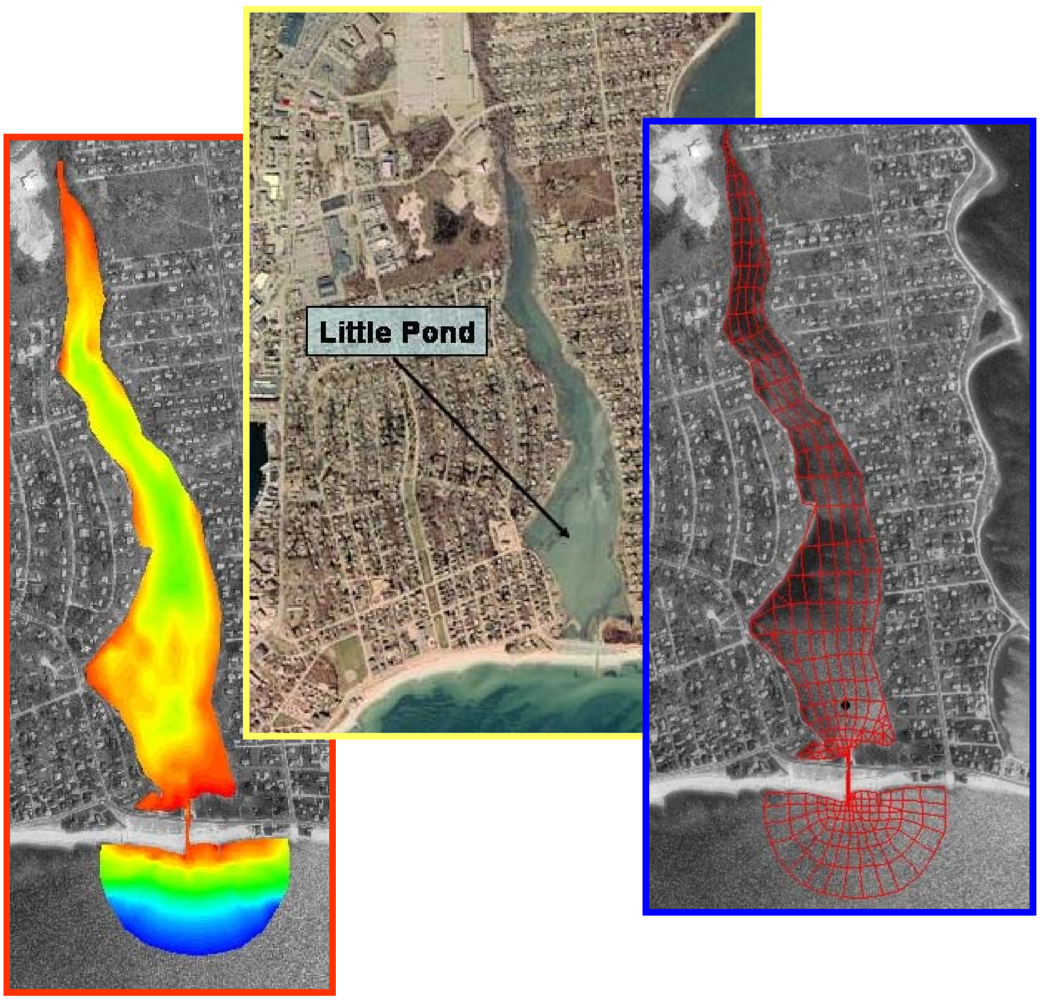
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Little Pond System, Falmouth, Massachusetts



University of Massachusetts Dartmouth
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Massachusetts Department of
Environmental Protection

FINAL REPORT – JANUARY 2006

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First and foremost is the significant time and effort in data collection and discussion spent by members of the Falmouth PondWatch Water Quality Monitoring Program, particularly Barbara Peri and Paul Affsa. These individuals gave of their time to collect nutrient samples from this system over many years and without this information, the present analysis would not have been possible. In addition, over the years, the Town of Falmouth Departments of Engineering and Planning have worked tirelessly with SMAST Coastal Systems Staff and engineers from Applied Coastal Research and Engineering towards the development of a restoration and management strategy for this system. This partnership has resulted in the reconstruction of the tidal inlet and phase I of the maintenance dredging of the tidal channel. These Town Departments along with the DPW have also provided important support to the present MEP effort. The technical team would like to specifically acknowledge the efforts of A. Lowell, B. Currie and G. Calise for helping to coordinate the Falmouth efforts with the MEP.

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I. INTRODUCTION

The Little Pond embayment system is located within the Town of Falmouth, on Cape Cod Massachusetts. The system has a southern shore bounded by water from Vineyard Sound (Figure I-1). The watershed for this great salt pond system is also distributed fully within the Town of Falmouth. The present configuration of the Little Pond embayment results from the drowning by rising sea level of valleys formed primarily via post-glacial erosion by groundwater fed rivers and streams. At present, Little Pond is a tidal embayment with a groundwater fed stream discharging to its headwaters. This situation is mirrored in almost all of the salt ponds on this stretch of the southern coast of Cape Cod. As is typical with other Falmouth embayments (Great, Green, and Bournes Pond) Little Pond is separated from Vineyard Sound by a barrier beach, which was naturally breached and is now artificially maintained by jetties. The beach and the opening to the embayment is a very dynamic geomorphic feature due to the influence of littoral transport processes. Over the past century the Little Pond inlet has experienced varying degrees of occlusion thereby affecting tidal exchange and circulation within the salt pond. By example, Bournes Pond became very restricted and finally completely isolated from Vineyard Sound waters in the late 1970's/early 1980's and was re-opened with a fixed inlet in mid 1980's. Currently, the inlet to Little Pond is periodically dredged to maintain the small tidal channel into the pond.

Similar to the Great, Green and Bournes Pond embayment systems, Little Pond is a mesotrophic (moderately nutrient impacted) to eutrophic (nutrient-rich) shallow coastal salt pond. The embayment is located within a glacial outwash plain, the Mashpee Pitted Plain, consisting of material deposited after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~18,000 years ago. The outwash material is highly permeable and varies in composition from well sorted medium sands to course pebble sands and gravels extending down to about 17 m below mean sea level (Millham and Howes, 1993). As such, direct rainwater run-off is typically rather low for these finger ponds and therefore, most freshwater inflow to these estuarine systems is via groundwater discharge or groundwater fed surface water flow (e.g. stream to the head of Little Pond, Coonamessett River to Great Pond, Backus River to Green Pond etc.). Little Pond acts as a mixing zone for terrestrial freshwater inflow and saline tidal flow from Vineyard sound, however, the salinity characteristics of the salt pond varies with the volume of freshwater inflow as well as the effectiveness of tidal exchange with Vineyard Sound.

Little Pond, along with the other salt pond embayments along the south coast of Falmouth, constitutes an important component of the Town's natural and cultural resources. In addition, the large length to width ratio (8:1) greatly increases the potential for direct discharges from homes situated on the shore and decreases the travel time of groundwater from the watershed recharge areas to bay regions of discharge. The nature of enclosed embayments in populous regions brings two opposing elements to bear: as protected marine shoreline they are popular regions for boating, recreation, and land development; as enclosed bodies of water, they may not be readily flushed of the pollutants that they receive due to the proximity and density of development near and along their shores. In particular, Little Pond as well as the Great, Green and Bournes Pond embayment systems along the Falmouth shoreline are at risk of eutrophication from high nitrogen loads in the groundwater and runoff from their watersheds.

Figure I-1. Study region proximal to the Little Pond embayment system for the Massachusetts Estuaries Project nutrient analysis. Tidal waters enter the salt pond through one inlet to Vineyard Sound. Freshwaters enter from the watershed primarily through 1 surface water discharge (stream to the head of Little Pond) and direct groundwater discharge.

The primary ecological threat to Little Pond resources is degradation resulting from nutrient enrichment. Loading of the critical eutrophying nutrient, nitrogen, to the embayment waters has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The nitrogen loading to this and other Falmouth salt ponds, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Town of Falmouth has been among the fastest growing towns in the Commonwealth over the past two decades and does not have centralized wastewater treatment throughout the entire Town. At present Little Pond is beyond its ability to tolerate additional nitrogen inputs. It is presently showing habitat degradation consistent with nitrogen overloading. Although the Little Pond watershed is approaching build-out, nitrogen related degradation will likely increase slightly with further water quality degradation, unless nitrogen management is initiated. Fortunately, as Little Pond nitrogen loads are near their build-out rates, management options can be clearly defined and implemented with a high degree of certainty for restoration.

To this end, as the primary stakeholder to the Little Pond embayment systems, the Town of Falmouth was one of the first communities to become concerned over perceived degradation of embayment waters. The Town of Falmouth (via the Planning Office) has long recognized the potential threat of nutrient over-enrichment of its coastal salt ponds and embayments. In the mid-1980's the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. Nutrient limits were set for nitrogen in each of the Town's embayments. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts, much like the approach of MEP and the associated TMDL process. To acquire baseline water quality data necessary for ecological management of Falmouth's coastal salt ponds and harbors, a citizen-based water quality monitoring program was initiated by the Town of Falmouth. Falmouth PondWatch, was established to provide on-going nutrient related embayment health information in support of the By-law. The water quality monitoring program was based on a collaborative effort between scientists, citizens and representatives of the Town of Falmouth. As originally conceived, the monitoring program focused on data collection in three original ponds, Oyster Pond, Little Pond and Green Pond. By 1990, the scope of water quality data collection expanded to include two additional ponds, Great/Perch Pond and Bournes Pond. In 1992, the scope of data collection was once again expanded to include West Falmouth Harbor in order to evaluate the effects from a nutrient enriched wastewater plume generated by the Falmouth Wastewater Treatment Facility.

The Falmouth PondWatch Program, as the water quality monitoring effort came to be known, continues to play an active role in the collection of baseline water quality data to this day, though it has evolved beyond its original mandate of providing basic environmental data relative to the Coastal Pond Overlay Bylaw (Nutrient Bylaw). The Pond Watch Program brings together, as requested by Town boards, ecological information relative to specific water quality issues. Additionally, as remediation plans for various systems are implemented, the continued monitoring satisfies demands by State regulatory agencies and provides quantitative information to the Town relative to the efficacy of remediation efforts. Lastly, the PondWatch Program has grown into being a repository of environmental data on Falmouth's coastal ponds. The database includes basic water quality monitoring data in addition to special project data on watershed nutrient loading and watershed delineation, circulation characteristics of the ponds, wetland delineations and plant and animal distributions.

The common focus of the Falmouth PondWatch Program effort has been to gather site-specific data on the current nitrogen related water quality throughout Falmouth's coastal embayments (e.g. Little Pond, Great, Green, and Bournes Pond embayment systems) and determine the relationship between observed water quality and 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 PondWatch Program in Little Pond developed a data set that elucidated the long-term trend of declining water quality and its relation to watershed based nutrient loading. The MEP effort builds upon the Falmouth water quality monitoring program, and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for the Little Pond embayment system, including any major sub-embayments, of which there are none for the Little Pond system. This is unlike the adjacent Great Pond system which has Perch Pond as a sub-embayment connected to the main bay of Great Pond

Falmouth's Planning Office continues to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present MEP evaluation is part of that continuing effort. Unfortunately, PondWatch monitoring has documented that most regions within the Town's coastal ponds, including Little Pond, are currently showing water quality declines and are beyond the limits set by the By-law. Based on the wealth of information obtained over the many years of study of these coastal ponds, in addition to the nutrient analyses undertaken as a precursor to the Massachusetts Estuaries Project, the Little Pond embayment system was included in the first round prioritization of the Massachusetts Estuaries Project to provide state-of-the-art analysis and modeling. However, given that the MEP was able to fully integrate the Towns' on-going data collection and modeling effort, minimal additional municipal funds were required for MEP tasks.

The critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning and nitrogen management alternatives development needed by the Town of Falmouth. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Town Falmouth to develop and evaluate the most cost effective nitrogen management alternatives to restore these valuable coastal resource which are currently being degraded by nitrogen overloading.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

Coastal embayments throughout the Commonwealth of Massachusetts (and along the U.S. eastern seaboard) are becoming nutrient enriched. 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 declining ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities. At its higher levels, enhanced loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human use.

Similar to nutrients, bacterial contamination is related to changes in land-use as watershed become more developed. Regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to culture, economy, and tax base of Massachusetts's coastal communities.

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

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

The Massachusetts Estuaries Project represents the newest generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MA 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 for watershed-embayment management for communities throughout Southeastern Massachusetts.

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

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

The major Project goals are to:

- develop a coastal TMDL working group for coordination and rapid transfer of results,
- determine the nutrient sensitivity of 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 regulatory needs.

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

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

The Linked Model has been applied for watershed nitrogen management in ca. 15 embayments throughout Southeastern Massachusetts. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options.

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

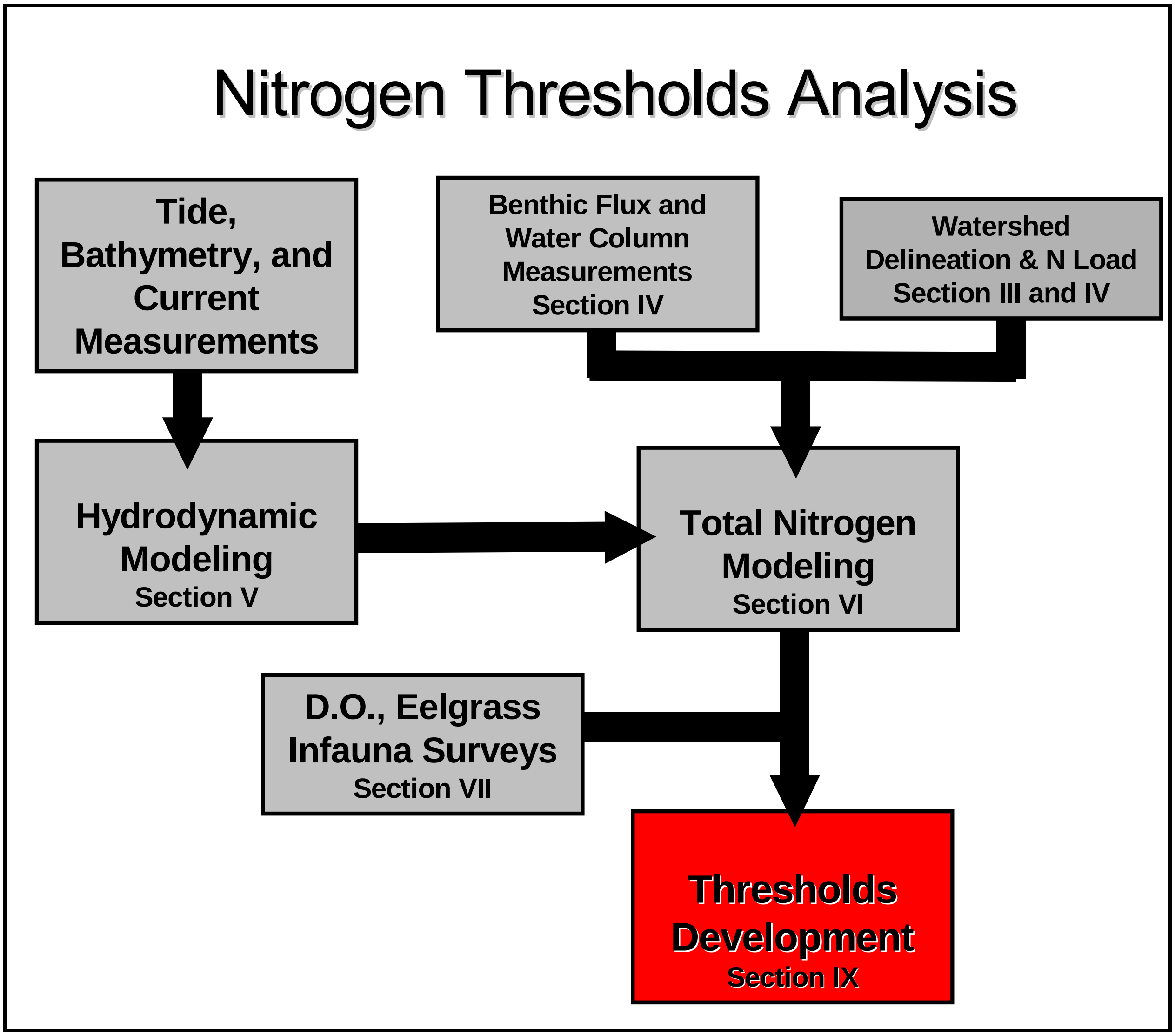


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach. Section numbers refer to sections in this MEP report where the specified information is provided.

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

- Monitoring - multi-year embayment nutrient sampling
- Hydrodynamics -
  - embayment bathymetry
  - site specific tidal record
  - current records (in complex systems only)
  - hydrodynamic model

- Watershed Nitrogen Loading
  - watershed delineation
  - stream flow (Q) and nitrogen load
  - land-use analysis (GIS)
  - watershed N model
- Embayment TMDL - Synthesis
  - linked Watershed-Embayment N Model
  - salinity surveys (for linked model validation)
  - rate of N recycling within embayment
  - D.O record
  - Macrophyte survey
  - Infaunal survey

I.2 SITE DESCRIPTION

The coastal salt ponds of Falmouth, including Little Pond, are oriented north-south, and open to Vineyard Sound via inlets. The configuration of the Little Pond and adjacent embayments results from the drowning by rising sea level of valleys formed primarily by post-glacial erosion by groundwater fed rivers and streams. At present, Little Pond is a tidal embayment with a groundwater fed stream discharging to its headwaters. This situation is mirrored in almost all of the salt ponds on this stretch of the southern coast of Cape Cod. As is typical with other Falmouth embayments (Great, Green, and Bournes Pond) Little Pond is separated from Vineyard Sound by a barrier beach, which was naturally breached and is now artificially maintained by jetties. The beach and the opening to the embayment is a very dynamic geomorphic feature due to the influence of littoral transport processes. Over the past century the Little Pond inlet has experienced varying degrees of occlusion thereby affecting tidal exchange and circulation within the salt pond. By example, Bournes Pond became very restricted and finally completely isolated from Vineyard Sound waters in the late 1970's/early 1980's and was re-opened with a fixed inlet in mid 1980's. Currently, the inlet to Little Pond is periodically dredged to maintain the small tidal channel into the pond.

Similar to the Great, Green and Bournes Pond embayment systems, Little Pond is a shallow coastal salt pond located within a glacial outwash plain, the Mashpee Pitted Plain, consisting of material deposited after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~18,000 years ago. The outwash material is highly permeable and varies in composition from well sorted medium sands to course pebble sands and gravels extending down to about 17 m below mean sea level (Millham and Howes, 1993). Depth to bedrock is approximately 75 m below sea level (O'Hara & Oldale 1987). The permeable nature of the upper outwash results in low rates of direct rainwater run-off and most freshwater inflow to these estuarine systems is via groundwater discharge or groundwater fed surface water flow (e.g. stream to the head of Little Pond, Coonamessett River to Great Pond, Backus River to Green Pond etc.). Little Pond acts as a mixing zone for terrestrial freshwater inflow and saline tidal flow from Vineyard sound, however, the salinity characteristics of the salt pond varies with the volume of freshwater inflow as well as the effectiveness of tidal exchange with Vineyard Sound.

The Little Pond Estuary is a relatively recent ecological system. After the formation of the valley which holds the present Little Pond about 18,000 years B.P. until 4,500-3,000 years B.P., sea level was too low to support tidal exchange. Only over the past 4,500-3,000 has "Little Pond" existed as an estuarine system. After the initial entry of tidal waters to the valley, coastal processes formed the barrier beach and restricted inlet. As sea level has risen, the estuary has migrated inland to its present location.

The habitat quality of Little Pond is linked to the level of tidal flushing through its inlet to Vineyard Sound. Although the salt pond embayment systems of Falmouth bounding Vineyard Sound exhibit slightly different hydrologic characteristics (river dominated versus tidally dominated), the tidal forcing for these systems is generated from Vineyard Sound. Vineyard Sound, adjacent the barrier beach separating the Little Pond embayment system from the ocean, exhibits a moderate to low tide range, with a mean range of about 0.5 m at the southern inlet of the pond. Since the water elevation difference between Vineyard Sound and Little Pond 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). The inlet is affected significantly by longshore sand transport (west to east), where shoaling can impede hydrodynamic exchange. The inlet is presently armored with jetties and the channel armored with riprap and features a significant scour channel between these structures. Little Pond is long and narrow, with length-to-breadth ratio of approximately 8-to-1. Little Pond, for example, is nearly 1400 meters in length with a width of approximately 170 meters. Depths within the deeper scour channel at the inlet are approximately 2 meters, with the upper (northern) reaches of the Pond frequently less than 1 meter deep.

The present Little Pond inlet was not formed naturally, but relocated as a result of road construction (See Section V). The inlet was located to the east of its present location in maps from 1846, 1880 and 1893. Between 1880 and 1893 a road was constructed along the length of the barrier beach to the inlet, but there is no evidence of jetties in this region at that time. By 1936 the road on the barrier had been extended fully across to the Maravista Peninsula, probably in the 1920's, but tidal exchange was maintained. However, Millham (1993) reconstructing recent changes to Little Pond, indicated that after the roadway extension, tidal flushing was restricted. He stated that "Anecdotal evidence of the fresh or brackish water conditions in the pond before 1964 were offered by several local individuals who fished (trout and perch) and trapped the pond prior to 1964. The reports give some details of the control of the pond level by the old culvert, which was apparently set at a level to prevent the entrance of seawater during most high tides. However, during spring tides and under wind forcing conditions, salt water from Nantucket Sound did flow into the pond inlet". In 1964 the Town of Falmouth placed dual culverts and jetties in the present location and pond tidal exchange was enhanced (Millham 1993). These structures were again replaced in February 1995 to repair damage from Hurricanes and to provide more consistent tidal exchange, at which time the inlet cross-section was altered to be wider to allow greater water exchange with Vineyard Sound under open channel conditions (from Sound to Pond). The tidal exchange of waters from Little Pond with Vineyard Sound water is driven by a relatively small tidal difference between the pond and the sound (<0.5m). It appears that in recent times that Little Pond ecological systems have varied significantly as a result of varying tidal exchange due to inlet changes.

At present, sedimentation of the "channel" between Vineyard Sound and the lower basin of Little Pond, as well as in the culvert, is resulting in less than maximum tidal exchange. 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. MEP tidal data indicate significant tidal damping through the Little Pond inlet. Due to significant tidal damping through the overall Little Pond System, enhancements in system flushing times may be realized if future modifications to the inlet are considered as a part of nutrient management.

Little Pond is a shallow mesotrophic (moderately nutrient impacted) to eutrophic (nutrient-rich) coastal pond on the southern coast of Falmouth. For the MEP analysis, the Little Pond system was analyzed individually as a stand-alone system. Similar to other salt ponds in Falmouth (e.g. Great/Perch Pond, Green Pond, and Bournes Pond) Little Pond is an estuary with focused freshwater input at the headwaters and tidal exchange of marine waters from Vineyard Sound (tide range of approximately 0.5 m) at its southern inlet. The Little Pond Estuarine System was partitioned into two regions: an 1) upper narrow portion, considered the head of the estuary, which receive discharge from the headwater stream and 2) a lower portion that includes the main basin and the tidal inlet (see Figure I-3). Little Pond is a true estuary, acting as the mixing zone of terrestrial freshwater inflow and saline tidal waters from Vineyard Sound. Salinity in the pond ranges from approximately 30 ppt at the Vineyard Sound inlet to less than 10 ppt at the northern end.

Figure I-3. Partitioning of the Little Pond embayment system relative to basin boundary volumes.

Given the present hydrodynamic characteristics of the Little Pond embayment system, it appears that estuarine habitat quality is dependent on both the level of nutrient loading to embayment waters and tidal characteristics. In Little Pond, some enhancements to tidal flushing may be achieved via inlet or channel modification resulting in some mediation of the nutrient loading impacts from the Little Pond watershed. The details of such are a part of the MEP analysis described in this report.

Nitrogen loading to the Little Pond embayment system was determined relative to the upper and lower portions of the estuary as depicted in Figure I-3. The watershed to Little Pond is fully within the Town of Falmouth. Based upon land-use and the watershed being fully with in Falmouth, it appears that nitrogen management for Pond restoration may likely be more rapidly developed and implemented than otherwise. As management alternatives are being developed and evaluated, it is important to note that strong gradients define the nutrient characteristics of each pond and as such the associated habitat impacts. There is a strong gradient in nitrogen level and health in Little Pond, with highest nitrogen and lowest environmental health in the headwaters of the system and lowest nitrogen and greatest health near the inlet to Vineyard Sound. The upper reach of Little Pond is presently showing poor water quality and “Eutrophic” conditions. Eelgrass is absent from this region and periodic fish kills have been reported, resulting from oxygen depletion.

I.3 NITROGEN LOADING

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

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

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

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

Unfortunately, almost all of the estuarine reach within Little Pond is near or beyond its ability to assimilate additional nutrients without impacting ecological health. Nitrogen levels are elevated throughout the system and only sparse eelgrass has been observed for over a decade in the lower basin of Little Pond. The result is that nitrogen management of the primary sub-embayments is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed "eutrophication" and when the nutrient loading is primarily from human activities, "cultural eutrophication". Although the influence of human-induced changes has increased nitrogen loading to the system and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within Little Pond could potentially occur without man's 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" for water quality modeling of the Little Pond system; however, a thorough understanding of estuarine circulation is required to accurately determine nitrogen concentrations within the system. Therefore, water quality modeling of tidally influenced estuaries must include a thorough evaluation of the hydrodynamics of the estuarine system. Estuarine hydrodynamics control a variety of coastal processes including tidal flushing, pollutant dispersion, tidal currents, sedimentation, erosion, and water levels. Numerical models provide a cost-effective method for evaluating tidal hydrodynamics since they require limited data collection and may be utilized to numerically assess a range of management alternatives. Once the hydrodynamics of an estuary system are understood, computations regarding the related coastal processes become relatively straightforward extensions to the hydrodynamic modeling. The spread of pollutants may be analyzed from tidal current information developed by the numerical models.

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

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all nitrogen entering Falmouth's salt ponds is transported by freshwater, predominantly groundwater, either through direct discharge or after discharging to streams flowing to estuarine waters. Concentrations of total nitrogen and salinity of Vineyard Sound source waters and throughout the Little Pond system was taken from the Falmouth PondWatch Monitoring Program (supported by the Town of Falmouth and associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the system were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Little Pond system for the Town of Falmouth. A review of existing water quality studies is provided (Section II). The development of the watershed delineations and associated detailed land use analysis for watershed based nitrogen loading to the coastal system is described in Sections III and IV. In addition, nitrogen input parameters to the water quality model are described. Since benthic flux of nitrogen from bottom sediments is a critical (but often overlooked) component of nitrogen loading to shallow estuarine systems, determination of the site-specific magnitude of this component also was performed (Section IV). Nitrogen loads from the watershed and sub-watershed surrounding the estuary were derived from Cape Cod Commission data and offshore water column nitrogen values were derived from an analysis of monitoring stations in Vineyard 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 each embayment was performed that included a review of existing water quality information, temporal changes in eelgrass distribution, dissolved oxygen records and the results of a benthic infaunal animal analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed for restoration of each embayment in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined threshold for restoration in a given salt pond. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for the Little Pond system. Finally, analyses of the Little Pond system was 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 (Section IX).

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

In most marine and estuarine systems, such as the Little Pond embayment system in Falmouth, 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 eutrophication management approaches via the reduction of nitrogen loads has also 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. 2003).

Until recently, these tools for predicting loads and concentrations tended to be generic in nature, and overlooked some of the site-specific characteristics associated with a 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 Little Pond System.

The Town of Falmouth, Massachusetts, has long recognized the potential threat of nutrient over-enrichment of its coastal salt ponds and embayments. In the mid-1980's the Town enacted an innovative Nutrient Overlay By-law that tied watershed development to water quality within the adjacent embayment. The goal was to keep nitrogen concentrations in the receiving systems below thresholds that were projected to cause water quality shifts. A water quality monitoring program, Falmouth PondWatch, was established to provide on-going nutrient related embayment health information in support of the By-law. Little Pond was among the first three Ponds (Oyster Pond, Little Pond, Green Pond) to undergo water quality monitoring in the Town of Falmouth. These approaches were primarily initiated for planning as development within coastal watersheds progressed. Falmouth's Planning Department has continued to enhance its tools for gauging future nutrient effects from changing land-uses. The GIS database used in the present study is part of that continuing effort. Unfortunately, monitoring has documented that most regions within the Town's coastal ponds are currently showing water quality declines and are beyond the limits set by the By-law.

Data generated by the Falmouth PondWatch program has yielded clear indications of impairment to the Little Pond system and has assisted the Town in the development of initial management options for improving the ecological health of the system. Specifically, the PondWatch program water quality database for Little Pond assisted the Town of Falmouth in the design and permitting of the redesign of the inlet between Little Pond and Vineyard Sound thereby allowing increased flushing of the overall system, as the channel from Vineyard Sound to the Little Pond lower basin was opened (dredged). The new inlet to Little Pond was completed in February of 1995. The inlet cross-section was widened to allow greater water exchange with the small tides in Vineyard Sound (<0.5m), however, the total cross-section was only slightly increased to prevent the potential of increased flooding of upland areas during storm tides. Despite initial management efforts for improving the habitat health of Little Pond, as has been observed in other Falmouth salt ponds in the vicinity of Little Pond (e.g. Great Pond, Green Pond, Bournes Pond), Little Pond exhibits high nutrient levels and periodic oxygen depletions in its upper reaches and water column nutrient levels exceed those historically specified by the Falmouth Nutrient Overlay Bylaw.

Based on the summary of results from the 1994 and 1995 water quality sampling seasons Little Pond shows characteristics of a eutrophic coastal salt pond with high nutrient loads. At the time of sampling in 1994 and 1995 nitrogen levels in the pond consistently exceeded limits set by the above mentioned Nutrient Bylaw. The high nitrogen levels reported within the system were primarily responsible for periodic and often severe low oxygen events experienced in the pond. Additionally, the effect of regular oxygen stress within Little Pond resulted in impoverished or non-existent benthic animal communities by the end of both the 1994 and 1995 summer seasons. Eelgrass beds were reported to have all but disappeared reflecting limited watercolumn transparency due to increased phytoplankton production. As well, macroalgal blooms leading to floating mats on the water surface have led to further declines in oxygen conditions. These conditions have persisted in the intervening years and accurately describe the present water quality of Little Pond.

At the time of the 1995 summer sampling season, the enlarged, modified inlet to Little Pond had only been in place for approximately five months and the strong gradient in nitrogen, oxygen and turbidity from the inlet to the headwaters of the pond remained similar to previous years. As of the summer 1995 sampling period, there appeared to be a reduction in macro-algae in the pond but water clarity and nutrient levels were similar to 1994 levels. Similarly, although PondWatch sampling suggested that oxygen levels may have been improving slightly, an oxygen mooring placed at a central location in the pond during what is generally considered the lowest oxygen period indicated short term anoxia. The observed oxygen levels measured by the mooring in 1995 were similar to oxygen levels obtained by moored oxygen instruments deployed in Little Pond in previous years. According to the of results from the 1994 and 1995 water quality sampling seasons, the new inlet had not created a significant change in the nutrient water quality of the pond. At the time, this was not an unexpected finding given the short interval between installation of the widened inlet and the 1995 summer field sampling. More importantly, dredging of the channel through the flood tidal delta had not yet been undertaken.

Concurrent with the water quality monitoring of Little Pond as initiated by the Falmouth PondWatch program, estuarine research was being conducted on the system yielding an additional level of understanding regarding primary production within this salt pond system, the ponds oxygen status and freshwater flows (groundwater and surfacewater) to the pond. As early as the summer of 1989 a study was undertaken in Little Pond to ascertain the effects of sampling frequency on measurements of seasonal primary production and oxygen status in near-shore coastal ecosystems (Taylor and Howes, 1994). During the June to December 1989 measurement period oxygen levels near the sediment surface (~15 cm) showed large and high frequency fluctuations with 5 to 8 sustained periods of hypoxia (DO < 50% atmospheric equilibrium. Additionally, oxygen levels in the summer of 1989 showed reduced levels during a week long event in mid-June, a 3 week event between July and August, and several day events in September, October and November. The most sustained oxygen depletion was correlated with a large six week bloom occurring in July and August of 1989. During and after the bloom the physical and biological processes of oxygen re-supply to the pond water column were barely able to keep up with the oxygen demand of the environment. Though the study did not specifically and directly attribute the low oxygen conditions to nutrient loading, it clearly revealed that the Little Pond system was experiencing periods of oxygen stress indicating a level of habitat impairment.

Additional research was undertaken within the Little Pond system towards refining patterns of freshwater flow as both a groundwater and surface water input and the effect of flow patterns on nutrient transport to the system. In short, groundwater was shown to dominate the freshwater budget to Little Pond, accounting for greater than 95 percent of the total annual input (Millham and Howes, 1994). Furthermore, the groundwater portion of the freshwater budget was nearly equally partitioned between direct groundwater seepage to the embayment waters and groundwater seepage to a stream with final discharge as surfacewater flow. As is common with many embayments on Cape Cod, freshwater inputs showed a rapid decrease toward the mouth of the estuary and greater than 80 percent of the freshwater input being introduced to the Little Pond system within the upper half. This characteristic will ultimately have management implications relative to the effectiveness of nutrient load removals as groundwater/surfacewater discharges focus the introduction of watershed generated nutrients in the head of the embayment system.

As part of on-going research and engineering efforts related to Little Pond, the geologic history of the pond was determined as was the recent history of shoreline change (See Section V). The short and long term trends in coastal processes as relate to the ecological health of Little Pond set an important background for the present restoration and management of this system. In addition, the previous studies of Little Pond dissolved oxygen status, groundwater and surface water hydrology provide an important basis for comparison to recent MEP measurements and help to establish the degree of stability of current conditions. In addition, for the MEP modeling analysis, the data from the previous studies were evaluated relative to the needs of the Linked Watershed-Embayment Model.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

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

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the Little Pond system under evaluation by the Project Team. The Little Pond estuarine system is composed of a simple estuary, originating from sea level flooding of a linear valley without tributary branches. Further watershed modeling was undertaken to sub-divide the overall watershed to the Little Pond system into functional sub-units based upon: (a) defining inputs from contributing areas to each major portion within the embayment system, (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to evaluate the contributing areas to public water supply wells in the Sagamore flow cell on Cape Cod. Model assumptions for calibration were matched to surface water inputs and flows from current (2002 to 2003) stream gage information.

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

III.2 MODEL DESCRIPTION

Contributing areas to the Little Pond system and local freshwater bodies were delineated using a regional model of the Sagamore Lens (Walter and Whealan, 2005). The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al.*, 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 1994), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the main basins of the Little Pond system and also to determine portions of recharged water that may flow through freshwater 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. Layers 1-7 are stacked above NGVD 29 and layers 8 to 20 extend below. Layer 18 has a thickness of 40 feet and layer 19 extends to 240 feet below sea level. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics. The rewetting capabilities of MODFLOW-2000, which allows drying and rewetting of model cells, was used to simulate the top of the water table, which varies in elevation depending on the location in the Lens. In the portion of the Sagamore Lens in which the Little Pond system resides, water elevations are generally less than +40 ft and, therefore, over much of the study area the uppermost layers of the model are inactive.

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. Little Pond is situated on the westernmost edge of the very-coarse grained Mashpee Pitted Plain deposits (Masterson, et al., 1996). Lithologic data used to determine hydraulic conductivities used in the groundwater model were obtained from a variety of sources including well logs from USGS, local Town records and data from previous investigations. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water level and streamflow data collected in May 2002.

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

III.3 LITTLE POND CONTRIBUTORY AREAS

Revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the Little Pond embayment system (Figure III-1). Model outputs of MEP watershed boundaries were “smoothed” to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the pond and coastal shorelines, (c) to include water table data in the lower regions of the watersheds near the coast (as available), and (d) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. Overall, five sub-watershed areas, including one freshwater pond, were delineated within the watershed to the Little Pond embayment system. The recharge area to Morse Pond is shown in Figure III-1 in order to help clarify the shape of the Little Pond watershed, but Morse Pond is not part of the Little Pond watershed. The MEP delineation includes 10 yr time of travel boundaries.

Table III-1 provides the daily discharge volumes for various watersheds as calculated by the groundwater model; these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models and for comparison to measured surface water discharges. The overall estimated groundwater flow into Little Pond from the MEP delineated watershed is 4,869 m3/d. This flow is nearly identical to the 4,800 m3/d average (4,100 to 5,900 m3/d range) measured by Millham and Howes (1994b) from a variety of direct measurement methods.

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

The MEP watershed area for the Little Pond as a whole is less than 0.1% (0.5 acres) larger than the 1995 CCC delineation. However, the area within the watershed is slightly different. This difference is attributable to a slightly different location of the Buzzards Bay/Vineyard Sound groundwater divide and more southern boundary location for the main portion of the watershed. The change in the southern boundary is largely due to a better consideration of the fresh ponds in the groundwater modeling as compared to the CCC watershed (see Figure III-2). Subwatersheds were not delineated in the CCC watershed.

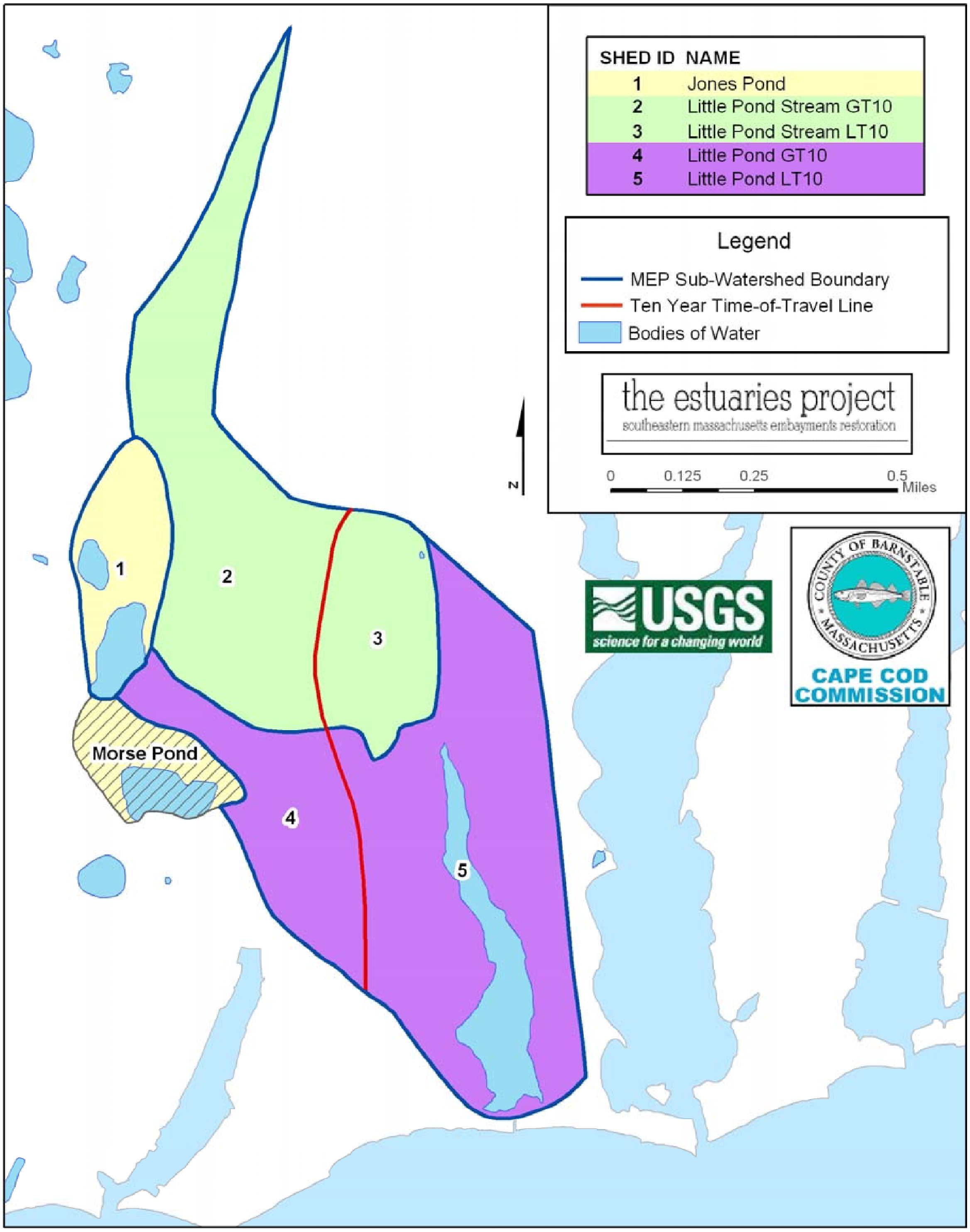


Figure III-1. Watershed and sub-watershed delineations for the Little Pond estuary system. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a “10” in the watershed names (above). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI). Morse Pond watershed is included in this figure to clarify the shape of the Little Pond watershed but flows to a separate system.

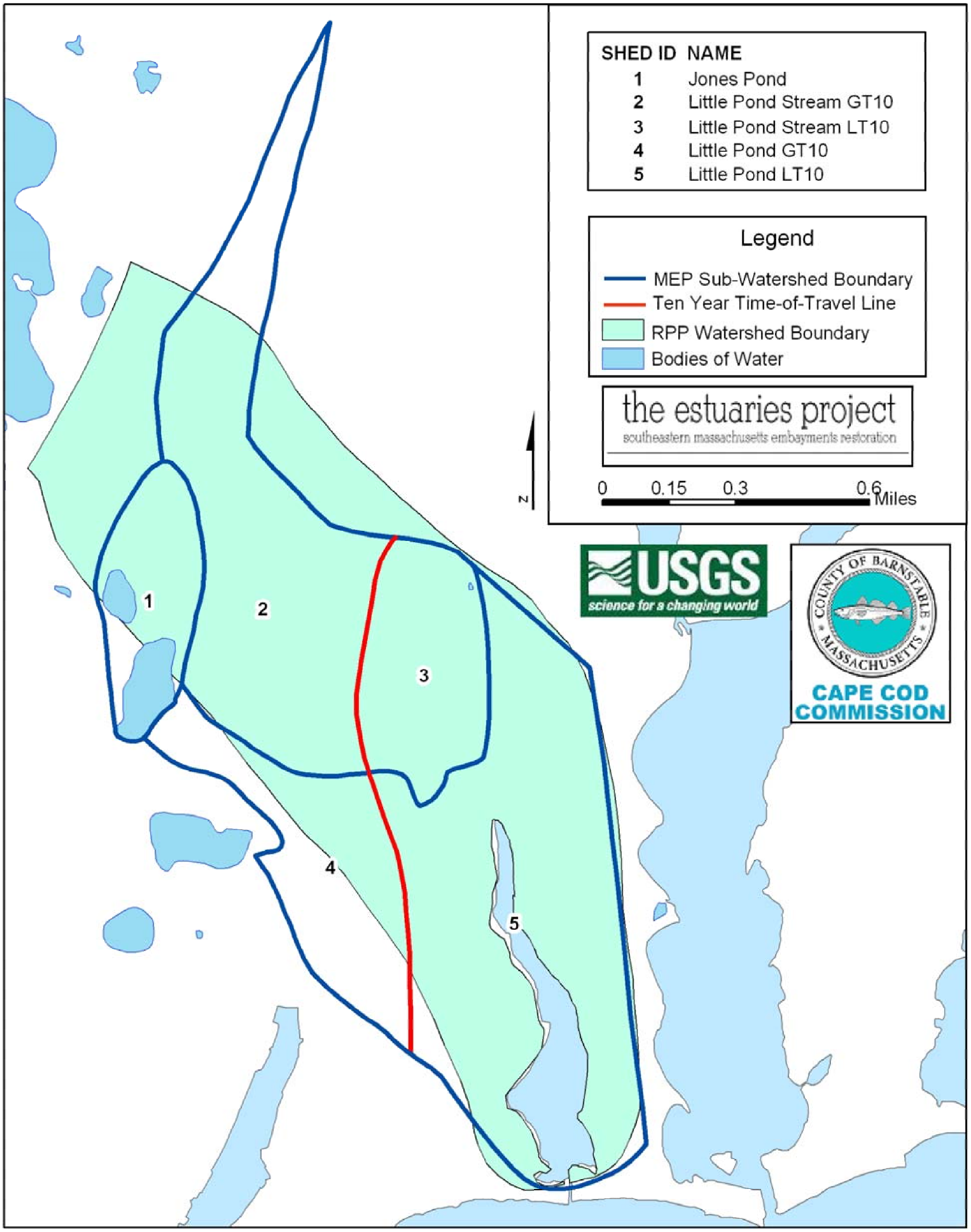


Figure III-2. Comparison of previous and current Little Pond watershed and subwatershed delineations.

Table III-1. Daily groundwater discharge from each of the sub-watersheds to the Little Pond Estuary, as determined from the USGS groundwater model.

Watershed	Watershed #	Discharge	
		ft^3/day	m^3/day
Jones Pond	1	7,439	211
Little Pond Stream GT10	2	56,822	1,609
Little Pond Stream LT10	3	22,662	642
Little Pond GT10	4	25,638	726
Little Pond LT10	5	59,379	1,682
Whole System		171,940	4,869

note: 58% of Jones Pond flow discharges into the Little Pond watershed; the remainder discharges outside of the watershed or to Morse Pond.

The evolution of the watershed delineations for Little Pond 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, 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 downgradient 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 Little Pond (Section V.1).

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

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) to the Little Pond embayment system based upon the sub-watershed delineations discussed above (Section III). The Little Pond watershed was sub-divided to define contributing areas to Little Pond Stream, Jones Pond, and the estuarine reach of Little Pond and further sub-divided into regions greater and less than 10 year groundwater travel time from the receiving estuary for a total of five sub-watersheds in all. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to the ponds and embayment.

The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges to the watershed have reached the embayment. This involves a temporal review of land use changes and the time of groundwater travel provided by the USGS watershed model. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel (LT10) and greater than 10 year time of travel (GT10) watersheds (Table IV-1), reviewing Falmouth land use development records and 2001 land use in the time of travel watersheds, and reviewing water quality modeling, it was determined that Little Pond is currently in balance with its watershed load. The bulk (66%) of the annual watershed nitrogen loading is situated within 10 years flow to Little Pond. In addition, the average year built of single family residences in regions greater than 10 year travel time is 1962 or 43 years ago; this suggests that the nitrogen from the majority of development in the greater than 10 year watersheds should currently be observed in Little Pond. Therefore, the distinction of less than 10 year and greater than 10 year time of travel regions within a sub-watershed (Figure III-1) was eliminated and the number of sub-watersheds was reduced to three. 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).

Table IV-1. Percentage of unattenuated nitrogen loads in less than 10 time of travel sub-watersheds to Little Pond. Note that within the >10 yr time of travel sub-watershed, the average age of homes is 43 years.

	LT10	GT10	TOTAL	%LT10
WATERSHED	kg/yr	kg/yr	kg/yr	
Little Pond Stream	2028	1694	3722	54%
Little Pond Salt	3774	1236	5010	75%
LITTLE POND TOTAL	5802	2930	8732	66%

In order to determine nitrogen loads from the watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes & Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon sub-watershed-specific land-uses and pre-determined nitrogen loading rates. For the Little Pond embayment system, the model used Falmouth -specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local watershed-specific data (such as parcel by parcel water use). Determination of the nitrogen loads required obtaining watershed-specific information regarding wastewater, fertilizers, runoff from impervious surfaces and atmospheric deposition. The primary regional factors were derived for southeastern Massachusetts from direct measurements. The resulting nitrogen loads represent the "potential" nitrogen load to each receiving embayment, since attenuation during transport has not yet been included.

Natural attenuation of nitrogen during transport from land-to-sea (Section IV.2) was determined based upon a site-specific study within the freshwater portion of Little Pond Stream. Attenuation during transport through the one fresh pond in the watershed (Jones Pond) was determined through comparison with other Cape Cod lake studies and other MEP Reports. Attenuation during transport through Jones Pond was assumed to equal 50% based on refined monitoring of selected Cape Cod lakes. Available historic data collected from individual fresh ponds reviewed in other MEP reviews has confirmed the appropriateness of this assumption. Internal nitrogen recycling was also determined throughout the tidal reaches of the Little Pond embayment; measurements were made to capture the spatial distribution of sediment nitrogen regeneration from the sediments to the overlying watercolumn. 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 Database Preparation

Estuaries Project technical staff obtained digital parcel and tax assessors data from the Town of Falmouth. Digital parcels and land use data are from 2001 and 2003, respectively, and were obtained from the Town of Falmouth Planning Department. The land use database contains traditional information regarding land use classifications (MADOR, 2002) plus additional information developed by the Town regarding impervious surfaces (building area, driveways, and parking area) on individual lots. The parcel coverages and assessors database were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS).

Figure IV-1 shows the land uses within the study area. Within the Little Pond study area, land use is classified as one of eight general land use types under the Massachusetts Assessors land uses classifications (MADOR, 2002). These eight land use categories and their respective MADOR numbers are:

1) Residential (100's less than 130)
2) Commercial (300's less than 380)
3) Industrial (400's)
4) Undeveloped (130, 131, 132, 390, 392)
5) Mixed use (013, 031)
6) Recreational/golf course (380)
7) Public service/government (including road rights-of-way) (900's)
8) Ponds

In the Little Pond sub-watersheds, the predominant land use based on area is either public service/government or residential, with these two land uses accounting for >60% of each sub-watershed. The percentage of residential area varies between 8% in the Jones Pond sub-watershed and 64% in the Little Pond estuarine reach sub-watershed ("Little Pond Salt"); 62% of the Jones Pond sub-watershed is classified as public service/government. However, over the entire Little Pond watershed, single family residences (land use code = 101) are clearly the predominant land-use, 86% on an areal basis. Only 10% of the area of both Little Pond sub-watersheds is presently classified as undeveloped (Figure IV-2). Half (53% or 16 acres) of the undeveloped land in the Little Pond Stream sub-watershed is classified as residentially developable, while 34% of the undeveloped land in the Little Pond estuarine reach sub-watershed is assigned the same classification; there is no land classified as undeveloped in the Jones Pond sub-watershed. Commercial properties in the Little Pond watershed are located mostly along Route 28/Teaticket Highway, which runs north to south through the main portion of the watershed and have the highest concentration in the Little Pond Stream sub-watershed (28% of the sub-watershed area).

In order to estimate wastewater flows within the study area, MEP staff obtained parcel by parcel water use information from the Town of Falmouth Water Department. This information included two years of water use information with the final reading in May 2003. Water use information was linked to the parcel and assessors data using GIS techniques. Water use for each parcel was converted to an annual volume for purposes of the nitrogen loading calculations. No wastewater treatment facilities (WWTFs) currently exist in the watershed, but 52 parcels are connected to the municipal sewer system. Forty of the 52 parcels are located in the sub-watershed to the estuarine reach of Little Pond (watershed #3). According to the town-supplied water use information, the 52 parcels have a combined average water use of 44,696 gallons per day.

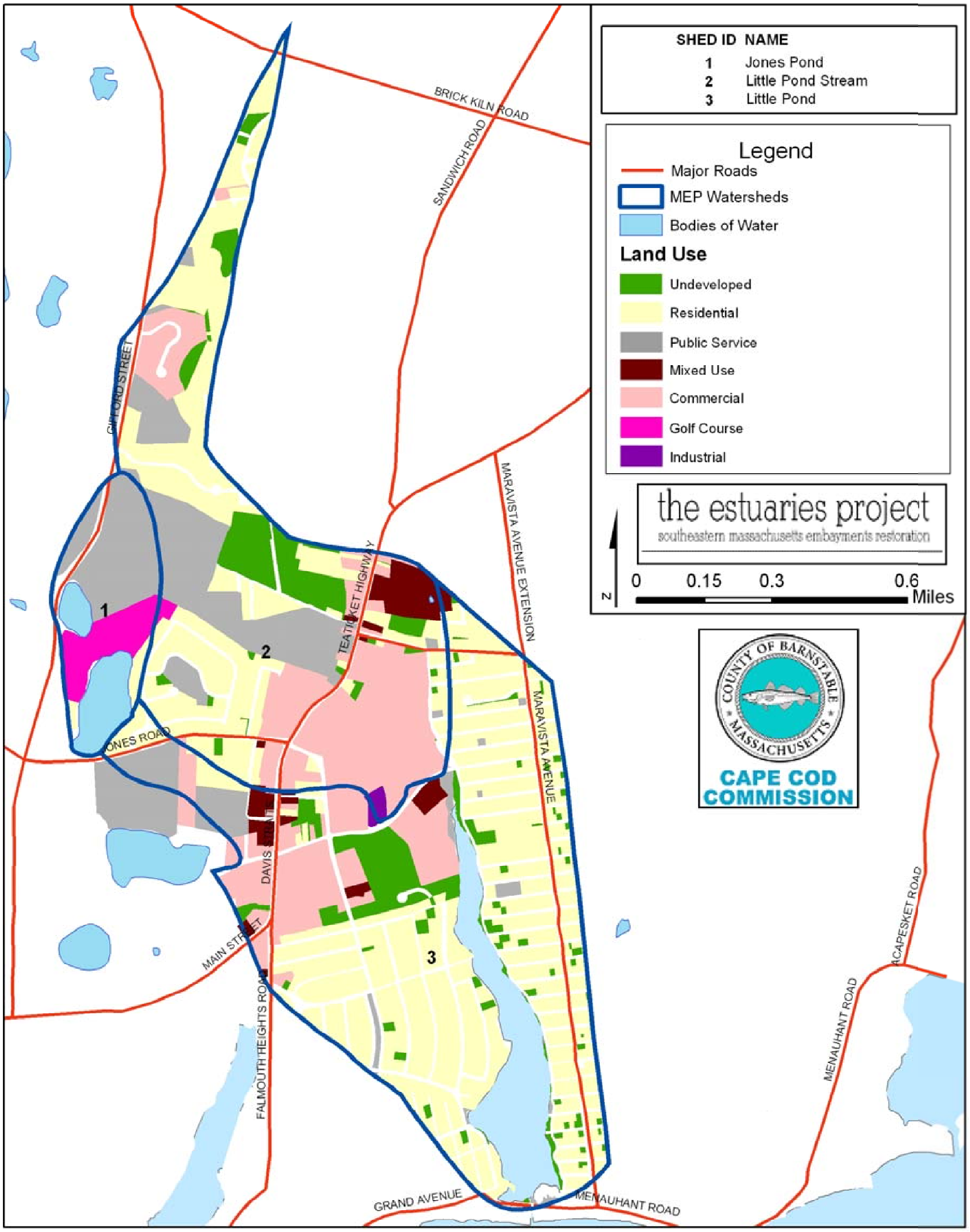


Figure IV-1. Land-use coverage in the Little Pond watershed. Watershed data encompasses portions of the Town of Falmouth and land use classifications are based on Town of Falmouth Planning Department records.

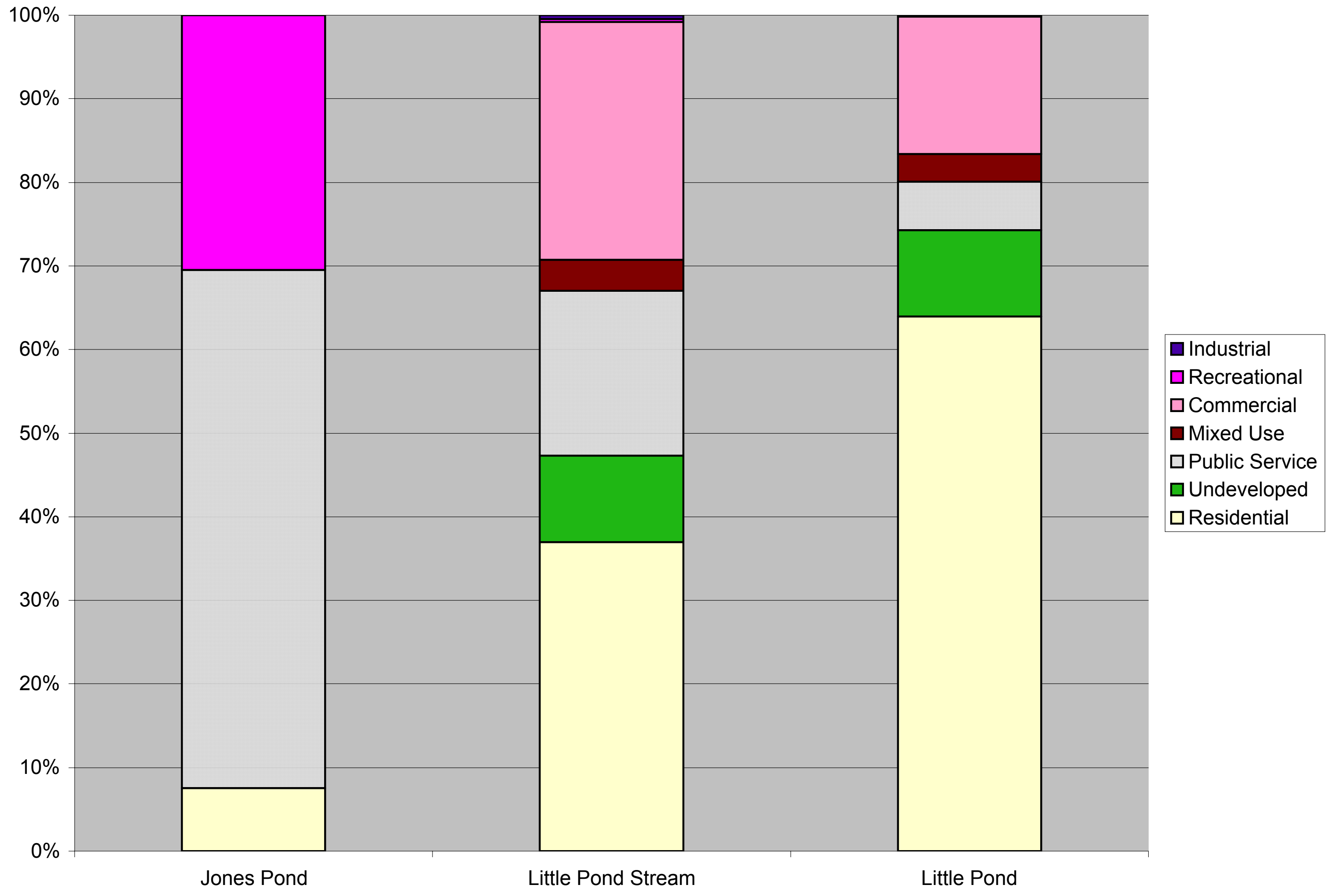


Figure IV-2. Distribution of land-uses within the major sub-watersheds to Little Pond.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

Similar to many other watershed nitrogen loading analyses, the Massachusetts Estuaries Project septic system nitrogen loading rate is fundamentally based upon a per Capita Nitrogen load to the receiving aquatic system. Specifically, the MEP septic system wastewater nitrogen loading is based upon directly measured septic system and per capita loads determined on Cape Cod or in similar geologic settings (Nelson et al. 1990, Weiskel & Howes 1991, 1992, Koppelman 1978, Frimpter et al. 1990, Brawley et al. 2000, Howes and Ramsey 2000, Costa et al. 2001). Variation in per capita nitrogen load has been found to be relatively small, with average annual per capita nitrogen loads generally between 1.9 to 2.3 kg person-yr^{-1}. However, given the seasonal shifts in occupancy in many of the watersheds throughout southeastern Massachusetts, census data yields accurate estimates of total population only in specific watersheds (see below). To correct for this uncertainty, the MEP employs a water-use approach. The water-use approach (Weiskel and Howes 1992) is applied on a parcel-by-parcel basis within a watershed, where usually an average of multiple years annual water meter data is linked to assessors parcel information using GIS techniques. The parcel specific water use data is converted to septic system nitrogen discharges (to the receiving aquatic systems) by adjusting for consumptive use (e.g. irrigation) and applying a wastewater nitrogen concentration. The water use approach focuses on the nitrogen load, which reaches the aquatic receptors down-gradient in the aquifer. All losses within the septic system are incorporated. For example, information developed at the DEP Alternative Septic System Test Center at the Massachusetts Military Reservation on Title 5 septic systems have shown nitrogen removals between 21% and 25%. Multi-year monitoring from the Test Center has revealed that nitrogen removal within the septic tank was small (1% to 3%), with most (20 to 22%) of the removal occurring within five feet of the soil adsorption system (Costa et al. 2001). Aquifer studies indicate that further nitrogen loss during aquifer transport is negligible (Robertson et al. 1991, DeSimone and Howes 1996).

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

The resulting nitrogen loads, based upon the above approach have been validated in a number of long and short term field studies where integrated measurements of nitrogen discharge from watersheds could be directly measured. For example, Weiskel and Howes (1991, 1992) conducted a detailed watershed/stream tube study that monitored septic systems, leaching fields and the transport of the nitrogen in groundwater to adjacent Buttermilk Bay. This monitoring resulted in estimated annual per capita nitrogen loads of 2.17 kg (as published) to 2.04 kg (if new attenuation information is included). The selected "effective N loading coefficient" also agrees with available watershed nitrogen loading analyses conducted on other Cape Cod estuaries. Aside from the concurrence observed between modeled and observed nitrogen concentrations in the estuary analyses completed under the MEP, analyses of other estuaries completed using this effective septic system nitrogen loading coefficient, the modeled loads also match observed concentrations in streams in the MEP region. Modeled and measured nitrogen loads were determined for a small sub-watershed to West Falmouth Harbor (Smith and Howes 2006) where a small stream drained the aquifer from a residential neighborhood. In this effort, the measured nitrogen discharge from the aquifer was within 5% of the modeled N load. A second evaluation was conducted by surveying nitrogen discharge to the Mashpee River in reaches with swept sand channels and in winter when nitrogen attenuation is minimal. The measured and observed loads showed a difference of less than 8%, easily attributable to the low rate of attenuation expected at that time of year and under the ecological situation (Samimy and Howes, unpublished data).

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

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

The independent validation of the water quality model (Section VI) and the reasonableness of the freshwater attenuation (Section IV.2) add additional weight to the nitrogen loading coefficients used in the MEP analyses and a variety of other MEP embayments. While the MEP septic system nitrogen load is the best estimate possible, to the extent that it may underestimate the nitrogen load from this source reaching receiving waters, the MEP approach provides a safety factor relative to other higher loads that are generally used in regulatory situations to add a safety factor for the protection of impacted resources. 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 directly proportional to the septic system nitrogen level, but rather is related to how the nitrogen load from septic systems compares to the nitrogen loads from all other sources that reach the estuary (i.e. attenuated loads).

While almost all of the developed parcels within the overall Little Pond watershed have corresponding water use accounts 11 (1% of all parcels) did not. Nine of the eleven parcels are residential, while two are commercial. All are assumed to utilize private wells for drinking water. In order to complete the nitrogen loading, the average water use from parcels with water use accounts (Table IV-2) was applied to the parcels assumed to be on private wells. Average water use was also used for determining nitrogen loads from new development determined in the buildout analyses.

Table IV-2. Average Water Use in Little Pond Watershed

Land Use	State Class Codes	Parcels in watershed with Water Use	Water Use (gallons per day)			
			Watershed Avg	Watershed Range	Town Avg	Town Range
Residential	101	1,143	102	0 to 894	153	0 to 79,618
Commercial	300 to 389	71	1,279	0 to 29,115	911	0 to 29,115
Industrial	400 to 439	0	0	none	1,229	0 to 13,583

Note: No industrial land uses with water use exist in the Little Pond watershed. All data for analysis supplied by the Town of Falmouth.

It should be noted that within each category of land use presented in Table IV-2 (i.e., residential, commercial, and industrial) are a variety of different types of uses. For example, included within the commercial category are low water users, like small offices or retail with one or two employees, and large water users, like small motels with a dozen or more rooms. As such, the ranges presented in Table IV-2 are rather broad. Nonetheless, the ranges employed in this analysis are very similar to those previously observed in other MEP watershed water use analyses conducted in the Towns of Chatham and Mashpee.

In order to provide an independent validation of the residential water use average within the study area, MEP staff reviewed US Census population values and compared that information with DEP design flows in Massachusetts on-site septic system regulations (*i.e.*, 310 CMR 15, Title 5). Title 5 assumes that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so based on this flow each person generates 55 gpd of wastewater. Average occupancy within the Town of Falmouth during the 2000 US Census was 2.36 people per household. If 2.36 were multiplied by 55 gpd, 130 gpd would be calculated as the average residential wastewater flow in Falmouth.

Average single-family residential flow (101 land use code) in the Little Pond watershed is 102 gpd, which would convert to 1.86 people per household. In previously completed MEP studies, average population and average water use based on an individual Title 5 flow have generally agreed fairly well. Since they did not in this watershed, MEP staff reviewed more refined US Census information and water use information for each parcel within the watershed. Besides reviewing data on town and state levels, the US Census also develops information for smaller areas (i.e., tracts and blocks). The Little Pond watershed straddles the boundary between Census blocks 147 and 148; block 147 has an average household size of 2.22, while block 148 has an average household size of 2.05. Both of these household sizes are less than the town average and support a lower flow in the Little Pond watershed and block 148 is only 10% higher than the estimate based upon the water use data.

As an additional check on the measured water uses, MEP staff also reviewed the average and range of residential flows within each sub-watershed. This review found that although the range of residential flows (minimum and maximum) is smaller in this watershed than for the whole town, there is a greater concentration of low water uses (especially in the Little Pond sub-watershed #3) than for the rest of Falmouth. Since no information was available to suggest that the water uses in the overall watershed are inappropriate, MEP staff decided to use the watershed-specific average water use for the nine developed residential parcels (~1% of the single family residences) without water use and for the 46 additional parcels included in the buildout analysis.

There are 69 properties classified as commercial developments in the overall Little Pond watershed and 9 classified as developable commercial properties (390 land use code). Of the 69 developed properties, 35 are connected to the town's municipal sewer system. All nine of the developable commercial properties are located within the North Davis Straits Service Area identified in the town's Wastewater Facilities Plan (Stearns and Wheler, 2001). Because the town planning department has developed building footprint areas, MEP staff were able to develop average water use per 1,000 ft2 of commercial building (141 gpd/1,000 ft2) and determine the percentage of each commercial lot that is occupied by a building (18%). This information was used to develop estimates for the two developed commercial properties without water use and for future water use for the nine developable parcels. However, because the developable commercial properties are within the North Davis Straits Service Area and this area was approved by DEP and the Cape Cod Commission for 0.2 million gallons per day of flow to the sewer system, it was assumed that wastewater from these future developments would be transported outside of the Little Pond watershed and therefore has not been included in the presented build-out nitrogen load. However, if the wastewater nitrogen load was included in the buildout assessment, these parcels would increase the unattenuated and attenuated load to Little Pond by approximately 2%. Since loads associated with impervious surfaces and fertilizer applications from these properties, would contribute to loading of Little Pond after development, they have been are included in the buildout loads.

Nitrogen Loading Input Factors: Residential Lawns

In most southeastern Massachusetts watersheds, nitrogen applied to the land to fertilize residential lawns is the second major source of nitrogen to receiving coastal waters after wastewater associated nitrogen discharges. However, residential lawn fertilizer use has rarely been directly measured in previous watershed-based nitrogen loading investigations. Instead, lawn fertilizer nitrogen loads have been estimated based upon a number of assumptions: a) each household applies fertilizer, b) cumulative annual applications are 3 pounds per 1,000 sq. ft., c) each lawn is 5000 sq. ft., and d) only 25% of the nitrogen applied reaches the groundwater (leaching rate). Because many of these assumptions had not been rigorously reviewed in over a decade, the MEP Technical Staff undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the Watershed Nitrogen Loading Sub-Model.

The initial effort was to determine nitrogen fertilization rates for residential lawns in the Towns of Falmouth, Mashpee and Barnstable. The assessment accounted for proximity to fresh ponds and embayments. Based upon ~300 interviews and over 2,000 site surveys, a number of findings emerged: 1) average residential lawn area is ~5000 sq. ft., 2) half of the residences did not apply lawn fertilizer, and 3) the weighted average application rate was 1.44 applications per year, rather than the 4 applications per year recommended on the fertilizer bags. Integrating the average residential fertilizer application rate with a leaching rate of 20%

results in a fertilizer contribution of N to groundwater of 1.08 lb N per residential lawn for use in the nitrogen loading calculations. It is likely that this still represents a conservative estimate of nitrogen load from residential lawns. However, it should be noted that professionally maintained lawns were found to have the higher rate of fertilization application and hence higher estimated loss to groundwater of 3 lb/lawn/yr.

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for atmospheric deposition, impervious surfaces and natural areas are from the MEP Embayment Modeling Evaluation and Sensitivity Report (Howes and Ramsey 2001). The factors are similar to those utilized by the Cape Cod Commission's Nitrogen Loading Technical Bulletin (Eichner and Cambareri, 1992) and Massachusetts DEP's Nitrogen Loading Computer Model Guidance (1999). The recharge rate for natural areas and lawn areas is the same as utilized in the MEP-USGS groundwater modeling effort (Section III). Factors used in the nitrogen loading analysis for the Little Pond watershed are listed in Table IV-3.

Table IV-3. Primary Nitrogen Loading Factors used in Little Pond MEP analysis. General factors are from MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Falmouth data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

Nitrogen Concentrations:	mg/l
Wastewater (Septic System effluent)	35
Wastewater (pre-discharge from aquifer)	26.25
Road Run-off	1.5
Roof Run-off	0.75
Direct Precipitation on Embayments and Ponds	1.09
Natural Area Recharge	0.072
Wastewater Coefficient	23.63
Fertilizer:	
Average Residential Lawn Size (ft^2)*	5,000
Residential Watershed Nitrogen Rate (lbs/lawn)*	1.08
Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined by site-specific information	

Recharge Rates:	in/yr
Impervious Surfaces	40
Natural and Lawn Areas	27.25
Water Use/Wastewater:	
For Single Family Residence Parcels wo/water accounts and Buildout additions:	102 gpd
For Commercial Properties wo/water accounts and Buildout additions:	141 per 1,000 ft^2 of building
For Parcels w/water accounts:	Measured annual water use

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 separately and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each sub-watershed and the sum of the area of the parcels within each watershed. The resulting "parcelized" watersheds are shown in Figure IV-3. This review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (golf courses, condominiums, etc.) were also assigned at this stage. In addition, the golf course superintendent for the one course in the watershed was contacted to provide fertilizer application rates, but these were not available; instead average application rates from fertilization rates at three other golf courses in Falmouth were used . It should be noted that small shifts in nitrogen loading due to the above assignment procedure, has a negligible effect on the total nitrogen loading to the Little Pond estuary. The effort was undertaken to better define the sub-embayment loads to enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

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

The three individual sub-watershed assessments were then integrated to generate nitrogen loading tables relating to the each of the individual estuaries and their major components: Little Pond Salt, Little Pond Stream, and Jones Pond. The sub-embayments represent the functional embayment units for the Linked Watershed-Embayment Model's water quality component.

For management purposes, the aggregated sub-embayment watershed nitrogen loads are partitioned by the major types of nitrogen sources in order to focus development of nitrogen management alternatives. Within the Little Pond system the major types of nitrogen loads are: wastewater, fertilizer, impervious surfaces, direct atmospheric deposition to ponds and estuarine waters, and recharge within natural areas (Table IV-4). The output of the watershed nitrogen loading model is the annual mass (kilograms) of nitrogen added to the contributing area of component sub-embayments (and freshwater ponds and streams), by each land use category (Figures IV-4 a-c). This annual watershed nitrogen input is then adjusted for natural nitrogen attenuation during transport to the estuarine system before use in the embayment water quality sub-model. Natural attenuation within the Little Pond Stream is also directly measured (Section IV.2) and compared to the unattenuated and attenuated annual watershed nitrogen load from the land-use sub-model.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally formed from glacial kettle hole depressions that penetrate the groundwater table revealing what some call "windows on the aquifer." The typical hydrologic condition of these kettle ponds is to have groundwater flowing in along the upgradient shore and pond water recharging back to the aquifer along the downgradient shore. In some cases, outflow from the pond may be via a natural stream or a channel dug for propagation of herring. Additional freshwater inflow occurs through direct atmospheric deposition and surface water flows. The residence time of water in these systems is related primarily to the rate of inflow and the volume of the pond basin. Nitrogen within the ponds is available to the pond ecosystem, which can produce significant nitrogen removal through denitrification and burial of refractory forms. The general result is a reduction in the mass of nitrogen flowing back into the groundwater system along the downgradient side of the pond or through a stream outlet and resulting in a reduction in the eventual discharge into the downgradient embayment. This removal or attenuation of nitrogen by natural systems is termed "natural attenuation" and is a fundamental part of the functioning of the linked watershed-estuarine system. The Nitrogen Load Summary Table IV-4 includes both the unattenuated (nitrogen load discharged to each sub-watershed) and attenuated nitrogen loads. Based upon direct measurements of ponds and rivers and similar studies on Cape Cod (see below), nitrogen attenuation in the ponds was set conservatively at 50% in the Linked Watershed-Embayment Model.

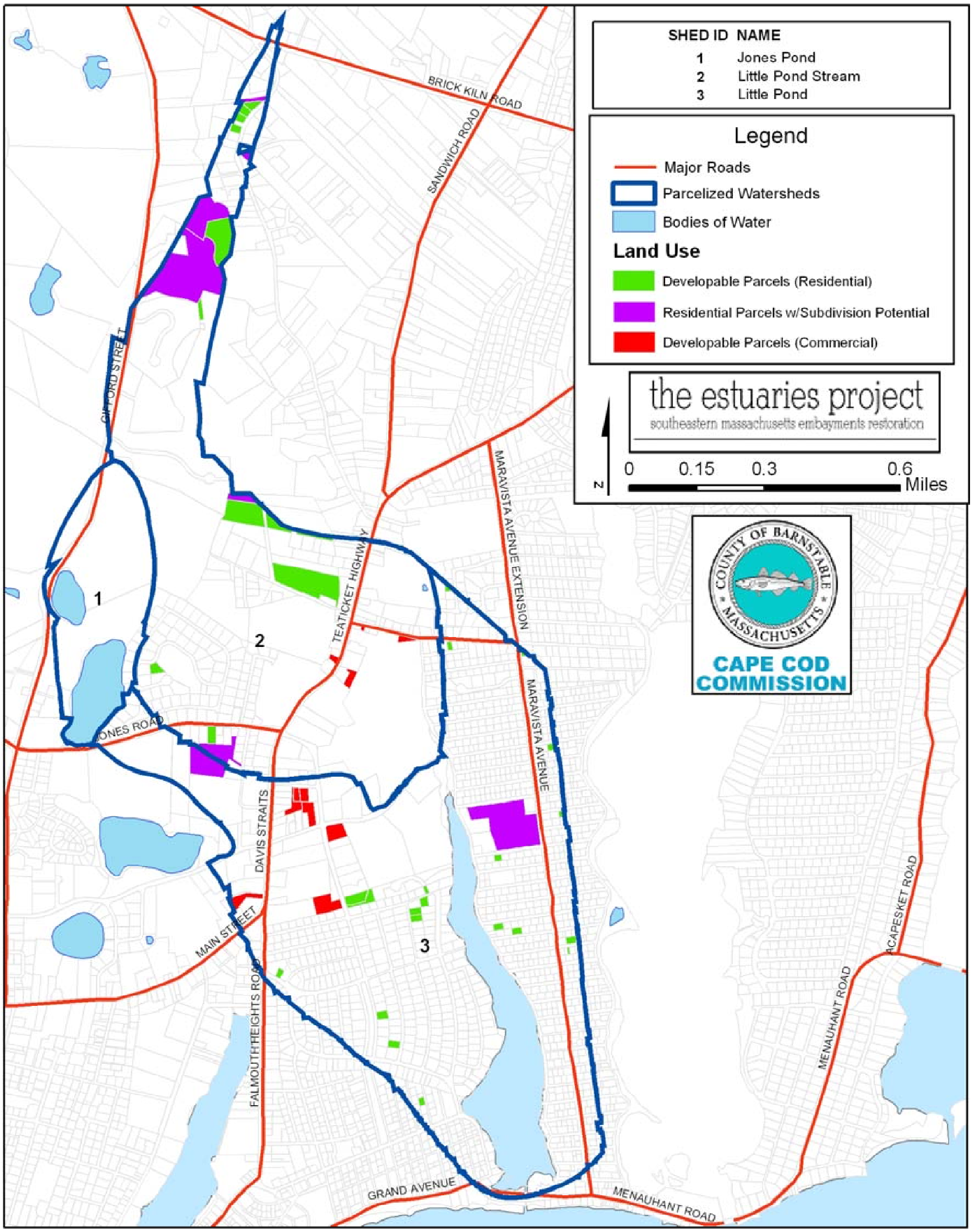
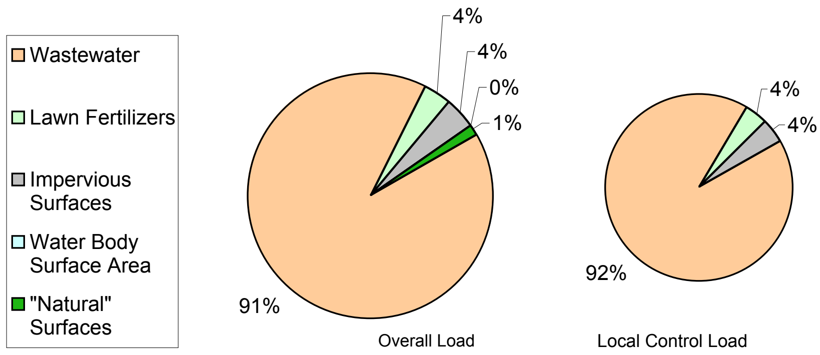


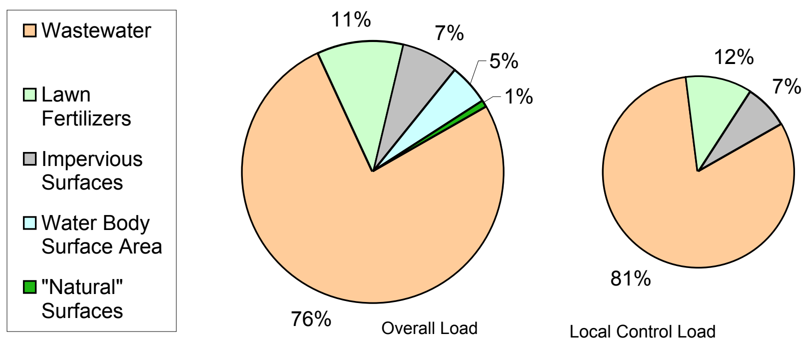
Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Little Pond watersheds. Note the limited area remaining for new development.

Table IV-4. Nitrogen loads discharged to the watershed of the Little Pond Estuarine System and for the direct groundwater discharge watershed to the estuarine reach (Little Pond Salt) and the Little Pond Stream. Attenuation of system nitrogen loads occurs as nitrogen moves through Jones Pond and the Little Pond stream during transport to the estuary. Build-out nitrogen loads are based upon amount of developable land, zoning and projections of development. Septic refers to on-site septic system treatment of wastewater from residential and commercial properties. Atmospheric deposition to water bodies is comprised of 46 kg yr^{-1} to freshwater ponds and 213 kg yr^{-1} directly to the estuarine waters of Little Pond.

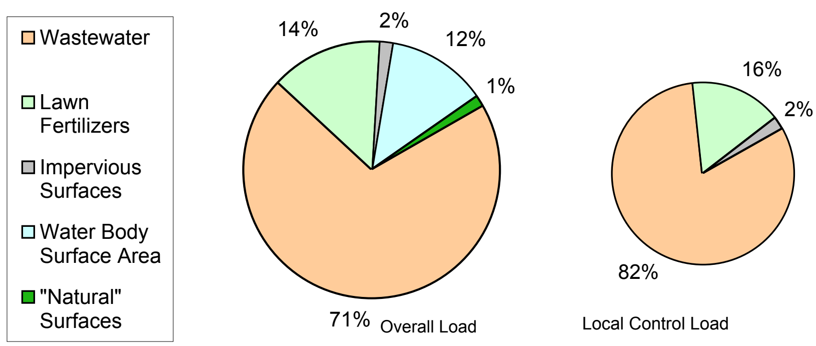
		Little Pond N Loads by Input:						% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Septic	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Little Pond System		7310	699	510	259	93	208		8870		7569	9078		7754
Little Pond Salt	4 + Jones Pond	3933	559	359	259	42	131		5152		4966	5283		5097
Jones Pond	1	261	53	7	46	5	0	58%	372	50%	186	372	50%	186
Little Pond Stream	3	3377	141	151	0	51	76		3719	30%	2603	3795	30%	2657



a. Little Pond Stream



b. Little Pond Salt



c. Jones Pond

Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) to the (a) Little Pond Salt sub-watershed, (b) Little Pond Stream sub-watershed, and (c) Jones Pond. “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 (i.e. no atmospheric deposition).

MEP analyses typically review available pond water quality data to check this standard attenuation assumption. Nitrogen attenuation was estimated directly, based on watershed nitrogen loading rates to the ponds coupled with pond residence time and nitrogen concentrations. Generally, this review begins by reviewing data collected by volunteers through the annual water quality snapshots of the Cape Cod Pond and Lake Stewardship (PALS) program. The PALS snapshots are a collaborative project between Cape Cod Commission and the Coastal Systems Program at SMAST. The samples for the PALS snapshot are collected between mid-August and the end of September and at least one pond in each town is sampled. These annual snapshots began in 2001. Citizen volunteers collect dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at selected depths. Water samples are analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH.

In the Little Pond watershed, Jones Pond is the only freshwater pond with a delineated watershed (see Figure IV-1). In a typical MEP review, staff review available bathymetric information and water quality data to check the reliability of the 50% nitrogen attenuation factor for ponds in each watershed. In this case, PALS data is not available for Jones Pond. Previous MEP analysis of ponds with PALS and other data has shown that the 50% attenuation rate is generally an appropriate and conservative assumption for nitrogen attenuation in freshwater ponds on Cape Cod.

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 attenuated nitrogen load to respective downgradient watersheds. The apportioning of nitrogen load was based on the percentage of discharging pond shoreline bordering each downgradient sub-watershed. For Jones Pond, 58% of its attenuated nitrogen load flows into the Little Pond Salt sub-watershed, while the remainder flows outside of the watershed.

Buildout

In order to gauge potential future nitrogen loads resulting from continuing development, the potential number of residential, commercial, and industrial lots within each sub-watershed was determined from the GIS database (Figure IV-3). Typically MEP review begins by looking at parcels classified by the town assessor as developable (e.g., state class codes 130, 131 for residential development), reviews all municipal overlay districts (*e.g.*, water resource protection districts) and existing zoning, and coordinates this assessment with the town planning department. In the Little Pond watershed, Falmouth Planning Department had previously developed individual parcel buildout estimates and these estimates were used to determine additional development within each sub-watershed. A nitrogen load for each parcel was determined using the factors presented in Table IV-3 as discussed above. A summary of potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-4. However, only the attenuated nitrogen loads from the Land-use Sub-model were used for the water quality modeling, as the unattenuated rates of nitrogen loading would not permit model validation to conditions within embayment waters under any realistic physical conditions.

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 Little Pond 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. 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, a portion of the watershed nitrogen load 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 cases of the Little Pond embayment system watersheds, most of the freshwater flow and transported nitrogen passes through a surface water system and frequently multiple systems prior to entering the estuaries, 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 2001). Similarly, MEP analysis of the Quashnet River indicates that nitrogen transport from the inland watershed regions to the estuary results in a natural attenuation predominantly associated with riverine processes of 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. Similarly, the small tidal basin of Frost Fish Creek in the Town of Chatham showed ~20% nitrogen attenuation or watershed nitrogen load prior to discharge to Ryders Cove. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements of upper watershed attenuation were undertaken as part of the MEP Approach. MEP conducted long-term measurements of natural attenuation relating to surface water discharges to the head of the Little Pond embayment system in addition to the natural attenuation measures by fresh kettle ponds, addressed above. The additional site-specific study conducted by the MEP in the major surface water flow system (e.g. the stream discharging to the headwaters of the estuary) builds on and is consistent with a previous flow study conducted in 1990 (Millham and Howes 1994 a, b). The earlier effort was part of a study to evaluate methods for determining watersheds to coastal embayments situated in sandy aquifers, which used Little Pond as an experimental systems. A significant part of the effort was to determine the rates and pathways of freshwater inflow to tidal embayments and the processes which control the spatial distribution of groundwater entry. As discussed above, the direct measurements of total freshwater discharge determined from multiple method agree well with the freshwater discharge estimated from the watershed hydrologic model. Similarly, the stream discharge measured in 1990 to the headwaters of the Little Pond Estuary is in close agreement with the MEP record discussed below. It should be noted that in both cases continuous records were obtained over annual cycles at the same location. These earlier studies were undertaken to help develop the process-level information required to support linkage of watershed nitrogen loading models to estuarine hydrodynamic/water quality model for estuarine management.

Using the directly measured stream nitrogen load to the estuarine waters of Little Pond allows quantification of watershed based nitrogen attenuation, when compared to the nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the stream discharging into the head of Little Pond (at Spring Bars Road) provides a direct integrated measure of all of the processes presently attenuating nitrogen in the upgradient watershed contributing freshwater to the gauging site. These upper watershed regions account for more than half of the entire watershed area of the Little Pond embayment system. Flow and nitrogen load were measured at the stream gaging site for 16 months of record (Figure IV-5). During the study period, velocity profiles were completed on the stream every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

Determination of stream flow 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.

←

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, velocity measurements across the entire stream cross section were not averaged and then applied to the total stream cross sectional area. Instead, each stream subsection has a calculated stream discharge value and the summation of all the sub-sectional stream discharge values yields the total calculated discharge for the stream.

Periodic measurement of flows over the entire stream gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauges. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given river. These hourly stage 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 each of the surface water discharges flowing into each individual salt pond.

The annual flow record for the surface water flow into Little Pond was merged with the nutrient data sets generated through the weekly water quality sampling to determine nitrogen loading rates to the head (tidally influenced) of Little Pond. Nitrogen discharge from the stream was calculated using the paired daily discharge and daily nitrogen concentration data to determine the mass flux of nitrogen through the gauging site. For a given gauging location, 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 the pond currently reduces (percent attenuation) nitrogen loading to the Little Pond embayment system.

Figure IV-5. Location of Stream gauge (red triangle) in the Little Pond embayment system.

IV.2.2 Surface Water Discharge and Attenuation of Watershed Nitrogen: Stream Discharge to the Upper Estuarine Reach of Little Pond

Unlike other coastal salt ponds on the south shore of Falmouth (e.g. Great Pond, Green Pond) the stream discharging to the head of Little Pond does not originate from a specific freshwater pond, such as Coonamessett Pond, that has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. The stream flowing into Little Pond is groundwater fed, however, as a surface water feature it will still generate a certain level of biological attenuation as water flows through any wetlands and the streambed. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the stream into Little Pond above the gauge site and the measured annual discharge of nitrogen to the tidal portion of the Little Pond embayment system Figure IV-5.

At the stream gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater portion of the stream that carries the flows and associated nitrogen load to the head of the Little Pond Estuary. As the stream is tidally influenced at the point where it discharges to the head of Little Pond, the gauge was located just above the zone of tidal influence, such that freshwater flow could be measured during any portion of a tidal period. To confirm that freshwater was being measured salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average salinity was determined to be 0.22 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on the stream to Little Pond was installed on August 1, 2002 and was set to operate continuously for 16 months such that a complete hydrologic year would be captured in the flow record. Due to a need for additional velocity measurements to develop a robust rating curve, stage data collection was extended until January 13, 2004 for a total deployment of 18 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the stream gauge site based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Before using the continuously measured stage data to determine volumetric flow, any potential tidal influence on stage was checked by examining stage over multiple tidal periods. No tidal periodicity was observed in the stage data. 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 estuarine portion of the stream (Figure IV-6 and Table IV-5). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site and a comparison of the flow developed by the MEP was compared to a previous study undertaken in 1990 (Millham and Howes 1994 a, b) in order to judge the difference, if any, in annual flow .

The stream discharging into Little Pond is one of the smaller surface freshwater discharges along the south shore of Falmouth, with a discharge only ~10 percent of the flow in the Coonamessett River discharging to Great Pond and ~47 percent of the flow in Bournes Brook discharging to the head of Bournes Pond.

The annual freshwater flow record for the stream 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 by the MEP from the stream to Little Pond was within 10% of the long-term average modeled flows. Similarly, the MEP 2002-03 flows were within 5% of the 1989-90 flows measured in the previous study. Therefore, the MEP watershed and river datasets appear to be in balance and the flow rates consistent with other efforts.

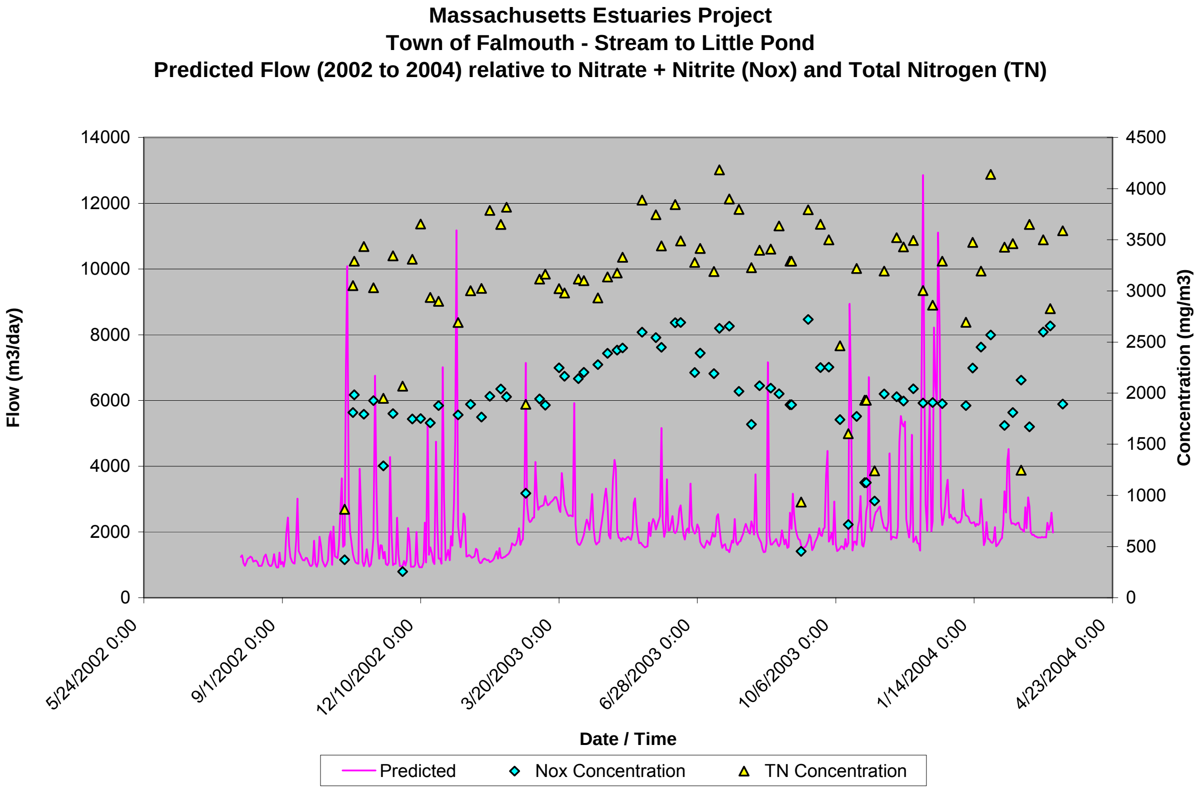


Figure IV-6. Predicted stream discharge (solid pink line) to Little Pond, nitrate+nitrite (blue diamonds) and total nitrogen (yellow triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to the Little Pond Estuary (Table IV-5).

Total nitrogen concentrations within the stream outflow were relatively high, 2.99 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 6.05 kg/day and a measured total annual TN load of 2,209 kg/yr. In the stream to Little Pond, nitrate was the predominant form of nitrogen (63%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater ponds and to the river was not completely taken up by plants within the pond or stream ecosystems. The high concentration of inorganic nitrogen in the outflowing stream waters also suggests that plant production within the upgradient freshwater ecosystems is not nitrogen limited.

From the measured nitrogen load discharged by the stream to the estuary 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 Bay. Based upon lower nitrogen load (2,209 kg yr^{-1}) discharged from the freshwater stream to Little Pond compared to that added by the various land-uses to the associated watershed (3719 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 40.6% (i.e. 40.6% of nitrogen input to watershed does not reach the estuary). This level of attenuation is also greater than the integrated attenuation rate determined from the watershed nitrogen model of 30% (Table IV-4). This is expected given the conservative assumptions of nitrogen attenuation used in the model. The observed attenuation rate in the Little Pond Stream is nearly identical to the observations in a generally similar system (most of nitrogen passing through a river, rather than ponds), Quashnet River, 39%. The directly measured nitrogen loads from the stream was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-5. Comparison of water flow and nitrogen discharge from stream discharging to the head of Little Pond. The "Stream" data is from the MEP stream gauging effort. Watershed flow data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Stream Discharge to Little Pond[(a)]	Data Source
Total Days of Record	365[(b)]	(1)
Flow Characteristics		
Stream Average Discharge (m3/day)	2006	(1)
Contributing Area Average Discharge (m3/day)	2251	(2)
Discharge Stream 2002-03 vs. Long-term Discharge	90%	
Nitrogen Characteristics		
Stream Average Nitrate + Nitrite Concentration (mg N/L)	1.894	(1)
Stream Average Total N Concentration (mg N/L)	2.988	(1)
Nitrate + Nitrite as Percent of Total N (%)	63%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	6.05	(1)
TN Average Contributing Area Attenuated Load (kg/day)	2603	(2)
TN Average Contributing UN-attenuated Load (kg/day)	3719	(3)
Attenuation of Nitrogen in Pond/Stream (%)	41%	(4)

(a) Flow and N load to stream discharging to Little Pond includes Long Pond contributing area.
(b) September 10, 2002 to September 10, 2003.

(1) MEP gage site data
(2) Calculated from MEP watershed delineations to Long Pond and Mares Pond for flow to Little Pond; the fractional flow path from each sub-watershed which contribute to the flow in the stream to Little Pond; and the annual recharge rate.
(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.

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

Table IV-6. Summary of annual volumetric discharge and nitrogen load from the stream discharging to the head of Little Pond based upon the data presented in Figures IV-6 and Table IV-5.

EMBAYMENT SYSTEM	PERIOD OF RECORD	DISCHARGE (m3/year)	ATTENUATED LOAD (Kg N/yr)	
			Nox	TN
Stream to Little Pond (MEP)	Sept. 10, 2002 to Sept. 9, 2003	739323	1399	2209
Stream to Little Pond (Millham & Howes 1994)	Sept. 10, 1989 to Sept. 9, 1990	700234		
Stream to Little Pond (CCC)	Water Balance (recharge to watershed)	821631		

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 Little Pond embayment predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the watercolumn (once it entered), then predicting watercolumn nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayment from the surrounding watersheds it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton “particles”. Most of these “particles” remain in the watercolumn for sufficient time to be flushed out to a downgradient larger waterbody (like Vineyard 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. Also, in longer residence time systems (greater than 8 days) these nitrogen rich particles may die and settle to the bottom. In both cases (grazing or senescence), a fraction of the phytoplankton with their associated nitrogen “load” become incorporated into the surficial sediments of the bays.

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

Once organic particles become incorporated into surface sediments they are decomposed by the natural animal and microbial community. This process can take place both under oxic (oxygenated) or anoxic (no oxygen present) conditions. It is through the decay of the organic matter with its nitrogen content that bioavailable nitrogen is returned to the embayment watercolumn for another round of uptake by phytoplankton. This recycled nitrogen adds directly to the eutrophication of the estuarine waters in the same fashion as watershed inputs. In some systems that 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. In contrast, in some depositional basins nitrogen fluxes can be near zero of even negative as a result of combined settling and denitrification. It is during these warmer months that estuarine waters are most sensitive to nitrogen loadings. Failure to account for this recycled nitrogen generally results in significant errors in determination of threshold nitrogen loadings. In addition, since the sites of recycling can be different from the sites of nitrogen entry from the watershed, both recycling and watershed data are needed to determine the best approaches for nitrogen mitigation.

IV.3.2 Method for determining sediment-watercolumn nitrogen exchange

For the Little Pond 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 8 sites in Little Pond (Figure IV-7) in July 2002. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample.

Rates of nitrogen release were determined using undisturbed sediment cores incubated for 24 hours in temperature-controlled baths. Sediment cores (15 cm inside diameter) were collected by SCUBA divers and cores transported by small boat to a shore side field lab. Cores were maintained from collection through incubation at *in situ* temperatures. Bottom water was collected and filtered from each core site to replace the headspace water of the flux cores prior to incubation. The number of core samples from each site (Figure IV-7) per incubation were as follows:

Little Pond Benthic Nutrient Regeneration Cores

- Station LP-1 1 core (Upper Region of Pond)
- Station LP-2 1 core (Upper Region of Pond)
- Station LP-3 1 core (Upper Region of Pond)
- Station LP-4 1 core (Upper Region of Pond)
- Station LP-5 1 core (Lower Region of Pond)
- Station LP-6 1 core (Lower Region of Pond)
- Station LP-7 1 core (Lower Region of Pond)
- Station LP-8 1 core (Lower Region of Pond)

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

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

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

Figure IV-7. Little Pond embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-7.

IV.3.3 Rates of Summer Nitrogen Regeneration from Sediments

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

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

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

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

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and some denitrification of produced inorganic nitrogen before it can "escape" to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-8).

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

In order to determine the net nitrogen flux between watercolumn and sediments, all of the above factors were taken into account. The net input or release of nitrogen within a specific embayment was determined based upon the measured ammonium release, measured nitrate uptake or release, and estimate of particulate nitrogen input. Dissolved organic nitrogen fluxes were not used in this analysis, since they were highly variable and generally showed a net balance within the bounds of the method.

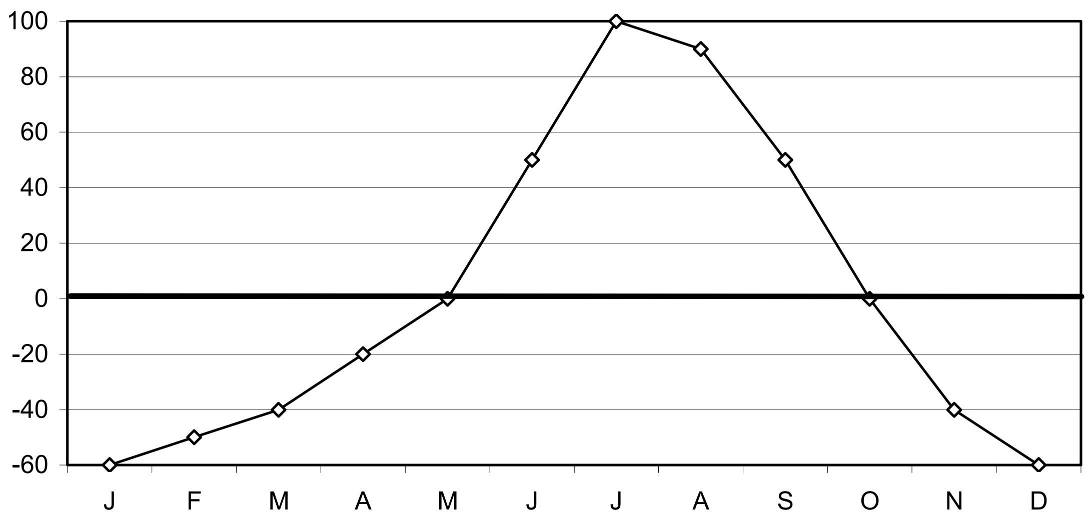


Figure IV-8. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment sampling was conducted within the upper and lower portions of Little Pond in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-7). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. Sampling was also undertaken to capture variability due to shallow regions versus deeper depositional sites. 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 and the average summer particulate carbon and nitrogen concentration within the overlying water. Two levels of settling were used. If the sediments were organic rich and fine grained and the hydrodynamic data showed low tidal velocities, then a water column particle residence time of 8 days was used (based upon phytoplankton and particulate carbon studies of poorly flushed basins). If the sediments indicated a coarse grained sediments and low organic content and high velocities, then half this settling rate was used. Adjusting the measured sediment releases was essential in order not to over-estimate the sediment nitrogen source and to account for those sediment areas that are net nitrogen sinks for the aquatic system. This approach was validated in outer Cape Cod embayments (Town of Chatham) by examining the relative fraction of sediment carbon turnover (total sediment metabolism) accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins is driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments.

Net nitrogen release or uptake from the sediments within the Little Pond embayment system for use in the water quality modeling effort (Chapter VI) are presented in Table IV-7. Rates of net sediment nitrogen release were similar to the three adjacent estuaries (Great, Green, and Bournes Ponds). Areas of highest summer chlorophyll a (Table VII-2) and fine-grained sediments tend to support the highest rates of nitrogen release, most likely due to their higher organic deposition rates, hence higher rates of nitrogen recycling. The rates in these systems were similar to those reported for the nearby Vineyard Sound estuary, Popponesset Bay, which ranged from 85 to - 17 mg N m^{-2} d^{-1}. Sediment nitrogen regeneration rates were determined at 2 sites in Little Pond (LP-2, LP-3) in a study in 1989-90 (D. Schlezinger unpublished). These data showed sediment regeneration rates of 32.7 (± 26.3) mg N m^{-2} d^{-1} and 30.9 (± 1.3) mg N m^{-2} d^{-1}, respectively. These compare well to the MEP 2002 values of 56.63 (Sta 7) and 32.78 (Sta 4) at similar locations compared to those from more than 1 decade earlier. The similarity in rates is consistent with the relative constancy of the watercolumn nitrogen levels over time (see Section VI).

Table IV-7. Rates of net nitrogen return from sediments to overlying waters of the Little Pond embayment system. These values are combined with basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July-August rates. The spatial variability in rates tends to follow the depositional environment and bathymetry of the basins (i.e. distribution of organic deposition).

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1})			
	Mean	S.E.	N	Station
Little Pond Estuary				
Upper	-1.23	-5.79	6	1
	-10.59	-21.12	6	2
	51.02	23.88	6	3
	32.78	18.52	6	4
Lower	-42.03	-23.56	6	5
	-19.46	-17.66	6	6
	56.63	14.08	6	7
	-18.22	-17.92	6	8

Station numbers refer to Figure IV-7.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

A hydrodynamic study was performed for Little Pond, which is located within the town of East Falmouth, Massachusetts, along the coast of Vineyard Sound. The aerial photograph in Figure V-1 shows the general study area. Little Pond has only one main basin (no tributaries) that extends northward about 0.84 mi (1.35 km) from the coastline. The pond covers approximately 45 acres. The system is very shallow, with a mean depth of 1.8 feet. The deepest section of Little Pond is midway up the pond, with a mean depth of 4.3 feet.

Circulation in the system is dominated by tidal exchange with Vineyard Sound. From measurements made in the course of this study, the average tide range in Little Pond is approximately 1.5 feet. The flow restrictions caused by the inlet channel and associated culvert, produces major reductions in the tide range from Vineyard Sound. The reductions are on the order of 1.0-1.5 feet.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry and tide data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry survey of Little Pond was performed to determine the variation of embayment and channel depths throughout the system. This survey was conducted to address any changes that may have occurred since the installation of the new box culvert and jetties at the inlet, which post-date the previous bathymetry surveys. In addition to the bathymetric survey, tides were recorded at two locations within the system for 29 days. This tide and bathymetry data are necessary to run and calibrate the hydrodynamic model of the system.

A numerical hydrodynamic model of the system was developed in the second portion of this study. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from a gauge located offshore of the entrance to Little Pond was used to define the open boundary condition that drives the exchange of water through the entrance of Little Pond. Data from the tide gauge within the pond was used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the real, physical system.

The calibrated computer model of the hydrodynamic characteristics of Little Pond was used to compute the flushing rates of each region of the embayment. Though water quality in an embayment cannot be directly inferred by use of the computed flushing rate alone, it can serve as a useful indicator of the flushing performance of an embayment relative to other systems. The ultimate utility of this hydrodynamic model is as input into a constituent transport model, where water quality constituents like nitrogen are modeled to determine the real water quality dynamics of a system.

Figure V-1. Aerial photograph of the Little Pond and adjacent embayments of Great Pond and Falmouth Harbor, Falmouth, MA.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE SYSTEM

The southern coast of Cape Cod in the vicinity of Little Pond is a relatively quiescent region of coastal processes. Although natural wave and tidal forces continue to reshape the shoreline, day-to-day conditions have limited impact on the shoreline migration and/or inlet stability. For typical wave conditions, longshore transport of sand is from west-to-east along the south coast of Falmouth, due primarily to the predominant local wind-driven waves. In contrast to the mild day-to-day conditions, infrequent hurricane events such as the hurricanes of 1938, 1944, and 1954, as well as Hurricane Bob in 1991, all caused significant overwash and transport of beach sediments. In addition, northeast storm events (causing waves to approach the Falmouth shoreline from the east and southeast) create a sediment transport reversal from typical conditions, where the longshore sediment transport is from east-to-west. The effect of this sediment transport reversal can often be seen by observing the sand impounded by the groins found along the shoreline, where typical summer conditions will impound sand along the west side of the groins and easterly storm events during the winter will impound sand along the east side of the groins. For those years in which there are a significant number of easterly storm events, the net alongshore sand transport direction becomes unclear. However, over the long-term the trend appears to be a west-to-east transport of sand on the order of 4,000 cubic yards per year (based on an analysis of observed volumetric accretion at the Waquoit Bay west jetty between 1938 and 1961, performed by the U.S. Army Corps of Engineers, 1964). More recent dredging volumes at Great, Green, and Bournes Ponds entrances indicate that longshore transport may be as low as 1,000 cubic yards per year (personal communication with Barnstable County Dredge personnel, 2004). Similarly, estimates of accumulation adjacent to the Little Pond entrance indicated longshore transport rates between 200 and 600 cubic yards per year (Ramsey and Fields, 1993). For comparison, longshore sediment transport rates along U.S. beaches exposed to Atlantic Ocean waves (e.g. the Cape Cod National Seashore, the New Jersey Coast, and the east coast of Florida) typically are two orders of magnitude higher, between 100,000 and 500,000 cubic yards per year.

Due to the quiescent wave environment and small tide range in the vicinity of the Little Pond, inlet migration is less of a concern than other areas of Cape Cod. According to FitzGerald (1993), inlets along the southern shore of Cape Cod required the construction of jetties to keep them open and navigable. Similar-sized coastal ponds on the southern shore of Martha's Vineyard have unstable inlets that close and are periodically opened to allow lowering of the pond level and some exchange of saltwater with the ocean (e.g. Edgartown Great Pond and Tisbury Great Pond). Relative to the systems on Martha's Vineyard, the inlet to Little Pond can be considered stable, with some observed historic migration of the inlet. The inlet was located to the east of its present location in maps from 1846, 1880 and 1893. Between 1880 and 1893 a road was constructed along the length of the barrier beach to the inlet but there is no evidence of jetties in this region at that time (Figure V-2). By 1936 the road on the barrier had been extended fully across to the Maravista Peninsula, probably in the 1920's, but tidal exchange was maintained. However, Millham (1993) reconstructing recent changes to Little Pond indicated that after the roadway extension, tidal flushing was restricted. He stated that "Anecdotal evidence of the fresh or brackish water conditions in the pond before 1964 were offered by several local individuals who fished (trout and perch) and trapped the pond prior to 1964. The reports give some details of the control of the pond level by the old culvert, which was apparently set at a level to prevent the entrance of seawater during most high tides. However, during spring tides and under wind forcing conditions, salt water from Nantucket Sound did flow into the pond inlet". In 1964 the Town of Falmouth placed dual culverts and jetties in the present location and pond tidal exchange was enhanced (Millham 1993). These structures were again replaced in 1995 to repair damage from Hurricanes and to provide more consistent tidal exchange.

Since the addition of jetties to Little Pond and its adjacent embayments, the cross-section of each inlet has remained relatively stable, allowing for effective tidal circulation through the flow constrictions at the entrances. The inlet stability afforded by the jetty systems inhibits periodic inlet closures that can cause large ecological shifts to estuarine plant and animal communities, such as has occurred in nearby Bournes Pond circa 1980.

For Little Pond, the entrance has a long history of shoaling, where beach material adjacent to the east jetty tends to infill the entrance channel during easterly storm events. Much of the observed accretion along both the updrift and downdrift shorelines likely is caused by the long-term influence of the 1957 beach nourishment program at Falmouth Heights (see Section V.2.3 for more details). Sand deposited in the inlet entrance has historically caused inlet blockage, as exemplified by the 36 inlet closures between August 1988 and May 1993 (Millham and Howes 1994, Howes and Goehringer 1993). The DPW has maintained the entrance by dredging sand deposited on a bar near the seaward end of the east jetty, as well as sand deposited within the inlet channel. Figures V-3 and V-4 show shoaling in the entrance channel and typical small-scale dredging efforts needed to keep the Little Pond entrance channel open, respectively.

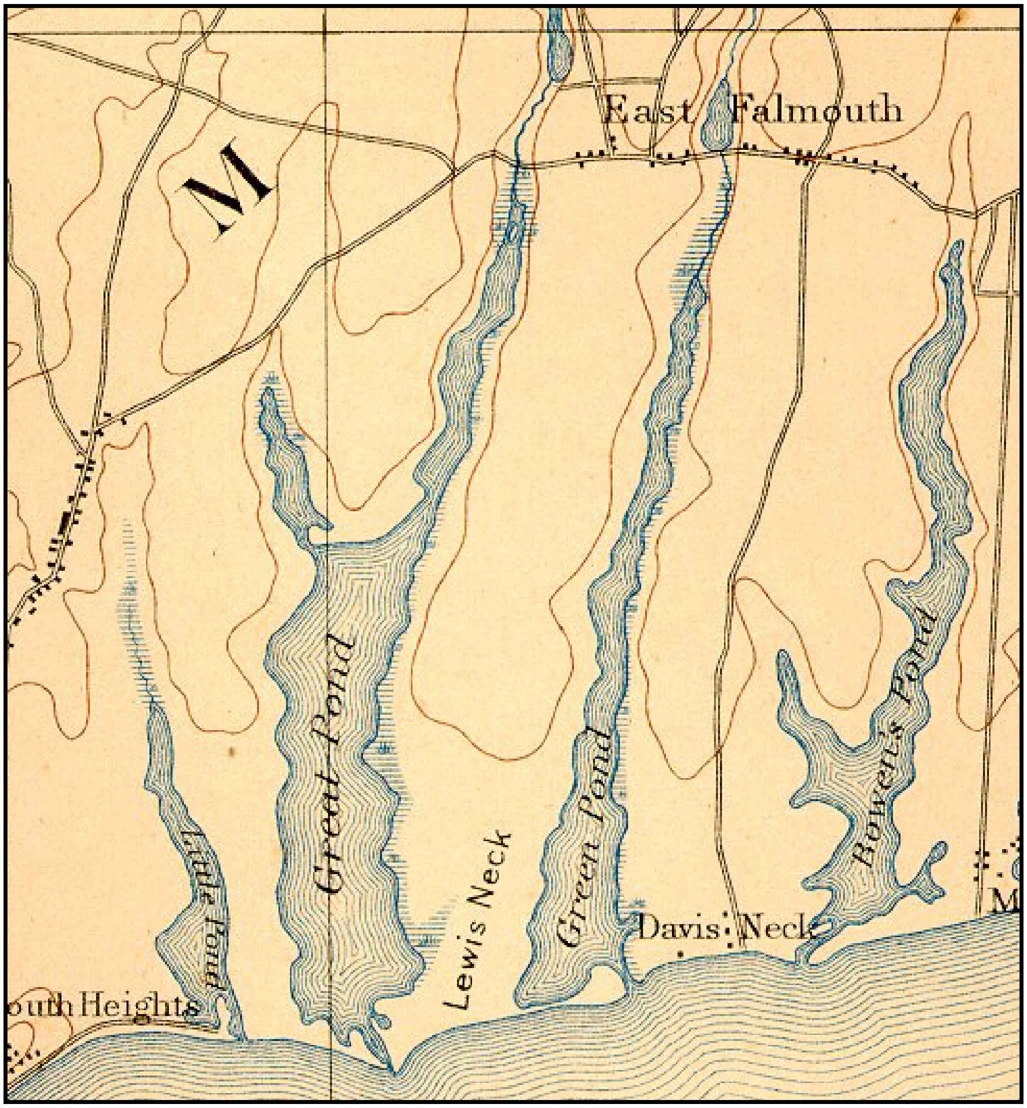


Figure V-2. A portion of the U.S.G.S. 1893 map showing Great, Green, and Bowen's (Bournes) Ponds. This map depicts the condition of these inlets prior to the installation of jetties.

In 1995, tidal flushing within Little Pond was improved by the replacement of the two corrugated metal culverts, with a single larger box culvert. The larger culvert improved tidal flushing; however, shoaling continues to be a problem within the outer portion of the inlet channel. Due to the larger tidal prism, dredging frequency has been reduced significantly relative to the pre-1995 conditions. In addition, sediment moving through the inlet channel has formed a well-defined flood shoal north of the inlet channel. The location of this shoal is beyond the limits of the existing maintenance dredging permits. Continued development of this shoal has inhibited tidal flushing during periods when the inlet channel is functioning properly.

Figure V-3. Shoaling of the Little Pond entrance channel in 2004, where littoral drift caused beach sediment from the east side of the inlet to partially block the channel.

Figure V-4. Typical dredging required to maintain the Little Pond entrance channel.

V.2.1 Coastal Processes and Inlet Stability

Since inlet stability is partially governed by longshore coastal sediment transport, understanding the regional long-term shoreline change and littoral movement of sand is critical for evaluating stability of the entrance to Little Pond. As discussed above, the observed longshore transport rates are relatively low, primarily as a result of the quiescent wave environment of Nantucket and Vineyard Sounds. Although the amount of sand moving along the coast is small, the tidal prism through Little Pond inlet also is relatively small. Since the reconstruction of the jetty system in 1995, the inlet has generally reached equilibrium, where the tidal velocities through the main channels are sufficient to prevent significant shoaling. However, infrequent storm events continue to periodically to block or partially block the inlet as local littoral drift transports material from east-to-west into the inlet channel. Continued small-scale maintenance dredging is anticipated to keep the inlet open following easterly storm events. The general west-to-east littoral drift along Falmouth's south shore continues to supply the Bristol Beach area with sediment from the 1957 Falmouth Heights beach nourishment. Due to this influx of sediment, the jetty to the east side of the inlet is typically filled beyond entrapment, allowing for more frequent shoaling problems. Overall, material supplying the beach to the east side of the entrance channel is derived from beaches to the west, where long-term sediment transport carries sediment from west-to-east around the longer west jetty and settling within the shadow zone to the east.

In addition, it appears that the south coast of Falmouth (between Falmouth Harbor and the west entrance to Waquoit Bay) has generally equilibrated to changes in local coastal sediment transport caused by the construction of shoreline armoring. Extensive armoring of the Falmouth shoreline began in the late 1800s and early 1900s with construction of the railroad to Woods Hole, the old stone dock, the Falmouth Harbor jetties, and the Waquoit Bay east jetties. This shoreline armoring continued through the mid-1900s with the construction of stone groin fields, which often replaced existing wooden structures. In the Little, Great, Green, and Bournes Ponds region, these wooden and stone structures were constructed to protect Menauhant Road and waterfront dwellings (Figures V-5 and V-6). The remnants of wooden groins and bulkheads can be found along much of Falmouth's south coast (Figure V-7).

As shown in Figures V-5, V-6, and V-8, hurricanes can have a significant impact on both the shoreline and the inlets. Due to the relatively quiescent wave and tide regime within this region, the impact of infrequent storms, primarily a result of storm surge, can be dramatic. According to historic flooding information (U.S. Army Corps of Engineers, 1988), the storm surge level in the Little Pond area was 11 feet NGVD (more than 10 feet above mean sea level). Due to this elevated water level, the series of low-lying barrier beaches that separate Nantucket Sound from the coastal ponds were overtopped, often carrying beach sediment into the estuaries. These infrequent storms can reshape the shoreline in ways that would require many years or decades under the typical wave, wind, and tide regime of the Falmouth south coast. During the twentieth century, the severe hurricanes influencing the Falmouth shoreline include the hurricanes of 1938, 1944, and 1954, as well as Hurricane Bob in 1991. Of these storms, the Hurricane of 1944 had the largest storm surge along the south shore of Falmouth (U.S. Army Corps of Engineers, 1988).

REMAINS OF BUILDINGS of Wellsmere Inn, Maravista, Falmouth.

Figure V-5. Photograph of the Wellsmere Inn in Maravista immediately after the 1944 Hurricane, which had the largest storm surge in the region of Little Pond of 20th century storms. Note the wood bulkheads and concrete seawall that were utilized to armor the shoreline.

ROAD BED at Maravista, Falmouth.

Figure V-6. Photograph of Menauhant Road in Maravista immediately after the 1944 Hurricane. Note the stone revetment armoring the roadway.

Figure V-7. Photograph of the shoreline west of Menauhant Beach taken in 2004 showing remnants of a timber bulkhead and groin, as well as more recent stone structures.

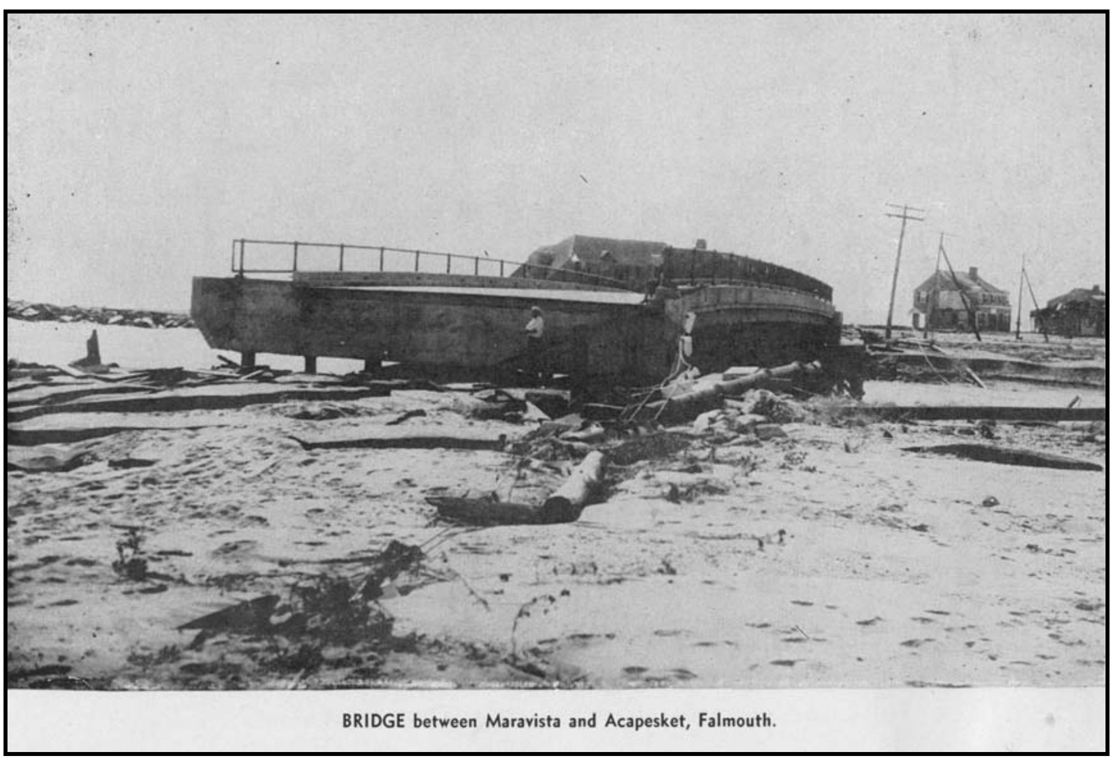


Figure V-8. Photograph of the old Great Pond bridge immediately after the 1944 Hurricane. The photograph shows that storm overwash eroded the roadway and approach ramps to the bridge. Note the Great Pond western jetty at the left side of the photograph.

V.2.2 Shoreline Change Analysis

Shoreline change maps can effectively be used to evaluate the effects of long-term coastal processes. In addition, shoreline change maps also can indicate the effects of short-term changes that often occur as the result of anthropogenic (e.g. development of extensive shore protection structures) or natural (e.g. inlet migration) processes. Prior to developing conclusions and/or management recommendations that depend on shoreline change estimates, such as involving inlet management, it is critical to understand potential errors and uncertainties associated with this type of analysis. Understanding the limitations of shoreline change data is critical for developing appropriate management strategies for shorelines and inlets in areas with relatively low shoreline migration rates, such as Falmouth's south coast.

The Massachusetts Coastal Zone Management Office (MCZM) recently updated their shoreline change analysis (Thieler et al., 2001) to incorporate more recent shoreline information. Relative to Little Pond, the updated Massachusetts Shoreline Change Project included analysis along much of the south coast of Falmouth. The analysis included shorelines from the mid-1800's to the three most recent shorelines available from the MCZM dataset from 1938, 1975, and 1994. Based on the maps, the long-term shoreline change rates for this stretch of the Falmouth coast (from the mid-1800s to 1994) were less than 0.5 ±0.4 feet per year of erosion, indicating a stable shoreline.

A recent report published by the Coastal Resources Working Group (CRWG, 2003), a citizens group focused on long-term management of the Falmouth shoreline, used the updated MCZM shoreline data set to analyze the shoreline between Nobska Point and the Waquoit Bay jetties, which includes the barrier beach of Little Pond. They determined that recent shoreline change in this region averaged about 2.4 feet of erosion annually from 1975 to 1994 (or about 46 feet of landward movement over this time period). An erosion rate of this magnitude would suggest significant coastal erosion and the associated longshore transport of beach-derived sediments. For the inlet to Little Pond, the large littoral drift indicated by the shoreline retreat rate would be expected to cause severe shoaling problems, as well as potential inlet stability concerns.

As a result of the potential implications to the management of Little Pond, the CRWG shoreline analysis was examined in detail. Initial analysis suggested that the high rate of shoreline retreat in the region of Little Pond is inconsistent with much of the available historical data and is uncharacteristic of south-facing shorelines in the quiescent wave environments of Nantucket and Vineyard Sounds. For example, the recent erosion rate for Falmouth from the MCZM data set (1975 to 1994) is nearly identical to the long-term erosion rate reported for the bluffs along the Cape Cod National Seashore, where Geise and Aubrey (1988) reported recession rates of 2.54 feet annually. Unlike the east facing bluffs along the Cape Cod National Seashore, Falmouth's south coast is not exposed to open Atlantic Ocean wave conditions and the erosional forces associated with that environment. In addition, many of the groins and jetties constructed between the early 1900s and the mid-1950s do not extend 50 feet beyond the existing high water line; therefore, these groins would have been completely buried in the beach in the mid-1970's according to the 46 feet of shoreline retreat indicated in the data set utilized by the CRWG (i.e. the two most recent shoreline available from the Massachusetts Coastal Zone Management shoreline change information). A review of shoreline data indicates that recent shoreline change in this region is consistent with the above observations and with the longer term retreat rate of 0.5 ft yr^{-1}.The discrepancy appears to derive from the statewide MCZM data set not providing the necessary accuracy to evaluate recent shoreline change along Falmouth's south coast for the purpose of developing a long-term coastal management plan. Because shoreline change is important to beach and inlet management and inaccurate rates could potentially lead to misinterpretation of regional coastal processes and improper decisions regarding long-term coastal management, MEP conducted a further analysis to derive recent rates of shoreline change. This analysis is presented below.

- Analysis of the CRWG report indicates an improper assessment of the error in the shoreline change analysis. The report implies that the errors in the shoreline change analysis for all time periods are 0.4 feet per year; however, this is incorrect based on the technical report for the shoreline change analysis (Thieler, et al., 2001). Specifically, the technical report indicates shoreline position errors of ±8.5 meters (±28 feet) exist for each data set. For the 1975 to 1994 shoreline change predictions, the root-mean-square error (RMS error) would be approximately ±2.1 feet per year, not the ±0.4 feet per year reported (the ±0.4 feet per year error is only appropriate for the entire time period from the mid 1800s to 1994). This misinterpretation of the errors associated with shoreline change predictions would incorrectly indicate that much of the measured shoreline change between 1975 and 1994 was actual shoreline migration, rather than error associated with the analysis technique. In general, the error in shoreline change rate predictions is higher for short time periods. Therefore, if the shorelines were properly evaluated, the recent 1975 to 1994 shoreline change would be correctly presented as averaging -2.4 feet ±2.1 feet along the south coast of Falmouth. The potential error in this short-term analysis is nearly identical to the observed shoreline change.
- Construction of shore protection structures along the Falmouth shoreline was not limited to the time period of the 1930s to 1960s as implied in the CRWG report. Structures that existed at the time of the 1938/1948 shoreline included numerous groins between Nobska Point and Trunk River, the Old Stone Dock groins along Shore Drive, the Falmouth Harbor jetties, numerous groins and seawalls between Great Pond and Bournes Pond, and the Waquoit Bay jetties. Once constructed, these structures immediately altered the longshore transport of sediments along the south coast of Falmouth. To evaluate how the Falmouth shoreline has responded to the existence of coastal engineering structures, a more appropriate time period for assessment is from 1938/1948 to the most recent shoreline available, not the 1975 to 1994 time period selected for the analysis in the CRWG report.

Due to the concerns regarding potential errors in determining an appropriate shoreline change rate, MEP reviewed the existing shoreline data sets. As part of the review process, recent imagery was downloaded from the MassGIS website and these readily available aerial photographs were compared to assess the horizontal control. These data were integrated with the interpreted shoreline data that was provided by the Massachusetts Office of Coastal Zone Management (MCZM). The 2001 aerial photography was flown in April, 2001; the 1994 aerial photos were flown in September/October 1994. To evaluate the horizontal control of the two aerial photograph sets, a field survey using differential GPS was conducted to locate a series of common features visible on both sets of orthophotos. This analysis indicated that horizontal control issues exist for both the 1994 and 2001 orthophotos; however, the errors appear to fall within the acceptable range of ±3 meters (±10 feet) for control points (for more information see Anders and Byrnes, 1991 or Crowell, et al., 1991).

In addition to horizontal control, interpretation of the shoreline from aerial photographs also can lead to non-random errors regarding mapped shoreline positions. Due to the poor quality of the 1994 Falmouth orthophotos, interpretation of the shoreline from these images appears to be a problem. For Falmouth shorelines, it appears very difficult to select a high water line from the 1994 imagery, primarily because the orthophotos appear overexposed. The State's 2001 aerial photography is of much higher quality, where the high water shoreline is typically discernable on the beach. Based on a cursory review of the 1994 shoreline overlaid on both the 1994 and 2001 orthophotos, it appears that incorrect identification of the high water line from the 1994 photographs is the cause of the over-prediction of recent shoreline erosion rates. This conclusion is further supported by a comparison of the 1994 shoreline interpreted from the orthophotos and the results of the differential GPS survey conducted in 2004 (survey described below). A cursory analysis of the 1994 and 2004 shorelines indicated an average apparent shoreline *accretion* of about 10 feet (with a maximum accretion of 68 feet) between Falmouth Harbor and Menauhant Beach. Since there does not appear to be a recent large-scale sediment source that would be responsible for an accreting shoreline, the most reasonable conclusion is that the 1994 shoreline was placed too far landward.

When the 1994 shoreline is excluded, three outer coast shoreline surveys (1938, 1975, and 2004) were used to quantify recent shoreline change between Falmouth Harbor and Menauhant Beach during the time period from 1938 to 2004. The 1938 shoreline survey was interpreted from aerial photography by the U.S. Coast and Geodetic Survey (USC&GS; predecessor to NOS) and vector data were provided online at the Shoreline Data Explorer website (http://www.ngs.noaa.gov/newsys_ims/shoreline/index.cfm). This 1938 shoreline was used in favor of the 1938 shoreline currently in the MCZM database, since the horizontal control for the NOAA digitized shoreline appears to be more accurate. It should be noted that both 1938 shorelines (MCZM and NOAA) were digitized from the same source. The 1975 shoreline was provided in digital format by the Massachusetts Coastal Zone Management (MCZM) Office, as part of the Massachusetts Shoreline Change Project. The 1975 digital shoreline data were digitized and assembled from aerial photographs by previous investigators (Theiler et al., 2001). The 2004 shoreline survey was developed by Applied Coastal personnel using a Trimble Pro/XR differential GPS. The region surveyed for this study is shown in Figure V-9.

All digital data were reviewed for accuracy and shoreline structure consistency. A review of metadata provided by MCZM regarding the quality of the 1975 data set indicated that the accuracy of the data was relatively low. This conclusion was supported by the consistency review of the digital data set. As a result, the overall time period (1938 to 2004) was deemed to yield the most accurate representation of shoreline change conditions in the region of Little Pond. Fortunately, this 66-year span effectively represents the period of time that the south coast of Falmouth has been influenced and/or governed by coastal engineering structures.

As indicated above, when determining shoreline position change, all data contain inherent errors associated with field and laboratory compilation procedures. These errors should be quantified to gauge the significance of measurements used for research/engineering applications and management decisions. Table V-1 summarizes estimates of potential error associated with shoreline data sets used for this study. Because individual errors are considered to represent standard deviations, root-mean square error estimates are calculated as a realistic assessment of combined potential error. Using these estimates, the total root mean square (RMS) estimate for the 1938 to 2004 time period is $\pm$29 feet, or approximately 0.4 ft/yr.

Figure V-9. The 2001 aerial photograph showing the extent of the Falmouth shoreline surveyed by Differential GPS in 2004.

Table V-1. Estimates of potential error associated with shoreline position surveys.

Cartographic / Interpretation Errors (1938 Shoreline Survey)	
Inaccurate location of control points on map relative to true field location	up to ±10 ft
Placement of shoreline on map	□16 ft
Line width for representing shoreline	±10 ft
Digitizer error	±3 ft
Operator error	±3 ft
Delineating high-water shoreline position	±16 ft
GPS Survey Errors (2004 shoreline survey)	
Delineating high-water shoreline	±10 ft
Total Potential RMS Error Between 1938 and 2004	**±28.8 ft** (**±0.44 ft/yr**)
Sources: Shalowitz, 1964; Ellis 1978; Anders and Byrnes, 1991; Crowell et al., 1991.	

Change calculations, 1938-2004, were made at 30-meter intervals along the outer coast between Falmouth Harbor and Menauhant Beach, MA using the Automated Shoreline Analysis Program (ASAP) for ArcGIS 8.3. Shore-normal transects were developed using average shoreline angles determined at each analysis point. All transects used for determining change rates were visually inspected to ensure suitability for analysis and shoreline structure avoidance.

Shoreline change calculated between 1938 and 2004 showed a relatively stable shoreline for the majority of the southern coast of Falmouth. During this time interval, change rates ranged from about -4.15 ft/yr to +1.43 ft/yr, with an average rate over the study area of about -0.36 feet/yr, where change denoted with a minus represents erosion and change denoted with a plus represents accretion. Maximum erosion rates for the study area were recorded near the inlets to Green and Bournes Ponds (-3.06 ft/yr and -4.15 ft/yr, respectively), while the most stable and/or accreting portion of the beach for this time interval was observed along the coast adjacent to Falmouth Heights (rates ranging from -0.23 ft/yr to +1.27 ft/yr). The change transects and data distribution for this time interval are shown in figures V-10 and V-11, respectively. Overall, 75% of shoreline change calculated within the study area during this time period ranged between -1.0 and +1.0 ft/yr. The average shoreline change in the region of Little Pond (i.e. between Falmouth Harbor and Menauhant Beach) appears to be slightly erosional; however, the magnitude of shoreline recession is actually smaller than the error estimates associated with the shoreline datasets (±0.44 ft/yr).

The erosion “hot-spots” identified along the outer coast adjacent to Green and Bournes Ponds represent regions associated with inlet processes and/or jetty construction, which do not appear representative of processes affecting the remainder of the coast. In contrast, the majority of the shoreline (86%) shows an average change rate of -0.055 ft/yr. This estimate was derived from the data distribution generated, excluding the “hot spots” (Figure V-12). Therefore, the data indicates that from 1938-2004 the southern coast of Falmouth between Falmouth Harbor and Menauhant Beach can be classified as a relatively stable shoreline. A small portion of this “stability” (e.g. east of the Great Pond jetties) is due to ‘hard’ shore protection measures that prevent further landward migration of the shoreline. In addition, beach nourishment performed in 1957 effectively stabilized the Falmouth Heights beach region. The classification of the shoreline in the region of Little Pond is consistent with the observations of locations of fixed structures (jetties and groins) and with shoreline changes in other areas of Cape Cod during this period.

V.2.3 Inlet Management Implications

For the Little Pond entrance, the influence of shoreline change and the related longshore sediment transport rates directly influence the stability of the existing inlet systems. Current management practices (e.g. post 1995) at the Little Pond inlet consist of periodic dredging to maintain the existing channels. This maintenance dredging is performed on an “as needed basis” following blockage or partial blockage of the inlet following easterly storms. The sand dredged from the inlet typically is placed on the beach immediately west (updrift) of each inlet. This placement location represents the least costly alternative; however, passing inlet sediment to downdrift shorelines (beaches to the east of each inlet) would supply these areas with needed littoral sediments as well. If future dredging places material on downdrift beaches, placement of material immediately east of the east jetty should be avoided to prevent inlet shoaling.

Approximately 120,000 cubic yards of beach nourishment was placed along the Falmouth Heights shoreline in 1957 as part of the navigation improvement project for Falmouth Harbor (U.S. Army Corps of Engineers, 1964). Based on shoreline change data (from -0.23 ft/yr to +1.27 ft/yr), much of this material can still be found on the beach between the Falmouth Heights bluffs and Little Pond inlet, where the shoreline has shown accretion or minimal erosion between 1938 and 2004. Continued migration of the beach fill will likely cause ongoing maintenance dredging issues at Little Pond inlet. However, maintenance dredging requirements are relatively minor. As the Town continues its maintenance dredging of the Little Pond inlet it should pursue appropriate permits to maintain a channel across the flood shoal at the same elevation as the box culvert invert to ensure proper tidal flushing.

Figure V-10. The 2001 aerial photograph showing scaled transects that indicate computed shoreline change rates between 1938 and 2004.

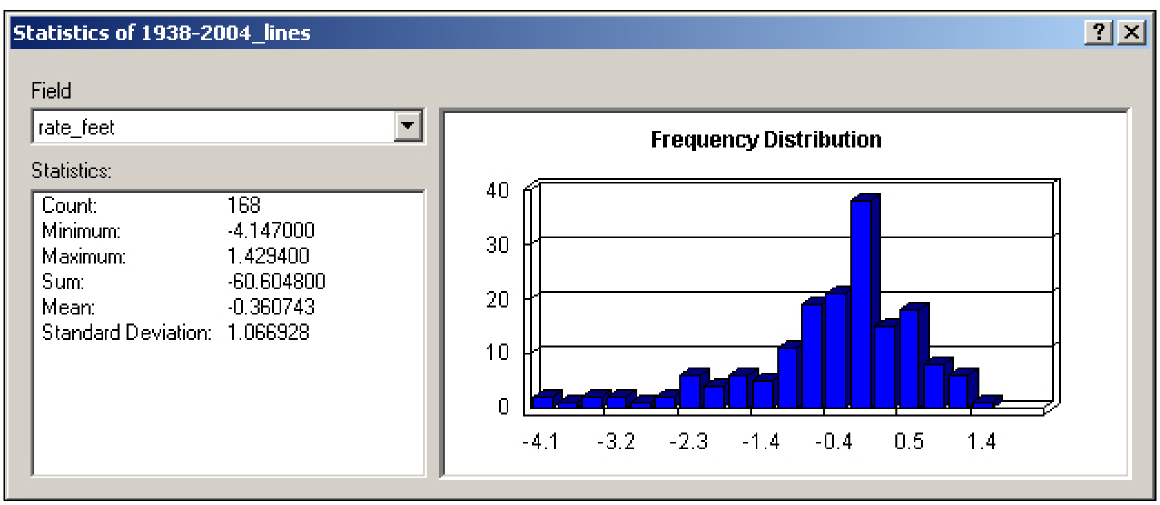


Figure V-11. Shoreline change data distribution calculated along the south coast of Falmouth (Falmouth Harbor to Menauhant Beach) between 1938 and 2004.

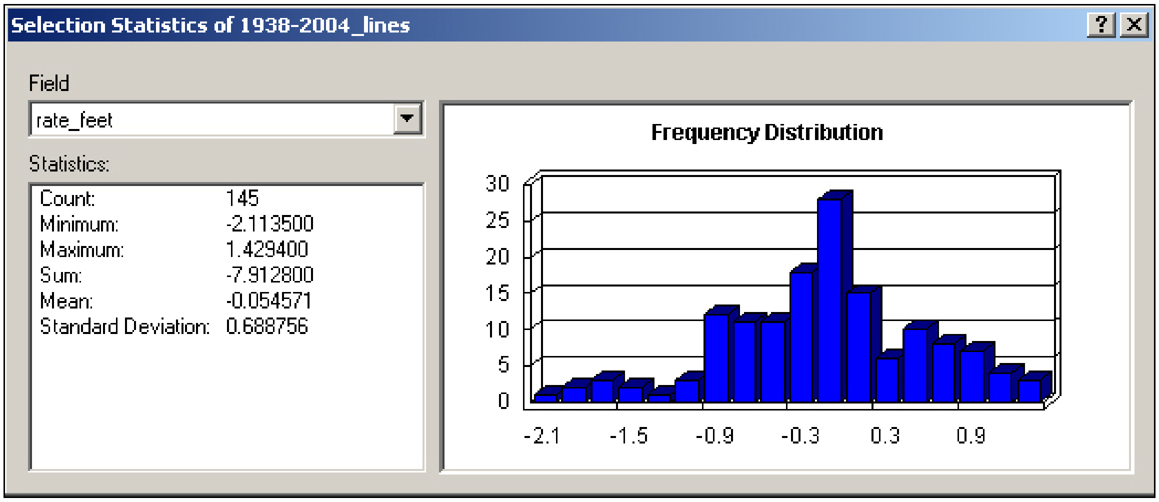


Figure V-12. Shoreline change data distribution calculated along the south coast of Falmouth (Falmouth Harbor to Menauhant Beach) between 1938 and 2004, excluding areas identified as erosion "hot-spots".

V.3 FIELD DATA COLLECTION AND ANALYSIS

A precise description of embayment geometries and hydrodynamic forcing processes is required for the development of numerical models. To support the MEP hydrodynamic and water quality modeling effort in Little Pond, the embayment bathymetry and water elevation variations were measured.

Bathymetry data was collected throughout the estuarine reach of Little Pond. This survey was conducted to address any changes that may have occurred since the installation of the new box culvert and jetties at the inlet, which post-date previous surveys. Tidal elevation measurements were used for both forcing conditions and to evaluate tidal attenuation through the inlet into the system. Figure V-13 shows the location of the tide gauges.

V.3.1 Bathymetry

Bathymetry, or depth, of Little Pond was measured during field a survey in August 2003. The survey was completed using a small vessel equipped with a precision fathometer interfaced to a differential GPS receiver. The fathometer has a depth resolution of approximately 0.1 foot and the differential GPS provides x-y position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder and GPS were logged to a data recorder.

GPS positions and echo sounder measurements were merged to produce data sets consisting of water depth as a function of x-y horizontal position (in Massachusetts Mainland State Plane, 1983). These data were combined with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NGVD 1929 vertical datum (NGVD29). The resulting xyz files were input to mapping software to calculate depth contours for the system shown in Figure V-14. The bathymetry was supplemented by existing data from NOAA collected in 1942 to define the offshore region.

Figure V-13. Little Pond with tide gauge locations labeled as W1 (forcing tide) and W2 (Little Pond tide)

V.3.2 Water Elevation Measurements and Analysis

Changes in water surface elevation were measured using internal recording tide gauges. These tide gauges were installed on fixed platforms (such as pier pilings or screw anchors) to record changes in water pressure over time. Variations in the water surface can be due to tides, wind set-up, or other low frequency oscillations of the sea surface. The tide gauges were installed in 2 locations in Little Pond (Figure V-13) on July 25, 2003 and recovered on September 11, 2003. Data records span at least 29 days to yield an adequate time period for resolving the primary tidal constituents.

The tide gauges used for the study were Brancker XR-420 TG. Data recording was set for 10-minute intervals, with each observation resulting from an average of 60 1-second pressure measurements on 10-minute intervals. These instruments use strain gauge transducers to sense variations in pressure, with resolution on the order of 1 cm (0.39 inches) head of water. Each gauge was calibrated prior to installation to assure accuracy.

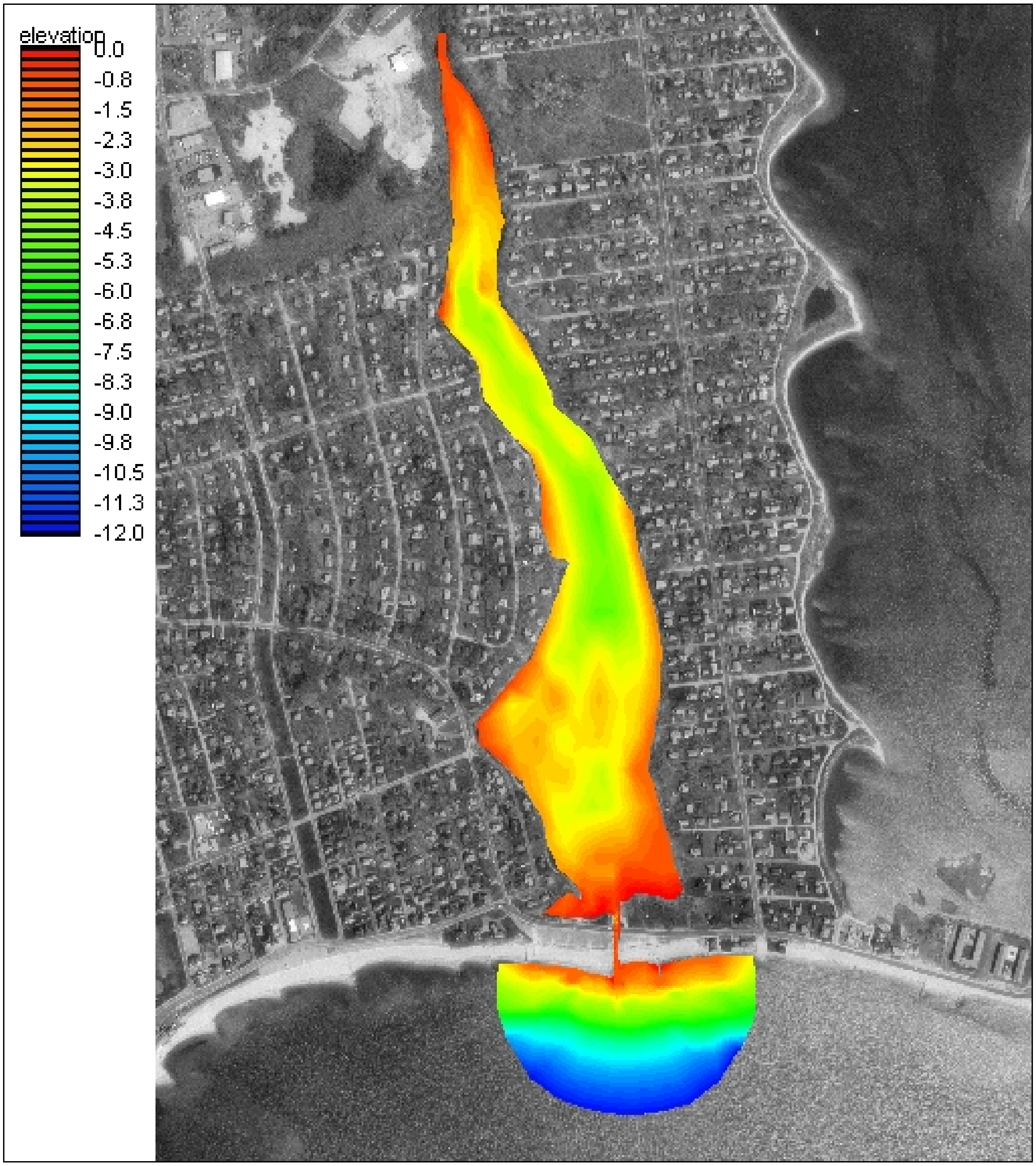


Figure V-14. Little Pond bathymetry showing depth contours of the numerical grid for the Little Pond embayment at 0.25-foot contour intervals relative to NGVD29.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric readings were obtained from the NOAA buoy in Buzzards Bay (site BUZM3), interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in water pressure above the instrument. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (for example, feet of water above the tide gauge). Sensors were surveyed into local benchmarks to provide vertical rectification of the water level;

these survey values were used to adjust the water surface to a known vertical datum. The result from each gauge is a time series representing the variations in water surface elevation relative to NGVD29. Figure V-15 present the water levels at each gauge location.

Figure V-15. Tidal elevation observations offshore Little Pond relative to the Little Pond embayment. (Upper) location W1 and (Lower) location W2 within Little Pond. Locations are shown in Figure V-13.

Figure V-15 shows the tidal elevation for the period July 25 through September 11, 2003 at offshore gauge and in Little Pond. The curves have a predominant 12.42-hour variation around the lunar semi-diurnal (twice-a-day), or M_2, tidal constituent. Modulation of the lunar and solar tides, results in the spring-neap fortnightly cycle, typically evidence by a gradual increase and decrease in tide range. The neap (or minimum) tide range was approximately 1.4 feet, occurring August 21. The spring (maximum) tide range was approximately 2.6 feet, and occurred on August 9.

Analyses of the tide data provided insight into the hydrodynamic characteristics of the system. Harmonic analysis of the tidal time series produced tidal amplitude and phase of the major tidal constituents, and provided assessments of hydrodynamic ‘efficiency’ of the system in terms of tidal attenuation. This analysis also yielded an assessment of the relative influence of non-tidal, or residual, processes (such as wind forcing) on the hydrodynamic characteristics of the offshore waters and the Little Pond embayment.

Harmonic analyses were performed on the time series for each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-2 presents the amplitudes of the eight largest 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 0.56 feet at the offshore gauge. The range of the M_2 tide is twice the amplitude, or 1.12 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.25 feet. The N_2 (12.66-hour period) semi-diurnal tide, also contributes significantly to the total tide signal with an amplitude of 0.20 feet. The M_4 and M_6 tides are higher frequency harmonics of the M_2 lunar tide (exactly half the period of the M_2 for the M_4, and one third of the M_2 for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 is approximately 30% of the amplitude of the M_2 in the offshore gauge (about 0.17 feet). The M_6 amplitude is relatively small throughout the system (less than 0.07 feet). The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon. The observed astronomical tide is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency.

Table V-2. Tidal Constituents, Little Pond July-September 2003

AMPLITUDE (feet)								
Period (hours)	M2 12.42	M4 6.21	M6 4.14	S2 12.00	N2 12.66	K1 23.93	O1 25.82	Msf 354.61
Offshore	0.56	0.17	0.07	0.09	0.20	0.25	0.24	0.03
Little Pond	0.13	0.00	0.01	0.01	0.04	0.14	0.15	0.05

Table V-2 also shows how the constituents vary as the tide propagates through the inlet into the pond. The most significant reduction is in the M_2 amplitude between the Vineyard Sound (offshore) gauge and Little Pond. Usually, a portion of the energy lost from the M_2 tide is transferred to higher harmonics, and is observed as an increase in the amplitude of the M_4 and M_6 constituents over the length of the estuary. However, in the Little Pond system M_2, M_4 and M_6 are all clearly smaller than the amplitudes due to the damping from inlet channel geometry restricting flow and a creating frictional drag through inlet

Table V-3 presents the phase delay of the M_2 tide through the inlet to gauge in Little Pond. Phase delay is another indication of tidal damping resulting from the inlet, and results with a later high tide in Little Pond (Figure V-16). The greater the frictional effects, the longer the delay between locations. The delay in Little Pond is 149 minutes. In general, the delays increase with increasing distance from the offshore gauge. However in this case the inlet is producing the significant damping and hence large phase delay.

Table V-3. M_2 Tidal Attenuation, July-September 2003 (Delay in minutes relative to Vineyard Sound).

Location	Delay (minutes)
Offshore (Vineyard Sound)	--
Little Pond	149.4

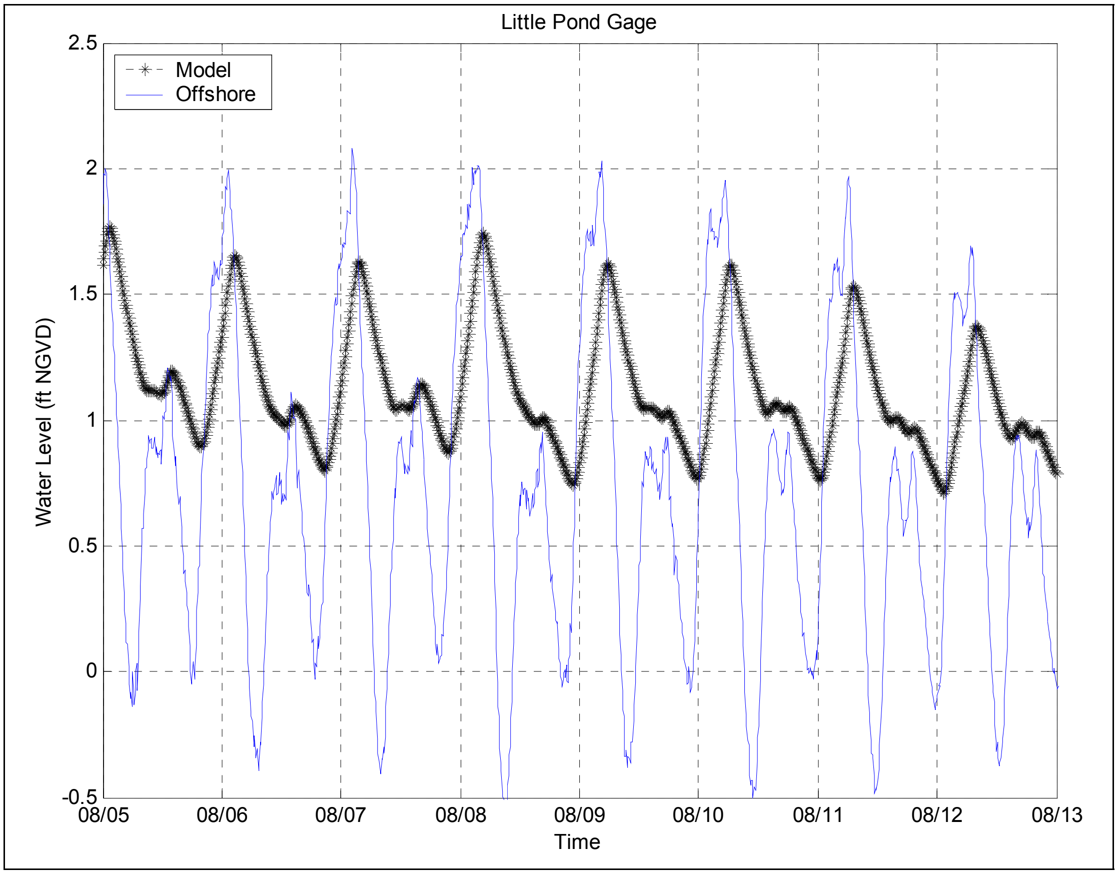


Figure V-16. Comparison of water surface elevation observations for Vineyard Sound (offshore), and in Little Pond. Damping effects are seen as a decrease in the tidal amplitude, as well as a lag in the time of high and low tides from Vineyard Sound.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Vineyard Sound is a relatively shallow semi-enclosed basin, therefore the water surface responds readily to wind-forcing. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large. This analysis calculated the energy (or variance) of the original water elevation time series, and compared these energy values to that of the purely tidal signal (re-created by summing the contributions from the 23 known harmonic constituents). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. The results of this analysis for are presented in Table V-4.

Table V-4. Percentages of Tidal versus Non-Tidal Energy, Little Pond, 2003.				
	Total Variance (ft^2·sec)	Total (%)	Tidal (%)	Non-tidal (%)
Offshore	0.312	100	93.0	7.0
Little Pond	0.052	100	70.3	29.7

The variability analysis shows that a majority of the change in water surface elevation in Vineyard Sound was due to tidal processes. However, in Little Pond more than one-quarter of the energy was the result of non-tidal processes. The significant increase in non-tidal energy is due to tidal damping and frictional losses through the inlet. Local effects of wind blowing across the pond surface will increase the energy of non-tidal processes, however in Little Pond this is not felt to be a significant percentage of the non-tidal effects. These results indicate that hydrodynamic circulation in the embayment is dependent primarily upon tidal processes, with a secondary, but significant contribution from other sources.

V.4 HYDRODYNAMIC MODELING

For the modeling of Little Pond, Applied Coastal utilized a state-of-the-art computer model to evaluate tidal circulation and flushing in these systems. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers.

V.4.1 Model Theory

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

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

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

V.4.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using the shorelines within Little Pond from 1994 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of Little Pond based on the tide gauge data collected near the entrance to Little Pond in Vineyard Sound. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several model calibration simulations for the system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.4.2.1 Grid Generation

The grid generation process was aided by the use of the SMS package. A 1994 digital aerial orthophoto and the bathymetry survey data were imported to SMS, and a finite element grid was generated to represent the embayment and inlet. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of adjoining salt marshes. The bathymetry data was interpolated to the developed finite element mesh of the system. The completed grid consists of 1003 nodes, which describes 312 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -12.0 ft (NGVD 29), along the offshore boundary to Vineyard Sound. In the model grid, a typical marsh plain elevation of +0.2 ft (NGVD 29) was used, based on spot surveys of the limited areas of marsh. The completed grid mesh of Little Pond is shown in Figure V-17.

The finite element grid for each system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties in Little Pond. Areas of marsh were included in the model because they represent a key portion of the estuary along the northern extents of Little Pond, and may have an effect on hydrodynamics within the system. Fine grid resolution was required to simulate the channel constrictions and the inlet structures which have a significant impact the estuarine hydrodynamics. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary.

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in inlet were designed to provide a more detailed analysis in these regions of rapidly varying flow. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as midsections of Little Pond and the offshore boundaries. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

Figure V-17. Hydrodynamic model grid mesh for Little Pond.

V.4.2.2 Boundary Condition Specification

Two types of boundary conditions were employed for the RMA-2 model of the Little Pond system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. A tidal boundary condition was specified at the inlet. Tide gauge (TDR) measurements provided the required data. The rise and fall of the tide in Vineyard Sound is the primary driving force for estuarine circulation in this system. For the boundary a dynamic (time-varying) water surface elevation condition was specified every model time step (10 minutes) to represent the tidal forcing.

V.4.2.3 Calibration

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

Calibration of the hydrodynamic model requires a close match between the modeled and measured tides in the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, an approximate seven-day period (14 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.3.2. The seven-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents

The calibration was performed for a seven-day period beginning August 5, 2003 at 1800 EDT. This representative time period included the spring tide range of conditions, where the tide range and tidal currents are greatest.

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

V.4.2.3.1 Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where velocities are relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude damping and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient, and is applied to grid areas by user specified material types. Initially, Manning's friction coefficients between 0.020 and 0.08 were specified for all element material types. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels and marsh plains with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy/silty bottom found in Little Pond, versus the rock lined channel in the inlet, which provides greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-5. The extents of each material type are shown in Figure V-18.

Table V-5. Manning's Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-18.	
System Embayment	Bottom Friction
Offshore	0.033
Culvert	0.050
Inlet Channel	0.042
Little Pond	0.030

V.4.2.3.2 Turbulent Exchange Coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 50 and 80 lb-sec/ft^2.

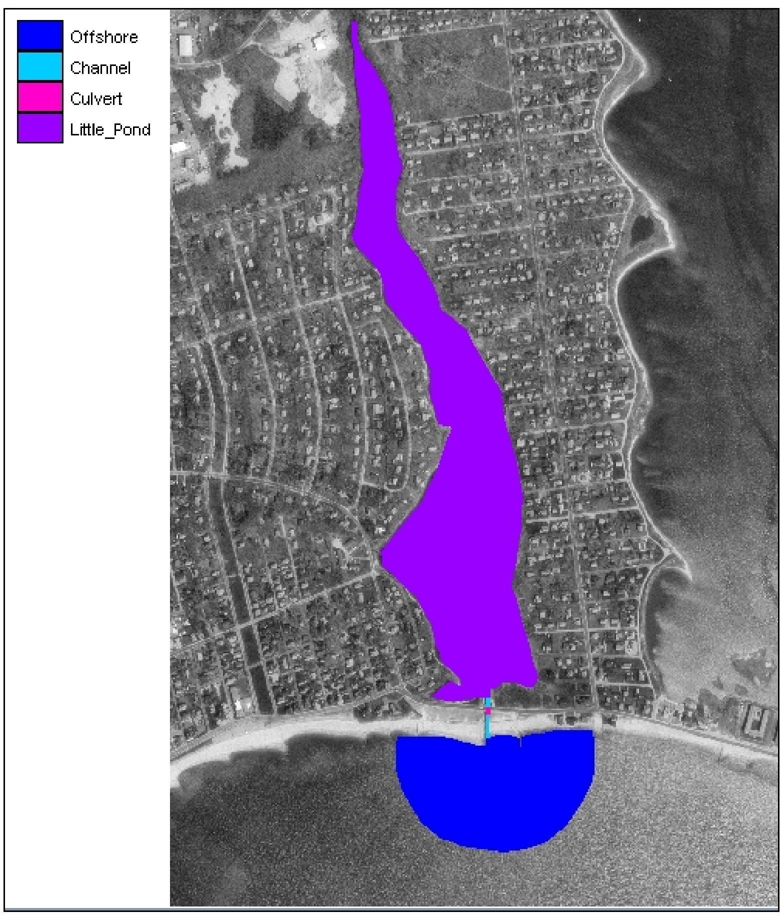


Figure V-18. Hydrodynamic model grid material properties. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.4.2.3.3 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model predictions for the first tide gauge (TDR) deployment was achieved using the aforementioned values for friction and turbulent exchange. Figure V-19 illustrates the seven-day calibration simulation for Little Pond. Modeled (solid line) and measured (dotted line) tides are illustrate the tide record.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-6 for the calibration period differ from those in Table V-3 because constituents were computed for only the seven-day section of the 26-days represented in Table V-3. Table V-6 compares tidal constituent height and phase for modeled and measured tides at the TDR locations.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.02 ft in Little Pond, which is within the accuracy of the tide gauges (0.032 ft). Similarly, time lag errors were typically less than the time increment resolved by the model (0.10 hours or 10 minutes), indicating good agreement between the model and data.

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

Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Little Pond	0.17	0.01	0.01	0.23	68.6	110.9
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Little Pond	0.16	0.02	0.01	0.25	67.9	107.9
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Little Pond	-0.01	0.01	0.00	0.02	-1.3	-3.2

The hydrodynamic model was then verified over a separate seven-day period, beginning August 21, 2003 at 2100 EDT, to ensure the coefficients established during calibration would accurately represent the estuary process beyond the calibration period. The verification period represented the variation from neap tide range of conditions toward spring tide conditions. The verification resulted in excellent agreement between modeled and measured tides. Figure V-20 show the comparison of modeled and measured tides, while Table V-7 shows the constituent agreement.

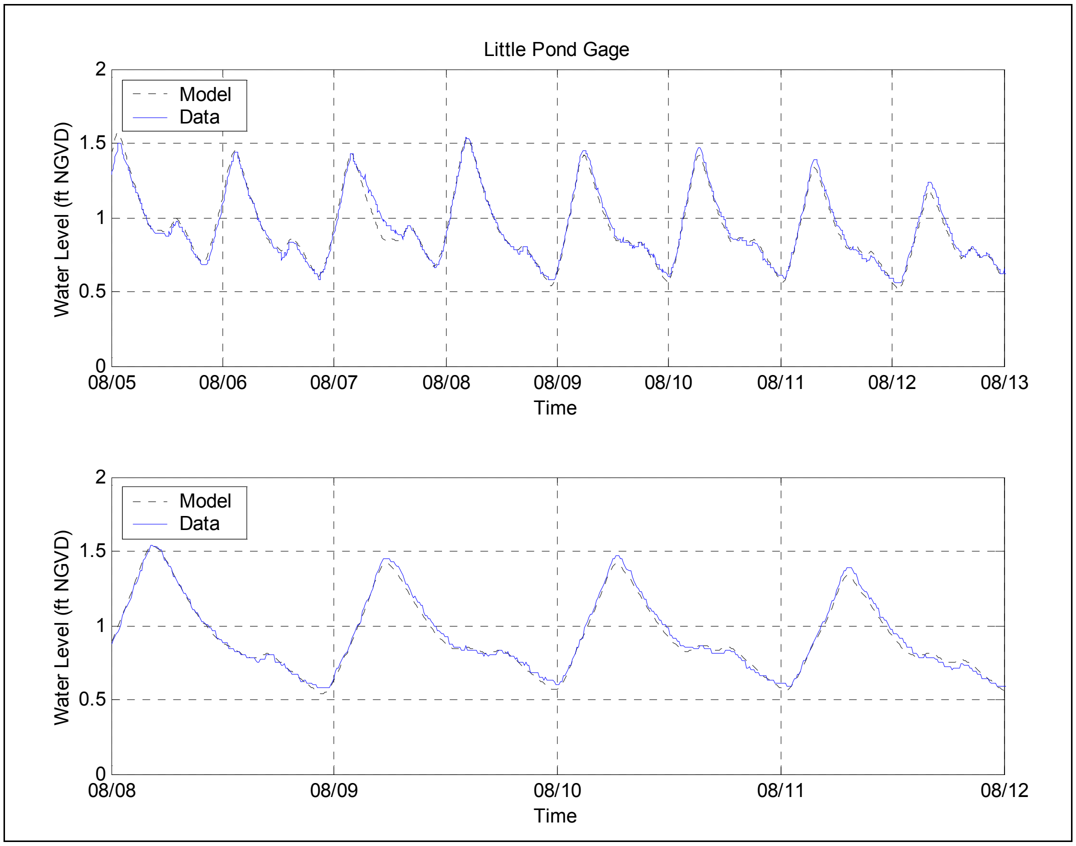


Figure V-19. Comparison of model output and measured tides for the gauge site in Little Pond. Differences between the observed (solid line) and modeled (dashed line) tidal elevation are generally imperceptible.

Table V-7. Tidal constituents for measured water level data and verified model output for Little Pond.

Model verification run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Little Pond	0.13	0.01	0.01	0.18	-16.7	83.1
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Little Pond	0.13	0.01	0.01	0.18	-17.1	83.3
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Little Pond	0.01	0.00	0.00	0.02	-0.9	0.2

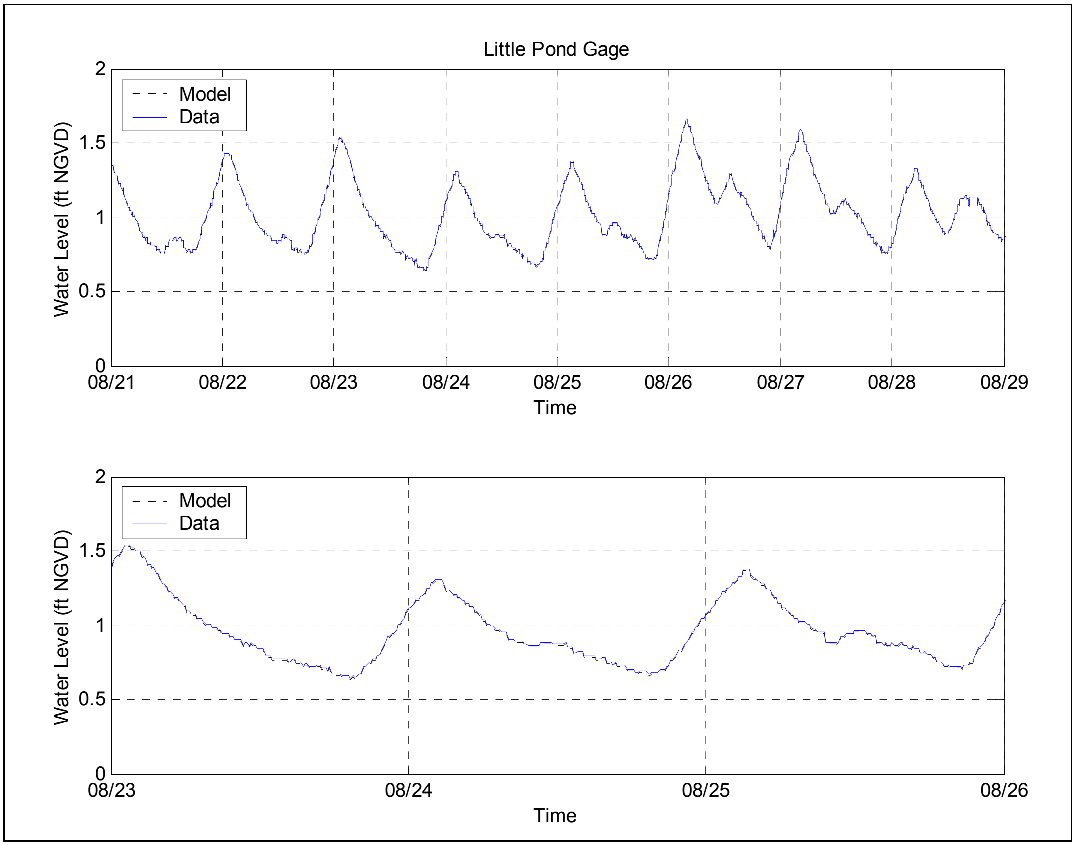


Figure V-20. Comparison of model output and measured tides for the gauge site in Little Pond.

V.4.2.4 Model Circulation Characteristics

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

Examining the results from the model run of Little Pond shows flood velocities in the channels are slightly larger than velocities during maximum ebb. The maximum velocities occur in the inlet with Vineyard Sound, the maximum depth-averaged flood velocities in the model are approximately 6.5 feet/sec, while maximum ebb velocities are about 5.5 feet/sec. A close-up of the model output is presented in Figures V-21 and V-22, which shows contours of velocity magnitude, along with velocity vectors which indicate the direction of flow, for a single model time-step.

Figure V-21. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

Figure V-22. Example of hydrodynamic model output for a single time step where maximum ebb velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

In addition to depth averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. For the flushing analysis in the next section, flow rates were computed across the inlet. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-23. Maximum flow rates occur during flood tides in this system, an indication that this estuary system is flood dominant, and likely a sediment sink (a system that accumulates sediment). During spring tides, the maximum flood flow rates reach 105 ft^3/sec through the inlet. Maximum ebb flow rates are less, approximately 65 ft^3/sec.

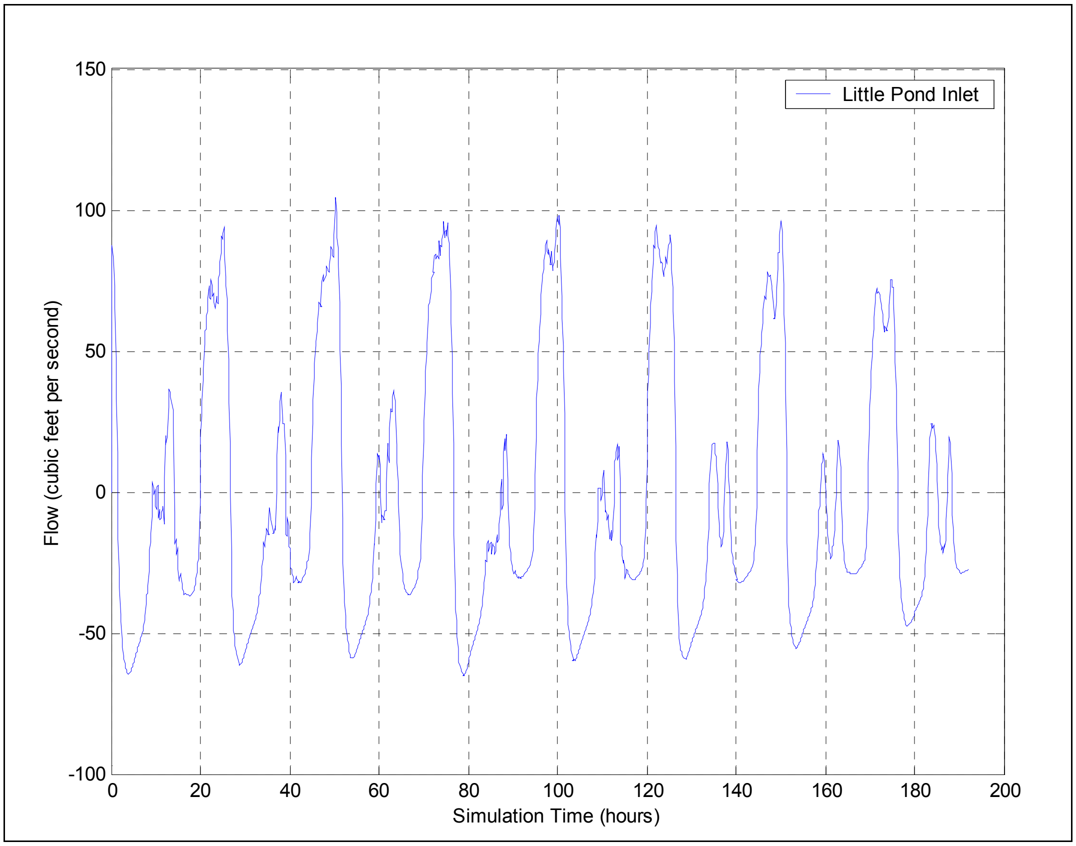


Figure V-23. Time variation of computed flow rate for inlet to Little Pond. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Plotted time period represents three tide cycles (12.42 h cycle). Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

V.5 FLUSHING CHARACTERISTICS

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

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).

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 offshore provide the only means of reducing the high nutrient levels. This is a valid approach in the case of Little Pond, since Vineyard Sound has relatively higher quality water then the Pond.

The rate of pollutant/nutrient loading and the quality of adjacent offshore waters 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 system (see Section VI).

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism). Units then were converted to days. The volume of the entire estuary was computed as cubic feet.

The residence time was averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and is listed in Table V-8. The modeled time period used to compute the flushing rate was the modeled calibration period, and included the transition from spring to neap tide conditions. The model calculated flow crossing specified grid lines along the inlet to compute the tidal prism volume. Since the 7.25-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system embayment.

Table V-8. Embayment mean volume and average tidal prism during simulation period.		
Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Little Pond	6,986,660	785,605

Table V-9. Computed residence times for Little Pond system.	
Embayment	System Residence Time (days)
Little Pond	4.6

The computed flushing rate for the Little Pond shows that system takes approximately 4.6 days for the volume of the system to be exchanged. This suggests that the system has marginal tidal flushing. This method assumes all the water in the system is exchanged, while in reality the water in the southern end of the system is most likely to be exchanged . Meaning that water quality continues to decrease in the upper portion of the system.

Generally, possible errors in computed residence times can be linked to two sources: the bathymetry information and simplifications employed to calculate residence time. In this study, the most significant errors associated with the bathymetry data result from the process of interpolating the data to the finite element mesh, which was the basis for all the flushing volumes used in the analysis. In addition, limited topographic measurements were available on the marsh plains. Minor errors may be introduced in residence time calculations by simplifying assumptions. Flushing rate calculations assume that water exiting the estuary does not return on the following tidal cycle. For regions where a strong littoral drift exists, this assumption is valid. However, water exiting a small sub-embayment on a relatively calm day may not completely mix with estuarine waters. In this case, the “strong littoral drift” assumption would lead to an under-prediction of residence time. Since littoral drift along the coast of Vineyard Sound is typically strong due to local winds inducing tidal mixing within the regional estuarine systems, the “strong littoral drift” assumption will cause only minor errors in residence time calculations. Based on our knowledge of estuarine processes, we estimate that the combined errors due to bathymetric inaccuracies represented in the model grid and the “strong littoral drift” assumption are within 10% to 15% of “true” residence times.

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

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayment

Extensive field measurements and hydrodynamic modeling of the embayment were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a calibrated model output representing the transport of water within the system embayment. Files of node locations and node connectivity for the RMA-2V model grid were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 10-tidal cycle period in August 2003. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model had reached a dynamic "steady state", and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayment

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

VI.1.3 Measured Nitrogen Concentrations in the Embayment

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. Typically, seven years of data (collected between 1997 and 2004) were available for stations monitored by Falmouth PondWatch with SMAST Staff in the Little Pond system.

Table VI-1. PondWatcher measured data, and modeled Total Nitrogen concentrations for the Little Pond system used in the model calibration plots of Figure VI-2. All concentrations are given in mg N/L. “Data mean” values are calculated as the average of the separate yearly means.

Sub-Embayment	WQ Station	1997 mean	1998 mean	1999 mean	2000 mean	2001 mean	2002 mean	2003 mean	2004 mean	data mean	s.d. all data	N	model min	model max	model average
Little Pond Head	LP Head	2.049	3.186	2.156	2.833	2.674	1.752	2.373	1.345	2.321	0.746	29	1.920	2.501	2.236
Little Pond upper	LP1	0.854	0.777	0.851	1.096	0.904	1.067	0.942	0.977	0.942	0.266	54	0.939	0.969	0.955
Little Pond mid	LP2	0.654	0.745	0.962	1.006	0.900	0.931	0.829	1.082	0.898	0.267	54	0.827	0.849	0.837
Little Pond lower	LP3	0.486	0.722	0.742	0.734	0.831	0.744	0.853	0.883	0.745	0.237	58	0.623	0.771	0.705

Figure VI-1. Estuarine water quality monitoring station locations in the Little Pond 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 Little Pond estuary system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of the Little Pond. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other Cape Cod embayments, including systems in Falmouth (Ramsey *et al*., 2000); Mashpee, MA (Howes *et al*., 2004) and Chatham, MA (Howes *et al*., 2003).

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

VI.2.1 Model Formulation

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

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

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

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

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

VI.2.2 Water Quality Model Setup

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

Based on measured stream flow rates from SMAST and groundwater recharge rates from the USGS (Section IV), the hydrodynamic model was set-up to include the latest estimate of surface water flow from the Little Pond Stream. The Little Pond Stream has a measure flow rate of 0.83 ft^3/sec (2,026 m^3/day), which is 4.6% of the volume of the average tide prism of the Little Pond.

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

VI.2.3 Boundary Condition Specification

Mass loading of nitrogen into each model included 1) sources developed from the results of the watershed analysis, 2) estimates of direct atmospheric deposition, 3) summer benthic regeneration, 4) point source input developed from measurements of the Little Pond Stream. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed direct atmospheric deposition load for Little Pond was evenly distributed at grid cells that formed the perimeter of the embayment. Benthic regeneration load was distributed among another sub-set of grid cells which are in the interior portion of each basin.

The loadings used to model present conditions in Little Pond estuary systems are given in Table VI-2. Watershed and depositional loads were taken from the results of the analysis of Section IV. Summertime benthic flux loads were computed based on the analysis of sediment cores in Section IV. The area rate (g/sec/m^2) of nitrogen flux from that analysis was applied to the surface area coverage computed for each sub-embayment (excluding marsh coverages, when present), resulting in a total flux for each embayment (as listed in Table VI-2). Due to the highly variable nature of bottom sediments and other estuarine characteristics of coastal embayments in general, the measured benthic flux for existing conditions also is variable. For present conditions, some regions (generally in deeper basins) of the embayment have a relatively high rate of summer time nitrogen release, while other regions show a low rate of release or even an slight uptake. This pattern is typical of these shallow estuaries, such as adjacent Great, Green and Bournes Ponds.

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 Vineyard Sound was set at 0.280 mg/L, based on SMAST data from the Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Vineyard Sound.

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Little Pond	13.022	0.584	1.581
Surface Water Sources			
Little Pond Stream	6.052	-	-

VI.2.4 Model Calibration

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

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

Embayment Division	E m^2/sec
Lower Little Pond	3.0
Upper Little Pond	2.0
Little Pond Stream	0.12
Inlet	7.0
Vineyard Sound	10.0

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

For model calibration, the mid-point between maximum modeled TN and average modeled TN was compared to mean measured TN data values, at each of the 4 PondWatch water-quality monitoring stations. The calibration target would fall between the modeled mean and maximum TN because the monitoring data are collected, as a rule, during mid ebb tide.

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for the system. The model fit is exceptional for the Little Pond model, with root mean square error (rms) of 0.05 mg/L and an R^2 correlation coefficient of 0.99.

A contour plot of calibrated model output is shown in Figure VI-3 for Little Pond. In the figure, color contours indicate nitrogen concentrations throughout the model domain. The output in the figure show average total nitrogen concentrations, computed using the full 5-tidal-day model simulation output period.

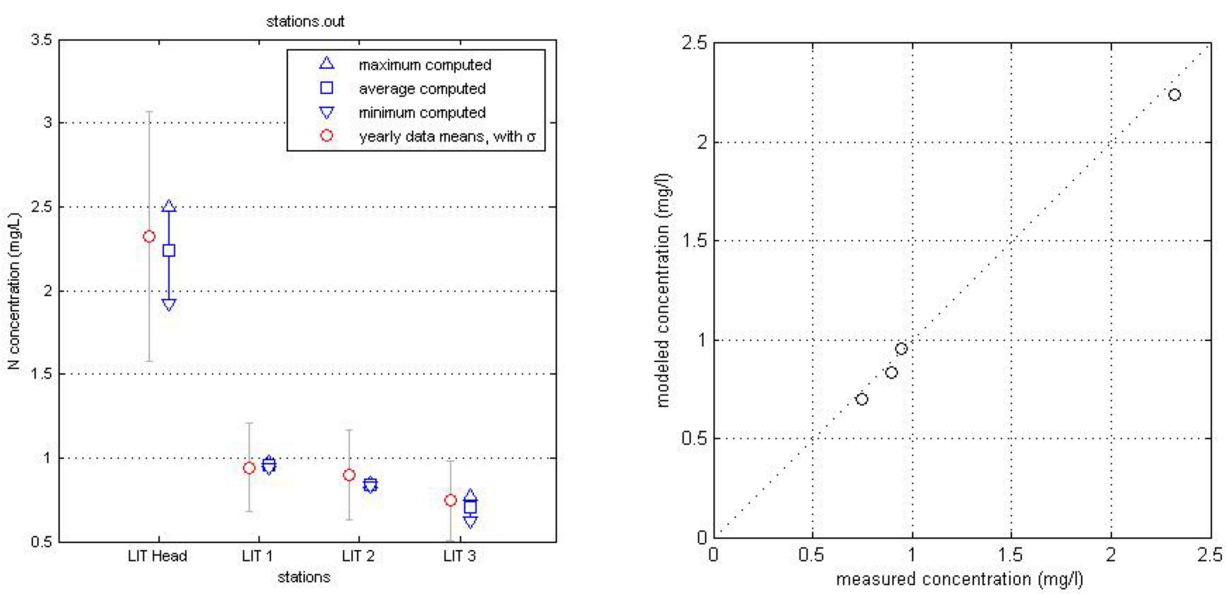


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in Little Pond. For the left plot, station labels correspond with those provided in Table VI-1 and Figure VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed concentration for the same period (square markers). Measured data are presented as the July-August mean (1997-2004) at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

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 Little Pond system using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, at the freshwater stream discharges, and groundwater inputs. The open boundary salinity was set at 29.6 ppt. For surface water steams and groundwater inputs salinities were set at 0 ppt. Surface water stream flow rates for the streams were the same as those used for the total nitrogen model, as presented earlier in this section. Groundwater input used for the model was 1.45 ft^3/sec (3,025 m^3/day). Groundwater flows were distributed evenly in each model through the use of several 1-D element input points positioned along each model's land boundary.

Comparisons of modeled and measured salinities are presented in Figure VI-4, with contour plots of model output shown in Figure VI-5. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in Little Pond. The rms error of the models was 1.3 ppt, and correlation coefficient was 0.95. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical systems.

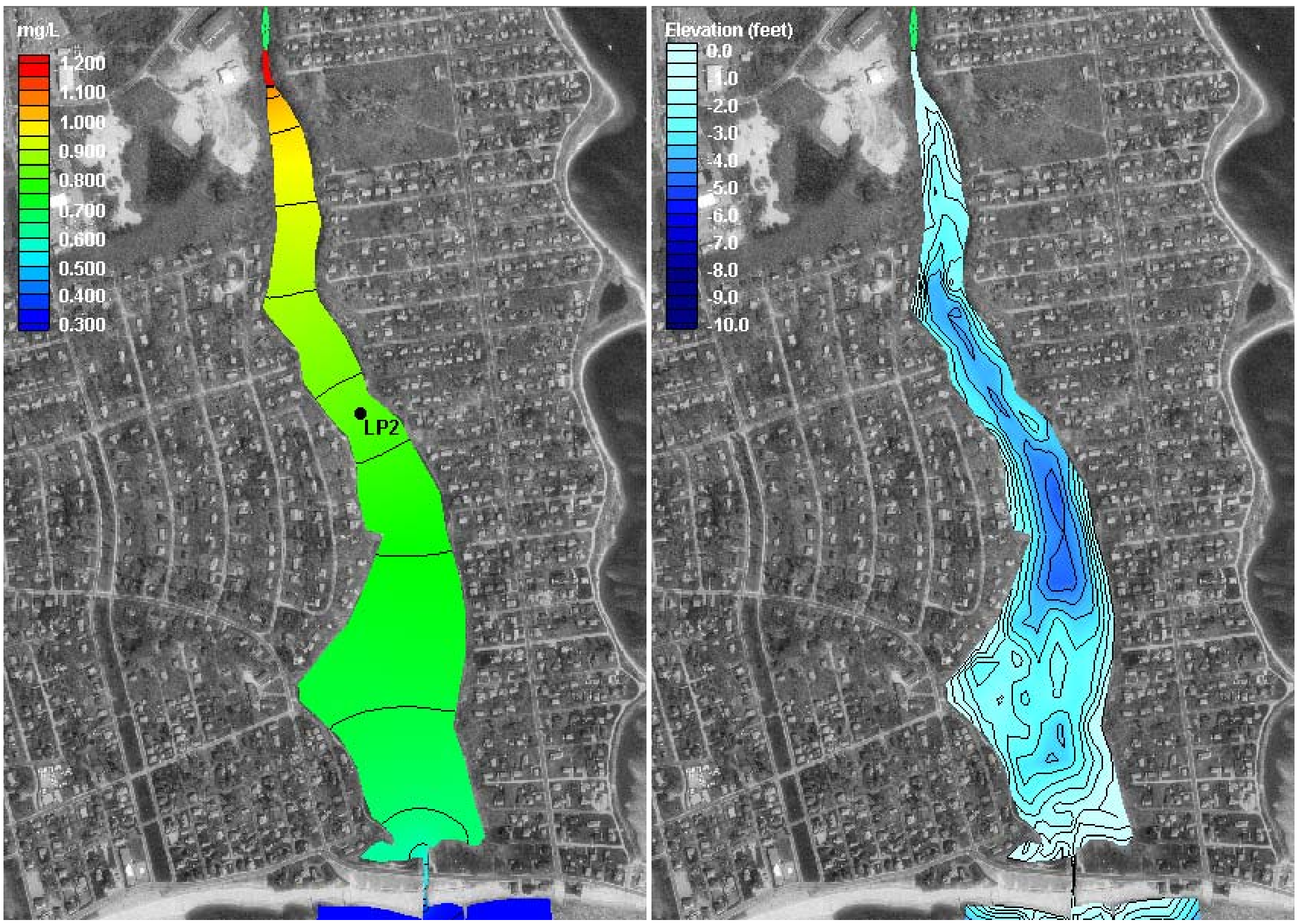
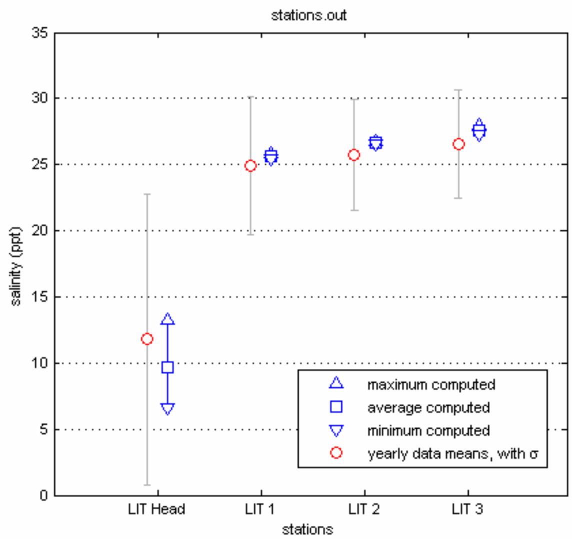


Figure VI-3. Contour plots of average total nitrogen concentrations from results of the present conditions loading scenario (left) and the bathymetry (right), for Little Pond. The approximate location of the sentinel threshold station for Little Pond (LP2) is shown.



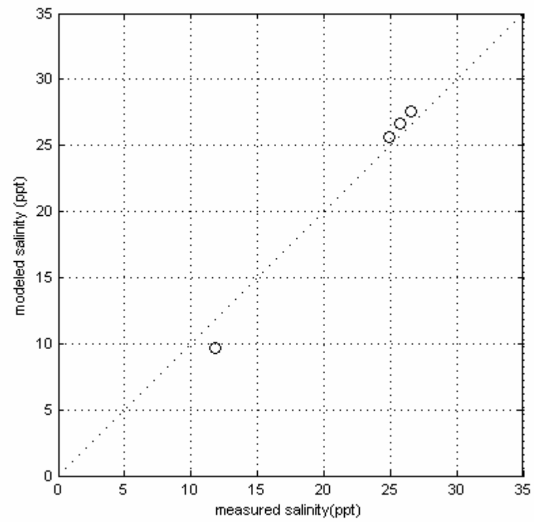


Figure VI-4. Comparison of measured and calibrated model output at stations in Little Pond. For the left plots, stations labels correspond with those provided in Table VI-1 and Figure VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the July-August mean (1997-2004) at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

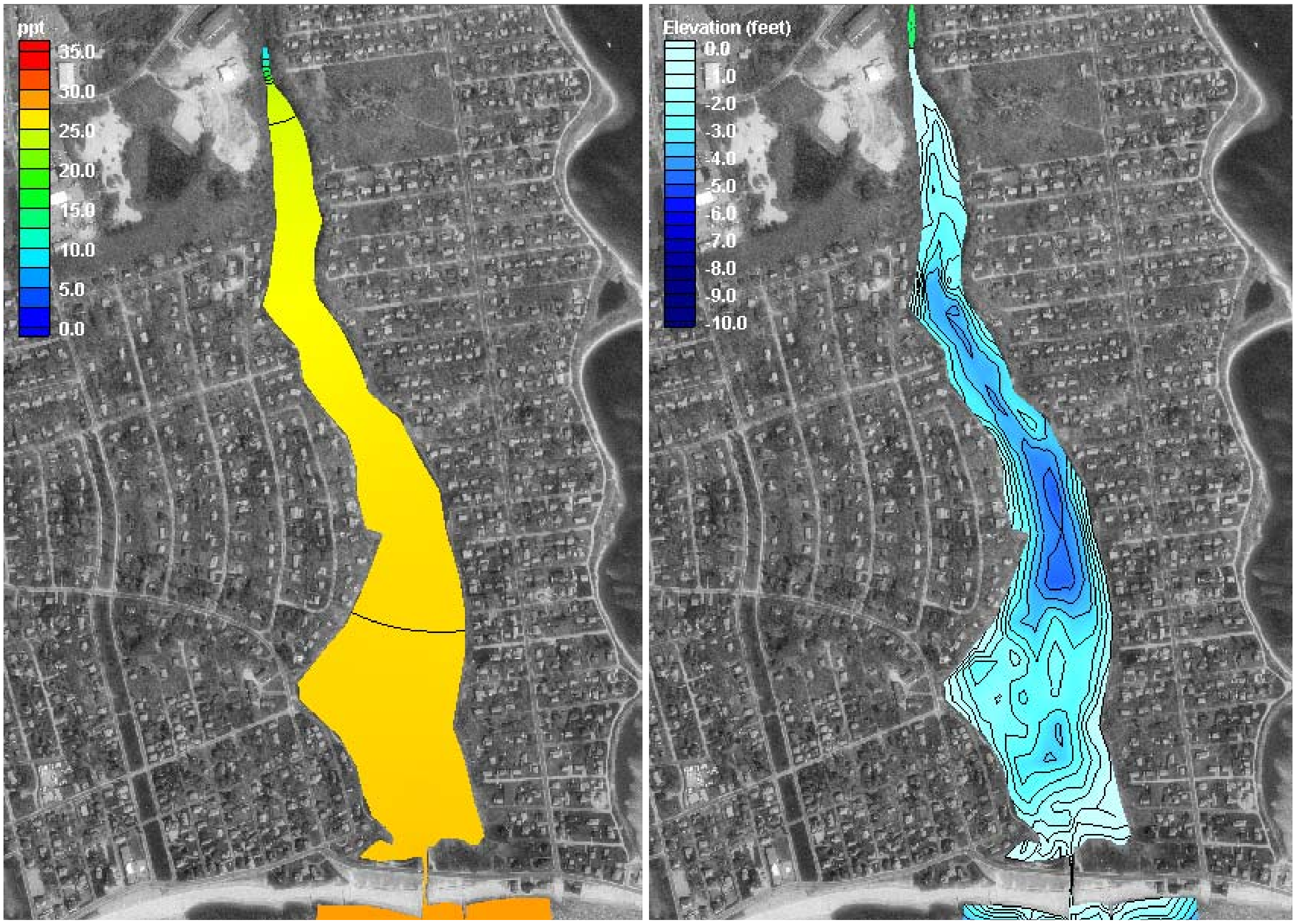


Figure VI-5. Contour plots of modeled salinity (ppt) and bathymetry in Little Pond.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Little Pond 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
Little Pond	13.022	13.381	+2.8%	1.145	-91.2%
Surface Water Sources					
Little Pond Stream	6.052	7.279	+20.3%	0.386	-93.6%

VI.2.6.1 Build-Out

In general, certain sub-watersheds would be impacted more than others. However, overall the increase in nitrogen load projected for Little Pond at build-out is relatively small, as its watersheds are currently nearly fully developed. For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions by over 90%.

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Little Pond, 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)
Little Pond	13.381	0.584	1.607
Surface Water Sources			
Little Pond Stream	7.279	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality model of Little Pond was run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. Total N concentrations increased in the stream sub-watershed to the system (21%) and the lower groundwater sub-watershed (5%). The increase in the stream sub-watershed is due in part to the change in the watershed load and in part to a shift in the projected stream load from the measured value to the modeled future value. This latter shift has a slightly lower attenuation rate (30%) than that measured under current conditions (41%) as projections of shifts in attenuation under increasing load can be difficult and the lower attenuation rate is conservative. Color contours of model output for the build-out scenario are present in Figure VI-6. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-3, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Little Pond system. Sentinel threshold stations are in bold print.				
Sub-Embayment	WQ Station	present (mg/L)	build-out (mg/L)	% change
Little Pond Head	LPHead	2.236	2.700	+20.7%
Little Pond - upper	LP1	0.955	1.030	+7.9%
Little Pond - mid	**LP2**	**0.837**	**0.888**	**+6.1%**
Little Pond - lower	LP3	0.705	0.737	+4.7%

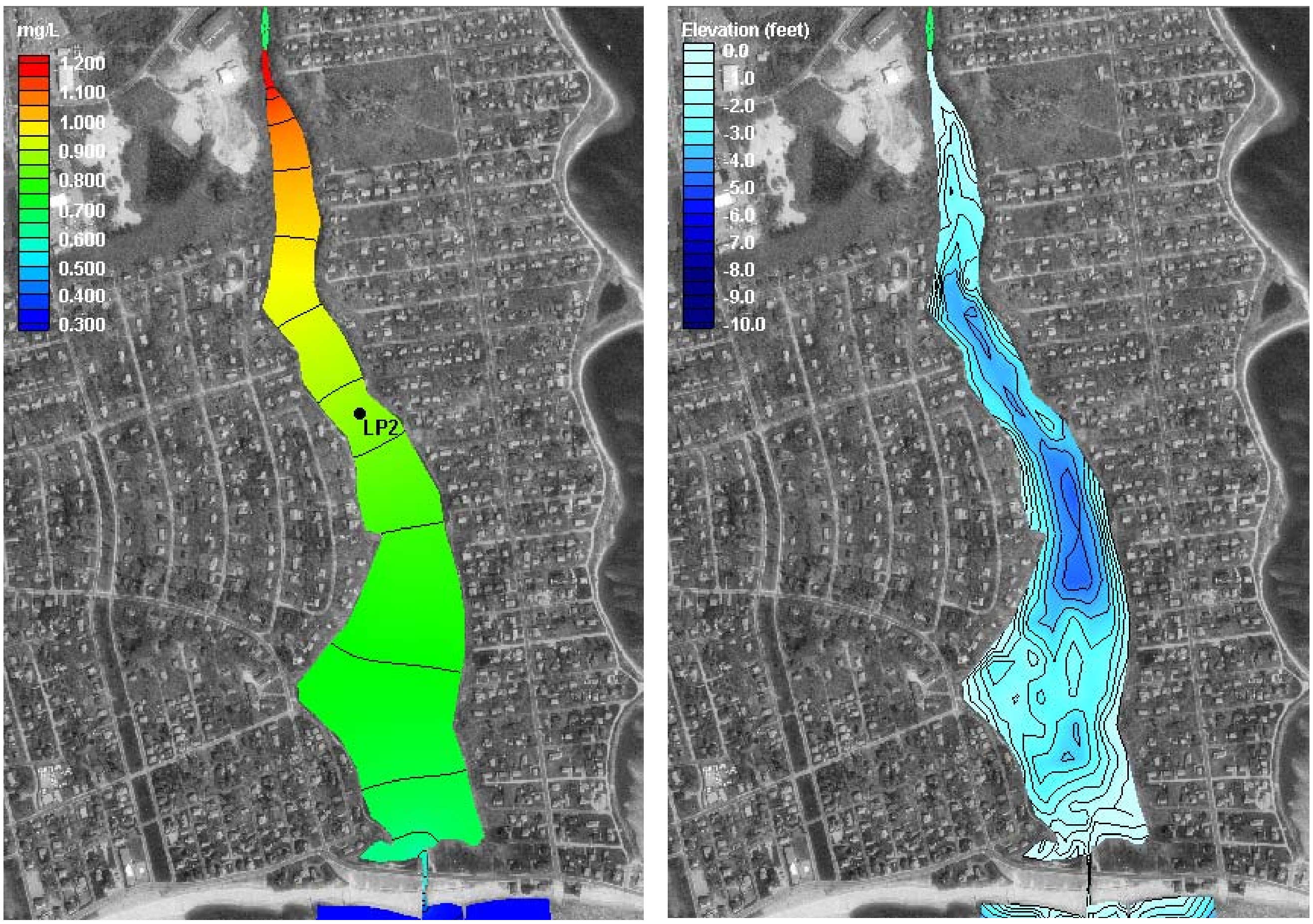


Figure VI-6. Contour plots of modeled total nitrogen concentrations (mg/L) in Little Pond, for projected build-out loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Little Pond (LP2) is shown.

VI.2.6.2 No Anthropogenic Load

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load ("no load") scenario is shown in Table VI-7. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in the above Section). 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 Little Pond, 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)
Little Pond	1.145	0.584	0.428
Surface Water Sources			
Little Pond Stream	0.386	-	-

Following development of the nitrogen loading estimates for the no load scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Again, total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from “no load” was significant as shown in Table VI-8, with reductions greater than 50% occurring the all portions of the systems. Results for each system are shown pictorially in Figure VI-7.

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic (“no load”) scenario, with percent change, for the Little Pond system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). Sentinel threshold stations are in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	no-load (mg/L)	% change
Little Pond Head	LPHead	2.236	0.244	-89.1%
Little Pond - upper	LP1	0.955	0.336	-64.9%
Little Pond - mid	**LP2**	**0.837**	**0.336**	**-59.9%**
Little Pond - lower	LP3	0.705	0.329	-53.3%

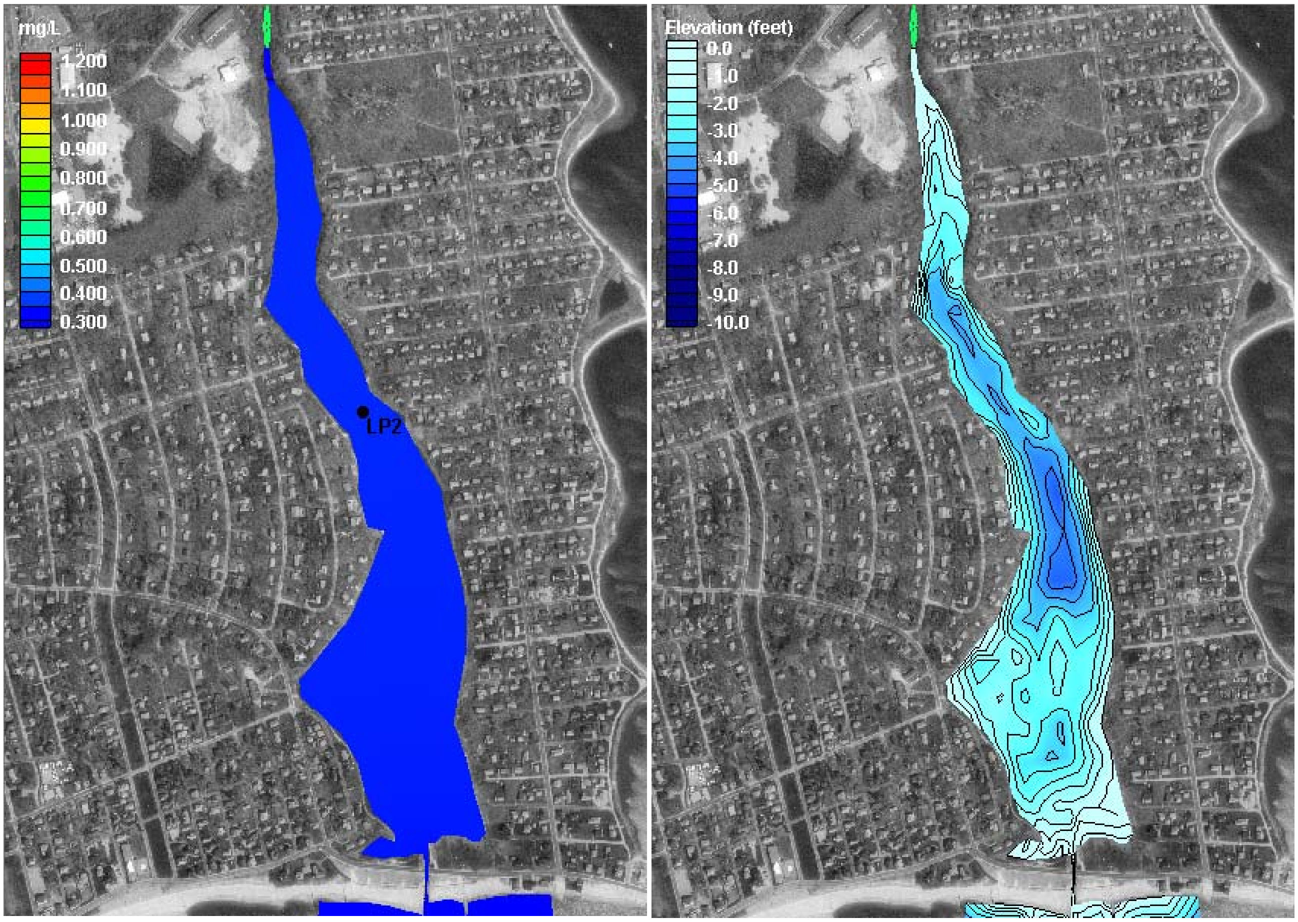


Figure VI-7. Contour plots of modeled total nitrogen concentrations (mg/L) in Little Pond, for no anthropogenic loading conditions, and bathymetry. The approximate location of the sentinel threshold station for Little Pond (LP2) is shown.

VII. ASSESSMENT OF EMBAYMENT NUTRIENT RELATED ECOLOGICAL HEALTH

The nutrient related ecological health of an estuary can be gauged by the nutrient, chlorophyll, and oxygen levels of its waters and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Little Pond embayment system, our assessment is based upon data from the water quality monitoring database and our surveys of eelgrass distribution, benthic animal communities and sediment characteristics, and dissolved oxygen records conducted during the summer and fall of 2002. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within the upper portion of the Little Pond system, as well as closer to the inlet to Little Pond, to record the frequency and duration of low oxygen conditions during the critical summer period. These stations are above and below the sentinel station in Little Pond (See Section VIII). 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 Little Pond System was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. Within the Little Pond System, temporal changes in eelgrass distribution provides a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing-new inlet) in nutrient enrichment.

In areas that do not support eelgrass beds, benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill (Hampson, 1978), benthic population studies in Buzzards Bay (e.g. Hampson, 1989) and New Bedford (SMAST, unpublished data), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the level of diversity (H') and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

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

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

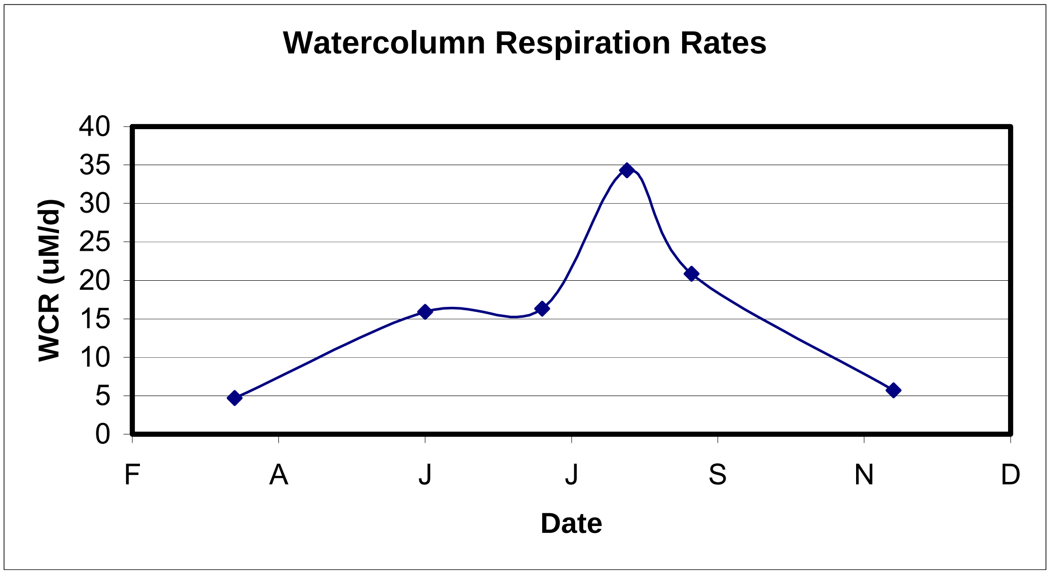


Figure VII-1. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

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

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 28 day deployment period that these parameters were below/above various benchmark concentrations (Tables VII-1, VII-2). These data indicate both the temporal pattern of minimum or maximum levels of these critical nutrient related constituents, as well as the intensity of the oxygen depletion events and phytoplankton blooms. However, it should be noted that the frequency of oxygen depletion needs to be integrated with the actual temporal pattern of oxygen levels, specifically as it relates to daily oxygen excursions. The 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 indicates that the upper tidal reaches of the Little Pond System is eutrophic.

The level of oxygen depletion and the magnitude of daily oxygen excursion and chlorophyll a levels indicate highly nutrient enriched waters and impaired habitat quality at all mooring sites within each estuary (Figures VII-3 through VII-6). The oxygen data is consistent with high organic matter loads from phytoplankton production (chlorophyll a levels) indicative of nitrogen enrichment and eutrophication of these estuarine systems. The oxygen records further indicate that both the upper and lower tidal reaches of each estuary have large daily oxygen excursions, which further supports the assessment of a high degree of nutrient enrichment. In general, the magnitude of oxygen over-saturation (high values) and depletion (low values) are similar in both upper and lower embayments. This is consistent the observed chlorophyll levels (Figures VII-5, VII-6). Similarly, both the significant dissolved oxygen excursion and high chlorophyll levels at both sites are consistent with the watercolumn total nitrogen data (Table VI-1) which showed high levels (>0.7 mgN/L) and only a small gradient from the lower basin (0.745 mg N/L) to the upper station (0.942 mg N/L) above the upper mooring site.

Specifically, the dissolved oxygen records indicate that the upper region of Little Pond is currently under seasonal oxygen stress, consistent with nitrogen enrichment (Table VII-1). That the cause is eutrophication is supported by the high levels of chlorophyll a, with blooms of >20 µg/L occurring at both sites and concentrations >15 ug/L occurring over ~15% of the deployment period (Table VII-2). Oxygen conditions and chlorophyll a levels were only slightly improved in the lower versus upper Little Pond locations. The upper and lower sites showed oxygen depletions below 3 mg/L on 5% and 2% of the 28 day sampling period and were below 6 mg/L 27% and 21% of time, respectively. The dissolved oxygen, chlorophyll a and total nitrogen data show a consistent pattern of a highly nitrogen enriched embayment with only a modest longitudinal gradient. The impairment of the habitat quality in Little Pond is clearly directly related to its eutrophic condition. Further evidence of the degree of this impairment is related to its periodic macroalgal blooms and fish kills and in the eelgrass and infaunal community distributions discussed below.

Figure VII-2. Aerial Photograph of the Little Pond system in Falmouth showing locations of Dissolved Oxygen mooring deployments conducted in the Summer of 2002. The sentinel station is located about mid way between the mooring sites.

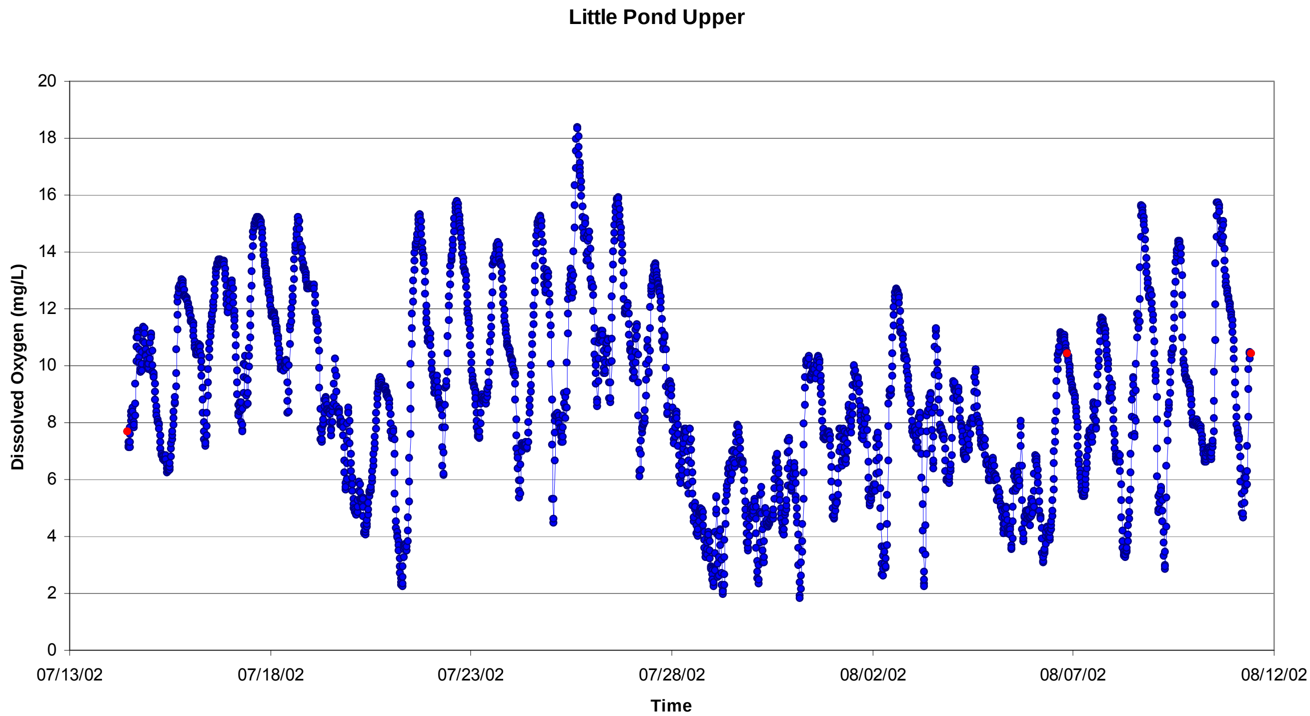


Figure VII-3. Bottom water record of dissolved oxygen at Little Pond Upper station, Summer 2002. Calibration samples represented as red dots.

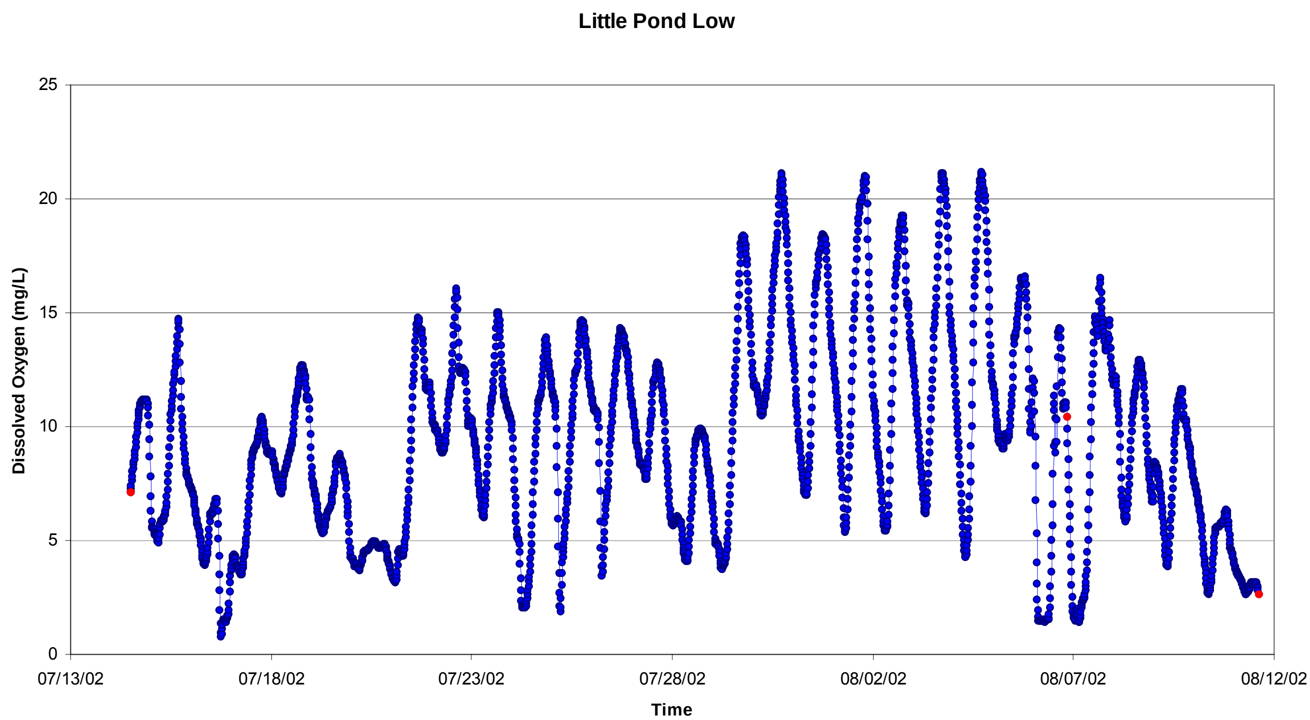


Figure VII-4. Bottom water record of dissolved oxygen in Little Pond Low station, Summer 2002. Calibration samples represented as red dots.

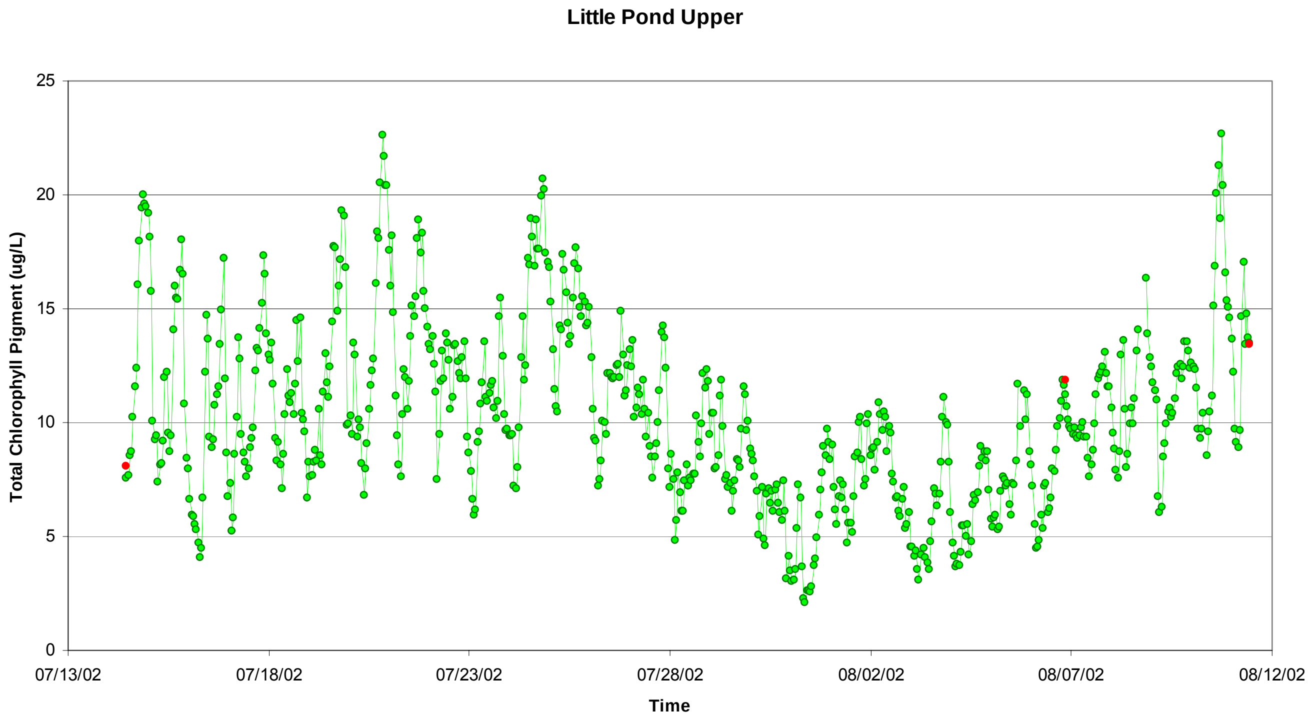


Figure VII-5. Bottom water record of Chlorophyll-*a* at Little Pond Upper station, Summer 2002. Calibration samples represented as red dots.

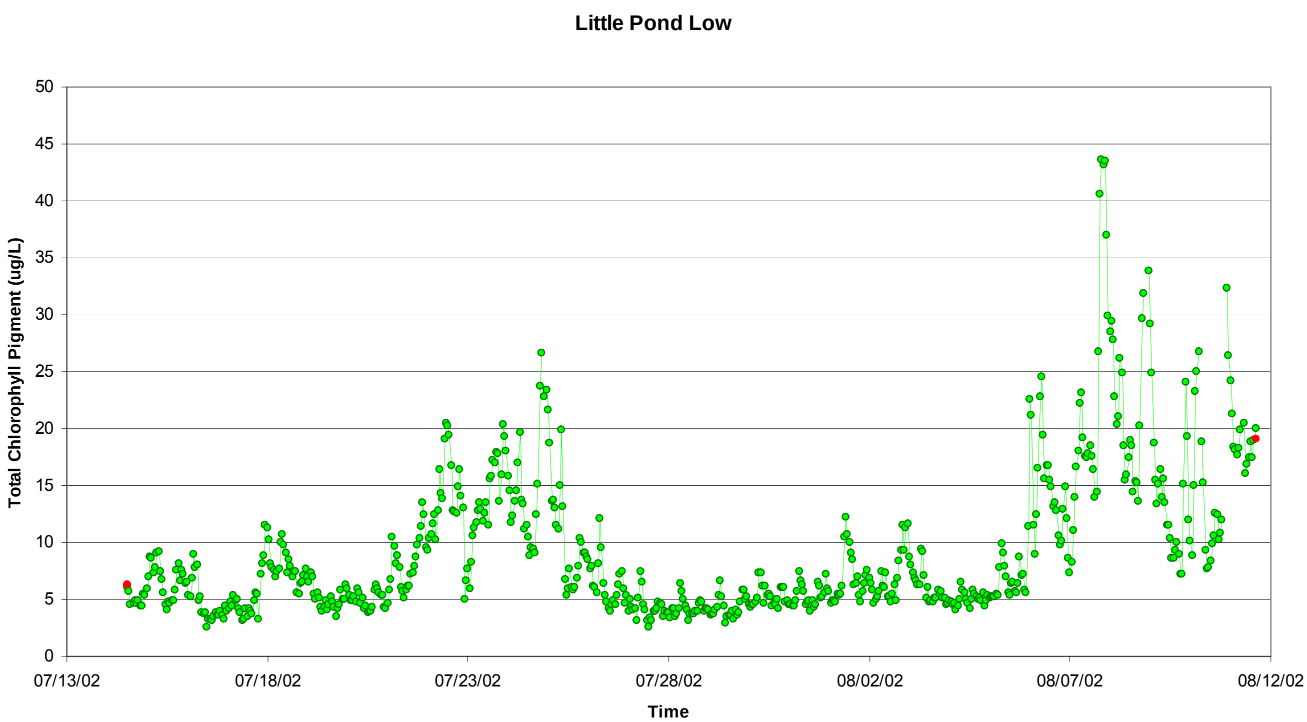


Figure VII-6. Bottom water record of Chlorophyll-*a* in Little Pond Low station, Summer 2002. Calibration samples represented as red dots.

Table VII-1. Percent of time during deployment of in situ sensors that bottom water oxygen levels were below various benchmark oxygen levels. Even short term oxygen declines below 3 mg/L result in a high level of stress to communities.

Massachusetts Estuaries Project Town of Falmouth: 2002	**Dissolved Oxygen: Continuous Record, Summer 2002**				
	Deployment Days	**< 6 mg/L (% of days)**	**< 5 mg/L (% of days)**	**< 4 mg/L (% of days)**	**< 3 mg/L (% of days)**
Little Pond Upper	28.1	27%	19%	11%	5%
Little Pond Lower	28.0	21%	13%	6%	2%

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.

Embayment System	**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)**
Little Pond								
Upper	7/14/2002	8/11/2002	28	**92%**	**47%**	**12%**	**2%**	**0%**
		Mean		1.61	0.29	0.19	0.11	N/A
		S.D.		2.82	0.35	0.17	0.06	N/A
Lower	7/14/2002	8/11/2002	28.1	**70%**	**30%**	**16%**	**7%**	**3%**
		Mean		0.46	0.34	0.20	0.13	0.12
		S.D.		0.73	0.44	0.19	0.14	0.13

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted for the Little Pond System by the DEP Eelgrass Mapping Program as part of the MEP Technical Team. Surveys were conducted in 1995 and 2001, as part of this program. Additional analysis of available aerial photos from 1951 was used to reconstruct the eelgrass distribution when watershed development was at a low level and much of that development was seasonal housing. The 1951 data were only anecdotally validated, while the 1995 and 2001 maps were field validated. Eelgrass observations were also made throughout Little Pond by SMAST Staff, during sampling for mooring calibrations, benthic regeneration and infaunal community analysis in 2002. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figures VII-7 and VII-8); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

At present, eelgrass is present within the Little Pond System. Based on the 2001 eelgrass survey conducted by the DEP Eelgrass Mapping Program, the remaining eelgrass appears to be limited to an area within the lower basin (in the region of LP-3, Figure VI-1, VII-7), just upgradient of the Little Pond inlet. In addition, to the DEP mapping, this has been confirmed by the multiple MEP staff conducting the infaunal and sediment sampling and the mooring studies. The eelgrass is not present as a “bed” or even significant patches, rather the plants are sparsely distributed. In essence, eelgrass is present, but does not represent a significant habitat feature at this time. The current decline of eelgrass beds relative to historical distributions is expected given the high chlorophyll a and low dissolved oxygen levels and high watercolumn nitrogen concentrations within this system. However, it appears that a substantial area of the lower portion of the Little Pond system did support eelgrass beds in 1951. This temporal pattern is also seen in other adjacent embayments, Great Pond and Green Pond. The pattern of the bed loss is consistent with the pattern of nitrogen related habitat quality which is currently observed within the System. It appears that as the Little Pond system became nutrient enriched, chlorophyll a increases caused a decreases in light penetration to the point that these sites could no longer support eelgrass beds. This appears to have been the pattern of loss in Little Pond, as the upper region of the 1995 distribution is in the shallows bordering the deeper water of the basin, rather than reaching into it. Based on the MEP analysis of chlorophyll, D.O., light penetration, nutrient concentrations, and historical patterns of bed loss, the Technical Team is confident that if nitrogen loading were to decrease, eelgrass could first be restored in the lower portion of the main basin. With further reductions it may be possible for beds to be restored to the 1951 pattern assuming other factors are not at play relative to eelgrass bed loss in Little Pond. The likelihood of this restoration pattern is supported by the present existence of some eelgrass in this system.

It is significant that eelgrass was not detected in the upper regions of the Little Pond system in the 1951 data. It appears that these areas, like the upper reaches of Great Pond, Green Pond and Bournes Pond are not generally supportive of eelgrass habitat. The underlying ecological and biogeochemical cause of this upper estuarine reach habitat type is not currently known, but it may relate to sediment type or the fact that these reaches are typically the discharge sites for the major streams entering these systems.

In systems like Little Pond, the general pattern is for highest nitrogen levels to be found within the innermost basins, with concentrations declining moving toward the tidal inlet. This pattern is also observed in nutrient related habitat quality parameters, like phytoplankton, turbidity, oxygen depletion, etc. This is also true of Little Pond, but the gradient is only modest compared to many other systems. However, the consequence is that eelgrass bed decline typically follows a pattern of loss in the innermost basins (and sometimes also from the deeper waters of other basins) first. The temporal pattern is a “retreat” of beds toward the region of the tidal inlet.

Other factors which influence eelgrass bed loss in embayments may also be at play in the Little Pond system, though the loss seems completely in-line with nitrogen enrichment. However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss is not related to mooring density or docks/piers and the systems does not contain these features at present. Similarly, shellfishing in the basins is not common at the present time. It should also be noted that in some estuarine systems a high presence of total suspended solids (TSS) may be a contributing factor to eelgrass bed loss, however, this has not been the case within the embayments of the MEP region that have been studied to date.

It is not possible to determine a general idea of short- and long-term rates of change in eelgrass coverage from the mapping data, since there is only one survey with eelgrass “beds”, the recent maps are only sparsely distributed plants. However, it is possible to utilize the 1951 coverage data to gauge the amount of eelgrass bed that might be recovered with nitrogen management. It appears that when nitrogen loading was at 1951 levels that on the order of 27 acres of eelgrass was present in Little Pond and that this eelgrass was sufficiently dense to allow detection on aerial photographs (Table VII-3).

The relative pattern of the eelgrass distribution and oxygen, chlorophyll a and nitrogen data is consistent with the results of the benthic infauna analysis and the observed eelgrass loss is typical of nutrient enriched shallow embayments (see below).

Figure VII-7. Eelgrass bed distribution within the Little Pond System. The 1995 coverage is depicted by the green outline inside of which circumscribes the eelgrass beds. The yellow (2001) areas were mapped by DEP. All data was provided by the DEP Eelgrass Mapping Program.

Department of Environmental Protection

Eelgrass Mapping for Little Pond, Falmouth

1951 Historic Eelgrass Mapping

(not field-verified)

Composite of 3 Datasets

Legend

1951 Historic Eelgrass Resource

1995 extent of Eelgrass Resource

2001 extent of Eelgrass Resource

0 125 250 500 750 1,000 Meters

Scale Bar

Figure VII-8. Eelgrass bed distribution within the Little Pond System. The 1951 coverage is depicted by the dark green outline (hatched area) inside of which circumscribes the eelgrass beds. In the composite photograph, the light green outline depicts the 1995 eelgrass coverage and the yellow outlined areas circumscribe the eelgrass coverage in 2001. The 1995 and 2001 areas were mapped by DEP. All data was provided by the DEP Eelgrass Mapping Program.

Table VII-3. Changes in eelgrass coverage in the Little Pond Embayment System within the Town of Falmouth over the past half century (C. Costello). Note that the 1995 and 2001 areas do not represent significant eelgrass beds, but rather enclose the zone of sparsely distributed surviving eelgrass.

Embayment	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Little Pond	26.76	15.1	6.15	77%

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 8 locations throughout the Little Pond System (Figure VII-9). In some cases multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds, benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, the Little Pond System 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 Study was fully consistent with the nitrogen data and oxygen and chlorophyll a records and the loss of eelgrass from this system, showing nitrogen related habitat impairment and only a relatively modest gradient between the upper and lower reaches. The infaunal community data indicated that Little Pond is presently moderately to severely degraded throughout its estuarine reach. The Lower basin stations (LP-6, 7, 8) nearest the tidal inlet showed generally higher numbers of individuals, higher diversity and evenness than the upper estuarine reach stations (LP-2,3,4). On average, the lower reach had more than 2 times the individuals (615 vs. 257) distributed among 1.7 times the species resulting in a more productive and diverse community. The evenness scores appear similar, however, it relates more to the small number of species rather than to the habitat quality in this comparison. While the trend is clear, it is important to note that these indicators from the lower basin are still much lower than indicative of higher quality systems where diversities generally range from 2.5 to 3 and above and evenness is generally >0.7. Equally important, the dominant species in both the upper and lower reaches are indicative of poor quality habitat. The infaunal populations throughout the Pond are dominated by small polychaete worms, specifically *Capitella* and *Streblospio*. These species are typical of highly organically loaded sediments. Their life histories allow them to persist in a stressful environment. The lack of larger longer-lived species is equally supportive of the conclusion that Little Pond is a significantly impaired system.

Table VII-4. Benthic infaunal community data for the Little Pond embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.018 m2).

Sub-Embayment	Location	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Little Pond System						
	Sta. 1	8	697	5.13	1.21	0.40
	Sta. 2	6	227	3.99	1.08	0.42
	Sta. 3	4	216	3.82	1.59	0.79
	Sta. 4	3	328	2.99	0.81	0.51
	Sta. 5	3	696	2.80	0.37	0.24
	Sta. 6	7	668	5.14	1.30	0.46
	Sta. 7	8	560	5.20	1.30	0.43
	Sta. 8	7	616	6.32	1.57	0.56

Figure VII-9. Aerial photograph of the Little Pond system showing location of benthic infaunal sampling stations (red symbol).

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

VIII.1 ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were collected to support threshold development for the Little Pond System by MEP Team 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 long-term baseline Falmouth water quality monitoring program collected by PondWatch. At present, Little Pond, both upper and lower reaches, is showing significantly impaired habitat quality. The lower reach is only moderately better habitat than the upper reach, in spite of its proximity to the tidal inlet and high quality waters of Vineyard Sound. All of the habitat indicators are consistent in the evaluation of the whole of Little Pond being classified as a significantly impaired estuarine habitat (Chapter VII, Table VIII-1).

Eelgrass: Currently, there is only sparsely distributed eelgrass in the central region of the lower basin in Little Pond. The larger distribution in 1995 diminished to 2001 and the extensive eelgrass beds of 1951 are no longer present. However, the presence of sparse eelgrass in Little Pond in 2001 indicates that eelgrass recovery within the lower basin of estuary is a good target for nitrogen management. The system's pattern of eelgrass bed loss and its current lack of eelgrass beds, indicates that the lower basin is moderately to significantly impaired and the upper tributary has not generally been supportive of eelgrass habitat. The pattern of eelgrass loss from the lower basin supports nitrogen loading as the primary mechanism. The loss occurred first in the deeper regions, consistent with decreased light penetration as increasing nitrogen inputs stimulated phytoplankton production. Eelgrass recovery following nitrogen management would likely follow the reverse of this pattern, with beds first being re-established in the marginal areas and moving finally to the deeper regions.

Water Quality: The upper and lower reaches of Little Pond are currently under seasonal oxygen stress, consistent with nitrogen enrichment (Table VII-1). That the cause is eutrophication is supported by the high levels of chlorophyll a, 15 ug/L to >20 ug/L (Table VII-2). Oxygen conditions and chlorophyll a levels indicated nutrient related stress throughout Little Pond with a moderate improvement with decreasing distance to the tidal inlet. Oxygen depletions below 5 mg/L were common in both upper and lower reaches, 19% and 13% of deployment time respectively, and depletions below 3 mg/L were observed (5% and 2% of deployment, respectively). The observed chlorophyll a levels and oxygen depletions within Little Pond waters during summer is consistent with the high level of total nitrogen found throughout the Pond's waters. Falmouth PondWatch data indicates average summer total nitrogen levels ranging from 0.745 in the lower basin nearest the tidal inlet to 0.942 mg N/L in the upper portion of the upper estuarine reach. These are relatively high total nitrogen values and when compared to the adjacent offshore waters (0.28 mgN/L) the extent of nitrogen loading to the estuarine waters of Little Pond is clear. The levels and distribution of chlorophyll a, dissolved oxygen and total nitrogen within Little Pond is consistent with the observed eelgrass losses (above) and the significantly impaired infaunal animal communities (below).

Infaunal Communities: The Infauna Study indicated that all areas are presently significantly impaired to severely degraded (Table VII-4). The upper reach of Little Pond is currently impoverished in infaunal species (~4 per site), has relatively low numbers (~250 per sample) and is dominated by stress indicator species. The lower reach of Little Pond is only moderately better having more individuals (~600 per sample) and more species (~7 per sample). However, the lower reach, like the upper reach, is dominated by organisms indicative of high organic matter loading, stressful conditions and impaired habitat. The gradient from upper to lower estuary is only sufficient to classify the upper region as significantly impaired/severely degraded to significantly impaired in the lower basin. The infaunal habitat classification within Little Pond is supported by all of the infaunal criteria (numbers of individuals, numbers of species, diversity, evenness and habitat indicator species). The limited infaunal species in Little Pond were dominated by small polychaete worms (*Capitella*, *Steblospio*, etc.). These species are adapted to conditions of high organic matter loading and sulfidic sediments and generally are good indicators of nitrogen enrichment (eutrophic conditions). As is typical when conditions support these opportunistic stress tolerant species, they can occur in relatively high densities. Both the upper and lower reaches of Little Pond supported an infaunal community dominated by opportunistic stress tolerant species, but with lower numbers of organisms distributed among fewer species in the upper reach, indicating a higher level of stress in this region. The infaunal community based classification in the upper and lower reaches of Little Pond is fully supported by the water quality and eelgrass data discussed in the text above.

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

Health Indicator	Estuary: Little Pond	
	Upper	**Lower**
Dissolved Oxygen	SI/SD[1]	SI/SD[1]
Chlorophyll	SI	SI
Macroalgae [2]	SI	SI
Eelgrass	SI[3]	MI/SI[4]
Infaunal Animals	SI/SD[5]	SI[5]
Overall:	**SI/SD**	**SI**

1 – periodic oxygen depletions to <3 mg/L and frequently <4 mg/L.
2 – macroalgal floating accumulations during summer in lower basin periodically occur.
3 – no evidence that upper region of this estuarine reach is supportive of eelgrass.
4 – estuarine reach is supportive of eelgrass, but shift from beds in 1951 to only sparsely distributed plants by 2001/2002.
5 – low diversity, modest numbers of individuals dominated by stress indicator species.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment; SD = Severe Degradation
-- = not applicable to this estuarine reach

VIII.2 THRESHOLD NITROGEN CONCENTRATIONS

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

The restoration goal for Little Pond based upon the present and historical conditions is to restore eelgrass habitat to the lower basin and infaunal community habitat to the upper estuarine reach. The nitrogen loading threshold for Little Pond was developed from the habitat indicator information in Section VII and thresholds development from adjacent Great Pond, Green Pond and Bournes Ponds which are structurally similar to Little Pond. In particular, Bournes Pond is very similar in its basin depths and shares the common tidal waters of Vineyard Sound. Little Pond has both historic evidence of eelgrass beds (1951) and recent surveys indicating sparsely distributed eelgrass within the lower basin. The historic presence of eelgrass beds and the present survival of eelgrass plants supports the restoration of eelgrass as the management target for Little Pond. However, eelgrass appears to have been confined mainly to the lower basin with fringing beds in the shallows up to PondWatch sampling station LP-2 (Figures VI-1, VIII-1). The distribution of eelgrass in the 1951 and 1995 data indicate that eelgrass beds were found primarily within the ~1 meter depths and not in the small basin at ~2 meters depth. This is similar to the findings for Bournes Pond where a threshold at the sentinel station to achieve the fringing beds at similar depths to Little Pond was set at 0.45 mg N L^{-1}. The restoration of the full lower basin of Little Pond should also be achieved when the sentinel station reaches 0.45 mg N L^{-1} as the lower basin will have lower nitrogen levels than the sentinel station. At present, the lower basin has moderately higher quality habitat than in the upper reach in which the sentinel station is located and this moderate gradient of improvement moving toward the tidal inlet will persist as future nitrogen management is implemented. Since the sentinel station is about 1/2 of the distance from the headwaters of the Pond, the region above this station is targeted for restoration of infaunal animal habitat (i.e. the reach above the upper limit of the 1951 eelgrass distribution). In studies of Great Pond, Green Pond and Bournes Ponds and Popponesset Bay, infaunal animal habitat quality was found to be moderate to high at 0.5 – 0.6 mg N L^{-1} and excellent at $\leq$0.5 mg N L^{-1}. Achieving the threshold level at the sentinel station will produce a level of total nitrogen in the upper estuarine reach (Station LP-1 to LP-2) of <0.5 mg N L^{-1} supportive of high quality benthic animal habitat.

Discussion: Within the Little Pond Estuary the most appropriate sentinel station (LP-2) was about ½ of the distance from the headwaters to the tidal inlet (Figure VIII-1). This location was selected because (1) it was in the upper region where eelgrass bed coverage was documented in the 1951 analysis, (2) restoration of nitrogen conditions supportive of eelgrass at this location will necessarily result in even higher quality conditions throughout the entire lower basin, and (3) restoration of nitrogen concentrations at this site should result in conditions similar to 1951 within the upper tributary, which will be supportive of high quality habitat for benthic infaunal communities (confirmed as described below). For embayment restoration, an additional requirement within the upper 1/2 of the estuary was to ensure that TN in this region has been reduced to levels supportive of healthy infaunal community habitat when the eelgrass threshold was met for the lower 1/2 of the embayment.

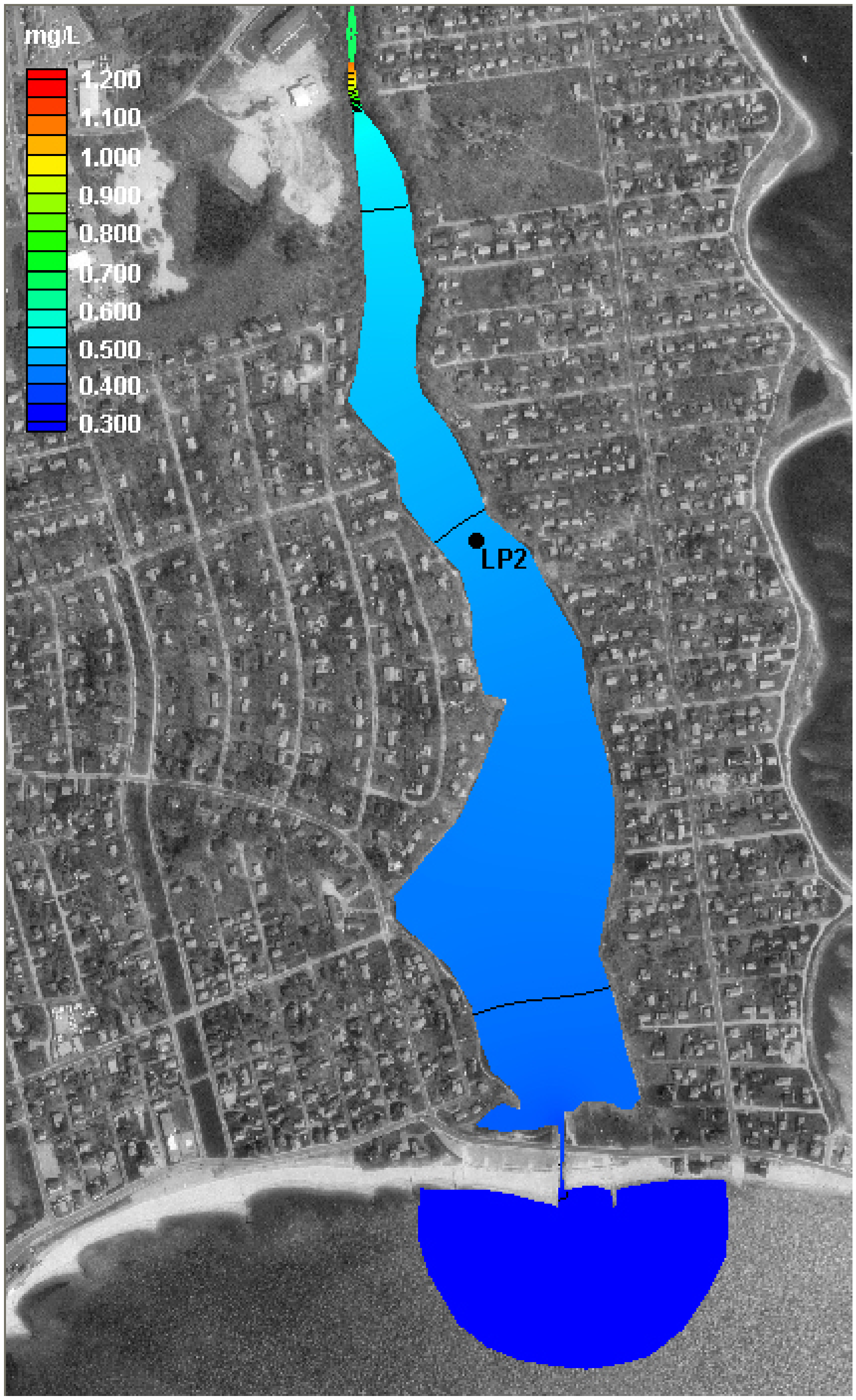


Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Little Pond system, for threshold conditions (0.45 mg/L at water quality monitoring station LP2). The approximate location of the sentinel threshold station for Little Pond (LP2) is shown.

The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.45 mg TN L^{-1} for Station LP2 and TN <0.45 mg TN L^{-1} in the lower basin. This threshold level is consistent with the findings that (1) eelgrass beds have been lost in the lower basin which currently supports a tidally averaged TN of 0.837 mg TN L^{-1} at LP2 and 0.705 mg TN L^{-1} in the lower basin (LP3), (2) sparse eelgrass can be still be found within the lower basin at tidally averaged TN of 0.61 mg TN L^{-1}, and (3) the eelgrass beds in Bournes Pond (threshold 0.45 mg TN L^{-1}) at water depths similar to those in the lower basin of Little Pond, which is important for light penetration. Based upon these data, the threshold TN level at the sentinel station was set at 0.45 mg TN L^{-1} to achieve eelgrass habitat recovery throughout the lower basin and to re-establish the marginal beds seen in 1995 near LP-2. These marginal beds along the edges of the basin where the estuary narrows existed outside of the relatively deeper basins within this portion of Little Pond. Due to the shallow depths of these margins and the small tide range within the system, eelgrass restoration likely will occur at slightly higher TN values than observed regionally (e.g. Stage Harbor in Chatham). The small tide range increases the duration of light penetration to the bottom compared to similar estuaries with larger tide ranges. Therefore, restoration of eelgrass beds along the margins immediately north of where the estuary narrows down should occur when TN levels are lowered to 0.45 mg TN L^{-1}.

Given the lack of healthy infaunal habitat in Little Pond, the infaunal requirements are based upon Great Pond and Bournes Pond, where TN levels below 0.45 mg TN L^{-1} support moderate to healthy habitat, with healthy infauna habitat requiring TN <0.5 mg TN L^{-1}. This result also is the same found to support moderate to healthy habitat by MEP for Popponesset Bay. The significantly impaired upper and mid reaches of Little Pond currently have very high TN levels, >0.8 mg TN L^{-1}. Based upon sequential reductions in watershed nitrogen loading in the analysis described in the section below (VIII-3), the sentinel station achieved an average TN level of 0.45 mg L^{-1}. This indicates that significant eelgrass habitat restoration would occur within the regions of the 1951 coverage which will achieve even lower TN levels, TN levels generally thought to be highly supportive of eelgrass beds. Although infaunal habitat is not the primary habitat restoration driving thresholds development for Little Pond, restoration of infaunal habitat throughout Little Pond should be achieved upon reaching the threshold TN level at the sentinel station. The result will be high quality infaunal habitat system wide and restoration of eelgrass habitat in the lower basin. Upon reaching the TN threshold at the sentinel station, the upper tributary (LPHEAD), which is currently showing signs of significant impairment bordering on severely degraded infaunal habitat (>2.0 mg TN L^{-1}) and located in the immediate mixing zone just below the entrance of Little Pond Stream (at LPHEAD), will have TN levels <1.1 mg L^{-1}. In the uppermost portion of the upper reach of Little Pond, LP-1 will be below the nitrogen level associated with high quality infaunal habitat, <0.5 mg TN L^{-1}. This yields a range of TN in the upper reach of 0.45 mg N/L at LP-2 and <0.5 mg N/L at LP-1. This indicates that upon reaching the threshold nitrogen level, restoration of infaunal habitat throughout the upper reach will be achieved. It should be emphasized that these infauna values were not used for setting nitrogen thresholds in this embayment system. These values merely provide a check on the acceptability of conditions in the tributary systems when the threshold level is attained at the sentinel station. The results of the Linked Watershed-Embayment modeling, when the nitrogen threshold is attained (Section VIII-3), yield TN levels in these regions within the acceptable range. Therefore, it appears that achieving the nitrogen target at the sentinel location is restorative of eelgrass habitat throughout the lower Little Pond basin and marginal beds along the narrowing edges of the estuary and restorative of infaunal habitat throughout the estuary.

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 Little Pond. 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 Little Pond. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the community. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

As shown in Table VIII-2, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required 100% removal of septic load (associated with direct groundwater discharge to the embayment) for the systems' lower watersheds. In addition, a portion of the septic load entering the pond from the fresh water stream also must be removed to meet the threshold nitrogen concentrations. For the load reduction scenario evaluated, the Little Pond Stream sub-watershed required removal of approximately 60% of the septic load. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figure VIII-1.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Little Pond 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
Little Pond	10.419	0.000	-100.0%
Surface Water Sources			
Little Pond Stream	5.496	2.198	-60.0%

Tables VIII-3 and VIII-4 provide additional loading information associated with the thresholds analysis. Table VIII-3 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-2. Removal of 100% of the septic load from the lower watershed of Little Pond results in an 80% reduction in total nitrogen load. For the Little Pond Stream, septic load reduction of 60% resulted in total attenuated watershed load reduction of 55%. 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 Vineyard Sound.

Table VIII-3. Comparison of sub-embayment ***total attenuated watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Little Pond 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
Little Pond	13.022	2.603	-80.0%
Surface Water Sources			
Little Pond Stream	6.052	2.755	-54.5%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Little Pond 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)
Little Pond	2.603	0.584	0.670
Surface Water Sources			
Little Pond Stream	2.755	-	-

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel station is shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel station, a reduction in TN concentration of approximately 50% is required for Little Pond.

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

Table VIII-5. Comparison of model average total N concentrations from present loading and the modeled threshold scenario, with percent change, for the Little Pond system. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Little Pond Head (fresh water)	LPHEAD	2.236	1.036	-53.7%
Little Pond - upper	LP1	0.955	0.492	-48.5%
Little Pond - mid	**LP2**	**0.837**	**0.449**	**-46.3%**
Little Pond - lower	LP3	0.705	0.408	-42.1%

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.

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

As part of the MEP analysis, nitrogen management alternatives are evaluated towards the goal of reaching the total nitrogen level at the sentinel station to achieve restoration of embayment habitats supportable by present and historic data. In the case of Little Pond nitrogen management alternatives were conducted related to addressing the restriction to tidal flushing imposed by the present inlet and channel configuration. These alternatives are but one of many potential management pathways, but also serve to indicate the application of the Linked Watershed-Embayment model now that it is fully parameterized, calibrated and validated.

IX.1 FLUSHING IMPROVEMENTS TO LITTLE POND BY MODIFICATIONS TO THE INLET

Water quality improvements may be possible by improving tidal exchange in an estuary. Little Pond could benefit possibly from flushing improvements. Tidal attenuation is high, where the average tide range is less than 74% of the offshore range. Attenuation in this system is primarily caused by an undersized inlet. In contrast, for Great Pond and Green Pond tide attenuation is between 6% and near zero, respectively, compared to the range offshore in Vineyard Sound. Although culvert and jetty improvements were completed in 1995, the overall size of the improved culvert was limited by the existing inlet channel width (i.e. the jetty spacing). Reconfiguration of the inlet channel, as well as culvert modifications, would be required to have a significant impact on tidal flushing. A more detailed analysis of inlet stability, maintenance requirements, and potential environmental impacts would be required to fully assess inlet widening options.

Widening Little Pond inlet would improve tidal flushing. To quantitatively assess inlet improvements, a model simulation was executed to simulate Little Pond hydrodynamics with an improved 20 ft-wide inlet, which is twice the width of the existing jettied inlet.

Hydrodynamic model results for existing and improved inlet conditions are presented in Figure IX-1. In the top plot, tide attenuation is apparent by the higher elevation of the low tides, and also by the time delay of the tide signal inside the pond. In the bottom plot of this figure, tidal attenuation is reduced for the proposed 20 ft-wide inlet, to the point where there is little difference between the range and phase of both tide signals.

Based on model output, the average tidal prism increases by 42% with the improved inlet. Average volumes of Little Pond for existing conditions and for the 20ft-wide inlet scenario are presented in Table IX-1. As a result of the increased tidal prism volume and the reduced mean tide volume of the system, the computed system residence time decreases to 3.2 days from 4.6 days for existing conditions.

Water quality model runs were performed using the hydrodynamic model output of the proposed 20 ft-wide Little Pond inlet. First, present loading conditions were modeled with the widened inlet. Second, loading conditions necessary to meet the nitrogen loading thresholds described in Chapter VIII were developed for the widened inlet condition. Results from the existing loading conditions with the improved hydrodynamics of the widened inlet are presented in Table IX-2, and plotted in Figure IX-2. Although TN concentrations are significantly reduced (i.e., up to an 18% reduction in the lower portion of the Pond), the reduction is not large enough to meet the threshold limits set for Little Pond (TN of 0.45 mg/L at water quality monitoring station LP2).

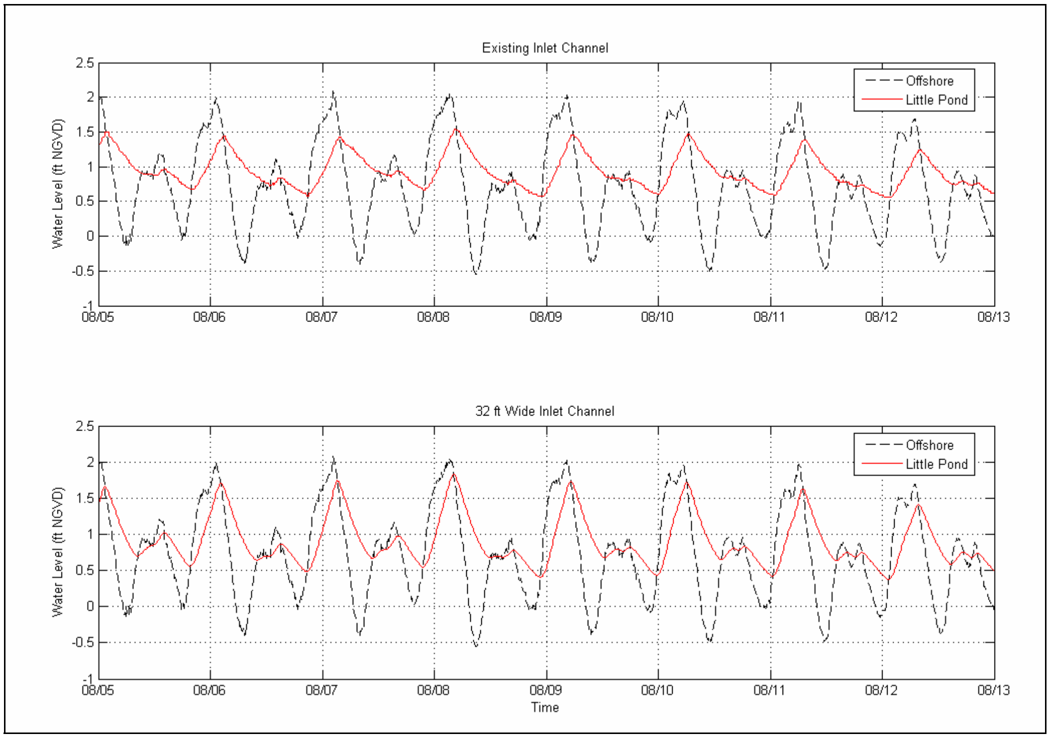


Figure IX-1. Plots showing a comparison of typical tides for modeled existing conditions (top plot) and proposed improved 20 ft-wide inlet (bottom plot) to Little Pond.

Table IX-1. Average high, mid and low tide volumes, with mean tide prism for Little Pond, for existing inlet conditions, and for the proposed 20 ft-wide inlet modification.

	existing inlet ft^3	20 ft-wide inlet ft^3	% change
Mean High Tide Volume	7,479,620	7,611,570	1.8%
Mean Tide Volume	6,986,660	6,930,340	-0.8%
Mean Low Tide Volume	6,573,610	6,253,470	-4.9%
Mean Prism Volume	785,605	1,082,070	37.8%

Table IX-2. Comparison of model average total N concentrations from present loading and the widened inlet channel (20 ft) scenario with present loading, with percent change.

Sub-Embayment	monitoring station	present (mg/L)	Channel mod, present (mg/L)	% change
Little Pond Head (fresh water)	LPHead	2.236	2.202	-1.5%
Little Pond - upper	LP1	0.955	0.838	-12.3%
Little Pond - mid	**LP2**	**0.837**	**0.714**	**-14.7%**
Little Pond - lower	LP3	0.705	0.577	-18.1%

Figure IX-2. Contour Plot of modeled total nitrogen concentrations (mg/L) in Little Pond, for present loading conditions, and widened inlet channel (20 ft). The approximate location of the sentinel threshold station for Little Pond (LP2) is shown.

However, threshold concentrations can be achieved for the widened inlet scenario as shown in Table IX-3, IX-4, and Figure IX-3. Similar to the threshold evaluation in Chapter VIII, the nitrogen load reductions within the system necessary to achieve the threshold nitrogen concentrations required 100% removal of septic load (associated with direct groundwater discharge to the embayment) required for the system's lower watershed. In addition, a portion of the septic load entering the pond from the fresh water stream also must be removed to meet the threshold nitrogen concentrations. For the load reduction scenario evaluated, the Little Pond Stream sub-watershed required removal of approximately 30% of the septic load.

Therefore, a combination of removing septic loading to the lower watershed, removing 30% of the freshwater stream load, and increasing the width of the inlet would improve the system to a level that meets the selected restoration threshold. Widening the inlet would certainly make the threshold limit more practically attainable, where significantly less nitrogen load (30 percent) would need to be removed within the watershed. Potential environmental and regulatory implications exist for reconfiguration of the inlet; therefore, a complete analysis of the costs, benefits, and impacts of this strategy would be required prior to further consideration of this option. From an engineering cost perspective alone, it likely is cheaper to modify the inlet than to sewer a large portion of the upper watershed.

Table IX-3. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading for the widened inlet threshold scenario of Little Pond. 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
Little Pond	10.419	0.000	-100.0%
Surface Water Sources			
Little Pond Stream	5.496	3.847	-30.0%

Table IX-4. Comparison of model average total N concentrations from present loading and the modeled widened inlet threshold scenario (lower watershed) with widened inlet channel (20 ft) scenario, with percent change.

Sub-Embayment	monitoring station	present (mg/L)	Channel mod, 20-ft wide (mg/L)	% change
Little Pond Head (fresh water)	LPHead	2.236	1.570	-29.8%
Little Pond - upper	LP1	0.955	0.527	-44.9%
Little Pond - mid	**LP2**	**0.837**	**0.454**	**-45.7%**
Little Pond - lower	LP3	0.705	0.393	-44.2%

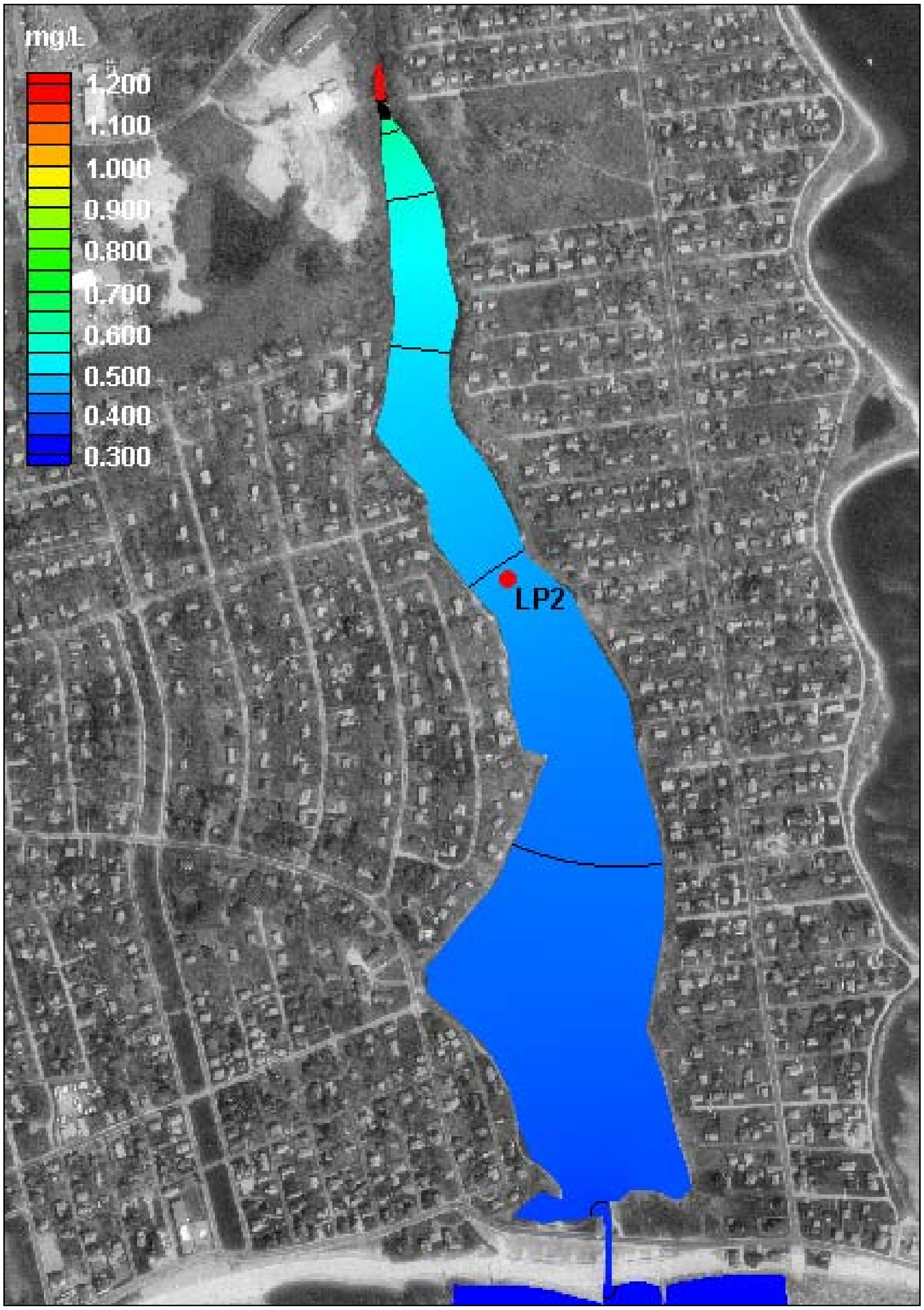


Figure IX-3. Contour Plot of modeled total nitrogen concentrations (mg/L) in Little Pond, for widened inlet threshold scenario (lower watershed) loading conditions, and widened inlet channel (20 ft). The approximate location of the sentinel threshold station for Little Pond (LP2) is shown.

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