and Nantucket, and the South Channel lobe that extended east toward Georges Bank (Oldale and Barlow, 1986; Oldale, 1992). After initial retreat of the ice sheet, approximately 18,000 years BP, a second series of moraines was formed on Cape Cod, with associated outwash plains. The Falmouth Moraine and associated outwash deposits form the watershed of present day Bass River. While the watershed was formed on the order of 15,000- 8,000 years ago, the estuary of Bass River is a much more recent formation, likely 2,000 - 4,000 years ago as sea level flooded the present basin.

The valley of Bass River itself appears to have formed from outflow or groundwater sapping derived from waters of Glacial Lake Cape Cod moving toward Nantucket Sound. The "lagoon", often referred to as Davis Beach is composed of a highly altered shallow open water basin bordered by remnants of the salt marsh to the south on the backside of the barrier beach and remaining salt marsh forming the basin of the upper tidal reach. While the salt marsh historically was comprised of a main tidal channel bordered by salt marsh to either side, that tidal channel was dredged and presently is the open water lagoon which is bordered to the north by sea walls and riprap that bounds a highly developed residential neighborhood. Portions of the relic tidal channel still exists through the remaining salt marsh on the backside of the barrier beach and terminates in a salt marsh that is hydraulically connected to the open water basin by a bridge opening where Loring Road crosses over the tidal creek. Bass River is composed of several significant basins. The major tidal basins of the Bass River system are Mill Pond, Follins Pond, Kelleys Bay, Dinahs Pond and Grand Cove, which along with the main channel of Bass River and the lagoon, comprise the Bass River Estuarine System. The uppermost basin of Mill Pond is the shallowest of the basins in the Bass River system (~1.0 m) and is brackish (~15 ppt), while the rest of the tidal reach exhibits nearly marine salinities (generally $\geq$25 ppt).

The Bass River system, while dominated by open water, still supports a limited number of tidal wetlands, primarily in the region of the inlet to the Bass River system and in the vicinity of the area considered Davis Beach as well as just north of the Route 28 bridge crossing. However, in most other areas of the system, as one moves up-gradient towards Route 6, the shoreline of Bass River can be primarily characterized as upland (with forest and single family residential development). The tidal river, Bass River, is functionally divided into an upper section comprised of Mill Pond, Follins Pond and Kelleys Bay, a middle section that is mainly a broad stretch of tidal river with the associated tributary sub-basin of Grand Cove and a lower river that extends from the inlet to the Route 28 bridge and includes the lagoonal basin and salt marsh considered Davis Beach.

The Massachusetts Estuaries Project (MEP) was able to determine that as far back as 1859, Mill Pond was connected to the Bass River, hence Nantucket Sound (i.e. it was tidal to some degree). This connection is seen in drawings and maps: 1858 and 1880 Map (personal communication, M. Rukstalis Historical Society of Old Yarmouth)[1]. As such, Mill Pond is clearly a functional estuarine basin and needs to be assessed and managed as such.

The Bass River system is one of the largest estuaries on Cape Cod. Its watershed is distributed amongst both the Towns of Yarmouth and Dennis. Currently Bass River represents a discharge zone for 2 of the major groundwater flow cells on Cape Cod. A large portion of the overall watershed includes the sub-watersheds contributing direct groundwater discharge to the estuary as well as to Hamblin Brook that flows into the most up-gradient region of Mill Pond. A subwatershed was also developed to Fresh Pond and the small brook that flows to Grand Cove and the middle portion of the tidal Bass River up-gradient of Route 28. Although land-uses closest to an embayment generally have greater impact than those in the inland 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 Bass River System, will require the Town of Yarmouth and Dennis to be active in nutrient management throughout the watershed to the overall system. However, management will be complicated since the watershed to Bass River exists within the confines of two distinct municipal boundaries and not all of the watershed nitrogen sources to the Bass River System reside within one town.

The large number of sub-basins (Davis Beach, Grand Cove, Kelleys Bay, Dinahs Pond, Follins Pond, Mill Pond) comprising the Bass River System greatly increases the shoreline and decreases the travel time of groundwater from the sites of watershed nitrogen inputs to estuarine regions of discharge. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine 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 Bass River system and its sub-embayments, along the southern shore of the Town of Yarmouth and Dennis, is at risk of eutrophication (over enrichment) from high nitrogen loads in the groundwater and runoff from their watersheds, in a similar fashion as nearby Parkers River and Lewis Bay. However, the physical structure of the Bass River System, with a long tidal reach that extends many kilometers away from the inlet as well as a series of large upper pond basins increases the sensitivity of this embayment system to nitrogen enrichment.

The Bass River Embayment System is a complex (drowned river valley + lagoonal) estuary exchanging tidal waters with the high quality waters of Nantucket Sound through a single inlet that is "fixed" by jetties (Figure I-1). The barrier spit supporting the inlet is used for

[1] The connection was not clear in the 1880 Atlas of Barnstable County, A. Gamble 1998, United Book Press of Baltimore Maryland, but as the connection can be seen in all subsequent maps the 1880 map is almost certainly in error.

Figure I-1. Major components of the Bass River Embayment System assessed by the Massachusetts Estuaries Project. Tidal waters enter the main channel of the Bass River system through a single inlet from Nantucket Sound. Freshwaters enter from the watershed primarily through direct groundwater discharge to the estuary and through 2 small surface water discharges (Fresh Pond Brook and Hamblin Brook into Mill Pond).

recreation, primarily Riviera/Smugglers Beach to the west and Davis Beach to the east. Due to its length, the Bass River Estuary contains a gradient in the salinity of waters throughout its tidal reaches, generally 31 parts per thousand (ppt) near the inlet and 25 ppt in Follins Pond. The interaction between high salinities in the lower portions of the system close to the inlet and the lower salinity regime in the upper portions of the system influenced by freshwater inflows reflects the varying dominance of tidal flows in structuring these systems. Prior to development and armoring of the tidal inlet, sediment transport and deposition associated with coastal processes including coastal storms likely caused the inlet to migrate and certainly resulted in periods of lower tidal flows due to deposition of sands within the tidal channel. This may have lead to periodic alteration of the salinity gradient when compared to the present condition.

Currently, the Towns of Yarmouth and Dennis periodically dredge the inlet channel to keep the inlet navigable and to maintain tidal flows.

Tidal forcing for this embayment system is generated from Nantucket Sound. Nantucket Sound exhibits a moderate to low tide range, with a mean range of about 2.5 to 3.5 ft. Since the water elevation difference between Nantucket Sound and the Bass River System is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, 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 indicated only minimal tidal damping through the inlet into the main tidal channel of the Bass River system down gradient of the Route 28 bridge crossing as well as above Route 6 and into Kelleys Bay. It appears that the tidal inlet is operating efficiently being periodically dredged to maintain navigation and flushing. In contrast, tidal damping was observed for the portion of the system upgradient of the N. Dennis Road culvert into Mill Pond (present tide range 0.9 ft). The tide propagates to the sub-embayment of Mill Pond with attenuation, where as there is little to no attenuation within Kelleys Bay / Follins Pond portion of the system, consistent with well-flushed conditions throughout.

The primary ecological threat to the Bass River system as a coastal resource is degradation resulting from nutrient enrichment. Although the enclosed estuarine system has some bacterial contamination issues related to stormwater run-off from the watershed and likely animal sources primarily associated with wetlands or undeveloped areas along the shores of Mill Pond, these do not appear to be having large system-wide impacts. Bacterial contamination causes periodic closures of shellfish harvest areas throughout the Bass River system and within only two specific areas, Mill Pond (SC35.6) and a small tributary cove (SC34.3), causes permanent closures. In contrast, loading of the critical eutrophying nutrient, nitrogen, to the Bass River System has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to the Bass River Estuary, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater.

The Towns of Yarmouth and Dennis, along with other towns on Cape Cod, have been among the fastest growing towns in the Commonwealth over the past two decades and do not have a centralized wastewater treatment system. The Town of Yarmouth does operate a regional septage treatment facility for the disposal of pump out from local septic systems located throughout the town of Yarmouth, however, as such the vast majority of the developed areas in the Bass River watershed are not connected to any municipal sewerage system. Therefore, wastewater treatment and disposal is primarily through privately maintained on-site septic systems. As present and future increased levels of nutrients impact the coastal embayments of Yarmouth and Dennis, water quality degradation will continue, with further harm to invaluable environmental resources, as evidenced by the July 2009 fish kill within the adjacent Parkers River Estuary.

As the primary stakeholder to the Bass River System, the Towns of Yarmouth and Dennis were among the first communities to become concerned over perceived degradation of coastal waters. Concern over declining habitat quality within its embayments led directly to the establishment of a comprehensive water quality monitoring program aimed at determining the degree to which waters of both Town's embayments may be impaired. The Town of Yarmouth and the Town of Dennis (through the Dennis Water District) Water Quality Monitoring Programs were provided technical assistance by the Coastal Systems Program at SMAST-UMD. Over the past several years the Yarmouth Program has operated in a coordinated manner with the Town of Dennis water quality monitoring program as a result of the shared embayment of Bass River. In addition to assessing the health of the estuaries in Yarmouth and Dennis, the water quality monitoring program provides the required quantitative watercolumn nitrogen data (2003-2008) for validation of the MEP's Linked Watershed-Embayment Approach used in the present study. Entry into the MEP and TMDL compliance monitoring depends upon Town supported water quality monitoring, as guided by SMAST.

The common focus of the Yarmouth and Dennis Water Quality Monitoring effort has been to gather site-specific data on the current nitrogen related water quality throughout the Bass River System and determine its relationship to watershed nitrogen loads. This multi-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program and adds several additional layers of high end data collection linking watershed characteristics to estuarine function. The MEP approach includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each major sub-embayment. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater planning and nitrogen management alternatives development needed by both the Towns of Yarmouth and Dennis. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Town of Yarmouth and Dennis 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 Bass River System and its associated watershed has been significantly altered by human activities over the past ~100 years or more (see Section I.2, below). As a result, the present nitrogen "overloading" appears to result partly from alterations to the geomorphology and ecological systems. These alterations subsequently affect nitrogen loading and transport within the watershed and influence the degree to which nitrogen loads impact the estuary. Therefore, restoration of this system should focus on managing nitrogen through both management of nitrogen loading within the watershed and restoration/management of processes which serve to lessen the amount or impact of nitrogen entering the estuary, for example hydrodynamic solutions.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the 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 Yarmouth and Dennis) are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries.

Municipalities are seeking guidance on the assessment of nitrogen sensitive embayments, as well as available options for meeting nitrogen goals and approaches for restoring impaired systems. Many of the communities have encountered problems with "first generation" watershed based approaches, which do not incorporate estuarine processes. The appropriate method must be quantitative and directly link watershed and embayment nitrogen conditions. This "Linked" Modeling approach must also be readily calibrated, validated, and implemented to support 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 MEP approach was selected after extensive review by the MassDEP and USEPA and associated scientists and engineers. It has subsequently been applied to more than 30 estuaries and reviewed by other state agencies, municipalities, non-profit environmental organizations, engineering firms, scientists and private citizens. Over the course of the extensive reviews, the MEP approach has proven to be robust and capable of yielding quantitative results to support management of a wide variety of estuaries.

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide 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, TMDLs for bacterial contamination have also been conducted in concert with the nutrient effort (particularly if there is a 303d listing). In these cases, the MEP (through SMAST) has produced a Technical Analysis and Report to support a bacterial TMDL for the system from which MassDEP develops the TMDL. 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 70 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment's model "alive" to address future municipal needs.

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

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

The Linked Model has been applied for watershed nitrogen management of more than 30 embayments throughout southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options. The MEP Technical Team, through SMAST-UMD, has conducted more than 200 scenarios to date.

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

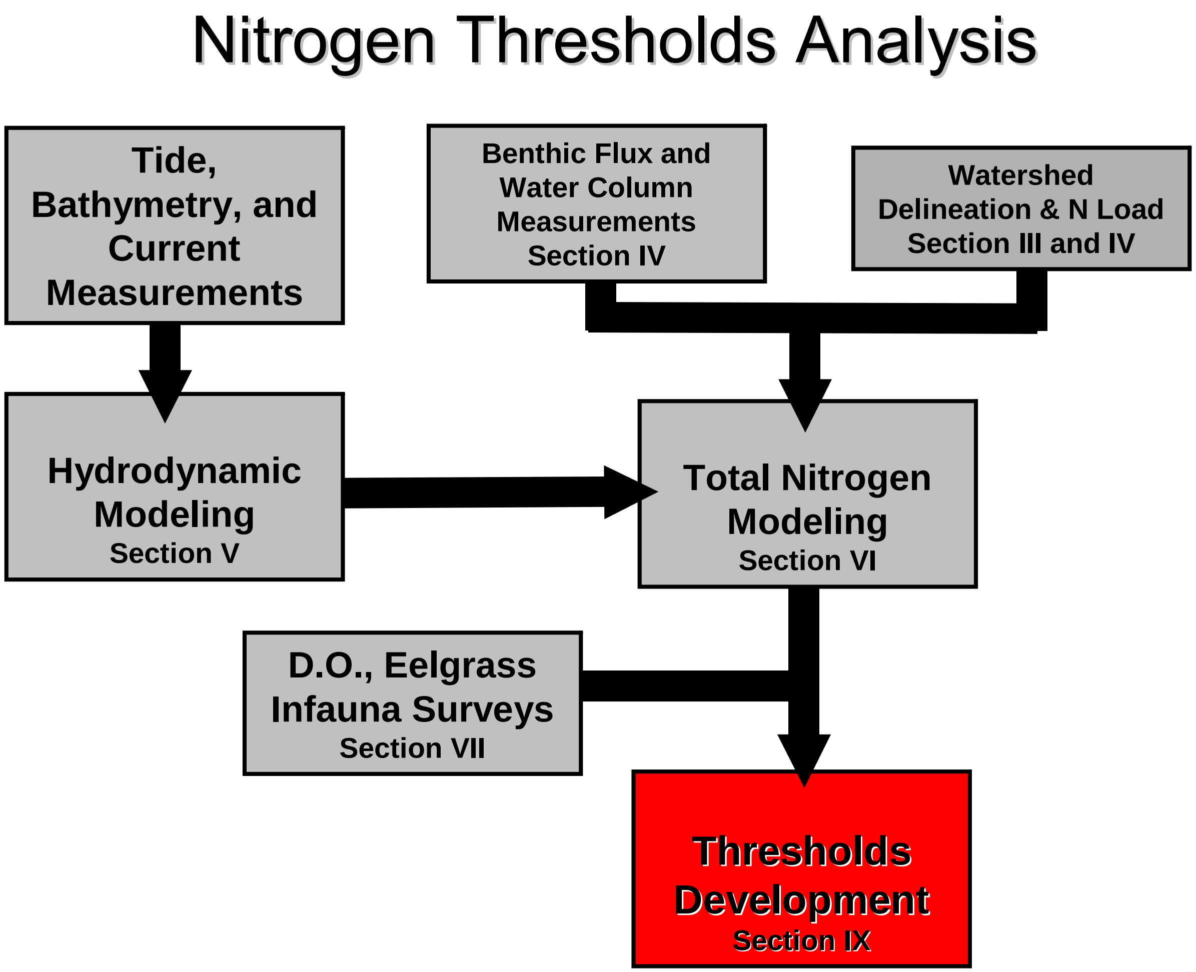

Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Note that the approach is not a single model, but a series of models linked by scientists and engineers who validate outputs and inputs.

I.2 NUTRIENT LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Bass River System, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer minerals (Weiskel and Howes 1992). Since even Cape Cod "rivers" are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through 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 Bass River Estuary follow this general pattern, where the primary nutrient of eutrophication in this system is nitrogen.

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

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

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts has been the site of intensive efforts in this area (Eichner et al., 1998, Costa et al., 1992 and in press, Ramsey et al., 1995, Howes and Taylor, 1990, and the Falmouth Coastal Overlay Bylaw). While each approach may be different, they all focus on changes in nitrogen loading from watershed to embayment, and aim at projecting the level of increase in nitrogen concentration within the receiving waters. Each approach depends upon estimates of circulation within the embayment; however, few directly link the watershed and hydrodynamic models, and virtually none include internal recycling of nitrogen (as was done in the present 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 Bass River System monitored by the Towns of Yarmouth and Dennis. Data from the Water

Quality Monitoring Program combined 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, the upper reaches of the Bass River System are at or beyond their ability to assimilate additional nutrients without impacting their ecological health. The MEP analysis clearly indicates that the system is presently impaired by nitrogen overloading and does not meet the Commonwealth's Water Quality Standards. Nitrogen levels are elevated throughout the System and while some eelgrass exists in Dinah's Pond, it was coated with epiphytes. Additionally, other areas of the upper Bass River system such as Follins Pond and Kelleys Bay were observed to have a variety of macroalgae at all the sites surveyed by the MEP. It is important to note that the present nitrogen enrichment of the Bass River Estuarine System results from the combination of increasing nitrogen loading to its contributing watershed coupled to reduced flushing of nitrogen due to tidal restriction in the upper portions of the system. The MEP analysis evaluates both of these processes and any efficient management plan will likely include modifications to both loading and flushing.

Nitrogen related habitat degradation within the Bass River Estuary shows a gradient of high to low impairment moving from the inland reaches of Mill Pond, Follins Pond and Kelleys Bay to the tidal inlet. The result is that nitrogen management of the primary sub-embayments to the Bass River system is aimed at restoration (particularly in the upper portions of the system), 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 system and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within certain Bass River 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 sub-embayment into a "pristine" system.

I.3 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Bass River System, including the tributary sub-embayments of Mill Pond, Follins Pond, Dinah's Pond, Kelleys Bay and Grand Cove. 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 watershed sourced nitrogen entering the Bass River System is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Bass River system were provided by the Town of Yarmouth and Town of Dennis (Dennis Water District) Water Quality Monitoring Program (a coordinated effort between the Towns and the Coastal Systems Program at SMAST). Measurements of the salinity and nitrogen distributions throughout estuarine waters of the Bass River System (2003-2008) were used to calibrate and validate the water quality model (under existing loading conditions).

I.4 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Bass River System for the Towns of Yarmouth and Dennis. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Town of Yarmouth, Town of Dennis and Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Nantucket Sound (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. 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) also discussed in Section VI. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration/protection of the River in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined system threshold for restoration or protection. This latter assessment represents only one of many solutions and is produced to assist the Towns in developing a variety of alternative nitrogen management options for this system. Finally, analyses of the Bass River System were undertaken relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions related to the railroad bridge down gradient of the Route 6 bridge. The results of the nitrogen modeling for specific Bass River scenarios are presented in Section IX with references provided in Section X.

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. This has the concomitant effect of increased rates of oxygen consumption and periodic depletion of dissolved oxygen, especially in bottom waters, as well as limiting the growth of desirable species such as eelgrass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic infaunal communities (animals living in the sediments). This habitat change causes a shift in infaunal communities from high diversity deep burrowing forms (which include economically important species), to low diversity shallow dwelling organisms. This shift alone causes significant degradation of the resource and a loss of productivity to both the local shell fisherman and to the sport-fishery and offshore fin fishery. Both the sport-fishery and the offshore fin fishery are dependant upon highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. This process is of degradation 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 Bass River 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 Bass River System. As the MEP approach requires substantial amounts of site specific data collection, part of the program is to review previous data collection and modeling efforts. These reviews are both for purposes of “data mining” and to gather additional information on an estuary’s habitat quality or unique features.

A limited number of studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Bass River System over the past two decades.

Flushing Characteristics of Upper Bass River, MA. (Woods Hole Group, November, 1994): In 1994, Woods Hole Group, Inc. was contracted to produce a calibrated hydrodynamic model to evaluate the flushing characteristics of the Upper Bass River system. The model was developed to include Kelleys Bay, Dinahs Pond, Follins Pond, and Mill Pond, as well as the connecting channels between the sub-embayments. Field measurements of water levels, bathymetry, and creek flow were used as input and used to calibrate the two-dimensional, depth-integrated hydrodynamic (RMA-2) model. According to a summary of the work completed by WHG, four tide gauges were deployed for a 43-day period to measure tidal elevations and phases. One gauge was deployed at the railway within Kelleys Bay, while the remaining three were deployed within Kelleys Bay (north), Follins Pond, and Mill Pond. Resolution of the dampening and time lag within the system was provided by the comparisons between the railway gauge and the remaining gauges. Bathymetry was collected to map the bay and pond seafloor elevations using a coupled DGPS and fathometer system. Additionally, creek flow was measured within Weir Creek using a Marsh-McBirney current meter.

Although the field data collected were used to characterize the hydrodynamics of the upper portion of Bass River, the model admittedly did not produce a complete picture of how the system functions. The RMA-2 model was applied to produce water levels and tidal currents at specific locations within the system. The use of the RMA-2 model required a customized finite element mesh to characterize the system, definition of a dynamic seaward tidal boundary condition (i.e., at the railway), specification of bathymetric data, and calibration of the model. According to WHG, the model was quantitatively calibrated using harmonic analysis. The measured and modeled water levels were calibrated until an acceptable tolerance was achieved. Ultimately, the model was employed to calculate residence times for each sub-embayment. This provided an estimate on how often each sub-embayment flushed in 1994.

Town of Yarmouth / Dennis Water Quality Monitoring Program (2003-2008) – Over the past seven years nutrient sampling of Bass River has been undertaken at a maximum of 14 stations (Figure II-1) throughout the system. Starting in 2003, the Town of Yarmouth Natural Resources Department and the Dennis Water District partnered with SMAST-Coastal Systems Program scientists to develop a unified sampling program for all the estuaries in both the Towns inclusive of Bass River in order to establish the baseline water quality monitoring record needed for the execution of the MEP analysis on the Bass River estuarine system. The Town of Yarmouth Natural Resources Department and the Dennis Water District working with SMAST staff coordinated and executed the water quality surveys of the Bass River System. This sampling effort began in 2003 and included all 14 stations (BR-1 through BR-14). All stations were sampled over a total of six sampling events per summer for the first three years of the program (2003-2005). With three years of consistently collected base data in hand, sampling at a reduced number of stations and reduced number of events (4 per summer) was continued in the summer of 2006 through 2008.

For the Bass River system as well as the other estuarine systems of Cape Cod, the focus of the effort has been to gather site-specific data on the current nitrogen related water quality throughout the estuarine reach of the system to support assessments of habitat health. These baseline water quality data are a pre-requisite to entry into the MEP and the conduct of its Linked Watershed-Embayment Approach. Throughout the water quality monitoring period, sampling was undertaken between 4 and 6 times per summer between the months of June and September. The Town based Water Quality Monitoring Program for Bass River including Mill Pond developed the baseline data from sampling stations distributed throughout the system as well as the main tidal channel of Bass River between Route 6 and Route 28 and the tributary sub-system of Dinah's Pond, Grand Cove and the basin on the backside of Davis Beach. An offshore station located just outside the mouth of the Bass River system was sampled on the inflowing tide in order to establish boundary water quality conditions for the MEP analysis (Figure II-1). As remediation plans for this and other various systems in the Town of Yarmouth and Dennis are implemented, monitoring will have to be resumed or continued to provide quantitative information to the Towns relative to the efficacy of remediation efforts.

Figure II-1. Town of Yarmouth / Dennis Water Quality Monitoring Program. Estuarine water quality monitoring stations sampled by the Town of Yarmouth Natural Resources Department/staff from the Dennis Water District/SMAST staff.

Implementation of the MEP's Linked Watershed-Embayment Approach incorporates the quantitative water column nitrogen data (2003-2008) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff. The MEP effort also builds upon previous watershed delineation and land-use analyses as well as historical eelgrass surveys. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Bass River Estuarine System. The MEP has incorporated data from appropriate previous studies to enhance the determination of nitrogen thresholds for the Bass River System and to reduce costs of restoration for the Towns of Yarmouth and Dennis.

Regulatory Assessments of Bass River Estuary Resources - The Bass River Estuary contains a variety of natural resources of value to the citizens of Yarmouth and Dennis as well as to the Commonwealth. As such, over the years surveys have been conducted to support protection and management of these resources. The MEP gathers the available information on these resources as part of its assessment, and presents them here (Figures II-2 through II-6) for reference by those providing stewardship for this estuary. For the Bass River Estuary these include:

- Mouth of River designation - MassDEP (Figure II-2)
- Designated Shellfish Growing Area – MassDMF (Figure II-3a, II-3b, II-3c)
- Shellfish Suitability Areas - MassDMF (Figure II-4)
- Anadromous Fish Runs - MassDMF (Figure II-5)
- Estimated Habitats for Rare Wildlife and State Protected Rare Species – NHESP (Figure II-6a and II-6b)

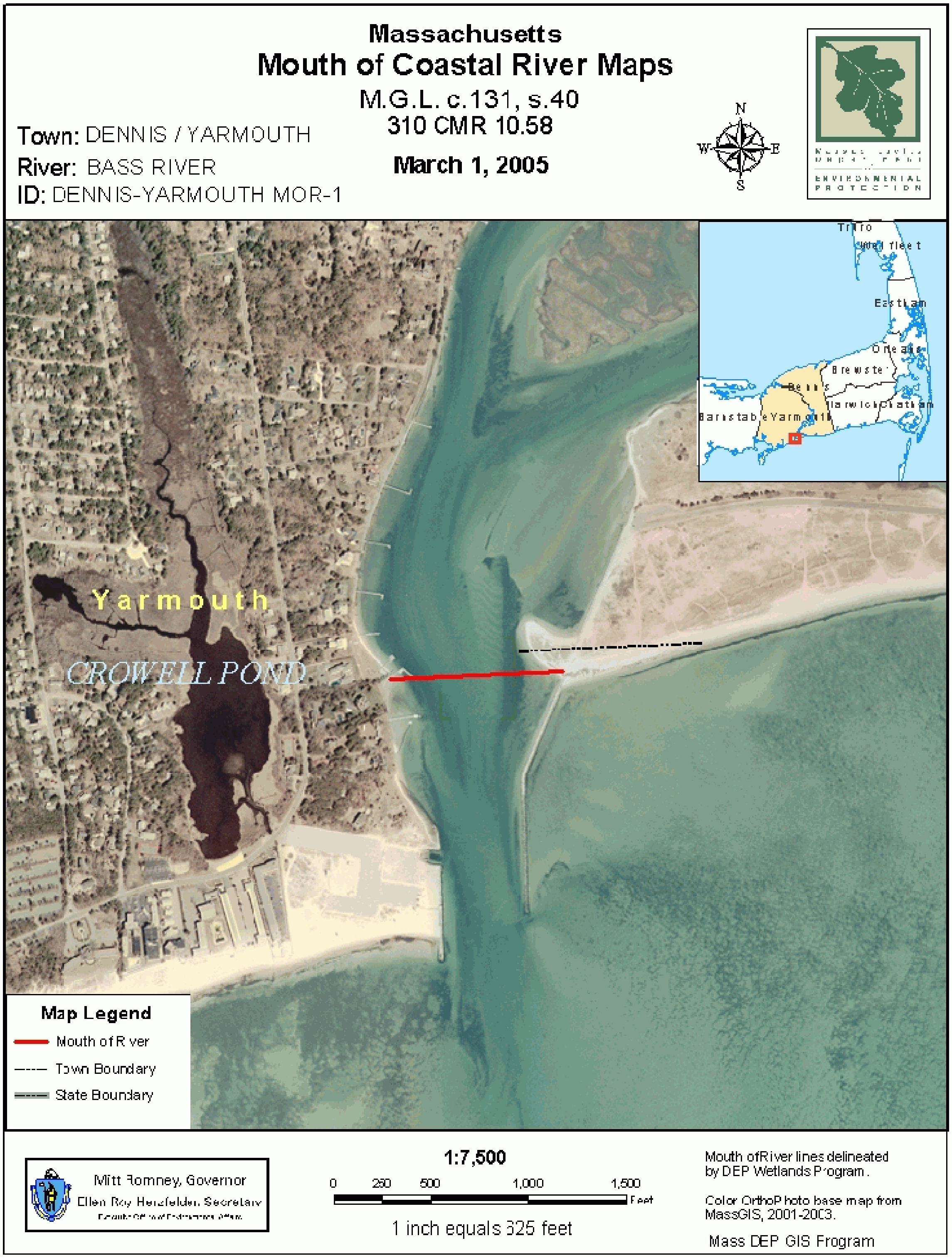

Figure II-2. Regulatory designation for the mouth of "River" line under the Massachusetts River Act (MassDEP). Upland adjacent the "river front" inland of the mouth of the river has restrictions specific to the Act.

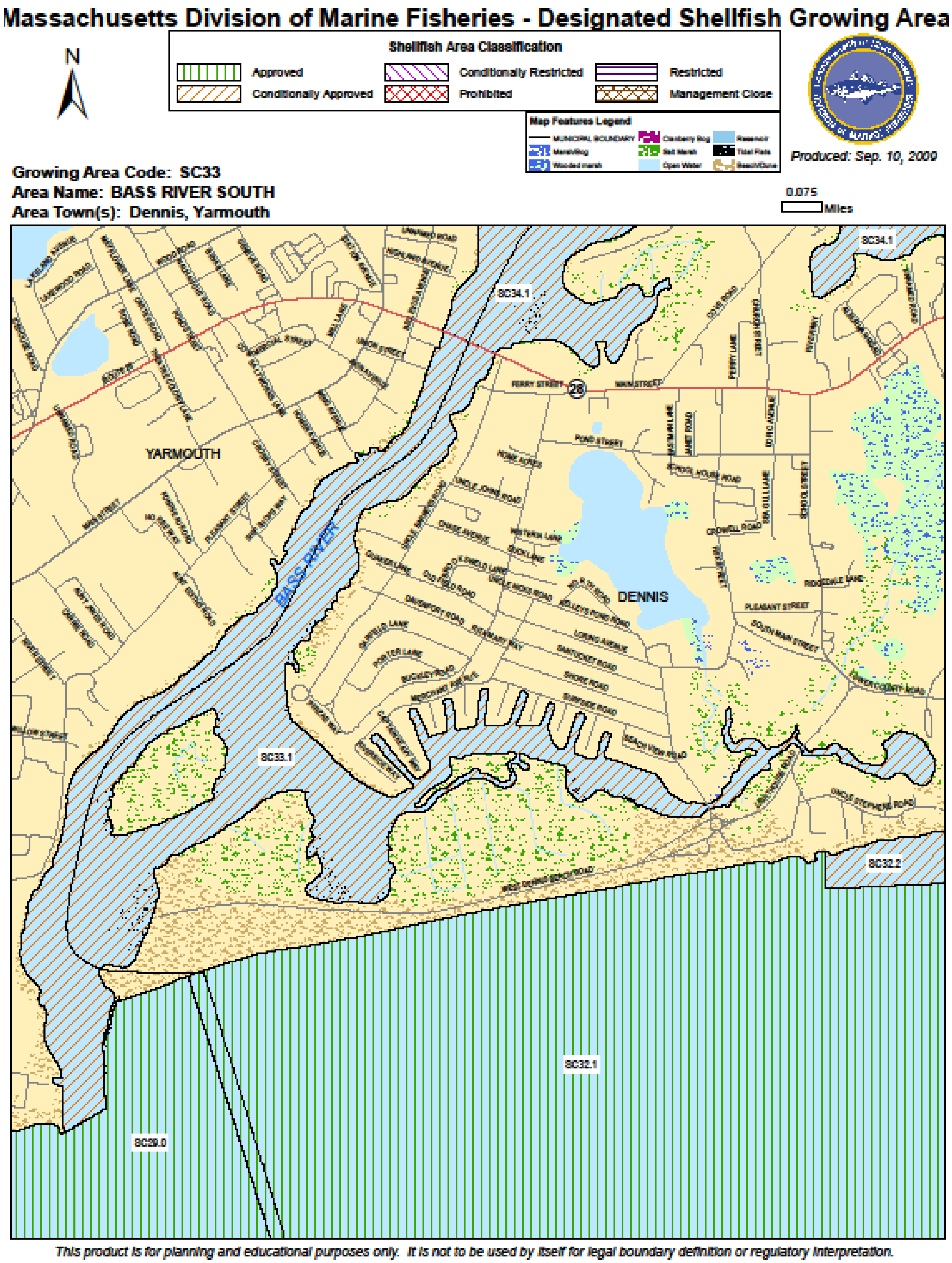

Figure II-3a. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

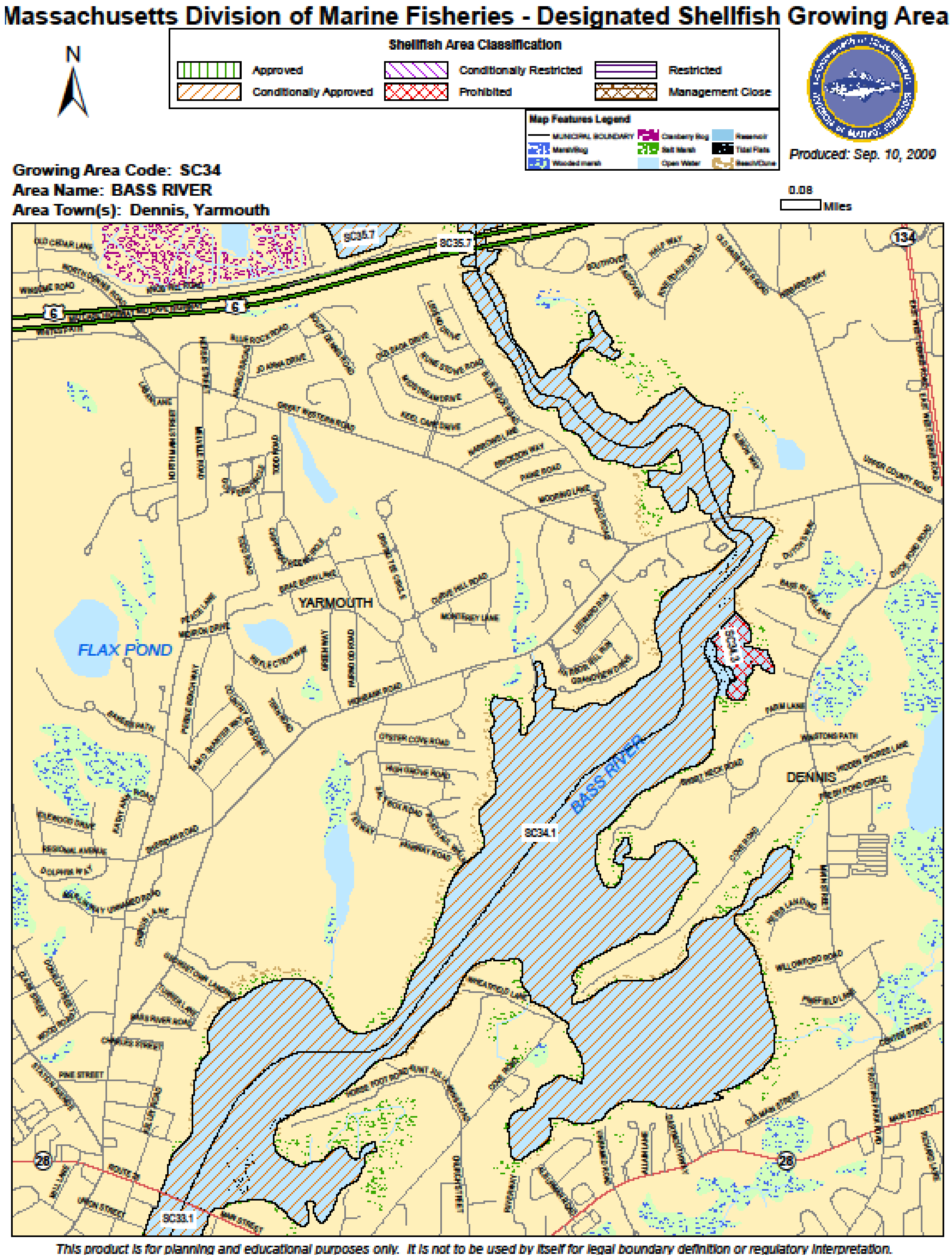

Figure II-3b. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas

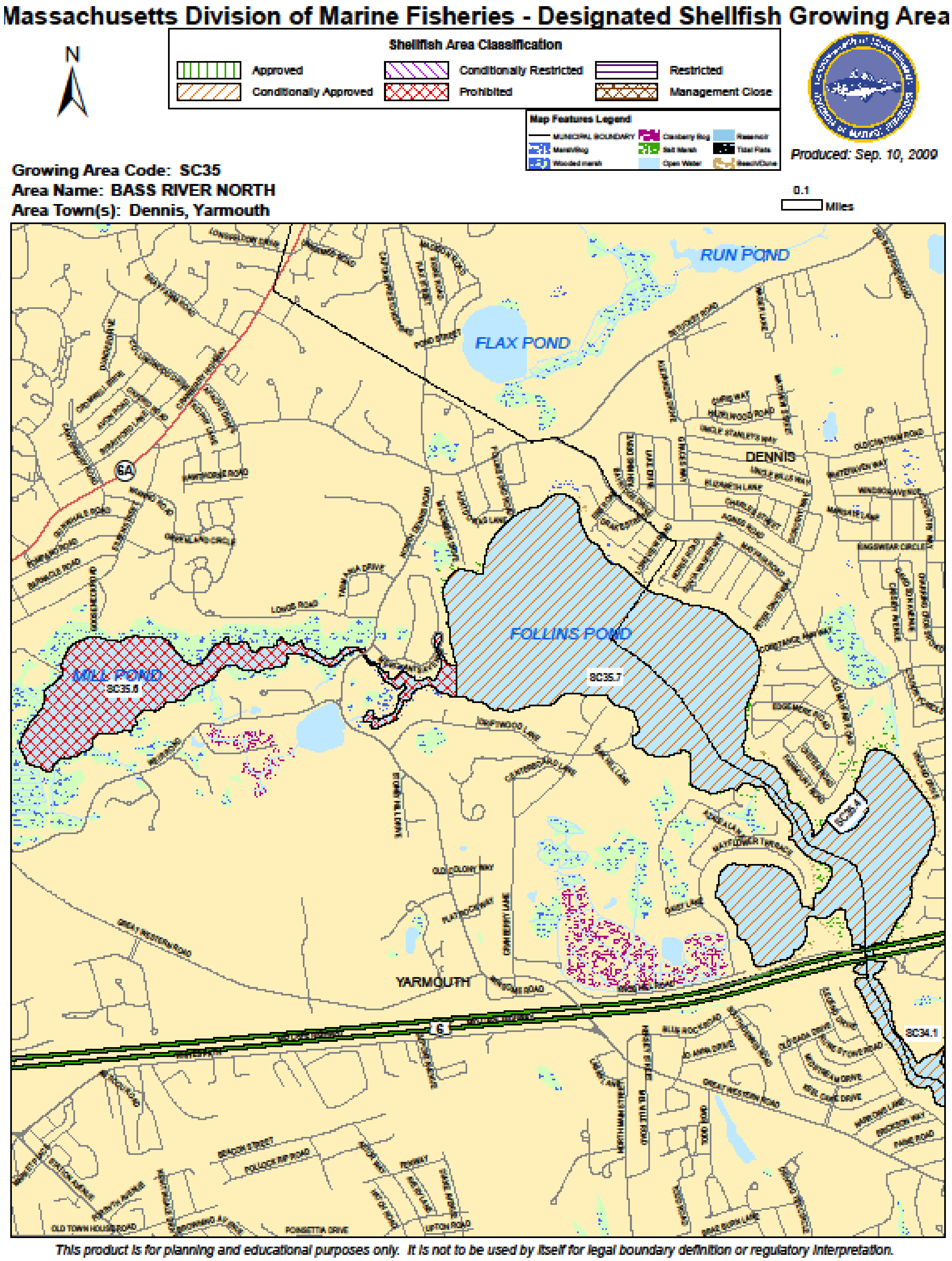

Figure II-3c. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas

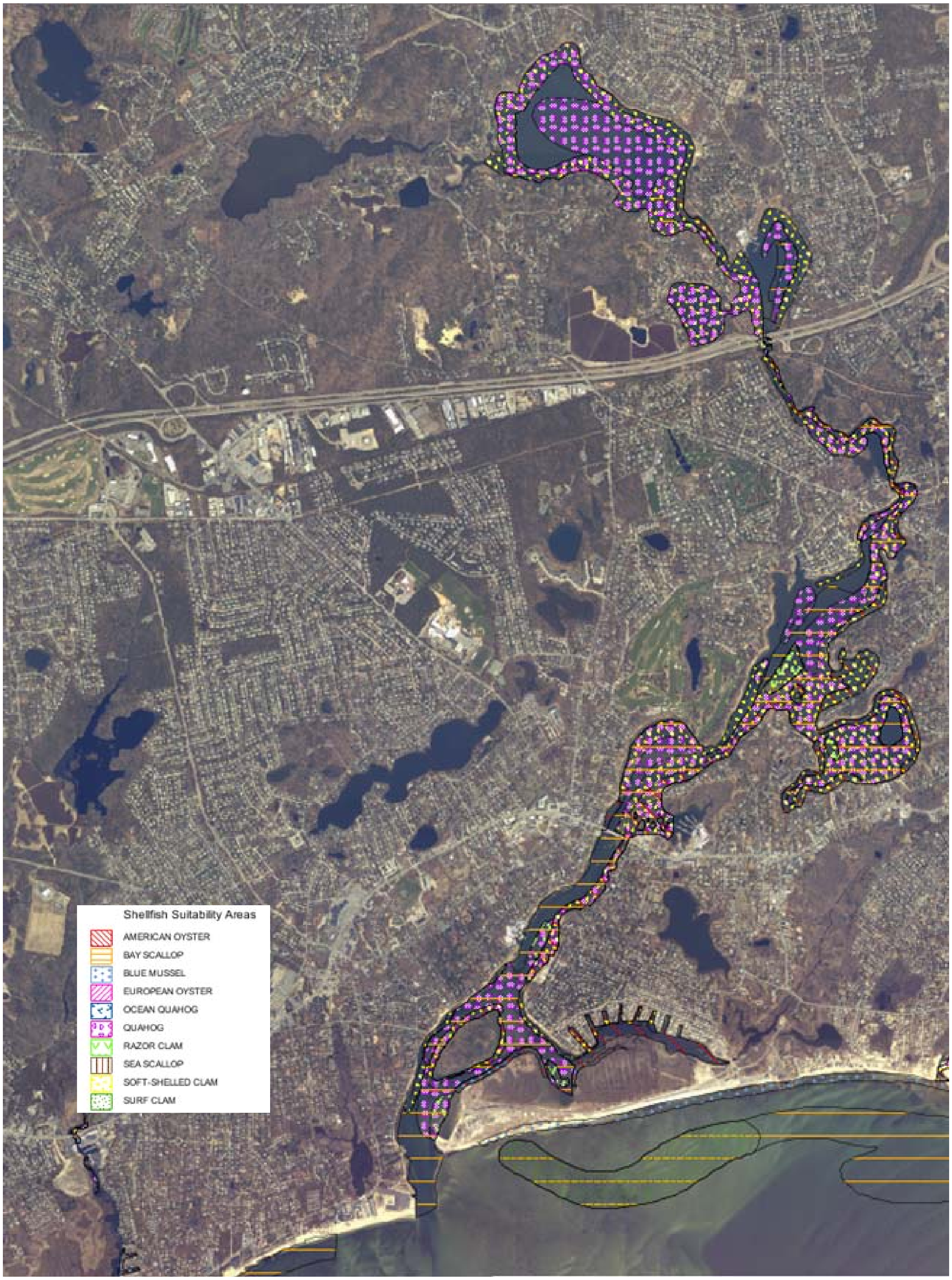

Figure II-4. Location of shellfish suitability areas within the Bass River Estuary (inclusive of Follins Pond and Grand Cove) as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence".

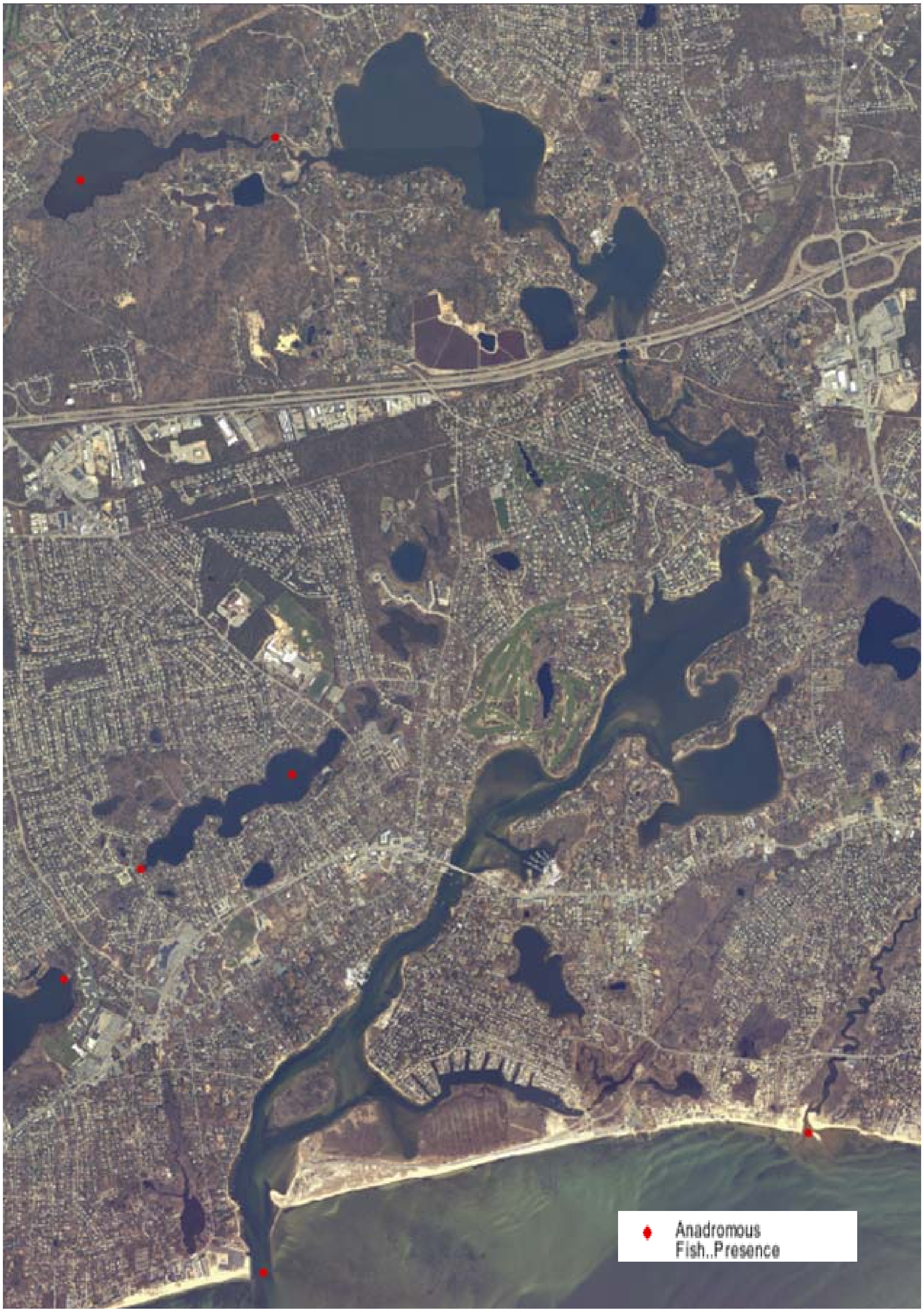

Figure II-5. Anadromous fish runs within the Bass River Estuary as well as the Mill Pond portion of the Bass River Estuarine System as determined by Mass Division of Marine Fisheries. The red diamonds show areas where fish were observed.

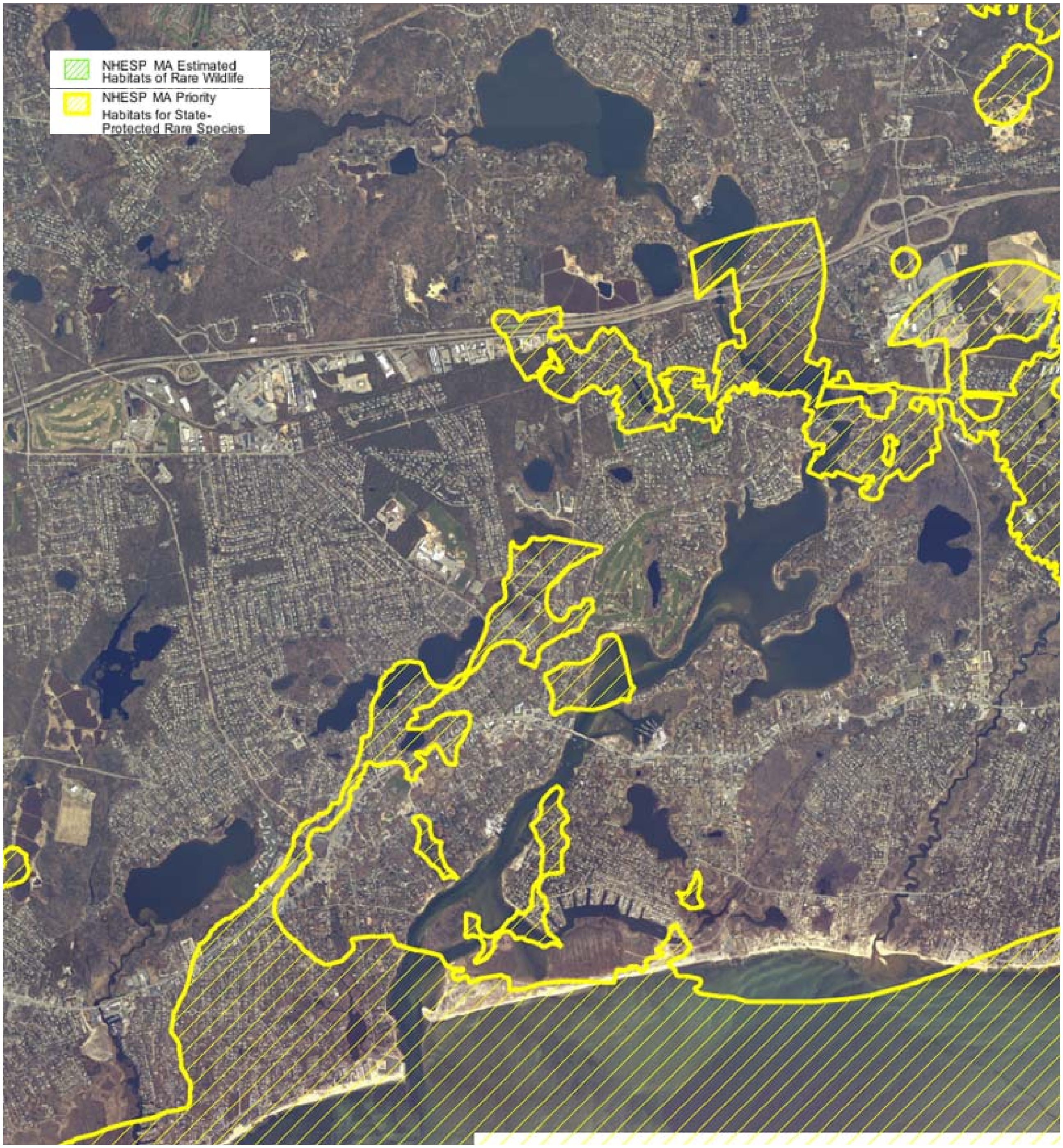

Figure II-6. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Bass River Estuarine System as determined by - NHESP.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation. These questions include surface water/groundwater interactions, groundwater travel times, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Bass River embayment system. The Bass River watershed is located on the boundary between Yarmouth and Dennis, 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 Bass River embayment system under evaluation by the Project Team. The Bass River estuarine system is a complex estuary with a number of road crossings and upper reaches far removed from its inlet. Watershed modeling was undertaken to sub-divide the overall watershed to the Bass River system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These travel distributions within each sub-watershed are used as a procedural check to gage the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in both the Sagamore and Monomoy flow cells on Cape Cod; the Bass River is a discharge boundary for both flow cells. Model assumptions for calibration of the Bass River Estuary included surface water discharges measured as part of the MEP stream flow program (2003 to 2005).

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

III.2 MODEL DESCRIPTION

Contributing areas to the Bass River system and its various sub-watersheds, such as Dinah’s Pond, Mill Pond, Fresh Pond and Kelley’s Bay, were delineated using regional models of the Sagamore and Monomoy Lens flow cells (Walter and Whealan, 2005). Since the Bass

River is a discharge boundary between the two flow cells, both models are used for the contributing area delineations. 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 Bass River system and its sub-watersheds and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers, while the Monomoy Flow Model grid consists of 164 rows, 220 columns, and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet in both models. The top 17 layers of the of both models extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 519 feet below NGVD 29 in the Sagamore and up to 525 ft below NGVD 29 in the Monomoy); since bedrock is 200 to 300 feet below NGVD 29 in the Bass River area the two lowest model layers were active in this area of the model. Because the Monomoy water table has a lower maximum elevation, the upper three layers of its model are dry. 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 both flow cells.

The glacial sediments that comprise the aquifer of both the Sagamore and Monomoy Lens consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. The Bass River system watershed is generally located in the Harwich Outwash Plain Deposits, although a portion of the Sandwich Moraine extends along the southern side of Mill Pond and terminates along the western shore of Follins Pond (Oldale, 1974). Modeling and field measurements of contaminant transport at the Massachusetts Military Reservation have shown that similar deposited materials are highly permeable (*e.g*., Masterson, *et al*., 1996). Given their high permeability, direct rainwater run-off is typically rather low for this type of watershed system. Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters in the groundwater models were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and stream flow data collected in 1989-1990 as well as 2003.

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

III.3 BASS RIVER SYSTEM CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Bass River embayment system, including Mill Pond, Dinah's Pond, and other sub-estuaries (Figure III-1) were determined by the United States Geological Survey (USGS). Model outputs of the watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (d) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model and (e) to address streamflow measurements collected as part of the MEP. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineation includes 10 yr time of travel boundaries. Overall, forty-two (42) sub-watershed areas, including eight to freshwater ponds, were delineated within the Bass River study area.

Table III-1 provides the daily freshwater discharge volumes for various sub-watersheds as calculated from the groundwater model; these volumes were used in the salinity calibration of the tidal hydrodynamic models and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to the directly measured surface water discharges. The overall estimated freshwater flow into the Bass River system from the MEP delineated watershed is 79,024 m3/d. This flow includes inflow from Long Pond, which has a surface water discharge to Parkers River (Howes, et al., in press) and groundwater discharge from Pine Pond and the large wellhead recharge areas that are located along the northeastern boundary of the overall watershed to the Bass River system.

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

The MEP watershed area for the Bass River system as a whole is 9% smaller (975 acres) than the 1998 CCC delineation. This calculation accounts for the portions of the subwatersheds that flow out of the Bass River overall watershed; the difference in area between the CCC and MEP watersheds without accounting for outflow is 1,193 acres. The majority of the difference is largely attributable to changes along the eastern boundary of the system watershed; portions of the NW Dennis Wells recharge area flow out of the system watershed and the boundary is located a bit further west than the CCC delineation. The MEP watershed delineation also includes interior sub-watersheds to various components of the Bass River system, such as ponds and public water supply wells that were not included in the CCC delineation. These refinements are another benefit of the update of the regional groundwater model (Walter and Whealan, 2005).

The evolution of the watershed delineations for the Bass River system has allowed increasing accuracy as each new version adds new hydrologic data to that previously collected;

the model allows all this data to be organized and to be brought into congruence with adjacent watersheds. The evaluation of older data and incorporation of new data during the development of the model is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model 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. The MEP watershed delineation was used to develop the watershed nitrogen loads to each of the aquatic systems and ultimately to the estuarine waters of the Bass River system (Section V.1).

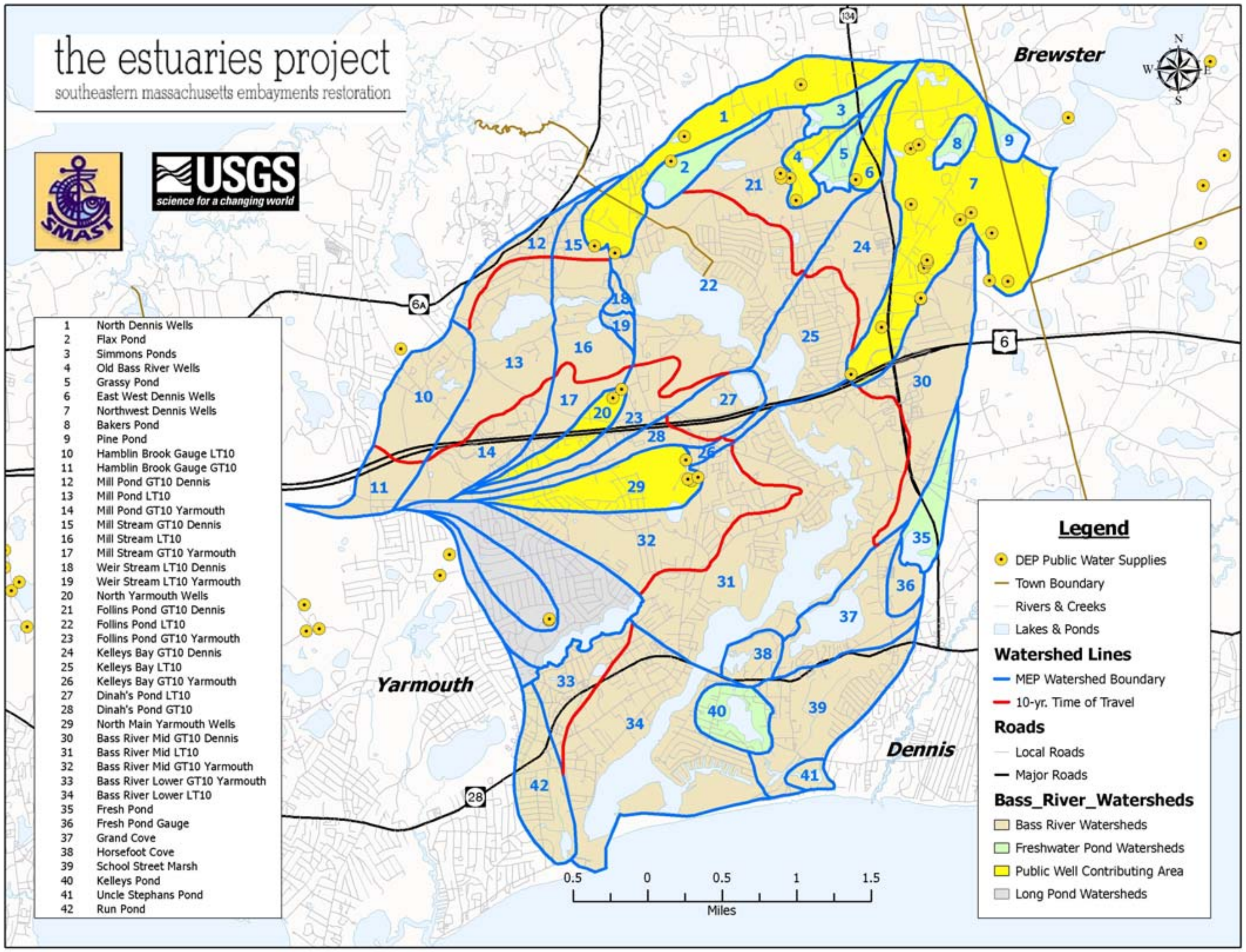

Figure III-1. Watershed delineation for the Bass River estuary system. Subwatershed delineations are based on USGS groundwater model output with modifications to better address pond and estuary shorelines and MEP stream gage measurements. Ten-year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watershed groups (e.g., Mill Pond) were selected based upon the functional estuarine sub-units in the water quality model (see Section VI). Watersheds to Long Pond are shared with the Parkers River system and are detailed in the Parkers River MEP Report (Howes, et al., 2009).

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

Watershed	#	Watershed Area (acres)	% contributing to Estuaries	Discharge	
				m^3/day	ft^3/day
N Dennis Wells	1	426	100	3,272	115,534
Flax Pond	2	100	100	769	27,167
Simmons Ponds	3	91	100	702	24,775
Old Bass River Wells	4	89	100	685	24,184
Grassy Pond	5	73	100	559	19,724
East West Dennis Wells	6	79	100	605	21,367
NW Dennis Wells	7	967	70	5,161	182,243
Bakers Pond	8	45	70	238	8,400
Pine Pond	9	44	41	137	4,836
Hamblin Brook Gage LT10	10	293	100	2,245	79,295
Hamblin Brook Gage GT10	11	124	100	950	33,549
Mill Pond GT10 N	12	113	100	869	30,694
Mill Pond LT10	13	369	100	2,833	100,054
Mill Pond GT10 S	14	237	100	1,821	64,312
Mill Stream GT10 N	15	67	100	516	18,211
Mill Stream LT10	16	238	100	1,825	64,438
Mill Stream GT10 S	17	127	100	973	34,365
Weir Stream	18	22	100	167	5,893
Muddy Creek	19	24	100	185	6,535
N Yarmouth Wells	20	109	100	834	29,457
Follins Pond GT10 D	21	361	100	2,773	97,935
Follins Pond LT10	22	666	100	5,113	180,580
Follins Pond GT10 Y	23	190	100	1,458	51,502
Kelleys Bay GT10 D	24	262	100	2,014	71,113
Kelleys Bay LT10	25	295	100	2,262	79,894
Kelleys Bay GT10 Y	26	19	100	143	5,065
Dinah's Pond LT10	27	123	100	942	33,267
Dinah's Pond GT10	28	116	100	891	31,477
N Main Yarmouth Wells	29	274	100	2,106	74,362

Table III-1 (continued). Daily groundwater discharge to each of the sub-watersheds in the watershed to the Bass River system estuary, as determined from the USGS groundwater model.

Watershed	**#**	**Watershed Area (acres)**	**% contributing to Estuaries**	**Discharge**	
				m^3/day	**ft^3/day**
Bass R Mid GT10 D	30	478	100	3,666	129,468
Bass R Mid LT10	31	1062	100	8,152	287,901
Bass R Mid GT10 Y	32	512	100	3,929	138,752
Bass R Lower GT10 Y	33	174	100	1,338	47,258
Bass R Lower LT10	34	645	100	4,954	174,933
Fresh Pond	35	128	79	775	27,363
Fresh Pond Gage	36	61	100	470	16,582
Grand Cove	37	211	100	1,621	57,242
Horsefoot Cove	38	75	100	572	20,209
School Street Marsh	39	405	100	3,105	109,651
Kelleys Pond	40	144	100	1,104	38,979
Uncle Stephans Pond	41	28	100	217	7,658
Run Pond	42	212	100	1,629	57,515
Long Pond Well & Long Pond*	A	650	95	4,444	156,955
TOTAL BASS RIVER SYSTEM (adjusted)*				79,024	2,790,695

Notes: 1) watershed areas are unadjusted to account for outflow, 2) discharge volumes are adjusted to account for flow of recharge out of the watershed and are based on 27.25 in of annual recharge; 3) percentage of outflow is determined by length of down gradient pond shoreline or watershed boundary; 4)*more discharge from Long Pond into Bass River occurred during the time of MEP water quality monitoring; the connection of Long Pond to Seine Pond was been enhanced after monitoring was completed (Howes, et al., 2009) and flow has been increased, so more of the Long Pond watershed recharge flows into Seine Pond and less into Bass River; this increase/change is included in buildout scenarios; watershed flows in this table are based on the more restricted connection between Long Pond and Seine Pond.

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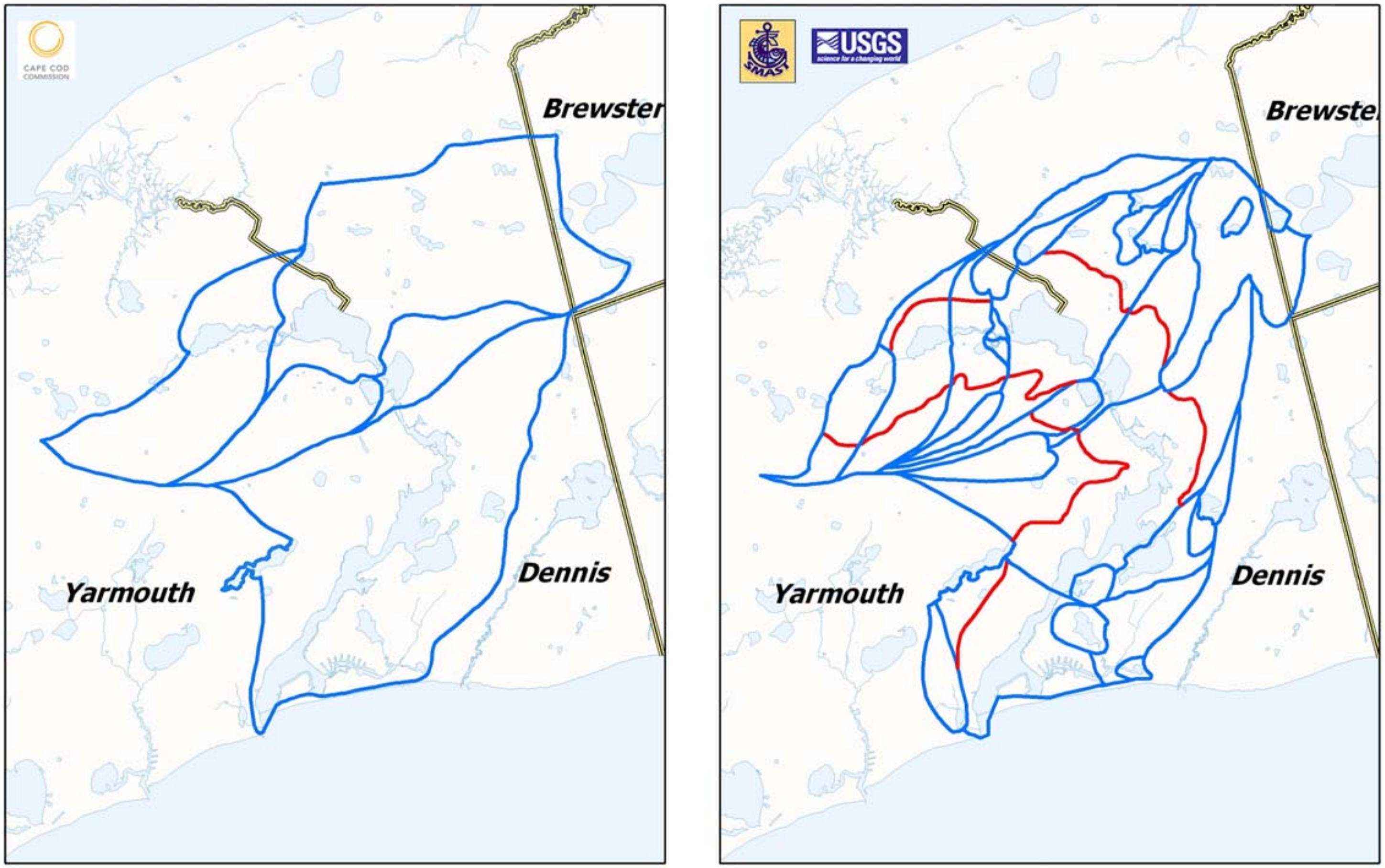


Used in 1996, 2001 & 2009 Regional Policy Plans (based on delineation in Eichner, et al., 1998)

Delineated by USGS for MEP Analysis

Red lines indicate ten year time-of-travel lines

Figure III-2. Comparison of MEP watershed and sub-watershed delineations used in the current analysis and the Cape Cod Commission delineation (Eichner, et al., 1998), which has been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009.

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

In order to determine watershed nitrogen loading inputs to the Bass River estuary system, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of estuary and its watersheds (Section III). The Bass River watershed was sub-divided to define contributing areas or subwatersheds to each of the major inland freshwater systems (which can attenuate nitrogen loads) and to each major portion of the estuary. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches estuary waters in less than 10 years or greater than 10 years. A total of 42 subwatersheds were delineated in the overall Bass River watershed, including watersheds to the following freshwater ponds: Flax, Grassy, Bakers, Pine, Fresh, Kelleys, and a combined watershed for Northern and Southern Simmons Ponds. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each portion of the estuary (see Chapter III).

The initial task in the MEP land use analysis is to gage whether or not nitrogen discharges to the watershed have reached the estuary. This involves a temporal review of land use changes, the time of groundwater travel provided by the USGS watershed model, and review of data at natural collections points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. Ten-year time of travel subwatersheds in the Bass River watershed have been delineated for ponds, streams and the estuary itself. Simple review of less than and greater than watersheds indicates that 57% of the unattenuated nitrogen load from the whole watershed is within less than 10 year travel time to the estuary (Table IV-1). This percentage is relatively low compared to other estuary watersheds, but review of land uses within the watershed shows 1) that most of the development in the Dennis portion of the watershed is within the 10 year time of travel and 2) review of land sales in both towns shows that more than 60% of parcels have been in the same ownership for 10 years or more. Since many of the parcels with new ownership may have existed for longer than 10 years, but have a recorded date of sale within the last 10 years, this analysis is somewhat conservative and suggests that the majority of development, and its nitrogen loads, beyond the 10 year groundwater travel time has also reached the estuary. In addition, the recharge areas to the public water supplies, which are generally located in the greater than 10 year travel time watersheds, would tend to capture their nitrogen loads and redistribute it to the higher density areas, which also tend to be in the less than 10 year travel time. 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) and that the distinction between time of travel in the subwatersheds is not important for modeling existing conditions. Overall and based on the review of all this information, it was determined that the Bass River estuary is currently in balance with its watershed load.

Table IV-1. Percentage of unattenuated nitrogen loads in less than ten year time-of-travel subwatersheds to Bass River.

WATERSHED	LT10	GT10	TOTAL	%LT10
Name	kg/yr	kg/yr	kg/yr	
Mill Pond	4,038	5,181	9,220	44%
Mill Stream	897	2,593	3,490	26%
Follins Pond	8,937	5,049	13,987	64%
Kelley Bay	4,536	4,871	9,407	48%
Bass River Mid	16,790	13,891	30,680	55%
Bass River Low	19,354	9,957	29,311	66%
Bass River Whole System	54,553	41,543	96,095	57%

Notes: loads have been corrected to 1) include division of portions of nitrogen load from ponds and wellhead protection areas to downgradient subwatersheds, 2) exclude nitrogen loads that are discharged outside of the Bass River system watershed from ponds or wellhead protection areas on the system watershed boundaries, and 3) include nitrogen loads from Long Pond under the more restricted conditions that existed during the collection of water quality data used for the Bass River assessment (current conditions allow more of the Long Pond load, which is in the GT10 area of Bass R Low, to discharge out of the system and into Parkers River). Loads include atmospheric loading on the estuary surface waters.

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

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) within the Bass River watershed was determined based upon a site-specific study of streamflow and assumed and measured attenuation in the upgradient freshwater ponds. Streamflow was characterized at Hamblin Brook and in the stream discharging from Fresh Pond. Subwatersheds to these stream discharge points allowed comparisons between field collected data from the streams and ponds and estimates from the nitrogen-loading sub-model. Nitrogen attenuation in individual ponds is generally estimated based on available information. Attenuation through the ponds is conservatively assumed to equal 50% unless available monitoring and pond physical data is reliable enough to calculate a pond-specific attenuation factor. Streamflow and associated surface water attenuation is included in the MEP's nitrogen attenuation and freshwater flow investigation, presented in Section IV.2.

Natural attenuation during stream transport or in passage through fresh ponds of sufficient size to effect groundwater flow patterns (area and depth) is a standard part of the data collection effort of the MEP. In the present effort, eight freshwater ponds have delineated subwatersheds within the Bass River watershed. If smaller aquatic features that have not been included in this MEP analysis were providing additional attenuation of nitrogen, nitrogen loading to the estuary would only be slightly (~10%) overestimated given the distribution of nitrogen sources within the watershed.

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary without flowing through one of these interim pond and stream measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Bass River Estuarine System; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying water-column. Nitrogen regeneration focused on summer months, the critical nitrogen management interval and the focal season of the MEP approach and application of the Linked Watershed-Embayment Management Model (Section IV.3).

IV.1.1 Land Use and Water Use Database Preparation

Since the watershed to Bass River includes portions of the towns of Yarmouth, Dennis, and Brewster, Estuaries Project staff obtained digital parcel and tax assessor's data from the towns to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data for all three towns in the watershed are from 2009. These land use databases contain traditional information regarding land use classifications (MassDOR, 2008) plus additional information developed by the towns. This effort was completed with the assistance from GIS staff from the Cape Cod Commission (CCC).

Figure IV-1 shows the land uses within the Bass River estuary watershed. Land uses in the study area are grouped into nine land use categories: 1) residential, 2) commercial, 3) industrial, 4) agricultural, 5) mixed use, 6) undeveloped (including residential open space), 6) public service/government, including road rights-of-way, 8) freshwater ponds, and 9)

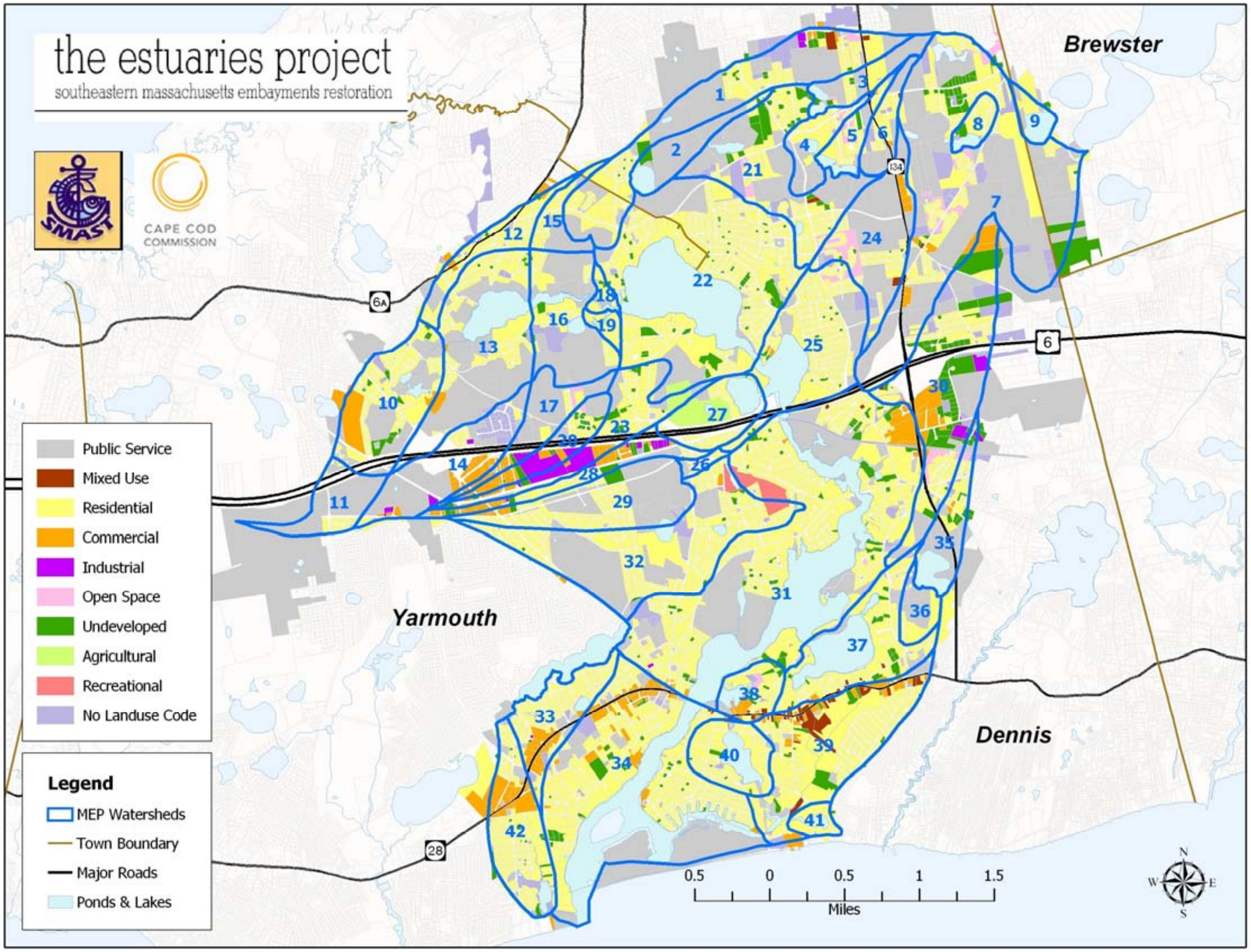


Figure IV-1. Land-use in the Bass River system watershed and subwatersheds. Watershed extends over portions of the Towns of Yarmouth, Dennis, and Brewster. Land use classifications are based on respective town assessor classifications and MADOR (2008) categories. Base assessor and parcel data for Yarmouth, Dennis, and Brewster are all from the year 2009.

unclassified properties. These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MADOR, 2008). “Public service” in the MADOR system is tax-exempt properties, including lands owned by government (e.g., wellfields, schools, golf courses, open space, roads) and private groups like churches and colleges. Unclassified parcels are properties without any assessor land use classifications.

Public service land uses are the dominant land use type in the overall Bass River watershed and occupy 44% of the watershed area (Figure IV-2). Examples of these land uses are lands owned by town and state government (including golf courses, open space, and wellhead protection lands), housing authorities, and churches. Residential land uses occupy the second largest area with 38% of the watershed area. It is notable that land classified by the town assessor as undeveloped is 8% of the overall watershed area. The Mill Pond and Kelleys Bay subwatersheds are where most of the public service lands are; parcel examples in these subwatersheds include the Bayberry Hills Golf Course, the former Town of Yarmouth landfill, Wixon Middle School, and the protected lands around the Dennis Water District public water supply wells.

In all the subwatershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type, ranging between 71% and 84% of all parcels in these subwatersheds and 82% of all parcels in the Bass River system watershed. Single-family residences (MassDOR land use code 101) are the dominant type of residential parcel; these represent 91% to 96% of residential parcels in the individual subwatershed groupings and 94% of the residential parcels throughout the Bass River system watershed.

In order to estimate wastewater flows within the Bass River study area, MEP staff also obtained parcel-by-parcel water use data from the Town of Yarmouth and the Dennis Water District. Five years of water use information (2001 through 2005) was obtained from the Town of Yarmouth, Department of Public Works (George Allaire, DPW Director, 2/06), while three years of water use was obtained from the Dennis Water District. The water use data was linked to the respective town parcel databases by the CCC GIS Department staff. Measured water use is used to estimate wastewater-based nitrogen loading from the individual parcels; average water use for each parcel is used for parcels with multiple years of data. The final wastewater nitrogen load for each parcel is based upon the measured water-use, wastewater nitrogen concentration, and consumptive loss of water before the remainder is treated in a septic system (see Section IV.1.2). All parcels are assumed to use on-site septic systems unless additional information is available.

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

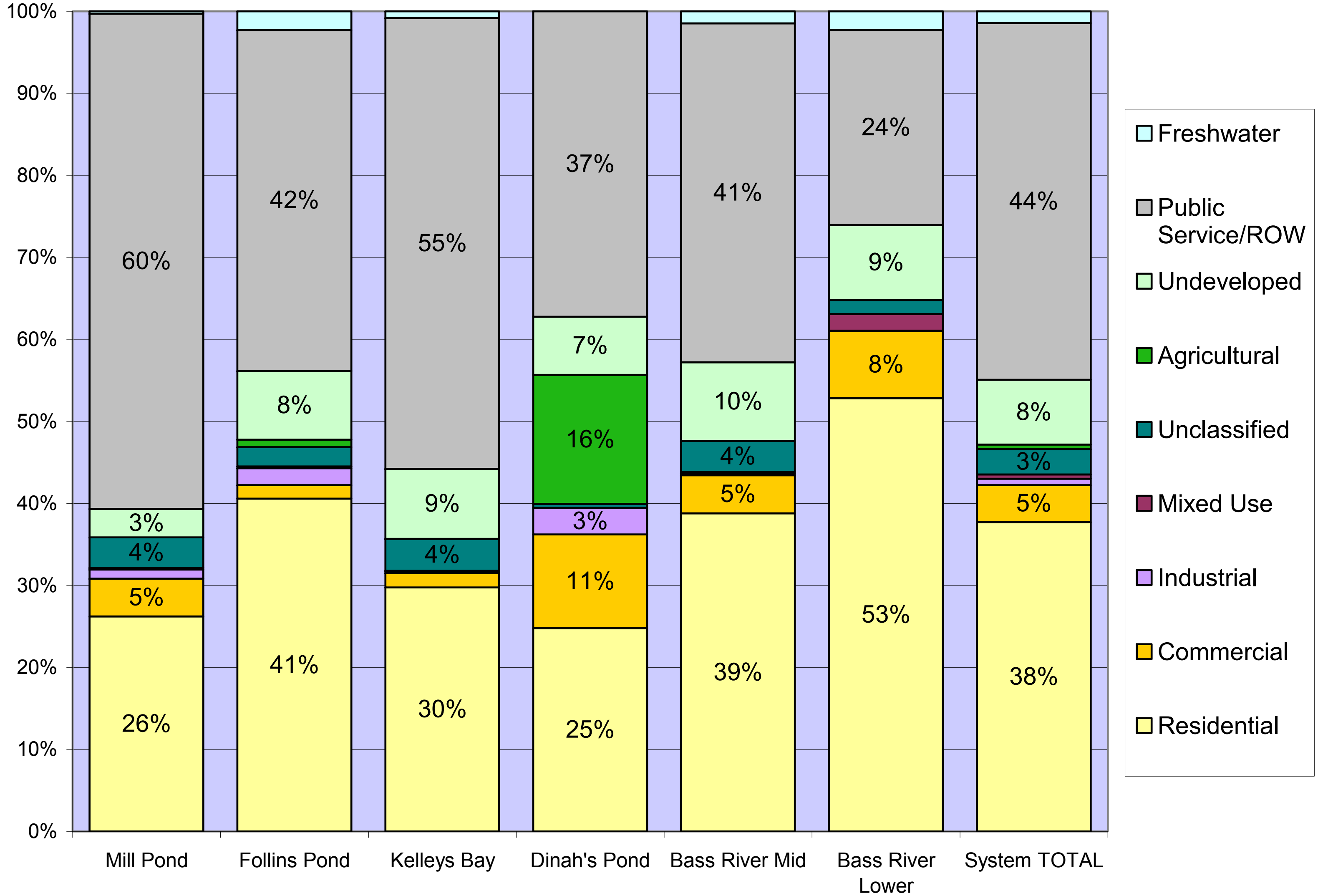


Figure IV-2. Distribution of land-uses by area within the Bass River system watershed and six component subwatersheds. Land use categories are generally based on town assessor's land use classification and grouping recommended by MADOR (2008). Unclassified parcels do not have an assigned land use code in the town assessor's databases. Only percentages greater than or equal to 3% are shown.

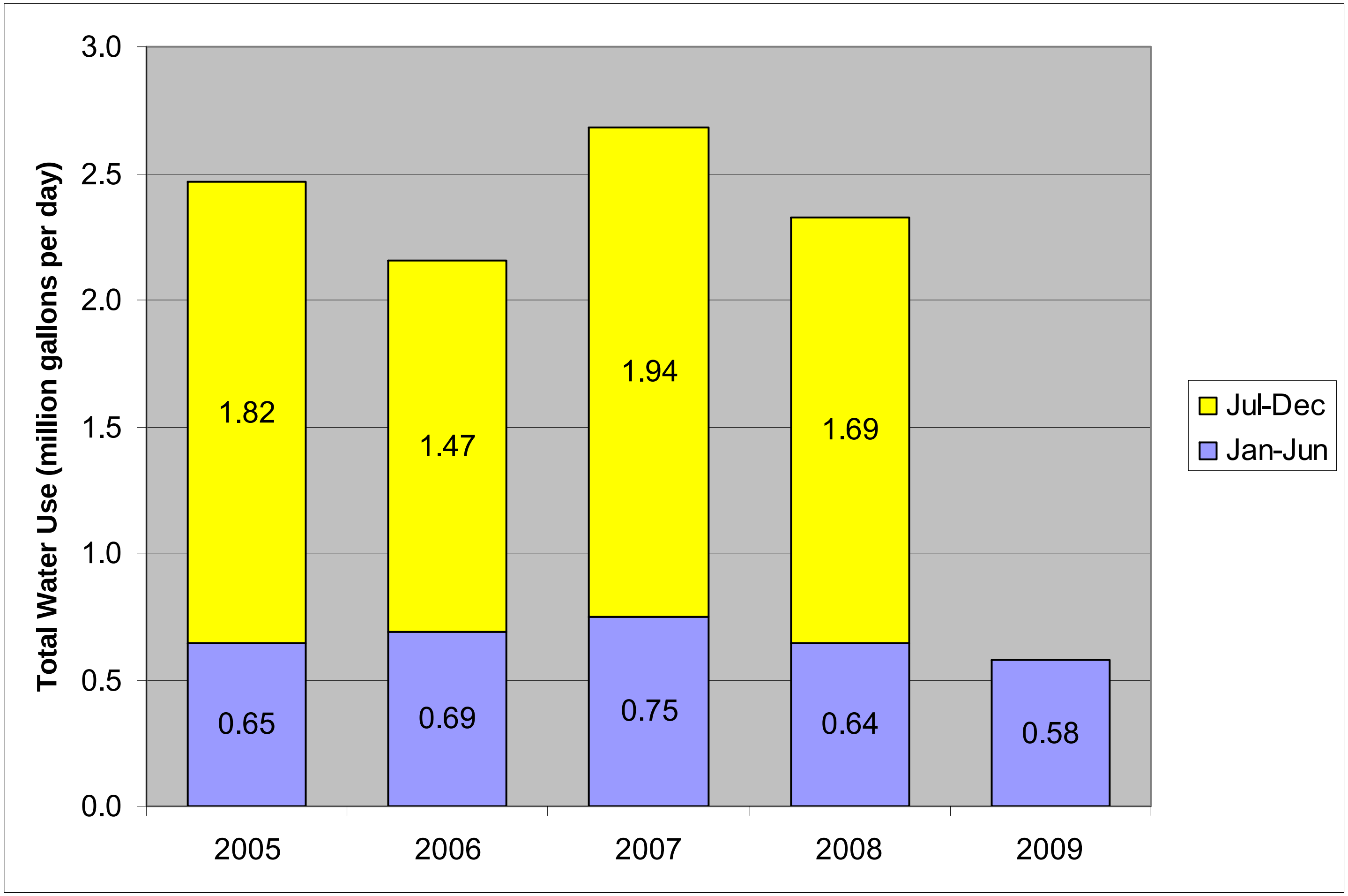


Figure IV-3. Dennis Seasonal Water Uses (2005-2009). Water use was only available for the first portion of 2009 during the development of the watershed nitrogen loading model. In order to address this, MEP staff utilized three years of data based on July to June calendar years to calculate average water use for parcels. However, since the July to December 2006 period was exceptional low, this period was replaced with the comparable 2007 period when calculating the average parcel water use. Overall, substitution of this period results in total average flows for the July to December period that are approximately 2% higher than a simple average of 2005, 2007, 2008 total water use for this period.

than the next closest match (2005). As a result, the Town of Dennis individual parcel water uses are based on average water use from July 2008 to June 2009, July 2007 to June 2008, and a combined July to December 2007 and January to June 2007.

MEP staff also received state Groundwater Discharge Permit (GWDP) nitrogen effluent data from the MassDEP (personal communication, Brian Dudley, 4/09) and alternative, denitrifying septic system total nitrogen effluent data from the Barnstable County Department of Health and the Environment (personal communication, Sue Rask and Brian Baumgaertel, 3/09). The only GWDP in the Bass River watershed is the effluent discharge from the Town of Yarmouth Septage Treatment Facility. Effluent flow from the septage treatment facility is discharged at one of two locations: the Bayberry Golf Course as irrigation water or at the disposal site south of Buck Island Road. The BCDHE has monitoring performance data on 54 innovative/alternative septic systems in the Bass River watershed: 15 are in the Yarmouth portion and 39 are in the Dennis portion. The reporting data from these two agencies was used to develop wastewater nitrogen loads for each of these sites.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

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

However, given the seasonal shifts in occupancy and rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use approach is applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to 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 that reaches the aquatic receptors downgradient in the aquifer.

All nitrogen losses within the septic system are incorporated into the MEP analysis. For example, information developed at the MassDEP Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa et al. 2001). Downgradient studies of septic system plumes in similar soils indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

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

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

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

Overall, the MEP water use approach for determining septic system nitrogen loads has been both calibrated and validated in a variety of watershed settings. The approach: (a) is consistent with a suite of studies on per capita nitrogen loads from septic systems in sandy soils and outwash aquifers; (b) has been validated in studies of the MEP Watershed "Module", where there has been excellent agreement between the nitrogen load predicted and that observed in direct field measurements corrected to other MEP Nitrogen Loading Coefficients (e.g., stormwater, lawn fertilization); (c) the MEP septic nitrogen loading coefficient agrees with specific studies of consumptive water use and nitrogen attenuation between the septic tank and the discharge site; and (d) the watershed module provides estimates of nitrogen attenuation by freshwater systems that are consistent with a variety of ecological studies. It should be noted that while points b-d support the use of the MEP Septic N Coefficient, they were not used in its development. The MEP Technical Team has developed the septic system nitrogen load over many years, and the general agreement among the number of supporting studies has greatly enhanced the certainty of this critical watershed nitrogen loading term.

The independent validation of the water quality model (Section VI) and the reasonableness of the freshwater attenuation (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters provides a safety factor relative to other higher loads that are generally used for septic systems in regulatory situations. The lower concentration results in slightly higher amounts of nitrogen mitigation (estimated at 1% to 5%)) needed to lower embayment nitrogen levels to a nitrogen target (e.g. nitrogen threshold, cf. Section VIII). The additional nitrogen removal is not proportional to the septic system nitrogen level, but is related to the how the septic system nitrogen mass compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

In order to provide an independent validation of the average residential water use within the Bass River watersheds, MEP staff reviewed 2000 US Census population values for the Towns of Yarmouth and Dennis. Since Brewster occupies such a small portion of the watershed, it was not included in this validation analysis. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), this is based upon the estimate that on average each person generates 55 gpd of wastewater. Based on data collected during the 2000 US Census, average occupancy within Yarmouth is 2.15 people per housing unit with 73% year-round occupancy of available housing units, while average occupancy in Dennis is 2.13 people per housing unit with 53% year-round occupancy. Measured average water use for single-family residences with municipal water accounts in the Bass River MEP study area is 180 gpd. If this flow is multiplied by 0.9 to account for consumptive use the study area average wastewater generation per residence is 162 gpd.

In order to provide a check on the per residence wastewater generation from the measured water use, MEP staff estimated wastewater generation from Census data and the Title 5 estimate of per capita daily wastewater generation. While this is a less accurate estimate than that based upon measured water use, it does provide an independent check on the water-use estimate. Yarmouth and Dennis 2000 Census based average occupancies were averaged since they are approximately the same, 2.15 and 2.13 people per housing unit, respectively. Multiplying this occupancy by the state Title 5 estimate of 55 gpd of wastewater per capita results in an average estimated water use per residence of 118 gpd, uncorrected for seasonal occupancy. Estimates of summer populations on Cape Cod derived from a number of approaches (*e.g.*, traffic counts, garbage generation, WWTF flows) suggest average population increases from two to three times year-round residential populations measured by the US Census. If it is assumed that seasonal properties are occupied at twice the year-round occupancy for three months in Yarmouth, the estimated average town-wide water use would be 147 gpd, whereas if the seasonal properties are occupied at three times the year-round occupancy for three months, the estimated average water use would be 178 gpd. If the same calculation is completed based on Dennis occupancy and seasonal properties, the respective flows would be 162 gpd and 212 gpd. Given that the average wastewater generation for the shared Bass River watershed is toward the high end of the Yarmouth range and the low end of the Dennis range, this analysis suggests that the average water use is yielding representative average wastewater estimates for the Bass River watershed.

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

Water use information exists for 91% of the 9,542 developed parcels in the Bass River watershed. Parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have a listed account in the water use databases. Of the 837 developed parcels without water use accounts, 669 (80%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and are assigned the Bass River study area average water use of 180 gpd in the watershed nitrogen loading modules. Another 48 developed parcels without water use are parcels classified as other types of residential properties (e.g., multi-family or condominiums). Average water use for similar non-single family residential properties in the study area is 352 gpd. This average is assigned to the 48 non-single family residential properties without water use.

Alternative Septic Systems

As mentioned previously, there are 54 alternative, denitrifying septic systems in the Bass River study area that have total nitrogen effluent data in the Barnstable County Department of Health and the Environment database (personal communication, Sue Rask and Brian Baumgaertel, 3/09). These systems have 1 to 85 measurements. In order to reasonably reflect the impact of these systems, project staff only altered the standard MEP wastewater factor for those systems with five or more measurements. This approach reduced the total number of denitrifying septic systems to 31 that were included in the watershed nitrogen loading model. Among these systems, average total nitrogen effluent concentrations ranged between 6 and 82 ppm. Project staff used these site-specific, average measured effluent total nitrogen concentrations and the average measured water use from the town records to calculate average annual loads from each of these sites. These loads were incorporated into the watershed nitrogen loading module for the Bass River.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of watershed nitrogen loading to estuaries is usually fertilized areas: lawns, golf courses, and cranberry bogs. Residential lawns are usually the predominant source within this category. In order to add this source to the nitrogen loading model for the Bass River system, MEP staff reviewed available regional information about residential lawn fertilizing practices and incorporated site-specific information for the following golf courses: Dennis Pines, Dennis Highlands, Bayberry Hills, Bass River, and Blue Rock. An estimated nitrogen load is also included for the cranberry bogs and athletic fields in the watershed. The area of athletic fields was determined by review and digitizing of aerial photographs; these were assigned the same application rate as residential lawns. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts, while MEP staff contacted the golf course superintendents to obtain course-specific fertilizer application rates.

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

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

Given the importance of the fertilizer leaching rate to nitrogen loading from residential lawns, the MEP fertilizer leaching rate of 20% recently received a detailed review prepared by Horsley Witten Group Inc. The task was to independently determine a nitrogen fertilizer leaching rate from turf grass specific to the permeable soils typical of the watersheds to southeastern Massachusetts estuaries, and then compare it to the MEP analysis. The analysis used both the results of previous studies and new data collected subsequent to the initiation of the MEP. The results indicated a leaching rate of 19% and the study concluded that, "the MEP leaching rate estimate of 20% is reasonable" (Horsley Witten Group, 2009).

In order to obtain a site-specific estimate of nitrogen loading from the publicly-owned Bayberry Hills Golf Course and Bass River Golf Course, MEP staff contacted Rick Lawlor, Superintendent to obtain current (2/09) information about fertilizer application rates. Golf courses usually have different fertilizer application rates for different turf areas, usually higher annual application rates for tees and greens (~3 to 4 pounds per 1,000 square feet) and lower rates for fairways and roughs (~2 to 3.5 pounds per 1,000 square feet). At both of these two golf courses, Mr. Lawlor reported the same annual nitrogen application rates (in pounds per 1,000 ft2) for the various turf areas: greens, 3.7; tees, 3.0; fairways, 3.0, and rough, 1.5. Mr. Lawlor also reported that the fertilizers used are all controlled-release forms and noted that monitoring on the portion that receives spray discharge from the town septage treatment facility includes six years worth of leaching data collected at lysimeters.

MEP staff also contacted Mike Cummings, Superintendent of both the Dennis Pines and Dennis Highlands golf courses to obtain current (6/10) fertilizer application information on these two publicly-owned courses. Mr. Cummings reported that the following nitrogen application rates (in pounds per 1,000 ft2) are used at Dennis Pines: greens, 2.2; tees, 2.5; fairways, 3.5, and rough, 3.5. He also reported that the following nitrogen application rates (in pounds per 1,000 ft2) are used at Dennis Highlands: greens, 4.0; tees, 3.5; fairways, 3.5, and rough, 3.5. MEP staff was unsuccessful in obtaining similar site-specific information for Blue Rock Golf Course and this was assigned average nitrogen application rates from 16 golf courses surveyed during the course of the MEP.

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

Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture that has been conducted in southeastern Massachusetts (Howes and Teal, 1995). Based on this study, only the bog loses measurable nitrogen, the forested upland releases only very low amounts. For the watershed nitrogen loading analysis, the areas of active bog surface are based on a GIS coverage maintained by MassDEP for Water Management Act purposes. Cranberry bogs are located in five Bass River subwatersheds: Dinah's Pond, Mill Stream, NW Dennis Wells, Pine Pond and Hamblin Brook.

Nitrogen Loading Input Factors: Yarmouth Landfill and Septage Spray Irrigation /Bayberry Hills Golf Course

The site of the Yarmouth landfill, which is located just to the south of Route 6 and west of Station Avenue, is also the site of a portion of the Bayberry Hills Golf Course, which is constructed on top of the capped landfill. The Golf Course also receives some of its irrigation water as treated effluent from the Town of Yarmouth Septage Treatment Facility. In addition to all these uses, the location of the site near the outer edge of the Bass River watershed and straddling two subwatersheds adds to the challenge of definitively assigning nitrogen loads from this site to each of the potentially impacted subwatersheds.

MEP staff began the development of a nitrogen loading estimate for this site by reviewing MassDEP groundwater discharge permit data for the Town of Yarmouth Septage Treatment Facility (personal communication, Brian Dudley, 4/09). The treatment facility utilizes two discharge locations for treated effluent: spray irrigation at the holes 2-8 at the Links portion of the Bayberry Hills Golf Course and fields at a town disposal site south of Buck Island Road. Flow and nitrogen concentration data was provided for the years 2004 through 2007. The Buck Island Road discharge site is located in the Parker's River watershed and is addressed in the MEP report for that system (Howes, et al., in press, Parkers River MEP report). MEP staff reviewed the monthly effluent flows directed to each discharge location (Figure IV-4). Total annual discharge from the septage treatment facility ranged between 14.1 and 19.6 million gallons between 2004 and 2007. Annual effluent discharge flows to the Bayberry Hills site between 2004 and 2007 ranged between 9.4 and 15.4 million gallons. Utilizing the reported total nitrogen concentrations, MEP staff determined that annual loads at the Bayberry Hills site during this same period varied between 261 kg and 579 kg with a four-year average of 374 kg/y.

As mentioned above, the Bayberry Hills Golf Course applies nitrogen fertilizers to the greens, tees, fairways, and roughs for the portion of the golf course constructed on top of the capped landfill. The following nitrogen application rates (in pounds per 1,000 ft2) were reported for the course's turf areas: greens, 3.7; tees, 3.0; fairways, 3.0, and rough, 1.5. After digitizing the corresponding turf areas on the portions of the golf course within the Bass River watershed and applying the appropriate recharge and turf attenuation factors, MEP staff determined that the golf course annually adds 342 kg of nitrogen to the Bass River watershed.

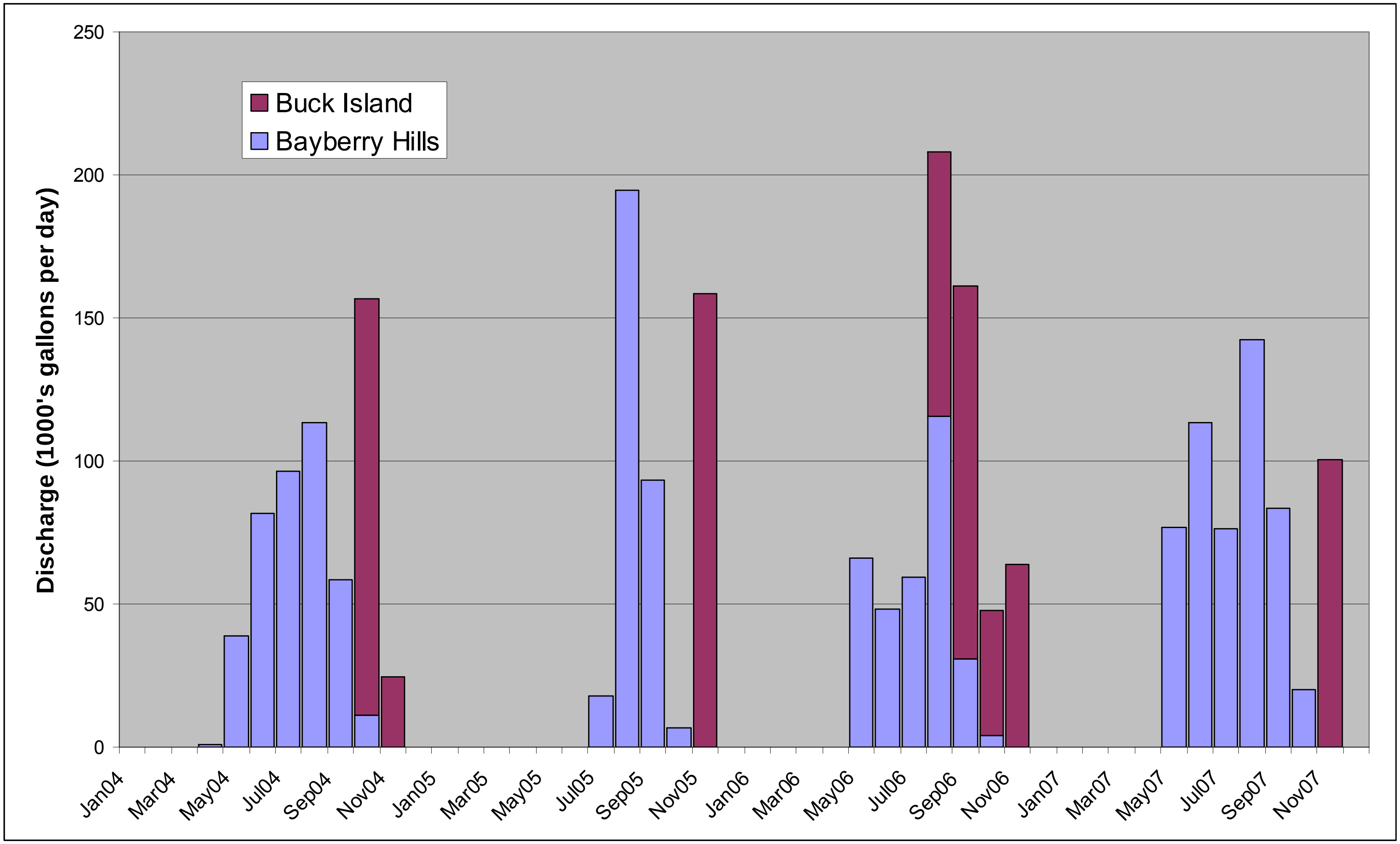


Figure IV-4. Total effluent discharge from the Town of Yarmouth Septage Treatment Facility (2004-2007). Effluent is discharged at two locations: a portion of the Bayberry Hills Golf Course that is built on the capped town landfill and a discharge area south of Buck Island Road. Monthly effluent discharge at both locations is shown. Only the Bayberry Hills site is located within the Bass River watershed. Annual nitrogen loads were determined from reported effluent flow and total nitrogen concentrations and averaged to produce the annual load used in the MEP watershed nitrogen loading calculations. Data provided by MassDEP (personal communication, B. Dudley, 4/09).

In addition to the septage facility effluent and golf course fertilizers, this site also contains the capped Town of Yarmouth landfill. The project to cap the landfill and install a golf course on top of the cap was completed in 2000. MEP staff obtained and reviewed water quality data collected between 2005 and 2010 from the groundwater well monitoring network around the landfill from the Town of Yarmouth Department of Public Works (personal communication, Mona Solamonte, May 2010).

The groundwater monitoring data includes nitrate-nitrogen, but does not include total nitrogen or ammonium-nitrogen data. Based on groundwater monitoring data from the Town of Brewster landfill (Cambareri and Eichner, 1993) that includes ammonium-nitrogen, nitrate-nitrogen and other typical inorganic contaminant measurements, MEP staff determined a relatively strong relationship between ammonium-nitrogen and alkalinity concentrations (NH4-N = 0.0352*ALK - 0.3565; r2 = 0.82). This relationship was used to determine estimated ammonium-nitrogen concentrations from the Yarmouth landfill based on the alkalinity concentrations. The estimated ammonium-nitrogen and measured nitrate-nitrogen concentrations were summed to provide an estimate of dissolved inorganic nitrogen. Although nitrate-nitrogen and ammonium-nitrogen concentrations are not a complete measure of all nitrogen species, landfills do not tend to release significant portions of dissolved organic nitrogen (Pohland and Harper, 1985). As an aside, the alkalinity readings from the Yarmouth wells are relatively consistent and generally do not show any significant upward or downward trend between 2005 and 2010.

In order to develop a nitrogen load for the site, MEP staff determined average alkalinity concentrations between 2005 and 2010 for wells in the landfill monitoring network and selected wells with average concentrations greater than 100 mg/l CaCO3 (Figure IV-5). These wells are clearly impacted and review of well logs, available water table contours and geologic cross-sections in the area (e.g., SEA, 1995) show that they are in the predominant groundwater flow path (both direction and depth) from the site. As a point of reference, USGS monitoring of alkalinity concentrations in groundwater throughout Cape Cod during the late 1970's found an average alkalinity concentration of 7.2 mg/l CaCO3 (Frimpter and Gay, 1979). All seven of the selected wells are located within the MEP Bass River watershed.

Staff then determined the average nitrogen concentration for each of the wells and averaged these concentrations to determine an average nitrogen concentration associated with groundwater flowing from the area of the landfill. The overall average estimated total nitrogen concentration is 10 mg/l. Using the average recharge rate, the area of solid waste (SEA, 1995), and this concentration, an annual nitrogen load of 1,575 kg was determined for the site.

Based on these assessments, the golf course, landfill, and septage effluent disposal collectively add 2,291 kg/yr of nitrogen to the Bass River watershed. However, one could make the case that the landfill monitoring is reflective of nitrogen loading impacts from all three uses, in which case the total annual load is closer to the 1,575 kg estimated for the landfill. MEP staff decided that the nitrogen load for the site would include the nitrogen load from the golf course and the landfill, but would exclude the load from the septage effluent. This approach would provide some acknowledgement that the landfill monitoring likely reflects the cumulative impact from the site while also acknowledging the uncertainties and limitations inherent in the landfill monitoring data. Also utilizing one source as a conservative assumption will simplify future management evaluations. Since the site is located on the watershed boundary between the Weir Road Gage GT10 subwatershed (#11) and Mill Pond GT10S subwatershed (#14), the landfill load was divided among these subwatersheds based the area of solid waste and the golf course load was divided based on the various turf areas within each subwatershed.

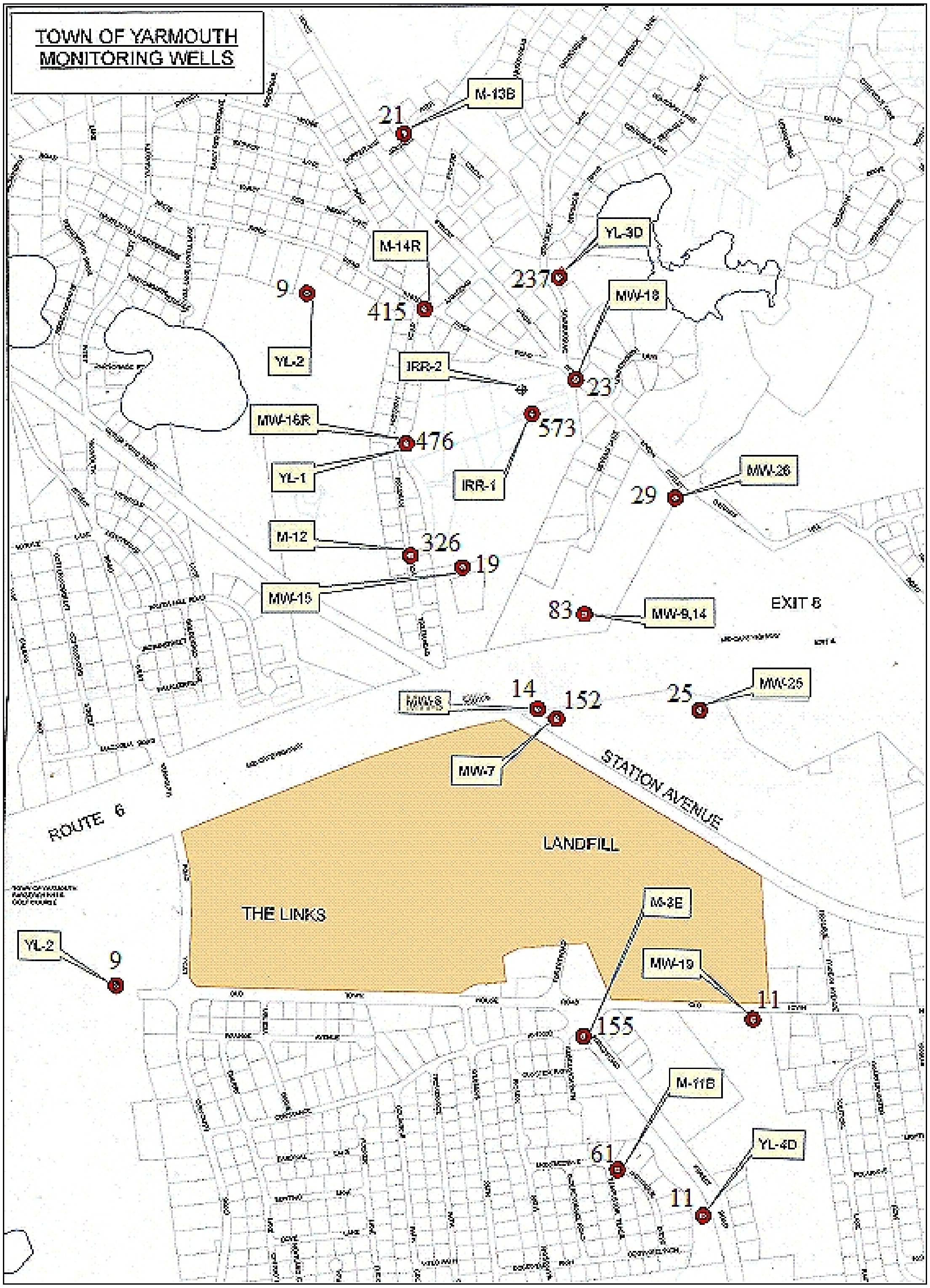


Figure IV-5. Average Alkalinity concentrations (mg/l CaCO3) in monitoring wells near the Town of Yarmouth landfill (2005-2010). Since samples from these wells are only monitored for nitrate-nitrogen, MEP staff used average alkalinity concentrations to estimate ammonium-nitrogen concentrations and used to the sum of nitrate-nitrogen and ammonium-nitrogen to estimate total nitrogen. Only wells with average alkalinity concentrations greater than 100 mg/l CaCO3 were used to help determine a nitrogen load from the site.

It is acknowledged that this approach for estimating a nitrogen load from the landfill area includes a number of assumptions. A detailed assessment of all the available data is beyond the scope of the MEP, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP to include conservatism in nitrogen loading estimates when uncertainty exists in the data. A more refined evaluation and assessment of the established monitoring well network, including, at a minimum, analysis of total nitrogen concentrations, would help to refine this assessment and future management options.

Nitrogen Loading Input Factors: Dennis Landfill

A portion of the capped Town of Dennis landfill is primarily located with the Bass River Mid GT10 D subwatershed (subwatershed #30). In order to assess the nitrogen load coming from the Dennis Landfill, MEP staff obtained groundwater monitoring data from MassDEP (Mark Dakers, SERO, personal communication, 9/15/10). This data consists of contaminant concentrations from groundwater samples collected from 14 wells semi-annually between March 2005 and March 2010 (*e.g.*, 10 sampling runs).

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

After determining the estimated nitrogen concentrations in the 14 wells regularly monitored in the Dennis landfill monitoring network, MEP staff reviewed the location of the regularly monitored wells and the well construction details including the screen depths shown in the well logs. This review found that there are only a few wells that are screened at appropriate depths to accurately measure the impact of the landfill. This review also found that the predominant groundwater flow direction from the landfill is toward the south-southeast away from the Bass River and toward Swan Pond. However, review of some of the groundwater monitoring data from westernmost wells suggests that flowpaths from the landfill arc over a wide range and impacts from the landfill are also measured in flowpaths headed toward Bass River.

Given some of the uncertainties with the available data, MEP reviewed the alkalinity concentrations in the downgradient monitoring wells and, as with the Yarmouth landfill assessment, selected those wells with average alkalinity concentrations greater than 100 mg/l CaCO3 and averaged the estimated nitrogen concentrations for the three wells that met this criterion. Based on this approach, the average total nitrogen concentration from the Dennis Landfill is 5.64 ppm. Using this concentration and the standard MEP recharge rate with the 14 acres of capped solid waste that exist within the watershed results in an annual nitrogen load of 220 kg from the Dennis landfill.

It is acknowledged that this approach for estimating a nitrogen load from the Dennis landfill includes a number of assumptions and is likely conservative. A detailed assessment of all the available data is beyond the scope of the MEP, but staff balanced reasonable estimates of the various factors based on the general MEP guidance from MassDEP to include conservatism in nitrogen loading estimates when uncertainty exists in the data. A more refined evaluation and assessment of the established monitoring well network, including, at a minimum, analysis of total nitrogen concentrations, would help to refine this assessment and future management options.

Nitrogen Loading Input Factors: Other

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

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

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information is linked to the parcel coverages, parcels are assigned to various watersheds based initially on whether at least 50% or more of the land area of each parcel is located within a respective subwatershed. Following the assigning of boundary parcels, all large parcels are examined individually and are split (as appropriate) in order to obtain less than a 2% difference between the total land area of each subwatershed and the sum of the area of the parcels within each subwatershed. The resulting "parcelized" watersheds to Bass River are shown in Figure IV-6.

Table IV-2. Primary Nitrogen Loading Factors used in the Bass River MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Yarmouth, Dennis, and Brewster-specific data.

Nitrogen Concentrations:	**mg/l**
Road Run-off	1.5
Roof Run-off	0.75
Natural Area Recharge	0.072
Direct Precipitation on Embayments and Ponds	1.09
Wastewater Coefficient	23.63
Town of Yarmouth Septage Treatment Effluent Discharge as Spray Irrigation at Bayberry Hills	
Average effluent Flow (million gallons per year)[2]	11.6
Average Total Nitrogen load (kg/yr)[1]	375
Town of Yarmouth Landfill Area	
Area of capped solid waste (acres)	58
Estimated TN concentration (mg/l)	10
Estimated Total Nitrogen Load (kg/yr)	1,575
Fertilizers:	
Average Residential Lawn Size (sq ft)[2]	5,000
Residential Watershed Nitrogen Rate (lbs/lawn)[2]	1.08
Cranberry Bogs nitrogen application (lbs/ac)	31
Cranberry Bogs nitrogen attenuation	34%
Nitrogen Fertilizer Rate for golf courses, determined from site-specific information; other areas assumed to utilize residential application rate	

Recharge Rates:	**in/yr**
Impervious Surfaces	40
Natural and Lawn Areas	27.25
Water Use/Wastewater:	
Existing developed single-family residential parcels wo/water accounts and buildout single-family residential parcels:	180 gpd[3]
Existing developed other residential parcels wo/water accounts and buildout other residential parcels:	352 gpd[4]
Existing developed parcels w/water accounts:	Measured annual water use
Commercial and Industrial Buildings without/WU and buildout additions[5]	
Commercial	
Wastewater flow (gpd/1,000 ft2 of building):	74
Building coverage:	15%
Industrial	
Wastewater flow (gpd/1,000 ft2 of building):	21
Building coverage:	10%
Average Single Family Residence Building Size from watershed data (sq ft)	1,450

Notes:
1) Averages for the septage effluent based on monitoring data from 2004 to 2007
2) Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.
3) Based on average flow in all single-family residences in the watershed
4) Based on average flow in all residences that are not single-family residences in the watershed
5) based on existing water use and water use for similarly classified properties throughout the watershed

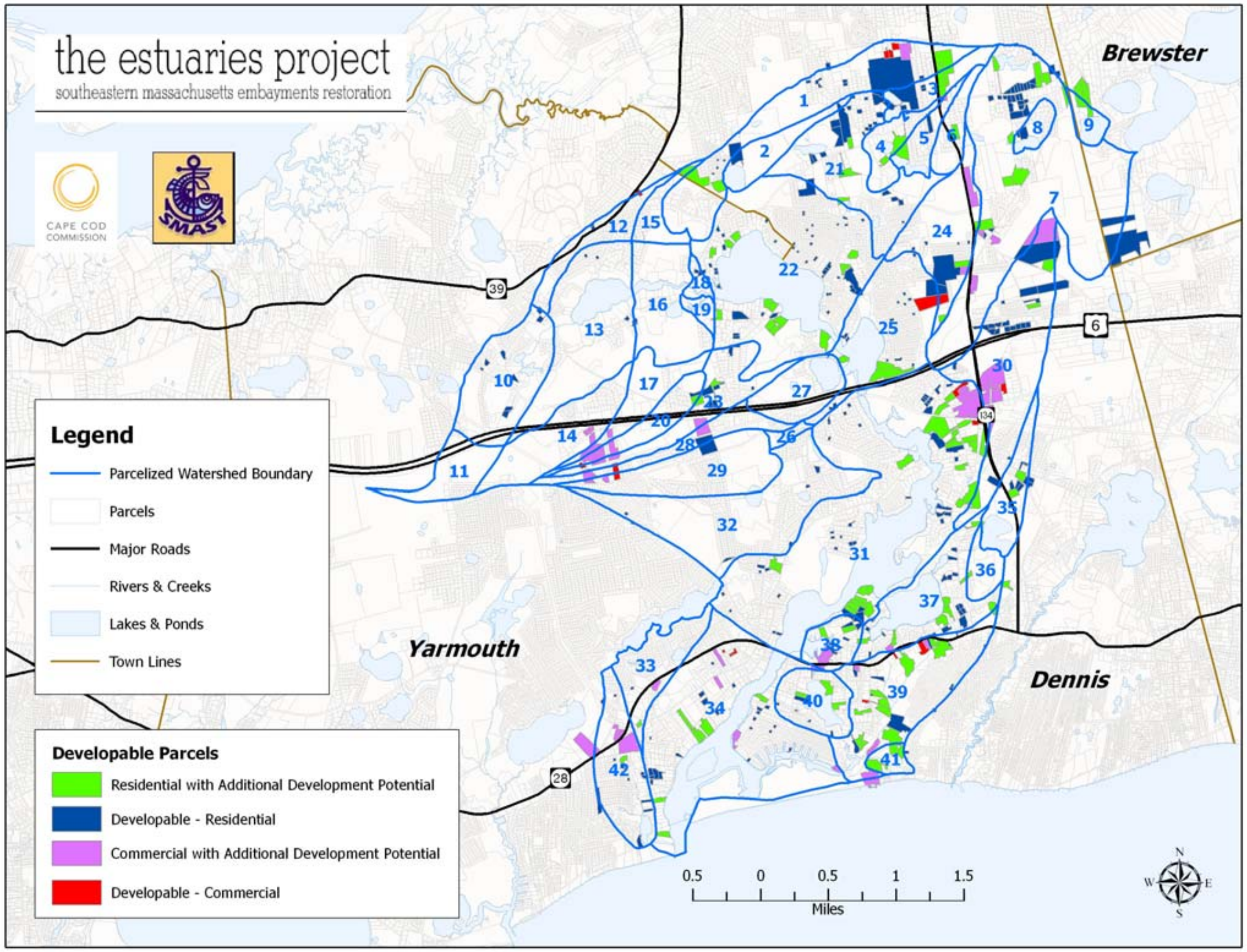


Figure IV-6. Parcels, Parcelized Watersheds, and Developable Parcels in the Bass River watersheds. Parcels colored green and purple are developed parcels (residential and commercial, respectively) with additional development potential based on current zoning, while parcel colored blue and red are corresponding undeveloped parcels classified as developable by the town assessor. The parcelized watersheds are drawn to minimize the division of properties for management purposes while achieving a match of area with the modeled watersheds of 2% or less. Developable parcels are based on town assessor classifications and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. All buildout results were reviewed with town staff.

The review of individual parcels straddling watershed boundaries includes corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Each of the towns provided GIS coverages of building footprints for the roof area calculations; Dennis and Yarmouth coverages are from 2009, while Brewster's coverage is from 2005. Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) is also assigned at this stage. It should be noted that small shifts in nitrogen loading due to the above assignment procedure generally have a negligible effect on the total nitrogen loading to the Bass River estuary. The assignment effort is undertaken to better define sub-estuary loads and enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

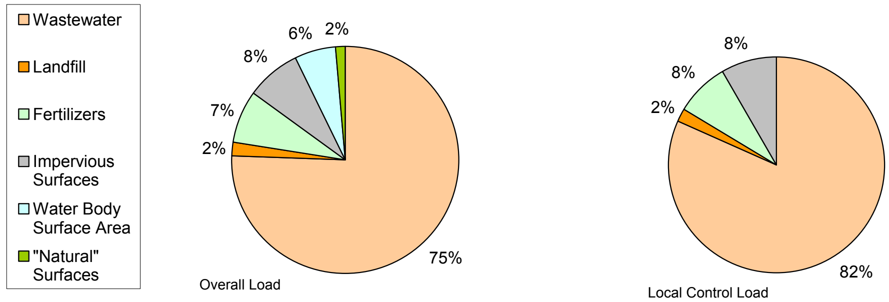
Following the assignment of all parcels, all relevant nitrogen loading data is assigned to each subwatershed. This step includes summarizing water use, parcel area, frequency, private wells, and road area. Individual sub-watershed information is then integrated to create the Bass River Watershed Nitrogen Loading module with summaries for each of the individual 42 subwatersheds. The subwatersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Bass River study area, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), fertilizers (including contributions from cranberry bogs and golf courses), impervious surfaces, direct atmospheric deposition to water surfaces, the landfills, and recharge within natural areas (Table IV-3). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-7). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model.

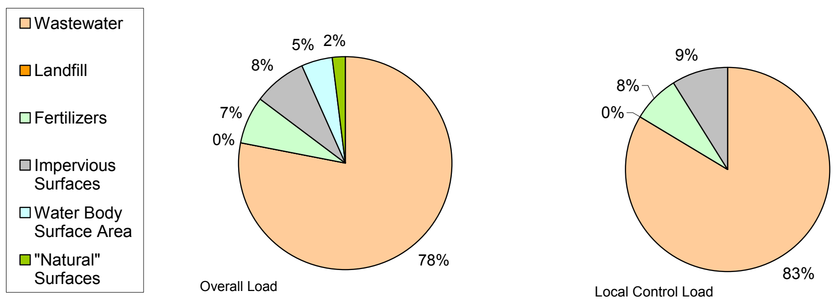
One of these attenuation adjustments occurs in the freshwater ponds. Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond is used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment is based on the percentage of discharging shoreline bordering each downgradient sub-watershed. In the Bass River study area, this occurs for ponds completely within the watershed (*e.g.*, Flax Pond) and the ponds located along the outer boundary of the River watershed (*e.g.*, Pine Pond). At Flax Pond, for example, the pond has a downgradient shoreline of 1,553 feet; 51% of that shoreline discharges into the Follins Pond LT10 subwatershed (watershed 22 in Figure IV-1) and 49% discharges into the North Dennis Wells subwatershed (watershed 1 in Figure IV-1). This breakdown of the discharge from Flax Pond means that 51% of the attenuated nitrogen load that leaves the pond reaches Follins Pond and the remainder is captured by the North Dennis wellfield. Similar pond-specific calculations were completed wherever pond flows and nitrogen loads were divided among a number of downgradient receiving subwatersheds.

Table IV-3. Bass River Watershed Nitrogen Loads. Attenuated N loads are measured & assigned attenuation rates for upgradient streams and freshwater ponds. Stream attenuation is based on measured loads (Section IV.2), pond attenuation is assigned a standard MEP nitrogen attenuation of 50% or as possible is based on water quality data from Cape Cod PALS. All attenuations in each sub-watersheds may not be shown, but cumulative results are reflected in attenuated nitrogen loading totals. All loads are kg N yr[1].

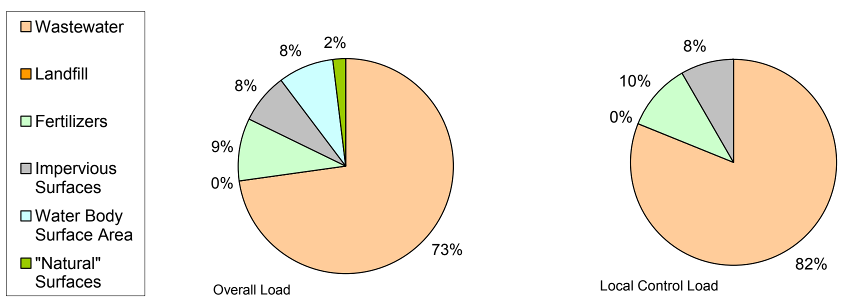
		Bass River N Loads by Input (kg/y):							% of Pond Outflow	***Present N Loads***			***Buildout N Loads***		
Name	Watershed ID#	Wastewater	Landfill	Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Bass River SYSTEM TOTAL	33,34,42 + School St Marsh + Bass R Mid	72512	1795	7293	7292	5594	1453	8247		95,939		84,068	104,186		91,047
Bass River Lower subtotal	33,34,42 + School St Marsh	23402	0	1651	2391	1634	233	1014		29,311		22,072	30,326		22,581
Bass R Lower Estuary Surface						1093				1093		1093	1093		1093
Run Pond Estuary Surface						81				81		81	81		81
School Street Marsh TOTAL		3995		380	462	223	74	534		5,134		4,419	5,668		4,920
School St Marsh Estuary Surface						52				52		52	52		52
cle Stephans Pond Estuary Surface						30				30		30	30		30
Bass River Mid TOTAL	29,30,31,32,38 + Grand Cove + Kelleys Bay	49109	1795	5642	4902	3960	1220	7233		66,628		61,996	73,861		68,466
Bass River Mid subtotal	29,30,31,32,38 + Grand Cove	23317	220	2310	2186	2036	456	3104		30,524		29,000	33,628		31,946
Bass R Mid Estuary Surface						1342				1342		1342	1342		1342
Horsefoot Cove Estuary Surface						60				60		60	60		60
Grand Cove TOTAL	37 + Fresh Pond	2418		147	193	502	56	232		3,315		3,053	3,547		3,250
Grand Cove Estuary Surface						391				391		391	391		391
Fresh Pond Gauge TOTAL		442		29	43	110	25	86		649	4%	387	735	4%	438
Kelleys Bay TOTAL	24,25,26,29 + Dinah's Pond + Follins Pond	25793	1575	3332	2715	1924	764	4129		36,104		32,996	40,232		36,520
Kelleys Bay subtotal	24,25,26,29 + Dinah's Pond	7339	0	676	776	433	182	1133		9,407		9,326	10,541		10,429
Kelleys Bay Estuary Surface						284				284		284	284		284
Dinah's Pond TOTAL	27,28	1300		75	177	113	32	71		1,696		1,696	1,767		1,767
Dinah's Pond Estuary Surface						113				113		113	113		113
Follins Pond TOTAL	2,3,5,6,18,19, 21,22,23 + Mill Stream	18453	1575	2656	1939	1491	582	2995		26,696		23,670	29,692		26,091
Follins Pond subtotal	2,3,5,6,18,19, 21,22,23	10185	0	1304	1059	1161	277	1545		13,987		13,424	15,532		14,853
Muddy Creek Estuary Surface						9				9		9	9		9
Weir Stream Estuary Surface						11				11		11	11		11
Follins Pond Estuary Surface						950				950		950	950		950
Mill Stream TOTAL	15,16,17 + Wells + Mill Pond	8269	1575	1352	880	329	305	1450		12,710		10,246	14,160		11,239
Mill Stream subtotal	15,16,17 + Wells	2533	0	499	251	76	131	540		3,490		3,453	4,031		3,993
Mill Stream Estuary Surface						51				51		51	51		51
Mill Pond TOTAL	12,13,14 + Weir Rd	5736	1575	852	629	253	174	910		9,220		6,793	10,129		7,245
Mill Pond Estuary Surface						253				253		253	253		253
Hamblin Brk Gauge TOTAL	10,11	1804	1159	516	193	0	61	704		3,733	65%	1,307	4,437	65%	1,553



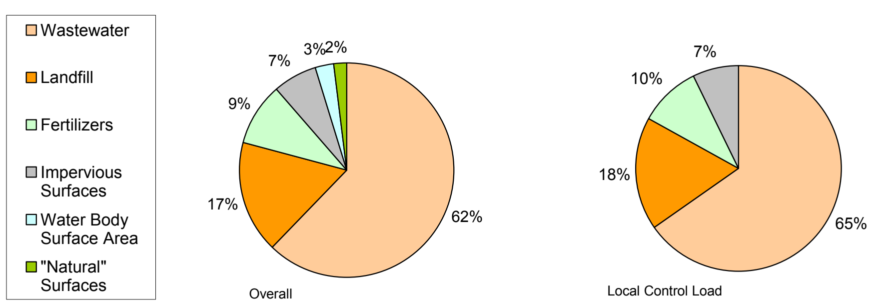
A. Whole Bass River System



B. Kelleys Bay System



C. Follins Pond System



D. Mill Pond System

Figure IV-7 . Land use-specific unattenuated nitrogen loads (by percent) to the a) whole Bass River watershed, b) Kelleys Bay subwatershed, c) Follins Pond subwatershed, and d) Mill Pond subwatershed. “Overall Load” is the total nitrogen input within the watershed, including from natural surfaces plus atmospheric deposition, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control.

Freshwater Pond Nitrogen Loads

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

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

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

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

Within the Bass River watershed, there are seven freshwater ponds with delineated watersheds: Flax, Grassy, Bakers, Pine, Fresh, Kelleys, and a combined watershed for North and South Simmons ponds. In addition, there is also Long Pond in Yarmouth, which is shared with the Parkers River estuary watershed. Except for Pine and Long, all the other ponds are located within the Town of Dennis; Pine is located within the Town of Brewster. None of these ponds, except Long, have available pond-wide bathymetric data (Eichner, et al., 2003). As such, a reasonable pond-specific nitrogen attenuation rate cannot be developed for these ponds, even though Flax, North Simmons, Bakers, and Pine have been regularly sampled via the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots and locally-supported volunteer pond sampling programs.

The PALS Snapshots are regional volunteer pond sampling supported for the last nine years by SMAST and the Cape Cod Commission, with free laboratory services provided by the Coastal Systems Program Laboratory at SMAST. In addition, both Dennis and Brewster have created local volunteer pond sampling programs that included regular sampling throughout multiple summers. All sampling runs in Bass River watershed ponds have generally followed PALS protocols (Eichner *et al*., 2003), which means that sampling has included field collection of temperature and dissolved oxygen profiles and sampling has generally occurred at standardized depths that provide some evaluation of potential sediment nutrient regeneration. PALS water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH.

Flax, North Simmons, Bakers, and Pine have all been sampled since the original PALS Snapshot in 2001. In addition, all of them have extensive supplemental datasets: Pine has a total of 48 sampling runs (Eichner, 2009a), while Flax, North Simmons, and Bakers have 39, 43, and 42 sampling runs, respectively (Eichner, 2009b). Unfortunately, given the lack of bathymetry, data is insufficient to assign pond-specific nitrogen attenuation factors to any of these freshwater ponds. The standard MEP freshwater pond 50% nitrogen attenuation rate is incorporated into the Bass River watershed nitrogen loading module of the linked watershed-estuary model for all these ponds.

Long Pond was previously reviewed in the Parkers River MEP assessment (Howes, et al, in press) since it is shared by both watersheds. Long Pond is 24.4 hectares (60.5 acres) with a maximum depth of 9.2 m. Given the area of its watershed and its volume based on it bathymetric map, it has a turnover time of 0.32 years. Long has had samples collected by staff from the Yarmouth Division of Natural Resources all eight PALS Snapshots between 2001 and 2008. Based on these results, the average surface TN concentration in Long Pond is 0.5 ppm (n=11). Using this information and the estimated watershed nitrogen load, MEP staff determined that 89% of the watershed nitrogen load into Long Pond is attenuated or removed by natural processes.

The amount of nitrogen discharging into Bass River from Long Pond is complicated by its surface water discharge (through Forest Road Brook) to Seine Pond in the Parkers River estuary watershed. During the water quality collection period for the MEP assessment of Parker River (September 2003 and September 2004), average flow through Forest Road Brook was measured as 353 m3/d (Howes, et al, in press, Parkers River MEP Report). Following the monitoring period and in fall 2008, Town of Yarmouth Department of Natural Resources staff modified the Brook and increased the flow to an estimated 1995 m3/d. This change means that under current conditions, but after the Bass River monitoring period, more of the nitrogen in Long Pond is discharged into Parkers River and less is discharged into Bass River.

Buildout

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

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

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

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

Following the completion of the initial buildout assessment for the Bass River watersheds, MEP staff reviewed the results with town officials. MEP staff reviewed the initial Yarmouth buildout results with Terry Sylvia and Karen Greene of the Town of Yarmouth Community Development Department in June 2010. The initial Dennis buildout results were reviewed with Dan Fortier, Town of Dennis Town Planner in June 2010. And the initial Brewster results were reviewed with Sue Leven, Town of Brewster Town Planner also in June 2010. Suggested changes from all reviews were incorporated into the final buildout for Bass River.

All the parcels with additional buildout potential within the Bass River watershed are shown in Figure IV-4. Each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for wastewater and impervious surfaces. Residential additions also include lawn fertilizer nitrogen additions. All wastewater loads are assumed to come from on-site septic systems. Cumulative unattenuated buildout loads are indicated in a separate column in Table IV-3. Buildout additions within the Bass River watersheds will increase the unattenuated loading rate by 9%.

The buildout analysis also contains a change from existing conditions in the conceptualization of the nitrogen loads from Long Pond. As discussed in the Freshwater Ponds section above, this change is based on modifications to the Forest Road Brook connections to Seine Pond accomplished by the Town of Yarmouth Department of Natural Resources. Existing conditions showed that an estimated annual load of 990 kg came from Long Pond to Bass River during the Bass River MEP water quality measurement period. Based on the increased flow in the channel between Long Pond and Seine Pond, the estimated annual attenuated nitrogen load from Long Pond into the Bass River watershed has decreased to 525 kg. This change is incorporated into the Bass River buildout scenario.

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 relative to the tidal flushing and nitrogen cycling within the embayment basins. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Bass River System being investigated under this nutrient threshold analysis 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 from source to estuarine waters is through groundwater flow in sandy outwash aquifers (such as the developed regions of the Bass River watershed). The lack of nitrogen attenuation in these aquifer systems results from the lack of biogeochemical conditions needed for supporting nitrogen sorption and denitrification. However, in most watersheds in southeastern Massachusetts, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the watershed for the Bass River embayment system, a portion of the freshwater flow and transported nitrogen passes through several surface water systems (e.g. Hamblin Brook and a small creek discharging to Grand Cove from the up-gradient Fresh Pond) prior to entering the estuary, producing the opportunity for significant nitrogen attenuation.

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

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach in the Bass River embayment system. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the estuary in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted in the 2 major surface water flow systems in the Bass River watershed, 1) Hamblin Brook discharging to the brackish Mill Pond at the head of the Bass River system and 2) Fresh Pond Brook, a small stream discharging to Grand Cove which is a tributary sub-basin of the Bass River. Regarding Weir Creek, the hydraulic connection discharging brackish water from Mill Pond to Follins Pond, a gage was deployed in the culvert through which the “creek” passes in order to estimate the freshwater portion of the flow and confirm the sub-watershed delineation to Mill Pond. Nitrogen loading and attenuation estimation was not attempted at this location because Mill Pond functionally serves as a brackish extension of the Bass River Estuarine System (Figure IV-8). The Creek from Fresh Pond is less significant than the other two surface water features mentioned and periodically went dry during low flow periods of the hydrologic year.

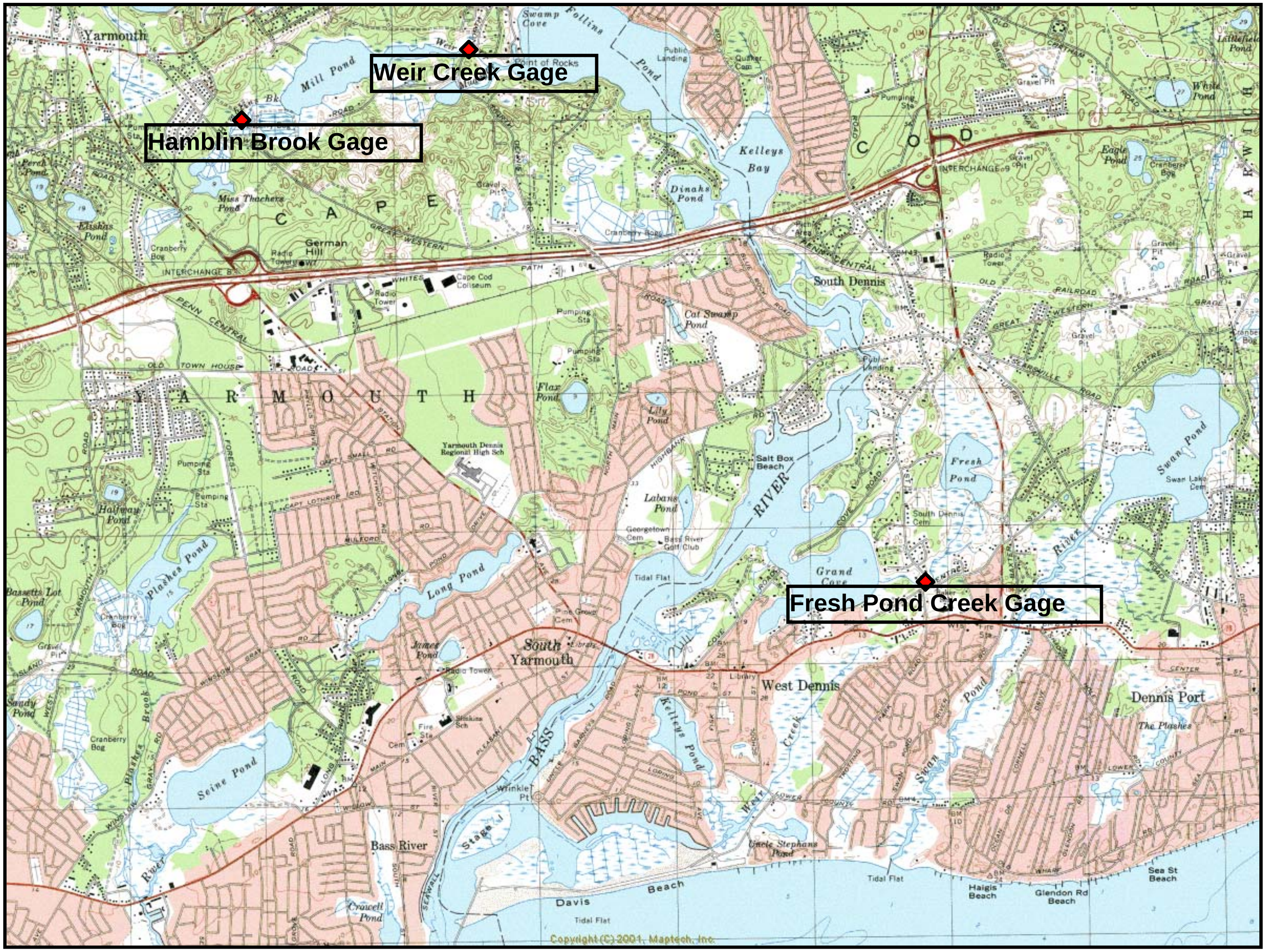


Figure IV-8. Location of Stream gages (red diamonds) in the Bass River embayment system. The Weir Creek gage was established to measure nitrogen loads from the brackish sub-embayment of Mill Pond to Follins Pond.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater streams discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area upgradient from the various gaging sites. Flow and nitrogen load were measured at the gages in each freshwater stream site for between 16 and 22 months of record depending on the stream gaging location (Figures IV-9a,b to IV-10a,b). During each study period, velocity profiles were completed on each surface water inflow every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

Determination of stream flow at each gage was calculated and based on the measured values obtained for stream cross sectional area and velocity. Stream discharge was represented by the summation of individual discharge calculations for each stream subsection for which a cross sectional area and velocity measurement were obtained. Velocity measurements across the entire stream cross section were not averaged and then applied to the total stream cross sectional area.

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

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

where by:

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

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

Periodic measurement of flows over the entire stream gage deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gages. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given river/stream/creek/brook. These hourly stages values where then entered into the stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. In the case of tidal influence on stream stage, the diurnal low tide stage value was extracted on a day-by-day basis in order to resolve the stage value indicative of strictly freshwater flow. The lowest low tide stage values for any given day were utilized in the stage – discharge relation in order to compute daily flow as this stage value is most representative of freshwater flow. A complete annual record of stream flow (365 days) was generated for the surface water discharges flowing into the Bass River embayment system.

The annual flow record for the surface water flow at each gage was merged with the nutrient data set generated through the weekly water quality sampling performed at the gage locations to determine nitrogen loading rates to the Bass River system. Nitrogen discharge from the streams was calculated using the paired daily discharge and daily nitrogen concentration data to determine the mass flux of nitrogen through a specific gaging site. For each of the stream gage locations, weekly water samples were collected (at low tide for a tidally influenced stage) in order to determine nutrient concentrations from which nutrient load was calculated. In order to pair daily flows with daily nutrient concentrations, interpolation between weekly nutrient data points was necessary. These data are expressed as nitrogen mass per unit time (kg/d) and can be summed in order to obtain weekly, monthly, or annual nutrient load to the embayment system as appropriate. Comparing these measured nitrogen loads based on stream flow and water quality sampling to predicted loads based on the land use analysis allowed for the determination of the degree to which natural biological processes within the watershed to each gaged stream currently reduces (percent attenuation) nitrogen loading to the overall embayment systems.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Hamblin Brook to Mill Pond

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, Hamblin Brook, that discharges into the head of Mill Pond at the top of the Bass River system, does not have an up-gradient pond from which the brook discharges. Rather, this small stream appears to be groundwater fed and emanates from a wooded and somewhat boggy area up-gradient of Cheyenne Lane and Weir Road. There may historically have been a small pond up-gradient of the brook, however, recent aerial photography indicates that whatever pond may have existed in the past is now grown over. The stream outflow leaving the boggy areas between Cheyenne Lane and Weir Road travels through a small wooded area just prior to discharging directly into Mill Pond. The stream outflow from the bog/wetland and the wooded area up gradient of the gage may serve to contribute to the attenuation of nitrogen and also provides for a direct measurement of the nitrogen attenuation. 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 bog/wetland and wooded areas above the gage site and the measured annual discharge of nitrogen to Mill Pond and the Bass River relative to the gage, Figure IV-7.

At the Hamblin Brook gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the channel that carries the flows and associated nitrogen load to the Mill Pond at the head of the Bass River system. As the lower reach of Mill Pond is slightly tidally influenced, the stage record from the gage was checked to make sure there was no tidal influence in the record. To confirm that freshwater was being measured the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average salinity of the water samples taken from Hamblin Brook was determined to be 0.1 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage on Hamblin Brook was installed on July 27, 2005 and was set to operate continuously for 16 months such that a complete hydrologic year would be captured in the flow record (Table IV-4). Stage data collection continued until November 15, 2006 for a total deployment of 16 months.

Table IV-4. Comparison of water flow and nitrogen load discharged by surface waters (freshwater) to the Bass River Estuary. The "Stream" data are from the MEP stream gaging effort. Watershed data are based upon the MEP watershed modeling effort (Section IV.1) and the USGS watershed delineation (Section III).

Stream Discharge Parameter	Hamblin Brook Discharge[a] Mill Pond-Bass River	Weir Creek Discharge Follins Pond-Bass River	Creek from Fresh Pond Discharge[a] Grand Cove-Bass River	Data Source
Total Days of Record	365[b]	365[c]	365[d]	(1)
Flow Characteristics				
Stream Average Discharge (m3/day)	2,993	16,098	1,107	(1)
Contributing Area Average Discharge (m3/day)	3,195	15178	1246	(2)
Discharge Stream 2004-05 vs. Long-term Discharge	-6.75%	5.71%	-12.56%	
Nitrogen Characteristics				
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.532	0.106	0.124	(1)
Stream Average Total N Concentration (mg N/L)	1.181	1.041	0.955	(1)
Nitrate + Nitrite as Percent of Total N (%)	45%	10%	13%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	3.53	--	1.06	(1)
TN Average Contributing UN-attenuated Load (kg/day)	10.22	--	1.77	(3)
Attenuation of Nitrogen in Pond/Stream (%)	65%	--	40%	(4)

(a) Flow and N load to streams discharging to Bass River, includes apportionments of Pond contributing areas.
(b) September 1, 2005 to August 31, 2006.
(c) October 2, 2003 to October 1, 2004.
(d) October 16, 2003 to October 15, 2004
(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the streams to Bass River;
and the annual recharge rate.
(3) As in footnote (2), with the addition of pond and stream conservative attentuation rates.
(4) Calculated based upon the measured TN discharge from the rivers vs. the unattenuated watershed load.

Table IV-5. Summary of annual volumetric discharge and nitrogen load from Hamblin Brook and the Creek from Fresh Pond to Grand Cove (based upon the data presented in Figures IV-9 through IV-10 and Table IV-4. Only flow was assessed in Weir Creek, the hydraulic connection between Mill Pond, the uppermost estuarine reach of Bass River and Follins Pond sub-basin in the estuary.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Bass River (Mill Pond) Hamblin Brook to Mill Pond MEP	**September 1, 2005 to August 31, 2006**	**1,092,485**	**581**	**1290**
Bass River (Mill Pond) Hamblin Brook to Mill Pond CCC	Based on Watershed Area and Recharge	**1,166,175**	--	--
Bass River (Follins Pond) Weir Creek to Follins Pond MEP	**October 2, 2003 to October 1, 2004**	**5,875,722**	--	--
Bass River (Follins Pond) Weir Creek to Follins Pond CCC	Based on Watershed Area and Recharge	**5,539,970**	--	--
Bass River (Grand Cove) Fresh Pond Creek to Grand Cove MEP	**October 16, 2003 to October 15, 2004**	**404,037**	**50**	**386**
Bass River (Grand Cove) Fresh Pond Creek to Grand Cove CCC	Based on Watershed Area and Recharge	**454,790**	--	--

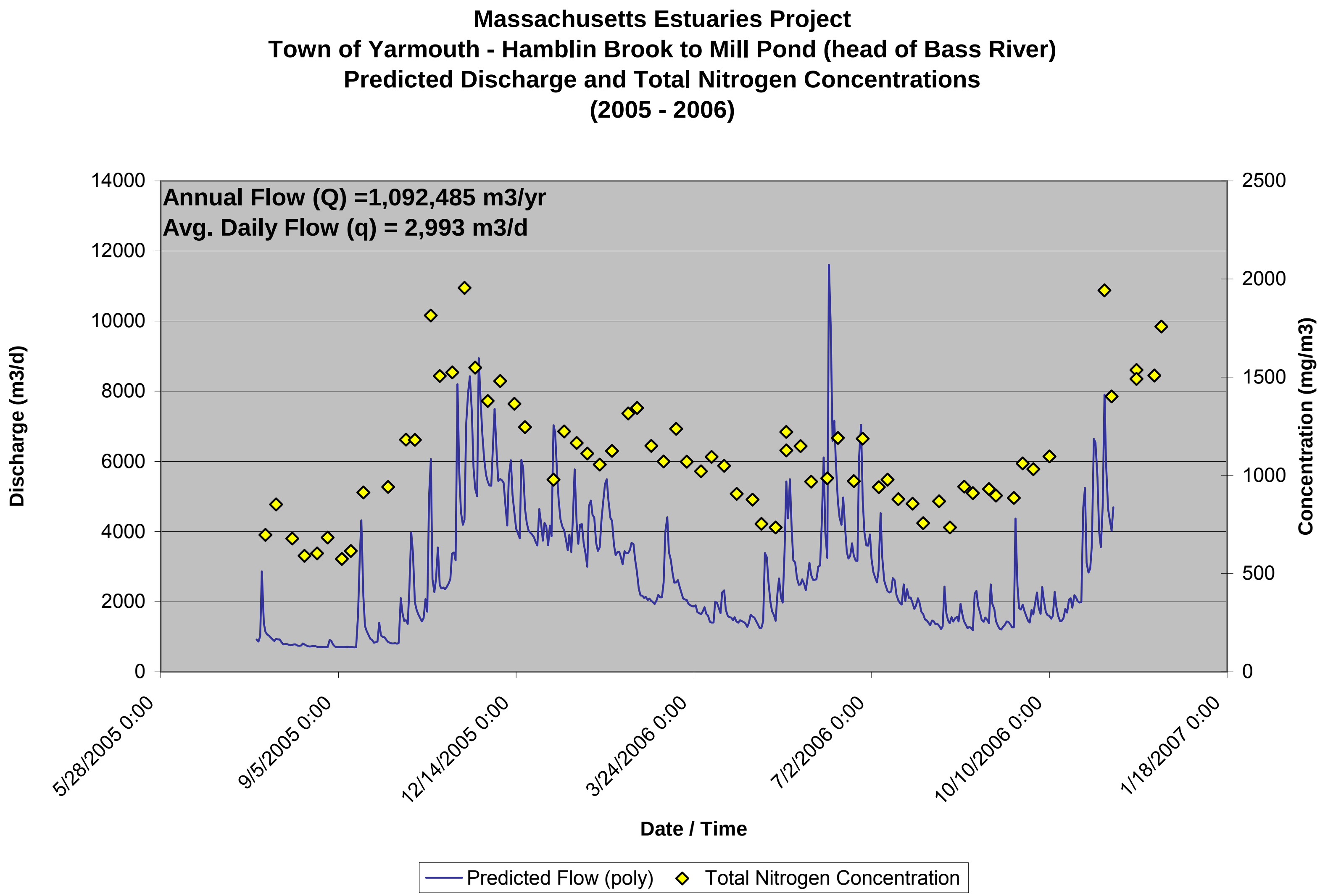


Figure IV-9a. Hamblin Brook discharge (solid blue line) and total nitrogen (yellow symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Mill Pond and Bass River (Table IV-4).

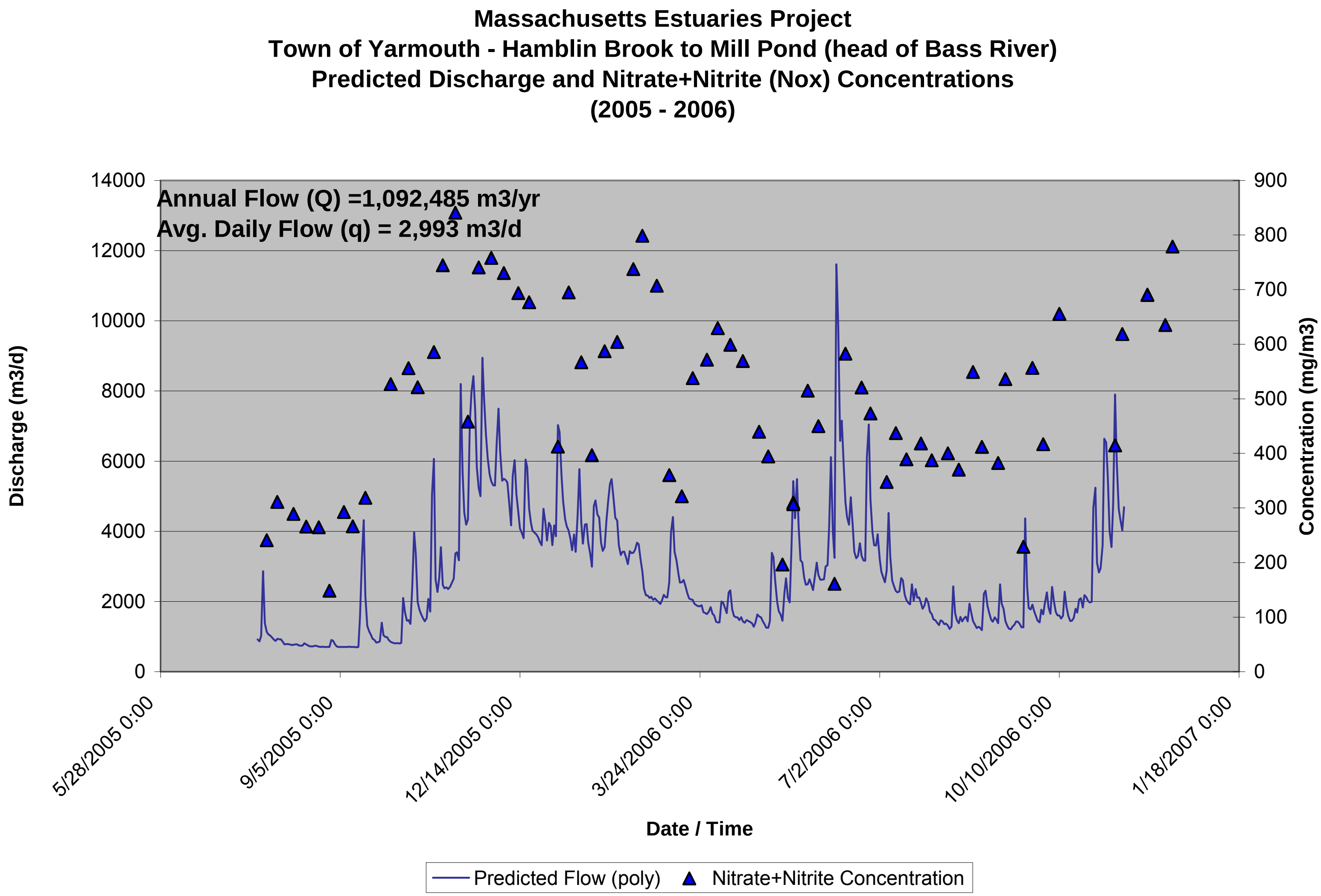


Figure IV-9b. Hamblin Brook discharge (solid blue line) and nitrate+nitrite (blue triangle) concentration for determination of annual volumetric discharge and nitrogen load from the upper watershed to Mill Pond and the Bass River (Table IV-4).

Surface freshwater flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Hamblin Brook site based upon these flow measurements and measured water levels at the gage site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the head of Mill Pond at the top of the Bass River system and reflective of the biological processes occurring in the stream channel and small bog/wetland and wooded area contributing to nitrogen attenuation (Figure IV-9a and 9b and Table IV-4 and IV-5). In addition, a water balance was constructed based upon the U.S. Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gage site.

The annual freshwater flow record for Hamblin Brook measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from Hamblin Brook was 6.75% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September 2005 and ending in August 2006 (low flow to low flow) was 2,993 m^3/day compared to the long term average flows determined by the USGS modeling effort (3,195 m^3/day). The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Hamblin Brook discharging from the sub-watershed indicate that the Brook is capturing the upgradient recharge (and loads) accurately.

Total nitrogen concentrations within the Hamblin Brook outflow were moderate to high, 1.181 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 3.53 kg/day and a measured total annual TN load of 1,290 kg/yr. In Hamblin Brook, nitrate made up slightly less than half of the total nitrogen pool (45%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the bog/wetland areas upgradient of the gage was partially taken up by plants within the bog/wetland or stream ecosystems. In addition, the level of remaining nitrate in the stream discharge suggests the possibility for additional uptake by freshwater systems might be accomplished up-gradient of the gage and prior to discharge to Mill Pond. Opportunities for enhancing nitrogen attenuation could be considered within the bog/wetland area or along the freshwater reach of Hamblin Brook.

From the measured nitrogen load discharged by Hamblin Brook to Mill Pond, upgradient of Follins Pond and the Bass River 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 Mill Pond and the Bass River estuary. Based upon lower total nitrogen load (1,290 kg yr^{-1}) discharged from the freshwater Hamblin Brook compared to that added by the various land-uses to the associated watershed (3,733 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 65% (i.e. 65% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is expected given the nature of the upgradient bog/wetland/wooded areas capable of attenuating nitrogen. The directly measured nitrogen load from the brook was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Weir Creek Discharge to Follins Pond (head of Bass River)

Weir Creek is a brackish stream discharging to Follins Pond in upper portion of the Bass River system. However, Weir Creek is not a traditional stream in that it is really a hydraulic connection between Mill Pond (the most upgradient estuarine reach of the Bass River system) and Follins Pond which is less brackish and has higher salinity. The Weir Creek gage site was located at the North Dennis Road crossing and was immediately down gradient of the culvert passing under N. Dennis Road. This gage was deployed as a way to confirm the sub-watershed to Mill Pond and get an estimate of "fresh" surfacewater flow to Follins Pond and informing the salinity calibration of the hydrodynamic model. Weir Creek captures groundwater discharged from its associated watershed and is also the main channel for water flowing out of Mill Pond (head of Bass River) to Follins Pond but also passes tidal water into Mill Pond on the flooding tide. As such, Weir Creek can not be regarded as a purely freshwater "stream". Any measure of freshwater can only be achieved by making salinity adjustment to the measured flow.

The freshwater flow carried by Weir Creek to the estuarine waters of Follins Pond and the Bass River was determined using a continuously recording vented calibrated water level gage. As this surface water system was clearly tidally influenced, the creek discharge was checked to confirm the extent of tidal influence and whether freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on weekly water quality samples collected from the gage site. Average measured sample salinity was found to be 12.5 ppt, clearly tidally influenced. As such, a salinity adjustment was made to the flows using the average sample salinity and the boundary salinity of the downgradient water quality monitoring station in Follins Pond. While the gage location was less than ideal for making measurements of freshwater flow, there was no other location that could be utilized for measuring freshwater flows out of Mill pond. That being the case, the Weir Creek gage location was deemed acceptable for obtaining an estimate of freshwater flow measurements, with the caveat that the site is clearly tidally influenced. Calibration of the gage was checked monthly. The gage was installed on September 11, 2003 and was set to operate continuously for 16 months such that at least one summer season would be captured in the flow record. Stage data collection continued until July 27, 2005 for a total deployment of 22 months.

Stream flow (volumetric discharge) was measured every 4 to 6 weeks or as stream flow conditions allowed as there were times when clearly the stages and flows were not indicative of low tide conditions. Flow measurements were made using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gage site based upon these flow measurements and the measured water levels at the gage site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. In addition, a water balance was constructed based upon the U.S. Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the gage site.

The annual freshwater flow record for Weir Creek measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from Weir Creek was 5.7% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning October 2, 2003 and ending in October 1, 2004 (low flow to low flow) was 16,098 m^3/day compared to the long term average flows determined by the USGS modeling effort (15,178 m^3/day). The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in Weir Creek discharging from the sub-watershed to Mill Pond indicated that the Creek is capturing the upgradient recharge accurately. The directly measured freshwater flow from Weir Creek was used in the Linked Watershed-Embayment hydrodynamic and water quality modeling described in Section V and VI.

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Creek Discharge from Fresh Pond to Grand Cove (sub-basin of Bass River)

The creek from Fresh Pond is a small freshwater feature discharging to the tributary sub-basin of Grand Cove situated in the middle portion of the Bass River system. Unlike many of the freshwater ponds on Cape Cod, Fresh Pond has “stream” outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, Fresh Pond Creek, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the sub-watershed nitrogen load to the estuary and the level of nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wooded area between the pond and Grand Cove as well as the streambed associated with the creek. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to Fresh Pond Creek above the gage site and the measured annual discharge of nitrogen to the Grand Cove tributary sub-basin as water flowed past the gage location.

The Fresh Pond Creek gage site was located at the Main Street crossing where it bends 90 degrees and becomes Old Main Street in the Town of Dennis. The gage was located upgradient of the road crossing in a wooded area that separates Fresh Pond from Grand Cove. The creek captures groundwater discharged from its associated watershed and also serves as a conduit for water to flow from Fresh Pond to Grand Cove. As this is a very small creek, flow was intermittent throughout the deployment period with no flow during the low flow period of the hydrologic year and times during the winter when it was frozen solid or to near solid.

The freshwater flow and nitrogen load carried by the creek from Fresh Pond to the estuarine waters of Grand Cove and the Bass River system was determined using a continuously recording vented calibrated water level gage. As surface water systems can at times be tidally influenced, the creek discharge was checked to confirm freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on weekly water quality samples collected from the gage site. Measured sample salinity was found to be $\leq$0.1 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage was installed on September 11, 2003 and was set to operate continuously for 16 months such that at least one summer season would be captured in the flow record. Stage data collection continued until January 4, 2005 for a total deployment of 17 months.

Stream flow (volumetric discharge) was measured every 4 to 6 weeks or as stream flow conditions allowed using a Marsh-McBirney electromagnetic flow meter. Stream flow was intermittent, as the creek periodically went dry during the lowest flow months of the hydrologic year and at times was too frozen over in the winter to make flow measurements. A rating curve was developed for the gage site based upon these flow measurements and the measured water levels at the gage site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to the Bass River system (Figure IV-10a,b). In addition, a water balance was constructed based upon the U.S. Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the gage site.

The annual freshwater flow record for Fresh Pond Creek measured by the MEP, taking into consideration periods during the year when the brook was dry or frozen over, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The data indicate that the freshwater discharge from the creek was 12.56% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning October 16, 2003 and ending in October 15, 2004 (low flow to low flow) was 1,107 m^3/day compared to the long term average flows determined by the USGS modeling effort (1,246 m^3/day). The negligible difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in the Creek discharging from the sub-watershed to Fresh Pond indicated that the Creek is capturing the upgradient recharge (and loads) accurately. The difference is likely due to the fact that for portions of the year the creek was dry and had no measurable flow.

Total nitrogen concentrations within the Fresh Pond Creek outflow were moderate to high, 0.955 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 1.06 kg/day and a measured total annual TN load of 386 kg/yr. In the Fresh Pond Creek, nitrate made up a small fraction (13%) of the total nitrogen pool to Grand Cove (tributary sub-basin to middle Bass River), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to Fresh Pond upgradient of the gage was significantly taken up by plants within the overall Fresh Pond ecosystem. This is consistent with the fact that particulate organic nitrogen (PON) and dissolved organic nitrogen (DON) represents 6% and 84% respectively of the total nitrogen pool entering Grand Cove via Fresh Pond Creek (PON+DON = 90% of TN load). That being the case, it is not likely much more DIN from Fresh Pond could be eliminated by enhancing natural processes to reduce levels of nitrate and nitrite in Fresh Pond. If anything, managers may want to control levels of inorganic nitrogen reaching Fresh Pond from the upgradient watershed in order to try and decrease levels of organic nitrogen production in Fresh Pond and subsequent loading to Grand Cove, which in turn becomes organic nitrogen available to be recycled into inorganic forms via biogeochemical processes that naturally occur in estuarine sediments.

From the measured nitrogen load discharged by Fresh Pond Creek to Grand Cove and the Bass River 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 through Fresh Pond with eventual discharge to the middle estuarine reaches of the Bass River estuary. Based upon lower total nitrogen load (386 kg yr^{-1}) discharged from the Fresh Pond Creek compared to that added by the various land-uses to the associated watershed (649 kg yr^{-1}), the integrated attenuation in passage through Fresh Pond prior to discharge to the estuary is 40% (i.e. 40% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other surface water systems evaluated under the MEP is expected given the nature of the upgradient Fresh Pond capable of attenuating large quantities of inorganic nitrogen. The directly measured nitrogen load from the Fresh Pond Creek was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

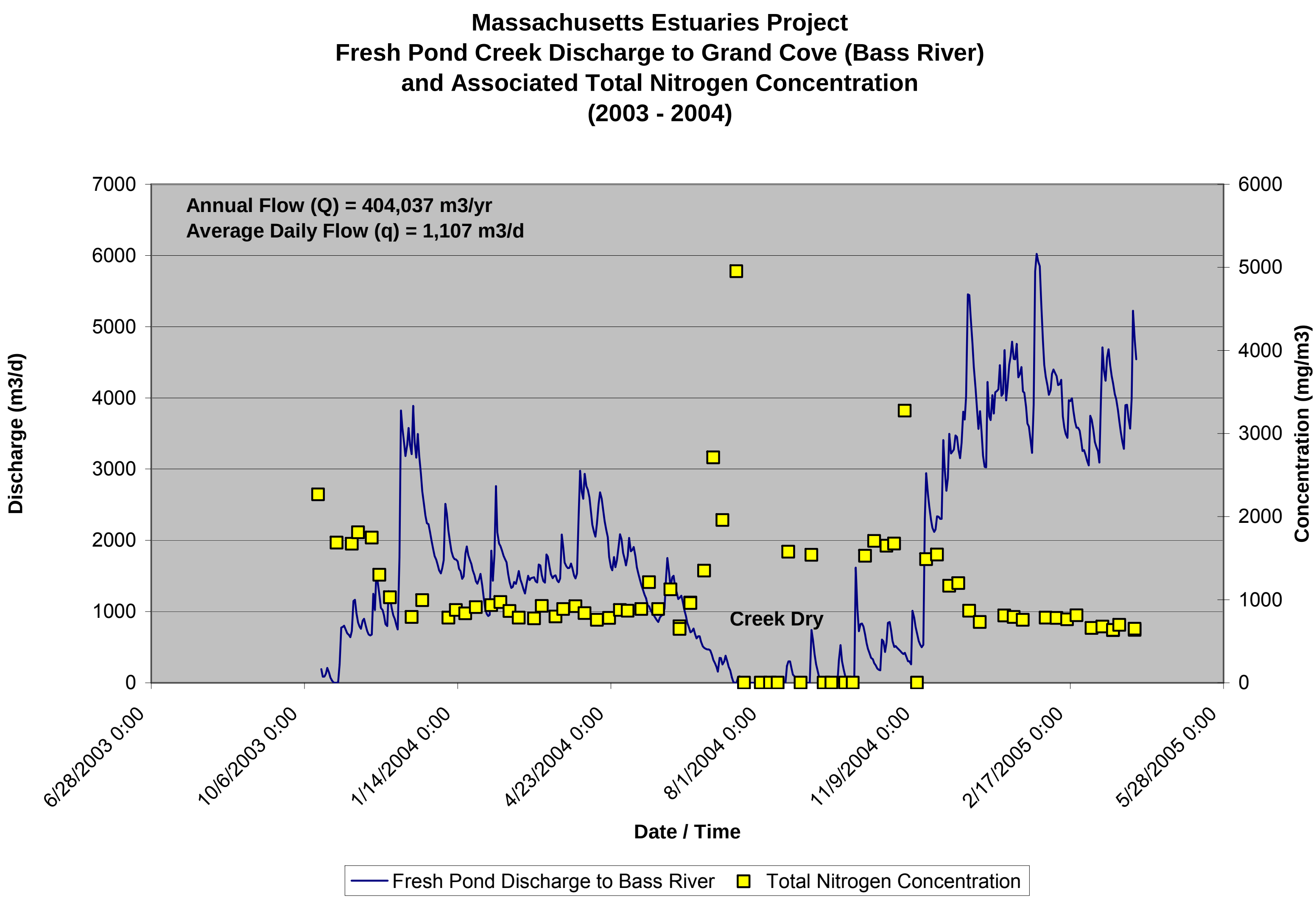


Figure IV-10a. Creek discharge from Fresh Pond (solid blue line) and total nitrogen (yellow symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of Fresh Pond discharging to Grand Cove in the Bass River (Table IV-4).

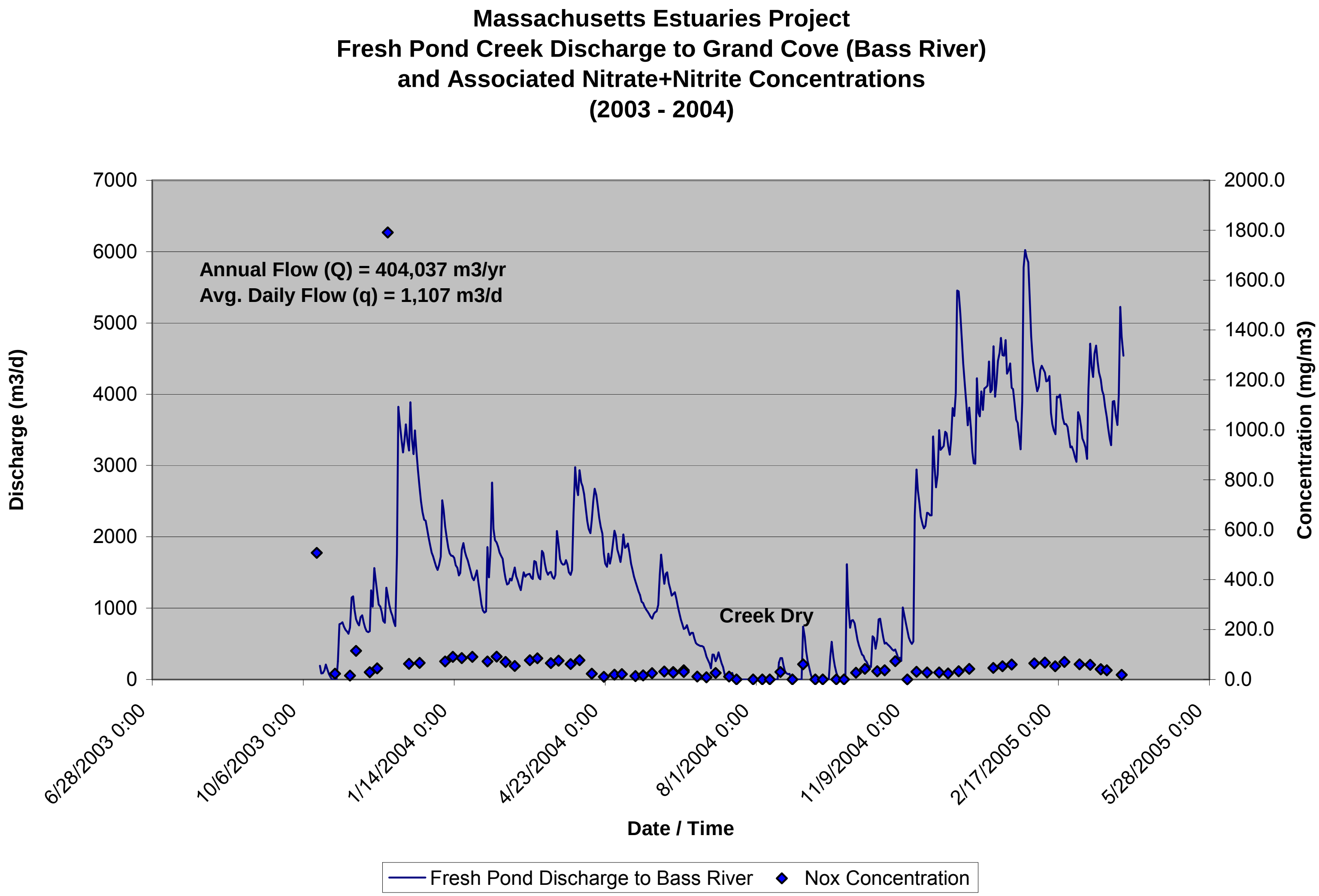


Figure IV-10b. Creek discharge from Fresh Pond (solid blue line) and nitrate+nitrite (blue triangle) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of Fresh Pond discharging to Grand Cove in the Bass River (Table IV-4).

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

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

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

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

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with salt marsh tidal creeks, the sediments can be a net sink for nitrogen even during summer (e.g. Mashapaquit Creek Salt Marsh, West Falmouth Harbor; Centerville River Salt Marsh). Embayment basins can also be net sinks for nitrogen to the extent that they support relatively oxidized surficial sediments, such as found within much of the bordering region to the nearby Lewis Bay main basin. In contrast, regions of high deposition like Hyannis Inner Harbor, which is essentially a dredged boat basin, typically support anoxic sediments with elevated rates of nitrogen release during summer months. The consequence of this deposition is that these basin sediments are unconsolidated, organic rich and sulfidic nature (MEP field observations).

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

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Bass River embayment system, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from a total of 43 sites in the Bass River system. Cores were spatially distributed throughout the entire system, inclusive of Mill Pond and other tributary sub-basins such as Grand Cove and Davis Beach, in order to capture representative nutrient flux rates. All the sediment cores for this system were collected in July-August 2005. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shoreside lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from core sites to replace the headspace water of each core prior to incubation. The number of core samples from each estuarine component (Figure IV-11a and 11b) are as follows:

Bass River Benthic Nutrient Regeneration Cores

• BSR-1	1 core	(Lower Bass River – Main Channel)
• BSR-2	1 core	(Lower Bass River – Davis Beach)
• BSR-3	1 core	(Lower Bass River – Davis Beach)
• BSR-4	1 core	(Lower Bass River – Davis Beach)
• BSR-5	1 core	(Lower Bass River – Tributary Channel)
• BSR-6	1 core	(Lower Bass River – Main Channel)
• BSR-7	1 core	(Lower Bass River – Main Channel)
• BSR-8	1 core	(Lower Bass River – Tributary Cove)
• BSR-9/10	2 cores	(Lower Bass River – Grand Cove)
• BSR-11	1 core	(Lower Bass River – Grand Cove)
• BSR-12	1 core	(Lower Bass River – Grand Cove)
• BSR-13	1 core	(Lower Bass River – Grand Cove)
• BSR-14	1 core	(Lower Bass River – Main Channel)
• BSR-15	1 core	(Lower Bass River – Main Channel)
• BSR-16	1 core	(Lower Bass River – Main Channel)
• BSR-17	1 core	(Lower Bass River – Main Channel)
• BSR-18	1 core	(Lower Bass River – Main Channel)
• BSR-19	1 core	(Lower Bass River – Main Channel)
• BSR-20	1 core	(Lower Bass River – Main Channel)
• BSR-21	1 core	(Lower Bass River – Main Channel)
• BSR-22	1 core	(Lower Bass River – Main Channel)
• BSR-23	1 core	(Lower Bass River – Tributary Cove)
• BSR-24	1 core	(Lower Bass River – Main Channel)
• BSR-25	1 core	(Upper Bass River – Follins Pond)
• BSR-26	1 core	(Upper Bass River – Follins Pond)
• BSR-27	1 core	(Upper Bass River – Follins Pond)
• BSR-28	1 core	(Upper Bass River – Follins Pond)
• BSR-29	1 core	(Upper Bass River – Follins Pond)
• BSR-30	1 core	(Upper Bass River – Follins Pond)
• BSR-31	1 core	(Upper Bass River – Follins Pond)
• BSR-32	1 core	(Upper Bass River – Follins Pond)
• BSR-33	1 core	(Upper Bass River – Kelleys Bay)
• BSR-34	1 core	(Upper Bass River – Dinah's Pond)
• BSR-35	1 core	(Upper Bass River – Dinah's Pond)
• BSR-36	1 core	(Upper Bass River – Dinah's Pond)
• BSR-37	1 core	(Upper Bass River – Kelleys Bay)
• BSR-38	1 core	(Upper Bass River – Kelleys Bay)
• BSR-39	1 core	(Upper Bass River – Follins Pond)
• BSR-40	1 core	(Upper Bass River - Follins Pond)
• BSR-41	1 core	(Upper Bass River - Mill Pond)
• BSR-42	1 core	(Upper Bass River - Mill Pond)
• BSR-43	1 core	(Upper Bass River - Mill Pond)

Figure IV-11a. Bass River embayment system sediment sampling sites (blue symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

Figure IV-11b. Bass River embayment system sediment sampling sites (blue symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

Sampling was distributed throughout the primary component basins of the Bass River Estuary and the results were used for calculating the net nitrogen regeneration rates for the water quality modeling effort.

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

Chemical analyses were performed by the Coastal Systems Analytical Facility at the School for Marine Science and Technology (SMAST) at the University of Massachusetts in New Bedford, MA. (508-910-6325 or d1white@umassd.edu). The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

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

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

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-12).

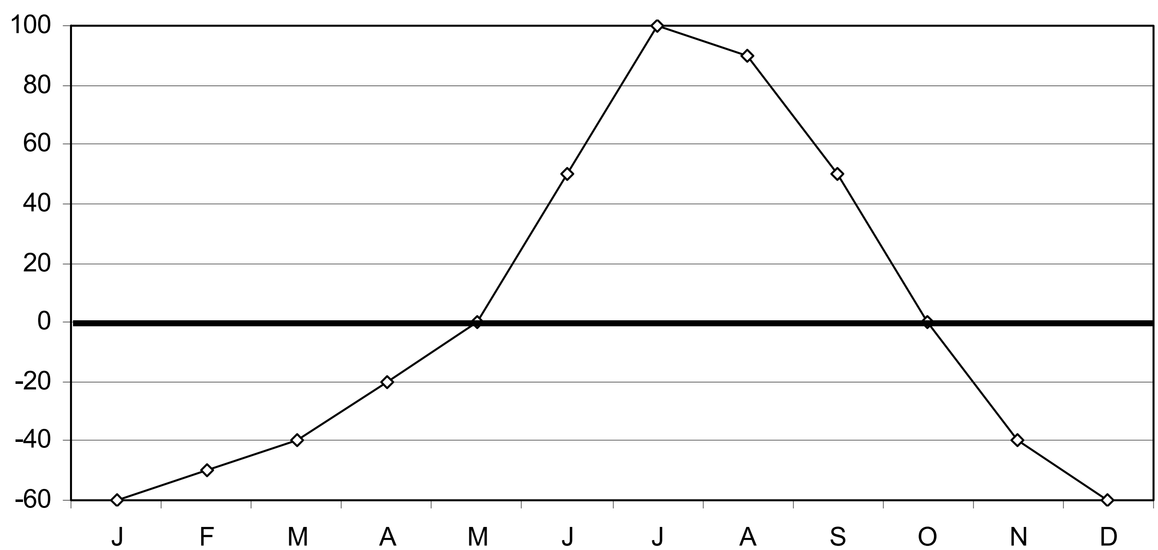


Figure IV-12. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

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

Sediment sampling was conducted throughout the primary component basins (e.g. Mill Pond, Follins Pond, Kelleys Bay, Dinah's Pond, Grand Cove) which comprise the Bass River Estuary in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model. The distribution of cores in each sub-basin, harbor and cove was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and sediment type and an analysis of each site's tidal flow velocities. The maximum bottom water flow velocity at each coring site was determined from the hydrodynamic model. These data were then used to determine the nitrogen balance within each sub-embayment.

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

Net nitrogen release or uptake from the sediments within the Bass River Embayment System were comparable to other similar embayments with similar configuration and flushing rates in southeastern Massachusetts. In addition, the spatial pattern of sediment N release was also similar to other systems, with the salt marsh basins and creeks showing net nitrogen uptake, the embayment depositional basins with oxidized surficial sediments showing low rates of net nitrogen uptake and the depositional areas within the tidal river showing net nitrogen release. Sediment nitrogen release rates were generally higher in the open water depositional basins with soft organic rich mud with a thin oxidized surface layer. These basins tended to have similar basin morphologies, tidal velocities and sediment characteristics and similar rates of net nitrogen release (Follins Pond, 46 mg N $m^{-2}d^{-1}$; Kelleys Bay, 75.1 mg N $m^{-2}d^{-1}$; and Grand Cove, 80.9 mg N $m^{-2}d^{-1}$). These rates are similar to other sub-embayments on Cape Cod, for example, the depositional main basin of East Bay (Centerville River Estuary) and the lower basin of Rock Harbor (Orleans/Eastham), both of which support benthic regeneration rates of 59.1 mg N m^{-2} d^{-1} and 80.8 mg N m^{-2} d^{-1}, respectively. Additionally, the drown kettle basins within Pleasant Bay, Meetinghouse Pond (79.5 mg N m^{-2} d^{-1}), Areys Pond (107.3 mg N m^{-2} d^{-1}), Lonnies Pond (22.7 mg N m^{-2} d^{-1}), Quanset Pond (98.0 mg N m^{-2} d^{-1}), and Paw Wah Pond (120.7 mg N m^{-2} d^{-1}) also have similar basins and net rates of nitrogen release. The net uptake in Dinah Pond with impaired eelgrass (-30 mg N m^{-2} d^{-1}) is nearly identical to the eelgrass areas in Trapps Pond (Sengekontaket Pond Estuary) which have net release rates -25.7 to -5.9 mg N m^{-2} d^{-1}. It is also notable that the brackish basins comprising the upper tidal reaches in the Madaket Harbor-Long Pond Embayment System show comparable rates to the brackish basin of Mill Pond, 6-14 mg N m^{-2} d^{-1} versus 6.1 mg N m^{-2} d^{-1}, respectively. Overall, it is clear that the multiple component basins of Bass River presently support rates of summertime sediment nitrogen release typical of these types of basins in other estuaries, with similar structure and sediment characteristics, that are also tributary to Nantucket/Vineyard Sound and Pleasant Bay.

Net nitrogen release rates for use in the water quality modeling effort for the main basins of the Bass River Embayment System (Chapter VI) are presented in Table IV-6. There was a clear spatial pattern of sediment nitrogen flux, with moderate net release of nitrogen in the main depositional basins, uptake of nitrogen within the brackish water basin (Mill Pond) and uptake in the eelgrass basin (Dinah Pond) The sediments within the Bass River Embayment System showed nitrogen fluxes typical of similarly structured systems within the region and appear to be in balance with the overlying waters and are consistent with the level of nitrogen loading to this system and its rates of tidal flushing.

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

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			Station I.D. *
	Mean	S.E.	# sites	BSR-#
Bass River Estuarine System				
Mill Pond	6.1	30.0	3	41, 42, 43
Follins Pond	46.0	9.0	9	25 - 32, 39, 40
Kelleys Bay	75.1	25.7	3	33, 37, 38
Dinah Pond	-30.0	8.4	3	34, 35, 36
Upper River	17.4	9.0	13	7, 8, 14 - 24
Grand Cove	80.9	22.2	5	9, 10 11, 12, 13
Lower Marsh	20.1	29.5	3	2, 3, 4
Lower River	-16.6	5.2	3	1, 5, 6

* Station numbers refer to Figure IV-11a and 11b.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

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

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

Coastal embayments like the Bass River 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 Bass River system, which is located in the Town of Yarmouth, on Cape Cod. A section of a topographic map in Figure V-1 shows the general study area. The Bass River system has many attached sub-systems, with the three main sub-divisions: 1) the main river reach, 2) Weir Creek, 3) Grand Cove, 4) Kelleys Bay, 5) Dinahs Pond, 6) Follins Pond and 7) Mill Pond. The entire Bass River system has a surface coverage of 1,150 acres, including the attached sub-embayments. Follins Pond is the largest sub-embayment of the River system, with a 215-acre coverage.

Circulation in the Bass River system is dominated by tidal exchange with Nantucket Sound. The River is connected to Nantucket Sound through a broad, structured inlet. The east inlet jetty is over 2,100 foot-long, extending from Davis Beach Road, and 1,200 ft beyond the beach into the Sound. A survey of tidal flows, performed at the inlet using an Acoustic Doppler Current Profiler (ADCP), indicates that the east jetty is permeable enough to permit approximately 15% of the total flow entering the River system on a flood tide to flow though the structure. Over the length of the system, there is considerable attenuation of the tide range. Between the inlet and Kelleys Bay, north of the Route 6 crossing, the average tide range is reduced from 3.4 feet to 1.9 feet, a reduction of 1.5 feet or 44%. This reduction is caused by frictional losses along the 6.25 mile-long reach of the River, to the culvert entrance of Mill Pond at the head of the system.

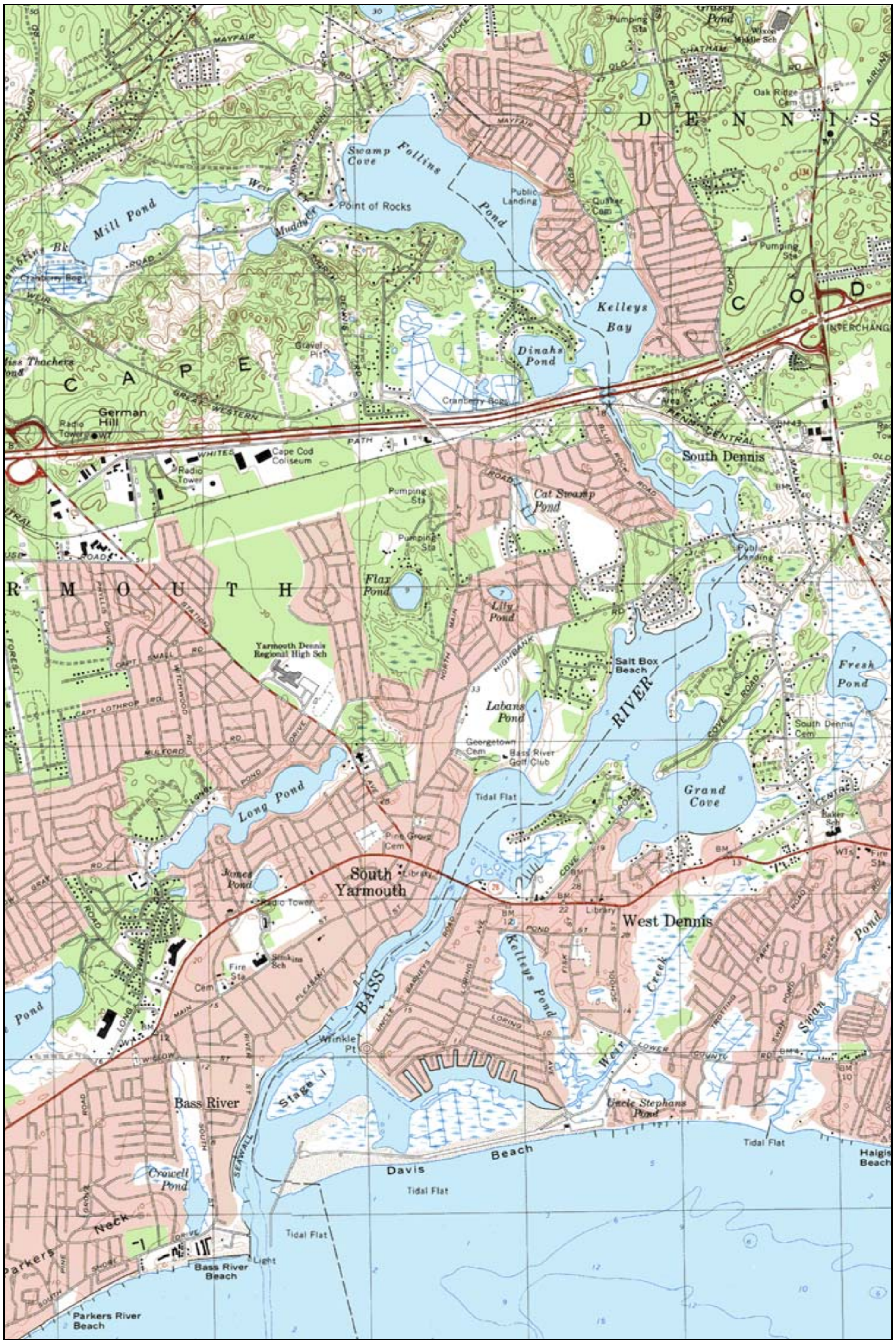


Figure V-1. Topographic map detail of the Bass River, from Nantucket Sound to Mill Pond.

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 Bass River was performed to determine the present variation of embayment and channel depths throughout the system. In addition to bathymetry, tides were recorded at seven locations within the River system for at least a complete lunar month (29.5 days). These tide data were necessary to run and calibrate the hydrodynamic model of the system. Finally, an Acoustic Doppler Current Profiler (ADCP) survey was completed during a single tide cycle to measure ebb and flood velocities across two 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 Bass River system was developed in the second portion of this analysis. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from offshore, in Nantucket Sound, were used to define the open boundary conditions that drive the circulation of the model at the system inlet, and data from the 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 the River was used to compute the flushing rates of selected sub-sections. Though water quality in an embayment cannot be directly inferred by use of computed flushing rates alone, they can serve as useful indicators of embayment flushing performance relative to other areas in the same system. The ultimate utility of this hydrodynamic model is as input into a constituent transport model, where water quality constituents like nitrogen are modeled to determine the real water quality dynamics of a system.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE ESTUARINE SYSTEM

V.3 DATA COLLECTION AND ANALYSIS

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

V.3.1 Bathymetry Data Collection

Bathymetry data in Bass River were collected during October 2004. The October 2004 survey employed a single-beam acoustic fathometer mounted to a motor boat. Positioning data were collected using a differential GPS. The survey design included gridded transects at roughly 100 meter spacings in the larger basins of the system (e.g. Grand Cove), and finer 50 meter spacings at the inlet and in sections where the river confines tidal flow to a narrow channel (e.g., at the bridge crossings). Survey paths and measured depths are shown in Figure V-2. The resulting bathymetric surface created by interpolating the data to a finite element mesh is shown in Figure V-3. All bathymetry was tide corrected, and referenced to the NGVD 29 vertical datum, using survey benchmarks located in the project area.

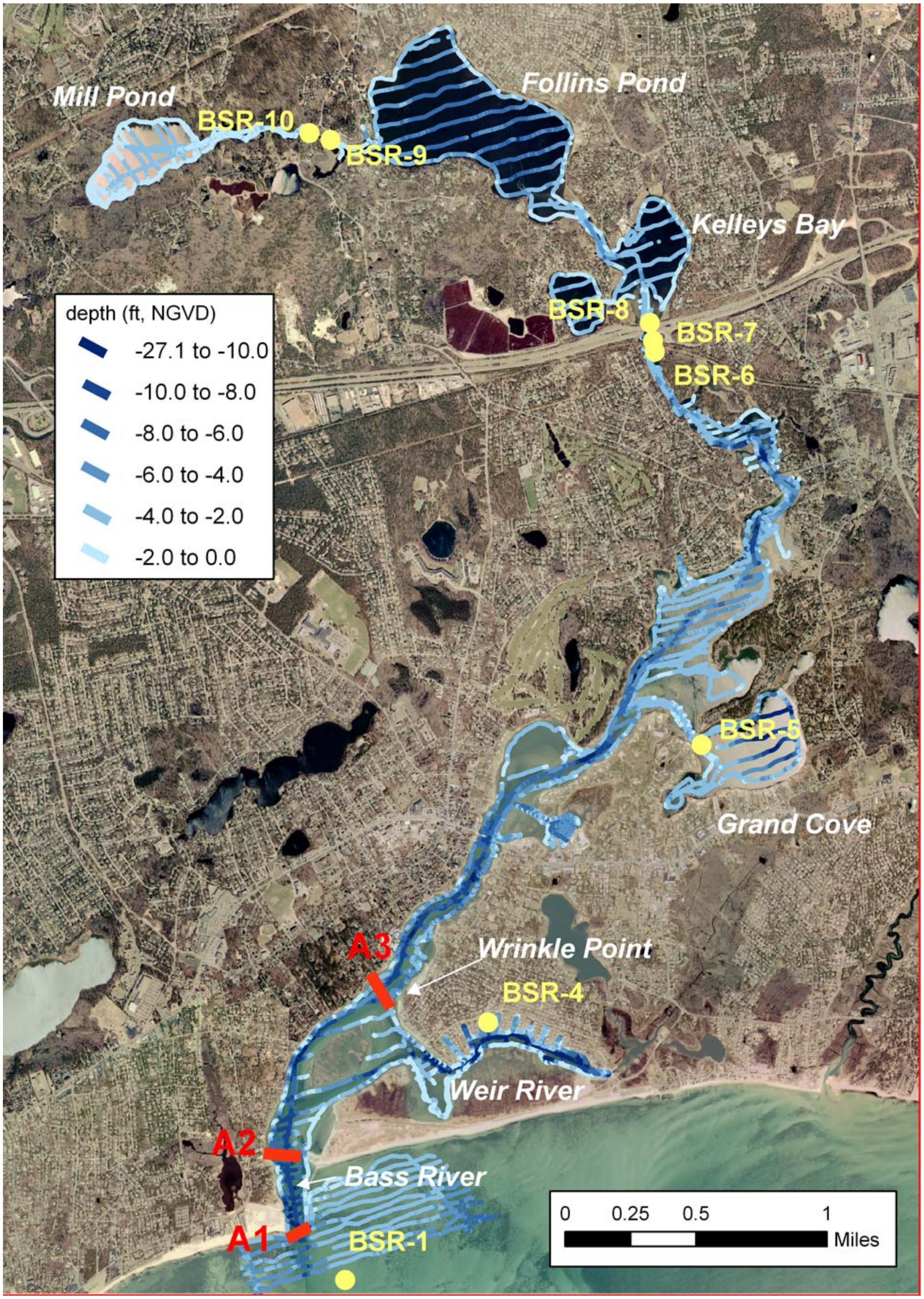


Figure V-2. Transects (blue-shaded lines) from the bathymetry survey of Bass River and markers (yellow circles) show the locations of the tide recorders deployed for this study. Red transects represent locations of ADCP surveys for model validation.

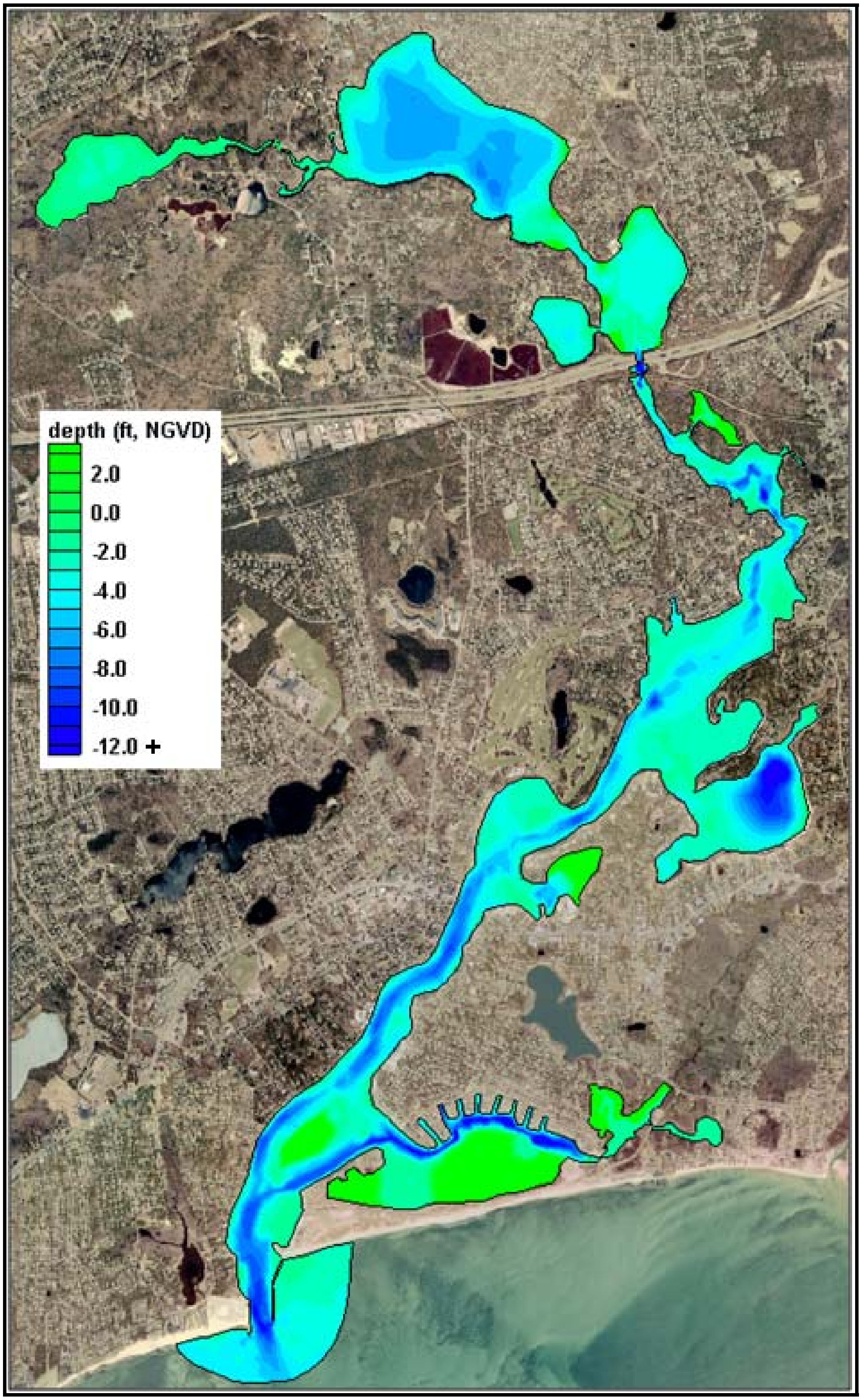


Figure V-3. Plot of interpolated finite-element grid bathymetry of the Bass River system, shown superimposed on 2005 aerial photos of the system locale. Bathymetric contours are shown in color at one-foot intervals.

Results from the survey show that the deepest point within the river is located in the small area between the railroad bridge and the Route 6 crossing, and is -27.1 ft NGVD. Other deep regions of the River system include the dredged channel in Weir Creek (located in the lower portion of the Bass River system{Dennis}), which has a maximum depth of -15.8 feet, and Grand Cove, which has a maximum depth of -16.0 feet, and an average depth of -3.6 feet.

V.3.2 Tide Data Collection and Analysis

Tide data records were collected at seven stations in the Bass River estuary: 1) offshore the inlet, 2) Weir Creek, 3) Grand Cove, 4) south of the railroad bridge, 5) north of the railroad bridge, 6) Kelleys Bay and 7) Mill Pond at the head of the system. The locations of the stations are shown in Figure V-2. The Temperature Depth Recorders (TDR) used to record the tide data were deployed for a 30-day period beginning September 17. The elevation of each gauge was surveyed relative to NGVD 29. Duplicate offshore gauges were deployed to ensure data recovery, since the offshore tide record is crucial for developing the open boundary condition of the hydrodynamic model of the Bass River system. Data from the gauges inside the system were used to calibrate the model.

Plots of the tide data from six representative gauges are shown in Figure V-4 for the entire 40-day deployment. The spring-to-neap variation in tide can be seen in these plots. From the plot of the data from offshore Bass River inlet, the tide reaches its maximum spring tide range of approximately 5.4 feet around October 17. About seven days earlier the neap tide range is much smaller, as small as 1.6 feet.

A visual comparison in Figure V-5 between tide elevations at four stations in Bass River shows that there is a reduction in the tide range as the tide propagates to the upper reaches of the system. The loss of amplitude with distance from the inlet is described as tidal attenuation. Frictional mechanisms dissipate tidal flow energy, resulting in a reduction of the height of the tide. Tide attenuation is accompanied by a time delay (or phase lag) in the time of high and low tide (relative to the offshore tide), which becomes more pronounced farther into an estuary. Not including Mill Pond, which is further restricted by a culvert, the tide lag is greatest in Follins Pond, as seen in Figure V-5, where high tide occurs approximately 130 minutes after high tide in Nantucket Sound.

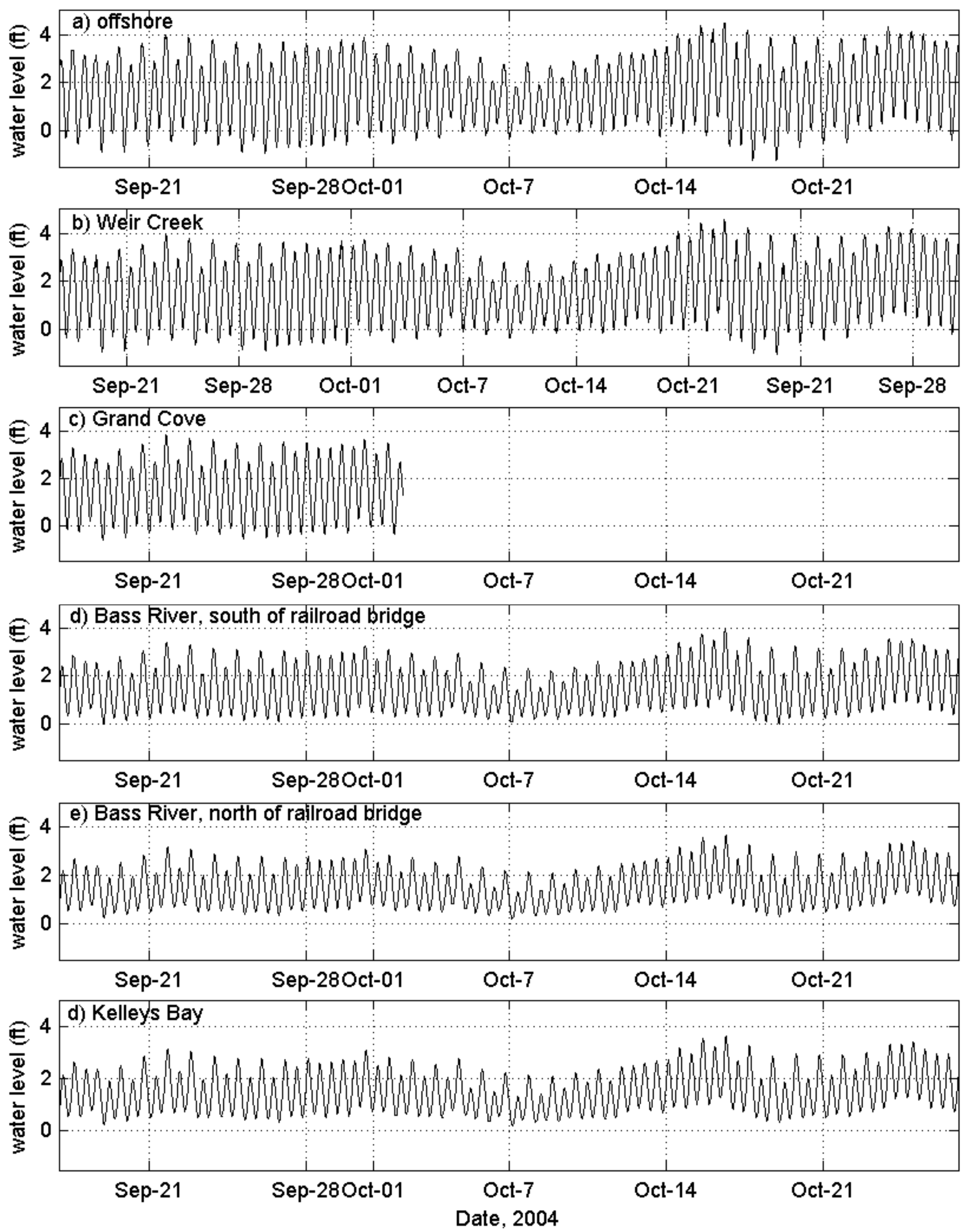


Figure V-4. Plots of observed tides for the Bass River system, for the 40-day period between September 17 and October 27, 2004. The top plot shows tides offshore Bass River inlet, in Nantucket Sound. Tides recorded at Weir Creek, Grand Cove, up- and downstream of the railroad bridge, and in Kelleys Bay are also shown. All water levels are referenced to the National Geodetic Vertical Datum of 1927 (NGVD).

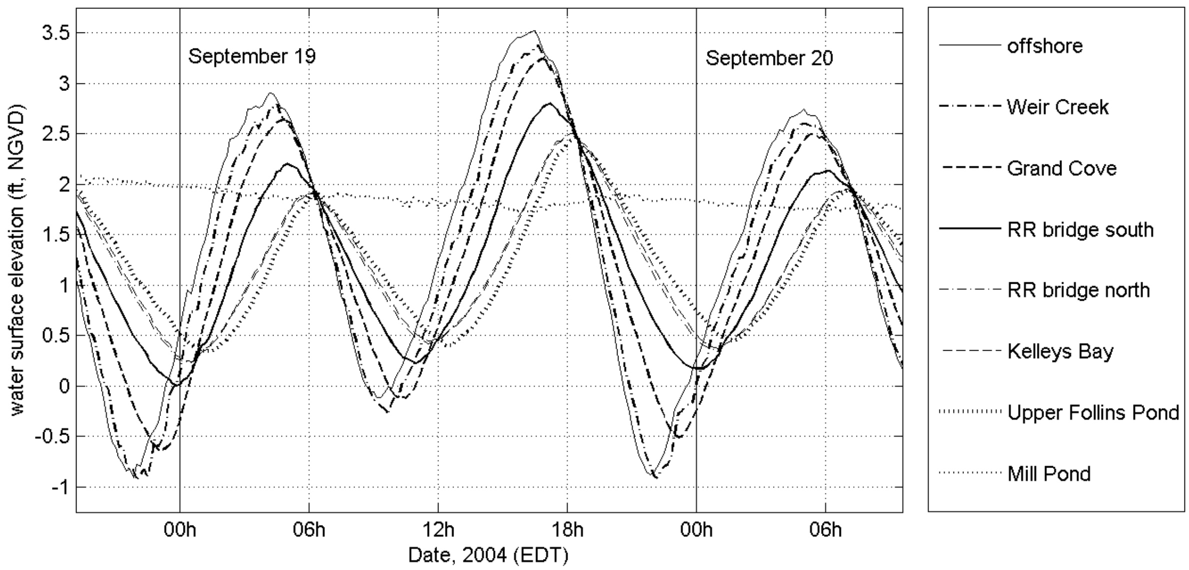


Figure V-5. Plot showing two tide cycles tides at three stations in the Bass River system plotted together. Demonstrated in this plot is the frictional damping effect caused by flow restrictions along the river's length. The damping effects are seen only as a lag in time of high and low tides from Nantucket Sound. The maximum time lag of low tide between the Sound and upper Follins Pond in this plot is 225 minutes (3.8 hours).

Standard tide datums were computed from the 40-day records. These datums are presented in Table V-1. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available, however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW.

As the tide propagates from Nantucket Sound to the upper reaches of the River attenuation of the tide occurs. This is observed as a reduction in the tide range and also as a delay in the time of high and low tide during each tide cycle. The mean tide range in Nantucket Sound is 3.4 feet. In Kelleys Bay the mean tide range is reduced to 1.8 feet by frictional losses along the length of the River and by the railroad bridge.

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

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

A harmonic analysis was performed on the time series from each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The observed astronomical tide is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-6. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of eight tidal constituents in the Bass River 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 1.4 ft throughout the system. The total range of the M_2 tide is twice the amplitude, or 3.2 f for the offshore gauge. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 period for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 has an amplitude of 0.1 feet near the system inlet, but is reduced in Kelleys Bay near the head of the system. The M_6 has a very small amplitude in the system (less than 0.1 feet at all gauge stations). There is a similar change in the M_2 across the system.

Table V-1. Tide datums computed from a 28-day period from the tide records collected in the Bass River system. Datum elevations are given relative to NGVD 29.

Tide Datum	Offshore	Weir Creek	Grand Cove	RR Bridge – south	RR Bridge – north	Kelleys Bay
Maximum Tide	4.5	4.6	3.9	4.3	4.0	3.9
MHHW	3.5	3.5	3.6	3.2	3.1	3.1
MHW	3.2	3.2	3.2	3.0	2.8	2.8
MTL	1.5	1.5	1.6	1.8	1.9	1.9
MLW	-0.2	-0.2	-0.1	0.7	1.0	1.0
MLLW	-0.4	-0.4	-0.3	0.6	0.9	0.9
Minimum Tide	-0.9	-0.9	-0.5	0.3	0.6	0.6

For all the other included constituents, except for the fortnightly M_{sf}, amplitudes decrease with distance into the system. The other major tide constituents also show little variation across the system. The diurnal tides (once daily), K_1 and O_1, possess amplitudes of approximately 0.3 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 0.3 feet, respectively. The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon, and has an amplitude of 0.3 ft.

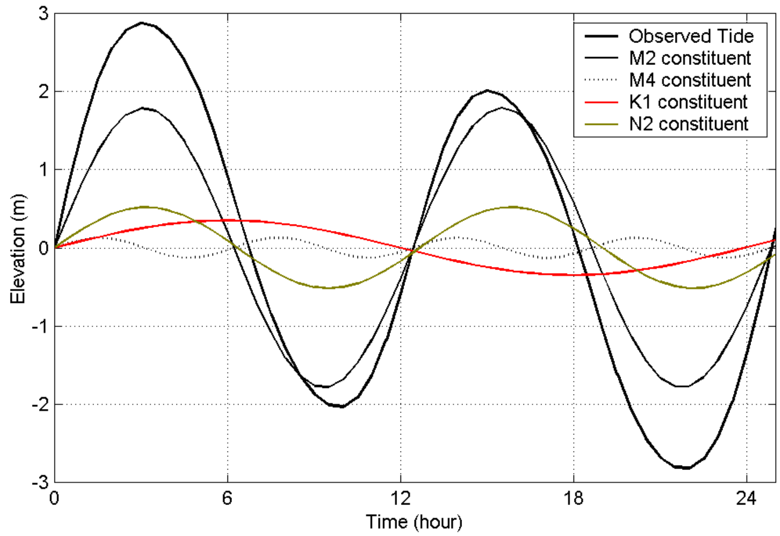


Figure V-6. Example of an observed astronomical tide as the sum of its primary constituents.

Table V-2. Major tidal constituents determined for gauge locations in Bass River, September 18 through October 18, 2004.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	M_{sf}
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Nantucket Sound (offshore)	1.60	0.14	0.06	0.24	0.34	0.25	0.34	0.25
Weir Creek	1.55	0.12	0.06	0.22	0.32	0.25	0.33	0.27
RR Bridge – south	1.06	0.05	0.02	0.15	0.20	0.21	0.27	0.26
RR Bridge - north	0.85	0.06	0.02	0.12	0.15	0.19	0.25	0.27
Kelleys Bay	0.84	0.05	0.02	0.11	0.15	0.19	0.25	0.27

Though there is little change in constituent amplitudes across the length of the main basin of the River, the phase change of the tide is easily seen from the results of the harmonic analysis. Table V-3 shows the delay of the M_2 at different points in the Bass River system, relative to the timing of the M_2 constituent in Nantucket Sound, offshore the inlet. The analysis of the data from Kelleys Bay show that there is a 134 minute delay between the inlet and the farthest reach of the system. Further delays occur at the Mill Pond TDR station, a sub-embayment which is separated from Follins Pond by a culvert. Compared to other locations instrumented in this study, Mill Pond shows the greatest tidal attenuation (Figure V-5).

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

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

Station	Delay (minutes)
Weir Creek	14.5
RR Bridge – south	77.2
RR Bridge – north	128.3
Kelleys Bay	134.3

Table V-4 shows that the variance of tidal energy decreases for stations that are farther from the inlet. The analysis also shows that tides are responsible for more than 90% of the water level changes for all gauges in the Bass River system. The remaining variance was the result of atmospheric forcing, due to winds, or barometric pressure gradients.

Table V-4. Percentages of Tidal versus Non-Tidal Energy for stations in Bass River, September to October 2004.

TDR LOCATION	Total Variance (ft^2)	Tidal (%)	Non-tidal (%)
Nantucket Sound (offshore)	1.55	97.0	3.0
Weir Creek	1.42	96.5	3.5
RR Bridge – south	0.69	93.6	6.4
RR Bridge - north	0.47	91.3	8.7
Kelleys Bay	0.46	90.9	9.1

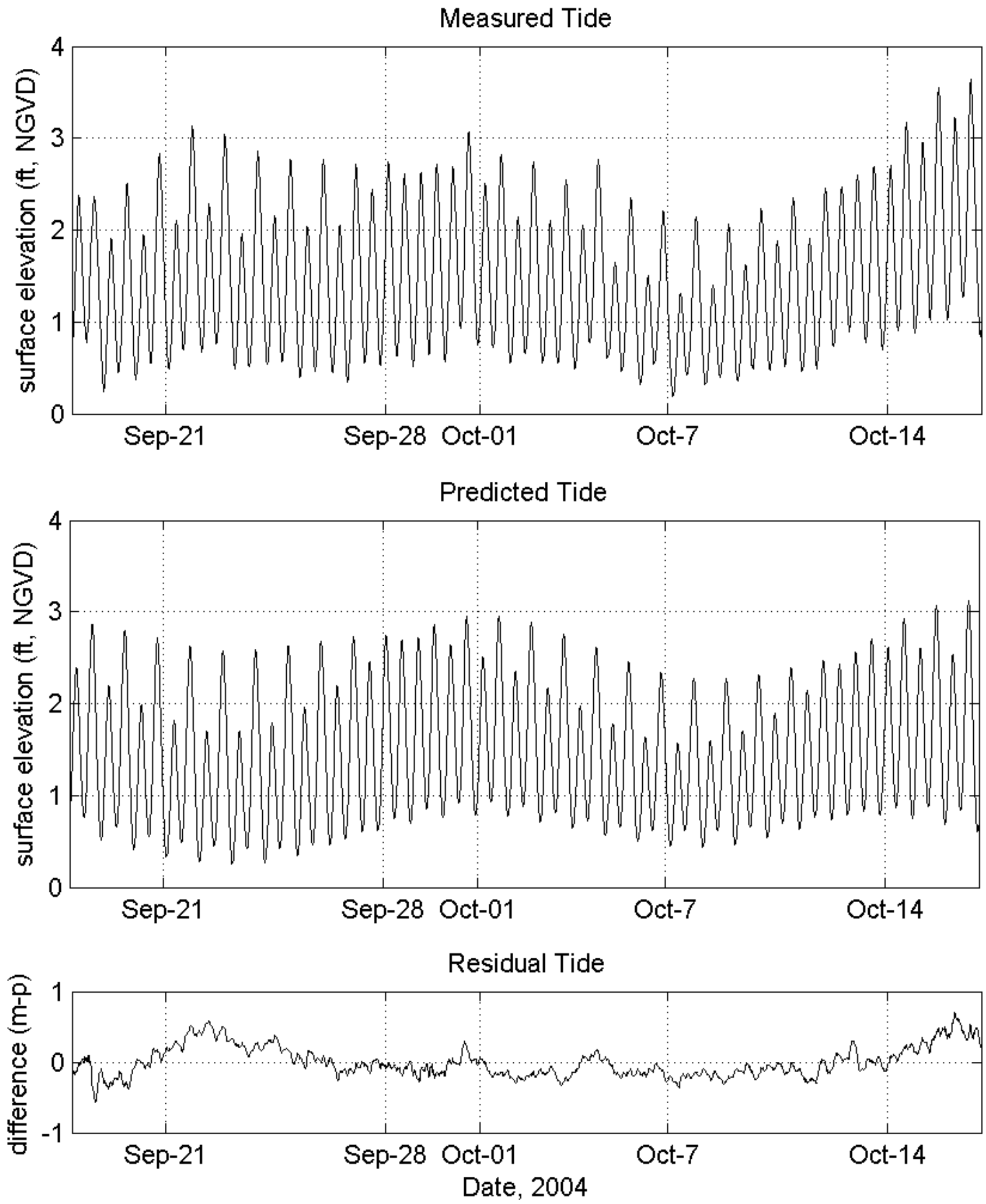


Figure V-7. Plot showing the comparison between the measured tide time series (top plot), and the predicted astronomical tide (middle plot) computed using the 23 individual tide constituents determine in the harmonic analysis of the Kelleys Bay gauge data. The residual tide shown in the bottom plot is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

V.2.3 ADCP Data Analysis

Cross-channel current measurements were surveyed through a complete tidal cycle in the Bass River system on October 13, 2004 to resolve spatial and temporal variations in tidal current patterns. The survey was designed to observe tidal flow across three transects in the system at hourly intervals. The two main transects of the survey (indicated in Figure V-2) were located 1) just inside the inlet, downstream of the confluence of Weir Creek, and 2) at Wrinkle Point. An auxiliary third transect between the tips of the inlet jetties. The data collected during this survey provided information that was necessary to model properly validate the hydrodynamic model of the Bass River system.

Figures V-8 through V-13 show color contours of the current measurements observed during the flood and ebb tides at three of the 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, between western shore and the jetty, positive along-channel flow is to the north, and positive cross-channel flow is moving to east. In Figure V-8, the lower left panel shows depth-averaged currents across the channel projected onto a 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.

Inside the inlet, maximum measured currents in the water column were between 1.5 and 2.1 ft/sec (0.9 and 1.2 knots). Maximum flood flows in the morning of October 13 were 6,740 ft^3/sec. In the afternoon, maximum ebb flows were 6700 ft^3/sec. Across the transect at Wrinkle Point, maximum measured currents over the entire measured tide cycle varied less, between 1.3 and 2.3 ft/sec. During maximum flood and ebb flows, the discharge rates were 4,620 ft^3/sec and 6,000 ft^3/sec, respectively. Between the jetties, the maximum measured currents were between 1.8 and 2.6 ft/sec (1.1 and 1.5 kts), and the maximum measured total flow rates were 5,420 ft^3/sec and 8,750 ft^3/sec for flood and ebb tides respectively.

During maximum flood, the discharge measured at the jetty tips was 80% of the tidal flow measured inside the inlet. This measurement indicates that the jetties are permeable, considering that the jetties represent 70% of the flow perimeter of the inlet, with the remaining 30% being the full width of the opening between the jetty tips. Based on the percentage of the inlet flow perimeter outlined by the jetties versus the line between the jetty tips, and that 80% of the measured flow into the Bass River flows over the jetties, the average permeability of the jetties (i.e., average over their entire length) is computed to be 29%. The actual permeability of the jetties is dependent upon the stage of the tide.

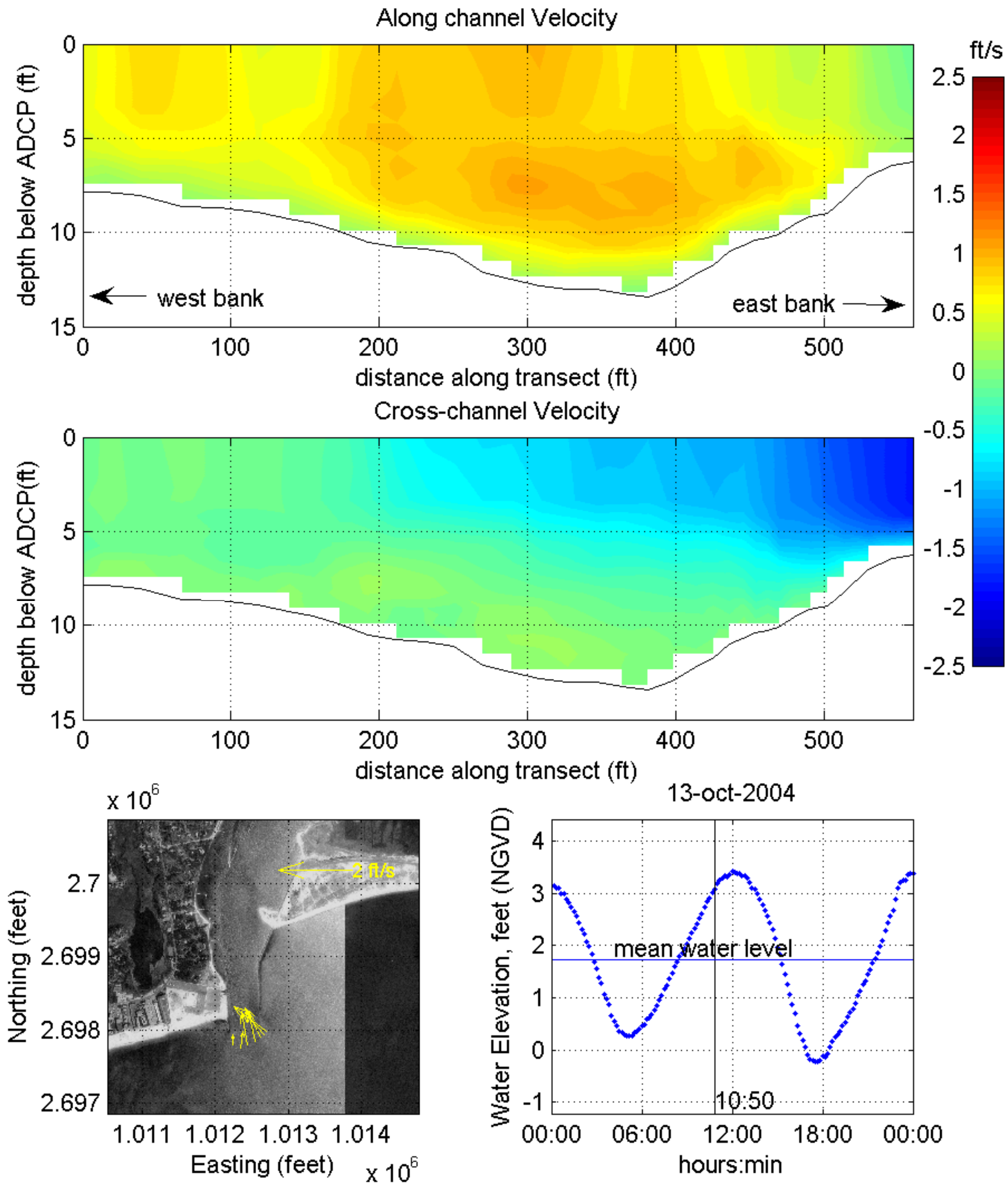


Figure V-8. Color contour plots of along-channel and cross-channel velocity components for transect line run between the Bass River inlet jetty tips, measured at 10:50 on October 13, 2004 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is 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 (MASS GIS) of the survey area. A tide plot for the survey day is also given.

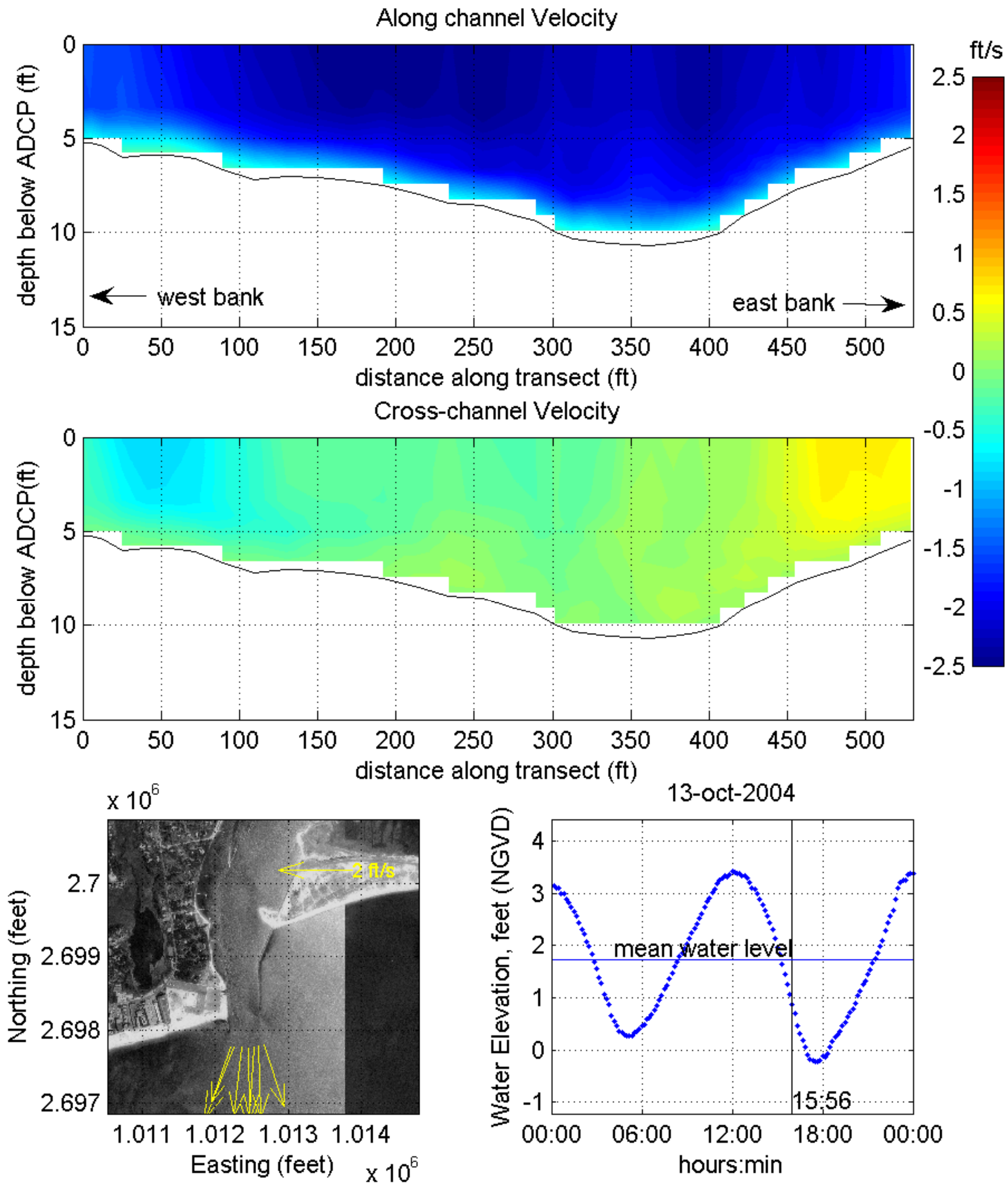


Figure V-9. Color contour plots of along-channel and cross-channel velocity components for transect line run between the Bass River inlet jetty tips, measured at 15:56 on October 13, 2004 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is 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 (MASS GIS) of the survey area. A tide plot for the survey day is also given.

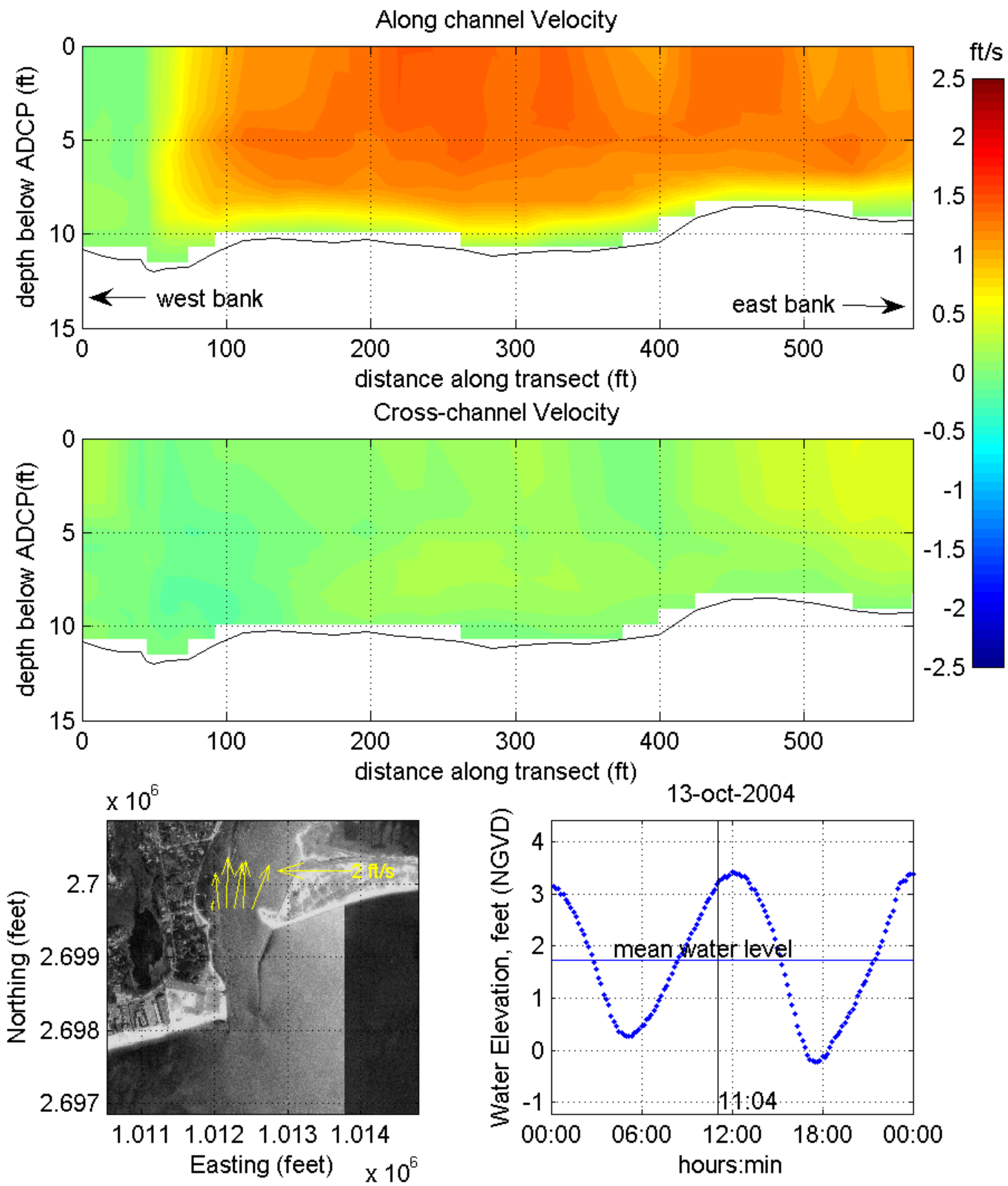


Figure V-10. Color contour plots of along-channel and cross-channel velocity components for transect line run First inside Bass River inlet, measured at 11:04 on October 13, 2004 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving 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 (MASS GIS) of the survey area. A tide plot for the survey day is also given.

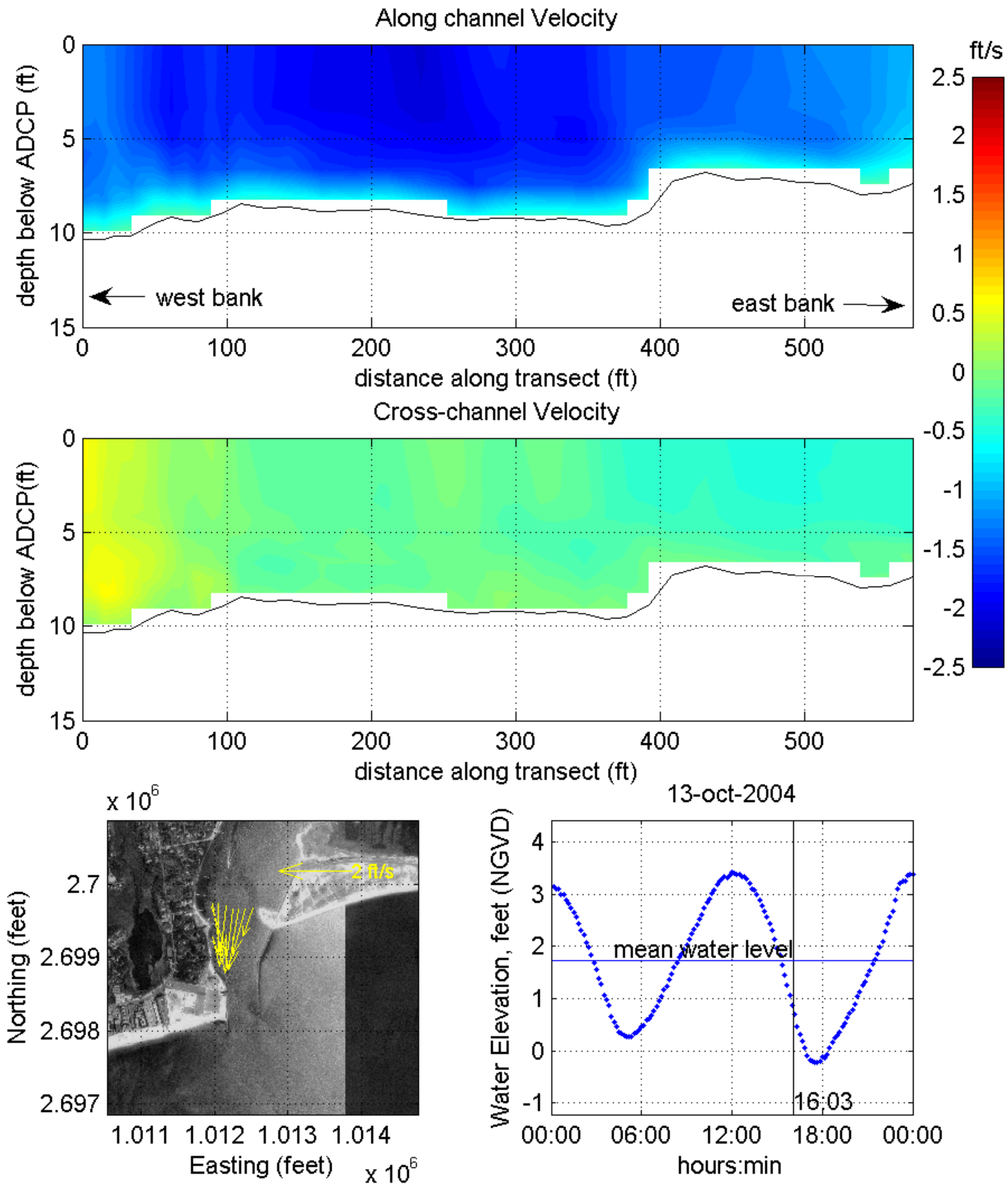


Figure V-11. Color contour plots of along-channel and cross-channel velocity components for transect line run inside Bass River inlet, measured at 16:03 on October 13, 2004 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the 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 (MASS GIS) of the survey area. A tide plot for the survey day is also given.

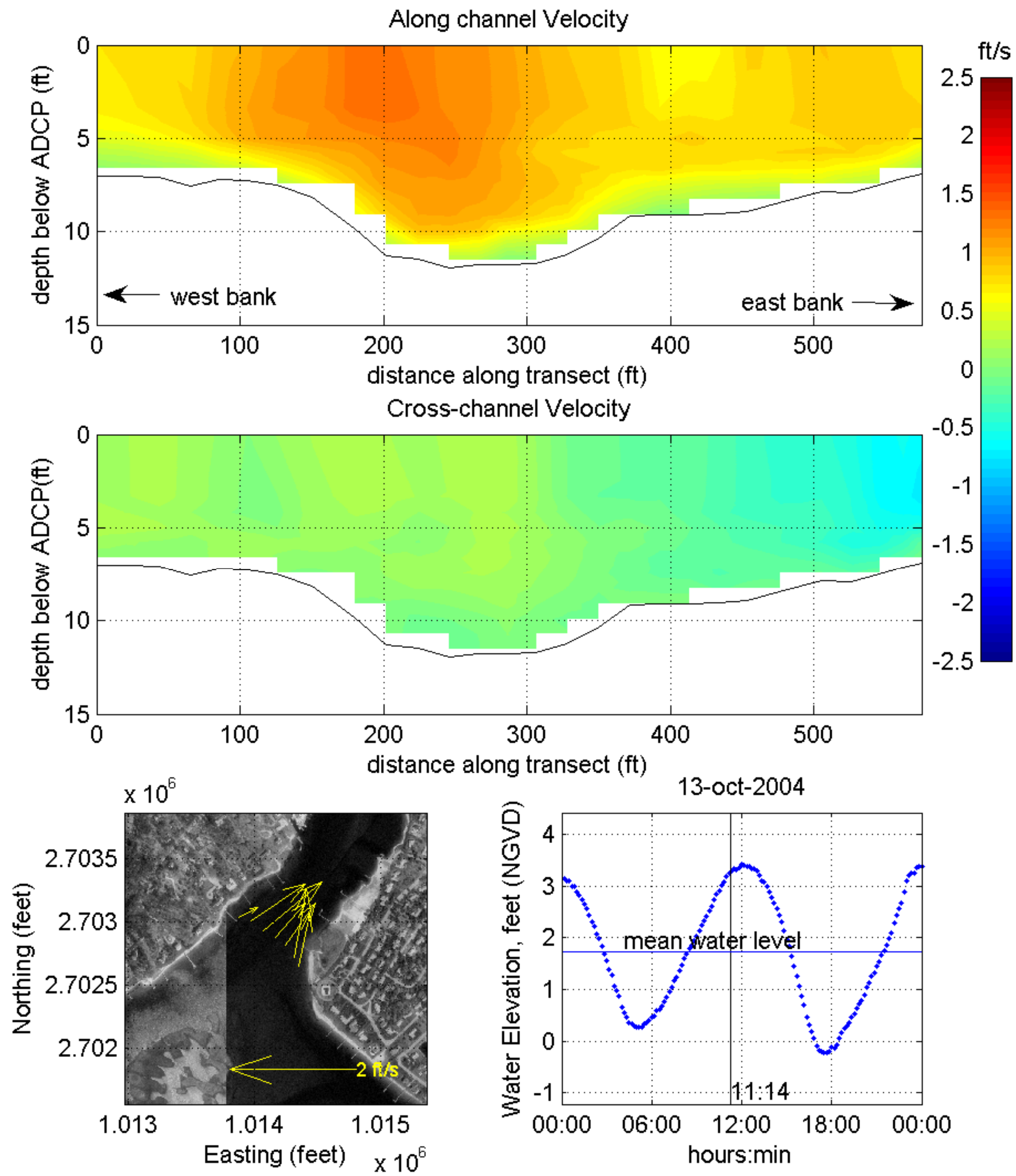


Figure V-12. Color contour plots of along-channel and cross-channel velocity components for transect line run at Wrinkle Point, measured at 11:14 on October 13, 2004 during the period of maximum flood tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, 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 (MASS GIS) of the survey area. A tide plot for the survey day is also given.

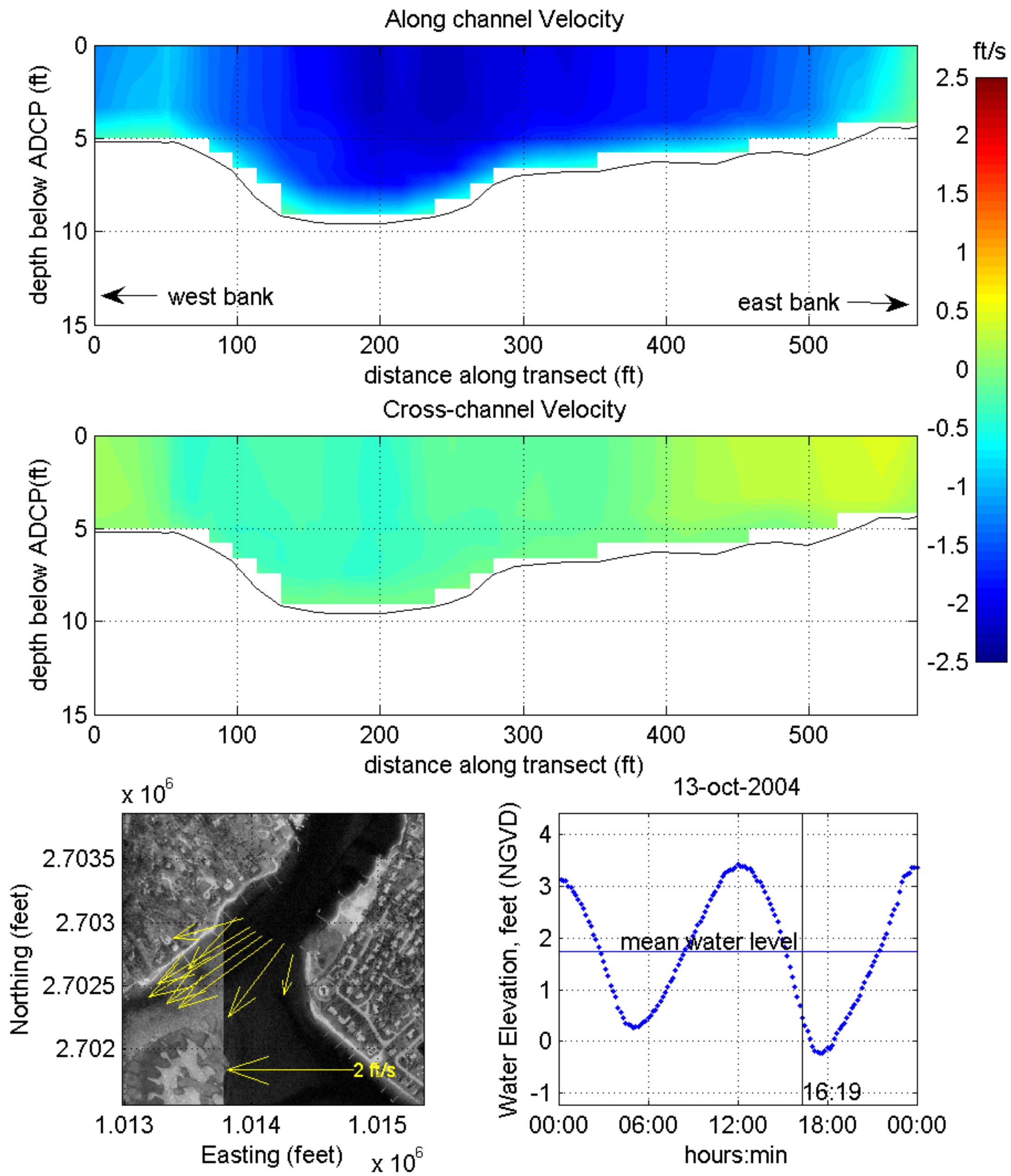


Figure V-13. Color contour plots of along-channel and cross-channel velocity components for transect line run at Wrinkle Point, measured at 16:19 on October 13, 2004 during the period of maximum ebb tide currents. Positive along-channel currents (top panel) indicate the flow is moving into the estuary, 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 (MASS GIS) of the survey area. A tide plot for the survey day is also given.

V.4 HYDRODYNAMIC MODELING

For modeling of Bass River, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in this system. 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 and the Islands, including West Falmouth Harbor, Popponesset Bay, Pleasant Bay (Howes, *et al*, 2006), Falmouth "finger" Ponds (Ramsey, *et al*, 2000), and Barnstable Harbor (Wood, *et al*, 1999), and Three Bays (Howes, *et al*, 2005).

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

V.4.2.1 Grid generation

The grid generation process was aided by the use of the SMS package. 2005 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 11,972 nodes, which describe 4,278 total 2-dimensional (depth averaged) quadratic elements, and covers 1,250 acres. The maximum nodal depth is -26.3 ft (NGVD). This deepest depth occurs in a scour hole that exists just upstream of the railroad bridge. The completed grid mesh of the Bass River system is shown in Figure V-14, and grid bathymetry was shown previously in Figure V-3.

The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties throughout the Harbor. 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 main and sub-systems inlet channels was designed to provide a more detailed analysis in these regions of rapidly varying flow (e.g., the inlet channel and the main river channel). Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in the attached sub-embayments, such as Kelleys Bay. Appropriate implementation of wider node spacing and larger elements was used to reduce 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 Bass River system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Tidal boundary conditions were specified at the inlet from Nantucket Sound. TDR measurements from a gauge deployed offshore the inlet provided the required data.

The 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 the model's offshore open boundary every model time step of 10 minutes, which corresponds to the time step of the TDR data measurements.

V.4.2.3 Calibration

After developing the finite element grid, and specifying boundary conditions, the model for the Bass River 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 10+) for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

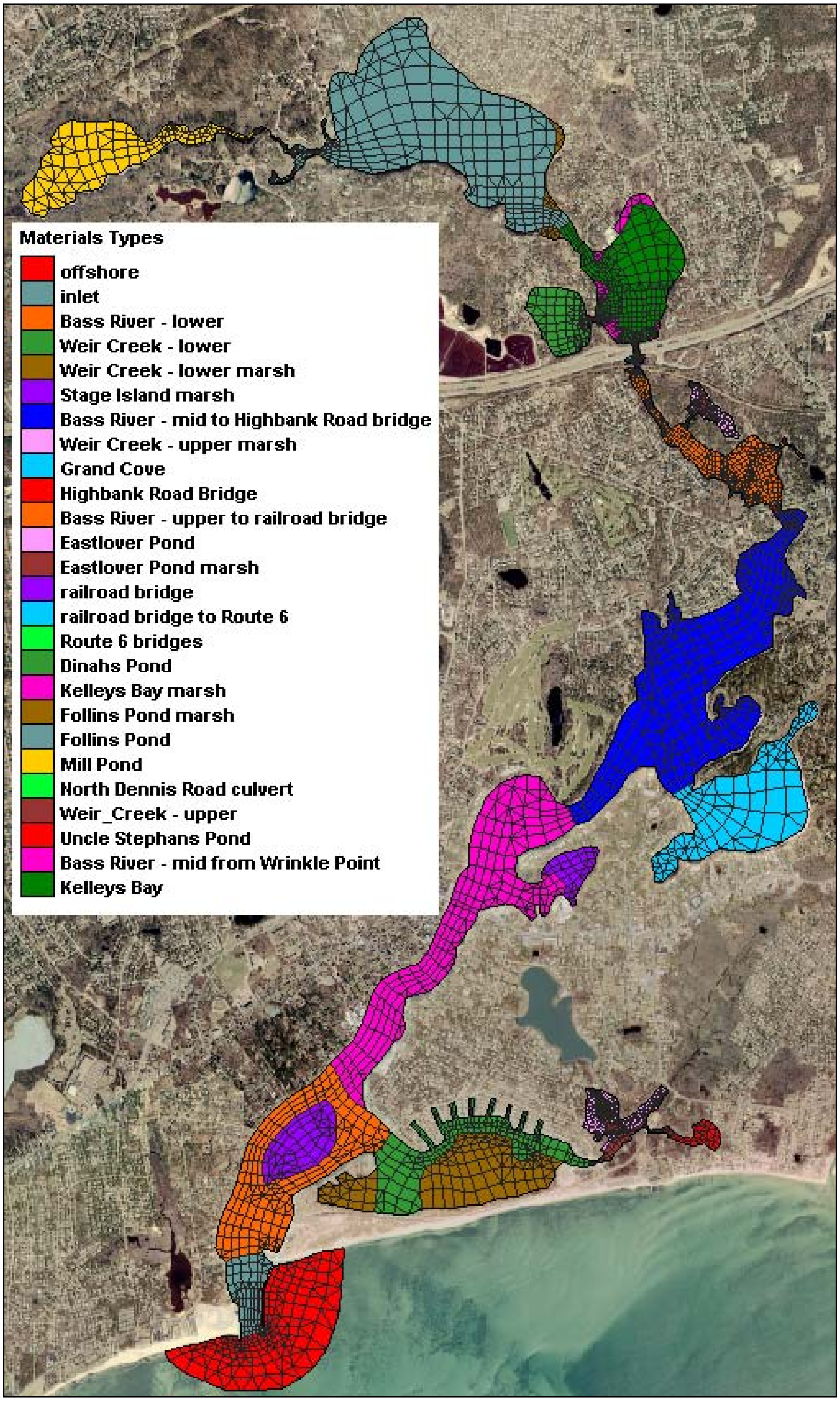


Figure V-14. Plot of hydrodynamic model grid mesh for the Bass River estuarine system of Yarmouth, Massachusetts. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

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 V.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 September 17, 2004 at 0200 EDT. 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).

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 0.025 were specified for all element material types. This values corresponds to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) (Henderson, 1966).

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

Table V-5. Manning's Roughness coefficients used in simulations of modeled sub-embayments. These embayment delineations correspond to the material type areas shown in Figure V-14.

System Embayment	Bottom Friction
Offshore	0.018
Inlet - jetties	0.018
Bass River – lower	0.018
Weir Creek – lower	0.018
Weir Creek – lower marsh	0.070
Stage Island marsh	0.070
Bass River – mid to Highbank Road	0.026
Weir Creek – upper marsh	0.070
Grand Cove	0.018
Highbank Road bridge	0.018
Bass River – upper to railroad bridge	0.026
	0.070
Eastlover Pond	0.018
Eastlover Pond Marsh	0.050
railroad bridge	0.030
railroad bridge to Route 6	0.040
Route 6 bridges	0.018
Dinahs Pond	0.070
Kelleys Bay Marsh	0.070
Follins Pond Marsh	0.025
Follins Pond	0.018
Mill Pond	0.100
North Dennis Road culvert	0.025
Weir Creek – upper	0.025
Uncle Stephans Pond	0.070
Bass River – mid from Wrinkle Point	0.026
Kelleys Bay	0.025

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 other channel 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 75 and 200 lb-sec/ft^2. In most cases, the Bass River 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 Marsh porosity processes

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

V.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-15 through and V-21 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.

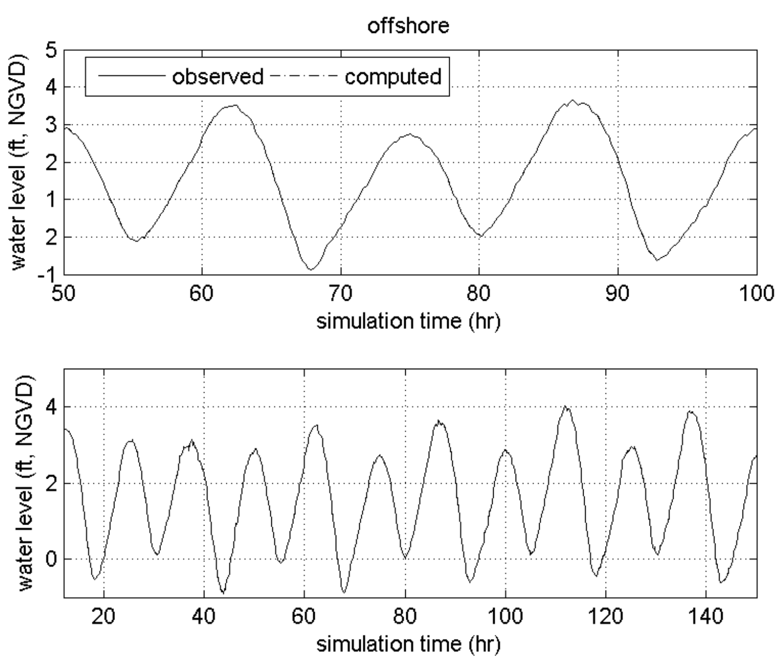


Figure V-15. Comparison of model output and measured tides for the TDR location offshore the inlet to Bass River, in Nantucket Sound. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

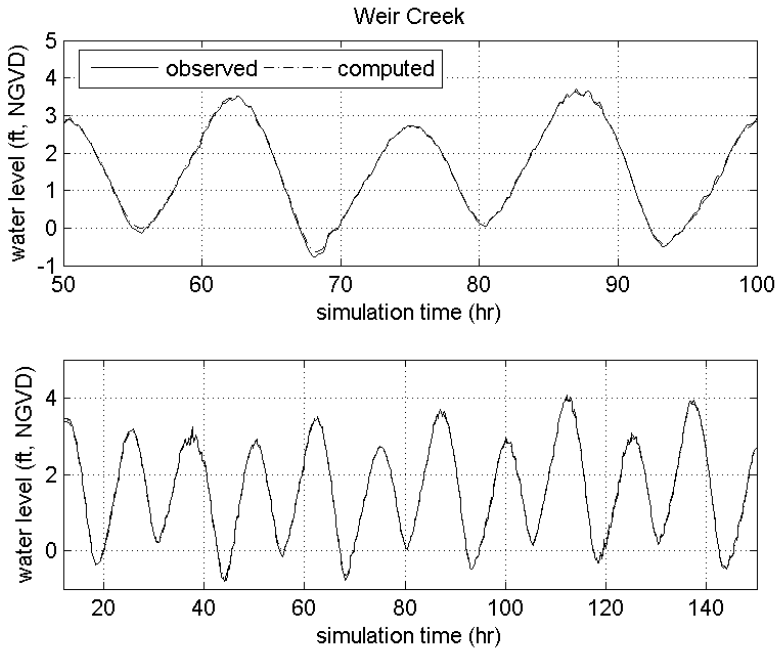


Figure V-16. Comparison of model output and measured tides for the TDR location at Weir Creek. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

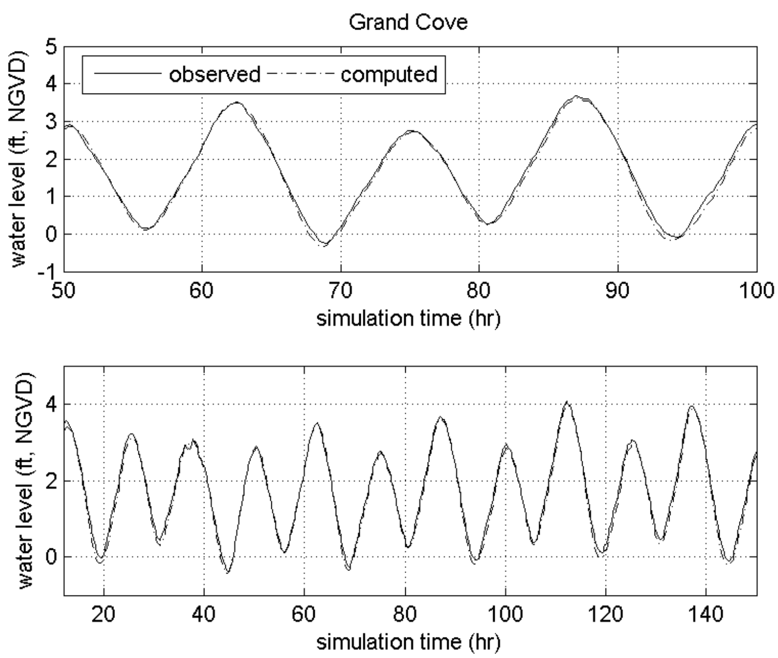


Figure V-17. Comparison of model output and measured tides for the TDR location in Grand Cove. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

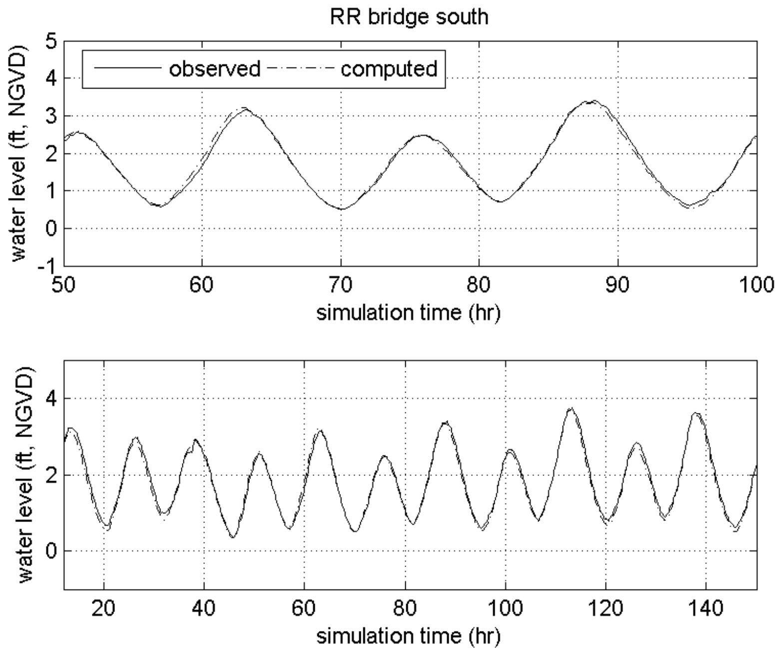


Figure V-18. Comparison of model output and measured tides for the TDR location downstream of the railroad bridge. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

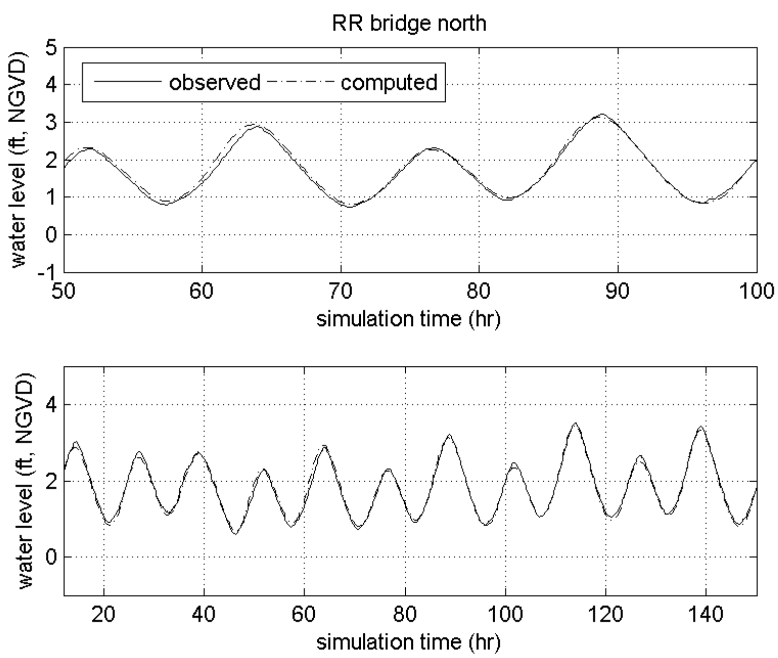


Figure V-19. Comparison of model output and measured tides for the TDR location upstream of the railroad bridge. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

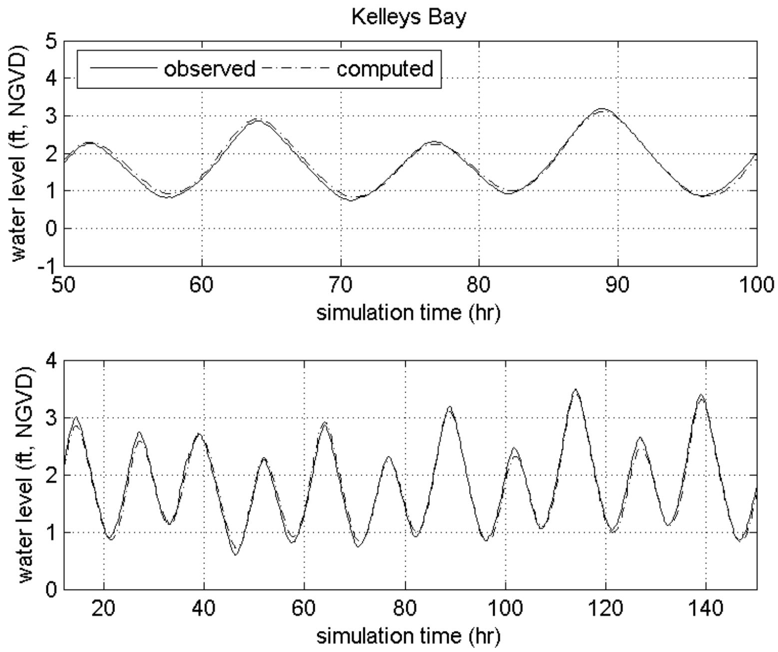


Figure V-20. Comparison of model output and measured tides for the TDR location at Kelleys Bay. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

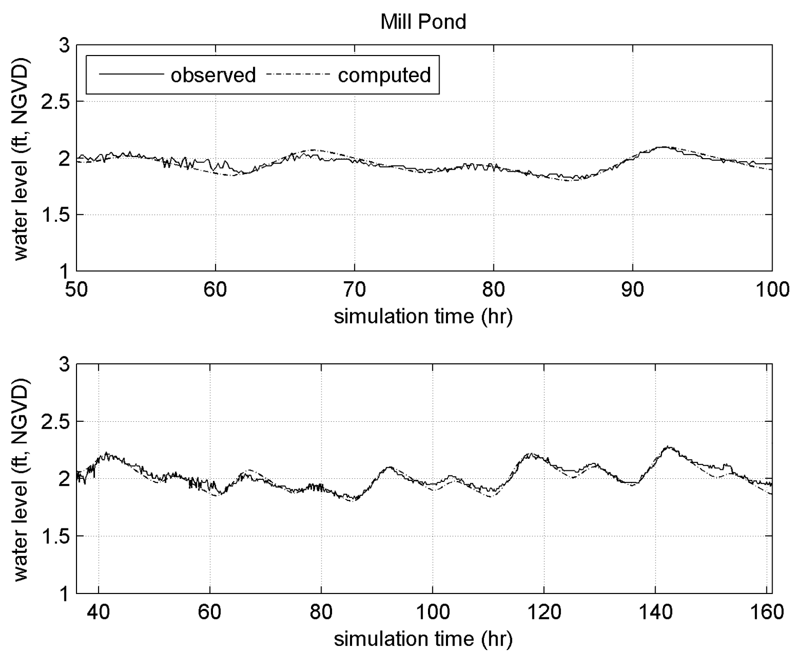


Figure V-21. Comparison of model output and measured tides for the TDR location in Mill Pond. The top plot is a 50-hour sub-section of the total modeled time period, shown in the bottom plot.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 (principle lunar semidiurnal constituent) was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled system. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6 for the calibration period differ from those in Table V-2 because constituents were computed for only the five-day section of the 40-days represented in Table V-2. 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.

Table V-6. Tidal constituents for measured water level data and calibrated model output, with model error amplitudes, for the Bass River 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*	1.64	0.18	0.08	0.52	308.2	8.2
Weir Creek	1.57	0.13	0.06	0.50	315.7	22.2
Grand Cove	1.48	0.07	0.04	0.49	321.6	28.22
RR bridge - south	1.07	0.04	0.02	0.41	343.2	293.1
RR bridge - north	0.85	0.04	0.01	0.37	6.2	341.8
Kelleys Bay	0.82	0.05	0.01	0.37	13.5	4.4
Mill Pond	0.08	0.01	0.00	0.08	97.5	119.6
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*	1.65	0.18	0.08	0.52	309.4	11.8
Weir Creek	1.60	0.15	0.07	0.49	316.6	26.5
Grand Cove	1.44	0.04	0.06	0.47	321.4	56.5
RR bridge - south	1.07	0.04	0.01	0.39	348.0	326.1
RR bridge - north	0.85	0.06	0.03	0.35	8.2	12.3
Kelleys Bay	0.85	0.05	0.03	0.34	11.3	10.5
Mill Pond	0.06	0.01	0.00	0.07	92.0	151.3
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
Nantucket Sound*	0.01	0.00	0.00	0.00	2.5	3.7
Weir Creek	0.03	0.02	0.01	-0.01	1.9	4.5
Grand Cove	-0.04	-0.03	0.02	-0.02	-0.4	28.7
RR bridge - south	0.00	0.00	-0.01	-0.02	9.9	34.1
RR bridge - north	0.00	0.02	0.02	-0.02	4.1	31.6
Kelleys Bay	0.03	0.00	0.02	-0.03	-4.4	6.4
Mill Pond	-0.02	0.00	0.00	-0.01	-11.4	32.8

*model open boundary

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.

V.4.2.4 ADCP verification of the Bass River system

An additional model verification check was possible by using collected ADCP velocity data to verify the performance of the Bass River 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 V.2. For the model ADCP verification, the River model was run for the period covered during the ADCP survey on October 13, 2004. Model flow rates were computed in RMA-2 at continuity lines (channel cross-sections) that correspond to two of the actual ADCP transects followed in each survey (i.e., inside the inlet at Wrinkle Point). The ADCP transect between the two jetty tips was not used for the model verification because there are not sufficient measurements from this transect to make a useful comparison to model output.

Comparisons of the measured and modeled volume flow rates in the Bass River system are shown in Figures V-22 and V-23. 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 both lines. For both transects the R^2 correlation coefficients between data and model results are equal or greater than 0.98. The RMS error computed from each transect is less than 670 ft3/sec, which is 8.2% of the maximum measured discharge rate. Correlation statistics between the modeled and measured flows for each ADCP transect are presented in Table V-7.

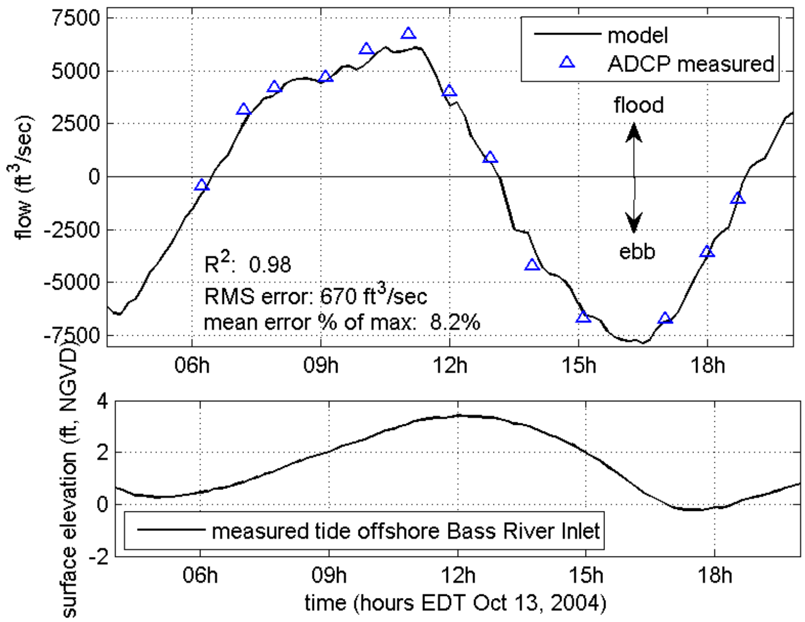


Figure V-22. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the Bass River Inlet, over a tidal cycle on October 13, 2004. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation offshore the River Inlet. (R^2 =0.98, E_{RMS}=670 ft^3/sec).

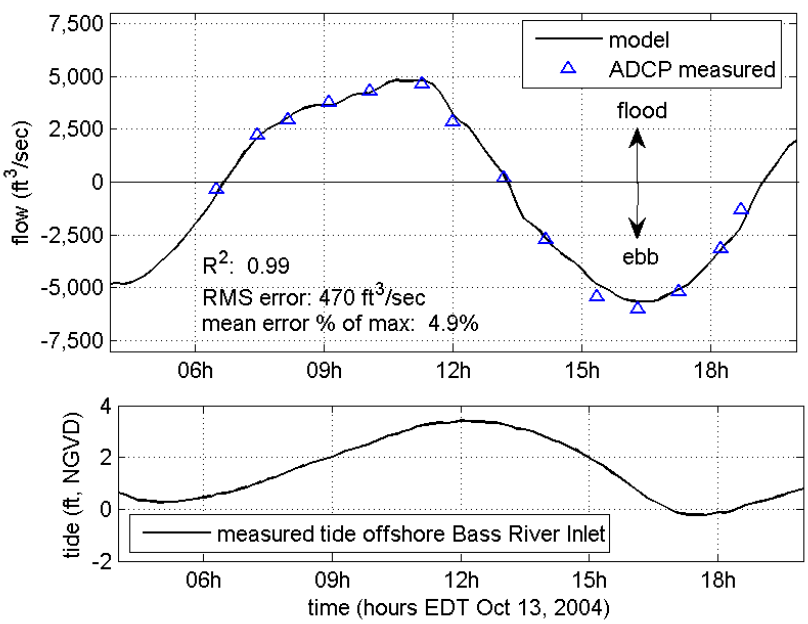


Figure V-23. Comparison of measured volume flow rates versus modeled flow rates (top plot) in Bass River at Wrinkle Point, over a tidal cycle on October 13, 2004. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation offshore the River inlet. (R^2=0.99, E_{RMS}=470 ft^3/sec).

Table V-7. Correlation statistics between modeled and measured total flow rates at the ADCP transects used in the model verification of the Bass River model.				
Transect	R^2 correlation	RMS error (ft^3/sec)	Max Error (ft^3/sec)	Min Error (ft^3/sec)
Bass River Inlet	0.98	670	1550	20
Bass River at Wrinkle Pt.	0.99	430	960	30

V.4.2.5 Model Circulation Characteristics

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

From the model run of the River, maximum flood velocities in the inlet channels are slightly larger than velocities during maximum ebb. Maximum depth-averaged flood velocities in the model are approximately 2.0 feet/sec at the narrows near the Bass River Golf Club , while maximum ebb velocities are about 1.7 feet/sec. Close-up views of model output are presented in Figure V-24 and V-25, 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-24), and for maximum flood velocities in Figure V-25.

In addition to depth-averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs at the two system inlets is seen in the plot of flow rates in Figure V-26. Maximum flow rates are roughly equal during flood and ebb tides. At the inlet, the modeled maximum flow rate during spring tides is 9,100 ft^3/sec

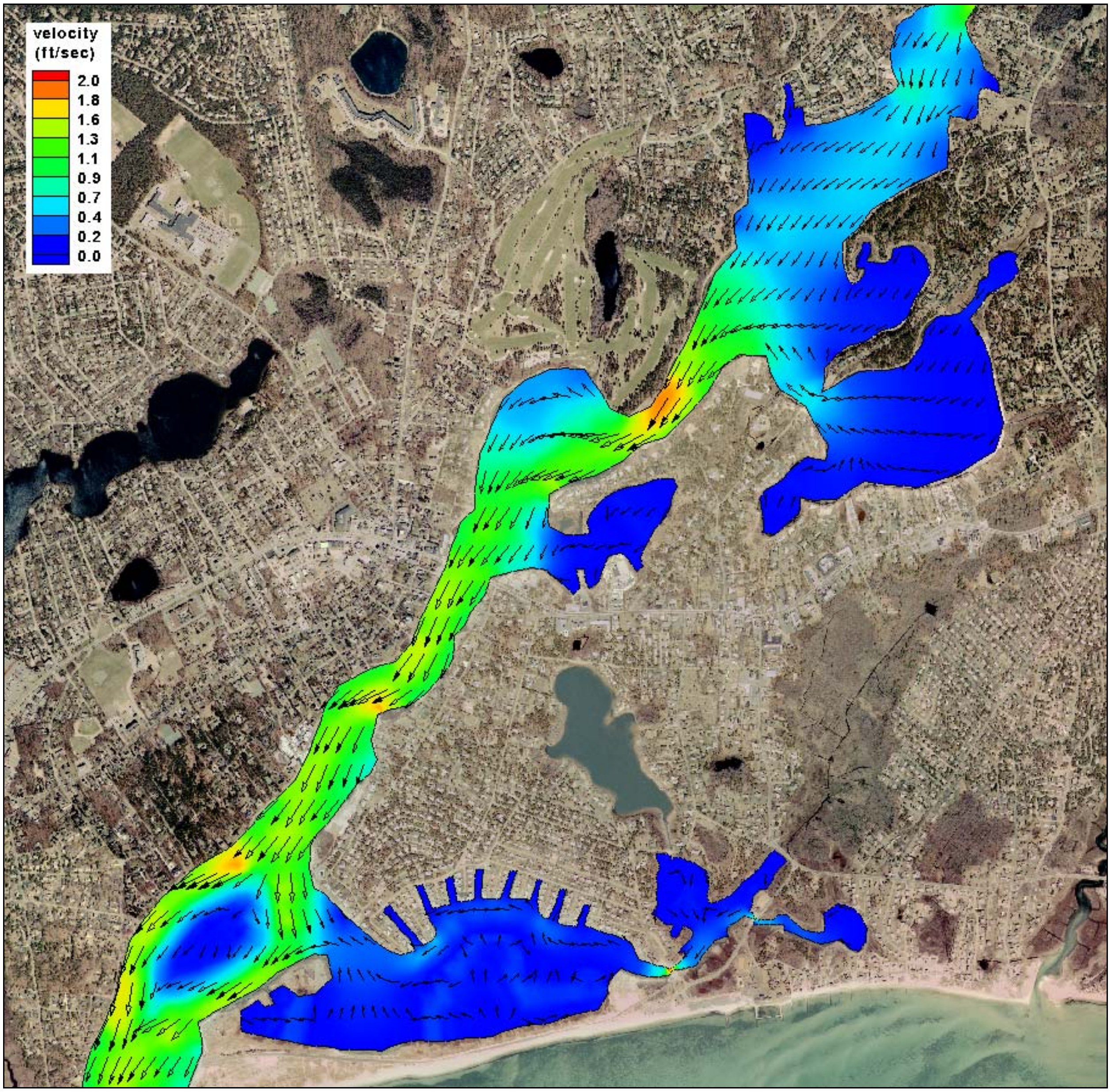


Figure V-24. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

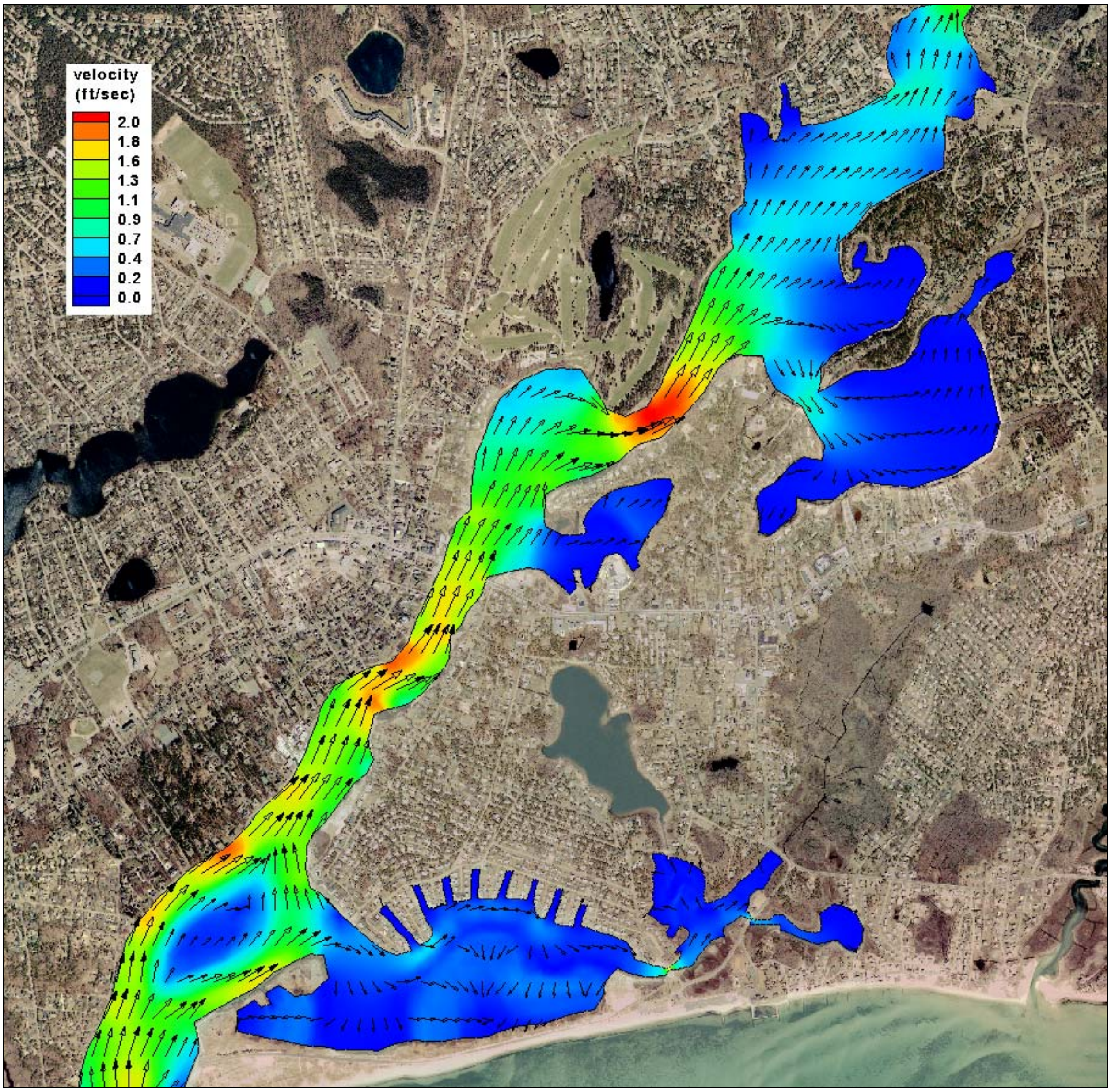


Figure V-25. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

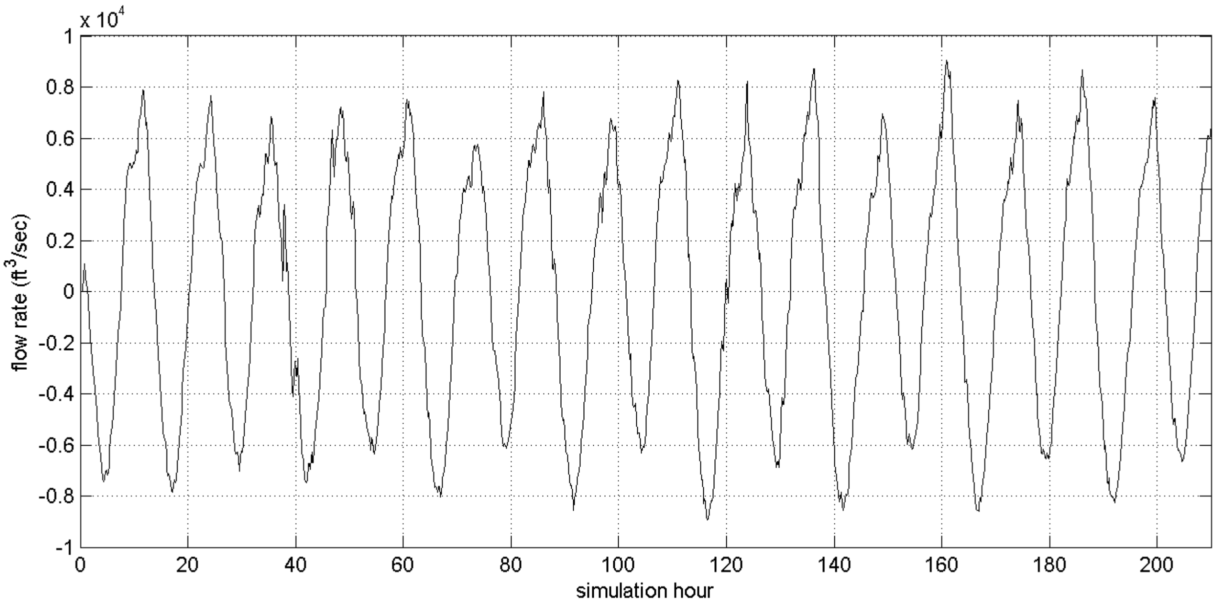


Figure V-26. Time variation of computed flow rates at the inlet to Bass River. Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

V.5 FLUSHING CHARACTERISTICS

Since the magnitude of freshwater inflow is much smaller in comparison to the tidal exchange through the inlet, the primary mechanism controlling estuarine water quality within the modeled Bass River system is tidal exchange. A rising tide offshore in Nantucket Sound creates a slope in water surface from the ocean into the upper-most reaches of the modeled system. Consequently, water flows into (floods) the system. Similarly, the estuary drains into the open waters of 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 River 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 Mill Pond as an example, the **system residence time** is the average time required for water to migrate from Mill Pond, through the mid-reach of the River, out through the inlet, and into Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Mill Pond to just Follins Pond (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 Bass River 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 is obtained from the calibrated hydrodynamic model in the following section of this report (Section VI) by extending the model to include pollutant/nutrient dispersion. The water quality model provides an additional valuable tool to evaluate the complex mechanisms governing estuarine water quality in the River 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 Bass River system, 2) Weir Creek, 3) Bass River north of Wrinkle Point, 4) Grand Cove, 5) Kelleys Bay to Mill Pond, 6) Dianahs Pond, 7) Mill Pond and Follins Pond and 8) Mill Pond. These system divisions follow the model material type areas designated in Figure V-10. Sub-embayment mean volumes and tide prisms are presented in Table V-8.

Residence times were averaged for the tidal cycles comprising a representative 7 lunar day period (14 tide cycles), and are listed in Table V-9. The modeled time period used to compute the flushing rates started September 17, 2004, similar to the model calibration period, and included the transition from neap to spring tide conditions. The RMA-2 model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume. Since the 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-8. Embayment mean volumes and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Bass River	249,984,000	103,395,000
Weir Creek	18,487,000	12,698,000
Bass River to Wrinkle Point	203,177,000	77,590,000
Grand Cove	24,582,000	11,999,000
Mill Pond to Kelleys Bay	90,465,000	23,957,000
Dinahs Pond	5,239,000	1,924,000
Mill Pond to Follins Pond	70,675,000	16,507,000
Mill Pond	9,180,000	502,000

Table V-9. Computed System and Local residence times for embayments in the Bass River system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Bass River	1.3	1.3
Weir Creek	10.2	0.8
Bass River to Wrinkle Point	1.7	1.4
Grand Cove	10.8	1.1
Mill Pond to Kelleys Bay	5.4	2.0
Dinahs Pond	67.2	1.4
Mill Pond to Follins Pond	7.8	2.2
Mill Pond	257.7	9.5

The computed flushing rates for the River system show that as a whole, the system flushes well. A flushing time of 1.3 days for the entire estuary shows that on average, water is resident in the system less than two days. System sub-embayments typically have local flushing times that are equal to or less than 2 days. Grand Cove has the shortest local flushing time, because this embayment has a small mean sub-embayment volume, relative to its tide prism. The highest local flushing rate for the system occurs in Mill Pond at the head of the River. For this sub-embayment, the local flushing rate is 9.5 days due to a small tide range.

The generally low local residence times in all areas of the Bass River 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 Kelleys Bay would likely be good as long as the water quality of the Harbor main basin was also good. Actual water quality would still also depend upon the total nutrient load to each embayment.

For the smaller sub-embayments of the Harbor 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 Bass River 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 Bass River System. These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayment

Extensive field measurements and hydrodynamic modeling of the embayment were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated model output representing the transport of water within the system embayment. Files of node locations and node connectivity for the RMA-2 model grid were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 11-tidal cycle period in September 2004. 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 Embayment

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

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Six years of data (collected between 2003 and 2008) were available for stations monitored by SMAST in the Bass River System.

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Bass River System. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of the Bass River System. Like RMA-2 numerical code, RMA-4 is a two-dimensional depth averaged finite element model capable of simulating time-dependent

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

Sub-Embayment	Monitoring station	2003 mean	2004 mean	2005 mean	2006 mean	2007 mean	2008 mean	mean	s.d. all data	N	model min	model max	model average
Mill Pond	BR-1	1.129	0.909	1.018	--	--	--	1.032	0.331	16	0.934	0.964	0.949
Follins Pond-Up	BR-2	0.930	0.569	0.740	0.893	1.084	--	0.804	0.233	25	0.729	0.769	0.751
Follins Pond-Lo	BR-3	0.845	0.605	0.761	0.838	0.949	1.002	0.807	0.227	27	0.723	0.766	0.747
Dinahs Pond	BR-4	0.727	0.814	0.924	0.811	0.959	0.919	0.843	0.181	31	0.664	0.722	0.696
Kelleys Pond	BR-5	0.663	0.789	0.860	0.734	0.881	0.900	0.790	0.137	30	0.589	0.753	0.695
Uppermost River	BR-6	0.684	0.864	0.841	0.739	0.834	0.832	0.796	0.162	31	0.464	0.727	0.607
Upper River	BR-7	0.570	0.372	0.471	0.621	0.804	--	0.529	0.177	26	0.422	0.629	0.523
Upper River	BR-8	0.460	0.346	0.349	0.605	0.736	0.659	0.485	0.171	30	0.407	0.591	0.493
Grand Cove	BR-9	0.588	0.403	0.471	0.628	0.763	0.738	0.564	0.164	30	0.492	0.548	0.520
Upper River	BR-10	0.423	0.436	0.343	0.481	0.694	0.676	0.479	0.157	30	0.343	0.550	0.438
Lower River	BR-11	0.393	0.329	0.310	0.423	0.443	--	0.367	0.096	51	0.316	0.509	0.389
Marsh-Lower	BR-12	0.402	0.398	0.380	0.435	0.440	0.496	0.418	0.075	26	0.323	0.461	0.372
Lower River	BR-13	0.414	0.349	0.321	0.383	0.411	0.384	0.370	0.088	58	0.306	0.440	0.340
Nearshore	BR-14	0.358	0.334	0.339	0.344	0.420	0.359	0.353	0.057	53	0.305	0.334	0.306

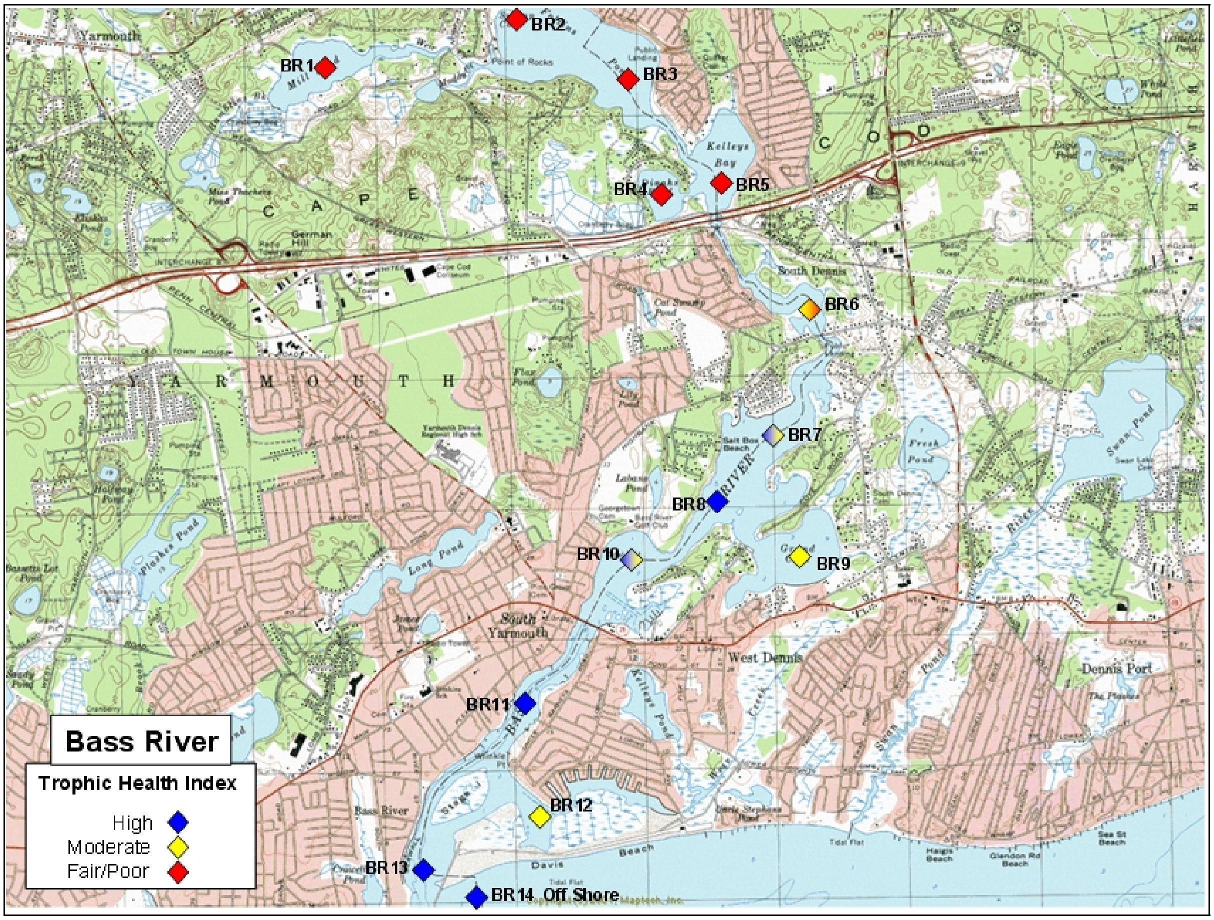


Figure VI-1. Estuarine water quality monitoring station locations in the Bass River System. Station labels correspond to those provided in Table VI-1.

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 (Ramsey *et al*., 2000); 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 SMAST and Cape Cod Commission watershed loading analysis (based on the USGS watersheds), as well as the measured bottom sediment nitrogen fluxes. Water column nitrogen measurements were utilized as model boundaries and as calibration data. Hydrodynamic model output (discussed in Section V) provided the remaining information (tides, currents, and bathymetry) needed to parameterize the water quality model of the system.

VI.2.1 Model Formulation

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

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

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

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

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

VI.2.2 Water Quality Model Setup

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

Based on groundwater recharge rates from the USGS, the hydrodynamic model was set-up to include ground water flowing into the system from the watersheds. Mill Pond Stream along with Mill Pond watersheds has groundwater flow rate into the system is 530,519ft^3/day (15,024 m^3/day), Follins Pond watershed has a groundwater flow rate of 494,890 ft^3/day (14,015 m^3/day), Dinah's Pond watershed has a groundwater flow rate of 63,949ft^3/day (1,811 m^3/day), Kelleys Bay has a groundwater flow rate of 270,591 ft^3/day (7,663 m^3/day), upper Bass River has a groundwater flow rate of 742,882 ft^3/day (21,038 m^3/day), Grand Cove has a groundwater flow rate of 101,626 ft^3/day (2,878 m^3/day), Horseshoe Foot Cove has a groundwater flow rate of 20,410 ft^3/day (578 m^3/day), Weir Creek Marsh has a groundwater flow rate of 156,747 ft^3/day (4,439 m^3/day), and lower Bass River watershed has a groundwater flow rate of 366,498 ft^3/day (10,379 m^3/day).

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

VI.2.3 Boundary Condition Specification

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

The loadings used to model present conditions in Bass River System are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverage, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, the benthic flux is generally positive with only two regions having a negative benthic flux (Dinah's Pond and lower Bass River). The upper portions of the Bass River system (Grand Cove, Kelley's Bay, and Follin's Pond) have benthic regeneration loading rates approaching and surpassing the watershed load.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary was 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 concentrations in Nantucket Sound were set at 0.305 mg/L, based on SMAST data from the Nantucket Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Nantucket Sound.

VI.2.4 Model Calibration

Calibration of the total nitrogen model proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled system by setting different values of *E* for each grid material type, as designated in Figure VI-2. Observed values of *E* (Fischer, *et al.*, 1979) vary between order 10 and order 1000 m^2/sec for riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents (from tides or atmospheric forcing). Generally, the relatively quiescent areas of Bass River (coves and marsh) require values of *E* that are lower compared to the riverine estuary systems evaluated by Fischer, *et al.*, (1979). Observed values of *E* in these calmer areas typically range between order 10 and order 0.001 m^2/sec (USACE, 2001). The final values of *E* used in each sub-embayment of the modeled systems are presented in Table VI-3.

These values were used to develop the “best-fit” total nitrogen model calibration. For the case of TN modeling, “best fit” can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Run Pond[1]	8.384	0.222	--
Bass River - Lower	36.764	2.995	-11.699
School Street Marsh	11.882	0.247	4.371
Bass River - Middle	67.674	3.841	29.285
Grand Cove	7.293	1.071	17.911
Dinah’s Pond	4.337	0.310	-2.016
Kelleys Bay	20.126	0.778	28.157
Follins Pond	34.121	2.658	39.596
Mill Pond and Stream	27.238	0.833	1.609

[1] The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.

Table VI-3. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for Bass River System.

Embayment Division	E m^2/sec
Nantucket Sound	125.0
Inlet	125.0
Bass River - Lower	125.0
Weir Creek - Lower	30.0
Weir Creek - Lower Marsh	3.0
Stage Creek Marsh	2.0
Bass River - Mid	125.0
Weir Creek - Upper Marsh	3.0
Grand Cove	22.0
Highbank Road Bridge	80.0
Bass River – Upper to RR Brg	100.0
Eastlover Pond	3.0
Eastlover Marsh	8.0
Rail Road Bridge	110.0
RR Brg to Route 6	110.0
Route 6 Bridge	110.0
Dinahs Pond	10.0
Kelley Bay Marsh	1.0
Follins Pond Marsh	1.0
Follins Pond	100.0
Mill Pond	40.0
North Dennis Road Culvert	40.0
Weir Creek Upper	9.0
Uncle Stephans Pond	9.0
Bass River Mid to Wrinkle Cove	125.0
Kelleys Bay	125.0

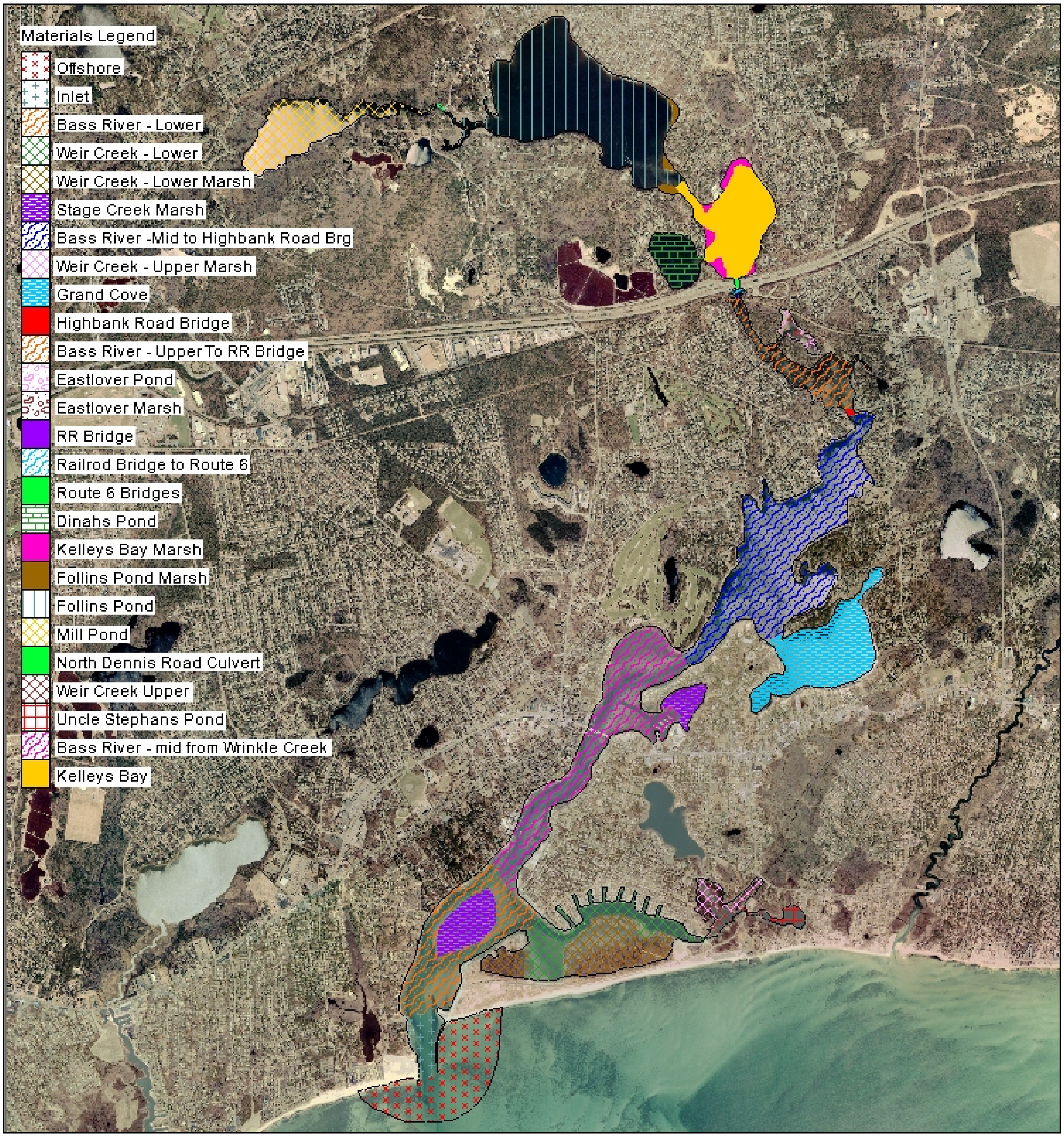


Figure VI-2. Map of Bass River water quality model longitudinal dispersion coefficients. Color patterns designate the different areas used to vary model dispersion coefficient values.

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

For model calibration, the mid-point between maximum modeled TN and average modeled TN was compared to mean measured TN data values, at each water-quality monitoring station. The calibration target would fall between the modeled mean and maximum

TN because the monitoring data are collected, as a rule, during mid ebb tide.

Also presented in this figure are unity plot comparisons of measured data versus modeled target values for the system. The model fit is exceptional for the Bass River System, with rms error of 0.06 mg/L and an R^2 correlation coefficient of 0.93.

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

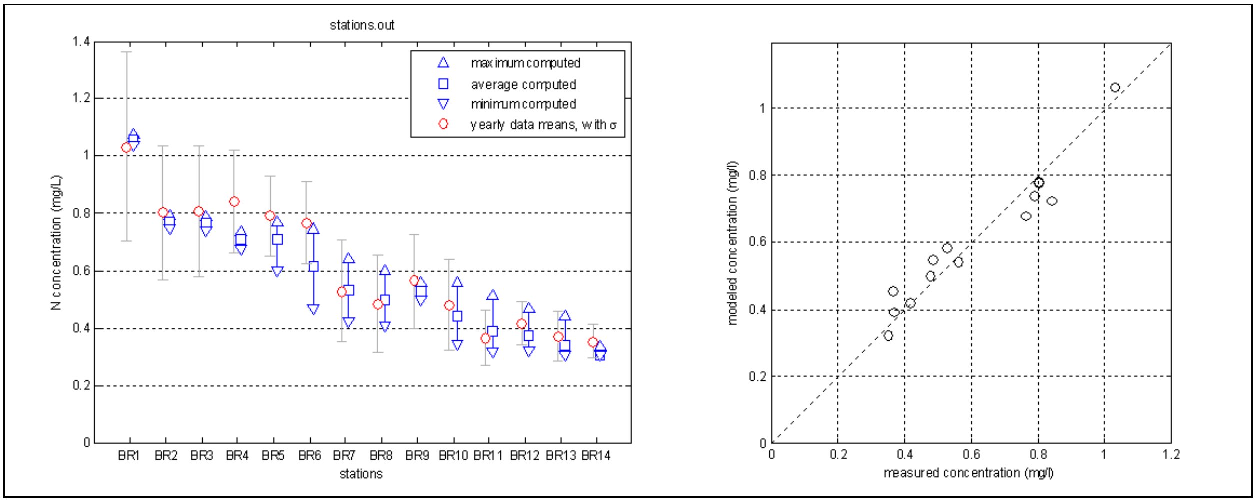


Figure VI-3. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Bass River System. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during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

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 Bass River System using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, and groundwater inputs. The open boundary salinity was set at 31.7 ppt. For groundwater inputs salinities were set at 0 ppt. The total groundwater input used for the model was 2,958,818 ft^3/day (83,792 m^3/day) distributed amongst the watersheds. Groundwater flows were distributed evenly within each watershed through grid cells that formed the perimeter along each watershed's land boundary.

Comparisons of modeled and measured salinities are presented in Figure VI-5, with contour plots of model output shown in Figure VI-6. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in Bass River System. The rms error of the models was 0.68 ppt, and correlation coefficient was 0.97. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical systems.

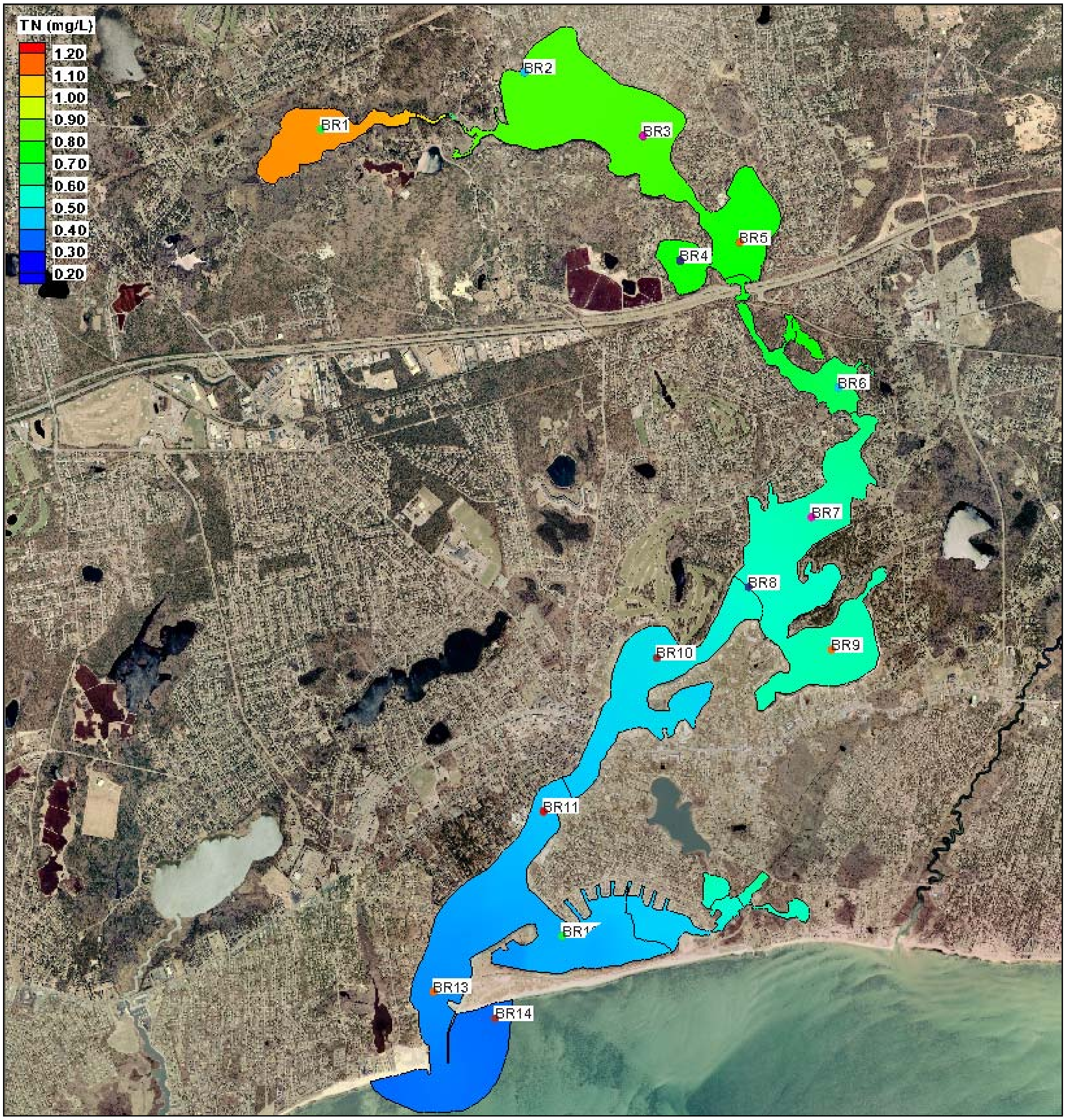


Figure VI-4. Contour plots of average total nitrogen concentrations from results of the present conditions loading scenario, for Bass River System. The approximate location of the sentinel threshold station for Bass River System (BR7) is shown.

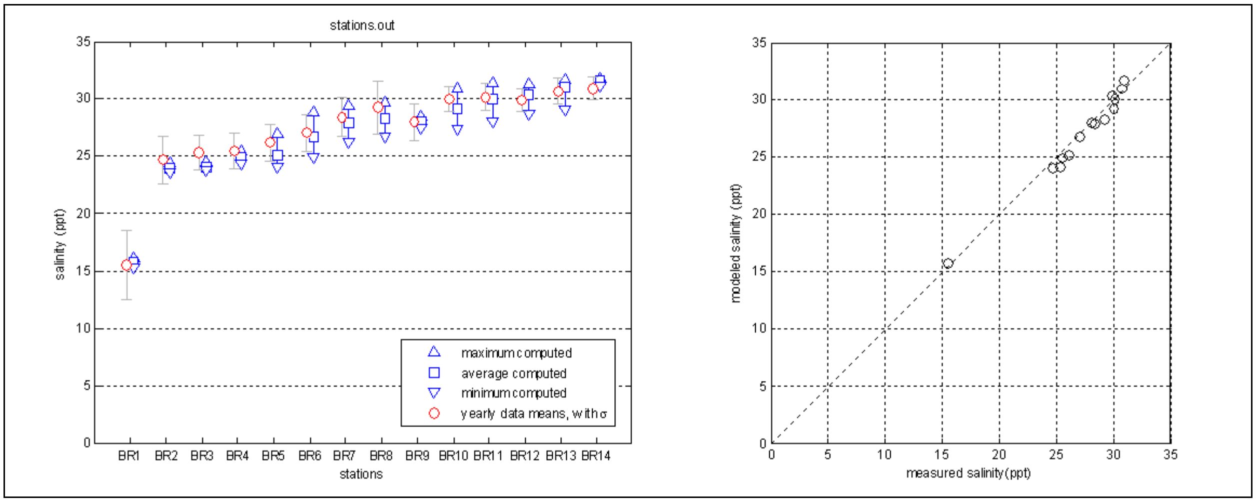


Figure VI-5. Comparison of measured and calibrated model output at stations in Bass River System. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle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

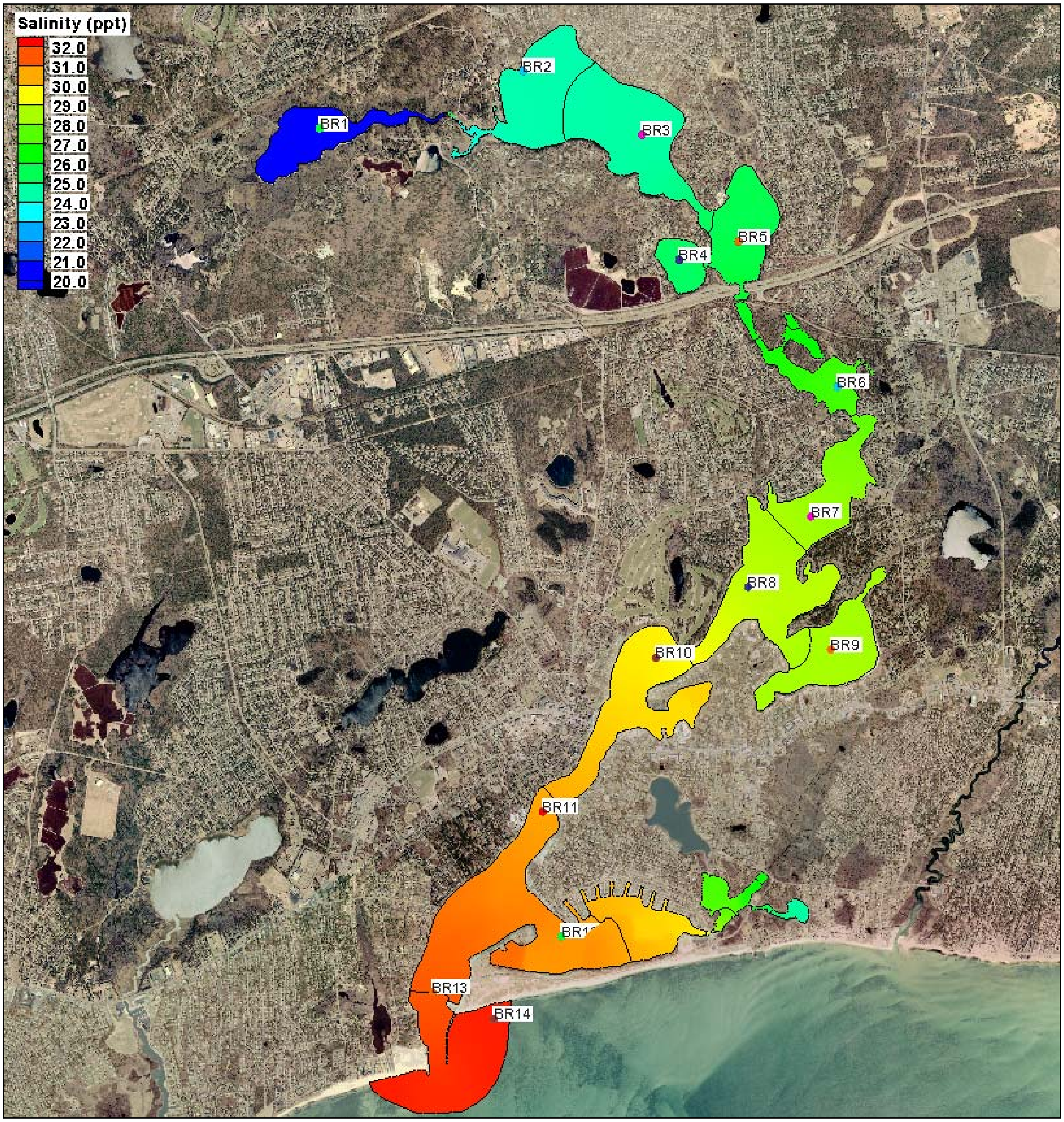


Figure VI-6. Contour plots of modeled salinity (ppt) in Bass River System.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

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

sub-embayment	present load (kg/day)	build out (kg/day)	build out % change	no load (kg/day)	no load % change
Run Pond	8.384	8.732	+4.2%	0.132	-98.4%
Bass River Lower	36.764	37.545	+2.1%	0.556	-98.5%
School Street Marsh	11.882	13.255	+11.6%	0.485	-95.9%
Bass River Middle	67.674	75.403	+11.4%	1.784	-97.4%
Grand Cove	7.293	7.833	+7.4%	0.323	-95.6%
Dinah's Pond	4.337	4.532	+4.5%	0.126	-97.1%
Kelleys Bay	20.126	22.918	+13.9%	0.627	-96.9%
Follins Pond	34.121	37.959	+11.2%	1.367	-96.0%
Mill Pond and Stream	27.238	30.104	+10.5%	1.019	-96.3%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be a increase in watershed nitrogen load to the Bass River as a result of potential future development. Specific watershed areas would experience large load increases, for example the loads to Kelleys Bay would increase 14% from the present day loading levels. For the no load scenarios, a majority of the load entering the watershed is removed; therefore, the load is significantly lower than existing conditions by over 90% overall.

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Bass River System, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Run Pond[1]	8.732	0.222	--
Bass River - Lower	37.545	2.995	-11.981
School Street Marsh	13.255	0.247	4.501
Bass River - Middle	75.403	3.841	30.308
Grand Cove	7.833	1.071	18.654
Dinah's Pond	4.532	0.310	-2.118
Kelleys Bay	22.918	0.778	29.373
Follins Pond	37.959	2.658	41.834
Mill Pond and Stream	30.104	0.833	1.740

[1] The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Bass River System was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. Total N concentrations increased the most in Mill Pond and Stream, with the water quality station in the pond showing a 18.2% increase in total nitrogen. The stations in Bass River show steady increase in nitrogen from the inlet to the head of the system. Color contours of model output for the build-out scenario are present in Figure VI-7. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-4, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Bass River System. Sentinel threshold station is in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Mill Pond	BR-1	0.949	1.121	+18.2%
Follins Pond-Up	BR-2	0.751	0.809	+7.7%
Follins Pond-Lo	BR-3	0.747	0.804	+7.7%
Dinahs Pond	BR-4	0.696	0.743	+6.8%
Kelleys Pond	BR-5	0.695	0.743	+6.9%
Uppermost River	BR-6	0.607	0.642	+5.9%
Upper River	**BR-7**	**0.523**	**0.548**	**+4.8%**
Upper River	BR-8	0.493	0.514	+4.3%
Grand Cove	BR-9	0.520	0.543	+4.4%
Upper River	BR-10	0.438	0.453	+3.4%
Lower River	BR-11	0.389	0.398	+2.3%
Marsh-Lower	BR-12	0.372	0.378	+1.9%
Lower River	BR-13	0.340	0.344	+1.1%
Nearshore	BR-14	0.306	0.307	+0.1%

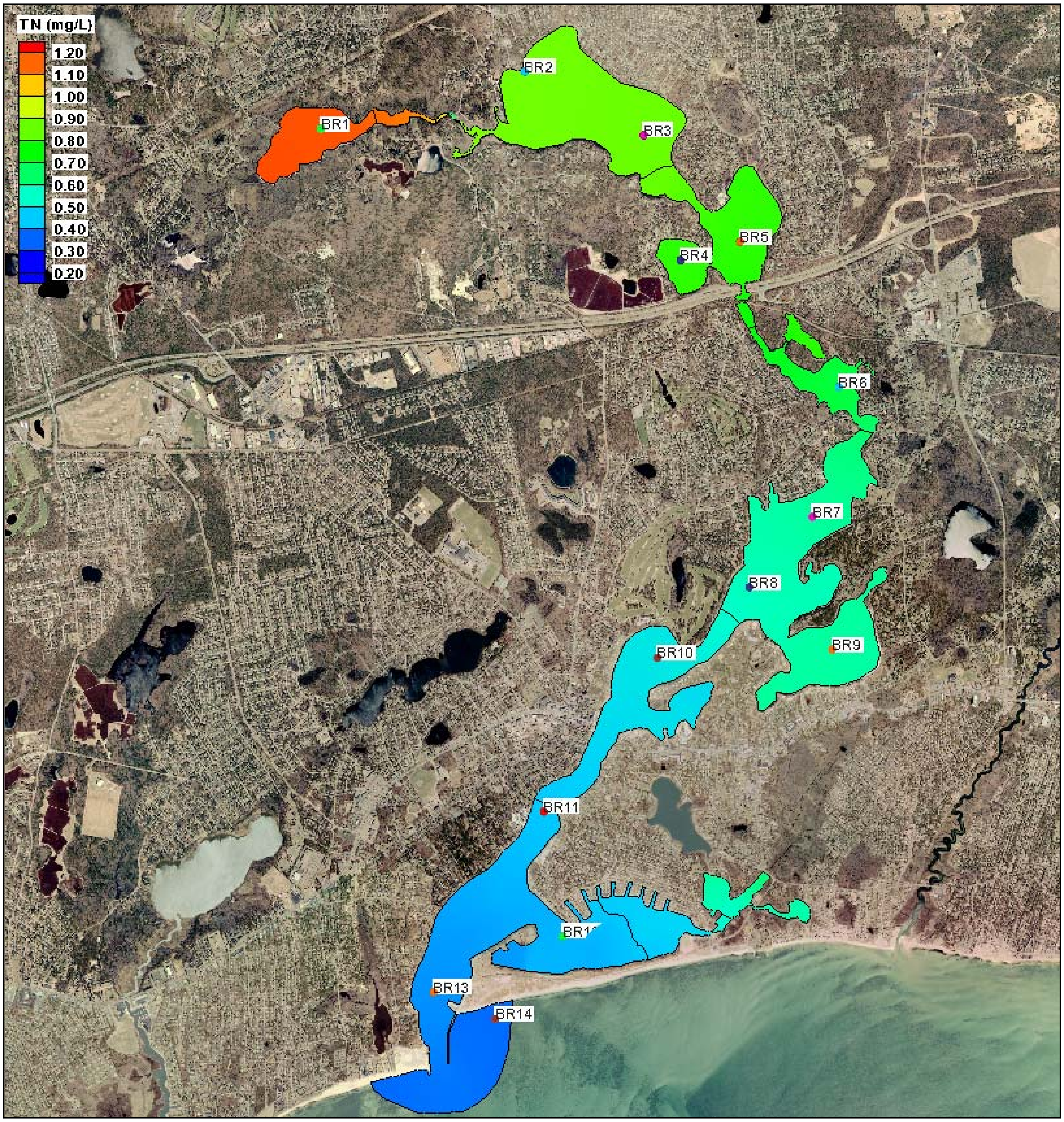


Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Bass River System, for projected build-out loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Bass River System (BR7) is shown.

VI.2.6.2 No Anthropogenic Load

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

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of Bass River System, with total watershed N loads, atmospheric N loads, and benthic flux			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Run Pond[1]	0.132	0.222	0.000
Bass River - Lower	0.556	2.995	-8.105
School Street Marsh	0.485	0.247	2.892
Bass River - Middle	1.784	3.841	19.182
Grand Cove	0.323	1.071	9.911
Dinah's Pond	0.126	0.310	-0.977
Kelleys Bay	0.627	0.778	15.591
Follins Pond	1.367	2.658	16.183
Mill Pond and Stream	1.019	0.833	0.422
[1] The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.			

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions ranging from less than 1% occurring just outside the inlet to Mill Pond and Stream with greater than 53% reduction in total nitrogen. Results for each system are shown pictorially in Figure VI-8.

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for the Bass River System. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold station is in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Mill Pond	BR-1	0.949	0.444	-53.2%
Follins Pond-Up	BR-2	0.751	0.421	-44.0%
Follins Pond-Lo	BR-3	0.747	0.420	-43.8%
Dinahs Pond	BR-4	0.696	0.404	-41.9%
Kelleys Pond	BR-5	0.695	0.407	-41.4%
Uppermost River	BR-6	0.607	0.383	-36.8%
Upper River	**BR-7**	**0.523**	**0.362**	**-30.8%**
Upper River	BR-8	0.493	0.354	-28.1%
Grand Cove	BR-9	0.520	0.365	-29.9%
Upper River	BR-10	0.438	0.339	-22.5%
Lower River	BR-11	0.389	0.326	-16.1%
Marsh-Lower	BR-12	0.372	0.321	-13.6%
Lower River	BR-13	0.340	0.313	-7.8%
Nearshore	BR-14	0.306	0.305	-0.4%

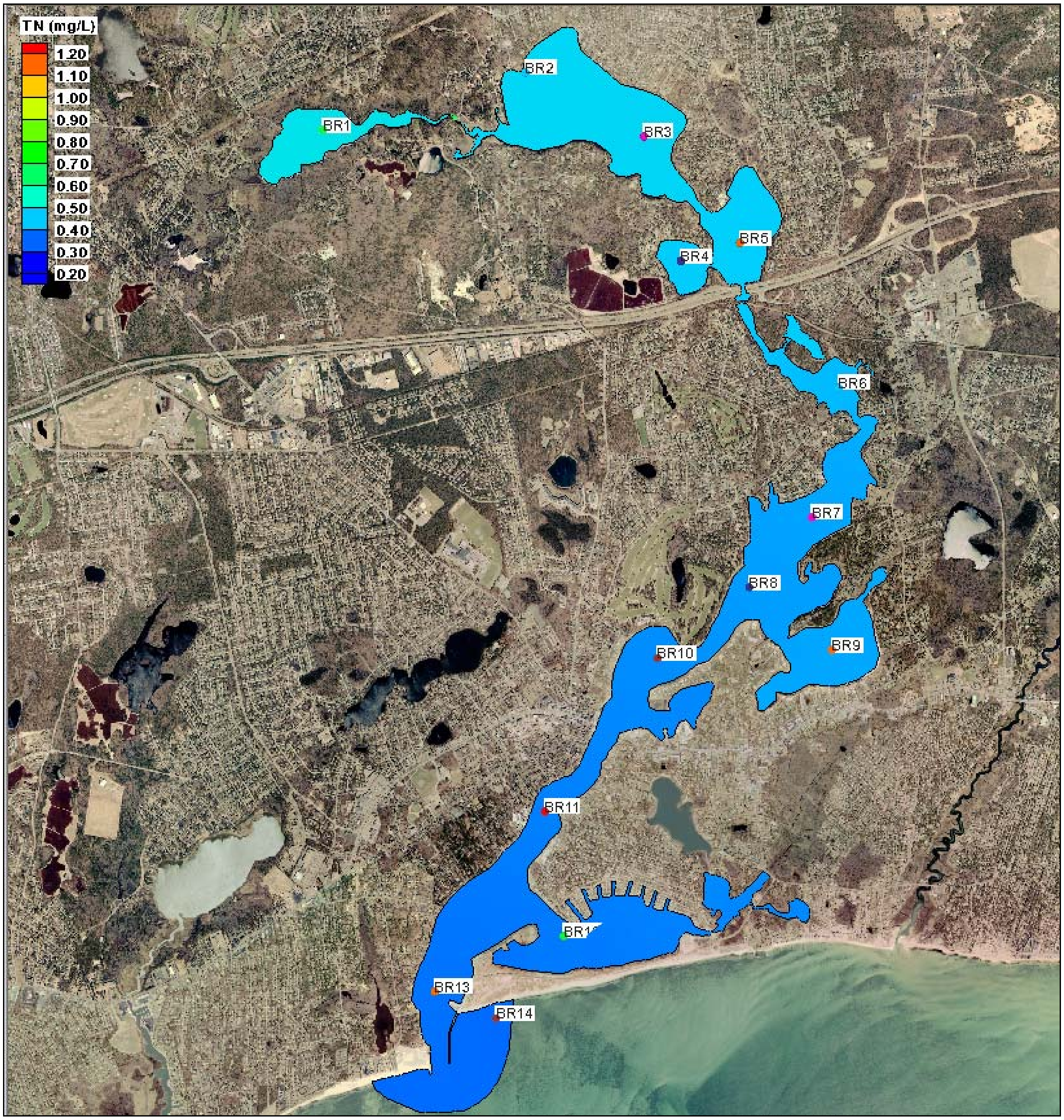


Figure VI-8. Contour plots of modeled total nitrogen concentrations (mg/L) in Bass River System, for no anthropogenic loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Bass River System (BR7) is shown.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gaged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Bass River embayment system (inclusive of Mill Pond at the head of the Bass River system) situated in the Towns of Yarmouth and Dennis, MA, the MEP assessment is based upon assorted data collected by the Towns of Yarmouth and Dennis as well as the MEP Technical Team and the MassDEP. Data from the water quality monitoring database was developed primarily by the Town of Yarmouth Water Quality Monitoring Program with some assistance from the Dennis Water District. The MassDEP and SMAST conducted surveys of eelgrass distribution and SMAST was entirely responsible for data collection on benthic animal communities, sediment characteristics, and dissolved oxygen records. Collection of these habitat related data sets was conducted during the summer and fall of 2005. These data were analyzed relative to recent changes within the watershed and have been used to form the basis of an assessment of this system's present nutrient related habitat quality. When coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, the full data set supports quantitative nitrogen threshold development for this system (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, assuming 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, the MEP approach focuses 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 at critical locations throughout the Bass River system (upper, central and lower sections) to capture gradients in oxygen depletion under worst case summertime conditions (Figure VII-2). Mooring deployments were conducted to record the frequency and duration of low oxygen conditions during the critical summer period and also to collect supporting information on phytoplankton biomass as Chlorophyll-a.

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 Bass River system was conducted for comparison to historic records as available (MassDEP Eelgrass Mapping Program, C. Costello). 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. However, it should be noted that certain systems like Lewis Pond in the adjacent Parkers River system, that are salt marsh dominated, generally do not support eelgrass for reasons related to ecosystem structure. Within the Bass River system, temporal changes in eelgrass distribution provide a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing from inlet management; nitrogen management) in nutrient enrichment.

In areas that do not support eelgrass beds (be it for natural or anthropogenic reasons), benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stress 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, USEPA[2] suggests that the chronic protective oxygen level to support growth of estuarine animals is 4.8 mg L^{-1}, with a limit for survival of juvenile and adult organisms of 2.3 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 Bass River Embayment 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 (see Figure VII-1 for example). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Bass River Embayment System (Figure VII-2). Measurements were made close to the

[2] USEPA 2000. Ambient Aquatic Life Water Quality Criteria for Dissolved Oxygen (Saltwater): Cape Cod to Cape Hatteras (133 p.).

sediment surface so as to quantify the oxygen environment affecting benthic animal communities. 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 deployment of ~30 days within the interval from July through mid-September. All of the mooring data from the Bass River embayment system was collected during the summer of 2005. Oxygen data from the Dennis/Yarmouth Water Quality Monitoring Program was used to provide inter-annual information on oxygen levels for integration with the detailed 2005 time-series data.

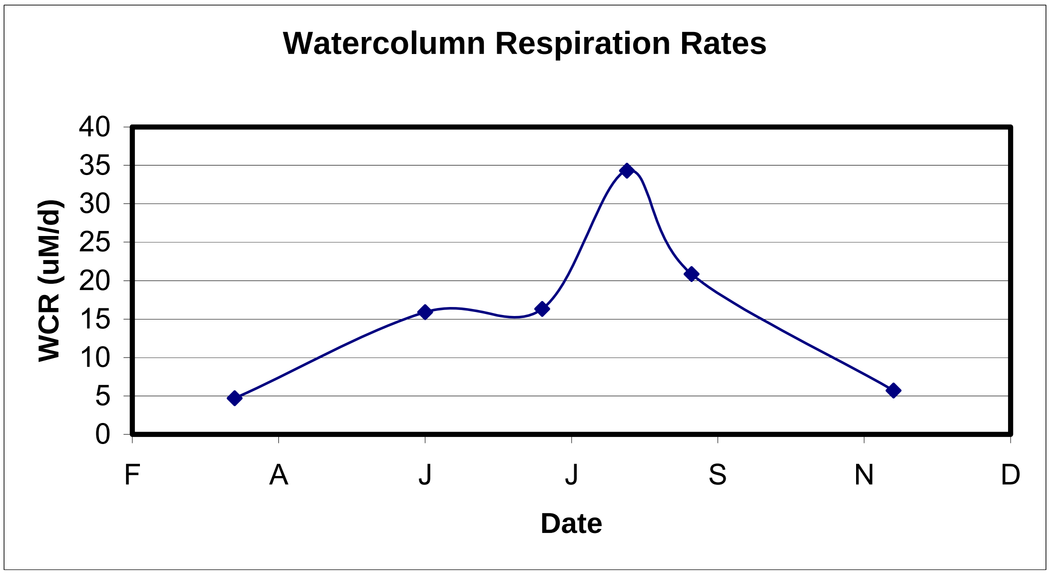


Figure VII-1. As an example, average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System are presented (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter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

Similar to other embayments in southeastern Massachusetts, the Bass River Estuary evaluated in this MEP assessment showed high frequency variation related primarily 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. Oxygen excursions result from oxygen consumption (night) and production (day) primarily by phytoplankton within the estuarine waters. Additional oxygen uptake results from the microbial decay of organic matter, which in the case of the Bass River Estuary is mainly from phytoplankton in the watercolumn and settling to bottom sediments. Oxygen levels in estuaries typically cannot be managed directly, but rather through management of nitrogen levels and mitigation of any direct organic matter inputs (e.g. outfalls).

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. However, the large number of oxygen samplings by the Yarmouth Water Quality Monitoring Program from 2002-2008 was sufficient to capture the minimum oxygen levels measured by the detailed time-series measurements. For example, the continuous oxygen record and the monitoring program yielded similar oxygen levels to the Water Quality Monitoring Program both

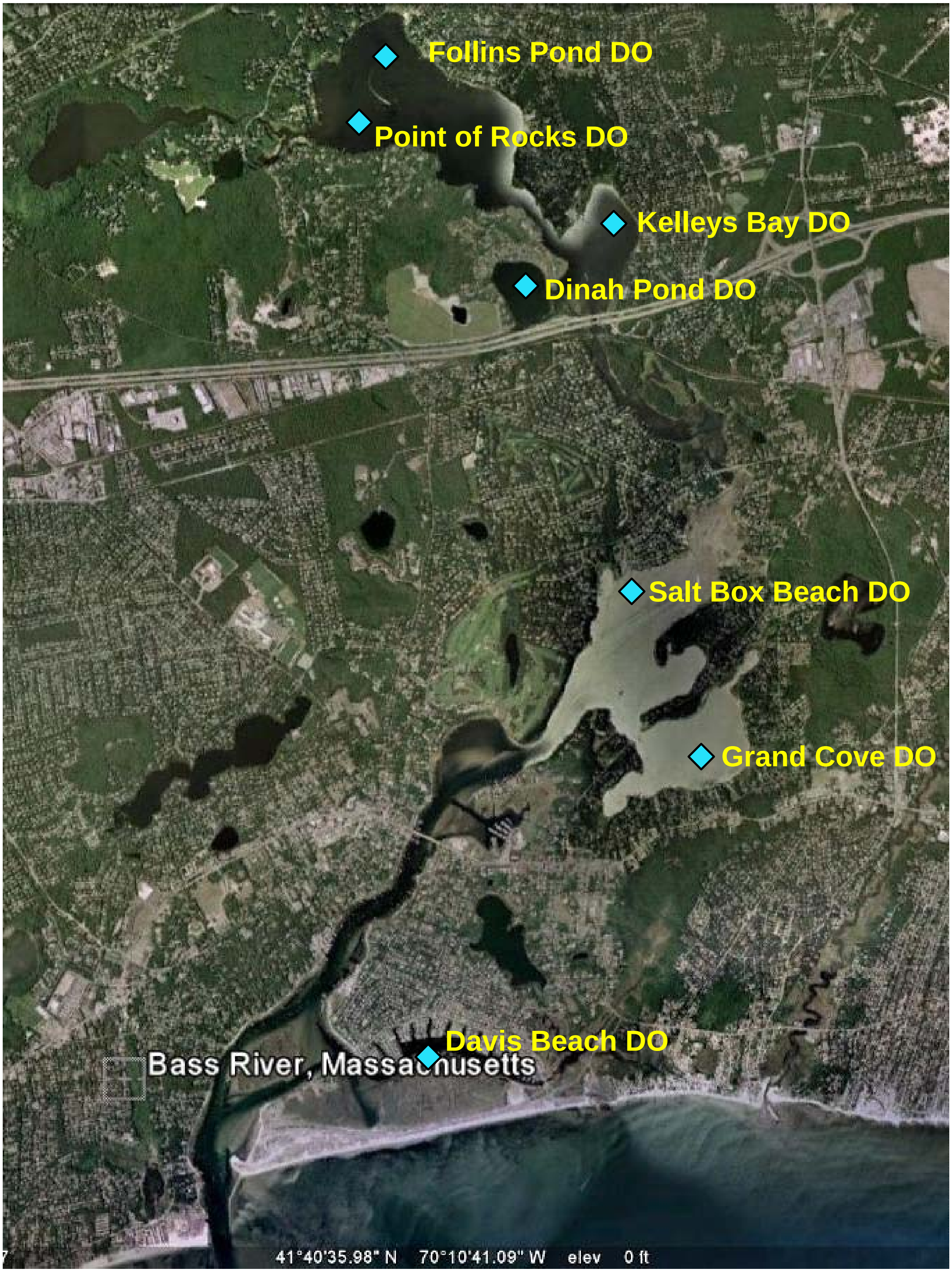


Figure VII-2. Location of time-series oxygen mooring, deployed summer 2005, in the Bass River Embayment System within the Towns of Yarmouth and Dennis.

showing minima of ~ 4mg L^{-1} in the mid river reach (Salt Box Beach, Figure VII-1) and <1 mg L-1 in Follins Pond, although in Grand Cove the time series captured lower oxygen levels than the monitoring program (3 mg L^{-1} versus 4.1 mg L^{-1}). The ability to better capture minimum oxygen levels by continuous measurement appears to be generally true for Cape Cod estuaries where periodic monitoring of oxygen and time-series oxygen recordings generally yield similar results, except that periodic low oxygen tends to be better captured in the continuous recordings. The agreement between the time-series oxygen mooring and the monitoring program at the 7 mooring locations throughout the Bass River System indicates that the monitoring data for the other basins within this estuary can be used in the assessment of those areas.

Dissolved oxygen and chlorophyll-a records were examined both for temporal trends and to determine the percent of the 27 day deployment period that these parameters were below or 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 throughout Bass River indicate high levels of nutrient enrichment and impaired habitat quality, particularly in the upper portions of the system above Route 6 (Figures VII-9, 11, 13, 15). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll-a levels) indicative of nitrogen enrichment and eutrophication of this estuarine basin. The large daily excursions in oxygen concentration in Follins Pond, Kelleys Bay and Dinahs Pond also indicate significant organic matter enrichment.

Table VII-1. Duration (percent of deployment time) that bottom water dissolved oxygen levels were below various benchmark levels within the lower, central and upper portions of the overall Bass River system. "Mean" represents the average duration of each event over the benchmark level and "S.D." its standard deviation. Data collected by the Coastal Systems Program, SMAST.

Mooring Location	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Davis Beach	**06/29/05**	**07/26/05**	**27.0**	**61%**	**30%**	**7%**	**0.0%**
			Mean	0.33	0.19	0.10	N/A
			Min	0.01	0.01	0.01	0.00
			Max	1.44	1.36	0.38	0.00
			S.D.	0.25	0.21	0.10	N/A
Grand Cove	**06/29/05**	**07/26/05**	**27.0**	**38%**	**18%**	**6%**	**0.1%**
			Mean	0.33	0.21	0.10	0.03
			Min	0.01	0.01	0.02	0.03
			Max	2.95	0.72	0.28	0.03
			S.D.	0.53	0.21	0.06	N/A
Salt Box Beach	**06/29/05**	**07/26/05**	**27.0**	**24%**	**7%**	**0.4%**	**0.0%**
			Mean	0.27	0.16	0.05	N/A
			Min	0.03	0.01	0.05	0.00
			Max	0.95	0.58	0.05	0.00
			S.D.	0.26	0.18	0.00	N/A
Kelleys Bay	**06/30/05**	**07/26/05**	**26.1**	**43%**	**23%**	**5%**	**0.2%**
			Mean	0.31	0.27	0.14	0.04
			Min	0.02	0.01	0.03	0.04
			Max	0.92	0.73	0.35	0.04
			S.D.	0.25	0.19	0.12	N/A
Dinahs Pond	**06/29/05**	**07/26/05**	**26.9**	**44%**	**32%**	**21%**	**12.5%**
			Mean	0.41	0.36	0.31	0.28
			Min	0.03	0.02	0.06	0.13
			Max	1.49	1.45	0.74	0.55
			S.D.	0.33	0.31	0.20	0.14
Follins Pond	**06/29/05**	**07/26/05**	**27.1**	**38%**	**24%**	**17%**	**9.7%**
			Mean	0.32	0.27	0.30	0.19
			Min	0.02	0.01	0.01	0.01
			Max	1.42	1.00	0.92	0.47
			S.D.	0.35	0.29	0.30	0.15
Point of Rocks	**06/29/05**	**07/26/05**	**27.0**	**40%**	**27%**	**18%**	**11%**
			Mean	0.47	0.44	0.53	0.34
			Min	0.04	0.01	0.06	0.03
			Max	2.83	2.74	1.74	1.56
			S.D.	0.63	0.69	0.49	0.47

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

Mooring Location	Start Date	End Date	Total Deployment (Days)	>5 ug/L Duration (Days)	>10 ug/L Duration (Days)	>15 ug/L Duration (Days)	>20 ug/L Duration (Days)	>25 ug/L Duration (Days)
Davis Beach	**6/29/2005**	**7/26/2005**	**26.96**	**72%**	**24%**	**4%**	**0%**	**0%**
Mean Chl Value = 7.6 ug/L			Mean	0.98	0.28	0.14	0.13	N/A
			Min	0.04	0.04	0.04	0.13	0.00
			Max	11.33	0.92	0.29	0.13	0.00
			S.D.	2.46	0.22	0.09	N/A	N/A
Grand Cove	**6/29/2005**	**7/26/2005**	**26.95**	**74%**	**18%**	**7%**	**3%**	**0%**
Mean Chl Value = 7.7 ug/L			Mean	0.54	0.20	0.18	0.10	0.04
			Min	0.04	0.04	0.04	0.04	0.04
			Max	3.67	0.67	0.54	0.17	0.04
			S.D.	0.63	0.18	0.14	0.05	N/A
Salt Box Beach	**6/29/2005**	**7/26/2005**	**26.96**	**81%**	**48%**	**20%**	**7%**	**3%**
Mean Chl Value = 10.5 ug/L			Mean	1.04	0.37	0.18	0.15	0.10
			Min	0.04	0.04	0.04	0.04	0.04
			Max	5.71	0.88	0.42	0.29	0.21
			S.D.	1.44	0.20	0.11	0.08	0.08
Kelleys Bay	**6/30/2005**	**7/26/2005**	**26.14**	**87%**	**42%**	**11%**	**2%**	**1%**
Mean Chl Value = 9.7 ug/L			Mean	1.03	0.24	0.17	0.14	0.10
			Min	0.13	0.04	0.04	0.08	0.08
			Max	3.54	0.63	0.42	0.17	0.13
			S.D.	0.79	0.15	0.11	0.04	0.03
Dinahs Pond	**6/29/2005**	**7/26/2005**	**26.94**	**30%**	**7%**	**2%**	**0%**	**0%**
Mean Chl Value = 5.2 ug/L			Mean	0.38	0.13	0.06	N/A	N/A
			Min	0.04	0.04	0.04	0.00	0.00
			Max	2.88	0.25	0.08	0.00	0.00
			S.D.	0.68	0.06	0.02	N/A	N/A
Follins Pond	**6/29/2005**	**7/26/2005**	**27.05**	**76%**	**35%**	**16%**	**5%**	**2%**
Mean Chl Value = 9.3 ug/L			Mean	0.54	0.19	0.12	0.12	0.10
			Min	0.04	0.04	0.04	0.04	0.04
			Max	3.67	0.63	0.29	0.21	0.17
			S.D.	0.72	0.15	0.09	0.07	0.06
Point of Rocks	**6/29/2005**	**7/26/2005**	**27.04**	**23%**	**3%**	**1%**	**0%**	**0%**
Mean Chl Value = 3.8 ug/L			Mean	0.17	0.13	0.17	N/A	N/A
			Min	0.04	0.04	0.17	0.00	0.00
			Max	0.92	0.25	0.17	0.00	0.00
			S.D.	0.18	0.09	N/A	N/A	N/A

The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). The clear evidence of oxygen levels above atmospheric equilibration in Kelleys Bay and Follins Pond is further evidence of nitrogen enrichment at a level consistent with habitat degradation.

Generally, the dissolved oxygen records throughout the Bass River Estuary showed moderate to severe depletions of oxygen (relative to the basin type) during the critical summer period. The greatest oxygen depletions were generally associated with the upper portions of the system above the Route 6 bridge crossing. The continuous D.O. records indicate that the upper region of the Bass River Embayment System, defined by the open water portion that is

Follins Pond and Kelleys Bay, shows regular oxygen depletion (below 6.0 mg/L) during summer with periodic depletions below 3.0 mg/L, consistent with its nitrogen and organic matter rich waters (Table VII-1, Figure VII-11, 13 and 15). The Bass River system also shows moderate oxygen depletions in the sub-basin of Grand Cove below the Rt. 6 Bridge (< 4.0 mg L-1) reach, as well as moderate to high chlorophyll levels. However, it is virtually certain that a portion of this enrichment stems from the River's role in transporting high nutrient, high phytoplankton, low oxygen waters from the upper most reach of the system (Mill Pond) and Follins Pond and Kelleys Bay to the lower River basins and finally Nantucket Sound on the ebb tide. The high turnover of water in the lower portion of the Bass River system closest to the inlet reduces that area's ability to build up nutrients. In addition, the inflow of high quality water from Nantucket Sound on the flooding tide, results in a lower section with relatively high water quality for a portion of the flood tide period. Even so, the mooring (Davis Beach) deployed in the small tidal tributary heading east from the inlet and separated from Nantucket Sound by a barrier beach showed that oxygen levels during the 27 day deployment period did regularly dip below the 4.0 mg L-1 oxygen threshold, an indication that this area is affected to a degree by nutrient enrichment, although this basin is significantly influenced by upgradient wetlands which naturally enhance nutrient levels and low oxygen within this basin. The result is that this lower basin is operating both as an estuarine basin and a salt marsh pond and management actions need to account for this intermediate ecological status.

Davis Beach – Bass River **(Figures VII-3 and VII-4):**

The Davis Beach mooring location was centrally located within the tidal tributary in the lower-most section of the Bass River system close to the inlet and the Nantucket Sound source water (Figure VII-2). Generally, the daily excursions in oxygen levels at this location were modest, ranging from levels only slightly in excess of air equilibration to periods of oxygen levels <4.0 mg L-1. Oxygen varied primarily with light (diurnal cycle) and the tides. As commonly occurs in Cape Cod waters, lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed when low tide occurred at the end of the photocycle (ca. 1500 hrs), thus integrating the effects of night-time oxygen uptake and low oxygen waters ebbing from the upper reaches of Bass River. Oxygen levels frequently declined to <6 mg L^{-1} and even <4 mg L^{-1}, with 7% of the 27 day record showing values below 4 mg L^{-1} (Table VII-1). Oxygen levels rarely climbed to above 8 mg L^{-1}, consistent with the modest phytoplankton biomass. Consistent with the oxygen data, chlorophyll-a was initially low (between 5 and 10 ug L^{-1}) but gradually increased during the deployment period to between 10 and 15 ug L^{-1}, indicative of greater phytoplankton production as the summer progressed. Towards the end of the deployment period chlorophyll-a levels declined to 5 - 10 ug L^{-1}. At the Davis Beach mooring location, chlorophyll-a levels infrequently exceeded the 15 ug L^{-1} benchmark 4% of the time (Table VII-2, Figure VII-4). Average chlorophyll levels over 10 ug L^{-1} have been used to indicate eutrophic conditions in embayments, the average at this location was 7.6 ug L^{-1}.

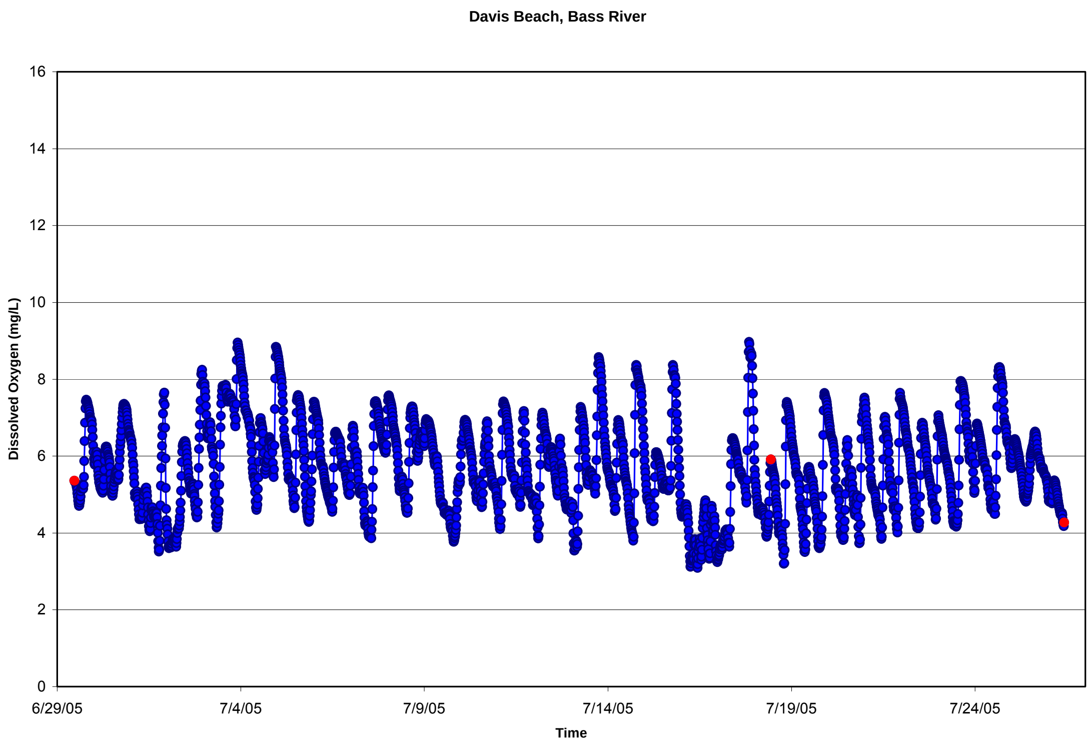


Figure VII-3. Bottom water record of dissolved oxygen at the Davis Beach - Bass River station, Summer 2005. Calibration samples represented as red dots.

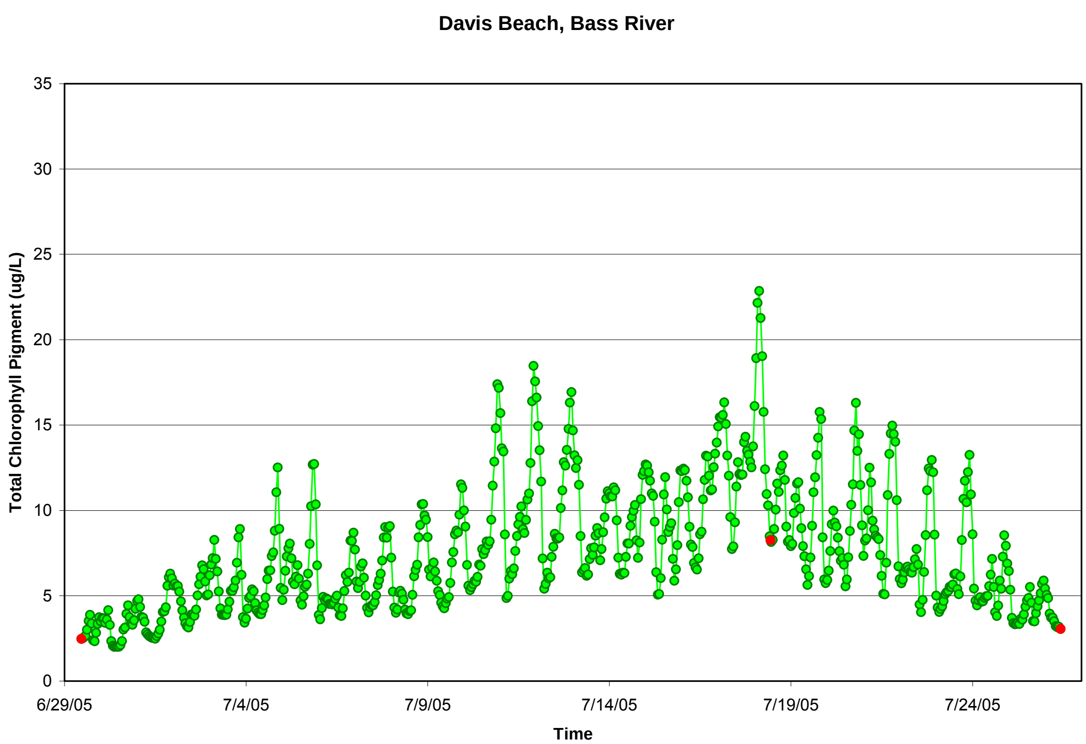


Figure VII-4. Bottom water record of Chlorophyll-*a* at the Davis Beach - Bass River mooring location, Summer 2005. Calibration samples represented as red dots.

***Grand Cove – Bass River* (Figures VII-5 and VII-6):**

The Grand Cove mooring location was centrally located within this tributary sub-basin within the middle reach of the Bass River System (Figure VII-2). High frequency data from Grand Cove showed clear daily excursions. The oxygen excursions resulted mainly from oxygen uptake associated with the diurnal cycle as well as tidal influence. Lowest dissolved oxygen was typically observed at high tide in the early morning. Highest dissolved oxygen was observed when low tide occurred at the end of the photocycle (ca. 1500 hrs) for reasons noted above. Dissolved oxygen occasionally declined to <4 mg L^{-1} (6 percent of the total deployment period) but never was it observed to drop below <3 mg L-1 (Figure VII-5, Table VII-1). The occurrence of oxygen levels over air saturation was pronounced. Oxygen levels regularly exceeded 10 mg L^{-1} and periodically exceeded 12 and even 14 mg L^{-1}. Given the larger excursions in dissolved oxygen and the decrease in concentrations to below 4 mg L^{-1}, the data indicate the effects of nitrogen enrichment of this basin. Consistent with the oxygen data, chlorophyll-a was initially low (between 5 and 10 ug L^{-1}) but increased during the early part of the deployment period to high levels (between 20 and 25 ug L^{-1}), indicative of a bloom and greater phytoplankton production as the summer progressed. Approximately mid way through the deployment period chlorophyll-a levels declined to between 5 and 10 ug L^{-1}, as the bloom dissipated. Although reduced, the daily excursion of dissolved oxygen remain relatively large possibly due to the presence of macroalgal accumulations in Grand Cove as observed by MEP survey teams, further evidence of nitrogen enrichment of this basin. At the Grand Cove mooring location, chlorophyll-a levels exceeded the 15 ug L^{-1} benchmark 7% of the time (Table VII-2, Figure VII-6). The overall average chlorophyll level was 7.7 ug L^{-1} over the deployment and 7.6 by the Water Quality Monitoring Program.

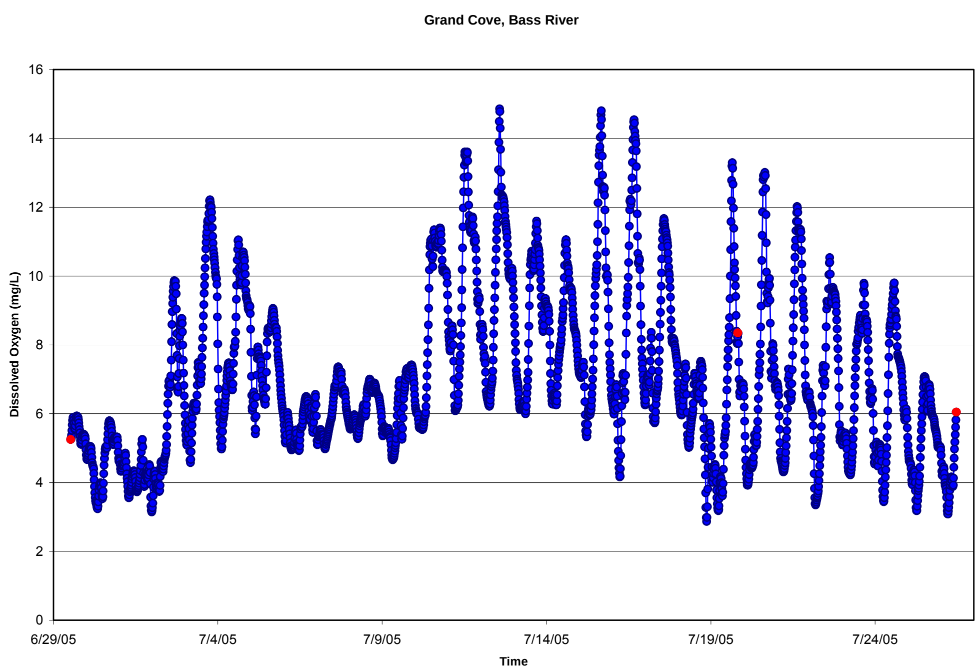


Figure VII-5. Bottom water record of dissolved oxygen at the Grand Cove – Bass River station, Summer 2005. Calibration samples represented as red dots.

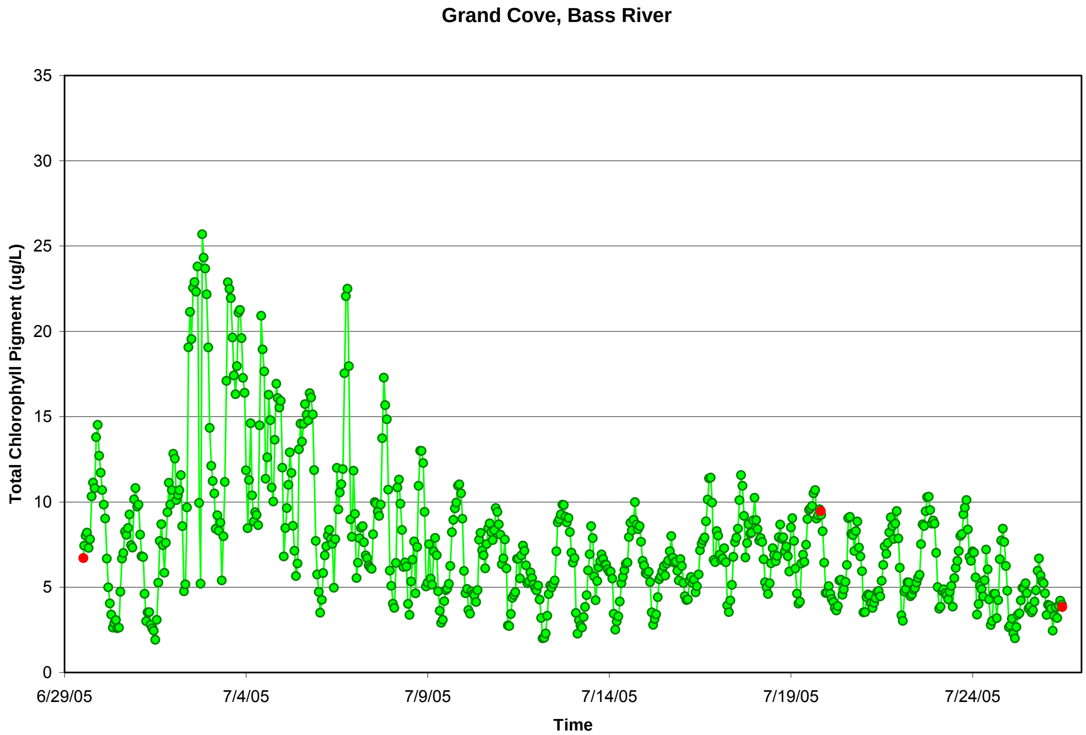


Figure VII-6. Bottom water record of Chlorophyll-*a* at the Grand Cove – Bass River mooring location, Summer 2005. Calibration samples represented as red dots.

***Salt Box Beach – Bass River* (Figures VII-7 and VII-8):**

The Salt Box Beach mooring was located within one of the main basins of open water in the Bass River system. This basin is situated immediately to the north of Grand Cove between Route 6 and Route 28 (Figure VII-2). As seen at the other mooring locations, large daily excursions in oxygen levels were observed at this location, ranging from levels in excess of air saturation to periods of oxygen depletion. However, levels of dissolved oxygen seldom dropped to 4 mg L^{-1} (Figure VII-7, Table VII-1). The occurrence of "excess" oxygen was more pronounced when compared to the Grand Cove and Davis Beach mooring location situated lower in the Bass River system. This is likely due to the high measured chlorophyll-a levels. Oxygen levels regularly exceeded 12 mg L^{-1} and periodically exceeded 16 and even 18 mg L^{-1}. The larger excursions in dissolved oxygen and the high measured chlorophyll concentrations are consistent with nitrogen enrichment and habitat impairment. Consistent with the large excursions observed in the oxygen data, chlorophyll-a concentrations were generally high during most of the deployment period (between 15 and 20 ug L^{-1}), indicative of significant phytoplankton production during the summer deployment period. At the Salt Box Beach mooring location, chlorophyll-a levels exceeded the 15 ug L^{-1} benchmark 20% of the time (Table VII-2, Figure VII-8). Average chlorophyll levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

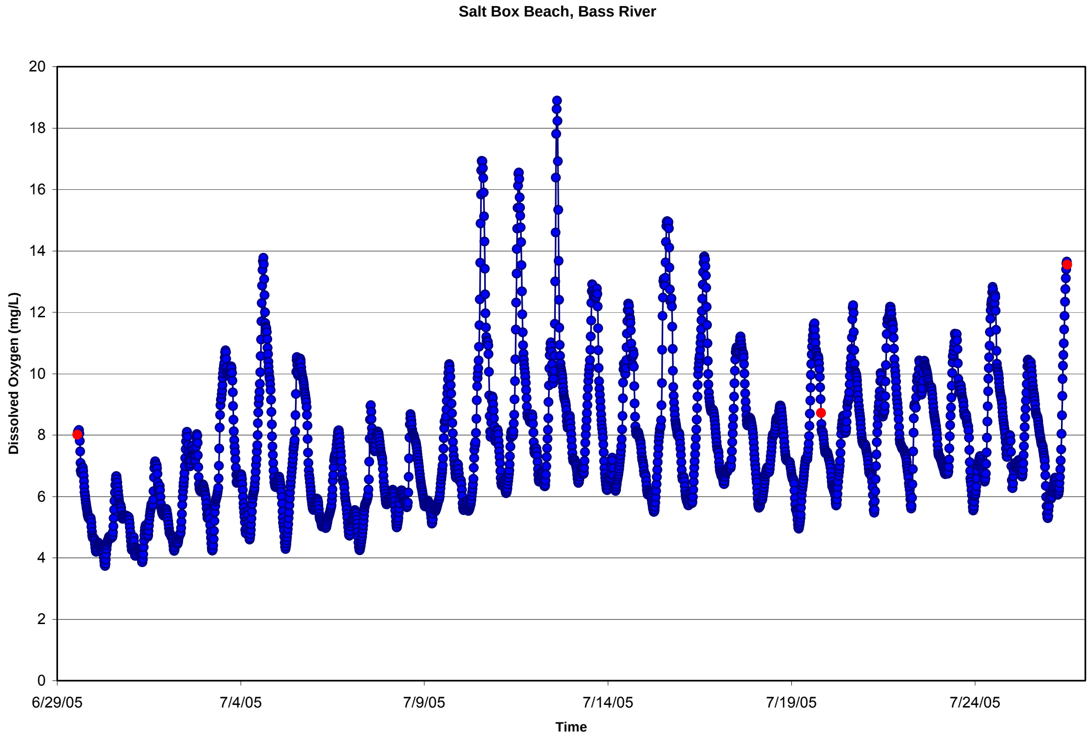


Figure VII-7. Bottom water record of dissolved oxygen at the Salt Box Beach - Bass River station, Summer 2005. Calibration samples represented as red dots.

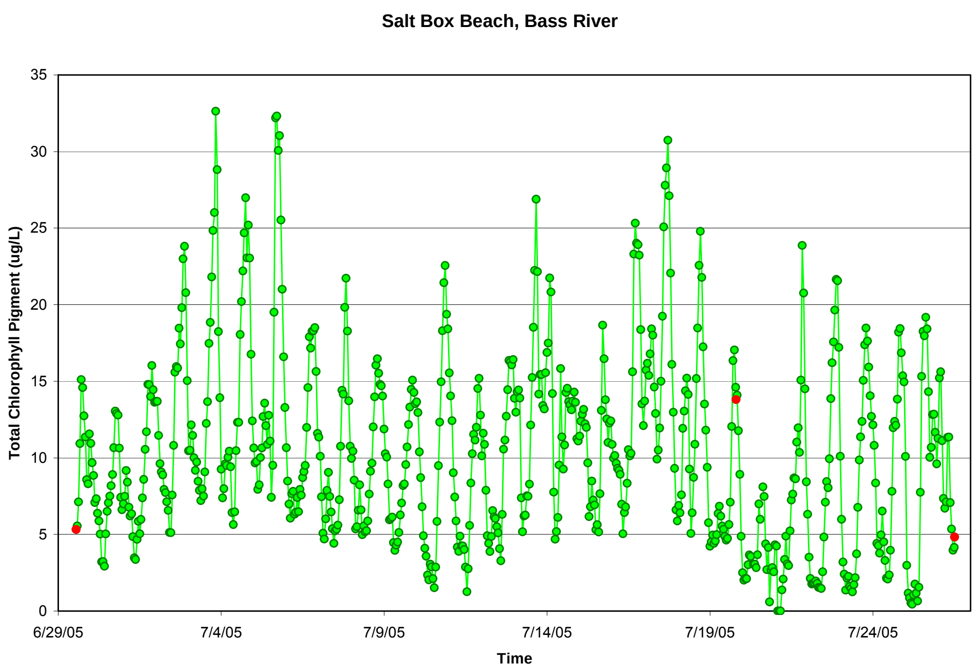


Figure VII-8. Bottom water record of Chlorophyll-*a* at the Salt Box Beach mooring location - Bass River, Summer 2005. Calibration samples represented as red dots.

Dinahs Pond – Bass River (Figures VII-9 and VII-10):

The Dinah's Pond mooring site was centrally located within this tributary sub-basin located off the main channel that is considered as Bass River (Figure VII-2). As was common in the basins in the upper reaches of this system, there were large daily excursions in oxygen levels, ranging from levels in excess of air equilibration to periods of oxygen depletion below 3 mg L^{-1} (12.4 percent of the deployment period) and even dropping occasionally below 2 mg L^{-1} (Figure VII-9, Table VII-1). Oxygen levels in Dinah's Pond regularly exceeded 10 mg L^{-1} and periodically exceeded 12 mg L^{-1}. However, unlike other basins chlorophyll-a levels were not excessive and it appears likely that much of the "excess" oxygen resulted from the eelgrass bed with high epiphyte growth that covered much of this basin. Chlorophyll-a levels were generally under 5 ug L^{-1} for most of the deployment period. Only towards the last 1/3 of the mooring deployment period were moderately high measured chlorophyll-a levels occasionally observed, consistent with the high dissolved oxygen levels and associated excursions. However, the high chlorophyll values represent a small percent of the total deployment period. It is important to note that unlike other mooring locations down gradient of Route 6, at the Dinah's Pond mooring location, chlorophyll-a levels exceeded the 15 ug L^{-1} benchmark only 2% of the time (Table VII-2, Figure VII-10). The chlorophyll-a levels and presence of eelgrass are also consistent with Secchi depths reaching to the bottom of this basin (i.e. light penetrates to the bottom).

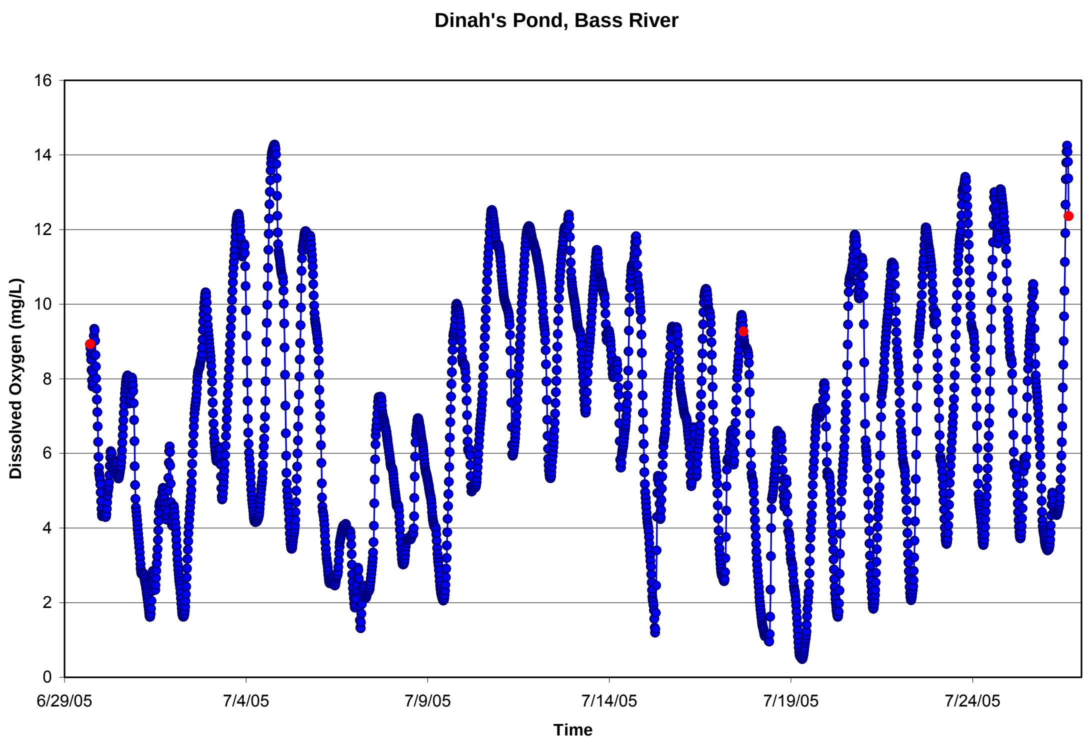


Figure VII-9. Bottom water record of dissolved oxygen at the Dinahs Pond - Bass River station, Summer 2005. Calibration samples represented as red dots.

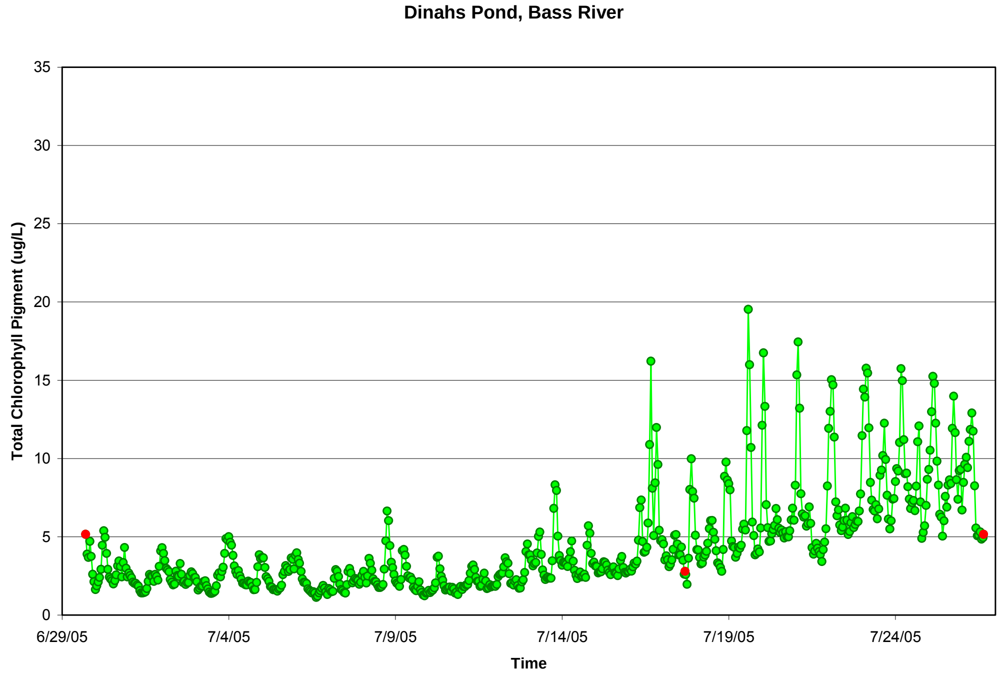


Figure VII-10. Bottom water record of Chlorophyll-*a* at the Dinahs Pond mooring location - Bass River, Summer 2005. Calibration samples represented as red dots.

Kelleys Bay – Bass River (Figures VII-11 and VII-12):

Kelleys Bay is a small basin located between Route 6 and Follins Pond within the same estuarine reach as Dinahs Pond. This basin is situated immediately to the north of Route 6 (Figure VII-2). As seen in adjacent Dinahs Pond, large daily excursions in oxygen levels were frequent, ranging from levels in excess of air equilibration to periods of oxygen depletion infrequently to < 4 mg L^{-1} over the 27 day deployment (Figure VII-11, Table VII-1). The occurrence of "excess" oxygen was equally pronounced when compared to the Dinahs Pond mooring location situated at approximately the same level but across the main channel of the Bass River system and in a slightly more restricted portion of the system. Oxygen levels regularly exceeded 10 mg L^{-1} and periodically exceeded 12 mg L^{-1}. The relatively high measured chlorophyll-a levels as well as the observed presence of macroalgae in the vicinity of the mooring location are likely contributors to the large oxygen excursions. Consistent with the large excursions observed in the oxygen data, chlorophyll-a concentrations were generally high during most the deployment period (between 15 and 20 ug L^{-1}) and even approached 30 ug L^{-1}, clearly indicative of significant phytoplankton production during the summer deployment period. At the Kelleys Bay mooring location, chlorophyll-a levels exceeded the 15 ug L^{-1} benchmark 11% of the time (Table VII-2, Figure VII-12). Average chlorophyll levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments. Given the high measured chlorophyll concentrations and the documented presence of macroalgae and oxygen excursions it appears that Kelleys Bay is presently over-enriched with nitrogen.

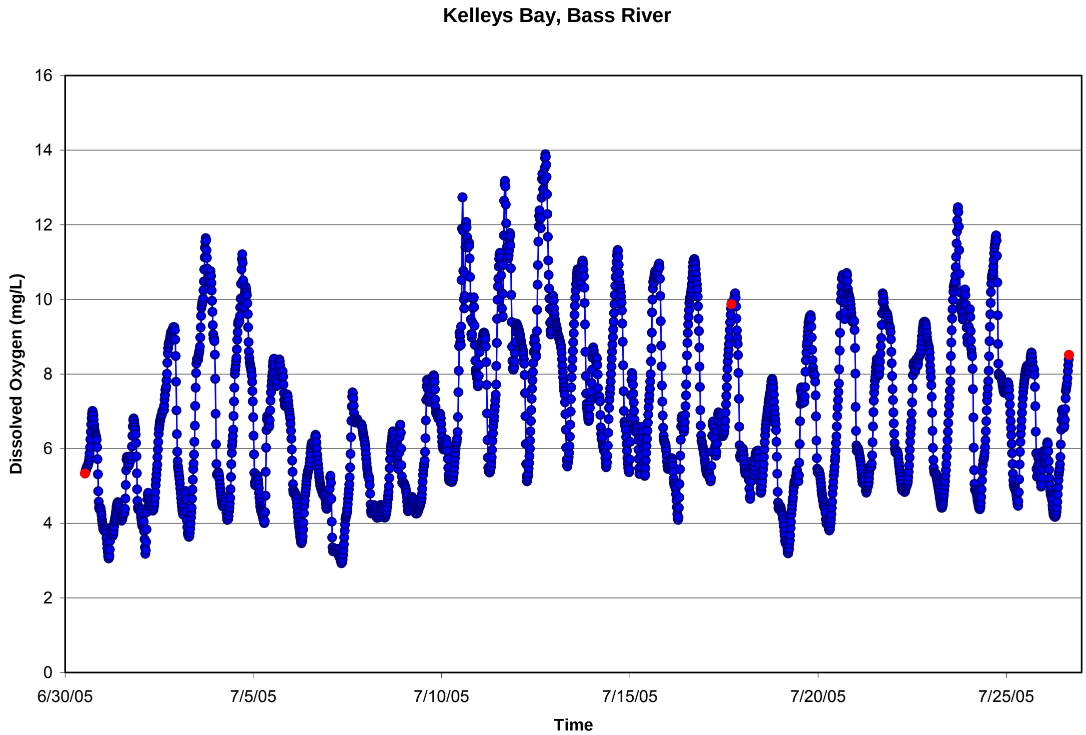


Figure VII-11. Bottom water record of dissolved oxygen at the Kelleys Bay - Bass River station, Summer 2005. Calibration samples represented as red dots.

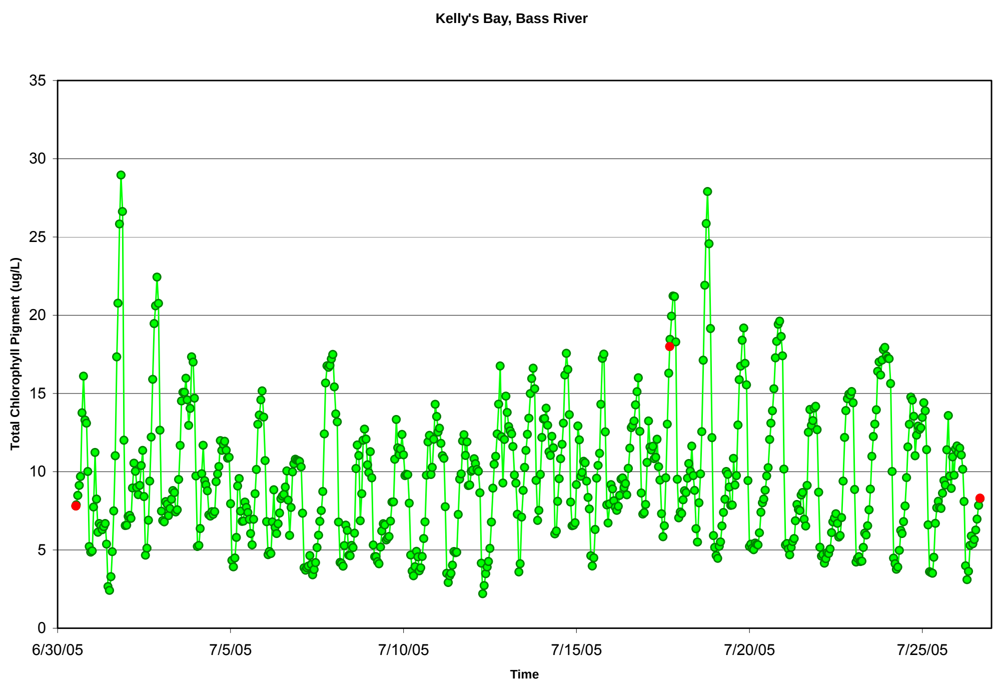


Figure VII-12. Bottom water record of Chlorophyll-*a* at the Kelleys Bay mooring location - Bass River, Summer 2005. Calibration samples represented as red dots.

Point of Rocks – Bass River **(Figures VII-13 and VII-14):**

The Point of Rocks mooring site was the uppermost most deployment location in the Bass River system. The mooring was positioned off Point of Rocks in Follins Pond down gradient of the Weir Creek discharge from Mill Pond. Similar to the other mooring deployment located in Dinahs Pond, high frequency data from the Point of Rocks mooring indicated clear daily excursions of approximately the same magnitude. As with other moorings lower in the Bass River system, the oxygen excursions resulted mainly from oxygen uptake associated with the diurnal cycle as well as tidal influence, though that may be less of a factor given the distance from the inlet. Lowest dissolved oxygen was typically observed at high tide in the early morning. Dissolved oxygen regularly declined to <4 mg L^{-1} and to <3 mg L^{-1} (18% and 11% of the deployment period, respectively) and was also observed to drop below <2 mg L-1 (Figure VII-13, Table VII-1). Oxygen levels regularly exceeded 10 mg L^{-1} but never exceeded 12 mg L^{-1} except for once during the entire 27 day deployment period. The larger excursions in dissolved oxygen and the decrease in concentrations to below 2 mg L^{-1} are consistent with the effects of nitrogen over-enrichment. Consistent with the oxygen data, chlorophyll-a was initially low (between 5 and 10 ug L^{-1}) but rapidly increased during the early part of the deployment period to between 10 and 15 ug L^{-1} with a few occurrences between 15 and 20 ug L^{-1}, indicative of a bloom and greater phytoplankton production as the summer progressed. Approximately 1/3 of the way through the deployment period chlorophyll-a levels clearly drop back to between 5 and 10 ug L^{-1} as the bloom dissipated. However, it is likely that the macroalgal accumulations in this location as observed by SMAST-MEP survey teams maintained a relatively large daily excursion of dissolved oxygen. At the Point of Rocks mooring location, chlorophyll-a levels exceeded the 15 ug L^{-1} benchmark only 1% of the time (Table VII-2, Figure VII-14), however, there was clear evidence of macroalgal accumulations at numerous sites throughout Follins Pond proximal to the Point of Rocks mooring location.

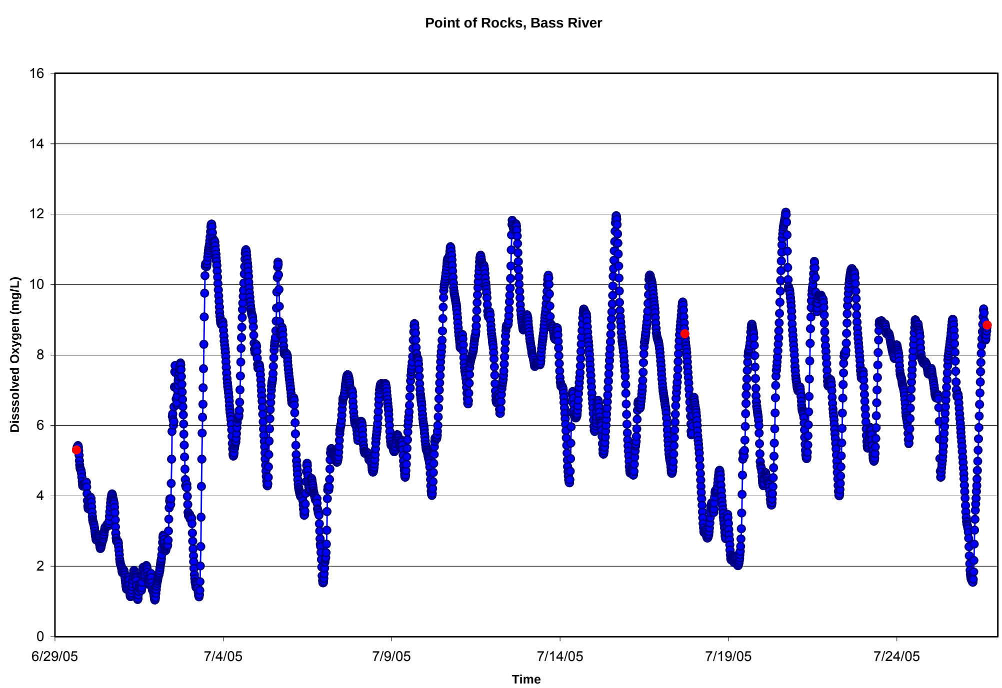


Figure VII-13. Bottom water record of dissolved oxygen at the Point of Rocks mooring location - Bass River station, Summer 2005. Calibration samples represented as red dots.

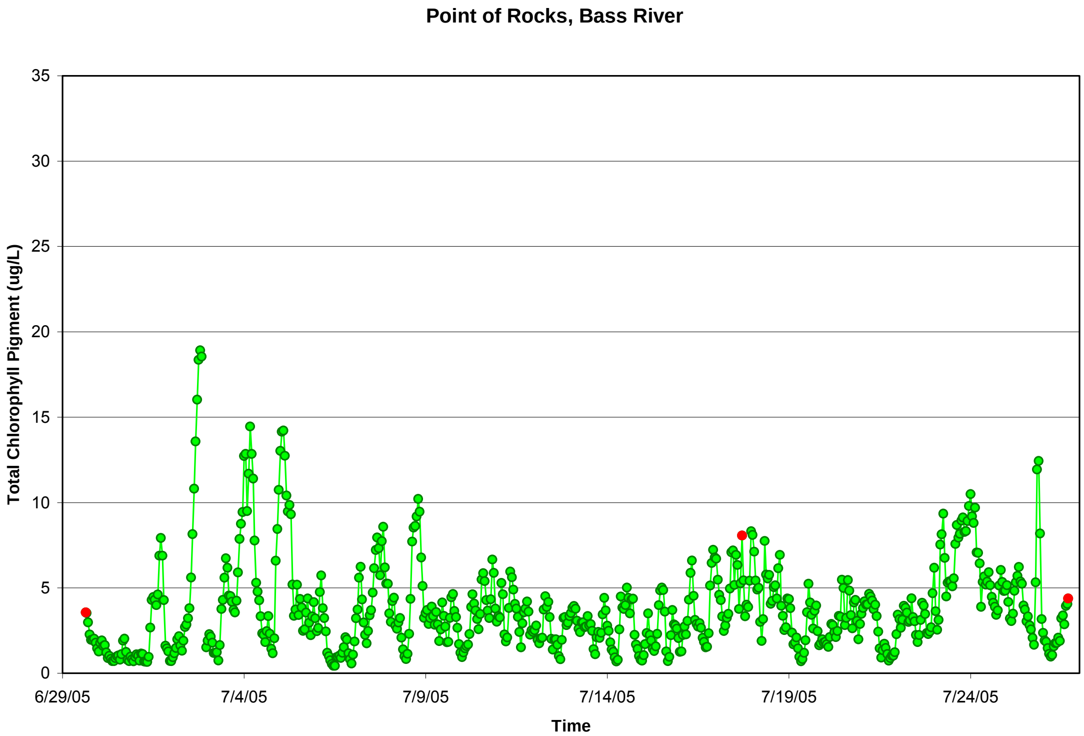


Figure VII-14. Bottom water record of Chlorophyll-*a* at the Point of Rocks mooring location - Bass River, Summer 2005. Calibration samples represented as red dots.

Follins Pond – Bass River (Figures VII-15 and VII-16):

The Follins Pond mooring was located within the uppermost portion of the basin known as Follins Pond at the top of the Bass River system. Follins Pond is immediately down gradient of Mill Pond which represents the head of the Bass River system. This basin is situated immediately to the north of Kelleys Bay. Similar to the Point of Rocks record, large daily excursions in oxygen levels were observed at this location as well, ranging from levels in excess of air equilibration to periods of oxygen depletion with base levels of dissolved oxygen regularly dropping below 4 mg L^{-1} and 3 mg L^{-1} (17% and 10% of the deployment period, respectively). For short periods during the 27 day deployment oxygen levels were also observed dropping below <2 mg L-1 (Figure VII-15, Table VII-1). The occurrence of "excess" oxygen was also similarly pronounced. This is likely due to the relatively high measured chlorophyll-a levels as well as the observed presence of macroalgae in the vicinity of the mooring location. Oxygen levels regularly exceeded 10 mg L^{-1} and periodically exceeded 12 mg L^{-1}. Consistent with the large excursions observed in the oxygen data, chlorophyll-a concentrations were generally high during most the deployment period (between 15 and 20 ug L^{-1}) and even approached 35 ug L^{-1}, clearly indicative of significant phytoplankton production during the summer deployment period. At the Follins Pond mooring location, chlorophyll-a levels exceeded the 15 ug L^{-1} benchmark 16% of the time (Table VII-2, Figure VII-8). Average chlorophyll levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments. The larger excursions in dissolved oxygen, the high measured chlorophyll concentrations and the documented presence of macroalgae are consistent with the impacts of nitrogen over-enrichment.

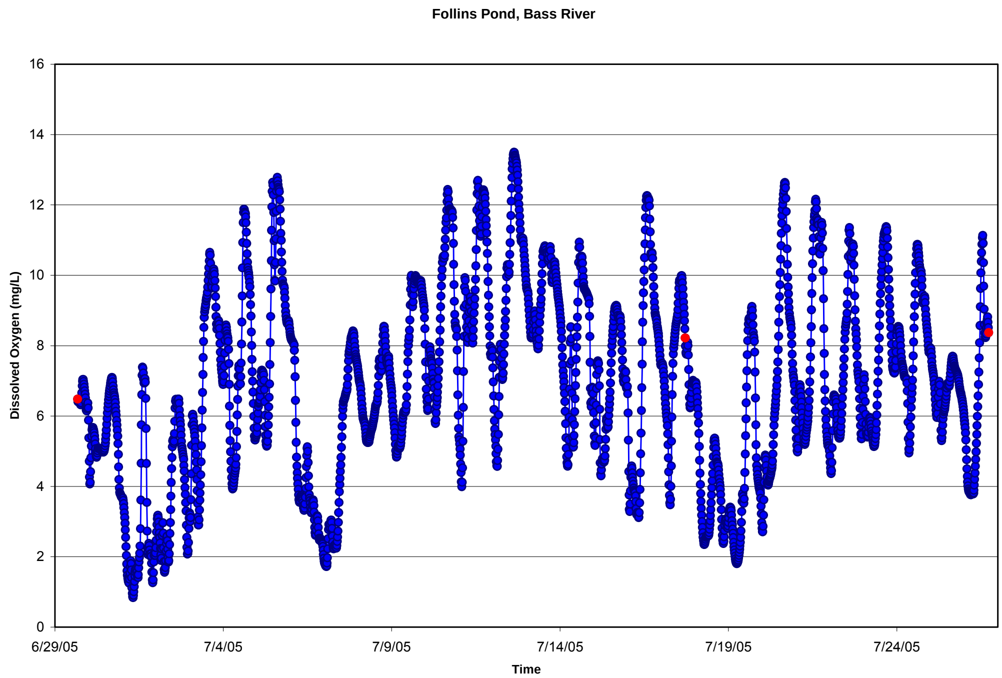


Figure VII-15. Bottom water record of dissolved oxygen at the Follins Pond - Bass River station, Summer 2005. Calibration samples represented as red dots.

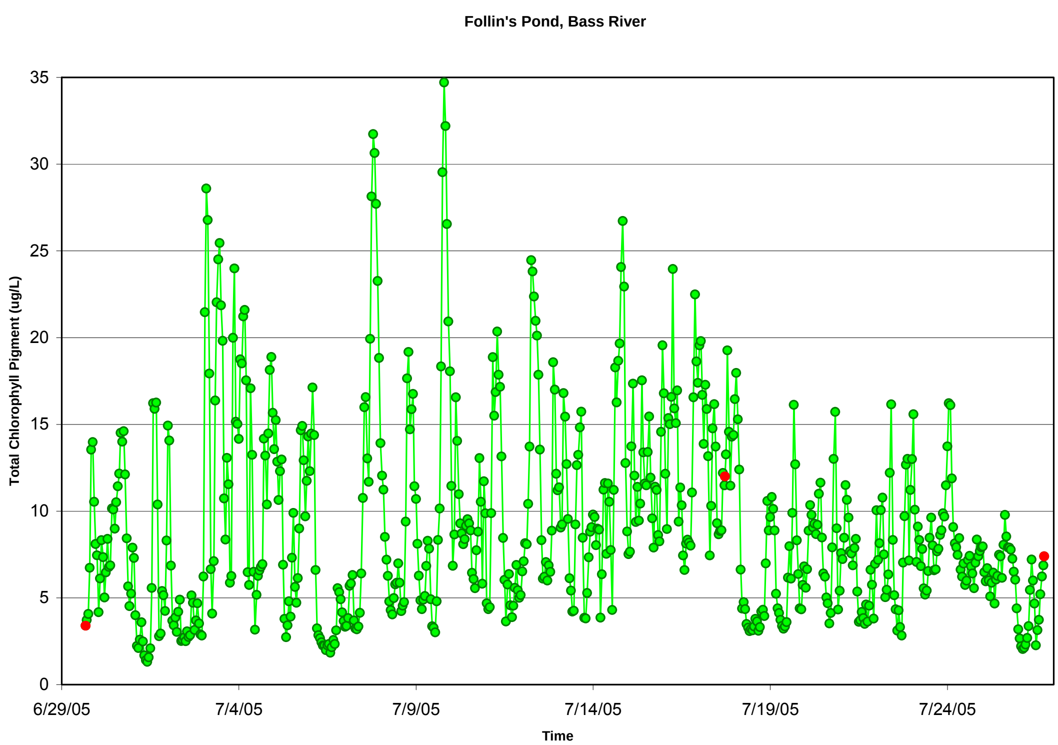


Figure VII-16. Bottom water record of Chlorophyll-*a* at the Follins Pond mooring location - Bass River, Summer 2005. Calibration samples represented as red dots.

Overall, the oxygen and chlorophyll data for the Bass River Estuary clearly indicate a system supporting sub-tidal habitats impaired by nitrogen as evidenced by both periodic low and high oxygen levels, high chlorophyll-a levels and macroalgal accumulations. The data indicate basins ranging from significantly impaired (Follins Pond) to areas with moderately impaired/high quality (Lower portions of Bass River from Salt Box Beach to the inlet). These observations are consistent with the observed levels of total nitrogen (TN) throughout the estuary (Section VI). The gradient in impairment follows the gradient in nitrogen enrichment, where Mill Pond, Follins Pond and Kelleys Bay have very high ebb tide TN levels ($\geq$0.75 mg L^{-1}) declining to the lower portions of Bass River nearest the tidal inlet (0.34 – 0.37 mg L^{-1}) that are better flushed with clean low nutrient water from Nantucket Sound. While the lower portion of Bass River supports lowest nitrogen levels within the system, the levels throughout the system are creating habitat incapable of supporting eelgrass beds and significant to moderate levels of impairment to benthic animal habitat (see Sections VII-3 & VII-4, below).

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data is a key part of the MEP Approach. Surveys were conducted in the Bass River Estuary, particularly within the main tidal channels (Bass River up to Route 6 bridge crossing and inclusive of Grand Cove) by the MassDEP Eelgrass Mapping Program (C. Costello). The most recent survey was conducted in 2001, as part of the MEP program with an earlier survey conducted in 1995. Additional analysis of available aerial photographs from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. The 1951 data were validated by the MEP Technical Team through discussion with the Town of Yarmouth, including individuals with long-term on-site knowledge of this system, particularly Follins Pond and the Kelley's Bay. The 2001 map was field validated by the MassDEP Eelgrass Mapping Program. The primary use of the data is to indicate (a) estuarine regions that have historically or presently support eelgrass habitat, and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figure VII-17); 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.

While there were extensive eelgrass beds in the lower Bass River System in 1951, there was only a small residual eelgrass bed bordering the mouth of the Bass River Estuary in the 2001 survey conducted by the MassDEP. During the 2005 MEP field data collection surveys, the MEP Technical Team also did not observe any significant eelgrass habitat in the lower reaches of the system. However, the MEP Technical Team did observe a relatively dense bed of eelgrass in Dinah's Pond while undertaking field surveys in 2005, as part of the benthic regeneration and benthic animal surveys and during the deployment and recovery of the instrument moorings. Eelgrass plants within the existing bed were heavy with epiphytes, a nitrogen related stressor of eelgrass. It is likely that the shallow nature of this basin and its partial isolation has allowed this residual bed to survive. The 1951 assessment conducted by MassDEP using high quality aerial photography indicated beds within the lower reach of the Bass River south of Route 6. Unfortunately, the 1951 aerial photos that covered the upper portions of the system were of poor quality and precluded being able to estimate the presence of eelgrass in the upper sections of the system. Nevertheless, the MEP Technical Team did contact the Yarmouth Natural Resources Department to see if any qualitative information existed regarding the historical presence of eelgrass in Bass River above the Route 6 bridge and Dinah's Pond. First and second hand accounts indicated that eelgrass was not present in Kelley's Bay and Follins Pond dating back to the 1950's and 1960's (personal communication, Conrad Caia – Town of Yarmouth Natural Resources Department). The fishermen and shellfishermen who were polled did indicate, however, the presence of “weeds”, further clarified by Mr. Caia to mean macroalgae, in the upper reaches of the Bass River system.

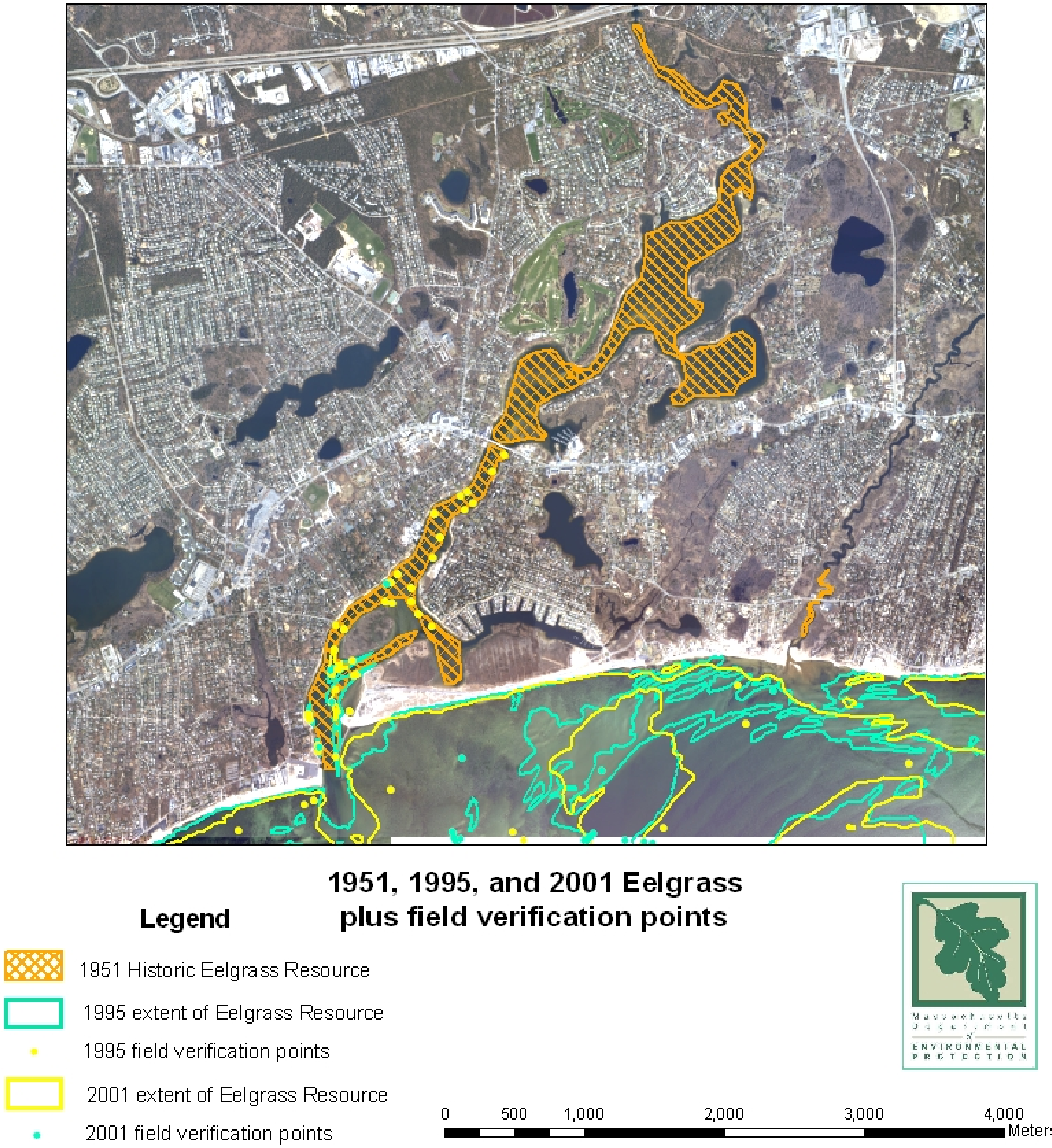


Figure VII-17. Eelgrass bed distribution associated with the Bass River Embayment System in 1951, 1995, 2001, as determined by the MassDEP Eelgrass Mapping Program (map courtesy of C. Costello). The green and yellow outlines circumscribe eelgrass beds as mapped in 1995 and 2001, respectively. The 1951 was determined from aerial photography and validated by descriptions provided by direct observers. Presently, there are no eelgrass beds within the Bass River system.

Overall, the historical distribution of eelgrass within the Bass River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins, with the exception of Dinah's Pond. This shallow tributary basin which is situated in the upper portions of Bass River and is connected to the overall system by a shallow tidal channel is somewhat anomalous but indicative that restoration of eelgrass throughout the lower portions of Bass River south of Route 6 should be achievable under lower nutrient loading conditions. At lower overall nitrogen loading, it would be expected that the lower River areas would have sufficient water clarity and oxygen levels to support eelgrass beds. However, the current absence of eelgrass within this system is expected given the high nitrogen levels and high chlorophyll levels in both the lower and upper basins (Table VII-3). Typically eelgrass beds exist at much lower nitrogen levels (0.35 - 0.45 mg N L^{-1}) than presently found in this system. The 1951 eelgrass distribution presently supports total nitrogen levels of 0.52 - 0.39 mg N L^{-1}. The few remaining eelgrass plants adjacent the tidal inlet where the average TN level is presently 0.34 mg L^{-1}, suggests some other factor like sediment transport and/or inlet maintenance may be limiting eelgrass at this location. The high nitrogen levels within the Bass River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates, which increases the nitrogen levels in the incoming tidal waters (0.3 mg L^{-1}) by several fold (see Section VI).

Table VII-3. Change in eelgrass coverage within the Bass River Embayment System, Towns of Yarmouth and Dennis, as determined by the MassDEP Eelgrass Mapping Program (C. Costello). Analysis indicates that >300 acres of eelgrass habitat could be recovered through implementation of nitrogen management.

Embayment	1951 (acres)	1995 (acres)	2001 (acres)	Percent Difference (1951 to 2001)
Bass River	354.7	14.9	3.1	**99%**

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 24 locations throughout the Bass River Embayment System (Figure VII-18 and 19). At many sites 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 stress 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 throughout most of the Bass River system, this estuary is clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities in the uppermost reaches of the system, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

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

Figure VII-18. Aerial photograph of the upper reach of the Bass River Estuarine System (inclusive of Mill Pond) showing locations of benthic infaunal sampling stations (red symbols). Station numbers relate to those in Table VII-4

Figure VII-19. Aerial photograph of the lower and mid reaches of the Bass River Estuarine System (inclusive of Grand Cove, BSR 9,10,11) showing locations of benthic infaunal sampling stations (red symbols). Station numbers relate to those in Table VII-4.

The infauna survey clearly indicated impaired habitat within various sub-basins of the Bass River System, as well as high quality benthic habitats. The Bass River reaches (mid and lower) extending from the tidal inlet to Rt. 6, were found to presently support high quality benthic habitat. These reaches had moderate to high numbers of individuals (356-1911), distributed among large numbers of non-organic enrichment species (25-31), with resulting very high community diversity (3.3-3.8) and Eveness (0.7-0.8). Based upon these metrics, these sites presently support some of the highest quality benthic animal habitat assessed by the MEP on Cape Cod. These sites also tended to have low to moderate levels of oxygen depletion and chlorophyll-a blooms and were generally not accumulating drift macroalgae. The benthic habitat within Weir Creek basin was of moderate to high quality but with lower diversity and Eveness consistent with it being wetland influenced (Table VII-4). These habitats presently show total nitrogen levels <0.5 mg L^{-1}, which has been found to support similarly high quality habitat in a variety of other Cape Cod estuaries (Popponesset Bay, Parkers River, Lewis Bay, Three Bays). For example in the adjacent Lewis Bay Estuary, the outer stations support several hundred individuals per grab distributed among 32 species. In addition, the community is composed of a variety of polychaete, crustacean and mollusk species, with high diversity and Evenness.

In contrast, the enclosed sub-basins of the Bass River system are presently supporting impaired benthic animal habitat. Mill Pond, Follins Pond and Grand Cove have communities of moderate to high numbers but few species (7-9), low diversity (1.2-1.5) and Evenness (0.41-0.55). They are generally dominated by organic enrichment indicators, consistent with high chlorophyll levels, moderate to significant oxygen depletion and accumulations of macroalgae. Dinahs Pond and Kelleys Bay showed slightly more impairment based upon their low numbers of individuals (<75). These metrics and indicators are consistent with significantly impaired benthic habitats. Their present nitrogen levels range from 0.52-0.95 mg L^{-1}. The lower reach of Follins Pond supported amphipod mats suggesting a transitional zone between the upper reaches and lower reaches of the system. Amphipod mats are typical of transitional environments and were the major communities to develop in Boston Harbor as nutrient loads began to diminish. These animal communities are consistent with the level of nitrogen enrichment and moderate chlorophyll levels and oxygen conditions within this estuarine basin. All of these parameters indicate a system that is supporting moderately impaired benthic habitat.

The Bass River System is similar to other nearby estuaries of similar structure, such as the Parkers River. The uppermost basin of the Parkers River estuary (Seine Pond) supports poor benthic habitat throughout the basin with few species and individuals (i.e. low secondary production), with an average of 6 species and 48 individuals per sample (similar to Dinahs and Kelleys Ponds). Similar to areas of Bass River, Seine Pond in the Parkers River system has very low diversity (1.25) and Eveness <1 (0.65) and is dominated by stress indicator species associated with organic matter enrichment. The lower reach of Parker's River currently supports higher species numbers (27 species) and population levels (2433 individuals), more in line with a high quality benthic animal habitat. However, the Diversity (2.94) and Eveness (0.61) indices suggest a moderate level of impairment.

Overall, the pattern of infaunal habitat quality throughout the Bass River system is consistent with measured dissolved oxygen concentration, chlorophyll, nutrients and organic matter enrichment in this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland influenced such as Weir Creek in the lowermost portion of the Bass River system or tidal embayment dominated, such as the semi-enclosed Ponds and the main reaches of the Bass River itself. Based upon this analysis it is clear that the upper regions of the Bass River

Embayment System are significantly impaired by nitrogen and organic matter enrichment while the lower basins are presently supporting high quality to moderately impaired benthic animal habitat.

The results of the infauna survey supports that nitrogen management threshold analysis (Chapter VIII) needs to include a lowering of the level of nitrogen enrichment in the "Ponds" within the Bass River System.

Table VII-4. Benthic infaunal animal community data for the Bass River embayment system (inclusive of Follins Pond and Dinahs Pond). Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (samples represent surface area of 0.0625 m2). Stations refer to map in Figure VII-18 and 19.

Bass River Basins	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)	Stations BSR
Upper Reach						
Mill Pond	8	1463	5	1.24	0.41	41,42,43
Follins Pond	9	298	6	1.54	0.55	26,28,29,31,40
Dinah Pond	7	70	9	1.64	0.60	34,36
Kelleys Bay	7	46	--[1]	1.74	0.68	37,38
Mid Reach						
Bass River	31	1911	15	3.27	0.67	7,8,14,15,17,20,22
Grand Cove	7	668	5	1.20	0.44	10,11
Lower Reach	.	.	.	.	.	
Bass River	25	356	19	3.81	0.82	5
Weir Crk Basin	17	1741	10	1.90	0.48	3,4

1- too few individuals extant in field sample to support this calculation.

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest (Figures VII-20 through VII-22) as well as the suitability of a system for the propagation of shellfish (Figure VII-23). As is the case with many systems on Cape Cod, all of the enclosed waters of Bass River, from the inlet to Point of Rocks, is conditionally approved for the taking of shellfish during specific times during the year, typically the cold winter months, indicating the system is generally supportive of shellfish communities. However, in the upper most reaches of the system, specifically Mill Pond, harvest of shellfish is prohibited year round indicating the presence of a persistent environmental contaminant. In the case of the Mill Pond closure, that is likely due to bacterial contamination. The major shellfish species with potential habitat within the Bass River Estuary are soft shell clams (*Mya*) and quahogs (*Mercenaria*) extending all the way up to Follins Pond (Figure VII-8). In addition, if habitat conditions improve there is also the potential for small grow areas for bay scallops, mostly in the area of Bass River below Route 6 with a small potential area in Kelleys Bay immediately up-gradient of Route 6.

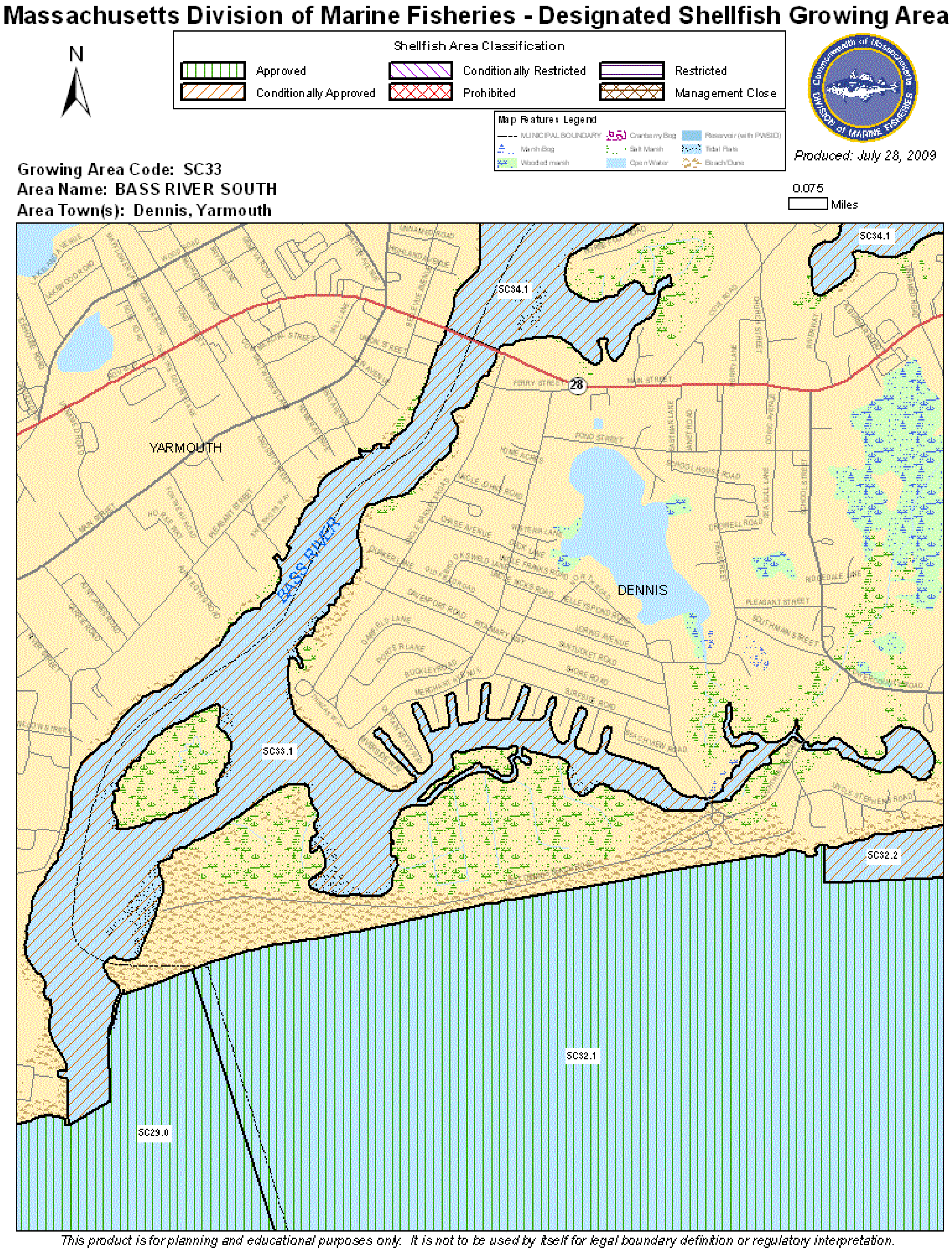


Figure VII-20. Potential shellfish growing areas within the southern portion of the Bass River system, Yarmouth and Dennis, MA.

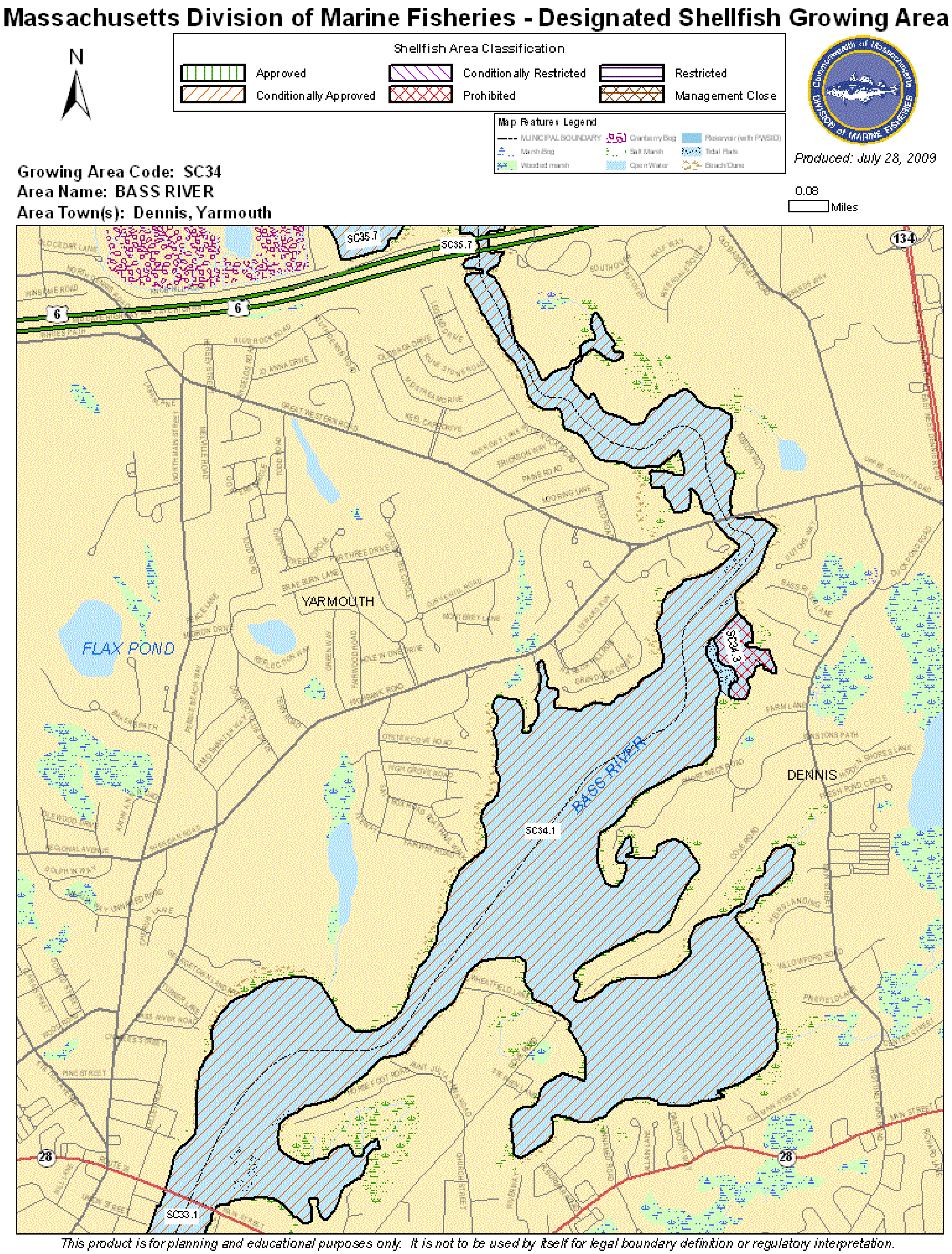


Figure VII-21. Potential shellfish growing areas within the central portion of the Bass River system, Yarmouth and Dennis, MA.

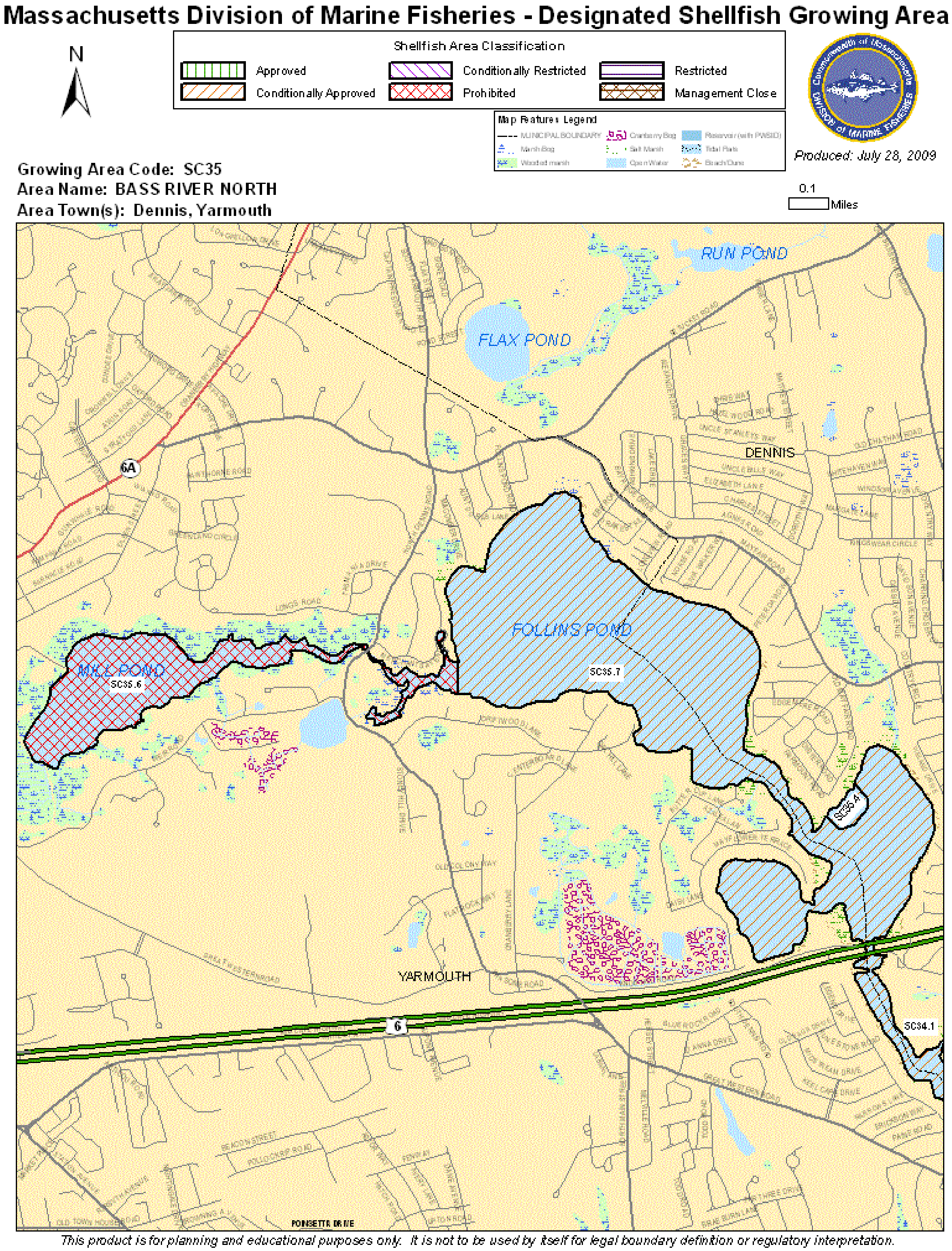


Figure VII-22. Potential shellfish growing areas within the northern portion of the Bass River system, Yarmouth and Dennis, MA.

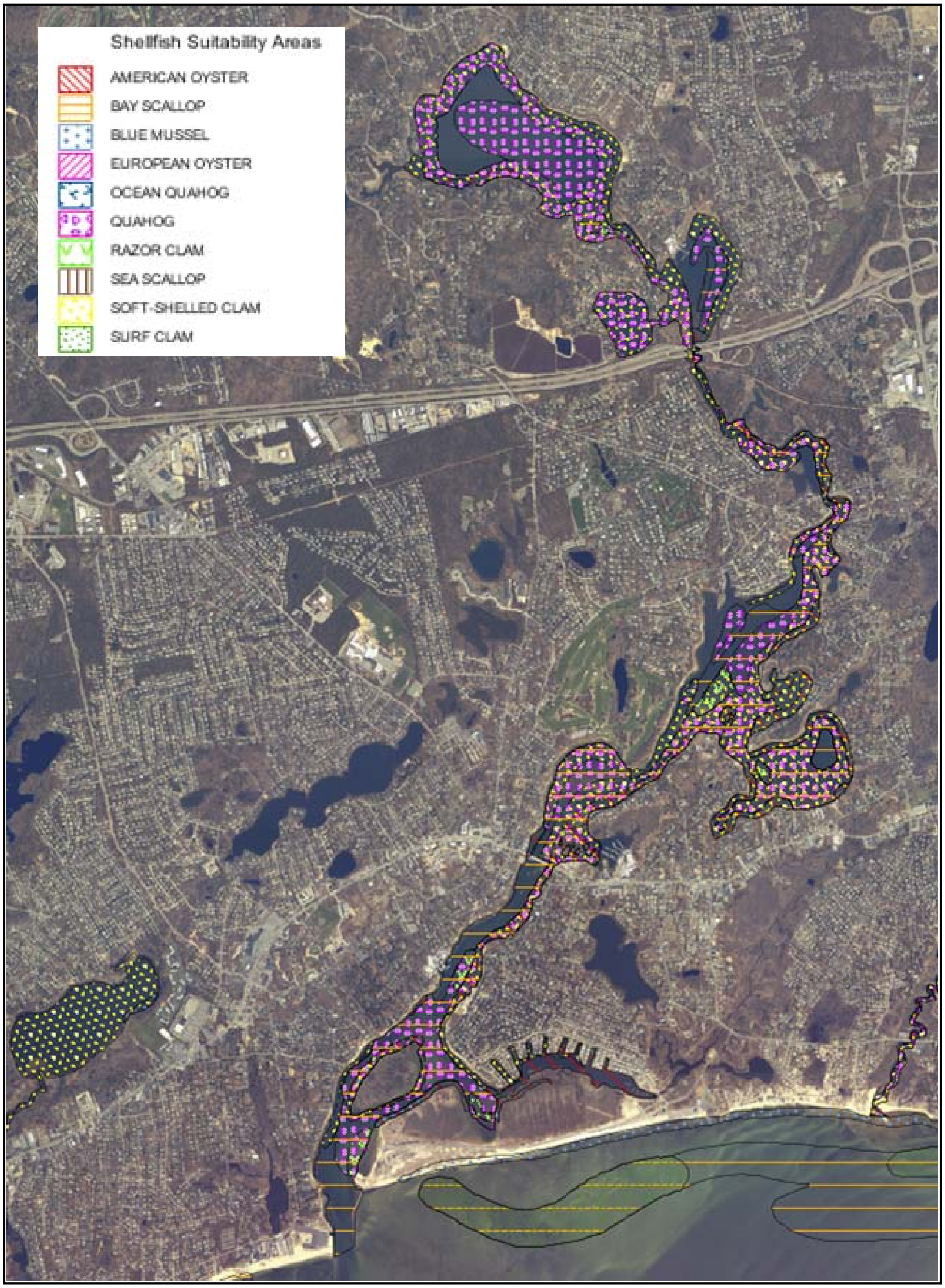


Figure VII-23. Potential shellfish growing areas within the Bass River system, Yarmouth and Dennis, MA. Primary species with potential suitable habitat are bay scallops, soft shell clams and quahogs. Source: Mass GIS.

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

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

The Bass River Embayment System is a complex estuary composed of 2 functional types of component basins: embayments (Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay, Grand Cove, Bass River) and a salt marsh influenced basin (Weir Creek). Each of these 2 functional components has different natural sensitivities to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of each system and the ability to support eelgrass beds and the types of infaunal communities that they support. At present, the Bass River is showing differences in nitrogen enrichment and habitat quality among its various component basins (Table VIII-1).

Overall, the system is showing some nitrogen related habitat impairment within each of its semi-enclosed component basins, however, there is a strong habitat quality gradient. Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay and Grand Cove are presently supporting significantly impaired infauna habitat. Since Mill Pond, Follins Pond and Kelleys Bay have not historically has not supported eelgrass, they are classified as "significantly impaired" basins due to loss of benthic animal habitat. Nitrogen enrichment (through inputs and tidal flushing) has resulted in phytoplankton blooms, periodic oxygen depletions, macroalgal accumulations and significantly reduced (Mill Pond, Follins Pond, Grand Cove) to virtual loss (Dinah Pond, Kelleys Bay) of benthic communities. The Bass River is also nitrogen enriched, but has less nitrogen enrichment based primarily on its structure and high water turnover. While the mid and lower reaches currently supports high quality benthic habitat, its loss of historical eelgrass coverage indicates that it has become a significantly impaired basin relative to eelgrass habitat. Finally, Weir Creek is a small shallow tidal basin with extensive wetlands in its upper reaches and as such has not historically supported eelgrass. Weir Pond has been deepened for navigation and currently functions as a wetland influenced basin with natural organic matter inputs and periodic low oxygen. As such, it is currently supporting moderately to highly productive diverse infaunal communities. However, based upon the high chlorophyll levels and some of the infaunal indicators, it may be showing some modest impairment of benthic habitat. Overall, the regions of significant and moderate habitat impairment (eelgrass or benthic infaunal) comprise >90% of the estuarine area of the Bass River Embayment System.

Eelgrass: The loss of historic eelgrass beds throughout the mid and lower basins of the Bass River Estuary is consistent with the observed nitrogen and the chlorophyll levels and functional basin types comprising this estuary. The Bass River basins below Rt. 6 supported eelgrass beds in 1951 under lower nitrogen loading conditions.

The historical distribution of eelgrass and its present absence within the Bass River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. Shallow wetland influenced basins, like Weir Creek typically do not typically support eelgrass beds. Similarly, the semi-enclosed basins of the upper estuary (Mill Pond, Follins Pond, Kelleys Bay) has likely been nutrient enriched with poor water clarity for many decades. While the presence of eelgrass within Dinah Pond was somewhat surprising, the heavy epiphyte growth over each of the plants is similar to that observed in other Cape Cod estuaries at similar shallow depths and nitrogen levels (e.g. Little Pond, Falmouth). The plants comprising this bed are not "healthy", but to the extent that they can persist, they may greatly accelerate the re-growth of eelgrass in Bass River once nitrogen levels are lowered.

In contrast, the lower tidal reaches of the Bass River with their lower nitrogen inputs might be expected to have sufficient water clarity and oxygen levels to support eelgrass beds. However, given the sensitivity of eelgrass to declining light penetration resulting from nutrient enrichment and secondary effects of organic enrichment and oxygen depletion, the current absence of eelgrass within this system is expected given the water depths, nitrogen levels and chlorophyll levels. Typically eelgrass beds exist at much lower nitrogen levels (threshold depending on water depth of 0.35 - 0.45 mg N L^{-1}) than presently found in the historic eelgrass areas within this system (0.52 - 0.39 mg N L^{-1}). Note that it appears that the reduction of the beds to a few remaining eelgrass plants within the Bass River tidal inlet where the average TN level is sufficiently low to support eelgrass, 0.34 mg L^{-1}, suggests that some other factor such as sediment transport and/or inlet maintenance may be limiting eelgrass at this location. The high nitrogen levels within the upper reaches of the Bass River Estuary (0.61 - 0.95 mg N L^{-1}) indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates, which increases the nitrogen levels in the incoming tidal waters (0.3 mg L^{-1}) by several fold (see Section VI). This pattern is similar to that in upper Parker's River (0.66 - 0.99 mg N L^{-1}), where similar levels and patterns of habitat impairment were found. As there is no evidence of eelgrass coverage within the upper Bass River (as noted above) these areas should not be considered for eelgrass restoration. In contrast, documented eelgrass within the lower tidal reaches makes restoration of this resource a primary target for overall restoration of the Parker's River Embayment System. Restoration of this habitat will require appropriate nitrogen management. Note that restoration of this eelgrass habitat will necessarily result in restoration of other resources throughout the system, particularly within the upper estuary. Mill Pond and Follins Pond are the discharge basins for much of the watershed nitrogen load to this estuary. The lower reach of the Bass River is the channel through which the nitrogen and organic matter enriched waters from the upper estuary is flushed out on ebbing tides. Nitrogen management focused on lowering nitrogen levels within the lower River will require lowering of nitrogen levels throughout the upper estuary. Therefore an improvement of infaunal habitats within these upper basins will result as part of improving eelgrass habitat in the lower River.

Based upon the above analysis, eelgrass habitat was selected as the primary nitrogen management goal for the lower reach of the Bass River and infaunal habitat quality the management target for upper basins, primarily Follins Pond (Mill Pond is brackish). These goals are the focus of the MEP management alternatives analysis presented in Chapter IX.

Water Quality: The tidal waters of the Bass River Embayment System are currently listed under this Classification as SA. The Bass River Estuary is not presently meeting the water quality standards for SA waters. The result is that as required by the Clean Water Act, TMDL processes and management actions must be developed and implemented for the restoration of resources within this estuary.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels within Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay and Grand Cove indicate high levels of nutrient enrichment and impaired habitat quality. The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of these estuarine basins. The large daily excursions in oxygen concentration in basins also indicate significant organic matter enrichment. However, the level of oxygen stress in Weir Creek needs to be evaluated in light of the fact that it is a wetland influenced basin. Salt marsh tidal basins are naturally organic matter enriched and typically have summertime low oxygen events (periodic hypoxia), while high quality embayment basins do not. Weir Creek is a highly modified basin which is now functioning as an intermediate between a salt marsh pond and an embayment basin.

The upper reaches of the Bass River Estuary generally show significant oxygen depletions and phytoplankton blooms and some basins have macroalgae accumulations. The largest upper basin, Follins Pond, had large daily excursions in oxygen levels ranging from levels in excess of air equilibration to below 4 mg L^{-1} and 3 mg L^{-1} (17% and 10% of the deployment period, respectively) and for short periods below 2 mg L-1 (Section VII-2). Oxygen levels regularly exceeded 10 mg L^{-1} and periodically exceeded 12 mg L^{-1}. Consistent with the large excursions observed in the oxygen data, chlorophyll a concentrations were generally high (between 15 and 20 ug L^{-1}) and even approached 35 ug L^{-1}, clearly indicative of significant phytoplankton production during the summer deployment period. Average chlorophyll levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments. The larger excursions in dissolved oxygen, the high measured chlorophyll concentrations and the documented presence of macroalgal accumulations are consistent with the impacts of nitrogen over-enrichment as indicated by the tidally averaged TN levels of 0.75 mg N L^{-1}.

Kelleys Bay and Dinahs Pond also had frequent large daily excursions in oxygen levels ranging from levels in excess of air equilibration to periods of oxygen depletion to < 4 mg L^{-1}. Oxygen levels regularly exceeded 10 mg L^{-1} and periodically exceeded 12 mg L^{-1}. High measured chlorophyll-a levels were found in Kelleys Bay with moderate levels in Dinah Pond (consistent with the light penetration in Dinah Pond). Kelleys Bay had macroalgal accumulations where Dinah Pond had primarily eelgrass epiphytes (growth of surface microalgae). Chlorophyll a concentrations in Kelleys Bay were generally high during most the deployment period (between 15 and 20 ug L^{-1}) and even approached 30 ug L^{-1}, clearly indicative of significant phytoplankton production during the summer deployment period. Given the high measured chlorophyll concentrations and the documented presence of macroalgae and oxygen excursions, it appears that Kelleys Bay is presently over-enriched with nitrogen. Moderate chlorophyll a levels but similarly large oxygen excursions and epiphyte growth support the same designation for Dinah Pond. The adjacent basins of Kelleys Bay and Dinah Pond presently support tidally averaged summertime TN levels of 0.70 mg N L^{-1}.

The major semi-enclosed basin in the mid/lower reaches of the Bass River is Grand Cove. Being in the mid reach of the estuary provides only a slightly better habitat quality for this system. Grand Cove also shows large daily oxygen excursions resulting mainly from oxygen uptake associated with the diurnal cycle as well as tidal influence. However depletions were modest, declining to <4 mg L^{-1} (6 percent of the total deployment period) but never reaching 3 mg L^{-1}. But the occurrence of oxygen levels over air saturation was pronounced, with levels regularly exceeding 10 mg L^{-1} and periodically 12 and even 14 mg L^{-1}. Consistent with the oxygen data, chlorophyll a was moderately elevated averaging ~7.6 ug L^{-1}. However the basin supports macroalgal accumulations, further evidence of nitrogen enrichment of this basin. These observations are consistent with the loss of historic eelgrass beds and impaired benthic habitats and the tidally averaged summertime TN level >0.5 mg N L^{-1} (0.52 mg N L^{-1}).

In contrast the mid and lower reaches of the Bass River support moderate levels of oxygen depletion (seldom dropping to 4 mg L^{-1}), lower daily excursions and chlorophyll levels, generally 4 - 10 ug L^{-1}. These reaches generally do not show macroalgal accumulations and support high quality benthic habitat. The strong horizontal gradient in water quality results mainly from the high nitrogen water entering from the upper estuary on the ebb tide and the low nitrogen water entering from the Nantucket Sound on the flood tide. Tidally averaged TN levels within the River range from 0.52 - 0.39 mg N L^{-1}, and lower right in the tidal inlet (0.34 mg N L^{-1}).

Overall, the pattern of high nitrogen, resulting in high phytoplankton biomass and periodic low oxygen depletion was found throughout the upper reaches and in Grand Cove grading to high water quality in the mid and lower reaches of the Bass River. The loss of eelgrass within the Bass River and Grand Cove is consistent with the observed water quality conditions. Similarly, the significant impairment of infaunal habitat within the upper Ponds and Grand Cove also reflect nitrogen enrichment. Management of nitrogen levels through reductions in watershed nitrogen inputs or increased tidal flushing are required for restoration of eelgrass and infaunal habitats within the Bass River Embayment System.

Infaunal Communities: In all areas and particularly those that do not support eelgrass beds, benthic animal indicators are 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 Infauna Survey clearly indicated significantly impaired benthic animal habitat within each of the upper basins and Grand Cove. Only Bass River and possibly Weir Creek showed high quality benthic habitat (Section VII-4).

The Bass River reaches (mid and lower) extending from the tidal inlet to Rt. 6, were found to presently support high quality benthic habitat. These reaches had moderate to high numbers of individuals (356-1911), distributed among large numbers of non-organic enrichment species (25-31), with resulting very high community diversity (3.3-3.8) and Eveness (0.7-0.8). Based upon these metrics, these sites presently support some of the highest quality benthic animal habitat assessed by the MEP on Cape Cod. These sites also tended to have low to moderate levels of oxygen depletion and chlorophyll a blooms and were generally not accumulating drift macroalgae. The benthic habitat within Weir Creek basin was of moderate to high quality but with lower diversity and Eveness consistent with its being wetland influenced (Table VII-4). These habitats presently show total nitrogen levels <0.5 mg L^{-1}, which has been found to support similarly high quality habitat in a variety of other Cape Cod estuaries (e.g. Lewis Bay).

In contrast, the enclosed sub-basins are presently supporting impaired benthic animal habitat. Mill Pond, Follins Pond and Grand Cove have communities of moderate to high numbers but few species (7-9), low diversity (1.2-1.5) and Evenness (0.41-0.55). They are generally dominated by organic enrichment indicators, consistent with high chlorophyll levels, moderate to significant oxygen depletion and accumulations of macroalgae. Dinah Pond and Kelleys Bay showed slightly more impairment based upon their low numbers of individuals (<75). These metrics and indicators are consistent with significantly impaired benthic habitats. Their present nitrogen levels range from 0.52-9.5 mg L^{-1}. The lower reach of Follins Pond supported amphipod mats suggesting a transitional zone between the upper reaches and lower reaches of the system. Amphipod mats are typical of transitional environments and were the major communities to develop in Boston Harbor as nutrient loads began to diminish. These animal communities are consistent with the level of nitrogen enrichment and moderate chlorophyll levels and oxygen conditions within this estuarine basin. All of these parameters indicate a system that is supporting moderately impaired benthic habitat.

Overall, the pattern of infaunal habitat quality throughout the Bass River system is consistent with measured dissolved oxygen concentration, chlorophyll, nutrients and organic matter enrichment in this system. Classification of habitat quality necessarily includes the structure of the specific estuarine basin, specifically as to whether a basin area is wetland influenced such as Weir Creek or tidal embayment dominated, such as the semi-enclosed Ponds and the main reaches of the Bass River itself. Based upon this analysis it is clear that the upper regions of the Bass River Embayment System are significantly impaired by nitrogen and organic matter enrichment while the lower basins are presently supporting high quality to moderately impaired benthic animal habitat.

The results of the infauna survey supports that nitrogen management threshold analysis (Section VIII.2) and indicates a need to lower the level of nitrogen enrichment within the semi-enclosed basins of the upper and mid reaches of the Bass River System (Table VIII-1).

Table VIII-1. Summary of nutrient related habitat quality within the Bass River Estuarine System within the Towns of Yarmouth and Dennis, MA, based upon assessments in Section VII. WQMP: Town of Dennis & Yarmouth Water Quality Monitoring Program.

	Upper Reach				Mid Reach		Lower Reach	
Health Indicator	**Mill Pond**	**Follins Pond**	**Dinah Pond**	**Kelleys Bay**	**Mid River**	**Grand Cove**	**Lower River**	**Weir Crk Basin**
Dissolved Oxygen	MI[1]	SI[2]	SI[3]	M/SI[4]	MI[5]	M/SI[6]	MI/H[7]	MI/H[8]
Chlorophyll	SI[9]	MI/SI[10]	MI[11]	MI/SI[12]	MI/SI[13]	MI[14]	MI/H[15]	MI/H[16]
Macroalgae	H/MI[17]	SI[18]	MI[19]	MI[19]	H[20]	MI[21]	H[22]	H[22]
Eelgrass	--[23, 33]	--[23]	MI[24]	--[23]	SI[25]	SI[25]	SI[25]	--[23]
Infaunal Animals	SI[26]	SI[27]	SI/SD[28]	SI/SD[28]	H[29]	SI[30]	H[31]	MI/H[32]
Overall:	**SI[34]**	**SI[35]**	**MI/SI[36]**	**SI[37]**	**SI[38]**	**SI[39]**	**SI[38]**	**H/MI[40]**

1- oxygen levels almost always > 4mg/L , and generally >5 mg/L, WQMP, levels >air saturation periodic.
2- periodic oxygen depletions to < 1 mg/L, <3 mg/L 10% of time, <5 mg/L ~25% of 27 day record, similar to WQMP BR-2 & BR-3 results, levels >air saturation periodic.
3- periodic oxygen depletions to < 1 mg/L, <3 mg/L 13% of time, <5 mg/L ~32% of 27 day record, generally similar to WQMP BR-4 results, levels >air saturation periodic.
4- generally oxygen >4 mg/L, infrequently below 4 mg/L (5% of record), <5 mg/L 23% of record, levels >air saturation periodic.
5- generally ~6 mg/L and above 5 mg/L 93% of record, rarely<4 mg/L: WQMP minimum= 4.3 mg/L
6- generally oxygen >4 mg/L, infrequently below 4 mg/L (6% of record), <5 mg/L 18% of record. Minimum 4.1 in WQMP and 3 mg/L in 27 day record.
7- generally >5 mg/L 98% of time, minimum 4.7 mg/L (133 samples WQMP), >6 mg/L 41% of time.
8- rare depletion <3 mg/L WQMP, generally >4 mg/L (93% of record & 93% WQMP samples), <5 mg/L 30% of record, wetland influenced. 9- blooms, overall average 24.7 ug/L in WQMP samplings
10- average ~10 ug/L and >15 ug/L 16% of record; WQMP average = 11.5 ug/L.
11- average 5.2 ug L^{-1} rarely ~15 ug L^{-1} over 27 day record ; WQMP average 9.3 ug L^{-1}
12- average ~10 ug/L and >15 ug/L 11% of record; WQMP average = 8.4 ug/L.
13- average ~10 ug/L and >15 ug/L 20% of record; WQMP average = 5.8 ug/L
14- average 7.7 ug/L and >15 ug/L 7% of record; WQMP average = 7.6 ug/L
15- average 3.9 ug/L WQMP average.
16- average 7.6 ug/L and >15 ug/L 4% of record; WQMP average = 4.9 ug/L
17- patchy surface mat, epiphytes on Ruppia, a brackish SAV (rooted submerged aquatic vegetation)
18- areas of dense drift algae, generally a branched form possibly Gracillaria, some *Ulva*.
19- drift algae generally sparse, some moderately dense patches
20- sparse drift algae, only BSR-20 had any significant accumulation, which appeared to be Ulva from upper basins.
21- areas of moderate accumulations of Ulva and filamentous and branched forms.
22- sparse drift algae with patches of attached *Codium*.
23- no evidence this basin is supportive of eelgrass.
24- areas of dense coverage, but heavy with epiphytes, no temporal data on changes in bed coverage
25- loss of extensive eelgrass coverage 1951-1995, no eelgrass in 2001/2006 MassDEP and MEP surveys.
26- high numbers of individuals, low diversity, dominated by single organic enrichment species (e.g. *Streblospio*).
27-moderate number of individuals, low diversity, main basin dominated by stress and organic enrichment indicators (e.g. tubificids, Capitella, Streblospio); lower basin by transitional indicators, amphipod mats.
28- low numbers of individuals (<75) and species (7), >50% of community is stress indicator species, *Capitella*.
29- high numbers of individuals, species (31), diversity (>3) and Evenness (~0.8) some deep burrowers, crustaceans, polychaetes and mollusk species.
30- high numbers of individuals and low species (7), diversity (~1) and Eveness (<0.5), patchy distribution with high numbers dominated by a cumacean, remainder of community dominated by organic enrichment indicators.
31- moderate-high numbers of individuals, species (25), diversity (>3) and Evenness (~0.7) some deep burrowers, crustaceans, polychaetes and mollusk species, some transitional species.
32- high numbers of individuals, moderate species (17), diversity (~2) & Evenness (~0.5), crustaceans, mollusk & polychaetes species, dominated by transitional species (amphipods & cumaceans), wetland influenced.
33- no eelgrass, but significant coverage by SAV, most likely *Ruppia*.
34- based on significantly impaired benthic habitat (dominated by 1 enrichment species) and high chlorophyll.
35- based upon significantly impaired benthic habitat, low oxygen and accumulations of drift algae.
36- based upon presence of eelgrass but with high epiphyte growth and significantly impaired benthic habitat.
37- based upon significantly impaired benthic habitat, low oxygen and moderate accumulations of drift algae.
38- based upon loss of historic eelgrass beds as documented by MassDEP, infauna habitat is high quality.
39- based upon loss of historic eelgrass beds as documented by MassDEP & significantly impaired infauna habitat.
40- based upon infauna habitat and absence of historic eelgrass beds and wetland influence.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation; -- = not applicable to this estuarine reach

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

The approach for determining nitrogen loading rates that will support acceptable habitat quality throughout an embayment system is to first identify a sentinel location within the embayment and secondly, to determine the nitrogen concentration within the water column that will restore the location to the desired habitat quality. The sentinel location is selected such that the restoration of that one site will necessarily bring the other regions of the system to acceptable habitat quality levels. Once the sentinel site and its target nitrogen level are determined (Section VIII.2), the Linked Watershed-Embayment Model is used to sequentially adjust nitrogen loads until the targeted nitrogen concentration is achieved (Section VIII.3.

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

The Bass River Embayment System presently supports a range of infaunal habitat quality. A gradient in nutrient related habitat degradation was observed from the most inland reaches of the overall system (Mill Pond, Follins Pond, Dinah Pond, Kelleys Bay) to the higher quality habitat within the Bass River and near the tidal inlet. While the basin of Weir Creek is partially naturally nutrient and organic matter enriched (as a wetland influenced basin), the existing benthic communities suggest a possible moderate level of impairment. However, this basin would be restored (to the extent that the present issue is anthropogenic in origin) as a consequence of improvements in other portions of the estuary. However, the primary habitat issues within the Bass River Embayment System relate to the loss of the eelgrass beds from the mid and lower reaches of the Bass River and Grand Cove, as well as the significantly impaired benthic animal habitat in the upper ponds. The loss of eelgrass classifies the mid and lower reaches (and Grand Cove) as "significantly impaired", although the River reaches presently support high quality infaunal communities. The impairments to both the infaunal habitat and the eelgrass habitat within the component basins of the Bass River Embayment System are supported by the variety of other indicators including oxygen depletion, chlorophyll, and TN levels, all of which support the conclusion that these impairments are the result of nitrogen enrichment, primarily from watershed nitrogen loading.

The habitat assessment data are also internally consistent. Overall, the oxygen and chlorophyll data for the Bass River Estuary clearly indicate a system supporting sub-tidal habitats impaired by nitrogen, ranging from highly stressed (Mill Pond, Follins Pond, Kelleys Bay) to moderately stressed (Dinah Pond and possibly Weir Creek). These observations are consistent with the high levels of total nitrogen (TN) throughout the estuary. The gradient in impairment follows the gradient in nitrogen enrichment, where the upper ponds have high ebb tide TN levels ($\geq$0.70 mg N L^{-1}) declining to the Lower River (0.39 mg N L-1) to the tidal inlet (0.34 mg N L^{-1}). While the lower River supports lowest nitrogen levels within the system, the levels are still higher than can support eelgrass beds in deep basins (see Sections VII-3 & VII-4).

The observed loss of eelgrass, moderate oxygen and chlorophyll levels and benthic community structure within the mid and lower Bass River reaches, suggests a system beyond the nitrogen threshold level that would be supportive of eelgrass, but currently supporting high quality infaunal habitat. The average nitrogen levels for these regions were 0.39 - 0.50 mg N

L^{-1}, the uppermost reach of the Bass River and Grand Cove appear to be above the level supportive of infaunal communities at 0.52 - 0.61 mg N L^{-1}) and well above levels supportive of eelgrass beds as found in a variety of MEP assessments of Cape Cod estuaries. Similarly, the total nitrogen levels at mid-ebb tide within the basins above Rt. 6 (0.61-95 mg N L^{-1}) are well above levels found in basins supportive of high quality benthic animal habitat. These upper basins have significant oxygen excursions and depletions, high chlorophyll a levels as well as accumulations of drift macroalgae (in places), consistent with basins significantly impaired by nitrogen enrichment. It is clear that a significant reduction in nitrogen loading or increase in tidal flushing (or both) will be required for restoration of these upper basins and for the whole of the Bass River Estuarine System.

The results of the water quality and infaunal surveys, coupled with the temporal trends in eelgrass coverage, clearly supports the need to lower nitrogen levels throughout the Bass River Estuary and specifically within the mid and lower reaches of the Bass River and Grand Cove to restore eelgrass habitat. Based on all indicators, the lowering of nitrogen levels will also be necessary to restore infaunal habitat within the upper basins. It is likely that restoration of the impaired infaunal habitats within these upper basins will be achieved with the restoration of eelgrass habitat within the mid and lower reaches of the River.

The eelgrass and water quality information supports the conclusion that eelgrass beds within the lower reach of the Bass River should be the primary target for restoration of the Parker's River Embayment System and that restoration requires appropriate nitrogen management. From the historical analysis, it appears that a large acreage of eelgrass (>300 acres) can be restored, it will be coupled with restoration of large areas of severely degraded benthic animal habitat within the upper estuary (above Rt. 6) as well as improved dissolved oxygen levels that cause periodic fish kills. Therefore, the sentinel station for the Bass River Estuary is located at the long-term water quality monitoring stations within the mid reach of the River (BR-6 & BR-7). These sites were selected based upon its location at the upper most extent of the documented eelgrass coverage in this estuary (Figure VII-6). The concept is to restore the fringing eelgrass beds along the River channel at BR-6 and extensive beds at BR-7 and below.

A single sentinel station was selected at the long term monitoring station, BR-7, for the re-establishment of the expansive beds at this location and in the region between this station and the tidal inlet, as well as the fringing beds within the river channel between BR-7 and BR-6. This determination is directly linked to analysis of the historical eelgrass coverage. The target nitrogen concentration (tidally averaged TN) for restoration of eelgrass at the sentinel location was determined to be 0.42 mg TN L^{-1}, with a secondary check to lower the River channel TN level to ~0.45 mg N L^{-1}. As there has not been significant eelgrass habitat within the Bass River Estuary for over a decade, this threshold was based upon comparison to other local embayments of similar depths and structure under MEP analysis. Similar nearby systems like the Bournes Pond Estuary, where eelgrass has historically been confined to the lower estuarine basin, has nitrogen concentrations supportive of fringing eelgrass at 0.45 mg TN L^{-1} and within the main open water stem of the channel to the upper estuary at lower level, 0.42 mg TN L^{-1}, (analogous to the region between BR-7 and the tidal inlet). The threshold within the main channel region of the Bournes Pond system is supported by the existence of healthy eelgrass beds at tidally averaged TN concentrations of 0.426 mg TN L^{-1} and the presence of eelgrass in patches (not beds & not high quality) at tidally averaged TN of 0.481 mg TN L^{-1}. Additionally, within the lower reach of the Green Pond Estuary, sparse eelgrass is found at tidally averaged TN levels of 0.41 mg TN L^{-1}. Similarly the threshold tidally averaged TN level for restoration of eelgrass in the lower Parker's River was 0.45 mg TN L^{-1}, but only fringing beds were targeted due to the basin configuration.

Although the nitrogen management target is restoration of eelgrass habitat (and associated water clarity, shellfish and fisheries resources), benthic infaunal habitat quality must also be supported as a secondary condition. Benthic animals are more tolerant of nutrient and organic matter enrichment than eelgrass, which requires clear waters and high oxygen levels. At present, in the regions with moderately to significantly impaired infaunal habitat within the Bass River Embayment System have average tidal total nitrogen (TN) levels of 0.52 to 9.5 mg mg N L^{-1}. The observed moderate impairment at this site is consistent with observations by the MEP Technical Team in other enclosed basins along Nantucket Sound (e.g. Perch Pond, Bournes Pond, Popponesset Bay) where levels <0.5 mg N L^{-1} were found to be supportive of healthy infaunal habitat and where moderately impaired habitat was found at ~0.6 mg N L^{-1}. Similarly, the Centerville River system showed moderate impairment at tidally averaged TN levels of 0.526 mg N L^{-1} in Scudder Bay (analogous to the salt marsh dominated Lewis Pond) and at 0.543 mg TN L^{-1} in the middle reach of the Centerville River. Similarly, moderate impairment was also observed at TN levels (0.535-0.600 mg N L^{-1}) within the Wareham River.

Based upon these observations, the MEP Technical Team concluded that an upper limit of 0.52 mg N L^{-1} tidally averaged TN would support healthy infaunal habitat in the embayment basins of the Bass River Estuary. However, it appears that due to the dense SAV in Mill Pond, its tidal restriction and brackish waters suggest a tidally averaged TN of <0.60 mg N L^{-1} is appropriate.

It should be emphasized that these secondary criteria values were not used for setting nitrogen thresholds in this embayment system. These values merely provide a check on the acceptability of conditions within the tributary basins at the point that the threshold level is attained at the sentinel station within the mid reach of the Bass River. The results of the Linked Watershed-Embayment modeling are used to ascertain that when the nitrogen threshold is attained, TN levels in these regions are also within the acceptable range. The goal is to achieve the nitrogen target at the sentinel location and restore eelgrass habitat within the lower reach of the Bass River and restore infaunal habitat throughout the System. The nitrogen loads associated with the threshold concentration at the sentinel location and secondary infaunal check stations are discussed in Section VIII.3, below.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The nitrogen thresholds developed in the previous section were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Bass River System. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for the Bass River System (BR-7 is located approximately at the midpoint of the Bass River System). It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required using: 1) removal of 100% of the septic nitrogen load from upper watersheds except for nitrogen load passing through the freshwater ponds within the watersheds (100% of the septic nitrogen load removed from watersheds 1, 4, 6, 7, 10, 11, 12, 13, 14, 15, 16, 17, 18, 19, 20, 21, 22, 23, 24, 25, 26, 27, 28, and 29) with 2) the removal of 80% of septic nitrogen loading from Bass River Middle Watersheds 31 and 32. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

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

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Run Pond[1]	7.014	7.014	0.0%
Bass River - Lower	29.858	29.858	0.0%
School Street Marsh	9.496	9.496	0.0%
Bass River - Middle	54.512	16.671	-69.4%
Grand Cove	6.159	6.159	0.0%
Dinah's Pond	3.559	0.000	-100.0%
Kelleys Bay	16.408	0.142	-99.1%
Follins Pond	27.085	0.822	-97.0%
Mill Pond and Stream	19.416	0.025	-99.9%

[1] The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.

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

Table VIII-3. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Bass River system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Run Pond	8.384	8.384	0.0%
Bass River - Lower	36.764	36.764	0.0%
School Street Marsh	11.882	11.882	0.0%
Bass River - Middle	67.674	29.833	-55.9%
Grand Cove	7.293	7.293	0.0%
Dinah's Pond	4.337	0.778	-82.1%
Kelleys Bay	20.126	3.860	-80.8%
Follins Pond	34.121	7.858	-77.0%
Mill Pond and Stream	27.238	7.847	-71.2%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Bass River system, with total watershed N loads, atmospheric N loads, and benthic flux			
sub-embayment	threshold load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Run Pond[1]	8.384	0.222	0.000
Bass River - Lower	36.764	2.995	-9.796
School Street Marsh	11.882	0.247	3.610
Bass River - Middle	29.833	3.841	24.042
Grand Cove	7.293	1.071	13.699
Dinah's Pond	0.778	0.310	-1.120
Kelleys Bay	3.860	0.778	17.337
Follins Pond	7.858	2.658	19.540
Mill Pond and Stream	7.847	0.833	0.607
[1] The nitrogen load from Run Pond was inputted into the system as a point source at the mouth of the inlet to the pond.			

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel stations is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station, a reduction in TN concentration of approximately 20% was required at station BR-7.

The basis for the watershed nitrogen removal strategy utilized to achieve the embayment thresholds may have merit, since this example nitrogen remediation effort is focused on watersheds where groundwater is flowing directly into the estuary. For nutrient loads entering the systems through surface flow, natural attenuation in freshwater bodies (i.e., streams and ponds) can significantly reduce the load that finally reaches the estuary. Presently, this attenuation is occurring due to natural ecosystem processes and the extent of attenuation being determined by the mass of nitrogen which discharges to these systems. The 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.

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Bass River system. Sentinel threshold station is in bold print.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Mill Pond	BR-1	0.949	0.609	-35.8%
Follins Pond-Up	BR-2	0.751	0.519	-30.9%
Follins Pond-Lo	BR-3	0.747	0.517	-30.7%
Dinahs Pond	BR-4	0.696	0.493	-29.1%
Kelleys Pond	BR-5	0.695	0.496	-28.6%
Uppermost River	BR-6	0.607	0.458	-24.5%
Upper River	**BR-7**	**0.523**	**0.419**	**-19.9%**
Upper River	BR-8	0.493	0.404	-17.9%
Grand Cove	BR-9	0.520	0.424	-18.4%
Upper River	BR-10	0.438	0.377	-13.9%
Lower River	BR-11	0.389	0.352	-9.5%
Marsh-Lower	BR-12	0.372	0.346	-7.0%
Lower River	BR-13	0.340	0.325	-4.3%
Nearshore	BR-14	0.306	0.306	-0.2%

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.

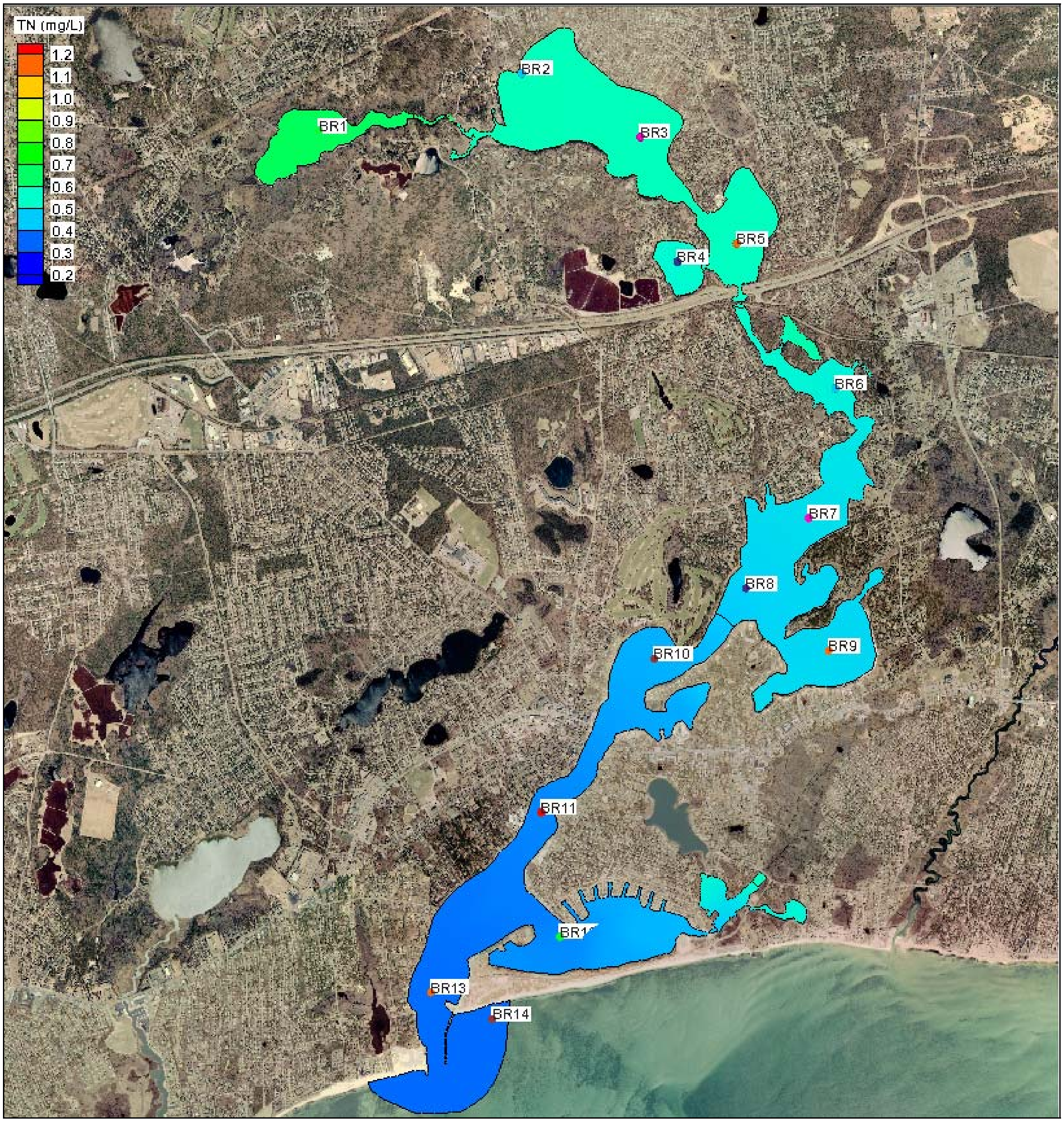


Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in Bass River system, for threshold conditions (0.42 mg/L at water quality monitoring station BR-7). The approximate location of the sentinel threshold station for Bass River (BR-7) is shown.

IX. ANALYSIS OF FLUSHING IMPROVEMENTS RESULTING FROM CULVERTS THROUGH RAILROAD BRIDGE

An analysis was performed to evaluate flushing improvements that would be possible if culverts were added to the railroad bridge crossing of the Bass River. A series of culvert options were investigated, including three where a single 20-foot-wide culvert was placed 30 feet west of the existing bridge span. Three additional modeled options included a second 20-foot-wide culvert placed 20 feet east of the bridge span. The locations of the modeled culverts are shown in Figure IX-1. The single and dual culvert options were simulated with invert depths at -0.5, -4.0 and -6.0 feet NGVD.

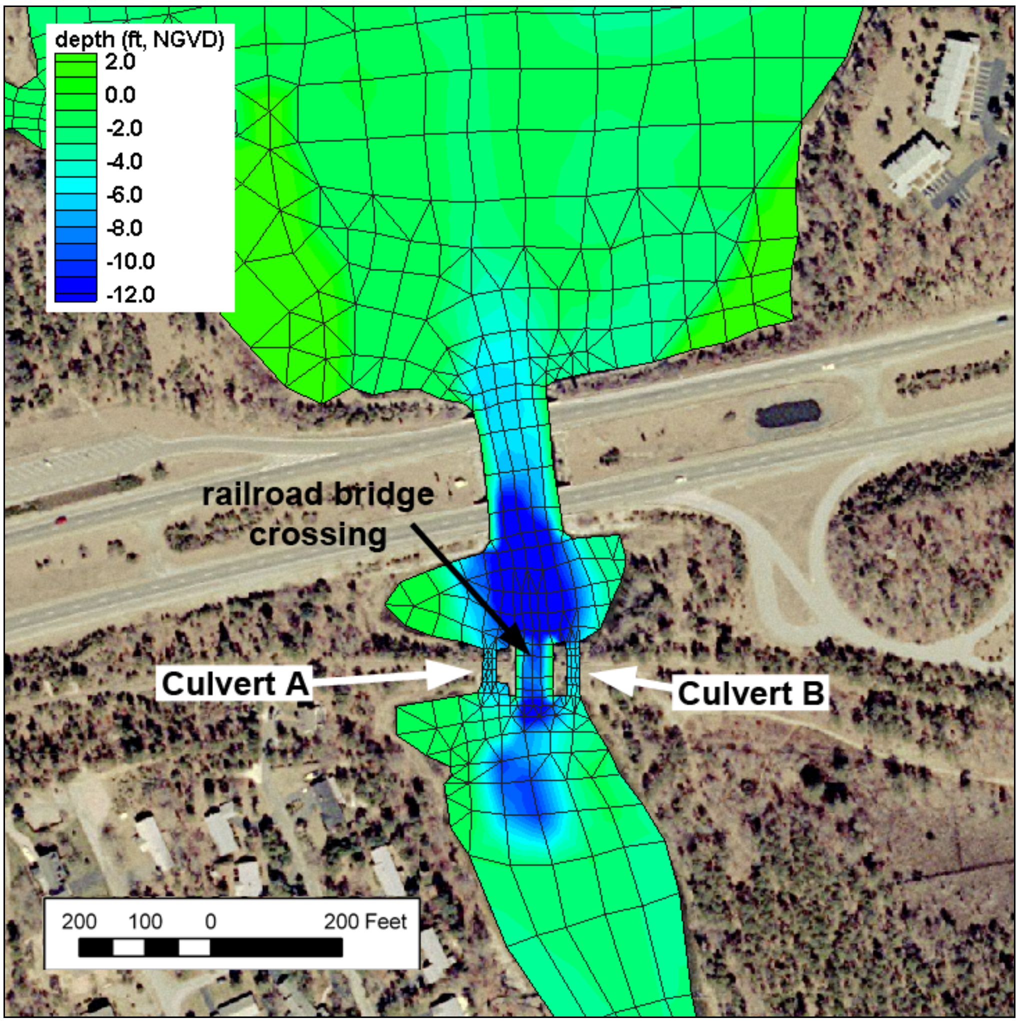


Figure IX-1. Detail of the modified Bass River system grid, showing the locations of the simulated culverts. For the single culvert scenarios, only culvert “A” was included in the model.

Results of the hydrodynamic simulations of the culvert scenarios are presented in Table IX-1 and IX-2, and are also shown in the plot of invert depth versus percent change in tidal prism presented in Figure IX-2. A comparison of the mean volume and tidal prism of the upper Bass River (the portion of the River north of the railroad bridge) is shown in Table IX-1 between each option and present conditions. The resulting changes to computed residence times are shown in Table IX-2.

Table IX-1. Comparison of mean volumes and average tidal prism for Bass River sub-embayments north of the railroad bridge for modeled culvert scenarios.				
Embayment	Mean Volume (ft^3)	Mean volume % change from present	Tide Prism Volume (ft^3)	Tide prism % change from present
Present bridge	90,989,000	-	24,287,000	-
One 20 ft culvert, -0.5 ft NGVD invert	91,126,000	+0.2%	24,896,000	+2.5%
One 20 ft culvert, -4.0 ft NGVD invert	91,074,000	+0.1%	25,408,000	+4.6%
One 20 ft culvert, -6.0 ft NGVD invert	91,074,000	+0.1%	25,665,000	+5.7%
Two 20 ft culverts, -0.5 ft NGVD invert	91,168,000	+0.2%	25,183,000	+3.7%
Two 20 ft culverts, -4.0 ft NGVD invert	91,143,000	+0.2%	26,473,000	+9.0%
Two 20 ft culverts, -6.0 ft NGVD invert	91,141,000	+0.2%	26,708,000	+10.0%

Table IX-2. Comparison of calculated residence times for Bass River sub-embayments north of the railroad bridge, for the modeled culvert scenarios.		
Embayment	System residence time (days)	Local residence time (days)
Present bridge	5.4	1.9
One 20 ft culvert, -0.5 ft NGVD invert	5.2	1.9
One 20 ft culvert, -4.0 ft NGVD invert	5.1	1.9
One 20 ft culvert, -6.0 ft NGVD invert	5.1	1.8
Two 20 ft culverts, -0.5 ft NGVD invert	5.2	1.9
Two 20 ft culverts, -4.0 ft NGVD invert	4.9	1.8
Two 20 ft culverts, -6.0 ft NGVD invert	4.9	1.8

It is apparent from the model results that any new culverts placed at the railroad bridge crossing will need to be large in order to make an appreciable change in the flushing of the upper portion of the River. Two culverts with -6.0 foot invert depths permit a tidal prism increase of 10%. For this same scenario, the local residence time of the upper River decreases by only 0.1 day, which is an indication that even this large double culvert option will improve flushing only marginally. The local residence time of the upper river is presently 1.9 days, which shows that it flushes very well as it exists today.

Model output plotted in Figure IX-3 shows that the largest modeled culvert option will increase the tide range north of the railroad bridge crossing by only 0.2 feet, or 11% of the present tide range (which is directly related to tide prism). This is further evidence that the railroad bridge is not a great flow restriction, and does not presently cause a great reduction in the tide range of the upper Bass River system. Based on these results, the small enhancement of flushing resulting from installation of additional culverts through the railroad bridge will not have a major effect on curtailing the effects of nutrient rich waters in the upper part of the Bass River Estuary. It is clear that a reduction in watershed nitrogen loading will be required for restoration of this system.

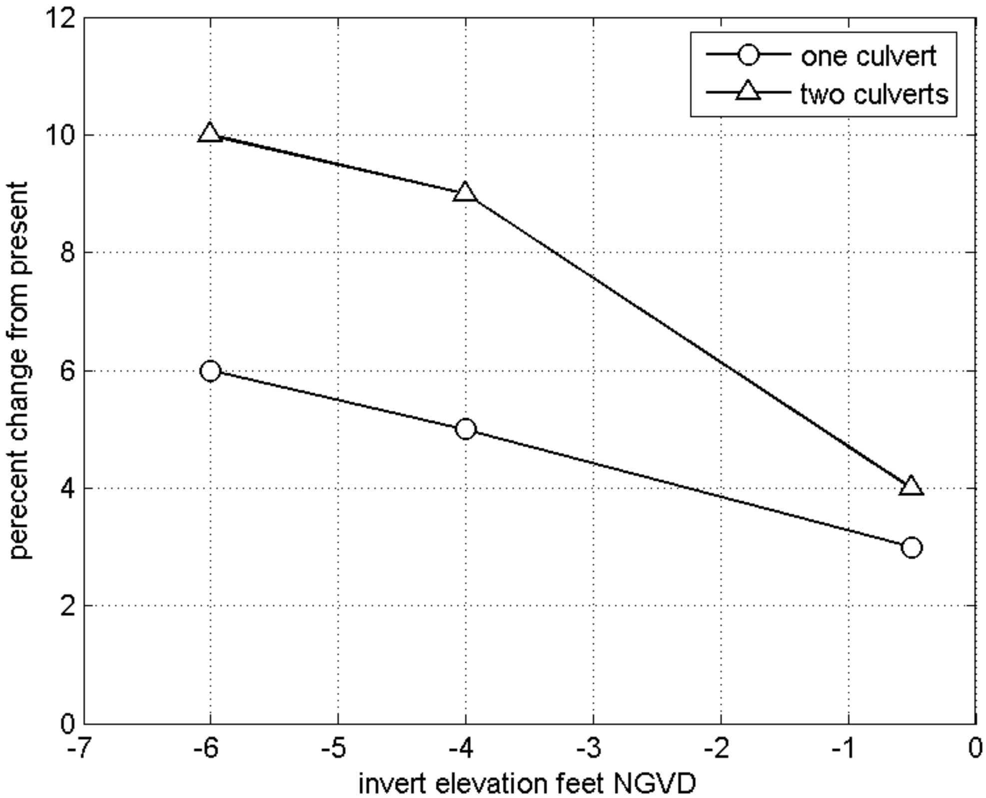


Figure IX-2. Plot of culvert invert elevation versus percent change in the upper Bass River tidal prism, for the modeled one- and two-culvert scenarios.

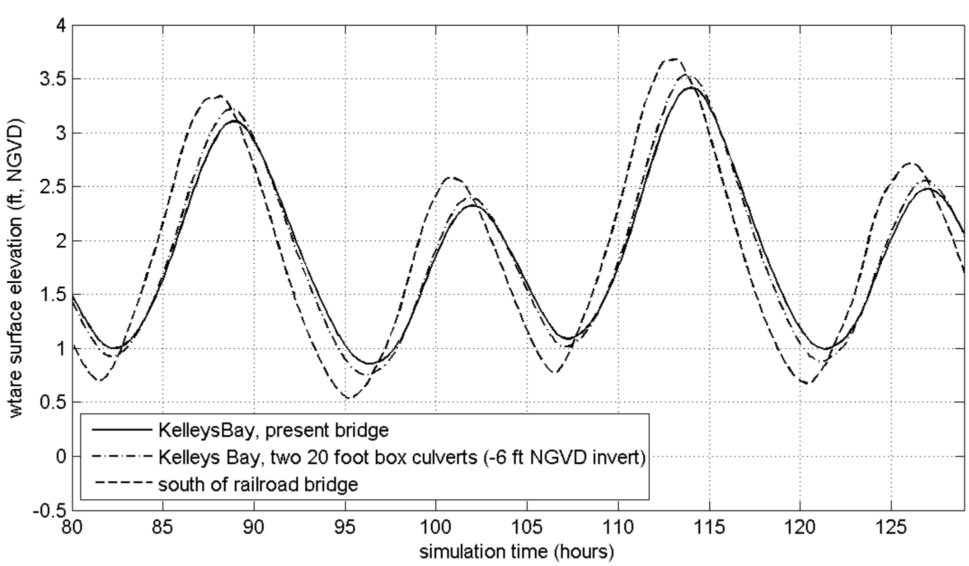


Figure IX-3. Comparison of hydrodynamic model output for present conditions (solid line) and for two 20 foot-wide culverts with invert depths of -6.0 ft NGVD (dot-dashed line).

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