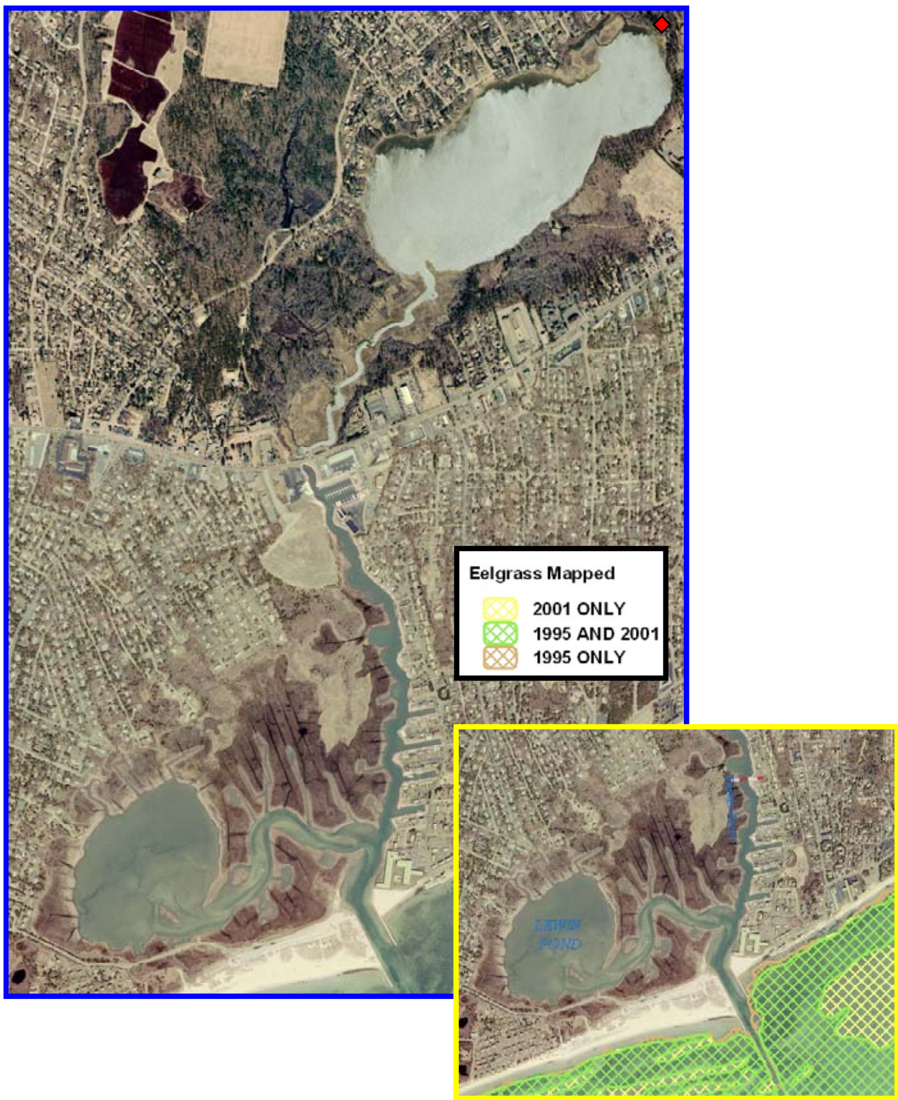
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Parkers River Embayment System, Yarmouth, Massachusetts



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

FINAL REPORT – May 2010

Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Parkers River Embayment System, Yarmouth, Massachusetts

FINAL REPORT – May 2010

Brian Howes
Roland Samimy
David Schlezinger
Ed Eichner

Sean Kelley
John Ramsey

Phil Detjens, "Jay"

Contributors:

US Geological Survey
Don Walters and John Masterson

Applied Coastal Research and Engineering, Inc.
Elizabeth Hunt and Trey Ruthven

Massachusetts Department of Environmental Protection
Charles Costello and Brian Dudley (DEP project manager)

SMAST Coastal Systems Program
Jenifer Bensen, Michael Bartlett, Sara Sampieri

Cape Cod Commission
Tom Cambareri

SMAST
University of Massachusetts Dartmouth
The School for Marine Science and Technology

Massachusetts Department of Environmental Protection

Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Parkers River Embayment System, Yarmouth, Massachusetts

Executive Summary

1. Background

This report presents the results generated from the implementation of the Massachusetts Estuaries Project's Linked Watershed-Embayment Approach to the Parkers River embayment system, a coastal embayment primarily within the Town of Yarmouth, Massachusetts. Analyses of the Parkers River embayment system was performed to assist the Town of Yarmouth 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 Yarmouth 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 Parkers 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 Parkers River embayment system.

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

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

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

The Town of Yarmouth has recognized the severity of the problem of eutrophication and the need for watershed nutrient management and is currently developing a Comprehensive Wastewater Management Plan which the Town plans to implement upon its completion. The Town of Yarmouth has been working with the Town of Barnstable that has also completed and implemented wastewater planning in other nearby regions not associated with the Parkers River system, specifically the Lewis Bay embayment system which it shares with Barnstable. Moreover, the Town of Yarmouth is working collaboratively with the Town of Dennis relative to the MEP nutrient threshold analysis of the Bass River system to the east of the Parkers River system. In this manner, this analysis of the Parkers River system will be combined in the future with the nutrient threshold analysis of Bass River and the already completed analysis of Lewis Bay in order to give the Town of Yarmouth the necessary results to plan out and implement a unified town-wide approach to nutrient management. The Town of Yarmouth with associated work groups have recognized that a rigorous scientific approach yielding site-specific nitrogen loading targets was required for decision-making and alternatives analysis. The completion of this multi-step process has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, which is a partnership effort between all MEP collaborators and the Towns. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns' nutrient management groups to predict the impacts on water quality from a variety of proposed management scenarios.

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

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

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

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

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

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

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

The nitrogen thresholds developed in Section VIII-2 were used to determine the amount of total nitrogen mass loading reduction required for restoration of eelgrass and infaunal habitats in the Parkers 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 Parkers 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 Parkers River embayment system in the Town of Yarmouth. Future water quality modeling scenarios should be run which incorporate the spectrum of strategies that result in nitrogen loading reduction to the embayment, however, within this report an additional hydrodynamic analysis was completed on a bridge widening scenario for the Route 28 bridge passing over the Parkers River. The MEP analysis has initially focused upon nitrogen loads from on-site septic systems as a test of the potential for achieving the level of total nitrogen reduction for restoration of each embayment system. The concept was that since nitrogen loads associated with wastewater generally represent 80% - 85% of the controllable watershed load to the Parkers 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 Parkers 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 Parkers River system showing some nitrogen related habitat impairment within each of its component basins, however, there is a strong gradient throughout this system. Seine Pond is a significantly impaired basin relative to infauna habitat, since it historically has not supported eelgrass. Nitrogen enrichment (resulting from inputs and restricted tidal exchange) has resulted in frequent large phytoplankton bloom, periodic hypoxia, macroalgal accumulations and virtual loss of benthic communities. The Parkers River is also nitrogen enriched, but has less nitrogen enrichment due primarily on its structure and high water turnover. While the lower reach currently supports only moderately impaired benthic habitat, its loss of historical eelgrass coverage indicates that it has become a significantly impaired basin relative to eelgrass habitat. Finally, Lewis Pond is a small shallow tidal basin surrounded by extensive tidal salt marsh and as such has not historically supported eelgrass. Lewis Pond currently functions as a salt marsh dominated basin with naturally high organic matter inputs and periodic low oxygen. As a salt marsh basin it is currently supporting moderately productive diverse infaunal communities. However, based upon the high chlorophyll levels and some of the infaunal indicators, currently it appears to be moderately impaired as benthic habitat. Overall, the regions of significant and moderate habitat impairment comprise >90% of the estuarine area of the Parkers River Embayment System.

Overall, the oxygen levels within the major sub-basins to the Parkers River Embayment System are indicative of high levels of nutrient enrichment and impaired habitat quality (Figures VII-3 and VII-5). The oxygen data are consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of this estuarine basin. The large daily excursions in oxygen concentration in both Seine Pond and Lewis Pond also indicate significant organic matter enrichment. While both Seine Pond and Lewis Pond share large daily excursions for dissolved oxygen, Lewis Pond generally shows a lower baseline dissolved oxygen concentration. However, the level of oxygen stress needs to be evaluated in light of the fact that Lewis Pond is a salt marsh tidal basin, while Seine Pond is a typical embayment basin. Salt marsh tidal basins are naturally organic matter enriched and typically have summertime low oxygen events (periodic hypoxia), while high quality embayment basins do not.

Generally, the dissolved oxygen records throughout the Parkers River Estuary showed moderate depletions (relative to the basin type) during the critical summer period. The greatest oxygen depletions were generally associated with the wetland dominated tributary basin of Lewis Pond, with higher oxygen levels maintained in the main embayment basin, Seine Pond. The continuous D.O. records indicate that the upper region of the Parkers River Embayment System, defined by the open water portion that is Seine Pond, shows regular oxygen depletion (below 6.0 mg/L) during summer with periodic depletions below 4.0 mg/L, consistent with its nitrogen and organic matter rich waters (Table VII-1, Figure VII-3). The Parkers River also shows oxygen depletions in the upper (3.6 mg L-1) and lower (4.4 mg L-1) reaches, as well as moderate to high chlorophyll levels. Oxygen levels in the Lewis Pond basin of the Parkers River Estuary were generally lower than those measured in Seine Pond despite being situated closer to the inlet to the overall system. The mooring deployment in Lewis Pond revealed very frequent oxygen depletions to <4 mg L^{-1} and periodically to <3 mg L^{-1}. However, it should be noted that these depletions occur within a salt marsh dominated shallow tidal basin, surrounded by extensive salt marsh. As such, Lewis Pond's oxygen depletion results primarily from the naturally organic matter and nutrient rich qualities of such an environment.

At present, the absence of eelgrass throughout the Parkers River Estuary is consistent with the observed nitrogen and the chlorophyll levels and functional basin types comprising this estuary. The lower Parkers River basin supported eelgrass beds in 1951 under lower nitrogen loading conditions. The historical distribution of eelgrass and its present absence within the Parkers River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. Shallow salt marsh basins, like Lewis Pond typically do not support eelgrass beds. Similarly, the tidally restricted upper estuary (upper River and Seine Pond) has likely been nutrient enriched with poor water clarity over many decades. In contrast, at lower overall nitrogen loading, it would be expected that the lower River areas would have sufficient water clarity and oxygen levels to support eelgrass beds. However, given the sensitivity of eelgrass to declining light penetration resulting from nutrient enrichment and secondary effects of organic enrichment and oxygen depletion, the current absence of eelgrass within this system is expected given the high nitrogen levels and high chlorophyll levels measured in all basins. Typically eelgrass beds exist at much lower nitrogen levels (0.35 - 0.45 mg N L^{-1}) than presently found in this system (0.66 - 0.99 mg N L^{-1}). The high nitrogen levels within the Parkers River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates.

The Infauna Survey clearly indicated impaired habitat within Seine Pond and to a lesser extent within the lower reach of the Parkers River and Lewis Pond. Seine Pond is currently supporting very poor benthic habitat throughout the basin. There are very few species and very few individuals (i.e. low secondary production), with an average of 6 species and 48 individuals per sample. These numbers are very low compared to other high quality areas in Cape Cod estuaries. In addition, Seine Pond benthic communities have very low Diversity (1.25) and Eveness <1 (0.65) and are dominated by stress indicator species associated with organic enrichment. In contrast to Seine Pond, the Parkers River lower reach currently supports species numbers and population levels more in line with a high quality benthic animal habitat. Although the Diversity (2.94) and Eveness (0.61) indices suggest a moderate level of impairment, as do the types of species present (e.g. dense amphipod mats and tubificids). Amphipod mats are typical of transitional environments and were the major communities to develop in Boston Harbor as nutrient loads began to diminish. These animal communities are consistent with the level of nitrogen enrichment and moderate chlorophyll levels and oxygen conditions within this estuarine basin. All of these parameters indicate a system that is supporting moderately impaired benthic habitat. Lewis Pond showed infaunal communities consistent with a salt marsh basin with relatively high quality benthic habitat. As expected from its marsh setting, the Lewis Pond benthic community consists mainly of species indicative of an organic rich environment which is normal for a salt marsh. The community was "patchy" with areas dominated by the stress indicator, tubificoides, although other areas were typical of salt marsh basins with high quality habitat.

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

3. Conclusions of the Analysis

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

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

Watershed nitrogen loads (Tables ES-1 and ES-2) for the Town of Yarmouth Parkers River embayment system was comprised primarily of wastewater nitrogen. Land-use and wastewater analysis found that generally about 80% - 85% of the controllable watershed nitrogen load to the embayment was from wastewater.

A major finding of the MEP clearly indicates that a single total nitrogen threshold can not be applied to Massachusetts' estuaries, based upon the results of the Great, Green and Bournes Pond Systems, Popponesset Bay System, the Hamblin / Jehu Pond / Quashnet River analysis in eastern Waquoit Bay, the analysis of the nearby Lewis Bay and Rushy Marsh system and the Pleasant Bay and Nantucket Sound embayments associated with the Town of Chatham. This is almost certainly going to be true for the other embayments within the MEP area, as well, inclusive of Parkers River.

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

Parkers River Threshold Nitrogen Concentrations

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

- The primary habitat issue within the Parkers River Embayment System relates to the loss of eelgrass beds from the lower portion of the main channel that constitutes a portion of the Parkers River system. This loss of eelgrass classifies this area of the system as "significantly impaired", although Lewis Pond, a sub-basin in the lower part of the overall system presently supports generally high quality infaunal communities. The impairments to both the infaunal habitat in Seine Pond and the eelgrass habitat within the lower portion of the system are supported by the variety of other indicators, oxygen depletion, chlorophyll, and TN levels, which support the conclusion that these impairments are the result of nitrogen enrichment, primarily from watershed nitrogen loading.

- The results of the water quality and infaunal data, coupled with the temporal trends in eelgrass coverage, clearly support the need to lower nitrogen levels within the Parkers River system in order to restore eelgrass habitat. The observed loss of eelgrass, presence of drift algae, moderate oxygen and chlorophyll levels and benthic community structure within the lower Parkers River basin, suggests a system beyond the nitrogen threshold level that would be supportive of eelgrass, but relatively close to the level for supporting high quality infaunal habitat. The average nitrogen levels for this lower reach were 0.663 mg N L^{-1} at mid ebb tide, slightly above the level supportive of infaunal communities (0.5-0.6 mg N L^{-1}), but well above levels supportive of eelgrass beds as found in a variety of MEP assessments of Cape Cod estuaries. Similarly, the total nitrogen levels at mid-ebb tide within Seine Pond (0.948-0.994 mg N L^{-1}) are well above levels found in basins supportive of high quality benthic animal habitat.

- With the sentinel station located at the uppermost extent of the historical eelgrass coverage (between the long-term water quality monitoring stations within the lower reach of the River (PR-2 & PR-3), the target nitrogen concentration (tidally averaged TN) for restoration of eelgrass at the sentinel location within the lower reach of the Parkers River was determined to be 0.42 mg TN L^{-1}. As there has not been eelgrass habitat within the Parkers River Estuary for over a decade, this threshold was based upon comparison to other local embayments of similar depths and structure under MEP analysis.

- Although the nitrogen management target is restoration of eelgrass habitat (and associated water clarity, shellfish and fisheries resources), benthic infaunal habitat quality must also be supported as a secondary condition. Benthic animals are more tolerant of nutrient and organic matter enrichment than eelgrass, which requires clear waters and high oxygen levels. At present, in the regions with moderately impaired infaunal habitat within the lower reach of the Parkers River, average ebb tide total nitrogen (TN) levels are in the range of 0.65 mg N L^{-1}. The observed moderate impairment at this site is consistent with observations by the MEP Technical Team in other enclosed basins along Nantucket Sound (e.g. Perch Pond, Bournes Pond, Popponesset Bay) where levels <0.5 mg N L^{-1} were found to be supportive of healthy infaunal habitat and where moderately impaired habitat was found at ~0.6 mg N L^{-1}.

- Based on all indicators, the lowering of nitrogen levels will also be necessary to restore infaunal habitat within Seine Pond and to a lesser extent within Lewis Pond. The MEP Technical Team concluded that an upper limit of 0.50 mg N L-1 tidally averaged TN would support healthy infaunal habitat in the Seine Pond sub-embayment basin of the Parkers River Estuary as well as the Parkers River above the Route 28 bridge. However, given the shallow nature of Lewis Pond and its function as primarily a salt marsh basin, a tidally averaged TN of <0.60 mg N L^{-1} was determined. This higher level stems from the much shallower basin of Lewis Pond. This infaunal nutrient threshold is reasonable by comparison to Scudder Bay which shares similar characteristics and the similar infaunal threshold for the shallow salt marsh dominated upper reach of the Mashpee River. It is likely that restoration of the impaired infaunal habitats within Seine Pond and the Parkers River will be achieved with the restoration of eelgrass habitat within the lower reach of the River.

- It should be emphasized that these secondary criteria values were not used for setting nitrogen thresholds in this embayment system. These values merely provide a check on the acceptability of conditions within the tributary basins (Seine Pond average of PR-1 & PR-5; Lewis Pond PR-4) at the point that the threshold level is attained at the sentinel station within the lower Parkers River..

For restoration of the Parkers River Embayment System, both the primary nitrogen threshold at the sentinel station and the secondary criteria within the sub-embayments need to be achieved. However, the secondary criteria established by the MEP are to merely provide a check on the acceptability of conditions within the tributary basins at the point that the threshold level is attained at the sentinel station. It should be emphasized that these secondary criteria values were not used for setting nitrogen thresholds in this embayment system. The results of the Linked Watershed-Embayment modeling are used to ascertain that when the nitrogen threshold is attained, TN levels in the regions associated with the secondary criteria are within the acceptable range. The goal is to achieve the nitrogen target at the sentinel location and restore eelgrass habitat throughout lower region of the Parkers River system as well as infaunal habitat throughout the embayment

It is important to note that the analysis of future nitrogen loading to the Parkers 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 Parkers River estuarine system is that restoration will necessitate a reduction in the present (2004) nitrogen inputs and management options to negate additional future nitrogen inputs.

Table ES-1. Existing total and sub-embayment nitrogen loads to the estuarine waters of the Parkers River system, observed nitrogen concentrations, and sentinel system threshold nitrogen concentrations. Loads to estuarine waters of the Parkers River system include both upper watershed regions contributing to the major surface water inputs.

Sub-embayments	Natural Background Watershed Load [1] (kg/day)	Present Land Use Load [2] (kg/day)	Present Septic System Load (kg/day)	Present WWTF Load [3] (kg/day)	Present Watershed Load [4] (kg/day)	Direct Atmospheric Deposition [5] (kg/day)	Present Net Benthic Flux (kg/day)	Present Total Load [6] (kg/day)	Observed TN Conc. [7] (mg/L)	Threshold TN Conc. [8] (mg/L)
PARKERS RIVER SYSTEM										
Seine Pond	0.293	3.57	16.992	-	20.562	1.096	-5.820	15.837	0.95-0.99	0.50
Upper Parkers River	0.926	3.7913	12.340	0.277	16.408	0.049	0.775	17.233	0.78	-
Power Parkers River	0.079	0.901	11.751	-	12.652	0.266	28.420	41.338	0.66	0.45
Lewis Pond	0.362	2.718	14.682	-	17.400	0.616	5.698	23.715	0.87	0.55
Parkers River System Total	**1.660**	**11.258**	**55.764**	**0.2767**	**67.022**	**2.027**	**29.074**	**98.123**	**0.66-0.99**	**0.45-0.55**

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 attenuated wastewater treatment facility discharges to estuary
4 composed of combined present land use, septic system, and WWTF 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 data collected between 2002 and 2008, ranges show the upper to lower regions (highest-lowest) of the indicated sub-embayment.
8 eelgrass threshold for sentinel station in lower Parkers River, benthic infauna threshold for sentinel stations located in Seine Pond and Lewis Pond.

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

Sub-embayments	Present Watershed Load [1] (kg/day)	Target Threshold Watershed Load [2] (kg/day)	Direct Atmospheric Deposition (kg/day)	Benthic Flux Net [3] (kg/day)	TMDL [4] (kg/day)	Percent watershed reductions needed to achieve threshold load levels
PARKERS RIVER SYSTEM						
Seine Pond	20.562	4.080	1.096	-2.227	2.949	-80.2%
Upper Parkers River	16.408	4.439	0.049	0.409	4.897	-72.9%
Power Parkers River	12.652	1.489	0.266	16.261	18.016	-88.2%
Lewis Pond	17.400	3.452	0.616	3.300	7.369	-80.2%
Parkers River System Total	**67.022**	**13.459**	**2.027**	**17.744**	**33.230**	**-79.9%**

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

ACKNOWLEDGMENTS

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

First and foremost we would like to recognize and applaud the significant time and effort in data collection and discussion spent by members of the Town of Yarmouth Water Quality Monitoring Program and SMAST “volunteers”. These individuals gave of their time to collect nutrient related water quality from this system, without which the present analysis would not have been possible.   Of particular note are George Allaire of the Department of Public Works, and Karl VonHone, Director of the Natural Resources Department and Coordinator of the Town of Yarmouth Monitoring Program. Similarly, many in the Town of Yarmouth helped in this effort, the Yarmouth Department of Public Works the Yarmouth Shellfish Department and the Yarmouth Planning Department. The MEP Technical Team would also like to thank Maureen Rukstalis from the Historical Society of Old Yarmouth for the time she instantly took to provide historical perspective on the geography of Seine Pond and the upper portion of Parker's River.

In addition to local contributions, technical, policy and regulatory support has been freely and graciously provided by Paul Niedzwiecki and Tom Cambareri of the Cape Cod Commission, which also supported the current report financially in order to keep the Town of Yarmouth moving forward with its nutrient management planning. We are also thankful for the long hours in the field and laboratory spent by the technical staff (Jen Antosca, Michael Bartlett, Sara Sampieri and Dahlia Medieros), interns and students within the Coastal Systems Program at SMAST-UMD.

Support for this project was provided by the Town of Yarmouth, Barnstable County and the University of Massachusetts-Dartmouth.

CITATION

Howes B., S. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2009). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Parkers River Embayment System, Yarmouth, Massachusetts, Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

I. INTRODUCTION

The Parkers River Embayment System is a complex estuary located within the Town of Yarmouth on Cape Cod, Massachusetts and which exchanges tidal waters with Nantucket Sound to the south (Figure I-1). The Parkers River Estuary is comprised of a tidal river connecting a large kettle pond, Seine Pond, to the Sound and a small lagoonal estuary formed perpendicular to the tidal river behind a barrier spit. The spit was formed from marine sands and gravel deposited by coastal processes during post-glacial sea-level rise. Drowning of the river valleys to form tidal rivers occurred gradually as a result of rising sea level following the last glaciation approximately 18,000 years BP. The lagoon is composed of a shallow open water basin, Lewis Pond (~0.5 m), surrounded by extensive emergent salt marsh. The salt marsh has filled much of the lagoon, but a small narrow tidal channel still connects the basin to the river. The major basin of the Parkers River system is Seine Pond. This uppermost basin is deeper than Lewis Pond, but is still relatively shallow (1-1.5 m). This system while dominated by open water still supports extensive wetlands, primarily in the region of the entrance to the upper Parkers River. However, the shoreline of Seine Pond is primarily characterized as upland (with forest and single family residential development). The tidal river, Parkers River, is functionally divided into an upper and lower river by the Route 28 bridge. Salt marsh dominates the shoreline along the upper river and the lower western shore of the river south of Route 28. This is in contrast to the densely developed eastern shore of the Parkers River system south of Route 28.

The Massachusetts Estuaries Project (MEP) was able to determine that as far back as 1795, Seine Pond was connected to the Parkers River, hence Nantucket Sound (i.e. it was tidal). This connection is seen in drawings and maps: 1795 drawing T. Thatcher; 1830, 1858 maps (personal communication, M. Rukstalis Historical Society of Old Yarmouth)[1]. It is also important to note that early maps denote Seine Pond as Swan Pond. However, later 20th Century USGS topographical maps consistently use the name Seine Pond, as such the MEP uses Seine Pond as the name of the upper basin.

The watershed of the Parkers River system is distributed entirely within the Town of Yarmouth. A large portion of the overall watershed includes the sub-watersheds contributing direct groundwater discharge to the estuary or to Plashes Brook that flows into the tidal river just below Seine Pond (Plashes Brook). In addition, Seine Pond receives freshwater inflow through a herring run from Long Pond. This run was recently improved to increase freshwater flow to enhance herring passage. Although land-uses closest to an embayment generally have greater impact than those in the inland portions of the watershed, which are subject to nitrogen attenuation during transport through natural aquatic systems (e.g. ponds, rivers, wetlands etc.) prior to discharge to the embayment, effective restoration of the Parkers River System, will require the Town of Yarmouth to be active in nutrient management throughout the watershed to the overall system. However, management will be simplified since all of the watershed nitrogen sources to the Parkers River System reside within the Town of Yarmouth.

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

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

The number of sub-embayments (Parkers River, Lewis Pond and Seine Pond) comprising the Parkers River System greatly increases the shoreline and decreases the travel time of groundwater from the sites of watershed nitrogen inputs to estuarine regions of discharge. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, the Parkers River system and its sub-embayments, along the southern shore of the Town of Yarmouth, is at risk of eutrophication (over enrichment) from high nitrogen loads in the groundwater and runoff from their watersheds, in a similar fashion as nearby Lewis Bay. However, the physical structure of the Parkers River System, with a large upper pond increases the sensitivity of this basin to nitrogen enrichment.

The Parkers River Embayment System is a complex (drown river valley + lagoonal) estuary exchanging tidal waters with the high quality waters of Nantucket Sound through a single inlet that is "fixed" by jetties (Figure I-1). The barrier spit is used for recreation, primarily Seagull Beach to the west and Thachers Beach to the east of the inlet. The Parkers River Estuary contains high salinity waters throughout its tidal reaches, generally 30 parts per thousand (ppt) near the inlet and 25 ppt in Seine Pond. The high salinities reflect the dominance of tidal flows, rather than freshwater inflows, in structuring these systems. Prior to development and armoring of the tidal inlet sediment transport and deposition associated with coastal processes including coastal storms likely caused the inlet to migrate and certainly resulted in periods of lower tidal flows due to deposition of sands within the tidal channel. This may have lead to periodic lowering of salinities compared to the present condition. Currently, both the east and west sides of the inlet are armored and stable and the Town of Yarmouth periodically dredges the inlet channel to keep the inlet navigable and maintain tidal flows.

The primary ecological threat to the Parkers River system as a coastal resource is degradation resulting from nutrient enrichment. Although the enclosed estuarine system has some bacterial contamination issues related to stormwater run-off from the watershed and likely animal sources primarily associated with wetland, these do not appear to be having large system-wide impacts. Bacterial contamination causes periodic closures of shellfish harvest areas within the Lewis Pond and Seine Pond (SC30 and SC31 growing areas) sub-embayments. In contrast, loading of the critical eutrophying nutrient, nitrogen, to the Parkers River System has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to the Parkers River Estuary, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater.

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

As the primary stakeholder to the Parkers River System, the Town of Yarmouth was among the first communities to become concerned over perceived degradation of its coastal waters. Concern over declining habitat quality within its embayments led directly to the establishment of a comprehensive water quality monitoring program aimed at determining the degree to which waters of the Town's embayments may be impaired. The Town of Yarmouth Water Quality Monitoring Program was provided technical assistance by the Coastal Systems Program at SMAST-UMD. Over the past several years the Yarmouth Program has operated in a coordinated manner with the Town of Barnstable water quality monitoring program as a result of the shared embayment of Lewis Bay. In addition to assessing the health of the estuaries in Yarmouth, the water quality monitoring program provides the required quantitative watercolumn nitrogen data (2002-2008) for validation of the MEP's Linked Watershed-Embayment Approach used in the present study. Entry into the MEP and TMDL compliance monitoring depends upon Town supported water quality monitoring, as guided by SMAST.

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

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

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

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

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

The Massachusetts Estuaries Project represents the next generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MassDEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts. The MEP approach was selected after extensive review by the MassDEP and USEPA and associated scientists and engineers. It has subsequently been applied to more than 30 estuaries and reviewed by other state agencies, municipalities, non-profit environmental organizations, engineering firms, scientists and private citizens. Over the course of the extensive reviews, the MEP approach has proven to be robust and capable of yielding quantitative results to support management of a wide variety of estuaries.

The Massachusetts Estuary Project is founded upon science-based management. The Project is using a consistent, state-of-the-art approach throughout the region's coastal waters and providing technical expertise and guidance to the municipalities and regulatory agencies tasked with their management, protection, and restoration. The overall goal of the Massachusetts Estuaries Project is to provide the DEP and municipalities with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system will serve as the basis for the development of Total Maximum Daily Loads (TMDLs). Development of TMDLs is required pursuant to Section 303(d) of the Federal Clean Water Act. TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an outline of an implementation plan. For this project, the DEP recognizes that there are likely to be multiple ways to achieve the desired goals, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, DEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly) through the Comprehensive Wastewater Management Planning process.

In appropriate estuaries, bacterial technical reports will be developed in support of a Cape Cod wide TMDL for bacterial contamination. As possible, these analyses of bacterial contamination will be conducted in concert with the nutrient effort (particularly if there is a 303d listing), as was the case for the Prince's Cove sub-embayment to the Three Bays system. Currently, the MEP (through SMAST) has not been tasked with a technical assessment of bacterial contamination in the Parkers River System for inclusion of this system into the Cape Cod wide bacterial TMDL that the MassDEP produced a few years ago.

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

The major MEP nitrogen management goals are to:

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

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

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

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

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

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

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

I.2 SITE DESCRIPTION

The Parkers River Embayment System is a complex (drown river valley + lagoonal) estuary exchanging tidal waters with the high quality waters of Nantucket Sound through a single inlet that is "fixed" by jetties (Figure I-1). The Parkers River Embayment System consists of a main tidal channel linking Nantucket Sound waters to two terminal salt ponds. The inlet connecting Nantucket Sound to Parkers River was originally a natural inlet that migrated and filled in as a function of natural longshore transport processes and was ultimately armored to the east (Thachers Beach) and west (Seagull Beach) in the early 1900's. For the MEP analysis, the Parkers River Estuarine System has been partitioned into four general sub-embayment groups: the 1) Lewis Pond 2) lower Parkers River below Route 28 3) upper Parkers River upgradient of Route 28 and 4) Seine Pond as depicted in Figure I-1.

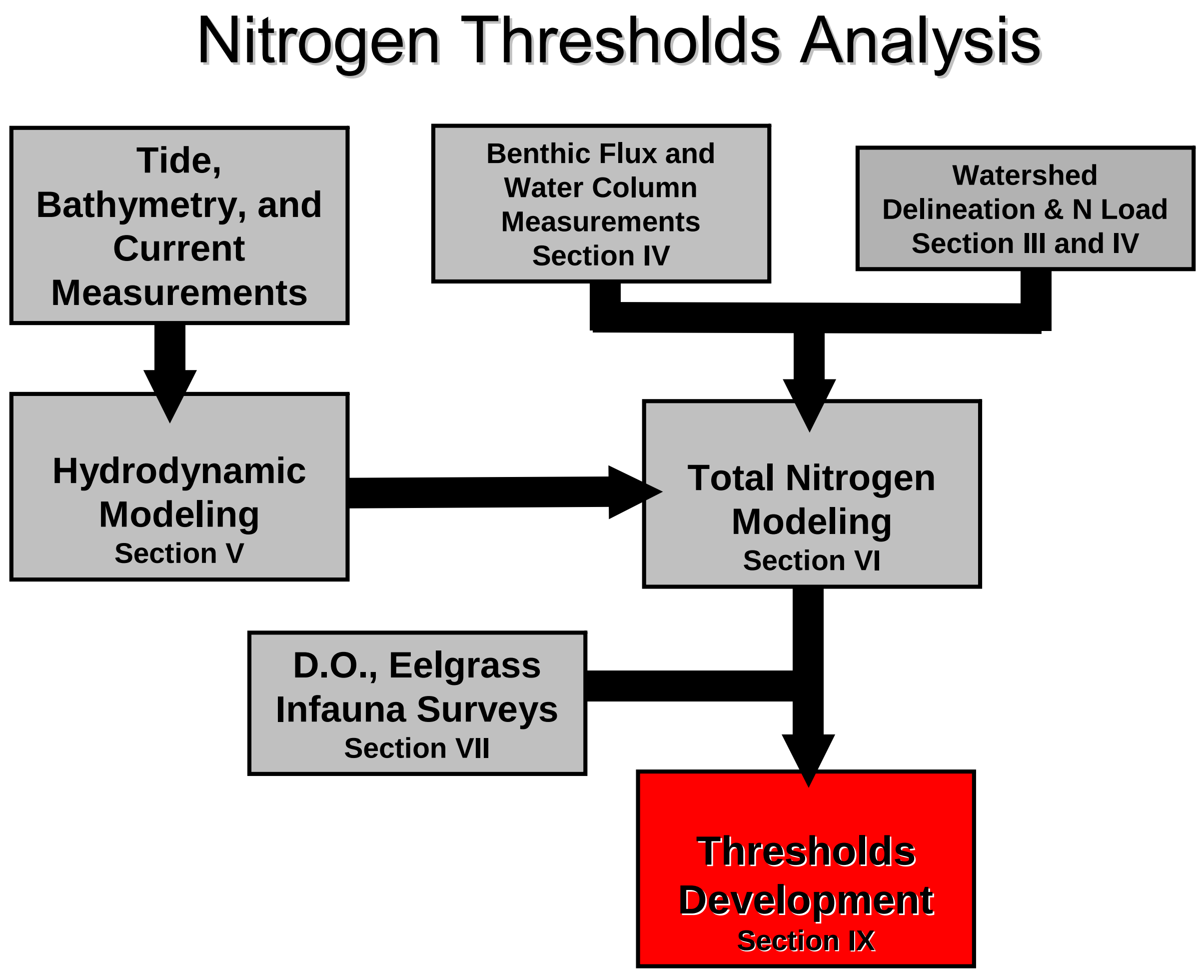


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Note that the approach is not a single model, but a series of models linked by scientists and engineers who validate outputs and inputs.

Within the overall Parkers River System is seen a diversity of estuarine habitats, including the main tidal channel portion of Parkers River which operates as tidal river, Lewis Pond with an associated tidal channel and surrounding large salt marsh area operating primarily as a salt marsh sub-system and the uppermost terminal basin of Seine Pond that is like a shallow kettle basin with some associated fringing wetland and some undeveloped wooded upland along the pond shore and serves as the main embayment basin. Most of the System's salt marsh area is associated with Lewis Pond and the western shore of Parkers River.

The Parkers River Estuary contains high salinity waters throughout its tidal reaches, generally 30 parts per thousand (ppt) near the inlet and 25 ppt in Seine Pond. The high salinities reflect the dominance of tidal flows, rather than freshwater inflows, in structuring these systems. Tidal forcing for this embayment system is generated from Nantucket Sound. Nantucket Sound exhibits a moderate to low tide range, with a mean range of about 2.5 to 3.5 ft. Since the water elevation difference between Nantucket Sound and the Parkers River System is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, Wellfleet Harbor is ~10 ft).

Tidal damping (reduction in tidal amplitude) through an embayment can range from negligible, indicating "well-flushed" conditions, or show tidal attenuation caused by constricted channels and marsh plains, indicating a "restrictive" system, where tidal flow and the associated flushing are inhibited. Tidal data indicated only minimal tidal damping through the inlet into the main tidal channel of the Parkers River system down gradient of the Route 28 bridge crossing and into Lewis Pond. It appears that the tidal inlet is operating efficiently being periodically dredged to maintain navigation and flushing. In contrast, tidal damping was observed for the portion of the system upgradient of the Route 28 Bridge into Seine Pond (present tide range 0.9 ft). The tide propagates to the sub-embayment of Seine Pond with attenuation, where as there is no attenuation within Lewis Pond, consistent with moderately well-flushed conditions throughout.

I.3 NUTRIENT LOADING

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

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

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

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

Unfortunately, the upper reaches of the Parkers River System are beyond their ability to assimilate additional nutrients without impacting their ecological health. The MEP analysis clearly indicates that the system is presently impaired by nitrogen overloading and does not meet the Commonwealth's Water Quality Standards. Nitrogen levels are elevated throughout the System and eelgrass beds have not been observed within the Parkers River system as far back as 1995. The large fish kill during July 2009 provides further dramatic evidence of habitat impairment. It is important to note that the present nitrogen enrichment of the Parkers River Estuarine System results from the combination of increasing nitrogen loading to its contributing watershed coupled to reduced flushing of nitrogen due to tidal restriction in the upper portions of the system. The MEP analysis evaluates both of these processes and any efficient management plan will likely include modifications to both loading and flushing.

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

I.4 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Parkers River System, including the tributary sub-embayments of Lewis Pond and Seine Pond. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for each of the systems. Once the hydrodynamic properties of each estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all watershed sourced nitrogen entering the Parkers River System is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Parkers River system were provided by the Town of Yarmouth Water Quality Monitoring Program (a coordinated effort between the Town of Yarmouth and the Coastal Systems Program at SMAST). Measurements of the salinity and nitrogen distributions throughout estuarine waters of the Parkers River System (2003-2008) 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 Parkers River System for the

Town of Yarmouth. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Town of Yarmouth and Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Nantucket Sound (Section IV). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section IV. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration/protection of the River in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined system threshold for restoration or protection. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for this system. Finally, analyses of the Parkers River System were undertaken relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging options to improve nitrogen related water quality. The results of the nitrogen modeling for each scenario have been presented in Section IX with references provided in Section X.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

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

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

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

A number of studies relating to nitrogen loading, hydrodynamics and habitat health have been conducted within the Parkers River System over the past decade.

Preliminary Watershed-Embayment Assessment. - Prior to the Massachusetts Estuaries Project, a preliminary analysis of the circulation and nitrogen loading of the Parkers River System was performed (ca. 2001) for the Bureau of Resource Protection, Massachusetts Department of Environmental Protection (Tetra Tech EM Inc. 2005). The study included an initial assessment of watershed nitrogen loading to the Parkers River Estuary and was based upon an earlier generation watershed loading model (Waquoit Bay Nitrogen Loading Model, Valiela et. al., 1996) that pre-dates the MEP Linked Watershed-Embayment Modeling Approach. The earlier loading analysis did not use the watershed developed by the USGS, but rather the watershed derived by the Cape Cod Commission (1998). It was also based upon available water table elevations. It should be noted that the present USGS based watershed developed for the MEP is 9% smaller (259 acres) than the 1998 Commission delineation. Additionally, there are some differences in spatial coverage (see Chapter III). The historical watershed loading estimates must be considered approximate, as land-use/N sources were based upon MASSGIS land-use data rather than local municipal assessors data and water-use. For example, an occupancy rate of 1.79 people per house for each system was used compared to the 2000 Census data for Barnstable showing a rate of 2.44 people per house. In addition, no assessment of seasonal occupancy was made. Equally important, no site-specific nitrogen attenuation measurements were made relative to the surface freshwater systems within the watershed (lakes, ponds, streams).

Given that this was a preliminary effort and did not use site-specific land-use or nitrogen attenuation factors, had limited data collection to support the hydrodynamic modeling and did not calibrate/validate the water quality model, and since the watershed delineation has been refined by the MEP and USGS using the West Cape groundwater model, the MassDEP has determined that the present MEP assessment and modeling effort supersedes its previous project and provides the accuracy needed for watershed nitrogen management planning, under the CWMP process.

Town of Yarmouth Water Quality Monitoring Program - The Town of Yarmouth partnered with SMAST-Coastal Systems Program scientists in 2002 to develop and implement a nutrient related water quality monitoring program of the Town's estuaries. The Town of Yarmouth coordinated with the Town of Barnstable for surveys of the Lewis Bay System, but was solely responsible for the data collection throughout the Parkers River Estuary. For Parkers River specifically, the focus of the effort has been to gather site-specific data on the current nitrogen related water quality throughout its estuarine reach to support assessments of habitat health. This baseline water quality data are a prerequisite to entry into the MEP and the conduct of its Linked Watershed-Embayment Approach. Water quality monitoring of the Parkers River System has been a coordinated effort between the Town of Yarmouth Natural Resources Department (Karl Von Hone) and The Coastal Systems Program at SMAST-UMD. The water quality monitoring program was initiated in 2002 with support from the Town of Yarmouth and continued uninterrupted through the summer of 2008. After the first three years of baseline water quality data were captured, the sampling program was reduced in terms of the number of sampling events conducted each summer. In the first three years of monitoring, six sampling events were undertaken each summer between June and early September. In subsequent years, the number of sampling events completed in a given summer was reduced from 6 to 4 events. The Yarmouth Water Quality Monitoring Program for Parkers River developed the baseline data from sampling stations distributed throughout the main tidal channel and tributary sub-systems such as Lewis Pond and the head of the system, Seine Pond (Figure II-1). Additionally, as remediation plans for this and other various systems are implemented throughout the Town of Yarmouth, continued monitoring is planned to provide quantitative information to the Towns relative to the efficacy of remediation efforts.

Implementation of the MEP's Linked Watershed-Embayment Approach incorporates the quantitative water column nitrogen data (2002-2008) gathered by the Monitoring Program and watershed and embayment data collected by MEP staff. The MEP effort also builds upon previous watershed delineation and land-use analyses, the previous embayment hydrodynamic modeling and historical eelgrass surveys. This information is integrated with MEP higher order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Parkers River Estuarine System. The MEP has incorporated all appropriate data from previous studies to enhance the determination of nitrogen thresholds for the Parkers River System and to reduce costs to the Town of Yarmouth.

Figure II-1. Town of Yarmouth Water Quality Monitoring Program. Estuarine water quality monitoring stations sampled by the Town and volunteers. Stream water quality stations depicted in Chapter 4 sampled weekly by the MEP.

Preliminary Assessment of Tidal Restriction by MEP Technical Team - As part of the MEP process the Massachusetts Estuaries Project Technical Team gives periodic updates of project status and key findings to the partnering towns relative to specific estuaries. During these update meetings between the MEP Technical Team and the Town of Yarmouth DPW and their wastewater consultants (Camp, Dresser & McKee) it became clear that remediating the tidal restriction at the Rt. 28 Bridge (Figure II-1) over the Parkers River would likely partially resolve the present level of nitrogen enrichment within Seine Pond. The Massachusetts Estuaries Project Technical Team with support from the Town of Yarmouth conducted a preliminary analysis of tidal flushing, relative to understanding the potential for improving water quality in Seine Pond by reducing the tidal restriction within the Parkers River. Technical Team Hydrodynamic Lead, John Ramsey, conducted the analysis[2]. The hydrodynamic results were presented as a draft of MEP Chapter 5 although refinements have being made since that time, none of which affected the earlier tidal restriction analysis. The results were also summarized in a companion MEP Technical Memorandum[3].

This preliminary analysis used the Technical Team's field calibrated and validated hydrodynamic model of the Parkers River estuary under present conditions to evaluate the effects that widening the opening of the Route 28 Bridge over the Parkers River (Figure II-1) would have on tidal flushing. The tidal restriction can be seen clearly in the reduced tidal range measured by the MEP gages in the upper versus lower portions of the estuary (see Chapter V).

The modeling results indicated that widening the culvert (to 30') had little affect for the areas south of the Route 28, but caused a large increase in the tidal prism in the upper estuary. The average tidal prism is increased by more than 35% in Seine Pond, from 3,460,000 ft3 to 4,688,000 ft3. This increased tidal prism is a clear sign that the wider bridge opening would increase the ease with which water is exchanged between the southern part of the system and the northern part. This improvement is further demonstrated by the decrease in residence times for Seine Pond (from 5.1 days to 3.8 days) and for the local residence time (from 2.4 days to 1.8 days). Due to the significant increase in flow through the enlarged opening, slightly less water was projected to flow into Lewis Bay. However, the change in tidal flushing would have a negligible impact on water quality. Although an increase in tidal flushing alone cannot be used to directly quantify improvement in water quality, the substantial increase in tidal prism to Seine Pond (>35%) resulting from the 30 ft culvert likely will cause a substantial improvement in water quality to the upper reaches of the estuarine system. However, it was noted that some watershed nitrogen management would likely still be needed to restore habitat quality within the upper estuary.

It should be noted that this 2008 analysis was preliminary and that it was used primarily to develop modeling scenarios that are presented in Chapter IX of the present MEP report. The data and results from this effort are fully incorporated into Chapter V of the present report.

---

[2] J. Ramsey and S. Kelley. July 2008. Preliminary hydrodynamic analysis of the Parkers River with analysis of potential tidal restriction/remediation related to the Rt. 28 Bridge.

[3] B. Howes, R. Samimy, & J. Ramsey. July 2008. Technical Memorandum Summarizing Parker River Bridge-Tidal Restriction.

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

- Mouth of River designation - MassDEP (Figure II-2)
- Designated Shellfish Growing Area – MassDMF (Figure II-3)
- 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 Parkers 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-3. Location of shellfish growing areas and their status relative to shellfish harvesting as determined by Mass Division of Marine Fisheries. Closures are generally related to bacterial contamination or "activities", such as the location of marinas.

Figure II-4. Location of shellfish suitability areas within the Parkers River Estuary as determined by Mass Division of Marine Fisheries. Suitability does not necessarily mean "presence".

Figure II-5. Anadromous fish runs within the Parkers River Estuary as determined by Mass Division of Marine Fisheries. The red diamonds show areas where fish were observed. The uppermost sites are within a freshwater pond, Long Pond, which is connected by a fish run to Seine Pond. The run was improved by the Town of Yarmouth in 2007-08 and represents a surface water pathway for nutrients to enter the Parkers River System.

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

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

The Massachusetts Estuaries Project team includes technical staff from the United States Geological Survey (USGS). The USGS groundwater modelers were central to the development of the groundwater modeling approach used by the Estuaries Project. The USGS has a long history of developing regional models for the six-groundwater flow cells on Cape Cod. Through the years, advances in computing, lithologic information from well installations, water level monitoring, stream flow measurements, and reconstruction of glacial history have allowed the USGS to update and refine the groundwater models. The MODFLOW and MODPATH models utilized by the USGS organize and analyze the available data using up-to-date mathematical codes and create better tools to answer the wide variety of questions related to watershed delineation. These questions include surface water/groundwater interactions, groundwater travel times, and drinking water well impacts that have arisen during the MEP analysis of southeastern Massachusetts estuaries, including the Parkers River embayment system. The Parkers River watershed, located in Yarmouth, Massachusetts, is situated along the southern edge of Cape Cod and is bounded to the south by Nantucket Sound.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Parkers River embayment system under evaluation by the Project Team. The Parkers River estuarine system is a complex estuary consisting of a drowned river valley estuary and lagoon and includes extensive wetland dominated components such as Lewis Pond and the upper reach of the River. Additional watershed modeling was undertaken to sub-divide the overall watershed to the Parkers River system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining the land areas with groundwater travel times that are greater and less than 10 years time-of-travel to the estuary. These travel distributions within each sub-watershed are used as a procedural check to gage the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters at the time of the MEP analysis. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the Sagamore flow cell on Cape Cod. Model assumptions for calibration of the Parkers River Estuary included surface water discharges measured as part of the MEP stream flow program (2003 to 2005).

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

III.2 MODEL DESCRIPTION

Contributing areas to the Parkers River system and its various sub-watersheds, such as Bassetts Lot Pond, Plashes Pond and Plashes Brook, were delineated using a regional model of the Sagamore Lens flow cell (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Parkers River system including sub-watersheds to Lewis Pond and Seine Pond and also to determine portions of recharged water that may flow through fresh water ponds and streams prior to discharging into coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below NGVD 29 and have a uniform thickness of 10 ft. The top of layer 8 resides at NGVD 29 with layers 1-7 stacked above and layers 8-20 below. Layer 18 has a thickness of 40 feet and extends to 140 feet below NGVD 29, while layer 19 extends to 240 feet below NGVD 29. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics (up to 519 feet below NGVD 29); since bedrock is approximately 300 feet below NGVD 29 in the Parkers River area the two lowest model layers were active in this area of the model. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens.

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

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

III.3 PARKERS RIVER SYSTEM CONTRIBUTORY AREAS

The refined watershed and sub-watershed boundaries for the Parkers River embayment system, including the Seine Pond and Lewis Pond sub-estuaries (Figure III-1) were determined by the United States Geological Survey (USGS). Model outputs of the watershed boundaries were "smoothed" to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), (d) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model and (e) to address streamflow measurements collected as part of the MEP. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. The MEP sub-watershed delineation includes 10 yr time of travel boundaries. Overall, seventeen (17) sub-watershed areas, including three freshwater ponds, were delineated within the Parkers River study area.

Table III-1 provides the daily freshwater discharge volumes for various sub-watersheds as calculated from the groundwater model; these volumes were used in the salinity calibration of the tidal hydrodynamic models and to determine hydrologic turnover in the lakes/ponds, as well as for comparison to the directly measured surface water discharges. The overall estimated freshwater flow into the Parkers River system, including Seine Pond and Lewis Pond, from the MEP delineated watershed is 22,873 m3/d. This flow includes discharge from freshwater ponds located along the overall watershed boundary to the Parkers River system and the inflow through the Herring ladder from Long Pond.

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

The MEP watershed area for the Parkers River system as a whole is 26% larger (737 acres) than the 1998 CCC delineation. The majority of the difference is largely attributable to the MEP delineation including the contributing areas to Bassetts Lot Pond, the Higgins Crowell 20 wellfield, and pond areas discharging into the watershed from the Lewis Bay watershed in the MEP delineation. The MEP area delineation also includes interior sub-watersheds to various components of the Parkers River system, such as ponds and wells. These refinements are another benefit of the update of the regional groundwater model (Walter and Whealan, 2005). Interior sub-watersheds to individual freshwater ponds and public water supplies were not delineated in the CCC watersheds.

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

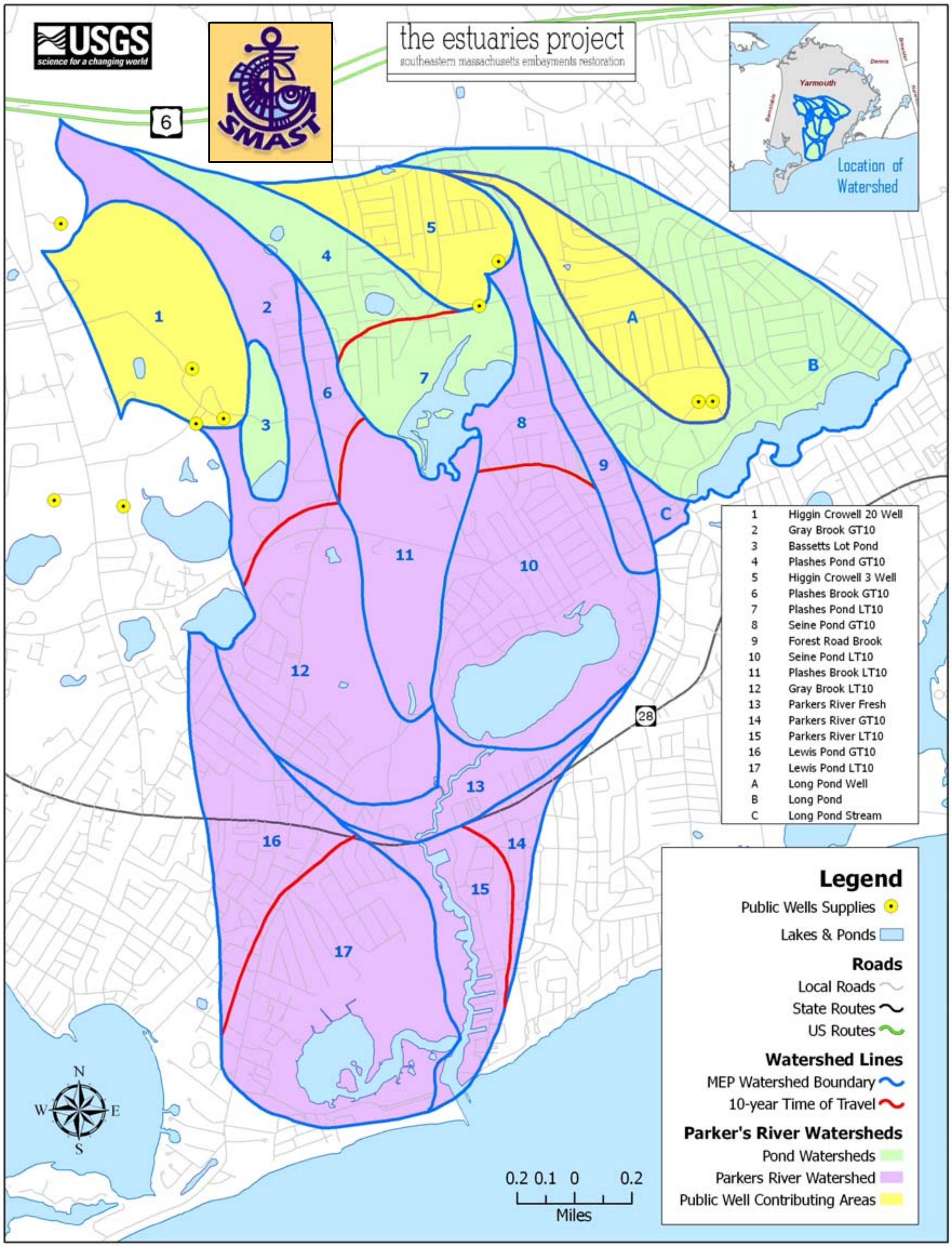


Figure III-1. Watershed delineation for the Parkers River estuary system. Subwatersheds based on USGS groundwater model output modified to better address pond and estuary shorelines and MEP stream gage measurements. Ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the watershed names (above). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI).

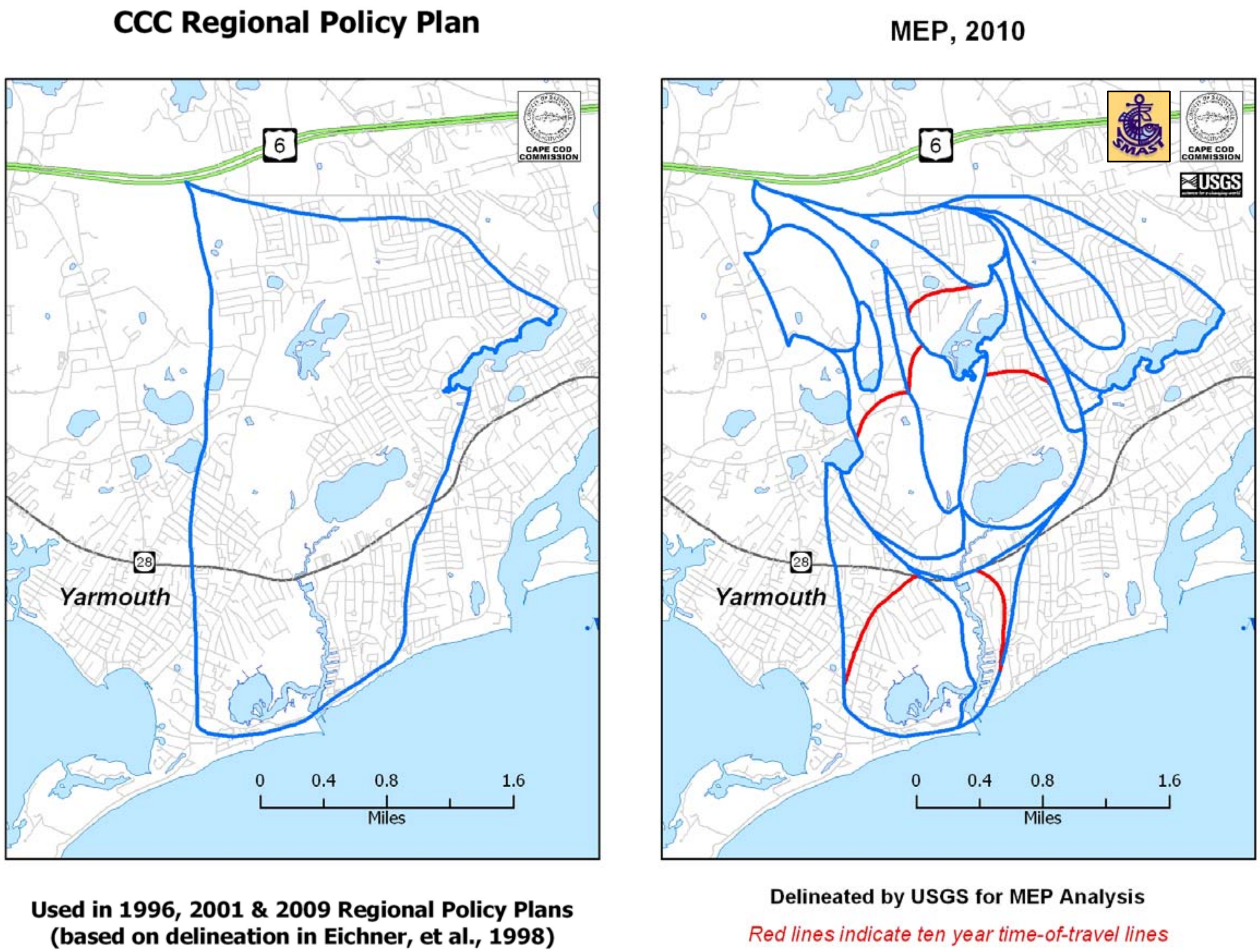


Figure III-2. Comparison of MEP watershed and sub-watershed delineations used in the current analysis and the Cape Cod Commission delineation (Eichner, et al., 1998), which has been used in three Barnstable County Regional Policy Plans (CCC, 1996, 2001, 2009).

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

Watershed	#	Watershed Area (acres)	% contributing to Estuaries	Discharge	
				m^3/day	ft^3/day
Higgins Crowell 20 Well	1	229	100	1,754	61,928
Gray Brook GT10	2	203	100	1,559	55,046
Bassetts Lot Pond	3	43	100	333	11,747
Plashes Pond GT10	4	129	100	989	34,910
Higgins Crowell 3 Well	5	144	100	1,109	39,159
Plashes Brook GT10	6	42	100	326	11,503
Plashes Pond LT10	7	154	100	1,178	41,617
Seine Pond GT10	8	83	100	634	22,372
Forest Road Brook	9	61	100	471	16,630
Seine Pond LT10	10	194	100	1,489	52,582
Plashes Brook LT10	11	185	100	1,422	50,227
Gray Brook LT10	12	294	100	2,256	79,679
Upper River	13	90	100	689	24,320
Parkers River GT10	14	47	100	358	12,640
Parkers River LT10	15	121	100	930	32,840
Lewis Pond GT10	16	192	100	1,473	52,015
Lewis Pond LT10	17	229	100	2,560	90,398
Long Pond Well	A	163	Varies depending on Well pumping and stream flow*	1,250	44,142
Long Pond	B	488		3,742	132,137
Long Pond Stream	C	17		131	4,621
From Lewis Bay watershed: Horse Pond, Big Sandy Pond, and Higgins Crowell Public Waters Supply recharge area				2,522	89,047
Sum of Watershed Areas				27,172	959,960
TOTAL PARKERS RIVER SYSTEM (MEP existing adjusted)*				22,873	807,759

Notes: 1) discharge volumes are based on 27.25 in of annual recharge over the unadjusted watershed area; 2) upgradient ponds often discharge to numerous downgradient subwatersheds, percentage of outflow is determined by length of downgradient shoreline to each subwatershed; 3) Lewis Bay watershed inflows based on MEP analysis (Howes, *et al.*, 2008); 4)* Long Pond watershed flows into Seine Pond was limited during MEP water quality collection phase, but was expanded afterwards; under MEP existing conditions flow from the combined Long Pond Well and Long Pond watershed is 353 m3/d, this was expanded to 2,466 m3/d; based on a comparison of 2001-05 pumping from the Long Pond wellfield and water use in the combined Long Pond/Long Pond Well watershed, an average of 27% of the watershed.

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

In order to determine watershed nitrogen loading inputs to the Parkers River estuary system, the MEP Technical Team developed nitrogen-loading rates (Section IV.1) to each component of estuary and its watersheds (Section III). The Parkers River watershed was sub-divided to define contributing areas or subwatersheds to each of the major inland freshwater systems and to each major portion of the estuary. Further sub-divisions were made to identify watershed areas where a nitrogen discharge reaches estuary waters in less than 10 years or greater than 10 years. A total of 20 subwatersheds were delineated in the overall Parkers River watershed, including watersheds to the following freshwater ponds: Plashes, Long, and Bassetts Lot. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to freshwater ponds and each portion of the estuary (see Chapter III).

The initial task in the MEP land use analysis is to gage whether or not nitrogen discharges to the watershed have reached the estuary. This involves a temporal review of land use changes, the time of groundwater travel provided by the USGS watershed model, and review of data at natural collection points, such as streams and ponds. Evaluation and delineation of ten-year time of travel zones are a regular part of the watershed analysis. Ten-year time of travel subwatersheds in the Parkers River watershed have been delineated for ponds, streams and the estuary itself. Because of the way the Parkers River and its tributaries flow from the inner portions of the watershed, flow and nitrogen loads from most of the upper portions of the watershed reach the estuary within 10 years. For example, nitrogen loads from the Plashes Pond watershed reach the Parkers River estuary in less than 10 years. MEP staff also reviewed land use development records for the age of developed properties in the watersheds. Based on the review of all this information, it was determined that the Parkers River estuary is currently in balance with its watershed load. Simple review of less than and greater than watersheds indicates that 72% of the unattenuated nitrogen load is within less than 10 year travel time to the estuary (Table IV-1). However, it is likely that a significant portion of the nitrogen load within the drinking water recharge areas are redistributed to areas of higher density, which are mostly located in the less than 10 year travel time areas. If adjustments are made to account for this, 79% of the unattenuated nitrogen load within the watershed is within 10 years travel time to the estuary. The overall result of the timing of development relative to groundwater travel times is that the present watershed nitrogen load appears to accurately reflect the present nitrogen sources to the estuaries (after accounting for natural attenuation, see below) and that the distinction between time of travel in the subwatersheds is not important for modeling existing conditions.

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

WATERSHED	LT10	GT10	TOTAL	%LT10
Name	kg/yr	kg/yr	kg/yr	
Seine Pond	5,487	4,056	9,543	57%
Gray Brook	4,934	1,486	6,420	77%
Parkers River Whole System	27,710	10,840	38,550	72%
Move public water supply recharge area nitrogen loads to higher density areas (LT10)				
Seine Pond	6,693	2,850	9,543	70%
Gray Brook	5,223	1,197	6,420	81%
Parkers River Whole System	30,410	8,140	38,550	79%
Notes: loads have been corrected to include portion of nitrogen added from ponds in the Lewis Bay watershed; internal subwatershed loads have also been adjusted to account for splitting of loads leaving Plashes Pond; no corrections have been made in Long Pond watershed loads; loads include atmospheric loading on the estuary surface waters				

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

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

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

Based upon the evaluation of the watershed systems, the MEP Technical Team used the Nitrogen Loading Sub-Model estimate of nitrogen loading for the subwatersheds that directly discharge groundwater to the estuary without flowing through one of these interim pond and stream measuring points. Internal nitrogen recycling was also determined throughout the tidal reaches of the Parkers 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

Estuaries Project staff obtained digital parcel and tax assessor's data from the Town of Yarmouth. Digital parcels and land use/assessors data for Yarmouth are from the year 2006. These land use databases contain traditional information regarding land use classifications (MassDOR, 2008) plus additional information developed by the town. Only Yarmouth data was necessary for the land use analysis since the watershed to Parkers River is completely contained within the Town of Yarmouth. This effort was completed with the assistance from GIS staff from the Cape Cod Commission (CCC).

Figure IV-1 shows the land uses within the Parkers River estuary watershed areas. Land uses in the study area are grouped into eight land use categories: 1) residential, 2) commercial, 3) industrial, 4) mixed use, 5) undeveloped, 6) agricultural, 7) public service/government, including road rights-of-way, and 8) freshwater features (e.g. ponds and streams). These land use categories, except the freshwater features, are aggregations derived from the major categories in the Massachusetts Assessors land uses classifications (MassDOR, 2008). "Public service" in the MassDOR system is tax-exempt properties, including lands owned by government (*e.g.*, wellfields, schools, golf courses, open space, roads) and private non-profit groups like churches and colleges.

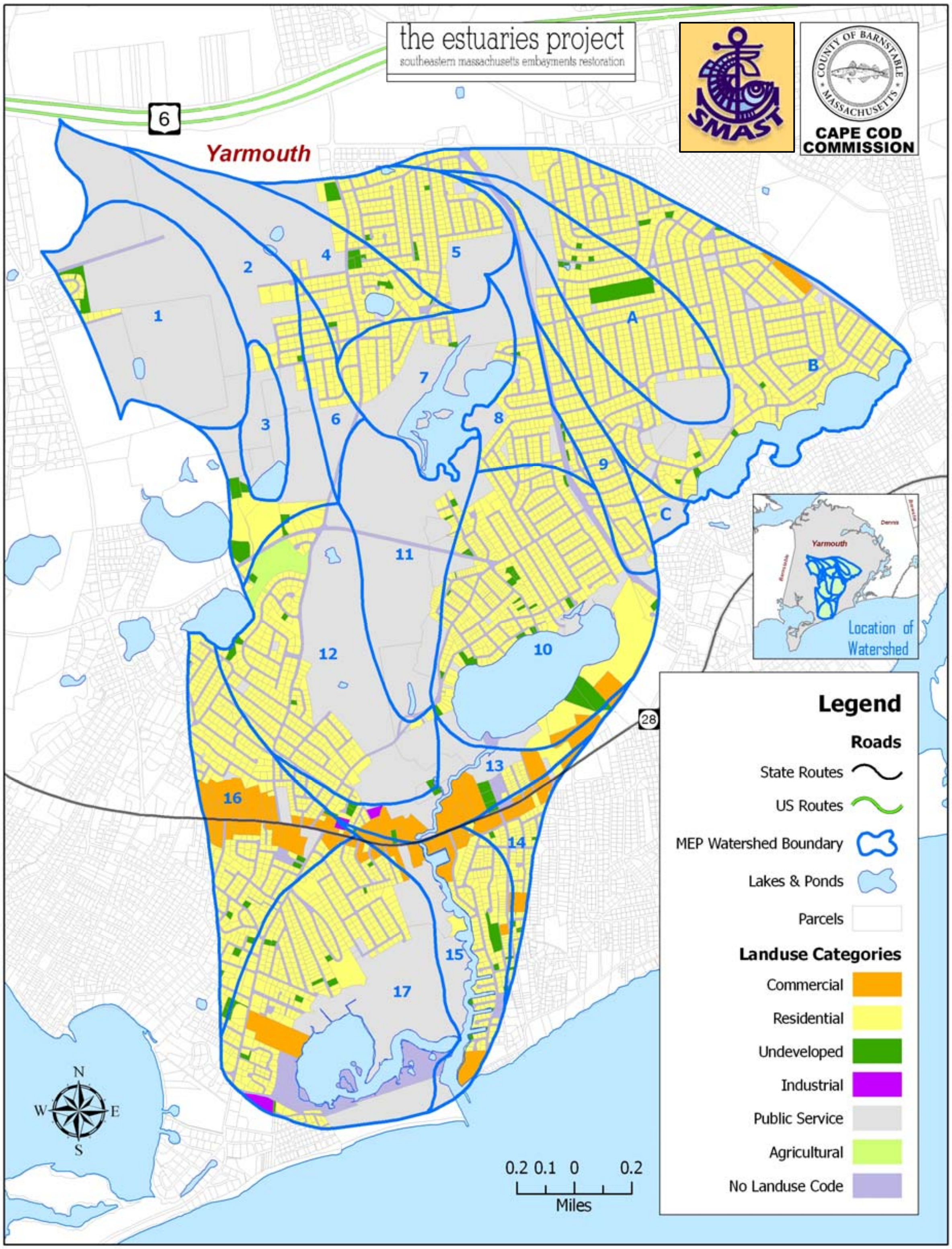


Figure IV-1. Land-use in the Parkers River system watershed. Watershed is completely within Town of Yarmouth. Land use classifications are based on 2006 Town of Yarmouth assessors' records provided by the town for the MEP analysis.

Public service land uses are the dominant land use type in the overall Parkers River watershed and occupy 49% of the watershed area (Figure IV-2). Examples of these land uses are lands owned by town and state government, housing authorities, and churches. Residential land uses occupy the second largest area with 40% of the watershed area. It is notable that land classified by the town assessor as undeveloped is only 2% of the overall watershed area. The Plashes Brook and Grays Brook subwatersheds are where most of the public service lands are; parcel examples in these watersheds include the Town of Yarmouth Septage Disposal site and the Town of Yarmouth lands protecting the public water supply recharge areas.

In all the subwatershed groupings shown in Figure IV-2, residential parcels are the dominant parcel type, ranging between 85% and 96% of all parcels in these subwatersheds and 92% of all parcels in the whole Parkers River watershed. Single-family residences (MassDOR land use code 101) are 66% to 100% of residential parcels in the individual subwatersheds and 94% of the residential parcels throughout the whole Parkers River watershed.

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

MEP staff also received state Groundwater Discharge Permit (GWDP) nitrogen effluent data from the MassDEP (personal communication, Brian Dudley, 4/09) and alternative, denitrifying septic system total nitrogen effluent data from the Barnstable County Department of Health and the Environment (personal communication, Sue Rask and Brian Baumgaertel, 3/09). The only GWDP in the Parkers River watershed is the effluent discharge from the Town of Yarmouth Septage Treatment Facility. Effluent flow from the septage treatment facility is discharged at one of two locations: the Bayberry Golf Course as irrigation water or at the disposal site south of Buck Island Road. The BCDHE has 48 innovative/alternative septic systems in their performance database for the Town of Yarmouth. Four of these IA systems are in the Parkers River watershed. The reporting data from these two agencies was used to develop wastewater nitrogen loads for each of these sites.

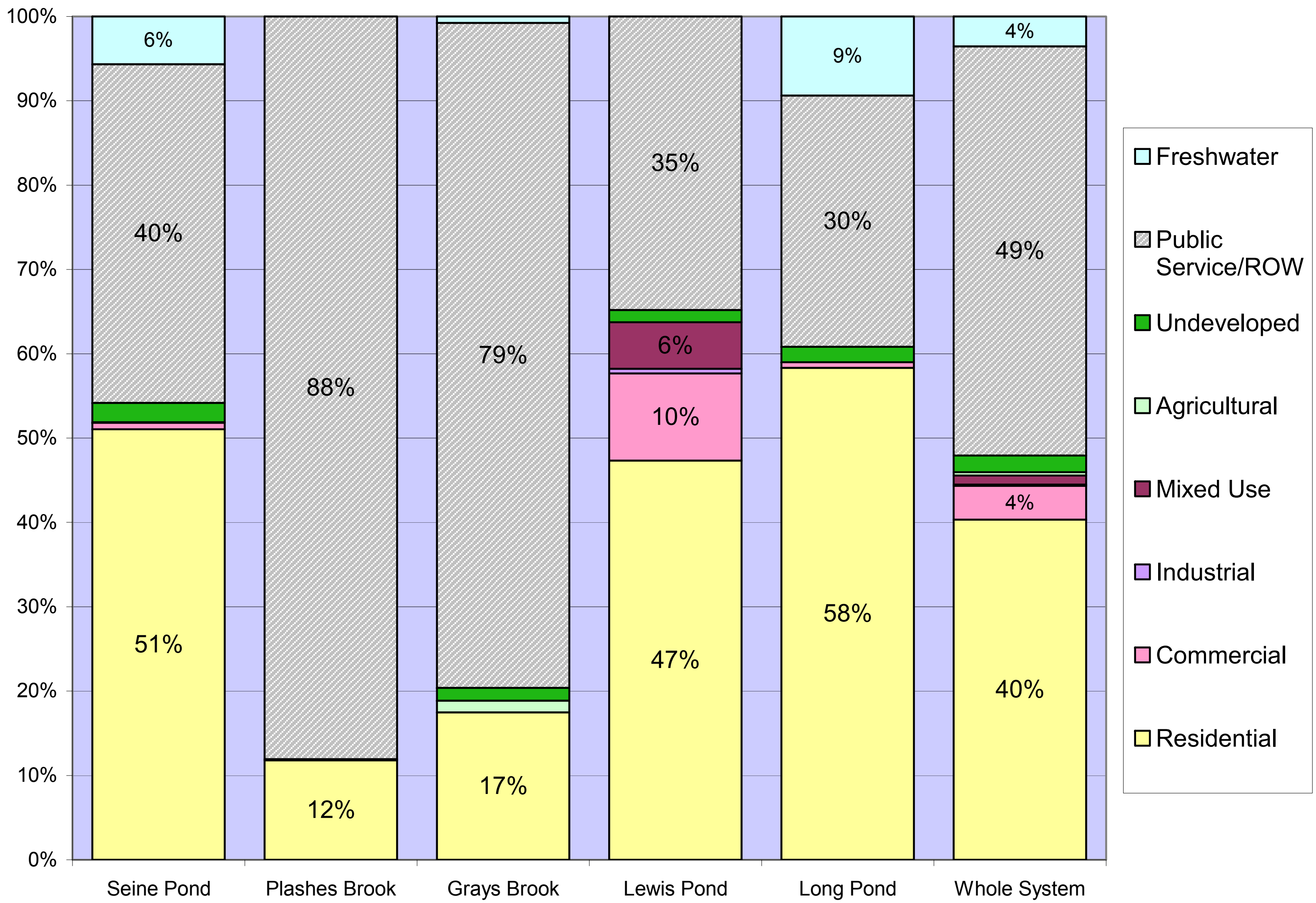


Figure IV-2. Distribution of land-uses by area within the whole Parkers River watershed and four component subwatersheds. Only percentages greater than or equal to 4% are shown. Land use categories are based on town assessor classifications.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

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

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

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

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

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

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

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

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

In order to provide an independent validation of the average residential water use within the Parkers River watersheds, MEP staff reviewed US Census population values for the Town of Yarmouth. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so for the purposes of Title 5 each person generates 55 gpd of wastewater. Based on data collected during the 2000 US Census, average occupancy within Yarmouth is 2.15 people per housing unit with 73% year-round occupancy of available housing units. Average water use for single-family residences with municipal water accounts in the Yarmouth MEP study area is 179 gpd. If this flow is multiplied by 0.9 to account for consumptive use, the study area average is 161 gpd.

If the state Title 5 estimate of 55 gpd of wastewater per capita is multiplied by average Yarmouth occupancy from the 2000 US Census, the average water use per residence would be estimated at 118 gpd. Estimates of summer populations on Cape Cod derived from a number of approaches (*e.g.*, traffic counts, garbage generation, WWTF flows) suggest average population increases from two to three times year-round residential populations measured by the US Census. If it is assumed that seasonal properties are occupied at twice the year-round occupancy for three months, the estimated average town-wide water use would be 148 gpd, while if the seasonal properties are occupied at three times the year-round occupancy for three months, the estimated average water use would be 178 gpd. Given that the average wastewater generation based on water use is within this range, this analysis suggests that the average water use is reasonably reflective of average wastewater estimates.

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

Water use information exists for 88% of the 4,416 developed parcels in the Parkers River watershed. Parcels without water use accounts are assumed to utilize private wells for drinking water. These are properties that were classified with land use codes that should be developed (*e.g.*, 101 or 325), have been confirmed as having buildings on them through a review of aerial photographs, and do not have a listed account in the water use databases. Of the 522 developed parcels without water use accounts, 463 (89%) are classified as single-family residences (land use code 101). These parcels are assumed to utilize private wells and were assigned the Yarmouth study area average water use of 179 gpd in the watershed nitrogen loading modules.

Wastewater Treatment Facilities

As mentioned previously, there is only one wastewater treatment facility requiring a state GWDP located in the Parkers River watershed: the Town of Yarmouth Septage Treatment Facility. The treatment facility utilizes two discharge locations for treated effluent: spray irrigation at the holes 2-8 at the Links portion of the Bayberry Hills Golf Course and fields at a town disposal site south of Buck Island Road. Effluent flow to both locations and total nitrogen concentration data for the effluent was provided to MEP staff by MassDEP staff (personal communication, Brian Dudley, 4/09). Flow and concentration data was provided for the years 2004 through 2007.

While the Buck Island Road discharge site is located within the Parkers River watershed, the portions of the golf course where the spray effluent discharge is utilized are not. MEP staff reviewed the monthly flows directed to each discharge location (Figure IV-3). Total annual discharge from the septage treatment facility ranged between 14.1 and 19.6 million gallons between 2004 and 2007. Annual effluent discharge flows to the Buck Island site between 2004 and 2007 ranged between 3.0 and 9.9 million gallons. Utilizing the reported total nitrogen concentrations, MEP staff determined that annual loads at the Buck Island site during this same period varied between 38 kg and 339 kg with a four-year average of 145 kg/y. This average load is used in the annual watershed nitrogen loading estimates for the Parkers River watershed.

Alternative Septic Systems

As mentioned previously, there are four alternative, denitrifying septic systems in the Parkers River study area that have total nitrogen effluent data in the Barnstable County Department of Health and the Environment database (personal communication, Sue Rask and Brian Baumgaertel, 3/09). These four systems have 2 to 7 measurements and have average total nitrogen effluent concentrations between 9.8 and 41.3 ppm. Project staff used these site-specific, average measured effluent total nitrogen concentrations and the average measured water use from the town Water Division records to calculate average annual loads from each of these sites. These loads were incorporated into the watershed nitrogen loading module for the Parkers River.

Nitrogen Loading Input Factors: Fertilized Areas

The second largest source of watershed nitrogen loading to estuaries is usually fertilized areas: lawns, golf courses, and cranberry bogs. Residential lawns are usually the predominant source within this category. In order to add this source to the nitrogen loading model for the Parkers River system, MEP staff reviewed available regional information about residential lawn fertilizing practices and incorporated site-specific information for the portion of the Bayberry Hills Golf Course and the cranberry bogs in the watershed. Cranberry bog nitrogen loading was determined based on previous studies conducted in southeastern Massachusetts, while MEP staff contacted the golf course superintendent to determine fertilizer application rates.

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

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

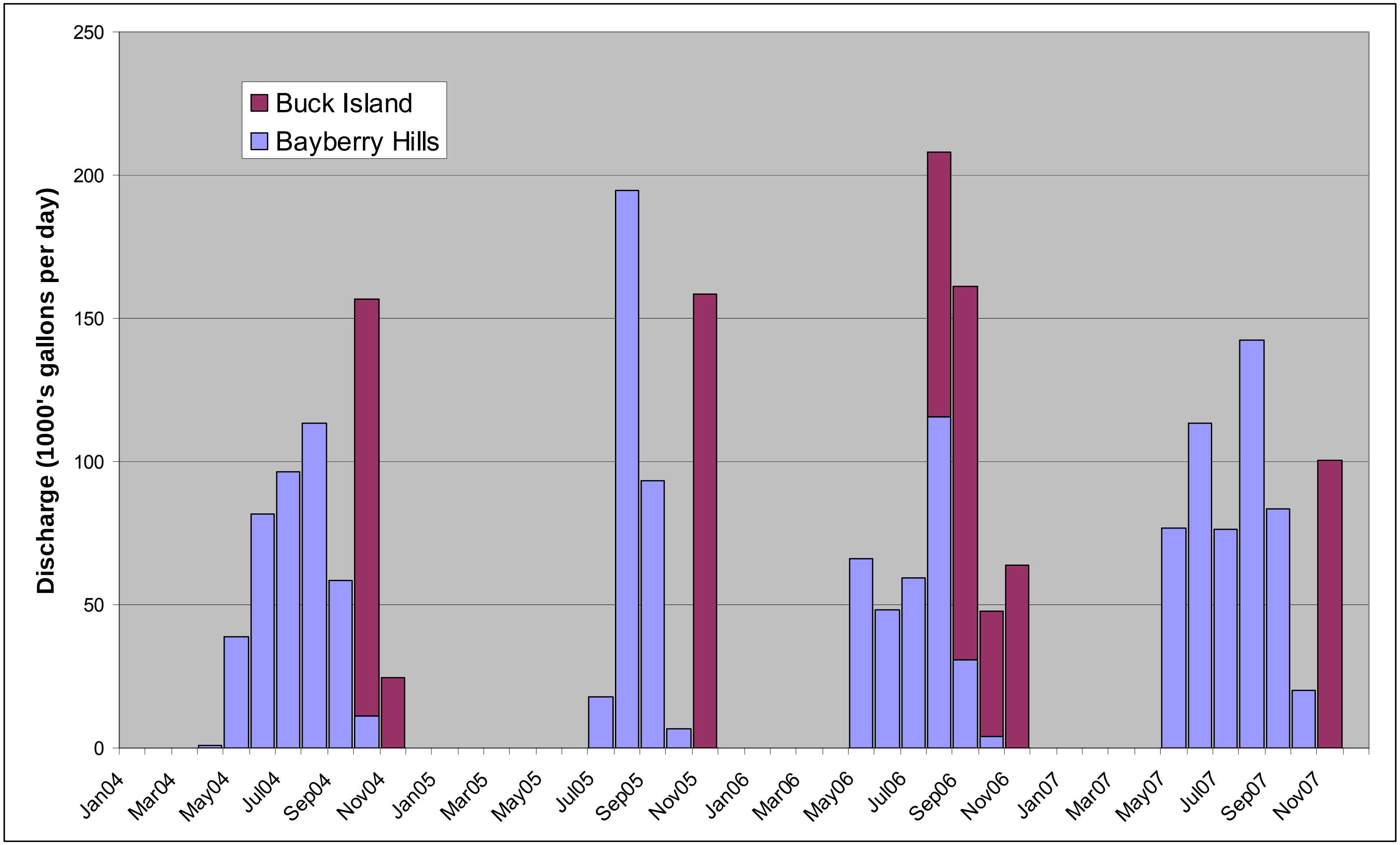


Figure IV-3. Total effluent discharge from the Town of Yarmouth Septage Treatment Facility (2004-2007). Effluent is discharged at two locations: a portion of the Bayberry Hills Golf Course that is built on the capped town landfill and a discharge area south of Buck Island Road. Monthly effluent discharge at both locations is shown. Only the Buck Island Road site is located within the Parkers River watershed. Annual nitrogen loads were determined from reported effluent flow and total nitrogen concentrations and averaged to produce the annual load used in the MEP watershed nitrogen loading calculations. Data provided by MassDEP (personal communication, B. Dudley, 4/09).

found to have the higher rate of fertilizer application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

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

As has been done in all MEP reviews, MEP staff reviewed the layout of the Bayberry Hills Golf Course from aerial photographs, classified the various turf types, and, using GIS, assigned these areas to the appropriate subwatersheds. The fertilized area within the Parkers River watershed is 48% of the total fertilized area of Bayberry Hills Golf Course. The golf course-specific nitrogen application rates were then applied to the respective turf areas, a standard MEP 20% leaching rate was applied, and annual load by subwatershed was calculated for the golf course.

Cranberry bog fertilizer application rate and percent nitrogen attenuation in the bogs is based on the only annual study of nutrient cycling and loss from cranberry agriculture that has been conducted in southeastern Massachusetts (Howes and Teal, 1995). Based on this study, only the bog loses measurable nitrogen, the forested upland releases only very low amounts. For the watershed nitrogen loading analysis, the areas of active bog surface are digitized based on review of aerial photographs. Cranberry bogs are only located in the Grays Brook subwatershed portion of the overall Parkers River watershed.

Nitrogen Loading Input Factors: Other

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

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

Table IV-2. Primary Nitrogen Loading Factors used in the Parkers River MEP analyses. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Yarmouth-specific data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

Nitrogen Concentrations:	mg/l	Recharge Rates:	in/yr
Road Run-off	1.5	Impervious Surfaces	40
Roof Run-off	0.75	Natural and Lawn Areas	27.25
Direct Precipitation on Embayments and Ponds	1.09	Water Use/Wastewater:	
Natural Area Recharge	0.072	Existing developed residential parcels wo/water accounts and buildout residential parcels:	179 gpd
Wastewater Coefficient	23.63		
Town of Yarmouth Septage Treatment Facility: Buck Island Road Discharge		Existing developed parcels w/water accounts:	Measured annual water use
Average effluent Flow (million gallons per year)	5.7		
Average Total Nitrogen load (kg/y)	145	Average Single Family Residence Building Size from watershed data (sq ft)	1,151
Fertilizers:		Commercial and Industrial Buildings without/WU and buildout additions (based on existing water use for similarly classified properties)	
		Commercial	
Average Residential Lawn Size (sq ft)*	5,000	Wastewater flow (gpd/1,000 ft2 of building):	1,000
Residential Watershed Nitrogen Rate (lbs/lawn)	1.08	Building coverage:	13.2%
Cranberry Bogs nitrogen application (lbs/ac)	31	Industrial	
Cranberry Bogs nitrogen attenuation	34%	Wastewater flow (gpd/1,000 ft2 of building):	1,000
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined from site-specific information		Building coverage:	14.5%
Note: none of the three lots classified by the town assessor as industrial uses have water use data; all existing and future buildout industrial lots are assigned the average			

IV.1.3 Calculating Nitrogen Loads

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

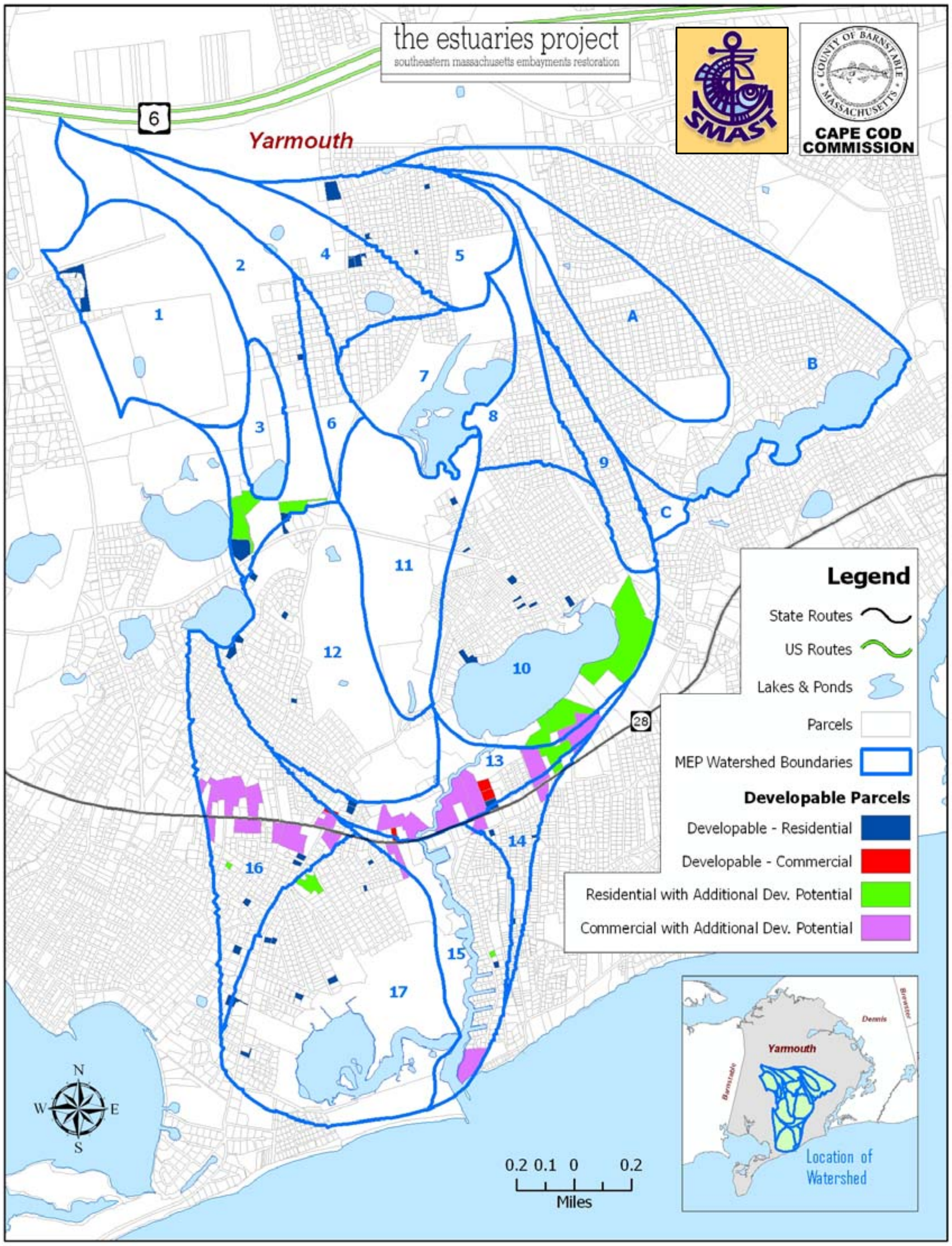


Figure IV-4. Parcels, Parcelized Watersheds, and Developable Parcels in the Parkers River watershed. Parcelized watersheds are based on the assignment of subwatershed boundary parcels within the MEP Watershed Nitrogen Loading Module; large parcels are split, but smaller parcels are generally assigned to subwatersheds where the majority of their area is located. Developable parcels are based on town assessor classifications and minimum lot sizes specified in town zoning; these parcels are assigned estimated nitrogen loads in MEP buildout calculations. All buildout parcels were reviewed and confirmed by Town of Yarmouth staff.

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 Parkers 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.

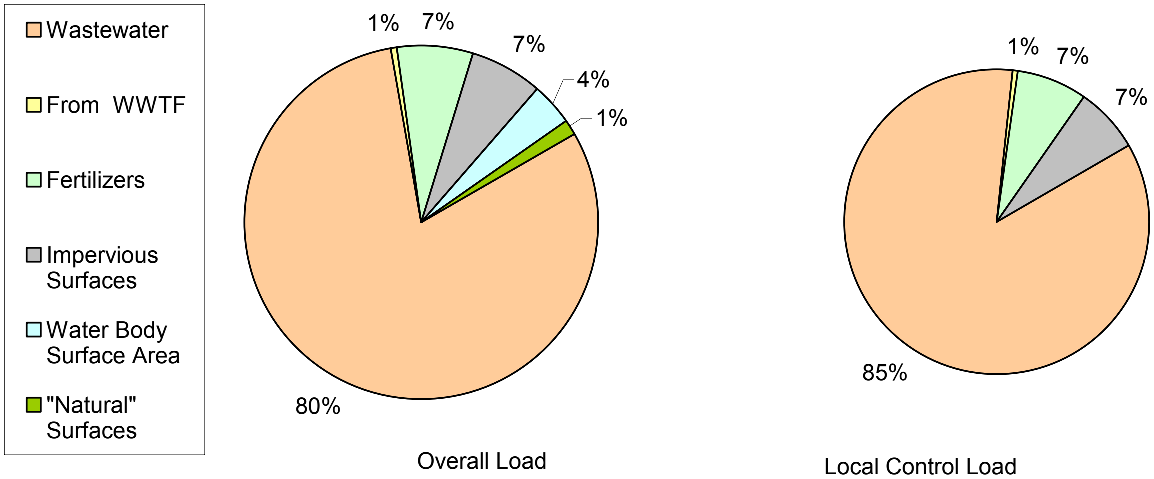
Following the assignment of all parcels, subwatershed modules were generated for each of the 20 sub-watersheds in the Parkers River study area. These subwatershed modules include each individual parcel and summarize, among other things: water use, parcel area, frequency, private wells, and road area. The individual sub-watershed modules are then integrated to create a Parkers River Watershed Nitrogen Loading module with summaries for each of the individual subwatersheds/subembayments. The subembayments represent the functional embayment 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 Parkers River study area, the major types of nitrogen loads are: wastewater (*e.g*., septic systems), wastewater treatment facilities, fertilizers (including contributions from cranberry bogs and golf courses), impervious surfaces, direct atmospheric deposition to water surfaces, and recharge within natural areas (Table IV-3). The output of the watershed nitrogen-loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments, by each source category (Figure IV-5). In general, the annual watershed nitrogen input to the watershed of an estuary is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model.

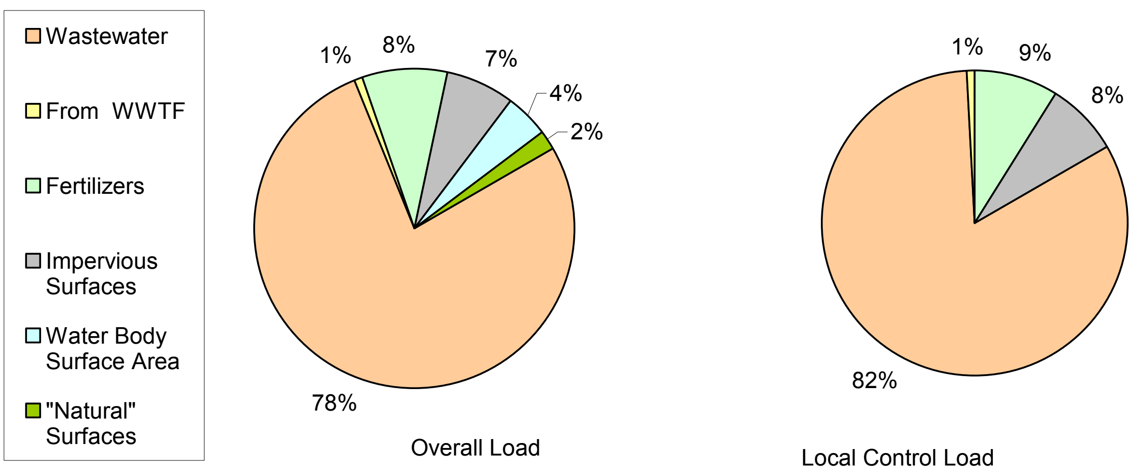
One of these attenuation adjustments occurs in the freshwater ponds. Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond was used to apportion the pond-attenuated nitrogen load to respective downgradient watersheds. The apportionment was based on the percentage of discharging shoreline bordering each downgradient sub-watershed. In the Parkers River study area, this occurs in both ponds completely within the watershed (*e.g.*, Plashes Pond) and the ponds located along the outer boundary of the River watershed (*e.g.*, Big Sandy Pond). At Plashes Pond, for example, the pond has a downgradient shoreline of 4,342 feet; 50% of that shoreline discharges into the Seine Pond GT10 subwatershed (watershed 8 in Figure IV-1) and 50% discharges to the Plashes Brook LT10 subwatershed (watershed 7 in Figure IV-1). This breakdown of the discharge from Plashes Pond means that 50% of the attenuated nitrogen load that leaves the pond reaches Seine Pond and the other half reaches Plashes Brook. Similar pond-specific calculations were completed wherever pond flows and nitrogen loads were divided among a number of downgradient receiving subwatersheds.

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

		Parkers River N Loads by Input (kg/y):							% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	From WWTF	Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Parkers River System		22821	145	1973	1881	1163	419	13272		28402		25221	41674		39295
Parkers River GT10	14	1106	0	44	65	0	5	713		1220		1220	1932		1932
Parkers River LT10	15	3183	0	86	114	0	16	1204		3398		3398	4603		4603
Parkers River Estuary Surface						97				97		97	97		97
Lewis Pond		5406	0	404	484	259	70	4674		6622		6576	11297		11242
Lewis Pond GT10	16	2830	0	180	235	0	21	4579		3266		3266	7844		7844
Lewis Pond LT10	17	2529	0	222	247	0	47	79		3046		3046	3125		3125
Big Sandy Pond	BSP	47	0	1	2	34	2	16	23%	86	50%	39	102	50%	47
Lewis Pond Estuary Surface						225				225		225	225		225
Upper River		13125	145	1439	1219	807	329	6681		17064		13930	23745		21421
Upper River	13	1047	0	36	50	0	15	5733		1148		1148	6881		6881
Gray Brook	GB	4600	145	686	467	216	241	488		6355		4841	6843		5293
Seine Pond	SP	7479	0	717	702	173	73	460		9143		7523	9603		8829
Seine Pond Estuary Surface						400				400		400	400		400
Upper River Estuary Surface						18				18		18	18		18



a. Parker River System



b. Upper River Subwatershed

Figure IV-5 . Land use-specific unattenuated nitrogen loads (by percent) to the a) whole Parkers River watershed and b) Upper River subwatershed. “Overall Load” is the total nitrogen input within the watershed, while the “Local Control Load” represents only those nitrogen sources that could potentially be under local regulatory control. Upper River subwatershed includes Seine Pond and Gray Brook.

Freshwater Pond Nitrogen Loads

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

Nitrogen attenuation in freshwater ponds has generally been found to be at least 50% in MEP analyses, so a conservative attenuation rate of 50% is generally assigned to all nitrogen from freshwater pond watersheds in the watershed model unless more detailed pond monitoring or studies are available. Detailed studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported a 50% attenuation factor as a reasonable, somewhat conservative rate. However, in some cases, if sufficient monitoring information is available, a pond-specific attenuation rate is incorporated into the watershed nitrogen loading modeling (e.g., Three Bays MEP Report, 2005). In order to review whether a nitrogen attenuation rate higher than 50% should be used for a specific pond, the MEP Technical Team reviews the available data on each pond, including available nitrogen concentrations, impacts of sediment regeneration, temperature profiles, and bathymetric information.

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

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

Within the Parkers River watershed, there are three freshwater ponds with delineated watersheds: Plashes, Bassetts Lot, and Long. Big Sandy Pond is also located on the western system watershed boundary. Among these ponds, only Long has available pond-wide bathymetric data (Eichner, et al., 2003). As such, a reasonable pond-specific nitrogen attenuation rate can only be developed for Long via the regional Cape Cod Pond and Lake Stewards (PALS) Snapshots.

The PALS Snapshots are regional volunteer pond sampling supported for the last eight years by SMAST and the Cape Cod Commission, with free laboratory services provided by the Coastal Systems Program Laboratory at SMAST. Sampling runs in Yarmouth ponds have generally followed PALS protocols (Eichner *et al*., 2003), which means that sampling has included field collection of temperature and dissolved oxygen profiles and sampling has generally occurred at standardized depths that provide some evaluation of potential sediment nutrient regeneration. PALS water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH.

In MEP analyses, available nitrogen concentrations from individual ponds are reviewed to establish whether sediment regeneration is a significant factor in a pond and, if not, the entire volume of the pond is used to determine a turnover time. Turnover time is how long it takes the recharge from the upgradient watershed to completely exchange the water in the pond or, in the case of a thermally stratified pond, exchange just the volume of the epilimnion. The total mass of nitrogen in the pond or epilimnion is adjusted using the pond turnover time to determine the annual nitrogen load returned to the aquifer through the downgradient shoreline. This mass is then compared to the nitrogen load coming from the pond's watershed to determine the nitrogen attenuation factor for the pond.

As mentioned above, only Long among the ponds within the Parkers River watershed has available bathymetric data and water quality data are generally limited. Long has had samples collected by staff from the Yarmouth Division of Natural Resources all eight PALS Snapshots between 2001 and 2008. Plashes had samples collected during the 2003 PALS Snapshot, while Big Sandy has samples collected during the 2003 and 2004 Snapshots. Bassetts Lot has not been sampled during the eight years of PALS Snapshots. Given the lack of bathymetry and limited water quality data, data are insufficient to assign pond-specific nitrogen attenuation factors to freshwater ponds in the Parkers River watershed except for Long Pond. The standard MEP freshwater pond 50% nitrogen attenuation rate was incorporated into the watershed nitrogen loading module of the linked watershed-estuary model for Plashes, Big Sandy, and Bassetts Lot (Table IV-4).

In order to estimate the natural nitrogen removal that occurs in Long Pond and the nitrogen load coming from Long Pond into Seine Pond, MEP staff reviewed the PALS Snapshot water quality results for Long Pond. Based on these results, the average surface TN concentration in Long Pond is 0.5 ppm (n=11). This concentration is based on surface TN concentrations measured in PALS Snapshots between 2001 and 2008. Using this information, MEP staff determined that 89% of the watershed nitrogen load into Long Pond is attenuated or removed by natural processes.

Table IV-4. Nitrogen attenuation by Freshwater Ponds in the Parkers River watershed based upon 2001 through 2008 town volunteer sampling by the Town of Yarmouth Natural Resources Division and Cape Cod Pond and Lakes Stewardship (PALS) program sampling. These data were collected to provide a site-specific check on nitrogen attenuation by these systems. All ponds in the watershed, except for Long Pond, are assigned the standard MEP nitrogen attenuation value of 50%.

Pond	PALS ID	Area acres	Maximum Depth measured by PALS m	Upper (U) /Whole (W) volume turnover time yrs	# of TN samples for N Attenuation calculation	N Load Attenuation %
Plashes	YA-653	44.8	2.0	No Bathymetric Info	1	Not calculated due to lack of bathymetry
Bassetts Lot	YA-686	7.2	-		0	
Big Sandy	YA-711	20.0	4.0		2	
Long	YA-657	60.5	9.2	0.32	11	89%

Data sources: all areas from CCC GIS; all maximum depths are maximum recorded depths from PALS Snapshots; number of Total Nitrogen samples available for attenuation calculations are surface concentrations from annual PALS Snapshot laboratory results provided by SMAST Coastal Systems Program Lab.

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) residential parcels classified by the town assessor as developable are 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 are identified, 3) developed residential parcels with areas greater than twice zoning's minimum lot size are identified, divided by the minimum lot size and the resulting number of new residential units is rounded down, and 4) results are discussed with town staff and/or board members and the analysis results are modified based on local knowledge. For example in step #1, an 81,000 square foot lot classified by the town assessor as a developable residential lot (land use code 130) is divided by the 40,000 square foot minimum lot size specified in town zoning and the result is rounded down to two additional residential lots at buildout.

Other provisions of the buildout assessment include undevelopable lots, commercial and industrial properties, and lots less than the minimum areas specified by zoning. Properties classified by the Town of Yarmouth assessors as "undevelopable" (*e.g.*, MassDOR codes 132, 392, and 442) are not assigned any development at buildout. Commercial and industrial properties classified as developable are not subdivided; the area of each parcel and the factors in Table IV-3 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. 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. It should be noted that this buildout approach does not include any modifications/refinements for lot line setbacks, wetlands, or road construction that might reduce the number of lots. The MEP buildout approach also does not include potential impacts associated with the higher densities usually associated with 40B affordable housing projects. The fourth step including the discussions with town planners, and, occasionally, town planning boards and wastewater consultants, often leads to additional insights on developments that are planned, especially developments planned on government or public service parcels, and updates to assessor classifications, including lands purchased by the town as open space.

Following the completion of the initial buildout assessment for the Parkers River watersheds, MEP staff reviewed the results with town officials. MEP staff reviewed the initial results with Terry Sylvia, Karen Greene, George Allaire, and the town's Wastewater Planning consultants in March 2009. Suggested changes from Terry Sylvia were incorporated into the final buildout for Parkers River.

All the parcels with additional buildout potential within the Parkers River watershed are shown in Figure IV-4. Each additional residential, commercial, or industrial property added at buildout is assigned nitrogen loads for wastewater and impervious surfaces. Residential additions also include lawn fertilizer nitrogen additions. All wastewater loads are assumed to come from on-site septic systems. Cumulative unattenuated buildout loads are indicated in a separate column in Table IV-3. Buildout additions within the Parkers River watersheds will increase the unattenuated loading rate by 47%; if the commercial additions along Route 28 are not included in the buildout increase, the overall increase in the unattenuated loading rate is 5%.

The buildout analysis also contains a change from existing conditions in the conceptualization of the Forest Road Brook flows and nitrogen loads from the Brook and Long Pond. This change is based on modifications to the Brook connections to Long Pond accomplished by the Town Department of Natural Resources. Flows and nitrogen concentrations for the existing MEP conditions were measured in the Forest Road Brook between September 2003 and September 2004 (see Section IV.2). In fall 2008, the Town DNR installed a control structure on Forest Road Brook near Clear Brook Road. Limited stream measurements (March to May 2008) showed significantly higher flows than measured during the MEP hydro-year (approximately three (3) times higher than MEP existing conditions flows)(personal communication, Steve Raneo, Yarmouth DNR, 5/09).

This change in the flows has been incorporated into the MEP buildout analysis of Parkers River. Existing conditions showed that an estimated 353 m3/d of flow came from Long Pond during the MEP water quality measurement period (see Section IV.2). Based on the changes in the channel between Long Pond and Seine Pond, the estimated flow from Long Pond has increased to 1995 m3/d (a nearly 6-fold increase). More data collection in the new channel configuration would help to clarify, but these results indicate that slightly more than 40% of the Long Pond watershed flow and attenuated nitrogen loads now discharge into Seine Pond.

Scenarios 3 and 4

At the request of Town of Yarmouth staff, MEP staff completed two additional buildout scenarios; labeled Scenarios 3 and 4, based on the assumption that the existing conditions run of linked watershed-estuary model is Scenario 1 and the run using buildout conditions is Scenario 2. These additional scenarios include the potential sewer district (or "sewershed") to

Parkers River that the Town of Yarmouth developed through the current town-wide comprehensive wastewater planning process (Figure IV-6). It should be noted that the Parkers River sewershed shown in Figure IV-6 extends slightly beyond the Parkers River watershed boundaries. It should also be noted that both alternative buildout scenarios include the expansion of flow from Long Pond to Seine Pond that is included in the standard MEP buildout scenario.

Scenario 3 includes collecting all the buildout wastewater flows within the Town-delineated sewershed, treating the wastewater to 5 ppm total nitrogen, and discharging the effluent at the current town septage effluent disposal site south of Buck Island Road. Total buildout flow collected by the proposed sewers within the Parkers River watershed is 1.19 million gallons per day (MGD), while collected flow from outside the watershed is 0.12 MGD. Total collected sewershed flow for Scenario 3 is 1.31 MGD.

Scenario 4 includes collecting all the buildout wastewater flows within the sewershed delineated for Scenario 3 plus the wastewater flows within the previous sewershed that the town delineated within the Lewis Bay watershed (Figure IV-7). All collected flows in this scenario are treated to 3 ppm total nitrogen and treated effluent is discharged at the current town septage effluent disposal site south of Buck Island Road. Total buildout flows collected by the proposed sewers within the Parkers River watershed in this scenario are 1.19 million gallons per day (MGD), while collected flow from outside the watershed is 0.74 MGD. Total collected sewershed flow for Scenario 4 is 1.93 MGD.

The primary increase in watershed nitrogen load for both Scenarios 3 and 4 is in the Gray Brook subwatershed (Table IV-5). Gray Brook attenuated annual nitrogen load increases from 5,293 kg under current buildout assumptions to 8,445 kg in Scenario 3 and 7,248 kg in Scenario 4. Overall, Parkers River watershed annual attenuated buildout nitrogen loads decrease from 39,221 kg under current buildout conditions to 17,623 kg in Scenario 3 and 16,426 kg in Scenario 4.

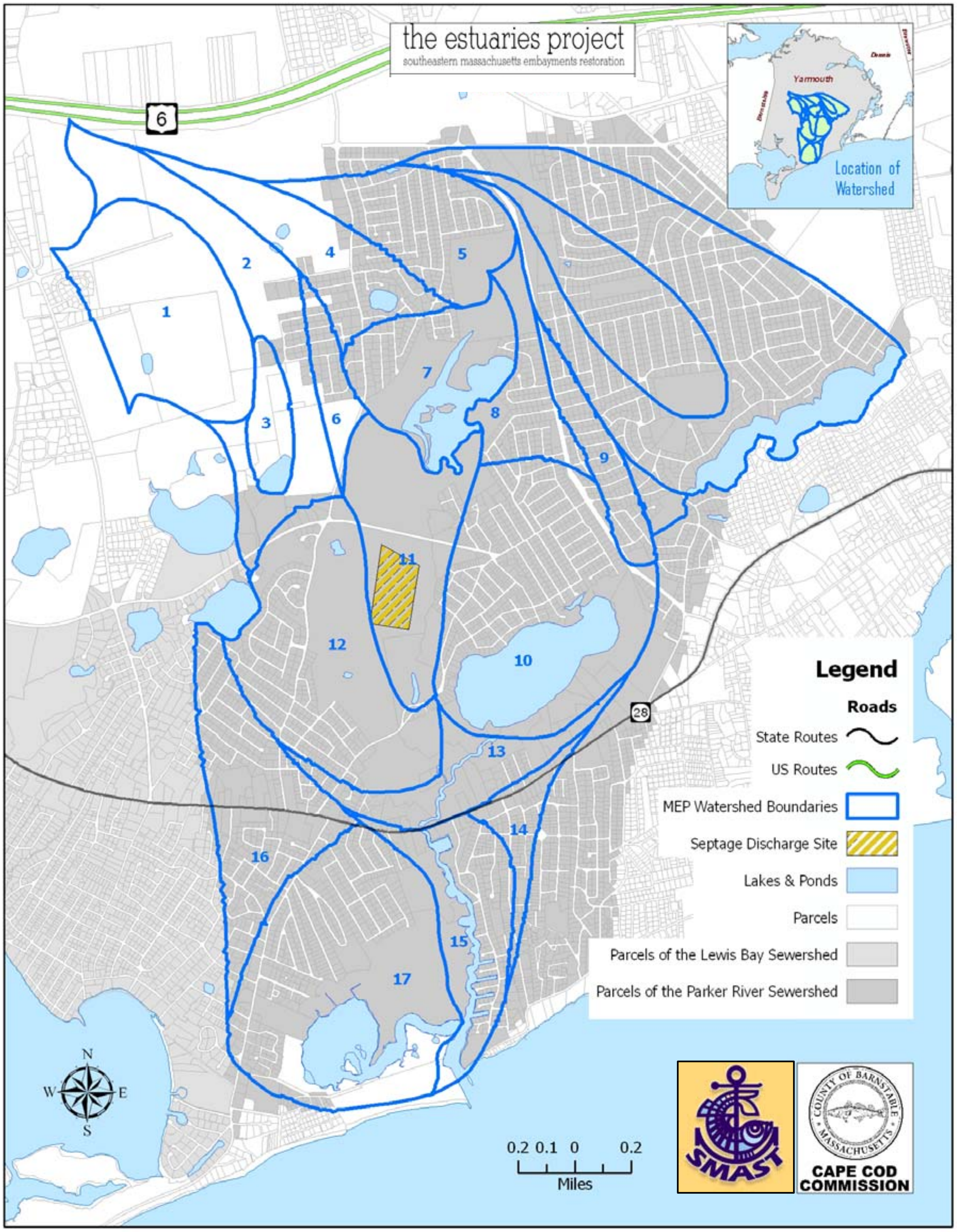


Figure IV-6. Buildout Scenario 3: Parkers River Potential Sewershed and Effluent Discharge Location. At the request of Town of Yarmouth officials, MEP staff prepared an analysis of the water quality impacts on Parkers River of collecting all wastewater flows from the indicated sewer collection area, treating the flows to an effluent concentration of 5 ppm total nitrogen, and discharging the effluent at the current septage effluent discharge site south of Buck Island Road (indicated). Sewershed delineated by the Town.

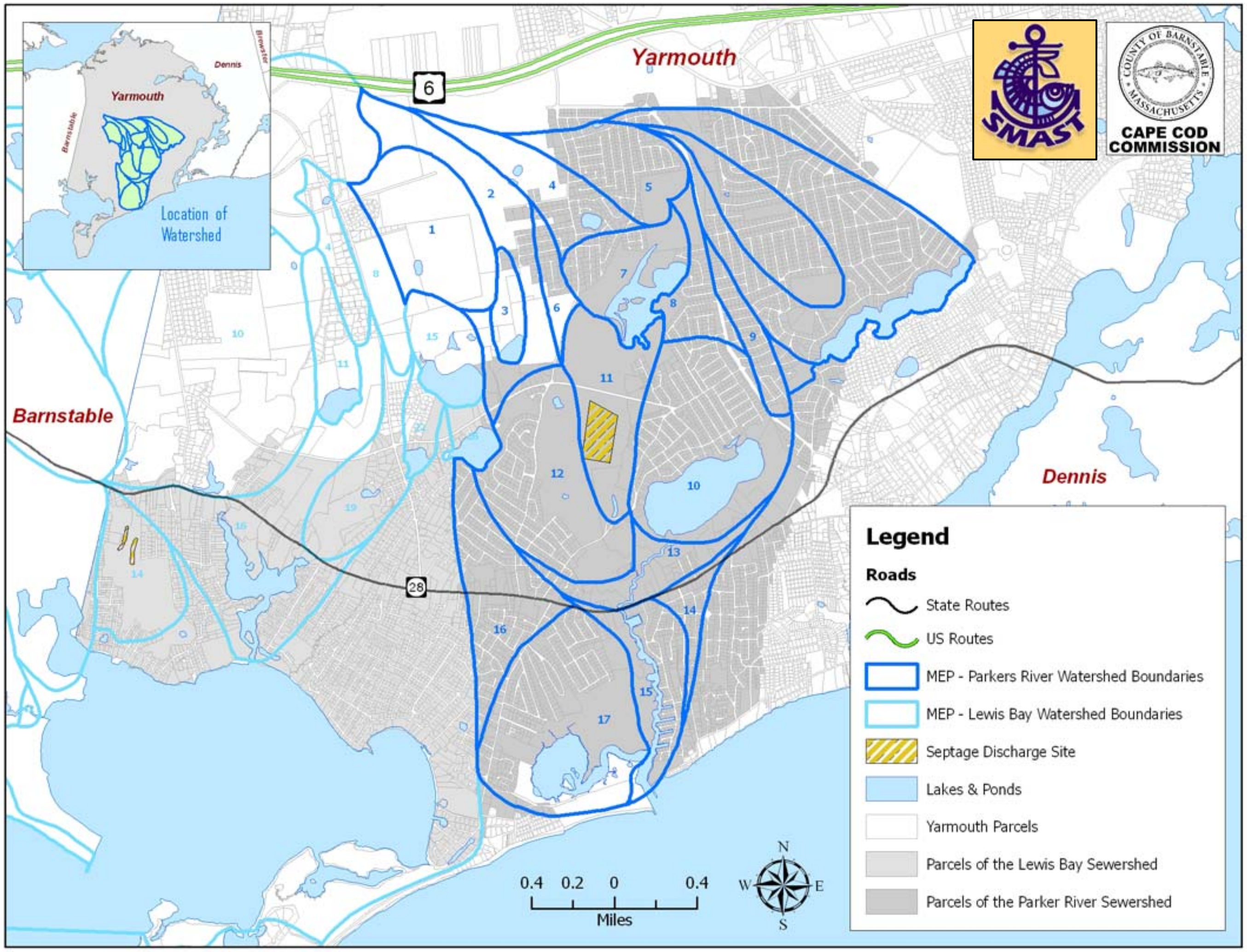


Figure IV-7. Buildout Scenario 4: Parkers River and Lewis Bay Sewersheds and Parkers River Effluent Discharge Location. At the request of Town officials, MEP staff prepared an analysis of the water quality impacts on Parkers River of collecting all wastewater flows from the indicated proposed sewer collection areas, treating the wastewater flows to an effluent concentration of 3 ppm total nitrogen, and discharging the effluent at the current septage effluent discharge site south of Buck Island Road (indicated). Both sewersheds delineated by the Town; Lewis Bay sewershed discussed in Lewis Bay MEP report (Howes, *et al.*, 2007).

Table IV-5. Parkers River Watershed Nitrogen Loads for Buildout Scenarios 3 and 4. Scenario 3 collects wastewater from a sewershed shown in Figure IV-6, treats it to 5 ppm total nitrogen, and discharges the effluent at the current Town septage discharge site south of Buck Island Road. Scenario 4 collects wastewater from a sewershed shown in Figure IV-6 plus the Lewis Bay sewershed shown in Figure IV-7, treats it to 3 ppm total nitrogen, and discharges the effluent at the current Town septage discharge site south of Buck Island Road. Attenuated nitrogen loads shown below are based on measured and assigned attenuation factors assigned to upgradient streams and freshwater ponds. Stream attenuation factors are based on measured loads (see Section IV.2), while all ponds except Long are assigned a standard MEP nitrogen attenuation of 50%. Long Pond has a nitrogen attenuation factor of 89%. All nitrogen loads are kg N yr^{-1}.

		% of Pond Outflow	Buildout N Loads			Buildout N Loads Scenario 3			Buildout N Loads Scenario 4		
Name	Watershed ID#		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Parkers River System			41674		39221	21790		17623	20742		16426
Parkers River GT10	14		1932		1932	121		121	121		121
Parkers River LT10	15		4603		4603	228		228	228		228
Parkers River Estuary Surface			97		97	97		97	97		97
Lewis Pond			11297		11242	6684		6630	6684		6630
Lewis Pond GT10	16		7844		7844	5417		5417	5417		5417
Lewis Pond LT10	17		3125		3125	940		940	940		940
Big Sandy Pond	BSP	23%	102	50%	47	102	50%	47	102	50%	47
Lewis Pond Estuary Surface			225		225	225		225	225		225
Upper River			23745		21347	14660		10547	13611		9350
Upper River	13		6881		6881	156		156	156		156
Gray Brook	GB		6843		5293	11609		8445	10561		7248
Seine Pond	SP		9603		8755	2476		1528	2476		1528
Seine Pond Estuary Surface			400		400	400		400	400		400
Upper River Estuary Surface			18		18	18		18	18		18

IV.2 ATTENUATION OF NITROGEN IN SURFACE WATER TRANSPORT

IV.2.1 Background and Purpose

Modeling and predicting changes in coastal embayment nitrogen related water quality is based, in part, on determination of the inputs of nitrogen from the surrounding contributing land or watershed. This watershed nitrogen input parameter is the primary term used to relate present and future loads (build-out, sewering analysis, enhanced flushing, pond/wetland restoration for natural attenuation, etc.) to changes in water quality and habitat health. Therefore, nitrogen loading is the primary threshold parameter for protection and restoration of estuarine systems. Rates of nitrogen loading to the sub-watersheds of the Parkers River System being investigated under this nutrient threshold analysis was based upon the delineated watersheds (Section III) and their land-use coverages (Section IV.1).

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

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving estuarine waters. If nitrogen attenuation is significant in one portion of a watershed and insignificant in another the result is that nitrogen management would likely be more effective in achieving water quality improvements if focused on the watershed region having unattenuated nitrogen transport (other factors being equal). In addition to attenuation by freshwater ponds (see Section IV.1.3, above), attenuation in surface water flows is also important. An example of the significance of surface water nitrogen attenuation relating to embayment nitrogen management was seen in the Agawam River, where >50% of nitrogen originating within the upper watershed was attenuated prior to discharge to the Wareham River Estuary (CDM 2000). Similarly, MEP analysis of the Quashnet River indicates that in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor

(Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach in the Parkers River embayment system. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the estuary in addition to the natural attenuation measures by fresh kettle ponds, addressed above (Section IV.1). These additional site-specific studies were conducted in the 2 major surface water flow systems in the Parkers River watershed, 1) Plashes Brook discharging to the main tidal channel of the Parkers River and 2) Forest Road Brook, a small stream discharging to Seine Pond which is the head of the Parkers River system.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the freshwater streams discharging to the estuary provides a direct integrated measure of all of the processes presently attenuating nitrogen in the contributing area upgradient from the various gauging sites. Flow and nitrogen load were measured at the gages in each freshwater stream site for between 17 and 20 months of record depending on the stream gauging location (Figures IV-8 to IV-10). During each study period, velocity profiles were completed on each river 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).

Figure IV-8. Location of Stream gages (red diamonds) in the Parkers River embayment system.

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

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

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

where by:

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

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

Periodic measurement of flows over the entire stream gage deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gages. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given river. 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 two low tide stage values for any given day were averaged and the average stage value for a given day was then entered into the stage – discharge relation in order to compute daily flow. A complete annual record of stream flow (365 days) was generated for the surface water discharges flowing into the Parkers River embayment system from Plashes Brook and the stream discharge to the head of Seine Pond.

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

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Plashes Brook

Plashes Pond, located up gradient of the Plashes Brook gage site (Winslow Gray Road crossing) is a small freshwater pond and unlike many of the freshwater ponds on Cape Cod, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, Plashes Brook, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the subwatershed nitrogen load to the estuary and the level of nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands, small impoundments and streambed associated with Plashes Brook. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to Plashes Brook above the gage site and the measured annual discharge of nitrogen to the tidal portion of the Parkers River system, Figure IV-8.

At the Plashes Brook gage site, a continuously recording vented calibrated water level gage was installed to yield the level of water in the freshwater portion of the Plashes Brook-Parkers River estuarine system that carries the flows and associated nitrogen load to the near shore waters of Nantucket Sound. As a portion of the lower reach of Plashes Brook is tidally influenced, the gage was located as far downgradient along the Plashes Brook reach such that freshwater flow could be measured at low tide. To confirm that freshwater was being measured the stage record was analyzed for any semi-diurnal variations indicative of tidal influence and salinity measurements were conducted on the weekly water quality samples collected from the gage site. Average low tide salinity was determined to be 0.1 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage on Plashes Brook was installed on April 30, 2004 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Stage data collection continued until November 7, 2005 for a total deployment of 19 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 Plashes Brook site based upon these flow measurements and measured water levels at the gage site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allows for the determination of nitrogen mass discharge to the main tidal channel of the Parkers River system, just north of the Route 28 bridge crossing, and reflective of the biological processes occurring in the stream channel and small up-gradient impoundment contributing to nitrogen attenuation (Figure IV-9 and Table IV-6 and IV-7). In addition, a water balance was constructed based upon the U.S. Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gage site.

The annual freshwater flow record for Plashes Brook measured by the MEP was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from Plashes Brook was 0.1% below the long-term average modeled flows. The average daily flow based on the MEP measured flow data for one hydrologic year beginning September and ending in August (low flow to low flow) was 3,107 m^3/day compared to the long term average flows determined by the USGS modeling effort (3,109 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 Plashes Brook

Table IV-6. Comparison of water flow and nitrogen load discharged by Rivers and Streams (freshwater) to the Parkers 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 USGS watershed delineation (Section III).

Stream Discharge Parameter	Plashes Brook Discharge[(a)] Parkers River	Forest Brook Discharge[(a)] Seine Pond	Data Source
Total Days of Record	365[(b)]	365[(c)]	(1)
Flow Characteristics			
Stream Average Discharge (m3/day) **	3,107	823	(1)
Contributing Area Average Discharge (m3/day)	3,109	821	(2)
Discharge Stream 2004-05 vs. Long-term Discharge	-0.06%	0.24%	
Nitrogen Characteristics			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.412	0.459	(1)
Stream Average Total N Concentration (mg N/L)	0.966	1.258	(1)
Nitrate + Nitrite as Percent of Total N (%)	43%	36%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	3.00	1.04	(1)
TN Average Contributing UN-attenuated Load (kg/day)	7.03	3.13	(3)
Attenuation of Nitrogen in Pond/Stream (%)	57%	67%	(4)

(a) Flow and N load to streams discharging to Parkers River / Seine Pond includes apportionments of Pond contributing areas.
(b) September 1, 2004 to August 31, 2005.
(c) September 24, 2003 to September 23, 2004.
** Flow is an average of annual flow for 2004-2005 in Plashes Brook
** Flow is an average of annual flow for 2003-2004 in Forest Brook
(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 Parkers 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-7. Summary of annual volumetric discharge and nitrogen load from Plashes Brook and the Forest Brook discharge to Seine Pond (head of Parkers River estuarine system based upon the data presented in Figures IV-6 through IV-9 and Table IV-6.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
Parkers River (main tidal channel) Plashes Brook MEP	**September 1, 2004 to August 31, 2005**	**1,134,083**	**467**	**1095**
Parkers River (main tidal channel) Plashes Brook CCC	Based on Watershed Area and Recharge	**1,134,785**	--	--
Seine Pond Forest Brook MEP	**September 24, 2003 to September 23, 2004**	**300,451**	**138**	**378**
Seine Pond Forest Brook CCC	Based on Watershed Area and Recharge	**299,665**	--	--

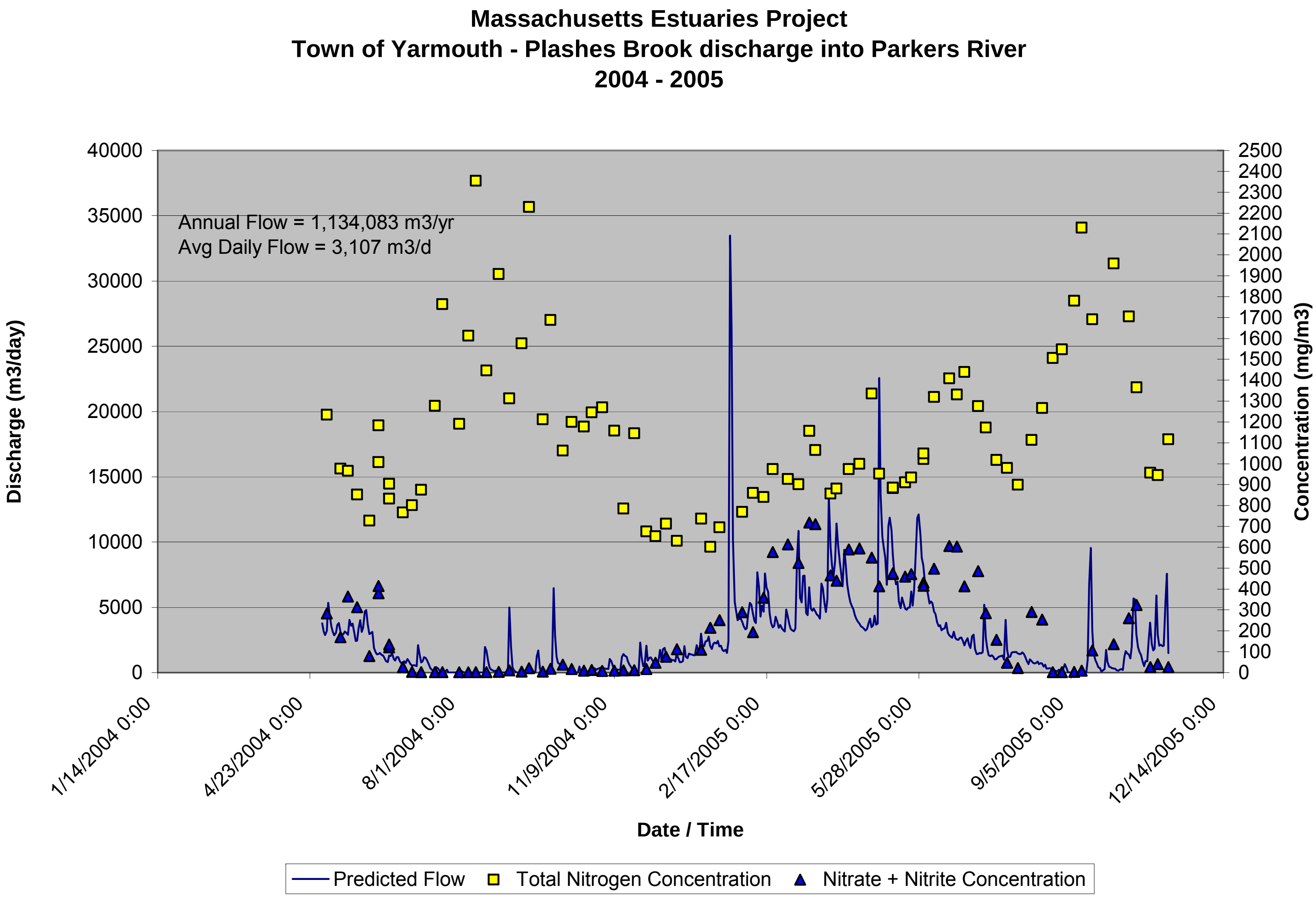


Figure IV-9. Plashes Brook discharge (solid blue line), nitrate+nitrite (blue triangle) and total nitrogen (yellow square) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Parkers River (Table IV-6).

discharging from Upper portion of the sub-watershed and the small freshwater impoundment upgradient of the stream gage would indicate that the Brook is capturing the upgradient recharge (and loads) accurately.

Total nitrogen concentrations within the Plashes Brook outflow were low to moderate, 0.967 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 3.01 kg/day and a measured total annual TN load of 1,097 kg/yr. In Plashes Brook, nitrate made up slightly less than half of the total nitrogen pool (43%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was partially taken up by plants within the pond or stream ecosystems. In addition, the level of remaining nitrate in the stream discharge suggests the possibility for additional uptake by freshwater systems might be accomplished. Opportunities for enhancing nitrogen attenuation could be considered either within Plashes Pond, the small impoundment between Plashes Pond and the Plashes Brook discharge to Parkers River or along the freshwater reach of Plashes Brook.

From the measured nitrogen load discharged by Plashes Brook to the Parkers River estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon lower total nitrogen load (1,097 kg yr^{-1}) discharged from the freshwater Plashes Brook compared to that added by the various land-uses to the associated watershed (2,566 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands prior to discharge to the estuary is 57% (i.e. 57% 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 network of up gradient ponds capable of attenuating nitrogen. The directly measured nitrogen load from the brook was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.3 Surface water Discharge and Attenuation of Watershed Nitrogen: Forest Brook Discharge to Seine Pond (Parkers River head)

Forest Road Brook is a small freshwater stream discharging to the headwaters of Seine Pond. The Forest Road Brook gage site was located at the Forest Road crossing and was immediately down gradient of the herring run. Forest Road Brook captures groundwater discharged from its associated watershed and is "artificially enhanced by surface water outflow from Long Pond associated with anadromous fish management (fish run). Until recently, 2008, outflow from Long Pond was intermittent and comprised about 40% of the Forest Road Brook flow. However, in 2008 the fish run was "improved" to increase flows to support fish passage. This management effort increased flow from Long Pond to Seine Pond by more than 5 fold.

At the time of the primary MEP data collection on Forest Road Brook the small watershed to the Brook and the configuration of control structures both up-gradient and down gradient of the gage location resulted in the brook having intermittent flow (periodically having flow too low to measure). In September 2008 the Town of Yarmouth undertook a restoration of the most down gradient control structure on the shore of Seine Pond as well as at the structure at the end of Clear Brook Road such that flow in the Brook would be maintained year round by outflow from Long Pond. The MEP Technical Team has considered the modification in the flow regime and has adjusted the quantification of flow and attenuation accordingly and this is reflected in the assessment of build-out conditions. It is yet undetermined the extent to which shifting outflow from Long Pond from groundwater recharge to surface water discharge may reduce nitrogen attenuation. It is clear that the increased flow to Seine Pond represents a minor additional nitrogen source to the Parkers River Estuary. However, to the extent that nitrogen attenuation occurs within the riparian zone and creek bed associated with the Forest Brook, the increased nitrogen load from Long Pond is mitigated prior to reaching Parkers River. The rate of nitrogen attenuation as determined prior to the 2008 modification was determined from the predicted nitrogen loading to the sub-watershed region contributing to Forest Brook above the gage site and the measured annual discharge of nitrogen to the tidal portion of the Parkers River system, Figure IV-8.

The freshwater flow and nitrogen load carried by Forest Road Brook to the estuarine waters of Seine Pond was determined using a continuously recording vented calibrated water level gage. As surface water systems can at times be tidally influenced, the brook discharge was checked to confirm freshwater flow could be measured at low tide in the estuary. To confirm that freshwater was being measured, salinity measurements were conducted on weekly water quality samples collected from the gage site. Measured sample salinity was found to be $\leq$0.1 ppt. Therefore, the gage location was deemed acceptable for making freshwater flow measurements. Calibration of the gage was checked monthly. The gage was installed on September 17, 2003 and was set to operate continuously for 16 months such that at least one summer season would be captured in the flow record. Stage data collection continued until March 31, 2005 for a total deployment of 18 months.

Stream flow (volumetric discharge) was measured every 4 to 6 weeks or as stream flow conditions allowed using a Marsh-McBirney electromagnetic flow meter. Stream flow was intermittent, as the Brook periodically went dry during the lowest flow months of the hydrologic year. A rating curve was developed for the gage site based upon these flow measurements and the measured water levels at the gage site. The rating curve was then used to convert the continuously measured stage data to daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. Integrating the flow and nitrogen concentration datasets allowed for the determination of nitrogen mass discharge to the Parkers River system (Figure IV-10 and Table IV-6 and IV-7). In addition, a water balance was constructed based upon the U.S. Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at the gage site.

The annual freshwater flow record for Forest Brook measured by the MEP, taking into consideration periods during the year when the brook was dry, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The data indicate that the freshwater discharge from the brook was comprised of ~60% groundwater discharge and ~40% Long Pond outflow. Measured flow in the brook under present conditions was obtained for one hydrologic year (September 2003 to September 2004). The average daily flow based on the MEP measured flow data was 823 m^3/day compared to the long term average flows determined by the USGS modeling effort (471 m^3/day). The additional measured flow can be ascribed to the intermittent outflows from Long Pond.

The difference between the long-term average flow based on recharge rates over the sub-watershed area and the MEP measured flow in the brook was considered to be negligible given the relatively small flow and associated load compared to other sources of load. However, the difference between the long-term average flow based on recharge rates over the watershed area and the MEP measured flow in brook discharging from the wooded portion of the watershed indicated that the intermittent interaction between the brook and Long Pond needed further clarification. This was further required given the hydraulic modifications that were recently completed in 2008 creating a constant interconnection between Forest Brook and Long Pond.

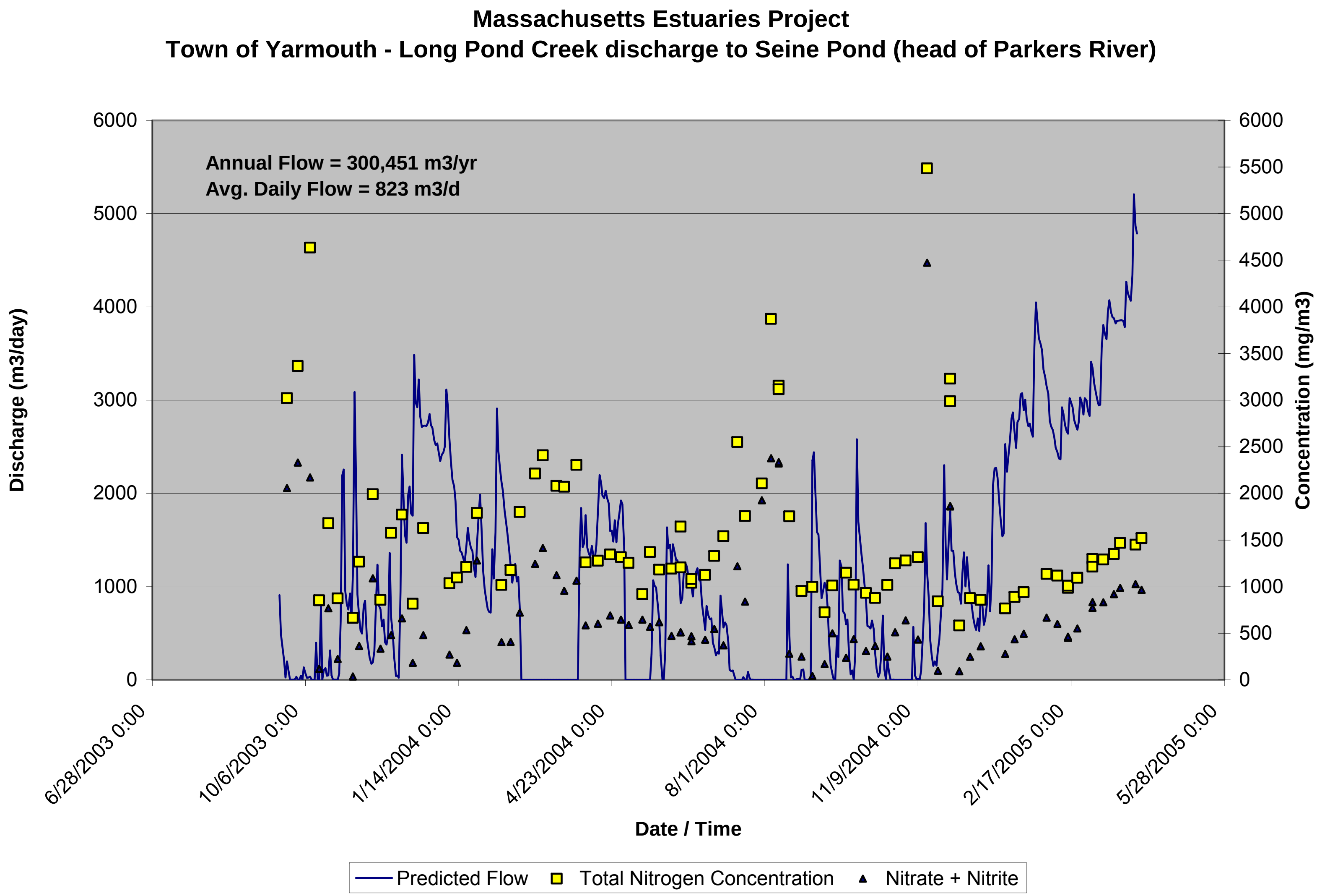


Figure IV-10. Forest Road Brook discharge (solid blue line), nitrate+nitrite (blue triangle) and total nitrogen (yellow squares) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to Parkers River / Seine Pond (Table IV-6).

In order to further refine the evaluation of freshwater flow and attenuated load of nutrients into Seine Pond from its watershed and Long Pond, the MEP Technical Team contacted both the Town Engineering Department responsible for the redesign of the control structures on Forest Brook as well as the Natural Resources Department that monitors the status of the anadromous fish run. These discussions allowed the MEP Technical Team to estimate the average water levels (hence volumetric flow) in the Forest Brook after the 2008 modifications. The objective for the MEP Technical Team was to refine the flow estimate and loading to Seine Pond via Forest Road Brook under the new Forest Road Brook flow regime. To do so the MEP Technical Team used stage and flow data from the MEP stream record to estimate the average flow conditions under the post-2008 Forest Road Brook structure. Initially, the MEP Technical Team compared flows determined in the last month of the MEP stream data record (March 2005) to flow measurements made by the Town of Yarmouth in March, April and May of 2008 to see how well the flows compared under “pre-modification” conditions for Forest Brook. Flow measurements were made by Town of Yarmouth Engineering Department personnel for the period March to May 2008. The measured flows determined by the Town averaged 4,396 m^3/day in March 2008, 3,636 m^3/day in April 2008 and 3,717 m^3/day in May 2008. The average daily flow for Forest Brook as determined by the MEP for March 2005 (3,841 m^3/day) compared favorably with the flows determined by the Town of Yarmouth indicating that it would be reasonable to use MEP measured stage and flows to try and extrapolate what the average flows would be under the new stage conditions in Forest Brook on an average annual basis. Based on the MEP stage and flow record for the period 2003-2005, the above flows occurred at stages between 0.55 and 0.57 feet (~6 inches of water in the brook). During the stream gage deployment period, on average stage in Forest Brook was 0.174 ft (~2 inches) which agreed with qualitative estimates by the personnel in the Natural Resources Department responsible for managing the anadromous fish run on Forest Road Brook. After the modifications to the control structures associated with the Brook, average stages were approximately 0.33-0.50 feet (4-6 inches, based on personal communication). Taking into consideration watershed areas, measured stream nitrogen concentrations and loads, land use analysis, nitrogen concentrations in Long Pond and measured stages and flows, it was deemed appropriate to use an average stage for the restructured Forest Road Brook of 0.33 feet (4 inches) to obtain an estimated average daily flow of ~2500 m^3/day. Utilizing this approach enabled the Technical Team to estimate the difference in flow and load conditions pre- vs. post-modification of the control structures. The increase in nitrogen discharged from the Brook resulting from the higher outflow from Long Pond was made part of the build-out loading, since it occurred after the data collection period for determining "existing conditions".

During the long-term gauging, total nitrogen concentrations within the brook outflow pre-restoration of Forest Brook were moderate for Cape Cod, 1.258 mg N L-1, yielding an average daily total nitrogen discharge to the estuary of 1.04 kg/day and a measured total annual TN load of 378 kg/yr. In the brook surface water system, nitrate represented only about one third (36%) of the total nitrogen pool. This relatively low contribution by nitrate results from the fact that the Long Pond inflow is low in nitrate, as the nitrate entering ponds is typically converted to organic forms. In addition, inorganic nitrogen entering the Brook from both Long Pond and via direct groundwater discharge, is partially taken up by plants associated with the stream bed ecosystems. Although the nitrate concentration in the brook water entering Seine Pond suggests the possibility that additional uptake by freshwater systems might be accomplished in this system, opportunities for enhancing nitrogen attenuation in this small brook may be limited due to the relatively short flow path and levels of inorganic nitrogen.

From the measured nitrogen load discharged by Forest Brook (inclusive of the intermittent flows from Long Pond) to Seine Pond and the nitrogen load determined from the watershed based land use analysis, it appears that there is nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon lower nitrogen load (378 kg yr-1) discharged from the freshwater Brook compared to that added by the various land-uses to the associated watershed (1,141 kg yr-1), the integrated attenuation in passage through ponds, streams and wetlands prior to discharge to the estuary is 67% (i.e. 67% of nitrogen input to watershed does not reach the estuary). This level of attenuation is comparable to other streams evaluated under the MEP and is expected given the intermittent hydraulic connection to Long Pond and the ponds capacity to attenuate nitrogen. The directly measured nitrogen loads from the brook was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter 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 Parkers River embayment systems. The mass exchange of nitrogen between water column and sediments is a fundamental factor in controlling nitrogen levels within coastal waters. These fluxes and their associated biogeochemical pools relate directly to carbon, nutrient and oxygen dynamics and the nutrient related ecological health of these shallow marine ecosystems. In addition, these data are required for the proper modeling of nitrogen in shallow aquatic systems, both fresh and salt water.

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

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

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

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

Failure to account for the site-specific nitrogen balance of the sediments and its spatial variation from the tidal creeks and embayment basins will result in significant errors in determination of the threshold nitrogen loading to the Parkers 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 Parkers 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 15 sites in the Parkers River system. Cores were collected from 8 sites within Seine Pond, 3 sites in the main tidal channel of the Parkers River, one core was collected from the tributary tidal channel leading into Lewis Pond and 3 cores were collected from Lewis Pond (Figure IV-11). All the sediment cores for this system were collected in July-August 2004. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

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

Parkers River Benthic Nutrient Regeneration Cores

- YPR-1 1 core (Seine Pond)
- YPR-2 1 core (Seine Pond)
- YPR-3 1 core (Seine Pond)
- YPR-4 1 core (Seine Pond)
- YPR-5 1 core (Seine Pond)
- YPR-6 1 core (Seine Pond)
- YPR-7 1 core (Seine Pond)
- YPR-8 1 core (Seine Pond)
- YPR-9 1 core (River Main Channel)
- YPR-14 1 core (River Main Channel)
- YPR-15 1 core (River Main Channel)
- YPR -10 1 core (Lewis Pond)
- YPR -11 1 core (Lewis Pond)
- YPR -12 1 core (Lewis Pond)
- YPR -13 1 core (Lewis Pond Channel)

Sampling was distributed throughout the primary component basins of the Parkers 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 Skippy's Pier 1 facility on the shore of the Parkers River, 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°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[4]. The laboratory follows standard methods for saltwater analysis and sediment geochemistry.

[4] Coastal Systems Analytical Facility, 508-910-6325 or d1white@umassd.edu

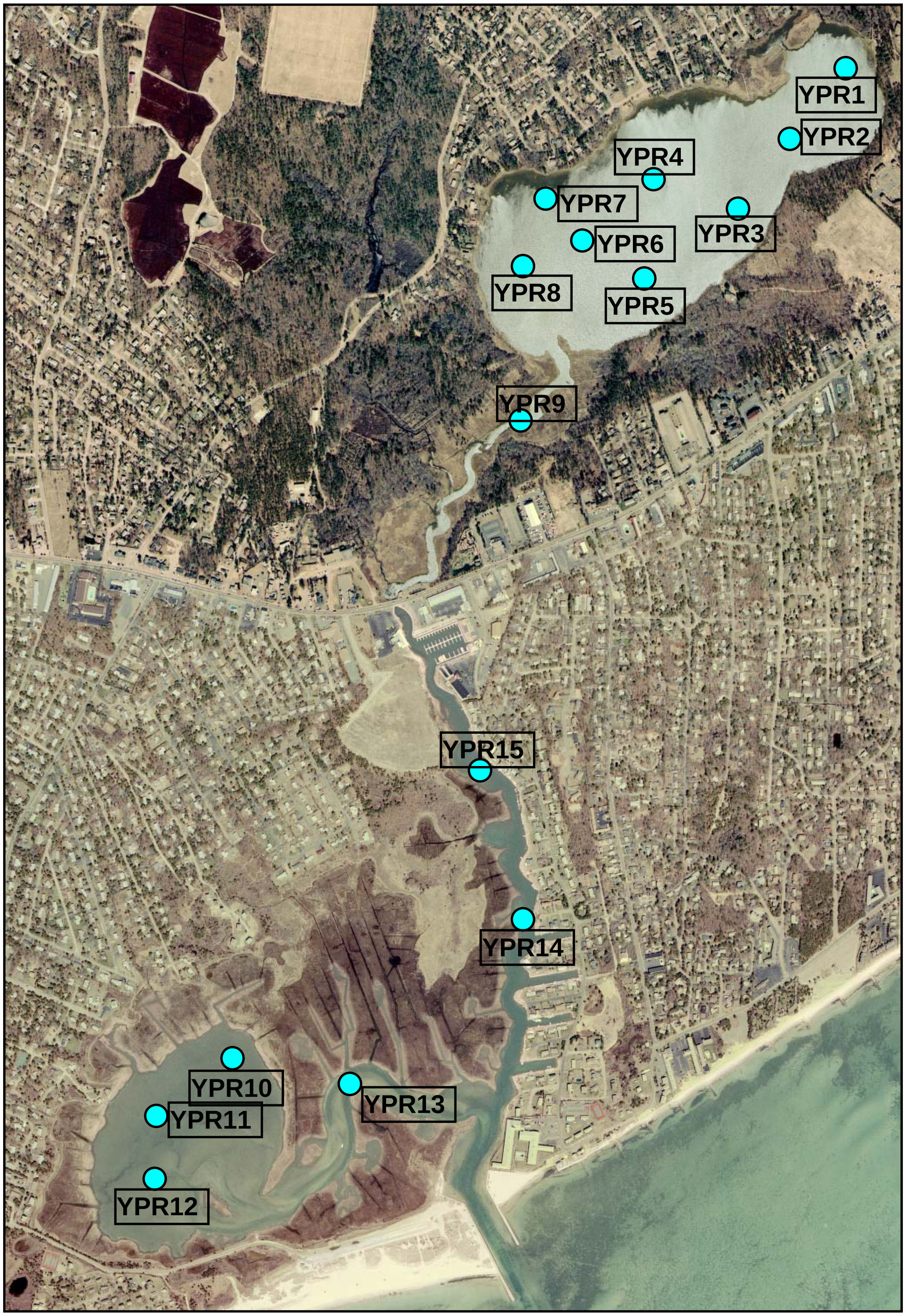


Figure IV-11. Parkers River embayment system sediment sampling sites (blue symbols) for determination of nitrogen regeneration rates. Numbers are for reference to station identifications listed above.

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 Parkers River System.

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

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

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

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

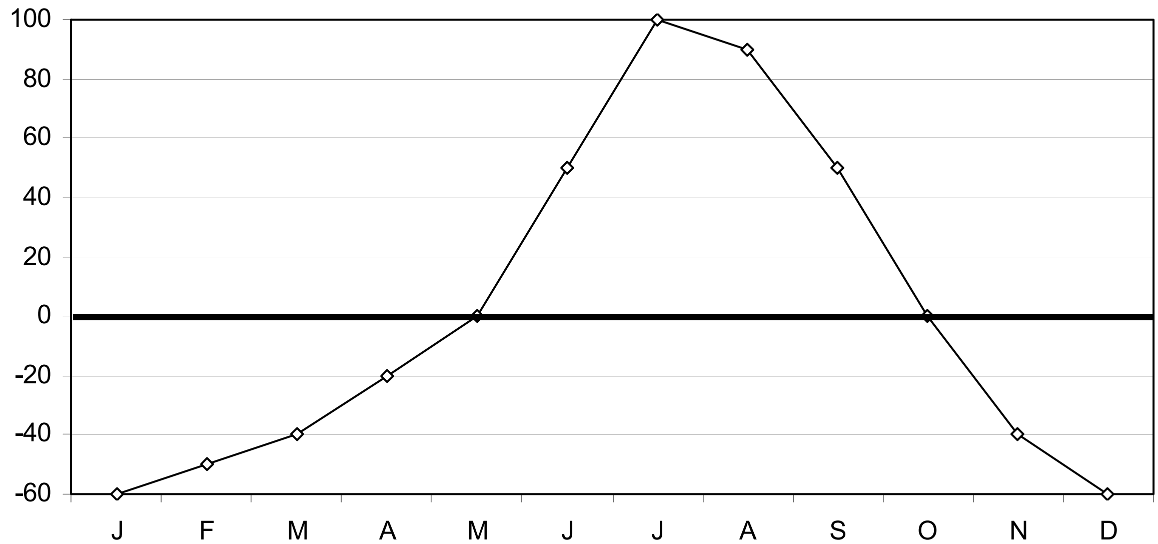


Figure IV-12. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

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

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

The magnitude of the settling of particulate organic carbon and nitrogen into the sediments was accomplished by determining the average depth of water within each sediment site, the average summer particulate carbon and nitrogen concentration within the overlying water and the tidal velocities from the hydrodynamic model (Chapter V). Two levels of settling were used. If the sediments were organic rich and fine grained, and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated coarse-grained sediments and low organic content and high velocities, then half this settling rate was used. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas which are net nitrogen sinks for the aquatic system. This approach has been previously validated in outer Cape Cod embayments (Town of Chatham embayments) by examining the relative fraction of the sediment carbon turnover (total sediment metabolism), which would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins 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 Parkers River embayment system were comparable to other similar embayments with similar configuration and flushing rates in southeastern Massachusetts. In addition, the spatial pattern of sediment N release was also similar to other systems, with the salt marsh basins and creeks showing net nitrogen uptake, the embayment depositional basins with oxidized surficial sediments showing low rates of net nitrogen uptake and the depositional areas within the tidal river showing net nitrogen release. Sediment nitrogen release was primarily within the Parkers River and ranged from 39.9 to 317 mg N mg N m^{-2} d^{-1}. The higher rate was in a marginal depositional area without an oxidized surface layer and with sulfidic sediments. A similar pattern was found in Lewis Bay, where the highly organic anoxic sediments of the dredged basin in the marina region of inner Hyannis Harbor showed high rates of nitrogen release. Similarly, the other areas with net nitrogen release were consistent with similarly structured components of other systems. By example, the lower main tidal channel of Rock Harbor (Orleans/Eastham) showed benthic regeneration rates of 80.8 mg N mg N m^{-2} d^{-1}, and The River within Pleasant Bay showed rates of 12.0 - 34.2 mg N m^{-2} d^{-1}. It should be noted that the most analogous system to the Parkers River main tidal channel assessed by the MEP to date (Centerville River main channel), has nearly identical rates of net nitrogen release as the upper Parkers River site, 32.3 - 36.6 mg N m^{-2} d^{-1} versus 39.9 mg N m^{-2} d^{-1}. The rates in Seine Pond and Lewis Pond are also typical of Cape Cod estuaries. Seine Pond is a headwater depositional basin that is nitrogen enriched (eutrophic) showing low net rates of nitrogen uptake, -16.9 mg N m^{-2} d^{-1}. Similar basins such as the Eel Pond within the Phinneys Harbor System, Eel River within Three Bays, lower Muddy Creek and Frost Fish Creek showed comparable flux rates as Seine Pond, -8.6, -6.4, -16.0, -5.1 mg N m^{-2} d^{-1}, respectively. The net uptake in Seine Pond likely results from its depositional nature and oxidized surficial sediments. Lewis Pond is a very shallow basin surrounded by extensive intertidal salt marsh. The pond is functioning as a salt marsh tidal basin and as such it is naturally organic matter enriched, although external loading appears to be causing some habitat decline within the open water basin. As a distinctly different component basin within the

Parkers River Estuary, Lewis Pond showed comparable rates (-11.8 mg N m^{-2} d^{-1}) to other similarly configured salt marsh basins on Cape Cod. For example, net nitrogen uptake within Mill Creek portion of Lewis Bay (-14.3 mg N m^{-2} d^{-1}), salt marsh basins in the Centerville River System, particularly Scudder Bay (-13.2 mg N m^{-2} d^{-1}) and lower Halls Creek (-11.1 mg N m^{-2} d^{-1}). Lewis Pond is consistent with a general pattern seen in a number of estuaries of similar structure within the MEP region (MEP Cockle Cove Technical Memorandum-Howes et al. 2006).

Net nitrogen release rates for use in the water quality modeling effort for the main basins of the Parkers River system (Chapter VI) are presented in Table IV-8. There was a clear spatial pattern of sediment nitrogen flux, with net uptake of nitrogen within the upper salt pond (Seine Pond) and within the shallow central salt marsh basin of Lewis Pond and net nitrogen release within the tidal river (Parkers River). The sediments within the Parkers River Estuary showed nitrogen fluxes typical of similarly structured systems within the region and appear to be in balance with the overlying waters and the nitrogen flux rates consistent with the level of nitrogen loading to this system and its relatively high flushing rate.

Table IV-8. Rates of net nitrogen return from sediments to the overlying waters of the component basins of the Parkers River System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July -August rates.

	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			
Location	**Mean**	**S.E.**	**# sites**	**i.d. ***
Parkers River Estuarine System				
Seine Pond Basin	-16.9	18.8	8	YPR 1,2,3,4,5,6,7,8
Upper Parkers River	39.9	1.5	1	YPR 9
Lower Parkers River	317.2	231.9	2	YPR 14,15
Lewis Pond Channel	-6.2	1.9	1	YPR 13
Lewis Pond Basin	-11.8	3.5	3	YPR 10,11,12

* Station numbers refer to Figure IV-11.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

This section summarizes field data collection effort and the development of hydrodynamic models for the Parker 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 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 various sub-embayments 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 residence 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. Nantucket Sound). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For the Parker 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 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.

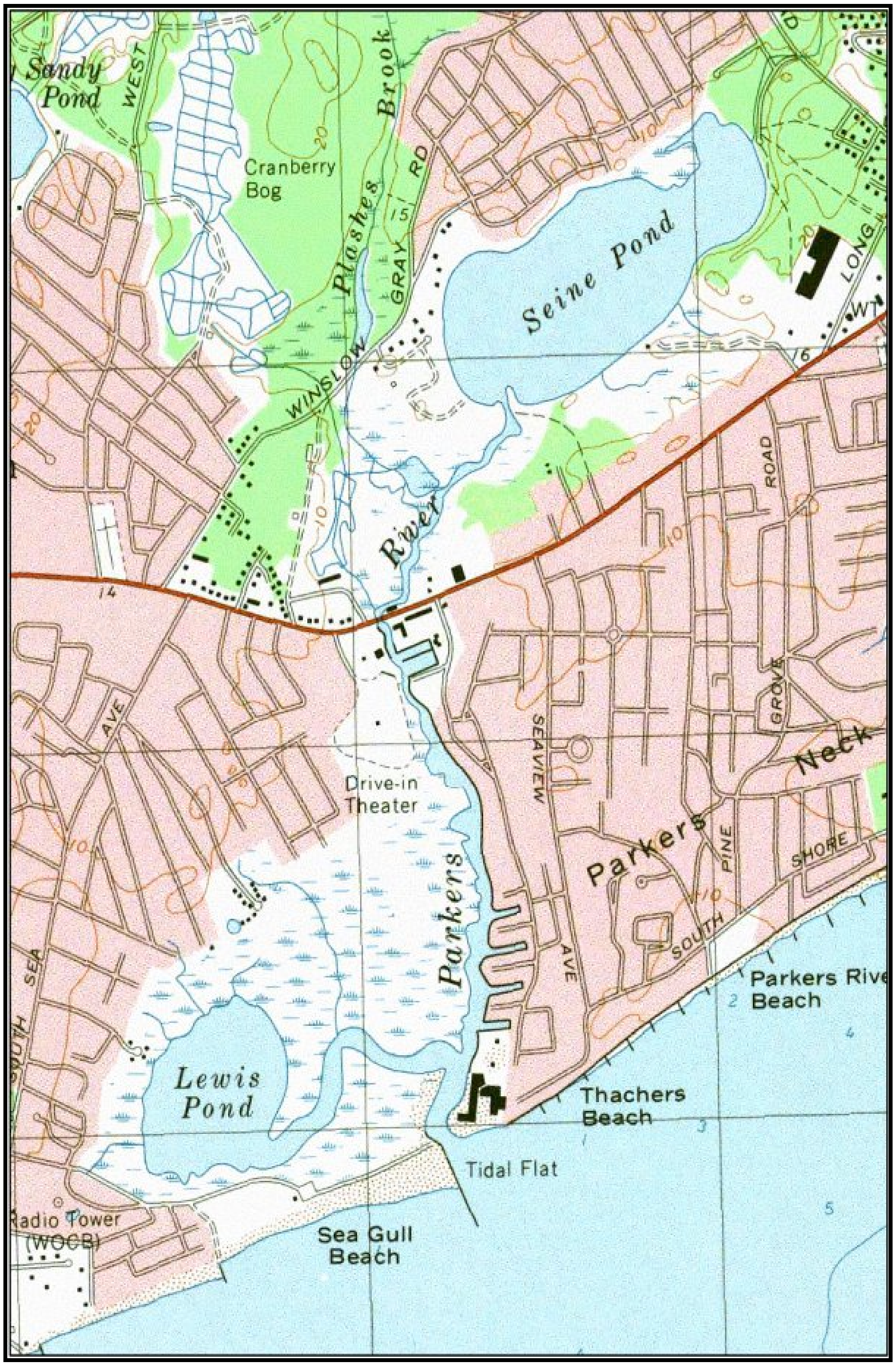


Figure V-1. Map of the Parker River estuary system (from United States Geological Survey topographic maps).

The Parker River estuary (Figure V-1) is a tidally dominated embayment system open to Nantucket Sound. Parker River has two main sub-embayments: Lewis Pond is a shallow sub-embayment (the mean depth is less than 1 foot), situated to the southwest side of the river near the mouth of the Parker River. Seine Pond is a shallow sub-embayment (with a mean depth of less than 2 feet) located at the northern tidal extent of the Parkers River system. The total length of the estuarine reach of mouth of Parkers River system to it northern tidal extent is approximately 2 miles. The Average depth of Parkers River is -4 ft, with maximum depths at -14 ft NGVD occurring in the river channel just north of its junction with Lewis Pond. The scour hole immediately north of the Route 28 culvert Is also approximately 14 feet deep.

Since the water elevation difference between Nantucket Sound and the inland reaches of Parkers River are the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) along the length of the river is negligible, indicating a system that flushed efficiently. Any issues with water quality, therefore, would likely be due other factors including nutrient loading conditions from the system's watersheds, and the tide range in Nantucket Sound.

Circulation in Parkers River 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 within Nantucket Sound at a gage station installed offshore of the mouth of the Bass River, and also at four stations located within the estuary (Figure V-2). All temperature-depth recorders (TDRs or tide gages) were installed for a 40-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 various sub-embayments 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

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 Nantucket Sound. 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 (gauging 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.

To complete the field data collection effort for this study, and to provide model verification data, a survey of velocities was completed at the inlet to Parker River. The survey was performed to determine flow rates at the inlet at discreet times during the course of a full tide cycle.

Figure V-2. Map of the study region identifying locations of the tide gages used to measure water level variations throughout the Parkers River estuary system. Five (5) gages were deployed for the 34-day period between November 4, and December 8, 2004. Each yellow dot represents the approximate locations of the tide gages: (Prv-1) represents the gage in Nantucket Sound (Offshore), (Prv-2) inside Lewis Pond, (Prv-3) in Parkers River south of Rt. 28 bridge, (Prv-4) in Parkers River north of the Rt. 28 bridge, (Prv-5) in Seine Pond.

V.2.1. Bathymetry

Bathymetry data (i.e., depth measurements) for the hydrodynamic model of the Parker River system was assembled from two recent hydrographic survey performed specifically for this study. Historical NOS survey data, where available, were used for areas in Nantucket Sound that were not covered by these more recent surveys.

The first of two hydrographic surveys were conducted November 4th 2004 (RWC 2004), were designed to collect coverage of the shallow water areas of Seine Pond and the Parker River as far south as the rt. 28 bridge. The second hydrographic survey, November 22nd 2004 (CRE, 2004), included coverage of the Parker River from the rt. 28 bridge south, including Lewis Pond. Survey transects in both cases were densest in the vicinity of the inlets, were 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 skiff with an installed precision fathometer (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 into the GPS Data Logger. A digital output produced a single data set consisting of water depth as a function of geographic position (latitude/longitude). The second survey was conducted from an outboard motorboat with an installed precision fathometer (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).

In both surveys, the raw measured water depths were merged with water surface elevation measurements to determine bathymetric elevations relative to the NGVD 1929 vertical datum. Once rectified, the finished processed data were archived as ‘xyz’ files containing x-y horizontal position (in Massachusetts 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.

V.2.2 Tide Data Collection and Analysis

Variations in water surface elevation were measured at a station in at four locations in the Parker River, and at a single station in Nantucket Sound. Stations within the Parker River were located in Lewis Pond (TG-2), in Parkers River south of the Rt. 28 culvert (Prv-3), the Parkers River north of the Rt. 28 culvert (Prv-4), and in Seine Pond (Prv-5). TDRs were deployed at each gauging station from November to December, 2004. The duration of the TDR deployment allowed time to conduct the ADCP and bathymetric surveys (RWC & CRE July 2004), as well as sufficient data to perform a thorough analysis of the tides in the system. The gauging station at Lewis Pond (Prv-2) went dry missing the bottom fraction of the tidal signal in the most extreme case signal (everything less than .3 ft NGVD). To compensate for the loss of data, the tidal signature from the gauging station located south of the rt. 28 bridge was used. Figure V-4 shows that when low tide was greater than 0.3 ft NGVD, tidal data showed little to no change in amplitude or phase congruently allowing for the data from station PRV-3 to be as safe estimate of tidal data at all points south in the Parker River estuary system.

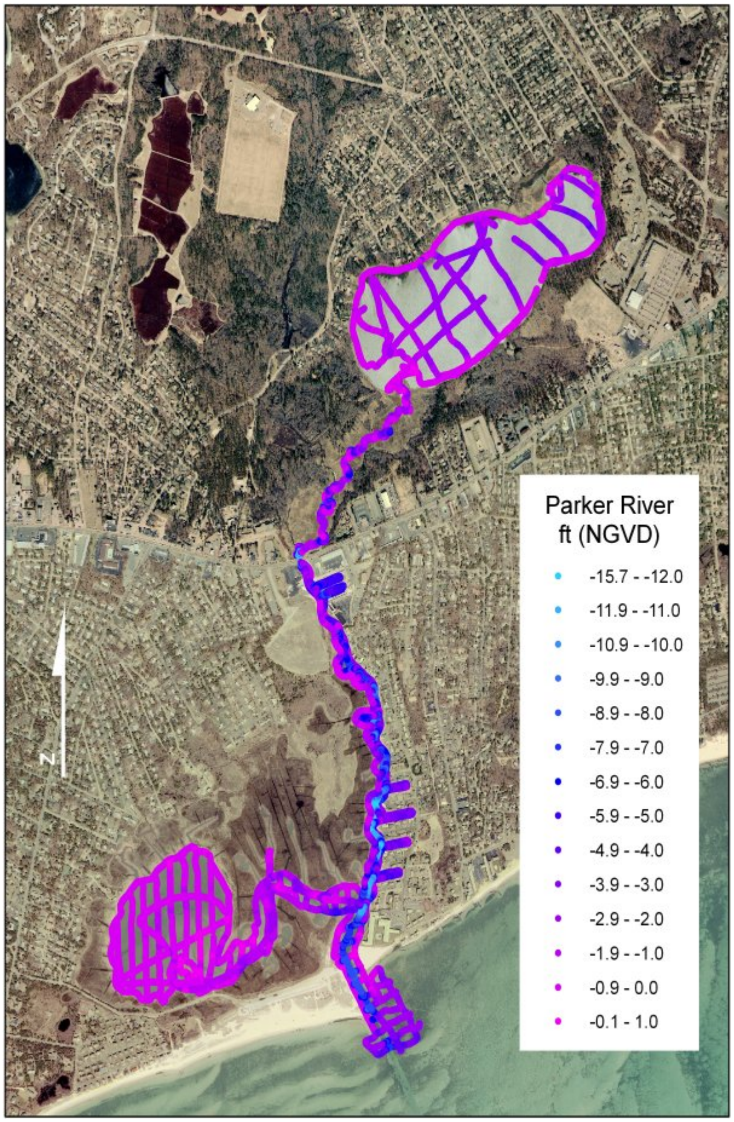


Figure V-3. Bathymetric data interpolated to the finite element mesh of hydrodynamic model.

The tide records from Parker River were corrected for atmospheric pressure variations and then rectified to the NGVD 29 vertical datum. Atmospheric pressure data, available in one-hour intervals from the NDBC Nantucket Sound C-MAN platform, were used to pressure correct the raw tide data. Final processed tide data from stations used for this study are presented in Figure V-5, for the complete 34-day period of the TDR deployment.

A tidal record longer than 29.5 days is 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.

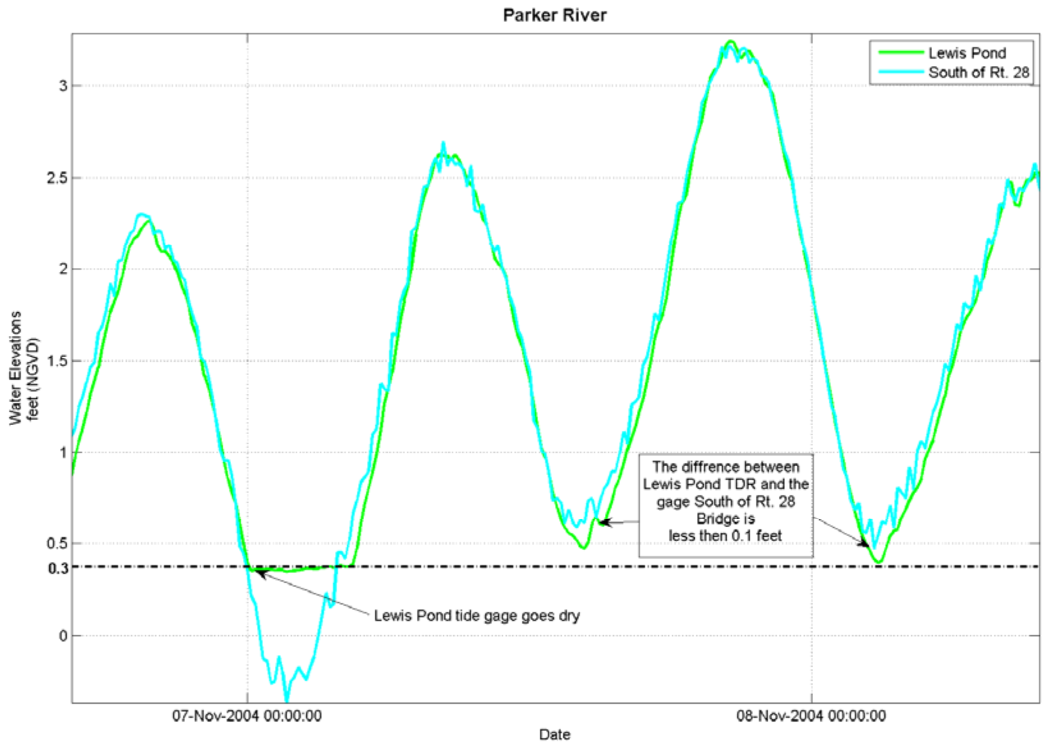


Figure V-4. Plot showing likeness of phase and amplitude at gauging stations located in Lewis Pond and South of Rt. 28 Bridge.

The loss of amplitude together with increasing phase delay with increasing distance from the inlet is described as tidal attenuation. Tidal 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. For Parkers River, a visual comparison in Figure V-6 between tide elevations at the four stations along the system demonstrates significant amplitude reduction between the tidal range of Nantucket Sound and the gauging stations north of Rt. 28. This provides an initial indication that flushing conditions north of the bridge are inhibited by the bridge and may be less then ideal.

To better quantify the tidal hydraulics of the system, the standard tide datums were computed from the 34-day record. These datums are presented in Table V-1. The Mean Higher High Water (MHHW) and Mean Lower Low Water (MLLW) levels represent the average of the highest and lowest water levels each day. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the average of all the high and low tides of a record. The Mean Tide Level (MTL) is simply the average of MHW and MLW. The tides in Nantucket Sound are semi-diurnal, meaning that there are typically two tide cycles in a day. There is usually a small variation in the level of the two daily tides. This variation can be seen in the differences between the MHHW and MHW, as well as the MLLW and MLW levels.

For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available; however, these datums still provide a useful comparison of tidal dynamics within the system.

From the computed datums, it further apparent that there is damping north of the Rt. 28 Bridge. Again, the absence of tide damping exhibited south of the Rt. 28 Bridge indicates that it the lower half of the system is flushing efficiently.

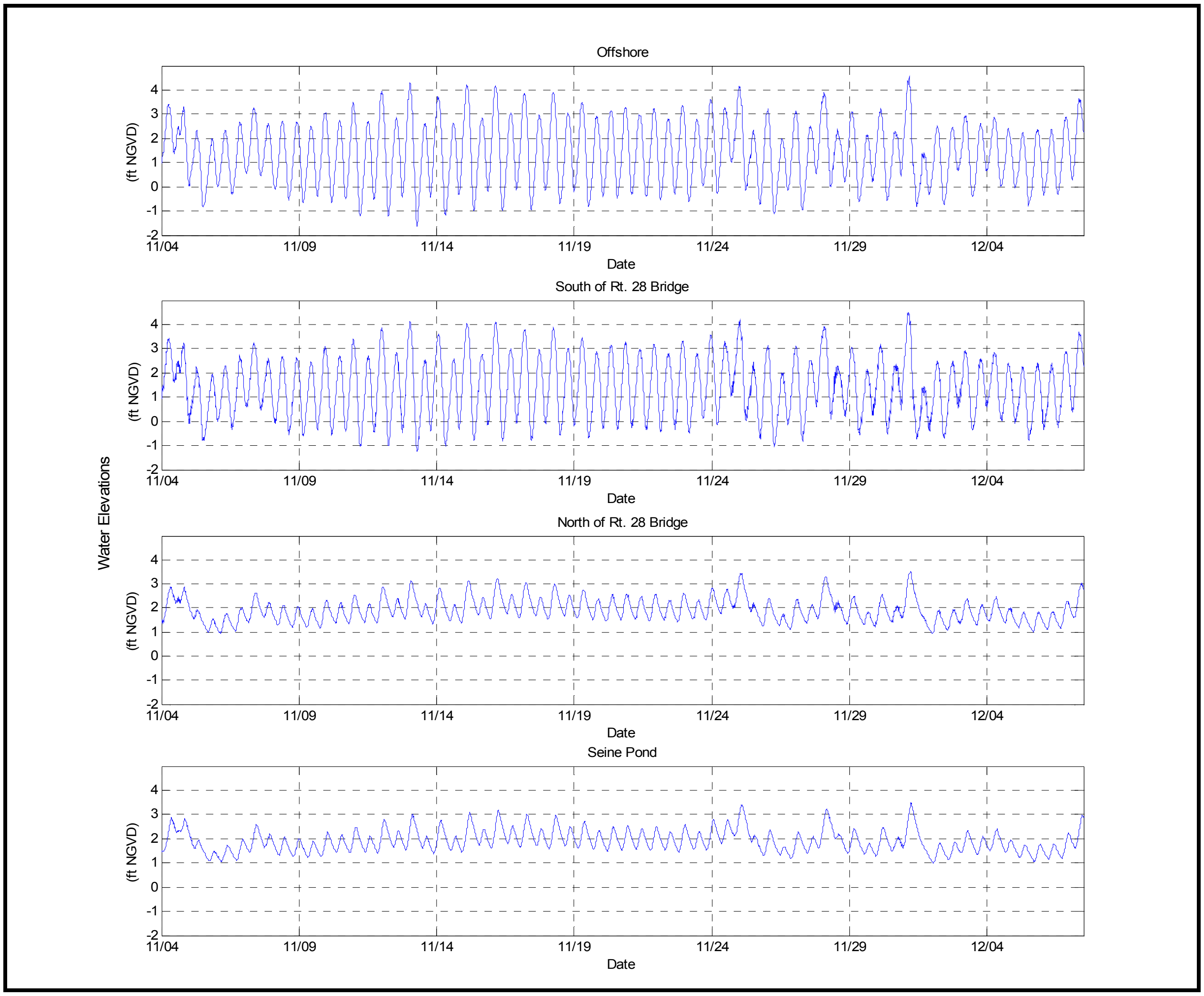


Figure V-5. Water elevation variations as measured at the four locations within the Parkers River estuary system, between November 4th and December 8th, 2004.

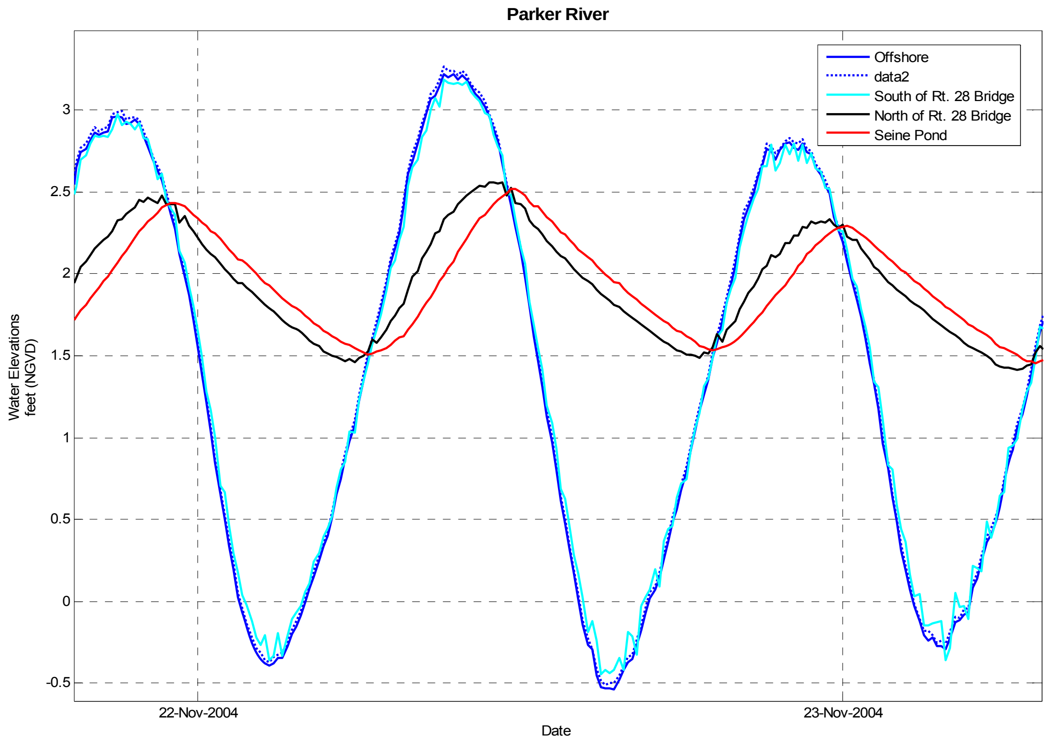


Figure V-6 Plot showing two tide cycles tides at the four stations in the Parkers River system plotted together inclusive of the second offshore gage (data2) deployed to insure 100 percent data recovery. The amplitude reduction north of the Route 28 culvert is significant.

Table V-1. Tide datums computed from records collected in the Parker River Estuary System November 4 - December 8, 2004. Datum elevations are given in feet relative to NGVD 29.

Tide Datum	Nantucket Sound	South of Rt. 28 Bridge	North of Rt. 28 Bridge	Seine Pond
Maximum Tide	4.56	4.49	3.53	3.48
MHHW	3.43	3.39	2.68	2.60
MHW	2.98	2.95	2.41	2.36
MTL	1.34	1.35	1.92	1.92
MLW	-0.31	-0.25	1.42	1.47
MLLW	-0.66	-0.57	1.32	1.37
Minimum Tide	-1.62	-1.24	0.92	0.99

A more thorough harmonic analysis was also performed on the time series data from each gauging 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-7.

Table V-2 presents the amplitudes of eight significant tidal constituents. The M_2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 1.58 feet in Nantucket Sound. The range of the M_2 tide is twice the amplitude, or about 3.16 feet. The diurnal (once daily) tide constituents, K_1 (solar) and O_1 (lunar), possess amplitudes of approximately 0.42 and 0.34 feet respectively and account for the semi-diurnal variance between high tides and low tides seen in Figure V-6. The N_2 tide, a lunar constituent with a semi-diurnal period, is the next largest tidal constituent (equal to in amplitude K_1) and is more then 3.5 time smaller then the main semi-diurnal constituent (M_2) with an amplitude of 0.42 feet. 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 and therefore have more influence on the shape of the tide signal than the other tidal constituents. The effect of the comparatively large amplitude can be seen most clearly in Figure V-6 as the sharply rising and falling semi-diurnal high and low tides.

Table V-2. Tidal Constituents, Parker River Estuary System November 4-December 8, 2004.

AMPLITUDE (feet)								
	M2	M4	M6	S2	N2	K1	O1	Msf
Period (hours)	12.42	6.21	4.14	12	12.66	23.93	25.82	354.61
Nantucket Sound	1.58	0.13	0.05	0.21	0.42	0.42	0.34	0.03
South of Rt. 28 Bridge	1.50	0.13	0.05	0.19	0.40	0.40	0.32	0.03
North of Rt. 28 Bridge	0.41	0.07	0.00	0.04	0.11	0.21	0.18	0.12
Seine Pond	0.37	0.06	0.01	0.04	0.10	0.20	0.17	0.12

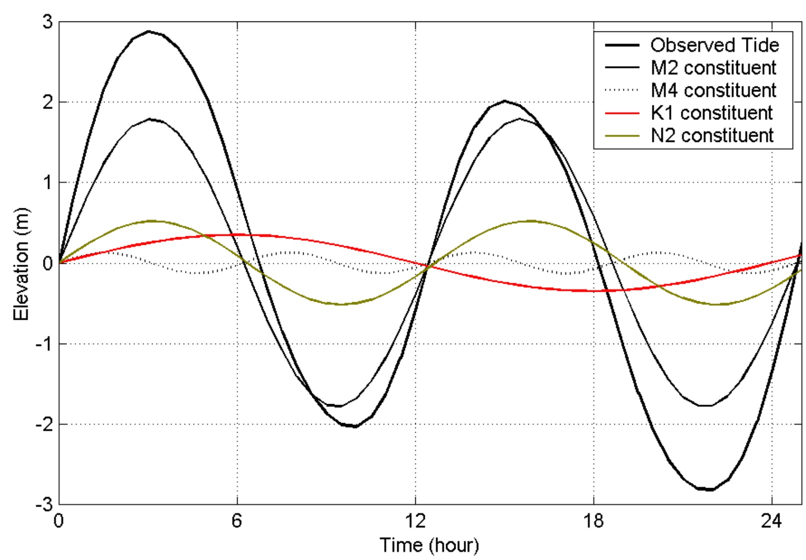


Figure V-7. Example of observed astronomical tide as the sum of its primary constituents. In this example the observed tide signal is the sum of individual constituents (M2, M4, K1, N2), with varying amplitude and frequency.

Table V-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 gage locations inside the Parker River. The difference in attenuation between stations on either side of the bridge is nearly 1.5 times larger than the difference in attenuation between the upper most reaches of the Parker River estuary system. This difference shows the culvert under the bridge is inhibiting the hydraulic efficiency of the northern reaches of the Parker River estuary system.

Table V-3. M_2 Tidal Attenuation, Parker River, November 4 - December 8, 2004. (Delay in minutes relative to Nantucket Sound)	
Location	Delay (minutes)
South of Rt. 28 Bridge	4
North of Rt. 28 Bridge	95
Seine Pond	157

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 Parker River systems 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-8 shows the comparison of the measured tide from Nantucket Sound, with the computed astronomical tide resulting from the harmonic analysis, and the resulting non-tidal residual.

Table V-4 shows that the percentage contribution of tidal energy was essentially equal in all parts of the system, which indicates that local effects due to winds and other non-tidal processes are minimal throughout the systems. The analysis also shows that tides are responsible for approximately 90% of the water level changes in Parkers River, south of Route 28. The remaining 10% was the result of atmospheric forcing due to winds, or barometric pressure gradients acting upon the collective water surface of Nantucket Sound and Parkers River. The total energy content of the tide signal in the southern part of the system does not change significantly, nor does the relative contribution of tidal vs. non-tidal forces. This is further indication that tide attenuation across the inlet and through the southern portion of the system is small. It is also an indication that the source of the non-tidal component of the tide signal is generated completely offshore, with no additional non-tidal energy input inside the system (e.g., from wind set-up).

The results from Table V-4 indicate that hydrodynamic circulation throughout Parker River is dependent primarily upon tidal processes. For the northern reach of Parkers River, the Route 28 culvert plays a dominant role in controlling the hydraulics. When wind and other non-tidal effects are 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 Nantucket Sound was used to force the model so that the effects of non-tidal energy are included in the modeling analysis.

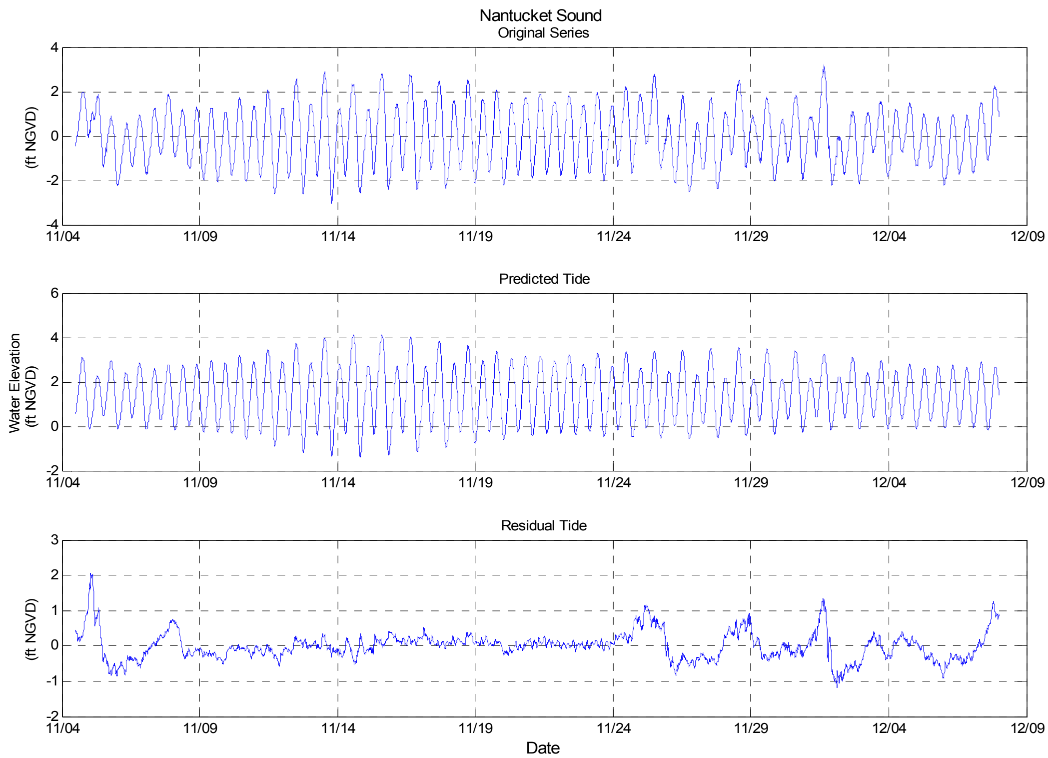


Figure V-8. Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured in Nantucket Sound (Prv-1).

Table V-4. Percentages of Tidal versus Non-Tidal Energy, Parkers River, 2004.

	Total Variance (ft^2)	Total (%)	Tidal (%)	Non-tidal (%)
Nantucket Sound	1.58	100	90.2	9.8
South of Rt. 28 Bridge	1.29	100	86.8	13.2
North of Rt. 28 Bridge	0.23	100	58.1	41.9
Seine Pond	0.21	100	54.4	45.6

V.3. HYDRODYNAMIC MODELING

This study of Parkers 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, 1990a). 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-2V for numerous flushing studies on Cape Cod, including Lewis Bay, Bass River and the Pleasant Bay estuary.

V.3.1 Model Theory

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

RMA-2 is a finite element model designed for simulating one- and two-dimensional depth-averaged hydrodynamic systems. The dependent variables are velocity and water depth, and the equations solved are the depth-averaged Navier Stokes equations. Reynolds assumptions are incorporated as an eddy viscosity effect to represent turbulent energy losses. Other terms in the governing equations permit friction losses (approximated either by a Chezy or Manning formulation), Coriolis effects, and surface wind stresses. All the coefficients associated with these terms may vary from element to element. The model utilizes quadrilaterals and triangles to represent the 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 it is unconditionally stable. Once the equations are solved, corrections to the initial estimate of velocity and water elevation are employed, and the equations are re-solved until the convergence criteria is met.

V.3.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The finite element grid was generated within the shoreline as determined by aerial photos. A time-varying water surface elevation boundary condition (measured tide) was specified just offshore of the entrance based on the tide gage data collected in Nantucket Sound. There were no freshwater recharge boundary conditions included in this model. Once the grid and boundary condition were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through numerous (10+) model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.3.2.1 Grid Generation

The grid generation process was simplified by the use of the SMS package. The aerial photos and bathymetry data were imported to SMS, and a finite element grid was generated to represent the estuary. Figure V-9 illustrates the finite element grid covering the Parkers River system. The entire mesh consists of two-dimensional (depth-averaged) elements.

The finite element grid provided the detail necessary to evaluate accurately the variation in hydrodynamic properties of each estuary. Fine resolution was required to simulate the channel constrictions which can significantly impact the estuarine hydrodynamics. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Reference water depths at each node of the model were interpreted from bathymetry data obtained in the field survey.

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability present. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in the vicinity of the inlet and culvert was chosen to provide a more detailed analysis in these regions of rapidly varying flow. Similarly, small spacing was needed to accurately reproduce flow through the main channel on either side of the culvert. More widely spaced nodes were defined for Lewis Pond and Seine Pond, where flow patterns did not change dramatically.

Figure V-9. Plot of numerical grid (yellow) used for hydrodynamic modeling of Parkers River.

Once the grid is constructed, the system is broken down into regions with each region given its own material type. The material types contain information primarily about element roughness and eddy viscosity which are used to calibrate the model as discussed in Section V.3.2.3 below. By dividing the system into regions and assigning each its own material type, the model can be more easily calibrated, with unique attributes assigned for each region of the system including the culvert. Figure V-10 shows the different material types now color coded. In total there were 7 material types used. There is one very small area which is not discernable in the figure. This was used to model the culvert beneath Route 28 which connects the southern and northern reaches of the system.

V.3.2.2 Boundary Condition Specification

Two types of boundary conditions were employed for the RMA-2 model: 1) "slip" boundaries and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified seaward of the inlet. The water elevations measured from Nantucket Sound provided the required data. Dynamic (time-varying) model simulations specified a new water surface elevation at the offshore boundary every 10 minutes.

V.3.2.3 Calibration

After developing the finite element grids, and specifying boundary conditions, the model was calibrated. Calibration ensured the model predicted accurately what was observed in nature during the field measurement program. Numerous model simulations were required (10+) to fine tune the model and complete the calibration. The calibrated model not only provides the basis for the flushing analysis, but is also a diagnostic tool that could be used to evaluate future changes proposed for the system (e.g. the effects of changing the culvert sizing under Route 28).

Calibration of the flushing model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured. A seven-day period was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.2.2. The seven-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions.

The calibration was performed for an seven-day period beginning 0:00 EST on November 8, 2004. This representative time period was selected because it included the range of tidal conditions typical in the estuary during the 30-day deployment period. To provide average tidal forcing conditions to the predictive water quality model, a time period was chosen that had minimal atmospheric pressure and/or wind effects. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire seven-day simulation. Other methods, such as dye and salinity studies, evaluate tidal flushing over relatively short time periods (less than one day). These short-term measurement techniques may not be representative of average conditions due to the influence of unique, short-lived atmospheric events (e.g. surge or wind setup occurring during a storm). 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.

Figure V-10. Plot of material types within the model domain. Each different colored region represents a unique material type.

V.3.2.3.1 Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient. Initially, Manning's friction coefficient between 0.025 and 0.05 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).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning's coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy channels in the entrance channel, versus the silty bottom of the shallow regions of Lewis Pond, which provided greater flow resistance. Final model calibration runs incorporated various values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were selected based on the Civil Engineering Reference Manual (Lindeburg, 1992) and values required to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-5.

V.3.2.3.2 Turbulent Exchange Coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is more swift, such as inlets and bridge constrictions. According to King (1990a), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). The model for Parkers River was relatively insensitive to turbulent exchange coefficients because there are no regions of strong turbulent flow. Primarily, this can be attributed to the moderate tide range and absence of strong changes in bathymetry throughout the system. Final calibrated turbulent exchange coefficients are shown in Table V-5.

Table V-5. Manning's Roughness and Eddy Viscosity coefficients used in simulations of modeled embayments. These delineations correspond to the material types shown in Figure V-10.

System Embayment	Bottom Friction	Eddy Viscosity
Offshore	0.025	20
South Parkers River	0.025	25
Lewis Pond	0.01	50
Marsh Plain	0.08	50
Culvert	0.04-0.35	50
North Parkers River	0.01	10
Seine Pond	0.01	10

V.3.2.3.3 Marsh Porosity Processes

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 vary 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. It is also useful to help ensure the stability of the model.

V.3.2.3.4 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model predictions for the first TDR deployment was achieved using values for friction and turbulent eddy viscosity listed above. Figures V-11 through V-14 illustrate tides during the seven-day calibration simulation. Modeled (solid line) and measured (each data point plotted as an 'o') tides are illustrated for each measurement location within the system. The 5th field location is offshore and was used as the boundary condition to drive the model. Figures V-11 through V-14 confirm visual agreement between modeled and measured tides throughout the system. The Lewis Pond gage can be seen to go dry throughout the deployment.

Although visual calibration revealed the modeled tidal hydrodynamics were reasonable, tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in Nantucket Sound. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6. The specific values of the constituents for the calibration period differ from those in Table V-2 because constituents were computed for only the seven-day section of the thirty-days represented in Table V-2. Table V-6 compares tidal constituent height and time lag for modeled and measured tides at each of the 3 locations inside the system. The data from Lewis pond was excluded from this analysis as the gage going dry prevents the accurate calculation of the constituents from the record. Time lag represents the time required for a constituent to propagate from Nantucket Sound to each location.

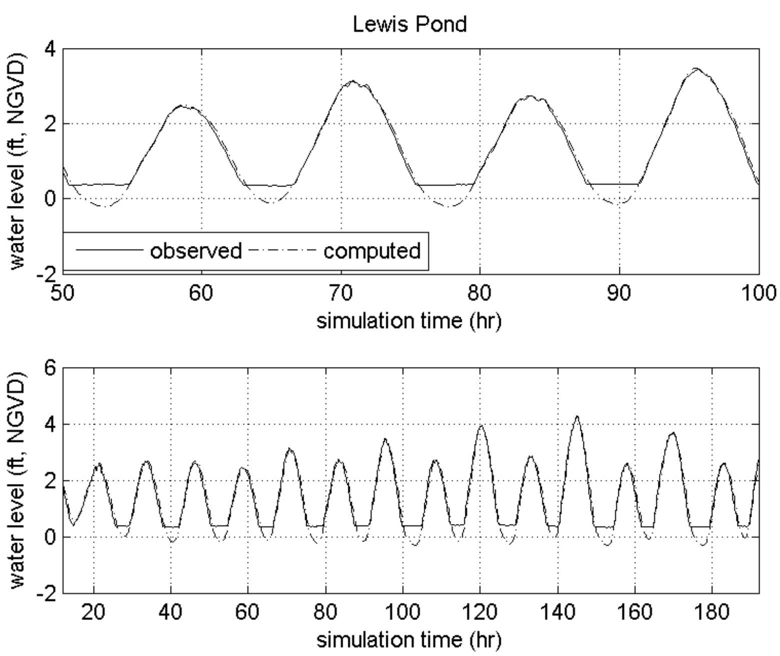


Figure V-11. Observed versus computed water level elevations for Lewis Pond. Top plot shows a subsection of the complete time covered by the bottom plot.

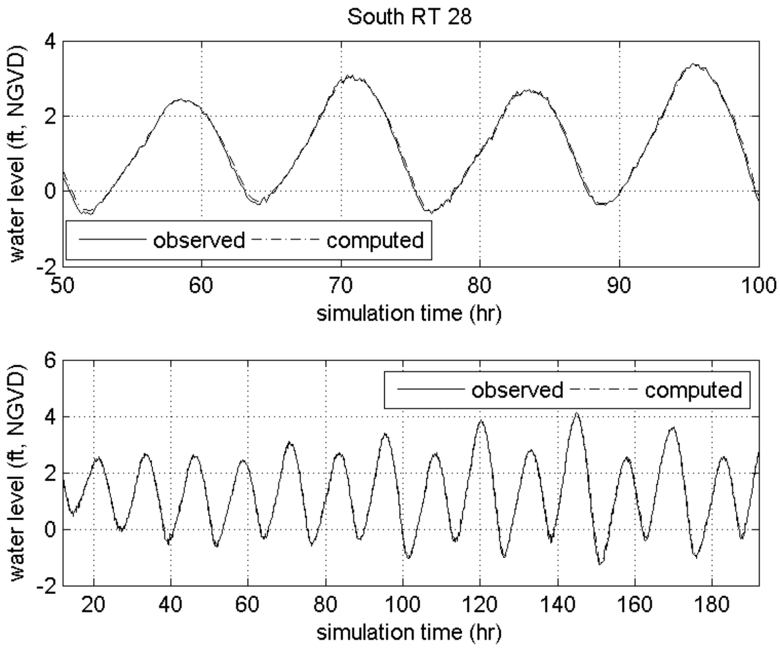


Figure V-12. Observed versus computed water level elevations south of the Rt. 28 culvert. Top plot shows a subsection of the complete time covered by the bottom plot.

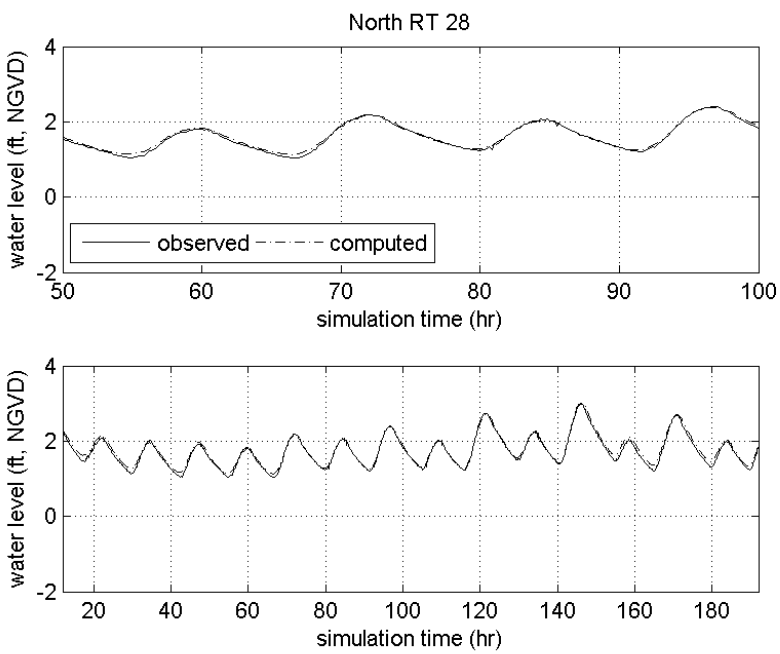


Figure V-13. Observed versus computed water level elevations north of the Rt. 28 culvert. Top plot shows a subsection of the complete time covered by the bottom plot.

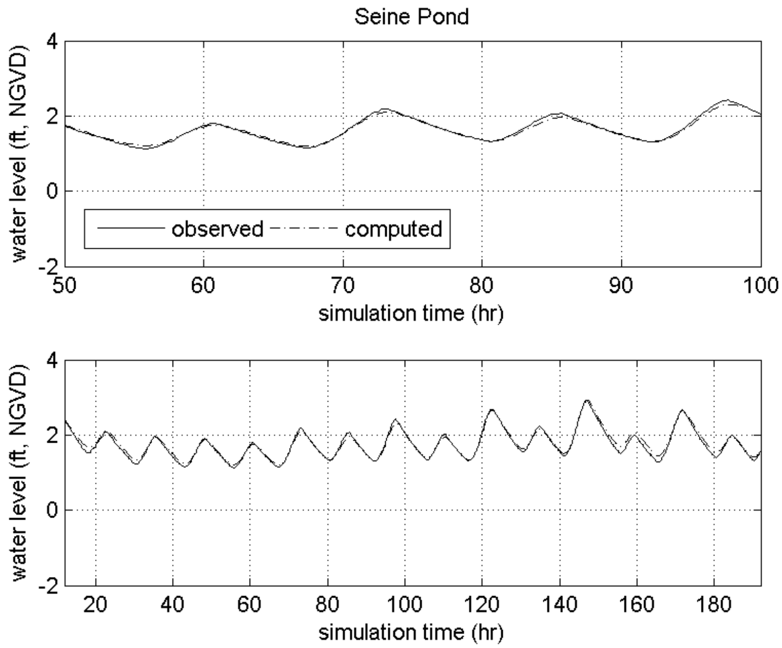


Figure V-14. Observed versus computed water level elevations for Seine Pond. Top plot shows a subsection of the complete time covered by the bottom plot.

The constituent calibration revealed excellent agreement between modeled and measured tides. Errors associated with tidal constituent height were on the order of 0.01 ft, which is less than the accuracy of the tide gages (0.032 ft). Time lag errors were typically less than the time increment resolved by the model (0.17 hours or 10 minutes), indicating good agreement between the model and data. Since tidal amplitude and phase attenuation was relatively minor, with the exception of Seine Pond constituent calibration required that the M_2 constituent propagate into the Ponds with minimal resistance. The tide attenuation at Seine Pond primarily results from the narrow culvert. The hydrodynamic model was able to predict accurately the effect of this constriction on flow properties.

The hydrodynamic model's ability to predict propagation of the secondary non-linear constituents through the estuary is important for understanding the attenuation of the tidal signal and the impact this has on estuarine circulation. Of primary interest is the M_4 constituent, which can be used to determine the flood dominance (sediment trapping characteristics) of an estuarine system. Proper prediction of M_4 provides confidence in the model's accuracy, since this indicates that the model is capable of simulating the tidal wave form and size. Similar to the model predictions for M_2, comparison of the information from Table V-6 indicates that the modeled phase of M_4 falls within approximately one time step of the observed value.

Table V-6. Tidal constituents for measured water level data and calibrated model output

Location	Constituent Amplitude (ft)				Phase (deg)	
Model calibration run						
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
South Parkers River	1.70	0.16	0.07	0.48	-89.7	-44.5
North Parkers River	0.43	0.07	0.00	0.24	-45.2	-164.6
Seine Pond	0.36	0.06	0.00	0.23	-11.2	-95.2
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
South Parkers River	1.74	0.17	0.07	0.48	-90.6	-52.2
North Parkers River	0.45	0.09	0.01	0.25	-44.1	-167.0
Seine Pond	0.40	0.08	0.01	0.23	-13.6	-97.9
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ϕM_2	ϕM_4
South Parkers River	-0.04	-0.01	0.01	0.01	1.9	8.0
North Parkers River	-0.02	-0.02	0.00	0.00	-2.4	2.4
Seine Pond	-0.04	-0.02	-0.01	0.00	4.9	2.8

V.3.2.4 Model Verification Using Horizontal ADCP Measurements

The calibration procedure used in the development of the finite-element models required a match between measured and modeled tides. To verify the performance of the Parkers River model, computed flow rates were compared to flow rates measure using an ADCP. The ADCP data survey efforts are described in Chapter III and the survey locations are shown in Figure V-2 above. For model verification, the model was run for a 3 day period ending with the end of the run corresponding to the day of the actual ADCP survey, December 4, 2004. Model flow rates were computed in RMA-2 at continuity lines (channel cross-sections) that correspond to the actual ADCP transects followed in each survey.

Comparisons of the measured and modeled volume flow rates in the Parkers River system are shown in Figures V-15 and V-16. For each figure, the top plot shows the flow comparison, and the lower plot shows the time series of tide elevation for the same period. Each ADCP point (blue circles shown on the plots) is a summation of flow measured along the ADCP transect. The first data plotted is labeled North of Inlet, which is located inside the inlet throat, immediately south of the confluence with Lewis Pond. The second transect line was immediately south of the marina near Route 28.

Data comparisons at both ADCP transect show good agreement with the model predictions. The calibrated model accurately describes the discharge magnitude at both locations. In general the model results record slightly larger values than the ADCP measurements. This is primarily due to the fact that the ADCP survey excludes flow from the extreme edges of the transect due to depth limitations from the survey boat itself. The R^2 correlation coefficient between data and model results are 0.89 for the southern transect (Figure V-15 below) and 0.81 for the northern transect (Figure V-15 below). There is a notable difference in the total flowrate passing each location, with flows near the inlet approaching 1000 ft^3/s while the maximum flowrate near Route 28 is only 200 ft^3/s.

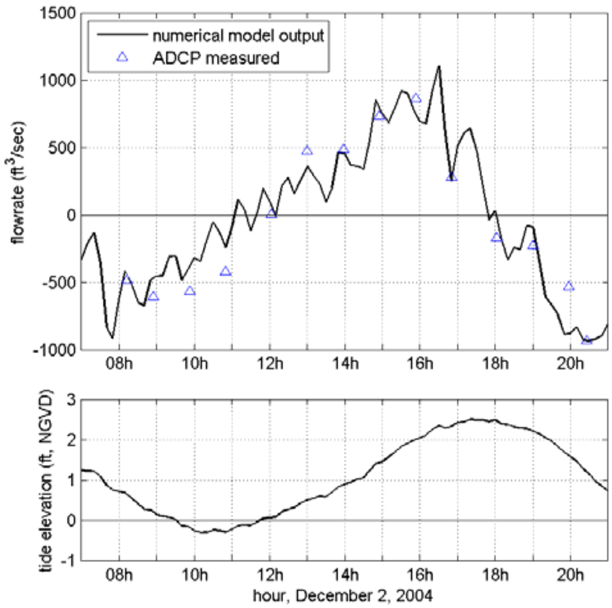


Figure V-15. Flow rate measurements collected on December 4, 2004 between the inlet and the confluence with Lewis Pond. The top plot shows measured and modeled flow rates while the bottom plot shows the offshore tide for the same time period. R^2 correlation is 0.89 and RMS error is 179 ft^3/sec.

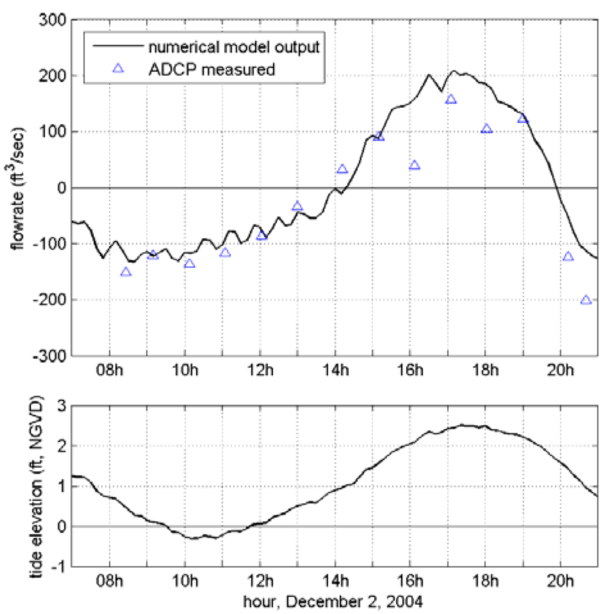


Figure V-16. Flow rate measurements collected on December 4, 2004 immediately south of the marina below Route 28. The top plot shows measured and modeled flow rates while the bottom plot shows the offshore tide for the same time period. R^2 correlation is 0.81 and RMS error is 50 ft^3/sec.

V.3.2.5 Model Circulation Characteristics

Tides in Parkers River affect sediment transport, pollutant dispersion, and water circulation. The calibrated hydrodynamic model provided an unparalleled tool to evaluate details of tidal circulation in the system. For example, field measurements of current flow within a system, using either single-point current meters or Lagrangian drifters, are intrinsically limited. Single point measurements are limited to small regions of the flow, and cannot account for spatial variations in the current throughout a region. Lagrangian drifters (drogues) follow the spatial track of the flow, but are limited to a single 'snapshot' of time at each location and do not resolve temporal variations in the flow. Numerical models offer both spatial and temporal coverage of circulation patterns that reveal the essence of the hydrodynamic behavior. Such insight is invaluable in evaluating tidal characteristics.

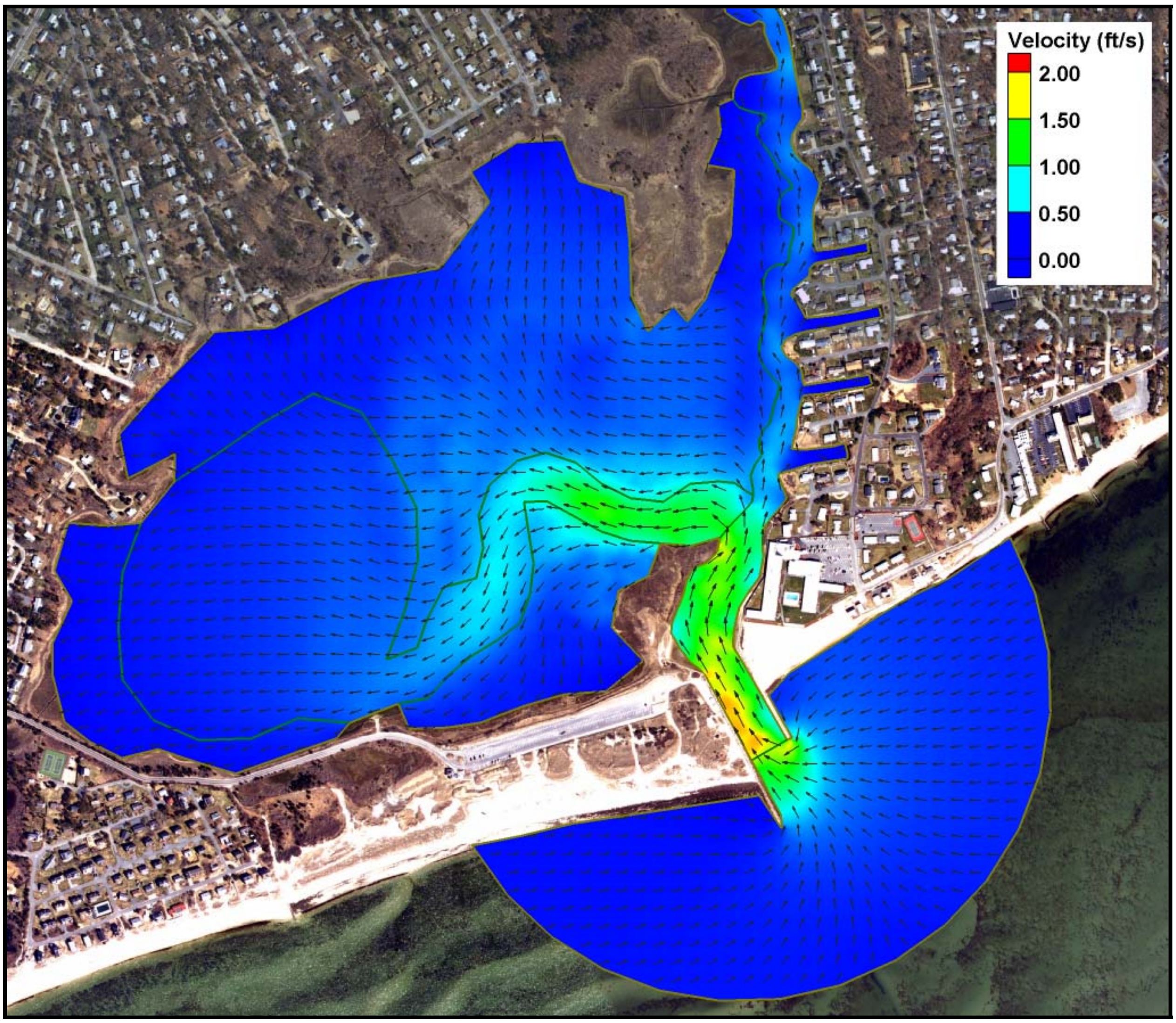


Figure V-17. Velocity contours and vectors during flood tide in the southern reach of the Parkers River System. Green outline represents the edge of the marsh plain and the channel banks.

Figure V-17 shows a typical flow pattern throughout the southern portion of the system during flood tide. Velocities increase as water passes through the constriction of the inlet. Velocities remain high up to the confluence with Lewis Pond. At this point velocities in the main channel north of the confluence decrease to under 1 ft/s while a majority of flow is moving west into Lewis Pond. This diversion of flow into the pond explains the almost order of magnitude difference in flow rates in the ADCP analysis above.

V.4. FLUSHING CHARACTERISTICS

Since freshwater inflow is negligible in comparison to the tidal exchange through each inlet, the primary mechanism controlling estuarine water quality within Parkers River is tidal circulation. A rising tide in Nantucket Sound creates a slope in water surface from the sound into the estuary. Consequently, water flows into (floods) the estuary. Similarly, the estuary drains into the bay on an ebbing tide. This exchange of water between the estuary and bay is defined as tidal flushing. The calibrated hydrodynamic model is a tool to evaluate quantitatively tidal flushing of the system, and was used to compute flushing rates (residence times) and tidal circulation patterns.

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

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

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

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. For example, the **system residence time** is the average time required for water to migrate from Seine Pond to Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Seine Pond to South Parkers River. 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. This is a valid approach in this case, since it assumes the bay has relatively higher quality water relative to the estuary.

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 pollutant/nutrient dispersion. The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Parkers River system.

Residence times reflect the lack of tidal damping through the harbor mouth as well as the bathymetry found in Parkers River and each sub-embayment. Since tidal waters flow freely into the system, the volume of water exchanged during a tidal cycle can be approximated by the surface area of the embayment multiplied by the tide range. For systems with little tidal damping, the bathymetry tends to control residence times, where shallow sub-embayments will exhibit lower residence times than deeper sub-embayments. This is most clearly illustrated by Seine Pond, which is the farthest north sub-embayment within the system. Seine Pond is connected to the southern reaches of the system by a shallow channel and small culvert. The tidal exchange through the culvert is severely restricted. This restriction causes the percentage of total volume exchanged with South Parkers River to be relatively low compared to other sub-embayments within the system.

The relatively long residence time for some sub-embayments (e.g. Seine Pond) revealed the inadequacy of using system residence time alone to evaluate water quality. By definition, smaller sub-embayments have longer residence times; therefore, residence times may be misleading for small, remote parts of the estuary. Instead, it is useful to compute a local residence time for each sub-embayment. A local residence time represents the time required for a water parcel to leave the particular sub-embayment. For instance, the local residence time for Seine Pond represents the time required for a water parcel to be flushed from the pond into the South Parkers River. Local residence times are computed as the volume of the sub-embayment divided by the tidal prism of that sub-embayment, and units are converted to days. Table V-8 lists local residence times for several areas within Parkers River.

Computed mean volumes and tide prisms for existing conditions are presented in Table V-7 for the entire Parkers River system and two subdivisions. System and Local residence times calculated using the volumes listed in Table V-7 are presented in Table V-8. It can be seen that local residence times based on the mean volume of each subdivision are significantly lower than residence times based on the volume of the entire estuary. For example, water in Seine pond on an average tidal cycle flushes through the entrance to Parkers River in 5.0 days, but flushes into the South Parkers River in 2.5 days. Generally, a local residence time is only useful where the adjacent embayment has high water quality.

Table V-7. Embayment mean volumes and average tidal prism during simulation period. Existing Conditions		
Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Parkers River (system)	28,565,000	15,461,000
Upper Parkers River	15,476,000	3,348,000
Seine Pond	14,214,000	2,959,000

Table V-8. Computed System and Local residence times for embayments in the system. Existing Conditions		
Embayment	System Residence Time (days)	Local Residence Time (days)
Parkers River (system)	1.0	1.0
Upper Parkers River	4.4	2.4
Seine Pond	5.0	2.5

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

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the embayments were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated hydrodynamic model representing the transport of water within the Parkers River system. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic model output used for the water quality model calibration was the 14.5 day (28 tide cycle) period beginning November 9, 2004 2245 EST. This period overlaps with that used in the flushing analysis presented in Chapter V. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model to reach a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

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

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in the area map presented in Figure VI-1. The multi-year averages present the “best” comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data are the minimum required to provide a baseline for MEP analysis. Seven years of data (collected between 2002 and 2008) were available at most stations.

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

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 Parkers River estuarine system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of Parkers River. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US

Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. The MEP Technical Team has utilized this model in water quality studies of other embayment systems in southeastern Massachusetts, including Pleasant Bay (Howes *et al*., 2006); New Bedford Harbor (Howes *et al*., 2008) and Edgartown Great Pond, MA (Howes *et al*., 2008).

Table VI-1. Measured data and modeled Nitrogen concentrations for the Parkers River estuarine system used in the model calibration plots of Figures VI-2 and VI-3. All concentrations are given in mg/L N. “Data mean” values are calculated as the average of the separate yearly means. Data represented in this table were collected in the summers of 2002 through 2008.

Sub-Embayment	MEP monitoring station	data mean	s.d. all data	N	model min	model max	model average
Lewis Pond	PR-4	0.868	0.227	36	0.563	1.515	0.859
Lower Parkers River	PR-3	0.663	0.167	32	0.309	0.760	0.491
Upper Parkers River	PR-2	0.776	0.216	37	0.395	1.022	0.802
Seine Pond - Lower	PR-1	0.948	0.225	34	0.819	1.046	0.965
Seine Pond - Upper	PR-5	0.994	0.229	24	0.953	1.059	1.007
Nantucket Sound	NTKS	0.294	0.062	4	-	-	-

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

VI.2.1 Model Formulation

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

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

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

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

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

VI.2.2 Water Quality Model Setup

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

For each model, an initial total N concentration equal to the concentration at the open boundary was applied to the entire model domain. The model was then run for a simulated month-long (28 day) spin-up period. At the end of the spin-up period, the model was run for an additional 14.5 tidal-day (348 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 Parkers River model.

VI.2.3 Boundary Condition Specification

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

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

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Nantucket Sound, offshore the River inlet, was set at 0.294 mg/L, based on SMAST data.

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	20.562	1.096	-5.820
Upper Parkers River	16.408	0.049	0.775
Lower Parkers River	12.652	0.266	28.420
Lewis Pond	17.400	0.616	5.698
System Total	67.022	2.027	29.074

VI.2.4 Model Calibration

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

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

For model calibration, the 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 near the modeled mean because the monitoring data are collected, as a rule, during mid ebb tide.

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for each system. Computed root mean squared (rms) error is less than 0.08 mg/L, which demonstrates a good fit between modeled and measured data for this system.

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

Embayment Division	E m^2/sec
lower Parkers River	16.5
Lewis Pond	0.3
lower system marsh plain	0.1
Route 28 Culvert	20.0
upper Parkers River	40.0
upper system marsh plain	5.0
Seine Pond	2.5

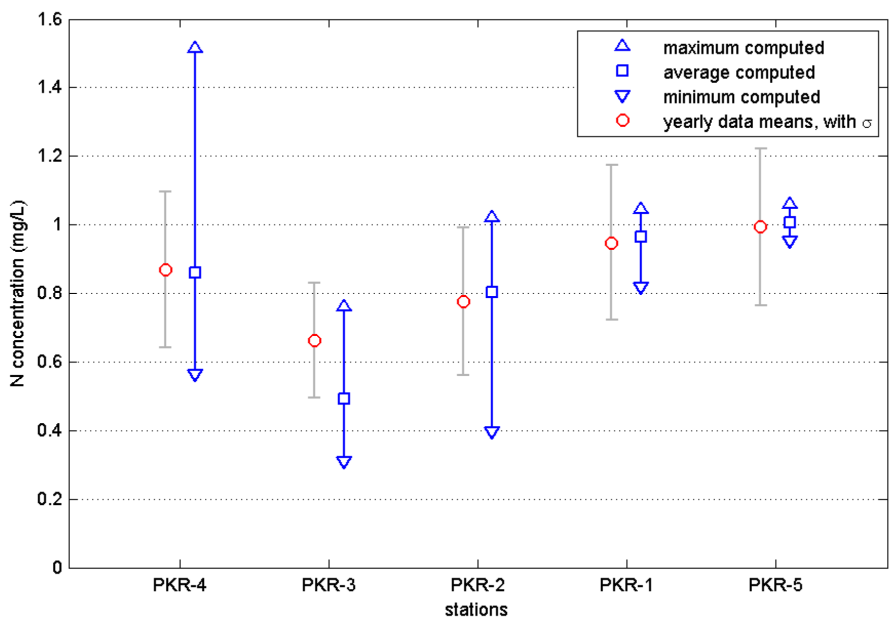


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Parkers River system. Station labels correspond with the MEP IDs provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation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

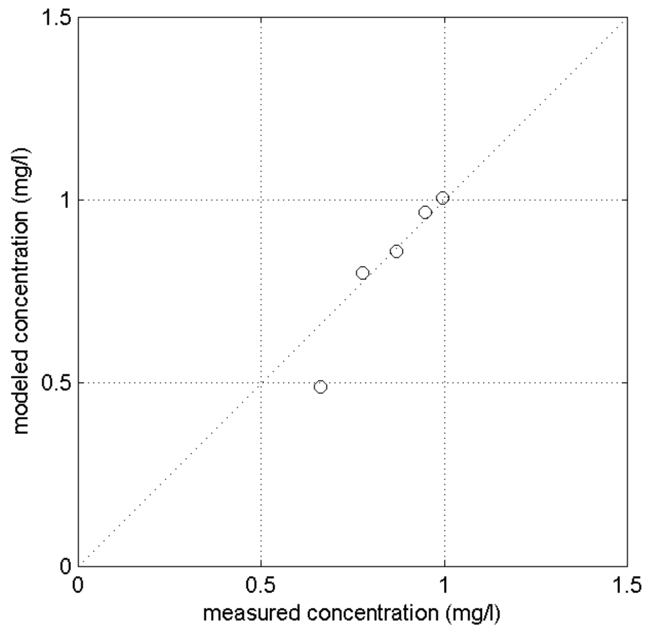


Figure VI-3. Model total nitrogen calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for the model are 0.57and 0.08 mg/L respectively.

A contour plot of calibrated model output is shown in Figures VI-4. In this figure, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full 14-tidal-day model simulation output period.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Parkers River system using salinity data collected at the same stations as the nitrogen data. Comparisons of modeled and measured salinities are presented in Figures VI-5 and VI-6, with contour plots of model output shown in Figure VI-7. The rms error of the model is 1.2 ppt.

The only required inputs into the RMA4 salinity model of the system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, and groundwater inputs. The open boundary salinity was set at 30.1 ppt. Surface water and groundwater input salinities were set at 0 ppt. Surface water stream inputs to the model include 0.34 ft^3/sec (823 m^3/day) for Forest Brook, 1.27 ft^3/sec (3,109 m^3/day) for Plashes Brook, and 3.36 ft^3/sec (8,224 m^3/day) for Gray Brook. Groundwater inputs used for the model were 1.65 ft^3/sec (4,038 m^3/day) for the Seine Pond basin watershed, 0.28 ft^3/sec (689 m^3/day) for the Upper Parkers River watershed, 0.53 ft^3/sec (1,288 m^3/day) for The lower Parkers River watershed, and 1.73 ft^3/sec (4,231 m^3/day) for the Lewis Pond basin. Groundwater flows were distributed evenly in the model through the use of several 1-D element input points positioned along the model's land boundary.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

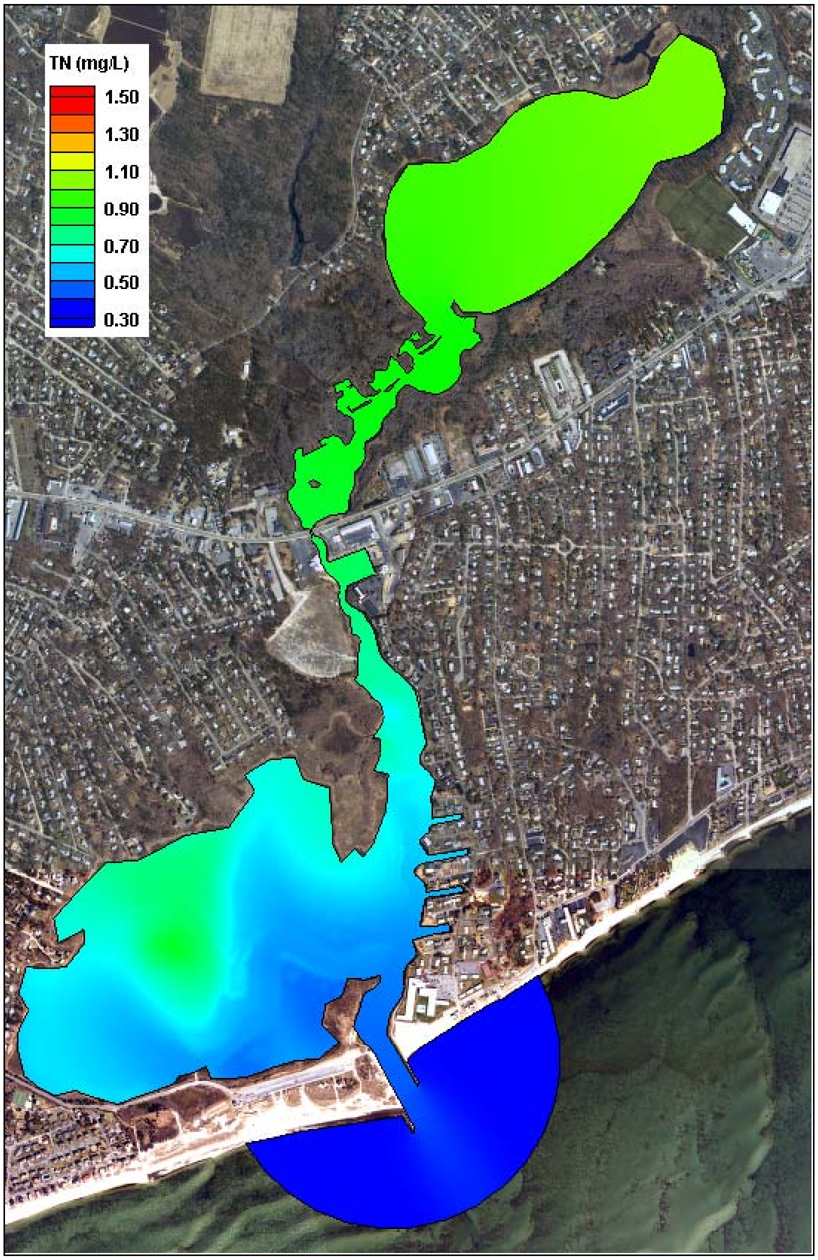


Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Parkers River system.

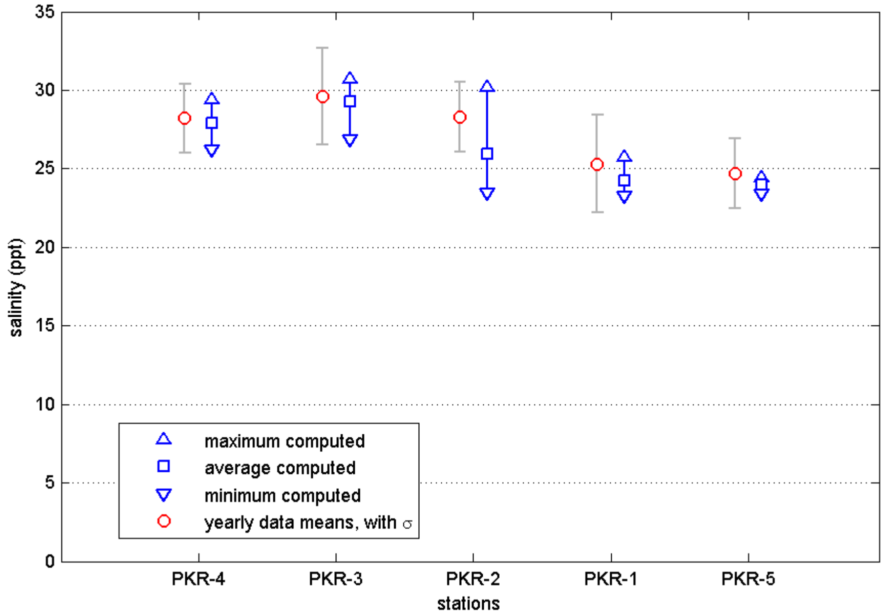


Figure VI-5. Comparison of measured and calibrated model output at stations in Parkers River. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with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

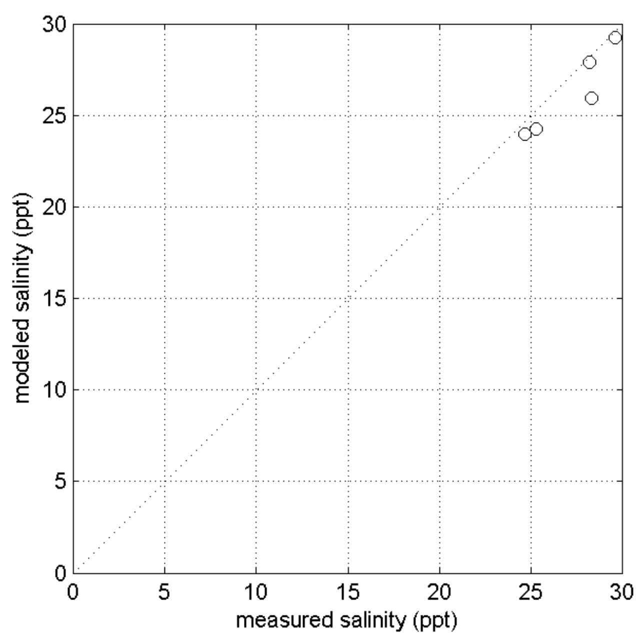


Figure VI-6. Model salinity target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) is 0.57 and RMS error for this model verification run is 1.2 ppt.

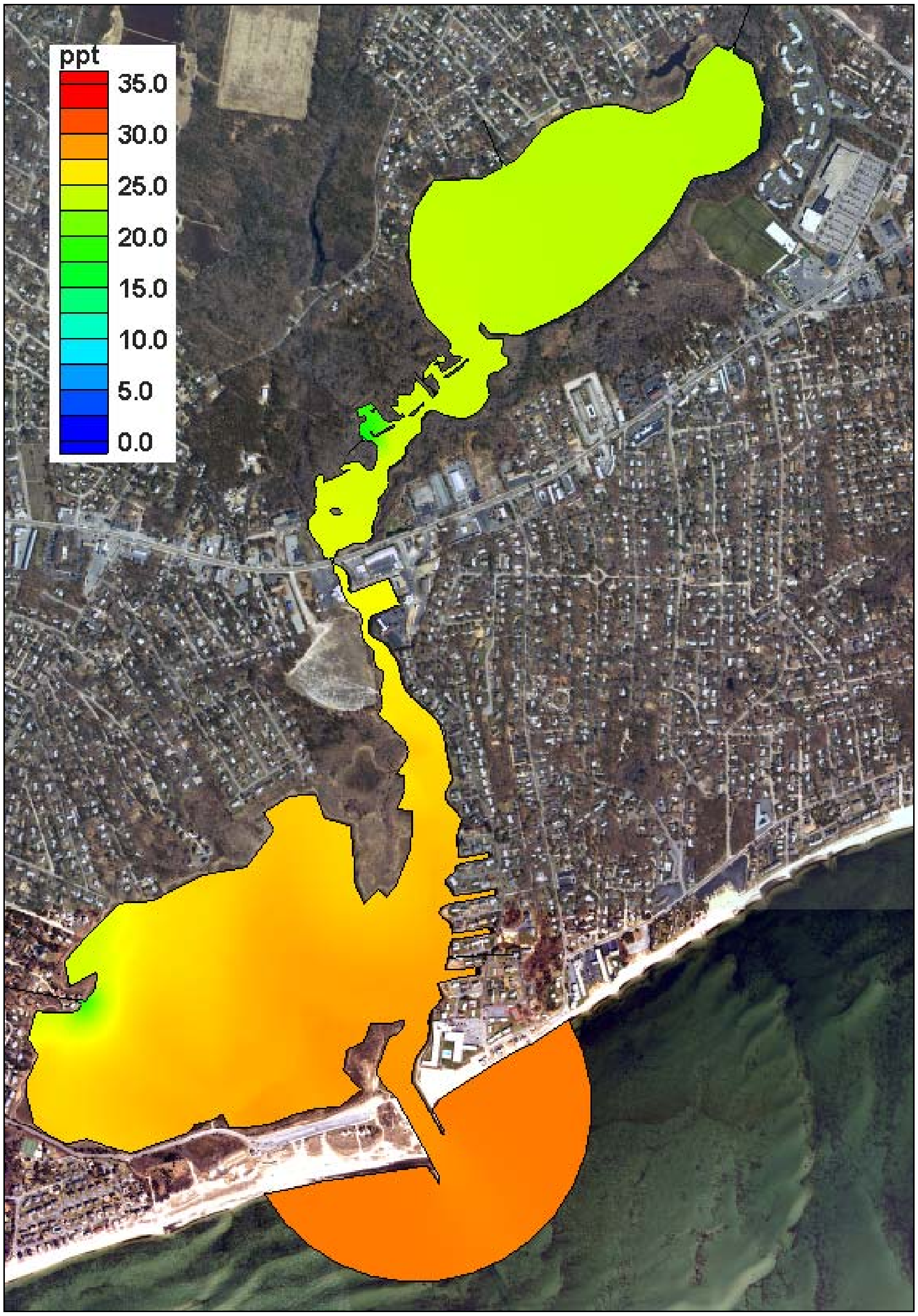


Figure VI-7. Contour Plot of average modeled salinity (ppt) in the Parkers River system.

VI.2.6.1 Build-Out

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

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 Parkers River system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.					
sub-embayment	present load (kg/day)	Build-out (kg/day)	build-out % change	no load (kg/day)	no load % change
Seine Pond	20.562	22.093	+7.4%	0.293	-98.6%
Upper Parkers River	16.408	33.351	+103.3%	0.926	-94.4%
Lower Parkers River	12.652	17.904	+41.5%	0.079	-99.4%
Lewis Pond	17.400	30.184	+73.5%	0.362	-97.9%
System Total	67.022	103.532	+54.5%	1.660	-97.5%

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. **Build-out** scenario sub-embayment and surface water loads used for total nitrogen modeling of the Parkers River system, with total watershed N loads, atmospheric N loads, and benthic flux.			
sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	22.093	1.096	-8.251
Upper Parkers River	33.351	0.049	1.024
Lower Parkers River	17.904	0.266	36.708
Lewis Pond	30.184	0.616	7.340
System Total	103.532	2.027	36.822

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of the system was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. For build-out, the increase in modeled TN concentrations is greatest in the Lewis Pond, where TN concentrations increase more than 59%. A contour plot showing average TN concentrations throughout the river system is presented in Figure VI-8 for the model of build-out loading.

Table VI-6. Comparison of model average total N concentrations from present loading and the **build-out scenario**, with percent change over background in Nantucket Sound (0.294 mg/L), for the Parkers River system.				
Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	build-out (mg/L)	% change
Lewis Pond	PR-4	0.859	1.197	+59.7%
Lower Parkers River	PR-3	0.491	0.576	+43.4%
upper Parkers River	PR-2	0.802	1.024	+43.8%
lower Seine Pond	PR-1	0.965	1.230	+39.5%
upper Seine Pond	PR-5	1.007	1.273	+37.4%

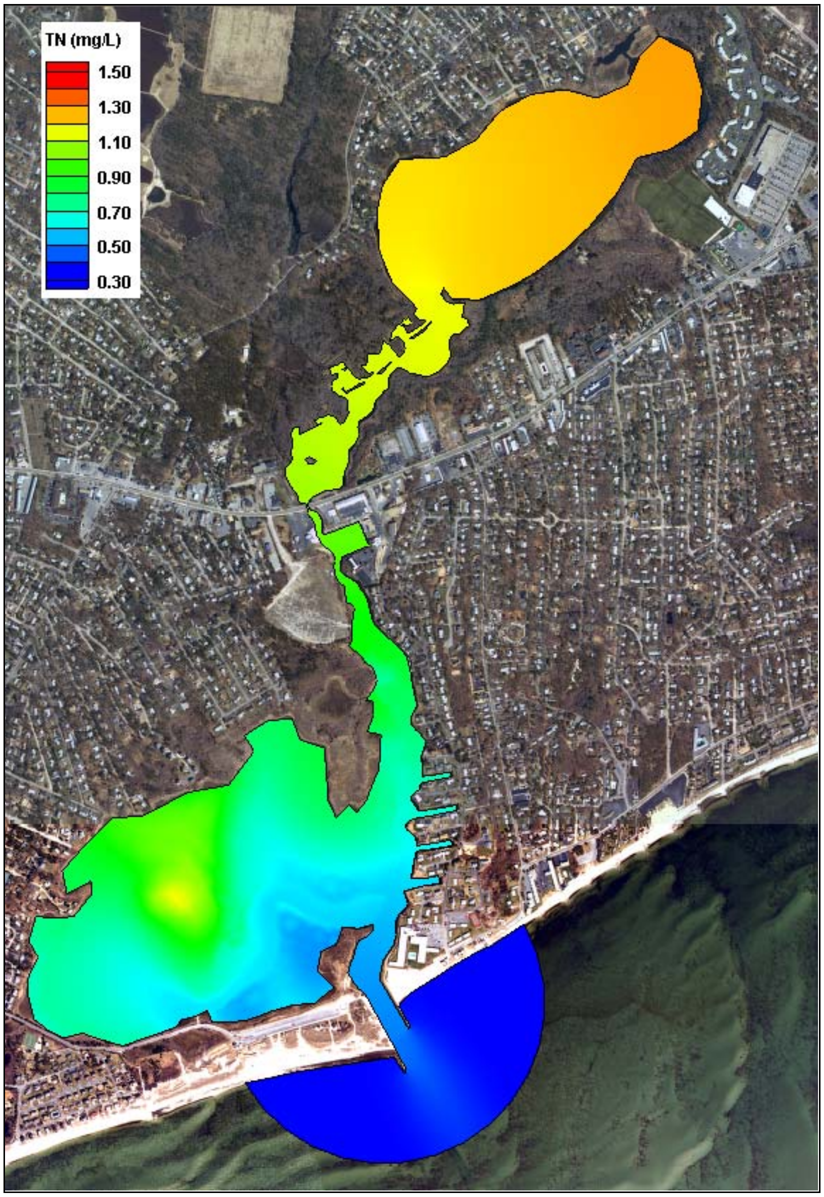


Figure VI-8. Contour plot of modeled total nitrogen concentrations (mg/L) in the Parkers River system, for projected build-out scenario loading conditions.

VI.2.6.2 No Anthropogenic Load

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

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	0.293	0.926	0.079
Upper Parkers River	1.096	0.049	0.266
Lower Parkers River	-1.438	0.328	13.583
Lewis Pond	-0.049	1.304	13.928
System Total	0.293	0.926	0.079

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations at each monitoring station. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was large, with all areas of the system experiencing reductions greater than 80%, compared to the background concentration of 0.294 in Nantucket Sound (Table VI-8). A contour plot showing TN concentrations throughout the system is shown pictorially in Figure VI-9.

Table VI-8. Comparison of model average total N concentrations from present loading and the "**No anthropogenic loading**" ("no load"), with percent change over background in Nantucket Sound (0.294 mg/L), for the Parkers River system.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	no-load (mg/L)	% change
Lewis Pond	PR-4	0.859	0.354	-89.3%
Lower Parkers River	PR-3	0.491	0.330	-81.6%
upper Parkers River	PR-2	0.802	0.370	-85.1%
lower Seine Pond	PR-1	0.965	0.377	-87.6%
upper Seine Pond	PR-5	1.007	0.379	-88.0%

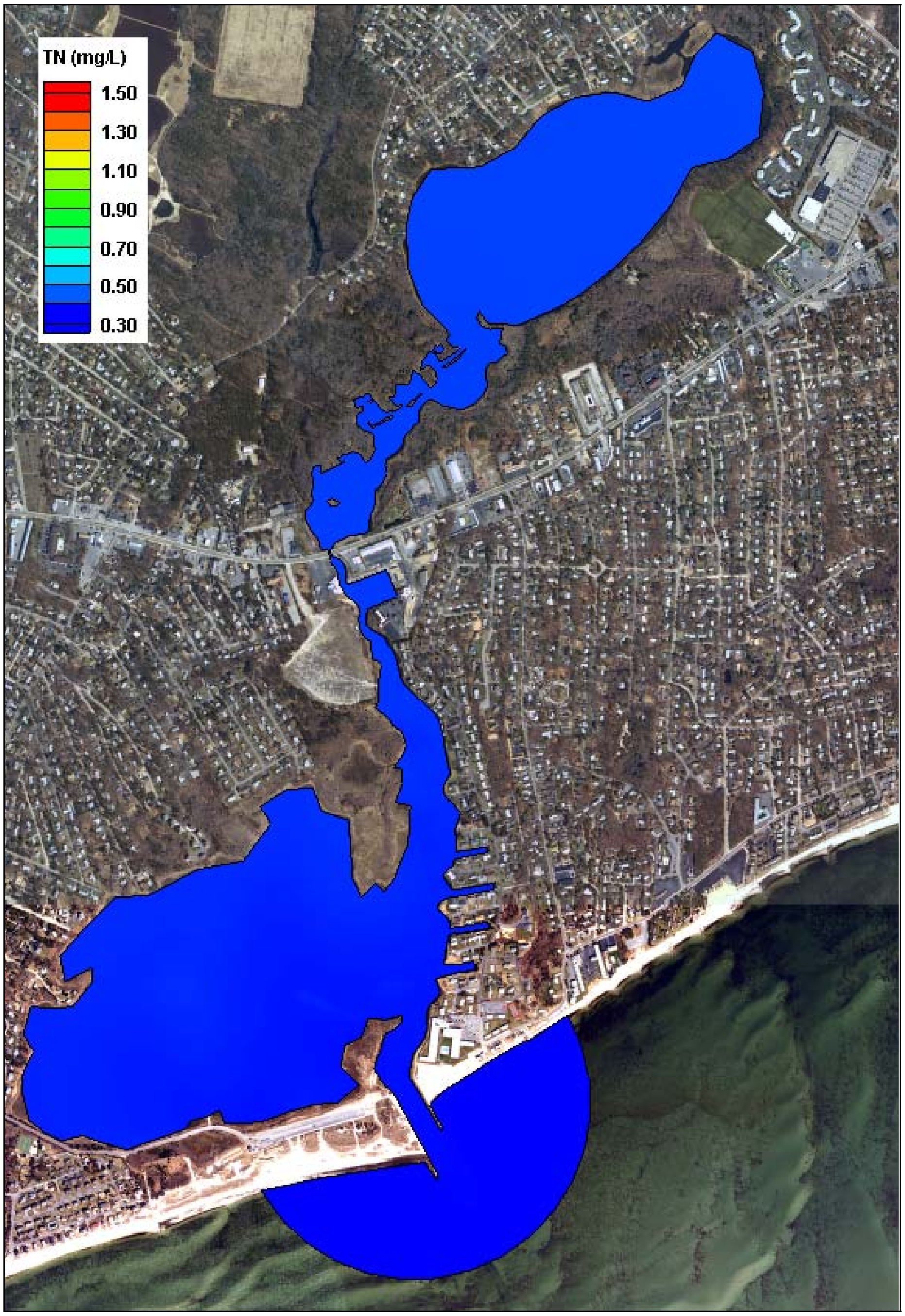


Figure VI-9. Contour plot of modeled total nitrogen concentrations (mg/L) in Parkers River, for no anthropogenic loading conditions.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gaged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Parkers River embayment system (inclusive of Seine Pond and Lewis Pond) in the Town of Yarmouth, MA, the MEP assessment is based upon data from the water quality monitoring database developed by the Town of Yarmouth Water Quality Monitoring Program and the MassDEP and SMAST surveys of eelgrass distribution, benthic animal communities, sediment characteristics, and dissolved oxygen records conducted during the summer and fall of 2004. These data were analyzed relative to recent changes within the watershed and have been used to form the basis of an assessment of this system's present nutrient related habitat quality. When coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, the full data set supports quantitative nitrogen threshold development for this system (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen thresholds determination, the MEP approach focuses on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within Seine Pond a salt pond comprising the uppermost basin of the Parkers River Estuary, as well as within the salt marsh dominated shallow basin of Lewis Pond. Mooring deployments were conducted to record the frequency and duration of low oxygen conditions during the critical summer period and also to collect supporting information on phytoplankton biomass as Chlorophyll-a.

The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within the Parkers River system was conducted for comparison to historic records as available (MassDEP Eelgrass Mapping Program, C. Costello). Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. However, it should be noted that certain systems like Lewis Pond, that are salt marsh dominated, generally do not support eelgrass for reasons related to ecosystem structure. Within the Parkers River system, temporal changes in eelgrass distribution provide a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing from inlet management; nitrogen management) in nutrient enrichment.

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

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, USEPA suggests that the chronic protective oxygen level to support growth of estuarine animals is 4.8 mg L^{-1}, with a limit for survival of juvenile and adult organisms of 2.3 mg L^{-1} (USEPA, 2000). Massachusetts State Water Quality Classification indicates that SA (high quality) waters maintain oxygen levels above 6 mg L^{-1}. The tidal waters of the Parkers River Embayment System are currently listed under this classification as SA. It should be noted that the classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (see Figure VII-1 for example). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Parkers River Embayment System (Figure VII-2). Measurements were made close to the sediment surface so as to quantify the oxygen environment affecting benthic animal communities. The sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Parkers River embayment system was collected during the summer of 2004.

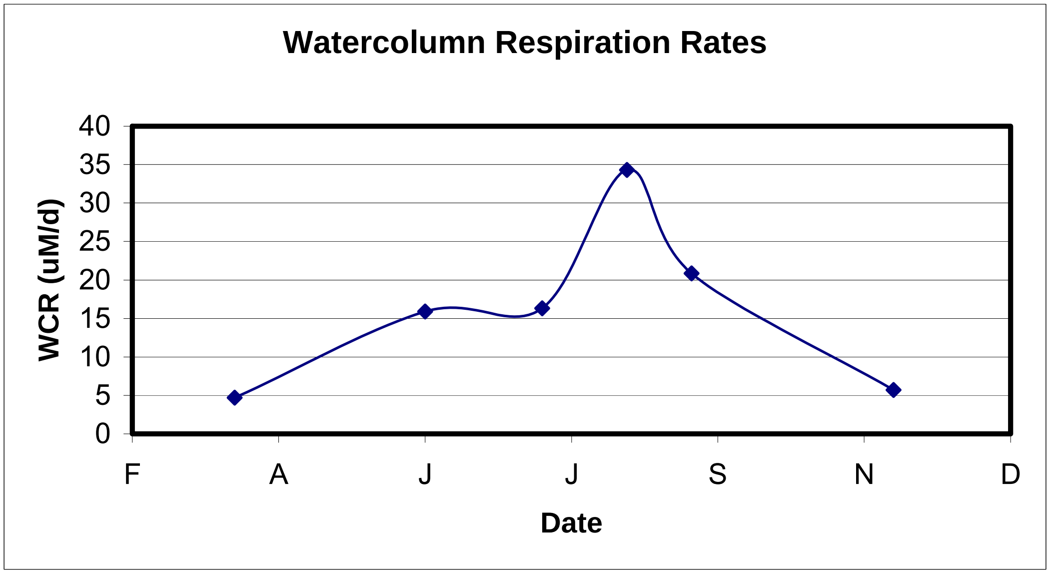


Figure VII-1. As an example, average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System are presented (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter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

Similar to other embayments in southeastern Massachusetts, the Parkers River Estuary evaluated in this MEP assessment showed high frequency variation related primarily to diurnal and sometimes tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. Oxygen excursions result from oxygen consumption (night) and production (day) primarily by phytoplankton within the estuarine waters. Additional oxygen uptake results from the microbial decay of organic matter, which in the case of the Parkers River Estuary is mainly from phytoplankton in the watercolumn and settling to bottom sediments. Oxygen levels in estuaries typically cannot be managed directly, but rather through management of nitrogen levels and mitigation of any direct organic matter inputs (e.g. outfalls).

The high degree of temporal variation in bottom water dissolved oxygen concentration at each mooring site underscores the need for continuous monitoring within these systems. However, the large number of oxygen samplings by the Yarmouth Water Quality Monitoring Program from 2002-2008 was sufficient to capture the minimum oxygen levels measured by the detailed time-series measurements. For example, in Seine Pond the continuous oxygen record and the monitoring program found minimum oxygen levels of 3.36 mg L^{1} and 3.6 mg L^{-1}, respectively and similarly in Lewis Pond found minimum oxygen levels of 2.48 mg L^{1} and 2.4 mg L^{-1}, respectively. The agreement between the time-series oxygen mooring and the monitoring program in Seine Pond and Lewis Pond indicates that the monitoring data for the other basins within this estuary can be used in the assessment of those areas.

Figure VII-2. Location of time-series oxygen mooring, deployed summer 2004, in the Parkers River Embayment System within the Town of Yarmouth.

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 51 day deployment period that these parameters were below or above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels within Seine Pond indicate high levels of nutrient enrichment and impaired habitat quality (Figures VII-3 and VII-5). The oxygen data are consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of this estuarine basin. The large daily excursions in oxygen concentration in both Seine Pond and Lewis Pond also indicate significant organic matter enrichment. While both Seine Pond and Lewis Pond share large daily excursions for dissolved oxygen, Lewis Pond generally shows a lower baseline dissolved oxygen concentration. However, the level of oxygen stress needs to be evaluated in light of the fact that Lewis Pond is a salt marsh tidal basin, while Seine Pond is a typical embayment basin. Salt marsh tidal basins are naturally organic matter enriched and typically have summertime low oxygen events (periodic hypoxia), while high quality embayment basins do not.

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

Mooring Location	Start Date	End Date	Total Deployme (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Seine Pond - Parker River	**8/1/2004**	**9/21/2004**	**51.1**	**19%**	**4%**	**1%**	**0%**
			Mean	0.23	0.16	0.09	N/A
			Min	0.02	0.03	0.02	0.00
			Max	0.72	0.44	0.18	0.00
			S.D.	0.20	0.14	0.06	N/A
Lewis Pond - Parker River	**8/1/2004**	**9/21/2004**	**51.1**	**47%**	**22%**	**5%**	**0%**
			Mean	0.55	0.22	0.10	0.03
			Min	0.02	0.02	0.01	0.01
			Max	1.95	0.90	0.32	0.05
			S.D.	0.52	0.20	0.08	0.03

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

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)
Seine Pond Parker River	**8/1/2004**	**9/21/2004**	**51.1**	**99%**	**99%**	**90%**	**70%**	**43%**
Mean Chl Value = 26.1 ug/L			Mean	50.58	25.27	1.92	0.87	0.50
			Min	50.58	24.04	0.04	0.04	0.04
			Max	50.58	26.50	13.38	7.96	3.63
			S.D.	N/A	1.74	3.47	1.74	0.79
Lewis Pond Parker River	**8/1/2004**	**9/21/2004**	**51.1**	N/A	N/A	N/A	N/A	N/A
Meter Malfunction			Mean	N/A	N/A	N/A	N/A	N/A
			Min	N/A	N/A	N/A	N/A	N/A
			Max	N/A	N/A	N/A	N/A	N/A
			S.D.	N/A	N/A	N/A	N/A	N/A

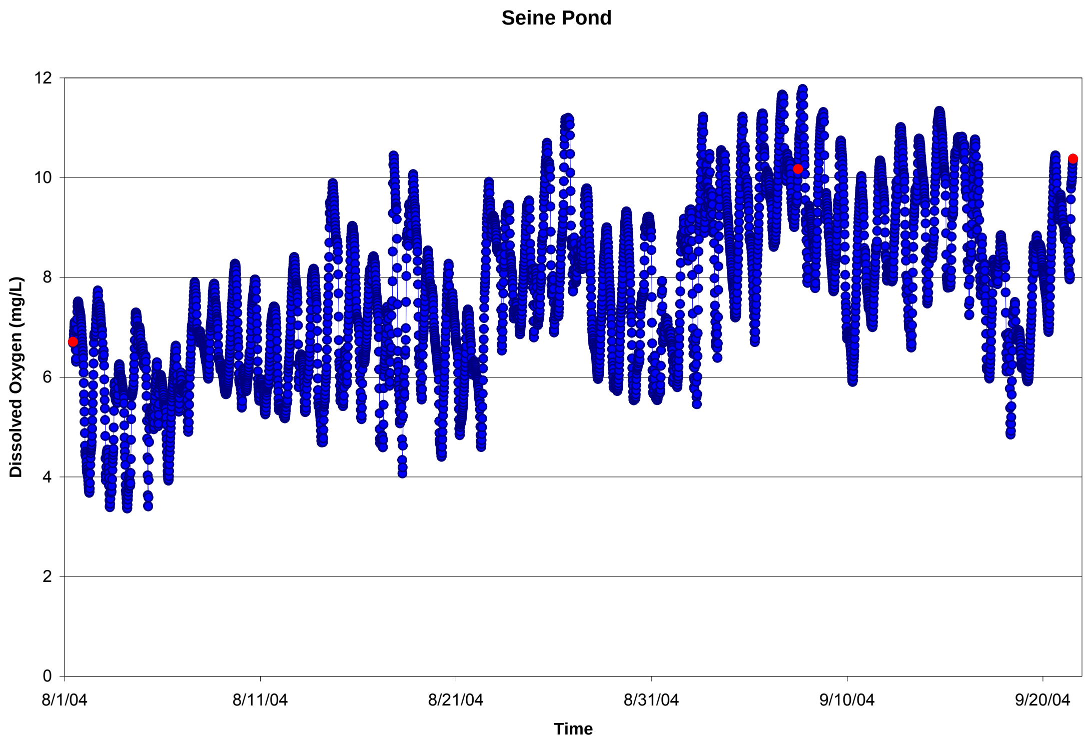


Figure VII-3. Bottom water record of dissolved oxygen at the Seine Pond - Parkers River station, Summer 2004. Calibration samples represented as red dots.

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

Generally, the dissolved oxygen records throughout the Parkers River Estuary showed moderate depletions (relative to the basin type) during the critical summer period. The greatest oxygen depletions were generally associated with the wetland dominated tributary basin of Lewis Pond, with higher oxygen levels maintained in the main embayment basin, Seine Pond. The continuous D.O. records indicate that the upper region of the Parkers River Embayment System, defined by the open water portion that is Seine Pond, shows regular oxygen depletion (below 6.0 mg/L) during summer with periodic depletions below 4.0 mg/L, consistent with its nitrogen and organic matter rich waters (Table VII-1, Figure VII-3). The Parkers River also shows oxygen depletions in the upper (3.6 mg L-1) and lower (4.4 mg L-1) reaches, as well as moderate to high chlorophyll levels. However, it is virtually certain that these enrichments stem primarily from the River's role in transporting high nutrient, high phytoplankton, low oxygen waters from Seine and Lewis Ponds to Nantucket Sound on the ebb tide. The high turnover of water in the Parkers River reduces its ability to build up nutrients. In addition, the inflow of high quality water from Nantucket Sound on the flooding tide, results in a basin with relatively high water quality for a portion of the flood tide period.

Oxygen levels in the Lewis Pond basin of the Parkers River Estuary were generally lower than those measured in Seine Pond despite being situated closer to the inlet to the overall system. The mooring deployment in Lewis Pond revealed very frequent oxygen depletions to <4 mg L^{-1} and periodically to <3 mg L^{-1}. However, it should be noted that these depletions occur within a salt marsh dominated shallow tidal basin, surrounded by extensive salt marsh. As such, Lewis Pond's oxygen depletion results primarily from the naturally organic matter and nutrient rich qualities of such an environment. However, it does appear from the high chlorophyll levels observed between 2002-2008 (average = 9.2 ug L-1) that there may be a moderate level of impairment to this basin from watershed nitrogen inputs. This assessment is also supported by the benthic animal community surveys noted in Section VII-4, below.

***Seine Pond – Parkers River* (Figures VII-3 and VII-5):**

The Seine Pond mooring location was centrally located within the basin of Seine Pond (Figure VII-2). There were large daily excursions in oxygen levels, ranging from levels in excess of air equilibration to periods of oxygen depletion. Oxygen varied primarily with light (diurnal cycle) and to a lesser extent with tides. Lowest oxygen was generally observed in the early morning. Highest dissolved oxygen was observed when low tide occurred at the end of the photocycle (ca. 1500 hrs). Oxygen levels frequently declined to <6 mg L^{-1} and even <5 mg L^{-1}, with 1% of the 51 day record showing values below 4 mg L^{-1} (Table VII-1). Equally indicative of eutrophic conditions, oxygen levels often climbed to above 8 mg L^{-1} and occasionally above 10 and 11 mg L^{-1}, consistent with the high phytoplankton biomass. Consistent with the oxygen data, chlorophyll a was very high over the entire study period (and in the 2002-08 monitoring period equaled 15 ug L^{-1}) exceeding the 25 ug L^{-1} benchmark 43% of the time (Table VII-2, Figure VII-5). Peak concentrations always occurred near low tide indicating in situ production as the source for the high chlorophyll a levels. Average chlorophyll levels over 10 ug L^{-1} have been used to indicate eutrophic conditions in embayments.

***Lewis Pond – Parkers River* (Figure VII-4):**

The Lewis Pond mooring location was centrally located within the basin of Lewis Pond (Figure VII-2). As in Seine Pond, high frequency data from Lewis Pond indicated large daily excursions, although levels did not typically exceed air equilibration. Instead, the oxygen excursions resulted mainly from oxygen uptake associated with the diurnal cycle, with some signs of tidal influence. Lowest dissolved oxygen was typically observed at high tide in the early morning. Highest dissolved oxygen was observed when low tide occurred at the end of the photocycle (ca. 1500 hrs). Dissolved oxygen frequently declined to <4 mg L^{-1} and periodically to <3 mg L-1 (Figure VII-4, Table VII-1). The occurrence of "excess" oxygen was not as pronounced as in Seine Pond, with levels only periodically exceeding 9 mg L^{-1}, consistent with the phytoplankton biomass. It was not possible to determine the degree to which chlorophyll a exceeded the 25 ug L^{-1} benchmark due to failure of the chlorophyll sensor on the mooring. However, the water quality monitoring data shows significantly lower chlorophyll levels in Lewis Pond (PR-4) of 9 ug L^{-1} compared to Seine Pond (PR-1, PR-5) of 12 - 15 ug L^{-1}.

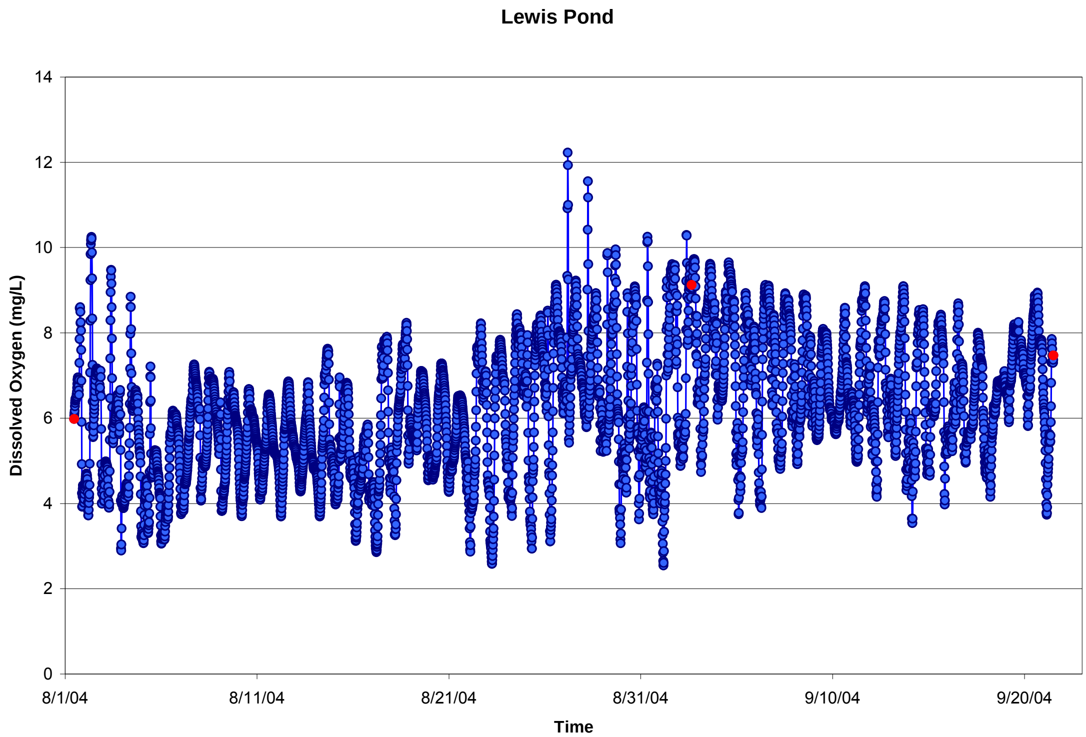


Figure VII-4. Bottom water record of dissolved oxygen at the Lewis Pond – Parkers River station, Summer 2004. Calibration samples represented as red dots.

The Lewis Pond portion of the Parkers River system is bordered by substantial areas of tidal salt marsh as well as more extensive salt marsh areas along the channel leading from the main stem of the Parker River to Lewis Pond. Salt marsh ponds, such as Lewis Pond, are by nature rich in organic matter and show periodic hypoxia in summer.

The assessment that low dissolved oxygen may be driven by the fact that Lewis Pond is functioning primarily as a shallow tidal salt pond is supported by the pattern of oxygen decline. Within Lewis Pond, the oxygen excursions were dominated by oxygen uptake, while in Seine Pond the excursions resulted from a combination of oxygen production and uptake.

The organic matter enriched sediments of salt marsh tidal creeks and basins, where the organic matter enriched sediments support high levels of oxygen uptake at night, typically show oxygen depletions. While oxygen depletion to 4 mg/L would indicate impairment in an embayment like the main basin of adjacent Lewis Bay or in Seine Pond, it is consistent with the organically enriched nature of tidal creeks. These observations are typical of other salt marsh dominated estuarine basins assessed by the MEP, for example the lower basin of Namskaket Marsh, a healthy salt marsh in Orleans, showed a nearly identical pattern of dissolved oxygen both in the level of the oxygen excursion and the extent of oxygen depletion. Similarly, Mill Creek within Lewis Bay (Barnstable and Yarmouth, MA), showed similar periodic oxygen depletions to 4 mg L^{-1}, but is functioning as a healthy yet nutrient rich salt marsh system. Given the significant salt marsh areas in the Lewis Pond portion of the Parkers River embayment system, the observed oxygen levels and the characteristics of the benthic community described in Section VII-4, it appears that this reach of the overall system is only moderately impaired due to its high chlorophyll levels and resident benthic animal community.

Overall, the oxygen and chlorophyll data for the Parkers River Estuary clearly indicate a system supporting sub-tidal habitats impaired by nitrogen, ranging from significantly impaired (Seine Pond) to moderately impaired (Lower River and Lewis Pond). These observations are consistent with the high levels of total nitrogen (TN) throughout the estuary. The gradient in impairment follows the gradient in nitrogen enrichment, where Seine Pond has very high ebb tide TN levels (>0.94 mg L^{-1}) declining to the Lower River nearest the tidal inlet (0.66 mg L^{-1}). While the lower portion of Parkers River supports lowest nitrogen levels within the system, the levels are still quite high and suggest a basin incapable of supporting eelgrass beds and with a moderate level of impairment to benthic animal habitat (see Sections VII-3 & VII-4, below). Lewis Pond also supports a high TN level (0.86 mg L^{-1}), but this is consistent with its function as a salt marsh basin.

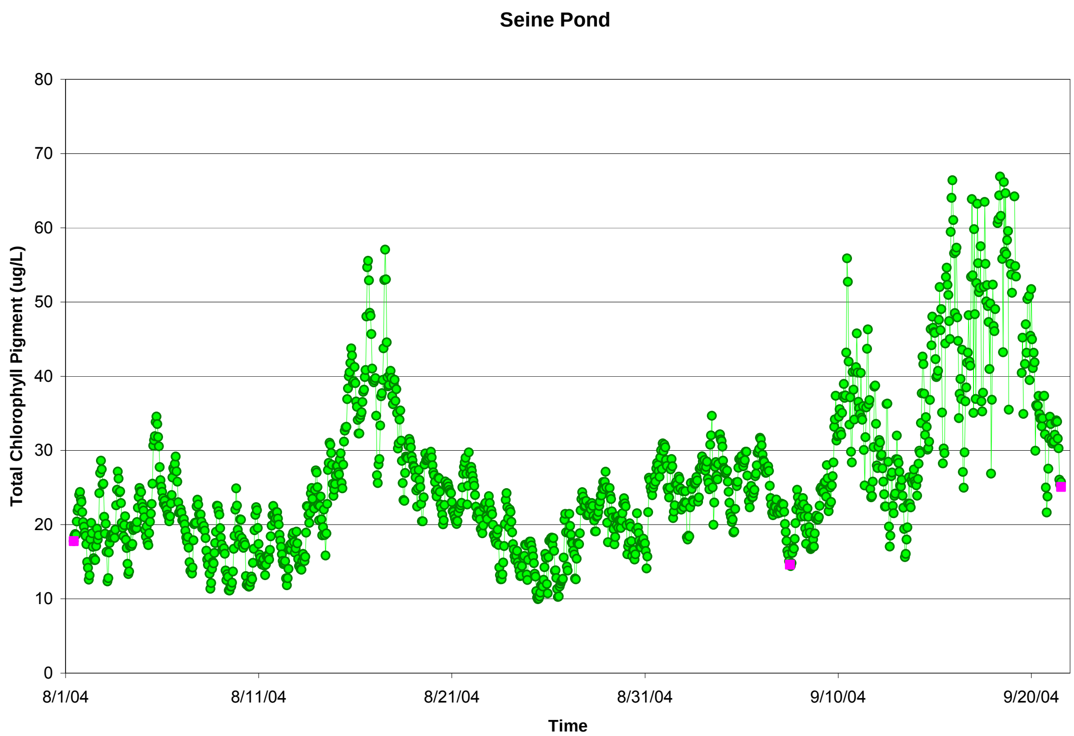


Figure VII-5. Bottom water record of Chlorophyll-*a* in Seine Pond - Parkers River, Summer 2004. Calibration samples represented as red dots.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data are a key part of the MEP Approach. Surveys were conducted in the Parkers River Estuary, particularly within the main tidal channels (Parkers River and channel to Lewis Pond) by the MassDEP Eelgrass Mapping Program (C. Costello). The most recent survey was conducted in 2001, as part of the MEP program with an earlier survey conducted in 1995. Additional analysis of available aerial photographs from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. The 1951 data were validated through discussion with the Town of Yarmouth, including individuals with long-term on-site knowledge of this system, particularly Lewis Pond and the main tidal channel. The 2001 map was field validated by the MassDEP Eelgrass Mapping Program. The primary use of the data is to indicate (a) estuarine regions that have historically or presently support eelgrass habitat, and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figures VII-6); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

While there were no eelgrass beds within the Parkers River Estuary in the 1995 and 2001 surveys, the MEP Technical Team confirmed both the lack of eelgrass in the tidal creeks to Seine Pond and Lewis Pond, as well as the main channel of the Parkers River system while undertaking field surveys in 2004-05. The eelgrass survey conducted by the MEP Technical Team was undertaken as part of the benthic regeneration and benthic animal surveys and during the deployment and recovery of the instrument moorings. The 1951 assessment indicated beds within the lower reach of the Parkers River. The 1951 analysis was based upon high quality aerial photos. To validate the eelgrass distribution of 1951, the Yarmouth Natural Resources Department was asked about historic eelgrass beds within the Parkers River Estuary (without access to the MassDEP map). First hand accounts indicated that eelgrass was not present Lewis Pond, but beds had been observed in patches within the lower Parkers River. These first hand accounts dated back to the 1940's and 1950's and matched with the independent assessment by MassDEP (Figure VII-6). In contrast to basins within the Parkers River Estuary, eelgrass is present offshore of the inlet as well as offshore of nearby Lewis Bay and Swan Pond River in the Town of Dennis. Based on the 2001 eelgrass survey conducted by the DEP Eelgrass Mapping Program offshore, there was evidence of a potential decline in the coverage of the offshore beds between 1995 and 2001 (Table VII-3). However, it is not possible at this time to determine if this represents an anthropogenically driven decline or natural variation at this site. Additional temporal sampling is planned to address this issue.

Overall, the historical distribution of eelgrass within the Parkers River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. Shallow salt marsh basins, like Lewis Pond typically do not support eelgrass beds. Similarly, the tidally restricted upper estuary (upper River and Seine Pond) has likely been nutrient enriched with poor water clarity over many decades. In contrast, at lower overall nitrogen loading, it would be expected that the lower River areas would have sufficient water clarity and oxygen levels to support eelgrass beds. However, the current absence of eelgrass within this system is expected given the high nitrogen levels and high chlorophyll levels in all basins. Typically eelgrass beds exist at much lower nitrogen levels (0.35 - 0.45 mg N L^{-1}) than presently found in this system (0.66 - 0.99 mg N L^{-1}). The high nitrogen levels within the Parkers River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates, which increases the nitrogen levels in the incoming tidal waters (0.3 mg L^{-1}) by several fold (see Section VI).

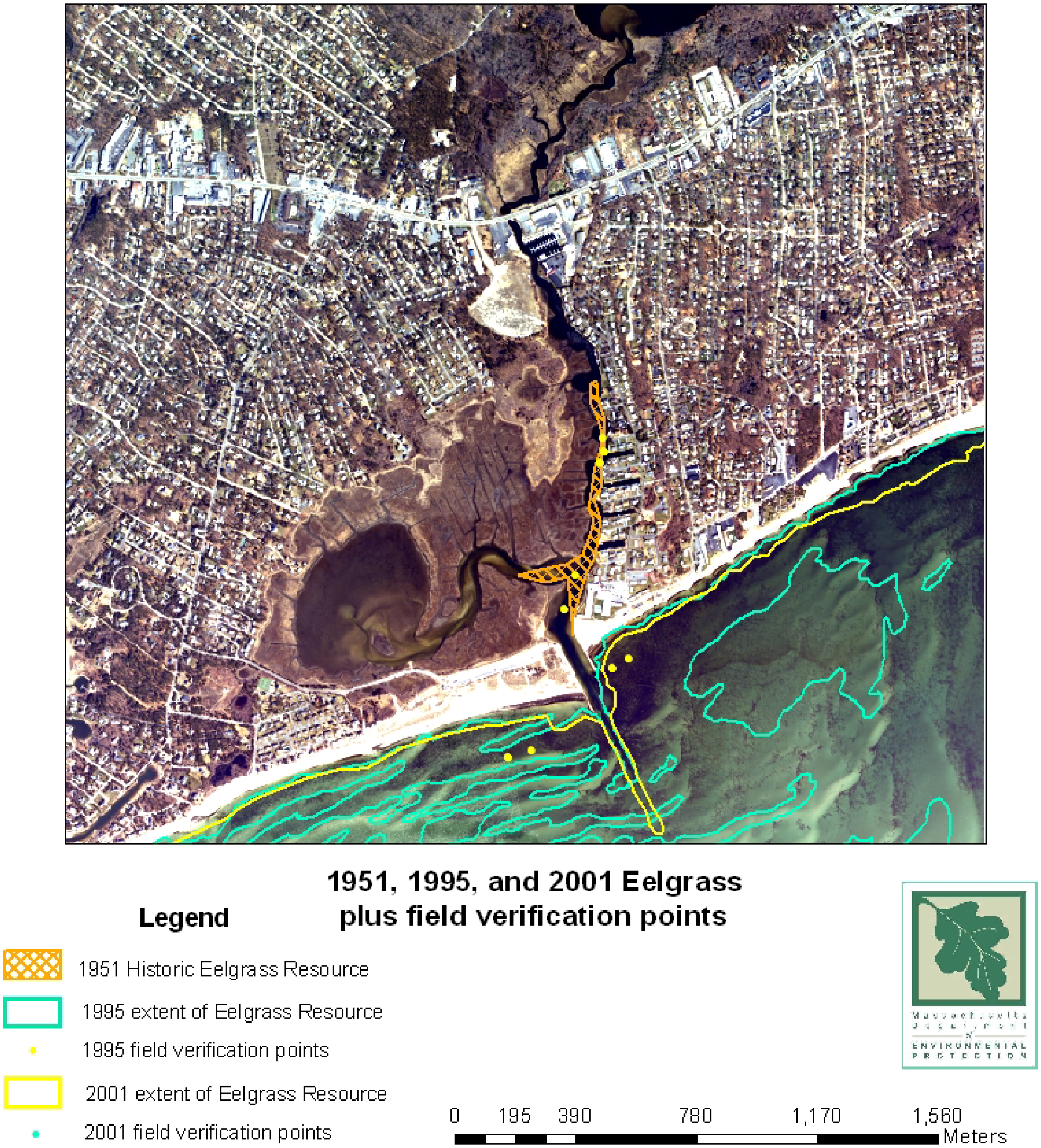


Figure VII-6. Eelgrass bed distribution associated with the Parkers River Embayment System in 1951, 1995, 2001, as determined by the MassDEP Eelgrass Mapping Program (map courtesy of C. Costello). The green and yellow outlines circumscribe eelgrass beds as mapped in 1995 and 2001, respectively. The 1951 was determined from aerial photography and validated by descriptions provided by direct observers. Presently, there are no eelgrass beds within the Parkers River system.

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

EMBAYMENT	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Parkers River	6.48	0.00	0.00	100%

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 10 locations throughout the Parkers River Embayment System (Figure VII-7). At many of the 10 locations sampled, multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stress conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, the Parkers River Estuary is clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information. The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. The highest quality habitat areas, as shown by the oxygen and chlorophyll records 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.

The infauna survey clearly indicated impaired habitat within Seine Pond and to a lesser extent within the lower reach of the Parkers River and Lewis Pond (Table VII-4). Seine Pond is currently supporting very poor benthic habitat throughout the basin. There are very few species and very few individuals (i.e. low secondary production), with an average of 6 species and 48 individuals per sample. These numbers are very low compared to other high quality areas in Cape Cod estuaries. For example in the adjacent Lewis Bay Estuary, the outer stations support several hundred individuals per grab distributed among 32 species. In addition, the community is composed of a variety of polychaete, crustacean and mollusk species, with high diversity and evenness. In contrast, Seine Pond has very low diversity (1.25) and Evenness <1 (0.65) and is dominated by stress indicator species associated with organic matter enrichment. These latter indicators are a small polychaete found in disturbed areas or areas with macroalgal accumulations, *Podarke obscura*, and the amphipod, *Ampelisca abdita*, generally associated with organic rich basins.

Figure VII-7. Aerial photograph of the Parkers River system (inclusive of Seine Pond and Lewis Pond) showing locations of benthic infaunal sampling stations (green symbols). Station numbers relate to those in Table VII-4.

Table VII-4. Benthic infaunal animal community data for the Parkers River embayment system (inclusive of Seine Pond and Lewis Pond). Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (samples represent surface area of 0.0625 m2). Stations refer to map in Figure VII-8, (N) is the number of samples per site.

	Total Actual # Species	Total Actual # Individuals	Species Calculated @ 75 Indiv.	Weiner Diversity (H')	Evenness (E)
Seine Pond - Main Basin					
Average	6	48	--[1]	1.25	0.65
s.d.	4	43	--[1]	0.92	0.21
N	10	10	--[1]	10	8
Lewis Pond - Main Basin					
Average	25	313	15	2.79	0.61
s.d.	8	163	2	0.18	0.03
N	3	3	3	3	3
Parker's River - Lower Reach					
Average	27	2433	15	2.94	0.61
s.d.	7	3156	10	1.59	0.29
N	2	2	2	2	2
1- too few individuals extant in field sample to support this calculation.					

In contrast to Seine Pond, the lower reach of Parkers River currently supports higher species numbers (27 species) and population levels (2433 individuals), more in line with a high quality benthic animal habitat. However, the Diversity (2.94) and Evenness (0.61) indices suggest a moderate level of impairment. A moderate level of impairment is also indicated by the types of species present, as the high numbers are due primarily to dense amphipod mats, although some areas had significant numbers of tubificids (stress indicators). Amphipod mats are typical of transitional environments and were the major communities to develop in Boston Harbor as nutrient loads began to diminish. These animal communities are consistent with the level of nitrogen enrichment and moderate chlorophyll levels and oxygen conditions within this estuarine basin. All of these parameters indicate a system that is supporting moderately impaired benthic habitat.

Lewis Pond showed infaunal communities consistent with a salt marsh basin with relatively high quality benthic habitat. The basin supports high numbers of species (25) and individuals (313) and moderate diversity (2.79) and evenness (0.61). As expected from its marsh setting, the community consists mainly of species indicative of an organic rich environment. The community was "patchy" with areas dominated by the stress indicator, tubificoides, although other areas were typical of salt marsh basins with high quality habitat. The observed benthic communities do appear to be consistent with the nutrient rich nature of the basin and the high chlorophyll levels, which may be adding additional stress to this basin. Given the patchiness of the community, the dominance of tubificoides in some samples and the high chlorophyll levels, it appears the Lewis Pond basin is presently supporting a high quality to moderately impaired habitat for benthic animals.

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

The results of the infauna survey supports that nitrogen management threshold analysis (Chapter VIII) needs to include a lowering of the level of nitrogen enrichment in Seine Pond and the Parkers River and potentially in Lewis Pond for restoration of nitrogen impaired benthic habitats.

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 (see Section II). The Massachusetts Division of Marine Fisheries has an extensive library of shellfish resources maps which indicate the current status of shellfish areas closed to harvest as well as the suitability of a system for the propagation of shellfish. As is the case with many systems on Cape Cod, a large portion of the Parkers River system is conditionally approved for the taking of shellfish during specific times during the year, typically the cold winter months, indicating the system is generally supportive of shellfish communities. However, in the upper most reaches of Seine Pond as well as a small section of the main tidal channel of the Parkers River system up gradient and down gradient of Route 28, harvest of shellfish is prohibited year round indicating the presence of a persistent environmental contaminant. In the case of the Seine Pond closure, that is likely due to bacterial contamination whereas in the lower portion of the system, closure is likely related to the required management closure of active marina basins. The major shellfish species with potential habitat within the Parkers River Estuary are soft shell clams (*Mya*) within Seine Pond and the upper reach of the Parkers River (and marginal areas of Lewis Pond) and quahogs (*Mercenaria*) within Lewis Pond (Figure VII-8). In addition, if habitat conditions improve there is also the potential for small grow areas for blue mussels.

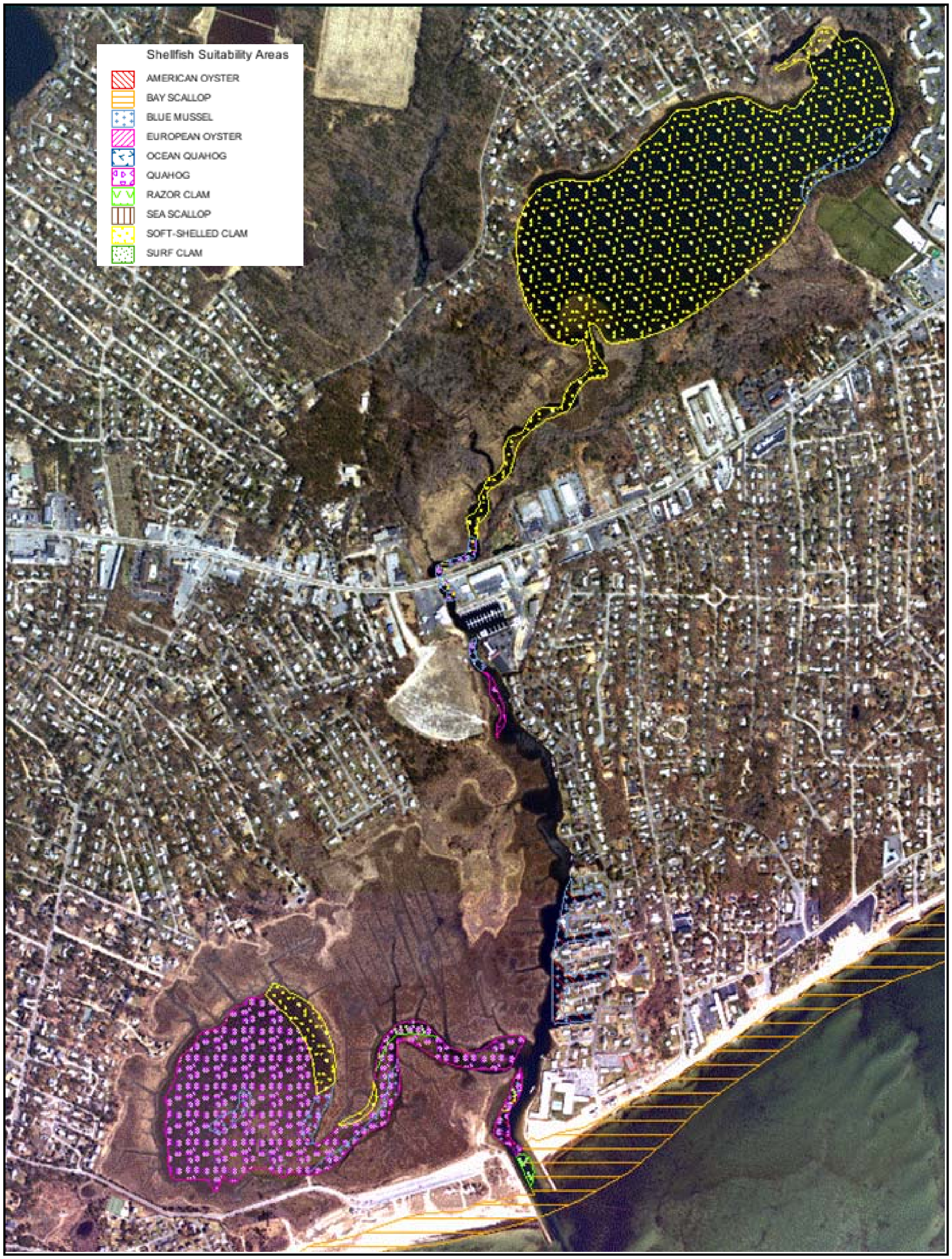


Figure VII-8. Potential shellfish growing areas within the Parkers River system, Yarmouth, MA. Primary species with potential suitable habitat are soft shell clams and quahogs. Source: Mass GIS.

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

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

The Parkers River Embayment System is a complex estuary composed of 2 functional types of component basins: embayments (Seine Pond, Parkers River) and a salt marsh pond/embayment (Lewis Pond). Each of these 2 functional components has different natural sensitivities to nitrogen enrichment and organic matter loading. Evaluation of eelgrass and infaunal habitat quality must consider the natural structure of each system and the ability to support eelgrass beds and the types of infaunal communities that they support. At present, the Parkers River is showing differences in nitrogen enrichment and habitat quality among its various component basins (Table VIII-1).

Overall, the system is showing some nitrogen related habitat impairment within each of its component basins, however, there is a strong gradient throughout this system. Seine Pond is a significantly impaired basin relative to infauna habitat, since it historically has not supported eelgrass. Nitrogen enrichment (resulting from inputs and restricted tidal exchange) has resulted in frequent large phytoplankton bloom, periodic hypoxia, macroalgal accumulations and virtual loss of benthic communities. The Parkers River is also nitrogen enriched, but has less nitrogen enrichment due primarily on its structure and high water turnover. While the lower reach currently supports only moderately impaired benthic habitat, its loss of historical eelgrass coverage indicates that it has become a significantly impaired basin relative to eelgrass habitat. Finally, Lewis Pond is a small shallow tidal basin surrounded by extensive tidal salt marsh and as such has not historically supported eelgrass. Lewis Pond currently functions as a salt marsh dominated basin with naturally high organic matter inputs and periodic low oxygen. As a salt marsh basin it is currently supporting moderately productive diverse infaunal communities. However, based upon the high chlorophyll levels and some of the infaunal indicators, currently it appears to be moderately impaired as benthic habitat. Overall, the regions of significant and moderate habitat impairment comprise >90% of the estuarine area of the Parkers River Embayment System.

Eelgrass: The absence of eelgrass throughout the Parkers River Estuary is consistent with the observed nitrogen and the chlorophyll levels and functional basin types comprising this estuary. The lower Parkers River basin supported eelgrass beds in 1951 under lower nitrogen loading conditions.

Table VIII-1. Summary of nutrient related habitat quality within the Parkers River Estuary within the Town of Yarmouth, MA, based upon assessments in Section VII. Seine Pond is a coastal salt pond connected to Nantucket Sound by the Parkers River, both of these estuarine components are typical embayment basins. In contrast, Lewis Pond is primarily a shallow tidal basin, surrounded by extensive salt marsh. YWQMP is the Yarmouth Water Quality Monitoring Program.

	Parkers River Embayment System			
	Parkers River		Seine Pond	Lewis Pond
Health Indicator	**Upper**	**Lower**		
Dissolved Oxygen	MI-SI[2]	MI[3]	MI-SI[4]	H-MI[1]
Chlorophyll	SI[5]	MI[6]	SD[7]	MI[8]
Macroalgae	--[9]	MI[10]	SI[11]	H[12]
Eelgrass	--[13]	SI[14]	--[13]	--[13]
Infaunal Animals	MI-SI[15]	MI[16]	SI-SD[17]	H-MI[18]
Overall:	**MI-SI[19]**	**SI[20]**	**SI[21]**	**MI[22]**

1 – primarily a salt marsh pond, frequent oxygen depletion to ≤4 mg/L, periodically to 3 mg/L; basin surrounded by extensive tidal saltmarsh resulting in natural organic enrichment.
2 – oxygen levels dominated by ebbing Seine Pond waters, minimum =3.6 mg/L (YWQMP, 2002-08)
3 – oxygen levels periodically depleted, water quality monitoring minimum (2002-08)=4.4 mg/L
4 -- oxygen depletions frequently <6 mg/L, infrequently to ≤4 mg/L, minimum=3.4 mg/L; YWQMP minimum D.O. (2002-08) = 3.6 mg/L & 1.9 mg/L at "deep" site
5 – moderate to high summer chlorophyll levels averaging 8 ug/L (YWQMP, 2002-2008)
6 – low to moderate summer chlorophyll levels averaging 4 ug/L (YWQMP, 2002-2008)
7 – very high chlorophyll, average 2004 = 26ug/L, 2002-08 = 12-15ug/L, frequent blooms to >40ug/L
8 – high chlorophyll levels, YWQMP average 2002-08 = 9 ug/L
9 -- drift algae sparse or absent, little surface microphyte mat.
10 -- patches of drift algae, *Ulva*, with some filamentous species and some algal mat.
11 -- dense patches of drift algae, *Ulva*, with some filamentous species mostly in the basin's lower half.
12 -- drift algae sparse or absent, small patches of SAV, *Ruppia* (common to salt marsh ponds)
13 – no evidence this basin is supportive of eelgrass.
14 -- MassDEP (C. Costello) indicates that eelgrass lost from this system between 1951-1995.
15 -- assessment based upon mouth of Seine Pond samples showing low diversity, eveness, low total numbers of species and individuals, the upper River is presently dominated by Seine Pond outflow.
16 -- high numbers of species and high number of individuals, dense amphipod mats indicative of disturbance and/or moderate levels of organic enrichment.
17 -- low numbers of species & individuals, low diversity & Eveness, dominated by organic enrichment and stress tolerant opportunistic species
18 -- Infauna: moderate numbers of individuals, high/moderate species, high diversity and Eveness; some organic enrichment indicators typical of salt marsh ponds and some deep burrowers, but dominated by opportunistic species indicative of organic matter overloading.
19 -- Moderate to Significant Impairment , primarily due to sustained high chlorophyll levels & periodic D.O. depletion. Dominated by outflows of low D.O., high organic matter waters from Seine Pond.
20 -- Significant Impairment based upon loss of eelgrass from system, 1951-1995.
21 -- Significant Impairment based primarily on the high sustained chlorophyll levels, periodic oxygen depletions and the depaupate benthic community dominated by stress indicator species.
22 -- Moderate Impairment based upon the elevated chlorophyll and infaunal community structure, particularly the dominance by tubificids.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation; -- = not applicable to this estuarine reach

The historical distribution of eelgrass and its present absence within the Parkers River Estuary is consistent with both the natural history of eelgrass and the present nitrogen, oxygen and chlorophyll levels within the different component basins. Shallow salt marsh basins, like Lewis Pond typically do not support eelgrass beds. Similarly, the tidally restricted upper estuary (upper River and Seine Pond) has likely been nutrient enriched with poor water clarity over many decades. In contrast, at lower overall nitrogen loading, it would be expected that the lower River areas would have sufficient water clarity and oxygen levels to support eelgrass beds. However, given the sensitivity of eelgrass to declining light penetration resulting from nutrient enrichment and secondary effects of organic enrichment and oxygen depletion, the current absence of eelgrass within this system is expected given the high nitrogen levels and high chlorophyll levels measured in all basins. Typically eelgrass beds exist at much lower nitrogen levels (0.35 - 0.45 mg N L^{-1}) than presently found in this system (0.66 - 0.99 mg N L^{-1}). The high nitrogen levels within the Parkers River Estuary indicate a high level of watershed nitrogen loading relative to the present tidal flushing rates, which increases the nitrogen levels in the incoming tidal waters (0.3 mg L^{-1}) by several fold (see Section VI). As there is no evidence of eelgrass coverage within Seine Pond, the upper reach of the Parkers River or Lewis Pond within the past 5 decades, they should not be considered for eelgrass restoration. In contrast, documented eelgrass within the lower Parkers River makes restoration of this resource a primary target for overall restoration of the Parkers River Embayment System. Restoration of this habitat will require appropriate nitrogen management. Note that restoration of this eelgrass habitat will necessarily result in restoration of other resources throughout the system, particularly within the upper estuary. Seine Pond and the upper Parkers River are the discharge basins for much of the watershed nitrogen load to this estuary. The lower reach of the Parkers River is the channel through which the nitrogen and organic matter enriched waters from the upper estuary is flushed out on ebbing tides. Nitrogen management focused on lowering nitrogen levels within the lower River will require lowering of nitrogen levels throughout the upper estuary. Therefore an improvement of infaunal habitats within these upper basins will result in part from improving eelgrass habitat in the lower River.

Based upon the above analysis, eelgrass habitat was selected as the primary nitrogen management goal for the lower reach of the Parkers River and infaunal habitat quality the management target for Seine Pond and Lewis Pond. These goals are the focus of the MEP management alternatives analysis presented in Chapter IX.

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

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels within Seine Pond indicate high levels of nutrient enrichment and impaired habitat quality. The oxygen data are consistent with high organic matter loads from phytoplankton production (chlorophyll-a levels) indicative of nitrogen enrichment and eutrophication of this estuarine basin. The large daily excursions in oxygen concentration in both Seine Pond and Lewis Pond also indicate significant organic matter enrichment. While both Seine Pond and Lewis Pond share large daily excursions for dissolved oxygen, Lewis Pond generally shows a lower baseline dissolved oxygen concentration. However, the level of oxygen stress needs to be evaluated in light of the fact that Lewis Pond is a salt marsh tidal basin, while Seine Pond is a typical embayment basin. Salt marsh tidal basins are naturally organic matter enriched and typically have summertime low oxygen events (periodic hypoxia), while high quality embayment basins do not.

Generally, the dissolved oxygen records throughout the Parkers River Estuary showed moderate depletions (relative to the basin type) during the critical summer period. The greatest oxygen depletions were generally associated with the wetland dominated tributary basin of Lewis Pond, with higher oxygen levels maintained in the main embayment basin, Seine Pond. D.O. records indicate that the upper region of the Parkers River Embayment System, defined by the open water portion that is Seine Pond, shows regular oxygen depletion (below 6.0 mg/L) during summer with periodic depletions below 4.0 mg/L, consistent with its nitrogen and organic matter rich waters. The Parkers River also shows oxygen depletions in the upper (3.6 mg L-1) and lower (4.4 mg L-1) reaches, as well as moderate to high chlorophyll levels. However, it is virtually certain that these enrichments stem primarily from the River's role in transporting high nutrient, high phytoplankton, low oxygen waters from Seine and Lewis Ponds to Nantucket Sound on the ebb tide. The high turnover of water in the Parkers River reduces its ability to build up nutrients. In addition, the inflow of high quality water from Nantucket Sound on the flooding tide, results in a relatively high water quality basin for a portion of the flood tide period.

Oxygen levels in the Lewis Pond basin of the Parkers River Estuary were generally lower than those measured in Seine Pond despite being situated closer to the inlet to the overall system. Lewis Pond had very frequent oxygen depletions to <4 mg L^{-1} and periodically to <3 mg L^{-1}. However, it should be noted that these depletions occur within a salt marsh dominated shallow tidal basin, surrounded by extensive salt marsh. As such, Lewis Pond's oxygen depletion results primarily from the naturally organic matter and nutrient rich qualities of such an environment. However, it does appear from the high chlorophyll levels observed between 2002 and 2008 (average = 9.2 ug L-1) that there may be a moderate level of impairment to this basin from watershed nitrogen inputs. This assessment is supported by the benthic animal community surveys noted in Section VII-4.

Overall, the pattern of high nitrogen, resulting in high phytoplankton biomass and periodic low oxygen depletion, was found throughout this system. The loss of eelgrass within the lower reach of the Parkers River is consistent with the observed water quality conditions. Similarly, the virtual loss of infaunal habitat within Seine Pond and moderate impairments within the Parkers River and Lewis Pond also reflect nitrogen enrichment. Management of nitrogen levels through reductions in watershed nitrogen inputs or increased tidal flushing are required for restoration of eelgrass and infaunal habitats within the Parkers River Embayment System.

Infaunal Communities: In all areas and particularly those that do not support eelgrass beds, benthic animal indicators are used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The Infauna Survey clearly indicated impaired habitat within Seine Pond and to a lesser extent within the lower reach of the Parkers River and Lewis Pond. Seine Pond is currently supporting very poor benthic habitat throughout the basin. There are very few species and very few individuals (i.e. low secondary production), with an average of 6 species and 48 individuals per sample. These numbers are very low compared to other high quality areas in Cape Cod estuaries. In addition, Seine Pond benthic communities have very low Diversity (1.25) and Eveness <1 (0.65) and are dominated by stress indicator species associated with organic enrichment. These latter indicators are a small polychaete found in disturbed areas or areas with macroalgal accumulations, *Podarke obscura*, and the amphipod, *Ampelisca abdita*, generally associated with organic rich basins.

In contrast to Seine Pond, the Parkers River lower reach currently supports species numbers and population levels more in line with a high quality benthic animal habitat. Although the Diversity (2.94) and Eveness (0.61) indices suggest a moderate level of impairment. As do the types of species present (e.g. dense amphipod mats and tubificids). Amphipod mats are typical of transitional environments and were the major communities to develop in Boston Harbor as nutrient loads began to diminish. These animal communities are consistent with the level of nitrogen enrichment and moderate chlorophyll levels and oxygen conditions within this estuarine basin. All of these parameters indicate a system that is supporting moderately impaired benthic habitat.

Lewis Pond showed infaunal communities consistent with a salt marsh basin with relatively high quality benthic habitat. The basin supports high numbers of species and individuals and moderate Diversity (2.79) and Eveness (0.61). As expected from its marsh setting, the community consists mainly of species indicative of an organic rich environment. The community was "patchy" with areas dominated by the stress indicator, tubificoides, although other areas were typical of salt marsh basins with high quality habitat. The observed benthic communities do appear to be consistent with the nutrient rich nature of the basin and the high chlorophyll levels, which may be adding additional stress to this basin. Given the patchiness of the community, the dominance of tubificoides in some samples and the high chlorophyll levels it appears the Lewis Pond is presently supporting a high quality to moderately impaired habitat for benthic animals.

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

The results of the Infauna Survey recommend that nitrogen management threshold analysis, needs to include a lowering of the level of nitrogen enrichment in Seine Pond and the Parkers River, and to a lesser extent in Lewis Pond, for restoration of nitrogen impaired benthic habitats.

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

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

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

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

The habitat assessment data are also internally consistent. Overall, the oxygen and chlorophyll data for the Parkers River Estuary clearly indicate a system supporting sub-tidal habitats impaired by nitrogen, ranging from highly stressed (Seine Pond) to moderately stressed (Lower River and Lewis Pond). These observations are consistent with the high levels of total nitrogen (TN) throughout the estuary. The gradient in impairment follows the gradient in nitrogen enrichment, where Seine Pond has very high ebb tide TN levels (>0.94 mg L^{-1}) declining to the Lower River nearest the tidal inlet (0.66 mg L^{-1}). While the lower River supports lowest nitrogen levels within the system, the levels are still quite high and indicate a basin incapable of supporting eelgrass beds and with a moderate level of impairment to benthic animal habitat (see Sections VII-3 & VII-4). Lewis Pond also supports a high TN level (0.86 mg L^{-1}), but this is in part the result of its function as a salt marsh basin.

The observed loss of eelgrass, presence of drift algae, moderate oxygen and chlorophyll levels and benthic community structure within the lower Parkers River basin, suggests a system beyond the nitrogen threshold level that would be supportive of eelgrass, but relatively close to the level for supporting high quality infaunal habitat. The average nitrogen levels for this lower reach were 0.663 mg N L^{-1} at mid ebb tide, slightly above the level supportive of infaunal communities (0.5-0.6 mg N L^{-1}), but well above levels supportive of eelgrass beds as found in a variety of MEP assessments of Cape Cod estuaries. Similarly, the total nitrogen levels at mid-ebb tide within Seine Pond (0.948-0.994 mg N L^{-1}) are well above levels found in basins supportive of high quality benthic animal habitat. Parallel measurements made in Seine Pond of oxygen depletion and very high chlorophyll a levels stemming from frequent phytoplankton blooms, as well as accumulations of drift macroalgae, are all consistent with a basin significantly impaired by nitrogen enrichment. It is clear that a significant reduction in nitrogen loading or increase in tidal flushing (or both) will be required for restoration of Seine Pond, the main embayment basin in the Parkers River Estuary.

In contrast, given the extensive salt marsh areas present in the Lewis Pond portion of the Parkers River embayment system, the observed oxygen levels and the characteristics of the benthic community described in Section VII-4, it appears that this reach of the overall system is only moderately impaired based upon its high chlorophyll levels and resident benthic animal community. While there is some impairment, the shallow nature of the pond and its ecological function as a salt marsh basin suggests that only modest nitrogen management will be needed for restoration.

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

The eelgrass and water quality information supports the conclusion that eelgrass beds within the lower reach of the Parkers River should be the primary target for restoration of the Parkers River Embayment System and that restoration requires appropriate nitrogen management. From the historical analysis, it appears that while only a modest acreage of eelgrass can be restored, it will be coupled with restoration of large areas of severely degraded benthic animal habitat within the upper estuary (above Rt. 28) as well as improved dissolved oxygen levels that cause periodic fish kills. Therefore, the sentinel station for the Parkers River Estuary is located between the long-term water quality monitoring stations within the lower reach of the River (PR-2 & PR-3). This site was selected based upon its location at the upper most extent of the documented eelgrass coverage in this estuary (Figure VII-6).

With the sentinel station located at the uppermost extent of the historical eelgrass coverage, the target nitrogen concentration (tidally averaged TN) for restoration of eelgrass at the sentinel location within the lower reach of the Parkers River was determined to be 0.42 mg TN L^{-1}. As there has not been eelgrass habitat within the Parkers River Estuary for over a decade, this threshold was based upon comparison to other local embayments of similar depths and structure under MEP analysis. The historic Parkers River eelgrass habitat appears to have been patchy and like other similar basins, found mainly within the more stable shallow areas. Similar nearby systems like the Bournes Pond Estuary, where eelgrass has historically been confined to the lower estuarine basin, has nitrogen concentrations supportive of eelgrass at 0.45 mg TN L^{-1} within the main stem of the channel to the upper estuary, with a lower level, 0.42 mg TN L^{-1}, within the open water basin of Israel's Cove. The higher threshold within the main channel region of the Bournes Pond system is supported by the existence of healthy eelgrass beds at tidally averaged TN concentrations of 0.426 mg TN L^{-1} and the presence of eelgrass in patches (not beds) at tidally averaged TN of 0.481 mg TN L^{-1}. Additionally, within the lower reach of the Green Pond Estuary, sparse eelgrass is found at tidally averaged TN levels of 0.41 mg TN L^{-1}. The threshold tidally averaged TN level for restoration of eelgrass in the lower Parkers River portion of the overall system (0.42 mg TN L^{-1}) results from the need to restore eelgrass within the margins of the tidal channel where light reaches the sediments at higher TN levels than in deeper areas.

Although the nitrogen management target is restoration of eelgrass habitat (and associated water clarity, shellfish and fisheries resources), benthic infaunal habitat quality must also be supported as a secondary condition. Benthic animals are more tolerant of nutrient and organic matter enrichment than eelgrass, which requires clear waters and high oxygen levels. At present, in the regions with moderately impaired infaunal habitat within the lower reach of the Parkers River, average ebb tide total nitrogen (TN) levels are in the range of 0.65 mg N L^{-1}. The observed moderate impairment at this site is consistent with observations by the MEP Technical Team in other enclosed basins along Nantucket Sound (e.g. Perch Pond, Bournes Pond, Popponesset Bay) where levels <0.5 mg N L^{-1} were found to be supportive of healthy infaunal habitat and where moderately impaired habitat was found at ~0.6 mg N L^{-1}. Similarly, the Centerville River system showed moderate impairment at tidally averaged TN levels of 0.526 mg N L^{-1} in Scudder Bay (analogous to the salt marsh dominated Lewis Pond) and at 0.543 mg TN L^{-1} in the middle reach of the Centerville River. Similarly, moderate impairment was also observed at TN levels (0.535-0.600 mg N L^{-1}) within the Wareham River.

Based upon these observations, the MEP Technical Team concluded that an upper limit of 0.50 mg N L-1 tidally averaged TN would support healthy infaunal habitat in the embayment basins of the Parkers River Estuary (Seine Pond and Parkers River). However, given the shallow nature of Lewis Pond and its function as primarily a salt marsh basin, a tidally averaged TN of <0.60 mg N L^{-1} was determined. This higher level stems from the much shallower basin of Lewis Pond compared to Scudder Bay and the similar infaunal threshold for the shallow salt marsh dominated upper reach of the Mashpee River.

It should be emphasized that these secondary criteria values were not used for setting nitrogen thresholds in this embayment system. These values merely provide a check on the acceptability of conditions within the tributary basins (Seine Pond average of PR-1 & PR-5; Lewis Pond PR-4) at the point that the threshold level is attained at the sentinel station within the lower Parkers River. The results of the Linked Watershed-Embayment modeling are used to ascertain that when the nitrogen threshold is attained, TN levels in these regions are also within the acceptable range. The goal is to achieve the nitrogen target at the sentinel location and restore eelgrass habitat within the lower reach of the Parkers River and restore infaunal habitat throughout the System. The nitrogen loads associated with the threshold concentration at the sentinel location and secondary infaunal check stations are discussed in Section VIII.3, below.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

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

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required nearly 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the entire system. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. For Example, removal of 95% of the septic load from the Parkers River watershed results in a 88% reduction in total watershed nitrogen load. No load reduction was necessary for the Cedar Pond watershed. Table VIII-4 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. In Table VIII-4, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent 'worst-case' summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to background concentrations in Cape Cod Bay, as discussed in Section VI.2.6.1.

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel station is shown in Table VIII-5. The percent change *P* over background presented in this table is caculated as:

$$P = (N_{threshold}-N_{present})/(N_{present}-N_{background})$$

where N is the nitrogen concentration at the indicated monitoring station for present and threshold conditions, and also in Nantucket Sound (background).

To achieve the threshold nitrogen concentrations at the sentinel station, reductions in TN concentrations of typically greater than 65% is required in the system, between the main harbor basin and the marsh.

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

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

Another management alternative that is available for the Parkers River system is the possibility of widening the culvert under Route 28, which connects between the upper and lower portions. The re-establishment of a larger tide range in the upper river and Seine Pond would increase tidal flushing, and help to reduce N concentrations in the upper system, and reduce the amount of watershed N load reduction necessary to achieve the target threshold concentrations. Widening scenarios are discussed in Chapter IX.

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

sub-embayment	present septic load (kg/day)	threshold septic load (kg/day)	threshold septic load % change
Seine Pond	16.992	0.510	-97.0%
Upper Parkers River	12.340	0.370	-97.0%
Lower Parkers River	11.751	0.588	-95.0%
Lewis Pond	14.682	0.73411	-95.0%
System Total	55.764	2.202	-96.1%

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

sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Seine Pond	20.562	4.080	-80.2%
Upper Parkers River	16.408	4.439	-72.9%
Lower Parkers River	12.652	1.489	-88.2%
Lewis Pond	17.400	3.452	-80.2%
System Total	67.022	13.459	-79.9%

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	4.080	1.096	-2.227
Upper Parkers River	4.439	0.049	0.409
Lower Parkers River	1.489	0.266	16.261
Lewis Pond	3.452	0.616	3.300
System Total	13.459	2.027	17.744

Table VIII-5. Comparison of model average total N concentrations from present loading and the threshold scenario, with percent change over background in Nantucket Sound (0.294 mg/L), for the Parkers River system. The threshold stations are shown in bold print.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	threshold (mg/L)	% change
Lewis Pond	**PR-4**	**0.859**	**0.448**	**-72.7%**
Lower Parkers River	PR-3	0.491	0.362	-65.6%
upper Parkers River	PR-2	0.802	0.456	-68.1%
lower Seine Pond	PR-1	0.965	0.493	-70.3%
upper Seine Pond	PR-5	1.007	0.504	-70.6%
Seine Pond sentinel station	**-**	**0.993**	**0.501**	**-70.5%**
Parkers River sentinel station	**-**	**0.638**	**0.410**	**-66.3%**

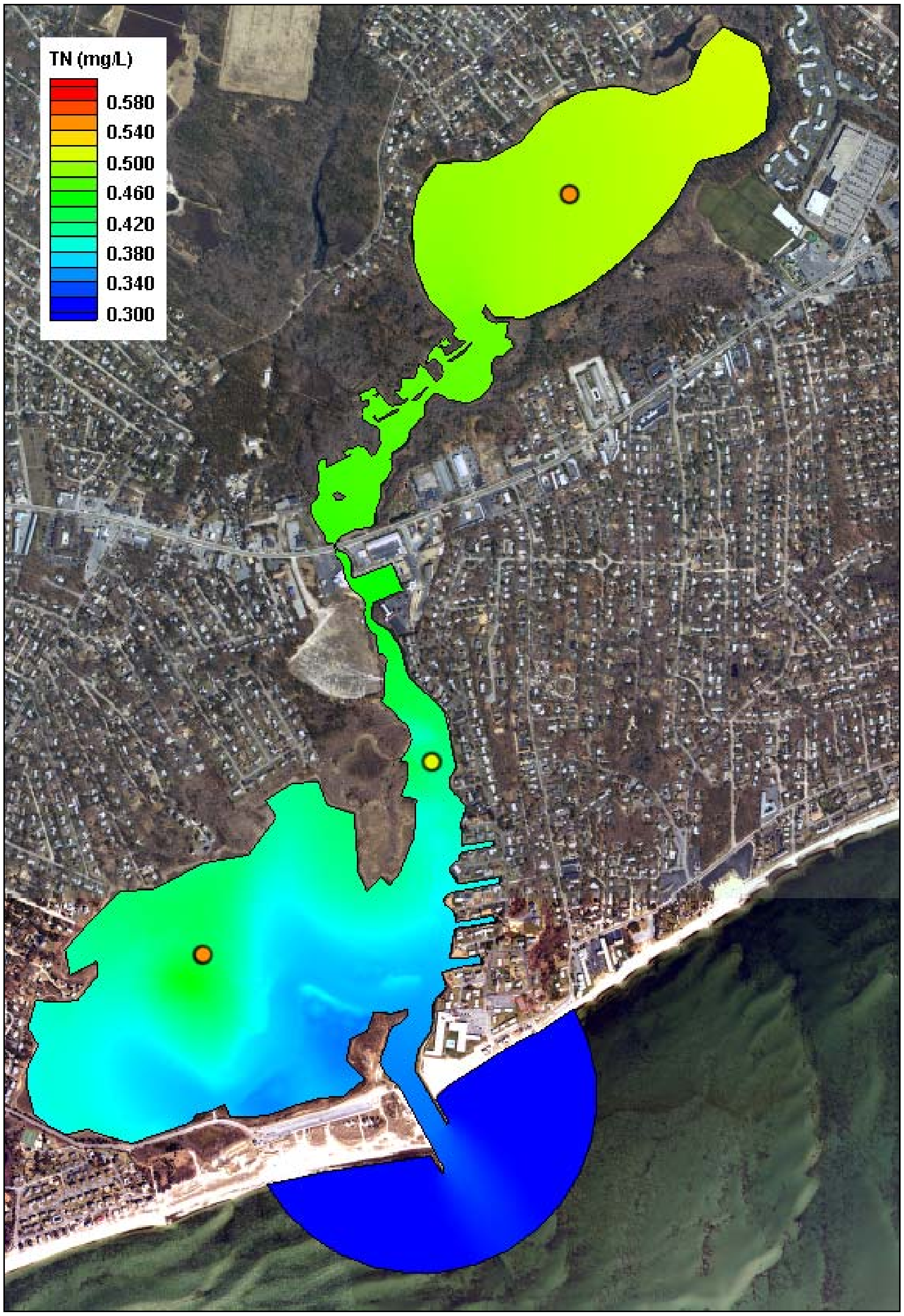


Figure VIII-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Parkers River estuary, for threshold conditions. Threshold station are indicated by the yellow dot (primary threshold of 0.42 mg/L Parkers River main channel), and the orange dots (0.50 mg/L in Seine Pond, and 0.60 mg/L in Lewis Pond).

IX. HYDRODYNAMIC AND WATER QUALITY MODEL SCENARIOS

Modeling of scenarios involving changes to the existing Route 28 Culvert between the upper and lower portions of Parkers River was performed to determine the possible benefit to increasing the tidal flushing of Seine Pond. The present culvert greatly restricts the tidal range of the upper system such that the tide range north of the culvert is only 30% of the total average range south of the road crossing. By increasing tidal flushing, it would be possible to reduce the amount of watershed N removal required to meet the threshold concentrations set in Chapter VIII.

For this analysis, the channel width and depth were alternately modified to determine changes to the maximum channel velocities and tide range using the hydrodynamic model. Using the results of this quasi-optimization analysis, the selected channel configuration was run using the water quality model in order to determine the changes to N concentrations that would results from the increased tidal flushing.

IX.1 HYDRODYNAMIC MODIFICATIONS TO THE ROUTE 28 CULVERT

A total of four alternate channel configurations were simulated with the hydrodynamic model. The range of modeled channel attributes included 1) a 30-foot crossing with a bottom elevation of -3.4 ft NAVD, 2) a 30-foot crossing with a bottom elevation of -6.4 ft NAVD, 3) a 30-foot crossing with a bottom elevation of –6.4 ft NAVD and a smooth unarmored channel bottom, and 4) a 40-foot crossing with a bottom elevation of -3.4 ft NAVD.

After completing each hydrodynamic scenario run, the maximum channel velocities were computed using the model output. As a general rule of thumb, maximum channel velocities should be no less than 3.0 feet per second (fps) in order to ensure that tidal flow will be able to keep the channel clear of sediment. This is important since it is difficult to otherwise maintain a channel by mechanical means when access is impeded by a structure, whether it is a bridge or a large culvert.

The resulting maximum modeled velocities from the four modeled scenarios (including present conditions) are plotted in Figure IX-1. All scenarios except Scenario 3 meet the minimum velocity threshold. Scenario 2 is the option that exceeds the minimum velocity threshold by the smallest amount, which indicates that it is the most optimum of the modeled scenarios.

After ruling out Scenario 3 and based on the minimum velocity threshold, Mean High Water (MHW) and Mean Low Water (MLW) values were computed for the remaining three scenarios to compare with present conditions. These values are presented in Table IX-1. In this comparison, Scenario 2 stands out due to its large tide range, which is 1.4 feet larger than the tidal range under present conditions. This increased tidal range results from a 0.4 foot increase in MHW and a 0.5 foot decrease in MLW. Scenario 2 allows the greatest increase in tide range with only a 0.4 foot increase in calculated MHW.

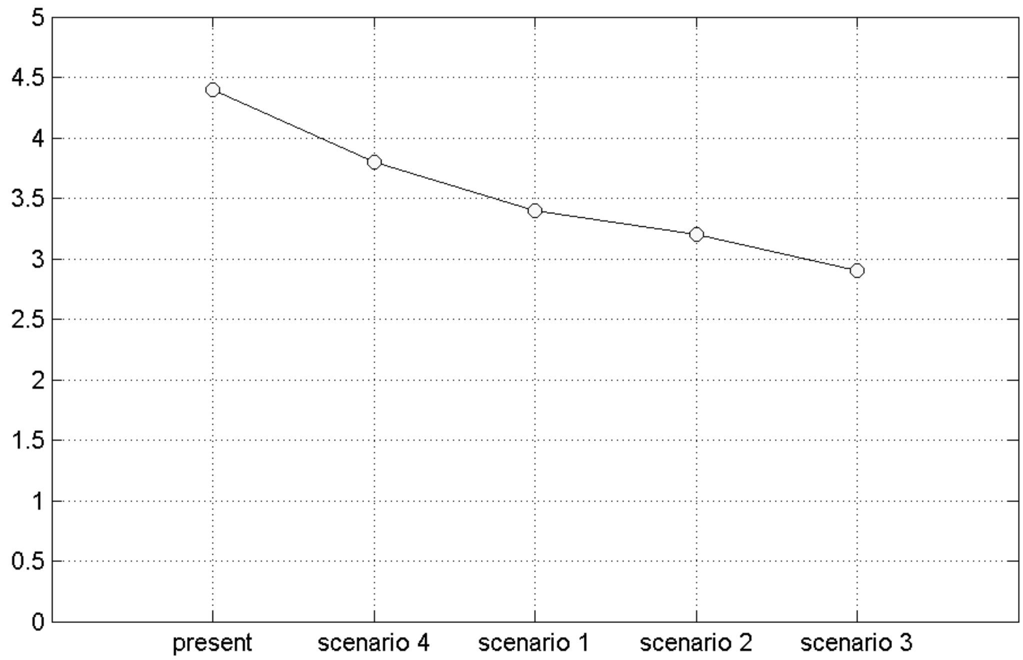


Figure IX-1. Maximum modeled culvert velocities (fps) for present conditions and four scenarios. The scenarios are 1) a 30-foot crossing with a bottom elevation of -3.4 feet (NAVD), 2) a 30-foot crossing with a bottom elevation of -6.4 feet, 2) a 30-foot crossing with a bottom elevation of -6.4 feet and a smooth unarmored channel bottom, and 4) a 40-foot crossing with a bottom elevation of -3.4 feet.

Table IX-1. Comparison of Mean High Water (MHW) and Mean Low Water (MLW) and mean tide range elevations computed from hydrodynamic model runs of present condition (calibration time period as presented in Chapter V) and the three scenarios, at the upper Parkers River tide station.

Scenario	MHW (ft, NAVD)	MLW (ft, NAVD)	tide range (feet)
Present	+1.4	+0.5	0.9
1) 30 foot-wide, -3.4 ft bottom	+1.6	+0.2	1.4
2) 30 foot-wide, -6.4 ft bottom	+1.8	-0.5	2.3
4) 40 foot-wide, -3.4 ft bottom	+1.7	0.0	1.7

Therefore, Scenario 2 is selected as the best option of the modeled scenarios for optimizing the Route 28 crossing channel, since it offers greater tidal flushing from the increased tide range with only a 0.4 foot increase in MHW, and because maximum tidal velocities exceed the velocity threshold of 3.0 ft/sec required to keep the channel clear of sediment. A comparison of the modeled tide for existing conditions and the selected culvert option is presented in Figure IX-2.

Computed system volumes and residence time for the optimized Route 28 crossing channel are shown below in Tables IX-2 through IX-5, along with a comparison to the computed values for existing conditions. The tide prism of the upper portion of the river (including Seine Pond) increases by nearly 80%, which results in a reduction of nearly 50% in the local residence time for the upper system.

Table IX-2. Embayment mean volumes and average tidal prism during hydrodynamic calibration simulation period for optimized 30 ft-wide culvert.		
Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Parkers River (system)	28,305,000	17,299,000
Upper Parkers River	15,269,000	5,595,000
Seine Pond	13,967,000	4,933,000

Table IX-3. Average tidal prisms and %change between present 18 ft-wide culvert and optimized 30 ft-wide culvert.			
Embayment	Tide Prism Volume (ft^3) 18' Culvert	Tide Prism Volume (ft^3) 30' Culvert	% Change
Parkers River (total)	15,461,000	17,493,000	+13.1
Upper Parkers River	3,348,000	5,984,000	+78.7
Seine Pond	2,959,000	5,299,000	+79.0

Table IX-4. Computed System and Local residence times for embayments in the system during hydrodynamic calibration simulation period for optimized 30 ft-wide culvert.		
Embayment	System Residence Time (days)	Local Residence Time (days)
Parkers River (system)	0.8	0.8
Upper Parkers River	2.4	1.3
Seine Pond	2.7	1.3

Table IX-5. Local Residence Times and % change between present 18 ft-wide culvert and optimized 30 ft-wide culvert.			
Embayment	Local Residence Time (days) 18' Culvert	Local Residence Time (days) 30' Culvert	% Change
Parkers River (system)	1.0	0.8	-20.0
Upper Parkers River	2.4	1.3	-45.8
Seine Pond	2.5	1.3	-48.0

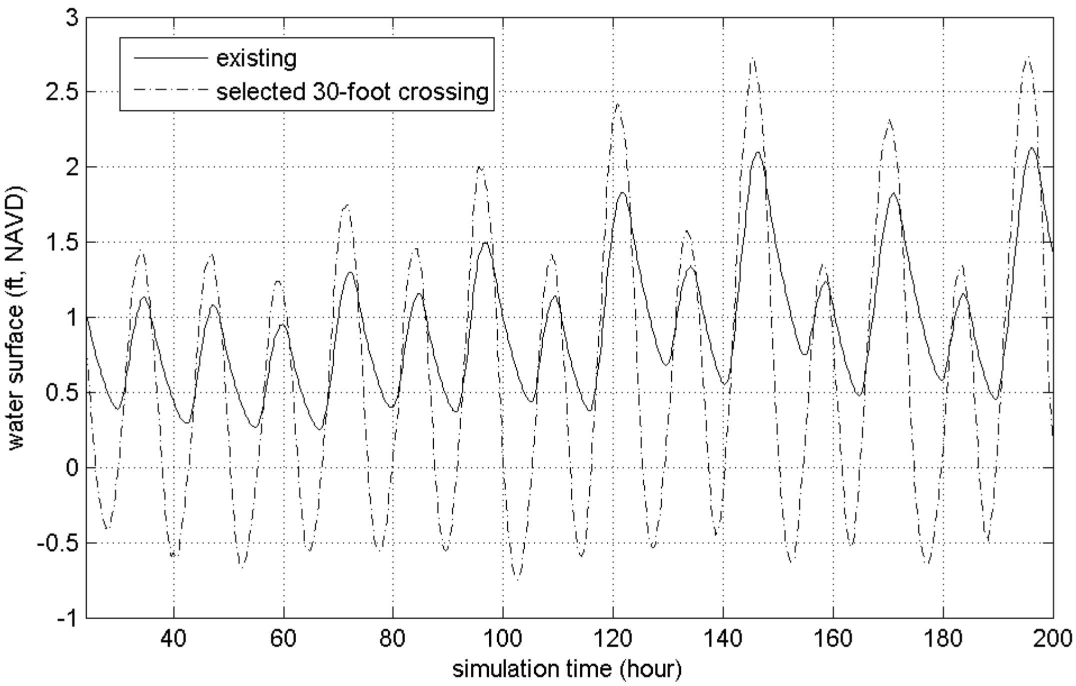


Figure IX-2. Comparison of modeled tides for existing conditions and the selected 30-foot-wide culvert option (channel depth at -6.4 ft NAVD).

IX.2 TOWN WATERSHED LOADING SCENARIOS USING OPTIMIZED CULVERT

The optimized 30 foot-wide culvert was used to model three watershed loading scenarios provided by the town, labeled Scenario 1, Scenario BO-3, and Scenario BO-4. In addition to the town supplied scenarios, an additional threshold loading scenario was determined in the following section of this chapter. This threshold was determined in a fashion similar to the scenario determined for the existing culvert, in Chapter VIII. The loading and results of the Town scenarios are presented in Tables IX-6 through 11. Color contour plot of each scenario are presented in Figures IX-3 through IX-5.

The results show that Town Scenario 1 meets the threshold concentration set in Chapter VIII, 0.42 mg/L at the mid Parkers River sentinel station, 0.50 mg/L at the mid Seine Pond sentinel station, and 0.60 mg/L at the Lewis Pond monitoring station. Scenario 1 includes present land use conditions with the pre-modification configuration of the channel between Long Pond and Seine Pond and the removal of all wastewater loads from the Parker River watershed. Scenarios BO-3 and BO-4 nearly achieve the required threshold concentrations, but would require some additional watershed load removal to reach the needed level of restoration.

Table IX-6. **Town Loading Scenario 1** sub-embayment loads used for total nitrogen modeling of the Parkers River system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	3.570	1.096	-2.294
Upper Parkers River	4.896	0.049	0.414
Lower Parkers River	0.901	0.266	16.414
Lewis Pond	4.762	0.616	3.324
System Total	14.129	2.027	17.858

Table IX-7. Comparison of model average total N concentrations from present loading and **Town Loading Scenario 1**, with percent change over background in Nantucket Sound (0.294 mg/L as discussed in Chapter VIII), for the Parkers River system. The threshold stations are shown in bold print.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	Scenario 1 (mg/L)	% change
Lewis Pond	PR-4	**0.859**	**0.457**	**-71.1%**
Lower Parkers River	PR-3	0.491	0.340	-76.5%
upper Parkers River	PR-2	0.802	0.377	-83.7%
lower Seine Pond	PR-1	0.965	0.394	-85.2%
upper Seine Pond	PR-5	1.007	0.400	-85.2%
Seine Pond sentinel station	**-**	**0.993**	**0.399**	**-85.0%**
Parkers River sentinel station	**-**	**0.638**	**0.360**	**-80.8%**

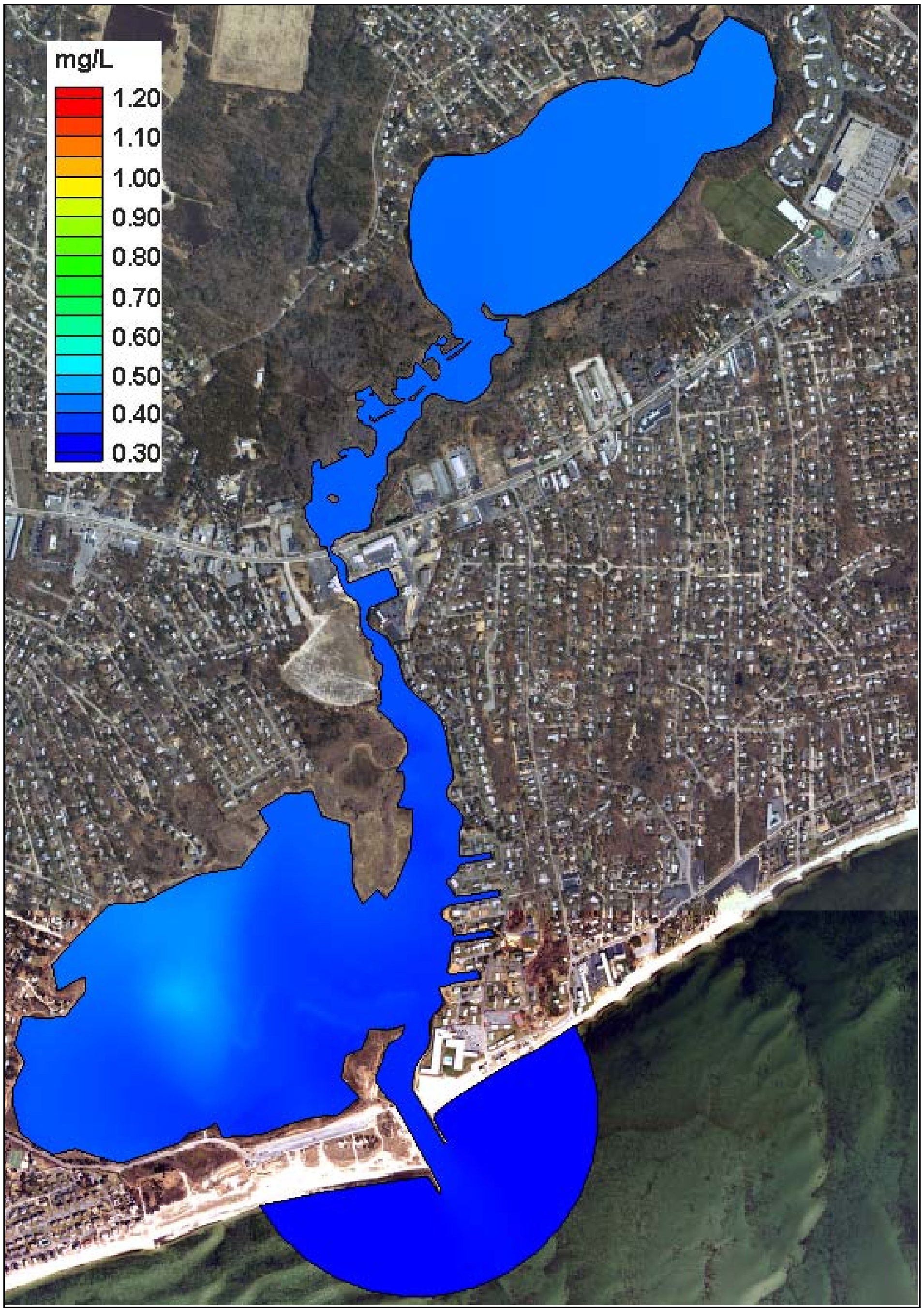


Figure IX-3. Contour plot of modeled total nitrogen concentrations (mg/L) in the Parkers River estuary, for **Town Loading Scenario 1**.

Table IX-8. **Town Loading Scenario BO-3** sub-embayment loads used for total nitrogen modeling of the Parkers River system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	4.185	1.096	-4.382
Upper Parkers River	23.564	0.049	0.629
Lower Parkers River	0.957	0.266	21.181
Lewis Pond	17.547	0.616	4.091
System Total	46.252	2.027	21.520

Table IX-9. Comparison of model average total N concentrations from present loading and **Town Loading Scenario BO-3**, with percent change over background in Nantucket Sound (0.294 mg/L as discussed in Chapter VIII), for the Parkers River system. The threshold stations are shown in bold print.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	Scenario BO-3 (mg/L)	% change
Lewis Pond	PR-4	**0.859**	**0.757**	**-18.1%**
Lower Parkers River	PR-3	0.491	0.386	-53.2%
upper Parkers River	PR-2	0.802	0.469	-65.6%
lower Seine Pond	PR-1	0.965	0.501	-69.2%
upper Seine Pond	PR-5	1.007	0.509	-69.8%
Seine Pond sentinel station	**-**	**0.993**	**0.509**	**-69.3%**
Parkers River sentinel station	**-**	**0.638**	**0.429**	**-60.9%**

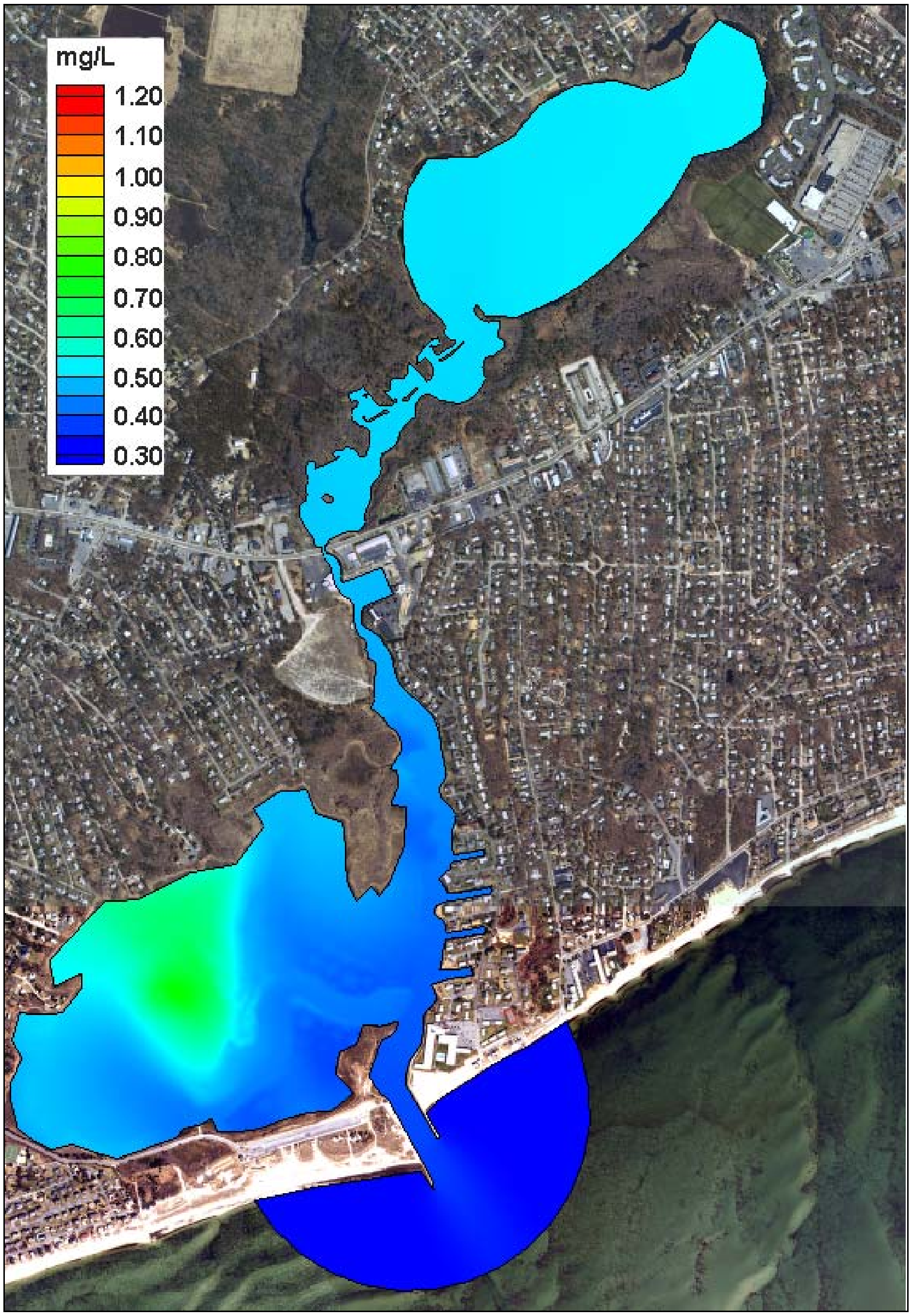


Figure IX-4. Contour plot of modeled total nitrogen concentrations (mg/L) in the Parkers River estuary, for **Town Loading Scenario BO-3**.

Table IX-10. **Town Loading Scenario BO-4** sub-embayment loads used for total nitrogen modeling of the Parkers River system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	4.185	1.096	-4.177
Upper Parkers River	20.284	0.049	0.608
Lower Parkers River	0.957	0.266	22.856
Lewis Pond	17.547	0.616	4.601
System Total	42.973	2.027	23.888

Table IX-11. Comparison of model average total N concentrations from present loading and **Town Loading Scenario BO-4**, with percent change over background in Nantucket Sound (0.294 mg/L as discussed in Chapter VIII), for the Parkers River system. The threshold stations are shown in bold print.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	Scenario BO-4 (mg/L)	% change
Lewis Pond	PR-4	**0.859**	**0.753**	**-18.9%**
Lower Parkers River	PR-3	0.491	0.383	-55.0%
upper Parkers River	PR-2	0.802	0.459	-67.5%
lower Seine Pond	PR-1	0.965	0.489	-70.9%
upper Seine Pond	PR-5	1.007	0.497	-71.5%
Seine Pond sentinel station	**-**	**0.993**	**0.497**	**-71.0%**
Parkers River sentinel station	**-**	**0.638**	**0.422**	**-62.7%**

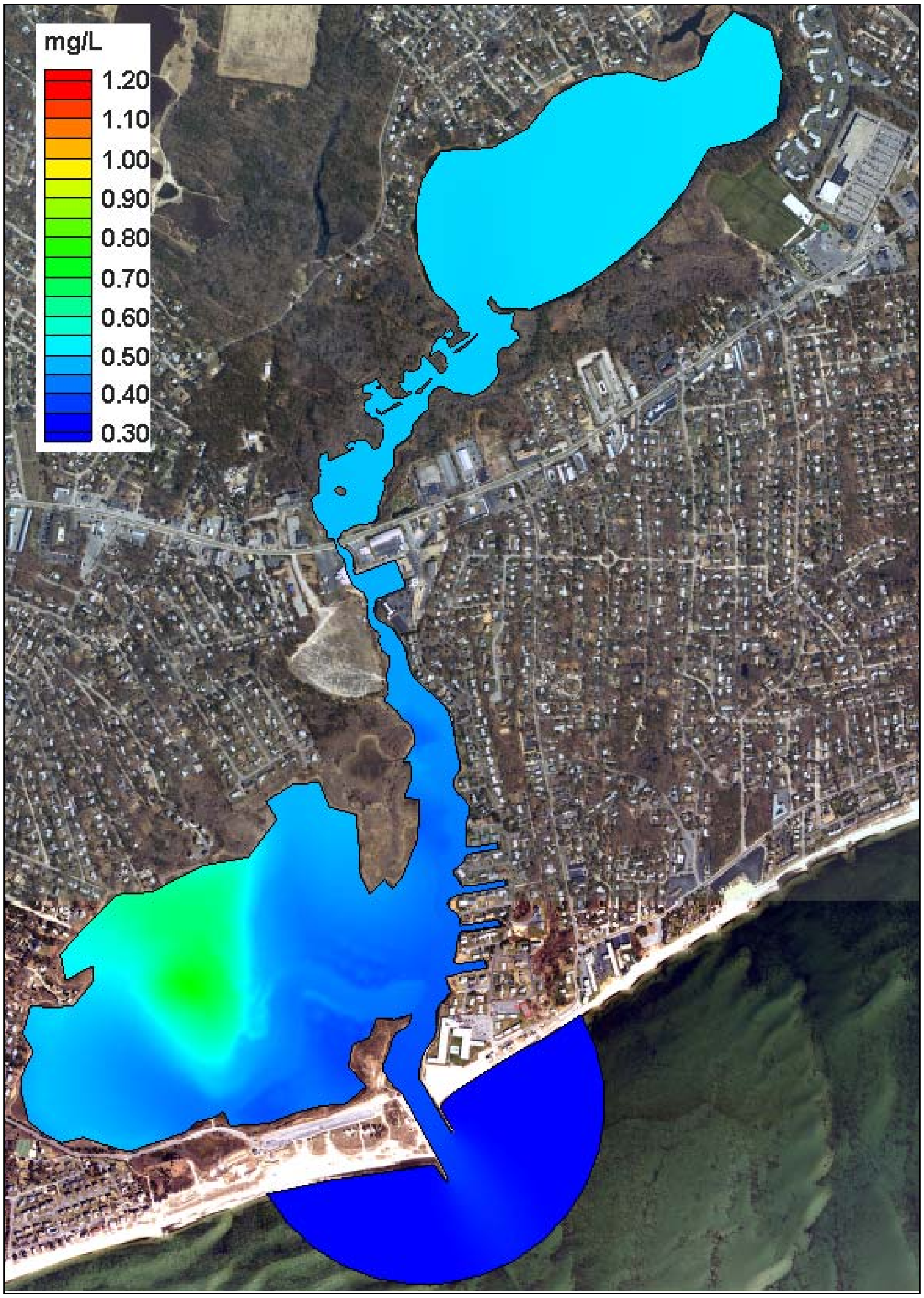


Figure IX-5. Contour plot of modeled total nitrogen concentrations (mg/L) in the Parkers River estuary, for **Town Loading Scenario BO-4**.

IX.3 OPTIMIZED CULVERT THRESHOLD LOADING

An alternate watershed loading threshold was determined for Parkers River using the optimized culvert. The loading and TN concentrations are presented in Tables IX-12 through IX-15. A color contour plot of resulting threshold load TN concentrations is presented in Figure IX-6. The results of the optimized threshold show that by modifying the Route 28 culvert, and improving tidal flushing in the upper portion of the system, the amount of watershed load removal required to achieve threshold TN concentrations is significantly less that the threshold determined in Chapter VIII, for the existing culvert. Instead of removing nearly 100% of the septic load to the system with the existing culvert, approximately 63% of the septic load needs to be removed with the modified culvert.

Table IX-12. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and **optimized culvert** threshold loading scenarios of the Parkers River System. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg/day)	Optimized threshold septic load (kg/day)	threshold septic load % change
Seine Pond	18.649	7.646	-59.0%
Upper Parkers River	16.022	4.319	-73.0%
Lower Parkers River	11.751	4.113	-65.0%
Lewis Pond	14.746	4.40	-70.0%
System Total	61.168	20.483	-66.5%

Table IX-13. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer, and WWTF loads) used for modeling of present and **optimized culvert** threshold loading scenarios of the Parkers River system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	Optimized threshold load (kg/day)	threshold % change
Seine Pond	20.611	11.216	-45.6%
Upper Parkers River	16.408	8.387	-48.9%
Lower Parkers River	12.652	5.014	-60.4%
Lewis Pond	17.400	7.122	-59.1%
System Total	67.070	31.740	-52.7%

Table IX-14. **Optimized Culvert Threshold** sub-embayment loads used for total nitrogen modeling of the Parkers River system, with total watershed N loads, atmospheric N loads, and benthic flux

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Seine Pond	11.216	1.096	-3.450
Upper Parkers River	8.387	0.049	0.534
Lower Parkers River	5.014	0.266	20.411
Lewis Pond	7.122	0.616	4.120
System Total	31.740	2.027	21.616

Table IX-15. Comparison of model average total N concentrations from present loading and the **Optimized Culvert Threshold**, with percent change over background in Nantucket Sound (0.294 mg/L as discussed in Chapter VIII), for the Parkers River system. The threshold stations are shown in bold print.

Sub-Embayment	monitoring station (MEP ID)	present (mg/L)	optimized threshold (mg/L)	% change
Lewis Pond	**PR-4**	**0.859**	**0.552**	**-54.3%**
Lower Parkers River	PR-3	0.491	0.376	-58.5%
upper Parkers River	PR-2	0.802	0.445	-70.3%
lower Seine Pond	PR-1	0.965	0.484	-71.8%
upper Seine Pond	PR-5	1.007	0.499	-71.2%
Seine Pond sentinel station	**-**	**0.993**	**0.497**	**-71.0%**
Parkers River sentinel station	**-**	**0.638**	**0.412**	**-65.6%**

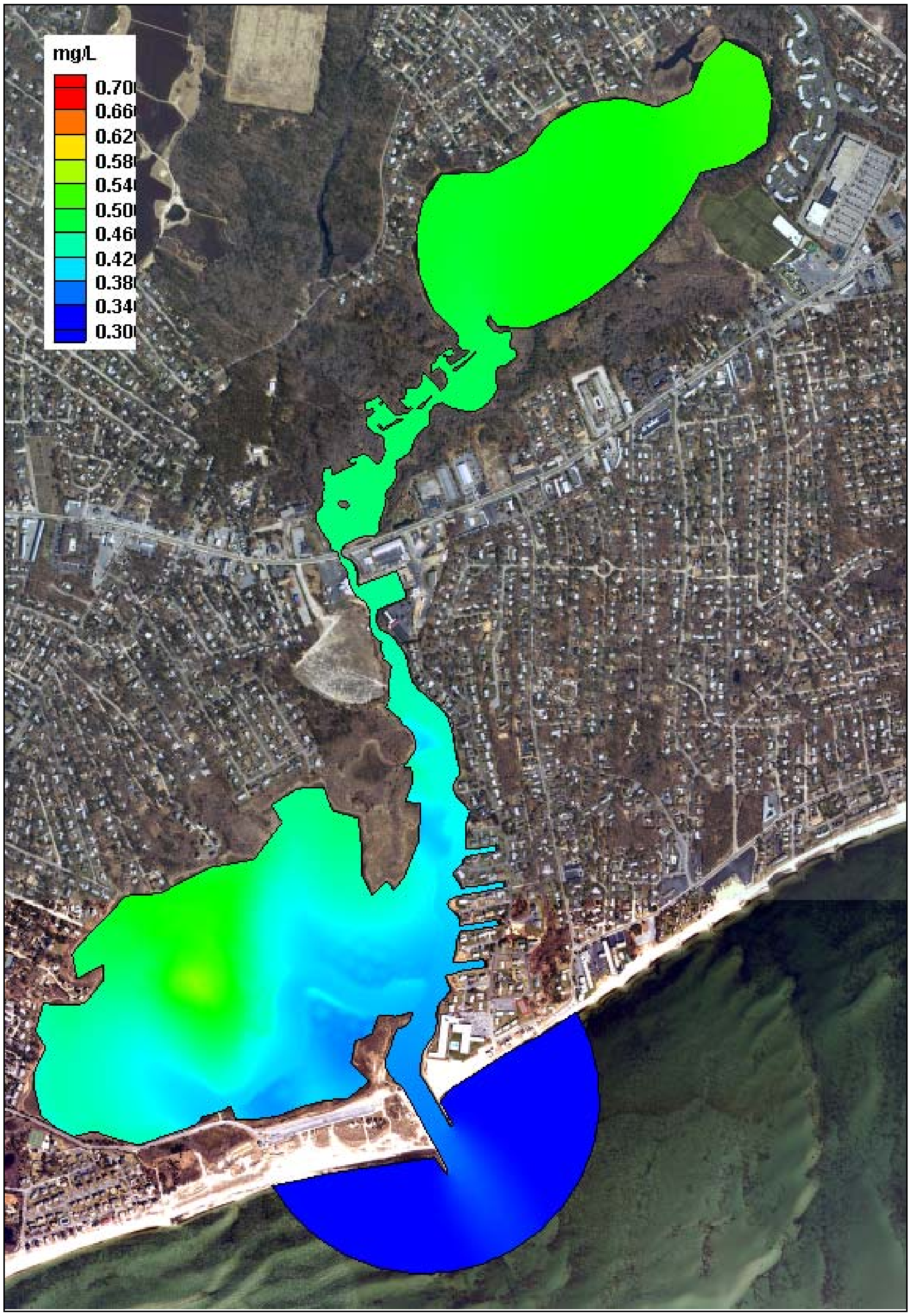


Figure IX-6. Contour plot of modeled total nitrogen concentrations (mg/L) in the Parkers River estuary, for the optimized culvert threshold.

IX.4 INITIAL ASSESSMENT OF SYSTEM RESPONSE DUE TO CHANGES IN TIDAL STAGES

A series of transects along the marsh plain were established on either side of Parkers River and along the shoreline of Seine Pond (Figures IX-7 and IX-8) for the purpose of surveying existing vegetative cover and species and assessing how these might change as a result of modifications to the Parkers River Route 28 bridge opening and ensuing changes in tidal stages throughout the system. Transects began at the upland border wherever possible or where identifiable marsh vegetation was visible and accessible. Transects ended at the water's edge. The dominant vegetation types along each transect were identified, recorded, dominance estimated and their distances measured. Vegetation transition points were staked and marked with a digital GPS unit. For Transects 1-6 and 19, stakes were used to survey the height of the marsh plain at each transition point. Because the marsh plain is flat and lies below the 4 ft. (NAVD) contour and significant amounts lie below the 2 ft. contour (Figure IX-9), changes predicted from the height and vegetation data for Transects 1-6 and 19 can reasonably be applied to the remainder of the transects in the Parkers River and around the shoreline of Seine Pond.

The current Mean High Water mark (MHW) for this system is 1.4 ft. NAVD. The Mean Low Water mark (MLW) is 0.5 ft. NAVD. The projected MHW and MLW marks after the more optimal Scenario 2 bridge modification (30 foot-wide, -6.4 ft bottom) are 1.8 ft. NAVD and -0.5 ft. NAVD, respectively. These predicted post-restoration changes in tidal range are then used to predict likely changes in the wetland vegetation for both Parkers River and Seine Pond.

In general, post-restoration high marsh vegetation will expand landward horizontally to the equivalent of approximately 0.3 ft. of vertical increase and replace other vegetation which will in turn retreat landward and likely decrease in aerial extent on the surface of the marsh plain. Similarly, as tidal inundation of the marsh plain increases, hydrologic conditions that support low marsh vegetation will improve. Consequently, the low marsh zone will likely expand into the present high marsh zone to some extent. The present marsh plain generally decreases in height along a longitudinal gradient from the area adjacent to the current bridge (Transects 1 and 2) to Transect 19 at Seine Pond (Figure IX-7). Therefore, after bridge replacement, increases in tidal inundation toward the upland edge of the marsh plain will be more significant toward Seine Pond than near the bridge.

The fringing wetlands of the Parkers River in Yarmouth, MA lie along a plain from the waters edge to the 4ft. (NAVD) contour and much of it lies at or below the 2 ft. contour (Figure IX-9). The vegetation is presently characterized primarily by high salt marsh vegetation dominated by salt marsh hay *Spartina patens* with spike grass *Distichlis spicata* mixed with the *Spartina patens* in many areas. Much of the area along the northwest shore of Parkers River is ditched and contains berms which criss-cross the marsh plain, suggesting that there were historically active cranberry bogs there, now abandoned. Vegetation in these areas include small shrubs and trees and the *Phragmites* reed, in addition to typical high marsh *Spartina patens* and *Distichlis spicata*

The wetlands along the shoreline of Seine Pond lie in a relatively narrow plain at or below the 2 ft. contour (Figure IX-9). Unlike the topography of the Parkers River shore, there is a steep rise from the 2 to the 4 ft. contour along most of this shore with little or no horizontal separation between the 2 and 4 ft. contours. Consequently, these contours are indistinguishable from each other in Figure IX-9. The vegetation here is characterized by intermittent narrow fringing stands of salt marsh with some expanded areas of marsh and large stands of *Phragmites* in the upper pond. There is also what appears to be an abandoned cranberry bog along the southeast shore of the pond adjacent to a commercial golf driving range.

Figure IX-7. Wetland transects, Parkers River and Seine Pond, Yarmouth, MA

Figure IX-8. Wetland transects, Parkers River and Seine Pond, Yarmouth, MA.

Figure IX-9. Topographic contours and vegetations transects, Parkers River and Seine Pond, Yarmouth, MA.

Parkers River

The area traversed by Transects T1 and T2 is mostly in the 2-4 ft. contour but with some of the plain in the <2 ft. contour (Figures IX-7, IX-9). The vegetation is characterized primarily by high salt marsh vegetation, dominated by *Spartina patens*. T1 is an irregularly shaped oval of elevated marsh plain in the center of the high marsh zone dominated by small shrubs, including marsh elder *Iva* and poison ivy, and a few small trees (Figure IX-7). There is a small elevation in this area with the estimated highest point marked by a stake, *Iva*01 (2.5 ft. NAVD) (Figure IX-7). T2 runs from the upland border (T2P1) at or very close to the 4 ft. contour to the edge of Parkers River (T2P6) and is characterized primarily by high salt marsh (*Spartina patens* and *Distichlis spicata*) with a large band of cattails *Typha* near the upland border (Figure IX-7). There is a narrow band of creek bank with tall *Spartina alterniflora* (salt marsh cord grass) close to the river's edge. The transect vegetation characteristics are detailed in Table IX-16.

Table IX-16. Vegetation in Transect T2, Parkers River.

TRANSECT SEGMENT	LENGTH (FT)	HEIGHT RANGE (FT NAVD)	DOMINANT VEGETATION DESCRIPTION
T2P1-T2P2	36	1.94-1.7	MIXED TREES SHRUBS AND HERBACIOUS PLANTS
T2P2-T2P3	100	1.7-1.5	100% *TYPHA*
T2P3-T2P4	300	1.5-1.13	*SPARTINA PATENS* DOMINANT WITH *DISTICHLIS* VARYING 0-50%
T2P4-T2P5	166	1.13-0.79	MIXED *PATENS*/*DISTICHLIS* (50/50) TO END OF HIGH MARSH
T2P5-T2P6	8	0.79-(-0.37)	CREEK BANK LOW MARSH SPARSE TALL *SPARTINA ALTERNIFLORA*

Post-restoration, some of the shrubs and trees in the elevated area of Transect 1 will likely disappear as the MHW mark rises from 1.4 to 1.7 ft. NAVD. High marsh vegetation (*Spartina patens* and *Distichlis spicata*) will migrate into this area as a result. Since the estimated highest point of this area is 2.5 ft. NAVD however, some of the trees and shrubs will probably persist.

Present MHW is 1.4 ft. NAVD which is at the seaward boundary of the Typha zone in Transect 2, T2P3 (Table IX-16). Post restoration MHW is predicted to be 1.7 ft. NAVD which would put it at the landward boundary of the Typha zone. As a result, the Typha zone will probably retreat landward and decrease in areal extent. Some of the small trees, shrubs and herbaceous plants above the present Typha zone may also retreat landward and decrease in numbers. The high marsh vegetation zone will likely expand and migrate landward to about 1.7 ft. NAVD. The low marsh vegetation zone will likely expand from creek bank areas landward and replace some of the present high marsh vegetation in segment T2P4-T2P5 (Table IX-16, Figure IX-7).

Transects T3, T4, T5 and T6 are in an area of what appears to be an abandoned cranberry bog. All transect points of the marsh plain are at or below the 2 ft. contour (Figures IX-7 and IX-9). There are numerous shallow, narrow ditches and small berms on the marsh plain in this area along the northwest shore of the river creating micro-topography and small scale differences in high vs. low marsh vegetation (Figure IX-7). The vegetation is dominated by high salt marsh but there is a mosaic of *Phragmites*, small shrubs and trees mixed with the

Spartina patens and *Distichlis*. There is a small amount of *Spartina alterniflora* (short form) in and along the edges of some of the small ditches. The details are described in Table IX-17.

Table IX-17. Vegetation in Transects T3, T4, T5 AND T6, Parkers River.

TRANSECT SEGMENT	LENGTH (FT)	HEIGHT RANGE (FT NAVD	DOMINANT VEGETATION DESCRIPTION
T3P1-T3P2	118	1.72-1.13	Old Bog, Mixed Shrub 30%, Phrag 50%, Small Trees 20% (Old Ditches Cris Cross Area)
T3P2-T3P3	230	1.13-2.35	*Patens* 50%-80% Mixed With Phrag 30%-50% And *Iva* 20% In The First 100 Ft
T3P3	4	2.35	Center Of Berm Shrubs/*Iva* 50% *Phragmites* 50%
T3P3-T3P4	100	2.35-1.13	*Patens* 80% *Distichlis* 20%
T3P4-T3P5	119	1.13-0.71	*Patens* 90% *Distichlis* 10% End Of High Marsh
T3P5-T3P6	4	0.71-0.38	Creek Bank Sparse Vegetation Very Little *Alterniflora*
T4P1-T4P2	126	1.49-0.81	*Patens* 60% *Distichlis* 40%
T4P2-T4P3	60	0.81-0.39	Low Marsh With 100% *Alterniflora* (Short) In Old Ditch In Bog System
T4P3-T4P4	7	0.39-?	Creek Bank Zone Sparse Vegetation *Alterniflora* 20% Rest Exposed Peat
T5P1-T5P2	87	1.56-1.4	Old Bog Surface *Patens* 30% *Iva* 30% Small Trees And Shrubs 20% *Distichlis* 20%
T5P2-T5P3	52	1.4-1.09	*Patens* 40% *Distichlis* 30% *Iva* 30%
T5P3-T5P4	35	1.09-0.79	*Patens* 70% *Distichlis* 30%
T5P4-WATER	12	0.79	Old Ditch/Some Berm No Creek Bank Vegetation
T6P1-T6P2	59	1.17-1.19	*Patens* 30% *Distichlis* 10% Phrag 60%
T6P2-T6P3	0-38	1.19	*Patens* 50% *Distichlis* 50% Old Ditch At 38 Ft. With *Alterniflora* 100%
T6P2-T6P3	38-138	1.19	*Patens* 50% *Distichlis* 50% <5% *Alterniflora* Ditch To Berm
T6P3	4	0.84	Center Of Berm
T6P3-T6P4	135.5	0.65	*Patens* 50% *Distichlis* 50%
T6P4-WATER	3	0.65	Some Tall *Alterniflora* <50% Rest Is Exposed Peat

Post restoration changes in the wetland vegetation in these transects will be similar to those described for transect T2. The MHW mark will increases from 1.4-1.7 ft. NAVD which will bring it up to the head of T3 and landward of T4-T6. High marsh vegetation will migrate landward and will replace some of the Phragmites and shrubs and trees in T3, T5 and T6 all of which will likely retreat landward and decrease in the extent of their areal coverage. The low marsh vegetation will also likely expand landward from its present coverage in creek bank areas to replace some of the adjacent high marsh vegetation.

Transect T7 is mostly in the < 2 ft. contour with some points between the 2 and 4 ft. contours (Figures IX-7 and IX-9). T7P1 is at or near the 4 ft. contour. The vegetation is characterized by high marsh near the upland border at T7P1 with a significant area of low marsh between T7P3 and T7P4 (Figure IX-7, Table IX-18). *Spartina alterniflora* (short) is abundant in this segment between 0-35 ft. (likely a small depression in the marsh plain probably due to some post-restriction subsidence of the peat) with some *Spartina patens* and *Distichlis* mix as well between 35 and 128 ft. The last 10 ft. of this segment are all *Spartina alterniflora* associated with a ditch. High marsh then dominates out to the creek bank. Table IX-18 contains a detailed description of this transect.

Table IX-18. Vegetation in Transect T7, Parkers River.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T7P1-T7P2	18	S*patens* 60% *Iva* 30% Small Trees 10%
T7P2-T7P3	14	*Patens* 50% *Distichlis* 50%
T7P3-T7P4	0-35	Low Marsh 100% *Alterniflora*
T7P3-T7P4	35-128	*Patens* 40%-60% *Distichlis* 30%-40% *Alterniflora* 30%
T7P3-T7P4	128-138	Ditch *Alterniflora* 100%
T7P4-T7P5	116	*Patens* 60% *Distichlis* 40%
T7P5-WATER	4	Creek Bank Tall *Alterniflora* 100%

Post- restoration changes are likely to be similar to those in Transects 2-6 with high marsh expanding landward to some extent. The expansion may be enough to push landward some of the small trees and *Iva* in the transect segment T7P1-T7P2 (Table IX-18). Low marsh will also expand from the creek bank area into the present high marsh zone in T7P4-T7P5 (Table IX-18). It may or may not reach the present area of low marsh described in segment T7P3-T7P4 (Table IX-18).

Most of Transect T8 is at or below the 2 ft. contour (Figures IX-1 and IX-9). The vegetation is characterized by high marsh with a section of low marsh and a pan area at 32-54 ft. of segment T8P2–T8P3, due probably to some subsidence of the marsh plain there (Figure IX-7, Table IX-19). There is no tall *S. alterniflora* on the creek bank. The details of this transect are in Table IX-19.

Table IX-19. Vegetation in Transect T8, Parkers River.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T8P1-T8P2	11	*Patens* 60% *Iva* 40%
T8P2-T8P3	0-32	*Patens* 50% *Distichlis* 50%
T8P2-T8P3	32-54	Low Marsh *Alterniflora* 100% Plus Pan
T8P2-T8P3	54-100	High Marsh *Patens* 80% *Distichlis* 20%
T8P2-T8P3	100-137	High Marsh *Patens* 60% *Distichlis* 40% No Creek Bank Vegetation

Post- restoration changes are likely to include the expansion of high marsh landward to some extent and possibly pushing landward some of the *Iva* in segment T8P1-T8P2 (Table IX-19). Low marsh will become established in and expand from the creek bank into some of the present high marsh zone in T8P2-T8P3. It may or may not reach the present area of low marsh at 32-54 ft. described in that segment (Table IX-19).

Transect T9 is entirely within the 2-4 ft. contour (Figures IX-7 and IX-9). The vegetation is dominated by high marsh plants near the upland border at T9P1 out to T9P3 with a narrow area of creek bank tall *Spartina alterniflora* in the creek bank zone (Figure IX-9, Table IX-20).

After bridge replacement, low marsh vegetation will likely expand landward from the creek bank area to present high marsh in segment T9P2-T9P3 (Table IX-20). High marsh may also expand to some extent landward in segment T9P1-T9P2 pushing landward and replacing some of the *Iva* in this segment (Table IX-20).

Table IX-20. Vegetation in Transect T9, Parkers River.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T9P1-T9P2	10	*Patens* 50% *Iva* 50%
T9P2-T9P3	105	*Patens* 50% *Distichlis* 50%
T9P3-WATER	3	Creek Bank *Alterniflora* 100%

Seine Pond

Narrow bands of fringing salt marsh vegetation are typical of much but not all of the wetlands around the shoreline of Seine Pond. There are expanded areas of salt marsh as well as *Phragmites* in the upper pond (Figures IX-7 and IX-8). For most of the shoreline, the 2 and 4 ft. contours are very close together, indicating that any expansion of marsh plain is not likely to go significantly beyond the 2 ft. contour (Figure IX-9).

Transect 19 is adjacent to the pond near the mouth of the Parkers River. It is entirely within the 2 ft. contour (Table IX-21). The current MHW mark is at 1.4 ft. NAVD which coincides with the head of this transect. The vegetation is dominated mostly by high marsh for most of its length, transitioning to low marsh toward the shore of the pond in segment T19P2-T19P3 and then to creek bank tall *Spartina alterniflora* at the water's edge in T19P4-edge of the water (Figure IX-7, Table IX-21).

Table IX-21. Vegetation in Transect T19, Seine Pond.			
TRANSECT SEGMENT	**LENGTH (FT)**	**HEIGHT RANGE (FT NAVD**	**DOMINANT VEGETATION DESCRIPTION**
T19P1-T19P2	63	1.4-1.06	*Patens* 20% *Distichlis* 20% *Iva* 40%
T19P2-T19P3	165	1.06-0.43	*Patens* 60% *Distichlis* 40% *Alterniflora* <5%
T19P3-T19P4	7	0.43-0.44	*Patens* 60% *Alterniflora* 40%
T19P4-EDGE OF WATER	3		Tall *Alterniflora* 100%

Post restoration changes will be similar to those described above for adjacent transects 2-6. The MHW mark will expand beyond the head of the transect and high marsh vegetation will expand landward as a result, pushing other vegetation such as *Iva* and other small shrubs and trees further landward. Low marsh will expand landward as well, increasing its % dominance in T19P2-T19P3 and T19P3-T19P4. It may also expand into present high marsh areas in T19P1-T19P2 (Table IX-21).

Transect T10 is almost entirely within the <2 ft. contour with a small segment in the 2-4 ft. contour (Figures IX-7 and IX-9). Vegetation is a mix of small trees and shrubs near the upland border at T10P1. High marsh dominates from T10P2 to T10P3, transitioning to low marsh toward the water's edge (Figure IX-7, Table IX-22). The details of this short transect are in Table IX-22 below.

Table IX-22. Vegetation in Transect T10, Seine Pond.		
TRANSECT SEGMENT	**LENGTH (FT)**	**DOMINANT VEGETATION DESCRIPTION**
T10P1-T10P2	15	Even Mix Of Small Trees Shrubs Grasses And *Iva*
T10P2-T10P3	28	*Patens* 80% *Distichlis* 20%
T10P3-EDGE OF WATER	10	100% *Alterniflora*

As the water level rises in the pond after bridge replacement, the amount of exposed shoreline will probably decrease to some extent and low marsh vegetation will move and expand landward. Consequently, adjacent high marsh vegetation will likely decrease and expand landward toward the upland border near the 4 ft. contour and will likely replace some of the small trees and shrubs currently in T10P1-T10P2 (Table IX-22).

Transect T11 is a narrow band of fringing low marsh approximately 11 ft. wide, all *Spartina alterniflora* (Figure IX-7). It is entirely within the 2 ft. contour (Figures IX-7 and IX-9). After restoration, the rise in water level along the shore will push the Spartina landward toward the upland border near the 4 ft. contour (Figures 1, 3).

Transect T12 is a small area of 100% *Phragmites* approximately 17 ft. wide from the upland border near the 2 and 4 ft. contours at T12P1 out to T12P2 (Figures IX-7 and IX-8). After bridge replacement, the areal extent of the Phragmites should decrease with the rise in water level. It will not expand beyond the upland border. Some low marsh vegetation may replace the Phragmites here depending on the extent of the water's rise.

Transect T13 is adjacent to a boardwalk over this section of marsh. The transect is entirely within the 2 ft. contour which is very close to and indistinguishable from the 4 ft. contour in Figure IX-9. The vegetation is dominated by *Phragmites* from the upland border at T13P1 out to approximately 270 ft. where there is a transition to high marsh and then to low marsh toward the water's edge (Figure IX-8). The vegetation details of T13 are in Table IX-23.

Table IX-23. Vegetation in Transect T13, Seine Pond.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T13P1-T13P2	270	*Phragmites* 100%
T13P2-T13P3	51	*Patens* 100%
T13P3-T13P4	10	*Alterniflora* 100%

Post restoration changes will likely include expansion of the high marsh into the present Phragmites zone as water levels rise in the pond. The areal extent of the Phragmites will decrease as it is pushed back toward the upland border by both high marsh expansion and rising water to the west (Figure IX-8). Low marsh vegetation will also expand into the present high marsh zone (Table IX-23).

Transect T14 is also entirely within the 2 ft. contour with both ends of the transect bordered by water (Figures IX-8 and IX-9). The vegetation is characterized by high marsh around the border areas near the water and low marsh in the interior where there is likely a small depression probably to subsidence of the marsh plain there (Figure IX-8, Table IX-24). There is a narrow band of creek bank tall *Spartina alterniflora* along the edge of the water at T14P4 (Figure IX-8, Table IX-24).

Table IX-24. Vegetation in Transect T14, Seine Pond.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T14P1-T14P2	65	*Patens* 90% *Distichlis* 10%
T14P2-T14P3	91	*Alterniflora* 100% Plus Pan
T14P3-T14P4	89	*Patens* 90% *Alterniflora* 10%
T14P4-EDGE OF WATER	7	Tall *Alterniflora* 100%

Rising water levels after bridge replacement are likely to expand the low marsh zone at both ends along this transect and consequently decrease the areal extent of the high marsh vegetation (Figure IX-8, Table IX-24).

As was the case with Transects 13 and 14, T15 is entirely within the 2 and 4 ft. contours (Figures IX-8 and IX-9) and is also on small peninsula. It begins at the upland border near the 2 and 4 ft. contours at T15P1 where there is a large tract of *Phragmites* from T15P1-T15P2. There is a transition at T15P2 to predominantly high marsh with some low marsh vegetation and pan area between T15P3-T15P4, probably due to a slight depression in the marsh plain from subsidence, and another tract of high marsh toward the water's edge (Figure IX-8, Table IX-25).

Table IX-25. Vegetation in Transect T15, Seine Pond.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T15P1-T15P2	95	*Phragmites* 100%
T15P2-T15P3	31	*Patens* 100%
T15P3-T15P4	38	*Alterniflora* 80% Plus Pan *Patens* 20%
T15P4-T15P5	67	*Patens* 90% *Distichlis* 10%

As water levels rise after bridge replacement, the high marsh vegetation in T15P2-T15P3 will expand into the Phragmites zone which will consequently decrease as it retreats landward toward the upland border. Low marsh vegetation will likely become established and expand into the present Phragmites zone from the water's edge on the north side of the peninsula (Figure IX-8). Low marsh will also expand from T15P3-T15P4 into the high marsh zone in segments T15P2-T15P3 and T15P4-T15P5 (Table IX-25, Figure IX-8).

The point marked "herrun" in Figure IX-8 is the outfall of the Anadromous fish run from Long Pond to the northeast.

Transect 16 marks a small area of *Phragmites* at the head of the pond adjacent to the Anadromous fish run (Figure IX-8). The transect is entirely within the 2 and 4 ft. contour (Figures IX-8 and IX-9). The transect is approximately 17 ft. long from T16P1-T16P2 and abuts the upland border near T16P1. The vegetation is 100% *Phragmites*.

Similar to T12, rising water levels after bridge replacement will decrease the areal extent of the Phragmites. Some Spartina grasses may become established there as well.

Transect 17 is characterized primarily by low marsh with some high marsh vegetation mixed in toward the water's edge (Figure IX-8). It is entirely within the 2 and 4 ft. contours (Figures IX-8 and IX-9). The details are in Table IX-26 below.

Table IX-26. Vegetation in Transect T17, Seine Pond.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T17P1-T17P2	28	*Alterniflora* 100%
T17P2-T17P3	23	*Patens* 80% *Alterniflora* 20%

Post-restoration changes will likely include expansion of the low marsh vegetation landward from the present water's edge as the water rises. It will diminish and possibly replace the high marsh vegetation currently present in segment T17P2-T17P3 (Table IX-26).

Transect 18 runs through what appears to be an abandoned cranberry bog adjacent to a commercial golf driving range off of Route 28. It is entirely within the 2 and 4 ft. contour except for a berm close to the shoreline of the pond (Figures IX-8 and IX-9). There is a mix of small tress and shrubs with *Phragmites* toward the upland border at T18P1 in segment T18P1-T18P2. *Spartina patens* appears with *Iva* and other small shrubs and trees in segment T18P2-T18P3 which ends at the berm (Figure IX-8). The berm separates this upper part of the transect from the remainder which is closer to the shore of the pond. Shoreward of the berm, *Spartina patens* dominates for about 35 ft. and transitions to creek bank tall *Spartina alterniflora* toward the water's edge (Figure IX-8, Table IX-27).

Table IX-27. Vegetation in Transect T18, Seine Pond.

TRANSECT SEGMENT	LENGTH (FT)	DOMINANT VEGETATION DESCRIPTION
T18P1-T18P2	126	Phrag 90% With Few Small Trees And Shrubs
T18P2-T18P3	172	*Patens* 60% *Iva* 30% Small Trees And Shrubs 10%
T18P3	4	Center Of Berm
T18P3-T18P4	15	*Patens* 70% *Iva* 30%
T18P4-T18P5	20	*Patens* 100%
T18P5-EDGE OF WATER	4	Tall *Alterniflora* 100%

Post-restoration changes are somewhat complicated by the presence of the berm. Low marsh vegetation will expand from the present creek bank area toward the berm and will likely replace much of the current high marsh vegetation presently abutting the berm (Figure IX-8, Table IX-27). Rising water will also penetrate the upper part of the transect through openings in the berm (Figure IX-9). Consequently, low marsh will be able to expand into this area in segment T18P2-T18P3 to some extent and replace some of the high marsh vegetation there. Rising waters will also allow high marsh vegetation to expand landward, increase its % dominance and possibly replace some of the shrubs, trees and Phragmites in this upper part of the wetland (Table IX-27).

X. REFERENCES

AFCEE (with Howes, B.L. & Jacobs Engineering). 2000. Ashumet Pond Trophic Health Technical Memorandum. AFCEE/MMR Installation Restoration Program, AFC-J23-35S18402-M17-0005, 210pp.

Aubrey Consulting Inc., 1996. Tidal Flushing within the East Bay/Centerville River Estuary: Existing Conditions and Effects of Proposed Dredging. Final Report, prepared for the Town of Barnstable, MA., 41 pp.

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

Brigham Young University, 1998. "User's Manual, Surfacewater Modeling System."

Burns, K., M. Ehrhardt, B. Howes and C. Taylor., 1993. Subtidal benthic community respiration and production rates near the heavily oiled coast of Saudi Arabia. Marine Pollution Bull. 27:199-205.

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

Cape Cod Commission, 1998. "Cape Cod Coastal Embayment Project." Barnstable, MA.

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

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

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

Cape Cod Commission. 1996. Regional Policy Plan. Cape Cod Commission, Barnstable, MA.

Cape Cod Commission. 2001. Regional Policy Plan. Cape Cod Commission, Barnstable, MA.

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

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

D'Elia, C.F, P.A. Steudler and N. Corwin. 1977. Determination of total nitrogen in aqueous samples using persulfate digestion. Limnology and Oceanography 22:760-764.

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

Draft Environmental Impact Report (DEIR)/Development of Regional Impact. Centerville River Dredging and Craigville Beach Nourishment, Woods Hole Group, Inc. December 2003

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

Eichner, E.M. and T.C. Cambareri, 1992. Technical Bulletin 91-001: Nitrogen Loading. Cape Cod Commission, Water Resources Office, Barnstable, MA. Available at: http://www.capecodcommission.org/regulatory/NitrogenLoadTechbulletin.pdf

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

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

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

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

Frimpter, M.H., J.J. Donohue, M.V. Rapacz. 1990. A mass-balance nitrate model for predicting the effects of land use on groundwater quality. U.S. Geological Survey Open File Report 88:493.

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

Hamersley, R.M. and B. Howes, 2004. Nitrogen Fluxes and Mitigation Strategies in the Audubon Skunknett River Wildlife Sanctuary. Report to the Town of Barnstable

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

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

Howes BL. 1998. Sediment metabolism within Massachusetts Bay and Boston Harbor relating to rates and controls of sediment-water column exchanges of nutrients and oxygen in 1997. Boston: Massachusetts Water Resources Authority. Report 1998-20. 80 p

Howes, B.L. and D.D. Goehringer. 1997. Falmouth's Coastal Salt Ponds. Falmouth Pondwatch Program, 1987-1996.

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

Howes, B.L., J.S. Ramsey and S.W. Kelley, 2001. Nitrogen modeling to support watershed management: comparison of approaches and sensitivity analysis. Final Report to MA Department of Environmental Protection and USEPA, 94 pp. Published by MADEP.

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, J. Wood, E. Eichner (2004). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Popponesset Bay, Mashpee and Barnstable, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, E. Eichner (2003). Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Stage Harbor, Sulphur Springs, Taylors Pond, Bassing Harbor and Muddy Creek, Chatham, Massachusetts. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.

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

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

Howes, B.L., H.E. Ruthven, E.M. Eichner, J.S. Ramsey, R.I. Samimy, D.R. Schlezinger. 2008. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Lewis Bay System, Towns of Barnstable and Yarmouth, MA. SMAST/DEP Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA

Howes, B.L. and J.M. Teal. 1995. Nitrogen balance in a Massachusetts cranberry bog and its relation to coastal eutrophication. Environmental Science and Technology 29:960-974.

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

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

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

Koppelman, L.E. (Ed.). 1978. The Long Island comprehensive waste treatment management plan. Vol II. Summary documentation report, Long Island Regional Planning Board, Hauppage, N.Y.

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

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

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

Masterson, J.P., Walter, D.A., Savoie, J., 1996, Use of particle tracking to improve numerical model calibration and to analyze ground-water flow and contaminant migration, Massachusetts Military Reservation, western Cape Cod, Massachusetts: U.S. Geological Survey Open-File Report 96-214, 50 p.

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

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

Murphy, J. and J.P. Reilly, 1962. A Modified Single Solution Method for the Determination of Phosphate in Natural Waters. Analytica Chemica Acta, v. 27, p. 31-36

Nelson, M.E., S.W. Horsley, T.C. Cambareri, M.D. Giggey and J.R. Pinnette. 1998. Predicting nitrogen concentrations in groundwater- An analytical model. Focus Conference on Eastern Groundwater Issues, National Water Well Association, Stamford, CT.

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

Oldale, R.N. 1974a. Geologic Map of the Hyannis Quadrangle, Barnstable County, Cape Cod, Massachusetts. US Geological Survey Map GQ-1158. US Geological Survey, Reston, VA.

Oldale, R.N. 1974b. Geologic Quadrangle Maps of the United States, Geologic Map of the Dennis Quadrangle, Barnstable County, Cape Cod, Massachusetts. US Geological Survey Map GQ-1114. US Geological Survey, Washington, DC.

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

Ramsey, J.S., B.L. Howes, S.W. Kelley, and F. Li, 2000. "Water Quality Analysis and Implications of Future Nitrogen Loading Management for Great, Green, and Bournes

Ponds, Falmouth, Massachusetts." Environment Cape Cod, Volume 3, Number 1. Barnstable County, Barnstable, MA. pp. 1-20.

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

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

Rhoads, D.C. and J.D. Germano. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142:291-308

Robertson, W.D., J.D. Cherry, and E.A. Sudicky. 1991. ³Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers.² Ground Water. 29(1): 82-92.

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

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

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

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

Stearns and Wheler. 2007. Town of Barnstable Final Wastewater Facilities Plan, Final Environmental Impact Report and Notice of Project Change. Hyannis, MA.

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

Tetra Tech EM Inc. 2005. Cape Cod Nutrient Loading Study: Watershed nitrogen loads to Lewis Bay, Parker's River and Swan Pond. Final Project Report, Project Number 01-03/MWI, pp. 76.

US Army Corps of Engineers, Engineer Research and Development Center, Waterways Experiment Station, Coastal and Hydraulics Laboratory, Users Guide To RMA4 WES Version 4.5, June 05, 2001.

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

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

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

Walter, D.A., A. Whealan (2004). Simulated Water Sources and Effects of Pumping on Surface and Ground Water, Sagamore and Monomoy Flow Lenses, Cape Cod, Massachusetts. USGS Scientific Investigations Report 2004-5181

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

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

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

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