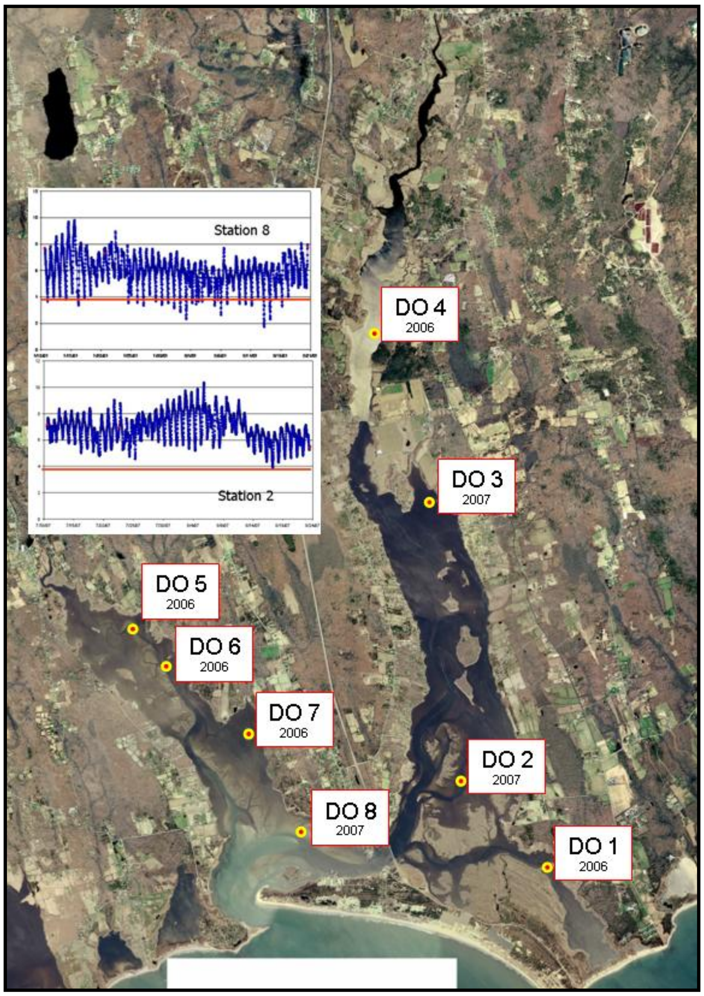
Massachusetts Estuaries Project

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



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

FINAL REPORT – May 2013

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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 Westport River embayment system, a coastal embayment within the Town of Westport, Massachusetts but whose watershed is shared amongst the Towns of Dartmouth, Freetown and Fall River in Massachusetts as well as the Towns of Tiverton and Little Compton in Rhode Island. Analyses of the Westport River embayment system was performed to assist the Town of Westport with up-coming nitrogen management decisions associated with the current and future wastewater planning efforts of the Town, as well as wetland restoration, anadromous fish runs, shell fishery, open-space, and harbor maintenance programs. As part of the MEP approach, habitat assessment was conducted on the embayment based upon available water quality monitoring data, historical changes in eelgrass distribution, time-series water column oxygen measurements, and benthic community structure. Nitrogen loading thresholds for use as goals for watershed nitrogen management are the major product of the MEP effort. In this way, the MEP offers a science-based management approach to support the Town of Westport's resource planning and decision-making process. The primary products of this effort are: (1) a current quantitative assessment of the nutrient related health of the Westport River embayment, (2) identification of all nitrogen sources (and their respective N loads) to embayment waters, (3) nitrogen threshold levels for maintaining Massachusetts Water Quality Standards within embayment waters, (4) analysis of watershed nitrogen loading reduction to achieve the N threshold concentrations in embayment waters, and (5) a functional calibrated and validated Linked Watershed-Embayment modeling tool that can be readily used for evaluation of nitrogen management alternatives (to be developed by the Town) for the restoration of the Westport 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. residential and agricultural 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 Westport River embayment system within the Town of Westport is at risk of eutrophication (over enrichment) from enhanced nitrogen loads entering through groundwater from the increasingly developed watershed to this coastal system as well as ongoing agricultural activity. Eutrophication is a process that occurs naturally and gradually over a period of tens or hundreds of years. However, human-related (anthropogenic) sources of nitrogen may be introduced into ecosystems at an accelerated rate that cannot be easily absorbed, resulting in a phenomenon known as cultural eutrophication. In both marine and freshwater systems, cultural eutrophication results in degraded water quality, adverse impacts to ecosystems, and limits on the use of water resources.

The Town of Westport has recognized the severity of the problem of eutrophication and the need for targeted watershed nutrient management with an initial focus on the largest sources on nitrogen load to the Westport River estuary. The MEP analysis of the Westport River system is yielding results which can be utilized by the Town of Westport to guide the development of a cost effective nutrient load reduction approach and will assist the Town of Westport as it moves forward in implementing a unified approach to nutrient management in the Westport River estuary. The Town of Westport with associated working groups has recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Towns. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns' nutrient management groups to predict and evaluate 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) and the Martha’s Vineyard Commission (MVC) have undertaken the task of providing a quantitative tool to communities throughout southeastern Massachusetts (the Linked Watershed-Embayment Management Model) for nutrient management in their coastal embayment systems. Ultimately, use of the Linked Watershed-Embayment Management Model tool by municipalities in the region results in effective screening of nitrogen reduction approaches and eventual restoration and protection of valuable coastal resources. The MEP provides technical guidance in support of policies on nitrogen loading to embayments, wastewater management decisions, and establishment of nitrogen Total Maximum Daily Loads (TMDLs). A TMDL represents the greatest amount of a pollutant that a waterbody can accept and still meet water quality standards for protecting public health and maintaining the designated beneficial uses of those waters for drinking, swimming, recreation and fishing. The MEP modeling approach assesses available options for meeting selected nitrogen goals that are protective of embayment health and achieve water quality standards.

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

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

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

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

For a comprehensive description of the Linked Model, please refer to the *Full Report: Nitrogen Modeling to Support Watershed Management: Comparison of Approaches and Sensitivity Analysis*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm. A more basic discussion of the Linked Model is also provided in Appendix F of the *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm. The Linked Model suggests which management solutions will adequately protect or restore embayment water quality by enabling towns to test specific management scenarios and weigh the resulting water quality impact against the cost of that approach. In addition to the management scenarios modeled for this report, the Linked Model can be used to evaluate additional management scenarios and may be updated to reflect future changes in land-use within an embayment watershed or changing embayment characteristics. In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries. Unlike many approaches, the Linked Model accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics and accommodates the spatial distribution of these processes. For an overview of several management scenarios that may be employed to restore embayment water quality, see *Massachusetts Estuaries Project Embayment Restoration Guidance for Implementation Strategies*, available for download at http://www.state.ma.us/dep/smerp/smerp.htm.

Application of MEP Approach: The Linked Model was applied to the Westport River embayment system by using site-specific data collected by the MEP and water quality data collected by both the Westport River Watershed Alliance (WRWA) as well as the Coalition for Buzzards Bay (CBB) Bay Watchers Program (assisted technically until 2008 by the University of Massachusetts-SMAST Coastal Systems Program, see Section II). Evaluation of upland nitrogen loading was conducted by the MEP, data was provided by the Town of Westport Planning Department as well as from similar departments in Dartmouth, Freetown, Fall River Tiverton and Little Compton to varying degrees of detail. Watershed boundaries for the Westport River system were delineated by the USGS. This land-use data was used to determine watershed nitrogen loads within the Westport River embayment system and the systems sub-embayments as appropriate (current and build-out loads are summarized in Table IV-3). Water quality within a sub-embayment is the integration of nitrogen loads with the site-specific estuarine circulation. Therefore, water quality modeling of this tidally influenced estuary included a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Once the hydrodynamics of the system was quantified, transport of nitrogen was evaluated from tidal current information developed by the numerical models.

A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the Westport River embayment system given the vigorous mixing in the lower portion of the system due to strong currents as well as the shallow waters of both the East and West Branches of the system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates. Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic model was then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis. Boundary nutrient concentrations in Buzzards Bay source waters were taken from water quality monitoring data. Measurements of current salinity distributions throughout the estuarine waters of the Westport 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, even though the East Branch is officially classified by the Commonwealth as SB. 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 Westport River embayment system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel stations chosen for the Westport 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 Westport River embayment system in the Town of Westport. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment. For illustrative purposes, the MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of the embayment system. The concept was that since nitrogen loads associated with wastewater generally represented the largest source (73%) of N-load in the lower portion of the system (harbor) and the second largest source of N-load behind agriculture in the East and West Branches (36% and 30% respectively) of the controllable watershed load to the Westport 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 Westport 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 Westport River Estuary is showing differences in nitrogen enrichment and habitat quality among its various component basins. Overall, the Estuary is showing some nitrogen related habitat impairment within some of its component basins, however, most of the system is supporting high quality to moderately impaired habitat, with regions of significant impairment resulting primarily from the loss of eelgrass coverage (e.g. mid reach East Branch) or degraded benthic animal habitat (upper East Branch).

Oxygen and chlorophyll a levels were generally consistent with the eelgrass and infaunal animal assessments and paralleled gradients in nitrogen enrichment. The upper and middle section of the East Branch of the Westport River Estuary has large daily oxygen excursions, with moderate to significant oxygen depletion consistent with the significant level of nitrogen enrichment. The salt marsh influenced lower East Branch showed lower nitrogen levels and less oxygen depletion than the upper and mid reaches. This parallels the level of nitrogen enrichment with the lower East Branch showing higher oxygen levels and The Let showing moderate oxygen depletions consistent with its function as a salt marsh basin. However, the chlorophyll and nitrogen levels within The Let indicated high water quality and supports both stable eelgrass beds and high quality benthic animal habitat. The observed levels of oxygen depletion within The Let (and to a lesser extent the lower East Branch) are typical of salt marsh ponds and therefore do not indicate impairment of this basin. The West Branch shows a similar gradient in oxygen depletion as the East Branch, but as it is less nitrogen enriched, the levels of depletion are smaller and less frequent than the East Branch. However, given the frequent large phytoplankton blooms within the upper West Branch and patches of moderately impaired benthic animal habitat with some macroalgal accumulations, it appears that this reach is just above its ability to assimilate additional nitrogen and is showing initial signs of impairment by nitrogen enrichment.

At present, eelgrass exists across a relatively large portion of the system, particularly in the southern portions of the east and west branch and Westport Harbor. However, a major decline in the aerial distribution of eelgrass beds in the Westport River Estuary has occurred primarily in the mid reach of the East Branch between 1951 and 2006. The loss of eelgrass at the uppermost portion of the coverage, with associated decline along the margins is consistent with nitrogen related eelgrass loss and the observed nitrogen levels and resulting chlorophyll a and dissolved oxygen depletions within this portion of the estuary. The condition of the uppermost eelgrass beds in the East and West Branches of the Westport River Estuary observed in 2006 is consistent with the level of nutrient enrichment and associated chlorophyll a and oxygen levels up-gradient from and within the eelgrass beds themselves. In addition, observations that the upper margins of the beds frequently show dense coverage by epiphytes and that the present beds within the coverage area in the mid region of the East Branch are not continuous, but patchy with moderate to sparse density of shoots, is indicative of impaired eelgrass habitat. These observations suggest that the upper region of the present coverage in the East Branch is above its ability to tolerate additional nitrogen enrichment and that continuing loss of coverage is expected in this region.

It is clear that eelgrass coverage is declining in the mid reach of the East Branch of the Westport River Estuary and that the habitat is moderately impaired (primarily by epiphyte growth) at the upper margins of the coverage in the West Branch. Significantly, all of the habitat and water quality indicators support the contention that the significant eelgrass loss and moderate impairments in these basins results from nitrogen enrichment. This is further supported by the timing and pattern of the loss/impairments. The result is that restoration of eelgrass habitat within the Westport Estuary is the primary management concern and nitrogen management is required, specifically targeting this resource.

Overall, the infauna survey measured generally diverse and productive benthic animal communities throughout most of the Westport River Estuary, consistent with the general absence of macroalgal accumulations and the “relatively” recent loss of eelgrass from the mid reach of the East Branch and presence of stable eelgrass beds within the lower basins of the estuary. While some basins are exhibiting impaired benthic animal habitat due to nitrogen enrichment (e.g. upper East Branch) most of the estuary is supporting high quality benthic animal habitat, particularly when the ecological structure of estuarine basin is taken into account (e.g. salt marsh influences).

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 with the caveat that the East Branch of the Westport River is actually classified as SB waters. The threshold was set relative to the restoration of eelgrass lost over the last several decades due to nutrient over-enrichment as well as the typical recreational use of the East Branch as an estuarine water body, consistent with an SA classification. In this system, high habitat quality was defined as supportive of eelgrass and supportive of diverse benthic animal communities. Dissolved oxygen and chlorophyll *a* were also considered in the assessment.

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Town of Westport Westport River embayment system was comprised primarily of agricultural and wastewater nitrogen. Land-use and wastewater analysis found that generally about 30-70% of the controllable watershed nitrogen load to the embayment was from wastewater depending on the sub-basin (East Branch, West Branch or the Harbor).

A major finding of the MEP clearly indicates that a single total nitrogen threshold can not be applied to Massachusetts' estuaries, based upon the results of embayment specific nutrient analyses undertaken by the MEP in over 50 estuarine systems across southeastern Massachusetts (inclusive of nearby Slocums River and Little River, New Bedford Harbor and Wareham River). This is almost certainly going to be true for the other embayments within the MEP area, as well, the Westport River estuary system.

The threshold nitrogen levels for the Westport River embayment system in the Town of Westport were determined as follows:

Westport River Threshold Nitrogen Concentrations

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

- In the Westport River System eelgrass exists in both the East and West Branches as well as in The Let and Westport Harbor. Within the East Branch, eelgrass historically existed near Upper Spectacle Island (between water quality stations E-56 and E-33). Eelgrass was documented in the region of Upper Spectacle Island in the 1951 analysis and in MassDEP field survey in 1995. However, eelgrass was not observed in 2001 or

2006 surveys in this region, at tidally averaged TN levels of 0.64 mg N L^{-1} (Station E-56). Instead, since 1995, eelgrass coverage has been limited to the lower basin of the East Branch, south of Great Island, including within The Let. Presently, the eelgrass beds just south of Great Island are heavy with epiphytes and while they persist, they are clearly impaired and diminishing at the current TN level, 0.51 mg N L^{-1} (Station E-41).

- Similar to the East Branch of the Westport River Estuary, the West Branch supported historically greater coverage of eelgrass than today. In 1951 eelgrass covered most the non-shoal areas of from Hicks Cove south, with a small fringing bed extending north along the eastern shore. At present, eelgrass still exists in fringing beds along the western shore but mainly near Westport Harbor and beds also exist along the eastern shore, but with dense epiphyte growth, except near the Westport Harbor basin. The unimpaired beds are associated with tidally averaged TN levels less than 0.421 mg N L^{-1} (Station W-9). The stable eelgrass beds within Westport Harbor are found at tidally averaged TN levels of 0.33-0.40 mg N L^{-1} (Stations E-26, W-6, N-12). Both Branches show the typical pattern of eelgrass loss associated with nitrogen loading, with eelgrass being lost from the uppermost regions of each basin and the deeper waters first, appearing to "retreat" toward the inlet.

- The results indicate that eelgrass has been lost from the Westport River Estuary in areas that presently support tidally averaged TN levels of 0.57 mg N L^{-1} and >0.50 mg N L^{-1} in the East and West Branches, respectively. At lower nitrogen levels eelgrass is persisting, but with epiphytes and losses of coverage from the upper and deeper areas of the beds. These sites are associated with 0.51 mg N L^{-1} and ~0.50 mg N L^{-1} in the East and West Branches, respectively, while "healthy" beds are found at lower concentrations, with <0.428 mg N L^{-1} and 0.421 mg N L^{-1} in the East and West Branches, respectively, and <0.400 mg N L^{-1} in Westport Harbor. It appears that in the Westport River Estuary, the TN level to support high quality eelgrass habitat may be greater than 0.43 mg N L^{-1}, but less than 0.50 mg N L^{-1}.

- The restoration of eelgrass habitat within the East and West Branches to documented historic coverages will require lowering the present nitrogen levels. In the East Branch it is clear that extensive eelgrass coverage within the immediate area south of Lower Spectacle Island has been lost, as seen in the 1951 and 1995 analyses compared to the more recent 2001 and 2006 surveys. To restore eelgrass to the 1951 and 1995 levels, TN concentration will need to be lowered to 0.49 mg L^{-1} at the Sentinel Station in the East Branch, determined to be the long-term water quality station E-33 (Section VI). Similarly within the West Branch, to prevent further loss of eelgrass and to restore eelgrass to 1951 and 1995 levels, tidally averaged TN at long-term water quality station W-12 needs to be similarly lowered to 0.48 mg L-1. Westport Harbor is presently supporting high quality eelgrass habitat.

It is important to note that the analysis of future nitrogen loading to the Westport 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 Westport River estuarine system is that restoration will necessitate a reduction in the present (Westport 2008, Dartmouth and Fall River 2009, Freetown 2005 and Tiverton 2010) nitrogen inputs and management options to negate additional future nitrogen inputs.

Table ES-1. Existing total and sub-embayment nitrogen loads to the estuarine waters of the Westport Harbor estuary system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations. Note: present load is based on summertime measured value of nitrogen flux at the Old County Road stream gauge on the freshwater portion of the Westport River.

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. (mg/L)
SYSTEMS										
North East Branch	7.071	93.789	9.299	--	103.088	4.360	-30.369	75.513	0.874 - 1.440	--
West Branch	3.416	26.362	6.540	--	32.901	11.154	-6.288	37.768	0.449 - 0.649	--
South East Branch	29.471	46.477	15.855	--	62.332	20.922	-16.675	63.385	0.626 - 0.864	--
The Let	2.63	4.312	1.447	--	5.759	1.968	11.811	19.538	--	--
Westport Harbor	1.422	3.660	6.592	--	10.252	8.226	-30.507	-12.028	0.534 - 0.538	--
Old County Road	63.805	114.353	48.260	--	162.614	--	-	162.614	--	--
Kirby Brook	5.986	13.167	7.786	--	20.953	--	-	20.953	--	--
Adamsville Brook	18.721	30.556	17.066	--	47.622	--	-	47.622	--	--
Angeline Brook	5.477	31.219	3.077	--	34.296	--	-	34.296	--	--
Snell Creek	1.866	3.581	4.556	--	8.137	--	-	8.137	--	--
System Total	**139.865**	**367.477**	**120.477**	**--**	**487.954**	**46.630**	**-72.027**	**457.798**	**--**	**0.48 – 0.49[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 sites located in the East Branch at water quality stations E33 (0.49 mg/L) and in the West Branch at W12 (0.48 mg/L)

Table ES-2. Present Watershed Loads, Thresholds Loads, and the percent reductions necessary to achieve the Thresholds Loads for the Westport Harbor estuary system, Town of Westport, Massachusetts.

Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed reductions needed to achieve threshold load levels
SYSTEMS						
North East Branch	103.088	93.030	4.360	-30.369	67.021	-9.8%
West Branch	32.901	32.901	11.154	-6.288	37.768	0.0%
South East Branch	62.332	46.477	20.922	-16.675	50.723	-25.4%
The Let	5.759	5.759	1.968	11.811	19.538	0.0%
Westport Harbor	10.252	10.252	8.226	-30.507	-12.028	0.0%
Old County Road	162.614	111.816	--		111.816	-31.2%
Kirby Brook	20.953	13.167	--		13.167	-37.2%
Adamsville Brook	47.622	47.622	--		47.622	0.0%
Angeline Brook	34.296	34.296	--		34.296	0.0%
Snell Creek	8.137	3.581	--		3.581	-56.0%
System Total	**487.954**	**398.901**	**46.630**	**-72.027**	**373.504**	**-18.3%**

(1) Composed of combined natural background, fertilizer, runoff, and septic system loadings and present load is based on summertime measured value of nitrogen flux at the Old County Road stream gauge on the freshwater portion of the Westport River.
(2) Target threshold watershed load is the load from the watershed needed to meet the embayment threshold concentration identified in Table ES-1.
(3) Projected future flux (present rates reduced approximately proportional to watershed load reductions).
(4) Sum of target threshold watershed load, atmospheric deposition load, and benthic flux load.

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I. INTRODUCTION

The Westport River Estuary is a complex estuarine system located in southeastern Massachusetts on the Massachusetts – Rhode Island State Boundary (Figure I-1). The system is comprised of 2 drown river valley estuaries (east and west branches), a coastal lagoon (Westport Harbor) and a relict tidal inlet (the Let). Westport Harbor, situated at the confluence of both the east and west branches, exchanges tidal waters with Buzzards Bay through a single unarmored tidal inlet to the southwest (Figure I-2). While the Westport River Estuary is located primarily within the Town of Westport, the watershed is located within both Massachusetts and Rhode Island. The watershed falls within the jurisdiction of 5 municipalities: most of the area is within the Town of Westport, with northern portions within the City of Fall River and eastern areas within the Town of Dartmouth. Rhode Island includes small areas of the western watershed region, within the Towns of Tiverton and Little Compton.

The present-day configuration of the Westport River embayment results from the drowning of a river valley by early Holocene post-glacial sea level rise. The surficial geology in which the Westport River Estuary is situated results from the Wisconsin glaciation. Upland areas have a compact, silty, bouldery till, while lowland areas are generally underlain by coarse gravel and medium to coarse outwash sands. Outwash deposits typically are less than 30 feet thick. Glacial scouring and subsequent deposition has resulted in a topography of gently rolling northwest to south west trending hills and broad stream valleys (Hoagland et al. 1987). The watershed boundaries are defined primarily by a bedrock morphology of several low ridges and valleys running roughly in a northward and northeastward direction (Zen, 1983). The mouth of the Westport River embayment is defined by bedrock outcrops on the west side of the inlet and Cherry and Webb / Horseneck Beach to the east. In addition to the bedrock morphology, large amounts of sand occur within the lower embayment and within and beyond the mouth of the estuary. The sand results from coastal processes, occurring as a dynamic and variable spit and bar system, which has strongly influenced the configuration and efficiency of the tidal inlet over at least the past 70 years (Fitzgerald, et al, 1993). These coastal processes that define the barrier beach along the system's southern shore also drove the lagoon that is Westport Harbor.

At present, the Westport River Estuary is a tidal embayment with marine waters entering from Buzzards Bay and freshwater entering via direct groundwater discharge and through a number of streams inflows. With a relatively large watershed and consequent substantial fresh surface water inputs, the Westport River Estuary has a variable salinity gradient that is strongly influenced by both short-term and seasonal rainfall patterns. The principal surface water inflows are the Westport River (which begins as the Shingle Island River up-gradient of Lake Noquochoke), which discharges into the head of the east branch and accounts for >67% of the total freshwater input to the east branch and Adamsville Brook discharging to the head of the west branch and accounting for 58% of the total freshwater inflow to this basin. Other streams that discharge to the estuary include, in order of diminishing freshwater contribution: Kirby Brook and Snell Creek that discharge to the east branch and Angeline Brook which discharges to the west branch. There are a few smaller more seasonal streams which account for minimal freshwater inflow.

The Westport River estuary and its watershed constitute one of the most important natural resources and cultural components of the Town of Westport (as well as the Town of Dartmouth to the east and Tiverton to the west). As a large estuary in a populous region, the Westport River brings two opposing elements to bear: 1) as a protected marine shoreline it is a popular region for boating, recreation and land-development; 2) as an enclosed body of water the

Westport River may not be readily flushed of pollutants that enter the estuary from its associated watershed. Pollutants, particularly nitrogen, entering from the watershed, particularly in the upper most reaches of both the east and west branches furthest away from the tidal inlet will tend to have the largest impact on the estuary. As a result, the Westport River, like many shallow coastal embayments in the region, has become nutrient enriched as a result of changing land-use from forest and fields to agricultural uses over the past centuries and more recently to residential and commercial development. Current nitrogen loading to the Westport River is from three principal sources: agricultural activities; atmospheric deposition of nitrogen compounds on the land and water surface and onsite disposal of wastewater. From the point of view of controllable nitrogen load, the principal sources are agricultural activities and onsite disposal of wastewater. Loading of the critical nutrient (nitrogen) to the embayment has increased over the last four decades with further increases certain unless nitrogen management is implemented. Presently, habitat degradation resulting from nutrient enrichment is the single major ecological threat to the aquatic resources of the Westport River Estuary.

Figure I-1. Location of the Westport River Estuary which drains into Buzzards Bay / Rhode Island Sound.

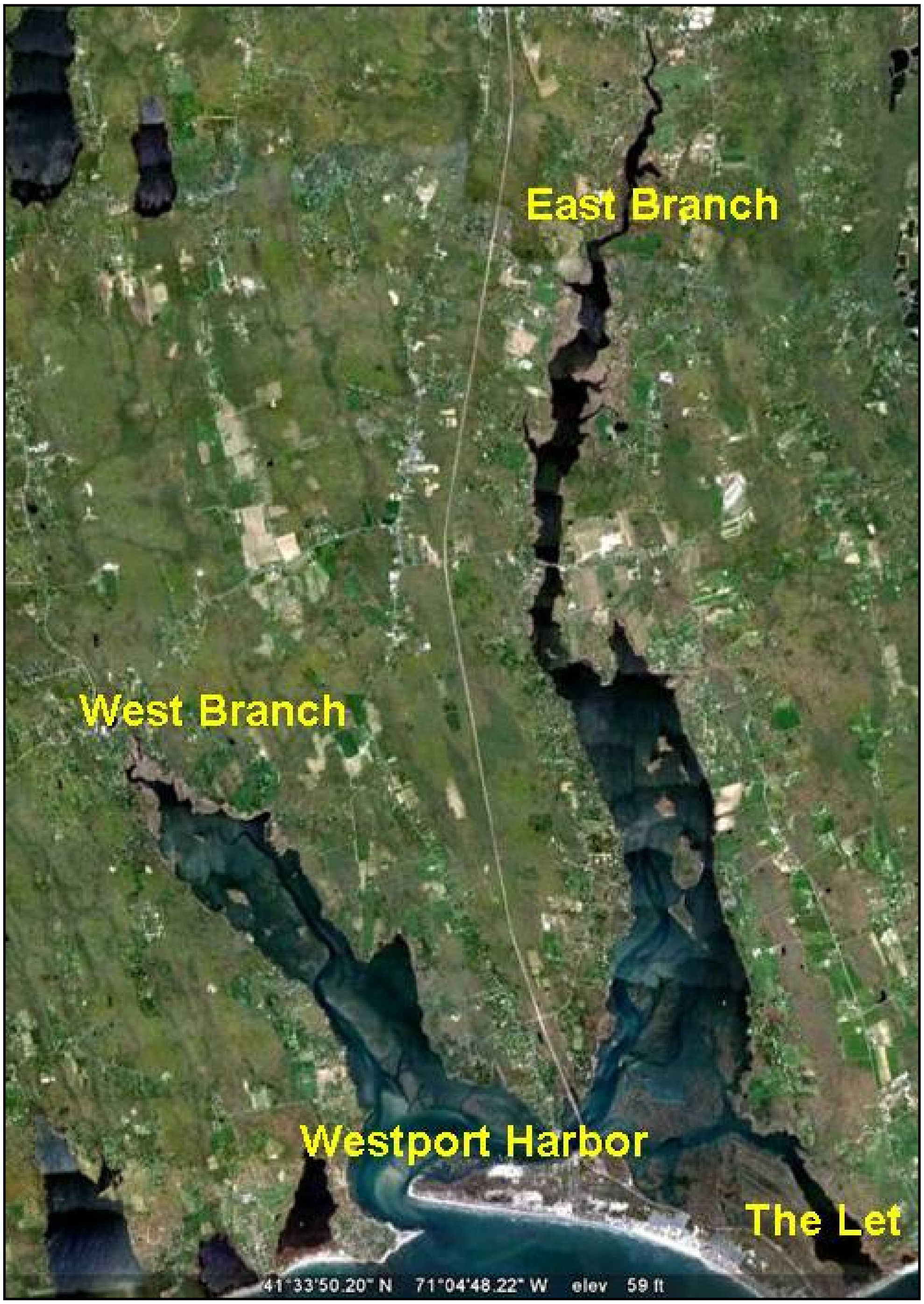


Figure I-2. Study region proximal to the East and West Branches of the Westport River Embayment System for the Massachusetts Estuaries Project assessment and modeling analysis. Tidal waters enter the Westport River lower basin (Westport Harbor) through a large inlet defined by Horseneck Point to the east and the "Knubble" (Westport Light) to the west. The primary freshwater discharges into the system are the Westport River entering at the head of the East Branch and Adamsville Brook entering at the head of the west branch of the Westport River Estuary. Smaller streams enter the system along the east and west shores of both branches. (Source: Google Earth).

Regular documentation of a decline in ecological health of the Westport River began in 1993 with the start of the BayWatchers Program in Westport and Dartmouth and other Buzzards Bay community's coastal waters. Water quality monitoring was undertaken by the Coalition for Buzzards Bay in concert with the Westport River Watershed Alliance. The BayWatchers data from 1993 to 2006 show the upper portions of the east branch of the Westport River to be among the poorest in nutrient related habitat quality of the more than 65 embayment segments surveyed throughout Buzzards Bay (Coalition for Buzzards Bay 2007). The present MEP effort arose directly from the efforts of concerned citizens, municipal officials and staff and local advocates along with the Community Preservation Committee to restore the health of the Westport River Estuary for the citizen's of the Town of Westport and surrounding communities.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. Nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over system assimilative capacity, which begins to cause declining ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities. At higher levels, enhanced nutrient loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human use. Similar to nutrients, bacterial contamination is related to changes in land-use as watersheds become more developed. Regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to culture, economy, and tax base of Massachusetts's coastal communities.

The primary nutrient causing the increasing impairment of the Commonwealth's embayments is nitrogen resulting from changing land-uses in coastal watersheds 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 are grappling with Comprehensive Wastewater Planning and/or environmental management issues related to the declining health of their estuaries. Municipalities are seeking guidance on the assessment of nitrogen sensitive embayments, as well as available options for meeting nitrogen goals and approaches for restoring impaired systems. Many of the communities have encountered problems with "first generation" watershed based approaches, which do not incorporate estuarine processes. The appropriate method must be quantitative and directly link watershed and embayment nitrogen conditions. This "Linked" Modeling approach must also be readily calibrated, validated, and implemented to support planning. Although it may be technically complex to implement, results must be understandable to the regulatory community, town officials, and the general public.

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

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

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

The major Project goals are to:

- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of each of the 70 out of 89 embayments in Southeastern MA
- provide necessary data collection and analysis required for quantitative modeling,
- conduct quantitative TMDL analysis, outreach, and planning,
- keep each embayment's model "alive" to address future regulatory needs.

The core of the Massachusetts Estuaries Project analytical method is the Linked Watershed-Embayment Management Modeling Approach. This approach represents the "next generation" of nitrogen management strategies. It fully links watershed inputs with embayment circulation and nitrogen characteristics (Figure I-3). 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.

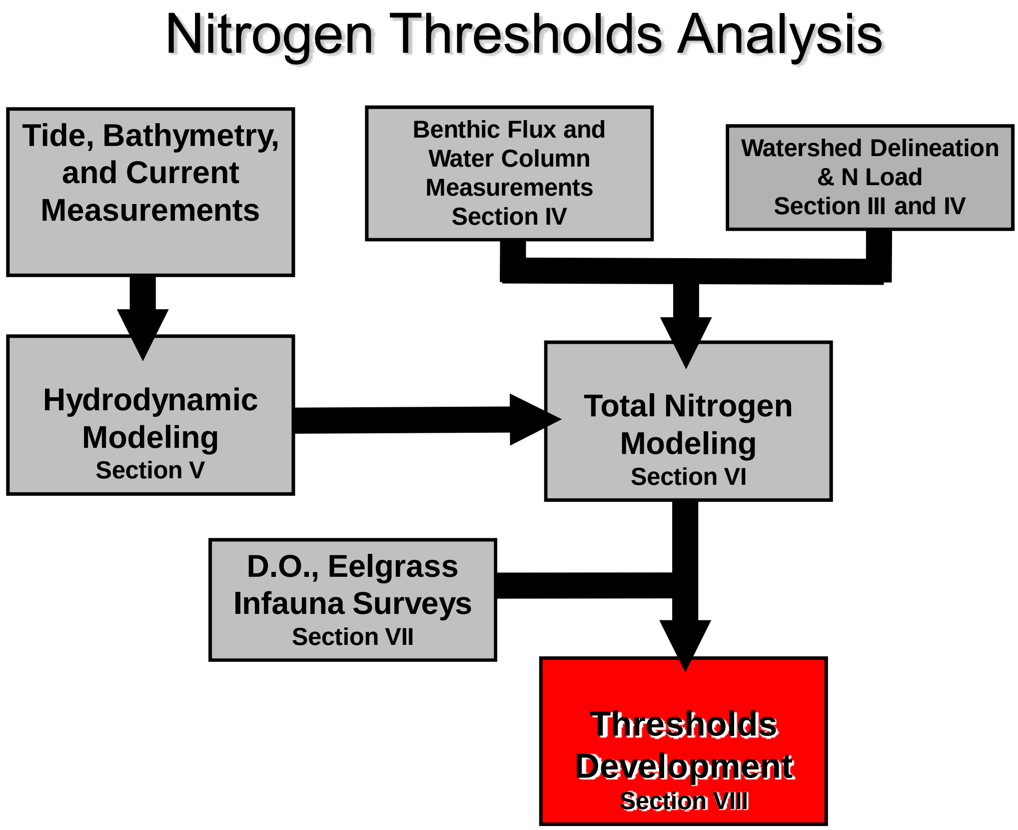


Figure I-3. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Section numbers refer to sections in this MEP Nutrient Technical Report where the specified information is provided. Note that this is a modeling "approach" with multiple inputs and outputs governed by QA/QC at each step, not a single model.

The Linked Model has been applied for watershed nitrogen management in ca. 50 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options.

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

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

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

I.2 SITE DESCRIPTION

The Westport River embayment is oriented roughly north to south and is open to Buzzards Bay on the south via a single tidal inlet. The configuration of the embayment results from post-glacial sea-level rise drowning of shallow valleys formed by both long-term and short-term processes. Long-term processes include erosion of the bedrock by chemical and physical weathering of the bedrock over the past 30 million years or more, which maintained a drainage network in the region. The bedrock underlying the soils of the region is crystalline bedrock of about 600 million years ago (Zen, 1983). The bedrock types include granite, gneiss, schist and other rocks (Murray 1990). Over the past 600,000 years a series of repeated continental glaciations have advanced and retreated across the region, with the most recent ending about 16,000 years ago. The glaciations lowered the local bedrock erosional surfaces by small amounts, leaving smoothed upland bedrock surfaces and a variety of glacial sediments covering much of the underlying bedrock structure. Bedrock outcrops occur throughout the watersheds and the exposed and buried bedrock topography is the primary control upon the local topography. The watershed shape and length is defined by a framework of roughly north-northeastward trending, low bedrock ridges with shallow intervening valleys. Where the low valleys meet the sea, the location of the Westport River mouth is anchored to the west by the configuration of the low bedrock ridges, outcrops and submarine reefs (FitzGerald et al, 1993) with a more dynamic barrier beach to the east. The east branch of the Westport River embayment is about 14.5 km from the head where the fresh Westport River discharges to the inlet. By comparison, the west branch is approximately 5 km in length from the Adamsville

Brook discharge to the inlet of the overall system. On average the west branch is approximately 1 km wide whereas the lower portion of the east branch is ~1.5 km and narrows moving north to ~1 km. Approximately 1.25 km south of the Hixbridge Road crossing, the east branch narrows significantly and appears as a tidal river ~200 to 300 meters across and even narrower as one approaches the Old County Road bridge crossing. The definition of the watershed basin predominantly by bedrock creates the crenulated margins, as opposed to the smooth watershed borders seen in sand outwash aquifers, like on Cape Cod. This difference in geology allows for the use of topographic techniques as part of delineating the contributing areas to these estuaries.

The soils and sediments of the two watersheds consist of a variable fabric of low permeability basal till which has been compressed by the weight of ice of a thickness of several hundred feet and by more permeable ablation or melt-out till that drapes the bedrock ridges and ridge slopes. Basal till usually forms the bottom-most sediment in the valley floors (Williams and Tasker, 1978; Larson, 1982). The area bedrock has a very low permeability, while the basal tills (locally known as hardpan) and ablation tills have low to moderately low permeability. The shallow valleys and lower elevations have a variety of glacial sediments in them usually described as stratified drift, which have substantially higher permeabilities than the glacial tills.

The combination of exposed and shallowly buried bedrock and generally low permeability glacial tills affects the upland hydrology of the watersheds to the east and west branch. The regions' bedrock and low permeability upland soils form a surface-water dominated regime where rainfall and snow melt tend to flow over the ground surface in a greater proportion than they percolate into the soils to become groundwater (Williams and Tasker, 1978; Bent 1995). The upland stream network is therefore well-developed and most upland groundwater flow is local, that is to the nearest stream.

In the shallow valleys with more permeable soils, the proportion of runoff in "undisturbed" settings is reduced and the role of groundwater in the valleys is proportionately more substantial. Streams that flow in till-dominated watersheds are likely to have "flashy" flow characteristics. That is the streams display both rapid rise and fall in amount of flow in response to rainfall or snowmelt. In the smaller tributary watersheds till-dominated streams will seasonally cease flowing for a week or more during late summer during a typical summer. Streams that flow in stratified drift dominated watersheds display more moderate flow characteristics and typically will flow year round even in dry summers. The largest stream, the Westport River drains more than half of the total Westport River embayment watershed. The main stream channel flows across valley sediments which are generally stratified drift and very permeable, while its short tributaries flow generally from till-dominated sub-watersheds (Williams and Tasker, 1978; Bent 1995).

Because of the layout of the two branch embayment, both branches are mixing zones for watershed derived freshwater and marine waters. Because of the length difference between the large length of the east branch and the much shorter west branch, as well as the difference in the volume of fresh surface water discharging to both branches and the significantly different watershed area of each branch, the tidal waters exhibit strong contrasts in salinity from headwaters to the tidal inlet. At the northern reach of the east branch of the estuary there is a highly variable but usually low salinity, which is maintained by flows from the Westport River, while the lower freshwater inflows result in higher salinities at comparable locations in the upper west branch. In both branches mixing processes result in predicable salinity gradients which increase towards the tidal inlet. The salinity gradients vary generally based upon surface water inflow rates with periods of high stream discharge resulting in a "freshening" of embayment waters and a brackish water zone that can extend southward to include much of the upper half of the estuary. Thus, the salinity regime of the upper half of the embayment is very dynamic and sensitive to recent rainfall patterns. However, given the relatively constant rainfall over the year, the salinity gradient is sufficiently consistent to support stable estuarine plant and animal communities.

I.3 NITROGEN LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In mixed stratified drift and glacial till watershed, such as in the watershed to the Westport River embayment system, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (Weiskel and Howes 1992). Since stream baseflow (flow provided by groundwater rather than surface water runoff) for the principal streams in the Westport River watersheds is provided by groundwater (Bent, 1995), much of the phosphorous generated in the watershed is retained by the soils and the watershed tends to release relatively moderate amounts of phosphorus to the Westport River. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through the stratified drift in the watershed valleys, both during surface flow and through groundwater systems (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 Westport River follow this general pattern, where the primary nutrient of eutrophication in these systems is nitrogen. This is consistent with the N/P ratios (inorganic N to inorganic P) generally less than 6 observed by the long-term water quality monitoring program (BayWatchers) that are much lower than the Redfield Ratio of 16 where <16 suggests N limitation.

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

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

Protection and restoration of coastal embayments from nitrogen overloading has resulted in a focus on determining the assimilative capacity of these aquatic systems for nitrogen. While this effort is ongoing (e.g. USEPA TMDL studies), southeastern Massachusetts has been the site of intensive efforts in this area (Eichner et al., 1998, Costa et al., 1992, 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 Westport River Estuary as monitored by the Coalition for Buzzards Bay BayWatchers Monitoring Program with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) to "refine" general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Even while the lower portions of the Westport River Estuary currently support relatively healthy habitat, the upper reaches of the Westport River Estuary (particularly the east branch) are supporting elevated nitrogen levels and are clearly resulting in impaired habitat. However, much of the benthic animal habitat within this estuarine system is presently unimpaired or only slightly impaired. Similarly, within the lower tidal reaches, eelgrass beds have been stable for many decades, although coverage within the upper reaches is declining. Together these habitat indicators indicate a system presently beyond its ability to assimilate additional nutrients without impacting ecological health (i.e. beyond its assimilative capacity). The result is that nitrogen management of this system should be aimed at restoration of impaired habitat through the management of potential increases in watershed nitrogen loading.

In general, nutrient over-fertilization is termed "eutrophication" and when the nutrient loading is primarily from human activities, "cultural eutrophication". Although the influence of human-induced changes has increased nitrogen loading to the system and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within the Westport River could potentially occur, wholly or in part, without human influence. It is critical to separate human induced versus natural processes in the nutrient threshold analysis. While this partitioning would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a "pristine" system.

I.4 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into Westport River. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for the system. Once the hydrodynamic properties of the estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

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 historic watershed delineation developed by the USGS and the BBP and refined by SMAST/MEP staff for watershed and sub-watershed areas designated by MEP. Land-uses were based upon available assessors data with significant QA by the Town staff. Almost all nitrogen entering the Westport River embayment is transported by surface water (streams) with a portion being transported via direct groundwater discharge to the estuary. Concentrations of total nitrogen and salinity of Buzzards Bay source waters and throughout the Westport River systems were measured by MEP Staff during the MEP field data collection campaigns in 2005 and 2006 and integrated with the Coalition for Buzzards Bay's BayWatcher data (2003-2009). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the system were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Westport River system for the Town of Westport. 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 “load” also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Southeastern Regional Planning and Economic Development District (SRPEDD) data with refinement using additional town specific databases. Offshore water column nitrogen values were derived from an analysis of a monitoring station in Buzzards Bay (Section IV and VI). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section VI. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of each embayment was performed that included a review of existing water quality information, temporal changes in eelgrass distribution, dissolved oxygen records and the results of a benthic infaunal animal analysis and other bioassays (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration in receiving estuarine system. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the Westport

River system, although future management alternatives are anticipated as part of the Town's restoration effort.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

Nutrient additions to aquatic systems cause shifts in a series of biological processes that can result in impaired nutrient related habitat quality. Effects include excessive plankton and macrophyte growth, which in turn lead to reduced water clarity, organic matter enrichment of waters and sediments with the concomitant increased rates of oxygen consumption and periodic depletion of dissolved oxygen, especially in bottom waters, and the limitation of the growth of desirable species such as eelgrass. Even without changes to water clarity and bottom water dissolved oxygen, the increased organic matter deposition to the sediments generally results in a decline in habitat quality for benthic infaunal communities (animals living in the sediments). This habitat change causes a shift in infaunal communities from high diversity deep burrowing forms (which include economically important species), to low diversity shallow dwelling organisms. This shift alone causes significant degradation of the resource and a loss of productivity to both the local shell fisherman and to the sport-fishery and offshore fin fishery, which are dependent upon these highly productive estuarine systems as a habitat and food resource during migration or during different phases of their life cycles. In addition, the diverse avian fauna which feed upon infauna or fish communities are also affected and their numbers and diversity declines. This overall nutrient driven process is generally termed “eutrophication” and in embayment systems, unlike in shallow lakes and ponds, it is not a necessarily a part of the natural evolution of a system.

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

These tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specifics for any given water body. 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 Westport 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.

In the case of the Westport River Estuary, there are historical studies relating to environmental functioning and habitat health conducted over the past three decades that have been used to inform the MEP habitat assessment and threshold development process. Among these are number of studies relating to nitrogen loading, hydrodynamics and habitat health, which are described briefly below.

Study to Determine the Causes, Types and Locations of Pollutants Contaminating the Westport River Estuary. - Prior to the Massachusetts Estuaries Project, a series of studies were completed by researchers from the Hydrology Research Group at Boston University. These studies were generally undertaken between 1984 and 1986. The goal was to address the growing concern over bacterial contamination in the Westport River Estuary and an associated increase in the number of shellfish bed closures. The report sought to identify sources of bacterial pollution in the Westport River watershed and estuary and develop recommendations for reducing the level and extent of bacterial contamination occurring in this system. Field work involved extensive sampling of groundwater flowing to the estuary as well as fresh surface water. Estuarine samples were also collected. All samples were assayed for both fecal coliform as well as total coliform and fecal streptococci bacteria. The results indicated that much of the bacterial contamination in the Westport River likely stemmed from agricultural activities, particularly associated with animal wastes, with a smaller proportion resulting from domestic wastewater from septic systems. Stormwater runoff was thought to be a major pathway for transport of these bacteria from watershed to estuarine waters. No significant sources were found within the estuary itself. While bacterial contamination is not the main focus of the MEP analysis, aspects of the early work undertaken by the Hydrology Research Group related to surface water flow and watershed characteristics served as useful references to the MEP developed stream flow volumes and watershed delineations.

Relationships between Suspended Sediment and the Movement of Bacteria in East Branch of the Westport River - This study, undertaken in parallel with the bacteria study described above, focused on understanding the relationship between surface water input of suspended sediments to the East Branch of the Westport River and the relationship of sediment load to bacterial contamination. Over the course of the study, efforts were made to quantify the flows and associated sediment loads for the major fresh surface water inflows to the East Branch as well as to examine tidal characteristics of the estuarine reach of the East Branch. Not surprisingly, the investigators determined that while sediment deposition occurs throughout the Westport River Estuary, suspended sediment load is most pronounced in the upper reaches of the East Branch. The observations were consistent with both the watershed characteristics and the high freshwater inflows to the upper east branch.

Hydrogeology and Contamination Investigation of the West Branch of the Westport River Watershed - This report, also completed by researchers from the Hydrology Research Group at Boston University over the same time frame, sought to identify sources of bacterial pollution in the watershed to the West Branch of the Westport River as well as the estuary and develop recommendations for reducing the extent of bacterial contamination occurring in this system. Field work involved extensive sampling of fresh surface water, mainly Adamsville Brook, Angeline Brook and Hulda Brook. The results indicated that much of the bacterial contamination occurring in the West Branch of the Westport River could be grouped into three main categories (not ranked by importance): 1) inadequate septic systems, 2) agricultural run-off (primarily associated with animal waste management) and 3) stormwater road run-off. According to this investigation, sub-surface and surface transport of septic leachate and road run-off are largely responsible for the degradation of Adamsville Brook discharging to the head of the West Branch of the estuary. Some of the remedial measures suggested in the report include upgrading septic systems to match local hydrogeologic conditions, implementation of Best Management Practices on local farms to reduce potential for bacterial contamination from animal husbandry activities and detention basins for management of road run-off. This report was a helpful benchmark for comparison to MEP developed stream flow volumes and watershed delineations for both Adamsville Brook and Angeline Brook.

Westport Point Stormwater Monitoring - The Westport River Watershed Alliance (WRWA) has been active in data collection related to the management and protection of the Westport River Estuary and its associated aquatic resources for several decades. Recently (2010) WRWA teamed with the SMAST Coastal Systems Program to conduct storm water monitoring at an outfall located at Westport Point. Concern over bacterial contamination within the Estuary has led WRWA to conduct multiple studies on pathogenic indicator bacteria levels within the estuary and to investigate sources of contamination. In the 2010 effort CSP scientists quantified storm water inputs of contaminants, including indicator bacteria, to the lower estuary from the storm water collection system on Westport Point. Both WRWA and the Westport "Massachusetts Estuaries Project" (MEP) Committee sought quantification of indicator bacteria and nitrogen loads from this storm water collection system to the estuary to determine if the constituent load had any potential effect on the receiving waters of Westport Harbor. The CSP Technical Team partnering with the Westport River Watershed Alliance undertook sampling and flow measurements of this storm water outfall in the vicinity of Lee's Wharf. The effort was discussed with the Town of Westport and the Department of Public Works provided significant logistical support to the project. The goal of the project was to sample the flow and contaminant levels from the initial flow to the end of flow at the Westport Point storm water outfall during multiple storm events. Multiple checks of the outfall during "dry" weather indicated that this storm water system does not have significant I/I and therefore does not discharge except during rain events or due to melt water inflows. Sampling occurred under wet conditions (greater than 0.25 inches of rainfall) and was undertaken during seven events of which three storm events, on March 23, April 17 and May 18, 2010, resulted in the necessary flows.

Based on data collected during the three viable storm events, it was possible to estimate the annual discharge of nutrients and bacteria from the outfall pipe to Westport Harbor. Annual rainfall for this region was available from New Bedford rainfall records. From 1961-2000 the average annual rainfall was approximately 47.8 inches. From this rainfall amount annual discharge from the outfall pipe at Westport Point was estimated based on the relationship between rainfall and discharge presented in the report. The estimate was considered approximate and probably overestimated the actual discharge because of threshold amounts of rain events required to trigger flow through the pipe. However, it provided an approximation of annual discharge with which the annual flux of nutrients and bacteria to the harbor could be estimated. Based on the rainfall and discharge data collected, the annual discharge of stormwater from the Westport Point outfall pipe to the harbor was estimated to be 55,450 gal. Using this estimate the investigators calculated the annual discharge from the outfall to Westport Harbor of total nitrogen (TN), total phosphorus (TP), total suspended solids (TSS), E. coli, Enterococcus and Fecal Coliform bacteria (Table 7). Results were based on average concentrations of these analytes from all 3 sampling events and the estimated water discharge.

These estimates indicate that the stormwater outfall pipe at Westport Point represents a minor input of nitrogen to the estuary, but a significant local source of indicator bacteria loading. Fortunately, the heavy metal and oil and grease discharges are generally below detection. Remediation efforts will clearly help to reduce this loading and will contribute to improving water quality within the harbor basin adjacent to the outfall and possibly the greater harbor area beyond.

Westport River Water Quality Monitoring Program - Beginning in 1993, summer measurement of nutrient levels (dissolved and particulate nitrogen; phosphorus); and other water quality indicators, (chlorophyll; secchi depth, dissolved oxygen and temperature) was begun in the Westport River embayment (east and West Branch) by the BayWatchers program instituted by the Coalition for Buzzards Bay (CBB), the Buzzards Bay Project and scientists now at SMAST-UMassD (then at WHOI). The Coalition's BayWatcher Program, with significant staff support from the Westport River Watershed Alliance, has collected the baseline water quality data necessary to support ecological management of the Westport River Embayment System. The BayWatcher monitoring was undertaken was a collaborative effort with CBB (Tony Williams) coordinating the field effort and chemical assays being completed by the SMAST Coastal Systems Analytical Facility. The Coastal Systems Analytical Facility is located in the School for Marine Science and Technology UMASS-Dartmouth, 706 S. Rodney French Blvd, New Bedford, MA, and the laboratory Points of Contact are Sara Sampieri 508-910-6325 (ssampieri@umassd.edu) or Mike Bartlett (mbartlett@umassd.edu). Use of the SMAST Analytical Facility ensured sufficient sensitivity and accuracy of the analytical protocols and that proper QA/QC procedures were followed to allow incorporation of the data into the MEP analysis. Baseline water quality data are a prerequisite to entry into the MEP. Implementation of the MEP's Linked Watershed-Embayment Approach necessarily incorporates the quantitative water column nitrogen data (2003-2009) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff.

The focus of the Coalition for Buzzards Bay BayWatcher Water Quality Monitoring Program effort has been to gather site-specific data on the current nitrogen related water quality throughout all the embayments tributary to Buzzards Bay (Figure II-1). The program was tailored to the gathering of data specifically to support evaluations relating observed water quality to habitat health. The BayWatcher Water Quality Monitoring Program in the Westport River Embayment System developed a data set that elucidated the long-term water quality of this system. The BayWatcher Program provided the quantitative water column nitrogen data (1992-2006) for validating the MEP's Linked Watershed-Embayment Approach (see Section VI).

After the first ten years of monitoring, the BayWatchers data set was reviewed and a summary report for the embayments tributary to Buzzards Bay was developed based upon measurements from 1992-1998 (published Coalition for Buzzards Bay, 1999). The results indicated clear gradients in nitrogen related water quality parameters, with high levels in the upper tidal reaches and lower levels moving toward the tidal inlet. In addition, it appeared that the East Branch was showing the effects of nitrogen enrichment while the West Branch was less enriched. This latter observation is consistent with the majority of the nutrient load entering the Westport River Estuary entering via the Westport River into the head of the East Branch. More recent 5-year running averages of the health indexes for the East and West Branches of the Westport River Estuary through the summer of 2005 show no evidence of an improvement in water quality throughout the Estuarine System. In comparison to other Buzzards Bay embayments sampled by the BayWatchers program, the Inner portion of the East Branch of the Westport River "Health Index" ratings were consistently in the lowest third of all 29 Buzzards Bay embayments. However, the lower tidal reaches and particularly Westport Harbor continue to support stable and productive estuarine plant and animal habitats.

The MEP effort integrated and built upon these previous watershed delineation and land-use analyses, river transport, embayment water quality and eelgrass surveys in the present effort. This information is integrated with MEP collected higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Westport River Embayment System. The incorporation of appropriate data from previous studies to which the MEP Technical Team had access both enhance the determination of nitrogen thresholds for the Westport River System and reduced costs for the Town of Westport.

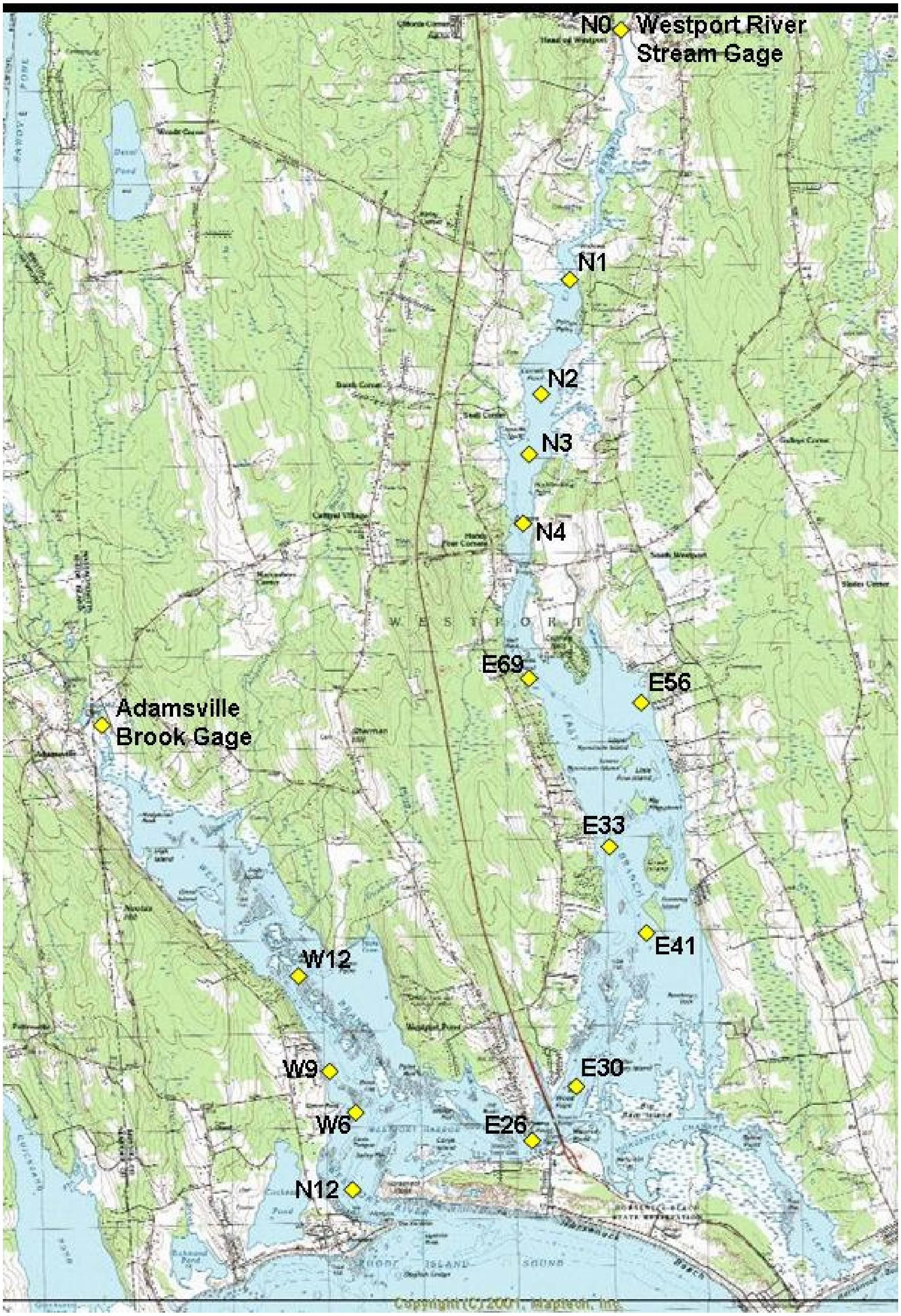


Figure II-1. BayWatcher Water Quality Monitoring Program of the Westport River Estuary, Town of Westport. Estuarine water quality monitoring stations sampled by the Coalition for Buzzards Bay and Westport River Watershed Alliance. Stream water quality stations depicted in Section IV were sampled weekly by the MEP staff.

Regulatory Assessments of Westport River Resources - The Westport River Estuary contains a variety of natural resources of value to the citizens of Westport 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 Westport River Estuary these include:

- Mouth of River designation - MassDEP (Figure II-2)
- Designated Shellfish Growing Area – MassDMF (Figure II-3a, 3b)
- 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-6)

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

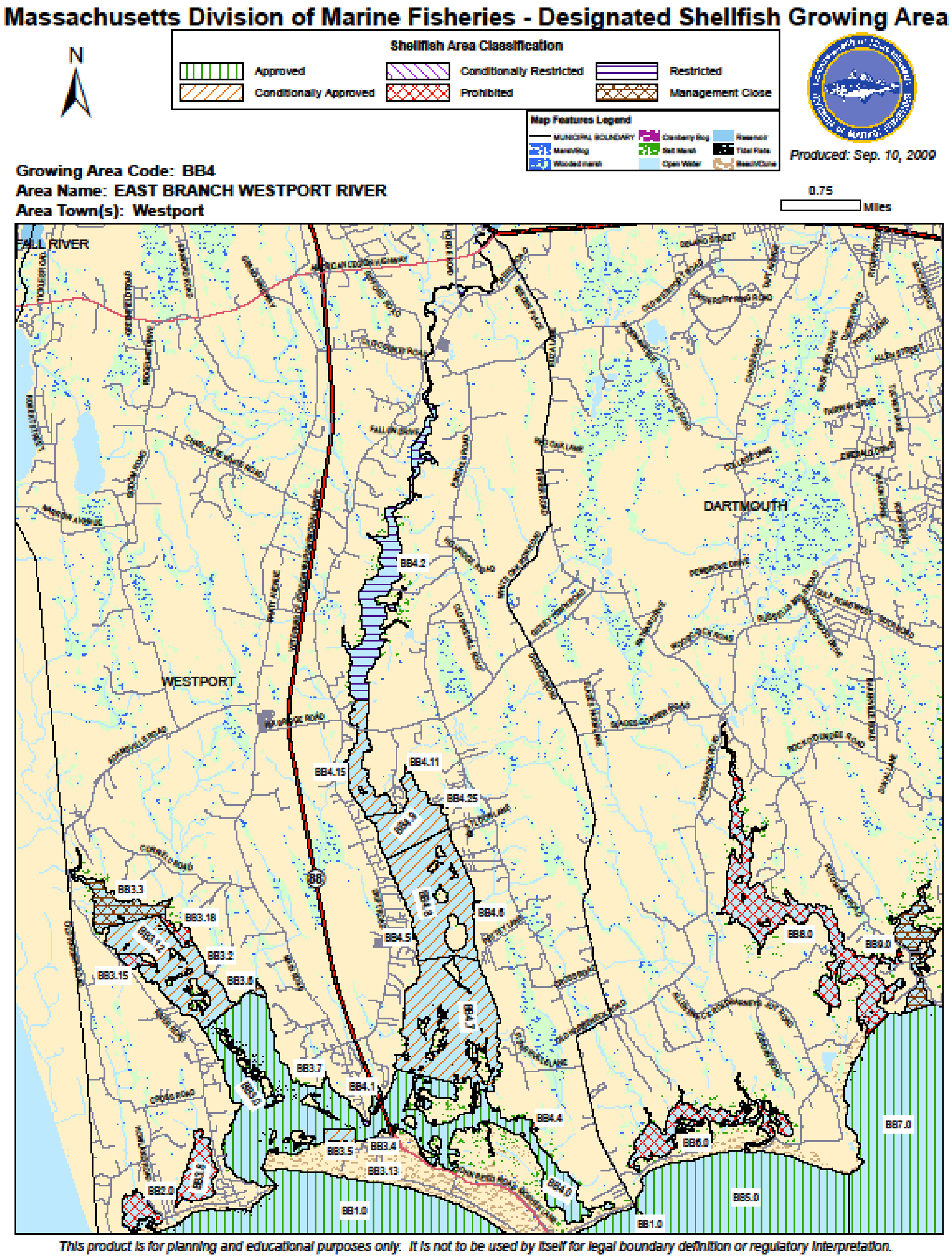


Figure II-3a. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or the presence of specific "activities", such as the location of marinas. Note system wide closures in adjacent estuaries, Allens Pond and Slocums and Little Rivers.

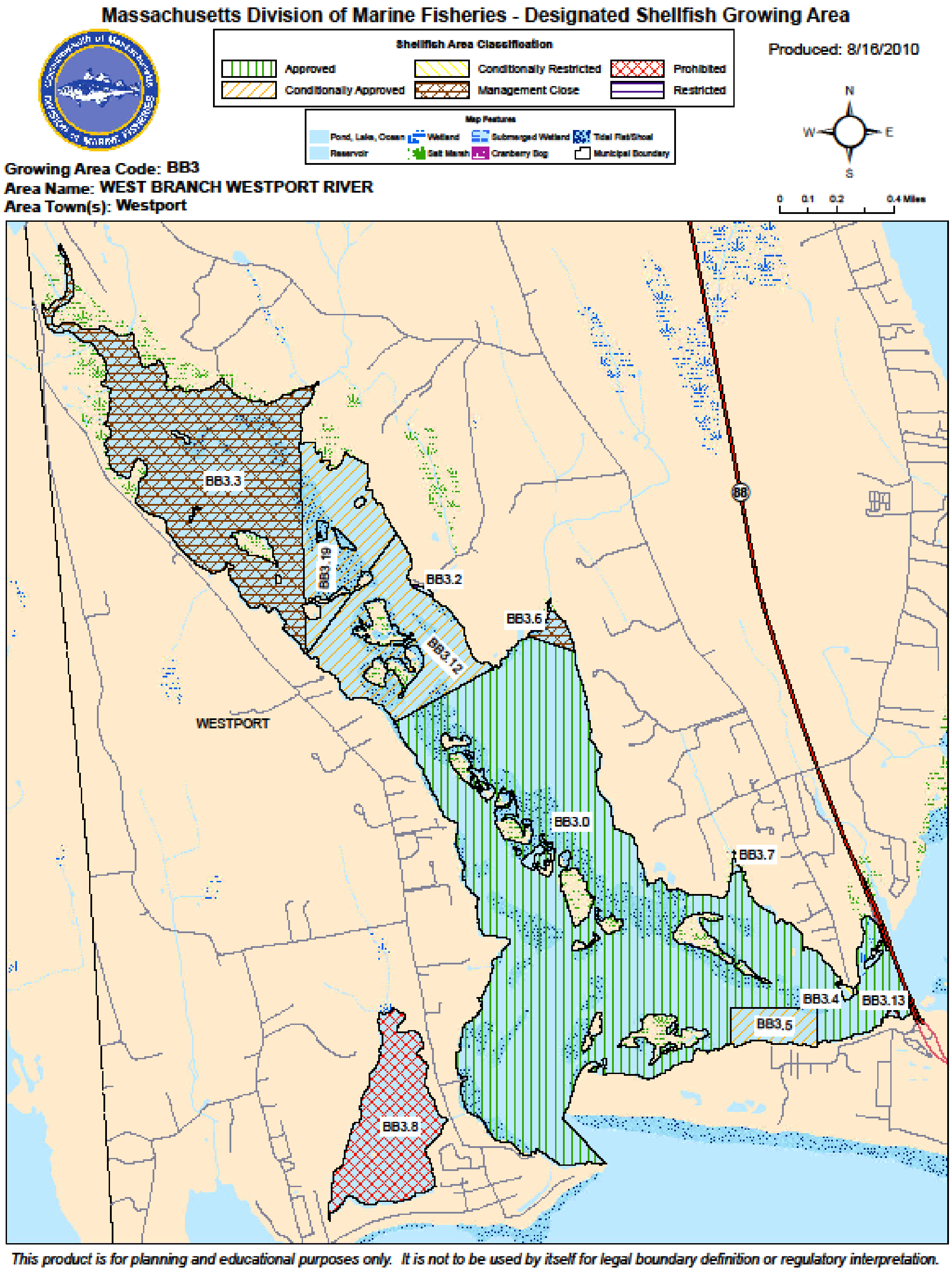


Figure II-3b. Location of shellfish growing areas in the West Branch of the Westport River Estuary and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or presence of specific "activities", such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

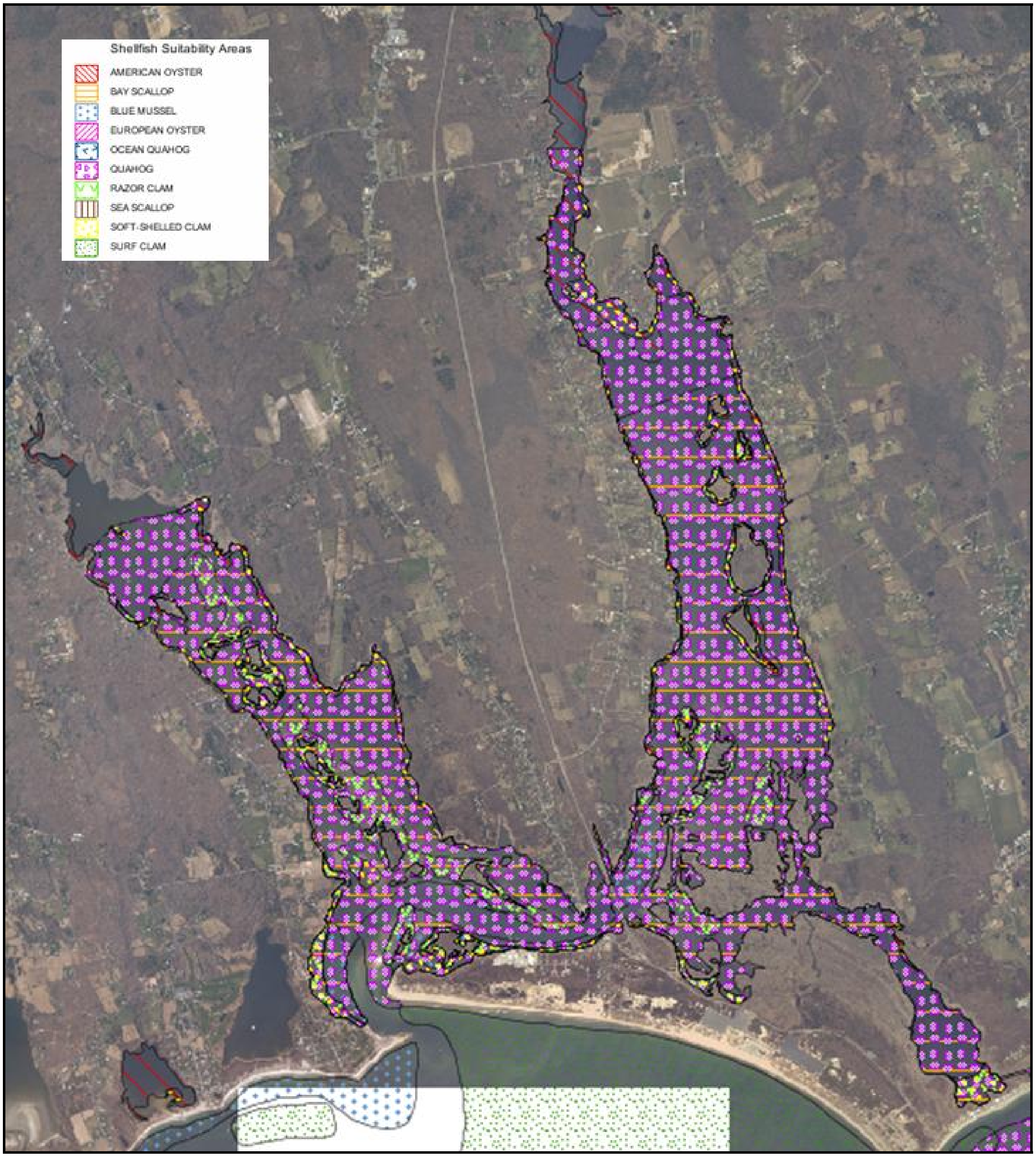


Figure II-4 Location of shellfish suitability areas within the Westport River Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean that a species of shellfish is present.

Figure II-5 Anadromous fish runs within the Westport River Estuary as determined by Mass Division of Marine Fisheries. The red diamonds show areas where fish were observed. The uppermost sites are within Grays Mill Pond located at the head of the West Branch, Cockeast Pond immediately west of the inlet and Forge Pond discharging into the East Branch represent potential spawning areas.

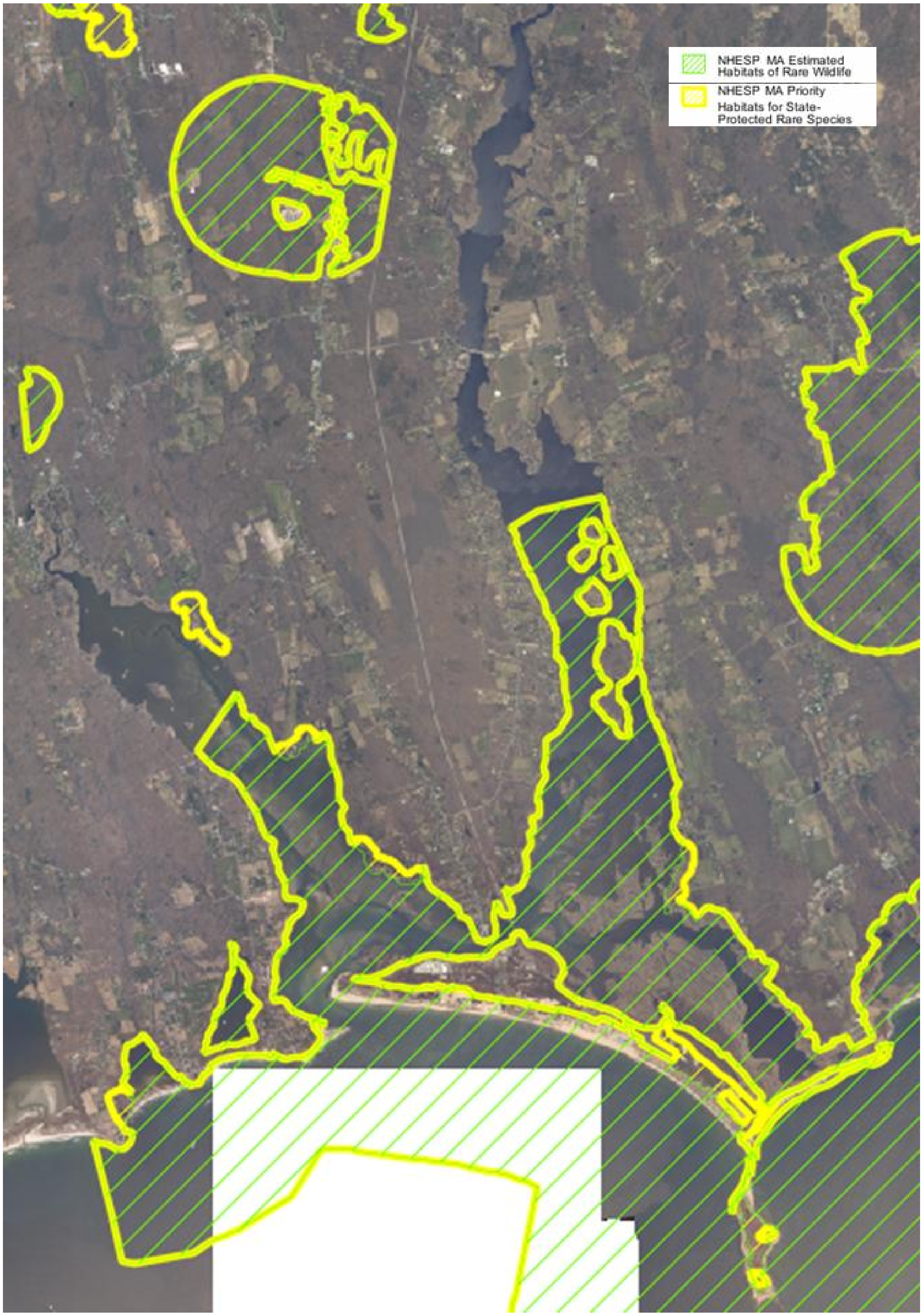


Figure II-6. Estimated Habitats for Rare Wildlife and State Protected Rare Species within the Westport River Estuary as determined by - NHESP.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Westport River study area is dominated by bedrock geology that is overlain with various glacial till deposits. Underlying the soils of the region is crystalline bedrock of about 600 million years old in an area known as the Narragansett Basin (Zen, *et al.*, 1983). Bedrock outcrops occur throughout the study area and the exposed and buried bedrock surface is the primary control upon the local topography. The watershed topography is defined by a framework of roughly north-south, low bedrock ridges with shallow intervening valleys. A series of repeated continental glaciations have advanced and retreated across the region over the past 600,000 years, with the most recent ending about 16,000 years ago. This most recent glaciation deposited outwash and till materials in varying thickness (2 to 60 m) over the bedrock (Willey *et al*., 1983; Williams and Tasker, 1978). It has been estimated that approximately half of the East Branch watershed area consists of till deposited over bedrock (Pivetz, *et al*., 1986). Tills tend to be unsorted, unstratified mixtures of clay, silt, sand, cobbles, and boulders and with varying composition from location to location. Tills tend to be more prevalent along the watershed ridges, while outwash deposits tend to be more prevalent in the valleys (Bent, 1995).

The combination of exposed and shallowly buried bedrock and generally low permeability glacial tills strongly influences watershed hydrology. The regions' bedrock and low permeability upland soils form a surface-water dominated regime where rainfall and snow melt tend to flow over the ground surface in a greater proportion than they percolate into the soils to become groundwater (Bent 1995). The stream network is therefore well-developed and most upland groundwater flow is local and discharges to the nearest stream. As a result, the freshwater entering the Westport River estuary from the watershed is primarily from streams and rivers and to a lesser extent through groundwater inputs. The result is that watershed delineation typically follows topography and topographic analysis was used in each of the watershed delineations that have been performed relative to the Westport River Estuary, including the present MEP analysis (Section III.2).

This local hydrologic regime is one that prevails over much of the glaciated Northeastern United States but is a marked contrast to the hydrology that prevails eastward from Mattapoisett to the tip of Cape Cod and on Martha's Vineyard and Nantucket, where more deeply buried bedrock is covered with relatively high permeability glacial sediments resulting in a groundwater-dominated freshwater regime. In those areas, watershed divides are determined primarily by the shape of the groundwater surface as determined by the recorded levels in groundwater wells and from pond and stream elevation records. These data are processed in computer models to produce the maps of the groundwater surface elevation and groundwater flow paths used to delineate the watersheds.

III.2 WATERSHED DELINEATION APPROACH

A watershed divide or boundary can be described as the line from which rainwater or snowmelt flows on the surface and through groundwater towards one stream, river or estuary, while rainfall and groundwater on the other side of the divide flow away to another water body. In addition, the water table, or the surface of the saturated sediments (aquifer), also tends to reflect the changes in surface elevation within bedrock and till dominated landscapes, but can be modified by layers of low hydraulic conductivity sediments within the aquifer. The technique of topographic inspection begins with developing an understanding of the watershed stratigraphy and hydrogeology to determine the validity of this method of watershed delineation. In the case of the Westport River Estuary the surficial till on high elevation areas and outwash in valleys and the dominance of bedrock in forming the watershed supports the use of this method. Analysis focuses on determining the pattern of lines of local maximum elevation upon a US Geological Survey 1:25,000 topographic map and draws watershed divides based upon the tendency of surface water and groundwater to flow downhill perpendicularly to the topographic contour lines. Divides drawn upon topographic maps can be confirmed by observing general patterns of groundwater flow and surface water flow during rainfall or snow melt or by measuring the flow of water in streams over a hydrologic cycle.

The initial watershed delineation for the Westport River Estuary was conducted in 1991 by the US Geological Survey as part of determining the watersheds for all the sub-embayments to Buzzard Bay for the Buzzards Bay Project, now the Buzzards Bay National Estuary Program (BBP, 1991). The boundaries were determined by the method of topographic inspection and focused on the outer boundary of each sub-embayment. MEP staff reviewed the delineation for the Westport River Estuary and generally found it to be sufficient for advancing a land-use analysis of this system. In order to complete the MEP assessment, however, subwatershed delineations were developed to address the major freshwater inputs from streams to the estuary and to provide nitrogen loadings at spatial scales matching the sub-embayment segmentation of the MEP tidal hydrodynamic model (e.g. East Branch, West Branch, The Let, Westport Harbor).

Fourteen (14) subwatersheds were delineated for the MEP analysis of the Westport River Estuarine System (Figure III-1) also using the method of topographic inspection. The subwatersheds include contributing areas to the following streams: Westport River above Route 177, the Westport River above the Old County Road bridge, Kirby Brook, Snell Creek, Angeline Brook, and Adamsville Brook. Delineation of these subwatersheds allows direct comparison between the expected discharge flows and nitrogen loads from the delineated areas and measured data from MEP stream gauges. This effort also supported quantification of nitrogen attenuation prior to discharge to estuarine waters. Attenuation is a critical element in the development of the inputs to the estuary water quality model (see section IV.2).

Based upon the delineated sub-watersheds and annual recharge, stream flows were determined for comparison to measured streamflows collected by the MEP (Table III-1). Annual recharge was based on a review of available precipitation data for the region. The National Oceanic and Atmospheric Administration (NOAA) maintains a long-term precipitation gauge at New Bedford, which is close to the Westport River watershed. Annual average precipitation at this site between 1961 and 2000 is 47.8 inches (CDM, 2006), while the average between 1971 and 2000 is 50.77 inches (NOAA, 2004). Review of unofficial NOAA data from this site between 2005 and 2010, the period associated with the MEP analysis, shows an average annual precipitation of 52.98 inches. Precipitation in the complete hydrologic year (2006) of MEP stream flow measurements had an annual precipitation rate of 63.82 inches, but the preceding and subsequent years were both less than the long term averages (48.2 and 44.47 inches, respectively). Given uncertain issues regarding watershed release/saturation indices and the unofficial nature of the NOAA data, MEP staff assessed that the near long term average at New Bedford (50.77 in/yr) was most appropriate annual precipitation rate for further analysis.

A portion of precipitation is utilized by plants on the land surface (transpiration) and a portion is evaporated back into the atmosphere. USGS recharge rates used in groundwater modeling on Cape Cod are approximately 60% of long term precipitation rates (*e.g.*, Walter and Whealan, 2005). USGS modeling of recharge in the Charles River basin, which is more similar to the geology of the Westport River has found recharge variations of 43 to 56% of precipitation with a strong reliance on measured streamflows for the development of the model (DeSimone, *et al.*, 2002). Given the uncertainty in many of the factors for developing the percentage of recharge, MEP staff conservatively assumed 60% of precipitation or 30.46 inches per year is an appropriate recharge rate in the Westport River watershed. This recharge rate is used to develop the long-term freshwater inflows in Table III-1 and is also use in the watershed nitrogen loading estimates (see Chapter 4). It should be noted that this recharge analysis is used for comparison of measured and modeled annual stream flow and for providing an independent check on stream watershed areas, but does not directly influence the nitrogen loading analysis for this system (Section IV).

Staff also contacted the City of Fall River to determine the average annual water withdrawal from the Copicut Reservoir. Ted Kaegael, Fall River Department of Public Works (personal communication, 10/10) provided monthly water withdrawals from the reservoir between mid-2005 through 2008. These withdrawals averaged 2.5 million m3/yr. Since these withdrawals are distributed as drinking water outside of the watershed, they are removed from the long-term flow estimate for the Westport River.

Using the Westport River watershed recharge rate and accounting for the Copicut Reservoir withdrawals, the overall estimated long-term freshwater inflow into the Westport River estuary from its MEP watersheds is 423,620 m3/d. The watershed to the Westport River extends across six (6) towns: Westport, Dartmouth, Fall River, and Freetown in Massachusetts and Tiverton and Little Compton in Rhode Island. Based on the details presented above, the MEP Technical Team concluded that the watershed delineations, as presented in Figure III-1 for the Westport River are suitable for the MEP nutrient threshold analysis.

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

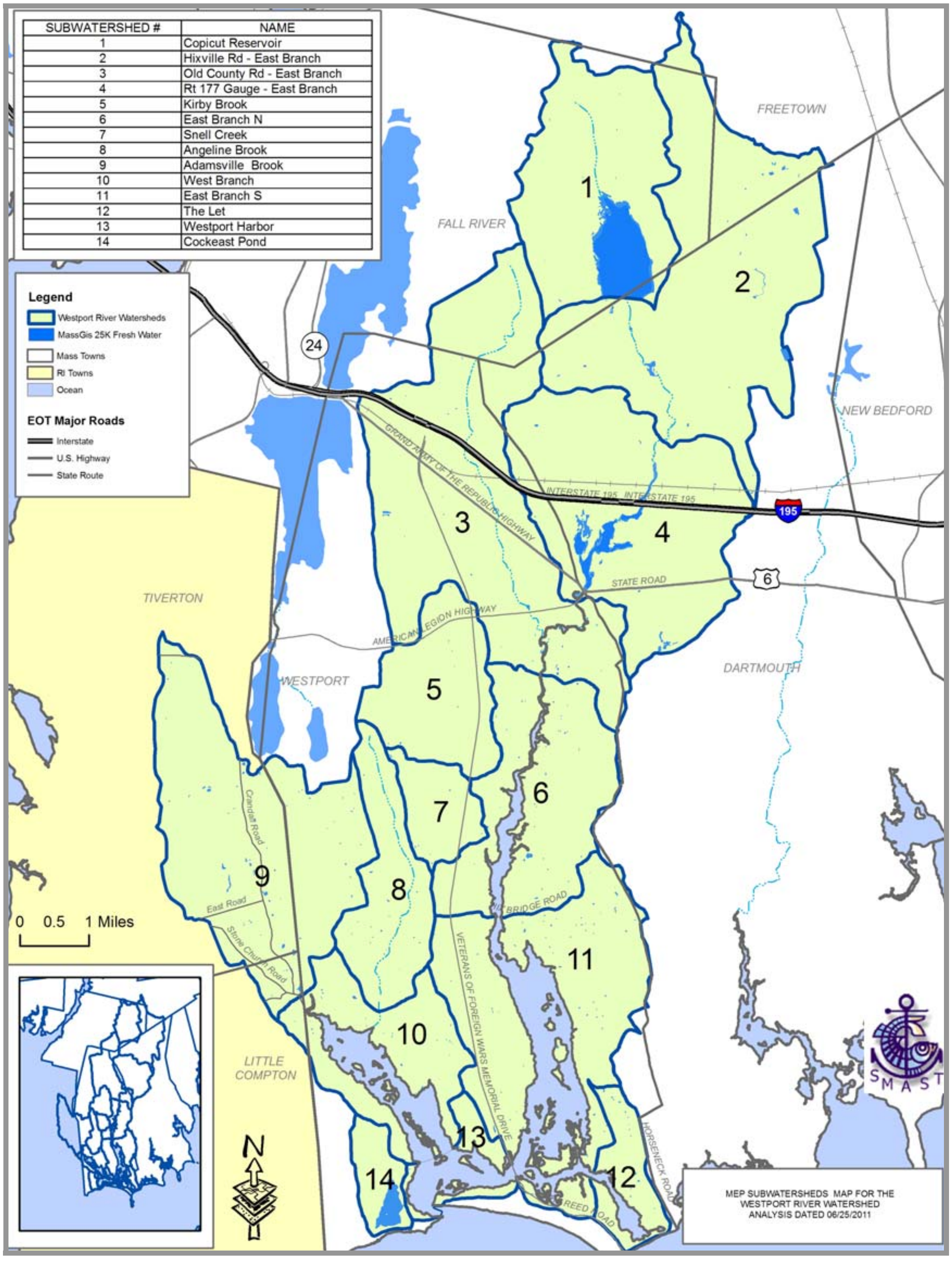


Figure III-1. Westport River Estuary watershed (outer boundary) and subwatershed delineations (numbered areas). The watershed boundary is based on 1991 USGS topographic delineation for the Buzzards Bay Project (BBP, 1991). Interior subwatershed delineations followed the USGS approach to determine contributing areas to MEP stream gauging locations and the basin segmentation used in the MEP estuary hydrodynamic model. The watershed includes portions of six towns: Westport, Dartmouth, Fall River, and Freetown in Massachusetts and Tiverton and Little Compton in Rhode Island.

Table III-1. Westport River MEP Subwatershed Areas and Estimated Long-Term Freshwater Recharge.

Watershed Name	#	Watershed Area (acres)	Discharge	
			m³/day	ft³/day
Copicut Reservoir	1	4,404	37,780	1,334,188
Hixville Road – East Branch	2	7,861	67,434	2,381,411
Old County Road - East Branch	3	7,945	68,153	2,406,804
Route 177 Gauge – East Branch	4	5,907	50,674	1,789,535
Kirby Brook	5	2,281	19,571	691,154
East Branch N	6	3,687	31,625	1,116,832
Snell Creek	7	956	8,199	289,529
Angeline Brook	8	2,121	18,194	642,528
Adamsville Brook	9	5,993	51,413	1,815,629
West Branch	10	2,272	19,494	688,416
East Branch S	11	4,877	41,837	1,477,459
The Let	12	874	7,499	264,820
Westport Harbor	13	479	4,109	145,125
Cockeast Pond	14	551	4,726	166,887
WESTPORT RIVER SYSTEM TOTAL			423,620	14,959,988

Notes:
1) discharge volumes are based on 30.46 inches of annual recharge over the watershed;
2) recharge is based on 60% of annual precipitation of 50.77 inches (1971-2000 average at nearest long-term NOAA gauge: New Bedford);
3) these flows do not include precipitation on the surface of the estuary;
4) total flows, but not subwatershed flows, include removal of drinking water from the Copicut Reservoir by the City of Fall River (250,328 cubic feet per day, personal communication, Ted Kaegael, Fall River Department of Public Works),
5) totals may not match due to rounding.

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 Westport River estuary system as well. Determination of watershed nitrogen inputs to this embayment system requires the (a) identification and quantification of the nutrient sources and their loading rates to the land or watershed, (b) confirmation that a watershed transported loads have 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 is generally small relative to the amount recycled. 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 Westport River estuary system, the MEP Technical Team developed nitrogen loading rates (Section IV.1) to each component of the estuary and its watersheds (Section III). The Westport River sub-watersheds were delineated to define contributing areas to each of the major streams and other significant freshwater systems and to each major portion of the estuary. A total of 14 sub-watershed areas were delineated within the Westport River study area, including watersheds to the following streams: Westport River at Route 177, Westport River at Old County Road, Kirby Brook, Snell Creek, Angeline Brook, and Adamsville Brook (see Section III). Freshwater inflow to the Westport River Estuary is dominated by stream discharges, which account for more than 70% of the freshwater inputs to the estuary.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the estuary. This generally involves a temporal review of land use changes, review of data at natural collection points, such as streams and ponds, and, in groundwater dominated systems, the time of groundwater travel provided by a USGS watershed model. The Westport River watershed system is a stream-dominated system because of its underlying geology, so this portion of the analysis focused on land use and associated nitrogen loading and data from stream gauges rather than time of travel. Review of the stream locations within the upper sub-watersheds generally show that most of the sub-watershed areas are within 3,500 ft of the main streams. If it is assumed that groundwater discharge is the only mechanism for transfer of nitrogen and water to the main stem of each stream and that groundwater travels at approximately 1 ft/d (a common assumption in outwash materials), the areas furthest from the primary stream channels (close to the sub-watershed boundaries) would take approximately 10 years to reach the main stem. Of course, these areas have significant wetland areas that extend to within hundreds of feet of the sub-watershed boundaries, likely surrounded with small tributaries with the wetlands feeding the main stem of the river. These wetlands and small tributaries are short travel-time conduits, which significantly shorten travel time, such that travel time from the boundary areas to the estuary is much less than 10 years. The data to support the10 year travel time estimates would need to be refined to project to a shorter travel time such as 3 year. For example, the travel estimates are complicated by the many small streams that penetrate the sub-watersheds, particularly those to the main stem of the Westport River. However, it might be useful to refine the travel times relative to specific nitrogen management alternatives to forecast the response in the estuary. Review of the lower, more groundwater dominated sub-watershed areas also show that the sub-watershed boundaries are also generally less than 3,500 ft from the estuary shoreline. MEP reviews in groundwater-dominated systems have shown that if most development is less than 10 years travel time to the estuary, then the watershed nitrogen loads are in relative balance with the estuary nitrogen concentrations. Therefore, there should be a high level of confidence that the present nitrogen load within the watershed to the Westport River Estuary accurately reflects the present nitrogen input to its estuarine waters (after accounting for natural attenuation, see below).

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used to develop nitrogen loads from most areas, while information developed from other detailed site-specific studies is applied to the remaining portions of the watershed. The Linked Watershed-Embayment Management Model (Howes and Ramsey, 2001) uses a land-use Nitrogen Loading Sub-Model based upon sub-watershed specific land uses and pre-determined nitrogen loading rates based on detailed source studies in southeastern Massachusetts. For the Westport River Embayment System, the model uses land-use data from the Towns of Westport, Dartmouth, and Freetown and the City of Fall River in Massachusetts and Tiverton and Little Compton in Rhode Island. This land-use data is transformed to nitrogen loads using both regional nitrogen loading factors and local watershed-specific data (such as parcel-by-parcel water use and agricultural animal counts in close coordination with the towns in the watershed). 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 within the Westport River watershed was determined based upon site-specific measurements of stream flow on each of the five major streams discharging to the estuary. Stream flow was characterized at the discharge of the Westport River at Route 177 and Old County Road, Kirby Brook, Snell Creek, Angeline Brook, and Adamsville Brook. A sub-watershed to these stream discharge points allowed comparison between field collected data from the streams and estimates from the nitrogen-loading sub-model. Stream flow and associated surface water attenuation is included in the MEP nitrogen attenuation and freshwater flow investigation, presented in Section IV.2. 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 system, the MEP Technical Team used the watershed Nitrogen Loading Model to estimate nitrogen loads for the sub-watersheds that directly discharge groundwater to the estuary without flowing through one of these interim stream measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Westport 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 the Westport River includes portions of the Towns of Westport, Dartmouth, Freetown and the City of Fall River in Massachusetts and the towns of Tiverton and Little Compton in Rhode Island, Estuaries Project staff obtained the most up-to-date, digital parcel and tax assessor's data from these municipalities to serve as a base for the watershed nitrogen loading model. Digital parcels and land use/assessors data for Westport are from 2008, Dartmouth and Fall River are from 2009, Freetown data are from 2005, and Tiverton are from 2010. Little Compton parcels were created within a GIS environment by MEP staff from digital pictures and 2010 assessors information that was provided by the town and linked to the created parcels. These land use databases contain traditional information regarding land use classification, although Massachusetts and Rhode Island use different land use numbering systems. For the purposes of this MEP assessment, Little Compton and Tiverton land use codes were translated to similarly described MassDOR (2009) land use codes to provide consistent coding within the database.

Figure IV-1 shows the land uses within the Westport River estuary watershed. Land uses in the study area are grouped into nine (9) land use categories: 1) residential, 2) commercial, 3) industrial, 4) agricultural, 5) recreational (golf courses and parks), 6) undeveloped, 7) forest land (Chapter 61 properties), 8) unclassified, and 9) public service/government, including road rights-of-way. These land use categories are generally aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MassDOR, 2009). Unclassified properties are those that do not have an assigned land use code in the town assessors databases. "Public service" properties in the MassDOR coding system are tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, open space, roads) and private non-profit groups like churches and colleges.

Residential land use categories are the dominant land use in the overall Westport River watershed. Residential land uses are 32% of the watershed area followed by public service/rights-of-way at 25%, undeveloped lands at 18% and agricultural lands at 14% (Figure IV-2). Residential land uses are the dominant land use in each of the contributing sub-watershed groupings shown except for the watersheds above Route 177, where public service is 38% of the watershed area. The large proportion of public service in this sub-watershed is primarily due to the Freetown/Fall River State Forest that occupies nearly 80% of the Copicut Reservoir (#1) sub-watershed. Agricultural or undeveloped land uses are the second-most predominant land use in the other sub-watershed groupings. Residential land uses vary from 3% to 45% of the area of the individual sub-watersheds.

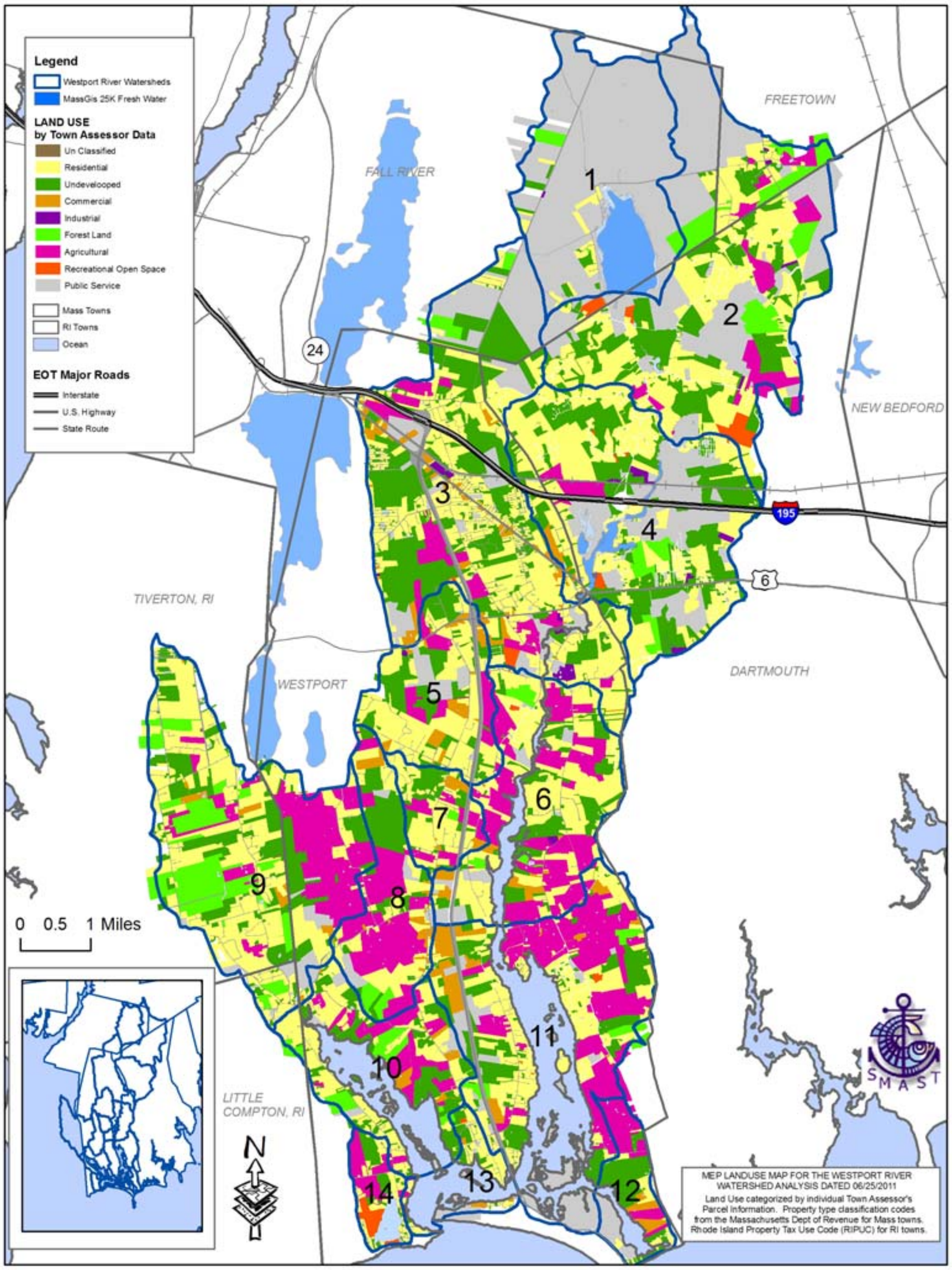


Figure IV-1. Land-use in the Westport River system watershed and sub-watersheds. The system watershed extends over portions of the towns of Westport, Dartmouth, and Freetown and the City of Fall River in Massachusetts and the towns of Tiverton and Little Compton in Rhode Island. Land use classifications are based on respective town assessor classifications and MADOR (2009) categories. Rhode Island land use classifications were converted to comparable Massachusetts classifications for this assessment. Digital parcels and land use/assessors data for Westport are from 2008, Dartmouth and Fall River are from 2009, Freetown are from 2005, and Tiverton are from 2010. Little Compton parcels were created within a GIS environment by MEP staff from digital pictures and 2010 assessors information was provided by the town and linked to the created parcels.

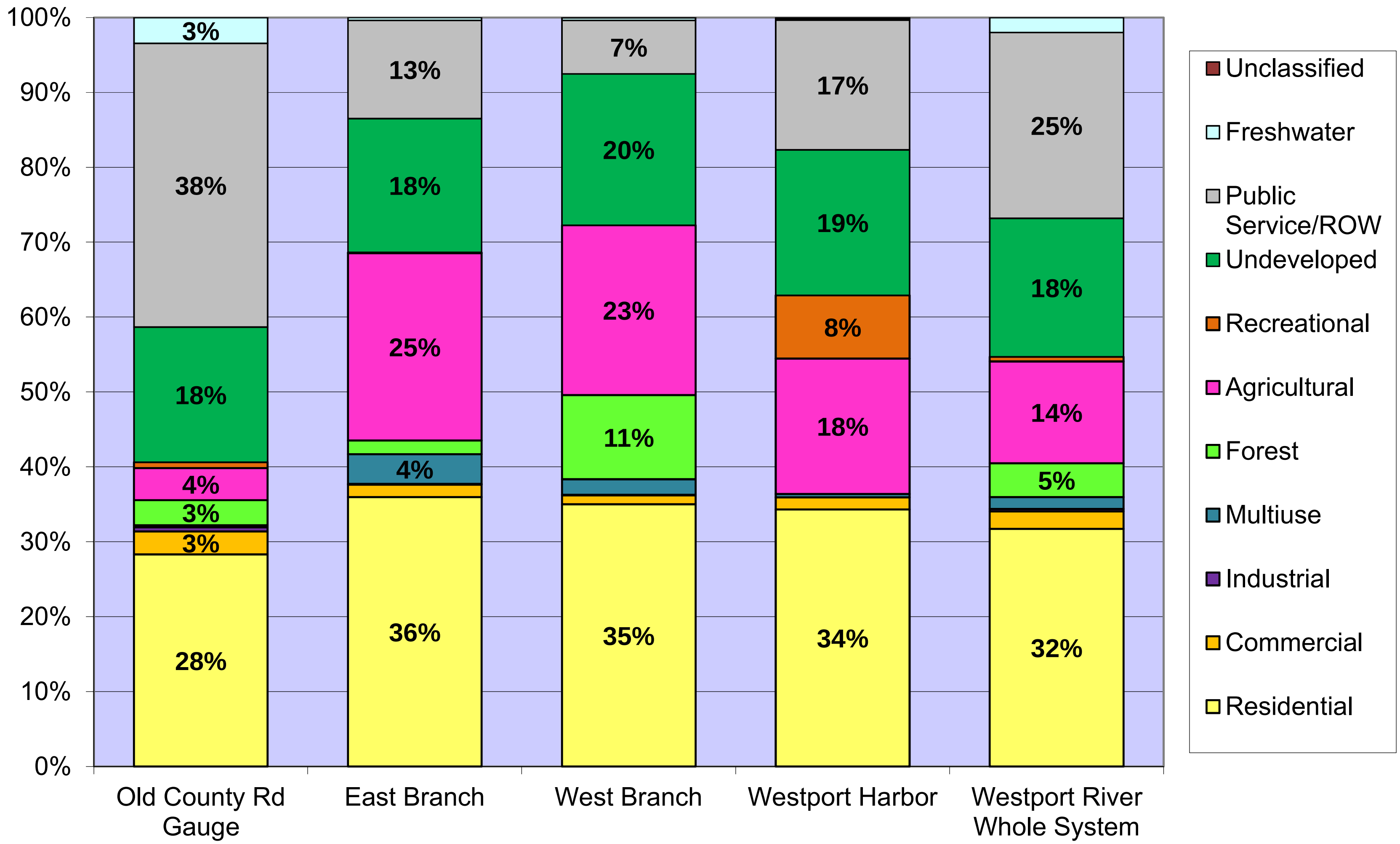


Figure IV-2. Distribution of land-uses by area within the Westport River system watershed and four component sub-watersheds. Land use categories are generally based on town assessors' land use classifications and groupings recommended by MADOR (2009); Rhode Island land uses were converted to comparable Massachusetts categories based on the Rhode Island descriptions. Only percentages greater than or equal to 3% are shown.

In all the sub-watershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type, ranging between 70% and 74% of all parcels in these sub-watersheds and 71% of all parcels in the Westport River system watershed. Undeveloped parcels are the second most frequent in these sub-watersheds. Review of average undeveloped lot size shows that there are larger undeveloped parcels in the West Branch watershed (average = 7.1 acres) than in the East Branch watershed (average = 4.4 acres). This finding is likely indicative of comparatively more isolated, already subdivided, undeveloped parcels in the East Branch watershed. Single-family residences (MassDOR land use code 101) are 77% to 98% of residential parcels in the individual sub-watersheds and 91% of the residential parcels throughout the Westport River system watershed.

Typically, in MEP analyses, project staff obtain parcel-by-parcel water use information to be used as a proxy for wastewater generation. In this watershed, this type of information was generally not available, but it is incorporated into the MEP watershed nitrogen loading model where it is available. The Town of Westport has a limited public water supply area, primarily located along Route 6; water is supplied to these properties from the City of Fall River. Three years of water use (2008-2010) was available for these properties and this information was incorporated into the MEP watershed nitrogen loading model. The Towns of Freetown, Little Compton and the City of Fall River do not have public water within the portions of their towns inside the Westport River watershed; the Town of Tiverton did not have public water information available (personal communication, David Robert, Town Assessor, 7/10). The Town of Dartmouth has public water available to portions of properties within sub-watersheds #2 and #4, but water use has not been linked to individual parcel databases (personal communication, Mike O'Reilly, Town of Dartmouth, 11/10). Previous discussions with Steve Sullivan, Dartmouth Water Department (personal communication, 6/07) indicated that a review of Dartmouth's 8,184 active accounts had an average water use of 188 gpd of water. This water use rate was used as the basis for assigning water use to all developed properties throughout the Westport River watershed.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

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

However, given the seasonal shifts in occupancy and past rapid population growth throughout southeastern Massachusetts, decennial census data yields accurate estimates of total population only in selected watersheds. To correct for this uncertainty and more accurately assess current nitrogen loads, the MEP employs a water-use approach. The water-use approach is applied on a parcel-by-parcel basis within a watershed, where annual water meter data is linked to assessor's parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (*e.g.*, irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down gradient in the aquifer.

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

In its application of the water-use approach to septic system nitrogen loads, the MEP 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, the MEP 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 use based on average residential occupancy to measured average water use. 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; (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 in 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 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 Westport River watersheds, MEP staff reviewed US Census population and housing information for the Towns of Westport, Dartmouth, Tiverton, and Little Compton. Fall River and Freetown were not included in this analysis because the Fall River portion of the watershed is largely undeveloped and Freetown occupies a relatively small portion of the watershed. The Massachusetts on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so for the purposes of Title 5 each person generates 55 gpd of wastewater. Based on data collected during the 2000 US Census, average occupancy within Westport, Dartmouth, Tiverton, and Little Compton were 2.63, 2.91, 2.51, and 2.44 people per housing unit with 88%, 94%, 94%, and 70% year-round occupancy of available housing units, respectively. Town specific 2010 Census results generally show a slight drop in both average occupancy and year-round occupancy, except for a slight increase in average occupancy in Dartmouth to 3.03 people per housing unit.

Comparison of estimated wastewater flows based on Census information are generally a bit lower than wastewater flow based on the Town of Dartmouth average water use (188 gpd), but incorporation of even small seasonal occupancy adjustments shows that the Town of Dartmouth average is reasonable for the Westport River study area. If the average 188 gpd water use is multiplied by 0.9 to account for consumptive use, the study area estimated wastewater flow average is 169 gpd. Since Dartmouth housing units are mostly occupied year-round (with only 10% seasonality), the average estimated wastewater flow for Dartmouth based on occupancy rates matches the reported wastewater flow estimate based on water use (160 gpd for the 2000 Census and 167 gpd for the 2010 Census). The other towns tend to have lower occupancy rates and resulting wastewater flow estimates, but addition of a doubling of occupancy for the mostly small percentage of seasonal dwellings in these towns brings their wastewater estimates based on occupancy/population within a reasonable range of the 169 gpd assumed for residences in the study area. With this reasonable (and somewhat conservative) seasonal adjustment, this analysis suggests that the 188 gpd average water use is reasonably reflective of average wastewater estimates and is appropriate for use in the MEP Westport River watershed nitrogen loading model.

At the outset of the MEP, project staff decided to utilize the water use approach for determining residential wastewater generation by septic systems because of the inherent difficulty in accurately gauging actual occupancy in areas impacted by seasonal population fluctuations, such as the Cape Cod portion of the MEP study area. Estimates of summer populations on Cape Cod derived from a number of approaches (*e.g.*, traffic counts, garbage generation, sewer use) suggest average population increases from two to three times year-round residential populations measured by the US Census. Previous evaluations of watersheds with available water use and sewer service areas have shown that water use provides a reasonable estimate of wastewater generation.

The average water use for the Westport River watershed is assigned to all developed parcels, but is modified for multi-family residential parcels in the MEP watershed nitrogen loading model. Parcels with land use codes for multiple residential units are assigned twice the average water use in the watershed model; these parcels include: condominiums (land use codes 102), two family residences (land use code 104), three family residences (land use code 105), multiple houses on one parcel (land use code 109), four to eight units (land use code 111), and more than eight units (land use code 112). These land uses make up less than 9% of the residential parcels throughout the watershed. All other developed properties in the area are also assigned the average Dartmouth water use; previous reviews of commercial and industrial development have shown that residential water use rates are conservative for most of these land uses.

Site-specific Wastewater Estimates

During MEP assessments, MEP staff integrate information on large wastewater treatment facilities and connections to municipal sewers for site-specific modification of nitrogen loads. A portion of the Westport River watershed is connected to the Town of Dartmouth sewer collection system. These parcels were identified in a GIS coverage provided by the Town (personal communication, Mike O'Reilly, 5/10). This coverage also identified parcels likely to be connected to the collection system when they are developed and these were accounted for in the buildout scenario. Wastewater flows from the parcels with sewer connections were removed from the Westport River watershed since the municipal treatment plant is located outside of the watershed. MEP staff also reviewed the MassDEP Groundwater Discharge Permit database and found that no GWDPs exist within the Westport River system watershed. GWDPs are required under MassDEP regulations for wastewater treatment systems with design flows greater than 10,000 gallons per day.

Nitrogen Loading Input Factors: Fertilized Turf Areas

The second largest source of estuary watershed nitrogen loading is usually fertilized turf areas: lawns and golf courses. Residential lawns are usually the predominant source within this category. In order to add this source to the watershed nitrogen loading model for the

Westport River system, MEP staff reviewed available regional information about residential lawn fertilizing practices. There is only one golf course within the Westport River watershed, Acoaxet Golf Course, so residential lawns are the predominant source of fertilizer turf nitrogen.

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 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 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 should also be noted that a recent data review of lawn fertilizer leaching in settings similar to those on Cape Cod confirmed that the 20% leaching rate is appropriate (HWG, 2009). It is likely that these load rates still represents a conservative estimate of nitrogen load from residential lawns. It should also be noted that professionally maintained lawns in the three town survey were found to have the higher rate of fertilizer application and hence higher estimated annual contribution to groundwater of 3 lb/lawn/yr.

Average nitrogen fertilizer application rates for the Acoaxet Golf Course were obtained by the Town of Westport Estuaries Project Committee (personal communication, Jim Whitin, 6/11). Fertilizers are applied on 17 acres of tees, greens, and fairways at 2 lbs/acre. Areas of the tees, greens, and fairways were digitized by Tim Gillespie of the Town of Westport Estuaries Project Committee (personal communication, 2/10).

Nitrogen Loading Input Factors: Agricultural Areas

As noted in the land use review above, agricultural areas are a significant percentage of the land area within the Westport River watershed. MEP staff obtained a review of active agricultural sites from the Town of Westport Estuaries Project Committee (personal communication, Tim and Gay Gillespie, 3/10). This review included various types of crops and estimated animal counts, but it was also acknowledged that it was not exhaustive. The Town of Dartmouth also provided an animal count for farms in their portion of the watershed that was previously obtained during the Slocums River MEP assessment (personal communication, Mike O'Reilly, 6/07).

During the parcel-by-parcel assignment of the nitrogen loads associated with the agricultural review from the Town of Westport into the MEP watershed nitrogen loading model, project staff noted that some agricultural areas and animal counts did not appear to be included in the inventory. In addition, it appeared that agricultural development in the other towns in the watershed existed, but was not recorded. In order to provide a better accounting of agricultural loads within the entire watershed, MEP Project Staff supplemented the Westport/Dartmouth agricultural/animal inventory with additional crop areas based on a review of aerial photographs and the assessors land use codes. Project staff also consulted with USDA staff regarding estimates of farm animals with the conclusion that additional animals should also be added to the agricultural inventory. Animal counts were supplemented based on a review of aerial photographs and USDA estimates. These additions assign 65% crop area to all parcels with agricultural land uses; this percentage is based on review of average crop area coverage of agricultural parcels in the Westport River watershed. The crop areas are assigned nitrogen application rates based on rates determined for various Massachusetts land use codes (previously used in the Slocum River MEP assessment (Howes, *et al*., 2007). The crop loading rates are derived from 1995 USDA guidance (summarized in Howes and Goehringer, 2000 and also used by the Buzzards Bay Project N loading guidance). When these loading rates were compared to additional N loading and crop yield optimization data that has developed since the guidance was published, it appears that optimal crop production was derived at higher application rates. Discussions with farmers, however, suggested that they tended to use application rates more within the range of the cited reference. The MEP Technical Team decided to continue to rely on the referenced ranges and select the higher end of the range for our calculations to adjust for the discussions with farmers. Agricultural nitrogen loads used the available inventory information and supplemented it with the additional information based on aerial photographs and town assessors' classifications. This effort was completed for all agricultural lots throughout the Westport River watershed. Care was taken to avoid double counting of areas identified in the town-specific inventories.

Cranberry bogs areas in the Westport River watershed were also identified. Bog areas in the MEP watershed nitrogen loading model are based on a GIS coverage maintained by MassDEP for Water Management Act purposes; this coverage identifies the surface areas of the bogs. Cranberry bogs only exist in the Hixville Road – East Branch sub-watershed (#2). Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on enhanced reviews of nitrogen export from cranberry bogs in southeastern Massachusetts (Howes and DeMoranville, 2009; Howes and Teal, 1995).

Nitrogen Loading Input Factors: Solid Waste Sites

MEP staff reviewed the MassDEP solid waste database and identified six landfill sites within the Westport River watershed and contacted MassDEP staff to obtain any available nitrogen monitoring data for these sites. Three of the sites have available monitoring information in the MassDEP files (personal communication, Mark Dakers, 10/10). Using the available information, MEP staff developed nitrogen loads for the following solid waste sites in the Westport River watershed: 1) Jarabeck Farm Landfill, 2) Crapo Hill Landfill, and 3) Westport Landfill.

<u>Jarabeck Farm Landfill</u>

The Jarabeck Farm Landfill is located within the Old County Road - East Branch sub-watershed (sub-watershed #3). According to MassDEP records, the site was a pig farm that began to accept incinerator ash and disposed of it on the farm. Four (4) monitoring wells have been installed at the site and have been sampled seven (7) times between July 1995 and December 2006. Available laboratory data show that groundwater samples were analyzed for nitrate-nitrogen (NO3-N) and other constituents, but were not analyzed for any other nitrogen components.

Based on a previous review of monitoring data from the groundwater plume associated with the Town of Brewster landfill (Cambareri and Eichner, 1993), MEP staff determined a relationship between ammonium-nitrogen and alkalinity concentrations (NH4-N = 0.0352*ALK - 0.3565; r2 = 0.82). This relationship was used to estimate ammonium-nitrogen concentrations from the available landfill monitoring data; alkalinity concentrations in the two down gradient wells on this site average 307 and 257 milligrams of $CaCO_3$ per liter, while NO3-N concentrations average 0.91 and 0.37 mg l^{-1}. 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).

Based on a review of the available water quality data and development of estimated ammonium-nitrogen concentrations, MEP staff determined average ammonium-N plus nitrate-N concentrations in two wells down gradient from the landfill at 10.2 mg l^{-1}. Using an estimate of 12 acres of solid waste based on the review of MassDEP files and the Westport River recharge rate, MEP staff developed an estimated annual total nitrogen load of 387 kg from the Jarabeck Farm Landfill.

It is acknowledged that this approach for estimating a nitrogen load from the Jarabeck Farm Landfill includes a number of assumptions, but these are appropriate based on the available data. 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 by including conservatism in nitrogen loading estimates when uncertainty existed 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.

Westport Landfill

The Westport (Hix-Bridge Road) Landfill is a municipal landfill located within the East Branch N sub-watershed (sub-watershed #6). Available MassDEP monitoring data were supplemented with post-closure monitoring information supplied by the Town (Tighe and Bond, 2009). Seven (7) monitoring wells are monitored on a regular schedule and have been sampled 21 times between November 1998 and December 2009. Available laboratory data show that groundwater samples were analyzed for nitrate-nitrogen (NO3-N) and other constituents, but were not analyzed for any other nitrogen components.

Based on a previous review of monitoring data from the groundwater plume associated with the Town of Brewster landfill (Cambareri and Eichner, 1993), MEP staff determined a relationship between ammonium-nitrogen and alkalinity concentrations (NH4-N = 0.0352*ALK - 0.3565; r2 = 0.82). This relationship was used to estimate ammonium-nitrogen concentrations from the available landfill monitoring data. MEP project staff reviewed the well logs, water table, and contaminant information and determined that three wells best represented the groundwater impact from landfill. Alkalinity concentrations in these three wells average 170 milligrams of $CaCO_3$ per liter with a range of averages from 37.5 to 412.8 mg $CaCO_3$ l^{-1}, while NO3-N concentrations average 0.1, 1.2 and 2.7 mg l^{-1}. 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).

Based on the water quality review and estimated ammonium-nitrogen, MEP staff determined average ammonium-N plus nitrate-N concentrations in three down gradient wells landfill at 7.0 mg l^{-1}. Using a digitized area of solid waste based on the review of available maps and aerial photos and using the Westport River recharge rate, MEP staff developed an estimated annual total nitrogen load of 277 kg from the Westport Landfill.

It is acknowledged that this approach for estimating a nitrogen load from the Westport landfill includes a number of assumptions, but these are appropriate based on the available data. 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 by including conservatism in nitrogen loading estimates when uncertainty existed 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.

Crapo Hill Landfill

The Crapo Hill Landfill is a municipal landfill shared by the Town of Dartmouth and the City of New Bedford through the Greater New Bedford Regional Refuse Management District and is located within the Hixville Road - East Branch sub-watershed (sub-watershed #2). According to MassDEP files, five (5) monitoring wells are monitored on a regular schedule and files contained results from eight (8) sampling runs between July 2008 and April 2010. Available laboratory data from the groundwater samples were analyzed for nitrate-nitrogen (NO3-N) and other constituents, but were not analyzed for any other nitrogen components.

Based on a previous review of monitoring data from the groundwater plume associated with the Town of Brewster landfill (Cambareri and Eichner, 1993), MEP staff determined a relationship between ammonium-nitrogen and alkalinity concentrations (NH4-N = 0.0352*ALK - 0.3565; r2 = 0.82). This relationship was used to estimate ammonium-nitrogen concentrations from the available landfill monitoring data. MEP project staff reviewed the well logs, water table, and contaminant information and determined that three wells best represented the groundwater impact from landfill. Alkalinity concentrations in these three wells average 104 milligrams of $CaCO_3$ per liter with a range of averages from 90 to 118.2 mg $CaCO_3$ l^{-1}, while NO3-N concentrations average 0.7, 3.8 and 5.6 mg l^{-1}. 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).

Based on the water quality review and estimated ammonium-nitrogen, MEP staff determined average ammonium-N plus nitrate-N concentrations in three down gradient wells landfill at 6.7 mg l^{-1}. Using a digitized area of solid waste based on the review of available maps and aerial photos and using the Westport River recharge rate, MEP staff developed an estimated annual total nitrogen load of 643 kg from the Crapo Hill Landfill.

It is acknowledged that this approach for estimating a nitrogen load from the Crapo Hill landfill includes a number of assumptions, but these are appropriate based on the available data. 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 by including conservatism in nitrogen loading estimates when uncertainty existed 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: Freshwater Wetlands

The data collected at the MEP gauge site at the lowermost reach of the freshwater portion of the Westport River generally produced measured nitrogen loads that were higher than what the preliminary MEP watershed nitrogen loading model indicated. Since the MEP assessment approach is data-driven, MEP staff began the process of exploring the cause of these higher nitrogen loads by re-reviewing all of the data leading to the preliminary watershed loads, including the watershed delineations, the nitrogen loading inputs, and re-reviewing the streamflow and concentration data. These steps confirmed the evaluations and provided support for nitrogen attenuation in the freshwater wetlands being less than in smaller river systems. A similar approach was taken in the Slocums River system, where MEP staff identified the extensive wetland and swamp lands surrounding most of the streams and rivers feeding into the estuary as the most likely cause of the high nitrogen loads in the Slocums River Estuary.

The nitrogen load assigned to freshwater wetlands bordering the Westport River is consistent with the nitrogen loading assigned to the freshwater wetlands bordering the Slocums/Paskamansett River as analyzed by the MEP in the Town of Dartmouth. It was clear from both the Westport River measurements and the Paskamansett River measurements that attenuation of nitrogen in these riverine wetlands was relatively low, with added nitrogen being transformed but not removed. Specifically, the indication of wetland "N saturation" is based upon atmospheric N deposition being only partially removed. This determination for the Westport River freshwater wetlands is reasonable based on the available similarly determined measurements for the adjacent Slocums River System and the observations developed during the assessments of other freshwater regions of other MEP system in similar settings (albeit, smaller systems).

The Westport River, Paskamansett River and Acushnet River have similar geology in their drainage basins, are structurally similar, have significant associated freshwater wetland resources adjacent to the rivers and are highly nitrogen enriched (TN's of 1.3 mg/L, 1.1 mg/L and 1.2 mg/L, respectively). The geology of these systems (particularly the Westport and Paskamansett Rivers) differs from those on Cape Cod and the islands, in that the watersheds tend to be topographically defined and even have bedrock outcrops, whereas Cape Cod is primarily glacial sands and till with watersheds primarily defined by differences in water elevation within a porous matrix. As such, based on site specific analysis that is a foundation of the MEP, some of the factors associated with surface water systems in the Westport and Dartmouth region are different than those developed for Cape Cod and the Islands.

In most of the Cape Cod streams, which are in different settings, application of the MEP N loading approach has produced very good agreement with measured stream N loads. These sandy aquifer-dominated systems typically support limited freshwater wetland areas and much lower stream flows. In these cases the N attenuation rates of 20% to 30% are typical, possibly due to the higher detention time. In contrast, the rivers along the northwestern edge of Buzzards Bay and specifically those associated with the Westport and Slocums River Estuaries, are underlain by bedrock and till, have comparatively high stream flows and extensive freshwater wetlands. In these systems, N contact time in the wetlands will be shorter and like freshwater ponds with short residence times, they should attenuate less nitrogen. In addition, reviews of river wetlands have indicated that they do have threshold effects like those seen in estuaries and ponds. This means that these freshwater wetlands can become loaded with nitrogen and act as transformers of nitrogen (changing nitrate+nitrite to organic forms), but not attenuators of nitrogen (e.g., USDA, 2011). This change appears to be related to the amount of nitrogen received, as well as inter-related factors such as hydraulic residence time, temperature, plant surface coverage, and plant density (e.g., Hagg et al., 2011; Kröger, et al., 2009; Alexander, et al., 2008).

It is important to note that the wetlands are not actually a nitrogen source, but they merely have a lower rate of nitrogen removal of the nitrogen deposited upon them, than in smaller low flow wetlands. The result is a low total combined attenuation of all nitrogen sources in the Westport River (fresh) being 15%. The results indicated that the large river/wetland systems of the Westport River and the Paskamansett River are operating in a similar manner.

The MEP results are also consistent with studies by other researchers that found that the ability of river wetlands to attenuate nitrogen is directly related to their hydraulic residence times (e.g., Jansson, et al., 1994; Perez, et al., 2011; Toet, et al., 2005) with longer residence times resulting in greater nitrogen reduction. Direct data in the overall MEP study area generally confirms this relationship with lower flow/longer residence times streams on the eastern portion of the overall MEP study area having greater nitrogen attenuation, as well as ponds and lakes, which have even longer residence times, having nitrogen attenuation rates of 50% or higher (e.g., Howes, et al., 2006).

In order to incorporate the nitrogen loading from the wetland areas in the Westport River watershed, MEP staff assigned the water surface nitrogen loading factor to the wetland areas identified in a MassGIS/MassDEP wetland coverage (Figure IV-3). This wetland coverage is based on a scanning of USGS 1:25,000 quadrangles. For the purposes of the MEP assessment, the treatment of these wetlands as water surfaces is appropriately conservative without further data to refine the spatial differences in residence times, plant communities/densities and the role of seasonal impacts along the various streams and rivers in the Westport River watershed system.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas in the Westport 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 Cape Cod Commission Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and the MassDEP Nitrogen Loading Computer Model Guidance Document (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the MEP nitrogen loading analysis for the Westport River watershed are summarized in Table IV-1.

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 sub-watershed. Project staff also checked this information against parcel-based rights-of-way.

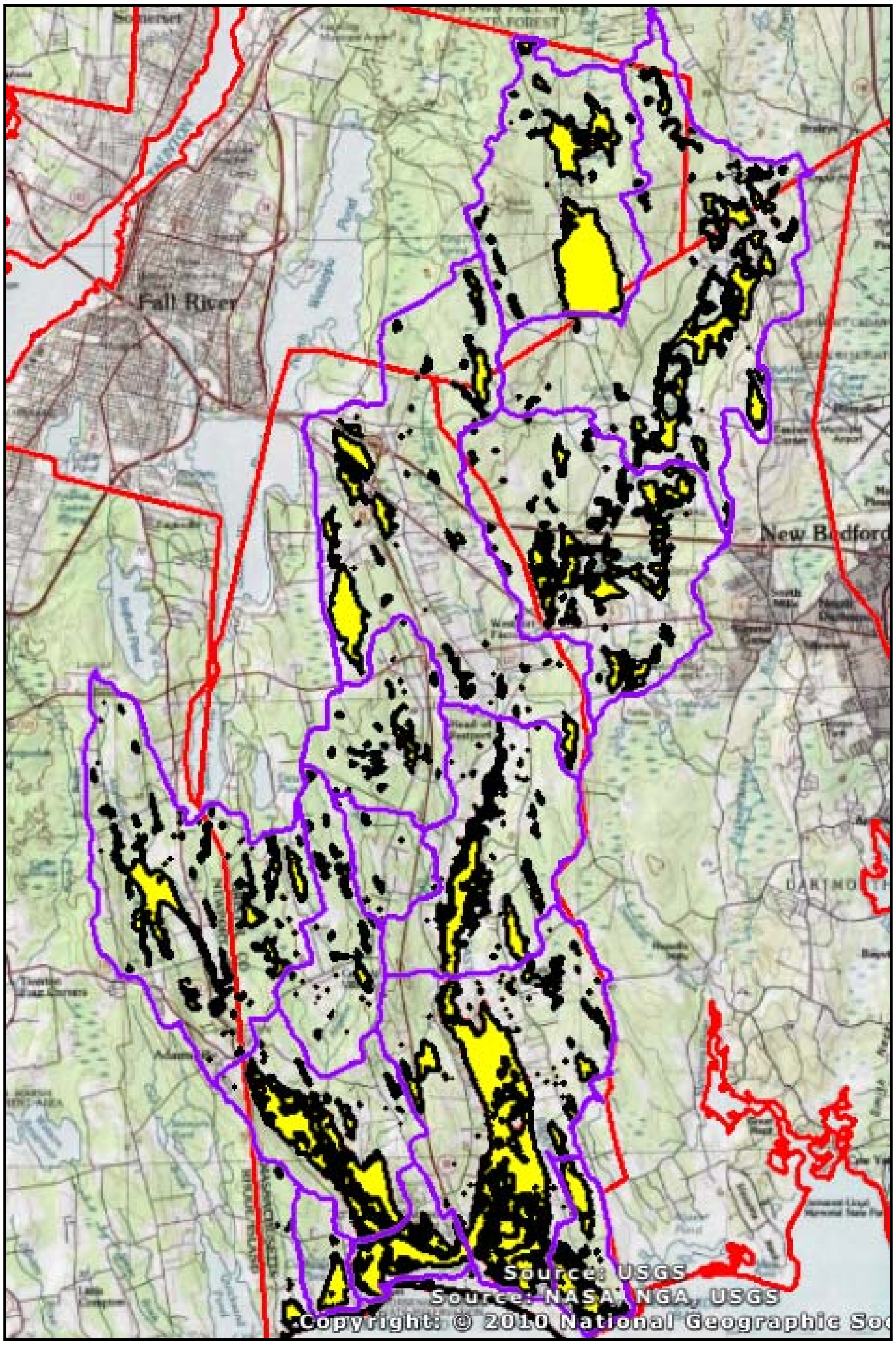


Figure IV-3. Wetland areas in the Westport River watersheds. All areas colored in yellow are wetlands areas delineated by MassGIS/MassDEP based on USGS quadrangles. All these areas were assigned a nitrogen loading concentration of 1.09 mg/l TN in the MEP watershed nitrogen loading model.

IV.1.3 Calculating Nitrogen Loads

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

The review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (condominiums, golf courses, etc.) was 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 Westport River estuary. The assignment effort was 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, sub-watershed modules were generated for each of the 14 sub-watersheds in the Westport River study area. These sub-watershed modules summarize, among other things: parcel area, parcel frequency by land use category, sewer connections, and road area. All relevant nitrogen loading data is assigned to each sub-watershed. Individual sub-watershed information is then integrated to create the Westport River Watershed Nitrogen Loading module with summaries for each of the individual 14 sub-watersheds. The sub-watersheds are generally paired with functional embayment/estuary units for the Linked Watershed-Embayment Model’s water quality component.

For management purposes, the aggregated estuary watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Westport River study area, the major types of nitrogen loads are: wastewater (*e.g.*, septic systems), landfills, agricultural, turf fertilizers (lawns and the golf course), impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-2). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-5). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model. In order to acknowledge uncertainties in stream or pond nitrogen loading measurements, attenuation rates are typically conservatively assigned in the nitrogen-loading model and may differ from the monitoring results discussed in Section IV-2.

Table IV-1. Primary Nitrogen Loading Factors used in the Westport River MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from watershed-specific data.

Nitrogen Concentrations:	mg/l	Recharge Rates:[2]	in/yr
Road Run-off	1.5	Impervious Surfaces	45.7
Roof Run-off	0.75	Natural and Lawn Areas	30.46
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing developed parcels wo/water accounts and buildout residential parcels	188 gpd[3]
Wastewater Coefficient	23.63		
Town of Westport Landfill Load (kg/yr)	277		
Crapo Hill Landfill Load (kg/yr)	643	Multi-family residential parcels	376 gpd
Jarabeck Farm Landfill Load (kg/yr)	387	Existing developed parcels w/water accounts:	Measured annual water use
Fertilizers:		Commercial and Industrial Buildings buildout additions[4]	
Average Residential Lawn Size (sq ft)[1]	5,000	Commercial	
Residential Watershed Nitrogen Rate (lbs/lawn)[1]	1.08	Wastewater flow (gpd/1,000 ft2 of building):	180
Nitrogen leaching rate	20%	Building coverage:	15%
Average Single Family Residence Building Size (based on other towns; sq ft)	1,500	Industrial	
		Wastewater flow (gpd/1,000 ft2 of building):	44
Farm Animals	kg/yr /animal	Building coverage:	5%
Horse	32.4	Crops	kg/ha/yr
Cow/Steer	55.8	Hay, Pasture	5
Goats	7.3	Corn, Vegetables, Vineyard, Fruit	34
Hogs	14.5	Crop N leaching rate	30%
Chickens	0.4	Cranberry Bog (leaching included)	6.9
Animal N leaching rate	40%		

Notes:
1) Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.
2) Based on precipitation rate of 50.77 inches per year (1971-2000 NOAA average for closest long-term precipitation gauge (New Bedford))
3) average based on all Town of Dartmouth municipal water accounts measured flow in all single-family residences in the watershed
4) Based on characteristics of Town of Falmouth land uses: existing water use and building coverage for similarly classified properties

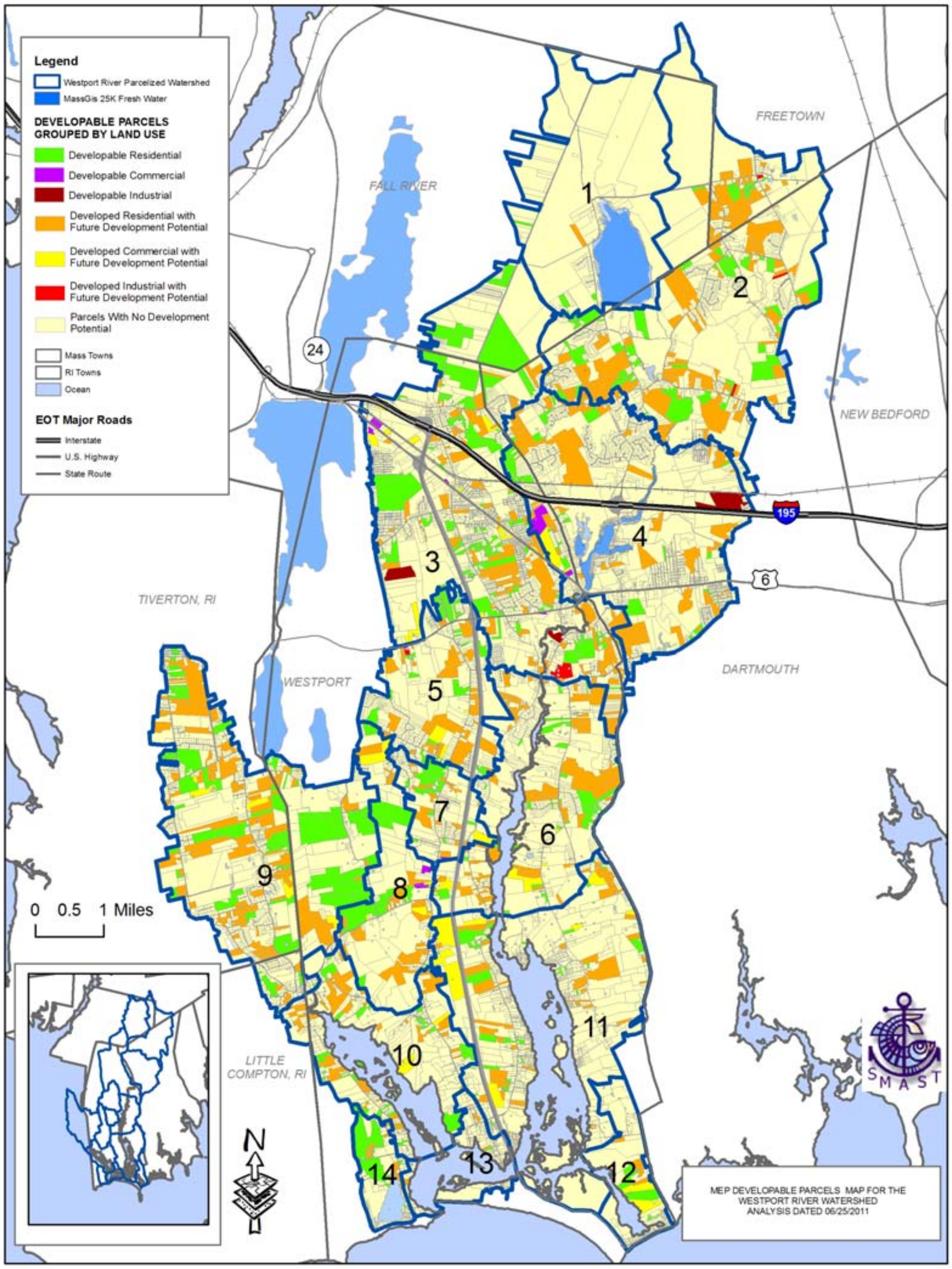
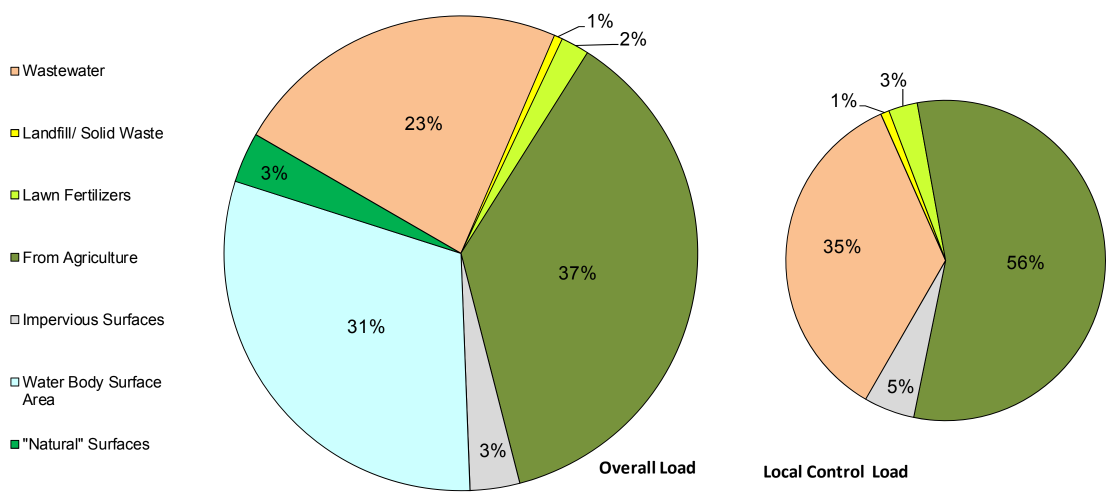


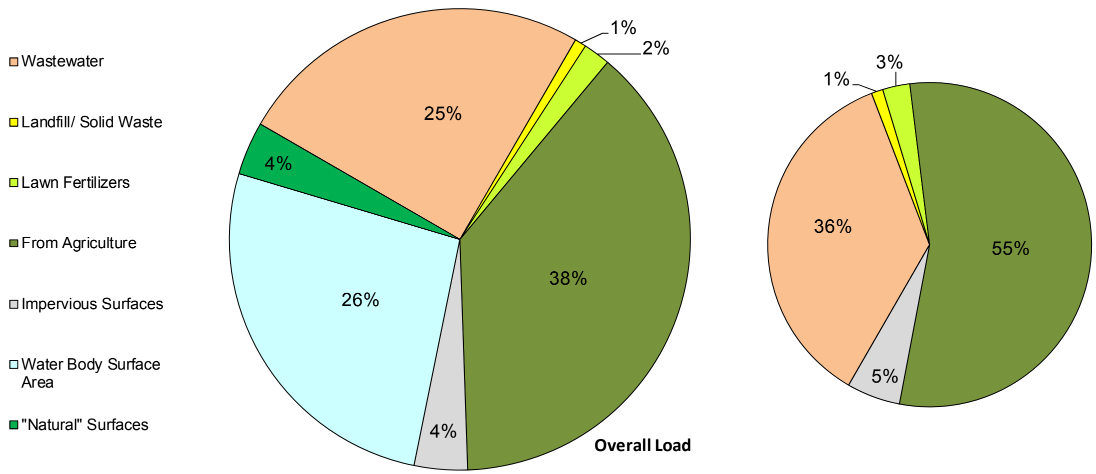
Figure IV-4. Parcels, Parcelized Watersheds, and Developable Parcels in the Westport River watersheds. Parcels colored orange, yellow, and red are developed parcels (residential, commercial, and industrial, respectively) with additional development potential based on current town zoning, while parcel colored green, purple, and brown are corresponding undeveloped parcels, respectively, classified as developable by the respective town assessors. 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. The amount of additional development is predominantly based on minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. All buildout results were reviewed with respective town staff and/or committees.

Table IV-2. Westport River Watershed Nitrogen Loads. Existing and buildout unattenuated and attenuated nitrogen loads are shown along with breakdowns in to component sources. Attenuated nitrogen loads are based on measured and assigned attenuation factors for streams. Old County Road gauge load is an annual load and is unadjusted for the lower measured flows during the summer. Stream attenuation factors are based on annual measured loads (see Section IV.2). All nitrogen loads are kg N yr-1.

		Westport River N Loads by Input (kg/y):								***Present N Loads***			***Buildout N Loads***		
Watershed Name	**shed ID#**	**Wastewater**	**Landfill/ Solid Waste**	**Lawn Fertilizers**	**From Agriculture**	**Impervious Surfaces**	**Water Body Surface Area**	**"Natural" Surfaces**	**Buildout**	**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**	**Atten %**	**Atten N Load**
Westport River System		52,460	1,306	4,373	83,990	7,678	69,219	7,767	57,205	226,793		210,422	283,999		263,755
East Branch Total		40,316	1,306	3,053	61,821	6,013	42,543	6,003	41,839	161,054		147,751	202,893		186,704
East Branch S Total		6,315	-	422	5,864	743	10,915	595	14,112	24,853		24,853	38,965		38,965
East Branch S	11	5787	0	387	5,312	676	10,113	476	11,386	22,751		22,751	34,137		34,137
The Let	12	528	0	35	553	67	802	118	2,726	2,102		2,102	4,829		4,829
East Branch N Total		34,001	1,306	2,631	55,957	5,270	31,628	5,408	27,727	136,201		122,898	163,928		147,738
Kirby Brook	5	2842	0	237	2,000	454	1,745	371	1,779	7,648		7,648	9,428		9,428
East Branch N	6	3394	277	230	30,844	436	1,835	612	4,411	37,627		37,627	42,038		42,038
Snell Creek	7	1663	0	116	348	197	487	159	2,295	2,970		2,970	5,265		5,265
Old County Rd Gauge Total		26,103	1,029	2,048	22,764	4,183	27,562	4,266	19,241	87,956	15%	74,653	107,197	15%	91,008
Copicut Reservoir	1	325	0	26	-	157	6,043	720	116	7,271		7,142	7,388		7,258
Hixville Rd - E Branch	2	5161	643	390	841	748	7,999	1319	4,106	17,101		17,101	21,207		21,207
Old County Rd - E Branch	3	15008	387	972	16,531	1873	7,031	1284	10,006	43,085		43,085	53,092		53,092
Rt 177 Gauge - E Branch	4	5609	0	660	290	1406	6,488	943	5,013	15,397		15,397	20,409		20,409
West Branch Total		9,739	-	1,156	21,694	1,435	9,355	1,599	14,792	44,977		41,909	59,769		55,715
Angeline Brook	8	1123	0	554	8,774	162	1,597	309	4,954	12,518		12,518	17,472		17,472
Adamsville Brook	9	6229	0	448	4,935	1013	6,996	829	6,578	20,449	15%	17,382	27,027	15%	22,973
West Branch	10	2387	0	154	7,986	260	762	461	3,260	12,009		12,009	15,270		15,270
Westport Harbor Total		2,406	-	165	475	230	301	166	575	3,742		3,742	4,316		4,316
Westport Harbor	13	1565	0	102	30	122	85	92	438	1,996		1,996	2,433		2,433
Cockeast Pond	14	841	0	63	445	108	216	74	137	1,746		1,746	1,883		1,883
East Branch N Estuary Surface	6						1,591			1,591		1,591	1,591		1,591
West Branch Estuary Surface	10						4,071			4,071		4,071	4,071		4,071
East Branch S Estuary Surface	11						7,636			7,636		7,636	7,636		7,636
The Let Estuary Surface	12						718			718		718	718		718
Westport Harbor Estuary Surface	13						2,477			2,477		2,477	2,477		2,477
Cockeast Pond Estuary Surface	14						526			526		526	526		526

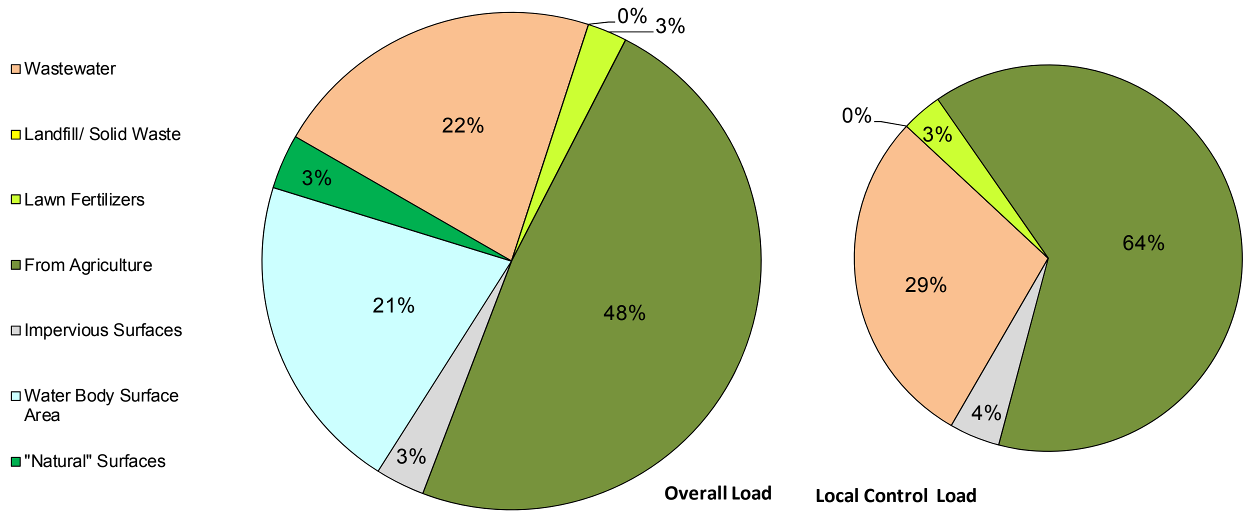


A. Whole System: Westport River Estuary

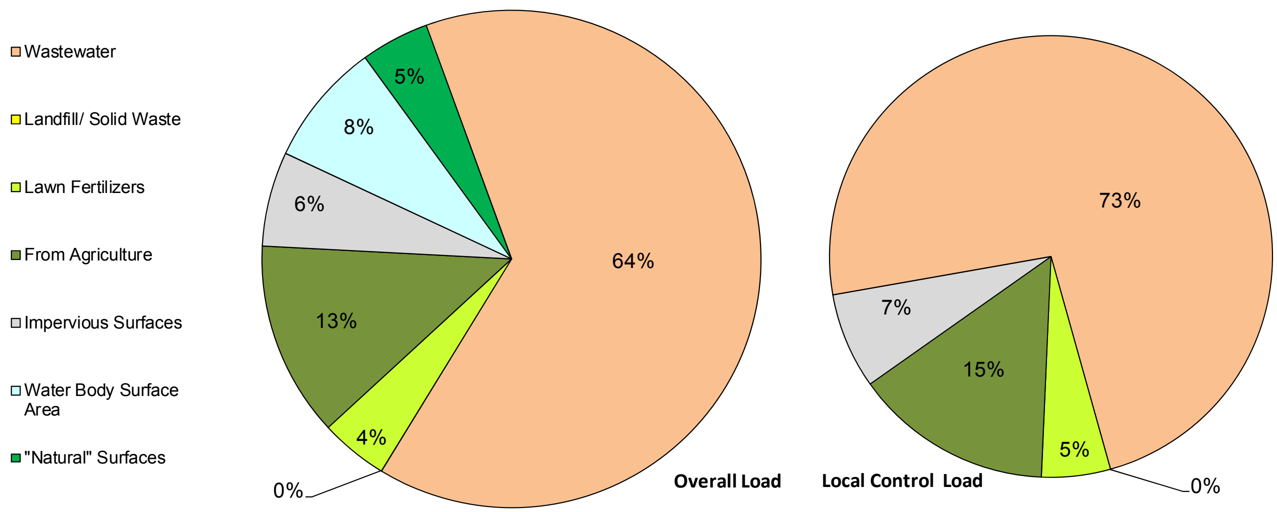


B. East Branch: Westport River Estuary

Figure IV-5. Land use-specific unattenuated nitrogen loads (by percent) to the a) whole Westport River watershed and b) the East Branch sub-watershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control. "Water body surface area" includes atmospheric inputs to both freshwater ponds and wetlands and the estuary surface.



C. West Branch: Westport River Estuary



D. Westport Harbor: Westport River Estuary

Figure IV-5 (continued). Land use-specific unattenuated nitrogen loads (by percent) to the c) the West Branch sub-watershed and d) the Westport Harbor sub-watershed. "Overall Load" is the total nitrogen input within the watershed, while the "Local Control Load" represents only those nitrogen sources that could potentially be under local regulatory control.

Freshwater Pond Nitrogen Loads

Freshwater ponds are generally watershed sites of natural nitrogen reduction (or attenuation) prior to the watershed nitrogen reaching an estuary. In southeastern Massachusetts, these ponds can be reservoirs created by the placement of a dam or kettle hole depressions left by retreating glaciers during the last continental glaciation. Kettle hole ponds intercept the surrounding groundwater table revealing what some call “windows on the aquifer.” Groundwater typically flows into these ponds along the up-gradient shoreline, then lake water flows back into the groundwater system along the down gradient shoreline. Occasionally these ponds will also have a stream outlet, which will “short circuit” flow, leading to more rapid residence times than if the pond was only groundwater fed. This same sort of shortened residence time is more common in river-fed reservoirs because water levels are more closely monitored to avoid upstream flooding.

As nitrogen loads flow into ponds either with groundwater or in a stream, the relatively more productive pond ecosystems tend to 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 is removed from the estuary watershed system, mostly through burial in the pond sediments and denitrification that returns it to the atmosphere. Following these reductions, the remaining (attenuated) loads flow downstream or back into the groundwater system and eventual discharge into the down gradient embayment or through a stream outlet directly to the estuary. Ponds with short residence times appear to have much more limited attenuation.

Nitrogen attenuation in freshwater ponds has generally been found to be between 30 and 80% in MEP analyses of ponds with extensive water quality monitoring and system characterization, but the primary ponds in the Westport River watershed (Lake Noquochoke and Copicut Reservoir) do not appear to have significant attenuation. They have no assigned attenuation factors in the MEP watershed nitrogen loading model. Review of the measured nitrogen loads on the freshwater portion of the Westport River results in an assigned nitrogen attenuation rate (see Section IV.2) that cumulatively impacts the nitrogen load reaching the estuary. This attenuation rate is similar to those that have been assigned to other large streams with monitoring through the MEP (*e.g.* Slocums and Waquoit Bay), but the apparent lack of significant attenuation in the ponds is something that may offer some opportunities for additional natural reductions in nitrogen loads flowing through them if the management of their residence times is adjusted. In order to review whether a pond-specific nitrogen attenuation rate should be used, additional information for assigning a nitrogen attenuation rate would be required: nitrogen concentrations, evaluation of impacts of sediment regeneration, temperature profiles, and bathymetric information. Information of this type would be necessary to discern how much of the attenuation currently assigned to the stream may be occurring in the ponds and whether there are alternative management strategies for the ponds that could lead to increased attenuation and lower loads to the Westport River estuary.

Buildout

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

It should be noted that the initial MEP 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 does not include potential impacts associated with the higher densities usually associated with Chapter 40B affordable housing projects. The fourth step, the discussions with town planners, and, town boards (and wastewater consultants), generated some additional insights on planned development, and included discussion of developments planned for government or public service parcels, and updates to assessor classifications, including lands purchased by the town as open space. This final step should continue as the Towns conduct nitrogen management planning and should include updates on parcels initially identified as developable or undevelopable and application of more detailed zoning provisions. As planning proceeds the Towns may request additional refined buildout scenarios to account for specific land-use shifts or projects that may be deemed likely within the watershed.

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 a 40,000 square foot minimum lot size specified in town zoning and the result is rounded down to two. As a result, two additional residential lots would be added to the sub-watershed in the MEP buildout scenario. This addition could then be modified during discussion of town staff.

Other provisions of the MEP buildout assessment include differentiated treatment of undevelopable lots, commercial and industrial properties, and lots less than the minimum areas specified by zoning. Properties classified by the town assessors as "undevelopable" (*e.g.*, MassDOR land use codes 132, 392, and 442) are not assigned any development at buildout (unless revised by the town review). The buildout also included the latest (3/11) protected open space coverage from the Buzzards Bay National Estuary Program (personal communication, Sarah Williams, Regional Planner, 3/11). 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 a developable residential property (130 land use code) will be assigned an additional residential dwelling in the MEP buildout scenario even though the minimum lot size in the area is 40,000 square feet. Most town zoning bylaws have a lower minimum lot size for pre-existing lots (usually 5,000 square feet) that will minimize instances of regulatory takings. Existing developed residential properties that are larger than zoning's minimum lot sizes are also assigned additional development potential only if enough area is available to accommodate at least one additional lot as specified by the zoning minimum.

Following the completion of the initial buildout assessment for the Westport River watersheds, MEP staff distributed the results to officials from each of the towns in the watershed and asked for their corrections/feedback. Staff from the towns of Freetown, Little Compton, and Tiverton, and the City of Fall River provided corrections to the initial buildout (personal communications from respectively: Laurie Carvalho Muncy, Planning/Land Use Administrator, 4/11; Denise M. Cosgrove, Clerk for the Board of Assessors, 5/11; David Robert, Town Assessor, 5/11; and Mike Labossiere, Water Division, 4/11). The Town of Westport Estuaries Project Committee held a number of meetings to review and correct the initial buildout and adjusted buildout on approximately 1,250 parcels and included a review of wetland restrictions, dead end street limits, and parcel access (personal communication, Tim Gillespie, 5/11). The Westport Committee generally assumed that “current use” forest lands protected under Chapter 61 will remain forest lands at buildout, although a few properties were identified for development. Chapter 61 forest lands, which are generally classified under land use code 601, are not generally assigned additional development in the Westport River MEP buildout. A team of Town of Dartmouth staff also completed a more detailed review, including removing wetland areas, removing any future residential development or redevelopment in business or industrial zones, and included an update of open space and protected lands (personal communication, Mike O’Reilly, Environmental Affairs Coordinator, 6/11). MEP staff also included restrictions based on the town Aquifer Protection Zone, which limits building footprint and covers most of the Westport River MEP watershed in Dartmouth, based on guidance from town staff. Buildout for Dartmouth also included wastewater corrections for connection of properties within sewer service areas. All town changes were incorporated into the MEP buildout estimates for the Westport River.

All the parcels with additional buildout potential within the Westport River watershed are shown in Figure IV-4. Overall, there are a projected 4,328 additional residences at buildout; 54% of this addition is within the Town of Westport. Each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for wastewater and impervious surfaces minus the sewer corrections in Dartmouth. Residential additions also include lawn fertilizer nitrogen additions. All wastewater loads in the watershed are assumed to come from on-site septic systems. Cumulative unattenuated buildout loads are indicated in a separate column in Table IV-2. Buildout additions within the Westport River watersheds will increase the unattenuated nitrogen loading rate by 26%.

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 Westport 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 being the case in the developed region of southeastern Massachusetts but more so on Cape Cod). 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) can be diminished by natural biological processes that represent removal (not just temporary storage). However, this potential natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes to varying degrees based on habitat and residence time. In the watershed for the Westport River embayment system, a portion of the freshwater flow and transported nitrogen passes through several surface water systems (e.g. the Westport River discharging to the head of the East Branch from the up-gradient Lake Noquachoke, Kirby Brook discharging to the East Branch, Snell Creek discharging to the East Branch, Adamsville Brook discharging to the head of the West Branch and Angeline Brook discharging to the West Branch) prior to entering the estuary, producing the opportunity for significant nitrogen attenuation under appropriate conditions (Figure IV-6).

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2000). Similarly, MEP analysis of the Quashnet River (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 Westport River embayment system. MEP conducted long-term measurements of natural attenuation relating to the most significant 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 5 major surface water flow systems in the Westport River watershed, 1) Westport River discharging from Lake Noquachoke to the head of the Westport River system, 2) Kirby Brook discharging from upland to the East Branch of the Westport River, 3) Snell Creek discharging from upland to the East Branch of the Westport River, 4) Adamsville Brook discharging from Adamsville Pond to the head of the West Branch and 5) Angeline Brook discharging from upland to the West Branch. Regarding Dunham's Brook and Hulda Brook, both located on the West Branch, given the small size of the drainage areas (0.95 and 0.22 sq-miles respectively) as well as the extremely small and intermittent flows by comparison to the other five major surface water discharges measured by the MEP, the technical team agreed to exclude those two small brooks from the gauging program.

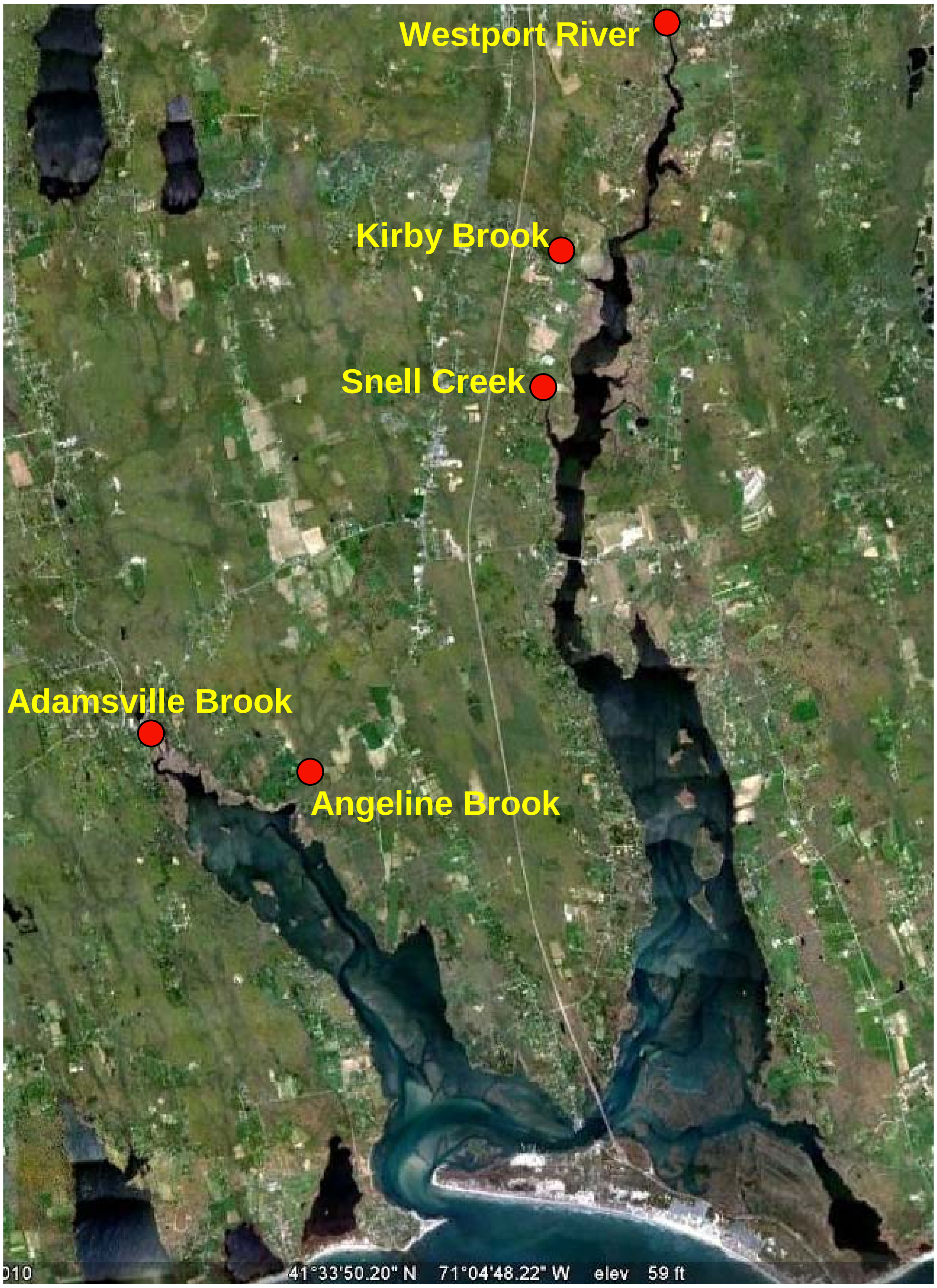


Figure IV-6. Location of Stream gauges (red dots) in the East and West Branches of the Westport River embayment system. The two main discharges are the Westport River from Lake Noquachoke, 67% of freshwater flowing into the East Branch, and Adamsville Brook to Adamsville Pond, 58% of freshwater flowing into the West Branch.

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 up-gradient from the various gauging sites. Flow and nitrogen load were measured at the gauges in each freshwater stream site for between 16 and 24 months of record depending on the stream gauging location (Figures IV-10 to IV-14). 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 gauge 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 gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauges. 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 Westport River embayment system.

The annual flow record for the surface water flow at each gauge was merged with the nutrient data set generated through the weekly water quality sampling performed at the gauge locations to determine nitrogen loading rates to the Westport 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 gauging site. For each of the stream gauge 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 gauged stream currently reduces (percent attenuation) nitrogen loading to the overall embayment systems.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Westport River Discharge to the East Branch of the Westport River Estuary

Like most surface water features in the MEP study region that typically emanate from a specific pond, the Westport River, which discharges into the head of the East Branch of the Westport River Estuary, does have a significant up-gradient lake from which the river discharges. Moreover, the Westport River is a complex network of tributary wetlands and ponds with associated rivers and brooks, all of which come together to form what is commonly considered the Westport River. Specifically, the Shingle Island River emanates from a large wetland in the upper portions of the Westport River watershed. The Shingle Island River flows into Lake Noquochoke which is source water for the Westport River. The Westport River down gradient of Lake Noquochoke flows into and out of Forge Pond and then receives flow from Bread and Cheese Brook which, flows into and out of both Trout Pond and Head Dam Pond. Collectively, all these tributary rivers, brooks and associated ponds which feed the Westport River provide significant potential for nitrogen attenuation and must be considered for accurate determination of the nitrogen load to the head of the East Branch of the Westport River. Based on numerous previous studies completed by the MEP on other systems in southeastern Massachusetts, the outflow from the ponds/lake/wetlands and the wooded areas up-gradient of the Westport River gauge very likely contribute to the attenuation of nitrogen and also provides for a direct measurement of the nitrogen attenuation. The combined rate of nitrogen attenuation by the biological processes that occur in the various surface water features was determined by comparing the present predicted (calculated from land use analysis) nitrogen loading to the sub-watershed region contributing to the ponds/lake/wetlands and wooded areas above the gauge site and the measured annual discharge of nitrogen to the head of the East Branch from the freshwater portion of the Westport River, Figure IV-6.

At the Westport River gauge site (established at the Old County Road bridge crossing), a continuously recording vented calibrated water level gauge was installed to yield the level of water in the channel that carries the flows and associated nitrogen load to the head of the East Branch of the Westport River estuarine system. As the lower reach of the Westport River is tidally influenced, the stage record from the gauge was checked to make sure there was no tidal influence in the record at low tide. To confirm that freshwater was being measured at low tide, the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity of the water samples taken from the Westport River at Old County Road at low tide was determined to be 0.3 ppt. Therefore, the gauge location was deemed acceptable for making freshwater flow measurements at low tide. Calibration of the gauge was checked monthly. The gauge on the Westport River at Old County Road was installed on

January 16, 2005 and was set to operate continuously for 16 months such that a complete hydrologic year would be captured in the flow record. Stage data collection continued until December 19, 2007 for a total deployment of 23 months.

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 Westport River (Old County Road gauge site) based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the head of the East Branch at the top of the Westport River estuarine system and reflective of the biological processes occurring in the stream channel and extensive network of lake/ponds/wetlands and wooded area contributing to nitrogen attenuation (Figure IV-10 and Table IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey/Buzzards Bay Project/MEP defined watershed delineations to determine long-term average freshwater discharge expected at each gauge site based on area and average recharge.

The annual freshwater flow record for the Westport River as measured by the MEP was compared to the long-term average flows determined by the USGS/BBP/MEP modeling effort (Table III-1). The measured freshwater discharge from the Westport River at the Old County Road gauge location was 0.2% above the long-term average modeled flows for the 2006-2007 hydrologic year. 2006-2007 flows were utilized as a point of comparison because the annual precipitation in the preceding year was unusually high, 31% over the 2001-2010 mean precipitation. Annual precipitation during the 2006-2007 gauge deployment period was only slightly under the long term mean (9%) and therefore more in line with average conditions. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 217,352 m^3/day compared to the long term average flows determined by the watershed modeling effort (216,953 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 Westport River discharging from the sub-watershed indicate that the Westport River is capturing the up-gradient recharge (and loads) accurately. It is important to note that while the annual flow measured at the stream gauge confirmed the accuracy of the watershed delineations, given the strong seasonal variations in the flow of the Westport River, it was necessary to use summer time flows and associated nitrogen concentrations in the calibration of the water quality model. This seasonal variation observed in the measured stream flow is not unusual given the geology of the watershed (stratified drift/till) compared to stream watersheds on Cape Cod that are dominated by sandy outwash sediments and are more groundwater dominated. Using summer time flows and total nitrogen concentrations enabled an appropriate comparison to the summer based water quality monitoring nitrogen concentrations obtained from the water quality station located in the upper portion of the East Branch of the Westport River as well as summertime benthic flux rates. While the average total nitrogen concentration on an annualized basis was measured at 0.938 mg N L^{-1}, summer time average total nitrogen concentrations under lower flow conditions were measured at 1.36 mg N L^{-1}, which compares favorably with summer time total nitrogen concentrations from the estuarine water quality monitoring program (1.4 mg N L^{-1}).

Total nitrogen concentrations within the Westport River outflow were moderate to high, 0.938 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 204 kg/day and a measured total annual TN load of 74,397 kg/yr. In the Westport River, nitrate made up well less than half of the total nitrogen pool (28%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the lake/ponds/wetland areas up-gradient of the gauge was partially taken up by plants within these different aquatic systems as well as the stream ecosystems. Given the relatively low levels of remaining nitrate in the stream discharge, the possibility for additional uptake by freshwater systems might be limited.

From the measured nitrogen load discharged by the Westport River to the head of the East Branch and the nitrogen load determined from the watershed based land use analysis, it appears that there is moderate to low nitrogen attenuation of upper watershed derived nitrogen during transport to the Westport River and the head of the East Branch of the Westport River estuary. Based upon lower total nitrogen load (74,397 kg yr^{-1}) discharged from the Westport River at Old County Road compared to that added by the various land-uses to the associated watershed (87,956 kg yr^{-1}), the integrated attenuation in passage through Lake Noquochoke, various ponds, streams and freshwater wetlands prior to discharge to the estuary is 15% (i.e. 15% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams evaluated under the MEP is slightly lower than expected given the nature of the up-gradient bog/wetland/wooded areas capable of attenuating nitrogen and potentially low residence times. The directly measured nitrogen load from the Westport River was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

It is important to note that while the attenuation determined for the Westport River at the Old County Road gauge site is low, it is consistent with an ancillary water quality study that was undertaken by Coastal Systems Program (SMAST/MEP) in the summer of 2008 whereby water quality sampling and flow measurements were completed up-gradient and down gradient of two specific surface water features located up-gradient of the Old County Road gauge location on the Westport River. These aquatic features (Trout Pond and Head Dam Pond) are highly altered pond/wetland areas that are flow through systems for water passing from the up-gradient portions of the Bread and Cheese Brook (which flows into the Westport River up-gradient of the Old County Road gauge) sub-watershed as well as the Westport River down gradient from Forge Pond (Figure IV-7, 8, 9). Based on meetings and discussions with the Town of Westport Conservation Commission Representative beginning in April 29, 2008, CSP scientists agreed to voluntarily conduct a baseline survey of water quality and flow through the Trout Pond and Head Dam Pond systems. The survey was undertaken to provide an initial estimate of water quality and nutrient characteristics to the Conservation Commission in its interest in mounting a restoration project for Trout Pond and Head Dam Pond. The limited base data set was collected to elucidate the merits of pond restoration in the context of habitat improvement and enhancing the biological function of the pond to increase the potential for improved natural attenuation of nutrients prior to nutrient load discharge to the head of the Westport River Estuary. While the water quality monitoring and associated flow measurements were limited to 6 discrete events over one summer, it did provide insight into the degree to which attenuation of inorganic nitrogen may be occurring up-gradient of the Old County Road gauge.

While accurate determination of the amount of removal or attenuation of nitrogen that occurs within a pond requires both long-term monitoring coupled to land-use loading analysis, it was possible to draw some preliminary conclusions regarding the potential attenuation occurring up-gradient of the Old County Road gauge site based upon the initial surveys of Trout Pond and Head Dam Pond in the summer of 2008. It should be stressed that the rates from the 2008 survey are incomplete and merely a preliminary assessment of the function of those two small features.

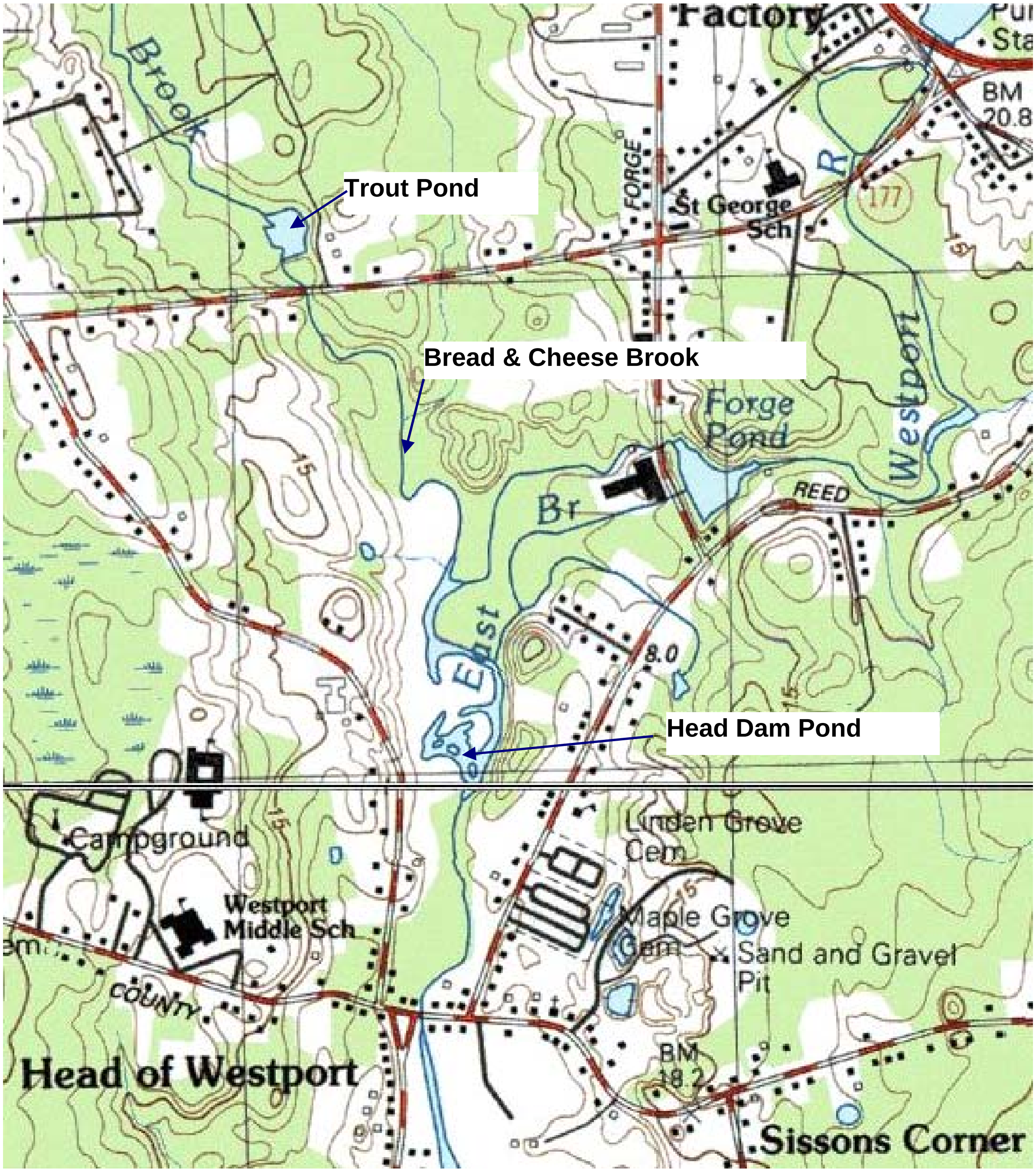


Figure IV-7. Locations for Head Dam Pond, Trout Pond and Forge Pond relative to each other. Flows discharging from Forge Pond and Trout Pond commingle in Head Dam Pond which discharges to the head of the East Brach of the Westport River Estuary System.

From the surface water inflow and outflow measurements it was possible to determine N and P loads into and out of each pond system. In the case of Trout Pond the assessment was difficult due to significant groundwater inflow to the pond (~1,210 m3/d) accounting for 1/3 of the total freshwater flow through the pond. As the land-use analysis of the pond was not part of the initial survey effort, it was not possible to quantify the total nitrogen load to Trout Pond. However, the data collected did reveal some important features relative to the potential for

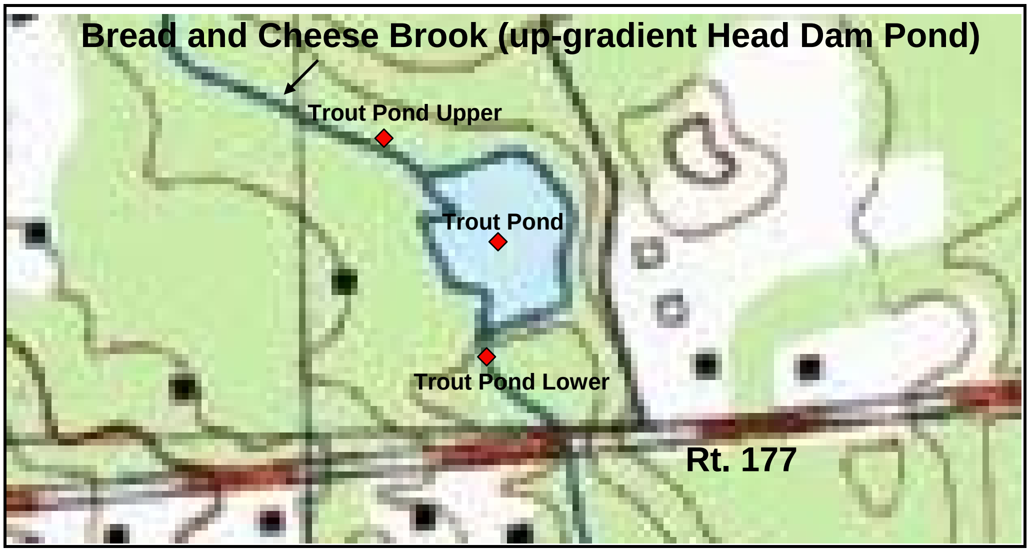


Figure IV- 8. Sampling locations in Trout Pond. Trout Pond Upper is the inflow of Bread and Cheese Brook and Trout Pond Lower is the outflow from the pond and discharges to Head Dam Pond.

enhancing nitrogen retention. First, nitrate represents about ½ of the nitrogen exiting the Trout Pond. Second, the amount exiting the pond is a moderate load if it were to reach the Westport River Estuary. The predominance of nitrate suggests that natural uptake within Trout Pond is not operating efficiently thereby potentially contributing to the lower N-attenuation level seen at Old County Road gauge site. The specific cause of the "passing through" of this easily removable nitrogen form is not presently clear, although the results suggest that some enhancement of natural attenuation may be possible in this system. It is possible that water entering the pond is "short-circuiting", that is flowing through a defined channel to the outflow, rather than mixing with the general pond waters, but this is also unclear. Additional site specific data collection would have to be undertaken to accurately determine the degree to which it is possible to enhance natural attenuation in this aquatic feature.

In contrast to Trout Pond, it was possible to generally balance the inflow and outflow to Head Dam Pond in the 2008 preliminary survey. As such, differences in nitrogen and phosphorus loads (in versus out) were generally representative of processes occurring within the pond, rather than due to an unquantified groundwater inflow. However, it is important to note that there is certainly groundwater inflow to Head Dam Pond, but it is small relative to the overall surface water inflow. This source still needs to be quantified to accurately determine the nutrient transformation processes within the Pond, but given the basic hydrologic balance, some minimum estimates could be made.

Overall, it appeared that Head Dam Pond was removing or attenuating <10% of the nitrogen and none of the phosphorus entering from surface water inflows. While accounting for groundwater inputs may increase this value, a preliminary look at the watershed indicated that the final attenuation value would likely remain low should additional data collection and land use analysis and synthesis be completed. The low estimate of attenuation was consistent with the physical structure of Head Dam Pond, where water enters at the head and flows through channels to the outflow at the breached dam, without any functional pond system being present. A second observation from the 2008 survey was that the overall nitrogen load discharged from the pond is significant. The low rate of observed nitrogen removal, the significant overall load discharged to down gradient systems and the elevated nitrate level supports the contention that additional nitrogen attenuation may be possible if pond restoration were undertaken. However, the 2008 was a preliminary assessment and is not conclusive given the need to close "data gaps" inherent to a preliminary and voluntary survey.

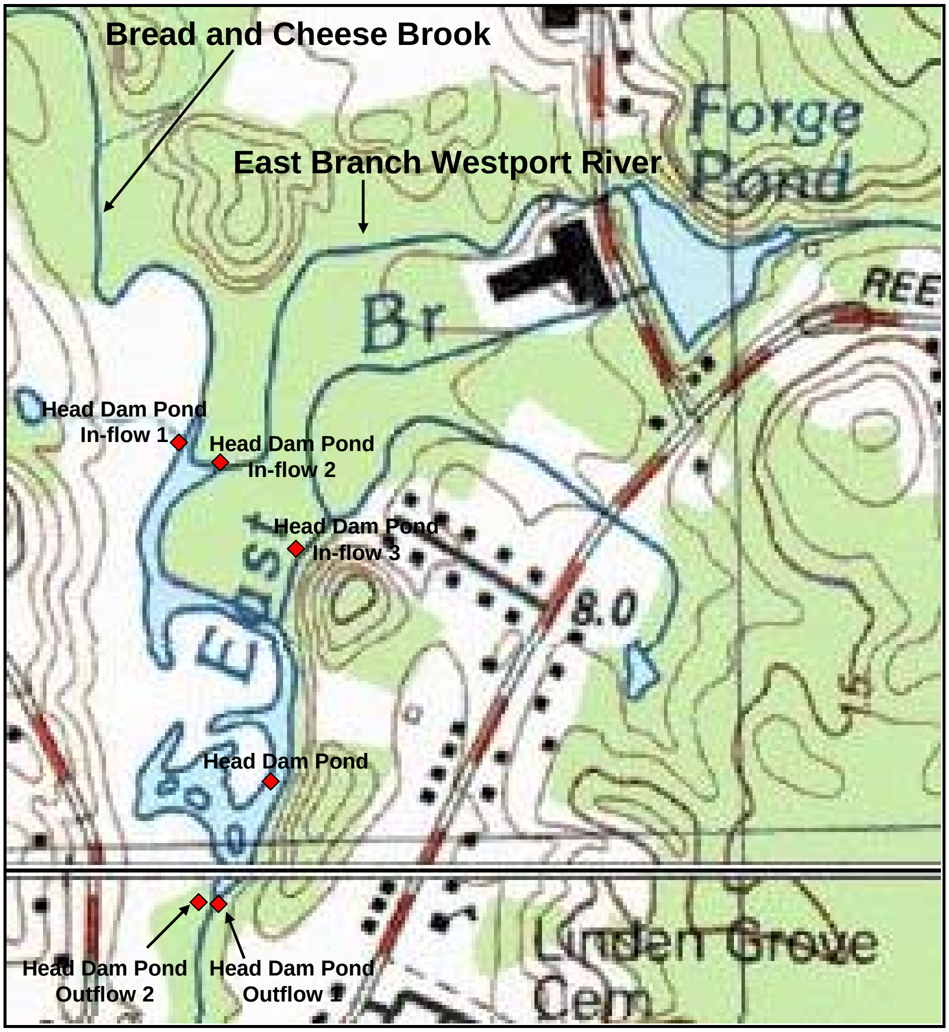


Figure IV-9. Trout Pond is located on the Bread and Cheese Brook up-gradient of Head Dam Pond. Bread and Cheese Brook and the Westport River join in Head Dam Pond which is a flow through pond/wetland located approximately one kilometer up-gradient of the MEP gauging station at Old County Road. Sampling locations in Head Dam Pond. Head Dam Pond Inflow 1 is the discharge from Trout Pond (Bread and Cheese Brook) while Head Dam Pond Inflow 2 and 3 is fundamentally the Westport River as it flows through Forge Pond, is intercepted by Head Dam Pond and ultimately discharges at the head of the estuarine reach of the East Branch of the Westport River embayment.

Table IV-3. Comparison of water flow and nitrogen load discharged by surface waters (freshwater) to the Westport River Estuary. The "Stream" data are from the MEP stream gauging effort. Watershed data are based upon the MEP watershed modeling effort (Section IV.1) and the combination of USGS watershed delineations and watershed delineation information provided by the Buzzards Bay Project. Delineations were reviewed by MEP Technical Team Members and sub-watershed delineations were developed by the MEP (Section III).

Stream Discharge Parameter	East Branch				West Branch		Data Source
	Westport River Discharge[a] (annual)	Westport River Discharge[a] (summer)	Kirby Brook Discharge[a] (annual)	Snell Creek Discharge[a] (annual)	Adamsville Brook Discharge[a] (annual)	Angeline Brook Discharge[a] (annual)	
Total Days of Record	365[b]	82[c]	365[b]	365[b]	365[b]	365[b]	(1)
Flow Characteristics							
Stream Average Discharge (m3/day)	217352	119569	23149	6643	46625	20974	(1)
Contributing Area Average Discharge (m3/day)	216953		19571	8199	51413	18194	(2)
Discharge Stream 2006-07 vs. Long-term Discharge	0.2%		15%	19%	9%	13%	
Nitrogen Characteristics							
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.267	0.423	0.254	0.58	0.22	0.93	(1)
Stream Average Total N Concentration (mg N/L)	0.938	1.361	0.903	1.158	0.980	1.588	(1)
Nitrate + Nitrite as Percent of Total N (%)	28%	31%	28%	50%	22%	59%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	203.83	189.39	20.9	7.69	45.71	33.31	(1)
TN Average Contributing UN-attenuated Load (kg/day)	240.98	--	20.95	8.14	55.93	34.3	(3)
Attenuation of Nitrogen in Pond/Stream (%)	15%	--	0.2%	6%	18%	3%	(4)

(a) Flow and N load to streams discharging to Westport River Estuary includes apportionments of Pond contributing area.
(b) September 1, 2006 to August 31, 2007.
(c) Average daily flow for summer months (June, July August, September) in 2006 and 2007.
(1) MEP gage site data
(2) Calculated from MEP watershed delineations to ponds upgradient of specific gages;
the fractional flow path from each sub-watershed which contribute to the flow in the streams to East and West Branch of the Westport 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-4. Summary of annual volumetric discharge and nitrogen load from the five major surface water discharges to the Westport River Estuarine System (based upon the data presented in Figures IV-10 through IV-14 and Table IV-3.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Westport River (MEP)	**September 1, 2005 to August 31, 2006** **September 1, 2006 to August 31, 2007** **Average 2005 - 2007**	99536641 **79333581** 89435111	27362 **21193** 24278	106692 **74397** 90545
Westport River (CCC)	Based on Watershed Area and Recharge	**79187845**	--	--
Kirby Brook MEP	**September 1, 2006 to August 31, 2007**	**8449207**	**2145**	**7629**
Kirby Brook (CCC)	Based on Watershed Area and Recharge	**7143415**	**--**	**--**
Snell Creek (MEP)	**September 1, 2006 to August 31, 2007**	**2424557**	**1402**	**2808**
Snell Creek (CCC)	Based on Watershed Area and Recharge	**2992635**	**--**	**--**
Adamsville Brook (MEP)	**September 1, 2006 to August 31, 2007**	**17018092**	**3840**	**16683**
Adamsville Brook (CCC)	Based on Watershed Area and Recharge	**18765745**	**--**	**--**
Angeline Brook (MEP)	**September 1, 2006 to August 31, 2007**	**7655524**	**7118**	**12158**
Angeline Brook (CCC)	Based on Watershed Area and Recharge	**6640810**	**--**	**--**

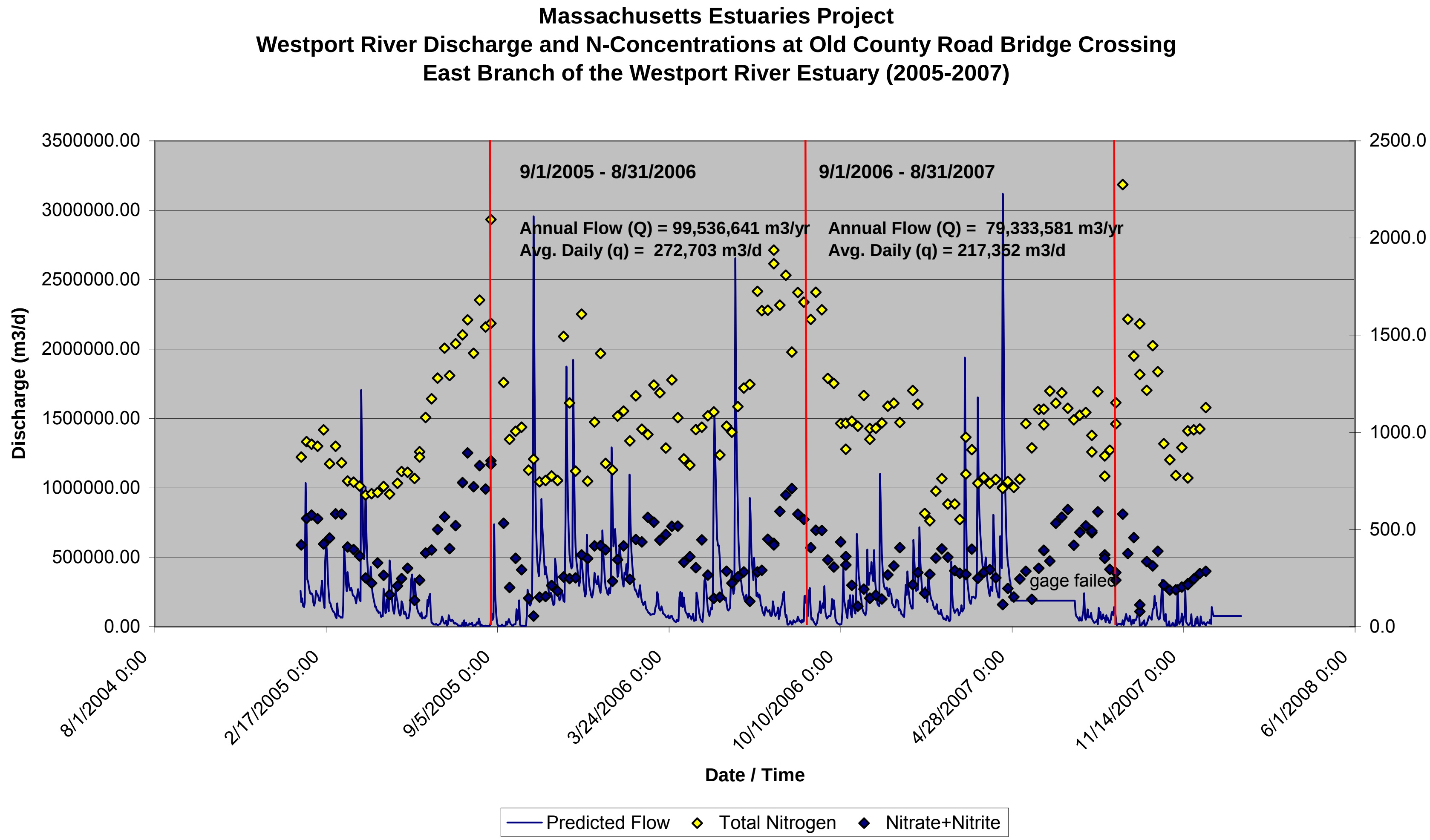


Figure IV-10. Discharge from Westport River (solid blue line), total nitrogen (yellow symbols) and NOx (blue symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of the Westport discharging to the head of the East Branch of the Westport River Estuary (Table IV-3).

A critical aspect of the preliminary data collection undertaken on Trout Pond and Head Dam pond is that based on the limited sampling and flow measurements undertaken in the summer of 2008, neither pond/wetland feature appear to be generating appreciable attenuation of inorganic nitrogen. This may be one reason for the low observed attenuation rate (11%) observed at the old County Road MEP gauge location. That being said and considering the number of flow through features (Trout Pond, Head Dam Pond, Forge Pond, Lake Noquochoke) up-gradient of the Old County Road discharge location for the Westport River, it may be worthwhile to undertake a detailed study of the potential for enhancing natural attenuation in the Westport River through restoration of the various up-gradient surface water features.

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Kirby Brook Discharge to East Branch of the Westport River

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, Kirby Brook, which discharges into the upper portion of the East Branch of the Westport River (above Hixbridge Road), does not have an up-gradient pond from which that brook discharges. Rather, this small stream appears to be groundwater fed and emanates from a wooded and somewhat boggy area up-gradient of Main Road and Route 88. The stream outflow leaving the boggy areas up-gradient of Main Road travels through a wooded upland area just prior to discharging directly into the East Branch. The stream outflow from the bog/wetland and the wooded area up-gradient of the gauge located at the Drift Road crossing of Kirby Brook 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 the biological processes occurring as the water in Kirby Brook flows to the estuary was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the bog/wetland and wooded areas above the gauge site and the measured annual discharge of nitrogen to East Branch of the estuary relative to the gauge, Figure IV-6.

The freshwater flow carried by Kirby Brook to the estuarine waters of the East Branch was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially 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 gauge site. Average measured sample salinity was found to be 0.2 ppt, clearly not tidally influenced. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relation. The Kirby Brook gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow. Calibration of the gauge was checked monthly. The gauge was installed on May 23, 2006 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 September 22, 2007 for a total deployment of 16 months.

Stream flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the upper portion of the East Branch at the top of the Westport River estuarine system and reflective of the biological processes occurring in the stream channel, wetlands and wooded areas contributing to nitrogen attenuation (Figure IV-11 and Table IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey/Buzzards Bay Project/MEP defined watershed delineations to determine long-term average freshwater discharge expected at the Kirby Brook gauge site based on area and average recharge.

The annual freshwater flow record for the Kirby Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/BBP/MEP modeling effort (Table III-1). The measured freshwater discharge from the Kirby Brook at the Drift Road gauge location was 15% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 23,149 m^3/day compared to the long term average flows determined by the watershed modeling effort (19,571 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 Kirby Brook discharging from the sub-watershed indicates that the Brook is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within the Kirby Brook outflow were moderate, 0.903 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 21 kg/day and a measured total annual TN load of 7,629 kg/yr. In the Kirby Brook, nitrate made up well less than half of the total nitrogen pool (28%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge was partially taken up by plants within these different aquatic systems. Given the relatively low levels of remaining nitrate in the stream discharge, the possibility for additional uptake by freshwater systems might be limited in the Kirby Brook sub-watershed.

From the measured nitrogen load discharged by the Kirby Brook to the upper portion of the East Branch and the nitrogen load determined from the watershed based land use analysis, it appears that there is insignificant nitrogen attenuation of upper watershed derived nitrogen during transport to the Kirby Brook and the upper portion of the East Branch of the Westport River estuary. Based upon the slightly lower total nitrogen load (7,629 kg yr^{-1}) discharged from the Kirby Brook at Drift Road compared to that added by the various land-uses to the associated watershed (7,648 kg yr^{-1}), the integrated attenuation in passage through the stream and up-gradient freshwater wetlands prior to discharge to the estuary is 0.2% (i.e. 0.2% 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 up-gradient wooded areas which lack significant up-gradient bogs or wetlands/ponds/lakes capable of attenuating nitrogen. The directly measured nitrogen load from the Kirby Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

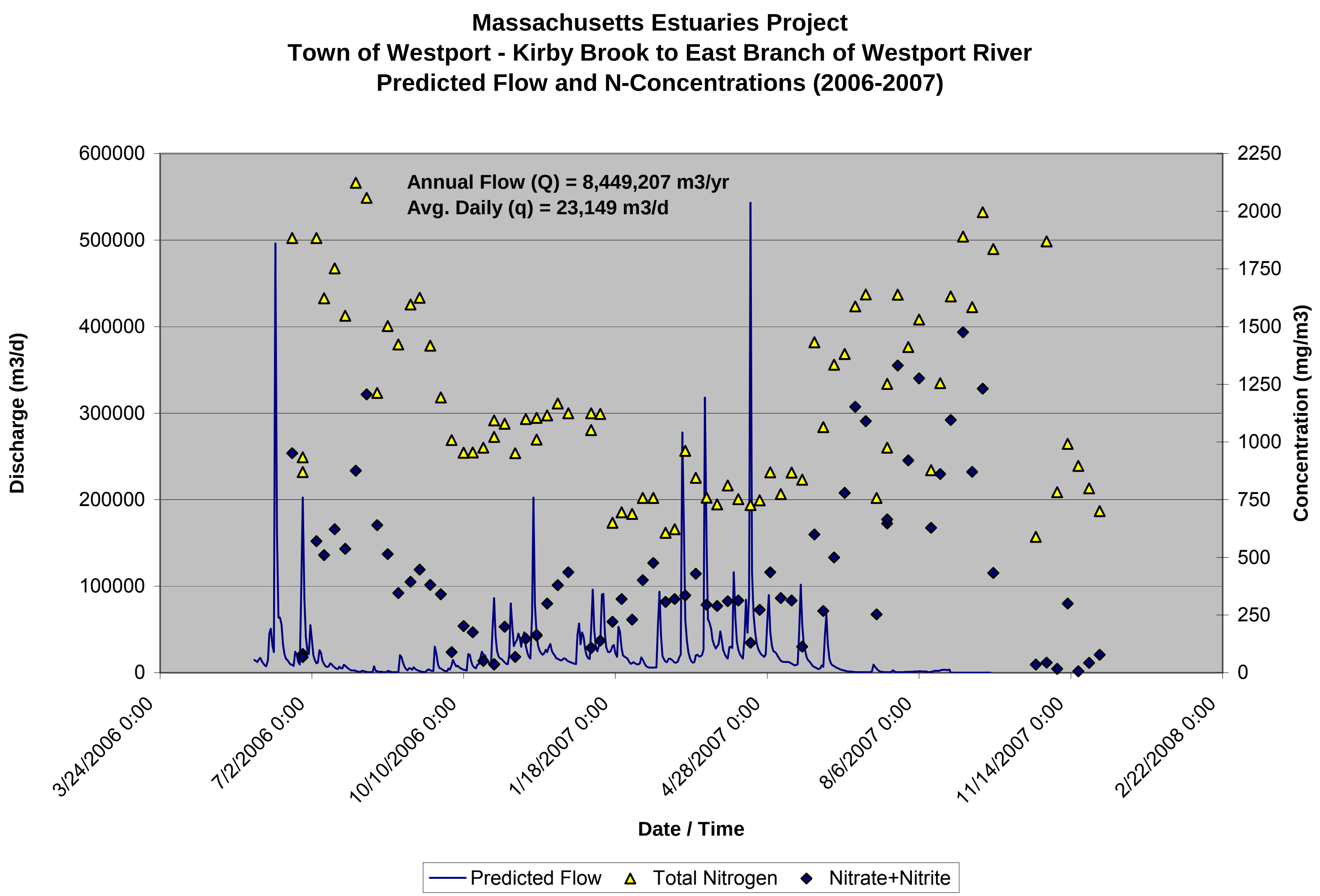


Figure IV-11. Discharge from Kirby Brook (solid blue line), total nitrogen (yellow symbols) and NOx (blue symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of Kirby Brook discharging to the upper portion of the East Branch of the Westport River Estuary (Table IV-3).

IV.2.4 Surface water Discharge and Attenuation of Watershed Nitrogen: Snell Creek Discharge to the East Branch of the Westport River Estuary

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, Snell Creek, which discharges into the upper portion of the East Branch of the Westport River (above Hixbridge Road), does not have an up-gradient pond from which that creek discharges. Rather, this small creek appears to be groundwater fed and emanates from a generally wooded area up-gradient of Main Road and Route 88. The outflow leaving the wooded areas up-gradient of Main Road travels through an upland environment just prior to discharging directly into the East Branch, down gradient of the Kirby Brook discharge point. The creek outflow from the wooded area up-gradient of the gauge located ~200 meters down gradient of the Drift Road crossing of Snell Creek 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 the biological processes occurring as the water in Snell Creek flows to the estuary was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the wooded areas and creek channel above the gauge site and the measured annual discharge of nitrogen to East Branch of the estuary relative to the Snell Creek gauge, Figure IV-6.

The freshwater flow carried by Snell Creek to the estuarine waters of the East Branch was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially 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 gauge site. Average measured sample salinity was found to be 0.2 ppt, clearly not tidally influenced. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relation. The Snell Creek gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow. Calibration of the gauge was checked monthly. The gauge was installed on May 23, 2006 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 September 22, 2007 for a total deployment of 16 months.

Flow in the creek (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the upper portion of the East Branch at the top of the Westport River estuarine system and reflective of the biological processes occurring in the stream channel and wooded areas contributing to nitrogen attenuation (Figure IV-12 and Table IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey/Buzzards Bay Project/MEP defined watershed delineations to determine long-term average freshwater discharge expected at the Snell Creek gauge site based on area and average recharge.

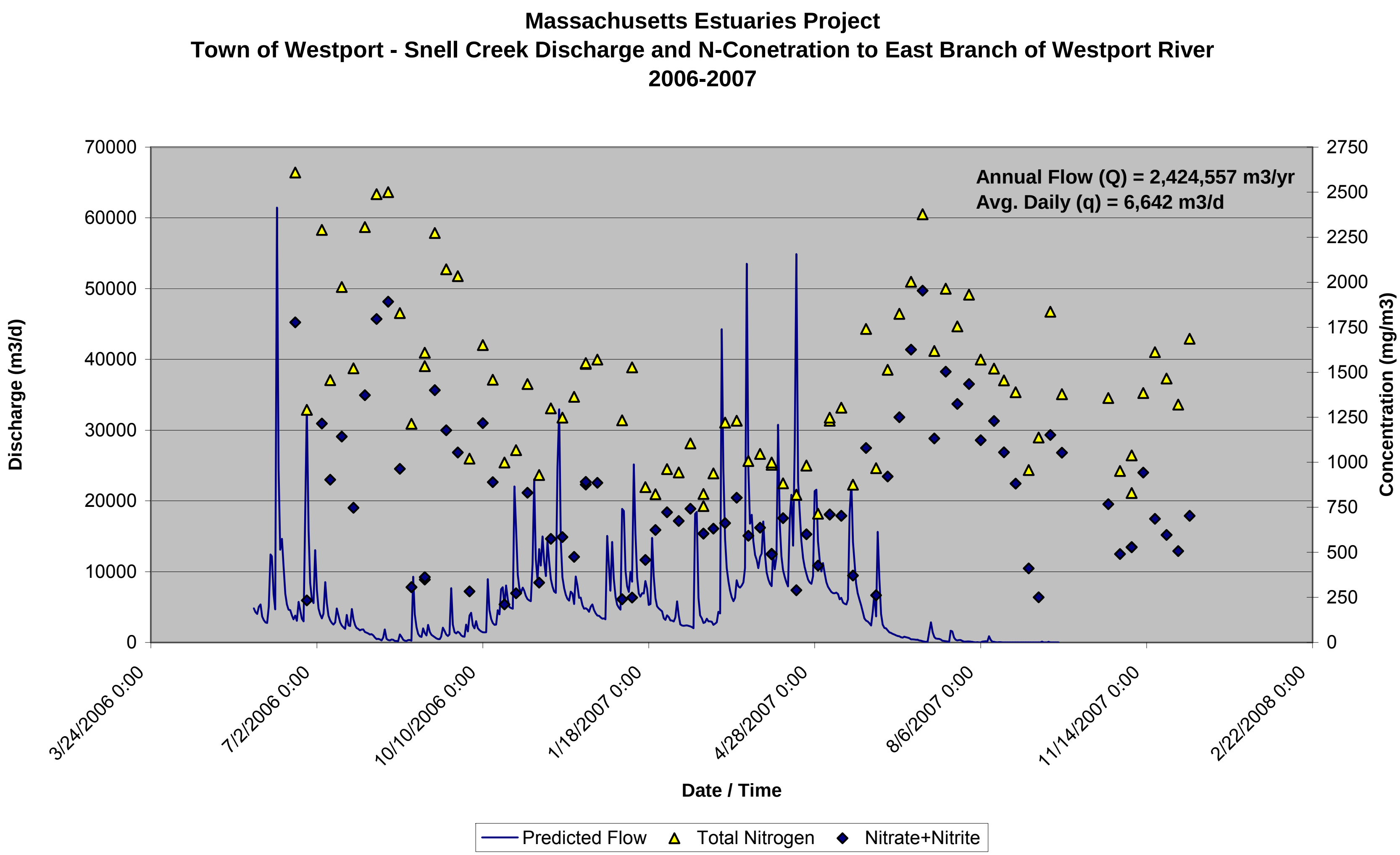


Figure IV-12. Discharge from Snell Creek (solid blue line), total nitrogen (yellow symbols) and NOx (blue symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of Snell Creek discharging to the upper portion of the East Branch of the Westport River Estuary (Table IV-3)

The annual freshwater flow record for Snell Creek as measured by the MEP was compared to the long-term average flows determined by the USGS/BBP/MEP modeling effort (Table III-1). The measured freshwater discharge from Snell Creek at the gauge location was 23% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 6,642 m^3/day compared to the long term average flows determined by the watershed modeling effort (8,199 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 Snell Creek discharging from the sub-watershed indicate that the creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within Snell Creek outflow were moderate to high, 1.16 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 7.69 kg/day and a measured total annual TN load of 2,808 kg/yr. In the Snell Creek outflow, nitrate made up approximately half of the total nitrogen pool (50%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge is only partially taken up by plants within the wooded upland and the creek bed. Given the moderate levels of remaining nitrate in the creek discharge, the possibility for additional uptake by freshwater systems may exist if the Snell Creek sub-watershed was appropriately structured or had potential aquatic features which could be enhanced or restored.

From the measured nitrogen load discharged by Snell Creek to the upper portion of the East Branch and the nitrogen load determined from the watershed based land use analysis, it appears that there is only modest nitrogen attenuation of upper watershed derived nitrogen during transport to Snell Creek and the upper portion of the East Branch of the Westport River estuary. Based upon slightly lower total nitrogen load (2,808 kg yr^{-1}) discharged from Snell Creek compared to that added by the various land-uses to the associated watershed (2,970 kg yr^{-1}), the integrated attenuation in passage through this small creek and up-gradient wooded upland prior to discharge to the estuary is 6% (i.e. 6% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams/creeks evaluated under the MEP is expected given the nature of the up-gradient sub-watershed areas which lack any appreciable wetlands/bogs/ponds capable of attenuating nitrogen. The directly measured nitrogen load from Snell Creek was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.2.5 Surface water Discharge and Attenuation of Watershed Nitrogen: Adamsville Brook Discharge to the West Branch of the Westport River

Similar to most surface water features in the MEP study region that typically emanate from a specific pond, Adamsville Brook discharges into the head of the West Branch of the Westport River directly from a small up-gradient pond (Grays Mill Pond). Adamsville Brook is generally groundwater fed, however, upgradient of Grays Mill Pond, the brook receives water from some very small ponds in the uppermost portions of the sub-watershed. Mostly, Adamsville Brook flows through wooded and somewhat boggy areas up-gradient of Grays Mill Pond (located immediately up-gradient of Adamsville Road). The stream outflow leaving the boggy areas up-gradient of Grays Mill Pond travels also through wooded uplands prior to discharging into Grays Mill Pond. The Adamsville Brook outflow from Grays Mill Pond up-gradient of the gauge which was located on the down gradient side of Adamsville Road serves 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 the biological processes occurring as the water in Adamsville Brook flows to the estuary was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the bog/wetland, wooded areas and pond above the gauge site and the measured annual discharge of nitrogen to the West Branch of the estuary relative to the gauge, Figure IV-6.

The freshwater flow carried by Adamsville Brook to the estuarine waters of the West Branch was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially 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 gauge site. Average measured sample salinity at low tide was found to be 1.6 ppt, indicating a slight tidal influence at the gauge location. As such, a small salinity adjustment was made to the flows in order to determine daily freshwater flows using the MEP developed stage-discharge relation. The Adamsville Brook gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow. Calibration of the gauge was checked monthly. The gauge was installed on May 23, 2006 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 December 4, 2007 for a total deployment of 18 months.

Flow in Adamsville Brook (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the head of the West Branch at the top of the Westport River estuarine system and is reflective of the biological processes occurring in Grays Mill Pond, the channel bed of the brook, wetlands and wooded areas contributing to nitrogen attenuation (Figure IV-13 and Table IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey/Buzzards Bay Project/MEP defined watershed delineations to determine long-term average freshwater discharge expected at the Adamsville Brook gauge site based on area and average recharge.

The annual freshwater flow record for the Adamsville Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/BBP/MEP modeling effort (Table III-1). The measured freshwater discharge from Adamsville Brook down gradient of Adamsville Road and Grays Mill Pond was 10% below/above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 46,625 m^3/day compared to the long term average flows determined by the watershed modeling effort (51,413 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 Adamsville Brook discharging from the sub-watershed indicate that the Brook is capturing the up-gradient recharge (and loads) accurately. Additional confirmation of the MEP determined stream flow for Adamsville Brook was based on a comparison to USGS generated data as reported in a 2008 Report (2008 Index Stream flows in Massachusetts) developed by the Massachusetts Department of Conservation and Recreation Office of Water Resources For the Massachusetts Water Resources Commission. In the report Quartile Flows (Q25, Q50, Q75) were developed for Adamsville Brook slightly up-gradient of the MEP gauge location. The quartile flows were reported as a monthly average based on data obtained from the USGS for the period 1960 to 2004. Based on

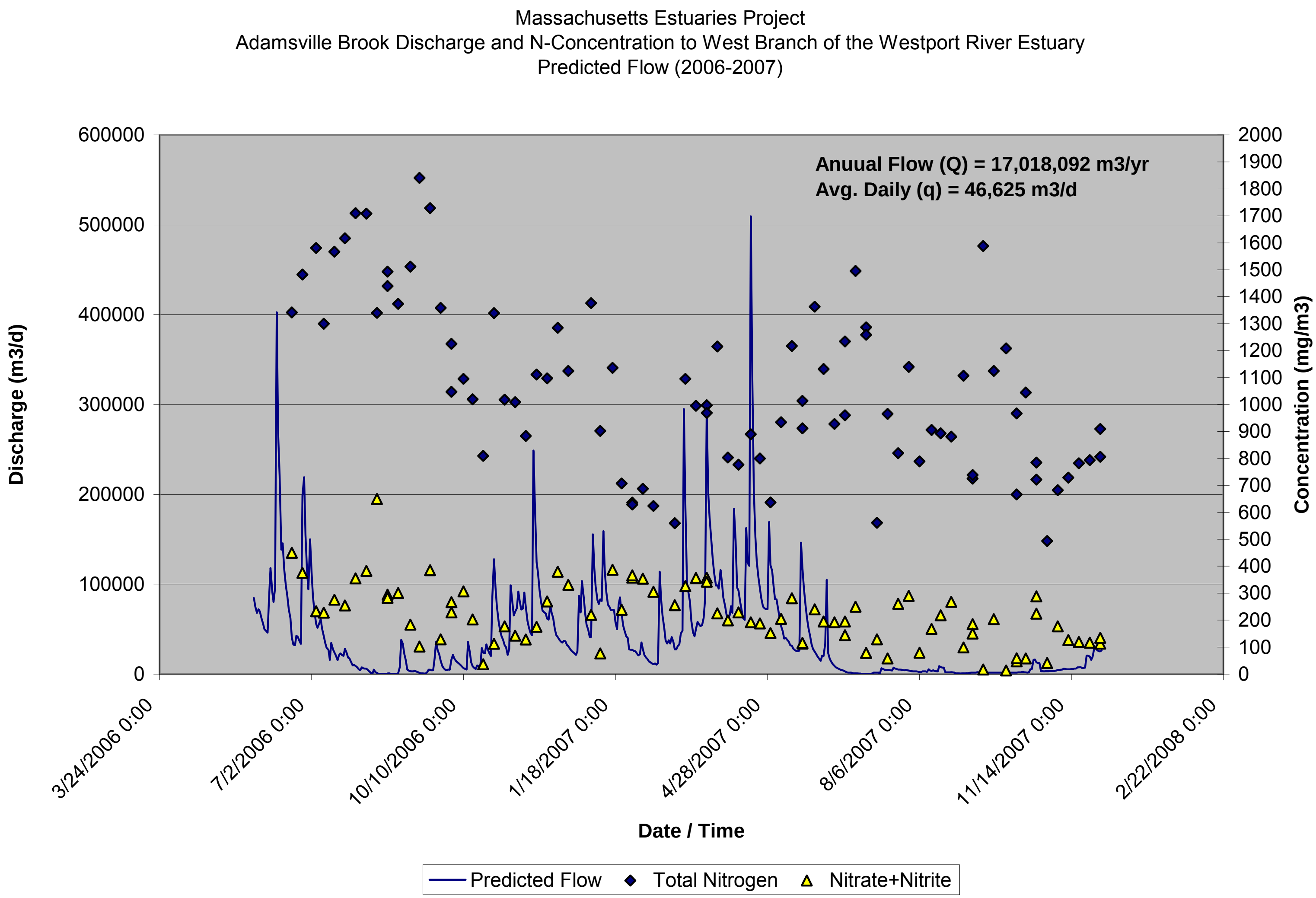


Figure IV-13. Discharge from Adamsville Brook (solid blue line), total nitrogen (blue symbols) and NOx (yellow symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of Adamsville Brook discharging to the head of the West Branch of the Westport River Estuary (Table IV-3).

these data, the Q25 and the Q50 flow for Adamsville Brook was 39,803 m^3/day and 73,350 m^3/day. The MEP determined flow of 46,625 m^3/day is well within the range of flows expected in Adamsville Brook.

Total nitrogen concentrations within the Adamsville Brook outflow were moderate to high, 0.980 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 45.7 kg/day and a measured total annual TN load of 16,683 kg/yr. In Adamsville Brook, nitrate made up well less than half of the total nitrogen pool (22%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas, Grays Mill Pond and the channel bed up-gradient of the gauge was partially taken up by plants within these different aquatic systems. Given the relatively low levels of remaining nitrate in the stream discharge, the possibility for additional uptake by freshwater systems might be limited in the Adamsville Brook sub-watershed.

From the measured nitrogen load discharged by Adamsville Brook to the upper portion of the West Branch 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 Adamsville Brook and the upper portion of the West Branch of the Westport River estuary. Based upon lower total nitrogen load (16,683 kg yr^{-1}) discharged from Adamsville Brook down gradient of Grays Mill Pond at Adamsville Road compared to that added by the various land-uses to the associated watershed (20,414 kg yr^{-1}), the integrated attenuation in passage through the stream and up-gradient Ponds and freshwater wetlands prior to discharge to the estuary is 18% (i.e. 18% 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 up-gradient pond/wetland/wooded areas capable of attenuating nitrogen. The directly measured nitrogen load from Adamsville Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

IV.2.6 Surface water Discharge and Attenuation of Watershed Nitrogen: Angeline Brook Discharge to the West Branch of the Westport River Estuary

Unlike most surface water features in the MEP study region that typically emanate from a specific pond, Angeline Brook, which discharges into the middle portion of the West Branch of the Westport River, does not have any up-gradient ponds from which that Brook discharges. Rather, this small brook appears to be groundwater fed and emanates from a generally wooded area up-gradient of Cornell Road and Adamsville Road. The outflow leaving the wooded areas up-gradient of Cornell Road travels through a sparsely developed upland environment just prior to discharging directly into the West Branch, down gradient of the Adamsville Brook discharge point. The brook outflow from the wooded area up-gradient of the gauge located at the Cornell Road crossing may serve to contribute to some attenuation of nitrogen, particularly if flows pass through any wetlands, and also provides for a direct measurement of the nitrogen attenuation. The combined rate of nitrogen attenuation by the biological processes occurring as the water in Angeline Brook flows to the estuary was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the wooded areas and brook channel above the gauge site and the measured annual discharge of nitrogen to the West Branch of the estuary relative to the Angeline Brook gauge, Figure IV-6.

The freshwater flow carried by Angeline Brook to the estuarine waters of the West Branch was determined using a continuously recording vented calibrated water level gauge. As this surface water system was potentially 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 gauge site. Average measured sample salinity was found to be 0.1 ppt, clearly not tidally influenced. As such, a salinity adjustment was not necessary in order to determine daily flows using the MEP developed stage-discharge relation. The Angeline Brook gauge location was deemed acceptable for making flow measurements and obtaining an estimate of annual freshwater flow. Calibration of the gauge was checked monthly. The gauge was installed on May 23, 2006 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 March 14, 2008 for a total deployment of 22 months.

Flow in the brook (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the gauge site based upon these flow measurements and the measured water levels at the gauge site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the middle portion of the West Branch of the Westport River estuarine system and reflective of the biological processes occurring in the channel bed and any associated wetlands or wooded areas contributing to nitrogen attenuation (Figure IV-14 and Table IV-3 and IV-4). In addition, a water balance was constructed based upon the U.S. Geological Survey/Buzzards Bay Project/MEP defined watershed delineations to determine long-term average freshwater discharge expected at the Angeline Brook gauge site based on area and average recharge.

The annual freshwater flow record for Angeline Brook as measured by the MEP was compared to the long-term average flows determined by the USGS/BBP/MEP modeling effort (Table III-1). The measured freshwater discharge from Angeline Brook at the gauge location was 13% above the long-term average modeled flows. The average daily flow based on the MEP measured flow data for the hydrologic year beginning September 2006 and ending in August 2007 (low flow to low flow) was 20,974 m^3/day compared to the long term average flows determined by the watershed modeling effort (18,194 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 Angeline Brook discharging from the sub-watershed indicate that the creek is capturing the up-gradient recharge (and loads) accurately.

Total nitrogen concentrations within Angeline Brook outflow were high, 1.58 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 33 kg/day and a measured total annual TN load of 12,158 kg/yr. In the Angeline Brook outflow, nitrate made up approximately half of the total nitrogen pool (59%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the wetland areas and stream bed up-gradient of the gauge is only partially taken up by plants within the wooded upland and the channel bed of the brook. Given the relatively high levels of remaining nitrate in the discharge water from the brook, the possibility for additional uptake by freshwater systems may exist if the Angeline Brook sub-watershed was appropriately structured to implement a pond or bog restoration for enhancing natural attenuation in the sub-watershed.

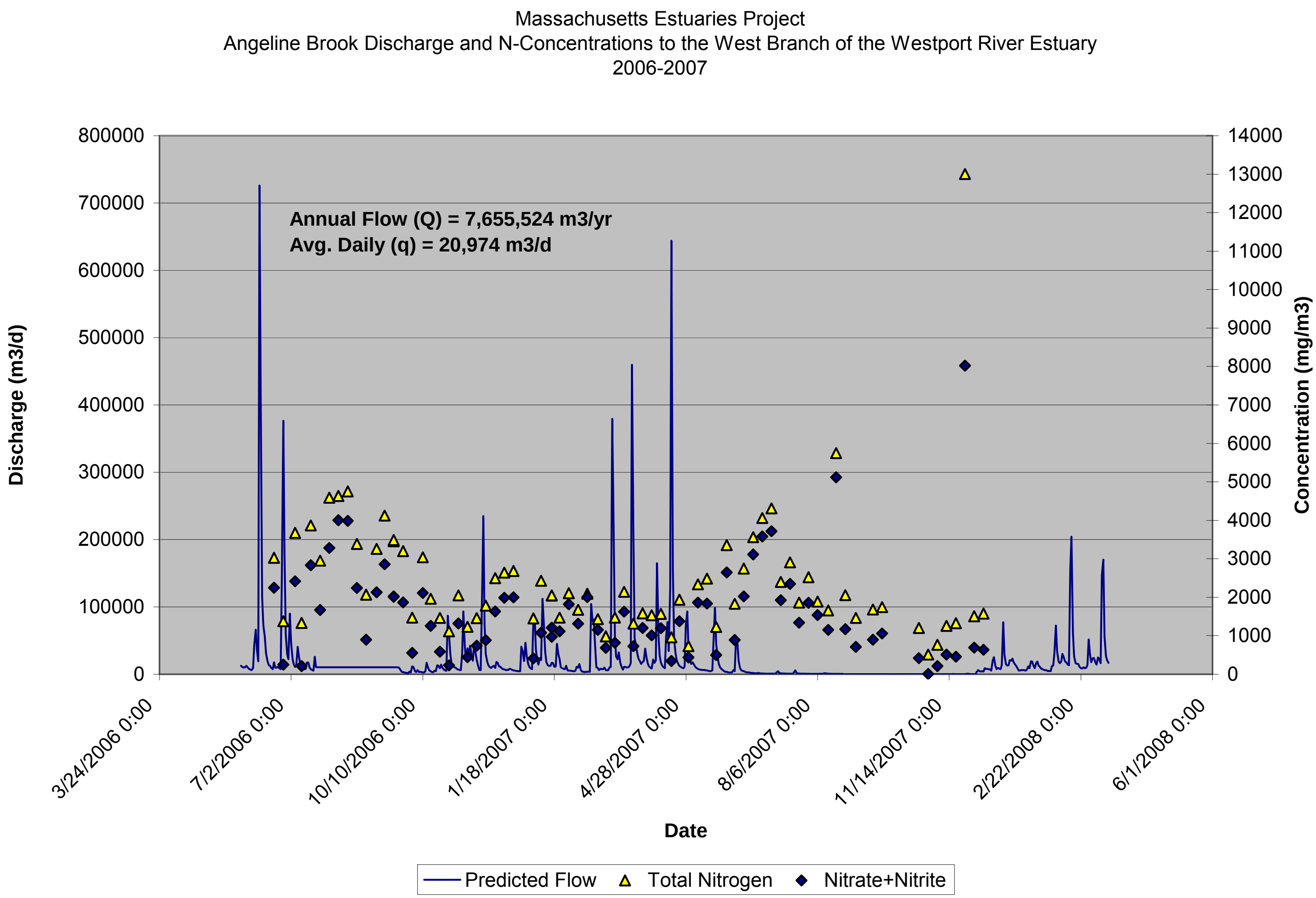


Figure IV-14. Discharge from Angeline Brook (solid blue line), total nitrogen (yellow symbols) and NOx (blue symbols) concentrations for determination of annual volumetric discharge and nitrogen load from the sub-watershed of Angeline Brook discharging to the middle portion of the West Branch of the Westport River Estuary (Table IV-3).

From the measured nitrogen load discharged by Angeline Brook to the middle portion of the West Branch and the nitrogen load determined from the watershed based land use analysis, it appears that there is only modest nitrogen attenuation of upper watershed derived nitrogen during transport to Angeline Brook and the estuarine receiving waters. Based upon lower total nitrogen load (12,158 kg yr^{-1}) discharged from Angeline Brook compared to that added by the various land-uses to the associated watershed (12,518 kg yr^{-1}), the integrated attenuation in passage through this small brook and up-gradient wooded upland prior to discharge to the estuary is 3% (i.e. 3% of nitrogen input to watershed does not reach the estuary). This level of attenuation compared to other streams/creeks/brooks evaluated under the MEP is expected given the nature of the up-gradient sub-watershed areas which lack any appreciable wetlands/bogs/ponds capable of attenuating nitrogen. The directly measured nitrogen load from Angeline Brook was used in the Linked Watershed-Embayment Modeling of water quality (see Section VI, below).

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 Westport 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-Water column 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 Westport River Estuary predominantly in highly bio-available 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 bio-available form nitrate. This nitrate and other bio-available forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the water column for sufficient time to be flushed out to a down gradient larger water body (like 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 bio-available nitrogen is returned to the embayment water column for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that have been investigated by SMAST and the MEP, recycled nitrogen can account for about one-third to one-half of the nitrogen supply to phytoplankton blooms during the warmer summer months. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. In contrast in some systems, with 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 Lewis Bay main basin in the Town of Barnstable, Cape Cod. In contrast, regions of high deposition like Hyannis Inner Harbor, also part of the Lewis Bay system but 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 Westport 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 Westport 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 64 cores from 62 sites in the Westport River system. Cores were spatially distributed throughout both the East and West Branches of the system, inclusive of the Let. All the sediment cores for this system were collected in July-August 2006. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shore side 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-15a,b,c,d) are as follows:

Westport River Benthic Nutrient Regeneration Cores

- WTP-1 1 core (East Branch-upper)
- WTP-2 1 core (East Branch-upper)
- WTP-3 1 core (East Branch-upper)

- WTP-4 1 core (East Branch-upper)
- WTP-5 1 core (East Branch-upper)
- WTP-6 1 core (East Branch-upper)
- WTP-7 1 core (East Branch-upper)
- WTP-8 1 core (East Branch-upper)
- WTP-9 1 core (The Let)
- WTP-10 1 core (The Let)
- WTP-11/12 2 core (The Let)
- WTP-13 1 cores (The Let)
- WTP-14 1 core (The Let)
- WTP-15 1 core (The Let)
- WTP-16 1 core (The Let)
- WTP-17 1 core (East Branch-Lower)

Westport River Benthic Nutrient Regeneration Cores (Continued)

- WTP-18 1 core (East Branch-Lower)
- WTP-19 1 core (East Branch-Lower)
- WTP-20 1 core (East Branch-Lower)
- WTP-21 1 core (East Branch-Lower)
- WTP-22 1 core (East Branch-Lower)
- WTP-23 1 core (East Branch-Lower)
- WTP-24 1 core (East Branch-Lower)
- WTP-25 1 core (East Branch-Lower)
- WTP-26 1 core (Westport Harbor)
- WTP-27 1 core (Westport Harbor)
- WTP-28 1 core (Westport Harbor)
- WTP-29 1 core (Westport Harbor)
- WTP-30 1 core (Westport Harbor)
- WTP-31 1 core (Westport Harbor)
- WTP-32 1 core (Westport Harbor)
- WTP-33 1 core (East Branch-Upper)
- WTP-34 1 core (East Branch-Upper)
- WTP-35 1 core (East Branch-Lower)
- WTP-36 1 core (East Branch-Lower)
- WTP-37 1 core (East Branch-Lower)
- WTP-38 1 core (East Branch-Lower)
- WTP-39 1 core (East Branch-Lower)
- WTP-40 1 core (East Branch-Lower)
- WTP-41 1 core (East Branch-Lower)
- WTP-42 1 core (East Branch-Lower)
- WTP-43 1 core (East Branch-Lower)
- WTP-44 1 core (East Branch-Lower)
- WTP-45 1 core (East Branch-Lower)
- WTP-46 1 core (East Branch-Lower)
- WTP-47 1 core (East Branch-Lower)
- WTP-48 1 core (East Branch-Lower)
- WTP-49 1 core (West Branch)
- WTP-50 1 core (West Branch)

- WTP-51 1 core (West Branch)
- WTP-52 1 core (West Branch)
- WTP-53 1 core (West Branch)
- WTP-54 1 core (West Branch)
- WTP-55 1 core (West Branch)
- WTP-56 1 core (West Branch)
- WTP-57 1 core (West Branch)
- WTP-58 1 core (West Branch)
- WTP-59 1 core (West Branch)
- WTP-60 1 core (West Branch)
- WTP-61 1 core (West Branch)
- WTP-62 1 core (West Branch)
- WTP-63/64 2 cores (West Branch)

Sampling was distributed throughout the primary component basins of the Westport 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 Tripps Marina facility on the shore of Westport Harbor, the cores were transferred to pre-equilibrated temperature baths. The headspace water overlying the sediment was replaced, magnetic stirrers emplaced, and the headspace enclosed. Periodic 60 ml water samples were withdrawn (volume replaced with filtered water), filtered into acid leached polyethylene bottles and held on ice for nutrient analysis. Ammonium (Scheiner 1976) and 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 Westport 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.

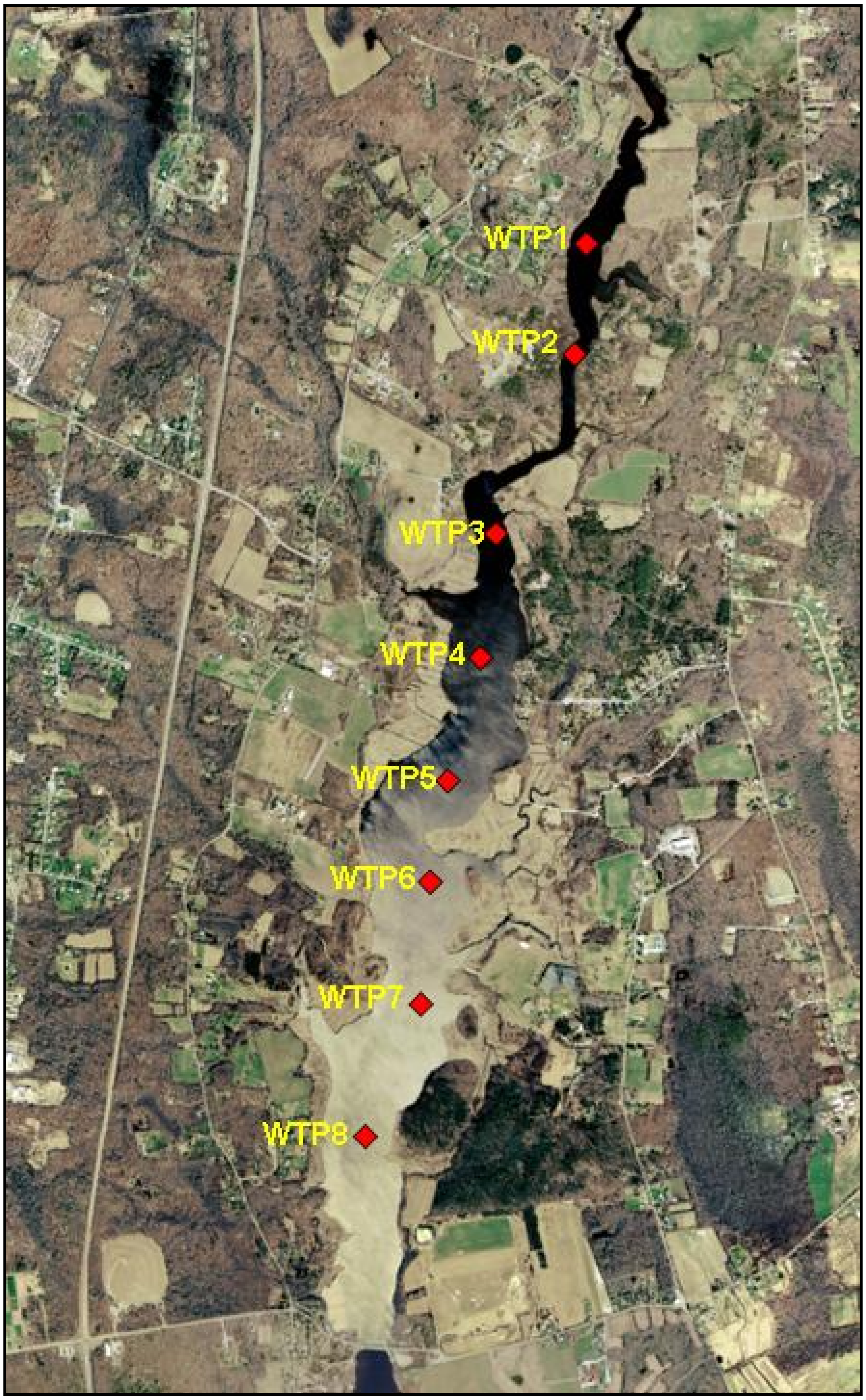


Figure IV-15a. East Branch (uppermost reach above Hix Bridge) of the Westport River embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

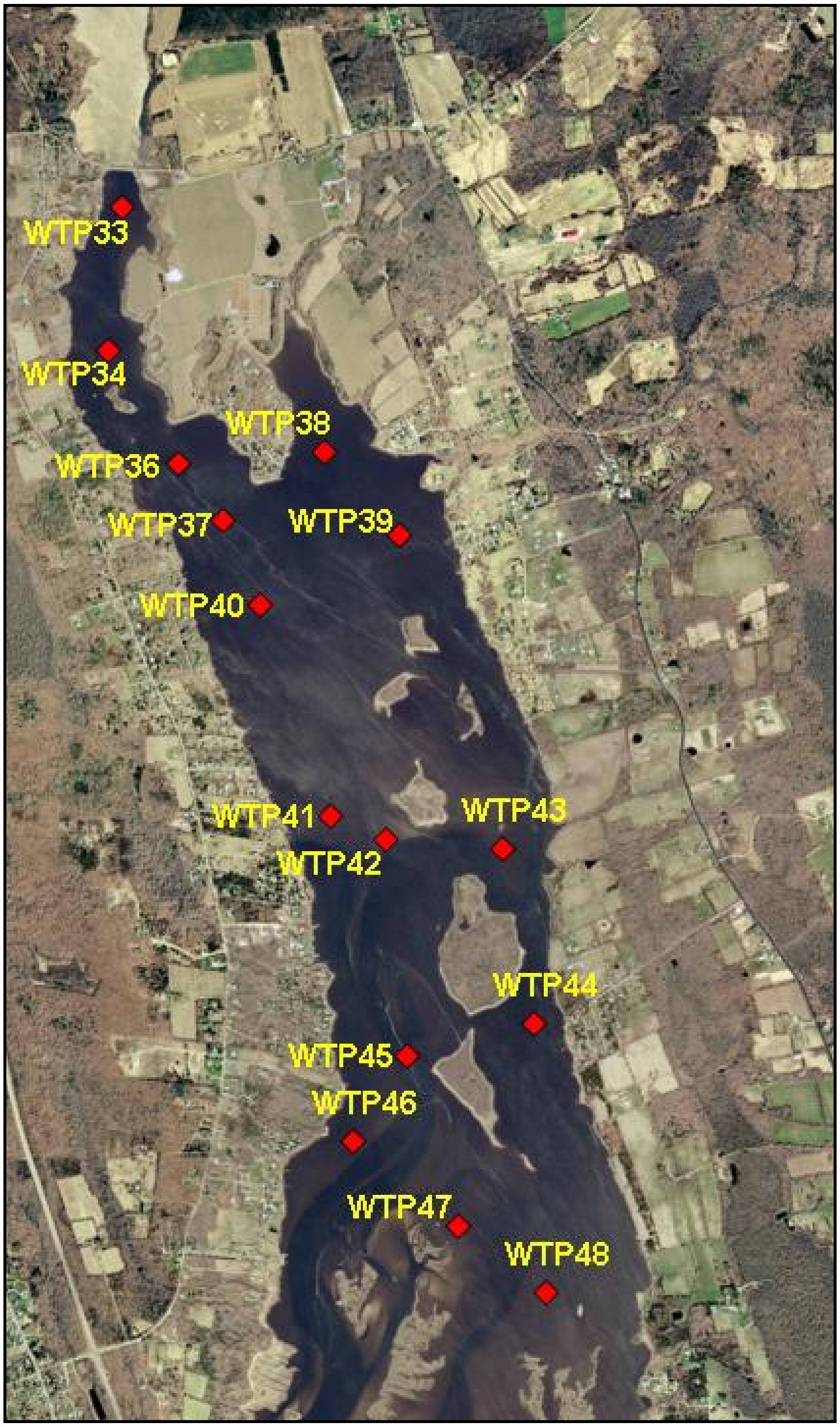


Figure IV-15b. East Branch (middle) of the Westport River embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

Figure IV-15c. Lower portion of the Westport River embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above. The Let is the salt marsh dominated area to the east and the inlet and Westport Harbor is to the west side of the image.

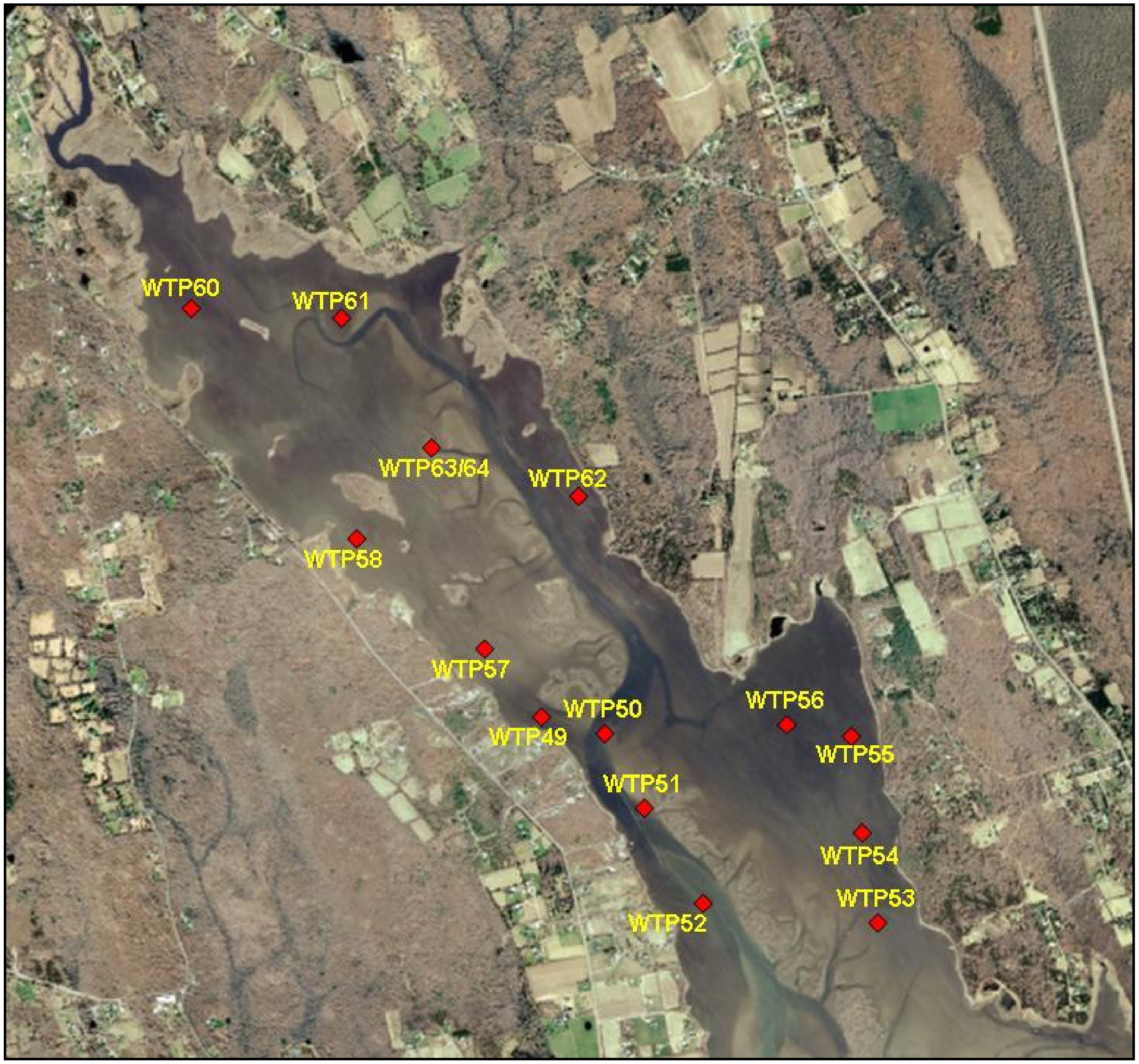


Figure IV-15d. West Branch of the Westport River embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

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-16).

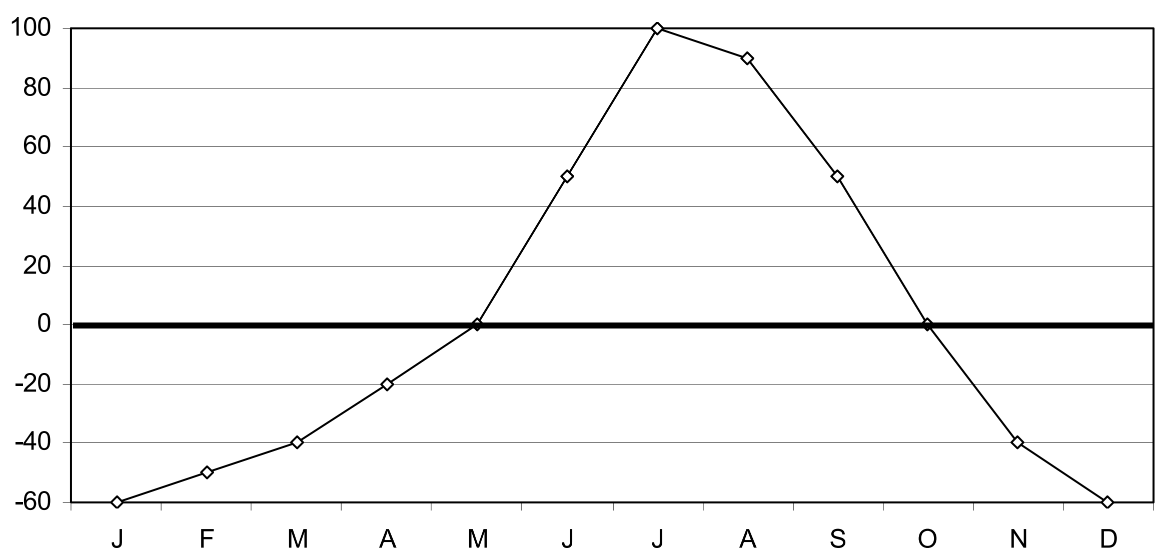


Figure IV-16. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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 major basins of the Westport River Estuary 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. East Branch, West Branch, Westport Harbor, the Let), which comprise the Westport 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 (Section 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 Westport 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. Overall, the wetland dominated tidal reaches tended to have low to moderate net nitrogen release (e.g. Lower East Branch, The Let, upper West Branch) while the upper reaches of the freshwater influenced East Branch and areas within and near Westport Harbor showed a low to moderate summertime net uptake of nitrogen. Moreover, regions of net uptake and release were contiguous and gradually graded into each other as expected based upon sedimentation and hydrodynamics.

The uppermost tidal reach of the East Branch of the Westport River (within the narrow portion of the drowned river valley channel from the Westport River) supports consistent moderate rates of net nitrogen uptake throughout, -25.8 mg N m^{-2} d^{-1}. These rates declined to -5.6 mg N m^{-2} d^{-1} in the mid reach. Similar rates were observed in the lower portion of the West Branch and contiguous regions of Westport Harbor, -8.4 mg N m^{-2} d^{-1} and -15.7 mg N m^{-2} d^{-1}, respectively. In contrast, salt marsh dominated reaches (e.g. upper West Branch and the region of the lower East Branch/Let) showed a low to moderate net nitrogen release, 3.2 - 20.5 mg N m^{-2} d^{-1}. It is important to note that the shifts in net uptake and release throughout the estuary are consistent with the sediment dynamics and observed sediment oxidation and type as well as the depositional characteristics of the environment in these specific areas. For example, as observed in other large estuarine systems, the sediments of the well flushed lower basin of Westport Harbor are sandy with a significant oxidized surface layer and support a moderate net uptake of nitrogen, -15.7 mg N m^{-2} d^{-1}.

The rates of net nitrogen uptake from the sediments throughout the Westport River System are comparable to that of other similarly configured estuaries in southeastern Massachusetts. The upper and mid reach of the East Branch (-25.8 mg N m^{-2} d^{-1}, -5.6 mg N m^{-2} d^{-1}) showed rates similar to the lower tidal river portion of the Bass River (-16.6 mg N m^{-2} d^{-1}) and the Eel River (-29.2 mg N m^{-2} d^{-1}) and Childs River (-45.2 mg N m^{-2} d^{-1}) in the Waquoit Bay System as well as the mid reach of the nearby Slocums River (-6.0 mg N m^{-2} d^{-1}). Similarly the low net nitrogen uptake in the lower reach of the West Branch (-8.4 mg N m^{-2} d^{-1}) had rates consistent with similarly configured basins, such as the lower reach of the Slocums River (-13.2 mg N m^{-2} d^{-1}), lower reach of the Eel River (-27.1 mg N m^{-2} d^{-1}) and lower reach of the Bumps River (-4.5 mg N m^{-2} d^{-1}).

The low to moderate nitrogen uptake in the sandy sediments of Westport Harbor (-15.7 mg N m^{-2} d^{-1}) are also consistent with measurements in other sandy high velocity basins with eelgrass. For example, the lower regions of Nantucket Harbor (-10.9 mg N m^{-2} d^{-1}), Chatham Harbor (-8.8 mg N m^{-2} d^{-1}), the Seapuit River in Three Bays (-37.7 mg N m^{-2} d^{-1}), Quissett Harbor (-12.2 mg N m^{-2} d^{-1}) and the outer basin of West Falmouth Harbor (-11.6 mg N m^{-2} d^{-1}). The salt marsh dominated reaches of the lower East Branch and The Let (8.9 mg N m^{-2} d^{-1}, 20.5 mg N m^{-2} d^{-1}) and the upper West Branch (3.2 mg N m^{-2} d^{-1}) did not reflect salt marsh creeks, which typically show significant nitrogen uptake, but rather salt marsh dominated basins which contain open water throughout the tidal cycle and are surrounded by significant salt marsh plain and typically show low to moderate net nitrogen release. These Westport River Estuary basins compared well with the lower marsh basin in Bass River (20.1 mg N m^{-2} d^{-1}), Polpis Harbor (14.6 to 65.9 mg N m^{-2} d^{-1}), Hamblin Pond (9.3 mg N m^{-2} d^{-1}), upper Halls Creek (31.6 mg N m^{-2} d^{-1}), Back River (6.5 - 22.1 mg N m^{-2} d^{-1}) and the upper reach of Green Pond (12.9 mg N m^{-2} d^{-1}).

Net nitrogen release rates for use in the water quality modeling effort for the main basins of the Westport River Embayment System (Section VI) are presented in Table IV-5. There was a clear spatial pattern of sediment nitrogen flux, with low to moderate net release of nitrogen in salt marsh influenced shallow basins and a relatively consistent low to moderate level of net uptake throughout the other reaches of the East and West Branches. The sandy oxidized sediments of Westport Harbor, which supports significant eelgrass resources showed a moderate net nitrogen uptake, which was nearly identical to other comparable estuarine basins throughout southeastern Massachusetts. The sediments within the Westport River Estuary 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-5. Rates of net nitrogen return from sediments to the overlying waters of component basins comprising the Westport River Estuarine System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Section VI). Measurements represent July -August rates.

	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			Station I.D. *
Location	**Mean**	**S.E.**	**# sites**	**WTP-#**
Westport River Estuarine System				
East Branch - upper **	-25.8	8.1	10	1-8, 33-34
East Branch - mid	-5.6	8.8	14	35-48
East Branch - lower	8.9	9.8	8	17-25
West Branch - upper	3.2	8.4	9	49, 57-64
West Branch- lower	-8.4	9.3	7	50-56
The Let	20.5	8.8	8	9-16
Westport Harbor	-15.7	8.3	6	27-32

* Station numbers refer to Figure IV-14a,b,c,d.
** Collected in the upper estuarine reach within the drowned channel of the Westport River, Fig. IV-14b

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

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

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

Estuarine water quality is dependent upon nutrient and pollutant loading and the processes that help flush nutrients and pollutants from the estuary (e.g., tides and biological processes). Relatively low nutrient and pollutant loading and efficient tidal flushing are indicators of high water quality. The ability of an estuary to flush nutrients and pollutants is proportional to the volume of water exchanged with a high quality water body (i.e. Buzzard’s Bay). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For the Westport River system, the most important parameters are the tide range along with the shape, length and depth of the estuary.

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

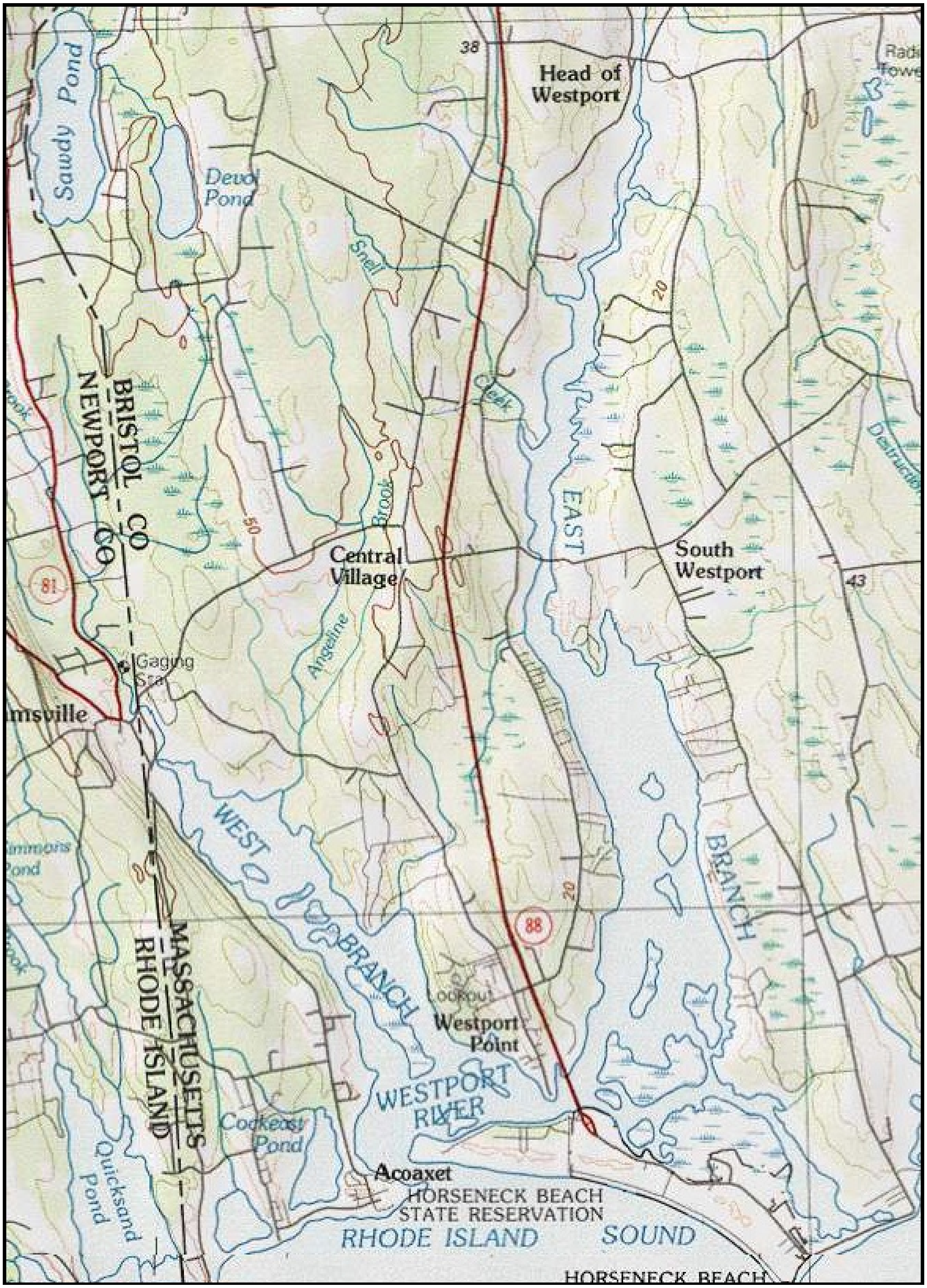


Figure V-1. Map of the Westport River estuary system (from United States Geological Survey topographic maps).

The Westport River system (Figure V-1) is a tidally dominated embayment with a southern opening to Buzzard's Bay. The Westport River system divides into two major sub-embayments, the East and West Branches. The East Branch is located east of the Route 88 Bridge, with the West Branch on the west side of the bridge. The East Branch also includes a sub-embayment known as The Let, located at the southeastern corner of the East Branch and connected by a mile long channel with a width ranging from roughly 400 feet to just over 800 feet. The East Branch stretches up to the Old County Road Bridge and continues further north; however this report uses the Old County Road Bridge as the northern boundary for the model because this was found to be the upper extent of the estuary based upon salinity and tidal influence. This section of the East Branch is roughly 7.2 miles long (north to south) and ranges from 5200 feet wide near the southern end, narrowing abruptly just south of the Hix Bridge (located roughly in the middle of the branch) to just over 1000 feet, and ending at roughly 38 feet wide at the Old County Road Bridge. The inlet to the Westport River Estuarine system is located on the West Branch side of the Route 88 Bridge, in the southwest corner of the West Branch. The 1400 foot wide inlet starts with the tide flowing in a northwesterly direction, but bends back on itself having the flood tide enter the system in a northeasterly direction. The West Branch stretches roughly 3.5 miles (southeast to northwest) and is predominately wide (around 3200 feet) for most of this length. It narrows sharply about 2.9 miles upstream to a width of 1200 feet, and continues narrowing to the uppermost extent of the model, ending at a width of about 30 feet. The entire system is roughly 3590 acres in size, with a maximum depth of roughly 20 feet around Hix Bridge and an average depth of 4.4 feet.

Since the water elevation difference between Buzzard's Bay and Westport River is the primary driving force for tidal exchange of this estuarine system, the local tide range limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) along the length of Westport River is minimal, indicating a system that flushes with minor energy loss. The exception to this trend occurs as a result of high stream water inflow and is limited to the most northern region of the East Branch. Any issues with water quality, therefore, would likely be due to other factors including nutrient loading conditions from the system's watersheds, the water quality of rivers and creeks emptying into the estuary, and the tide range in Buzzard's Bay.

Circulation in the Westport River estuarine system was simulated using the RMA-2 numerical hydrodynamic model. To calibrate the model, field measurements of water elevations and bathymetry were required. Tide data were acquired for the system at a gauge station installed in Buzzard's Bay and at six stations located within the estuary (Figure V-2). All temperature-depth recorders (TDRs or tide gauges) were installed for a 76-day period to measure tidal variations through one spring-neap tidal cycle. In this manner, attenuation of the tidal signal as it propagates through the harbor and each branch was evaluated accurately.

V.2 FIELD DATA COLLECTION AND ANALYSIS

Accurate modeling of system hydrodynamics is dependent upon measured conditions within the estuary for two important reasons:

- To define accurately the system geometry and boundary conditions for the numerical model
- To provide 'real' observations of hydrodynamic behavior to calibrate and verify the model results

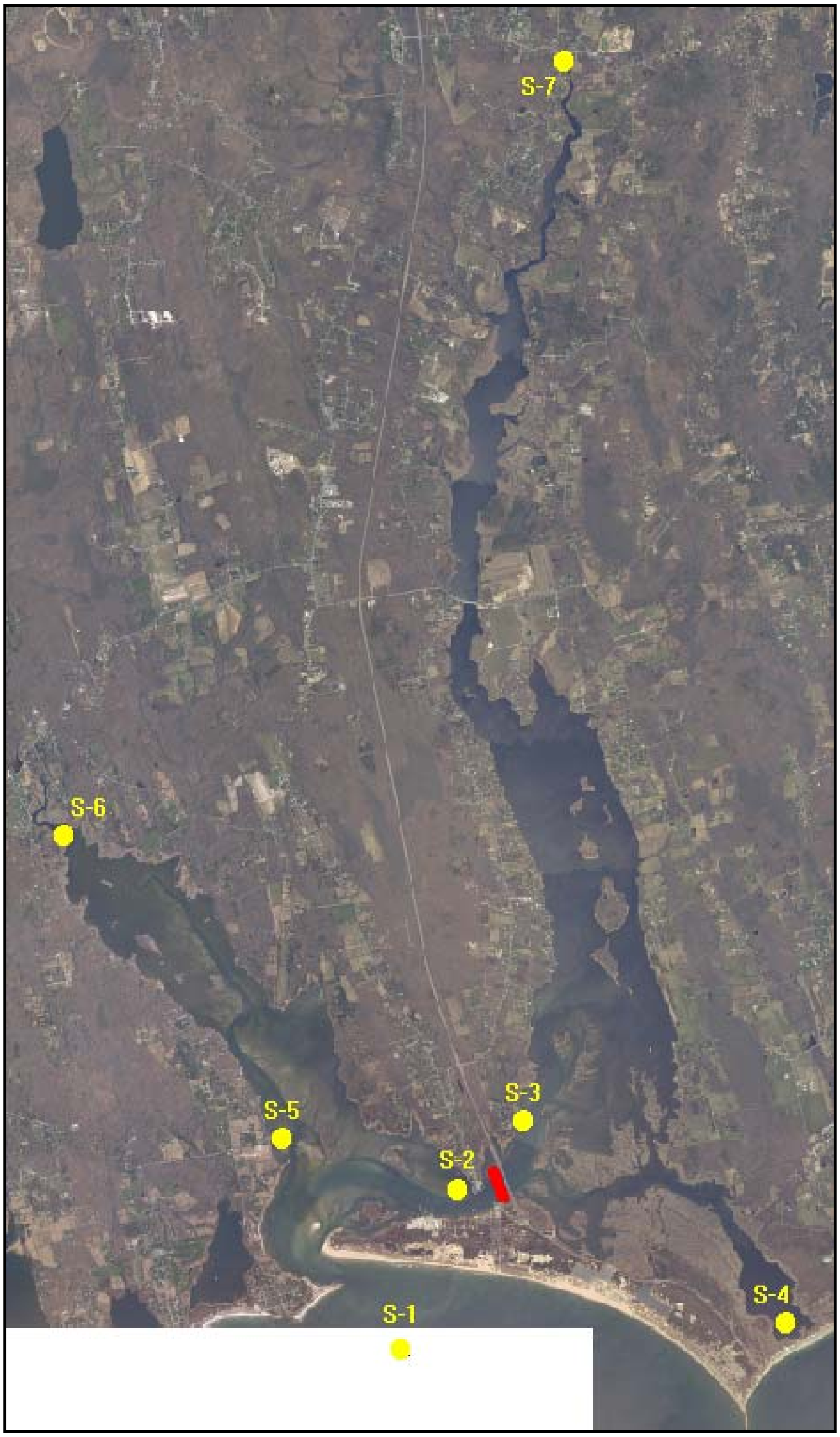


Figure V-2. Aerial photograph of the study region identifying locations of the tide gauges used to measure water level variations throughout the system. The seven (7) gauges were deployed for a 76-day period between May 3, and July 18, 2007. Each yellow dot represents the approximate locations of the tide gauges: (S-1) represents the Offshore gauge, (S-2) the West Harbor gauge, (S-3) the East Harbor gauge, (S-4) the gauge at The Let, (S-5) the Lower West Branch, (S-6) the Upper West Branch, and (S-7) the Upper East Branch. The ADCP transect line is shown in red.

System geometry is defined by the shoreline of the system, including all coves, creeks, and marshes, as well as accompanying depth (or bathymetric) information. The three-dimensional surface of the estuary is mapped as accurately as possible, since the resulting hydrodynamic behavior is strongly dependent upon features such as channel widths and depths, sills, marsh elevations, and inter-tidal flats. Hence, this study included an effort to collect bathymetric information in the field.

Boundary conditions for the numerical model consist of variations of water surface elevations measured in Buzzard's Bay. These variations result principally from tides, and provide the dominant hydraulic forcing for the system, and are the principal forcing function applied to the model. Additional pressure sensors were installed at selected interior locations to measure variations of water surface elevation along the length of the system (gauge locations are shown in Figure V-2). These measurements were used to calibrate and verify the model results, and to assure that the dynamic of the physical system were properly simulated.

V.2.1 Bathymetry

Bathymetry data (i.e., depth measurements) for the hydrodynamic model of the Westport River system was assembled from the three recent hydrographic surveys performed specifically for this study. NOAA Coastal Services LIDAR survey data and National Ocean Service (NOS) Hydrographic Data, where available, were used for areas of Westport River that were not covered by these more recent surveys.

The three hydrographic surveys were conducted June 15th, 19th, and 26th 2007 and collected bathymetry in Westport River and narrow portions of the estuary. Survey transects in all three cases were densest in the vicinity of the inlets, where the greatest variability in bottom bathymetry was expected. Bathymetry in the inlet is important from the standpoint that it has the most influence on tidal circulation in and out of the estuary. The first survey was conducted from a shallow draft outboard boat with a precision fathometer installed (with a depth resolution of approximately 0.1 foot), coupled together with a differential GPS to provide position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder (fathometer) and GPS were logged to a laptop computer, which integrated the data to produce a single data set consisting of water depth as a function of geographic position (latitude/longitude).

The raw measured water depths were merged with water surface elevation measurements to determine bathymetric elevations relative to the North American Vertical Datum of 1988 (NAVD88) vertical datum in feet. Once rectified, the finished, processed data were archived as 'xyz' files containing x-y horizontal position (in Massachusetts Mainland State Plan 1983 coordinates) and vertical elevation of the bottom (z). These xyz files were then interpolated into the finite element mesh used for the hydrodynamic simulations. The final processed bathymetric data from the survey are presented in Figure V-3.

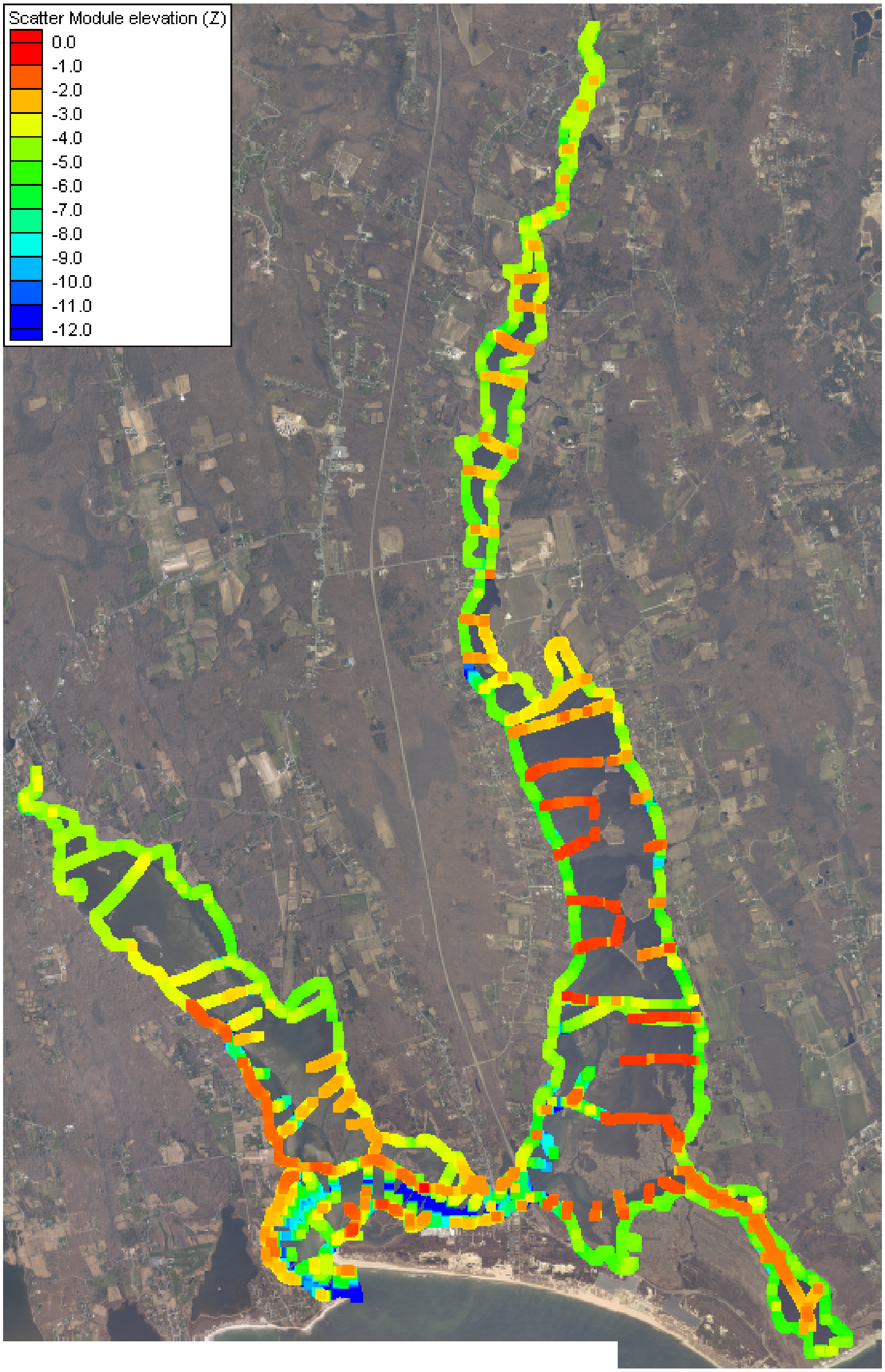


Figure V-3. Bathymetric data interpolated to the finite element mesh of hydrodynamic model.

V.2.2 Tide Data Collection and Analysis

Variations in water surface elevation were measured at stations in six locations in the Westport River estuary and at a station in Buzzard's Bay. The station location in Buzzard's Bay is located just offshore, south of the inlet (S-1). Stations within the Westport River estuary system were located in Westport Harbor on the western bank (S-2), on the eastern bank of Westport Harbor (S-3), at the southern edge of The Let (S-4), on the western bank of the West Branch (S-5), at the northern extent of the West Branch (S-6) and at the northern extent of the East Branch (S-7). TDRs were deployed at each gauge station from the beginning of May 3rd through July 18th 2007. The duration of the TDR deployment allowed time to conduct the bathymetric surveys, as well as sufficient data to perform a thorough analysis of the tides in the system.

The tide records from Westport River were corrected for atmospheric pressure variations and then rectified to the NAVD88 vertical datum. Atmospheric pressure data, available in one-hour intervals from the NDBC Buzzards Bay C-MAN platform, were used to pressure correct the raw tide data. Final processed tide data from the stations used for this study are presented in Figure V-4, for the complete 76-day period of the TDR deployment.

Tide records longer than 29.5 days are necessary for a complete evaluation of tidal dynamics within the estuarine system. Although a one-month record likely does not include extreme high or low tides, it does provide an accurate basis for typical tidal conditions governed by both lunar and solar motion. For numerical modeling of hydrodynamics, the typical tide conditions associated with a one-month record are appropriate for driving tidal flows within the estuarine system.

The loss of amplitude together with increasing phase delay with increasing distance from the inlet is described as tidal attenuation. Tide attenuation can be a useful indicator of flushing efficiency in an estuary. Attenuation of the tidal signal is caused by the geomorphology of the near-shore region, where channel restrictions (e.g., bridge abutments) and also the depth of an estuary are the primary factors which influence tidal damping in estuaries. A visual comparison of the seven stations throughout the Westport River estuary system (Figure V-5), demonstrates a reduction in the tidal efficacy of the system.

To better quantify the changes to the tide from the inlet to inside the system, the standard tide datums were computed from the 76-day records. These datums are presented in Table V-1. The Mean Higher High Water (MHHW) and Mean Lower Low Water (MLLW) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW. The tides in Buzzard's Bay 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

For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data were available; however, these datums still provide a useful comparison of tidal dynamics within the system. From the computed datums, it is further apparent that there is damping occurring in the Westport River Estuary System.

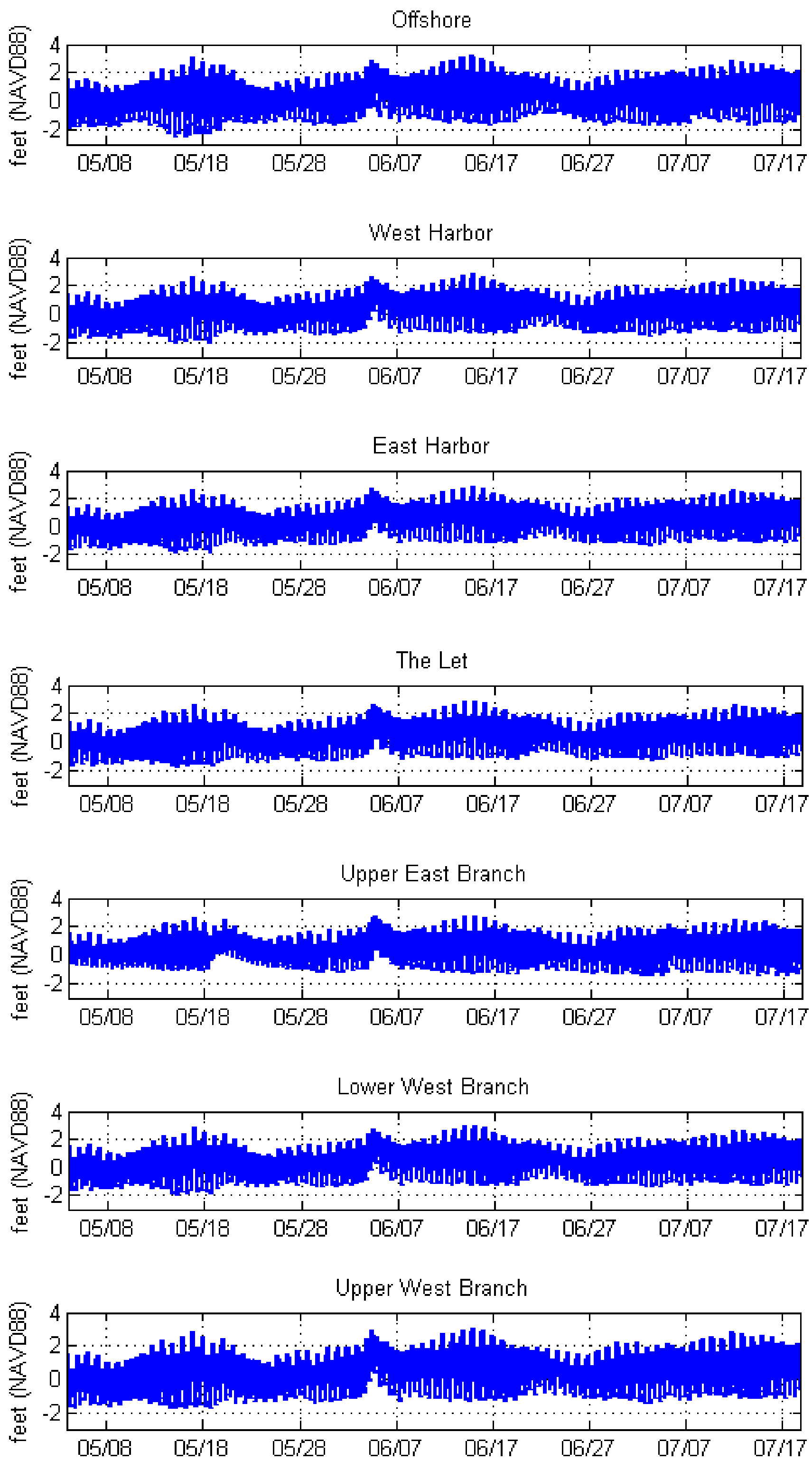


Figure V-4. Water elevation variations as measured at the seven locations of the Westport River system, from May 04th to July 18th 2007.

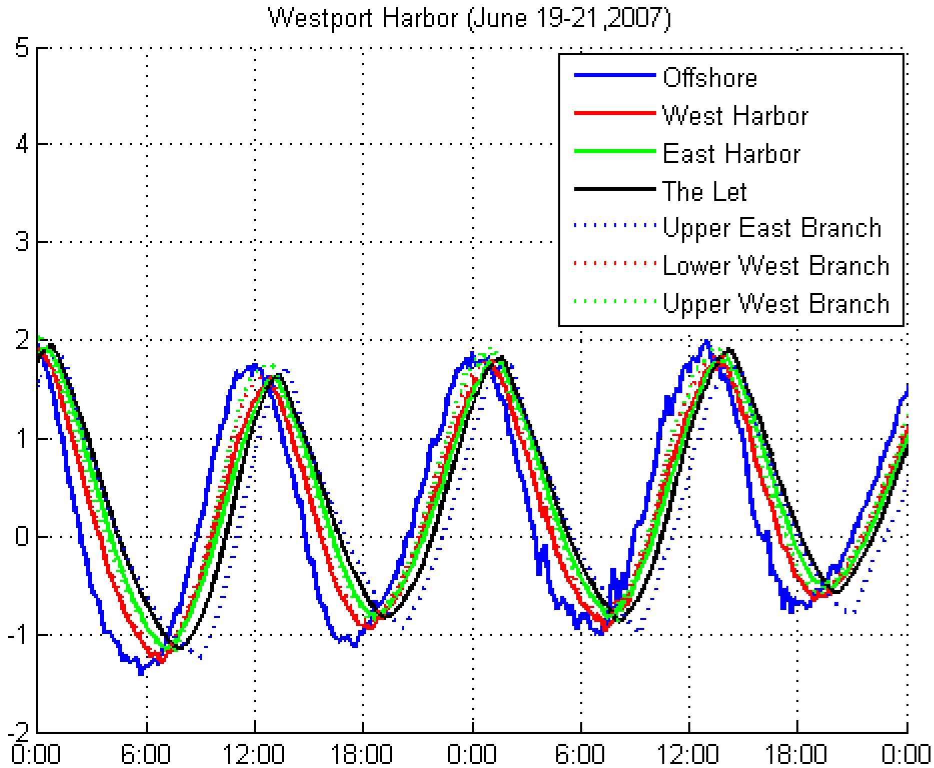


Figure V-5 Plot showing two tide cycles tides at seven stations in the Westport River system plotted together. Demonstrated in this plot is the phase delay and amplitude reduction as the tidal signal progresses through the estuarine system.

Table V-1. Tide datums computed from records collected in the Westport River Estuarine system May 3 - July 18, 2007. Datum elevations are given relative to NAVD88.

Tide Datum	Offshore	West Harbor	East Harbor	The Let	Upper East Branch	Lower West Branch	Upper West Branch
Maximum Tide	3.283	2.828	2.836	2.858	2.776	2.977	3.036
MHHW	2.081	1.866	1.910	1.903	1.903	1.935	2.035
MHW	1.822	1.603	1.645	1.636	1.643	1.671	1.761
MTL	0.221	0.215	0.292	0.282	0.351	0.263	0.317
MLW	-1.379	-1.174	-1.060	-1.072	-0.940	-1.145	-1.126
MLLW	-1.488	-1.264	-1.150	-1.147	-1.008	-1.236	-1.218
Minimum Tide	-2.496	-2.044	-1.850	-1.773	-1.455	-2.012	-1.748

A more thorough harmonic analysis was also performed on the time series data from each gauge station in an effort to separate the various component signals which make up the observed tide. The analysis allows an understanding of the relative contribution that diverse physical processes (i.e. tides, winds, etc.) have on water level variations within the estuary. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 tidal constituents, with periods between 4 hours and 2 weeks, result from this procedure. The observed tide is therefore the sum of an astronomical tide component and a residual atmospheric component. The astronomical tide in turn is 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.

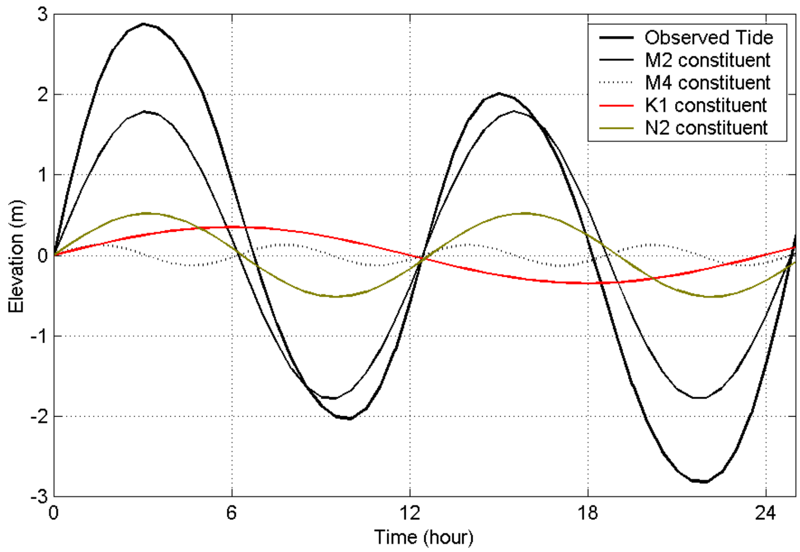


Figure V-6. Example of observed astronomical tide as the sum of its primary constituents. In this example the observed tide signal is the sum of individual constituents (M2, M4, K1, N2), with varying amplitude and frequency.

Table V-2 presents the amplitudes of seven significant tidal constituents. The M_2, or the familiar twice-a-day lunar, semi-diurnal, tide is the strongest contributor to the signal throughout the system. The M2 amplitude drops as the signal propagates around the inlet and into the system, showing some reduction in efficiency. The range of the M_2 tide is twice the amplitude, or about 2.92 feet offshore and 2.21-2.61 feet inside the system. The diurnal (once daily) tide constituents, K_1 (solar), O_1 (lunar), and S_2 (principal solar semidiurnal) possess amplitudes of approximately 0.27-0.29 feet, 0.17-0.19 feet, and 0.13-23 feet respectively. These constituents account for the lack of semi-diurnal variance one high/low tide to the next, as seen in Figure V-5. The M_4 tide, a higher frequency harmonic of the M_2 lunar tide (twice the frequency of the M_2), results from frictional dissipation of the M_2 tide in shallow water.

Table V-2. Tidal Constituents for the Westport River System. Data collected May 3 - July 18, 2007.

AMPLITUDE (feet)							
	M2	M4	M6	K1	S2	N2	O1
Period (hours)	12.42	6.21	4.14	23.93	12.00	12.66	25.82
Offshore	1.461	0.140	0.019	0.276	0.228	0.335	0.184
West Harbor	1.263	0.103	0.026	0.271	0.167	0.266	0.177
East Harbor	1.217	0.082	0.033	0.277	0.154	0.255	0.180
The Let	1.212	0.129	0.027	0.273	0.159	0.248	0.180
South West Branch	1.294	0.113	0.017	0.279	0.177	0.280	0.186
North West Branch	1.306	0.147	0.014	0.294	0.170	0.277	0.185
East Branch	1.107	0.210	0.046	0.266	0.133	0.227	0.170

Table V-3 presents the phase delay (in other words, the travel time required for the tidal wave to propagate throughout the system) of the M_2 tide at all tide gauge locations inside the system. The greatest delay occurs between the offshore gauge station and the East Branch gauge stations. The largest changes in phase delay occur between the East Harbor gauge station and East Branch. This suggests some amount of hydraulic inefficiency being caused by the constriction of the northern extent of the East Branch, as well as the influence of freshwater streamflow to East Branch.

Table V-3. M_2 Tidal Attenuation, Westport River Estuary System, May 3 - July 18, 2007 (Delay in minutes relative to the offshore station).

Location	Delay (minutes)
West Harbor	52
East Harbor	78
The Let	102
South West Branch	47
North West Branch	60
East Branch	127

The tide data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. Non-tidal 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 two river systems 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 are relative to hydrodynamic circulation within the estuary. Figure V-7 shows the comparison of the measured tide from Offshore, with the predicted tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-4 shows that the percentage contribution of tidal energy was the predominate driving force of the observed tidal signal in Westport River. The analysis also shows that tides are responsible for more than 89% of the water level changes, while the remaining 7% to 11% was the result of a combination of freshwater discharge from the five streams and atmospheric forcing, due to winds, or barometric pressure gradients acting upon the collective water surface of Buzzard's Bay and Westport River. The total energy content of the tide signal should carry over from one embayment to the next unless tidal flow is inhibited. The minor energy loss shown here demonstrates the conservation of the tidal energy in the estuarine system.

The results from Table V-4 indicate that hydrodynamic circulation throughout the Westport River Estuarine System is primarily dependent upon tidal processes. While wind and other non-tidal effects can be a less significant portion of the total variance, the residual signal should not be ignored. Therefore, for the hydrodynamic modeling effort described below, the actual tide signal from Buzzard's Bay was used to force the model so that the effects of non-tidal energy are included in the modeling analysis.

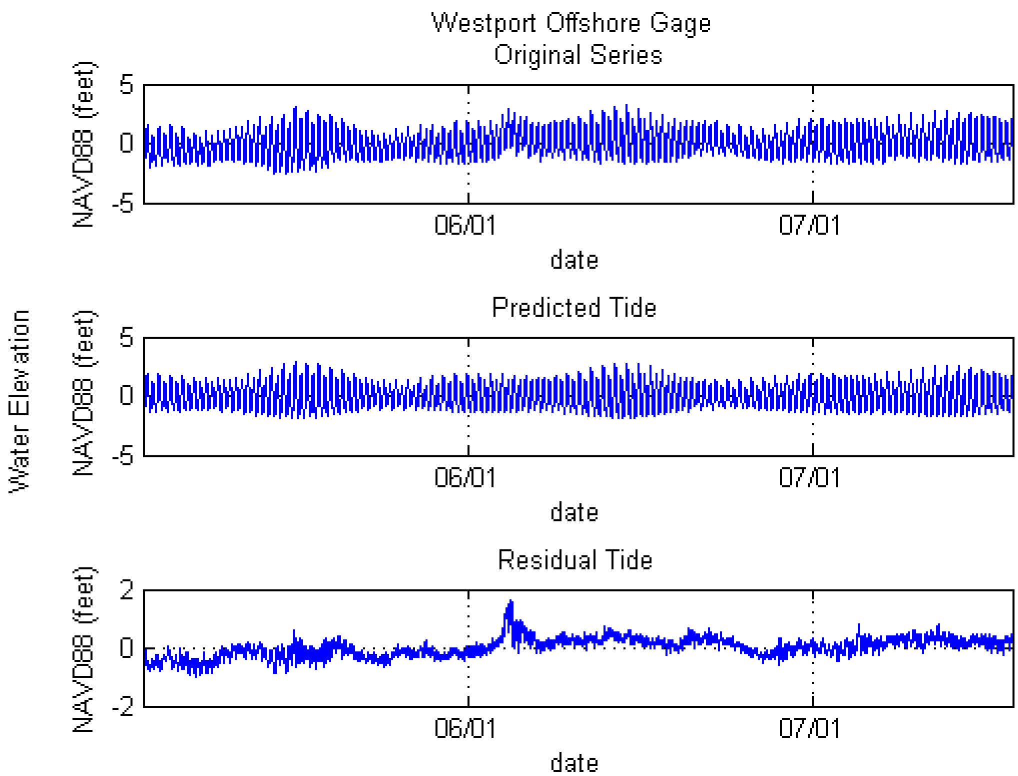


Figure V-7. Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured at the Offshore Gauge (S-1).

Table V-4 Percentages of Tidal versus Non-Tidal Energy, Westport River, 2007

Location	Total Variance (ft^2)	Total (%)	Tidal (%)	Non-tidal (%)
Offshore	1.267	100	92.7	7.3
West Harbor	0.939	100	92.3	8.7
East Harbor	0.876	100	89.9	10.1
The Let	0.978	100	89.4	10.6
South West Branch	0.993	100	90.8	9.2
North West Branch	1.017	100	89.8	10.2
East Branch	0.755	100	89.7	10.3

V.2.3 ADCP DATA ANALYSIS

Cross-channel current measurements were surveyed through a complete tidal cycle at a transect station in Westport Harbor on July 9, 2007 to resolve spatial and temporal variations in tidal current patterns (Figure V-2). The survey was designed to observe tidal flow across the harbor at hourly intervals. The data collected during this survey provided information that was necessary to check the hydrodynamics of the model at this location in the system. Along-channel flow is defined as flow east (positive) for this survey.

V.3 HYDRODYNAMIC MODELING

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

V.3.1 Model Theory

This study of Westport River utilized a state-of-the-art computer model to evaluate tidal circulation and flushing. 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 for estuary systems in southeast Massachusetts, including systems in Chatham, Falmouth's 'finger' ponds, and Popponesset Bay.

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 Brigham Young University through a package called the Surfacewater Modeling System or SMS (BYU, 1998). SMS is a front- and back-end software package that allows the user to easily modify model parameters (such as geometry, element coefficients, and boundary conditions), as well as view the model results and download specific data types. While the RMA model is essentially used without cost or constraint, the SMS software package requires site licensing for use.

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 criterion is met.

V.3.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of the finite element grid was generated using digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of the system based on the tide gauge data collected at the Offshore gauge location. 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 (15+) model calibration simulations for each system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid Generation

The grid generation process for the model was assisted through the use of the SMS package. The digital shoreline and bathymetry data were imported to SMS, and a finite element grid was generated to represent the estuary with 2814 elements and 8061 nodes (Figure V-8). All regions in the system were represented by two-dimensional (depth-averaged) elements. The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties within the estuary. Fine resolution was required to simulate the numerous marshy areas (e.g., scattered throughout the lower portions of the system) that significantly impact the estuarine hydrodynamics. The completed grid is made up of quadrilateral and triangular two-dimensional elements. Reference water depths at each node of the model were interpreted from bathymetry data obtained in the recent field surveys and the NOAA data archive. The final interpolated grid bathymetry is shown in Figure V-9. The model computed water elevation and velocity at each node in the model domain.

Grid resolution is governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each region. Smaller cross channel node spacing in the river channels was designed to provide a more detailed analysis in these regions of rapidly varying velocities and bathymetry. Widely spaced nodes were utilized in areas where velocity gradients were likely to be less acute; for example, in broad, deep channel sections in the model domain. Appropriate implementation of wider node spacing and larger elements reduces computer run time with no sacrifice of accuracy.

V.3.2.2 Boundary Condition Specification

Three types of boundary conditions were employed for the RMA-2 model: 1) "slip" boundaries, 2) freshwater inflow, and 3) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A freshwater inflow boundary condition was specified for Adamsville Brook (at the head of the West Branch), Angeline Brook (roughly 1.1 miles southeast of the head of the West Branch, entering the system from the east side of the West Branch), Westport River (at the Old County Road bridge near the Old County Road and Drift Road intersection), Kirby Brook (roughly 1.75 miles south of the Old County Road bridge, entering the East Branch from the west bank), and Snell Creek (another 0.9 miles south of the Kirby Brook boundary condition, also entering from the West Bank).

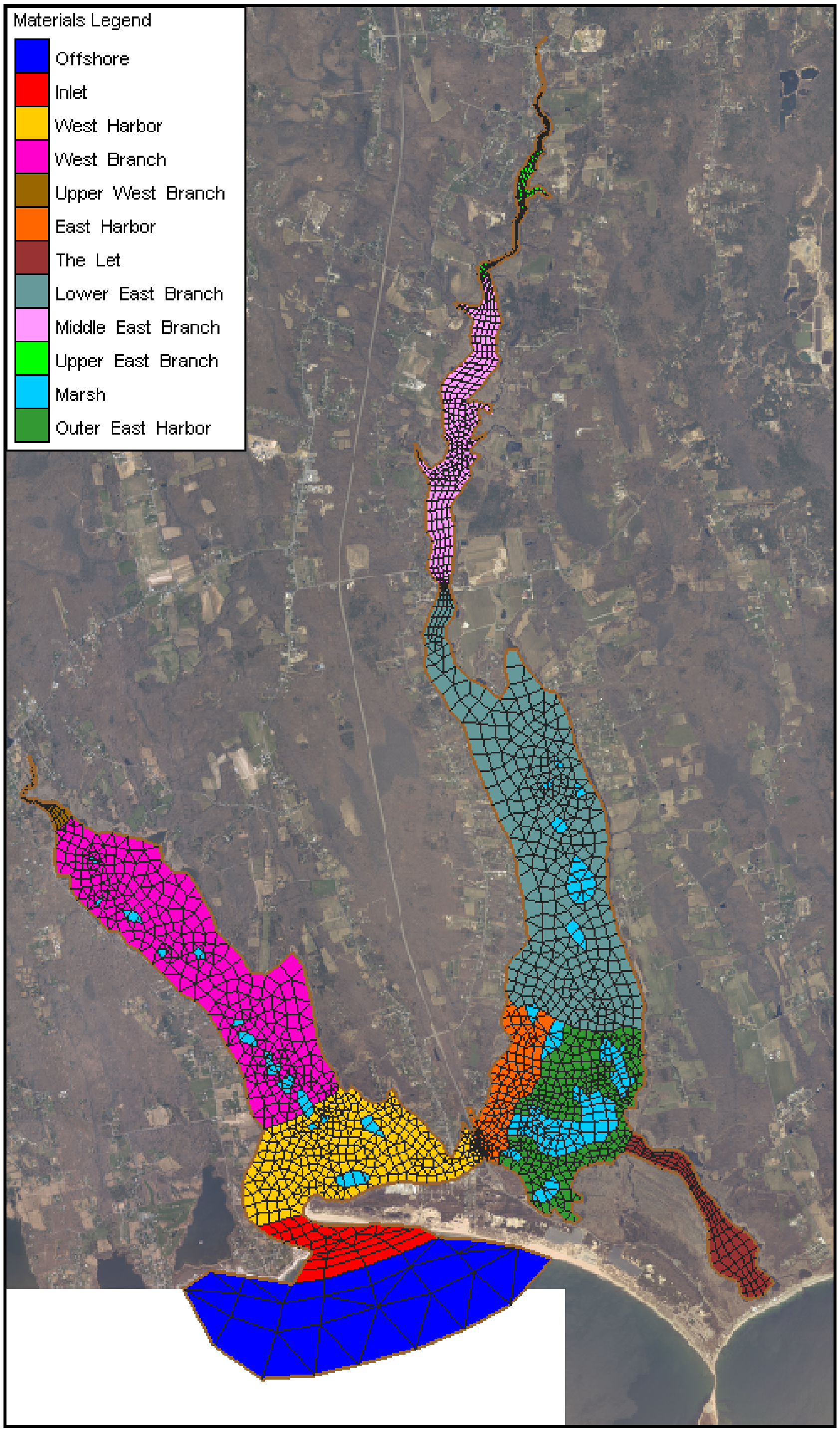


Figure V-8. The model finite element mesh developed for Westport River estuary system. The model seaward boundary was specified with a forcing function consisting of water elevation measurements obtained at the Offshore Gauge (S-1).

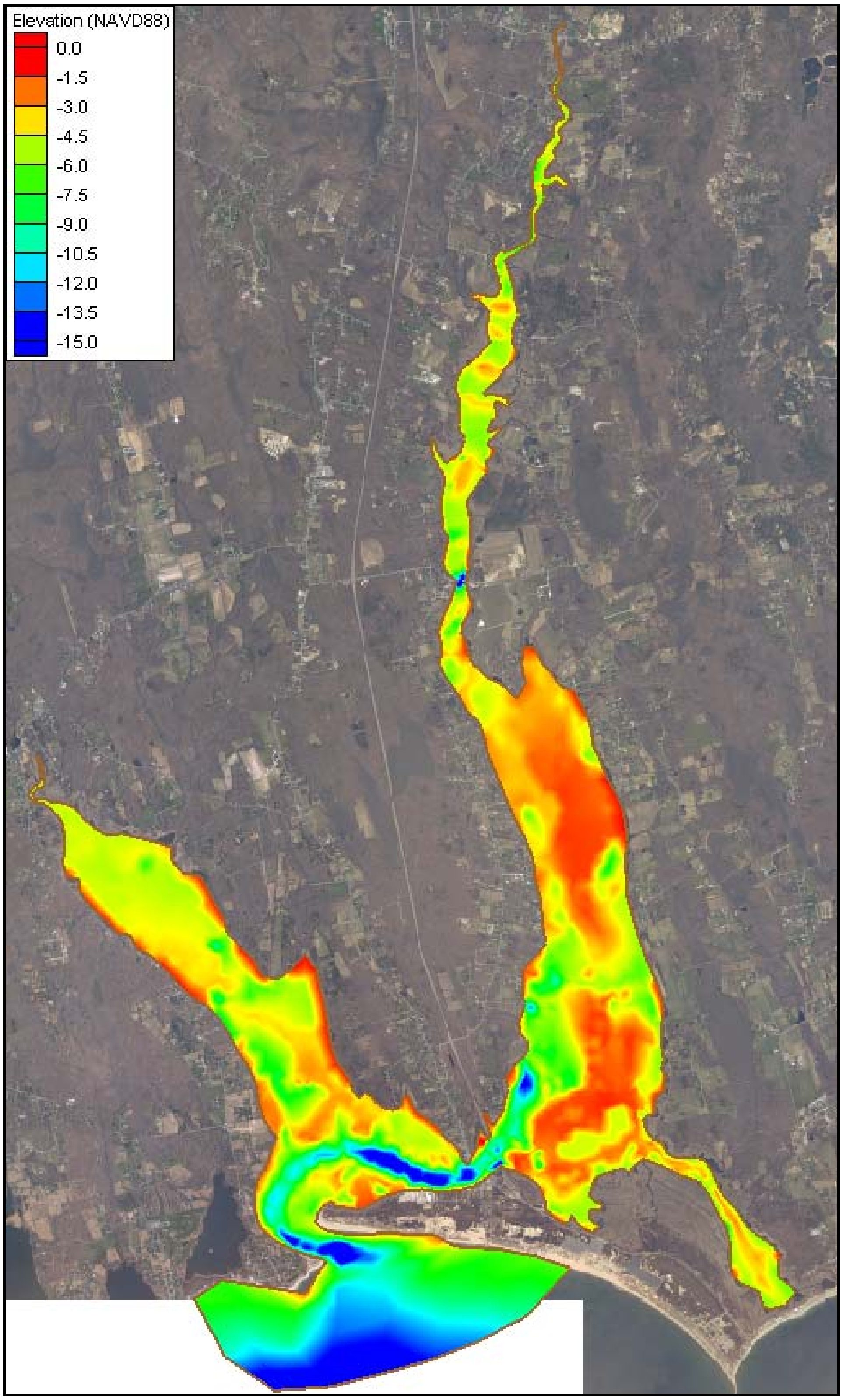


Figure V-9. Depth contours of the completed Westport River finite element mesh.

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

V.3.3 Calibration

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

Calibration of the flushing model required a close match between the modeled and measured tides at each gauge station. Initially, the model was calibrated by the visual agreement between modeled and measured tides. To refine the calibration procedure, water elevations were output from the model at the same locations in the estuary where tide gauges were installed, and the data were processed to calculate standard error as well harmonic constituents (of both measured and modeled data) over the fourteen-day calibration period. The amplitude and phase of four constituents (M_2, M_4, O_1, and K_1) were compared and the corresponding errors for each were calculated. The intent of the calibration procedure is to minimize the error in amplitude and phase of the individual constituents. In general, minimization of the M_2 amplitude and phase becomes the highest priority, since this is the dominant constituent. Emphasis is also placed on the M_4 constituent, as this constituent has the greatest impact on the degree of tidal distortion within the system, and provides the unique shape of the modified tide wave at various points in the system.

The calibration was performed for an approximate fourteen-day period, beginning 2250 hours EDT June 15, 2007 and ending 0000 hours EDT June 30, 2007. This time period included a 26-hour model spin-up period, and a 24-tide cycle period used for calibration. This representative time period was selected because it included tidal conditions where the wind-induced portion of the signals (i.e. the residual) was minimal, hence more typical of tidal circulation within the estuary. The selected time period also spanned the transition from spring (bi-monthly maximum) to neap (bi-monthly minimum) tide ranges, which is representative of average tidal conditions in the embayment system. Throughout the selected 13 day period after the spin-up, the tide ranged approximately 4.4 feet from minimum low to maximum high tides. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. Modeled tides were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibrated model was used to analyze existing detailed flow patterns and compute residence times.

V.3.3.1 Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where water depths can become shallow and velocities relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude attenuation and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient. First, Manning's friction coefficient values of 0.025 were specified for all elements. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels with pools and shoals with higher friction (Henderson, 1966). Final calibrated friction coefficients (listed in Table V-5) were largest for the marsh areas scattered throughout the system, where values were set at 0.075. Small changes in these values did not change the accuracy of the calibration.

Table V-5. Manning's Roughness coefficients used in simulations of modeled embayments.

Embayment	Bottom Friction
Offshore	0.025
Inlet	0.035
West Harbor	0.030
West Branch	0.030
Upper West Branch	0.030
East Harbor	0.030
The Let	0.045
Lower East Branch	0.020
Middle East Branch	0.020
Upper East Branch	0.020
Marsh	0.075
Outer East Harbor	0.035

V.3.3.2 Turbulent Exchange Coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swift, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). The model was mildly sensitive to turbulent exchange coefficients, with the Outer East Harbor and The Let being the most sensitive. In other regions where the flow gradients were not as strong, the model was much less sensitive to changes in the turbulent exchange coefficients. Typically, model turbulence coefficients (D) are set between 10 and 100 lb-sec/ft^2 (as listed in Table V-6).

V.3.3.3 Wetting and Drying/Marsh Porosity Processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain included in the model as part of both the East and West Branches of the Westport 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. The marsh porosity feature of RMA-2 is typically utilized in estuarine systems where the marsh plain has a significant impact on the hydrodynamics of a system.

Table V-6. Turbulence exchange coefficients (D) used in simulations of modeled embayment system.

Embayment	D (lb-sec/ft$^{2)}$
Offshore	100
Inlet	100
West Harbor	35
West Branch	75
Upper West Branch	75
East Harbor	35
The Let	100
Lower East Branch	20
Middle East Branch	15
Upper East Branch	15
Marsh	20
Outer East Harbor	100

V.3.3.4 Comparison of Modeled Tides and Measured Tide Data

Several calibration model runs (June 15 to June 29, 2007) were performed to determine how changes to various parameters (e.g. friction and turbulent exchange coefficients) affected the model results. These trial runs achieved excellent agreement between the model simulations and the field data. Comparison plots of modeled versus measured water levels at the seven gauge locations are presented in Figures V-10 through V-16. At all gauge stations, RMS errors were less than 0.18 ft (<2.16 inches) and computed R^2 correlation was better than 0.95 for every station. Errors between the model and observed tide constituents were less than 0.09 feet for all locations, suggesting the model accurately predicts tidal hydrodynamics within the Westport estuarine system. Measured tidal constituent amplitudes and time lags (ϕ_{lag}) for the validation time period (July 1 to July 14, 2007) are shown in Table V-7. The constituent values for the validation time period differ from those in Tables V-2 because constituents were computed for only 14 days, rather than the entire 78-day period represented in Tables V-2. Errors associated with tidal constituent height were on the order of hundredths of feet, which was an order of magnitude better than the accuracy of the tide gauge (±0.12 ft). Time lag errors were close to the time increment resolved by the model and measured tide data (10 minutes) for all gauges, indicating good agreement between the model and data.

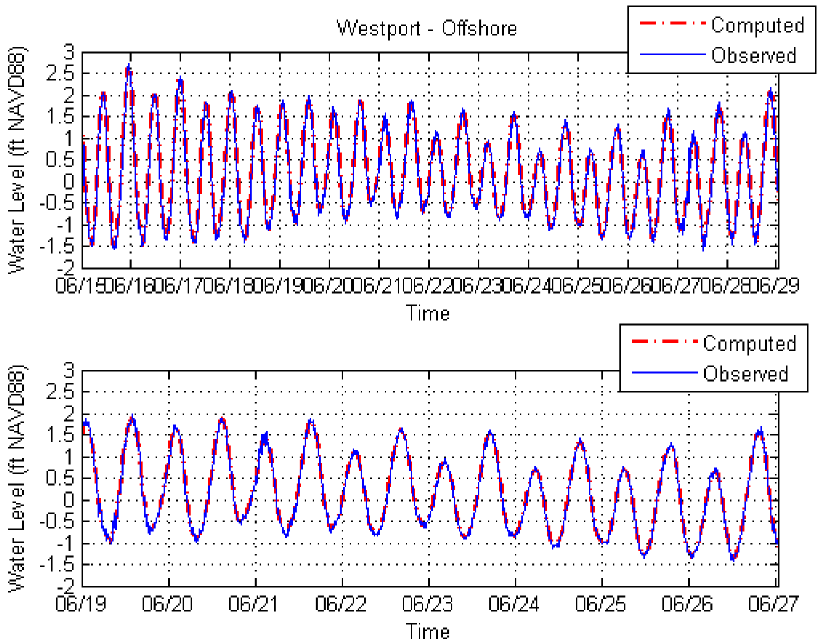


Figure V-10. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for the Offshore Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

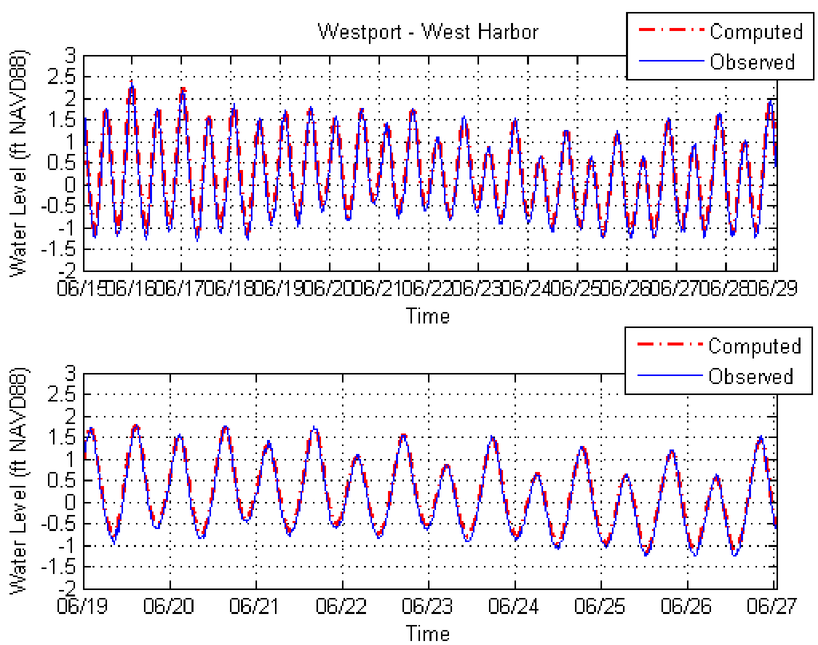


Figure V-11. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for the West Harbor Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

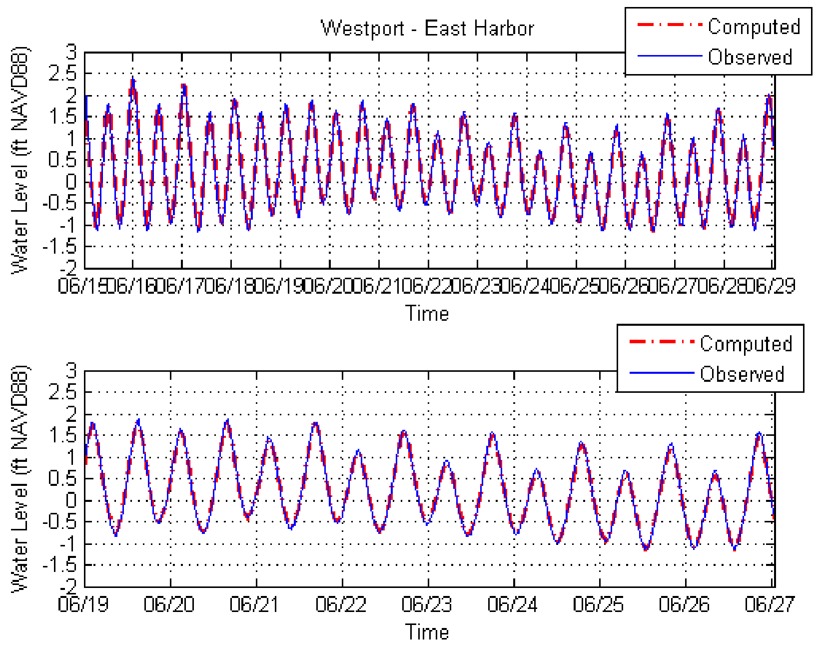


Figure V-12. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for the East Harbor Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

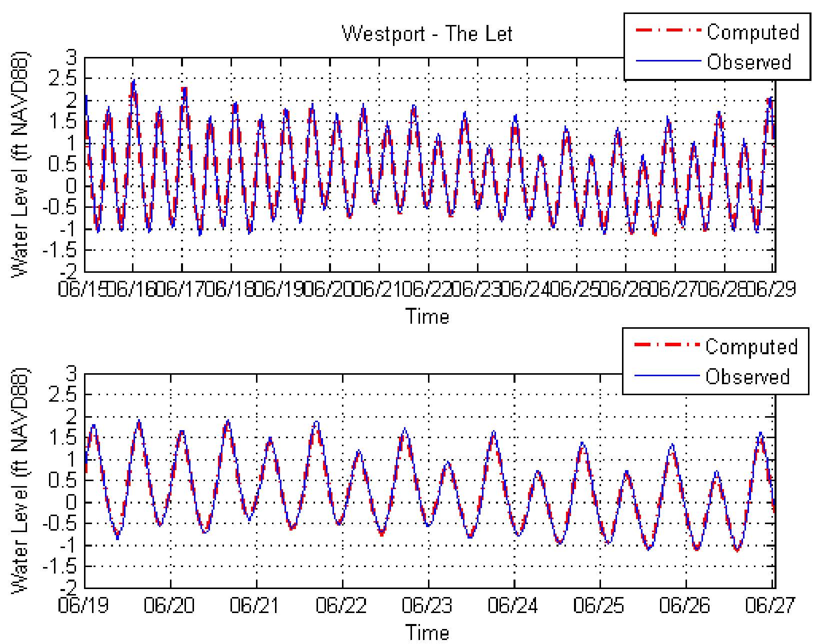


Figure V-13. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for The Let Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

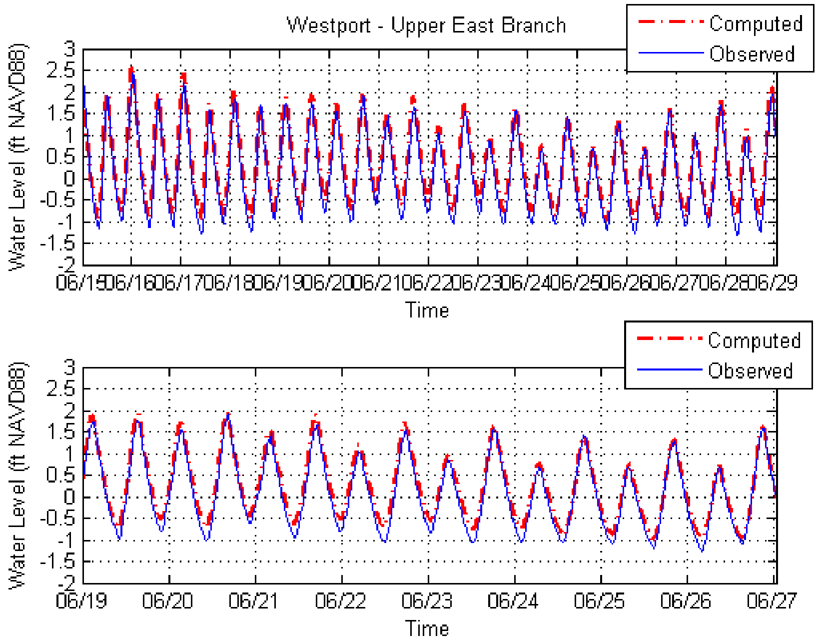


Figure V-14. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for the Upper East Branch Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

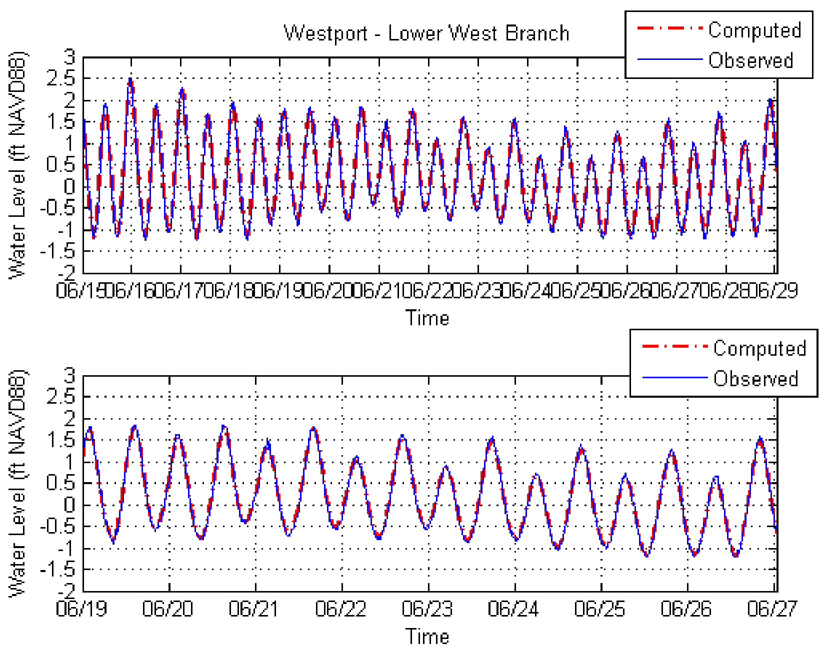


Figure V-15. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for Lower West Branch Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

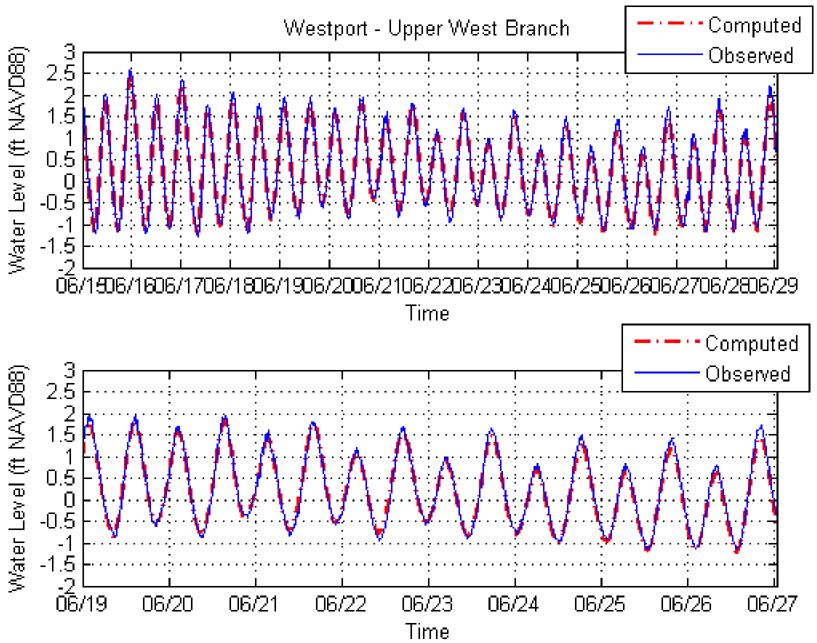


Figure V-16. Comparison of water surface variations simulated by the model (dashed red line) to those measured within the system (solid blue line) for the calibration time period, for Upper West Branch Gauge Station. The top plot shows the entire record with the bottom plot showing an 8-day segment.

V.3.4 ADCP Verification of the Westport River System

A model verification check was possible by using collected ADCP velocity data to verify the performance of the Westport 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 previously in this chapter. For the model ADCP verification, the Westport River model was run for period covered during the ADCP survey on July 9, 2007. Model flow rates were computed in RMA-2 at a continuity line (channel cross-section) that correspond to the actual ADCP transect followed in the survey (i.e., across the Route 88 Bridge).

Comparisons of the measured and modeled volume flow rates in the Westport River system are shown in Figure V-17. In the 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 inlet, and inside 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 the ADCP transect show exceptionally good agreement with the model predictions. The calibrated model accurately describes the discharge magnitude at the bridge. The R^2 correlation coefficient between data and model results is 0.96 with a RMS error of 14%.

Table V-7. Comparison of Tidal Constituents from the validated RMA2 model versus measured tidal data for the period July 1 to July 14, 2007.

Model Verification Run						
Location	Constituent Amplitude (ft)				Phase (degrees)	
	M_2	M_4	O_1	K_1	ΦM_2	ΦM_4
Offshore	1.394	0.135	0.159	0.266	99.75	-116.52
Harbor West	1.200	0.090	0.157	0.256	125.45	-86.28
Harbor East	1.169	0.084	0.159	0.256	136.92	-66.24
Upper East Branch	1.113	0.186	0.155	0.251	158.09	-77.56
The Let	1.158	0.100	0.157	0.257	145.92	-86.3
Lower West Branch	1.225	0.086	0.159	0.259	122.8	-101.26
Upper West Branch	1.236	0.083	0.159	0.261	126.35	-105.76
Measured Tidal Data						
Location	Constituent Amplitude (ft)				Phase (degrees)	
	M_2	M_4	O_1	K_1	ΦM_2	ΦM_4
Offshore	1.392	0.134	0.160	0.267	98.48	-119.9
Harbor West	1.248	0.102	0.162	0.267	123.31	-91.66
Harbor East	1.225	0.080	0.165	0.272	134.6	-75.58
Upper East Branch	1.199	0.171	0.160	0.282	158.15	-82.01
The Let	1.208	0.105	0.172	0.272	146.78	-93.07
Lower West Branch	1.271	0.092	0.163	0.267	121.11	-110.29
Upper West Branch	1.286	0.096	0.156	0.286	126.47	-130.47
Error						
Location	Constituent Amplitude (ft)				Phase (minutes)	
	M_2	M_4	O_1	K_1	ΦM_2	ΦM_4
Offshore	-0.002	0.000	0.002	0.001	-2.62	-3.5
Harbor West	0.048	0.012	0.005	0.011	-4.42	-5.56
Harbor East	0.055	-0.004	0.007	0.016	-4.8	-9.67
Upper East Branch	0.086	-0.014	0.004	0.031	0.11	-4.6
The Let	0.050	0.005	0.015	0.015	1.79	-7.01
Lower West Branch	0.046	0.005	0.004	0.008	-3.51	-9.35
Upper West Branch	0.051	0.013	-0.003	0.024	0.26	-25.57

V.3.5 Model Circulation Characteristics

The final calibrated and validated model serves as a useful tool for investigating the circulation characteristics of the Westport River estuary 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.

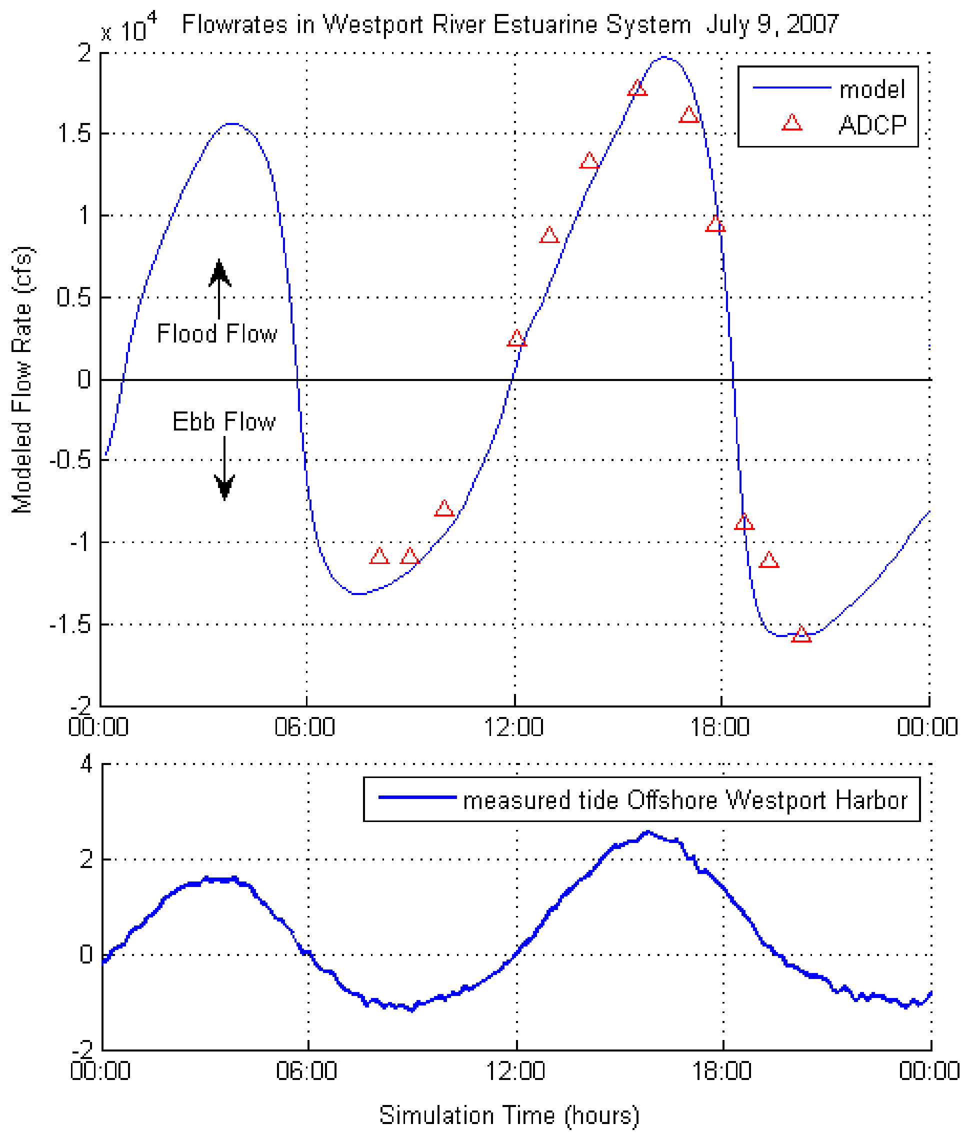


Figure V-17. Comparison of measured volume flow rates versus modeled flow rates (top plot) through the Route 88 Bridge over a tidal cycle July 9, 2007. The computed RMS error for this model run was 14% of maximum measured flow, with a R2 correlation coefficient of 0.96. Flood flows into the inlet are positive (+), and ebb flows out of the inlet are negative (-). The bottom plot shows the tide elevation offshore Westport River.

From the model run of the estuary system, maximum flood velocities at the Westport River inlet are slightly smaller than velocities during the ebb portion of the tide. Maximum depth-averaged velocities in the model are approximately 3.0 feet/sec for flooding tides, and 3.6 ft/sec for ebbing tides. A close-up of the model output is presented in Figure V-18, which shows contours of flow velocity, along with velocity vectors which indicate the direction and magnitude of flow, for a single model time-step, at the portion of the tide where maximum flood velocities occur at the inlet.

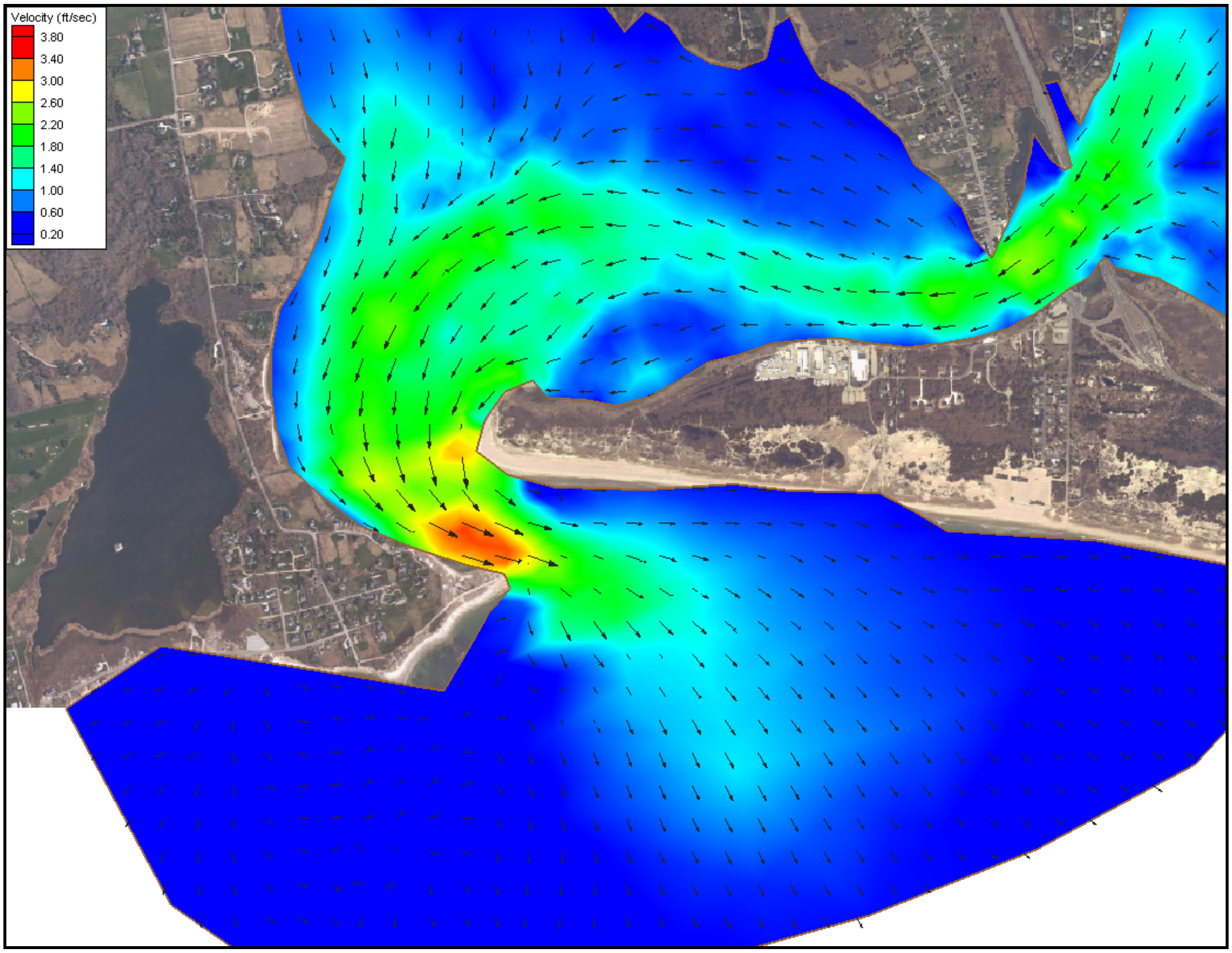


Figure V-18. Example of hydrodynamic model output in Westport River for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate flow velocity, and vectors indicate the direction and magnitude of flow.

In addition to depth averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. The variation of flow as the tide floods and ebbs through the Westport River Estuarine system is seen in Figures V-19 and V-20. During the simulation time period, maximum modeled flood tide flow rates through the Westport River inlet were 34,300 ft^3/sec and ebb tide flow rates were 31600 ft^3/sec. The higher flow rates for the flood tide occur despite the lower maximum velocity, this is due to there being a localized high velocity area hugging the outer bank during ebb flow (as shown in Figure V-18), while the flood velocities are more consistent across the channel. The massive influence of surfacewater on the flows in the upper portions of the system is demonstrated in Figure V-20. The maximum modeled flood tide flow rates through the Route 88 Bridge were 22,800 ft^3/sec and ebb tide flow rates were 17,800 ft^3/sec, the flood tide flow rates for the Hix Bridge were 3,880 ft^3/sec and the ebb tide flow rates were 3,130 ft^3/sec, and the flood tide flow rates for the Top of the West Branch were 39 ft^3/sec and the ebb tide flow rates were 73 ft^3/sec. The maximum modeled flood tide flow rates at the Top of the East Branch were 76 ft^3/sec, but this flow was in the direction of ebb tide, indicating that during the flood portion of the tide, there is merely a slowing of the incoming freshwater flow at the upper reaches of the East Branch The ebb tide flow rates were 101 ft^3/sec for the Top of the East Branch. This is due to the large surfacewater flow boundary condition at this bridge influencing the flow, but the fluctuation shows that this region is strongly influenced by the tide.

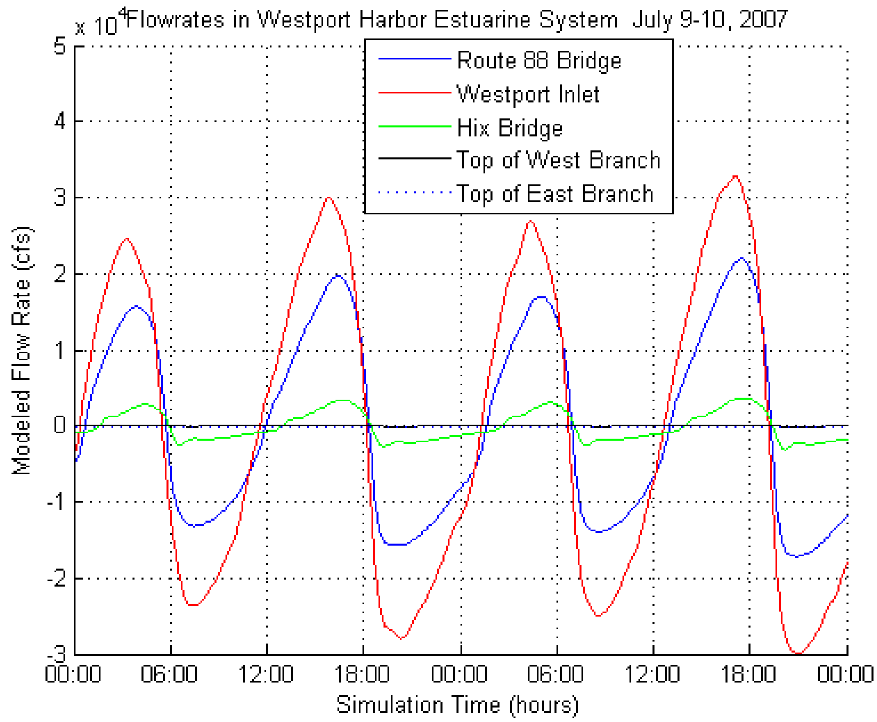


Figure V-19. Time variation of computed flow rates for transects across the Route 88 Bridge, the Westport Inlet, the Hix Bridge, the Top of the West Branch, and the Top of the East Branch. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are likewise large compared to neap tide conditions. Positive flow indicates flooding tide, while negative flow indicates ebbing tide.

V.4 FLUSHING CHARACTERISTICS

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

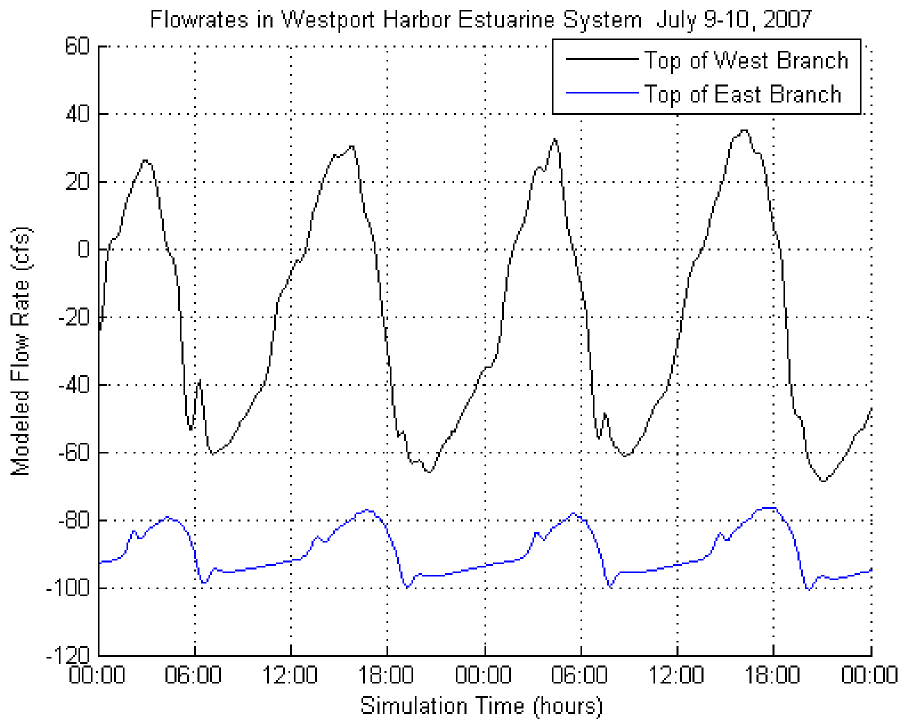


Figure V-20. Time variation of computed flow rates for transects across the Top of the West Branch and the Top of the East Branch, demonstrating the massive influence surfacewater has at the upper reaches of the system. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are likewise large compared to neap tide conditions. Positive flow indicates flooding tide, while negative flow indicates ebbing tide.

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 the head of the East Branch as an example, the system residence time is the average time required for water to migrate from the Old County Road bridge, through the lower portions of the East Branch into the harbor, and finally into Buzzard's Bay, where the local residence time is the average time required for water to migrate from the Old County Road bridge to the Hix Bridge (not all the way to the inlet and out of the system). 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 modeled system, this approach is applicable, since it assumes the main system has relatively low quality water relative to Buzzard’s Bay.

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality will be obtained from the calibrated hydrodynamic model by extending the model to include a total nitrogen dispersion model (Section VI). The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Westport River Estuary.

The volume of each sub-embayment, as well as their respective tidal prisms, was computed in cubic feet (Table V-8). Model divisions used to define the system sub-embayments for the two systems include 1) the whole of the Westport River system, 2) West Branch, 3) the whole of the East Branch east of the Route 88 Bridge, 4) The Let, and 5) the section of the East Branch north of Hix Bridge. The model computed total volume of each sub-embayment at every time step, and this output was used to calculate mean sub-embayment volume and average tide prism. Since the 13-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. Mean volumes and average tidal prism of the Westport River estuary system during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Westport River	1,360,330,000	522,282,000
West Branch	154,011,000	99,642,000
East Branch with The Let	349,205,000	226,276,000
The Let	27,383,000	16,976,000
East Branch North of Hix Bridge	66,691,000	36,515,000

Residence times were averaged for the tidal cycles comprising a representative 13 day period (24 tide cycles), and are listed in Table V-9. Residence times were computed for the entire estuary, as well selected 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 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 moderate local residence time (1.4 days) of the whole Westport River estuary system shows that the harbor area most likely flushes reasonably well. However, with the decrease in the local residence times (to about half as long) of the embayments as water progresses deeper into the system, it could be assumed that the flushing of the system as a whole is adequate. The extreme lengths of the system residence times inside the Let (almost 42 days) indicate poor flushing of the system in this relatively stagnant region.

Table V-9. Computed System and Local residence times for sub-embayments of the Westport River estuary system.

Embayment	Local Residence Time (days)	System Residence Time (days)
Westport River	1.4	1.4
West Branch	0.8	7.1
East Branch with The Let	0.8	3.1
The Let	0.8	41.7
East Branch North of Hix Bridge	0.9	19.4

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 Westport River estuary 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 in Buzzard's Bay is typically strong because of the effects of the local winds and tidal induced mixing, the "strong littoral drift" assumption should cause only 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 Westport River estuary 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 systems. Files of node locations and node connectivity for the RMA-2V 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 20-tidal cycle period in July 2007. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28 day spin-up period, to allow the model to reach a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayment

Three primary nitrogen loads to the 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 Westport River estuary system, consisting of the background concentrations of total nitrogen in the waters entering from Buzzard’s Bay. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the 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 summer 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. Generally, seven years of data (collected between 2003 and 2009) were available for stations monitored by SMAST in the Westport River estuary system, with a few data gaps.

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 Westport River estuary 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 Westport River estuary system. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other embayments in Southeastern Massachusetts, including systems in Falmouth (Ramsey *et al*., 2000); Mashpee, MA (Howes *et al*., 2004) and Chatham, MA (Howes *et al*., 2003).

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

Sub-Embayment	Monitoring station	Mean	s.d. all data	N	model min	model max	model average
Head Westport	N-0	1.440	0.266	22	1.340	1.346	1.344
Upper East Branch	N-1	1.102	0.295	27	0.889	0.958	0.919
Upper East Branch	N-2	1.009	0.278	27	0.840	0.910	0.879
Upper East Branch	N-3	0.874	0.200	23	0.777	0.906	0.855
Mid East Branch	N-4	0.864	0.223	25	0.647	0.897	0.798
Mid East Branch	E-69	0.851	0.227	44	0.587	0.851	0.735
Mid East Branch	E-56	0.794	0.279	23	0.538	0.712	0.616
Lower East Branch	E-33	0.700	0.186	20	0.441	0.693	0.554
Lower East Branch	E-41	0.626	0.172	19	0.406	0.575	0.492
Lower East Branch	E-30	0.538	0.173	21	0.302	0.518	0.414
Lower Rivers	E-26	0.534	0.192	22	0.293	0.485	0.389
Lower West Branch	W-12	0.649	0.253	41	0.383	0.595	0.491
Lower West Branch	W-9	0.501	0.117	17	0.296	0.511	0.394
Lower West Branch	W-6	0.449	0.081	13	0.286	0.476	0.364
Inlet	N-12	0.477	0.166	22	0.284	0.424	0.329

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 and addresses the seasonal fluctuations in Westport River stream inputs. 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 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.

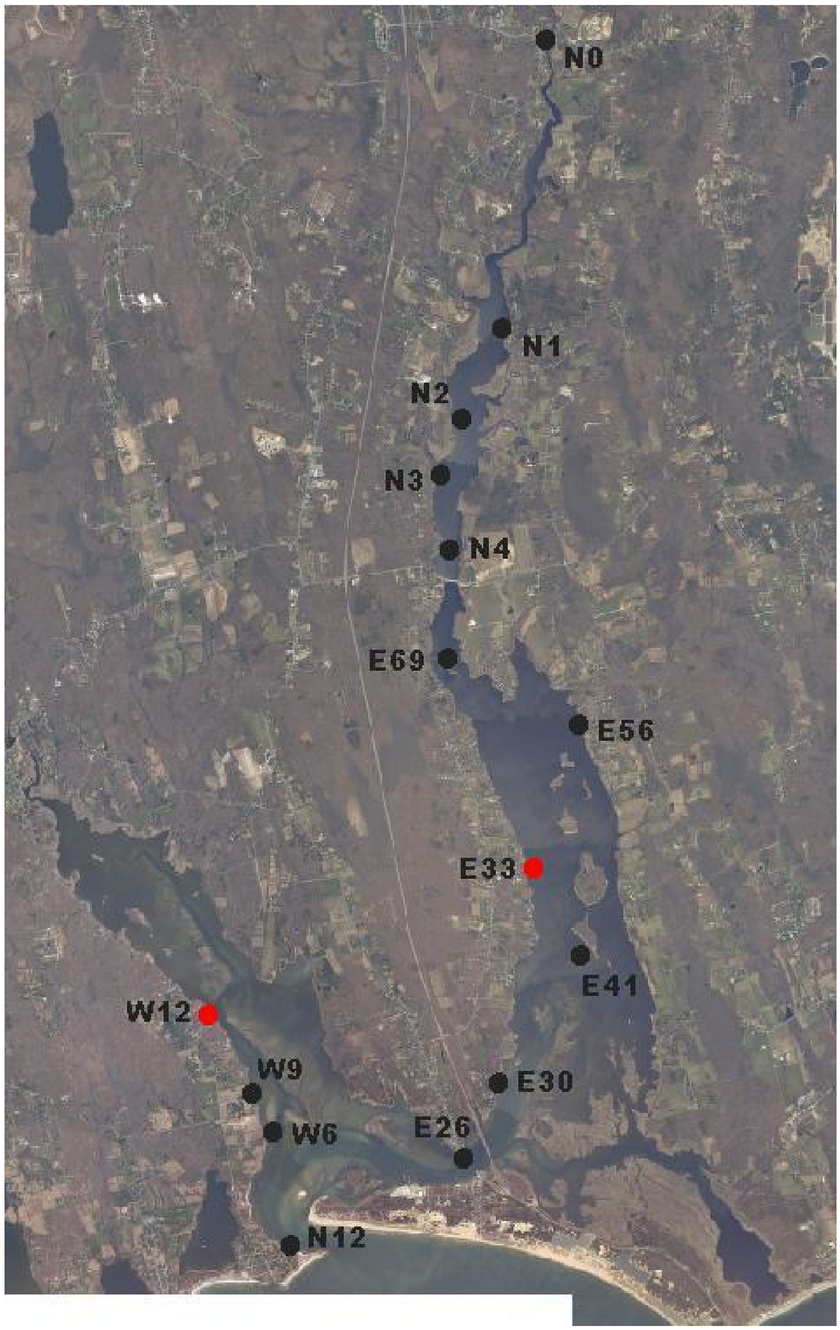


Figure VI-1. Estuarine water quality monitoring station locations in the Westport River estuary system. Station labels correspond to those provided in Table VI-1. The approximate locations of the threshold stations are depicted by the red symbols, at E33 for the East Branch and W12 for the West Branch.

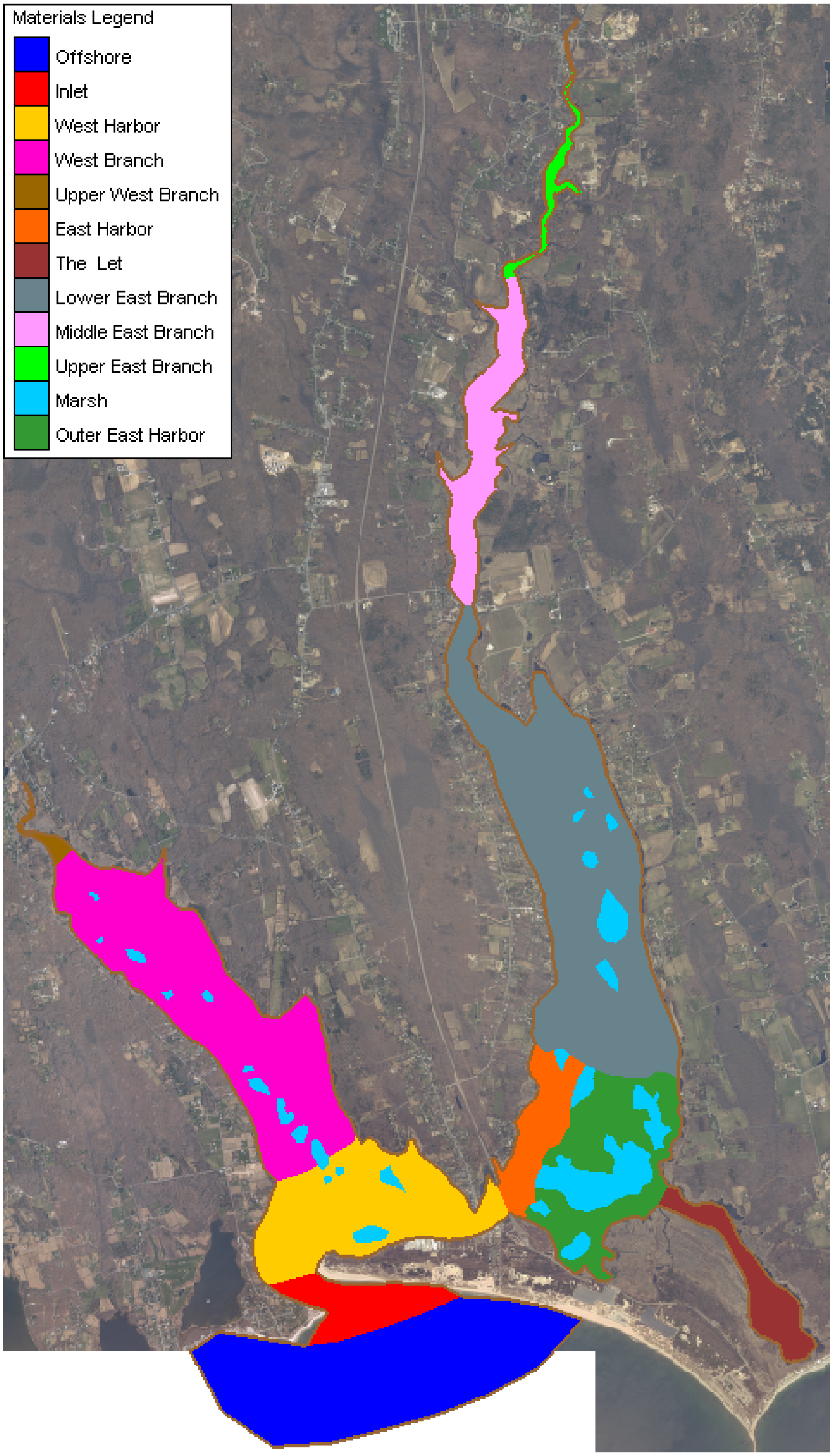


Figure VI-2. Map of the Westport River water quality model longitudinal dispersion coefficients. Color patterns designate the different areas used to vary model dispersion coefficient values.

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 Westport River estuary 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 Westport River estuary system were used for the water quality constituent modeling portion of this study.

Based on measured surface water flow rates from SMAST and watershed groundwater recharge rates, the hydrodynamic model was set-up to include the latest estimates of freshwater flows from the non-tidal reaches of the Westport River (north of the Old County Road Bridge), Kirby Brook, Snell Creek, Adamsville Brook, and Angeline Brook. Westport River has a mean measured summer flow rate of 48.8 ft^3/sec (119,569 m^3/day), which is only 0.4% of the average tidal prism of Westport River system; however it is 5.98% of the average tidal prism for the sub-embayment between the Old County Road bridge and Hix Bridge, which explains the localized impact this input has on the region versus the system as a whole. Kirby Brook has a mean measured annual flow rate of 9.5 ft^3/sec (23,100 m^3/day or 0.08% of the average tidal prism of the Westport River system), Snell Creek has a mean measured annual flow rate of 2.7 ft^3/sec (6,600 m^3/day or 0.02% of the average tidal prism of the Westport River system), Adamsville

Brook has a mean measured annual flow rate of 19.6 ft^3/sec (48,000 m^3/day or 0.17% of the average tidal prism of the Westport River system), and Angeline Brook has a mean measured annual flow rate of 8.6 ft^3/sec (21,000 m^3/day or 0.07% of the average tidal prism of the Westport River system). Among the gauged surface waters, only Westport River demonstrated a significant seasonal flow difference (see section IV.2). The overall groundwater flow rate into the system is 145.6 ft^3/sec (356,000 m^3/day) distributed amongst the watersheds.

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 spin-up period of 28 day. At the end of the spin-up period, the model was run for an additional 20 tidal-cycle (250 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 Westport River estuary 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, 3) summer benthic regeneration, and 4) point source inputs developed from measurements of the freshwater surface flows of the Westport River Estuarine System. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed and direct atmospheric deposition loads for Westport River were evenly distributed at grid cells that formed the outer edge of the embayment. Benthic regeneration loads were distributed among another sub-set of grid cells which are in the interior portion of each basin.

The loadings used to model present conditions in Westport River estuary system are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Due to the seasonal variability of TN flux in the Westport River (determined by two years measurements in 2006 amd 2007) the modeled summertime nitrogen load for the river was set at 80% of the mean annual load determined in 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, 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 primary portion of the total loading rate for the system comes from watershed loads.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Buzzard’s Bay was set at 0.282 mg/L, based on SMAST data from Buzzard’s Bay. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Buzzard’s Bay.

Table VI-2. Sub-embayment loads used for total nitrogen modeling of the Westport River estuary system, with total watershed N loads, atmospheric N loads, and benthic flux. Watershed loads at gauged streams are based on measured flows and concentrations. These loads represent **present loading conditions.** Note: present load is based on summertime measured value of nitrogen flux at the Old County Road stream gauge on the freshwater portion of the Westport River.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
North East Branch	103.088	4.360	-31.934
West Branch	32.901	11.154	-6.288
South East Branch	62.332	20.922	-19.869
The Let	5.759	1.968	11.811
Westport Harbor	10.252	8.226	-30.507
Old County Road	162.614	-	-
Kirby Brook	20.953	-	-
Adamsville Brook	47.622	-	-
Angeline Brook	34.296	-	-
Snell Creek	8.137	-	-

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

Embayment Division	E m^2/sec
Offshore	15.0
Inlet	15.0
West Harbor	15.0
West Branch	10.0
Upper West Branch	5.0
East Harbor	15.0
The Let	5.0
Lower East Branch	30.0
Middle East Branch	40.0
Upper East Branch	2.5
Marsh	30.0
Outer East Harbor	15.0

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 Westport River 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 portion of the estuary.

Comparisons between model output and measured nitrogen concentrations are shown in plots presented in Figure VI-3. In these plots, means of the summer water column data and a range of one standard deviation of the annual summer 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 excellent for the Westport River system, with rms error of 0.047 mg/L and an R^2 correlation coefficient of 0.96.

A contour plot of calibrated model output is shown in Figure VI-4 for Westport River estuary 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 10-tidal-day model simulation output period.

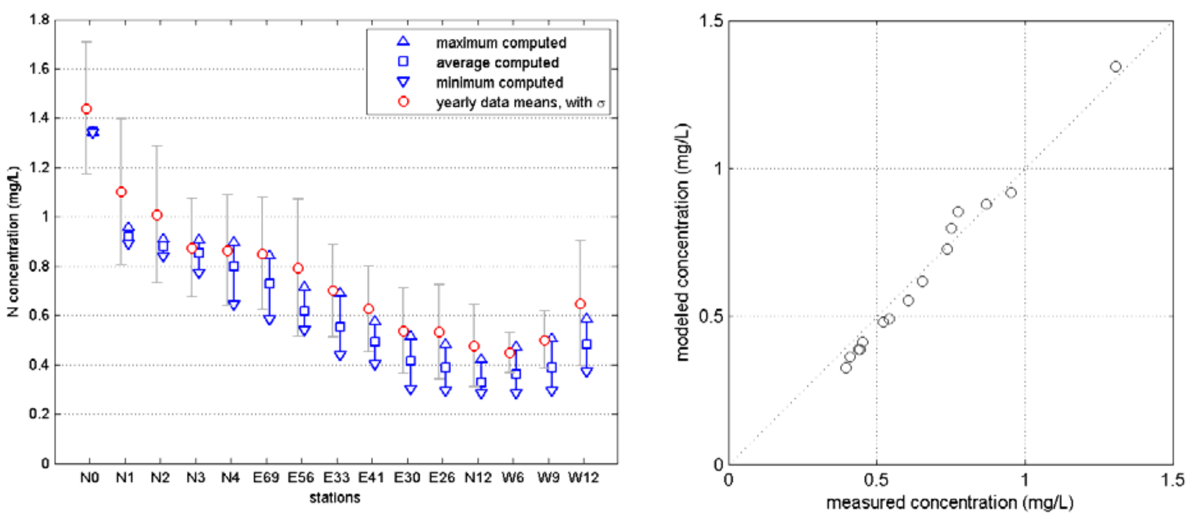


Figure VI-3. Comparison of measured summer total nitrogen concentrations and calibrated model output at stations in Westport River estuary system. For the left plot, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values 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 summer 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. Computed correlation (R^2) and error (rms) for each model are also presented.

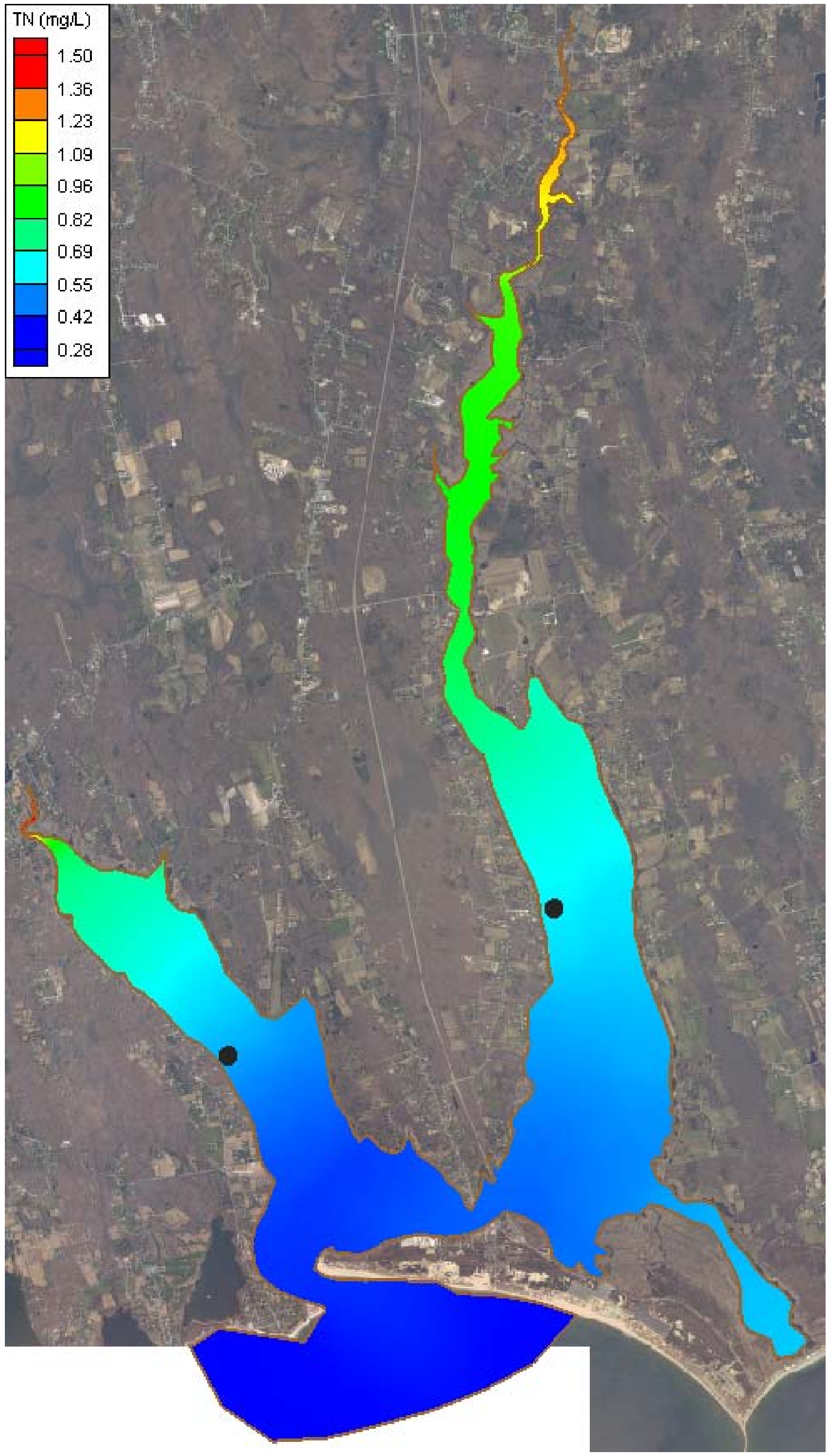


Figure VI-4. Contour plots of average summer total nitrogen concentrations from results of the present conditions loading scenario, for Westport River estuary system. The approximate locations of the sentinel threshold stations for Westport River estuary system are shown by the black symbols (W12 for the West Branch and E33 for the East Branch).

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 Westport River estuary system using summer 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, groundwater inputs, and measured surface water stream inputs. The open boundary salinity was set at 31.6 ppt. For groundwater and measured surface water inputs, salinities were set at 0 ppt. Total groundwater input used for the model was 145.6 ft^3/sec (356,000 m^3/day) distributed amongst the watersheds. Groundwater flows were distributed evenly in each sub-embayment through the use of several "rainwater" element input points positioned along each model's land boundary.

Comparisons of modeled and measured summer 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 reasonably represents salinity gradients in Westport River estuary system. The rms error of the models was 0.86 ppt, and the correlation was 0.99. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical systems.

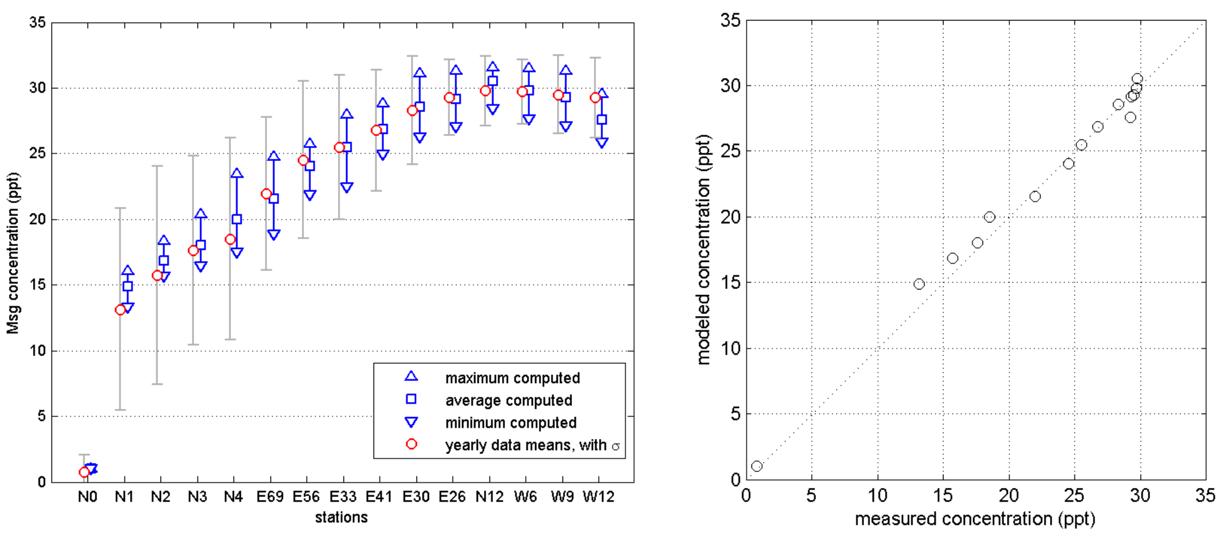


Figure VI-5. Comparison of measured and calibrated salinity model output at stations in Westport River estuary system. For the left plot, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total summer yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plot to the right, model calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

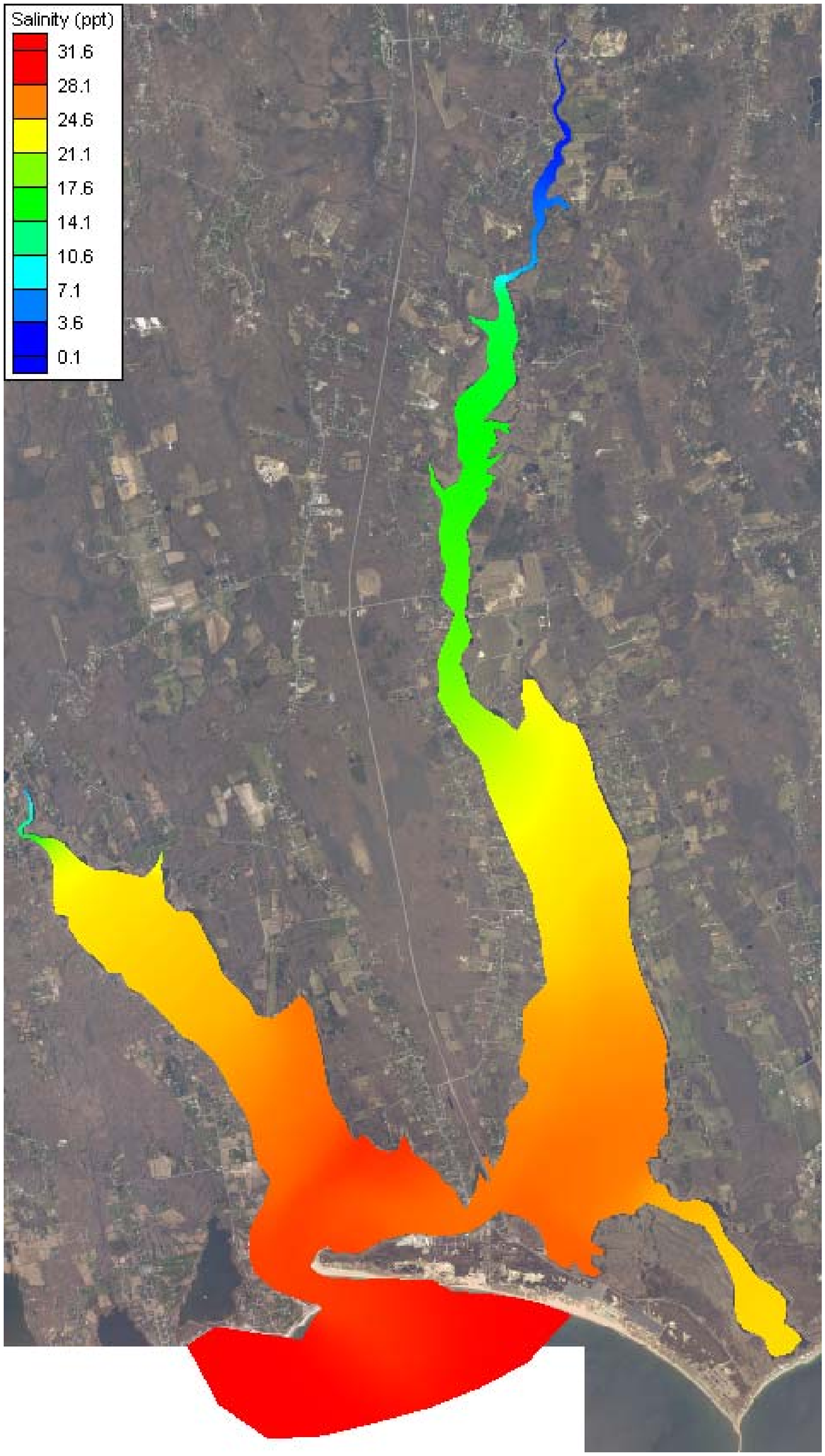


Figure VI-6. Contour plots of modeled summer salinity (ppt) in Westport River estuary system.

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 Westport River estuary system. Buildout loads are conservative because they are uncorrected for the measured seasonality of the Westport River freshwater input. 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
North East Branch	103.088	115.173	+11.7%	7.071	-93.1%
West Branch	32.901	41.836	+27.2%	3.416	-89.6%
South East Branch	62.332	93.526	+50.0%	29.471	-52.7%
The Let	5.759	13.230	+129.7%	2.630	-54.3%
Westport Harbor	10.252	11.825	+15.3%	1.422	-86.1%
Old County Road	162.614	199.470	+22.7%	63.805	-60.8%
Kirby Brook	20.953	25.830	+23.3%	5.986	-71.4%
Adamsville Brook	47.622	62.940	+32.2%	18.721	-60.7%
Angeline Brook	34.296	47.868	+39.6%	5.477	-84.0%
Snell Creek	8.137	14.425	+77.3%	1.866	-77.1%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. For example, the build-out scenario indicates that there would be more than a 50% increase in watershed nitrogen load to the South East Branch of the Westport River as a result of potential future development, but only an increase of 11.7% from the North East Branch. For the no load scenario, a majority of the load entering the watershed is removed; therefore, the load is lower than existing conditions by over 52% for the South East Branch of the Westport River and 93% for the North East Branch.

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

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 Westport River estuary 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)
North East Branch	115.173	4.36	-38.17
West Branch	41.836	11.15	-6.85
South East Branch	93.526	20.92	-24.68
The Let	13.230	1.97	12.84
Westport Harbor	11.825	8.23	-32.43
Old County Road	199.470	-	-
Kirby Brook	25.830	-	-
Adamsville Brook	62.940	-	-
Angeline Brook	47.868	-	-
Snell Creek	14.425	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of the Westport River estuary system was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (*i.e.*, Buzzard's Bay) remained identical to the existing conditions modeling scenarios. It should be noted that the buildout nitrogen loads are based on annual loads and do not include corrections for low seasonal loads measured in the freshwater input from the Westport River. As such, these loads are conservative and could be further refined as additional scenarios are developed. In the buildout scenario, total N concentrations increased the most in the West Branch (30%), while the change was slightly less at station N-4 in the East Branch (roughly 26%). 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. For typical systems, a total nitrogen concentration greater than 0.5 mg/L leads to negative impacts in benthic fauna.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Westport River estuary system. The sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Head Westport	N-0	1.344	1.649	+28.7%
Upper East Branch	N-1	0.919	1.091	+27.0%
Upper East Branch	N-2	0.879	1.037	+26.5%
Upper East Branch	N-3	0.855	1.006	+26.3%
Mid East Branch	N-4	0.798	0.932	+25.9%
Mid East Branch	E-69	0.735	0.854	+26.2%
Mid East Branch	E-56	0.616	0.705	+26.8%
Lower East Branch	**E-33**	**0.554**	**0.628**	**+27.1%**
Lower East Branch	E-41	0.492	0.550	+27.5%
Lower East Branch	E-30	0.414	0.451	+27.9%
Lower Rivers	E-26	0.389	0.419	+28.1%
Lower West Branch	**W-12**	**0.491**	**0.553**	**+29.5%**
Lower West Branch	W-9	0.394	0.426	+29.2%
Lower West Branch	W-6	0.364	0.388	+29.1%
Inlet	N-12	0.329	0.342	+28.8%

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.

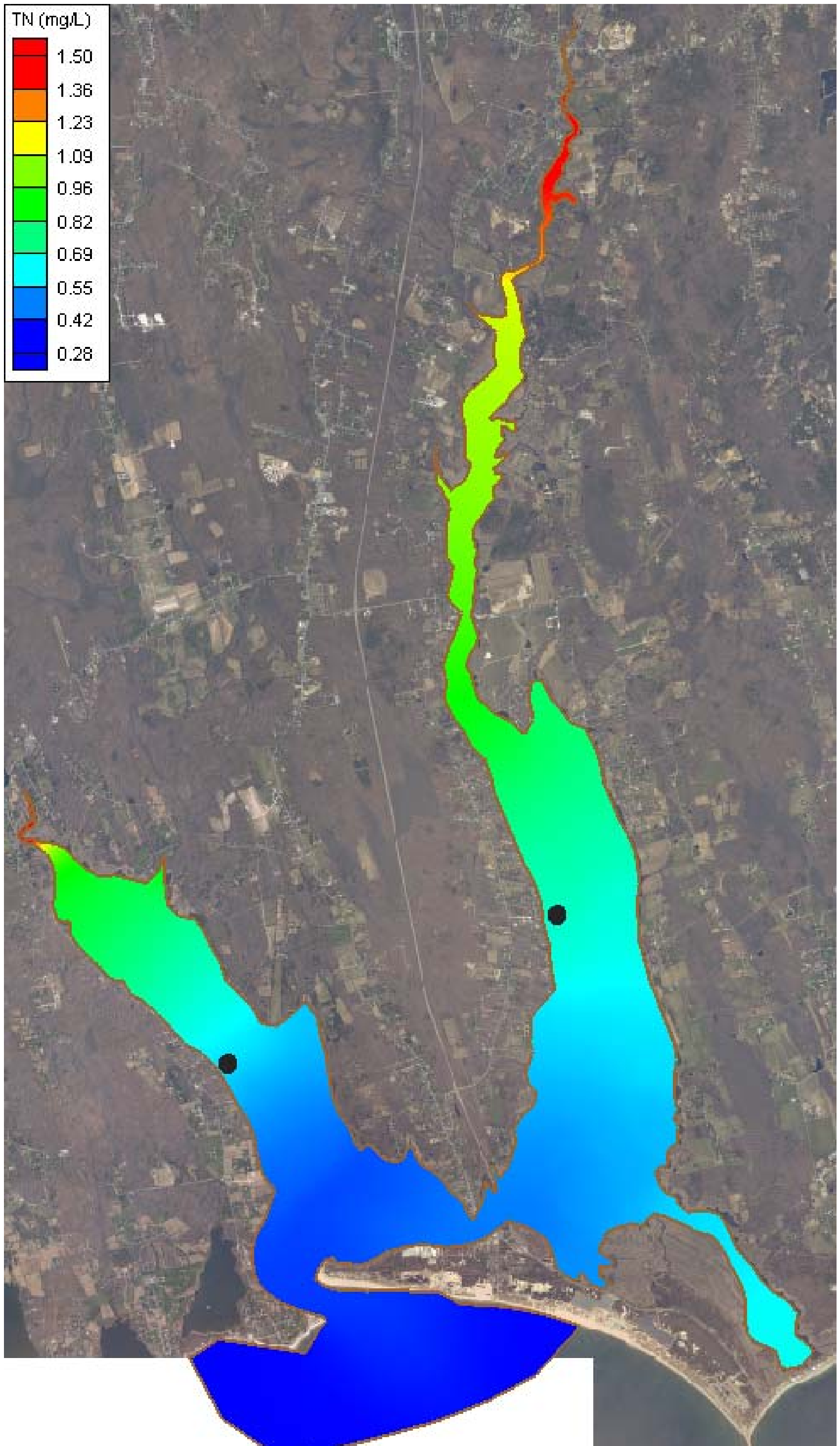


Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Westport River estuary system, for projected build-out loading conditions, and bathymetry. The approximate locations of the sentinel threshold stations for Westport River estuary system are shown by the black symbols (W12 for the West Branch and E33 for the East Branch).

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of Westport River estuary 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)
North East Branch	7.071	4.36	-15.89
West Branch	3.416	11.15	-4.67
South East Branch	29.471	20.92	-7.31
The Let	2.630	1.97	9.05
Westport Harbor	1.422	8.23	-25.45
Old County Road	63.805	-	-
Kirby Brook	5.986	-	-
Adamsville Brook	18.721	-	-
Angeline Brook	5.477	-	-
Snell Creek	1.866	-	-

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., Buzzard's Bay) 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 of 72% to almost 82%. Results for the 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 Westport River estuary system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). The sentinel threshold stations are in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Head Westport	N-0	1.344	0.526	-77.1%
Upper East Branch	N-1	0.919	0.400	-81.5%
Upper East Branch	N-2	0.879	0.392	-81.6%
Upper East Branch	N-3	0.855	0.389	-81.4%
Mid East Branch	N-4	0.798	0.383	-80.5%
Mid East Branch	E-69	0.735	0.375	-79.6%
Mid East Branch	E-56	0.616	0.358	-77.3%
Lower East Branch	**E-33**	**0.554**	**0.346**	**-76.5%**
Lower East Branch	E-41	0.492	0.334	-75.3%
Lower East Branch	E-30	0.414	0.316	-74.4%
Lower Rivers	E-26	0.389	0.310	-74.1%
Lower West Branch	**W-12**	**0.491**	**0.341**	**-71.7%**
Lower West Branch	W-9	0.394	0.313	-72.6%
Lower West Branch	W-6	0.364	0.304	-73.0%
Inlet	N-12	0.329	0.294	-73.8%

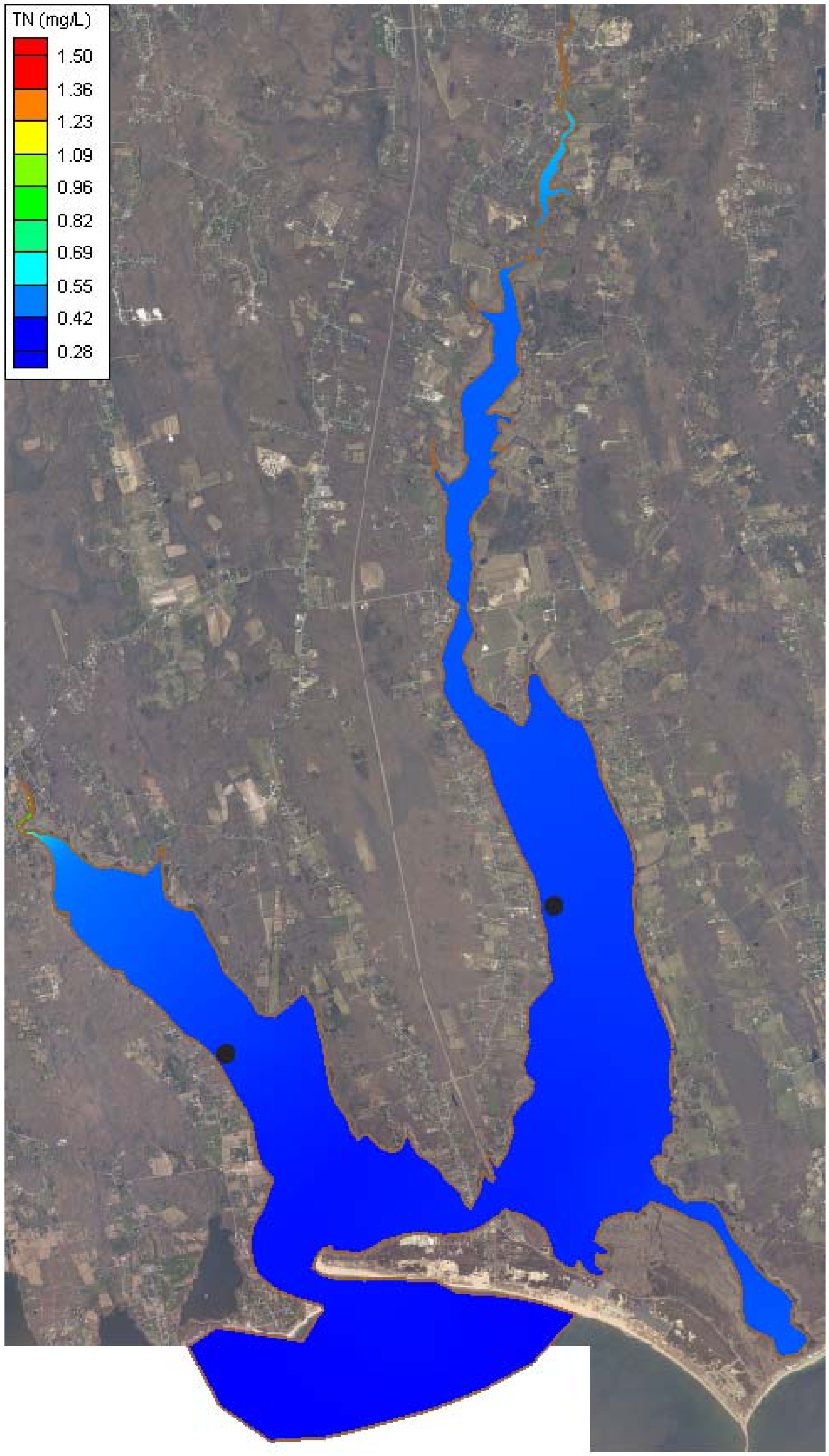


Figure VI-8. Contour plots of modeled total nitrogen concentrations (mg/L) in Westport River estuary system, for no anthropogenic loading conditions, and bathymetry. The approximate locations of the sentinel threshold stations for Westport River estuary system are shown by the black symbols (W12 for the West Branch and E33 for the East Branch).

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Westport River embayment system in the Town of Westport, MA, our assessment is based upon: 1) data from the water quality monitoring database developed by the BayWatchers (Coalition for Buzzards Bay and the Westport River Watershed Alliance) and MEP dissolved oxygen records obtained during the summers of 2006 and 2007, 2) surveys of eelgrass distribution, and 3) autumnal surveys of benthic animal communities and sediment characteristics. These data form the basis of a site-specific assessment of the present ecological health of the Westport River Estuary and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for this system (Chapter VIII). It should be noted that nitrogen enrichment occurs through 2 primary mechanisms, high rates of nitrogen entering from the surrounding watershed and/or less than optimal rates of flushing due to restriction of tidal exchange with the low nitrogen waters of Buzzards Bay / Rhode Island Sound. The Westport River has increasing nitrogen loading from its watershed from shifting land-uses. Fundamentally, restrictions of tidal exchange increase the sensitivity of an estuary to nitrogen inputs. As such, maximizing tidal exchange and circulation within the estuary should be a part of any planning for managing nitrogen enrichment.

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen threshold determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed eight (8) autonomous dissolved oxygen sensors in the Westport River system at locations that would be representative of the dissolved oxygen condition at critical points in the system, namely the uppermost portions of both the east and west branches as well as The Let. These are all areas furthest removed from the influence of inflowing waters from Buzzards Bay / Rhode Island Sound. The dissolved oxygen moorings were deployed to record the frequency and duration of low oxygen conditions during the critical summer period. The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within the Westport River system was conducted for comparison to historic records (MassDEP Eelgrass Mapping

Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. Analysis of inorganic N/P molar ratios within the watercolumn of the Westport River support this contention that nitrogen is the nutrient to be managed, as the ratio in the Westport River (<7 in upper East Branch, <5 throughout other basins) is far below the Redfield Ratio value (16) indicating that nitrogen additions will increase phytoplankton production as well as organic matter levels and turbidity within this system. Increased phytoplankton and organic matter levels increase oxygen consumption within the waters and sediments and increase the extent of oxygen depletion and habitat impairment. Within the Westport River system, temporal changes in eelgrass distribution provided a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing from a newly dredged inlet) in nutrient enrichment.

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 4 mg L^{-1}. Massachusetts State Water Quality Classification indicates that SA (high quality) waters be able to maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the east branch of the Westport River embayment are currently listed under the State Classification as SB (the west branch is classified as SA), however, the MEP analysis considered historic presence of eelgrass habitat and restoration of eelgrass lost over the last several decades due to nutrient over-enrichment as well as the typical recreational use of the East Branch as an estuarine water body, as it evaluated the dissolved oxygen data. The analysis of oxygen data was undertaken consistent with an SA classification as a goal for restoration as that is more conservative. It should be noted that the Classification represents the water quality that the embayment should support, not the existing level of water quality and that it is the designated water quality that is the target of TMDL's generated under the U.S. Clean Water Act. It is through the MEP and TMDL processes that site specific management targets are developed and under a Town's Comprehensive Wastewater Management Plan (CWMP) that management alternatives are designed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Westport River system (Figure VII-2). The dissolved oxygen sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Westport River system was collected during the summer of 2006 and 2007. Three moorings (DO2, DO3 and DO8) had to be re-deployed in the summer of 2007 due to instrument failure or vandalism.

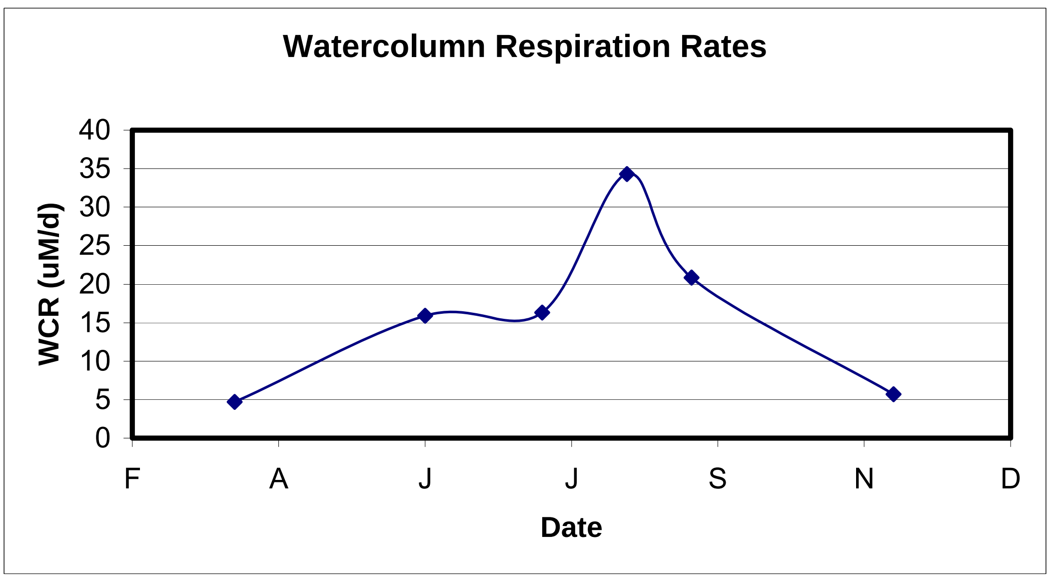


Figure VII-1. Example of typical average water column respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System, Cape Cod (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability. Similar to other embayments in southeastern Massachusetts, the Westport River system evaluated in this assessment showed high frequency variation, apparently related to diurnal and tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration at each mooring site, underscores the need for continuous monitoring within these systems.

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 28 – 43 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms.

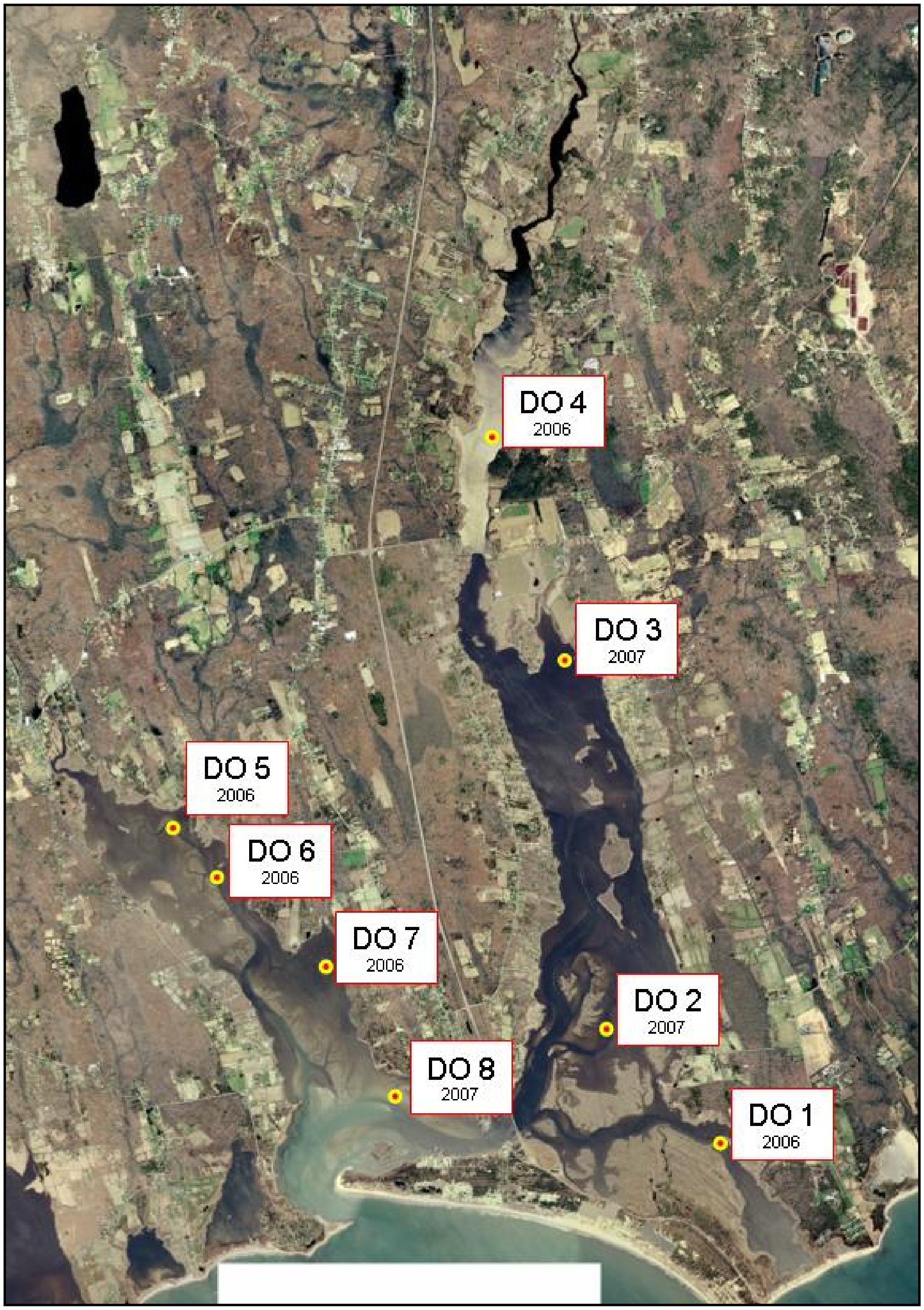


Figure VII-2. Aerial Photograph of the Westport River system (East and West Branches) in the Town of Westport showing the location of the continuously recording Dissolved Oxygen / Chlorophyll-a sensors deployed during the Summer of 2006 and 2007. All sensors were deployed in 2006, however, a few had to be redeployed in 2007 (DO2,3,8) due to instrument malfunction or vandalism.

However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate moderate to significant nutrient enriched waters within the upper portions (west and east branch respectively) of the Westport River system (Figures VII-3 through VII-15). The oxygen data is consistent with organic matter enrichment, primarily from phytoplankton production as seen from the parallel measurements of chlorophyll a. The measured levels of oxygen depletion and enhanced chlorophyll a levels follows the spatial pattern of total nitrogen levels in this system (Chapter VI), and the parallel variation in these water quality parameters is consistent with watershed based nitrogen enrichment of this estuarine system.

The oxygen record for the upper and mid east branch of the Westport River (DO3 and DO4) show that these basins have large daily oxygen excursions, consistent with their significant level of nitrogen enrichment. The levels of oxygen depletion parallel the level of nitrogen enrichment with the lower East Branch showing higher oxygen levels. However, The Let (DO1) showed oxygen depletions similar to the mid reach (DO4), greater than the lower basin of the East Branch. However, The Let functions primarily as a salt marsh pond, and as such is naturally organic matter enriched. The observed levels of oxygen depletion, although moderate for an open water embayment, are typical of salt marsh ponds and therefore do not indicate impairment of this basin. The West Branch shows a similar gradient in oxygen depletion as the East Branch, but as it is less nitrogen enriched, the levels of depletion are smaller and less frequent than the East Branch (DO5 and DO 6 versus DO7 and DO8). 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). Measured dissolved oxygen depletion and clear evidence of oxygen levels above atmospheric equilibration indicates that the upper portion of the west and east branch of the Westport River system show moderate to significant oxygen stress, respectively. The embayment specific results are as follows:

Westport DO/CHLA Mooring 1 – East Branch (Figures VII-3 and VII-4):

The DO1 instrument mooring was located at the southern end of the east branch of the Westport River system in an area dominated by salt marsh and commonly referred to as The Let. This mooring, while in the lower part of the system is still ~ 5.4 km from the inlet to the overall Westport River system. The DO1 mooring was centrally located in the shallow tidal channel but not in the terminal shallow basin of The Let. Modest daily excursions in oxygen levels were observed at this location, ranging from levels at air equilibration (7-8 mg L^{-1}) to hypoxic conditions where levels frequently decline to < 5 mg L^{-1} and to 4 mg L^{-1} (Figure VII-3, Table VII-1). The level of oxygen depletion is typical of salt marsh basins. Salt marshes are naturally nutrient and organic matter enriched and as such have significant oxygen depletion.

Oxygen in salt marsh channels may frequently become anoxic at night due to the high respiration rates stemming from the large amounts of organic matter being produced and deposited within the sediments.

The extensive eelgrass coverage in The Let is consistent with the moderate oxygen depletion, but not anoxia, in this basin and the high water clarity resulting from the low phytoplankton biomass. Over the 29 day deployment there were no phytoplankton blooms with chlorophyll-a levels generally between 2-8 ug L^{-1} (average 5.4 ug L^{-1}). Oxygen and chlorophyll levels are clearly indicative of high water quality conditions for this type of estuarine basin. Chlorophyll-*a* levels exceeded the 10 ug L^{-1} benchmark only 2% of the record (Table VII-2, Figure VII-4). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

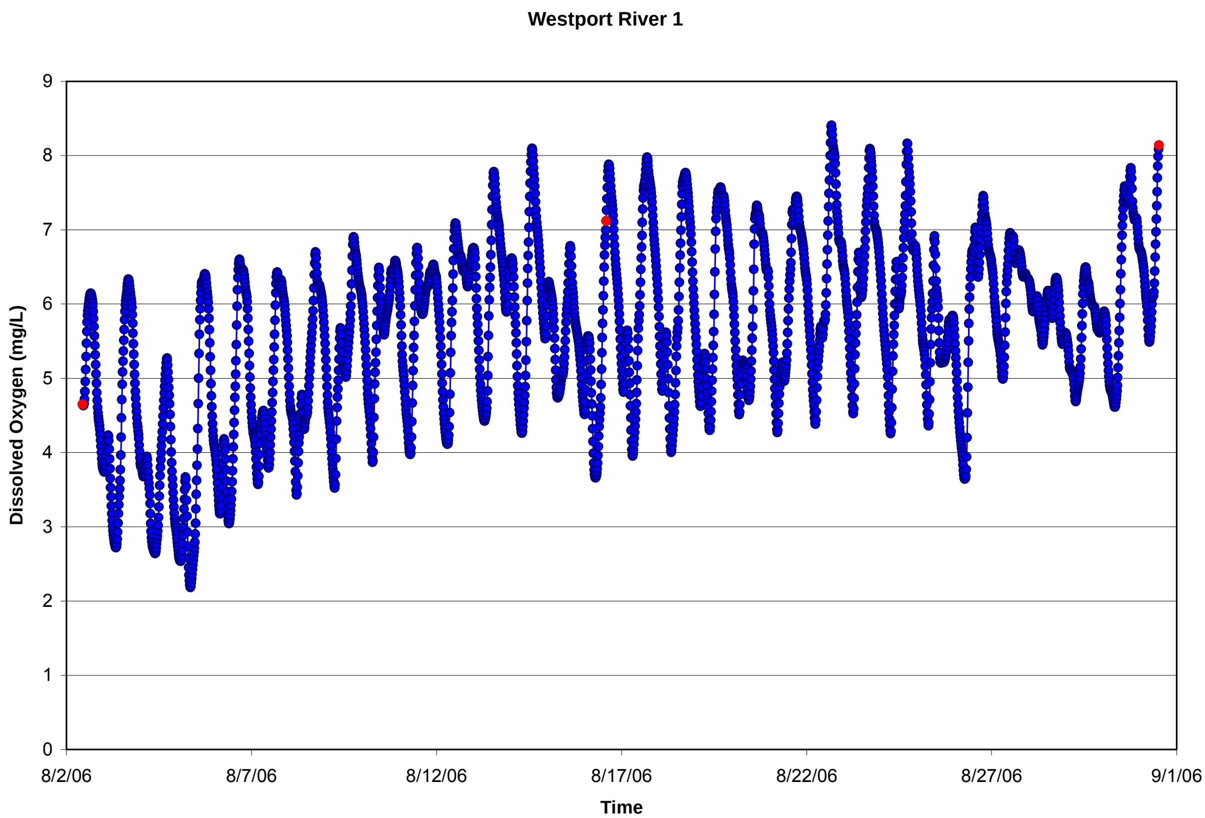


Figure VII-3. Bottom water record of dissolved oxygen at the Westport River 1 station (East Branch-lower), Summer 2006. Calibration samples represented as red dots.

Westport DO/CHLA Mooring 2- East Branch (Figures VII-5 and VII-6):

The DO2 instrument mooring was located at the lower reach of the East Branch of the Westport River Estuary in a tidal channel that flows through the salt marsh islands, creating ecological characteristics similar to those within The Let. The mooring was located up-gradient of the Route 88 bridge crossing to Horseneck Beach. As with the record obtained from the mooring in the Let, moderate daily excursions in oxygen levels were observed at this location, although with a smaller range. Oxygen generally ranged from levels at air equilibration (7-8 mg L^{-1}) to 5 mg L^{-1} and only once did oxygen decline to 4 mg L^{-1}. During approximately a three day period oxygen did approach 10 mg L^{-1}, however that was not characteristic of oxygen conditions during most of the deployment (Figure VII-5, Table VII-1). The smaller range in daily excursions may be the result of the sandy sediments containing little organic matter as well as the strong tidal flow in this part of the system.

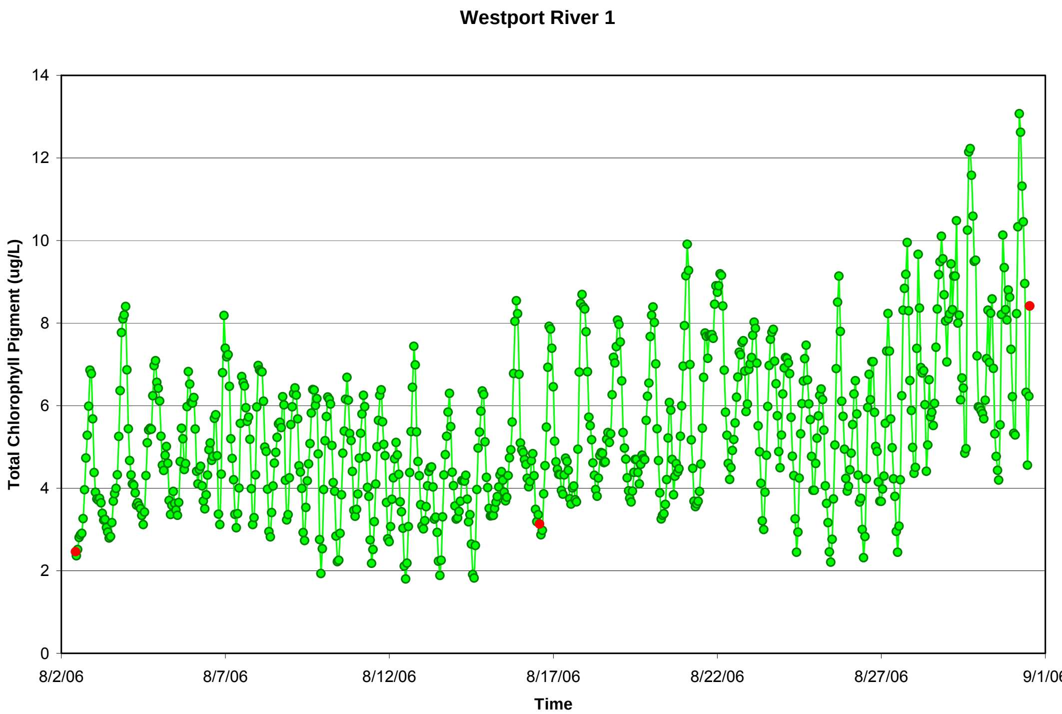


Figure VII-4. Bottom water record of Chlorophyll-*a* in the Westport River 1 station (East Branch-lower), Summer 2006. Calibration samples represented as red dots.

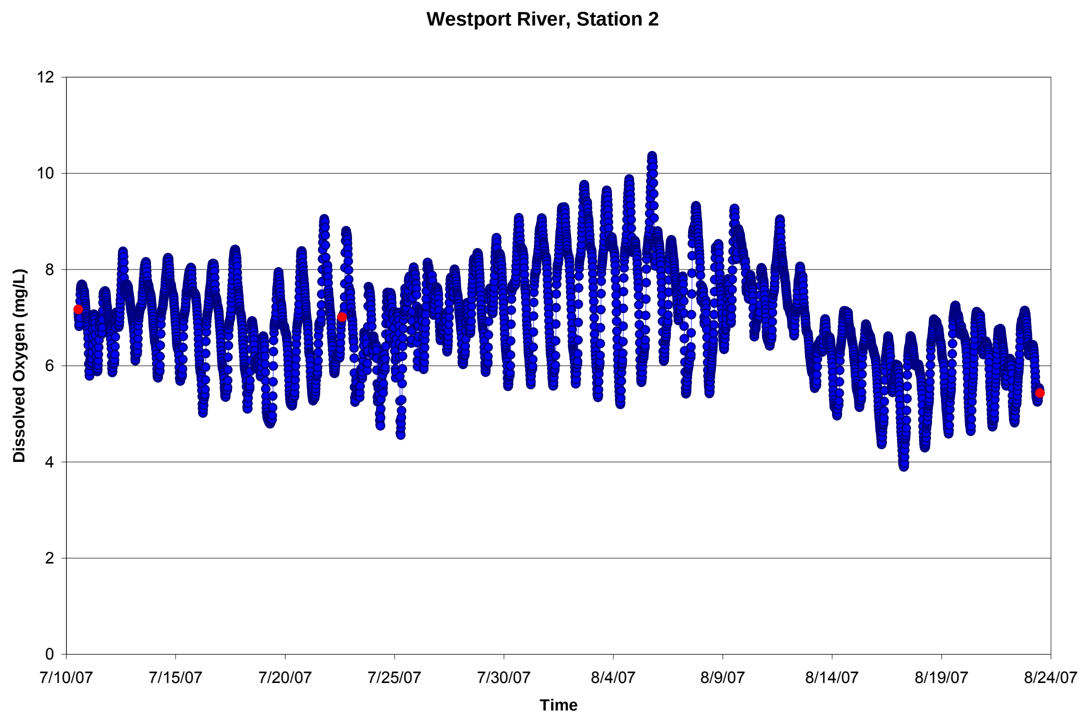


Figure VII-5. Bottom water record of dissolved oxygen at the Westport River 2 station (East Branch-mid), Summer 2007. Calibration samples represented as red dots.

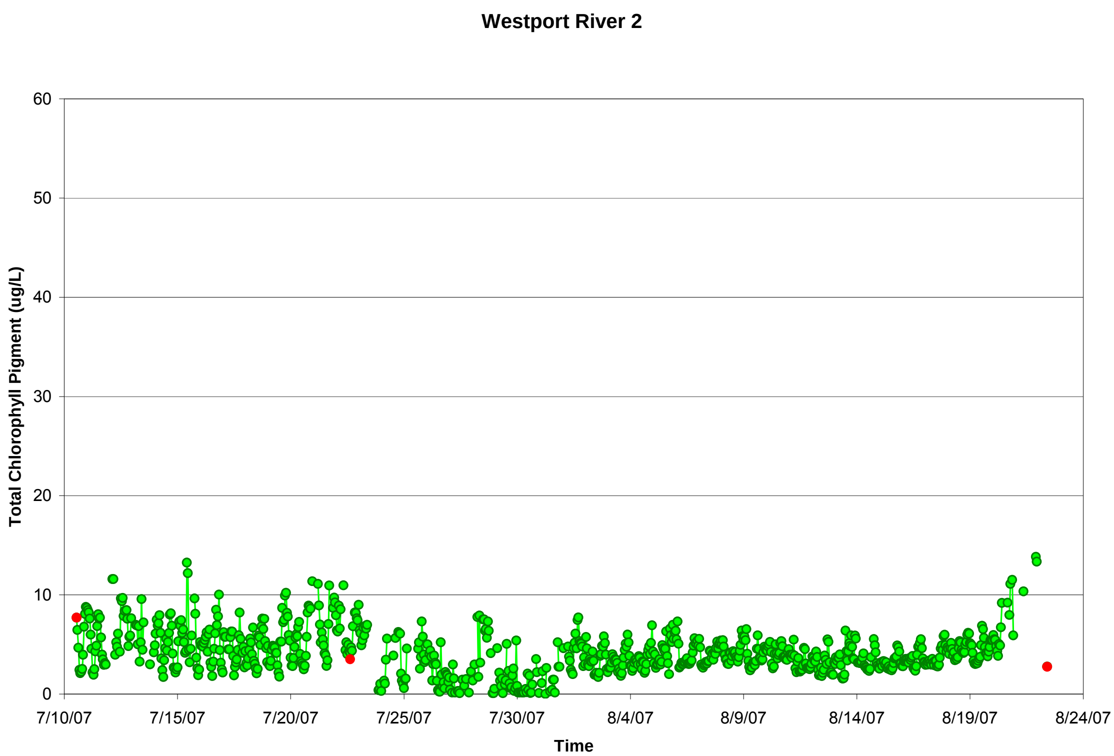


Figure VII-6. Bottom water record of Chlorophyll-*a* in the Westport River 2 station (East Branch-upper), Summer 2007. Calibration samples represented as red dots.

Over the 36 day deployment there were no phytoplankton blooms with chlorophyll-a levels generally between 2-8 ug L^{-1} (average 7.7 ug L^{-1}). Oxygen and chlorophyll levels are clearly indicative of high water quality conditions. Chlorophyll-*a* levels exceeded the 10 ug L^{-1} benchmark only 2% of the record (Table VII-2, Figure VII-6). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

Westport DO/CHLA Mooring 3- East Branch (Figures VII-7 and VII-8):

The DO3 instrument mooring was located within the mid reach of the East Branch of the Westport River (~ 1.5 km south of the Hix Bridge Road crossing) in the vicinity of a shallow cove located at the level where the east branch narrows significantly towards the head of the system. The mooring was located well away from the inlet (~7.8 km) connecting the estuary to waters from Buzzards Bay/Rhode Island Sound. Large daily excursions in oxygen levels were observed at this location compared to the oxygen records from moorings positioned lower in the east branch. Excursions ranged from levels at or above air equilibration to hypoxic conditions were frequent. Significant oxygen depletions periodically occurred with levels frequently declining to 3 mg L^{-1} and sometimes to 2 mg L^{-1} (Figure VII-7, Table VII-1). The oxygen dynamics indicate moderate to significant organic enrichment of this basin. The level and frequency of oxygen depletion presents a stress to benthic animal communities within this portion of the estuary. Oxygen levels regularly exceeded 8 mg L-1 and periodically exceeded 9 mg L-1.

Over the 44 day deployment there does appear to be a clear rise in the level of phytoplankton as represented by chlorophyll values which appear to increase approximately 1/3 and then decrease around 2/3 of the way through the deployment. During the most intense period of phytoplankton activity, chlorophyll-a values ranged from between 20 and 25 ug L^{-1}. Chlorophyll levels at this location in the east branch are clearly indicative of impaired conditions and nitrogen enrichment. Chlorophyll a averaged 13.1 ug L^{-1}, with levels exceeded the 10 and 20 ug L^{-1} benchmarks 67% and 11% of the time respectively (Table VII-2, Figure VII-8). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate nitrogen enriched conditions in open water embayments.

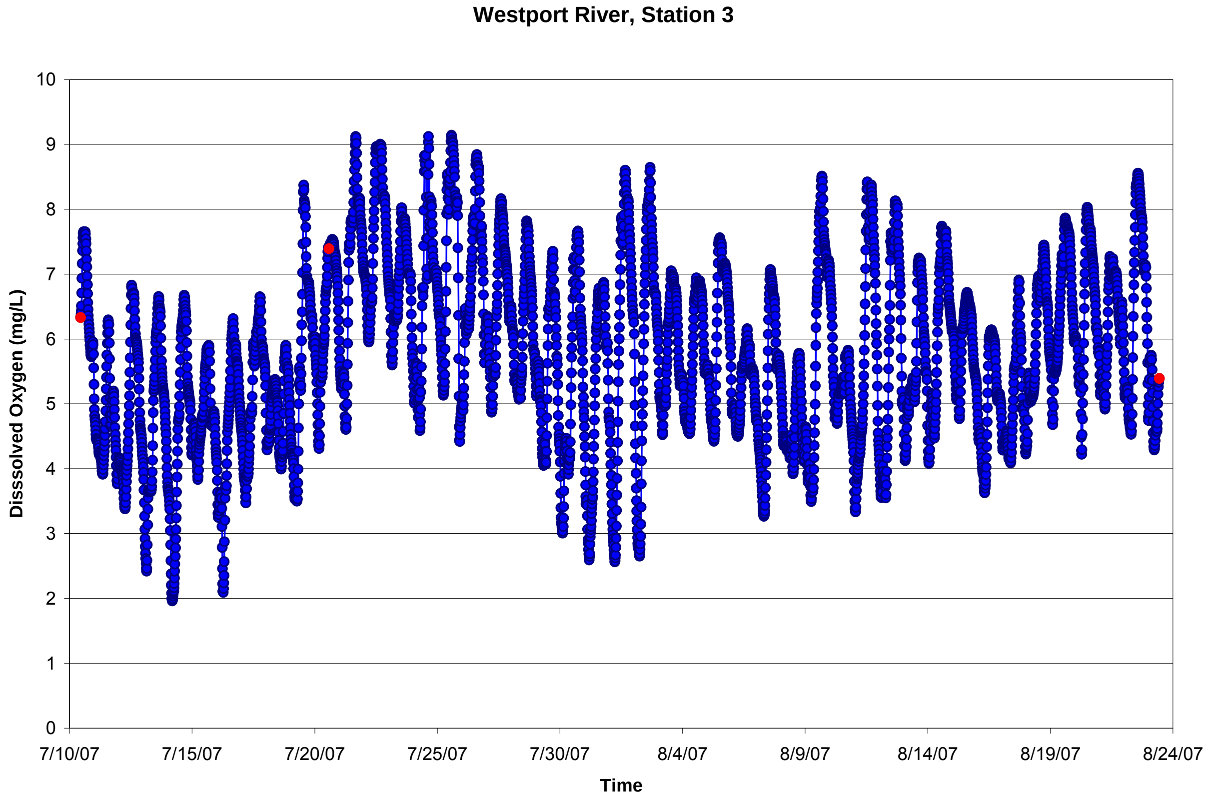


Figure VII-7. Bottom water record of dissolved oxygen at the Westport River 3 station (East Branch-mid), Summer 2007. Calibration samples represented as red dots.

Westport DO/CHLA Mooring 4- East Branch (Figures VII-9 and VII-10):

The DO4 instrument mooring was the northern most mooring location in the east branch of the Westport River estuarine system. The mooring was centrally located in the tidal river ~1.2 km up-gradient from the Hix Bridge Road crossing and ~ 3.7 km down gradient from the head of the east branch of the Westport River. This mooring was also approximately 10.5 km from the inlet connecting the estuarine waters of the upper estuary to low nutrient waters entering the system from Buzzards Bay/Rhode Island Sound. As was measured at the DO3 mooring location, large daily excursions in oxygen levels were observed at this location, ranging from levels at or above air equilibration to stressful oxygen conditions where levels frequently decline to 4 mg L^{-1} and even drop to near 3 mg L^{-1} (Figure VII-9, Table VII-1). The organic enrichment of this portion of the system is demonstrated by the steadily growing algal bloom observed during the deployment period.

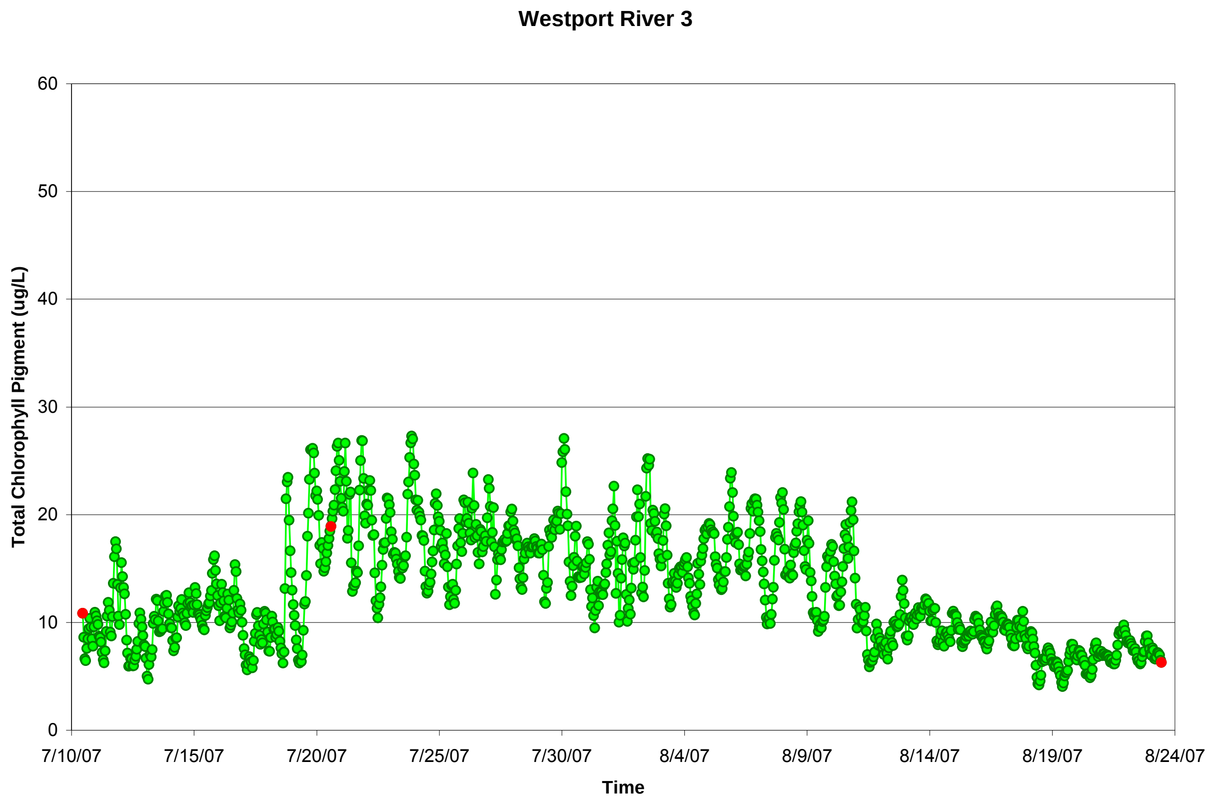


Figure VII-8. Bottom water record of Chlorophyll-*a* in the Westport River 3 station (East Branch-upper), Summer 2007. Calibration samples represented as red dots.

Oxygen levels regularly persisted between 7-8 mg L^{-1} but rarely exceeded 8 mg L^{-1}. These oxygen levels are likely the result of the combined effects of photosynthesis by high levels of phytoplankton present in this area during the deployment and the relatively shallow water that is easily mixed during windy days. Over the 28 day deployment there does appear to be a relatively high base level of phytoplankton represented by chlorophyll values that range between 10 and 20 ug L^{-1} during the first part of the deployment and steadily increase to very high chlorophyll values in the second half of the deployment. In the last few days of the deployment chlorophyll values ranged between 30 and 55 ug L^{-1}. Oxygen and chlorophyll levels at this location in the east branch are clearly indicative of impaired conditions and nitrogen enrichment (mooring chlorophyll average 20.2 ug L^{-1}) and chlorophyll-*a* levels exceeded the 10 and 20 ug L^{-1} benchmarks 95% and 40% of the time respectively (Table VII-2, Figure VII-10). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

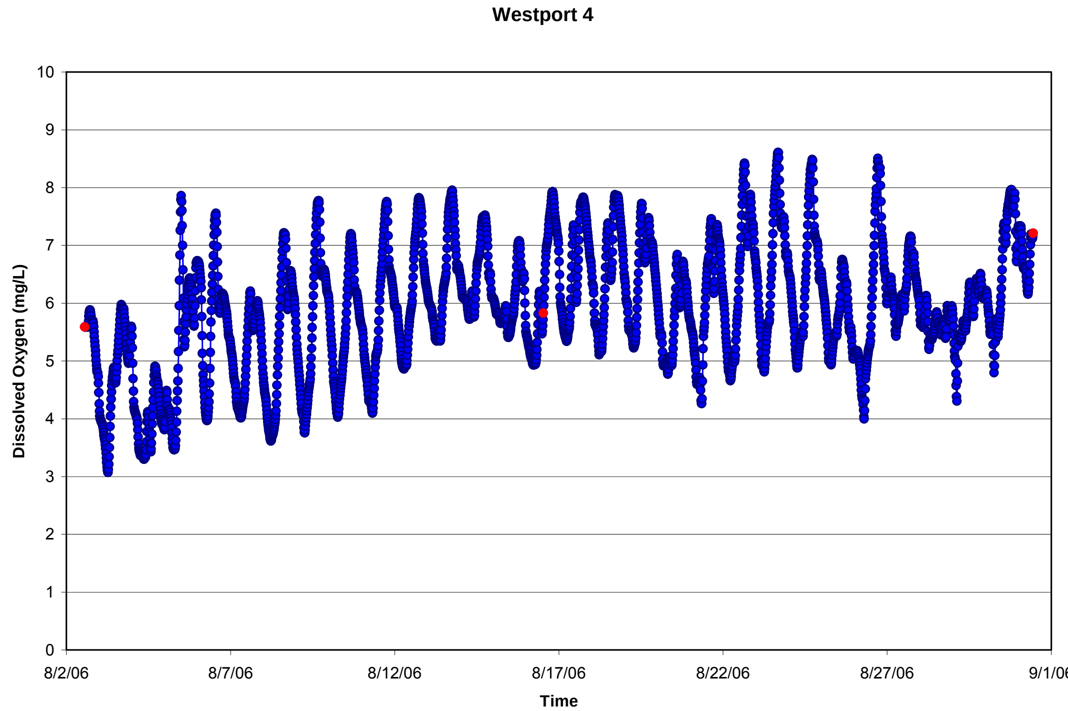


Figure VII-9. Bottom water record of dissolved oxygen at the Westport River 4 station (East Branch-upper), Summer 2006. Calibration samples represented as red dots.

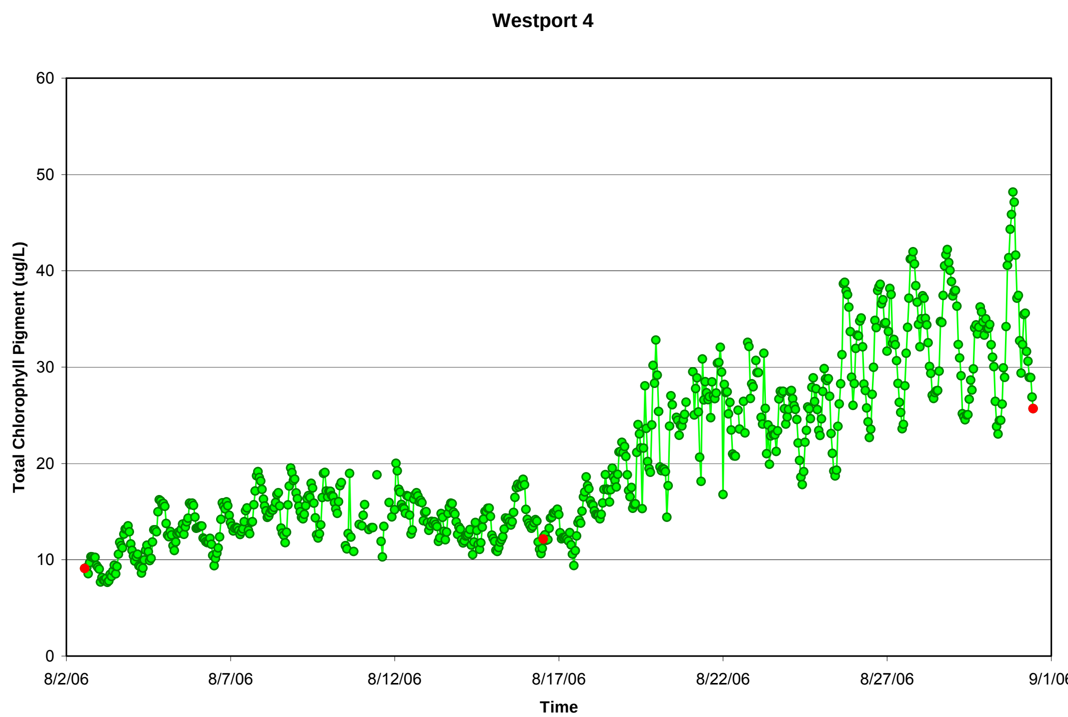


Figure VII-10. Bottom water record of Chlorophyll-*a* in the Westport River 4 station (East Branch-upper), Summer 2006. Calibration samples represented as red dots.

Westport DO/CHLA Mooring 5- West Branch (Figures VII-11 and VII-12):

The DO5 instrument mooring was the northern most mooring location in the west branch of the Westport River estuarine system. The position of the mooring was dictated by the location of the terminal end of the tidal channel as there were few areas in this part of the West Branch that had sufficient water depth for deployment of the meter such that it remained submerged at all stages of the tide. The DO5 mooring was situated approximately 800 meters from where Adamsville Brook discharges into the head of the west branch of the Westport River estuary and approximately 4.3 km from the inlet connecting the estuarine waters of the upper estuary to low nutrient waters entering the system from Buzzards Bay/Rhode Island Sound. This region of the West Branch supports significant salt marshes. Oxygen dynamics at this site were intermediate between the salt marsh pond, The Let, and the salt marsh influenced lower reach of the East Branch. DO5 recorded large daily excursions in oxygen levels, ranging from levels at or above air equilibration and frequent declines to 4 mg L^{-1} (Figure VII-11, Table VII-1).

Oxygen levels regularly persisted between 7-8 mg L^{-1} and occasionally exceeded 10 mg L^{-1}. These oxygen levels are likely the result of the combined effects of photosynthesis by moderate levels of phytoplankton present in this area and the patches of algae mat cover in the immediate vicinity of the mooring. Over the 28 day deployment there was a steady rise in the level of phytoplankton biomass, which appears to peak mid way through the deployment and then decline. During the most intense period of phytoplankton activity, chlorophyll-a values ranged widely from between 5 and 35 ug L^{-1} over a one week period. It is likely that this high variability results in part from movement of water masses with the twice daily tides. Oxygen and chlorophyll levels at this location in the west branch are clearly indicative of nutrient enriched conditions impaired conditions and nitrogen enrichment with chlorophyll-*a* levels averaging 12.4 ug L^{-1} and exceeding the 10 and 20 ug L^{-1} benchmarks 56% and 16% of the time respectively (Table VII-2, Figure VII-12). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate nitrogen enriched conditions in open water embayments. Even accounting for the influence of the surrounding salt marshes, it appears that some impairment of this basin exists due to the high levels of chlorophyll a.

Westport DO/CHLA Mooring 6- West Branch (Figures VII-13 and VII-14):

The DO6 instrument mooring was slightly south of the northern most mooring location (DO5) also within the upper reach of the West Branch of the Westport River Estuary. As was the case for DO5, the position of the DO6 mooring was dictated by the location of the tidal channel to ensure that the sensors would remain submerged at all stages of the tide. The DO6 mooring was situated approximately 1.7 km from where Adamsville Brook discharges into the head of the west branch of the Westport River estuary and approximately 3.5 km from the inlet connecting the estuarine waters of the upper estuary to low nutrient waters entering the system from Buzzards Bay/Rhode Island Sound. Similar to the DO5 mooring location, large daily excursions in oxygen levels were observed at this location, ranging from levels at or above air equilibration to stressful oxygen conditions where levels frequently approached 4 mg L^{-1} (Figure VII-13, Table VII-1). The nitrogen enrichment of this portion of the estuary is demonstrated by the large algal bloom observed during the deployment period.

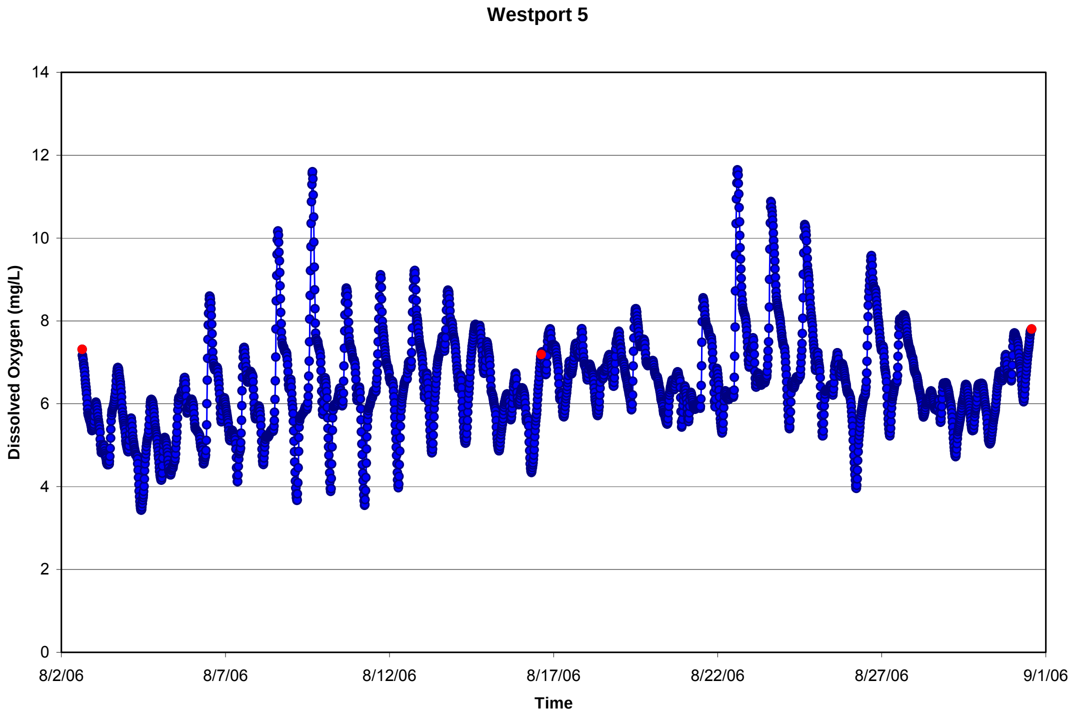


Figure VII-11. Bottom water record of dissolved oxygen at the Westport River 5 station (West Branch-upper), Summer 2006. Calibration samples represented as red dots.

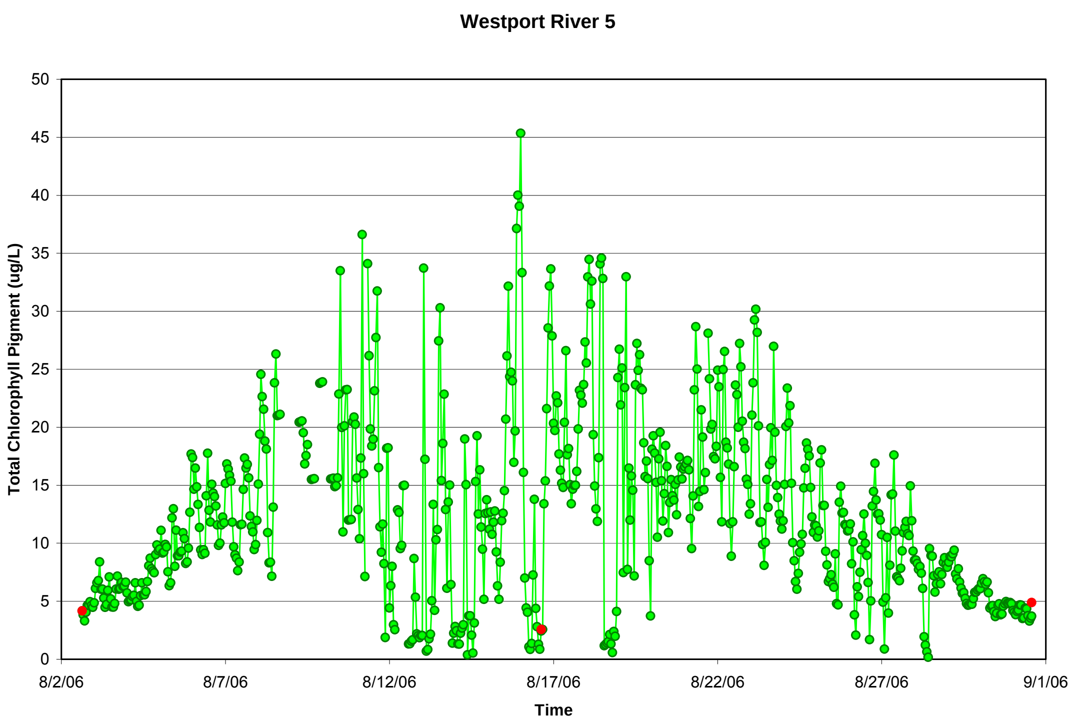


Figure VII-12. Bottom water record of Chlorophyll-*a* in the Westport River 5 station (West Branch-upper), Summer 2006. Calibration samples represented as red dots.

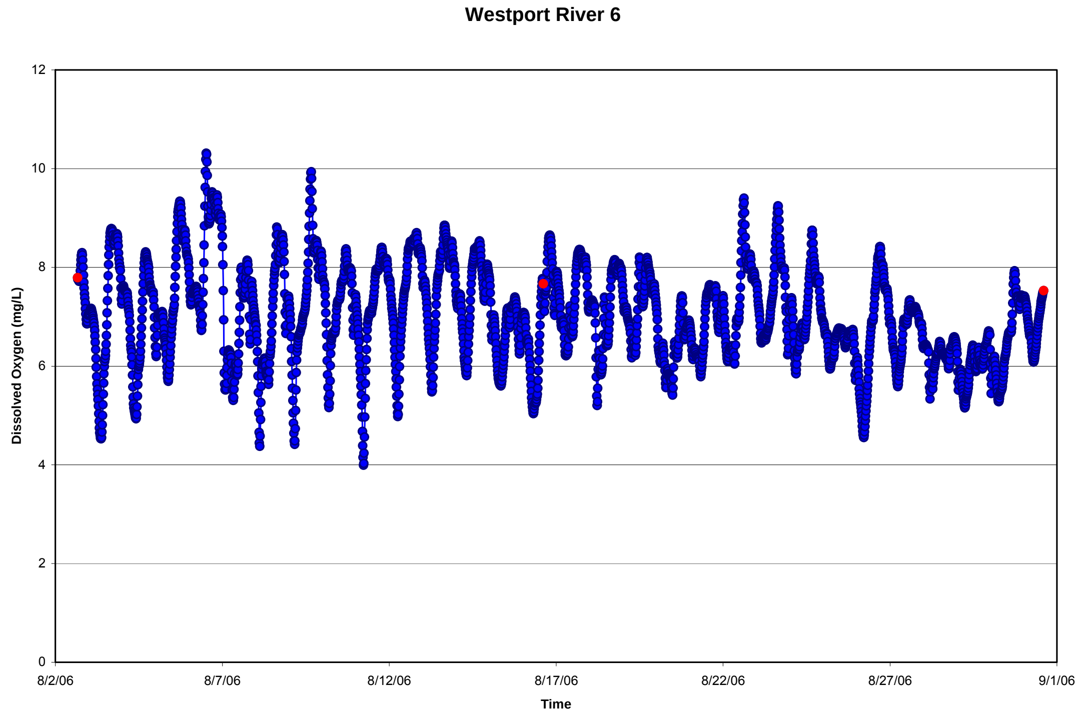


Figure VII-13. Bottom water record of dissolved oxygen at the Westport River 6 station (West Branch-mid), Summer 2006. Calibration samples represented as red dots.

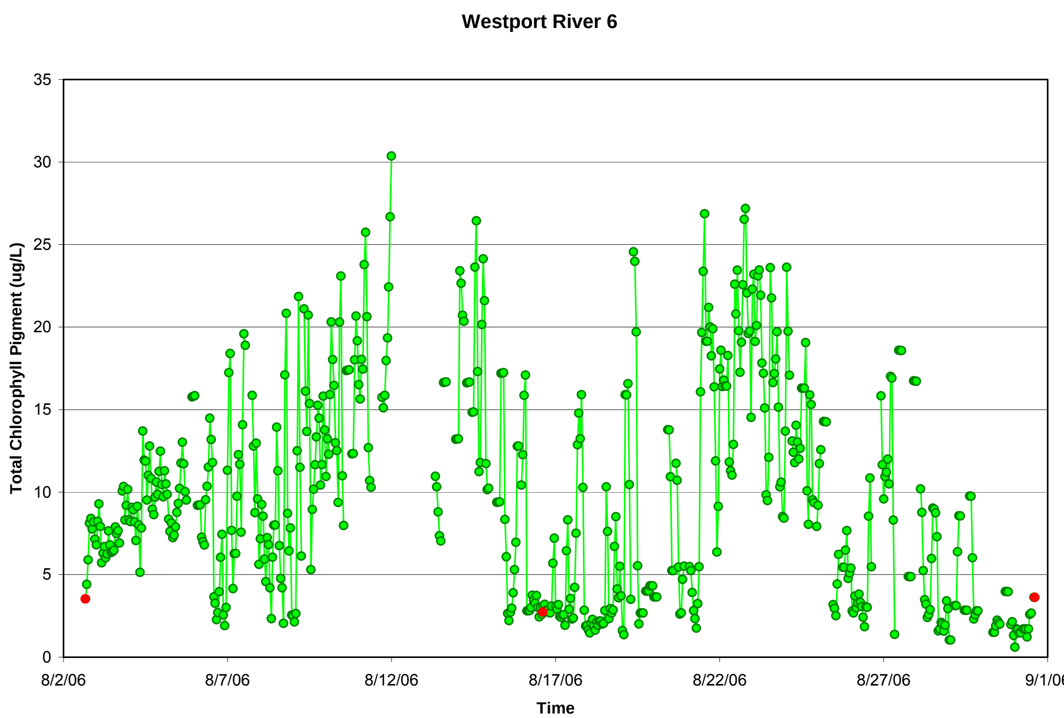


Figure VII-14. Bottom water record of Chlorophyll-*a* in the Westport River 6 station (West Branch-mid), Summer 2006. Calibration samples represented as red dots.

Oxygen levels regularly persisted between 7-8 mg L^{-1} and occasionally exceeded 9 mg L^{-1}. These oxygen levels are likely the result of the combined effects of photosynthesis by high levels of phytoplankton present in this area during the deployment and the dense macroalgae cover in the immediate vicinity of the mooring. Over the 28 day deployment there does appear to be two clear rises in the level of phytoplankton as represented by chlorophyll values which appear to peak approximately 1/3 and 2/3 of the way through the deployment with an obvious decrease in between. During the most intense period of phytoplankton activity, chlorophyll-a values ranged widely from between 5 and 25 ug L^{-1} during the first peak in phytoplankton activity and from 10 to 25 L^{-1} during the second peak in phytoplankton activity. Chlorophyll levels at this location in the west branch are clearly indicative of impaired conditions and nitrogen enrichment, even considering the influence of surrounding salt marshes. Chlorophyll a averaged 9.7 ug L^{-1}, with levels exceeded the 10 and 20 ug L^{-1} benchmarks 43% and 8% of the time respectively (Table VII-2, Figure VII-14). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate nitrogen enriched conditions in open water embayments.

Westport DO/CHLA Mooring 7- West Branch (Figures VII-15 and VII-16):

The DO7 instrument mooring was situated in the middle portion of the West Branch of the Westport River Estuary. The position of the mooring was towards the eastern edge of the west branch in the vicinity of Hicks Cove. The DO7 mooring was situated approximately 3 km down gradient from where Adamsville Brook discharges into the head of the west branch of the Westport River estuary and approximately 2.3 km from the inlet connecting the estuarine waters of the upper estuary to low nutrient waters entering the system from Buzzards Bay/Rhode Island Sound. As was measured at the DO5 and DO6 mooring locations, large daily excursions in oxygen levels were observed, ranging from levels at or above air equilibration to frequently 4 mg L^{-1} (Figure VII-15, Table VII-1). Nitrogen enrichment of this portion of the system is demonstrated by the soft organic enriched sediments and heavy epiphyte growth on remaining eelgrass plants that were documented in the vicinity of the DO7 mooring by MEP divers during the deployment period. However, oxygen levels were within the range anticipated for a shallow salt marsh influenced basin.

Oxygen levels regularly persisted between 7-8 mg L^{-1}, but consistently rose above 8 mg L^{-1} and occasionally exceeded 10 mg L^{-1}. Over the 29 day deployment there was a gradual increase in phytoplankton biomass, however, chlorophyll-a levels generally remained moderate to low, between 2-8 ug L^{-1}, except near the end of the deployment period when levels reach slightly above 10 ug L^{-1}. Chlorophyll levels are indicative of high water quality averaging 3.8 ug L^{-1} and exceeding the 10 ug L^{-1} benchmark <1% of the record (Table VII-2, Figure VII-16). Average chlorophyll-*a* levels over 10 ug L^{-1} have been used to indicate nitrogen enriched conditions in open water embayments.

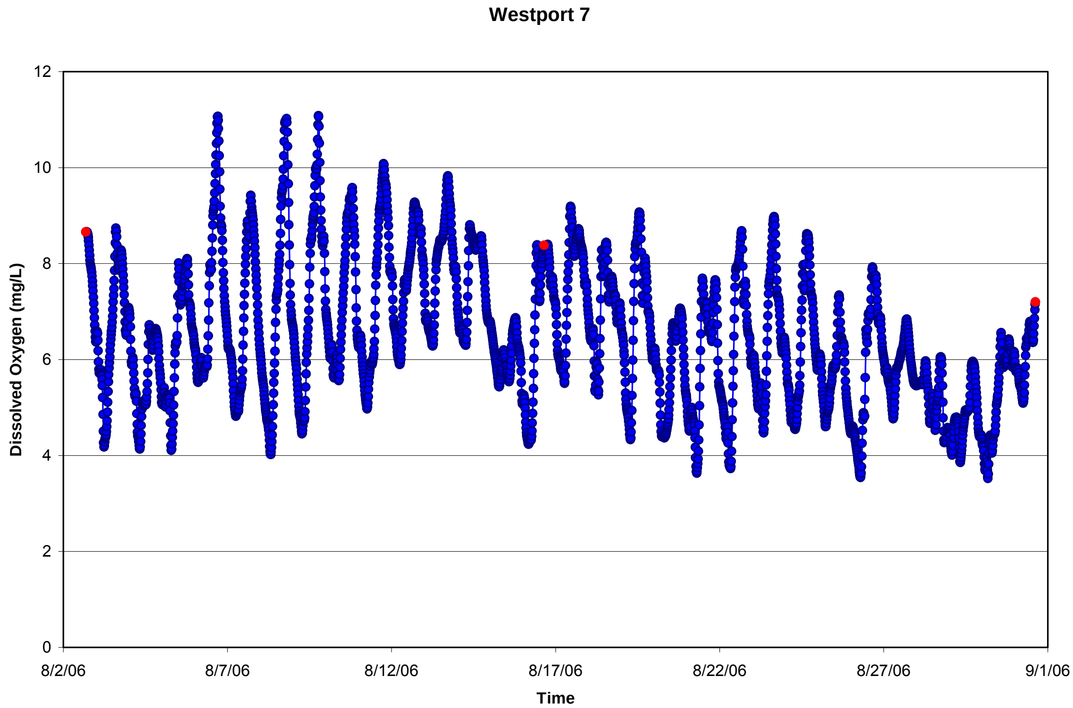


Figure VII-15. Bottom water record of dissolved oxygen at the Westport River 7 station (West Branch-mid), Summer 2006. Calibration samples represented as red dots.

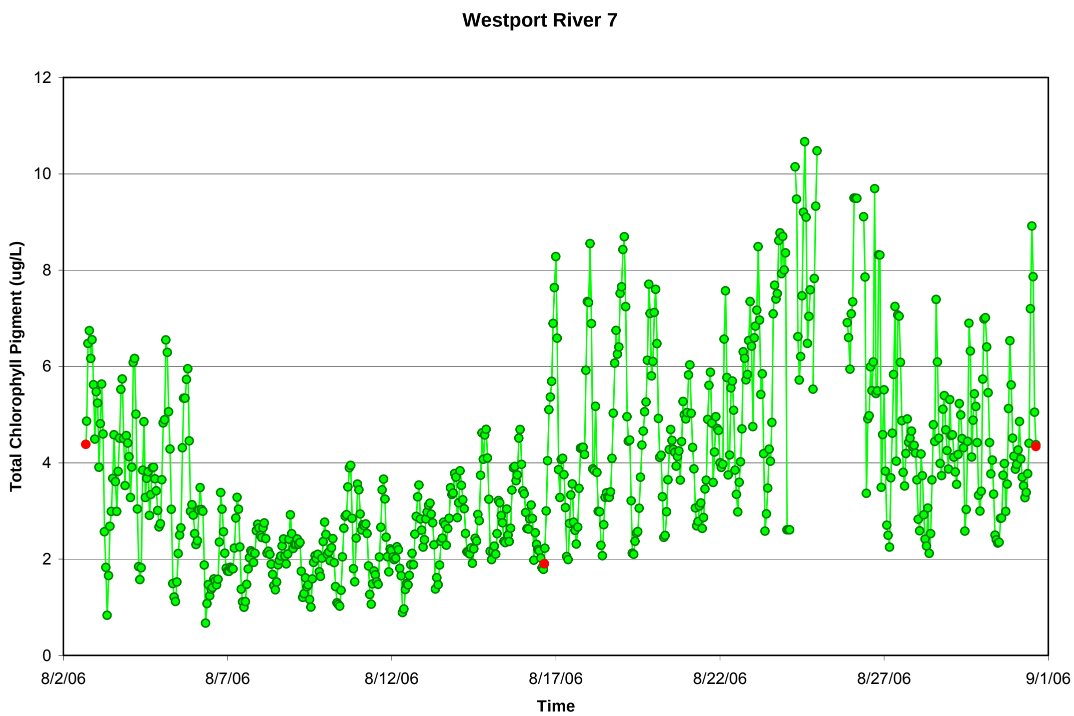


Figure VII-16. Bottom water record of Chlorophyll-*a* in the Westport River 7 station (West Branch-mid), Summer 2006. Calibration samples represented as red dots.

Westport DO/CHLA Mooring 8- West Branch (Figures VII-17 and VII-18):

The DO8 instrument mooring was situated within a small semi-confined basin at the lower reach of the west branch of the Westport River Estuary. The position of the mooring was towards the eastern edge of the west branch near, but separated from, Westport Harbor. As was measured by the other West Branch moorings, large daily excursions in oxygen levels were observed at this location, ranging from levels at or above air equilibration to stressful oxygen conditions where levels frequently decline to 4 mg L^{-1} (Figure VII-17, Table VII-1). The organic enrichment of this portion of the system is demonstrated by the organic enriched soft sediments at this site. From the basin configuration and circulation, it appears that this mooring is measuring "local" conditions and is not representative of the greater region of the lower West Branch. This conclusion is consistent with the generally low chlorophyll a levels (average 3.9 ug L^{-1}) yet the greatest oxygen depletion within the West Branch. Oxygen frequently decline to <4 mg L^{-1} and periodically to <3 mg L^{-1}. The oxygen levels would indicate significant impairment in this portion of the west branch, however, chlorophyll levels would be more indicative of high water quality. It is important to note that chlorophyll-*a* levels exceeded the 5 ug L^{-1} benchmark 33% of the time but did not exceed 10 ug L^{-1} (Table VII-2, Figure VII-18). Average chlorophyll-*a* levels over 10 ug L-1 have been used to indicate eutrophic conditions in embayments.

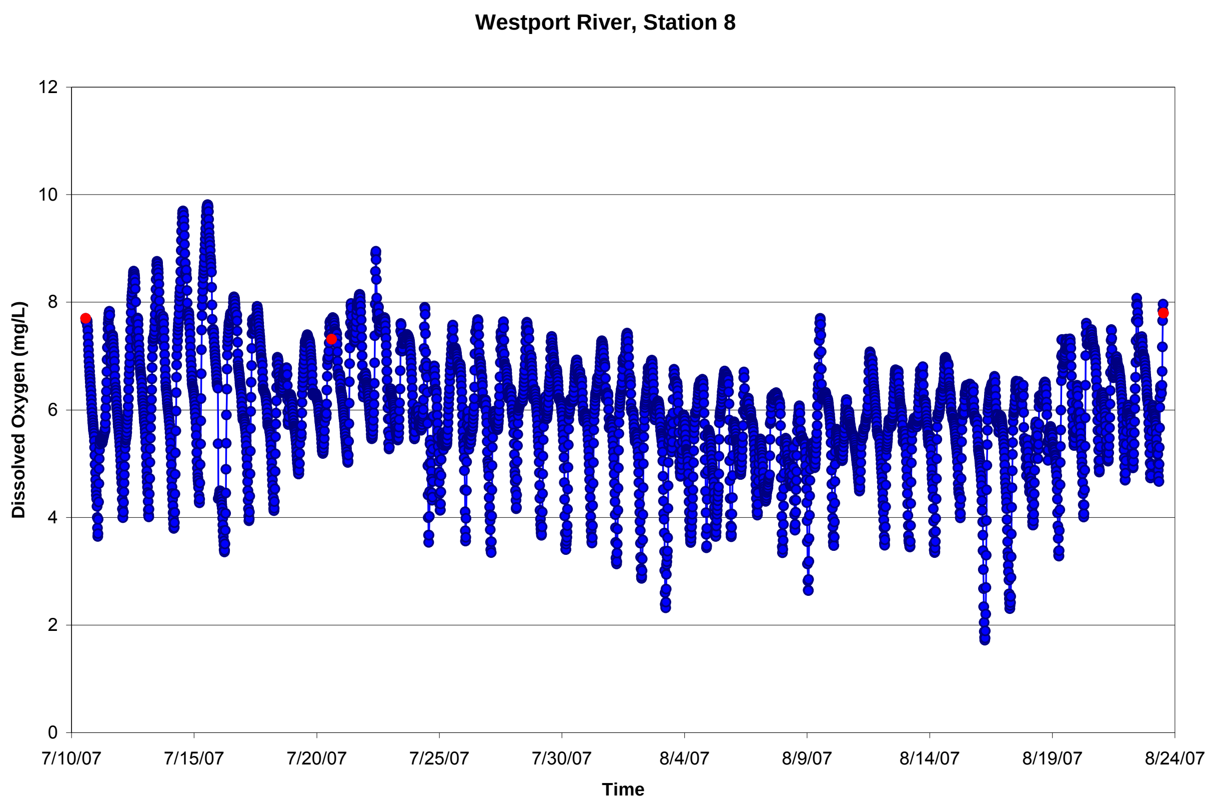


Figure VII-17. Bottom water record of dissolved oxygen at the Westport River 8 station (West Branch-lower but separate from Harbor), Summer 2007. Calibration samples represented as red dots.

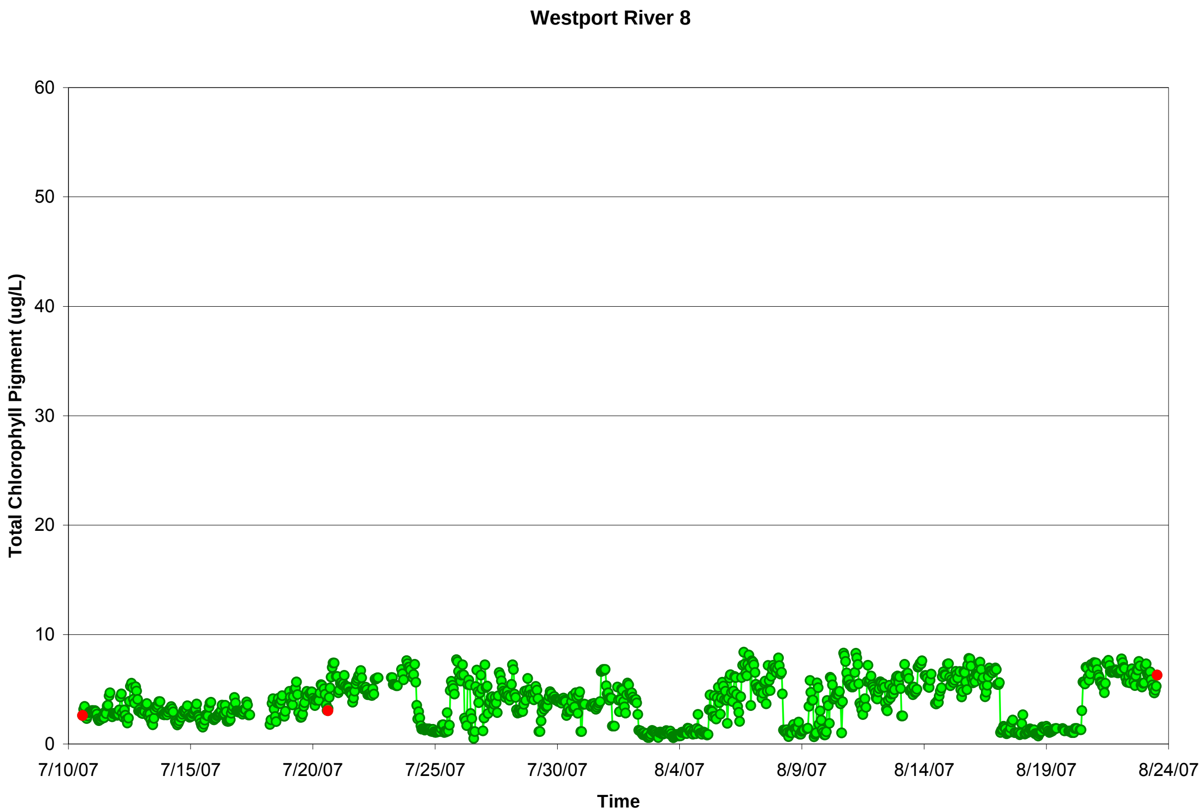


Figure VII-18. Bottom water record of Chlorophyll-*a* in the Westport River 8 station (West Branch-lower but separate from Harbor), Summer 2007. Calibration samples represented as red dots.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

For the Westport River Estuary, eelgrass distribution data was generated primarily by the MassDEP Eelgrass Mapping Program (C. Costello) through surveys undertaken in 1995, 2001 and 2006-07. Additional field verification was conducted by the SMAST/MEP Technical Team members during the conduct of MEP related field work in 2006. In the 2006 effort, eelgrass observations were made at 62 discrete points throughout the entire Westport River system. The MassDEP Eelgrass Mapping Program assembled the multi year survey data into its standard format for use in the MEP threshold analysis. Additional analysis by the MassDEP Eelgrass Mapping Program of available aerial photos from 1951 was used to reconstruct the eelgrass distribution prior to any substantial residential and commercial development of the watershed. The MassDEP evaluation of the Westport River 1951 aerial imagery determined it to be adequate for quantitative photo-interpretation of historic eelgrass distribution. In addition, survey work in 1984 (Costa, 1988) was assessed to help in the evaluation of the west branch eelgrass coverages. This 1984 data showed eelgrass coverage consistent with the 1951 and 1995 coverages develop by MassDEP, with low density beds and eelgrass epiphytes along the upper reach of the eastern shore and dense beds in the lower reach of the West Branch. The 1984 coverage did not extend as far up in the west branch as the 1951 MassDEP coverage but was more similar to the 1995 MassDEP eelgrass coverage. There was no 1984 coverage data collected in the East Branch. Based upon all of these results and the consistent analysis by MassDEP, the overall eelgrass analysis that follows is based on changes over the period 1951 – 2006/07 (Figure VII-19). The primary use of the data is to indicate: (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 2006; the period in which watershed nitrogen loading increased significantly. This temporal information is also used to determine the stability of the eelgrass community.

Table VII-1. Days and percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels within the East and West Branches of the Westport River embayment system.

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)
Westport River 1	**8/2/2006**	**8/31/2006**	**29.08**	**58%**	**29%**	**10%**	**3%**
			Mean	0.45	0.24	0.19	0.19
			Min	0.01	0.01	0.03	0.14
			Max	1.85	0.82	0.70	0.22
			S.D.	0.35	0.23	0.20	0.04
Westport River 2	**7/10/2007**	**8/23/2007**	**43.96**	**21%**	**4%**	**0%**	**0%**
			Mean	0.19	0.14	0.07	N/A
			Min	0.01	0.02	0.07	0.00
			Max	0.52	0.28	0.07	0.00
			S.D.	0.14	0.09	N/A	N/A
Westport River 3	**7/10/2007**	**8/23/2007**	**43.99**	**57%**	**31%**	**9%**	**2%**
			Mean	0.57	0.28	0.18	0.13
			Min	0.02	0.01	0.02	0.10
			Max	1.84	0.76	0.41	0.22
			S.D.	0.42	0.21	0.12	0.04
Westport River 4	**8/2/2006**	**8/31/2006**	**28.89**	**54%**	**20%**	**5%**	**0%**
			Mean	0.42	0.24	0.16	N/A
			Min	0.01	0.01	0.01	0.00
			Max	2.88	1.40	0.31	0.00
			S.D.	0.49	0.30	0.11	N/A
Westport River 5	**8/2/2006**	**8/31/2006**	**28.94**	**33%**	**9%**	**1%**	**0%**
			Mean	0.23	0.15	0.06	N/A
			Min	0.02	0.05	0.01	0.00
			Max	0.86	0.39	0.15	0.00
			S.D.	0.20	0.08	0.05	N/A
Westport River 6	**8/2/2006**	**8/31/2006**	**28.94**	**14%**	**2%**	**0%**	**0%**
			Mean	0.13	0.07	0.01	N/A
			Min	0.03	0.01	0.01	0.00
			Max	0.36	0.13	0.01	0.00
			S.D.	0.09	0.04	N/A	N/A
Westport River 7	**8/2/2006**	**8/31/2006**	**28.94**	**33%**	**9%**	**1%**	**0%**
			Mean	0.23	0.15	0.06	N/A
			Min	0.02	0.05	0.01	0.00
			Max	0.86	0.39	0.15	0.00
			S.D.	0.20	0.08	0.05	N/A
Westport River 8	**7/10/2007**	**8/23/2007**	**43.96**	**50%**	**17%**	**5%**	**1%**
			Mean	0.37	0.15	0.08	0.07
			Min	0.02	0.03	0.01	0.03
			Max	0.85	0.32	0.19	0.10
			S.D.	0.24	0.07	0.04	0.03

Table VII-2. Duration (days and % of deployment time) that chlorophyll a levels exceed various benchmark levels within the East and West Branches of the Westport 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)
Westport River 1	**8/2/2006**	**8/31/2006**	**29.17**	**50%**	**2%**	**0%**	**0%**	**0%**
Mean Chl Value = 5.4 ug/L			**Mean**	0.32	0.11	N/A	N/A	N/A
			Min	0.04	0.04	0.00	0.00	0.00
			Max	1.13	0.21	0.00	0.00	0.00
			S.D.	0.23	0.09	N/A	N/A	N/A
Westport River 2	**7/10/2007**	**8/23/2007**	**36.04**	**30%**	**2%**	**0%**	**0%**	**0%**
Mean Chl Value = 7.7 ug/L			**Mean**	0.13	0.07	N/A	N/A	N/A
			Min	0.04	0.04	0.00	0.00	0.00
			Max	0.50	0.13	0.00	0.00	0.00
			S.D.	0.12	0.03	N/A	N/A	N/A
Westport River 3	**7/10/2007**	**8/23/2007**	**44.00**	**99%**	**67%**	**37%**	**11%**	**2%**
Mean Chl Value = 13.1 ug/L			**Mean**	8.73	0.87	0.44	0.17	0.10
			Min	0.83	0.04	0.04	0.04	0.04
			Max	36.21	11.83	1.58	0.58	0.17
			S.D.	15.39	2.29	0.38	0.14	0.05
Westport River 4	**8/2/2006**	**8/31/2006**	**27.71**	**100%**	**95%**	**62%**	**40%**	**30%**
Mean Chl Value = 20.2 ug/L			**Mean**	27.71	3.78	0.69	1.01	0.40
			Min	27.71	0.08	0.04	0.04	0.04
			Max	27.71	13.50	10.67	6.00	1.79
			S.D.	N/A	5.62	2.11	1.69	0.45
Westport River 5	**8/2/2006**	**8/31/2006**	**27.83**	**82%**	**56%**	**34%**	**16%**	**7%**
Mean Chl Value = 12.4 ug/L			**Mean**	0.91	0.38	0.19	0.13	0.08
			Min	0.08	0.04	0.04	0.04	0.04
			Max	6.46	1.83	0.88	0.42	0.25
			S.D.	1.62	0.45	0.18	0.09	0.06
Westport River 6	**8/2/2006**	**8/31/2006**	**23.92**	**69%**	**43%**	**24%**	**8%**	**1%**
Mean Chl Value = 9.7 ug/L			**Mean**	0.59	0.25	0.16	0.09	0.06
			Min	0.04	0.04	0.04	0.04	0.04
			Max	4.13	1.42	0.50	0.17	0.08
			S.D.	1.16	0.32	0.13	0.04	0.02
Westport River 7	**8/2/2006**	**8/31/2006**	**27.83**	**23%**	**0%**	**0%**	**0%**	**0%**
Mean Chl Value = 3.8 ug/L			**Mean**	0.19	0.04	N/A	N/A	N/A
			Min	0.04	0.04	0.00	0.00	0.00
			Max	1.08	0.04	0.00	0.00	0.00
			S.D.	0.19	0.00	N/A	N/A	N/A
Westport River 8	**7/10/2007**	**8/23/2007**	**40.75**	**33%**	**0%**	**0%**	**0%**	**0%**
Mean Chl Value = 3.9 ug/L			**Mean**	0.24	N/A	N/A	N/A	N/A
			Min	0.04	0.00	0.00	0.00	0.00
			Max	2.00	0.00	0.00	0.00	0.00
			S.D.	0.33	N/A	N/A	N/A	N/A

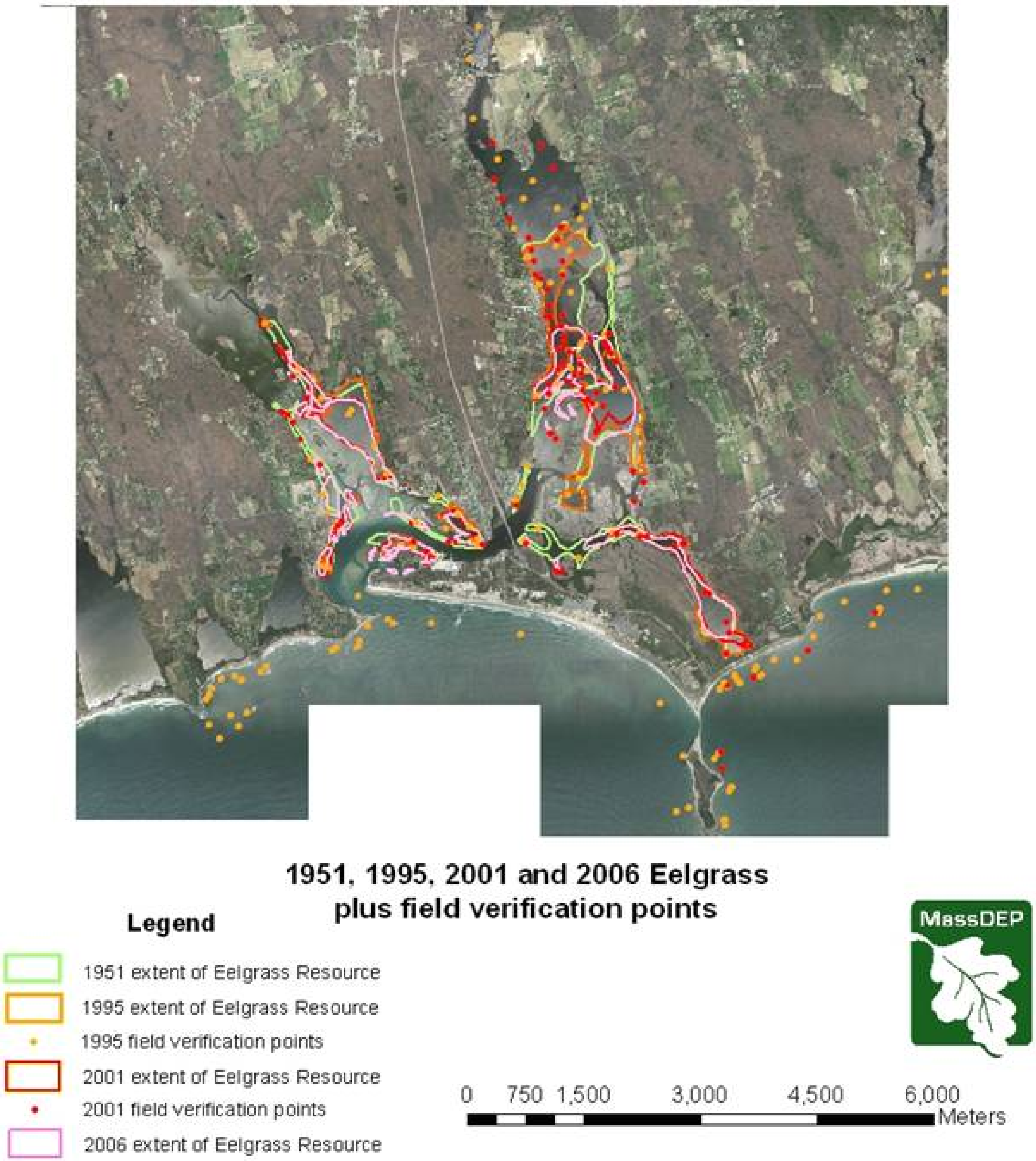


Figure VII-19. Eelgrass bed distribution within the Westport River System. Beds delineated in 1951 are circumscribed by the green outline, with 2006/7 outlined in blue. Beds delineated in 1995 are circumscribed by the red outline (map from the MassDEP Eelgrass Mapping Program). Eelgrass coverage was also noted in 2006 in SMAST-MEP sediment surveys.

At present, eelgrass exists across a relatively large portion of the system, particularly in the southern portions of the east and west branch and Westport Harbor (Figure VII-16). The results of the 2006/07 survey were confirmed by MEP staff conducting infaunal animal and sediment sampling and mooring studies in the summer and fall of 2006 and 2007. The major decline in the aerial distribution of eelgrass beds in the Westport River occurred primarily in the mid reach of the East Branch from 1951 to 2006. The loss of eelgrass at the uppermost portion of the coverage, with associated decline along the margins is consistent with nitrogen related eelgrass loss and the observed nitrogen levels and resulting chlorophyll a and dissolved oxygen depletions within this portion of the estuary. Generally, the eelgrass beds in the lower East Branch and The Let have been stable but with epiphytes at tidally averaged nitrogen levels <0.506 mg L^{-1}, with healthy beds within the East and West Branches at levels between 0.506-0.422 mg L^{-1} and 0.497-0.399 mg L^{-1}, respectively (tidally averaged TN presented in Section VI.1). Loss of eelgrass within the mid reach of the East Branch is occurring at tidally averaged TN levels of >0.506 and is showing stress in the West Branch (dense colonization by epiphytes) at levels >0.50 mg L^{-1}.

The condition of the uppermost eelgrass beds in the East and West Branches of the Westport River Estuary observed in 2006 is consistent with the level of nutrient enrichment up-gradient from and within the eelgrass beds themselves. Field observations by SCUBA diver of the eelgrass beds during the sediment survey indicated that the upper margins of the beds frequently showed dense coverage by epiphytes and that the present coverage in the mid region of the East Branch is patchy with moderate to sparse density of shoots, indicative of impaired habitat. These observations suggest that the upper regions of the present coverage in the East Branch is above its ability to tolerate additional nitrogen enrichment and that continuing loss of coverage is expected in this region.

Other key water quality indicators, dissolved oxygen and chlorophyll a, show similar levels of moderate enrichment with periodic oxygen depletions to 6-4 mg L^{-1} and chlorophyll levels averaging ~10 ug L^{-1} with periodic blooms in the 10-15 ug L^{-1} range. Given the sensitivity of eelgrass to declining light penetration resulting from nutrient enrichment and secondary effects of organic enrichment and oxygen depletion, the eelgrass with epiphytes and significantly impaired eelgrass habitat in the mid reach of the East Branch is to be expected. It is clear that eelgrass coverage is declining in the mid reach of the East Branch of the Westport River Estuary and that the habitat is moderately impaired (primarily by epiphyte growth) at the upper margins of the coverage in the West Branch. Significantly, all of the habitat and water quality indicators support the contention that the significant eelgrass loss and moderate impairments in these basins results from nitrogen enrichment. This is further supported by the timing and pattern of the loss/impairments. The result is that restoration of eelgrass habitat within the Westport Estuary is the primary management concern and nitrogen management is required targeting this resource. Since benthic animal habitat throughout the estuary is generally unimpaired, with impairment primarily in the upper reach of the East Branch, nitrogen management to restore eelgrass will also enhance benthic habitats.

Other factors which influence eelgrass bed loss in embayments can also be at play in the Westport River Embayment System, though the recent loss appears completely in-line with nitrogen enrichment. However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss could potentially be directly related to mooring density, as the system does support a permanent boat mooring area, however, much of the eelgrass loss has occurred in areas that really do not have boat moorings. While pier construction can cause impacts to eelgrass beds, the number of piers (docks) and the area they represent is insignificant given the area of eelgrass bed loss. It is also important to note that the lost eelgrass was primarily from open water areas as opposed to fringing beds along the shore which would be more susceptible to shading from piers. Boating pressure or small scale shell fishing may be adding additional stress in nutrient enriched areas such as the middle portion of the west branch, but it does not seem to be the overarching factor, especially given the shallow structure of this basin and the limited navigable water and other ecological indicators that point more towards the effect of nutrient over-enrichment.

It is not possible to determine quantitative short and long-term rates of change in eelgrass coverage from the mapping data, since there is only limited temporal data. However, based upon the 1951 and 1995 coverages, it appears that a minimum area of eelgrass habitat, on the order of 400 to 500 acres, could be recovered if nitrogen management alternatives were implemented (Table VII-3). It should be noted more importantly, that the health of the eelgrass that still exists in the Westport River would improve significantly with increased density, reduction in epiphytes and continued low levels of drift algae. Additionally, restoration of this eelgrass habitat will necessarily result in restoration of other resources throughout the Westport River Embayment System. With a reduction in nitrogen loading to the Westport River benthic infaunal habitat would be restored with an increase in shellfish habitat and shift toward larger longer lived deep burrowing organisms (see below discussion on benthic infaunal community survey).

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 28 locations within the Westport River Embayment System (Figure VII-20a,b,c,d), with replicate assays at almost every site. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as key community indices, diversity (H') and evenness (E). Particularly in areas that are naturally without eelgrass or have lost eelgrass coverage, 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).

Table VII-3. Temporal changes in eelgrass coverage in the Westport River Embayment System within the Town of Westport 1995 to 2006/7 (MassDEP, C. Costello).

EMBAYMENT	1951 (acres)	1995 (acres)	2001 (acres)	2006/7 (acres)	% Difference (1951 to 2006)
Westport River	1008.60	677.07	465.72	513.24	49%

Figure VII-20a. Aerial photograph of the Upper portion of the East Branch of the Westport River system showing location of benthic infaunal sampling stations (yellow symbol).

Figure VII-20b. Aerial photograph of the Middle portion of the East Branch of the Westport River system showing location of benthic infaunal sampling stations (yellow symbol).

Figure VII-20c. Aerial photograph of the Lower portion of the East and West Branches (inclusive of Westport Harbor and the Let) of the Westport River system showing location of benthic infaunal sampling stations (yellow symbol).

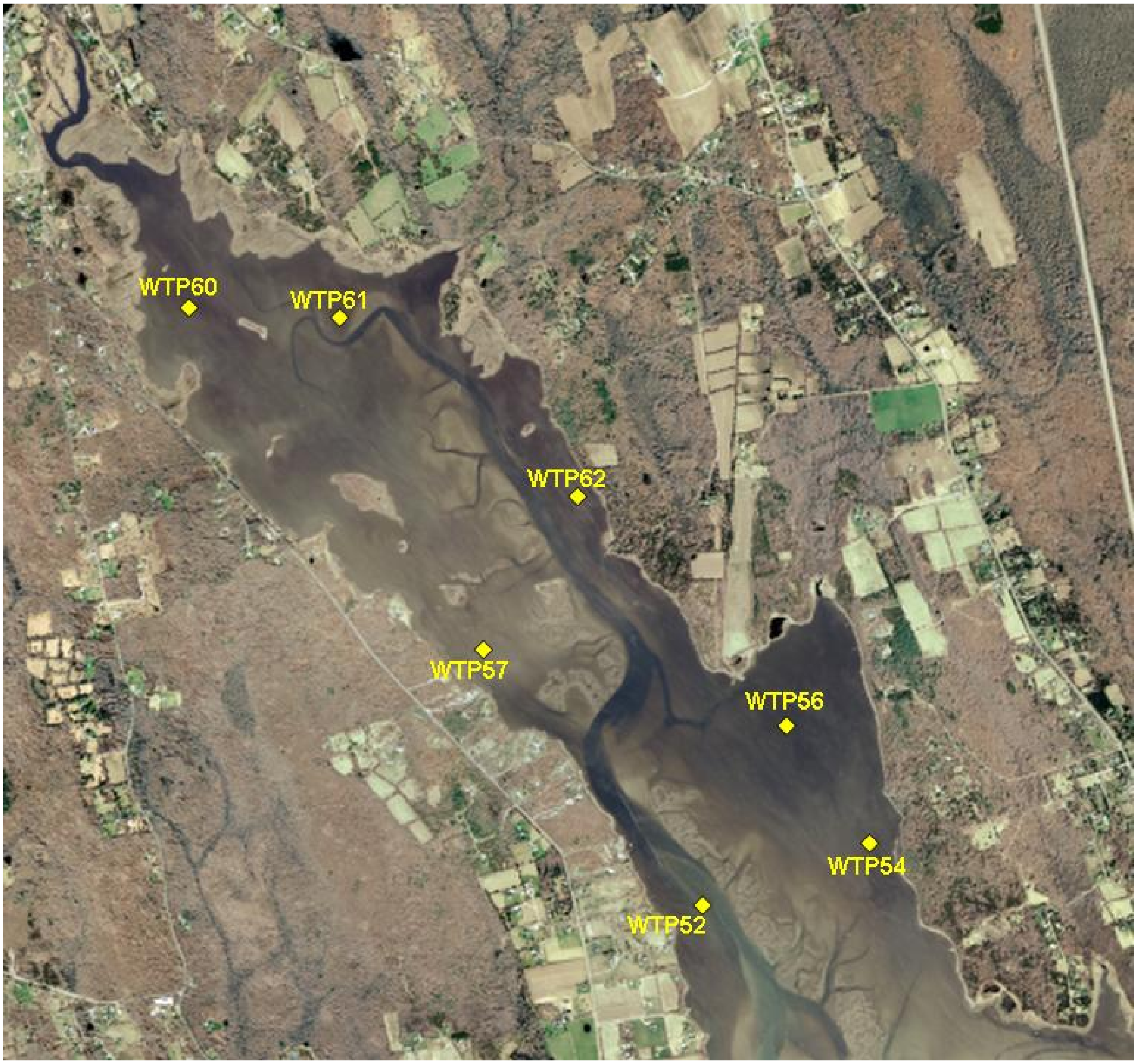


Figure VII-20d. Aerial photograph of the Upper and Middle portions of the West Branch of the Westport River system showing location of benthic infaunal sampling stations (yellow symbol).

Analysis of the evenness and diversity of the benthic animal communities are 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.

Overall, the infauna survey measured generally diverse and productive benthic animal communities throughout most of the Westport River Estuary, consistent with the general absence of macroalgal accumulations and the only relatively recent loss of eelgrass from the mid reach of the East Branch and presence of stable eelgrass beds within the lower basins of the estuary. While some basins are exhibiting impaired benthic animal habitat due to nitrogen enrichment (e.g. upper East Branch) most of the estuary is supporting high quality benthic animal habitat, particularly when the ecological structure of an estuarine basin is taken into account (e.g. salt marsh influences), Table VII-4.

Table VII-4. Benthic infaunal community data for the Westport River embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2). Stations refer to map in Figure VII-20a,b,c,d.

Basin	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)	Stations WTP# [1]
Westport River Estuary						
East Branch - Upper						
Mean =	7	506	6	1.59	0.57	2,4,6,8
S.E. =	1	138	0	0.13	0.05	
N =	7	7	7	7	7	
East Branch - Mid						
Mean =	16	554	11	2.58	0.66	36,38,40,41
S.E. =	2	114	1	0.18	0.04	43,45,47
N =	10	10	10	10	10	
East Branch - Lower						
Mean =	17	1078	12	2.61	0.64	16,17,22
S.E. =	3	537	3	0.42	0.06	
N =	4	4	4	4	4	
The Let						
Mean =	17	1699	12	2.99	0.74	9,12,15
S.E. =	1	413	1	0.18	0.04	
N =	5	5	5	5	5	
West Branch - Upper						
Mean =	19	1501	13	3.04	0.72	57,60,61,62
S.E. =	2	401	1	0.21	0.04	
N =	5	5	5	5	5	
West Branch - Lower						
Mean =	16	943	11	2.50	0.63	52,54,56
S.E. =	1	160	1	0.08	0.03	
N =	5	5	5	5	5	
Westport Harbor						
Mean =	12	593	10	2.24	0.74	27,29,31,32
S.E. =	4	332	3	0.22	0.07	
N =	5	5	3	5	5	

1- Station ID's refer to locations in Figures VII-17 a,b,c,d

The upper reach of the East Branch (to just below Hix Bridge) presently supports a benthic community with low Diversity and Evenness (H'= 1.59, E= 0.57) and is generally dominated by a polychaete worm typical of nutrient enriched southeastern Massachusetts estuaries, *Streblospio* (>60% of community). While the number of species was low (7 species), the numbers of individuals were moderate to high (~500 per sample). However, the low species numbers and community indices clearly indicate a moderate to significant level of habitat impairment resulting from the effects of nitrogen enrichment. This upper reach contrasts with the mid and lower reaches of the East Branch and The Let which support benthic animal communities with moderate to high Diversity (2.6 - 3.0) and Evenness (0.65 - 0.74), with moderate numbers of species (16-17 per sample) and high numbers of individuals (550 to >1000 per sample). Given the strong salt marsh influences on these basins, which tend to reduce species numbers and diversity even in "pristine" systems, it appears that these lower basins are not showing indications of excessive nutrient enrichment and are currently supporting high quality habitat. It should be noted that salt marshes are naturally nutrient and organic matter enriched and the benthic animal communities found within these lower basins are consistent with salt marsh influenced systems throughout the region. The upper and lower regions of the West Branch support benthic animal habitat similar to the lower basin of the East Branch and The Let. The benthic community was similarly configured and had high numbers of species (16-19) and individuals (>900) for these types of estuarine basins, with high Diversity and Evenness in the upper (H' > 3, E > 0.7) reach and moderate to high diversity and Evenness (H' = 2.5, E = 0.63) in the lower reach. Similar to the lower basins of the East Branch the species were indicative of salt marsh and eelgrass influences, with polychaete worms and amphipods being prevalent in a diverse productive benthic community. The communities observed in the lower East Branch and West Branch of the Westport River Estuary are in line with other ecologically similar estuarine basins that support high quality benthic habitats. For example, the wetland influenced upper reach of the Centerville River was found to have an average of 17 species, with 400 to >1000 individuals and comparable Diversity (H'= 2.6) and Evenness (E= 0.67). In the Phinneys Harbor System, Eel Pond is similarly configured to The Let and has comparable unimpaired benthic habitat with similar numbers of species (14), high numbers of individuals (650 to >1000), and generally high diversity (>3) and Evenness (>0.7).

Westport Harbor has high water quality and stable eelgrass beds. Its sediments are sandy and oxidized and have a low organic matter content. However, the high velocities of tidal waters during flood and ebb tides throughout much of the basin results in shifting sands, except in areas stabilized by eelgrass beds. The result is a benthic animal community with deep burrowers, with crustaceans and mollusks dominating and polychaete worms being less prevalent than in the other basins. However, due to the physical disturbance there are only moderate numbers of species (3-27), with moderate Diversity (2.24), but high numbers of individuals (~600) and Evenness (>0.7). Under similar conditions, a similar community was observed in Cotuit Bay (Three Bays) and Chatham Harbor, which support 16 and 8 species respectively, similar Eveness, but lower numbers (200-500) and a deep burrowing community.

It should be noted that estuaries with different geomorphologies, hydrodynamics and sediment dynamics, support high quality benthic habitat with similar levels of Diversity and Evenness to the unimpaired areas within the Westport River Estuary. However, these open basins typically have higher species numbers (≥25), but slightly lower numbers of individuals (500-800), although the actual species dominating the communities in all cases is similar. These differences stem primarily from the configuration of the basins, their depth and hydrodynamics.

The classification of habitat quality necessarily included the structure of the estuarine basin, specifically that it is fully representative of a tidal embayment, as opposed to a tidal river or salt marsh basin. The Westport River Estuary is a complex estuary composed of 3 types of basins: shallow open water basins with no eelgrass or surrounding wetland, shallow basins with significant associated salt marsh and eelgrass, and an estuarine lagoon with high tidal velocities and areas of shifting sands (Westport Harbor). Each of these 3 basins has different natural sensitivities to nitrogen enrichment and organic matter loading and each has its own benthic community indicative of an unimpaired or impaired habitat. Evaluation of infaunal habitat quality considered the natural structure of each system and the types of infaunal communities that they typically support. The benthic animal communities throughout most of the Westport River Estuary (except upper to mid East Branch) indicated generally healthy infaunal habitat, consistent with the tidally averaged nitrogen levels and levels of oxygen depletion in line with the ecosystem types represented. The general absence of macroalgal accumulations and sediments of consolidated sands and mud, with a visible oxidized surface layer is also consistent with the community measurements. Since eelgrass loss has occurred primarily in the mid reach of the East Branch indicating a significant level of impairment to eelgrass habitat, lowering the nitrogen to improve eelgrass habitat in this region will also likely be sufficient to restore infaunal animal habitat in the upper reach, as eelgrass is much more sensitive to nitrogen enrichment than infaunal communities.

Other Resource Characteristics:

In addition to benthic infaunal community characterization undertaken as part of the MEP field data collection, other biological resources assessments were integrated into the habitat assessment portion of the MEP nutrient threshold development process as developed by the Commonwealth and available. The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish (Figure VII-21). As is the case with some systems on Cape Cod, most of the enclosed waters of the east and west branches of the Westport River system are classified as conditionally approved for the taking of shellfish during specific periods of the year, indicating the system is impaired relative to the taking of shellfish. This is most likely due to bacterial concerns which would be a result of both human activity (septic systems in the watershed) as well as natural fauna.

Nevertheless, the Westport River estuarine system has also been classified as supportive of specific shellfish communities (Figure VII-22). The major shellfish species with potential habitat within the Westport River Estuary are mainly quahogs (*Mercenaria*) and bay scallops. Additionally, suitable habitat closer to shore and along the various marsh islands was identified for soft shell clams (*Mya*) as well as razor clams. It should be noted that the observed pattern of shellfish growing area is consistent with the observed organic rich sediments within the Westport River system. Moreover, improving benthic animal habitat quality should also expand the shellfish growing area within this system. This will not necessarily result in the opening of shell fishing beds as the underlying concern over bacterial levels in the Westport River system may still exist, though that can be closely monitored.

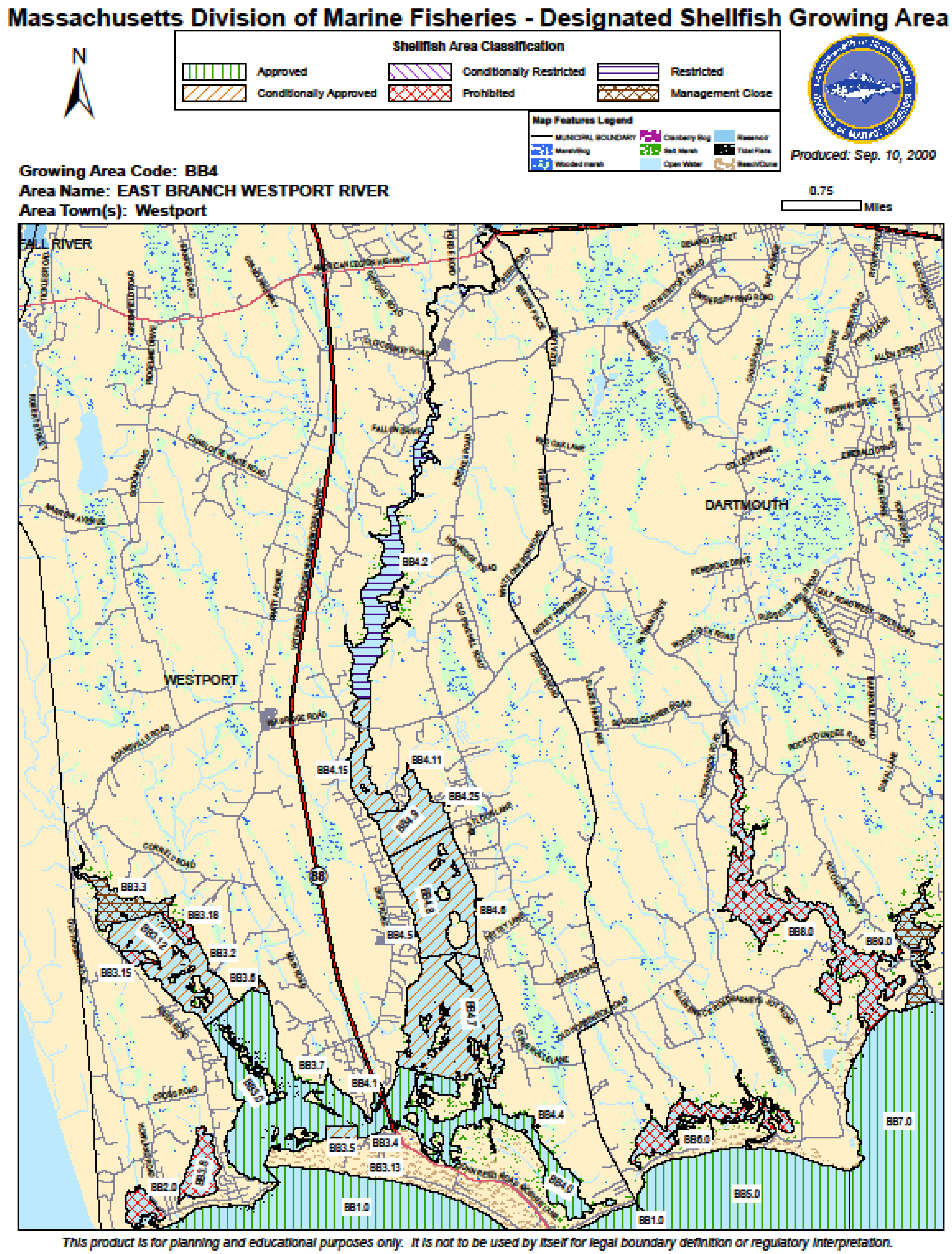


Figure VII-21. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

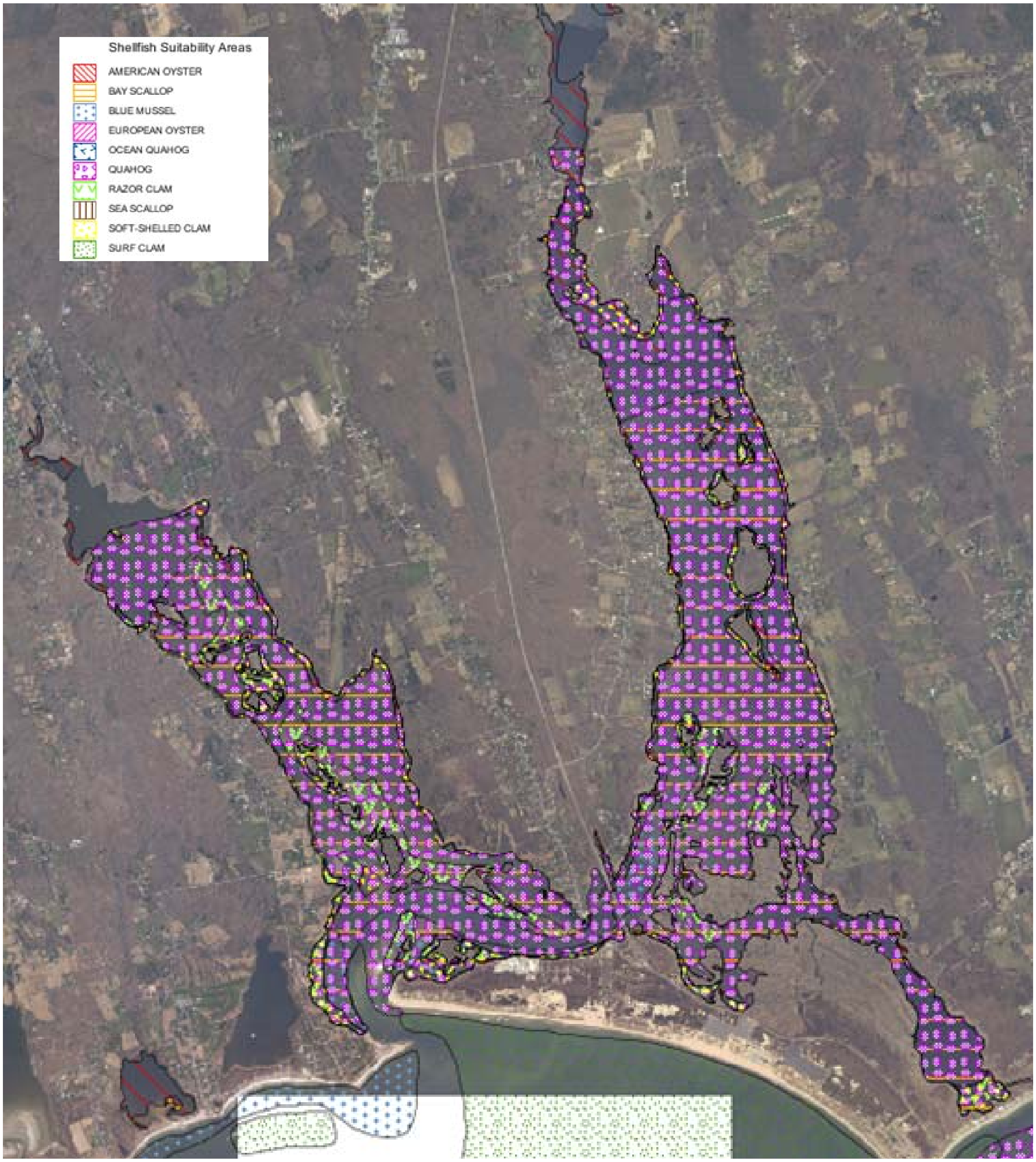


Figure VII-22 Location of shellfish suitability areas within the Westport River Estuary as determined by Mass Division of Marine Fisheries. The predominant shellfish in a fully functional habitat are quahogs and bay scallops. Bay scallops would likely see significant recovery under nitrogen management, as eelgrass habitat would be restored. Suitability does not necessarily mean that the species of shellfish is "present".

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 Westport River Estuary 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 BayWatcher Water Quality Monitoring Program.

The Westport River Estuary is a complex estuary composed of 3 functional types of basins: shallow open water basins with no eelgrass or surrounding wetland, shallow basins with significant associated salt marsh and eelgrass, and an estuarine lagoon with high tidal velocities and areas of shifting sands (Westport Harbor). Each of these 3 basin types has difference in their natural sensitivity to nitrogen enrichment and organic matter loading and each has its own benthic community indicative of an unimpaired or impaired habitat as well as different abilities to support stable eelgrass beds. Evaluation of habitat quality considered the natural structure of each component of the overall system and the types of infaunal communities and eelgrass coverages that they support. At present, the Westport River Estuary is showing differences in nitrogen enrichment and habitat quality among its various component basins (Table VIII-1).

Overall, the Estuary is showing some nitrogen related habitat impairment within some of its component basins, however, most of the system is supporting high quality to moderately impaired habitat, with regions of significant impairment resulting primarily from the loss of eelgrass coverage (e.g. mid reach East Branch) or degraded benthic animal habitat (upper East Branch). The benthic animal communities throughout most of the Westport River Estuary (except upper to mid East Branch) indicate generally healthy infaunal habitat, consistent with the tidally averaged nitrogen levels and levels of oxygen depletion and the ecosystem types represented. The general absence of macroalgal accumulations and sediments composed of consolidated sands and mud, with a visible oxidized surface layer, is also consistent with the benthic community characteristics. Since eelgrass loss has occurred primarily in the mid reach of the East Branch, indicating a significant level of impairment to eelgrass habitat, lowering the nitrogen to improve eelgrass habitat in this region will also likely be sufficient to restore infaunal animal habitat in the upper reach, as eelgrass is much more sensitive to nitrogen enrichment than infaunal communities.

Oxygen and chlorophyll-*a* levels were generally consistent with the eelgrass and infaunal animal assessments and paralleled gradients in nitrogen enrichment. The upper and middle section of the East Branch of the Westport River Estuary has large daily oxygen excursions, with moderate to significant oxygen depletion consistent with the significant level of nitrogen enrichment. The salt marsh influenced lower East Branch showed lower nitrogen levels and less oxygen depletion than the upper and mid reaches. This parallels the level of nitrogen enrichment with the lower East Branch showing higher oxygen levels and The Let showing moderate oxygen depletions consistent with its function as a salt marsh basin. However, the chlorophyll and nitrogen levels within The Let indicated high water quality and supports both stable eelgrass beds and high quality benthic animal habitat. The observed levels of oxygen

Table VIII-1. Summary of nutrient related habitat quality within the Westport River Estuary within the Town of Westport, MA, based upon assessments in Section VII. The West Branch is a shallow saltmarsh influenced tidal basin; The Let is primarily a shallow salt marsh tidal basin, surrounded by extensive tidal salt marsh; the East Branch and Westport Harbor are typical embayment basins. Note: WQMP refers to the BayWatcher Water Quality Monitoring Program.

Health Indicator	Westport River Embayment System						
	West Branch		East Branch			Westport Harbor	The Let
	Upper	Lower	Upper	Mid	Lower		
Dissolved Oxygen	H[1]	H[1]	MI-SI[2]	MI-SI[2]	H[3]	H-MI[4]	H[5]
Chlorophyll	MI[6]	H[7]	MI-SI[8]	MI-SI[8]	H[9]	H[10]	H[9]
Macroalgae	MI[11]	MI[12]	--[13]	--[13]	--[13]	--[13]	--[13]
Eelgrass	--[14]	H-MI[15]	--[14]	SI[16]	H-MI[15]	H[17]	H[17]
Infaunal Animals	H[18]	H-MI[19]	MI-SI[20]	H[18]	H[18]	H[21]	H[18]
Overall:	**H-MI[22]**	**H-MI[23]**	**SI[24]**	**SI[25]**	**H-MI[26]**	**H[27]**	**H[28]**

1 – salt marsh influence, oxygen depletion rarely to ≤4 mg/L (<1% of record), generally >5mg/L (>91% of record), and typically >6 mg/L (Moorings DO-5,6,7)
2 -- oxygen depletions frequently <5 mg/L, periodically to <4 mg/L and <3 mg/L, minimum=2 mg/L;
3 – salt marsh influence, oxygen almost always >5 mg/L (96% of record), generally >6 mg/L.
4 – oxygen levels frequently depleted to <5 mg/L 27%, <4 mg/L 15% of 95 samples, 2000-04 WQMP.
5 – primarily a salt marsh pond, periodic oxygen depletion to ≤4 mg/L, infrequently to <3 mg/L; basin surrounded by extensive tidal saltmarsh resulting in natural organic enrichment.
6 – moderate to high summer chlorophyll levels averaging 10-12 ug/L, blooms >15 ug/L ≥25% of record and >20 ug/L ~10% of record
7 – low summer chlorophyll levels averaging 4 ug/L, <5 ug/L 77% of record, did not reach 10 ug/L
8 – high chlorophyll levels, averaging 13-20 ug/L, >10 ug/L >67% of record and frequent blooms >20 ug/L, reaching ~40 ug/L.
9 – salt marsh influenced, low to moderate chlorophyll levels, averaging 5-8 ug/L <10 ug/L 98% of record.
10 – low chlorophyll, averaging 4.8 ug/L,<5 ug/L >71% and >10 ug/L 6% of 52 samples,1995-2009 WQMP
11 -- filamentous species and some algal mat.
12 -- some dense patches of drift filamentous species mostly within the western side of the basin.
13 -- drift algae sparse or absent
14 – no evidence this basin is supportive of eelgrass.
15 -- MassDEP (C. Costello) indicates that eelgrass coverage has been stable in this basin in all surveys since 1951, although there is some slight loss post-1995 and epiphytes on plants in the uppermost beds.
16-- MassDEP indicates significant and continuing loss of eelgrass coverage, 1995-2007.
17-- MassDEP indicates that eelgrass coverage has been stable in this basin 1951-2007.
18 -- Infauna: high numbers individuals (550 to >1000), moderate species (16-19), moderate-high Evenness (0.65->0.70) and diversity (~2.6->3.0) and; some organic enrichment indicators typical of salt marsh influenced basins and some areas with amphipod mats. Assessment takes into account the natural organic enrichment associated with surrounding salt marsh.
19 -- Infauna: high numbers of individuals (>900), moderate species (16), moderate diversity (2.5) and Evenness (0.63); some organic enrichment indicators typical of salt marsh influenced basins and some areas with amphipod mats, however patches of stress indicators (tubificids).
20 -- low numbers of species, diversity & Evenness, dominated by moderate to high numbers of organic enrichment species
21 -- Infauna: high numbers of individuals (~600) and Evenness (>0.7), moderate numbers of species and diversity with few organic enrichment indicators, instead dominated by deep burrowers, with mollusks and crustaceans prevalent. Major stressor is disturbance by high tidal velocities.
22 -- Low to moderate Impairment, primarily due to the high chlorophyll-*a* levels and blooms and presence of filamentous algal mats. Oxygen levels and infaunal communities reflective of salt marsh influence.
23 -- Low to Moderate Impairment, primarily due to some eelgrass coverage loss in uppermost beds, but dense epiphyte growth on existing beds coupled with some patches of impaired benthic animal habitat with macroalgal accumulations. among high quality.
24 -- Significant Impairment, indicated by stressed infaunal communities dominated by organic enrichment species, low oxygen and high chlorophyll levels.
25 -- Significant Impairment resulting from significant loss of eelgrass coverage coupled with high chlorophyll levels and moderate to low dissolved oxygen.
26 -- Low to Moderate Impairment resulting from some loss of eelgrass and some epiphyte growth, but with low chlorophyll, and high quality benthic animal habitat.
27 -- High quality, indicated by stable eelgrass coverage, high water quality and high quality benthic habitat.
28 -- Unimpaired as indicated by stable eelgrass coverage, high water quality and high quality benthic habitat consistent with a salt marsh dominated basin.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment; SD = Severe Degradation; -- = not applicable to this estuarine reach

depletion within The Let (and to a lesser extent the lower East Branch) are typical of salt marsh ponds and therefore do not indicate impairment of this basin. The West Branch shows a similar gradient in oxygen depletion as the East Branch, but as it is less nitrogen enriched, the levels of depletion are smaller and less frequent than the East Branch. However, given the frequent large phytoplankton blooms within the upper West Branch and patches of moderately impaired benthic animal habitat with some macroalgal accumulations, it appears that this reach is just above its ability to assimilate additional nitrogen and is showing initial signs of impairment by nitrogen enrichment.

For the Westport River, eelgrass distribution data from 1951, 1995, 2001 and 2006-07 was used to assess (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provided a view of temporal trends in eelgrass distribution from 1951 to 2006; the period in which watershed nitrogen loading increased significantly.

At present, eelgrass exists across a relatively large portion of the system, particularly in the southern portions of the east and west branch and Westport Harbor. However, a major decline in the aerial distribution of eelgrass beds in the Westport River Estuary has occurred primarily in the mid reach of the East Branch between 1951 and 2006. The loss of eelgrass at the uppermost portion of the coverage, with associated decline along the margins is consistent with nitrogen related eelgrass loss and the observed nitrogen levels and resulting chlorophyll-*a* and dissolved oxygen depletions within this portion of the estuary. Generally, the eelgrass beds in the lower East Branch and The Let have been stable but with epiphytes at tidally averaged nitrogen levels <0.506 mg L^{-1}, with healthy beds within the East and West Branches at levels between 0.506-0.422 mg L^{-1} and 0.497-0.399 mg L^{-1}, respectively (tidally averaged TN presented in Section VI.1). Loss of eelgrass within the mid reach of the East Branch is occurring at tidally averaged TN levels of >0.506 and is showing stress in the West Branch (dense colonization by epiphytes) at levels >0.50 mg L^{-1}.

The condition of the uppermost eelgrass beds in the East and West Branches of the Westport River Estuary observed in 2006 is consistent with the level of nutrient enrichment and associated chlorophyll-*a* and oxygen levels up-gradient from and within the eelgrass beds themselves. In addition, observations that the upper margins of the beds frequently show dense coverage by epiphytes and that the present beds within the coverage area in the mid region of the East Branch are not continuous, but patchy with moderate to sparse density of shoots, is indicative of impaired eelgrass habitat. These observations suggest that the upper region of the present coverage in the East Branch is above its ability to tolerate additional nitrogen enrichment and that continuing loss of coverage is expected in this region.

It is clear that eelgrass coverage is declining in the mid reach of the East Branch of the Westport River Estuary and that the habitat is moderately impaired (primarily by epiphyte growth) at the upper margins of the coverage in the West Branch. Significantly, all of the habitat and water quality indicators support the contention that the significant eelgrass loss and moderate impairments in these basins results from nitrogen enrichment. This is further supported by the timing and pattern of the loss/impairments. The result is that restoration of eelgrass habitat within the Westport Estuary is the primary management concern and nitrogen management is required, specifically targeting this resource. Since benthic animal habitat throughout the estuary is generally unimpaired, with impairment primarily in the upper reach of the East Branch, nitrogen management to restore eelgrass will also enhance benthic habitats. It should be noted that actions to restore lost eelgrass habitat will also enhance the health of the existing eelgrass beds within the Westport River Estuary resulting in increases in shoot density, reduction in epiphytes and continued low levels of drift algae. Additionally, restoration of this eelgrass habitat will necessarily result in restoration of other resources throughout the Westport River Embayment System. With a reduction in nitrogen loading to the Westport River, benthic infaunal habitat would be restored with an increase in shellfish habitat and shift toward larger longer lived deep burrowing organisms.

Overall, the infauna survey measured generally diverse and productive benthic animal communities throughout most of the Westport River Estuary, consistent with the general absence of macroalgal accumulations and the “relatively” recent loss of eelgrass from the mid reach of the East Branch and presence of stable eelgrass beds within the lower basins of the estuary. While some basins are exhibiting impaired benthic animal habitat due to nitrogen enrichment (e.g. upper East Branch) most of the estuary is supporting high quality benthic animal habitat, particularly when the ecological structure of estuarine basin is taken into account (e.g. salt marsh influences).

The upper reach of the East Branch presently supports a benthic community with low Diversity and Evenness and is generally dominated by a polychaete worm typical of nutrient enriched southeastern Massachusetts estuaries. While the number of individuals was moderate to high, there are low species numbers and community indices clearly indicate a moderate to significant level of habitat impairment resulting from the effects of nitrogen enrichment. This upper reach contrasts with the mid and lower reaches of the East Branch and The Let which support benthic animal communities with moderate to high Diversity and Evenness with moderate numbers of species and high numbers of individuals. Given the strong salt marsh influences on these basins, which tend to reduce species numbers and diversity even in "pristine" systems, it appears that these lower basins are not showing indications of excessive nutrient enrichment and are currently supporting high quality habitat. Similarly, the upper and lower regions of the West Branch support benthic animal habitat comparable to the lower East Branch and The Let. The benthic community was similarly configured according to similar metrics, with high numbers of species and individuals and moderate to high Diversity and Evenness. Similar to the lower basins of the East Branch the species were indicative of salt marsh and eelgrass influences, with polychaete worms and amphipods being prevalent in a diverse productive benthic community. The communities observed in the lower East Branch and West Branch of the Westport River Estuary are similar to other ecologically similar estuarine basins with high quality benthic habitats. Westport Harbor has high water quality and stable eelgrass beds and sandy oxidized sediments with a low organic matter content. However, the high velocities of tidal waters during flood and ebb tides throughout much of the basin results in shifting sands, except in areas stabilized by eelgrass beds. The result is a benthic animal community with deep burrowers, with crustaceans and mollusks dominating and polychaete worms being less prevalent than in the other basins.

The benthic animal communities throughout most of the Westport River Estuary (except upper to mid East Branch) indicated generally healthy infaunal habitat, consistent with the tidally averaged nitrogen levels and levels of oxygen depletion which were in line with the ecosystem types represented. The general absence of macroalgal accumulations and sediments of consolidated sands and mud, with a visible oxidized surface layer, is also consistent with the community measurements. Since eelgrass loss has occurred primarily in the mid reach of the East Branch, indicating a significant level of impairment to eelgrass habitat, lowering the nitrogen to improve eelgrass habitat in this region will also likely be sufficient to restore infaunal animal habitat in the upper reach, as eelgrass is much more sensitive to nitrogen enrichment than infaunal communities.

Based upon the above analysis, eelgrass habitat was selected as the primary nitrogen management goal for the mid reach of the East Branch and the lower reach of the West Branch of the Westport River Estuary, with associated restoration of impaired infaunal habitat within the upper reach of the East Branch. These goals are the focus of the MEP threshold analysis presented in Section VIII.3.

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

In the Westport River System eelgrass exists in both the East and West Branches as well as in The Let and Westport Harbor. Within the East Branch, eelgrass historically existed near Upper Spectacle Island (between water quality stations E-56 and E-33). Eelgrass was documented in the region of Upper Spectacle Island in the 1951 analysis and in MassDEP field survey in 1995. However, eelgrass was not observed in 2001 or 2006 surveys in this region, at tidally averaged TN levels of 0.64 mg N L^{-1} (Station E-56, Table VI-1). Instead, since 1995, eelgrass coverage has been limited to the lower basin of the East Branch, south of Great Island, including within The Let. Presently, the eelgrass beds just south of Great Island are heavy with epiphytes and while they persist, they are clearly impaired and diminishing at the current TN level, 0.51 mg N L^{-1} (Station E-41). Areas with TN levels of 0.64 mg L^{-1} in the East Branch do not have a history of supporting eelgrass beds. In contrast healthy eelgrass beds in the lower portion of the East Branch appear to be present at levels ~0.40-0.43 mg N L^{-1} (Stations E-30 and E-26). The tidally averaged level of TN supportive of high quality eelgrass habitat in this basin, therefore appears to be between 0.51 and 0.43 mg N L^{-1}.

Similar to the East Branch of the Westport River Estuary, the West Branch supported historically greater coverage of eelgrass than today. In 1951 eelgrass covered most the non-shoal areas of from Hicks Cove south, with a small fringing bed extending to the north along the eastern shore. At present, eelgrass still exists in fringing beds exists along the western shore but mainly near Westport Harbor and beds also exist along the eastern shore, but with dense epiphyte growth, except near the Westport Harbor basin. The unimpaired beds are associated with tidally averaged TN levels less than 0.421 mg N L^{-1} (Station W-9). The stable eelgrass beds within Westport Harbor are found at tidally averaged TN levels of 0.33-0.40 mg N L^{-1} (Stations E-26, W-6, N-12). Both Branches show the typical pattern of eelgrass loss associated with nitrogen loading, with eelgrass being lost from the uppermost regions of each basin and the deeper waters first, appearing to "retreat" toward the inlet.

The results indicate that eelgrass has been lost from the Westport River Estuary in areas that presently support tidally averaged TN levels of 0.57 mg N L^{-1} and >0.50 mg N L^{-1} in the East and West Branches, respectively. At lower nitrogen levels eelgrass is persisting, but with epiphytes and losses of coverage from the upper and deeper areas of the beds. These sites are associated with 0.51 mg N L^{-1} and ~0.50 mg N L^{-1} in the East and West Branches, respectively, while "healthy" beds are found at lower concentrations, with <0.428 mg N L^{-1} and 0.421 mg N L^{-1} in the East and West Branches, respectively, and <0.400 mg N L^{-1} in Westport Harbor. It appears that in the Westport River Estuary, the TN level to support high quality eelgrass habitat may be greater than 0.43 mg N L^{-1}, but less than 0.50 mg N L^{-1}. Given these results and the configuration and depth of the eelgrass areas of this system, a comparison to other estuaries was undertaken to refine the threshold.

These TN levels supportive of eelgrass habitat in the Westport River Estuary are higher than generally found in high quality eelgrass habitat such as within deeper systems (>2 m) like Stage Harbor (0.38 mg L-1) or West Falmouth Harbor and Phinneys Harbor (0.35 mg L^{-1}).

However, in shallow systems like most of the areas that support eelgrass in the Westport River Estuary (with eelgrass generally at <1 m depth), eelgrass beds are sustainable at higher TN (higher chlorophyll-*a*) levels than in deeper waters, because of the "thinner" water column that light has to pass through to support eelgrass growth (less water to penetrate). At comparable depths in Bournes Pond, eelgrass can be still be found (although heavy with epiphytes) at the mouth of the upper tributary at a tidally averaged TN concentration of 0.481 mg TN L^{-1}, while the more stable beds in the lower region of Israel's Cove have eelgrass at a tidally averaged TN of 0.429 mg TN L^{-1}. It should be added that eelgrass can persist at nitrogen levels that are non-supportive of healthy beds, and eelgrass within Hamblin Pond (sub-embayment to Waquoit Bay) persisted at a high TN level (0.5 mg L^{-1}) after eelgrass within the central portion of Waquoit Bay had disappeared, but the 0.5 mg N L^{-1} TN level was associated with diminishing eelgrass patches and was just beyond the level supportive of high quality habitat. These higher levels (~0.5 mg L-1) were also associated with impaired eelgrass areas in the Westport River System. All of the eelgrass information for the Westport River Estuary indicates that the eelgrass habitat is significantly impaired in the upper reaches of the East and West Branches and moderately impaired in the mid regions, with stable high quality eelgrass beds in the lower regions near Westport Harbor. The data also indicates that the nitrogen threshold level supportive of high quality eelgrass habitat is close to, but less than 0.50 mg N L-1, since eelgrass habitat in both the East and West Branches is only moderately impaired at this level.

The restoration of eelgrass habitat within the East and West Branches to documented historic coverages will require lowering the present nitrogen levels. In the East Branch it is clear that extensive eelgrass coverage within the immediate area south of Lower Spectacle Island has been lost, as seen in the 1951 and 1995 analyses compared to the more recent 2001 and 2006 surveys. To restore eelgrass to the 1951 and 1995 levels, TN concentration will need to be lowered to 0.49 mg L^{-1} at the Sentinel Station in the East Branch, determined to be the long-term water quality station E-33 (Section VI). Similarly within the West Branch, to prevent further loss of eelgrass and to restore eelgrass to 1951 and 1995 levels, tidally averaged TN at long-term water quality station W-12 needs to be similarly lowered to 0.48 mg L-1. Westport Harbor is presently supporting high quality eelgrass habitat.

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 infaunal habitats in the Westport River estuary system. Tidally averaged total nitrogen thresholds derived in Section VIII.1 were used to adjust the calibrated constituent transport model developed in Section VI. It is understood that the landfill load present in the Old County Road and East Branch North watershed areas is being mitigated at the present time, therefore this load was removed for the threshold analysis. Watershed nitrogen loads were lowered by reductions in septic effluent discharges until the nitrogen levels reached the threshold level at the sentinel stations chosen for the Westport River estuary system. It is important to note that load reductions can be produced by reduction of any or all sources. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment. A comparison between present septic and total watershed loading and the loadings for the modeled threshold scenario is provided in Tables VIII-2 and VIII-3.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required 71% removal of septic load (associated with direct groundwater discharge to the embayment) for the entire system. This load is being removed from the watersheds located on the upstream side of the Route 88 Bridge, excluding The Let. Focused nitrogen removal in the East Branch watersheds caused improvements in water quality to both the East and West Branches of the Westport River Estuary. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

Tables VIII-3 and VIII-4 provides additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. For example, removal of 100% of the septic load from the North East Branch watershed results in a 9.8% reduction in total watershed nitrogen load within the North East Branch Watershed. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent 'worst-case' summertime conditions.

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel station is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station, reductions in TN total watershed load of 18% are required in the system.

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 estuarine system.

Table VIII-2. Comparison of sub-embayment watershed septic loads (attenuated) used for modeling of present and threshold loading scenarios of the Westport River estuary. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms. Present septic loads at stream gauge locations are based on wastewater percentages from unattenuated loads; Old County Road present load is based on measured summer conditions (see Figure IV-3).

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
North East Branch	9.299	0.000	-100.0%
West Branch	6.540	6.540	0.0%
South East Branch	15.855	0.000	-100.0%
The Let	1.447	1.447	0.0%
Westport Harbor	6.592	6.592	0.0%
Old County Road	48.260	0.000	-100.0%
Kirby Brook	7.786	0.000	-100.0%
Adamsville Brook	17.066	17.066	0.0%
Angeline Brook	3.077	3.077	0.0%
Snell Creek	4.556	0.000	-100.0%
System Total	120.477	34.721	-71.2%

Table VIII-3. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Westport River estuary. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms. Present loads at stream gauge locations are based on measured loads; Old County Road present load is based on measured summer conditions (see Table IV-3).

sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
North East Branch	103.088	93.030	-9.8%
West Branch	32.901	32.901	0.0%
South East Branch	62.332	46.477	-25.4%
The Let	5.759	5.759	0.0%
Westport Harbor	10.252	10.252	0.0%
Old County Road	162.614	111.816	-31.2%
Kirby Brook	20.953	13.167	-37.2%
Adamsville Brook	47.622	47.622	0.0%
Angeline Brook	34.296	34.296	0.0%
Snell Creek	8.137	3.581	-56.0%
System Total	487.954	398.901	-18.3%

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

sub-embayment	threshold watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
North East Branch	93.030	4.360	-30.369
West Branch	32.901	11.154	-6.288
South East Branch	46.477	20.922	-16.675
The Let	5.759	1.968	11.811
Westport Harbor	10.252	8.226	-30.507
Old County Road	111.816	-	-
Kirby Brook	13.167	-	-
Adamsville Brook	47.622	-	-
Angeline Brook	34.296	-	-
Snell Creek	3.581	-	-
System Total	398.901	46.630	-72.027

Table VIII-5. Comparison of model average total N concentrations from present loading and the threshold scenario, with percent change, for the Westport River estuary. The threshold is 0.48 mg/L for W-12 and 0.49 for E-33.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Head Westport	N-0	1.344	0.920	-39.9%
Upper East Branch	N-1	0.919	0.702	-34.1%
Upper East Branch	N-2	0.879	0.688	-32.1%
Upper East Branch	N-3	0.855	0.678	-30.9%
Mid East Branch	N-4	0.798	0.647	-29.3%
Mid East Branch	E-69	0.735	0.605	-28.8%
Mid East Branch	E-56	0.616	0.524	-27.4%
Lower East Branch	**E-33**	**0.554**	**0.482**	**-26.7%**
Lower East Branch	E-41	0.492	0.438	-25.8%
Lower East Branch	E-30	0.414	0.382	-24.5%
Lower	E-26	0.389	0.363	-23.9%
Lower West Branch	**W-12**	**0.491**	**0.469**	**-10.6%**
Lower West Branch	W-9	0.394	0.378	-14.5%
Lower West Branch	W-6	0.364	0.350	-16.5%
Inlet	N-12	0.329	0.319	-20.4%

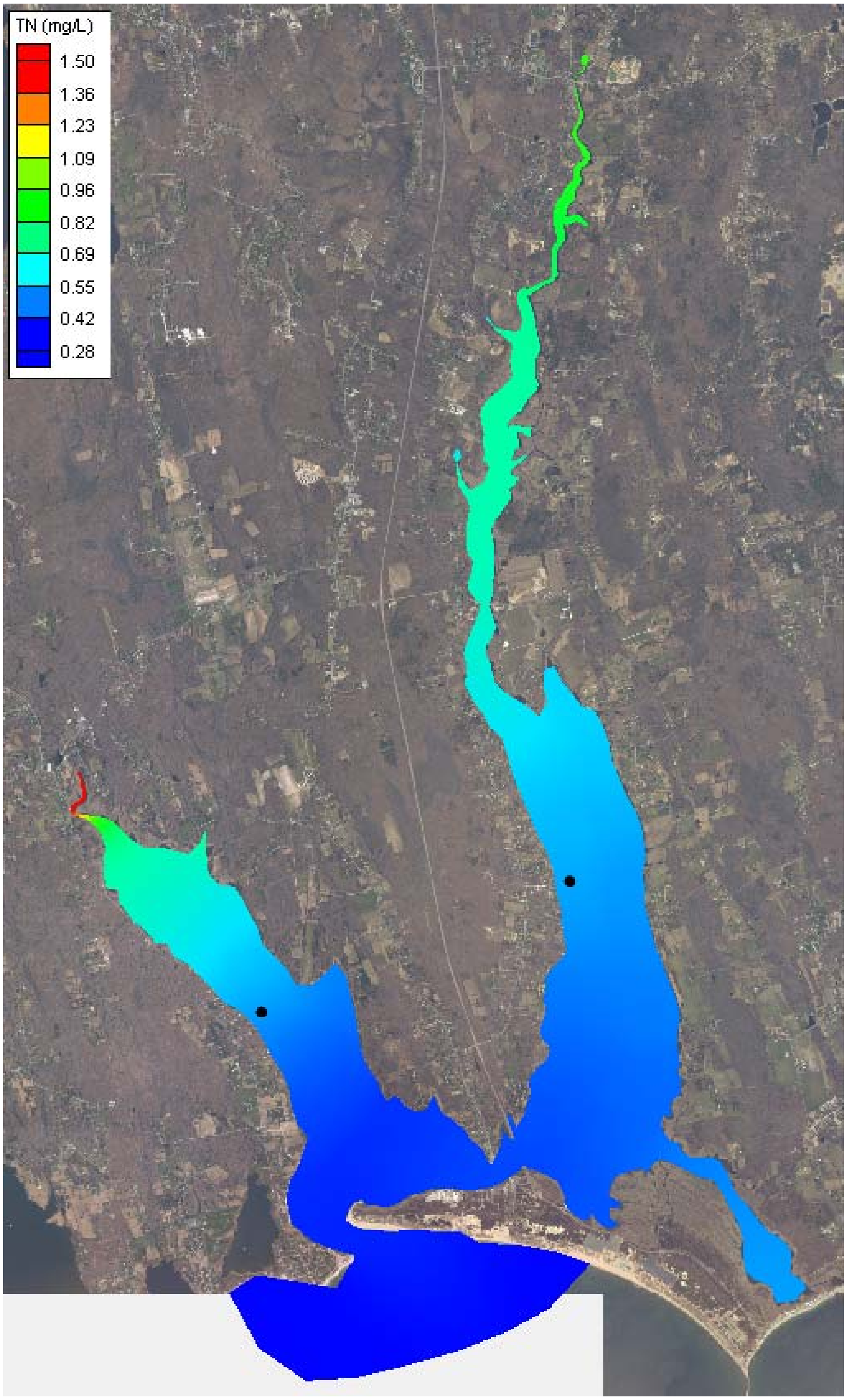


Figure VIII-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Westport River estuary system, for threshold conditions. Threshold or Sentinel Stations are shown by the dots, W-12 for the West Branch (Threshold TN 0.48 mg/L) and E-33 for the East Branch (Threshold TN 0.49 mg/L).

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