Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Great/Perch Pond, Green Pond and Bournes Pond, Falmouth, Massachusetts

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I. INTRODUCTION

The Great, Green and Bournes Pond embayment systems are located within the Town of Falmouth, on Cape Cod Massachusetts. Each of the separate systems has a southern shore bounded by water from Vineyard Sound (Figure I-1a and 1b). The watersheds for each salt pond system are distributed among the Towns of Falmouth and Sandwich (north). The present configuration of each embayment results from inundation of shallow valleys by rising sea level beginning approximately 3000 yr B.P. These systems were initially marine, but due to barrier beach development, evolved into brackish coastal lagoons after sea level rose to near present levels. Currently, the lower portions of each valley supports a groundwater fed stream/river that discharges to the headwaters of each of the salt ponds/estuaries.

All three embayments (Great Green and Bournes Pond) are separated from Vineyard Sound by a barrier beach, which was naturally breached and allows tidal exchange producing the estuarine characteristics of each of these Great Salt Ponds. The beach and the openings to each embayment are very dynamic geomorphic features due to the influence of littoral transport processes. Over time the inlets have experienced varying degrees of occlusion thereby affecting tidal exchange and circulation within each of the salt ponds. Bournes Pond became very restricted and finally completely isolated from Vineyard Sound waters in the late 1970's/early 1980's. Bournes Pond was re-opened with a fixed inlet in the mid 1980's. Over the past century the inlet to each estuary was stabilized with jetties and/or culverts to maintain tidal exchange in support of estuarine resources and/or navigation. Today each embayment exchanges tidal water with Vineyard Sound through discrete and maintained inlets that are periodically dredged.

The Great, Green and Bournes Pond embayment systems are shallow coastal salt ponds located within a glacial outwash plain, the Mashpee Pitted Plain. The Mashpee Pitted Plain consists of material that can be classified as primarily sands and gravel deposited after the retreat of the Cape Cod Lobe of the Laurentide Ice sheet ~15,000 years ago. The outwash material is highly permeable and varies in composition from well sorted medium sands to course pebble sands and gravels (Millham and Howes, 1994). 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. Coonamessett River, Backus River, Bournes Brook). All three systems act as mixing zones for terrestrial freshwater inflow and saline tidal flow from Vineyard sound, however, the salinity characteristics of each salt pond varies with the volume of freshwater inflow as well as the effectiveness of tidal exchange with Vineyard Sound.

These embayments constitute important components of the Town's natural and cultural resources. In addition, the large length to width ratio (~15:1) 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, 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-1a. Green Pond and Bournes Pond embayment systems along the southern shore of the Town of Falmouth, MA. Tidal waters enter the salt ponds through fixed tidal inlets to Vineyard Sound. Freshwaters enter from the watershed primarily through 2 surface water discharges (creek from Mill Pond to Green Pond) via the Backus River and (a creek from an upgradient cranberry bog to Bournes Pond) via Bournes Brook and direct groundwater discharge.

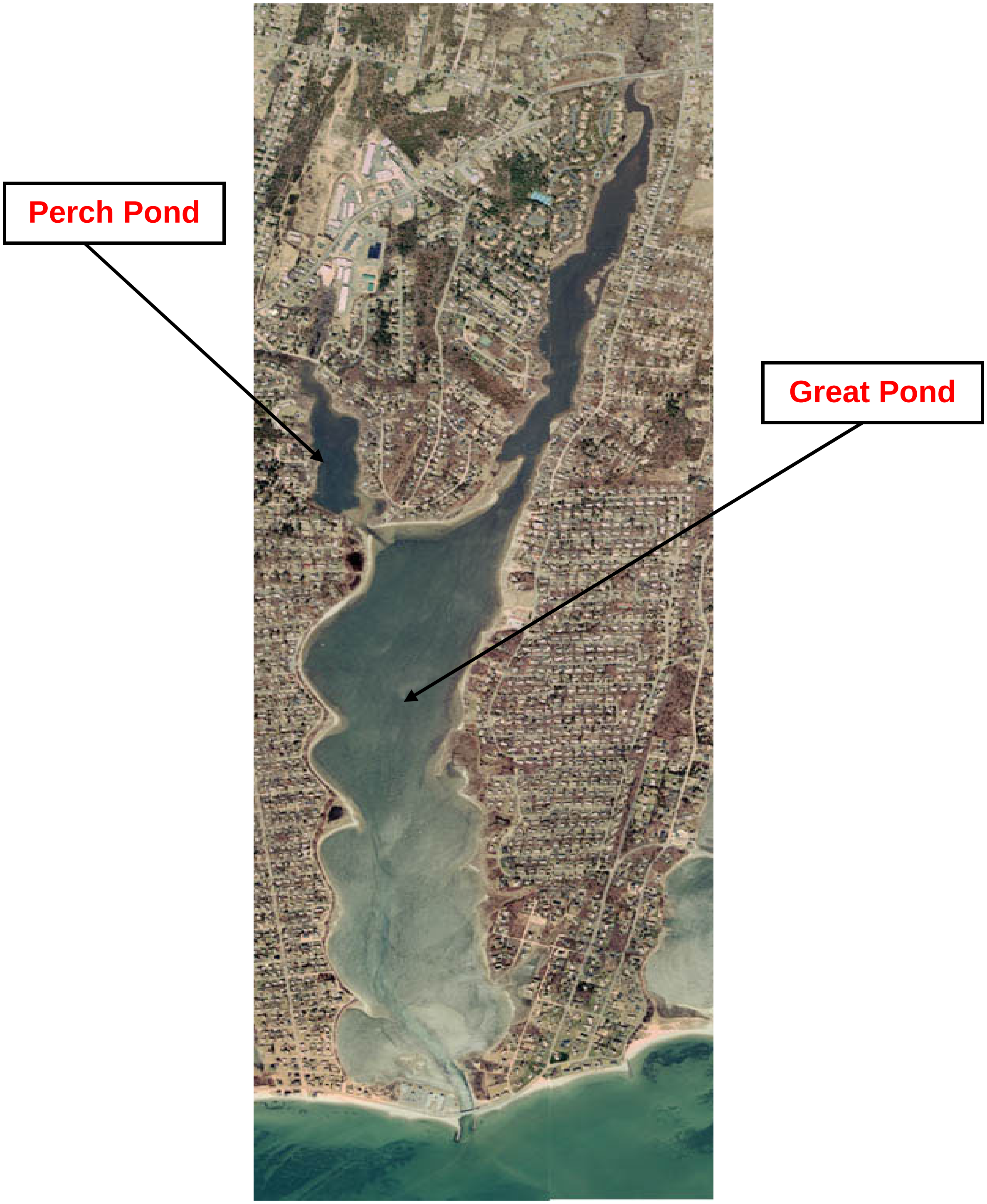


Figure I-1b. The Great/Perch Pond Embayment System along the southern shore of the Town of Falmouth, MA. Perch Pond is a drowned kettle pond communicating through a narrow tidal channel with Great Pond. Tidal waters enter Great Pond from Vineyard Sound through a fixed inlet. Freshwaters enter from the watershed primarily through 1 surface water discharge, the Coonamesset River, and through direct groundwater discharge.

The primary ecological threat to Great, Green and Bournes 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 these salt ponds, similar to almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. Over the past two decades, the Town of Falmouth has been among the fastest growing towns in the Commonwealth and does not have centralized wastewater treatment throughout the entire Town. As build-out continues, associated increases in nitrogen loading further degrades the habitats and resources within Falmouth's coastal embayments.

As the primary stakeholder to the Great, Green and Bournes 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. Three levels were established (1) High Quality (2) Stabilization and (3) Intensive Water Activity Areas (e.g. harbors, marinas) with upper limits of 0.32, 0.50 and 0.75 mg N L^{-1}, respectively. These planning thresholds were based upon the lake and estuarine literature at that time. With the establishment of upper limits as described above, a baseline water quality monitoring effort was necessary for ecological management of Falmouth's coastal salt ponds and harbors. A citizen-based water quality monitoring program developed by the Town of Falmouth, the Falmouth PondWatch Program, sought 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 WHOI 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, beginning in 1987. 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 provides quantitative information to the Town relative to the efficacy of remediation efforts. Lastly, the Pond Watch Program has grown into being a repository of environmental data on Falmouth's coastal ponds and has supported various process level research studies relating to nitrogen loading, freshwater inflow and ecological responses to nitrogen loading of shallow coastal embayments. These research studies have been published in multiple journal articles, technical reports, graduate theses, a book and management documents.

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 evaluation is part of that continuing effort. Unfortunately, monitoring has documented that most regions within the Town's coastal ponds are currently showing significant nitrogen related impairments of habitat quality and nitrogen levels 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 modeling analyses undertaken as a precursor to the Massachusetts Estuaries Project, the Great, Green, and Bournes Pond embayment systems were included in the first round prioritization of the Massachusetts Estuaries Project (MEP). The MEP was specifically developed to provide state-of-the-art analysis and modeling of coastal embayments in order to accurately and precisely determine embayment specific critical nitrogen thresholds. Given that the MEP was able to fully integrate the Towns' on-going data collection and modeling effort (primarily through PondWatch, various Engineering Department efforts and the Ashumet Plume Nitrogen Offset Program), no additional municipal funds were required for MEP tasks.

Although the common focus of the Falmouth Pond Watch Program effort has been to gather site-specific data on the current nitrogen related water quality throughout the Great, Green, and Bournes Pond embayment systems and determine its relationship to watershed nitrogen loads, various process-level studies had not yet been conducted (e.g. sediment nitrogen regeneration, stream nitrogen loading, etc.). Nonetheless, the Pond Watch effort has provided high quality water column nitrogen baseline information required for the MEP assessment and modeling approach. The MEP effort builds upon the Falmouth water quality monitoring program, and previous hydrodynamic and water quality analyses, to include high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each embayment system.

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. The MEP Technical Team has been working with the Town of Falmouth's Nitrogen Management Committee to coordinate the current MEP effort with the Town's on-going planning and implementation schedules. A key part of this interaction has been to develop nitrogen reduction scenarios for processing through the Linked Watershed-Embayment Management Model for Great, Green and Bournes Ponds (Section IX). 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 a 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 resources 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. The nutrients are primarily related to changes in watershed land-use associated with increasing population within the coastal zone over the past half century. Many of Massachusetts' embayments have nutrient levels that are approaching or are currently over this assimilative capacity, which begins to cause declines in their ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities. 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 uses. However like nutrients, bacterial contamination is related to changes in land-use as watershed becomes more developed. The regional effects of both nutrient loading and bacterial contamination span the spectrum from environmental to socio-economic impacts and have direct consequences to the culture, economy, and tax base of Massachusetts's coastal communities.

The primary nutrient causing the increasing impairment of the Commonwealth's coastal embayments is nitrogen and the primary sources of this nitrogen are wastewater disposal, fertilizers, and changes in the freshwater hydrology associated with development. At present there is a critical need for state-of-the-art approaches for evaluating and restoring nitrogen sensitive and impaired embayments. Within Southeastern Massachusetts alone, almost all of the municipalities (as is the case with the Town of 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 next 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 partners including the Cape Cod Commission (CCC), Martha's Vineyard Commission and Southeastern Regional Planning and Economic Development District (SRPEDD), 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 components and nitrogen parameters are drawn primarily from peer-reviewed research and highly evolved engineering studies from southeastern Massachusetts estuaries and watersheds. The MEP approach has been fully reviewed by EPA and DEP and others prior to the first municipal application of the MEP Approach. Furthermore, there is a full review process for each embayment specific Threshold Report, prior to review by the general public, municipal staff and their engineering consultants. The Linked Model has been applied for watershed nitrogen management in 15 embayments throughout Southeastern Massachusetts, some of which having undergone Linked Model nitrogen analysis prior to the initiation of the MEP. In these applications it has become clear that the Linked Model Approach's greatest assets are its ability to be clearly calibrated and validated, and its utility as a management tool for testing "what if" scenarios for evaluating watershed nitrogen management options.

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

Linked Watershed-Embayment Model Overview: The Model provides a quantitative approach for determining an embayment's: (1) nitrogen sensitivity, (2) nitrogen threshold loading levels (TMDL) and (3) response to changes in loading rate. The approach is fully field validated and unlike many approaches, accounts for nutrient sources, attenuation, and recycling and variations in tidal hydrodynamics (Figure I-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 are oriented north-south, and open to Vineyard Sound via fixed inlets. These inlets are affected significantly by longshore sand transport (west to east), where shoaling can impede hydrodynamic exchange at each mouth. All inlets are armored with jetties, with each featuring significant scour channels between these structures. The ponds are long and narrow, with length-to-breadth ratios of approximately 15-to-1. Green Pond, for example, is nearly two miles in length. Depths within the deeper scour channels at each inlet are approximately 8 feet, with the upper (northern) reaches of the Pond frequently less than 5 feet deep. Deeper areas in Perch Pond are a result of a geologic feature known as a "kettle pond", which was drowned by rising sea-level.

The Great, Green and Bournes Pond embayment systems exchange tidal water with Vineyard Sound through individual maintained inlets crossing the barrier beach that separates these coastal salt ponds from the ocean. The inlet to Great Pond has been stabilized with riprap in order to accommodate a bridge passing over the inlet to Great Pond as is also the case with Bournes Pond. The inlet connecting lower Green Pond to Vineyard Sound has been stabilized but does not have a bridge crossing, however a stabilized channel is located further up gradient where Menauhant Road crosses over Green Pond. Menauhant Road bridge crossing over Green Pond was reconstructed in 1995-1996 and a detailed hydrodynamic

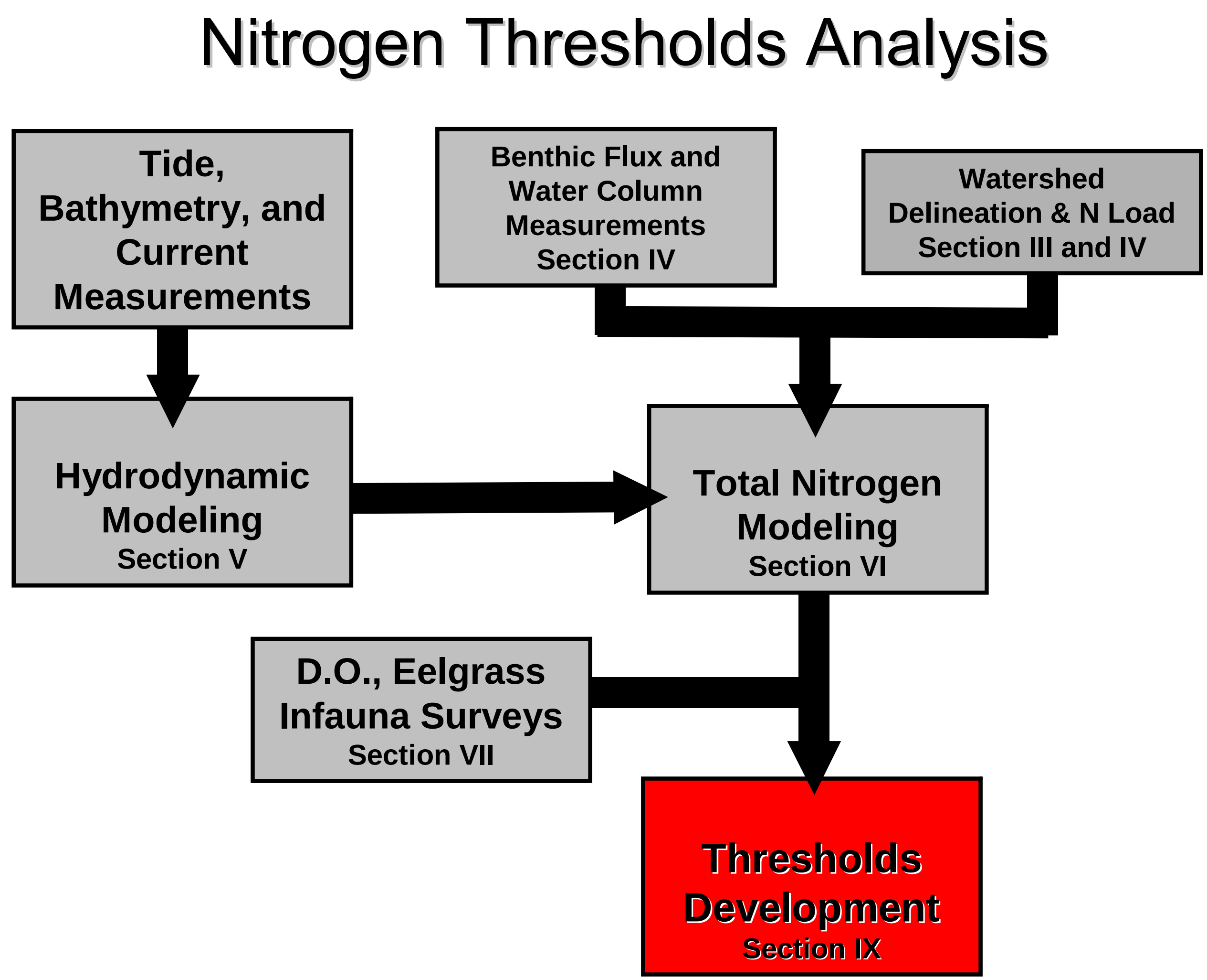


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach analysis was undertaken to investigate the effects of bridge reconstruction on embayment circulation. Subsequent to the bridge reconstruction, additional culverts were constructed under Menauhant Road between lower and upper Green Pond.

For the MEP analysis, the three systems (Great, Green, and Bournes Pond) were analyzed individually as stand alone systems. In the case of Great Pond, an additional sub-division of the overall Great Pond embayment had to be made in order to focus on the specific sub-embayment characteristics of Perch Pond that is hydraulically connected to the main body of Great Pond via a small channel. In the case of Bournes Pond, the overall embayment was further sub-divided into a sub-embayment capturing the water quality and hydrodynamic characteristics of Isreal's Cove. Each estuarine system has been partitioned into two general embayment groups: an 1) upper portion also considered the head of the estuary and 2) a lower portion that includes the mouth of the estuary (Figure I-3).

The three salt ponds (Great/Perch Pond, Green Pond, and Bournes Pond) in Falmouth are estuaries with focused freshwater inputs at the headwaters and tidal exchange of marine waters from Vineyard Sound (tide range of approximately 0.5 m) at their southern inlets. Perch Pond is a tributary to Great Pond and is predominantly influenced by the water quality of the much larger Great Pond through tidal exchange. The three main ponds are similar in length, but show a range of widths that result in their differing surface areas. Great/Perch Pond is the largest at 109 hectares (1 ha = 2.47 acres) with Bournes (62 ha) and Green (53 ha) being about half as large.

Great, Green, and Bournes Ponds are shallow mesotrophic (moderately nutrient impacted) to eutrophic (nutrient-rich) coastal ponds on the southern coast of Falmouth. These ponds are situated on the southern margin of the Mashpee Outwash Plain, consisting of deposits about 50 to 60 ft thick in the study area. Each of the three ponds is a true estuary, acting as the mixing zone of terrestrial freshwater inflow and saline tidal waters from Vineyard Sound. Salinity in the three ponds ranges from approximately 30 ppt at the Vineyard Sound inlets to less than 10 ppt at the northern ends

Although the three embayment systems bounding Vineyard Sound exhibit slightly different hydrologic characteristics, the tidal forcing for these systems is generated from Vineyard Sound. Vineyard Sound, adjacent the barrier beach separating the salt ponds from the ocean, exhibits a moderate to low tide range, with a mean range of 0.5 m at the southern inlets of the ponds. Since the water elevation difference between Vineyard Sound and Great, Green and Bournes Ponds 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 result is a greater sensitivity degradation of these embayments to nitrogen loading than similar basins along Cape Cod Bay or the outer Cape.

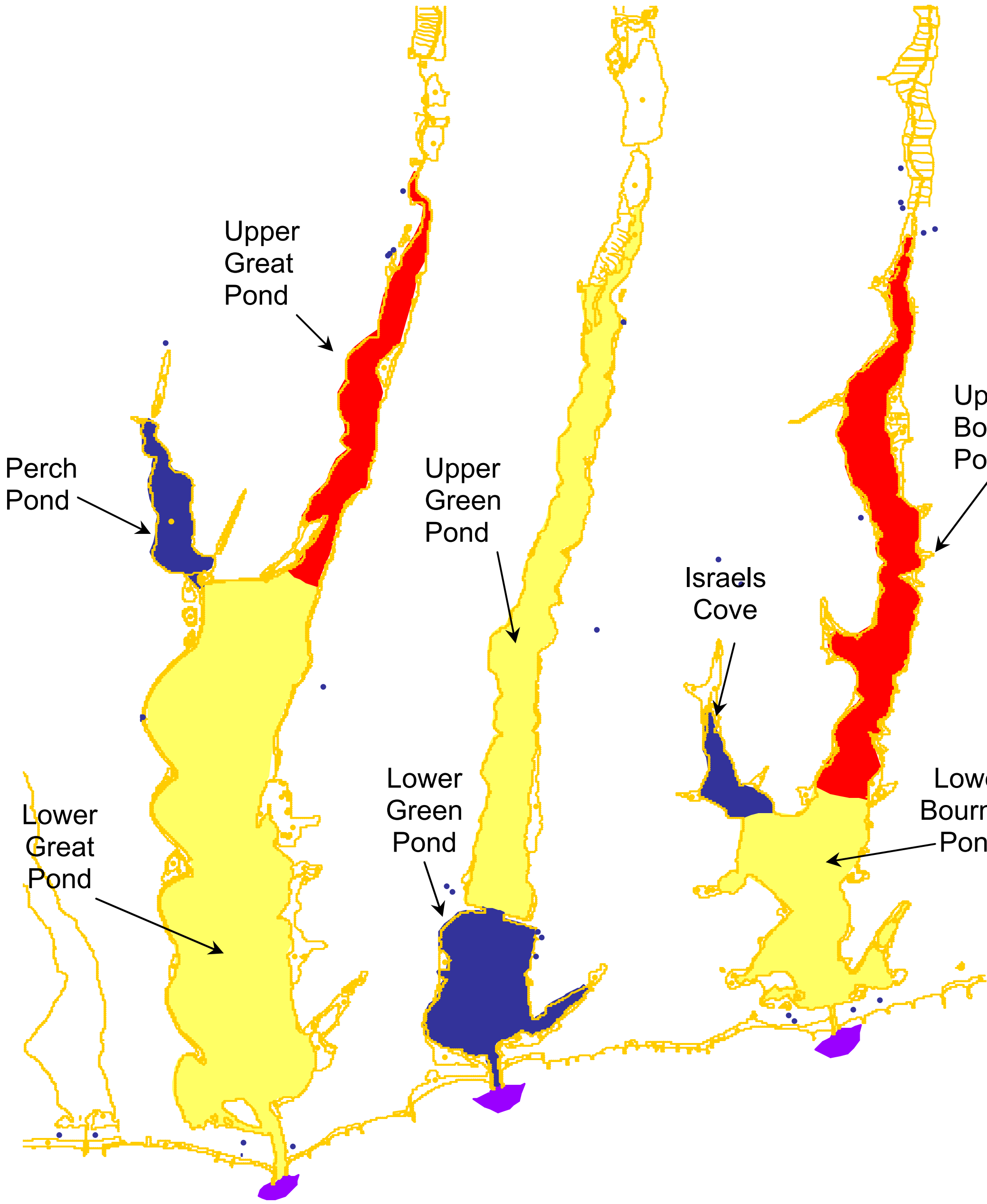


Figure I-3. Partitioning of each embayment system relative to basin boundary volumes and general regions of nitrogen related habitat quality.

Tidal damping (reduction in tidal amplitude) through an embayment can range from negligible indicating “well-flushed” conditions (relative to the driving tide range) or show tidal attenuation caused by constricted channels and marsh plains, indicating a “restrictive” system, where tidal flow and the associated flushing are inhibited. Tidal data collected by MEP throughout Great/Perch, Green and Bournes Ponds indicate only minimal tidal damping through the inlets. However, minor tidal attenuation (reduction in tide height) occurs through the existing Perch Pond entrance to Great Pond. It appears that the existing narrow, shallow channel that connects Perch Pond to Great Pond inhibits the exchange of tidal flow between the two ponds. Based on previous hydrodynamic modeling (Howes, Ramsey and Kelley, 2000), indications are that the Perch Pond flushing to Great Pond can be improved. Due to minimal tidal damping through the overall Great Pond system, only slight reductions in system flushing times may be realized if future modifications to the Great Pond inlet are considered as nutrient management alternative. This is similarly the case for tidal flushing of Green Pond and to a lesser degree for Bournes Pond.

It appears that the tidal inlets to the three systems are generally operating efficiently, possibly due to the Town’s active inlet maintenance program. Within each of the three systems, the tide propagates to the sub-embayments with negligible attenuation (with the exception of Perch Pond to Great Pond and Bournes Pond), consistent with generally well-flushed conditions. It must be noted, however, that though the tidal damping is generally minimal (with the exception of Bournes Bond), that does not necessarily translate to well flushed conditions throughout the entire length of each of the three salt ponds. This is in large part due to the geometry of these elongated finger ponds as well as the relatively small driving head (tidal height) in Vineyard Sound. As such, the lower portions of each pond generally exhibits relatively good flushing as opposed to the uppermost reaches of the ponds that are poorly flushed.

Green Pond appears to be hydrodynamically efficient, as the tide is damped negligibly between Vineyard Sound and the upper reaches, almost two miles upstream. Great Pond suffers some damping at the mouth, delaying the tide approximately 7 minutes, with more significant damping in its northern region. Perch Pond, a sub-embayment off the lower Great Pond basin, shows approximately half-hour delay in high water relative to Vineyard Sound. This delay may be due to a sand shoal at the Perch Pond mouth, which impedes the flow of water into the basin. Importantly, the results of the previous hydrodynamic analyses indicate Bournes Pond suffers from relatively greater frictional damping than adjacent Ponds, specifically between Vineyard Sound and the location of the tide gauge in lower Bournes Pond. This fact suggests the inlet at Bournes Pond is hydraulically less efficient than the Great Pond and Green Pond inlets.

Given the present hydrodynamic characteristics of the three embayments systems, it appears that estuarine habitat quality is more the result of present nutrient loading to bay waters than restricted tidal exchange within the component sub-embayments. In Bournes Pond, some enhancements to tidal flushing may be achieved via inlet modification resulting in some improvement of the nutrient loading impacts from the Bournes Pond watershed. The details of such are a part of the MEP analysis described in this report.

Nitrogen loading to the Great, Green, and Bournes Pond embayment systems was determined relative to the upper and lower portions of each salt pond as depicted in Figure I-3. Nitrogen loading was further evaluated relative to the Perch Pond sub-embayment to the Great Pond System and the Israels Cove sub-basin within the Bournes Pond System. The watersheds to the Great, Green and Bournes Pond Systems are primarily in the Town of Falmouth (approximately 84% of the watersheds are within the Town of Falmouth). In fact, the watersheds to these salt ponds encompass about 27% of the total Town land area, 9,632 acres (previous watershed study conducted by Ramsey and Howes, 2000) vs. 29,698 acres (Falmouth Planning Office, 1997). The remaining areas of the Ponds' upper watersheds are within the Towns of Bourne and Sandwich, but are nearly completely in conservation land or are part of the Massachusetts Military Reservation. Based upon land-use and the watersheds being predominantly in Falmouth, it appears that nitrogen management for Pond restoration can be fully conducted within the Town of Falmouth.

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. In previous studies for the Ashumet Plume Nitrogen Offset Program and DEP, as well as those conducted by PondWatch, it appears that there is a strong gradient in nitrogen level and health in Great Pond. The highest nitrogen and lowest environmental health is found within its headwaters and in Perch Pond while lowest nitrogen levels and greatest health is encountered in the lower portion of the main basin, near the inlet to Vineyard Sound. Both of the upper tributary basins to Great Pond are presently showing poor water quality and "Hyper-Eutrophic" conditions. Eelgrass is absent from these regions and periodic fish kills have been reported, resulting from oxygen depletion. Additionally, Perch Pond exhibits higher nitrogen levels than its adjacent source waters primarily due to the shoaling of its short inlet to the main basin of Great Pond. Perch Pond has experienced increased nitrogen levels, 1994 through 1997, compared to 1990 through 1993.

With regards to Green Pond, there is a strong gradient in nitrogen levels and environmental health within Green Pond, with highest nitrogen at the estuarine headwaters (approximately 1 mg N/L) and decreasing concentrations to a low in the lower basin between the Menauhant Road bridge and the inlet (approximately 0.5 mg N/L). As a result of high watershed nitrogen loading, Green Pond is currently showing significant/severe nitrogen related habitat quality impairments (N>0.7mg/L) over the entire upper two-thirds of its length. All eelgrass is absent in this region. There are macro-algal accumulations that smother shellfish and other bottom-dwelling animals. Large phytoplankton blooms (>20:g/L Chlorophyll a) are typical summer occurrences. These blooms result from the high nutrient availability and cause low watercolumn transparency (secchi depth <1 meter) and oxygen depletions to stressful levels (<4 mg/L). Fish kills related to periodic hypoxia occur almost every year.

Bournes Pond is the least nitrogen enriched of the three Falmouth ponds evaluated under the MEP nutrient threshold analysis. As a result, it supports sections with the highest water quality within the three systems. Nevertheless, Bournes Pond, like Green and Great Ponds, shows a strong longitudinal gradient in nitrogen and health, resulting from the distribution of its watershed nitrogen inputs and the exchanges with the high quality waters of Vineyard Sound. The upper basin of the pond, including the northerly third of the pond (marine reach of Bournes Brook), are currently showing poor water quality and "Eutrophic" conditions approaching that seen in upper Green and Great Ponds. The high nitrogen levels are associated with moderate to high chlorophyll *a* concentrations and moderate to high oxygen depletions. Water transparency within all but the most northerly reaches is sufficient to support benthic plant production. Most of the lower portion of Bournes Pond (more than half of the surface area) can be considered between Mesotrophic and Eutrophic. This can be seen in the modest chlorophyll a levels (<5:g/L), moderate oxygen depletions and moderate to poor water column transparencies. This region of the pond supports shellfish beds and does not appear to accumulate significant amounts of macro-algae. The MEP assessment of the present nitrogen related habitat quality of the Great, Green and Bournes Pond Systems presented in this report, builds on the previous studies mentioned above and incorporates additional measurements from continuous oxygen records, infaunal communities and temporal shifts in eelgrass distribution.

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 watersheds to the Great, Green and Bournes Pond embayment systems, phosphorus is highly retained during groundwater transport as a result of sorption to aquifer mineral (Weiskel and Howes 1991). Since even Cape Cod "rivers" are primarily groundwater fed, watersheds tend to release little phosphorus to coastal waters. In contrast, nitrogen, primarily as plant available nitrate, is readily transported through oxygenated groundwater systems on Cape Cod (DeSimone and Howes 1998, Weiskel and Howes 1992, Smith *et al.* 1991). The result is that terrestrial inputs to coastal waters tend to be higher in plant available nitrogen than phosphorus (relative to plant growth requirements). However, coastal estuaries tend to have algal growth limited by nitrogen availability, due to their flooding with low nitrogen coastal waters (Ryther and Dunstan 1971). Tidal reaches within the Great, Green and Bournes Pond Systems 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 this 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 Great, Green, and Bournes Pond Systems, as monitored by the Falmouth PondWatch Water Quality Monitoring Program, with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals) in order to "tune" general nitrogen thresholds typically used by the Cape Cod Commission, Buzzards Bay Project, and Massachusetts State Regulatory Agencies.

Unfortunately, almost all of the estuarine reaches within the Falmouth salt ponds that are the focus of this study appear to be near or beyond their ability to assimilate additional nutrients without impacting their ecological health. 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 systems and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within the Great, Green and Bournes Pond Systems could potentially occur without man's influence and must be considered in the MEP's 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, but restoration of critical resources should be the focus for management

I.4 WATER QUALITY MODELING

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

The MEP water quality evaluation examined the potential impacts of nitrogen loading into the Great, Green and Bournes Pond Systems, including the tributary sub-embayments of Perch Pond (tributary to Great Pond) and Israel's Cove (tributary to Bournes Pond). A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for each of the systems. Once the hydrodynamic properties of each estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP (Section III). Almost all nitrogen entering Falmouth's salt ponds is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Vineyard Sound source waters and throughout the Great, Green, and Bournes Pond Systems were taken from the Falmouth Pondwatch Water Quality 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 these Systems 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 Great, Green and Bournes Pond Systems 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 and the results of an infaunal community and eelgrass 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 these systems. Potential alterations in watershed nitrogen loading to each embayment were also conducted as additional "Scenario Runs" based upon detailed discussions with the Falmouth Nitrogen Management Committee. Finally, analyses of the Great, Green and Bournes Pond Systems was conducted 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 are presented in Section IX.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

In most marine and estuarine systems, such as the Great, Green, and Bournes Pond embayment systems 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 approach via the reduction of nitrogen loads generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission 1991, 1998; Howes et al. 2002).

Until recently, these tools for predicting loads and concentrations tended to be generic in nature, and overlook some of the specifics for any given water body. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using embayment specific measurements, thus enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including the Great, Green, and Bournes Pond Systems.

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. 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.

The Falmouth Pondwatch data was utilized to assess the overall nutrient related health of Falmouth's "finger ponds" (Little, Great, Green, and Bournes Ponds), where Howes and Goehringer,1996 concluded that increasing nutrient loading was resulting in "periodic dense algal blooms, malodorous conditions, and occasional fish kills from nutrient-related oxygen depletion in bottom waters." Based on this analysis, in 1994 the Massachusetts Highways Department hired Aubrey Consulting to evaluate potential water circulation improvements to the Menauhant Road causeway across Green Pond. A one-dimensional hydrodynamic analysis and an evaluation of culverts was performed (Aubrey Consulting, 1995). This study concluded that improved circulation would result along the regions immediately adjacent to the causeway if culverts were installed. The Massachusetts Highways Department included the culverts in their design effort for the bridge and the culverts were installed in 1996.

More recently, an additional source of nitrogen to three of the salt ponds on the south shore has been cause for concern. A plume of nitrogen rich groundwater was discovered approaching Green Pond and possibly Great Pond and Bournes Pond. The plume emanated from the Massachusetts Military Reservation's wastewater treatment facility which discharged secondarily treated effluent to rapid infiltration beds near the southeastern corner of MMR from 1936 through 1995. Although a new facility has come on-line which now discharges to Cape Cod Canal waters, decades of nitrogen discharge is still moving through groundwater towards Falmouth's coastal ponds. Since the plume (Ashumet Valley Wastewater Plume) would ultimately be discharging to already nitrogen-overloaded ecosystems, nitrogen remediation of plume waters was considered. Since the Ashumet Valley wastewater plume contains a large volume of contaminated water but at relatively low nitrogen levels, nitrogen removal is technically difficult and inefficient.

After evaluation of the plume remediation possibilities, an agreement was reached between MMR/AFCEE and the Town of Falmouth for management of nitrogen loading to the three salt ponds (Great, Green and Bournes Ponds) which could potentially receive plume nitrogen upon discharge. An innovative approach was developed whereby the Department of Defense would grant funding to the Town for nitrogen reduction, not of the Ashumet Valley Plume, but of other more readily addressed sources within the pond watersheds. Since all nitrogen inputs to the embayments impact ecological health regardless of the source, focusing on the more readily treatable sources (septic systems, fertilizers etc) should allow for a higher level and more rapid reduction in total nitrogen loading than merely treating the Ashumet Valley Plume. The Nitrogen Offset Program was established with $8.5 million for nitrogen source reductions within the watersheds of the three salt ponds potentially receiving MMR wastewater nitrogen through the Ashumet Valley Plume.

A major component of the MEP nutrient analysis is the evaluation of watershed based nutrient loading and hydrodynamics within the estuarine system. A watershed based nutrient loading model and a two-dimensional hydrodynamic and water quality model was previously developed in 2000 as part of a nutrient and hydrodynamic study of Great, Green, and Bournes Ponds. The study was undertaken as a collaboration between Applied Coastal Research and Engineering, Inc. and the then Center for Marine Science and Technology, University of Massachusetts at Dartmouth. The study was developed to include a detailed summary of the nutrient related water quality studies performed for the Town of Falmouth Nitrogen Offset Program and Horsley & Witten Inc. The nutrient and hydrodynamic study undertaken in 2000 was comprised of several tasks: Task 1 focused on evaluating the distribution and loading intensity of the various nitrogen sources within the watersheds of Great, Green and Bournes Ponds. Task 2 described the results of pond flushing studies and detailed the hydrodynamic modeling which formed the basis for the water quality modeling presented in Task 3 of the report. The results of the water quality modeling (Task 3) indicated which land areas and sources played the greatest role in the nitrogen degradation of these salt ponds.

Previous nitrogen loading analysis to Great, Green, and Bournes Ponds was undertaken under task 1 of the 2000 study. While the results of the watershed N loading analysis are primarily for parameterizing the embayment water quality model (Task 3), the analysis revealed several important factors relating to N sources and potential remediation. The analysis had 5 major findings relating to the N-Offset Program goals of nitrogen management for restoration of Great, Green and Bournes Ponds:

- Septic systems are the major single source of nitrogen to each of the salt ponds.
- The Ashumet Valley Wastewater Plume will only discharge to Great and Green Ponds. This plume will increase the N loading from the upper watershed over current conditions by 5% and 10%, for Great and Green Ponds, respectively. However, it will only increase the total N loading (upper and lower watersheds) to these salt ponds by 2% and 3%, for Great and Green Ponds, respectively.

- At maximum build-out and full discharge of the Ashumet Valley Wastewater Plume, total nitrogen loads from their watersheds are projected to increase over current conditions by 16% for Great Pond, 13% for Green Pond and 21% for Bournes Pond.
- Nitrogen loading from the lower watersheds accounts for most of the total N load to Great Pond-59%, Green Pond-70% and Bournes Pond-79%.

Since most of the nitrogen loading was found to be concentrated in the lower portions of the watersheds, which have little natural attenuation of nitrogen during transport to the salt ponds, engineered nitrogen remediation efforts should focus on these areas.

The previous study of Great, Green, and Bournes Ponds utilized a state-of-the-art computer model to evaluate tidal circulation and flushing. The particular model employed was the RMA-2V model developed by Resource Management Associates for the U.S. Army Corps of Engineers. It is a two-dimensional depth averaged model that solves momentum and flow continuity equations over several tidal cycles. The model is widely accepted and tested for analyses of estuaries or rivers. Prior to the use of the model in Great, Green, and Bournes Pond, Applied Coastal staff members had utilized RMA-2V for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, and the Pleasant Bay estuary.

The hydrodynamic modeling undertaken as task 2 produced information on the flushing characteristics throughout Great, Green and Bournes Ponds. The major hydrodynamic findings from task 2 of the historical hydrodynamic and water quality study for Great, Green, and Bournes Pond indicate:

- All of the ponds studied may be considered rapidly flushing systems, based upon their measured residence times.
- The relatively low residence time of upper Bournes Pond in comparison to upper Great Pond and upper Green Pond is due to natural water depth.
- Tides propagate from Vineyard Sound into each estuary, with little attenuation or amplitude damping and tides in all three ponds have flood-dominant characteristics.
- The greatest tide attenuation occurs in Bournes Pond. The restricted inlet causes a tide lag of approximately one-hour.
- Tide attenuation through Great Pond inlet and Green Pond inlet were negligible suggesting that improvements to these inlets will have a negligible impact on estuarine water quality.

The water quality modeling undertaken as task 3 produced information on the nutrient/habitat characteristics throughout Great, Green and Bournes Ponds. The upper reaches of each of the Great, Green and Bournes Ponds are currently showing poor nutrient related water quality as a result of nitrogen loading from the upper and lower watersheds. While the lower portions of each pond support at least moderate quality waters, only lower Bournes Pond exhibits a good level of environmental quality. The severely degraded environmental health of the upper portions of each of the Ponds is manifested in high chlorophyll a levels (>10 μg/L and typically >20 μg/L), periodic oxygen depletions to less than 4 mg/L, low water column transparency, and high nitrogen concentrations (>0.7 mg N/L). The nutrient overloaded nature of these systems is consistent with (a) the loss of eelgrass, (b) periodic fish kills due to oxygen depletion, and (c) periodic appearance of macro-algae. Each of the three ponds have total nitrogen concentrations above the levels set by the Falmouth Nutrient Overlay By-law. Since each of these coastal ponds show signs of degraded water quality, steps should be taken to limit additional nitrogen loading

For the MEP modeling analysis, the data from the previous studies were evaluated relative to the needs of the Linked Watershed-Embayment Model. Since the previous Applied Coastal and SMAST work on Great, Green, and Bournes Ponds utilized a similar modeling approach to the Linked Watershed-Embayment Model, much of the data incorporated into this previous analysis is useful for the updated MEP effort. Specifically, the tide and bathymetry data, as well as the hydrodynamic RMA-2V model, remain valid for the updated analysis. In addition, much of the Falmouth Pondwatch nutrient and salinity data has been incorporated into the water quality calibration and verification effort. Although benthic regeneration was evaluated as part of the original SMAST/Applied Coastal study of Great, Green, and Bournes Ponds, a more thorough analysis of this nitrogen loading component was required for the evaluation contained in this report. In addition, the previous watershed loading analysis has been superceded, due to recent improvements to the watershed delineations and the GIS-based land use evaluation.

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 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 Great, Green and Bournes Pond embayment systems.

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watersheds or contributing areas to the Great Pond, Green Pond, and Bournes Pond estuaries under evaluation by the Project Team. The Bournes Pond estuary system is composed of the main body of the estuary with one tributary basin, Israels Cove. Similarly, the Great Pond estuary system is composed of the main body of the estuary and a tributary basin, Perch Pond. The Green Pond estuary is a simple estuary, originating from sea level flooding of a linear valley without tributary branches. The watersheds to these estuaries and tributary basins were divided into functional sub-units based upon: (a) defining inputs from contributing areas to each major sub-embayment (basin) within the embayment system (for example, Perch Pond tributary to the Great Pond system), (b) defining contributing areas to major aquatic systems which might attenuate nitrogen passing through them on the way to the estuary (lakes >10 acres, streams, wetlands >10 acres), and (c) defining 10 year time-of-travel distributions within each sub-watershed in order 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 define the contributing areas to public water supply wells on 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 gauge 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 and 1994b). 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.

Biological attenuation of nitrogen (natural attenuation) occurs primarily within surface aquatic ecosystems (streams, wetlands, ponds) with little occurring within the main aquifer. Biological attenuation of nitrogen is predominantly through denitrification, sometimes directly from nitrate and sometimes indirectly after uptake by plants and remineralization and oxidation back to nitrate in the surface sediments. Burial of decayed plant matter containing nitrogen is almost always much less important than denitrification in reducing nitrogen transport. The freshwater ponds on Cape Cod provide important environments for the biological attenuation of nitrogen entering them and therefore also require that their contributing areas be delineated. Fresh ponds are typically “kettle” ponds, which are directly connected to the groundwater system and receive groundwater inflow through upgradient shores and discharge water into the aquifer in down gradient areas. The residence time of water within the ponds is a function of pond volume and inflow/outflow rates. Natural nitrogen attenuation is directly related, in part, to residence time.

III.2 MODEL DESCRIPTION

Contributing areas to each of the three Town of Falmouth salt ponds evaluated under this current nutrient threshold study (Great, Green and Bournes Pond) and local freshwater bodies were delineated using a regional model of the Sagamore flow cell. The USGS three-dimensional, finite-difference groundwater model MODFLOW-2000 (Harbaugh, *et al*., 2000) was used to simulate groundwater flow in the aquifer. The USGS particle-tracking program MODPATH4 (Pollock, 2000), which uses output files from MODFLOW-2000 to track the simulated movement of water in the aquifer, was used to delineate the area at the water table that contributes water to wells, streams, ponds, and coastal water bodies. This approach was used to determine the contributing areas to the Great, Green, and Bournes Pond Estuarine Systems and also to determine portions of recharged water that may flow through ponds and streams prior to discharging into the 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 sea level and have a uniform thickness of 10 ft. The top of layer 8 resides at sea level with layers 1-7 stacked above sea level to a maximum elevation of +70 feet. In regions like the Sagamore Lens in which the Great, Green, and Bournes Pond embayment systems reside, water elevations are greater than 60 ft at the top of the lens and therefore these uppermost layers are required for model operation. At depth within the aquifer, 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 glacial sediments that comprise the aquifer of the Sagamore flow cell consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence 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. While there are glacial morainal deposits comprising some regions of the aquifer of the Sagamore flow cell, these are generally located adjacent Buzzards Bay and are not found within the watersheds to Great, Green, and Bournes Pond. Most groundwater flow in the aquifer occurs in shallower portions of the aquifer dominated by coarser-grained sand and gravel deposits. Lithologic data used to determine hydraulic conductivities used in the 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, as well as 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. Since large portions of the watersheds to Great, Green, and Bournes Ponds are unsewered, 85% of the water pumped from wells was modeled as being returned to the ground via on-site septic systems.

III.3 GREAT POND, GREEN POND, AND BOURNES POND CONTRIBUTORY AREAS

The MEP watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the Great Pond, Green Pond, and Bournes Pond Embayment Systems and their major tributary basins (Perch Pond and Israels Cove for Great and Bournes Ponds, respectively and depicted in Figure III-1). Model outputs of MEP watershed boundaries were “smoothed” to correct for the grid spacing, to more accurately characterize the shorelines, and to refine the embayment segmentation to more closely match the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the MEP Technical Team. Overall, 33 sub-watershed areas were delineated within the watersheds to Great Pond (24 subwatersheds), Green Pond (4 subwatersheds), and Bournes Pond (5 subwatersheds) systems. 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. The MEP delineation includes subwatershed delineations to eleven freshwater ponds and 10 year time of travel boundaries. Contributing areas for fresh ponds were generally delineated if the pond was greater than 10 acres in area, i.e. about three groundwater model grid cells (400 ft X 400 ft each). The decision to use 3 model grid cells as minimum size criteria for ponds to which contributing areas would be developed was based partly on nitrogen attenuation considerations as well as computational complexity. Ponds with a surface area greater than or equal to 10 acres are likely to have the potential for nitrogen attenuation and as such warrant developing a sub-watershed delineation and performing a land use analysis in order to quantify the level of nitrogen attenuation. From a modeling point of view, including ponds less than 10 acres in size adds several degrees of computational complexity thereby making the groundwater models unwieldy with little if any measurable improvement in the watershed nitrogen loading analysis. The USGS determined that smaller ponds do not significantly intercept groundwater flows.

The watershed delineations completed for the MEP project are the third generation of delineations in less than 10 years for these three estuaries. Each delineation has included more sub-watershed detail and has been based on more refined data. Figure III-2 compares the delineation completed under the current effort with the delineations completed for the Cape Cod Commission in 1996 (Eichner, *et al*., 1998) and the USGS delineation in 2000 completed for SMAST to assist the Ashumet Plume Nitrogen Offset Program (Howes and Li, 2000). The delineation completed in 1996 was defined based on an average water level conditions water table map prepared by the Cape Cod Commission, while the 2000 delineation was completed by the USGS using a previous iteration of the Sagamore Lens groundwater model.

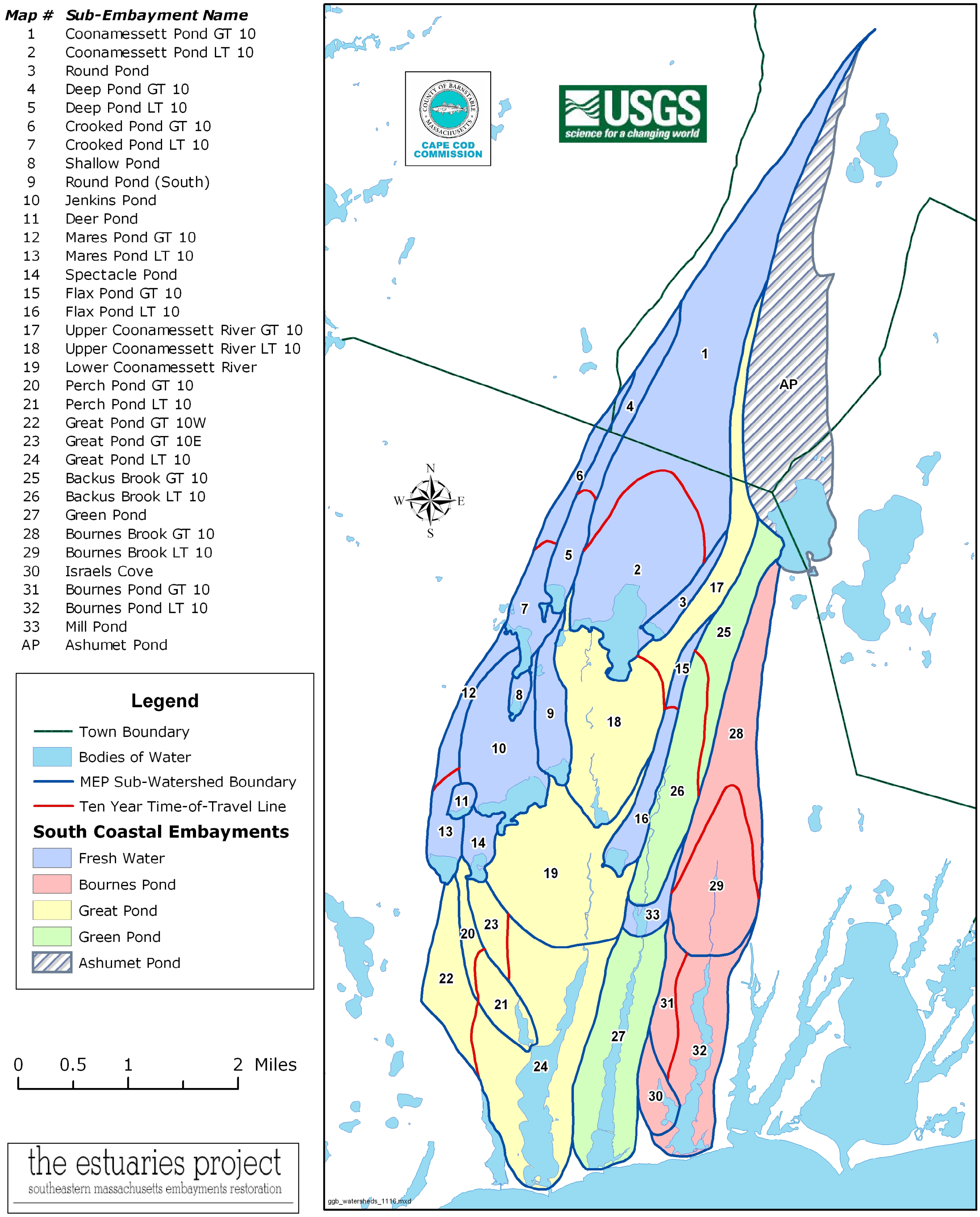


Figure III-1. Watershed and sub-watershed delineations for Great Pond, Green Pond and Bournes Pond. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the figure legend (above at left). Sub-watersheds to embayments were selected based upon the functional estuarine sub-units in the water quality model (see section VI).

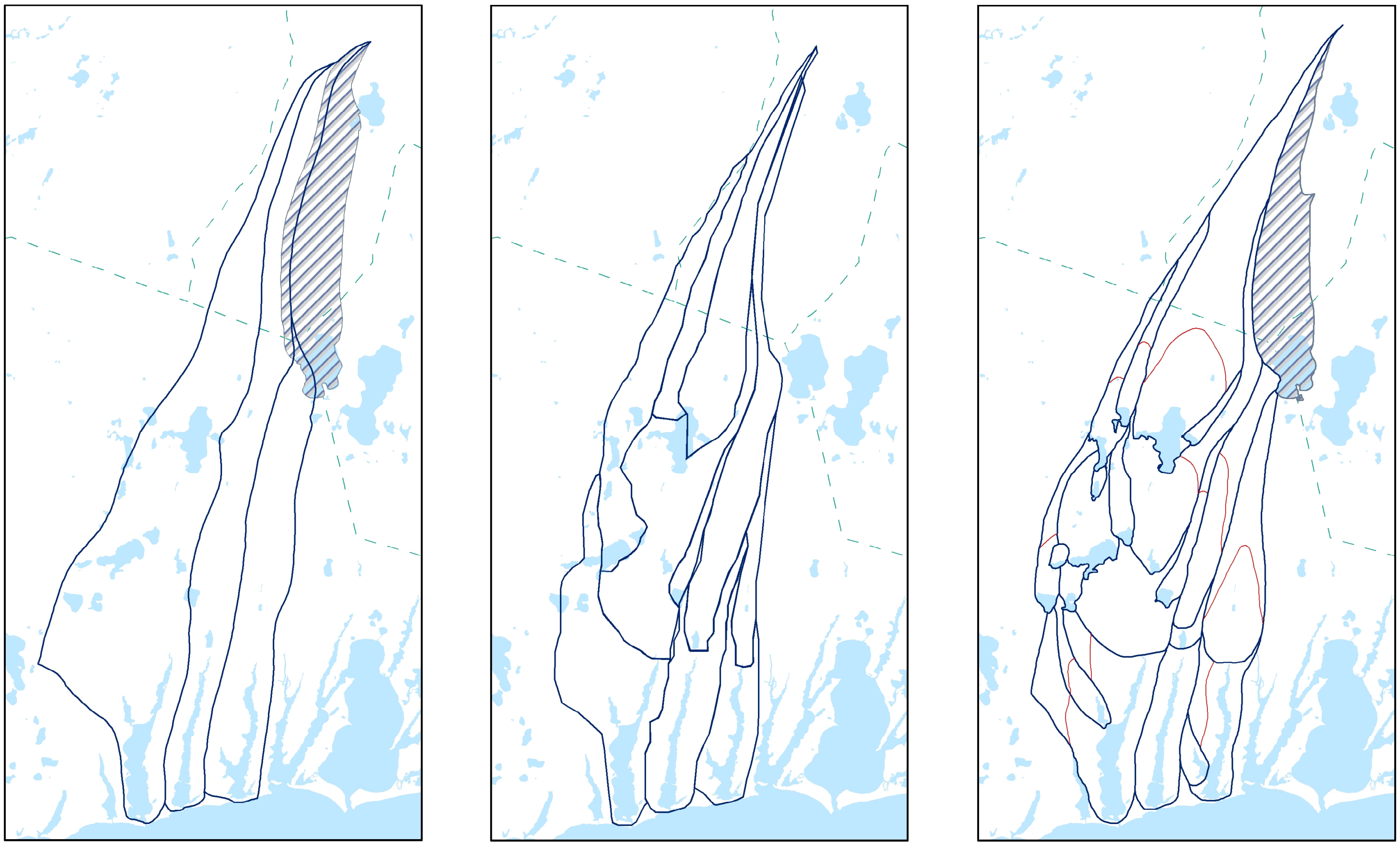
Table III-1. Daily groundwater discharge to the major streams and to the estuarine waters of Great Pond, Green Pond, and Bournes Pond Systems and their major tributary basins, as determined from the refined USGS groundwater model watershed outputs. Discharge values represents long-term average flows.

Watershed	Discharge	
	ft^3/day	m^3/day
Great Pond System		
Upper Coonamesset River	1,026,817	29,079
Lower Coonamesset River	539,361	15,275
Perch Pond	154,006	4,361
Great Pond	456,047	12,916
Great Pond Total	2,176,231	61,631
Green Pond System		
Backus Brook	243,616	6,899
Green Pond	224,955	6,371
Green Pond Total	468,571	13,270
Bournes Pond System		
Bournes Brook	206,454	5,847
Israels Cove	36,023	1,020
Bournes Pond	383,012	10,847
Bournes Pond Total	625,489	17,714

Table III-2 summarizes the percent difference in selected embayment watershed areas between watershed delineations utilized in previous assessments of Great Pond, Green Pond, and Bournes Pond and the newly delineated watersheds. In general, all of the watershed delineations show excellent agreement on the land-area contributing to the three estuaries combined. Differences result primarily in how the contributing areas are partitioned between each of the three systems. The MEP delineation (11,324 acres) is only 3% and 1% larger than the 2000 delineation (10,969 acres) for the Ashumet Plume Nitrogen Offset Program and 1996 delineation (11,186 acres) by the CCC, respectively.

Differences in the partitioning of the overall contributing area between each of the embayments showed greater differences among the delineations. While all approaches assign the largest watershed to the Great Pond System. The MEP watershed is 14% larger than previously derived by the earlier groundwater model and 41% larger than derived by the groundwater level approach. This latter difference is consistent with the need for large numbers of wells in coastal areas, for delineating watersheds by this approach rather than the current USGS model which includes water levels, stream and well discharges, and other hydrologic information. Similarly large area differences were found for Green Pond and Bournes Pond. The largest shift in the watershed delineation is the decrease in the Green Pond watershed area to 1,586 acres from either 2,978 or 2,558 acres. The Bournes Pond watershed area of 1,494 acres is close to the previously modeled area of 1,180 acres, but still much smaller than the 1996 watershed area of 2,370 acres. Overall, MEP watershed differences to Great Pond, Green Pond, and Bournes Pond ranged from –38 to +27% when compared to the Ashumet Plume Nitrogen Offset Program individual watersheds, with a combined area difference of 3%. The MEP watershed differences ranged from –47 to +41% when compared to the CCC individual watersheds, with a combined area difference of 1%.

CCC, 1996. **USGS, 2000.** **MEP, 2004.**



- Used in the 1996 & 2001 Regional Policy Plans. (Eichner, et al., 1998)

- Delineated by USGS (Howes and Li, 2000.)

- Based on USGS modeling and MEP refinement

Ashumet Pond Watershed

** Red lines indicate ten year time-of-travel lines*

Figure III-2. Comparison of previous and current Great Pond, Green Pond, and Bournes Pond watershed and subwatershed delineations.

Table III-2. Percent difference in delineated embayment watershed areas between old and newly revised delineations.

	CCC[1]	Ashumet Offset[2]	MEP[3]	MEP/Ashumet Offset difference		MEP/CCC difference	
	acres	acres	acres	%	acres	%	acres
Great Pond Total	5838	7231	8244	14%	1013	41%	2406
Green Pond Total	2978	2558	1586	-38%	-972	-47%	-1392
Bournes Pond Total	2370	1180	1494	27%	314	-37%	-876
OVERALL	11186	10969	11324	3%	355	1%	138

[1] Cape Cod Commission, Regional Policy Plan, 1996 & 2001 (delineation in Eichner, et al, 1998)
[2] Howes and Li, 2000
[3] This report (Green and Bournes Ponds receive a small portion of flow from Ashumet Pond; theland area associated with this flow is included in the above areas)

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

The MEP Technical Team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) to the Great, Green, and Bournes Pond embayment systems (Section III). The watersheds were sub-divided to define contributing areas to each of the major inland freshwater systems and to each major sub-embayment to Great, Green and Bournes Ponds and further sub-divided into regions greater and less than 10 year groundwater travel time from the receiving estuary, a total of 33 sub-watersheds in all. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to ponds and embayments.

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 view 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 in 1994 and 2001 in the time of travel watersheds, and reviewing water quality modeling, it was determined that each embayment was currently in balance with its watershed load. Therefore, the distinction of less than 10 year and greater than 10 year time of travel regions within a subwatershed (Figure III-1) was eliminated and the number of subwatersheds was reduced to 21. Although the percentage of nitrogen loads in the less than 10-year subwatersheds ranges between 48% and 100% among the three systems, the bulk of the total nitrogen load to each estuary is within 10 years flow to Great Pond (84%), Green Pond (86%), and Bournes Pond (66%). Within the regions of the watershed greater than 10 years travel time from the estuaries, most of the development occurred prior to 1990. In addition, it is important to note that even with Falmouth's rapid growth, almost all of the development has been within the 10 year time of travel zones or has occurred prior to 1995. 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 subwatersheds to Great Pond, Green Pond, and Bournes Pond

	LT10	GT10	TOTAL	%LT10
WATERSHED	kg/yr	kg/yr	kg/yr	
Upper Coonamessett River	5647	1053	6700	84%
Lower Coonamessett River	10038	119	10157	99%
Perch Pond	1628	845	2473	66%
GREAT POND TOTAL	25257	4939	30196	84%
Backus Brook	1530	1680	3210	48%
GREEN POND TOTAL	9920	1680	11601	86%
Bournes Brook	1276	1307	2583	49%
Israels Cove	845	0	845	100%
BOURNES POND TOTAL	4983	2543	7526	66%

Note: no subwatersheds were split for this analysis; some loads to ponds located in the <10 yr subwatersheds to coastal surface waters may take >10 yrs to reach the coast

In order to determine nitrogen loads from large watersheds, detailed individual lot-by-lot data is used for some portion of the loads, while information developed from other detailed studies is applied to other portions. The Linked Watershed-Embayment Management Model (Howes & Ramsey 2001) uses a land-use Nitrogen Loading Sub-Model based upon subwatershed-specific land-uses and pre-determined nitrogen loading rates. For the Great Pond, Green Pond, and Bournes Pond embayment systems, the model used Falmouth and Massachusetts Military Reservation (MMR)-specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local site-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 site-specific studies within the freshwater portions of the Coonamessett River, Backus Brook, and Bournes Brook and through the freshwater ponds. Attenuation during transport through each of the major fresh ponds was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Attenuation during transport through each of the major fresh ponds was assumed to equal 50% based on refined monitoring of selected Cape Cod lakes. Available historic data collected from individual fresh ponds in the Great Pond, Green Pond, and Bournes Pond watersheds confirmed the appropriateness of this assumption. Internal nitrogen recycling was also determined throughout the tidal reaches of the Great Pond, Green Pond, and Bournes Pond embayments;

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 staff obtained digital parcel and tax assessors data from the Town of Falmouth. Digital parcel line work is from 2001 and Falmouth's land use data was obtained from the Town of Falmouth Assessors Office and is from 2003. The land use database contains additional information about impervious surfaces (building area, driveways, and parking area) on individual lots. Land use information within the Massachusetts Military Reservation was also obtained from publicly available aerial photos. The parcel coverages and assessors database were combined for the MEP analysis by using the Cape Cod Commission Geographic Information System (GIS). Nitrogen loading from development after the land use data collection (2000, 2001) and prior to summer 2003 (the last embayment monitoring date) must be within the 1.5-2.5 yr travel time to affect the nitrogen loading estimate. MEP staff sought information on any large development occurring in this time window for separate inclusion into the land use analysis. Based upon these efforts and the time of travel constraint, this source of error is deemed negligible.

Figure IV-1 shows the land uses within the study area; assessors land uses classifications (MADOR, 2002) are aggregated into nine land use categories:

1) residential
2) commercial
3) industrial
4) undeveloped
5) mixed use
6) recreational/golf course
7) agricultural (including cranberry bogs)
8) ponds
9) public service/government (including road rights-of-way)

In the watersheds reviewed, the predominant land use based on area is either public service/government or residential, with these two land uses accounting for >70% of each embayment's watershed. Within the whole Great Pond watershed, public service/government land uses occupy 46% of the area, including 64% of the Upper Coonamessett River subwatershed (Figure IV-2). This high percentage is largely due to the inclusion of a portion of the Massachusetts Military Reservation. Residential land use represents the next largest fraction of the watershed at 28%. The whole Green Pond and Bournes Pond watersheds are predominantly residential land use at 46% and 44%, respectively, with public service accounting for 36% and 26% of their respective watershed areas. Commercial properties are scattered throughout the watersheds with the majority of parcels located along Route 28.

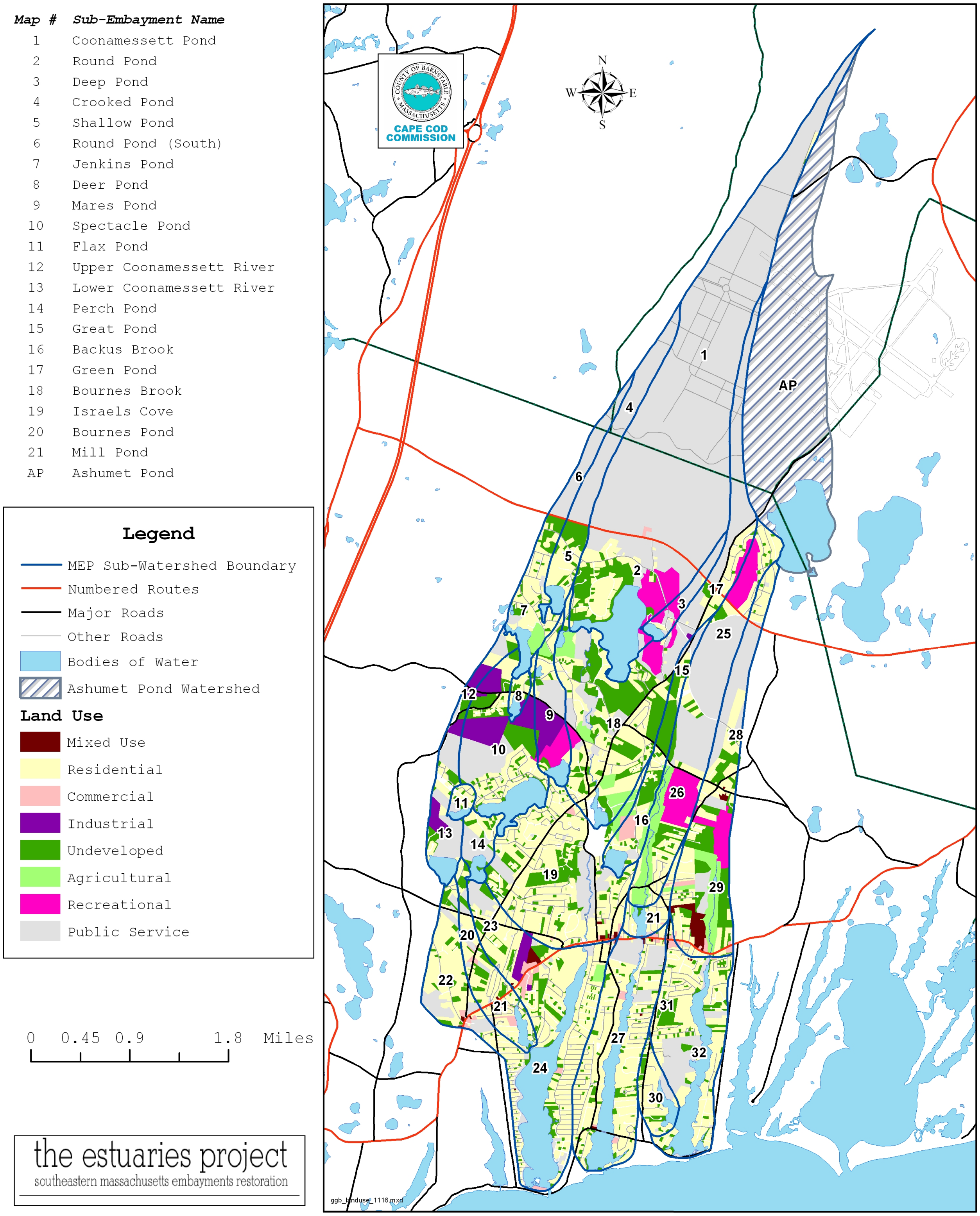


Figure IV-1. Land-use coverage in the Great Pond, Green Pond, and Bournes Pond watersheds. Watershed data encompasses portions of the Town of Falmouth and the Massachusetts Military Reservation, MA (northern portion of Great Pond watershed).

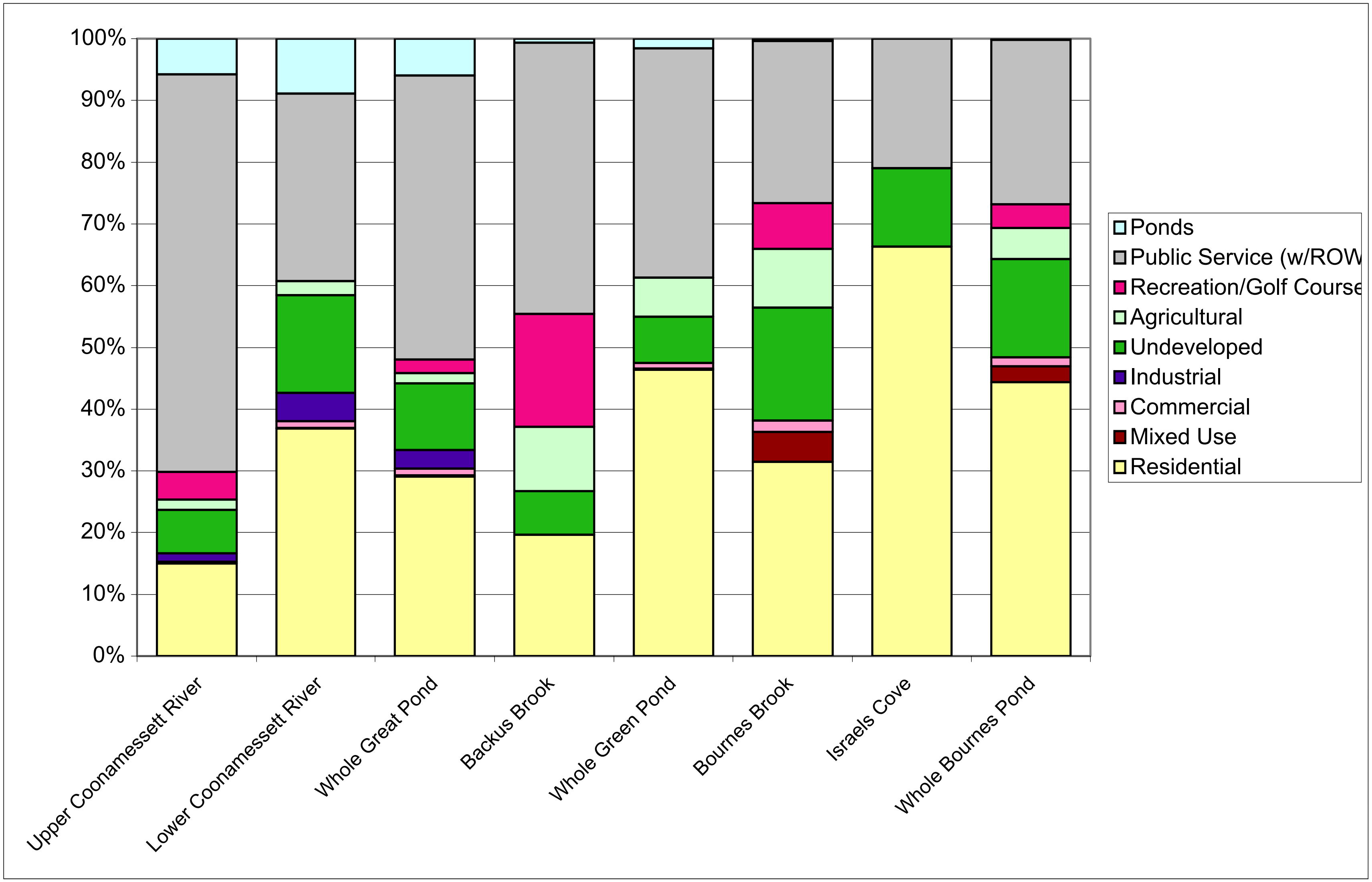


Figure IV-2. Distribution of land-uses within the major subwatersheds and whole watersheds to Great Pond, Green Pond, and Bournes Pond.

In order to estimate wastewater flows within the study area, MEP staff also obtained parcel by parcel water use information from the Town of Falmouth. This information included two years of water use information with the final reading in May 2003 for the majority of parcels, which are billed on a semiannual basis, and one complete year of data (mid-2002 to mid-2003) for approximately 300 accounts that are billed on a quarterly basis. 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 watersheds, but the nitrogen additions are included from the old MMR WWTF effluent discharge beds that are the source of the Ashumet Valley Plume of treated wastewater.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

All wastewater within the Great Pond, Green Pond, and Bournes Pond watersheds is returned to the aquifer through individual on-site septic systems. Wastewater based nitrogen loading from the parcels using on-site septic systems is based upon the measured water-use, nitrogen concentration in wastewater (35 mg N/L) and nitrogen loss estimates within the septic tank and soil adsorption system (25%). Loss in passage through the septic system used by MEP (Howes and Ramsey 2000, Weiskel and Howes 1991) is consistent with other regional studies (Brawley et al. 2000, Costa et al. 2001). The best quantitative information on Title 5 septic system nitrogen removals (21%-25%) was developed at the DEP's Alternative Septic System Test Center at MMR. Multi-year monitoring of Title 5 septic system performance revealed that nitrogen removal within the septic tank was small (1%-3%), with most of the removal occurring within the soil adsorption system (Costa et al. 2001).

Wastewater engineering studies conventionally assume 90% of water used in a town is converted to wastewater (*e.g.*, Massachusetts Water Resources Authority 1983, Stearns and Wheler, 1999). In order to check the reliability of parcel water use as a proxy for wastewater flow, average influent flow at three nearby WWTFs (Mashpee Commons, Willowbend, and the Town of Falmouth municipal facility) was compared to parcel water use within the respective service areas. These WWTFs, which are located to the east and west of the Great, Green and Bournes Pond watersheds have more diverse mixes of land uses within them, however, the analysis is useful as a local check of the 90% engineering assumption. The review of the WWTFs found that 79% of the Mashpee Commons (primarily commercial) water use is returned to the WWTF, 87% of the Willowbend water use is returned, and 87% of the water use in the Town of Falmouth sewer service area is returned to the WWTF. This analysis confirms that 90% return flow is an appropriate general adjustment when using water use in the nitrogen loading calculations within the Great Pond, Green Pond, and Bournes Pond watersheds.

While almost all of the developed parcels within the study watersheds have corresponding water use accounts 3% did not; all of these latter parcels are residential and 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. Of the 215 developed parcels in the study area without water use in the available database; 139 are in the Great Pond watershed, 9 are in the Green Pond watershed, and 67 are in the Bournes Pond watershed. Average water use was also used for determining nitrogen loads from new development determined in the buildout analyses.

In order to provide an independent validation of the residential water use average within the study area, MEP staff reviewed US Census population values. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd), so each person generates 55 gpd of wastewater. Average occupancy within the Town of Falmouth during the 2000 US Census was 2.36 people per household. If 2.36 is multiplied by 55 gpd, 130 gpd would be calculated as the average residential wastewater flow in Falmouth. This compares favorably with the study area residential wastewater estimate of 136 gpd, based on average water use (151 gpd) and the water use to wastewater conversion of 0.9. Alternatively, if 136 gpd is divided by 55 gpd, the resulting occupancy is 2.47 people per household. These analyses confirm that 151 gpd is an appropriate value for the few residential parcels without water use and for the analysis of future, buildout residential dwellings.

Table IV-2. Average Water Use in Great Pond, Green Pond, and Bournes Pond Watersheds.

Land Use	State Class Codes	# of Parcels in Study Area	Water Use (gallons per day)		
			Study Area Avg	Town Avg	Town Range
Residential	101	5,889	151	153	**0 TO 79,618**
Commercial	300 to 389	73	553	911	**0 TO 29,115**
Industrial	400 to 439	4	18	1,229	**0 TO 13,583**

Note: All data for analysis supplied by the Town of Falmouth.

Commercial and industrial building footprints were made available to Estuaries Project staff as part of an impervious surface GIS coverage provided by the Town of Falmouth Planning Department. MEP staff used this data to review existing water use for these properties based on square footage of building and to determine the building percentage as a portion of each commercial or industrial lot. Based on this analysis, project staff determined that the town-wide average commercial water use is 122 gpd/1,000 ft^2 of building, while the town-wide average industrial water use is 112 gpd/1,000 ft^2 of building. These values were used only in the buildout analysis to determine water use for all future commercial and industrial additions, since all existing commercial and industrial properties in the study area have public water supply accounts. It should be noted that within each category of land use presented in Table IV-2 (i.e., residential, commercial, and industrial) are 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 Town of Chatham and Mashpee. Buildout building areas were determined by assuming maximum lot coverage allowed under current town zoning regulations; this coverage is 35 to 40% in the zoning areas that overlap the Great Pond, Green Pond, and Bournes Pond watersheds. Based on this review of zoning, no industrial buildout additions were included in any portions of the watersheds, while 263,222 ft^2 of future commercial development could be added.

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. 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 Great, Green and Bournes Pond watersheds are listed in Table IV-3.

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 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 (small public water supplies, golf courses, etc.) were also assigned at this stage. DEP and Town records were reviewed to determine water use for small public water supplies (e.g. non-community public water supplies). Additionally, golf course superintendents for two golf courses in the study area were contacted to determine fertilizer application rates. 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 Great, Green and Bournes Pond Estuaries. 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.

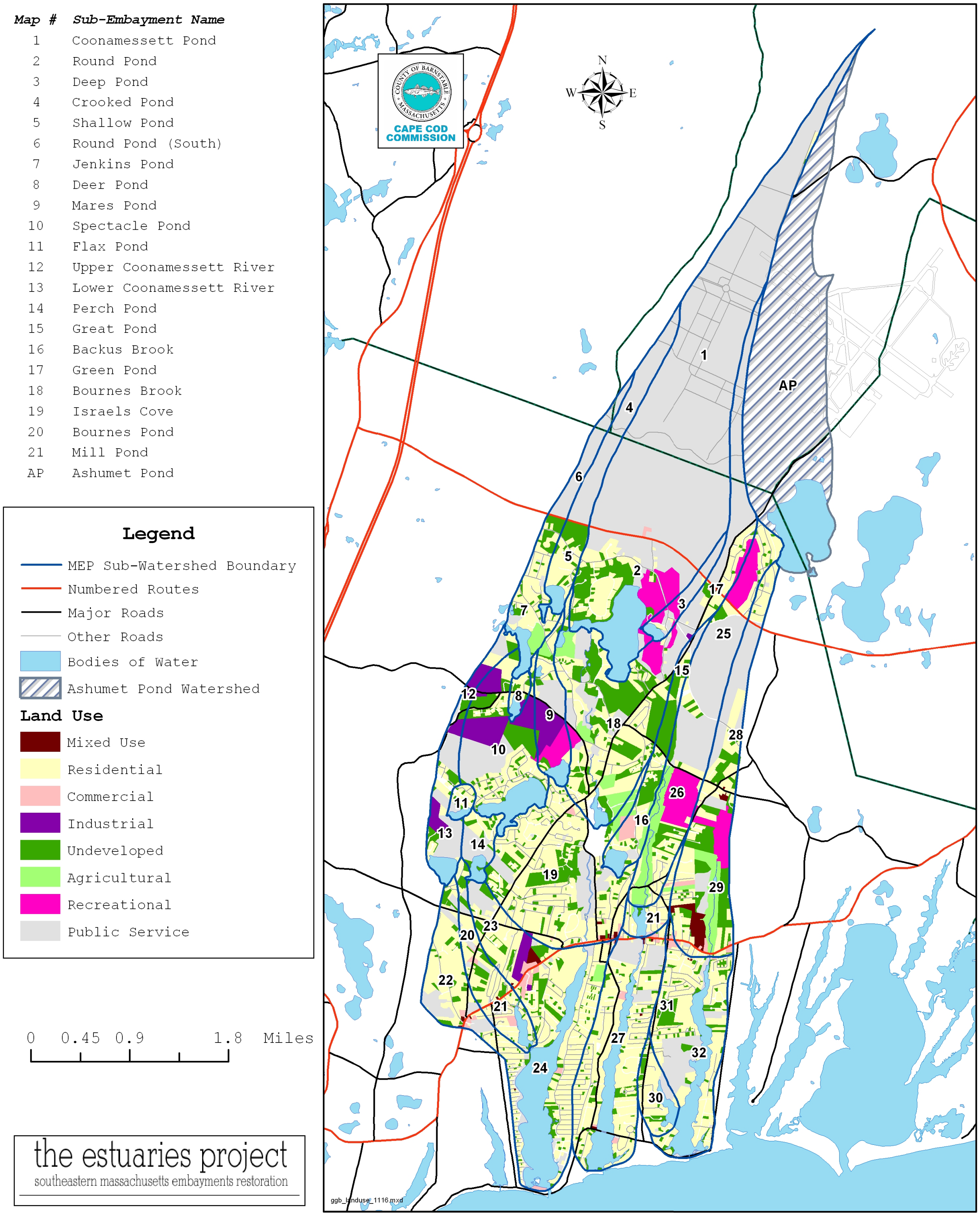


Figure IV-3. Parcels, Parcelized Watersheds, and Developable Parcels in the Great Pond, Green Pond, and Bournes Pond watersheds.

Table IV-3. Primary Nitrogen Loading Factors used in Great Pond, Green Pond, and Bournes 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	35
Road Run-off	1.5
Roof Run-off	0.75
Direct Precipitation on Embayments and Ponds	1.09
Natural Area Recharge	0.072
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 Parcels wo/water accounts:	gpd
Single Family Residence	151
Commercial Properties	122 per 1,000 ft^2 of building
Industrial Properties	112 per 1,000 ft^2 of building
For Parcels w/water accounts:	Measured annual water use
Wastewater volume determined by multiplying water use by 0.9	

Following the assignment of all parcels to individual watersheds, tables were generated for each of the 33 sub-watersheds summarizing water use, parcel area, frequency, sewer connections, private wells, and road area. As mentioned above, these tables were then condensed to 21 subwatersheds based upon the time of travel analysis (<10 yr vs. > 10 yr) discussed above.

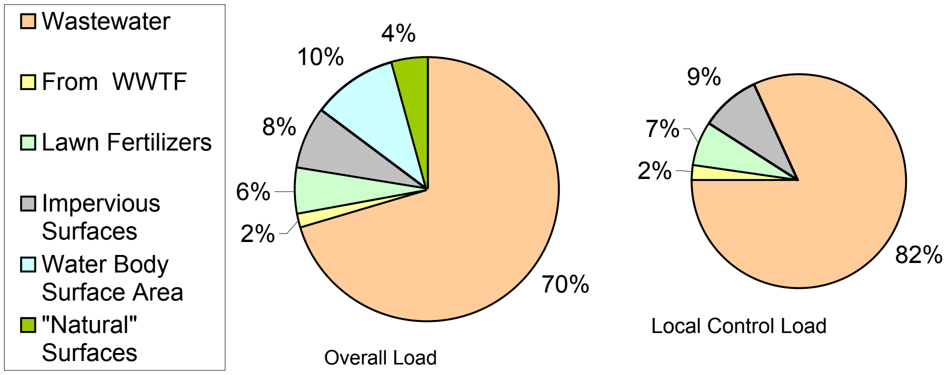
The 21 individual sub-watershed assessments were then integrated to generate nitrogen loading tables relating to the each of the individual estuaries and their major components: Great Pond main stem Perch Pond, Upper (fresh) and Lower (fresh) Coonamessett River; Green Pond Estuary and Backus Brook; and Bournes Pond main stem, Israel Cove and Bournes Brook. 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 Great, Green and Bournes Pond systems the major types of nitrogen loads are: wastewater (septic systems and the Ashumet Plume), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, 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 upper watershed to each estuary is also directly measured (Section IV.2) and compared to the attenuated annual watershed nitrogen load from the land-use sub-model.

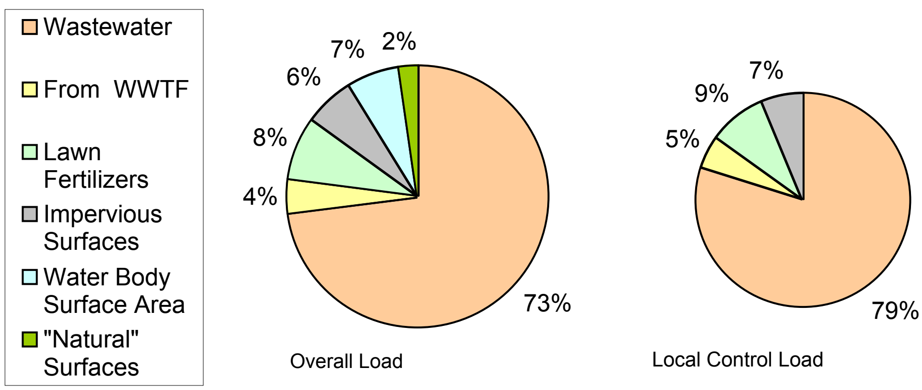
Table IV-4. Great Pond, Green Pond, and Bournes Pond Nitrogen Loads. Attenuation of system nitrogen loads occurs as nitrogen moves through freshwater ponds, marshes and stream systems during transport the estuaries.

		Great, Green, and Bournes Ponds N Loads by Input:								***Present N Loads***			***Buildout N Loads***		
Name	**Watershed ID#**	**Wastewater**	**From WWTF**	**Lawn Fertilizers**	**Impervious Surfaces**	**Water Body Surface Area**	**"Natural" Surfaces**	**Buildout**	**% of Pond Outflow**	**UnAtten N Load**	**Atten %**	**Atten N Load**	**UnAtten N Load**	**Atten %**	**Atten N Load**
Great Pond System	**1 to 15**	**21208**	**539**	**1700**	**2319**	**3128**	**1302**	**4781**		**30196**		**21833**	**34977**		**24945**
Great Pond	15	8092	0	425	746	1376	226	749		10866		10299	11615		10977
Perch Pond	14 + Mares + Spectacle Ponds	1881	0	92	188	230	82	433		2473		2041	2906		2419
Coonamesset River		11235	539	1183	1385	1522	994	3599		16857		9493	20457		11549
Upper Coonamessett River	12 + Coonamesset + Round + Deep + Round (S) Ponds	3586	0	829	726	894	666	2352		6700	30%	3616	9052	30%	5037
Lower Coonamesset River	13 + Spectacle + Round (S) + Jenkins + Flax + Crooked + Shallow Ponds	7649	539	354	659	629	328	1247		10157	30%	5877	11405	30%	6513
Ashumet Plume			539							539		539	539		539
Green Pond System	**16, 17, 21 + Ashumet Pond**	**8458**	**499**	**909**	**688**	**763**	**281**	**949**		**11598**		**8941**	**12546**		**9526**
Green Pond	17	5979	0	256	415	586	119	453		7356		7356	7809		7809
Mill Pond (MP)	21 + Backus Brk	2479	499	653	273	176	162	495	100%	4242	50%	1585	4737	50%	1716
Backus Brook	16 + Ashumet Pond	1639	499	613	202	106	151	461		3210	30%	2138	3671	30%	2367
Ashumet Pond (AP)		109	2	7	67	106	21	267	11%	313	50%	156	579	50%	290
Ashumet Plume			497							497		497	497		497
Bournes Pond System	**18, 19, 20 + Ashumet Pond**	**5573**	**1**	**485**	**502**	**723**	**242**	**1261**		**7526**		**6711**	**8787**		**7704**
Israels Cove	19	651	0	25	54	95	19	51		845		845	896		896
Bournes Pond	20	3031	0	139	250	589	89	428		4098		4098	4526		4526
Bournes Brook	18 + Ashumet Pond	1891	1	320	198	39	135	782		2583	30%	1769	3365	30%	2282
Ashumet Pond (AP)		40	1	3	24	39	8	97	4%	113	50%	57	210	50%	105

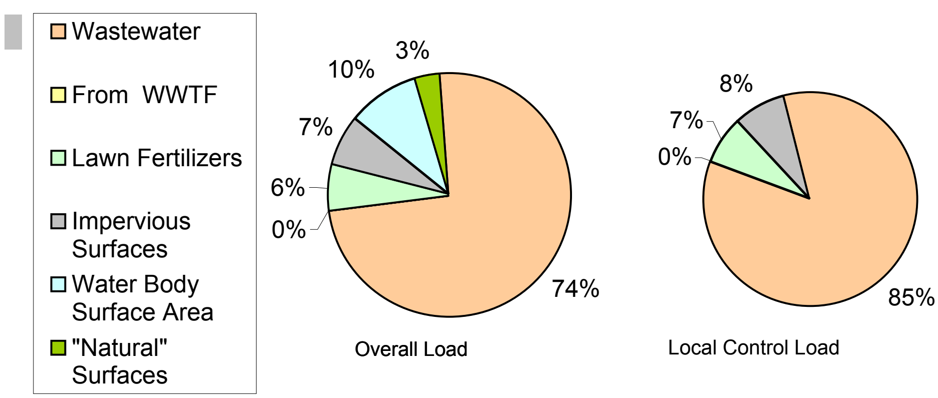
AP = Ashumet Pond, CP = Coonamessett Pond, CRP = Crooked Pond, DP = Deep Pond, FP = Flax Pond, JP = Jenkins Pond, MP = Mares Pond, RP = Round Pond, RPS = Round Pond (South), SP= Shallow Pond, SPP = Spectacle Pond



a. Great Pond System



b. Green Pond System



c. Bournes Pond System

Figure IV-4 (a-c). Land use-specific unattenuated nitrogen load (by percent) to the (a) Great Pond System, (b) Green Pond System, and (c) Bournes Pond System. “Overall Load” is the total nitrogen input to the aquifer or aquatic surfaces, while the “Local Control Load” represents only those nitrogen sources that stem only from activities within the watershed itself (i.e. no atmospheric deposition).

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally formed from 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 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 watershed-estuarine complex. The Nitrogen Load Summary Table IV-4 includes both the unattenuated (nitrogen load to each subwatershed) 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.

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 through the Cape Cod Pond and Lake Stewardship (PALS) program, which is a collaborative effort between the Cape Cod Commission and the Coastal Systems Program at SMAST. This data is part of annual regional snapshots of pond water quality collected between mid-August and the end of September. These regional 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 Great Pond, Green Pond, and Bournes Pond watersheds, there are 12 major freshwater ponds for which MEP delineated subwatersheds and conducted nitrogen loading analysis: Coonamessett Pond, Round Pond, Deep Pond, Crooked Pond, Shallow Pond, Round Pond (South), Jenkins Pond, Deer Pond, Mares Pond, Spectacle Pond, Flax Pond and Mill Pond (see Figure IV-1). PALS data is only available for Deep Pond and only from the 2001 snapshot (three samples). Additional data on Flax Pond (29 samples; AFCEE, 2000) was found after reviewing Massachusetts Military Reservation (MMR) documents in the Cape Cod Commission archives. It is unclear whether nutrient monitoring data for the other ponds in the study area has been collected.

In order to estimate nitrogen attenuation in the ponds, available physical and chemical data was analyzed . Available bathymetric information was reviewed relative to measured pond temperature profiles to determine the epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) in each pond. Of the ponds in the study area, bathymetric information is only available for Coonamessett, Deep, Jenkins, Mares, and Round ponds. An estimate of the volume of Flax Pond was obtained from AFCEE (2000). Of these, Crooked, Jenkins, and Mare are deep enough to develop strong temperature stratification, but temperature profile data is not available to confirm this, so turnover time calculations utilized the entire pond volume. Following this determination, the volume of this portion was determined and compared to the annual volume of recharge from each pond's watershed in order to determine how long it takes the aquifer to completely exchange the water in this portion of the pond (*i.e.*, turnover time). Using the total nitrogen concentrations collected only within the epilimnion, the total mass of nitrogen within this portion of the pond was determined. This mass was then adjusted using the pond turnover time to determine how much nitrogen is returned to the aquifer through the downgradient shoreline on an annual basis. In ponds with homothermic water columns, the nitrogen mass within the pond was based on the entire water volume.

Table IV-5 summarizes the pond attenuation estimates calculated for Deep (26%) and Flax (69%) ponds; these are the only ponds in the study area with sufficient information to compare pond watershed nitrogen loads and estimated pools of nitrogen based on pond nitrogen concentrations. However, a caveat to these attenuation estimates is that they are based upon nitrogen outflow loads from summer water column samples, and are not necessarily representative of the annual nitrogen loads that are transferred downgradient. More detailed annual studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) support a 50%-60% attenuation factor. This factor is also consistent with the freshwater pond attenuation factors used for the nitrogen balance for Great, Green and Bournes Ponds (embayments) in the Town of Falmouth (Howes and Ramsey, 2001). Significantly, direct measurement of the nitrogen discharge from the upper watershed to each of the estuaries indicated an integrated nitrogen attenuation rate for ponds plus streams consistent with a pond attenuation rate of 50%-60% (see Section IV-2, below). These direct measurements were from 16-18 month records of flow and nitrogen load at the discharge of the Coonamessett River to Great Pond, Backus Brook to Green Pond and Bournes Brook to Bournes Pond.. This site specific pond data and the stream nitrogen load measurements support a pond attenuation of 50% as a conservative estimate for the watersheds to the Great, Green, and Bournes Pond embayment systems.

Since groundwater outflow from a pond can enter more than one down gradient sub-watershed, the length of shoreline on the down gradient side of the pond was used to apportion the attenuated nitrogen load to respective down gradient watersheds. The apportioning of nitrogen load was based on the percentage of discharging pond shoreline bordering each downgradient sub-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 subwatershed was determined from the GIS database (Figure IV-3). Buildout of parcels were determined in consultation with the Falmouth Planning Department, including commercial and industrial parcel estimates. All municipal overlay districts (*e.g.*, water resource protection districts) and existing zoning were considered in the determination of minimum lot sizes. A nitrogen load for each parcel was determined for the existing development using the factors presented in Table IV-3 and 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 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.

Table IV-5. Nitrogen attenuation by Freshwater Ponds in the Great Pond, Green Pond, and Bournes Pond watersheds based upon late summer 2001 and 2002 Cape Cod Pond and Lakes Stewardship (PALS) program sampling and Massachusetts Military Reservation (MMR)-associated monitoring. These data were collected to provide a site specific check on nitrogen attenuation by these systems. The MEP Linked N Model for Great Pond, Green Pond, and Bournes Pond uses a value of 50% for the non-stream discharge systems.

Pond	PALS ID	Area acres	Maximum Depth m	Overall turnover time yrs	N Load Attenuation %
Coonamessett	FA-855	162	10.4	0.5	
Crooked	FA-884	34	12.8		
Deep	FA-857	21	8.5	0.3	26%
Deer	FA-884	9			
Flax	FA-937	22	8.8	0.5	69%
Jenkins	FA-918	89	15.5	1.0	
Mares	FA-938	29	16.8	0.6	
Round	FA-882	11	3.0	0.3	
Round (South)	FA-916	20			
Shallow	FA-904	12			
Spectacle	FA-939	19			
Mill Pond	FA-948	11			
				Mean	48%
*bathymetric and water quality data is unavailable for most ponds in the study area				s.d.	31%

Data sources: all areas from CCC GIS; Max Depth from MADFW, PALS monitoring, & MMR reports; Volume for turnover time calculations from MADFW bathymetric maps (www.mass.gov/dfwele/dfw/dfw_pond.htm) and estimates based on max depth; TN concentrations for attenuation calculation from PALS monitoring and MMR reports

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 each of the three Falmouth salt ponds (Great, Green and Bournes 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, nitrogen passes through a surface water ecosystem (pond, wetland, stream) on its path to the adjacent embayment. Surface water systems, unlike sandy aquifers, do support the needed conditions for nitrogen retention and denitrification. The result is that the mass of nitrogen passing through lakes, ponds, streams and marshes (fresh and salt) is diminished by natural biological processes that represent removal (not just temporary storage). However, this natural attenuation of nitrogen load is not uniformly distributed within the watershed, but is associated with ponds, streams and marshes. In the cases of the Great, Green and Bournes 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 in the upland watershed, which has natural attenuation predominantly associated with riverine processes, the integrated attenuation was 39% (Howes et al. 2004). In addition, a preliminary study of Great, Green and Bournes Ponds in Falmouth, measurements indicated a 30% attenuation of nitrogen during stream transport (Howes and Ramsey 2001). An example where natural attenuation played a significant role in nitrogen management can be seen relative to West Falmouth Harbor (Falmouth, MA), where ~40% of the nitrogen discharge to the Harbor originating from the groundwater effluent plume emanating from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Similarly, the small tidal basin of Frost Fish Creek in the Town of Chatham showed ~20% nitrogen attenuation of 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 each embayment system in addition to the natural attenuation measures by fresh kettle ponds, addressed above. These additional site-specific studies were conducted in each of the 3 major surface water flow systems (e.g. the Coonamessett River discharging to the tidal portion (head) of Great Pond, the creek from Mill Pond discharging to the head of Green Pond and the freshwater discharge from the cranberry bog draining into the head of Bournes Pond).

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the flow and nutrient load associated with the Coonamessett River (at Route 28), the creek from Mill Pond (at Route 28) and the creek from the cranberry bog to Bournes Pond (at Route 28) provide a direct integrated measure of all of the processes presently attenuating nitrogen in the sub-watersheds upgradient from the gauging sites. These upper watershed regions account for more than half of the entire watershed area to the Great, Green and Bournes Pond embayment systems. Flow and nitrogen load were measured at each site for 16 months of record (Figure IV-5). During the study period, velocity profiles were completed on each river every month to two months. The summation of the products of stream subsection areas of the stream cross-section and the respective measured velocities represent the computation of instantaneous stream flow (Q).

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. Velocity measurements made across the entire stream cross section were not averaged and then applied to the total stream cross sectional area.

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

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

where by:

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

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

Periodic measurement of flows over the entire stream gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauges. Water level data obtained every 10-minutes was averaged to obtain hourly stages for a given river. These hourly stages values where then entered into the Stage-discharge relation to compute hourly flow. Hourly flows were summed over a period of 24 hours to obtain daily flow and further, daily flows summed to obtain annual flow. In the case of tidal influence on stream stage, the diurnal low tide stage value was extracted on a day by day basis in order to resolve the stage value indicative of strictly freshwater flow. The two low tide stage values for any given day were averaged and the average stage value for a given day was then entered into the stage – discharge relation in order to compute daily flow. A complete annual record of stream flow (365 days) was generated for each of the surfacewater discharges flowing into each individual salt pond.

Figure IV-5. Location of Stream gauges (red triangles) in the Great, Green and Bournes Pond embayment systems.

The annual flow record for each surface water flow 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 Great Pond, Green Pond and Bournes Pond. Nitrogen discharge from a given 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 gaging 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 each pond currently reduces (percent attenuation) nitrogen loading to each salt pond embayment system.

IV.2.2 Surface water Discharge and Attenuation of Watershed Nitrogen: Coonamessett River to Great Pond (upper)

Coonamessett Pond is one of the larger ponds on Cape Cod and unlike many of the freshwater ponds, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, the Coonamessett River, may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. In addition, nitrogen attenuation also occurs within the wetlands and streambed associated with the Coonamessett River. The combined rate of nitrogen attenuation by these processes was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to the Coonamessett River above the gauge site and the measured annual discharge of nitrogen to the tidal portion of the upper tributary to Great Pond, Figure IV-5.

At the Coonamessett River gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater portion of the Coonamessett River that carries the flows and associated nitrogen load to the head of the upper tributary of Great Pond. As the Coonamessett River is tidally influenced upgradient of the Route 28 bridge, the gauge was located above the saltwater reach and such that freshwater flow could be measured during the low tide period. To confirm that freshwater was being measured at low tide, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 0.4 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly. The gauge on the Coonamessett River was installed on June 24, 2002 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to a need for additional low tide velocity measurements to develop a valid rating curve, stage data collection was extended until February 26, 2004 for a total deployment of 20 months. The 12-month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured at low tide every 4 to 6 weeks using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Coonamessett River 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 tidal influence on stage was filtered out of the record by examining stage at ebb slack tide. 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 Coonamessett River (Figure IV-6 and Table IV-6). In addition, a water balance was constructed based upon the US Geological Survey groundwater flow model to determine long-term average freshwater discharge expected at each gauge site.

The Coonamessett River is one of the larger surface freshwater discharges along the south shore of Falmouth, with a discharge ~2.5 fold higher than the combined flow of the Backus Brook and Bournes Brook and about 2/3 the annual discharge of the Quashnet River into Waquoit Bay. The annual freshwater flow record for the Coonamessett River measured by the MEP, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the Coonamessett River was 60% of the long-term average modeled flows. This value is consistent with the low groundwater levels during the initial months of the study period. Therefore, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the Coonamessett River outflow were relatively high, 0.851 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 22.63 kg/day and a measured total annual TN load of 8,259 kg/yr. In the Coonamessett River, nitrate was the predominant form of nitrogen (66%), 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 Coonamessett River to the estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the Bay. Based upon lower nitrogen load (8,259 kg yr^{-1}) discharged from the freshwater Coonamessett River compared to that added by the various land-uses to the associated watershed (16,857 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 51% (i.e. 51% of nitrogen input to watershed does not reach the estuary). This measured level of attenuation is also greater than the integrated attenuation rate determined from the watershed nitrogen model of 44% (Table IV-4). This is expected given the conservative assumptions of nitrogen attenuation used in the model. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-6. Comparison of water flow and nitrogen discharges from Coonamessett River to Great Pond, Backus Brook from Mill Pond discharging to Green Pond and Bournes Brook discharging from the cranberry bog upgradient of Route 28 to Bournes Pond. The “Stream” data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Bournes Brook Discharge to Bournes Pond©	Backus Brook Discharge to Green Pond[d]	Coonamesset River Discharge to to Great Pond[e]	Data Source
Total Days of Record	365[a]	365[a]	365[b]	(1)
Flow Characteristics				
Stream Average Discharge (m3/day)	3766	7211	26593	(1)
Contributing Area Average Discharge (m3/day)	5847	6899	44354	(2)
Discharge Stream 2002-03 vs. Long-term Discharge	0.64	1.05	0.60	
Nitrogen Characteristics				
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.543	0.062	0.565	(1)
Stream Average Total N Concentration (mg N/L)	0.874	0.528	0.851	(1)
Nitrate + Nitrite as Percent of Total N (%)	62%	12%	66%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/day)	3.29	3.81	22.63	(1)
TN Average Contributing Area Attenuated Load (kg/day)	4.85	4.34	26.01	(3)
TN Average Contributing UN-attenuated Load (kg/day)	7.08	11.62	46.18	(4)
Attenuation of Nitrogen in Pond/Stream (%)	54%	67%	51%	(5)

(a) from September 11, 2002 to September 11, 2003
(b) from September 10, 2002 to September 10, 2003
(c) Flow and N load to creek discharging into Bournes Pond include cranberry bog contributing area.
(d) Flow and N load to stream discharging to Green Pond includes Mill Pond contributing area.
(e) Flow and N load to Coonamessett River discharging to Great Pond includes the Coonamessett Pond and Flax Pond contributing areas.

(1) MEP gage site data
(2) Calculated from USGS-MEP watershed delineations to Coonamessett Pond and Flax Pond for flow to Great Pond, Mill Pond to Green Pond, and the bog to Bournes Pond; the fractional flow path from each sub-watershed which contribute to the flow in the Coonamessett River, stream to Green Pond, and creek to Bournes Pond; and the annual recharge rate.
(3) MEP watershed nitrogen loading sub-model, with pond and stream conservative attentuation rates.
(4) MEP watershed nitrogen loading sub-model, without pond and stream conservative attentuation rates.
(5) Calculated from the measured TN discharge from the rivers vs. the unattenuated watershed load, calculated as the mass of N removed versus the total mass load

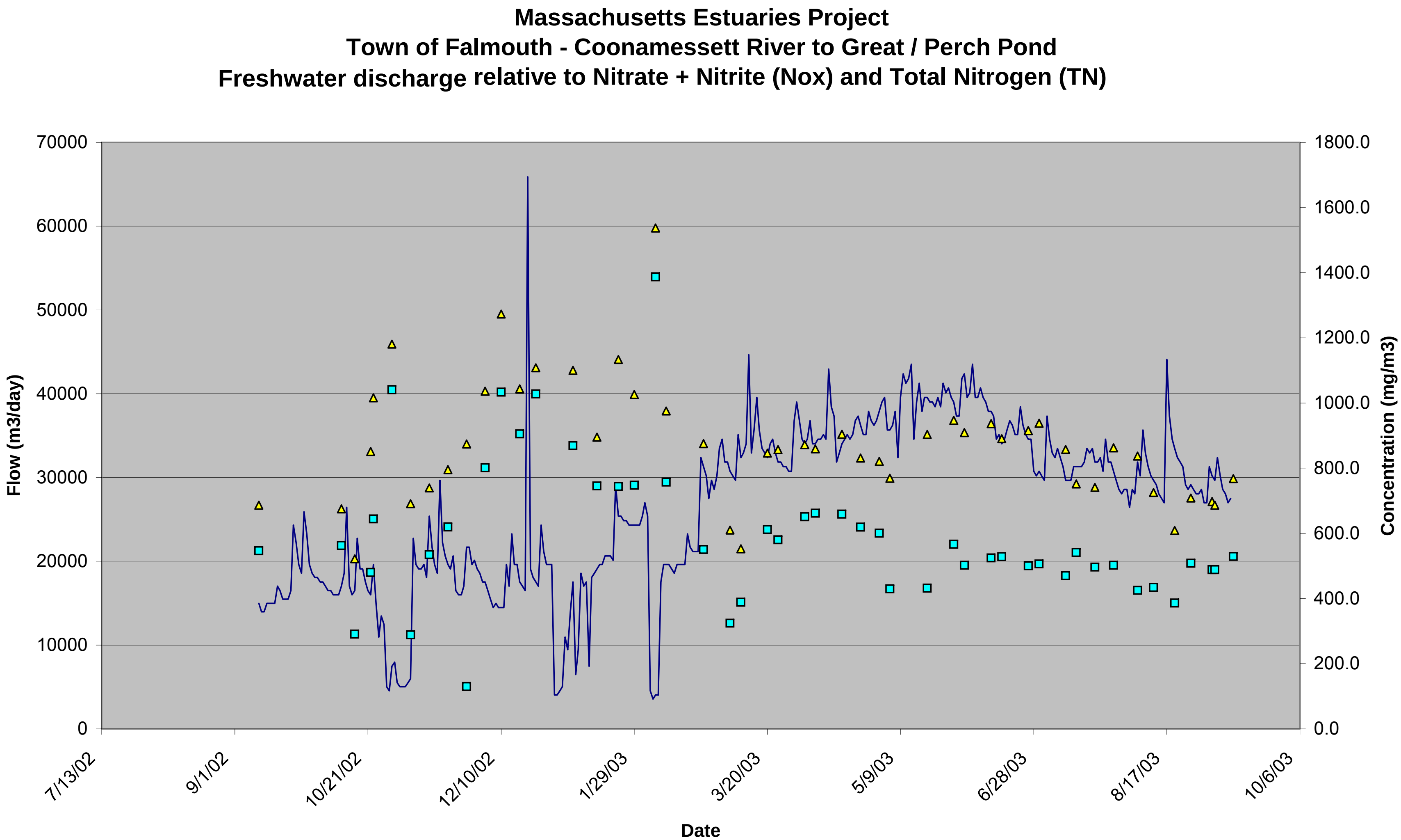


Figure IV-6. Coonamessett River discharge (solid blue line), nitrate+nitrite (blue boxes) and total nitrogen (yellow triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to the Great Pond Estuary (Table IV-6).

IV.2.3 Freshwater Discharge and Attenuation of Watershed Nitrogen: Backus Brook from Mill Pond to Green Pond (upper)

The upper watershed to Green Pond contains one significant surface water flow, Backus Brook, which discharges to the headwaters of Green Pond. Backus Brook passes through an impoundment, Mill Pond, just prior to discharging to Green Pond. Mill Pond appears to be a man-made system, formed by damming the lower Backus Brook, and has been part of the surface water system for over 125 years (Mill Pond can be seen on 1880 maps of Falmouth). Mill Pond (15.6 acres) is one of the larger ponds within the watershed to the Green Pond embayment system and unlike many of the freshwater ponds in the study area (e.g. Ashumet Pond, Jenkins Pond, etc.) it has stream outflow to the head of Green Pond, rather than discharging solely to the aquifer on the down-gradient shore. As for Backus Brook discharging from Mill Pond, this surface water outflow may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the integrated nitrogen attenuation within the upper Green Pond watershed. It is likely that nitrogen attenuation occurs both within the stream bed of Backus Brook and within Mill Pond with some additional attenuation associated with Backus Brook prior to the discharge to the estuary. 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 Backus Brook above the gauge site (Figure III-1, IV-5) and the measured annual discharge of nitrogen to the tidal portion of the upper tributary to Great Pond.

At the Backus Brook gauge site (immediately downgradient of Rt. 28), a continuously recording vented calibrated water level gauge was installed to yield the level of water in Backus Brook for the determination of freshwater flow (Figure IV-5). Early review of the stage record indicated that the gauge location was not tidally influenced. To confirm that freshwater was being measured at low tide, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 0.11 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked approximately monthly. The gauge on Backus Brook was installed on June 24, 2002 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to the desire to have simultaneous measurement of river discharge from the Coonamessett River and Bournes Brook (the creek flowing from the cranberry bog up-gradient of adjacent Bournes Pond), stage data collection was extended slightly until November 18, 2003 at which point the instrument failed and was removed from the river. The 12-month uninterrupted record (September 2002 to September 2003) used in this analysis encompasses the summer 2003 field season and corresponds to the same period of record for the Coonamessett River and Bournes Brook.

River flow (volumetric discharge) was measured approximately monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Backus Brook down gradient from Mill Pond based upon these flow measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to the headwaters of the Green Pond embayment system (Figure IV-7 and Table IV-6).

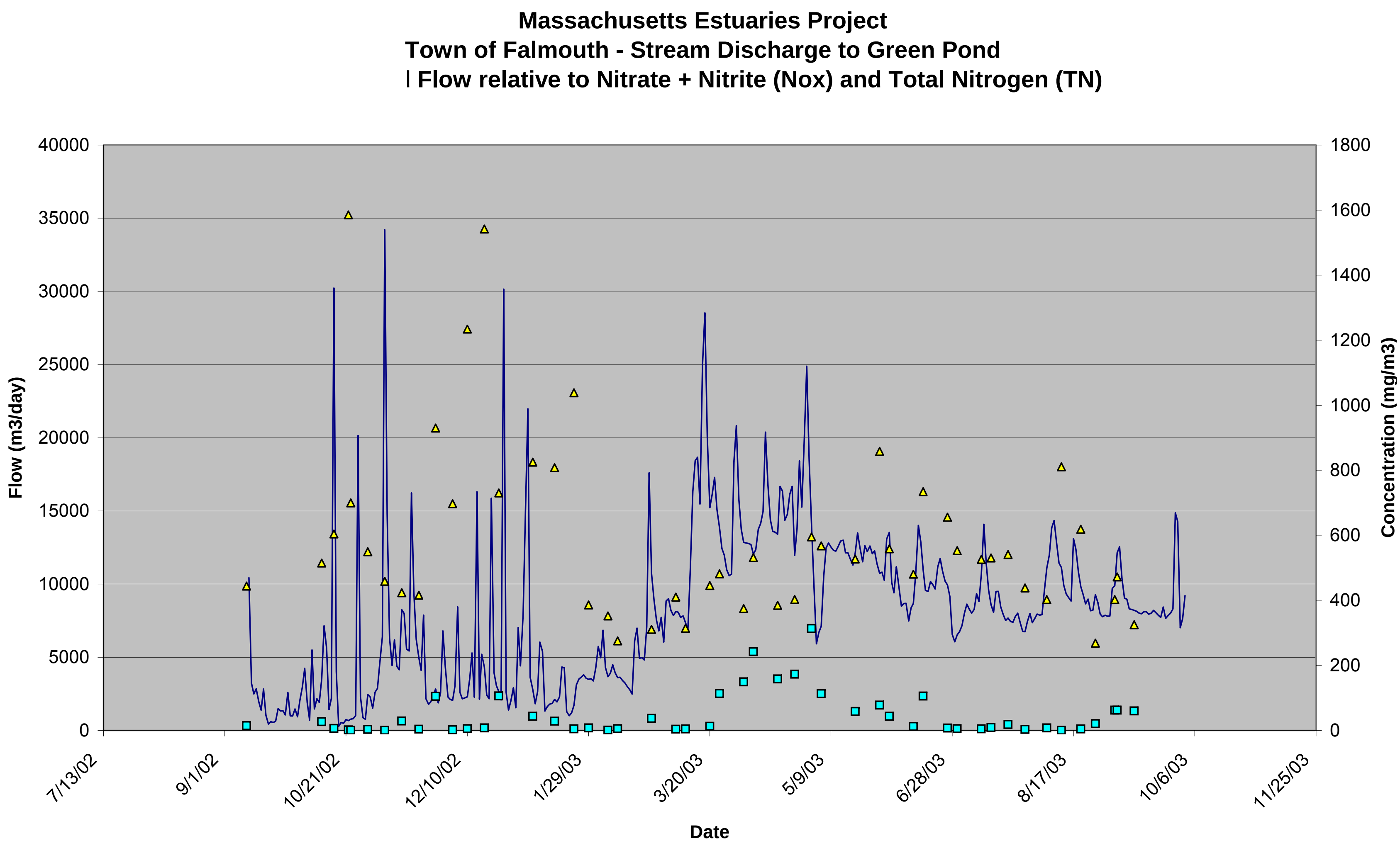


Figure IV-7. Backus Brook freshwater discharge (solid blue line), nitrate+nitrite (blue boxes) and total nitrogen (yellow triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to the Green Pond Estuary (Table IV-6).

Backus Brook is a moderate surface freshwater discharge compared to others along the south shore of Falmouth, with a discharge ~1/4 of the Coonamessett River and 2 times that of Bournes Brook. The annual freshwater flow record for Backus Brook measured by the MEP, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from Backus Brook was equal to the long-term average modeled flows. Therefore, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the Backus Brook outflow were relatively low, 0.528 mg N L^{-1} in comparison to total nitrogen concentrations determined for the Coonamessett River gauging location (0.851 mg N L^{-1}), yielding an average daily total nitrogen discharge to the estuary of 3.81 kg/day and a measured total annual TN load of 1391 kg/yr. In Backus Brook, nitrate was a small fraction of the total nitrogen (12%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the freshwater Mill Pond and to the river was taken up to a large extent by plants within the pond ecosystem. This is not surprising given the highly eutrophic state of Mill Pond.

From the measured nitrogen load discharged by the Backus Brook to the headwaters of the Green Pond estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon the measured lower nitrogen discharged from the Backus Brook load (1,391 kg yr^{-1}) compared to that added by the various land-uses to the associated watershed (4,242 kg yr^{-1}, Table IV-4), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 67% (i.e. 67% of nitrogen input to watershed does not reach the estuary). The measured level of integrated nitrogen attenuation in the Backus Brook (67%) was similar to that measured in the larger adjacent Coonamessett River (51%), most likely due to the final passage through Mill Pond. The measured level of attenuation in the Backus Brook is similar to the integrated attenuation rate determined from the watershed nitrogen model of 63% (Table IV-4). The slightly lower attenuation rate in the land use model is expected given the conservative assumptions of nitrogen attenuation used in the model. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.2.4 Freshwater Discharge and Attenuation of Watershed Nitrogen: Bournes Brook, Creek from Cranberry Bog to Bournes Pond (upper)

The upper watershed to Bournes contains one significant surface water flow, Bournes Brook, which discharges to the headwaters of Bournes Pond. Bournes Brook passes through a large cranberry bog system, just prior to discharging to Bournes Pond. The lack of a freshwater pond (like Mill Pond on the Backus River) in the lower region of the upper Bournes Pond watershed and the direct surface water discharge to the estuary may serve to decrease the attenuation of nitrogen. However, the surface water flow does allow for a direct measurement of the integrated nitrogen attenuation within the upper watershed. It is likely that nitrogen attenuation occurs within the streambed and associated freshwater wetlands of the Bournes Brook System. The integrated rate of upper watershed nitrogen attenuation was determined by comparing the present predicted nitrogen loading to the sub-watershed region contributing to Bournes Brook above the gauge site (Figures III-1, IV-5) and the measured annual discharge of nitrogen to the tidal portion of the upper tributary to Bournes Pond.

At the Bournes Brook gauge site located downgradient from the cranberry bogs (Figure IV-5), a continuously recording vented calibrated water level gauge was installed to yield the level of water for the determination of freshwater flow. As the lower reach of Bournes Brook is tidally influenced, the gauge was located at the culvert passing under Route 28, such that freshwater flow could be measured during the low tide period. To confirm that freshwater was being measured at low tide, salinity measurements were conducted on the weekly water quality samples collected from the gauge site. Average low tide salinity was determined to be 0.26 ppt therefore, the gauge location was deemed acceptable for making freshwater flow measurements. Calibration of the gauge was checked monthly throughout the study period. The gauge was installed on June 24, 2002 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to the desire to have simultaneous measurement of surface water discharge from the Coonamessett River and Backus Brook, stage data collection was extended until October 3, 2003. The 12-month uninterrupted record (September 2002 to September 2003) used in this analysis encompasses the summer 2003 field season and corresponds to the same period of record for the Coonamessett River to Great Pond and the Backus Brook discharging to the head of Green Pond.

River flow (volumetric discharge) was measured approximately monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Bournes Brook gauge site based upon these flow measurements and the measured water levels. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to the headwaters of the Bournes Pond embayment system (Figure IV-8 and Table IV-6).

Bournes Brook is a small surface freshwater discharge compared to others along the south shore of Falmouth, with a discharge 1/7 of the Coonamessett River and 1/2 that of Backus Brook in the adjacent Green Pond watershed. The annual freshwater flow record for Bournes Brook measured by the MEP, was compared to the long-term average flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from Bournes Brook was 64% of the long-term average modeled flows. This value is consistent with the low groundwater levels during the initial months of the study period. Therefore, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the Bournes Brook outflow were high, 0.874 mg N L^{-1} and similar to total nitrogen concentrations determined for the Coonamessett River gauging location (0.851 mg N L^{-1}). In contrast, Backus Brook averaged TN levels 40% (0.528 mg N L^{-1}) lower than Bournes Brook. Based upon measured flow and nitrogen levels, Bournes Brook discharges an average daily total nitrogen load to the Bourne Pond estuary of 3.29 kg/day and a measured total annual TN load of 1,201 kg/yr. In Bournes Brook, nitrate was a large fraction of the total nitrogen pool (62%), indicating that groundwater nitrogen (typically dominated by nitrate) discharging to the upgradient freshwater systems and to the river was not completely taken up by plants. 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. This finding is similar to that for the Coonamessett River, but again contrasts with Backus Brook, which supports a freshwater pond just upgradient of the discharge to estuarine waters.

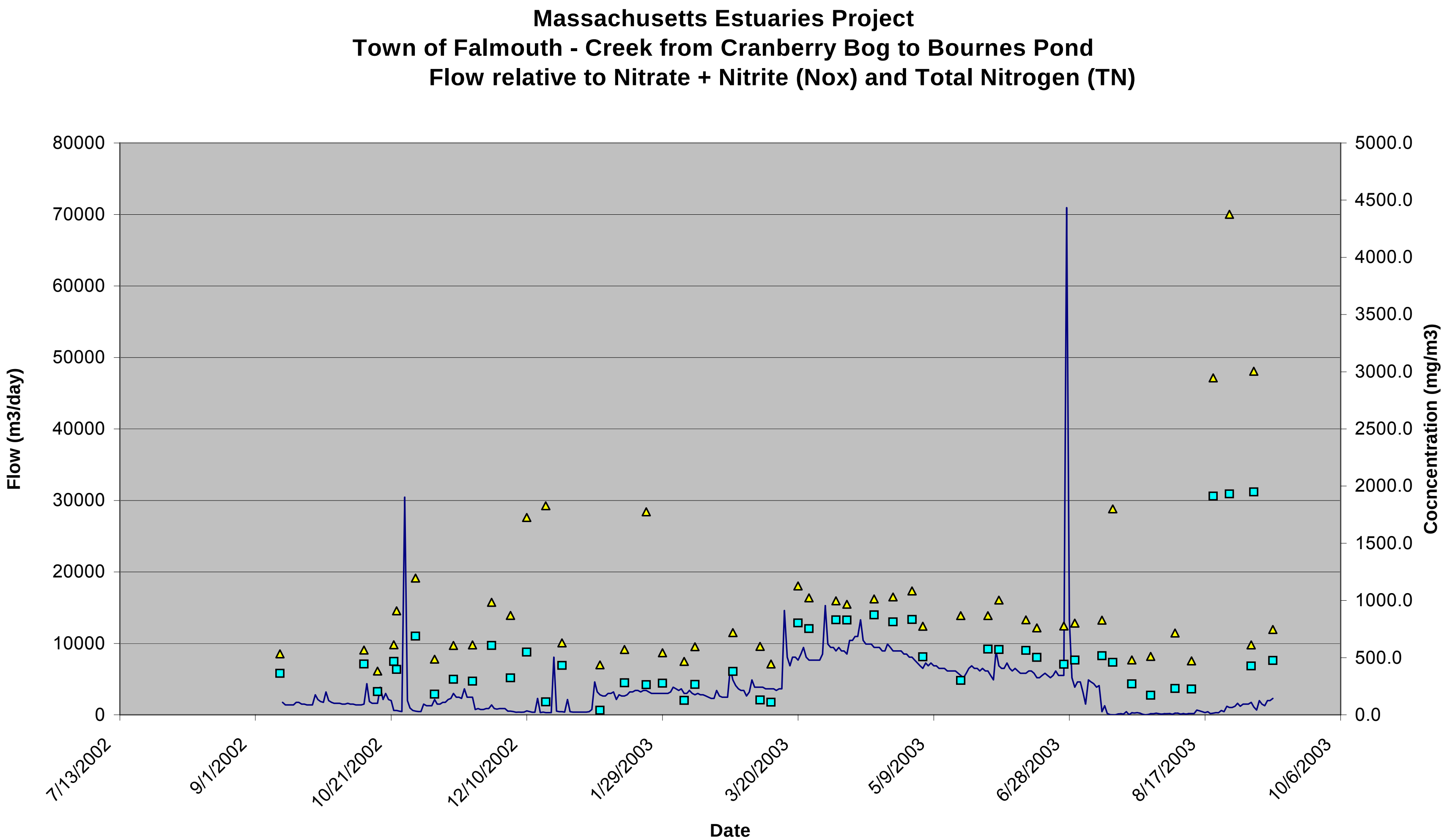


Figure IV-8. Bournes Brook freshwater discharge (solid blue line), nitrate+nitrite (blue boxes) and total nitrogen (yellow triangles) concentrations for determination of annual volumetric discharge and nitrogen load from the upper watershed to the Bournes Pond Estuary (Table IV-6).

Table IV-7. Summary of annual volumetric discharge and nitrogen load from the Coonamessett River discharging to the head of Great Pond, Backus Brook to the head of Green Pond and Bournes Brook to the head of Bournes Pond based upon the data presented in Figures IV-6, 7, 8 and Table IV-6.

System	Period	Discharge (m^3/yr)	Load (kg/yr)	
			NOx	TN
Coonamessett River to Great Pond	9/10/2002 to 9/10/2003	9706546	5481	8259
Backus Brook to Green Pond	9/11/2002 to 9/10/2003	2965314	163	1391
Bournes Brook to Bournes Pond	9/11/2002 to 9/10/2003	1374508	747	1201

From the measured nitrogen load discharged by Bournes Brook to the headwaters of the Bournes Pond estuary and the nitrogen load determined from the watershed based land use analysis, it appears that there is significant nitrogen attenuation of upper watershed derived nitrogen during transport to the estuary. Based upon the measured lower nitrogen load discharged from Bournes Brook (3.29 kg N d^{-1}, 1,201 kg yr^{-1}) compared to that added by the various land-uses to the associated watershed (7.08 kg N d^{-1}, 2,583 kg yr^{-1}), the integrated attenuation in passage through ponds, streams and freshwater wetlands is 54% (i.e. 54% of nitrogen input to watershed does not reach the estuary). The measured level of integrated nitrogen attenuation in the Bournes Brook System (54%) was similar to that measured in the larger adjacent Coonamessett River System (51%), but slightly lower than in the Backus Brook System (67%). The measured level of attenuation in the Bournes Brook System is also greater than the integrated attenuation rate determined from the watershed nitrogen model of 32% (Table IV-4). This is expected given the conservative assumptions of nitrogen attenuation used in the model. The directly measured nitrogen loads from the river was used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

IV.3 BENTHIC REGENERATION OF NITROGEN IN BOTTOM SEDIMENTS

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

IV.3.1 Sediment-Watercolumn Exchange of Nitrogen

As stated in above sections, nitrogen loading and resulting levels within coastal embayments are the critical factors controlling the nutrient related ecological health and habitat quality within a system. Nitrogen enters the Great, Green and Bournes Pond embayments predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the watercolumn (once it entered), 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 basins, particularly if they are deeper than the adjacent embayment (e.g. Perch Pond). 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. 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 Great, Green and Bournes Pond systems, 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 14 sites in Great/Perch Pond, 10 sites in Green Pond and 10 sites in Bournes Pond (Figures IV-9 and IV-10) in July 2002. Measurements of total dissolved nitrogen, nitrate + nitrite and 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 (see Figures IV-9 and IV-10) per incubation were as follows:

Great/Perch Pond Benthic Nutrient Regeneration Cores

- Station GRT-1 1 core (Upper Region of Tributary)
- Station GRT-2 1 core (Upper Region of Tributary)
- Station GRT-3 1 core (Upper Region of Tributary)
- Station GRT-4A/B 2 cores (Upper Region of Tributary)
- Station GRT-5 1 core (Lower Region of Tributary)
- Station GRT-6 1 core (Lower Region of Tributary)
- Station GRT-8 1 core (Perch Pond)
- Station GRT-9 1 core (Perch Pond)
- Station GRT-7 1 core (Main Basin)
- Station GRT-10 1 core (Main Basin)
- Station GRT-11 1 core (Main Basin)
- Station GRT-12/13 2 cores (Main Basin)
- Station GRT-14 1 core (Main Basin)
- Station GRT-15 1 core (Main Basin)

Green Pond Benthic Nutrient Regeneration Cores

- Station GP-1 1 core (Upper Region)
- Station GP-2 1 core (Upper Region)
- Station GP-3 1 core (Upper Region)
- Station GP-4A/B 1 core (Middle Region)
- Station GP-5 1 core (Middle Region)
- Station GP-6 1 core (Middle Region)
- Station GP-7 1 core (Middle Region)
- Station GP-8 1 core (Lower Region)
- Station GP-9 1 core (Lower Region)
- Station GP-10 1 core (Lower Region)

Bournes Pond Benthic Nutrient Regeneration Cores

- Station BP-1 1 core (Upper Region of Tributary)
- Station BP-2 1 core (Upper Region of Tributary)
- Station BP-3 1 core (Upper Region of Tributary)
- Station BP-4 1 core (Upper Region of Tributary)
- Station BP-5 1 core (Upper Region of Tributary)
- Station BP-6 1 core (Upper Region of Tributary)
- Station BP-7 1 core (Israels Cove)
- Station BP-8 1 core (Main Basin)
- Station BP-9 1 core (Main Basin)
- Station BP-10 1 core (Main Basin

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.

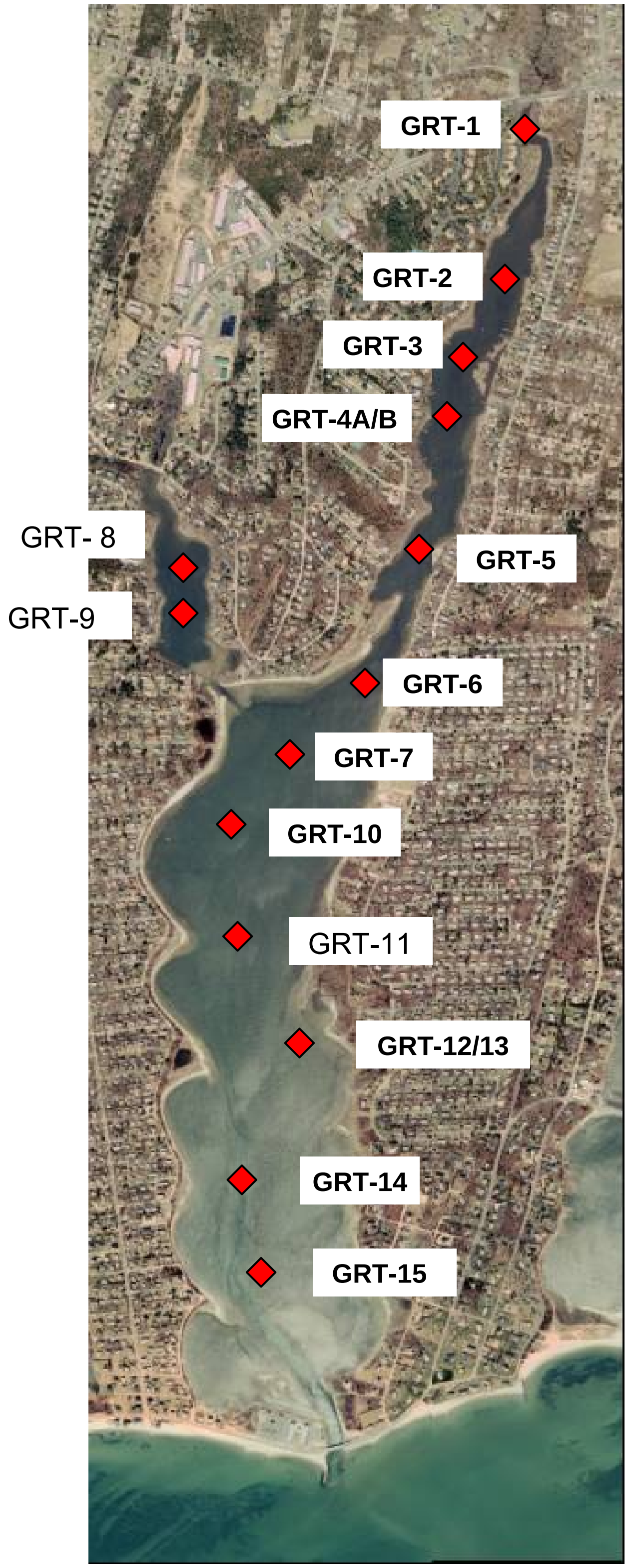


Figure IV-9. Great/Perch Pond embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-8.

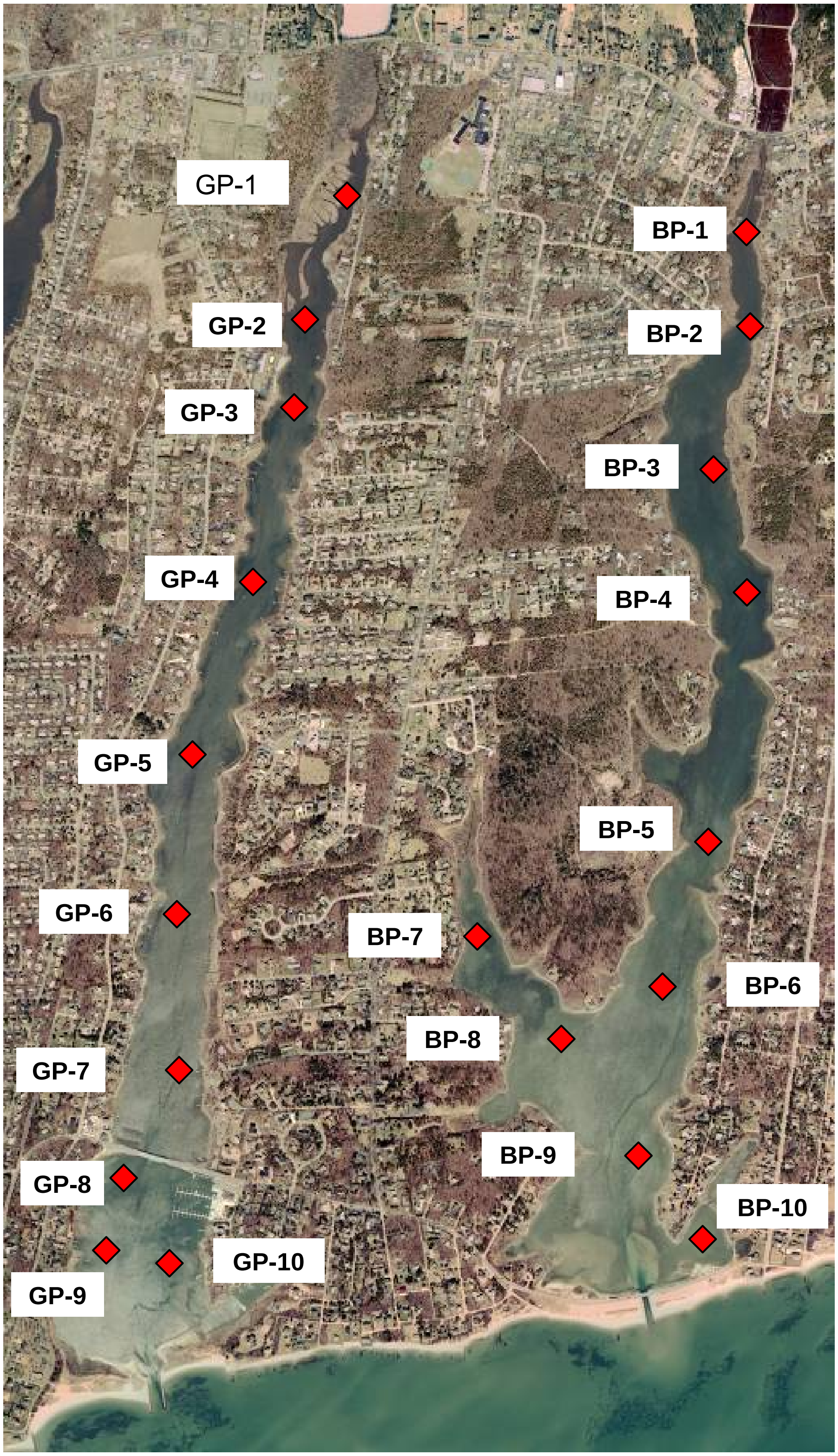


Figure IV-10. Green Pond and Bournes Pond embayment system sediment sampling sites (red symbols) for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-8.

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 Bournes 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.

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. 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-11).

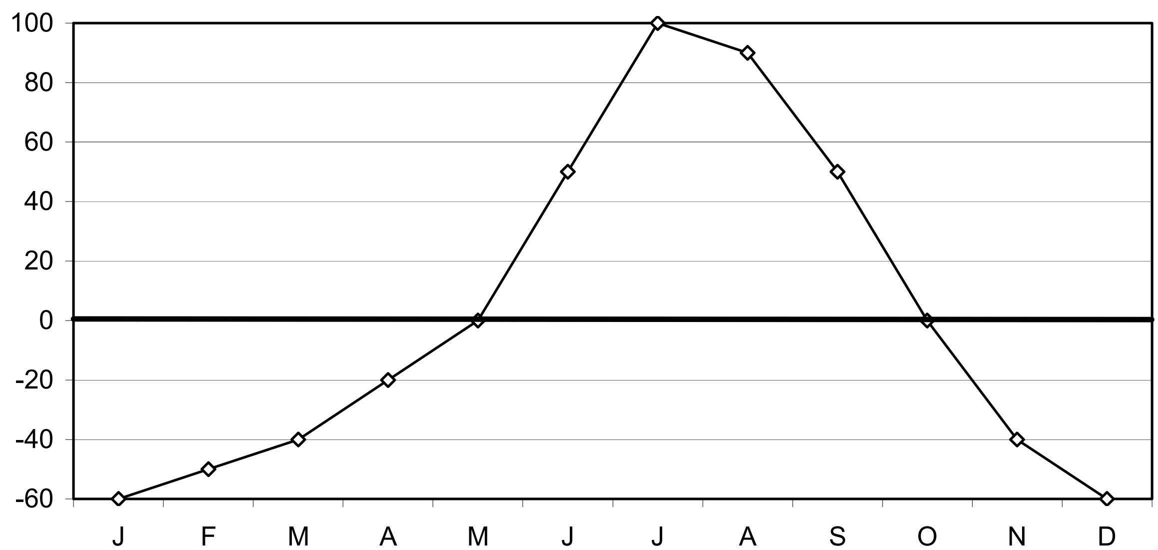


Figure IV-11. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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

In order to determine the net nitrogen flux between 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.

Sediment sampling was conducted within the upper and lower portions of Great, Green and Bournes Pond as well as each of the sub-embayments (Perch Pond and Israels Cove) in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-9 and 10). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. For each core the nitrogen flux rates (described in the section above) were evaluated relative to measured sediment organic carbon and nitrogen content and 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 the sediment carbon turnover (total sediment metabolism) that would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins is driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments.

Net nitrogen release or uptake from the sediments within the Great, Green and Bournes Pond embayment systems for use in the water quality modeling effort (Chapter VI) are presented in Table IV-8. The rates of net sediment nitrogen release were similar among the 3 estuaries. In addition, the smaller tributary systems of Perch Pond and Israels Cove appeared to be small net nitrogen sinks, consistent with the general patterns discussed above. The areas of highest summer chlorophyll a (Table VII-2) and fine-grained sediments tended to support the highest rates of nitrogen release, most likely due to their higher organic deposition rates, hence higher rates of nitrogen recycling. Rates in these systems were similar to those reported for the nearby Vineyard Sound estuary, Popponessett Bay, which ranged from 85 to - 17 mg N m^{-2} d^{-1}.

Table IV-8. Rates of net nitrogen return from sediments to the overlying waters of the Great, Green and Bournes Pond embayment systems. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent July -August rates.

Location	Sediment Nitrogen Flux (mg N m^{-2} d^{-1}) Mean	S.E.	N	Station
Great/Perch Pond Estuary				
Upper Tributary	100.7	28.9	4	1-4
Lower Tributary	56.5	13.7	2	5,6
Perch Pond	-20.2	6.2	2	8,9
Main Basin	-16.4	21.8	6	7-15
Green Pond Estuary				
Upper	12.9	5.3	3	1,2,3
Middle	54.5	7.9	4	4,5,6,7
Lower	30.5	26.5	3	8,9,10
Bournes Pond Estuary				
Upper Tributary	51.5	15.6	4	1,2,3,4
Lower Tributary	29.3	15.9	2	5,6
Israels Cove	-14.1	4.2	5	7
Main Basin	57.3	39.6	3	8,9,10

Station numbers refer to Figures IV-9 and IV-10.

V. HYDRODYNAMIC MODELING

V.1 INTRODUCTION

The coastal ponds within the greater Ashumet Valley watershed include Great, Green, and Bournes Ponds (Figure V-1). These estuaries include several sub-embayments that provide important recreational and environmental resources for the local community. Similar to many other Cape Cod estuaries, the quality of water in the system has become a concern. Increased nutrient (typically nitrogen) loading from leaching septic systems and use of fertilizers, as well as other sources of pollution contribute to water quality degradation.

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

Estuarine water quality is dependent upon the nutrient and pollutant loading and the processes which help flush nutrients and pollutants from the estuary (e.g., tides and biological processes). Relatively low nutrient and pollutant loading and efficient tidal flushing are indicators of high water quality. The ability of an estuary to flush nutrients and pollutants is proportional to the volume of water exchanged with a high quality water body (i.e. Nantucket Sound). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For Great, Green, and Bournes Ponds, the most important parameters are:

- Tide range
- Inlet configuration
- Estuary size, shape, and depth
- Longshore transport of sediment

The south shore of Falmouth exhibits a relatively small tide range, with a range of only about 1.5 ft. Since the water elevation difference between Nantucket Sound and each of the Ponds is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) through the Great Pond and Green Pond inlets is negligible indicating "well-flushed" systems. In contrast, the large relative size of Bournes Pond in comparison to the cross-section of its inlet is indicative of a "restrictive" system, where tidal flow and the associated flushing are inhibited. Based on the tidal characteristics alone, this might indicate that Great and Green Ponds are "healthier" than Bournes Pond; however, land development in the Great Pond and Green Pond watersheds likely provides a substantially higher nutrient load to these systems. Consequently, estuarine water quality may be more dependent on nitrogen loading than tidal characteristics for these three Ponds.

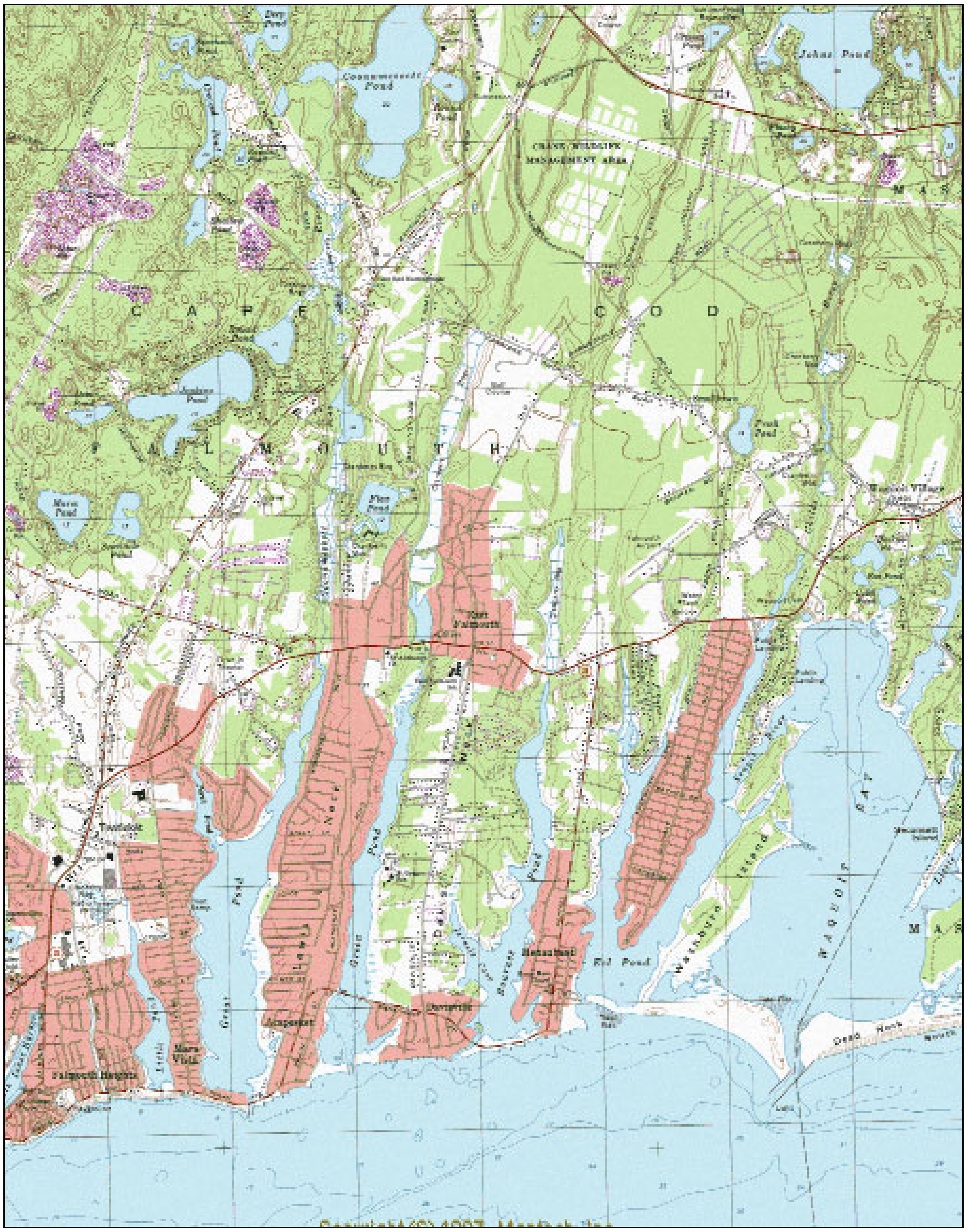


Figure V-1. Map of Greater Ashumet Valley watershed (from United States Geological Survey topographic map).

In addition to a small tide range, the length and width of these Ponds also influence tidal flushing characteristics. Since the three Ponds are relatively narrow (Falmouth's south coast Ponds are often referred to as "finger ponds"), with inlets at the south ends and small streams at the north ends, the tidal exchange with Nantucket Sound decreases with distance inland from the inlet. For example, water in lower Green Pond (south of the causeway) is exchanged or flushed with Nantucket Sound water relatively rapidly; however, water in the northern section of Green Pond requires a significantly longer time period to exchange with Nantucket Sound. A quantitative analysis of flushing times is provided in Section V.5.

On the south shore of Falmouth, inlets to estuaries often shoal due to littoral drift of sediment across the inlet mouth; therefore, tidal flushing may be limited by the inlet geometry. Dredging of all three Pond entrances was performed during the Spring of 1999 (after the bathymetry and tidal data were collected for this study); therefore, damping of the Nantucket Sound tide signal through each inlet should be minimized. However, as the littoral drift reduces the inlet cross-sections in the future, tidal exchange again can be reduced.

This section summarizes the development of hydrodynamic models for: Great Pond, Green Pond, and Bournes Pond. For each Pond, the calibrated model provides an understanding of water movement through the estuary. Tidal flushing information will be utilized as the basis for a quantitative evaluation of water quality. Nutrient loading data combined with measured environmental parameters within the various sub-embayments provide the basis for an advanced water quality model (see Ramsey et al. (1995) for an example). This type of model will provide a tool for evaluating existing estuarine water quality, as well as determine the influence of various methods for improving overall estuarine health.

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

To calibrate the hydrodynamic model, field measurements of water elevations and bathymetry were required. Tide data was acquired within Nantucket Sound (two gauges were installed between Green and Bournes Ponds), lower Great Pond, upper Great Pond, Perch Pond, lower Green Pond, upper Green Pond, lower Bournes Pond, and upper Bournes Pond. All nine (9) temperature-depth recorders (TDRs) were installed for a 30-day period to measure tidal variations through an entire neap-spring cycle. In this manner, attenuation of the tidal signal between Nantucket Sound and the various sub-embayments was evaluated accurately.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE SYSTEM

The southern coast of Cape Cod in the vicinity of Great, Green, and Bournes Ponds is a relatively quiescent region. 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 years with a significant number of easterly storm events, the net longshore sand transport direction remains unclear. However, it appears that the overall long-term trend is 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). 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 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 Great, Green, and Bournes Ponds inlets, inlet migration is less of a concern than other areas of Cape Cod. According to FitzGerald (1993), inlets to Great, Green, and Bournes Ponds 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 periodically are 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 inlets to Great, Green, and Bournes Ponds are stable, with some observed historic migration of each inlet, as well as infrequent closures prior to placement of jetties at each of the entrances. Figure V-2 shows the position of the inlets in 1893, prior to construction of jetties in this region. Jetties were added to Great Pond after the 1938 Hurricane, to Green Pond in the mid-1950s, and Bournes Pond in 1985. Since the addition of jetties, the cross-section of each inlet has remained relatively stable, allowing for effective tidal circulation through the flow constrictions at the entrances (as well as navigation in Great and Green Ponds). The inlet stability afforded by the jetty systems prevents infrequent inlet closures that can cause large ecological shifts to estuarine plant and animal communities.

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 entrances to Great, Green, and Bournes Ponds. 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 each of the three inlets also is relatively small. Since the construction of the jetty systems at each of the entrances, the inlets have generally reached equilibrium, where the tidal velocities through the main channels are sufficient to prevent significant shoaling. Recent annual dredging of these inlets has only been on the order of 1,000 cubic yards per year (personal communication with Barnstable County Dredge personnel, 2004).

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 Great, Green, and Bournes Ponds region, these wooden and stone structures were constructed to protect Menauhant Road and waterfront dwellings (Figures V-3 and V-4). The remnants of wooden groins and bulkheads can be found along much of Falmouth's south coast (Figure V-5).

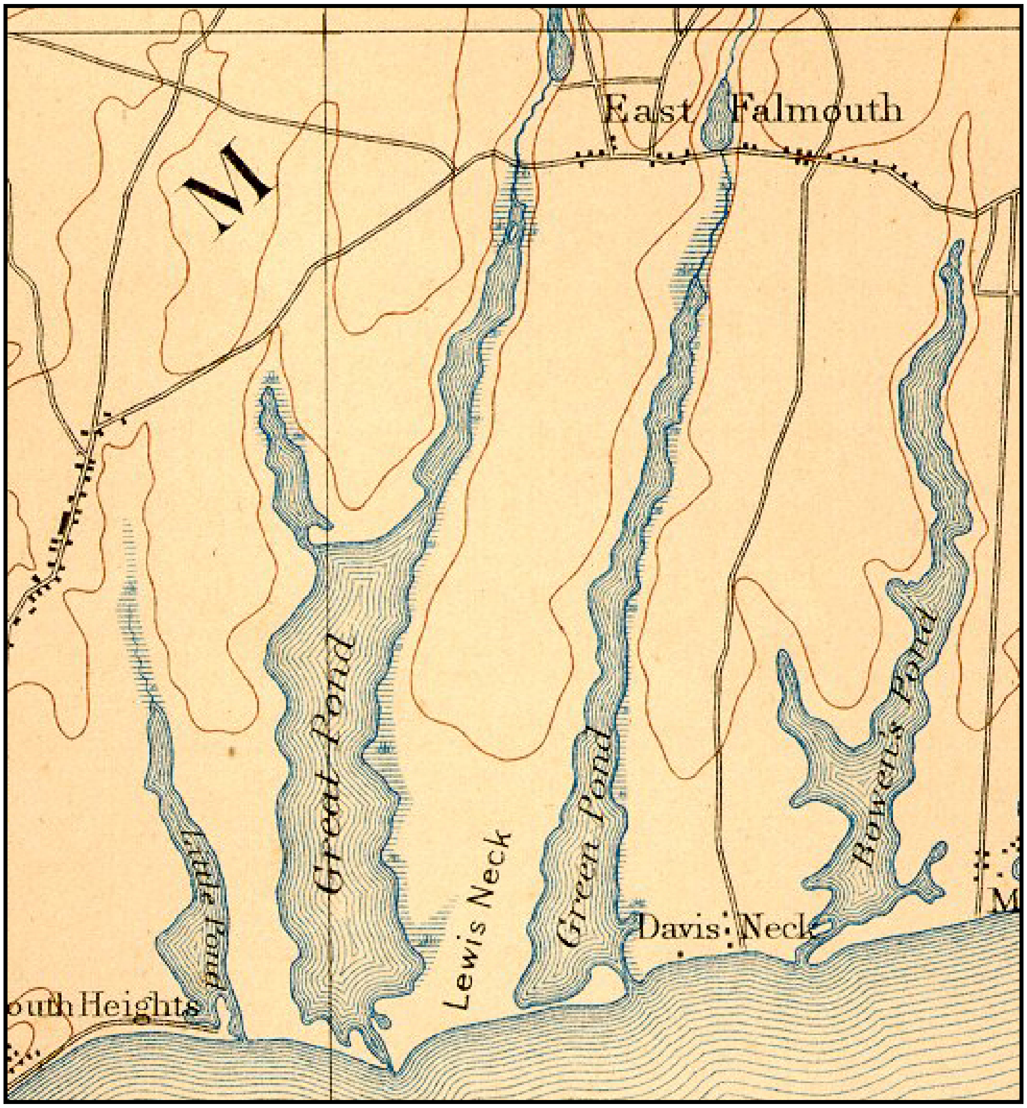


Figure V-2. A portion of the U.S.G.S. 1893 map showing Great, Green, and Bowen's (Bournes) Ponds. This map depicts the condition of these inlets prior to the installation of jetties.

As shown in Figures V-3, V-4, and V-6, 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 Green 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-3. Photograph of the Wellsmere Inn in Maravista immediately after the 1944 Hurricane. Note the wood bulkheads and concrete seawall utilized to armor the shoreline.

ROAD BED at Maravista, Falmouth.

Figure V-4. Photograph of Menauhant Road in Maravista immediately after the 1944 Hurricane. Note the stone revetment armoring the roadway.

Figure V-5. 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.

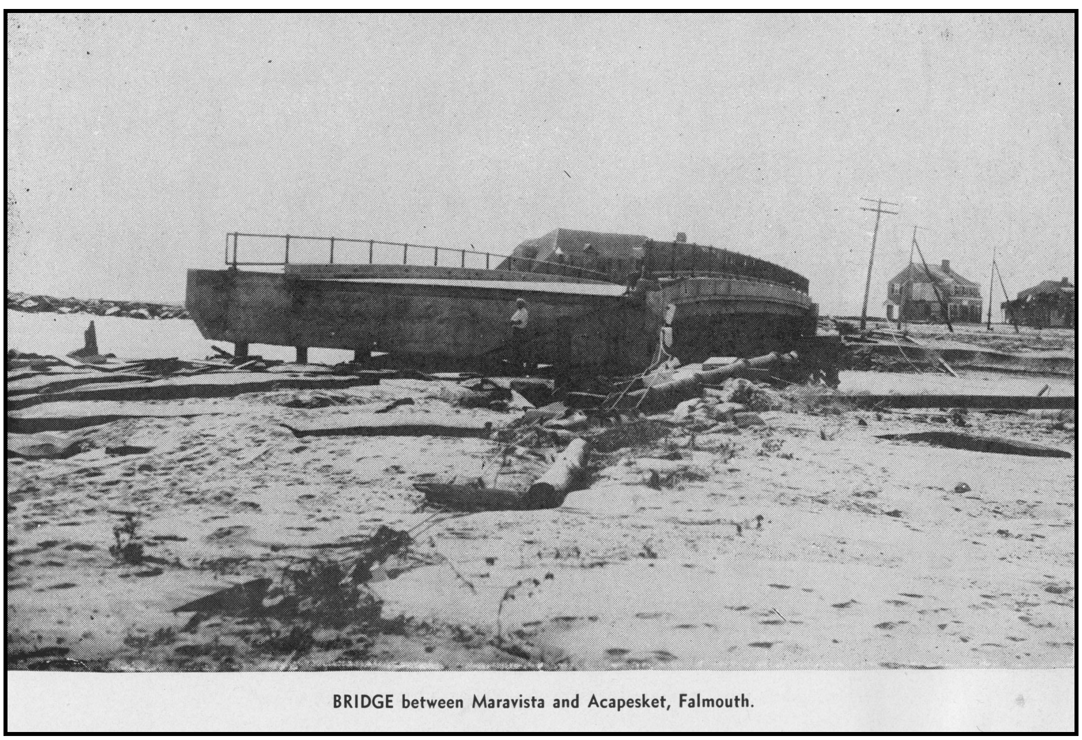


Figure V-6. Photograph of the old Great Pond bridge immediately after the 1944 Hurricane. The photograph shows that storm overwash eroded the roadway and approach ramps to the bridge. Note the Great Pond jetty at the left side of the photograph.

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, 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 (Theiler et al., 2001) to incorporate more recent shoreline information. Specifically, the updated Massachusetts Shoreline Change Project included a 1994 shoreline developed from orthophotos. Along much of the south coast of Falmouth, the three most recent shorelines available from the MCZM dataset are the 1938, 1975, and 1994 shorelines. 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. 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 inlets to Great, Green, and Bournes Ponds, 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. This finding in the CRWG report appears to contradict 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 (1990) 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 shoreline change data set utilized by the CRWG (the two most recent shoreline available from MCZM shoreline change information). A review of shoreline data indicates that this was not the case; therefore, the statewide MCZM data set likely does not provide the necessary accuracy to evaluate recent shoreline change along Falmouth's south coast for the purpose of developing a long-term coastal management plan.

To improve the Town's ability to manage their coastal resources, some of the other shortcomings in the CRWG analysis are presented below. All of these problems could potentially lead to misinterpretation of regional coastal processes and improper decisions regarding long-term coastal management include:

- The CRWG 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 (Theiler, 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 in the CRWG report (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 and Bournes Ponds, 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 to evaluate 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 rate shoreline change rate, a review of the existing shoreline data sets was performed. As part of the review process, recent imagery was downloaded from the MassGIS website and these readily available aerial photographs were compared to asses the horizontal control. In addition, the interpreted shoreline data were provided by 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 differential GPS was utilized to locate a series of common features visible on both orthophoto sets. 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 (see Anders and Byrnes, 1991 or Crowell, et al., 1991 for more information).

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, it appears very difficult to select a high water shoreline from the 1994 imagery, primarily because the orthophotos appear overexposed. The 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 overlayed on both the 1994 and 2001 orthophotos, it appears that incorrect identification of the high water line from the 1994 photographs caused an over-prediction of recent shoreline erosion rates. This conclusion is further supported by a comparison of the 1994 shoreline interpreted from the orthophotos and a 2004 shoreline determined using a differential GPS (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, it appears that the 1994 shoreline likely contains interpretation errors and is inappropriate to use for shoreline change analyses.

After excluding the 1994 shoreline, three outer coast shoreline surveys were available for quantifying historical shoreline change between Falmouth Harbor and Menauhant Beach during the time period from 1938 to 2004. Available data layers for this time period include 1938, 1975, and 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 MCZM Office, as part of the Massachusetts Shoreline Change project. Digital shoreline data for 1975 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-7. This shoreline was added to the data set because of the concerns associated with the existing 1994 shoreline.

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 information, in combination with a review of the digital data set required that the data be excluded from the analysis. As such, the overall time period (1938 to 2004) was used to represent shoreline change conditions for this study. 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.

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-7. 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.	

Shoreline change was evaluated for this study during the time period from 1938 to 2004. Change calculations 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-8 and V-9, 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 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 may not be representative of processes affecting the remainder of the coast. By excluding these higher erosion areas from the dataset, a data distribution that represents the majority (~86%) of the shoreline was developed to determine typical shoreline evolution over the past 66 years. The data distribution generated by excluding this analysis is shown in figure V-10. The average change rate representing the shoreline excluding the erosion “hot-spots” is about -0.055 ft/yr. Therefore, with the exception of the “hot spots”, 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.

Figure V-8. The 2001 aerial photograph showing scaled transects that indicate computed shoreline change rates between 1938 and 2004.

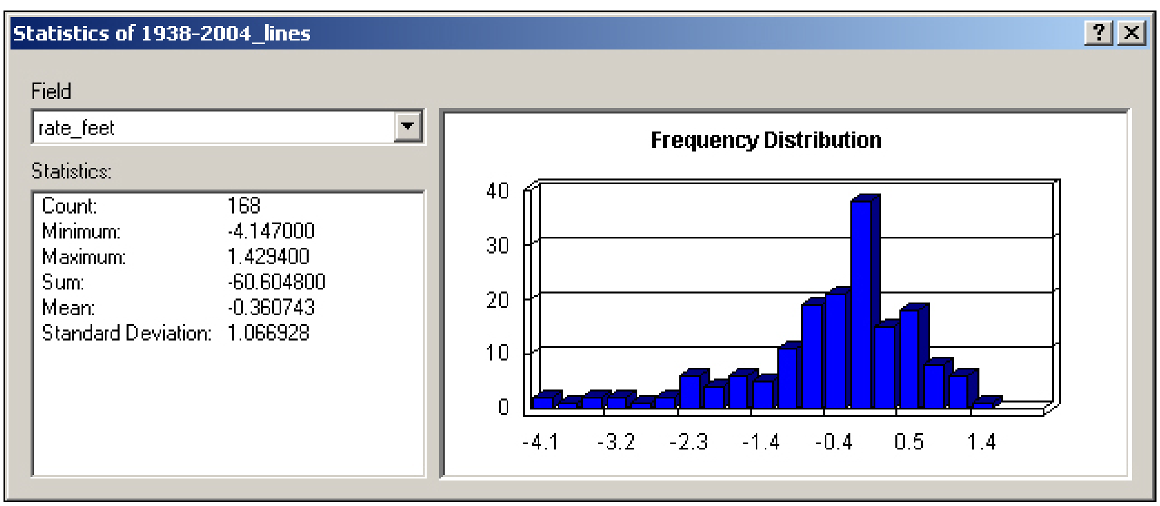


Figure V-9. Shoreline change data distribution calculated along the south coast of Falmouth (Falmouth Harbor to Menauhant Beach) between 1938 and 2004.

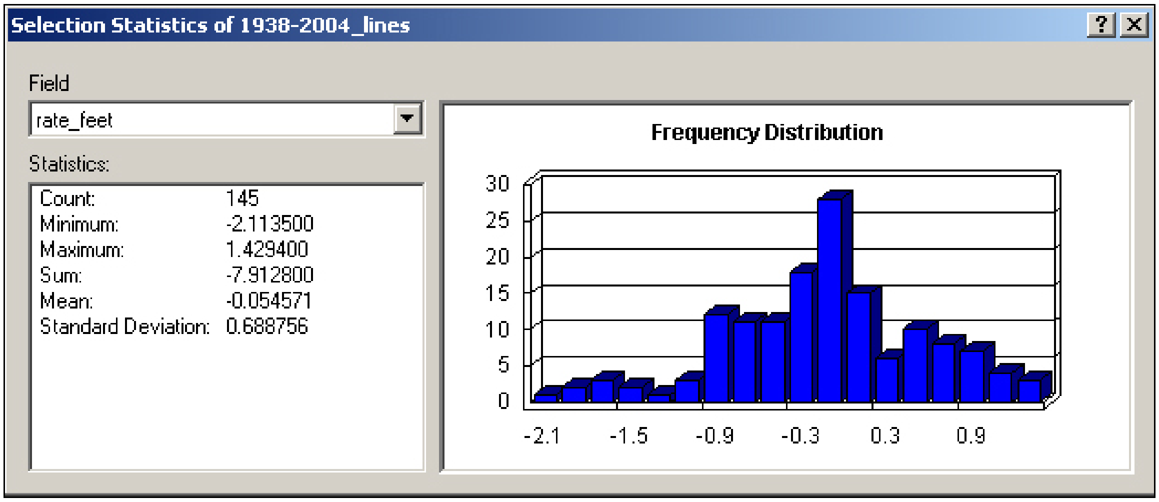


Figure V-10. Shoreline change data distribution calculated along the south coast of Falmouth (Falmouth Harbor to Menauhant Beach) between 1938 and 2004, excluding areas identified as erosion "hot-spots".

V.2.3 Inlet Management Implications

For the tidal inlets of Great, Green, and Bournes Ponds, the influence of shoreline change and the related longshore sediment transport rates directly influence the stability of the existing inlet systems. The "hot spot" erosion in the vicinity of both Green Pond and Bournes Pond likely represents adjustment of the shoreline following placement of the jetties. According to the shoreline change analysis, very little erosion has occurred to the east of Great Pond inlet. However, the size of these jetties and the orientation of the shoreline at this location indicate that downdrift impacts as a result of jetty construction would be expected. Due to the armoring of the roadway east of the inlet, shoreline retreat has been limited in this region. For Green Pond, placement of the jetty system occurred in the mid-1950s. Erosion of the updrift (west) barrier beach actually has been higher than the downdrift (east) beach, possibly due to loss of the updrift sediment supply caused by construction of the Great Pond jetties. Although structures existed for a previous inlet to Bournes Pond (west of the existing inlet), the existing jetty configuration was not completed until 1985. The previous inlet was located where the remnant road bridge still exists to the south of Menauhant Road.

Current management practices at Great, Green, and Bournes Ponds inlets consist of periodic dredging to maintain the existing channels. Recently, these inlets have been dredged on an annual basis; however, the impounded sediment volumes are small (generally less than 1,000 cubic yards). The sand dredged from each of these inlets has been placed to beaches west of each inlet. This placement location represents the most highly eroded areas adjacent to each inlet; however, passing inlet sediment to downdrift shorelines (beaches to the east of each inlet) would supply these areas with needed littoral sediments as well.

Due to the relatively minor erosion rates along much of Falmouth's south coast, only the regions adjacent to the three major tidal inlets (Great, Green, and Bournes Ponds) likely require the addition of beach nourishment to stabilize the shoreline. Projects such as the dune restoration planned for the shoreline immediately west of Bournes Pond inlet will continue to provide needed sediments to the littoral system. In addition, beach nourishment appears to be an effective means of shoreline stabilization in this region, due to the relatively small longshore transport rates. As an example, 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, 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 between 1938 and 2004. If designed properly, both dune restoration and beach nourishment projects can be constructed in a manner that will not affect dredging frequency and/or stability of the existing tidal inlets to Great, Green, and Bournes Ponds.

At the present time, the inlets to both Great and Green Ponds provide for safe navigation as well as efficient tidal circulation. Although Bournes Pond shows some signs of tidal attenuation through the narrow inlet channel, it appears that this entrance also allows relatively efficient tidal circulation. Any proposed alterations to the entrances of Great, Green, and/or Bournes Ponds should ensure that tidal flushing is not negatively impacted (i.e. there is no reduction in the inlet cross-sectional area).

V.3 FIELD DATA COLLECTION AND ANALYSIS

A requirement for the numerical model generation is precise descriptions of embayment geometry as well as hydrodynamic forcing processes. To this end, the bathymetry of the embayments and water elevation variations were measured. The bathymetry of the Great Pond system (including Perch Pond), Green Pond, and Bournes Pond were surveyed. The resulting depth measurements were used to create computational grids of each pond. Figure V-11 shows the depth contours of these grids, for the three ponds. In addition to the bathymetry surveys, tide gauges were installed at selected locations in Nantucket Sound and within each system to observe the rise and fall of water surfaces. These data were processed to provide input information required for the numerical model and, in addition, analyzed to provide insight into existing hydrodynamic conditions for each system.

This section will demonstrate that these Ponds, open to Nantucket Sound, have different hydrodynamic characteristics that are dependent primarily upon the geometry of each Pond's inlet. The hydrodynamic character of each Pond is not directly correlated to water quality issues; rather, Bournes Pond, which has the greatest tide attenuation of the three ponds, seems to enjoy relatively good water quality whereas Green Pond, with the least tide attenuation of the three Ponds considered, suffers from poor water quality. These results suggest that nitrogen loading into each system, not hydrodynamic characteristics, is the primary indicator of water quality, and that nitrogen loading to Green Pond, for example, is far in excess of the Pond's natural ability to flush such pollutants from its system.

V.3.1 Bathymetry

Bathymetry, or depth, of each Pond was measured during a series of field surveys in late February, 1999. The surveys were 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 position measurements accurate to approximately 1-3 feet. Digital data output from both the echosounder and GPS were logged to a laptop computer, which integrated the data to produce multiple data sets consisting of water depth as a function of geographic position (latitude/longitude). The surveys were performed within each Pond to develop plan view contour maps of Pond depth.

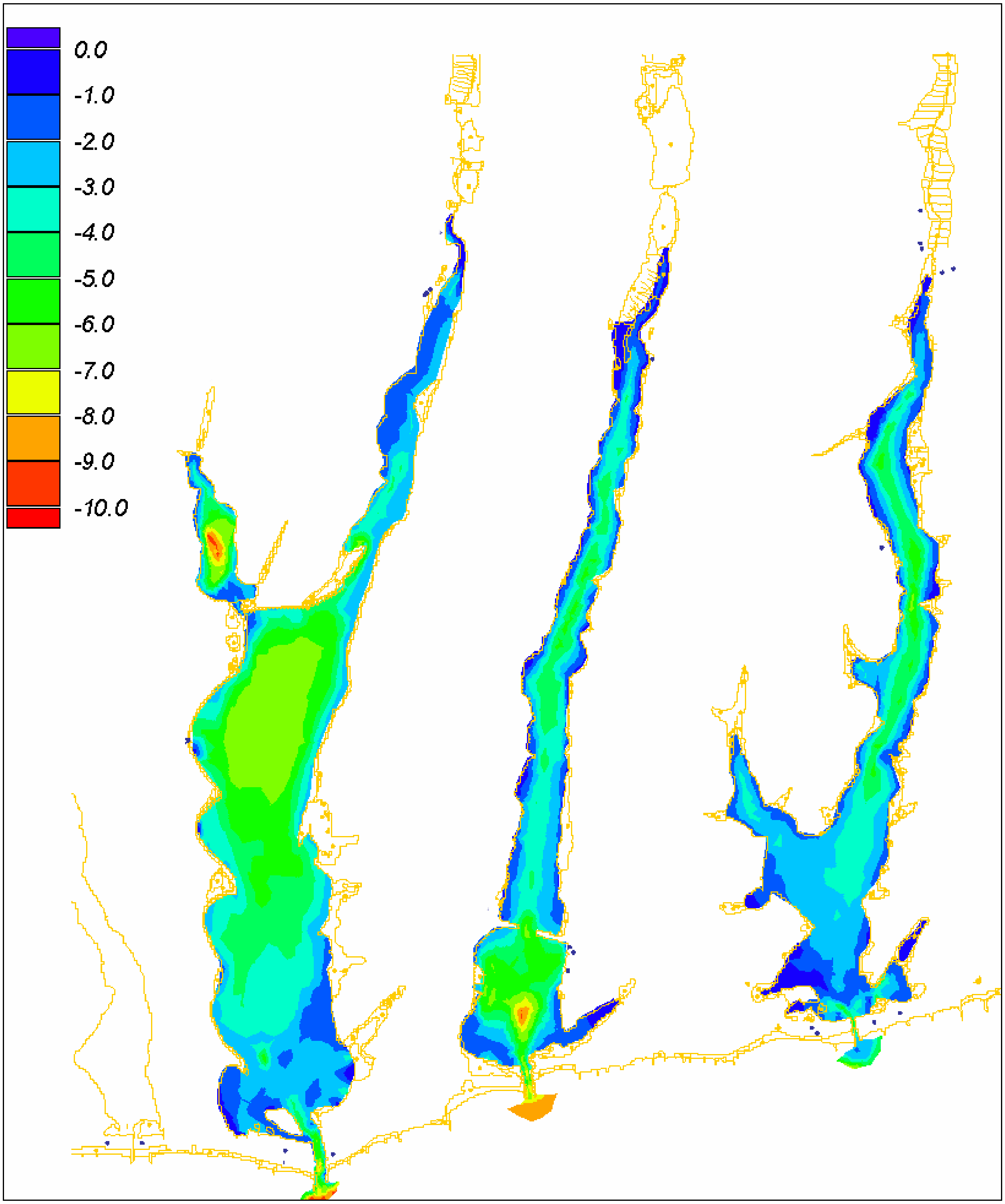


Figure V-11. Depth contour plots of the numerical grid for the hydrodynamic model at 0.5-foot contour intervals relative to NGVD29.

These data files were merged with water surface elevation measurements to correct the measured depths to the NGVD 1929 vertical datum. Once corrected, the data were then merged into larger 'xyz' files containing x-y horizontal position (in Massachusetts State Plan

1927 coordinates) and vertical elevation of the bottom (z) relative to NGVD29. These xyz files were then input to mapping software to calculate depth contours for each Pond.

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) to record changes in water pressure. These water surface variations can be due to tides, wind set-up, or other low-frequency oscillations of the sea. The tide gauges were installed in nine (9) locations throughout the study area (see Figure V-12) in mid-January 1999 and removed in late February, 1999. Data records span at least 28 days, an adequate time period to resolve the primary tidal constituents.

The tide gauges used for the study consisted of Brancker TG-205, Coastal Leasing Microtide, and Global Water WL-14 instruments. Data sampling was set for 10-minute intervals, with each 10-minute observation resulting from an average of 16 1-second pressure measurements. Each of these instruments use strain gauge transducers to sense variations in pressure, with resolutions on the order of 1 cm head of water. Each gauge was calibrated prior to installation to assure accuracy. Each gauge returned 100% of the desired data.

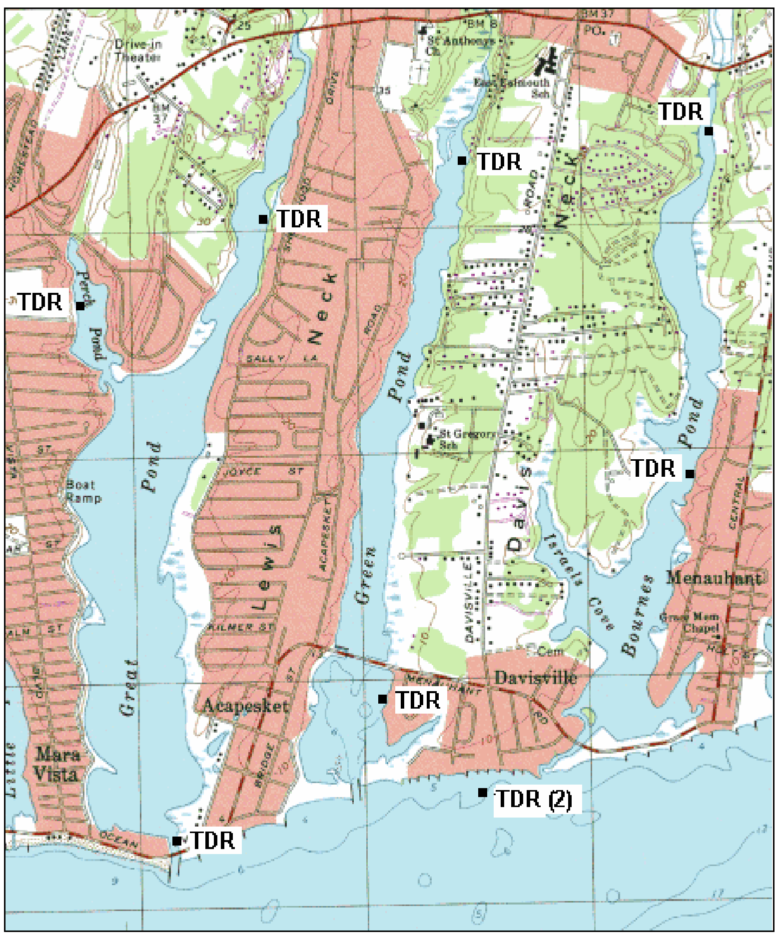


Figure V-12. Map of the study region identifying locations of the tide gauges used to measure water level variations throughout the system. Nine (9) gauges were deployed for one month between January and February, 1999. Each black square represents the approximate locations of the tide gauges.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric pressure readings were obtained from the NOAA station in Buzzards Bay, interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in variations 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). Several sensors had been 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 record representing the variations in water surface elevation relative to the NGVD 1929 vertical datum. Figure V-13 presents the time variation of water level in each of the ponds. Plots of all tide gauges are presented in Figures V-13 through V-15.

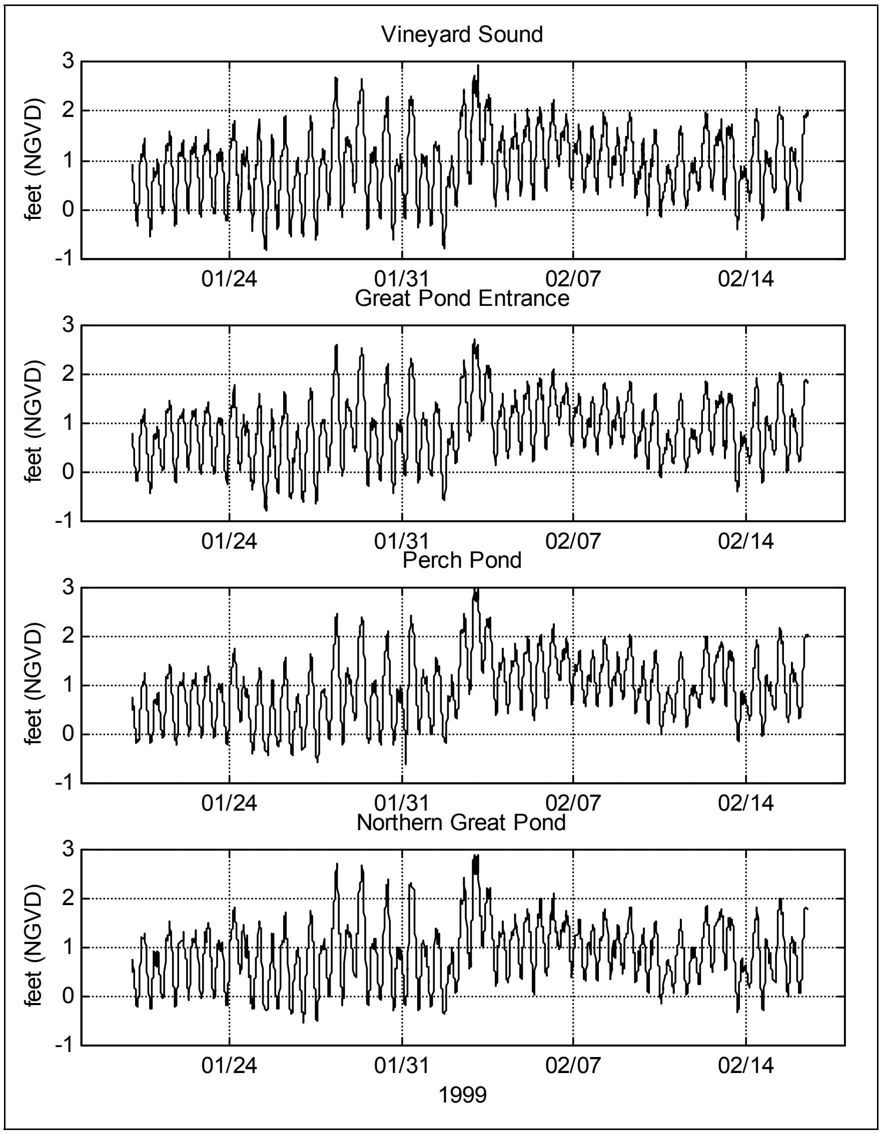


Figure V-13. Water elevation variations as measured at the four locations within the Great Pond.

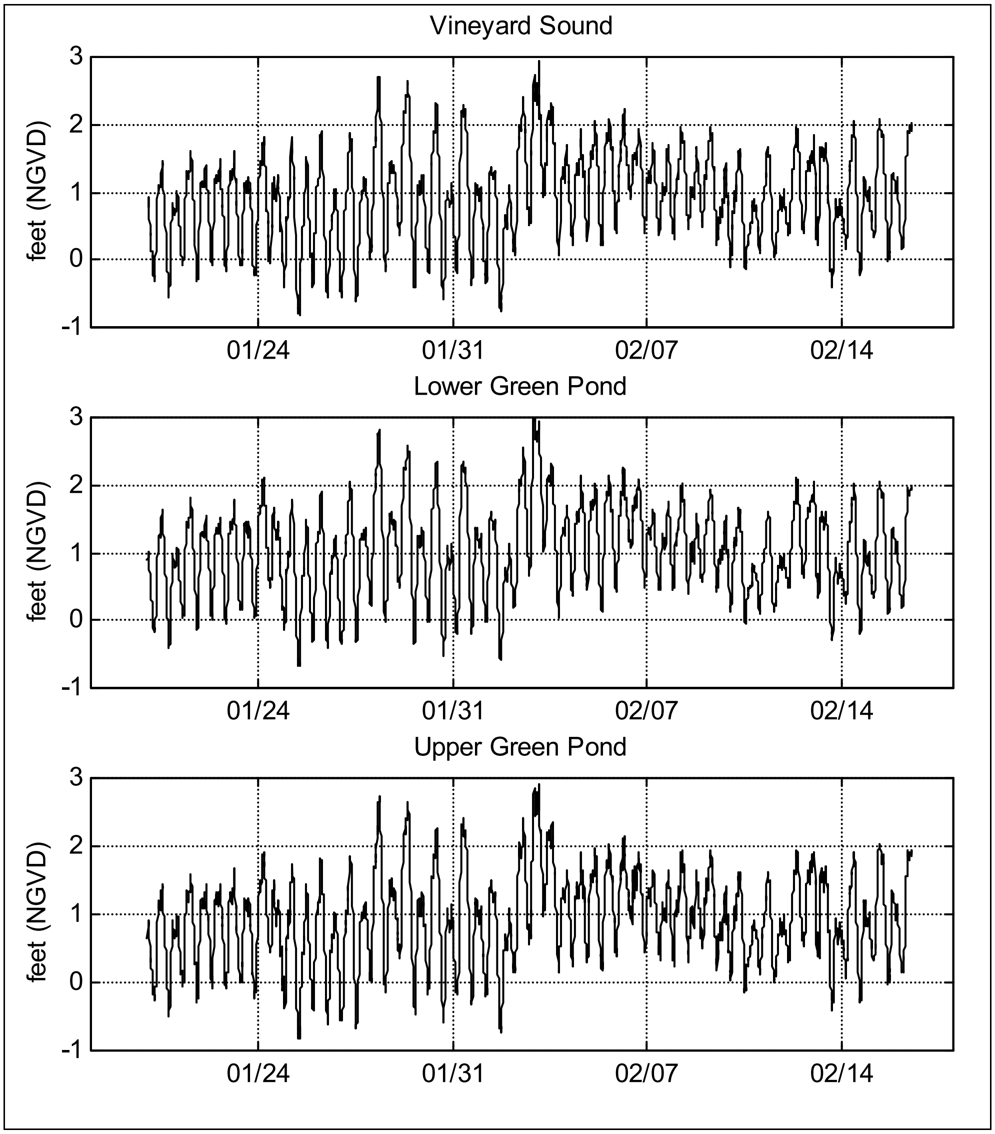


Figure V-14. Water elevation variations as measured at the four locations within the Green Pond.

V.3.2 Field Data Analysis

Analyses of the tide and bathymetric data provided insight into the hydrodynamic characteristics of each 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 each 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 each system. The results of these analyses will be discussed in this section.

The geometry of each system is similar, as the three 'finger' ponds were formed possibly as outwash channels from the retreating Laurentian ice sheet during the Wisconsinan glacial stage (approximately 18,000 years ago). Several theories on the origin of the Falmouth finger ponds exist (Oldale, 1992, Cape Cod and the Islands: The Geologic Story), despite the lack of agreement on their formation, it is evident from this recent bathymetry that the Ponds share many common features.

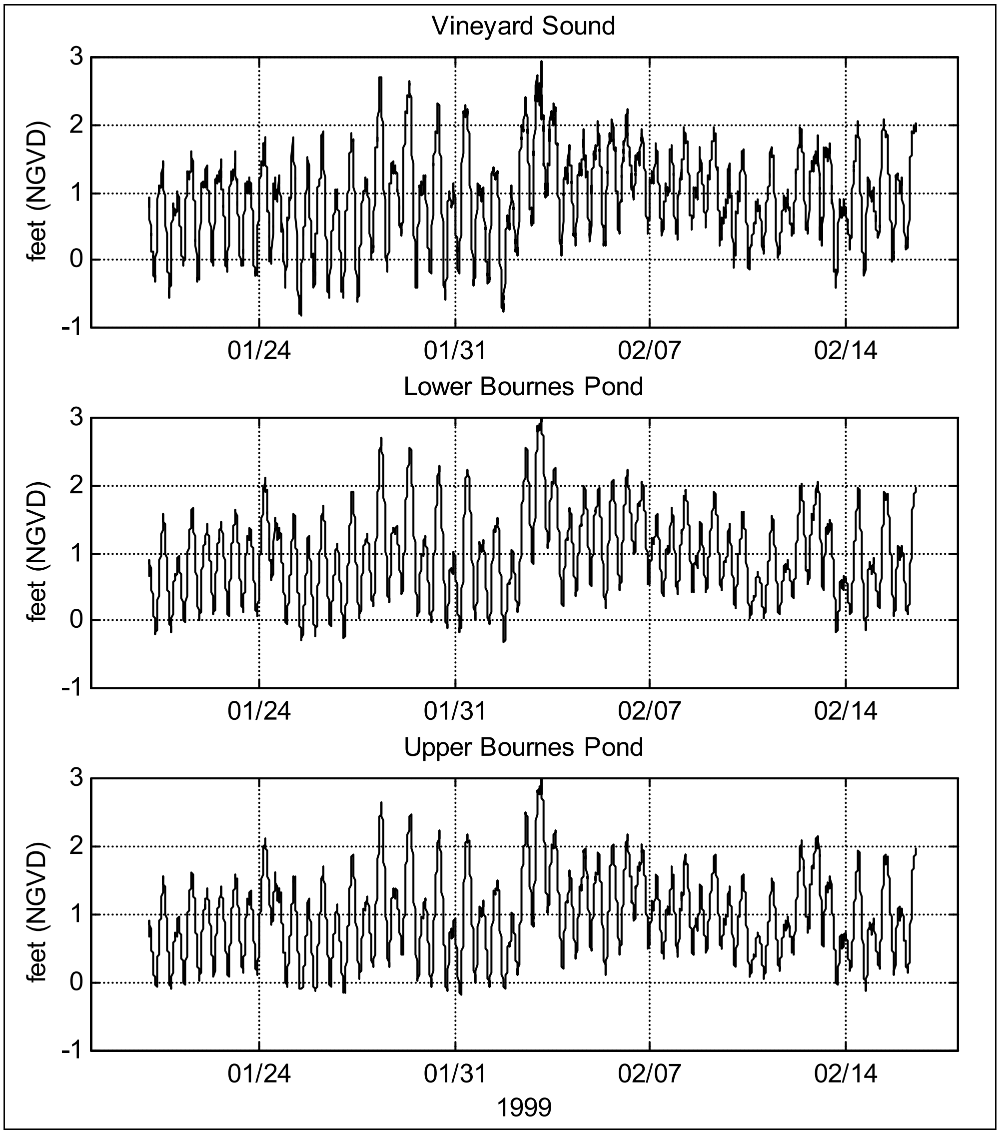


Figure V-15. Water elevation variations as measured at the four locations within the Bournes Pond.

The ponds are oriented north-south, and open to Nantucket Sound via inlets. These inlets are affected significantly by longshore sand transport (west to east), where shoaling can impede hydrodynamic exchange at each mouth. All inlets are armored with jetties, with each featuring significant scour channels between these structures. The ponds are long and narrow, with length-to-breadth ratios of approximately 15-to-1. Green Pond, for example, is nearly two miles in length. Depths within the deeper scour channels at each inlet are approximately 8 feet, with the upper (northern) reaches of the Pond frequently less than 5 feet deep. Deeper areas in Perch Pond are a result of a geologic feature known as a “kettle pond”.

Harmonic analyses were performed on the time series from each gauge location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 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 0.68 feet in Nantucket Sound. The range of the M_2 tide is twice the amplitude, or 1.36 feet. The diurnal tides, K_1 and O_1, possess amplitudes of approximately 0.2 feet. The N_2 tide, also of semi-diurnal period, rivals the diurnal constituents with an amplitude of 0.20 feet. The M_4 tide, a higher frequency harmonic of the M_2 lunar tide, results from frictional dissipation of the M_2 tide in shallow water. The M_4 is significant in Nantucket Sound, and is responsible for the unusual 'double high' tide signature local to the Falmouth shore. This M_4 constituent (0.18 feet) is approximately one-third the amplitude of the M_2.

Table V-2 also shows how the constituents vary as the tide propagates into the estuaries. Note the reduction in the M2 amplitude between Nantucket Sound and the upper reaches of each pond (Upper Green Pond, Perch Pond, and Upper Bournes Pond). The loss of amplitude with distance from the inlet describes tidal attenuation. Frictional mechanisms dissipate energy, resulting with a reduction in energy (or height). Note the relatively greater head loss in lower Bournes Pond versus lower Green Pond or the Great Pond mouth (Figure V-16 and V-17).

Table V-2. Tidal Constituents for Falmouth Ponds 1999.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	Msf
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Nantucket Sound	0.68	0.18	0.07	0.12	0.20	0.23	0.19	0.11
Lower Green Pond	0.67	0.19	0.07	0.09	0.20	0.22	0.17	0.11
Upper Green Pond	0.66	0.18	0.08	0.12	0.20	0.23	0.17	0.12
Great Pond Mouth	0.65	0.13	0.05	0.12	0.19	0.23	0.17	0.14
North Great Pond	0.64	0.13	0.06	0.12	0.19	0.22	0.17	0.11
Perch Pond	0.62	0.11	0.06	0.11	0.17	0.22	0.18	0.18
Lower Bournes Pond	0.63	0.09	0.05	0.07	0.14	0.22	0.18	0.12
Upper Bournes Pond	0.60	0.09	0.05	0.07	0.14	0.21	0.17	0.09

Standard tide datums were computed from the 28-day records. These datums are presented in Table V-3. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available, however, these datums still provide a useful comparison of tidal dynamics within the system. Mean Higher High (MHH) and Mean Lower Low (MLL) levels represent the mean of the daily highest and lowest water levels. Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. Mean Tide Level (MTL) is simply the mean of MHW and MLW. The most significant effects of tide attenuation in the three Ashumet Valley salt ponds are apparent in Bournes Pond and Perch Pond. For these embayments, attenuation is evident by a 0.1 ft increase in elevation of the mean tide elevation, as well as a 0.2 ft increase in the MLLW elevations.

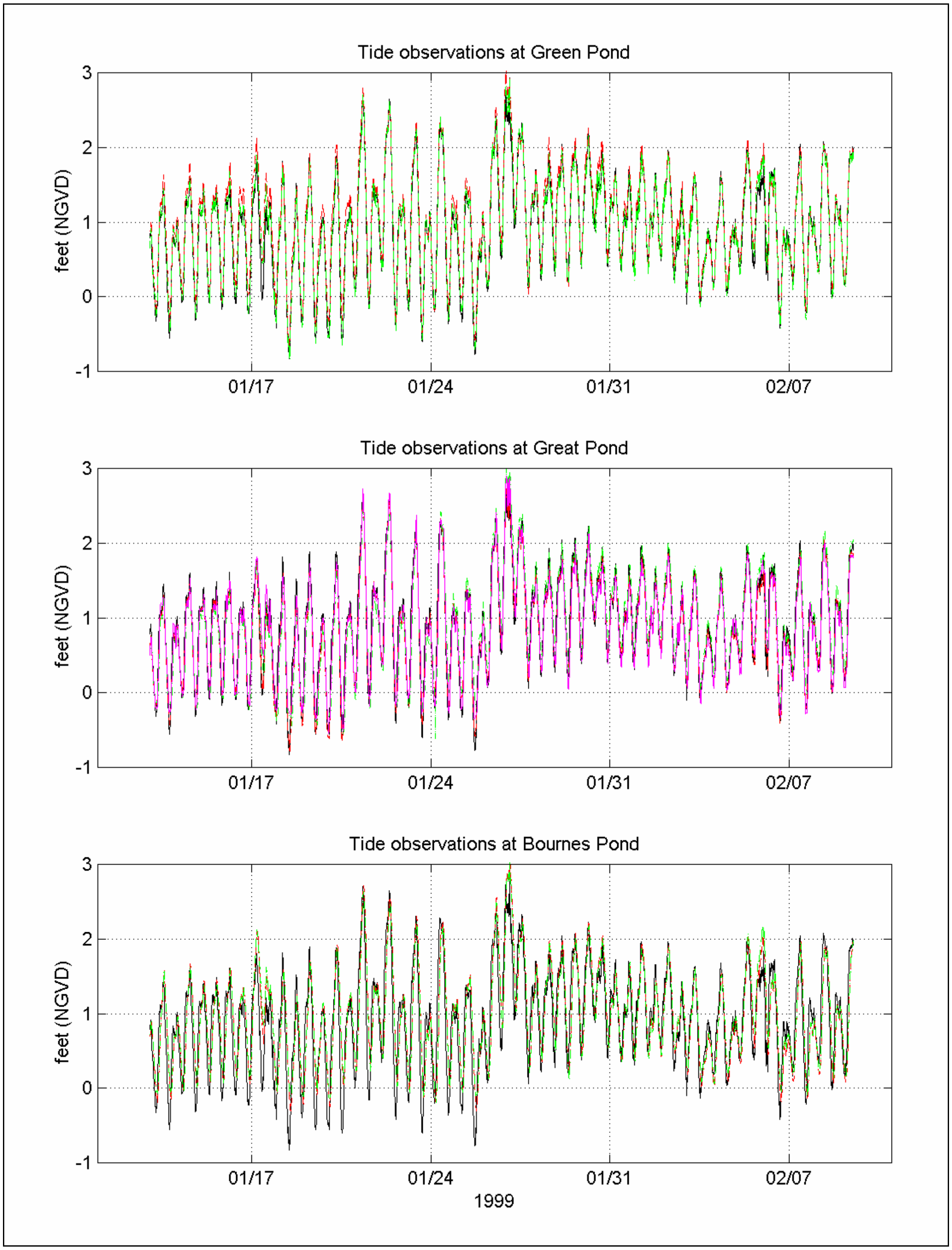


Figure V-16. Observations of water surface elevation variations in the Falmouth finger pond systems. Each plot shows the Nantucket Sound measurements overlaid with measurements obtained in each pond. The range of variation was approximately 3.5-4 feet during the deployment period January 12 through February 10, 1999; however, the average daily fluctuation was only about 1.5 feet.

Table V-3. Tide datums computed from 38-day records collected offshore, in Vineyard Sound, and in Great, Green and Bournes Ponds of Falmouth. Datum elevations are given relative to NGVD 29.								
Tide Datum	Offshore	Great P. Lower	Perch P.	Great P. Upper	Green P. Lower	Green P. Upper	Bournes P. Lower	Bournes P. Upper
Maximum Tide	2.9	2.7	3.0	2.9	2.9	2.9	3.0	3.0
MHHW	2.0	1.9	1.9	1.9	2.0	2.0	2.0	2.0
MHW	1.7	1.6	1.6	1.6	1.7	1.7	1.7	1.6
MTL	0.8	0.8	0.9	0.8	0.8	0.8	0.9	0.9
MLW	0.0	0.0	0.2	0.1	0.0	0.0	0.2	0.2
MLLW	-0.2	-0.1	0.0	-0.1	-0.2	-0.2	0.0	0.1
Minimum Tide	-0.8	-0.8	-0.6	-0.5	-0.8	-0.8	-0.3	-0.2

Table V-4 presents the phase delay of the M_2 tide at all tide gauge locations. Phase delay is another indication of tidal damping, and results with a later high tide at upper pond locations versus Nantucket Sound. The greater the frictional effects, the longer the delay between locations. These data suggest Green Pond is hydrodynamically efficient, as the tide is damped negligibly between Nantucket Sound and the upper reaches, almost two miles upstream. Great Pond suffers some damping at the mouth, delaying the tide approximately 7 minutes, and more significant damping in its northern region. Perch Pond, a sub-embayment off of the lower Great Pond basin, shows approximately half-hour delay in high water relative to Nantucket Sound. This delay may be due to a sand shoal at the Perch Pond mouth, which impedes the flow of water into the basin. Importantly, the results also indicate Bournes Pond suffers relatively greater frictional damping than adjacent Ponds, specifically between Nantucket Sound and the location of the gauge in lower Bournes Pond. This fact suggests the inlet at Bournes Pond is hydraulically less efficient than the Great Pond and Green Pond inlets.

Table V-4. M_2 Phase delays from Nantucket Sound.	
Location	Delay (minutes)
Lower Green Pond	0.32
Upper Green Pond	1.53
Great Pond Mouth	6.97
North Great Pond	17.41
Perch Pond	27.52
Lower Bournes Pond	62.61
Upper Bournes Pond	63.27

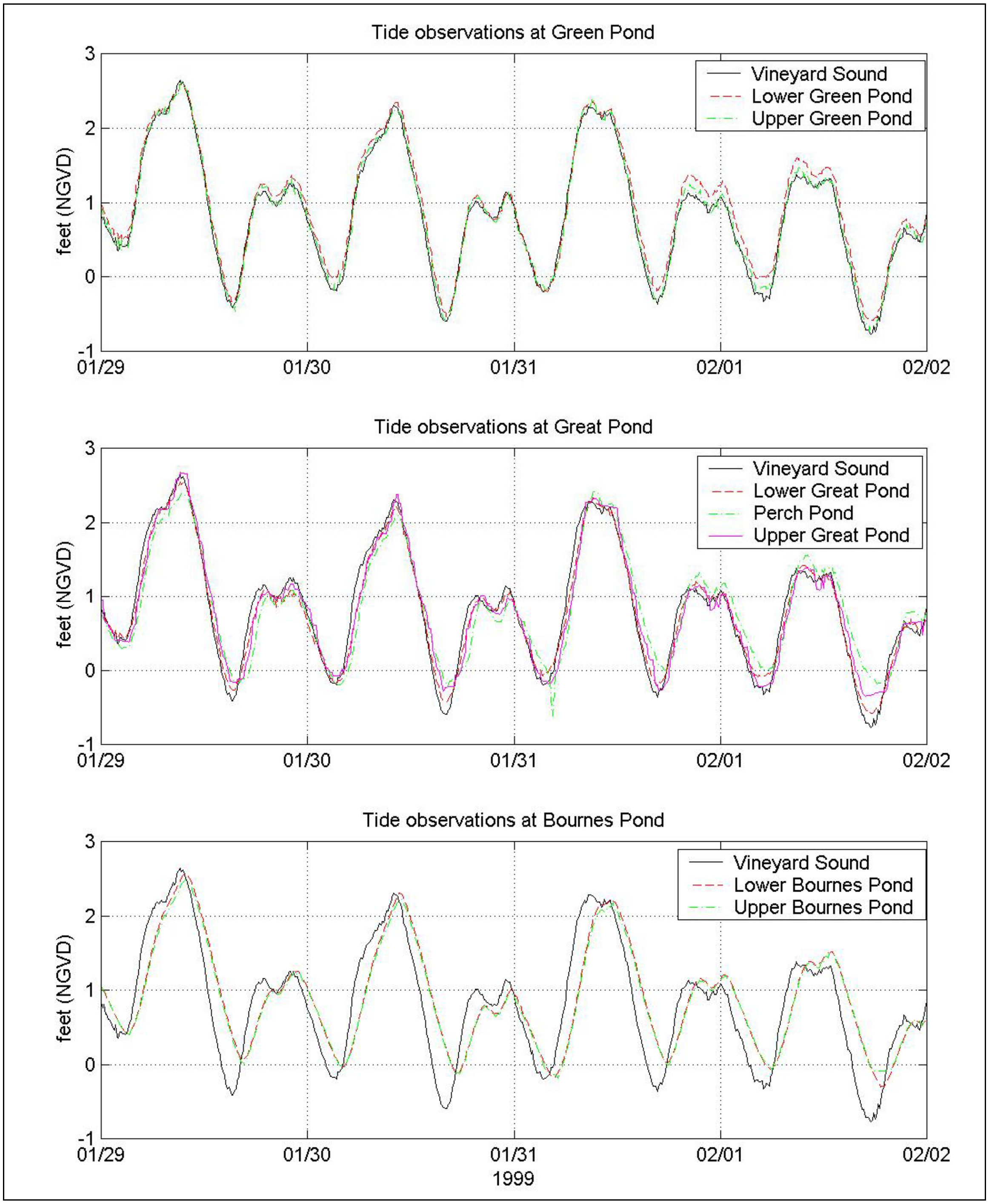


Figure V-17. View of water elevation variations for a four-day period during the deployment. Each plot depicts the Nantucket Sound signal overlaid with measurements obtained in the pond interiors. Note the reduced amplitude as well as the delay in times of high- and low tide relative to Nantucket Sound due to frictional damping through the pond systems.

Table V-5 shows the relative energy of tidal versus non-tidal processes at different locations in the systems. Non-tidal processes include wind responses, for example wind set-up and set-down, or sub-tidal oscillations originating in the Atlantic Ocean. Vineyard and Nantucket Sounds are relatively shallow and semi-enclosed, hence the water surface responds readily to variations in wind forcing, typically with water 'piling up' along the south Cape shore during southerly winds. At Falmouth, the island of Martha's Vineyard provides a wind block to the direct south; however, southwest and southeast winds can force significant variation in water surface elevations. To calculate these percentages, the signal variance (or energy) of each time series was computed. The results show that nearly one-quarter of the water surface variations in Nantucket Sound were due to non-tidal processes, and three-quarters of the signal were due to tidal processes. Figure V-18 shows that non-tidal residual processes can force water elevation changes greater than 2 feet within the ponds.

Table V-5. Percentages of Tidal versus Non-Tidal Energy (units of $ft^2 \cdot sec$).

Location	Total Variance	Total(%)	Tidal (%)	Non-tidal (%)
Nantucket Sound	0.443	100%	75.6	24.4
Lower Green Pond	0.437	100%	73.4	26.6
Upper Green Pond	0.434	100%	73.2	26.8
Great Pond Mouth	0.411	100%	72.7	27.3
North Great Pond	0.405	100%	71.2	28.8
Perch Pond	0.434	100%	64.6	35.4
Lower Bournes Pond	0.392	100%	68.7	31.3
Upper Bournes Pond	0.362	100%	67.4	32.6

The residual signals propagate into each of the embayments, where the percentage of non-tidal energy increases and the percentage of tidal energy decreases. This observation was due to several effects. One, tidal damping, or losses of amplitude through the inlet specifically, reduces tidal energy. Second, local effects of wind blowing across each Pond surface will increase the energy of non-tidal processes. These results indicate that hydrodynamic circulation in each of the Ponds is dependent primarily upon tidal processes, yet wind effects are of significant concern as well. Wind forces that affect Nantucket Sound are most important, as these effects propagate through the inlets to impact volume flow within each embayment.

Analysis of these data show that circulation in the ponds is governed primarily by tides, with a decreasing but still significant wind effect. The tide range is small relative to other Cape Cod locations, both along the south shore as well as north shore locations. Tidal damping within the Ponds decreases the water surface elevation variations relative to Nantucket Sound, as well as delays the time of high tide. Damping in Bournes Pond is significant, due likely to the undersized inlet opening, as well as in northern areas of Great Pond. Tidal damping reduces each pond's natural flushing ability; hence, Bournes Pond has the weakest flushing, with Great Pond and Green Pond flushing rates substantially stronger.

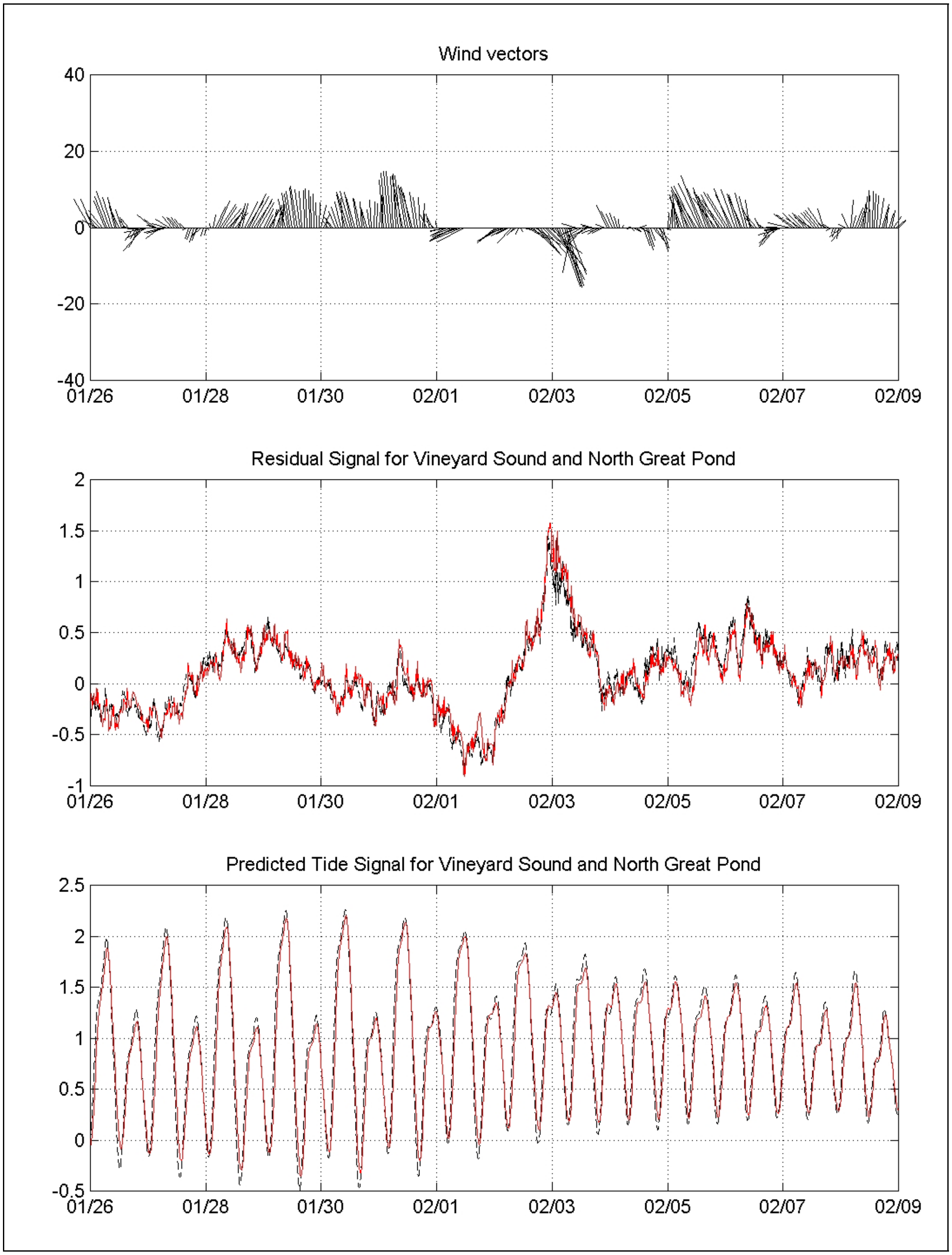


Figure V-18. Residual signal (middle plot) in Nantucket Sound and north Great Pond can be as great as 2 feet. The large gradient observed on February 3, 1999 coincided with strong (approximately 20 kts) southeast winds. These winds produced a 'piling up' of water along the southern shore of Falmouth; the elevation changes propagated into Great Pond. The bottom plot depicts the harmonic tides calculated from the harmonic analysis and shows the tidal range in the study area to be approximately 2.75 feet on January 29, 1999.

Water quality information, obtained annually in these ponds from the Falmouth Pond Watchers program, shows that Green Pond suffers from poor water quality, with Bournes Pond possessing the best water quality of the three systems under study. This is a surprising comparison, as Bournes Pond, with the greatest tide damping, has higher quality than Green pond, whose natural flushing capability is strong. One obvious ramification in this comparison is the amount of nutrient loading into each basin. Water quality, while a complex biochemical process, can be thought of simply as a balance between pollutant input versus flushing rate (or removal). A well-flushed system can tolerate larger pollutant inputs than a poorly-flushed system, and vice-versa. The data results from this study suggest that the pollutant loading to Green Pond is greater than the system's natural ability to flush such pollutants from its waters. Bournes Pond, despite its undersized inlet and resulting poor circulation, does not have pollutant inputs sufficient to impact negatively the Pond's water quality.

The M_4 constituent governs the shape of the tide, and relationships between M_2 and M_4 indicate whether an estuary is flood- or ebb-dominant. M_4 emerges as a harmonic of M_2 as a result of non-linear friction effects. M_4 is the quarter-diurnal (occurring 4 times daily) overtide of M_2 with a period (6.21 hours) equal to half the period of M_2 (12.42 hours). A relation between the phases of M_2 and M_4, *sea-surface phase*, can be used to classify an estuary as flood- or ebb-dominant (Friedrichs and Aubrey, 1988) based on the asymmetrical shape of the tide. Sea surface phases for all eight tide measurement locations demonstrated flood-dominance, causing a tendency to trap sediment. Flood-dominant systems trap sediment because current velocities are more swift when the tide is rising; therefore, more sediment is deposited within the system on the rising tide than can be transported out of the system on the falling tide. The relative height of M_4 and M_2 tidal constituents (M_4/ M_2), indicates the strength of the flood- or ebb-dominance. M_4/M_2 ranged from 0.14 (lower Bournes Pond) to 0.28 (lower Green Pond), indicative of moderate to strong flood-dominance compared to numerous estuaries studied by Friedrichs and Aubrey (1988). In comparison, the studied estuaries along the U.S. Atlantic coast exhibited M_4/M_2 ratios ranging from 0.003 (Townsend Inlet, NJ) to 0.26 (within North Channel in the Nauset, MA system). In addition, the Nantucket Sound tide itself indicates strong flood-dominance tendencies. Therefore, the sediment trapping characteristics primarily are due to the form of the forcing tide in Nantucket Sound and not inlet configuration or bathymetry of the ponds.

V.4. HYDRODYNAMIC MODELING

This study of Great, Green, and Bournes Ponds utilized a state-of-the-art computer model to evaluate tidal circulation and flushing. The particular model employed was the RMA-2 model developed by Resource Management Associates (King, 1990a). It is a two-dimensional, depth-averaged finite element model, capable of simulating transient hydrodynamics. The model is widely accepted and tested for analyses of estuaries or rivers. Applied Coastal staff members have utilized RMA-2V for numerous flushing studies on Cape Cod, including West Falmouth Harbor, Popponesset Bay, and the Pleasant Bay estuary.

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 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 finite element grid was generated within the shoreline developed for the Town's future Geographic Information System (GIS). A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of each Pond based on the tide gauge data collected in Nantucket Sound. Freshwater recharge boundary conditions were specified to approximate surface water inputs to the system. 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 numerous (20+) model calibration simulations for each Pond, 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 simplified by the use of the SMS package. The digital shoreline and bathymetry data were imported to SMS, and a finite element grid was generated to represent the estuary. Figure V-19 illustrates the finite element grids for Great Pond, Green Pond, and Bournes Pond. With the exception of the streams entering each Pond at the north end, the Ponds were represented by two-dimensional (depth-averaged) elements.

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

Figure V-19 illustrates the varying element sizes at the inlets to each pond. Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each Pond. Relatively fine grid resolution was employed where complex flow

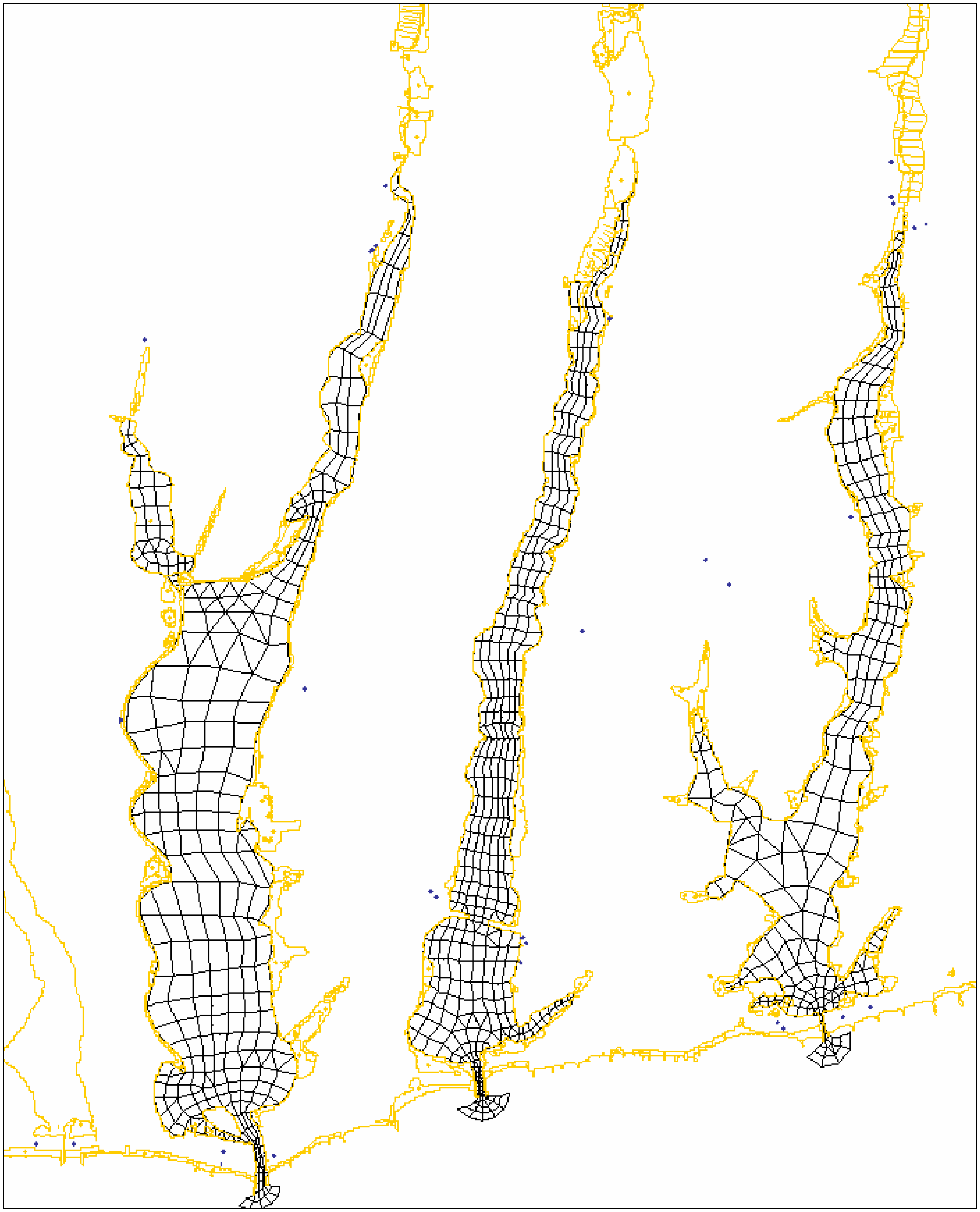


Figure V-19. Plot of numerical grids (black) used for hydrodynamic modeling for Great, Green, and Bournes Ponds, with shoreline (yellow).

patterns were expected. For example, smaller node spacing in the vicinity of each inlet was designed to provide a more detailed analysis in these regions of rapidly varying flow. Also, elements through the each inlet region were designed to account for the rapid changes in bathymetry caused by inlet shoaling and scour processes. Widely spaced nodes were defined for much of the lower Ponds, where flow patterns did not change dramatically. Appropriate implementation of wider node spacing and larger elements, reduced computer run time with no sacrifice of accuracy.

V.4.2.2 Boundary Condition Specification

Three types of boundary conditions were employed for the RMA-2 model: 1) "slip" boundaries , 2) freshwater inflow, and 3) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Based on field measurements, freshwater recharge was specified at the north end of each Pond. A tidal boundary condition was specified seaward of the inlet to each Pond. TDR measurements provided the required data. The rise and fall of the tide in Nantucket Sound is the primary driving force for estuarine circulation. Dynamic (time-varying) model simulations specified a new water surface elevation in Nantucket Sound every 10 minutes.

V.4.2.3 Calibration

After developing the finite element grids, and specifying boundary conditions, the model for each Pond was calibrated. Calibration ensured the model predicted accurately what was observed in nature during the field measurement program. The calibrated models provide a diagnostic tool to evaluate other scenarios (e.g. the effects of increasing the Bournes Pond inlet cross-section to improve flushing). Numerous model simulations were required (20+) for each estuary, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the flushing model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (e.g. lower Green Pond). Initially, a two-day period was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, a seven-day period 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.

The calibration was performed for a seven-day period beginning 0:00 EST on January 26, 1999. This representative time period was selected because it included the range of tidal conditions typical in the estuary during the 30-day deployment period. Since the tide gauges were deployed during the months of January and February, typical winter storm activity influenced tidal fluctuations. To provide average tidal forcing conditions to the predictive water quality model, a time period was chosen that had minimal atmospheric pressure and/or wind effects. Throughout the selected seven-day period, the tide range varied from approximately 1.0 to 1.7 ft. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. For instance, average residence times were computed over the entire seven-day simulation. Other methods, such as dye and salinity studies, evaluate tidal flushing over relatively short time periods (less than one day). These short-term measurement techniques may not be representative of average conditions due to the influence of unique, short-lived atmospheric events (e.g. the storm surge on February 3, 1999 exceeded 1.5 ft.). Modeled tides were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibrated model was used to analyze existing detailed flow patterns and compute residence times.

V.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. Initially, Manning's friction coefficient between 0.02 and 0.04 were specified for all elements. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels with pools and shoals with higher friction (Henderson, 1966).

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning’s coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy channels in the entrance channel of each Pond, versus the silty bottom of the shallow regions in the upper portions of each Pond, which provided 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 selected based on the Civil Engineering Reference Manual (Lindeburg, 1992) and values required to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-6.

Table V-6. Manning’s Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas.

System Embayment	Bottom Friction
Great Pond (entrance)	0.025
Great Pond (upper)	0.025
Green Pond	0.025
Bournes Pond (entrance)	0.03
Bournes Pond (upper)	0.03

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 more swift, such as inlets and bridge constrictions. According to King (1990a), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). The models of each Falmouth finger pond were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Primarily, this can be attributed to the small tide range along the south shore of Falmouth. Final calibrated turbulent exchange coefficients were set at a typical value of 0.5 times the element dimension for each of the three model grids.

V.4.2.3.3 Marsh Porosity Processes

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

V.4.2.3.4 Comparison of Modeled Tides and Measured Tide Data

A best-fit of model predictions for the first TDR deployment was achieved using the aforementioned values for friction and turbulent exchange. Figures V-20 through V-22 illustrate a two tidal cycle sub-section of the seven-day calibration simulation. Modeled (solid line) and measured (dotted line) tides are illustrated for the Great Pond (lower and upper), Perch Pond, Green Pond (lower and upper), and Bournes Pond (lower and upper). Only two tidal cycles are illustrated to focus on the details of the tide curve. Figures V20 through V-22 confirm visual agreement between modeled and measured tides within each coastal Pond.

Although visual calibration revealed the modeled tidal hydrodynamics were reasonable, tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in Nantucket Sound. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Tables V-7 through V-13 for the calibration period differ from those in Table V-2 because constituents were computed for only the seven-day section of the thirty-days represented in Table V-2. Tables V-7 through V-13 compare tidal constituent height and time lag for modeled and measured tides at Great Pond (lower and upper), Perch Pond, Green Pond (lower and upper), and Bournes Pond (lower and upper), respectively. Time lag represents the time required for a constituent to propagate from Nantucket Sound to each location.

The constituent calibration revealed excellent agreement between modeled and measured tides. Errors associated with tidal constituent height were on the order of 0.1 ft, which was only slightly larger than the accuracy of the tide gauges (0.032 ft). Time lag errors were typically less than the time increment resolved by the model (0.17 hours or 10 minutes), indicating good agreement between the model and data. Since tidal amplitude and phase attenuation was relatively minor, with the exception of Bournes Pond, constituent calibration required that the M_2 constituent propagate into the Ponds with minimal resistance. The tide attenuation at Bournes Pond primarily results from the narrow inlet. The hydrodynamic model was able to predict accurately the effect of the inlet on flow properties.

The hydrodynamic model’s ability to predict propagation of the secondary non-linear constituents through the estuary is important for understanding the attenuation of the tidal signal and the impact this has on estuarine circulation. Of primary interest is the M_4 constituent, which can be used to determine the flood dominance (sediment trapping characteristics) of an estuarine system. Proper prediction of M_4 provides confidence in the model’s accuracy, since this indicates that the model is capable of simulating the tidal wave form and size. Similar to the model predictions for M_2, comparison of the information from Tables V-7 through V-13 indicates that the modeled phase of M_4 falls within approximately one time step of the observed value.

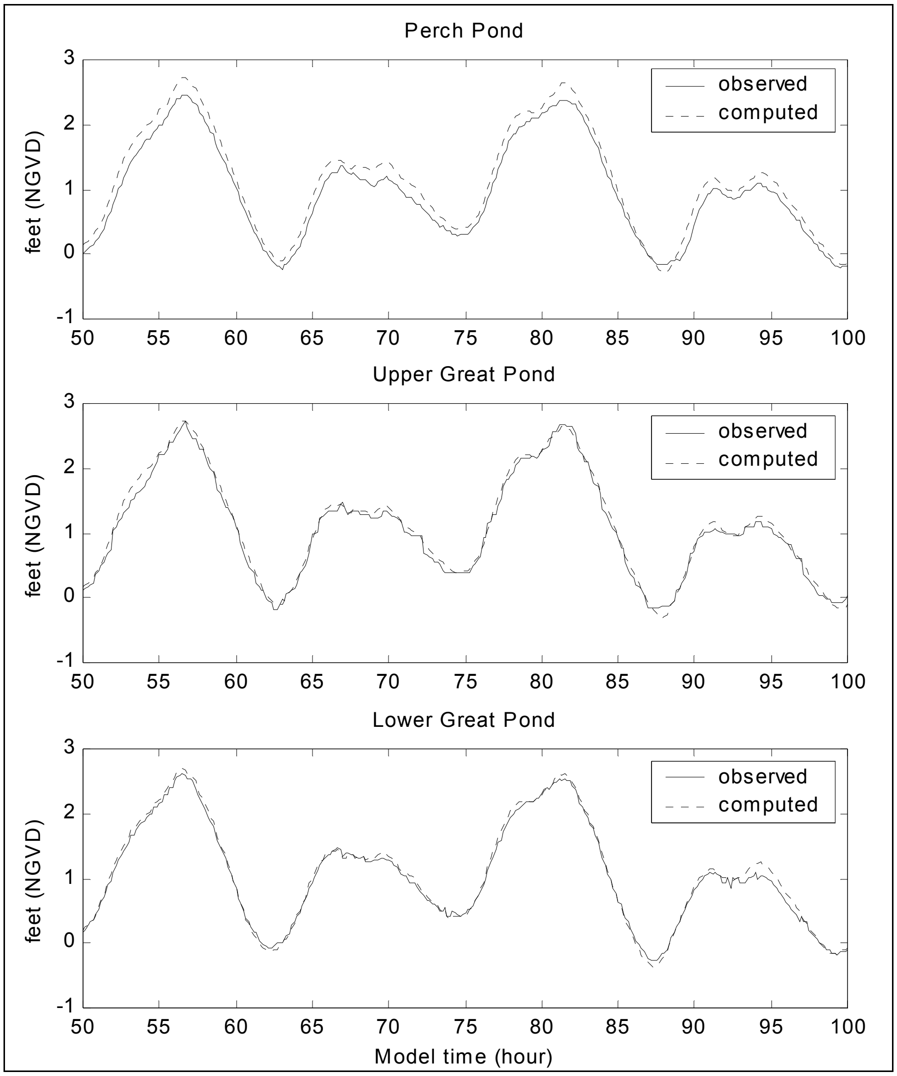


Figure V-20. Observed vs. computed water level elevations for tide gauges in Great Pond.

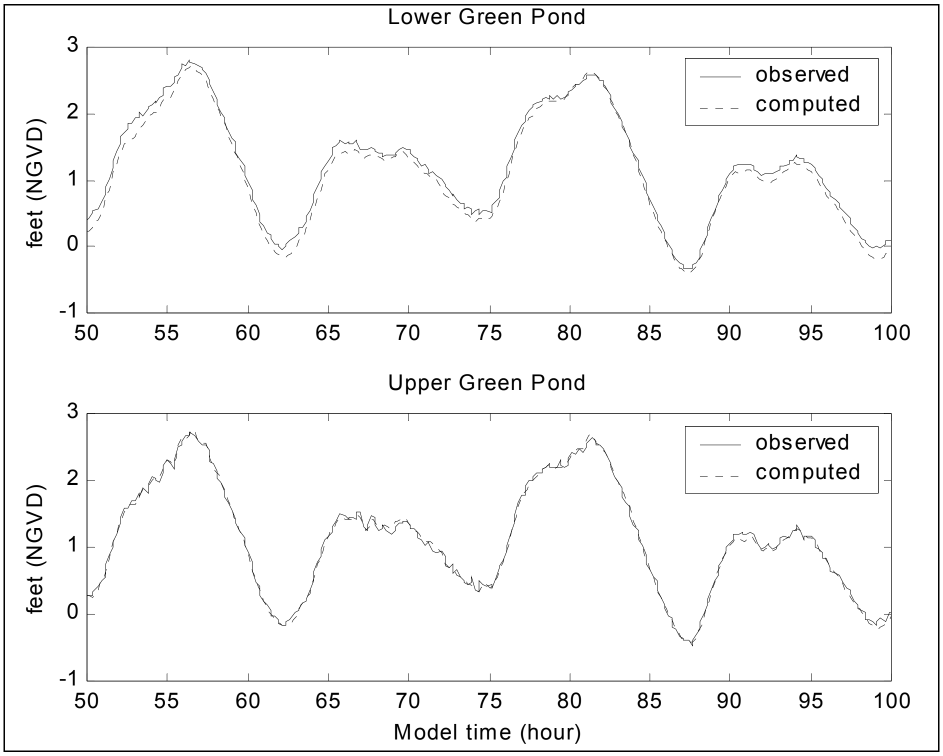


Figure V-21. Observed vs. computed water level elevations for tide gauges in Green Pond.

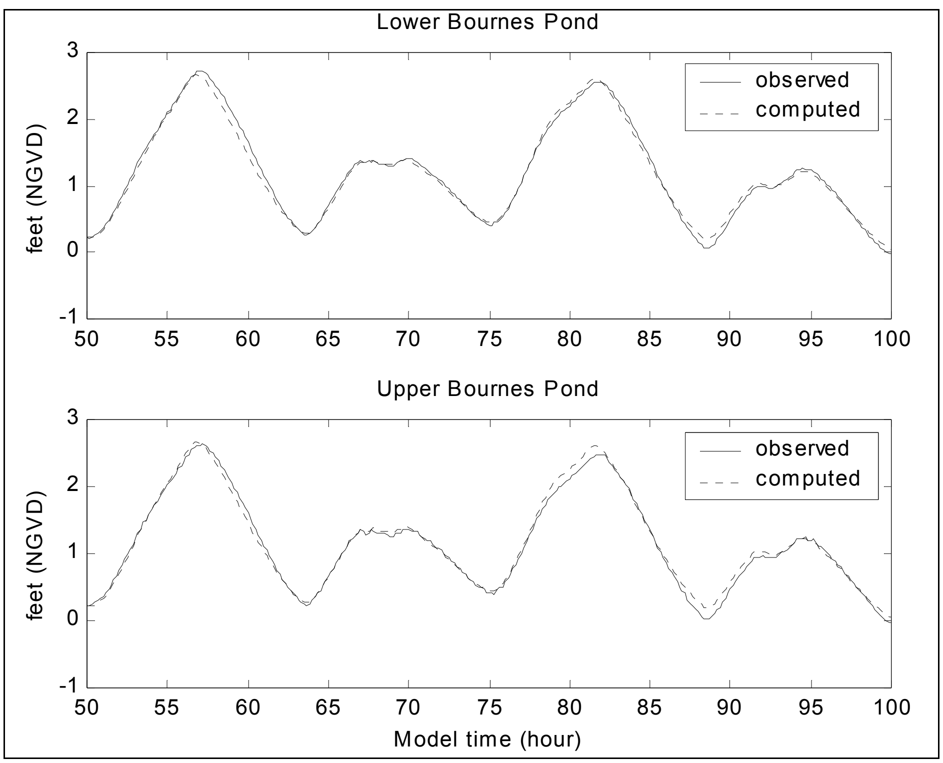


Figure V-22. Observed vs. computed water level elevations for tide gauges in Bournes Pond.

Table V-7. Tidal Constituent Calibration for Lower Great Pond.

Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.41	0.24	0.42	0.14	-0.01	0.10
M_2 (12.42)	0.85	0.16	0.88	0.18	-0.03	-0.02
M_4 (6.21)	0.14	0.27	0.18	0.19	-0.04	0.08
M_6 (4.14)	0.04	0.40	0.06	0.30	-0.02	0.10

Table V-8. Tidal Constituent Calibration for Upper Great Pond.

Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.41	0.81	0.43	0.62	-0.02	0.19
M_2 (12.42)	0.83	0.36	0.85	0.47	-0.02	-0.11
M_4 (6.21)	0.14	0.57	0.16	0.67	-0.02	-0.10
M_6 (4.14)	0.06	0.54	0.08	0.52	-0.02	0.02

Table V-9. Tidal Constituent Calibration for Perch Pond.

Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.40	0.96	0.43	0.69	-0.03	0.27
M_2 (12.42)	0.79	0.55	0.85	0.55	-0.06	0.00
M_4 (6.21)	0.11	1.05	0.15	0.82	-0.04	0.23
M_6 (4.14)	0.07	0.87	0.09	0.63	-0.02	0.24

Table V-10. Tidal Constituent Calibration for Lower Green Pond.

Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.42	0.23	0.43	0.07	-0.01	0.16
M_2 (12.42)	0.86	0.02	0.88	0.08	-0.02	-0.06
M_4 (6.21)	0.23	0.14	0.22	0.11	0.01	0.03
M_6 (4.14)	0.07	0.17	0.07	0.10	0.00	0.07

Table V-11. Tidal Constituent Calibration for Upper Green Pond.						
Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.42	0.19	0.43	0.11	-0.01	0.08
M_2 (12.42)	0.86	0.04	0.88	0.12	-0.02	-0.08
M_4 (6.21)	0.22	0.17	0.22	0.17	0.00	0.00
M_6 (4.14)	0.08	0.25	0.08	0.14	0.00	0.11

Table V-12. Tidal Constituent Calibration for Lower Bournes Pond.						
Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.42	2.24	0.40	2.00	0.02	0.24
M_2 (12.42)	0.75	1.16	0.70	1.34	0.05	-0.18
M_4 (6.21)	0.08	1.26	0.04	1.53	0.04	-0.27
M_6 (4.14)	0.05	1.78	0.05	1.82	0.00	-0.04

Table V-13. Tidal Constituent Calibration for Upper Bournes Pond.						
Constituent & Period (hours)	Measured		Modeled		Error	
	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)	H (ft)	ϕlag (hours)
K_1 (23.93)	0.40	2.25	0.40	2.00	0.00	0.25
M_2 (12.42)	0.72	1.17	0.70	1.34	0.02	-0.17
M_4 (6.21)	0.08	1.24	0.04	1.54	0.04	-0.30
M_6 (4.14)	0.05	1.79	0.05	1.83	0.00	-0.04

V.4.2.4 Model Verification Using Horizontal ADCP Measurements

Current measurements across the inlet channel of Great Pond were collected using a Horizontal Acoustic Doppler Current Profiler (H-ADCP),manufactured by RD Instruments (RDI). The HADCP was mounted on a frame that could be vertically leveled, and attached between the pilings of a private dock on the East side of the entrance to Great Pond. The instrument was oriented approximately westward (275.5°), providing a continuous horizontal profile of flow from the east bank to the west bank of the inlet.

The H-ADCP emits individual acoustic pulses from three transducers aligned horizontally. The instrument then listens to backscattered echoes reflected from ambient sound scatters at discrete intervals in the water column. The propagation delay, the time lag between emitted pulses, corresponds to a change in distance between the transducer and the sound scatterer, due to a Doppler shift. As particles move further away from the transducers, sound takes longer to travel back and forth. The time lag and the speed of sound in water are used to compute the velocity of the particle relative to the transducer. By combining the velocity components of the three directional beams, the current velocities are transformed using the unit's internal compass readings to an orthogonal earth coordinate system in terms of east and north current velocity.

Horizontal structure of the currents is obtained using a technique called ‘range-gating’. Received echoes are divided into successive segments (gates) based on discrete time intervals of pulse emissions. The velocity measurements for each gate are averaged over a specified width to produce a single velocity at the specified interval (‘bin’). A horizontal velocity profile is composed of measurements in successive bins across the channel.

The fixed H-ADCP current measurements result in a time series of a slice of the water column at a fixed depth location. The horizontal resolution was set to 1.64 ft, or approximately one velocity observation per every 20 inches of water depth. The first measurement bin was centered approximately 3 ft from the head of the H-ADCP, allowing for an appropriate blanking distance between the transducer and the first measurement. Velocity measurements for each ensemble are computed by averaging several velocity measurements together in time to increase the accuracy of the measurements. For this study, each ensemble was a 30 second average measured at a rate of 1 ping per second.

The H-ADCP was deployed for approximately 48 hours, from 13:00 May 5, 2003 to 15:00 May 7, 2003. The maximum tidal currents recorded during the deployment occurred during the evening flood tide on May 5, and the ebb tide on May 7. At the time of deployment, the tide was flowing in (flood); reaching high tide at 04:00 on May 6. Under peak flood conditions, speeds in the center of the channel reached a maximum of 3.8 ft/s flowing into Great Pond (Figure V-23). On May 7, high tide occurred at approximately 04:30, and water elevations began to drop (ebb). Maximum ebb tidal currents of 2.8 ft/s were measured flowing out of Great Pond on May 7 at approximately 08:14 (Figure V-24).

The horizontal distance of the H-ADCP profile is dependent on the width of the acoustic beam, instrument frequency, and the total water depth at the deployment site. The H-ADCP was deployed at approximately mid-depth; an average of 4 ft from the water surface to the instrument and 5 ft above the seabed. In this position the H-ADCP can measure a distance of approximately 60 ft before the signal is contaminated by reflection off the seabed or sea surface. The diminishing magnitude of the tidal currents (Figures V-23 and V-24) with increased distance from the instrument, mounted on the East bank of the inlet, is likely the result of contamination from the side lobe transducers. Due to the shallow depth of Great Pond inlet, a complete horizontal profile (from East bank to West bank) was not obtained.

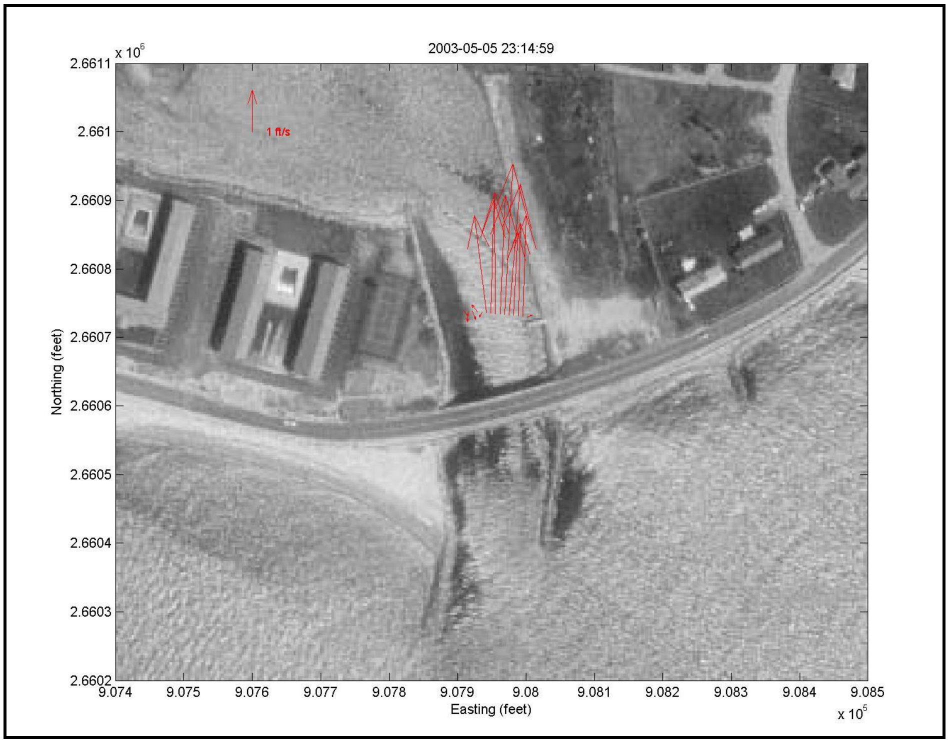


Figure V-23. Current velocity measurements collected during peak flood conditions on May 5 at 23:15 in the entrance to Great Pond. For comparison, the scale of 1 ft/s is shown in the upper right: The maximum speed was 3.8 ft/s.

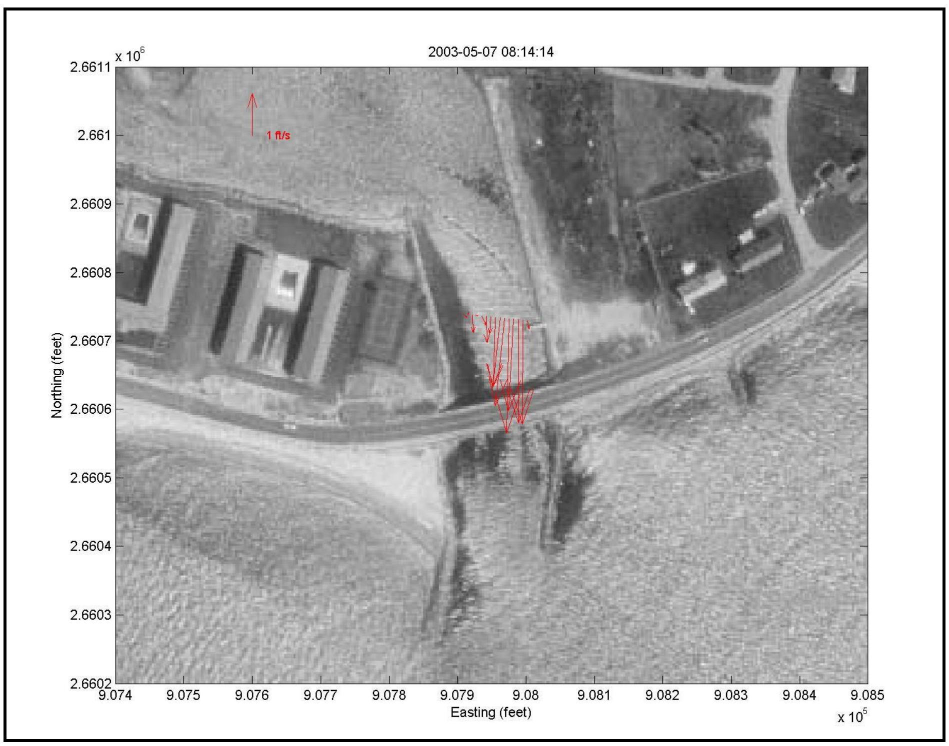


Figure V-24. Current velocity measurements collected during peak ebb conditions on May 7 at 08:14 in the entrance to Great Pond. For comparison, the scale of 1 ft/s is shown in the upper right: The maximum speed was 2.8 ft/s.

The H-ADCP velocity measurements provide an additional data source for model verification. The accuracy of all three of the Ashumet Valley ponds was first verified by modeling a time period different from the original calibration time period. With the H-ADCP current data, it is possible to verify the model's accuracy by using a variable that is independent from the tide elevations used to calibrate the model.

For the velocity verification of Great Pond, velocities from a single mid-channel bin from the H-ADCP were compared to velocities output from the hydrodynamic model, at the same mid-channel location. A plot of the comparison is presented in Figure V-25. The comparison shows that the model represents velocities in the inlet very skillfully, with an RMS error of 12.0%, and an R^2 value of 0.74. The successful velocity verification provides confidence that the model represents well the hydrodynamics of the real, physical system.

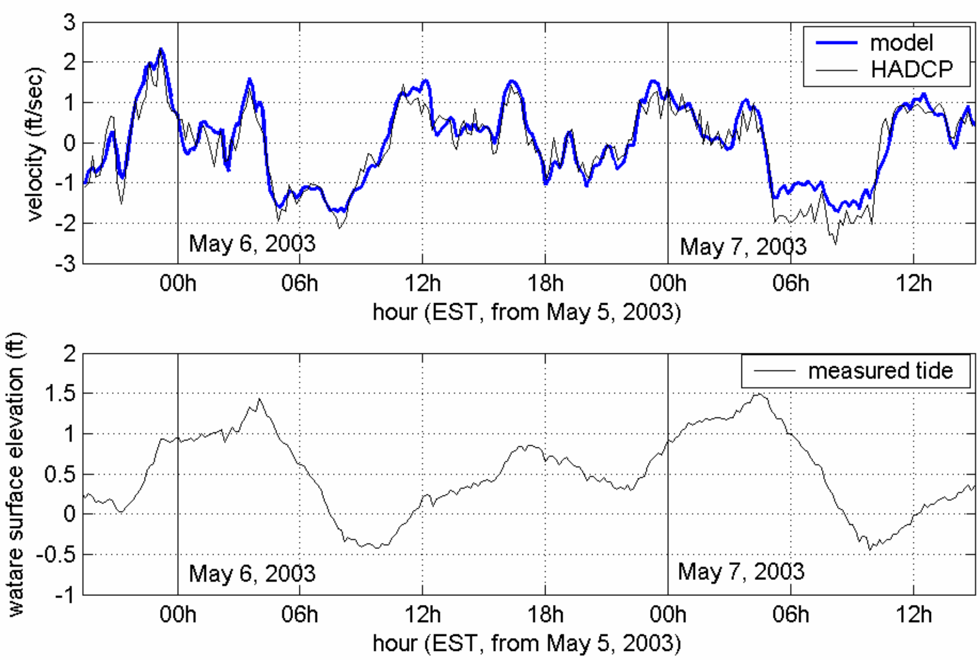


Figure V-25. Top plot depicts the Great Pond H-ADCP current measurement (thin black line) comparison with hydrodynamic model output (thick blue line). Coefficient of determination, R^2=0.74; rms error, E_{rms}=12.0%. The bottom plot presents the measured offshore tide (Vineyard Sound) for the same period.

V.4.2.5 Model Circulation Characteristics

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

In Section V.3.2, the tidal analysis revealed that all three Ponds indicate flood-dominant characteristics (Aubrey and Speer, 1985). The strong flood tide currents of short-duration, and corresponding weaker ebb flows over a longer duration were characteristic of flood-dominant estuaries. Typically, flood-dominance is an indicator of an estuary's tendency to trap and accumulate sediment. A simplified explanation of this complex phenomenon is stronger flood currents have the energy to drag suspended sediment into the system, whereas weaker ebb flows do not have sufficient energy to suspend and flush these sediments. A majority of the sediment most likely settles after flood tides and is not re-suspended on the ebb. However, it should be noted that the forcing tide within Nantucket Sound also exhibits strong flood dominant characteristics. In addition to sediment trapping potential within the estuaries, offshore tidal flow will tend to mobilize more sediment during the flood portion of the tidal cycle than the ebb.

The primary sediment source to each of the three Ponds is the predominant west-to-east littoral drift along Falmouth's beaches. Since the estuaries are flood-dominant, sand entering each inlet will continue to cause shoaling within the inlet throat. Since each inlet is jettied, this build-up of sand periodically will require dredging. To reduce "recycling" of sand, dredged material usually should be placed on the downdrift (east) side of the inlet. In addition to beach sediment, secondary sediment sources include bottom sand in the estuary, bank erosion and runoff, and biological decay. Although these sources may create flushing/water quality problems at certain locations within each Pond, they should not have a major impact on the overall ability of the estuaries to exchange water with Nantucket Sound.

Tidal circulation in each of the three Ponds is illustrated in Figures V-26 through V-28 illustrate two-dimensional flood tide current patterns at locations near each inlet. Tidal circulation patterns are indicative of flushing characteristics. Currents are represented by arrow vectors pointing in the direction of flow, with color contours indicating velocity magnitude. During flood and ebb, the strongest tidal currents occur through the tidal inlets of all three ponds. Due to the flow restriction created by the narrow entrance channel, Bournes Pond inlet exhibits flood velocities of approximately 7 ft/sec. (4.2 knots). Great Pond inlet and Green Pond inlet have maximum velocities of 4.5 and 2.9 feet per second (2.7 and 1.7 knots), respectively. The stronger tidal currents in Bournes Pond inlet may be responsible for decreased shoaling within the inlet channel, as well as the significant shoals north (flood shoal) and south (ebb shoal) of the entrance.

V.5. FLUSHING CHARACTERISTICS

Since freshwater inflow is negligible (based on measurements between November 1998 and March 1999, surface flow into the estuaries ranges from 0.7 to 19.8 cubic feet per second) in comparison to the tidal exchange through each inlet, the primary mechanism controlling estuarine water quality within Great, Green, and Bournes Ponds is tidal circulation. A rising tide in Nantucket Sound creates a slope in water surface from the ocean into the estuary. Consequently, water flows into (floods) the estuary. Similarly, the estuary drains into Nantucket Sound on an ebbing tide. This exchange of water between the estuary and the ocean is defined as tidal flushing. The calibrated hydrodynamic model is a tool to evaluate quantitatively tidal flushing of the Ponds, and was used to compute flushing rates (residence times) and tidal circulation patterns.

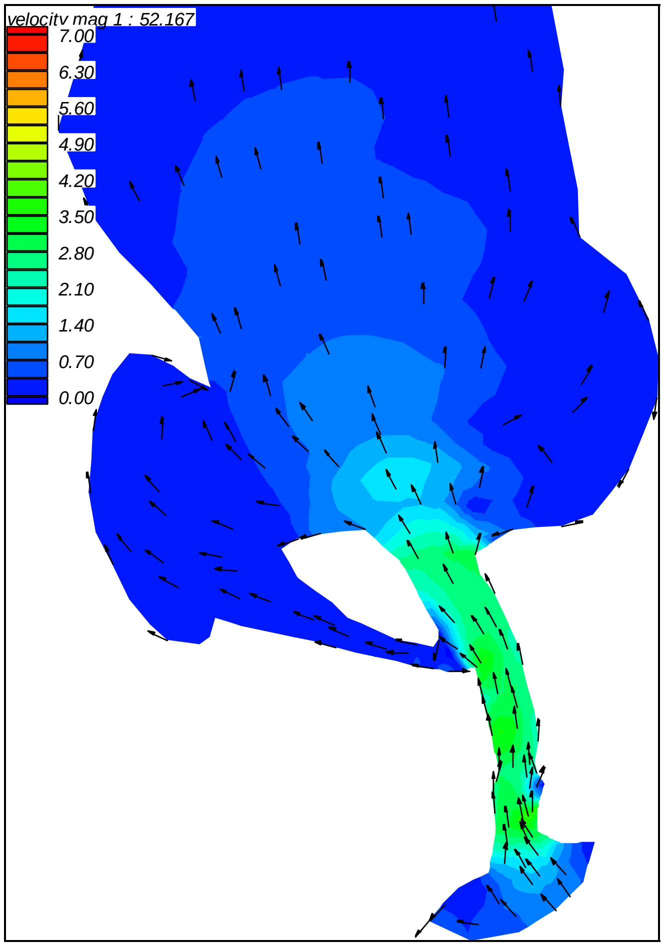


Figure V-26. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle within Great Pond. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

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

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

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

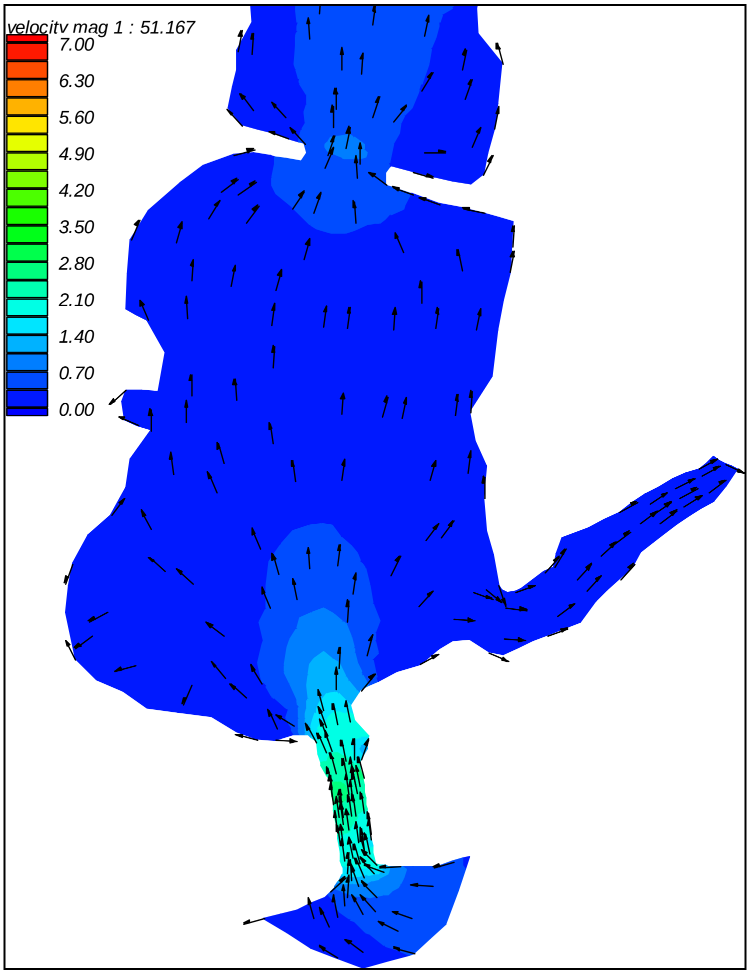


Figure V-27. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle within Green Pond. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Green Pond as an example, the **system residence time** is the average time required for water to migrate from upper Green Pond to Nantucket Sound, where the **local residence time** is the average time required for water to migrate from upper Green Pond to lower Green Pond. Local residence times for each sub-embayment are computed as:

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

where T_{local} denotes the residence time for the local sub-embayment, V_{local} represents the volume of the sub-embayment at mean tide level, P equals the tidal prism (or volume entering the local sub-embayment through a single tidal cycle), and t_{cycle} the period of the tidal cycle (again, 0.52 days).

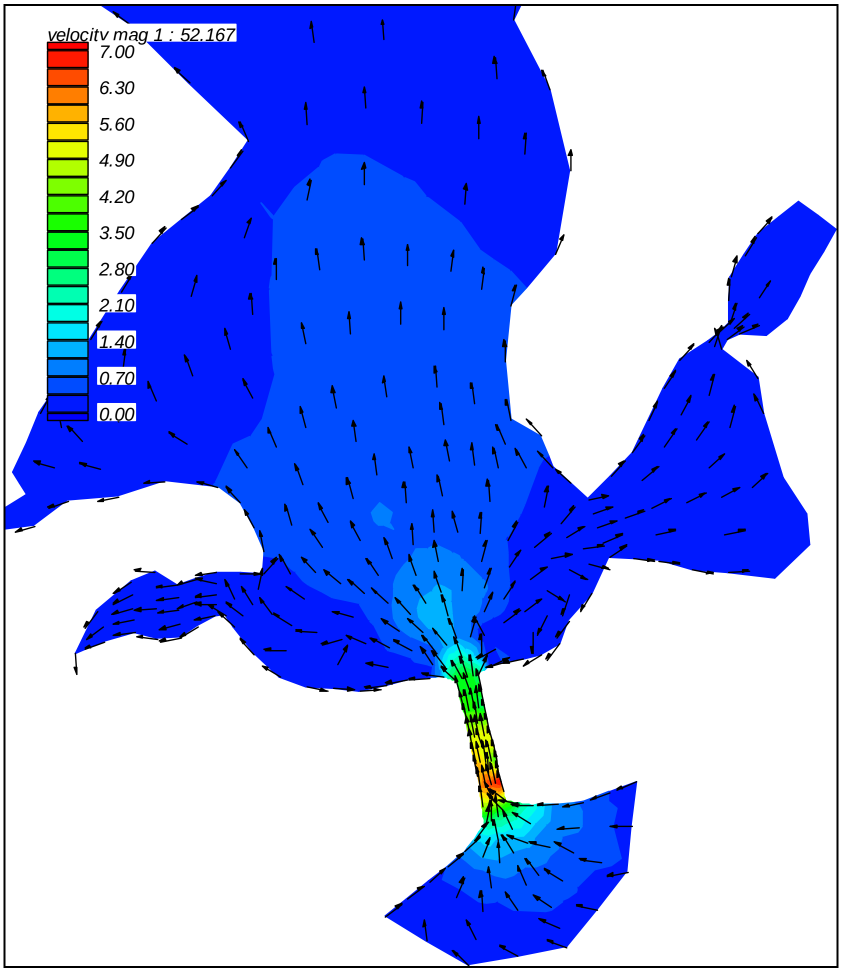


Figure V-28. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle within Bournes Pond. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

Residence times are provided as a first order evaluation of estuarine water quality. Lower residence times generally correspond to higher water quality; however, residence times may be misleading depending upon pollutant/nutrient loading rates and the overall quality of the receiving waters. As a qualitative guide, **system residence times** are applicable for systems where the water quality within the entire estuary is degraded and higher quality waters provide the only means of reducing the high nutrient levels. This is a valid approach in this case, since it assumes the sound has relatively higher quality water relative to the estuary.

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

Although the three Ponds are relatively shallow (large portions of each Pond are less than 4 feet deep at mean tide level), the small tide range reduces the tidal prism, thereby increasing residence times. The relatively short system residence times for all three Ponds indicates that the inlets from Nantucket Sound to each Pond do not significantly retard tidal flow. Based on the tidal constituent analysis tidal attenuation along the length of each Pond is negligible. The inlets at Great and Green Ponds cause less than 10 minutes of phase lag for the M_2 constituent (see Table V-13 for details). The only significant damping within the three Ponds is through the inlet at Bournes Pond, where the inlet alone causes more than a one-hour phase lag between Nantucket Sound and lower Bournes Pond.

Residence times reflect the lack of tidal damping through the inlets as well as the bathymetry found in each Pond and sub-embayment. Since tidal waters flow freely into each Pond, the volume of water exchanged during a tidal cycle can be approximated by the surface area of the embayment multiplied by the tide range. For systems with little tidal damping, the bathymetry tends to control residence times, where shallow sub-embayments will exhibit lower residence times than deeper sub-embayments. This is most clearly illustrated by Perch Pond, which is a relatively deep sub-embayment of Great Pond. This "kettle hole" is connected to Great Pond by a shallow channel. Although tidal exchange through the channel is adequate, the deep depths (in excess of 9.5 ft at some locations) cause the percentage of total volume exchanged with Great Pond relatively low compared to other sub-embayments with the three Ponds.

The relatively long residence time for some sub-embayments (e.g. Israels Cove) revealed the inadequacy of using system residence time alone to evaluate water quality. By definition, smaller sub-embayments have longer residence times; therefore, residence times may be misleading for small, remote parts of the estuary. Instead, it is useful to compute a local residence time for each sub-embayment. A local residence time represents the time required for a water parcel to leave the particular sub-embayment. For instance, the local residence time for Israels Cove represents the time required for a water parcel to be flushed from the Cove into Bournes Pond. Local residence times are computed as the volume of the sub-embayment divided by the tidal prism of that sub-embayment, and units are converted to days, Table V-14. Table V-15 lists local residence times for several areas within Great, Green, and Bournes Ponds. The basins utilized for residence time calculations are shown in Figure V-29.

Local residence times in Table V-15 are significantly lower than residence times based on the volume of the entire estuary. For example, flow entering upper Bournes Pond on an average tidal cycle flushes through Bournes Pond inlet in 4.21 days, but flushes into lower Bournes Pond in only 1.09 days (less than two tidal cycles). Generally, a local residence time is only useful where the adjacent embayment has high water quality. For the three Falmouth Ponds, the receiving waters that exchange tidal flow with the various sub-embayments show signs of ecological stress, indicative of poor water quality. Therefore, system residence times may be more appropriate for future planning scenarios.

Table V-14. Embayment mean volumes and average tidal prism during simulation period.		
Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Great Pond (total)	54,452,700	22,985,400
Great Pond (upper)	5,471,000	3,223,400
Perch Pond	3,775,200	1,495,700
Green Pond (total)	22,471,900	12,091,500
Green Pond (upper)	13,263,600	8,077,400
Bournes Pond (total)	22,408,300	10,620,400
Bournes Pond (upper)	6,046,800	2,752,500
Israels Cove	1,357,100	726,400

Table V-15. Computed System and Local residence times for embayments in the system.		
Embayment	System Residence Time (days)	Local Residence Time (days)
Great Pond (total)	1.23	1.23
Great Pond (upper)	8.74	0.88
Perch Pond	18.84	1.31
Green Pond (total)	0.96	0.96
Green Pond (upper)	1.44	0.85
Bournes Pond (total)	1.09	1.09
Bournes Pond (upper)	4.21	1.14
Israels Cove	15.96	0.97

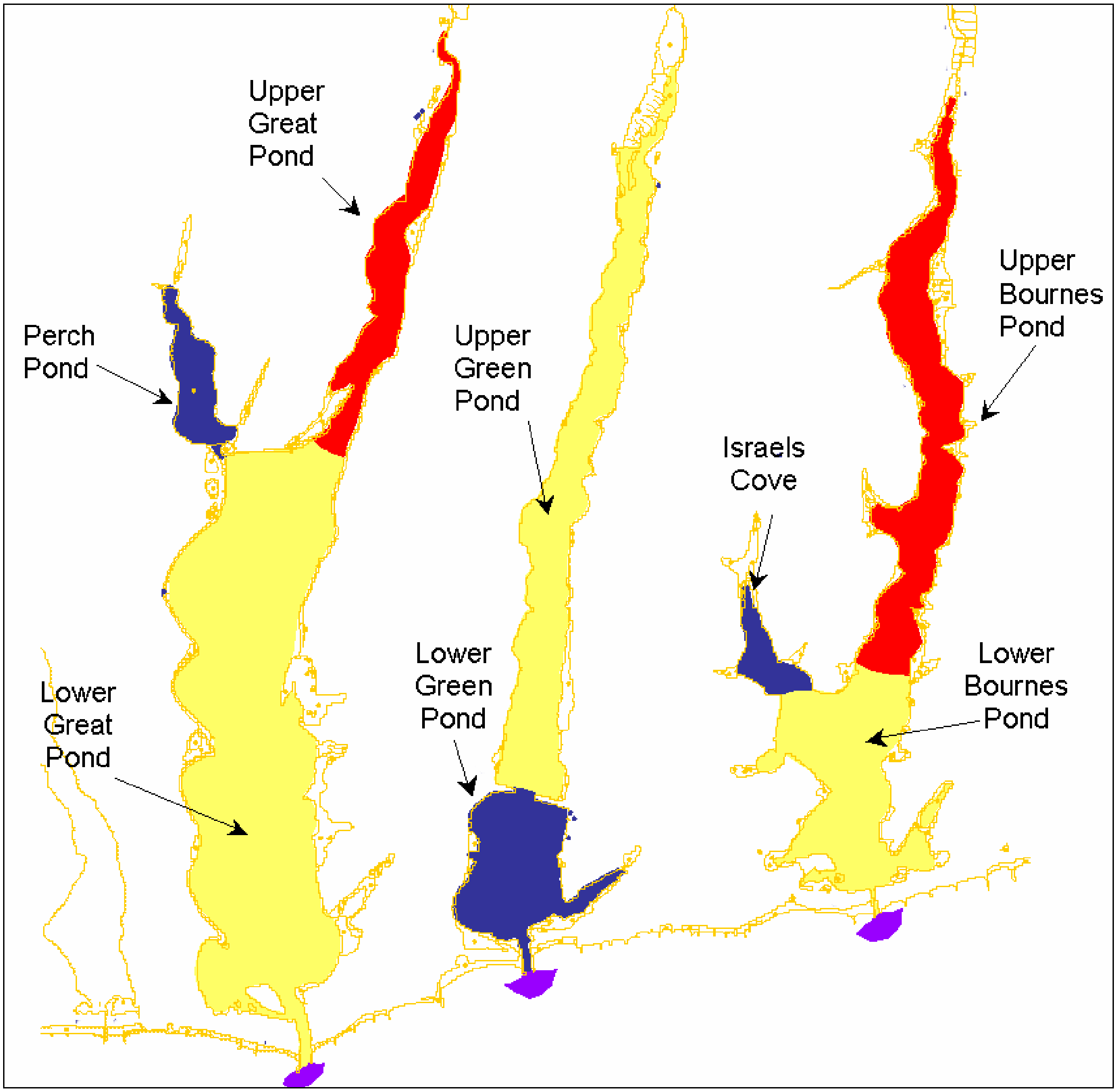


Figure V-29. Basins used to compute residence times for the system.

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 Ashumet Valley estuary system (Great, Green and Bournes ponds). These include the output from the hydrodynamics model, calculations of external nitrogen loads from the watersheds, measurements of internal nitrogen loads from the sediment (benthic flux), and measurements of nitrogen in the water column.

VI.1.1 Hydrodynamics and Tidal Flushing in the Embayments

Extensive field measurements and hydrodynamic modeling of the embayments were an essential preparatory step to the development of the water quality model. The result of this work, among other things, was a set of five files of calibrated model output representing the transport of water within each of the five embayment systems. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 10-tidal cycle period in winter 1999. Each modeled scenario (e.g., present conditions, build-out) required the model be run for a 28-day spin-up period, to allow the model had reached a dynamic “steady state”, and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

Three primary nitrogen loads to sub-embayments are recognized in this modeling study: external loads from the watersheds, nitrogen load from direct rainfall on the embayment surface, and internal loads from the sediments. Additionally, there is a fourth load to the Ashumet Valley sub-embayments, 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 Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in 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 1989 and 2003) were available for stations monitored by SMAST in the Ashumet Valley systems.

Table VI-1. Pond-Watcher measured data, and modeled Nitrogen concentrations for the Ashumet Valley systems used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means.

Sub-Embayment	monitoring station	data mean	s.d. all data	N	model min	model max	model average
Coonemessett River (fresh water)	GT1	0.855	0.187	51	-	-	0.851
Coonemessett River (estuarine)	GT2	0.881	0.218	100	0.834	0.915	0.875
Great Pond - upper	GT3	0.739	0.221	105	0.709	0.854	0.782
Perch Pond	GT4	0.895	0.239	101	0.802	0.902	0.859
Great Pond - mid	GT5	0.644	0.189	104	0.508	0.648	0.591
Great Pond - lower	GT6	0.543	0.181	104	0.280	0.493	0.339
Backus Brook (fresh water)	G1	1.364	0.361	64	-	-	0.528
Green Pond - upper	G2	0.988	0.340	138	0.821	1.025	0.932
Green Pond - upper	G2a	0.927	0.270	134	0.666	0.956	0.792
Green Pond - mid	G3	0.750	0.222	138	0.553	0.750	0.642
Green Pond - mid	G4	0.540	0.140	136	0.431	0.652	0.526
Green Pond - lower	G5	0.440	0.133	210	0.346	0.503	0.409
Bournes Brook (fresh water)	B1	0.928	0.422	52	-	-	0.874
Bournes Pond - upper	B2	0.880	0.291	109	0.846	0.940	0.901
Bournes Pond - mid	B3	0.670	0.264	105	0.496	0.753	0.643
Bournes Pond - lower	B4	0.482	0.142	101	0.322	0.555	0.426
Israels Cove	B5	0.674	0.194	100	0.591	0.663	0.633
Bournes Pond - lower	B6	0.387	0.110	100	0.283	0.497	0.340
Vineyard Sound	VS	0.280	0.065	196	-	-	0.280

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the three Ashumet Valley estuary systems. 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 Great, Green and Bournes Pond embayments. 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 three Ashumet Valley systems.

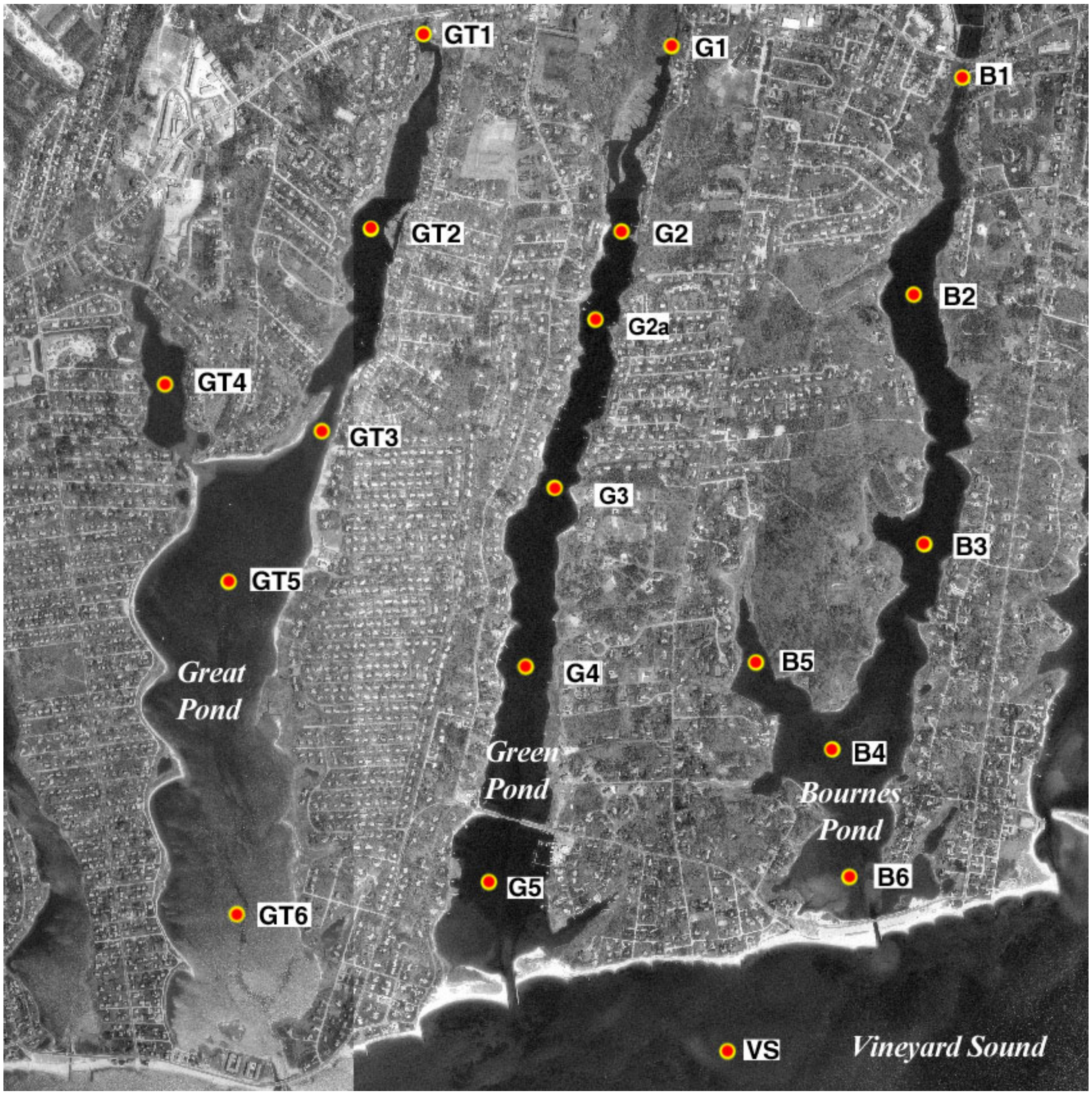


Figure VI-1. Estuarine water quality monitoring station locations in the Ashumet Valley estuary systems. Station labels correspond to those provided in Table VI-1.

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 Great, Green and Bournes Ponds.

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 three Ashumet Valley embayment systems also were used for the water quality constituent modeling portion of this study.

Based on measured flow rates from SMAST and groundwater recharge rates from the USGS, each hydrodynamic model was set-up to include the latest estimates of surface water flows from the Coonamessett River (to Great Pond), Backus Brook (to Green Pond) and Bournes Brook (to Bournes Pond). The Coonamessett River has a measure flow rate of 10.9 ft^3/sec (26,600 m^3/day), which is 2.1% of the volume of the average tide prism of the Great Pond. Backus Brook and Bournes Brook have average flows of 2.9 ft^3/sec and 1.5 ft^3/sec (7,200 m^3/day and 3,800 m^3/day) respectively. The Backus Brook discharge is 1.9% of the Green Pond tide prism, and the Bournes Brook flow is only 0.5% of the average prism in Bournes Pond.

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

VI.2.3 Boundary Condition Specification

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

The loadings used to model present conditions in the Ashumet Valley 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 sub-embayments (e.g., Green Pond) have almost twice the loading rate from benthic regeneration as from watershed loads. For other sub-embayments (e.g., Perch Pond), the benthic flux is relatively low or negative indicating a net uptake of nitrogen in the bottom sediments.

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary were specified. The model uses concentrations at the open boundary during the flooding tide periods of the model simulations. TN concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentration in Vineyard Sound was set at 0.280 mg/L, based on SMAST data from the Sound (station VS). 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 Ashumet Valley pond systems, with total watershed N loads, atmospheric N loads, and benthic flux. These loads represent **present loading conditions** for the listed sub-embayments.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux net (kg/day)
Great Pond	25.00	3.22	-0.27
Perch Pond	5.38	0.22	-1.39
Green Pond	18.55	1.61	55.60
Bournes Pond	9.61	1.61	28.45
Israels Cove	2.05	0.26	-0.32
Surface Water Sources			
Coonemessett River (Great Pond)	22.63	-	-
Backus Brook (Green Pond)	3.81	-	-
Bournes Brook (Bournes Pond)	3.29	-	-

VI.2.4 Model Calibration

Calibration of the three separate Ashumet Valley total nitrogen models 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 systems 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 Ashumet Valley embayment systems 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 the Ashumet Valley estuary systems.

Embayment Division	E m^2/sec
Great Pond Inlet	2.0
Great Pond	2.0
Perch Pond	0.10
Lower Coonamessett River	10.0
Coonamessett River	1.0
Green Pond Inlet	1.0
Lower Green Pond	100.0
Green Pond	15.0
Upper Green Pond	1.0
Backus Brook	0.10
Bournes Pond Inlet	5.0
Lower Bournes Pond	4.0
Israels Cove	0.15
Upper Bournes Pond	2.0
Bournes Brook	0.10

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 Pond-Watcher water-quality monitoring station. The calibration target would fall between the modeled mean and maximum TN because the monitoring data are collected, as a rule, during mid ebb tide.

Also presented in this figure are unity plot comparisons of measured data verses modeled target values for each system. Root mean squared (rms) errors are less than 0.08 mg/L for all three modeled systems. The model fits are exceptional for the Green and Bournes Ponds model, both with rms error of 0.04 mg/L or less and an R^2 correlation coefficient as high as 0.97.

Contour plots of calibrated model output are shown in Figures VI-3 through VI-5, for Great, Green and Bournes Ponds. In these figures, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full 5-tidal-day model simulation output period.

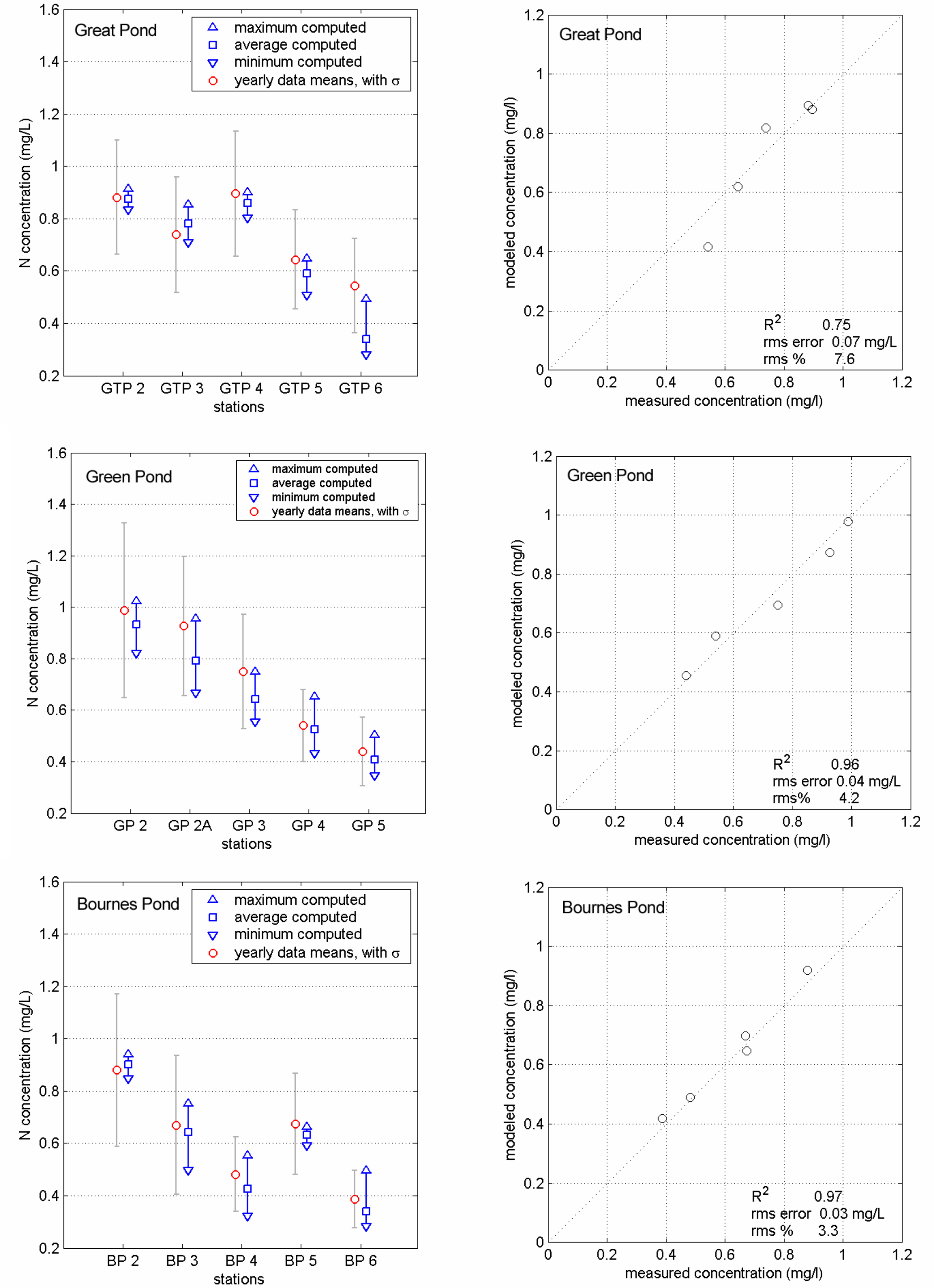


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Ashumet Valley systems. For the left plots, station labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed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

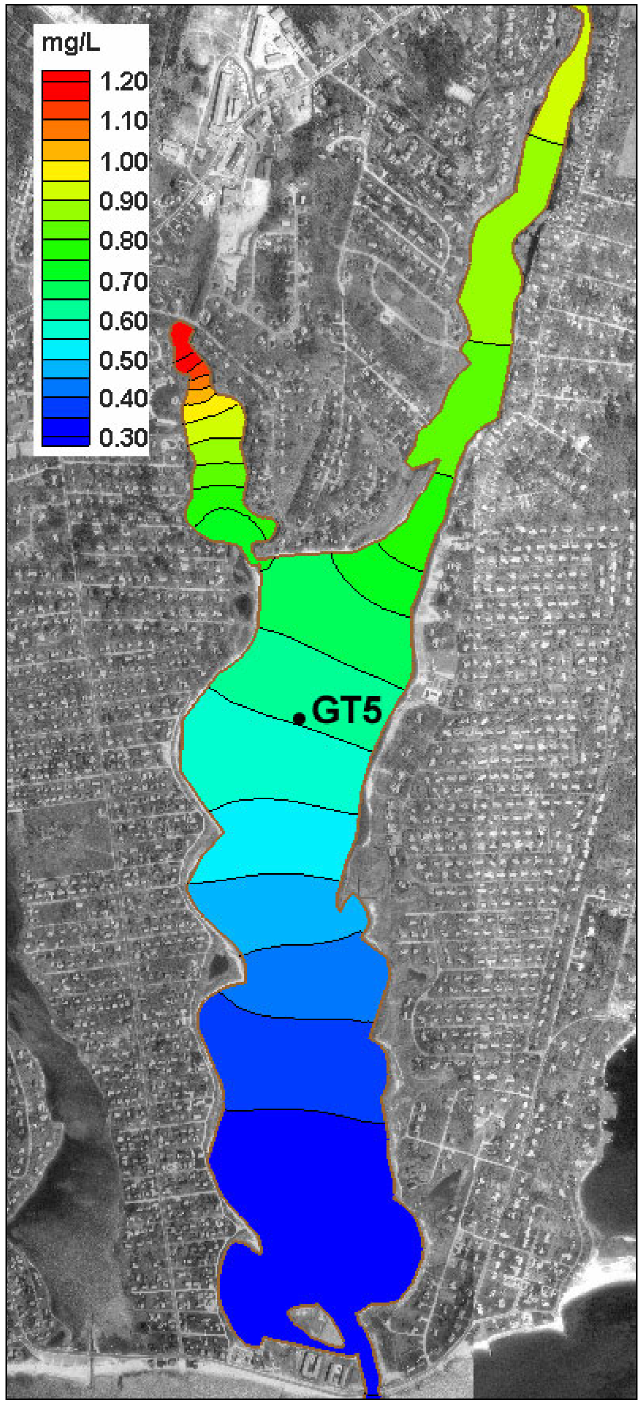


Figure VI-3. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Great Pond. The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

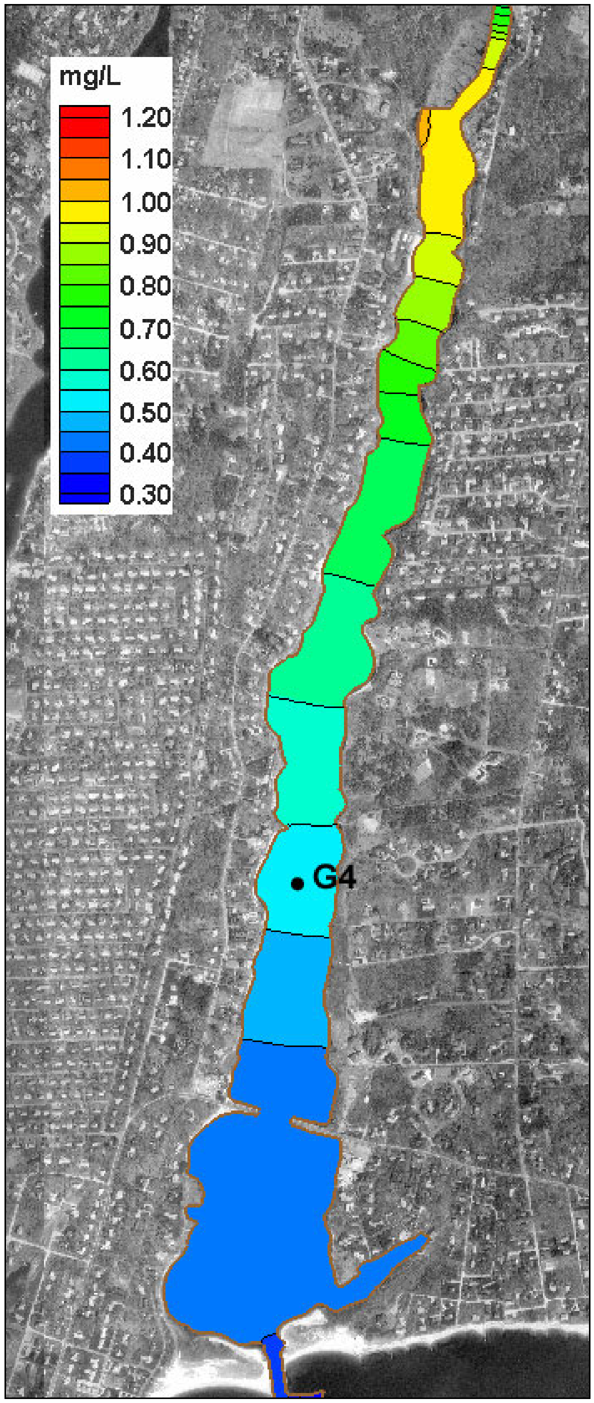


Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Green Pond. The approximate location of the sentinel threshold station for Green Pond (G4) is shown.

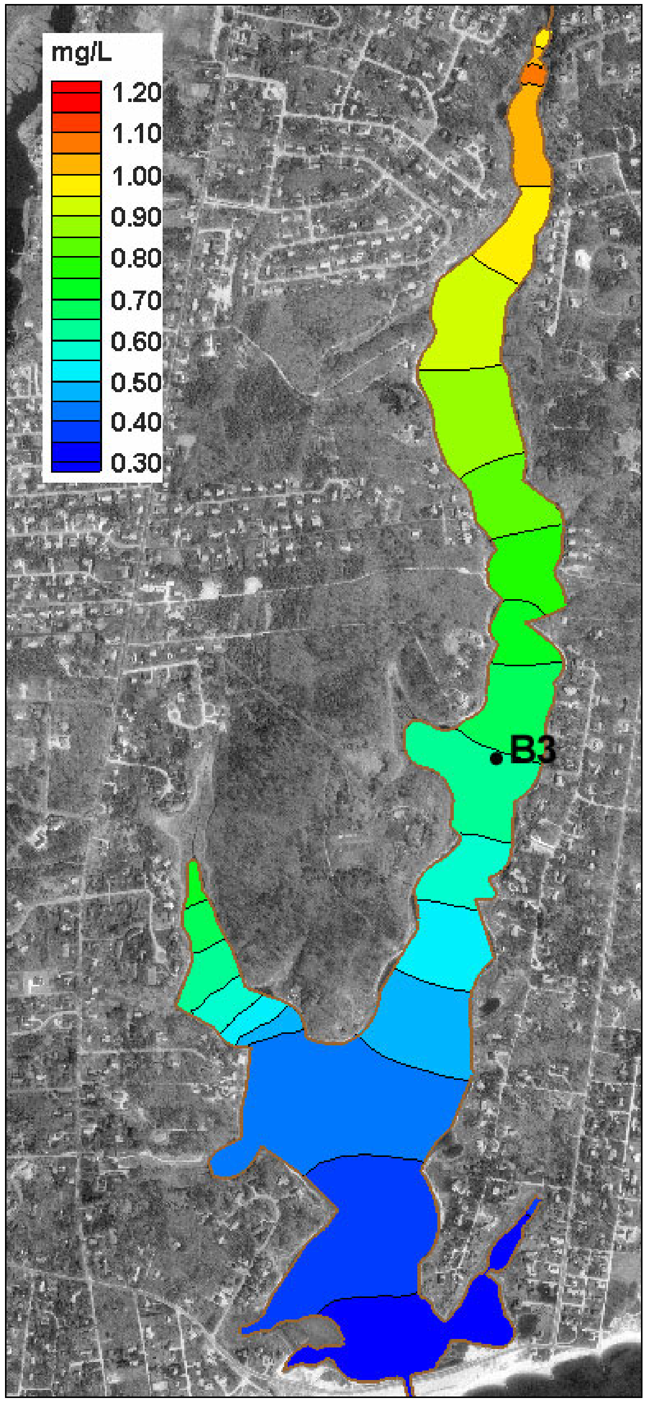


Figure VI-5. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Bournes Pond. The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

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 Ashumet Valley systems using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of each system, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary, at the freshwater stream discharges, and groundwater inputs. The open boundary salinity was set at 29.6 ppt. For surface water steams and groundwater inputs salinities were set at 0 ppt. Surface water stream flow rates for the streams were the same as those used for the total nitrogen model, as presented earlier in this section. Groundwater inputs used for each model were 2.46 ft^3/sec (6,000 m^3/day) for Bournes Pond, 2.31 ft^3/sec (5,700 m^3/day) for Green Pond, and 5.27 ft^3/sec (12,900 m^3/day) for Great Pond, with1.60 ft^3/sec (3,900 m^3/day) separately for Perch Pond. 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-6, with contour plots of model output shown in Figures VI-7 through VI-9 for each separate system. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model skillfully represents salinity gradients in the three Ashumet Valley estuary systems. The rms error of the three models is less than 1.5 ppt, and correlation coefficients are greater than 0.8. 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.

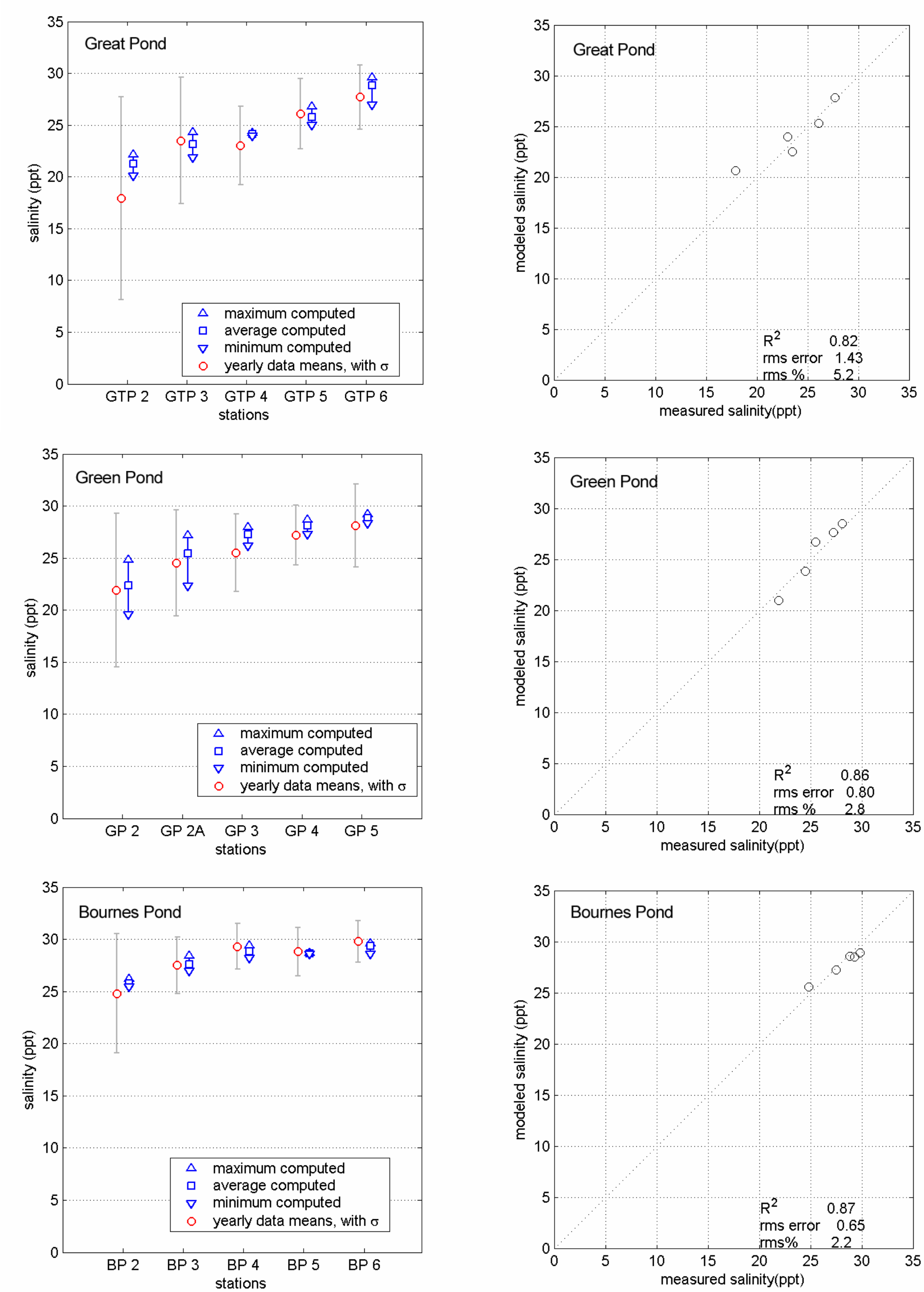


Figure VI-6. Comparison of measured and calibrated model output at stations in the Great Pond. For the left plots, stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed salinity for the same period (square markers). Measured data are presented as the total yearly mean at each station (circle markers), together with ranges that indicate ± one standard deviation of the entire dataset. For the plots to the right, model calibration target values are plotted against measured concentrations, together with the unity line. Computed correlation (R^2) and error (rms) for each model are also presented.

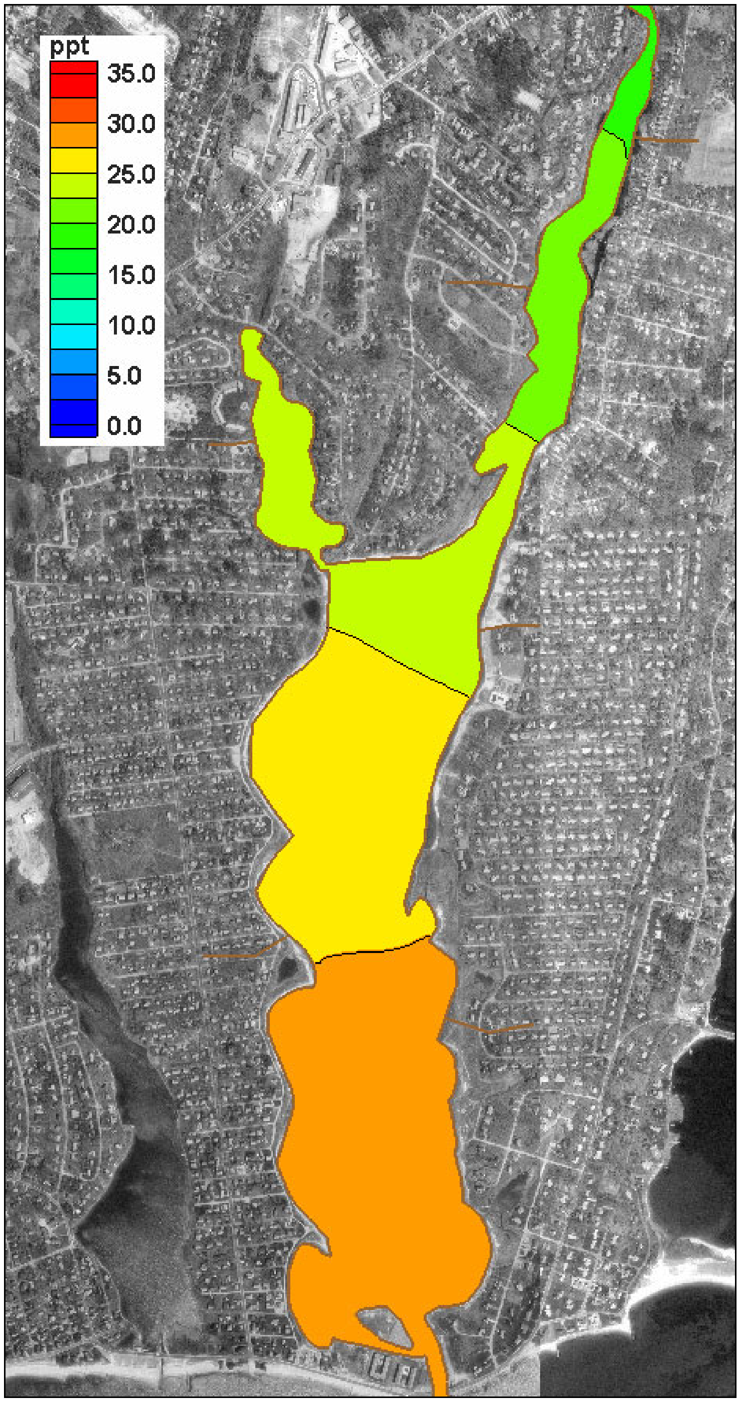


Figure VI-7. Contour Plot of modeled salinity (ppt) in Great Pond.

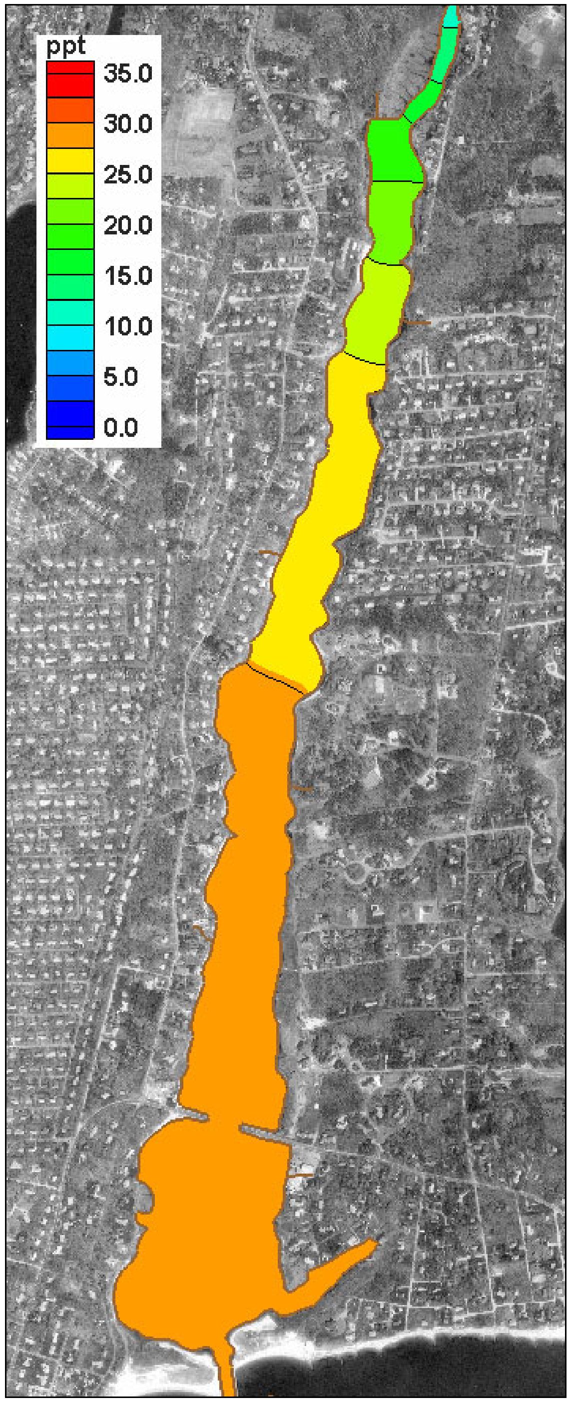


Figure VI-8. Contour Plot of modeled salinity (ppt) in Green Pond.

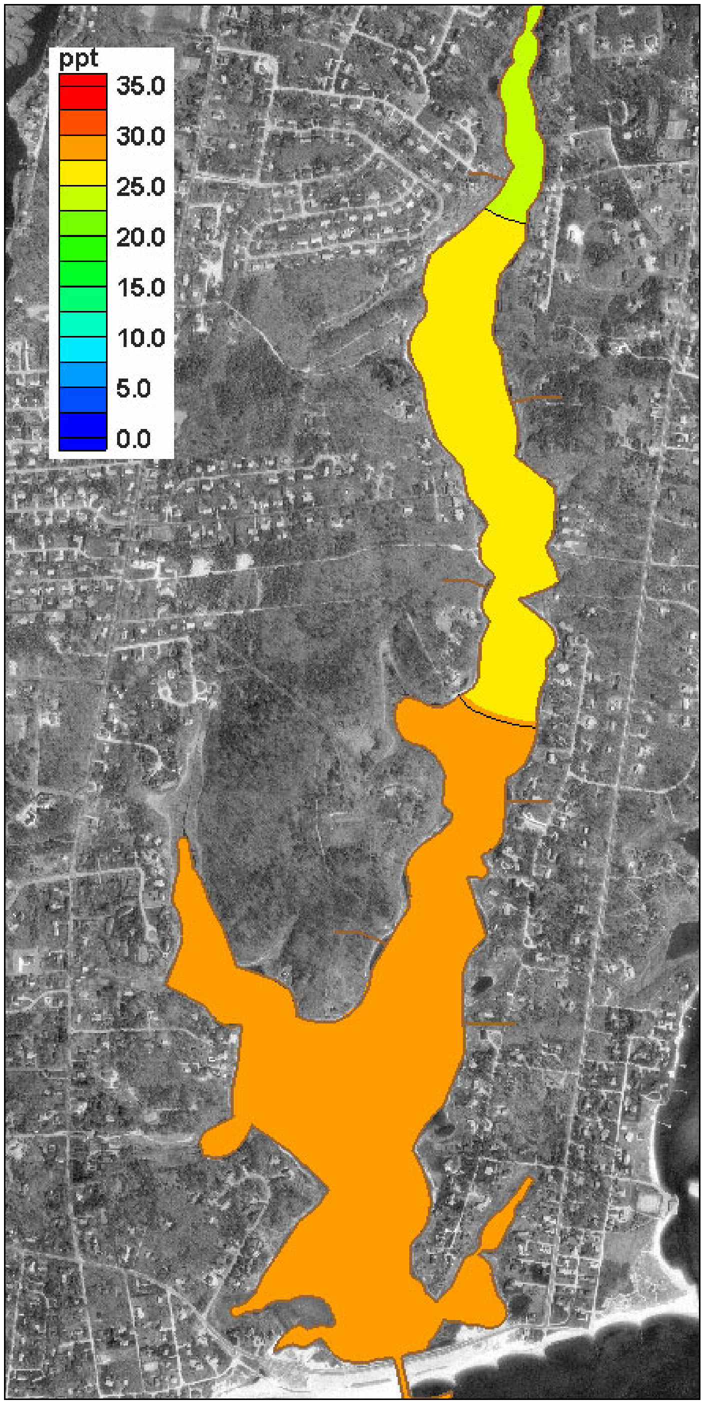


Figure VI-9. Contour Plot of modeled salinity (ppt) in Bournes Pond.

VI.2.6 Build-Out and No Anthropogenic Load Scenarios

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

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build-out, and no-anthropogenic ("no-load") loading scenarios of the Ashumet Valley systems. 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
Great Pond	25.00	26.85	+7.4%	2.66	-89.4%
Perch Pond	5.38	6.41	+19.3%	0.70	-87.0%
Green Pond	18.55	19.79	+6.7%	1.65	-91.1%
Bournes Pond	9.61	10.78	+12.2%	0.95	-90.2%
Israels Cove	2.05	2.19	+6.8%	0.21	-90.0%
Surface Water Sources					
Coonemessett River (Great Pond)	22.63	27.46	+21.3%	6.02	-73.4%
Backus Brook (Green Pond)	3.81	4.25	+11.7%	0.55	-85.6%
Bournes Brook (Bournes Pond)	3.29	4.33	+31.6%	0.54	-83.6%

VI.2.6.1 Build-Out

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be less than a 7% increase in watershed nitrogen load to the lower portion of the Green Pond as a result of potential future development. Other watershed areas would experience much greater load increases, for example the loads to Perch Pond and Bournes Brook would increase 22% and 31.6% respectively from the present day loading levels. A maximum increase in watershed loading resulting from future development would occur in the Coonamessett River, where the increase would be 4.83 kg/day, or 21% more than present conditions. For the no load scenarios, almost all of the load entering the watershed is removed; therefore, the load is generally lower than existing conditions by over 90%.

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

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Ashumet Valley systems, 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)
Great Pond	26.85	3.22	-0.39
Perch Pond	6.41	0.22	-1.55
Green Pond	19.79	1.61	59.3
Bournes Pond	10.78	1.61	30.36
Israels Cove	2.19	0.26	-0.36
Surface Water Sources			
Coonemessett River (Great Pond)	27.46	-	-
Backus Brook (Green Pond)	4.25	-	-
Bournes Brook (Bournes Pond)	4.33	-	-

Following development of the nitrogen loading estimates for the build-out scenario, the water quality models of each system were run to determine nitrogen concentrations within each sub-embayment (Table VI-6). Total nitrogen concentrations in the receiving waters (i.e., Vineyard Sound) remained identical to the existing conditions modeling scenarios. Total N concentrations increased the most in the upper portions of each system, with the largest change in Great Pond (13%) and the least change in Green Pond (6%). The small increase in the Green Pond build-out results emphasizes the fact that the combined upper and lower watershed loadings are already very close to build-out conditions. Color contours of model output for the build-out scenario are present in Figure VI-10 through VI-12. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figures VI-3 through VI-5, 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 Ashumet Valley systems. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Coonemessett River (estuarine)	GT2	0.875	0.990	+13.1%
Great Pond - upper	GT3	0.782	0.875	+12.0%
Perch Pond	GT4	0.859	1.017	+18.4%
Great Pond - mid	**GT5**	**0.591**	**0.648**	**+9.7%**
Great Pond - lower	GT6	0.339	0.349	+3.0%
Green Pond - upper	G2	0.932	0.985	+5.7%
Green Pond - upper	G2a	0.792	0.830	+4.8%
Green Pond - mid	G3	0.642	0.667	+3.9%
Green Pond - mid	**G4**	**0.526**	**0.543**	**+3.2%**
Green Pond - lower	G5	0.409	0.417	+2.1%
Bournes Pond - upper	B2	0.901	0.985	+9.3%
Bournes Pond - mid	**B3**	**0.643**	**0.685**	**+6.7%**
Bournes Pond - lower	B4	0.426	0.441	+3.6%
Israels Cove	B5	0.633	0.661	+4.3%
Bournes Pond - lower	B6	0.340	0.345	+1.7%

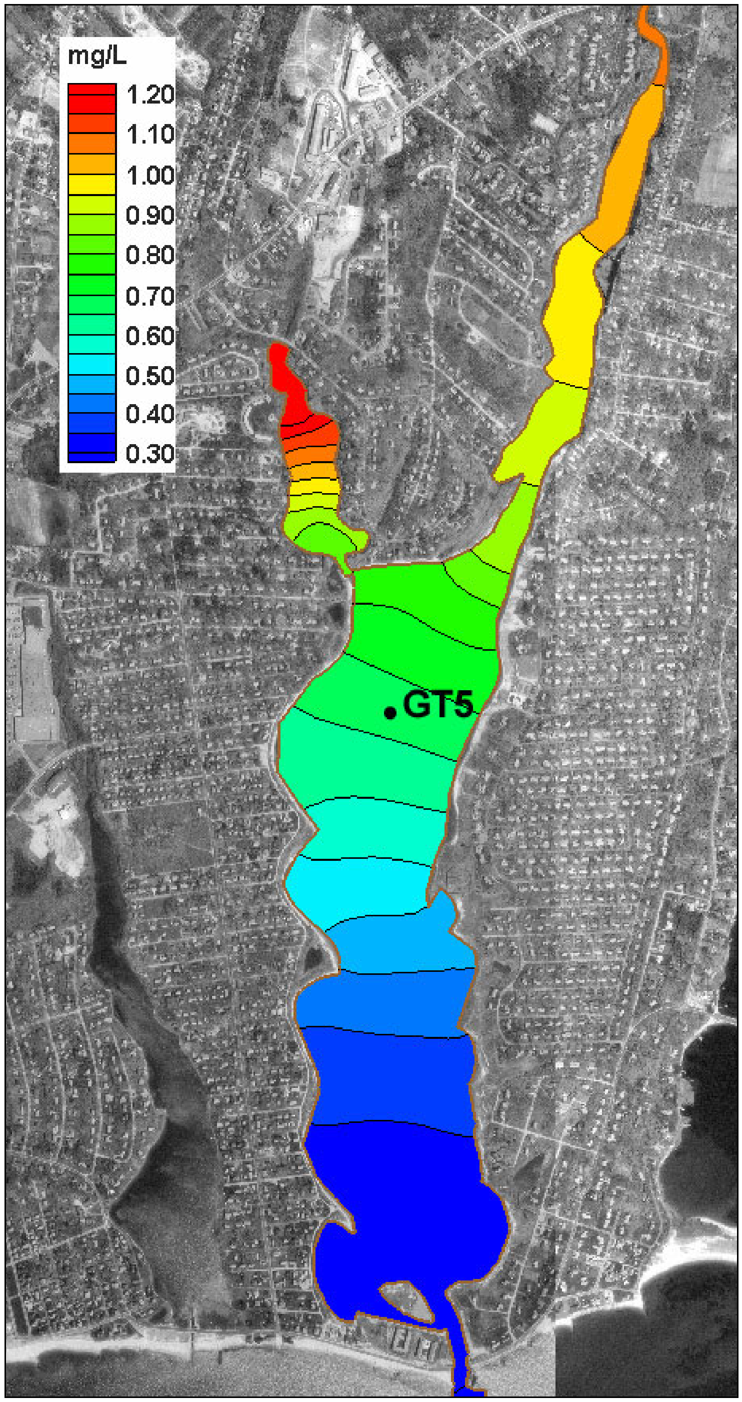


Figure VI-10. Contour plot of modeled total nitrogen concentrations (mg/L) in Great Pond, for projected build-out loading conditions. The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

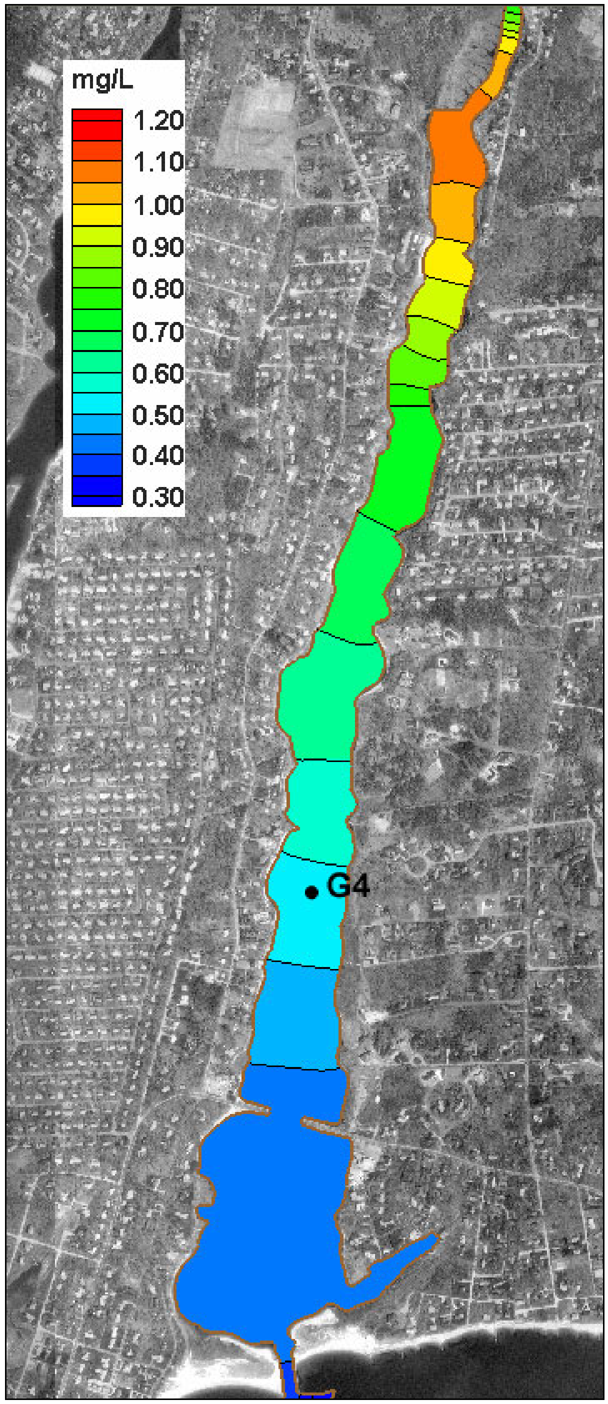


Figure VI-11. Contour plot of modeled total nitrogen concentrations (mg/L) in Green Pond, for projected build out loading conditions. The approximate location of the sentinel threshold station for Green Pond (G4) is shown.

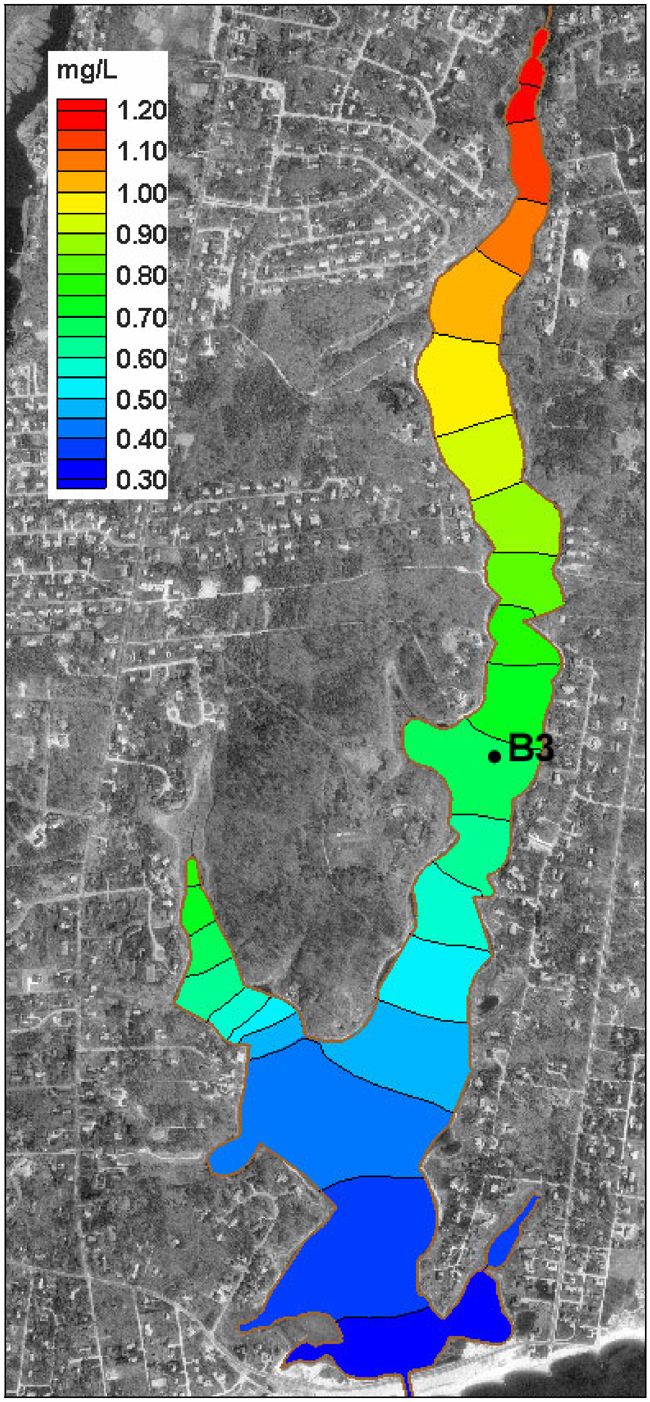


Figure VI-12. Contour plot of modeled total nitrogen concentrations (mg/L) in Bournes Pond, for projected build-out loading conditions. The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

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") scenarios is shown in Table VI-7. The benthic flux input to each embayment was reduced (toward zero) based on the reduction in the watershed load (as discussed in §VI.2.6.1). Compared to the modeled present conditions and build-out scenario, atmospheric deposition directly to each sub-embayment becomes a greater percentage of the total nitrogen load as the watershed load and related benthic flux decrease.

Table VI-7. "No anthropogenic loading" ("no load") sub-embayment and surface water loads used for total nitrogen modeling of the Ashumet Valley systems, 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)
Great Pond	2.66	3.22	0.40
Perch Pond	0.70	0.22	-0.49
Green Pond	1.65	1.61	20.47
Bournes Pond	0.95	1.61	17.6
Israels Cove	0.21	0.26	-0.13
Surface Water Sources			
Coonemessett River (Great Pond)	6.02	-	-
Backus Brook (Green Pond)	0.55	-	-
Bournes Brook (Bournes Pond)	0.54	-	-

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 45% occurring the upper portions of the systems. Results for each system are shown pictorially in Figures VI-13 through VI-15.

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 Ashumet Valley systems. 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
Coonemessett River (estuarine)	GT2	0.875	0.377	-57.0%
Great Pond - upper	GT3	0.782	0.366	-53.1%
Perch Pond	GT4	0.859	0.375	-56.3%
Great Pond - mid	**GT5**	**0.591**	**0.329**	**-44.3%**
Great Pond - lower	GT6	0.339	0.288	-14.9%
Green Pond - upper	G2	0.792	0.393	-57.9%
Green Pond - upper	G2a	0.642	0.404	-49.1%
Green Pond - mid	G3	0.526	0.383	-40.3%
Green Pond - mid	**G4**	**0.409**	**0.355**	**-32.4%**
Green Pond - lower	G5	0.792	0.323	-21.0%
Bournes Pond - upper	B2	0.901	0.485	-46.1%
Bournes Pond - mid	**B3**	**0.643**	**0.416**	**-35.3%**
Bournes Pond - lower	B4	0.426	0.342	-19.8%
Israels Cove	B5	0.633	0.402	-36.6%
Bournes Pond - lower	B6	0.340	0.307	-9.6%

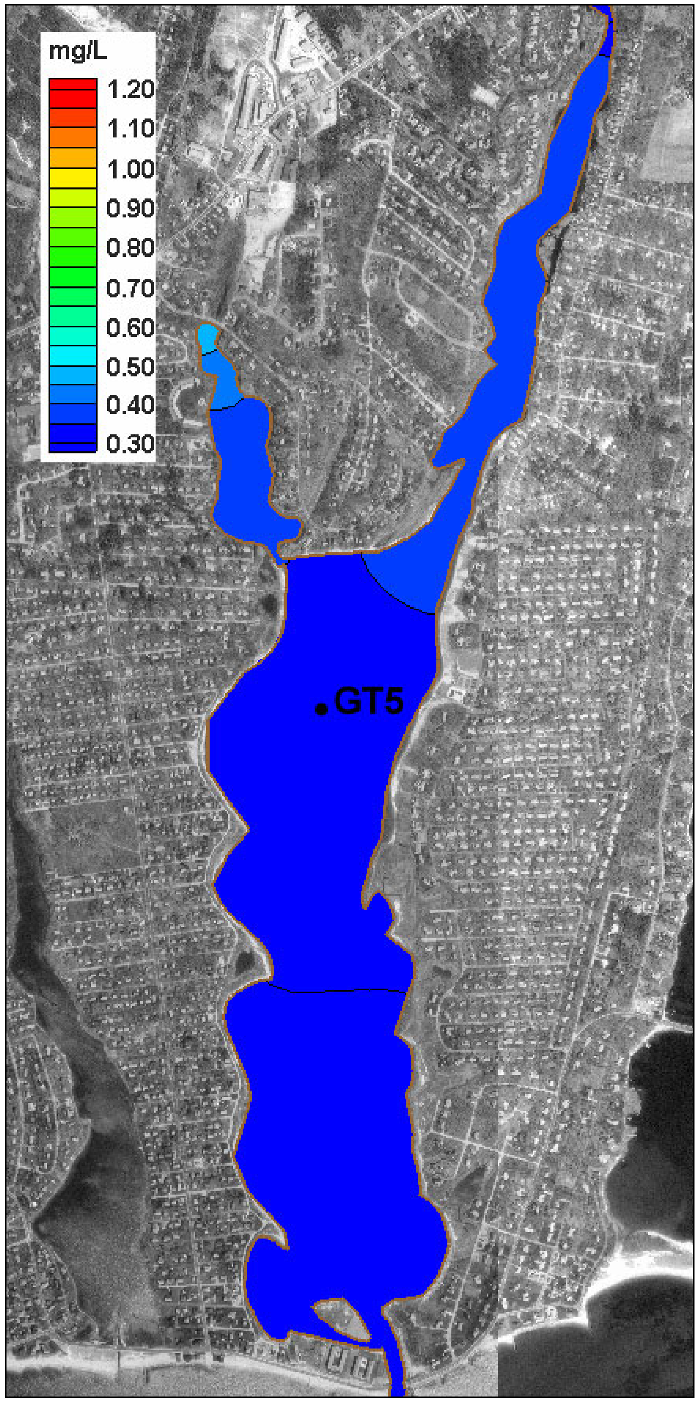


Figure VI-13. Contour plot of modeled total nitrogen concentrations (mg/L) in Great Pond, for no anthropogenic loading conditions. The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

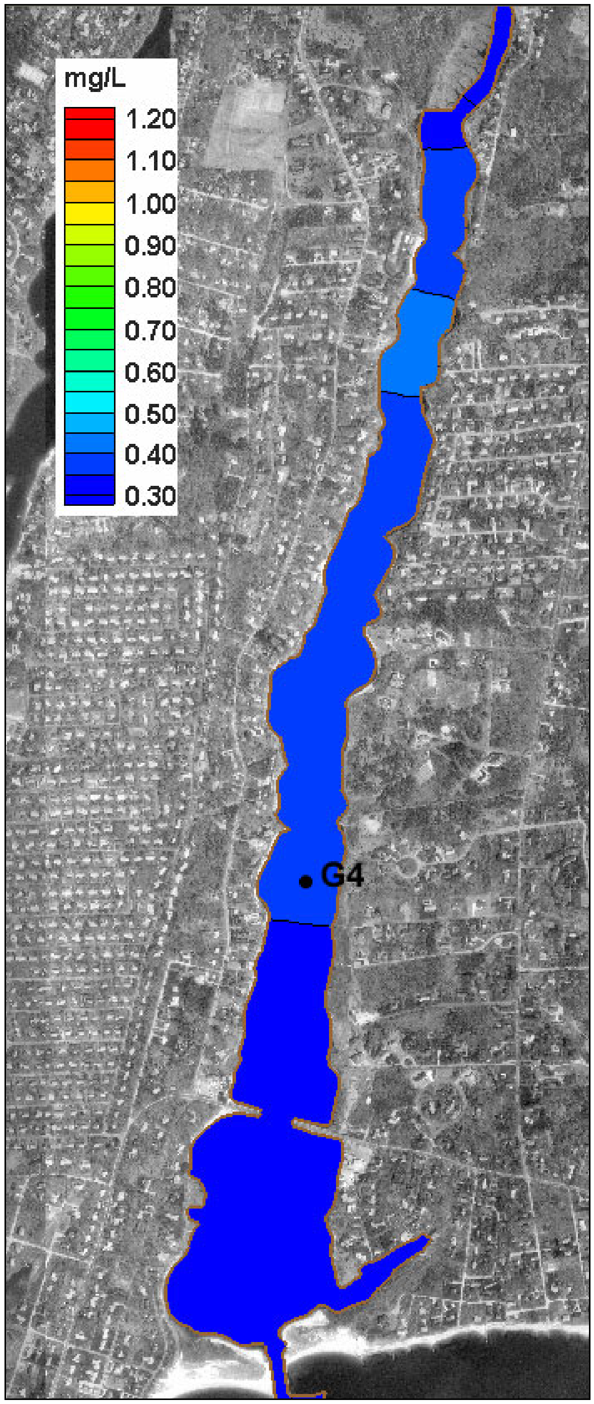


Figure VI-14. Contour plot of modeled total nitrogen concentrations (mg/L) in Green Pond, for no anthropogenic loading conditions. The approximate location of the sentinel threshold station for Green Pond (G4) is shown.

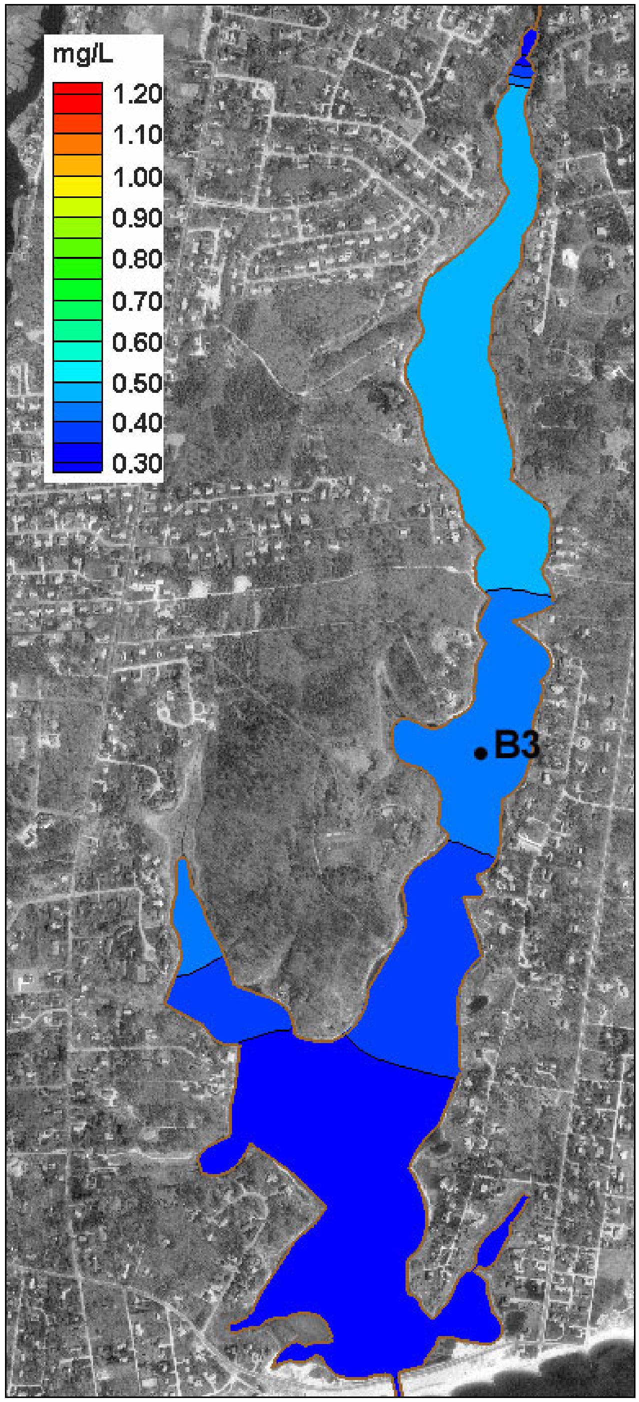


Figure VI-15. Contour plot of modeled total nitrogen concentrations (mg/L) in Bournes Pond, for no anthropogenic loading conditions. The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

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 Great, Green, and Bournes Pond embayment systems the MEP assessment is based upon data from the water quality monitoring database and MEP surveys of eelgrass distribution, benthic animal communities and sediment characteristics, and dissolved oxygen and chlorophyll a records during the summer of 2002. In addition to indicating each system's present health, these data, coupled with a full water quality synthesis and modeling effort, support the development of site-specific nitrogen thresholds for the management and restoration of nitrogen impaired regions of 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 that 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 portions of Great, Green and Bournes Ponds, as well as closer to the inlets in each embayment, to record the frequency and duration of low oxygen conditions during the critical summer period.

The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to shallow coastal embayments. Eelgrass is a fundamentally important species in the ecology of coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within each of the embayment systems 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 Great Pond, Green Pond, and Bournes Pond embayment systems, 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 stress indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes *et al.* 1997). These data are coupled with the density (number of individuals), level of diversity (H’), and evenness (E) of the benthic community 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 Great, Green and Bournes Ponds embayment systems 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, which consume oxygen from the water column, 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. 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 placed within key regions of the Great, Green, and Bournes Ponds systems (Figure VII-2a and 2b). The sensors (YSI 6600) were first calibrated in the laboratory and checked with standard oxygen mixtures. In addition, periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a 4 week minimum deployment within the interval from July through mid-September. All of the mooring data from the Great, Green and Bournes Ponds embayment systems was collected during 2002.

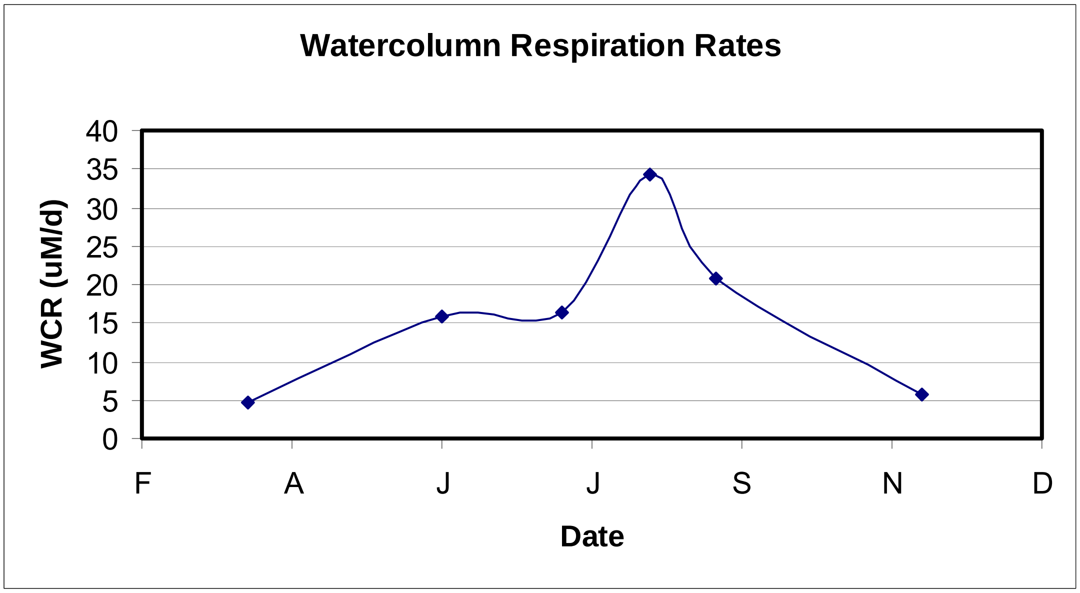


Figure VII-1. Average watercolumn respiration rates from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

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

Dissolved oxygen and chlorophyll a records were examined both for temporal trends and to determine the percent of the 25-28 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 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-15). 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 the upper tidal reaches of each estuary have the largest daily oxygen excursion, which further supports the assessment of a high degree of nutrient enrichment. The use of only the duration of oxygen below, for example 4 mg L^{-1}, can underestimate the level of habitat impairment in these locations. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae) production, oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems (generally ~7-8 mg L^{-1} at the mooring sites). The clear evidence of oxygen levels above atmospheric equilibration indicates that the upper tidal reaches of the Great Pond, Green Pond, and Bournes Pond Systems are eutrophic.

Figure VII-2a. Aerial Photograph of the Great / Perch Pond embayment system in Falmouth showing locations of Dissolved Oxygen mooring deployments conducted in Summer 2002.

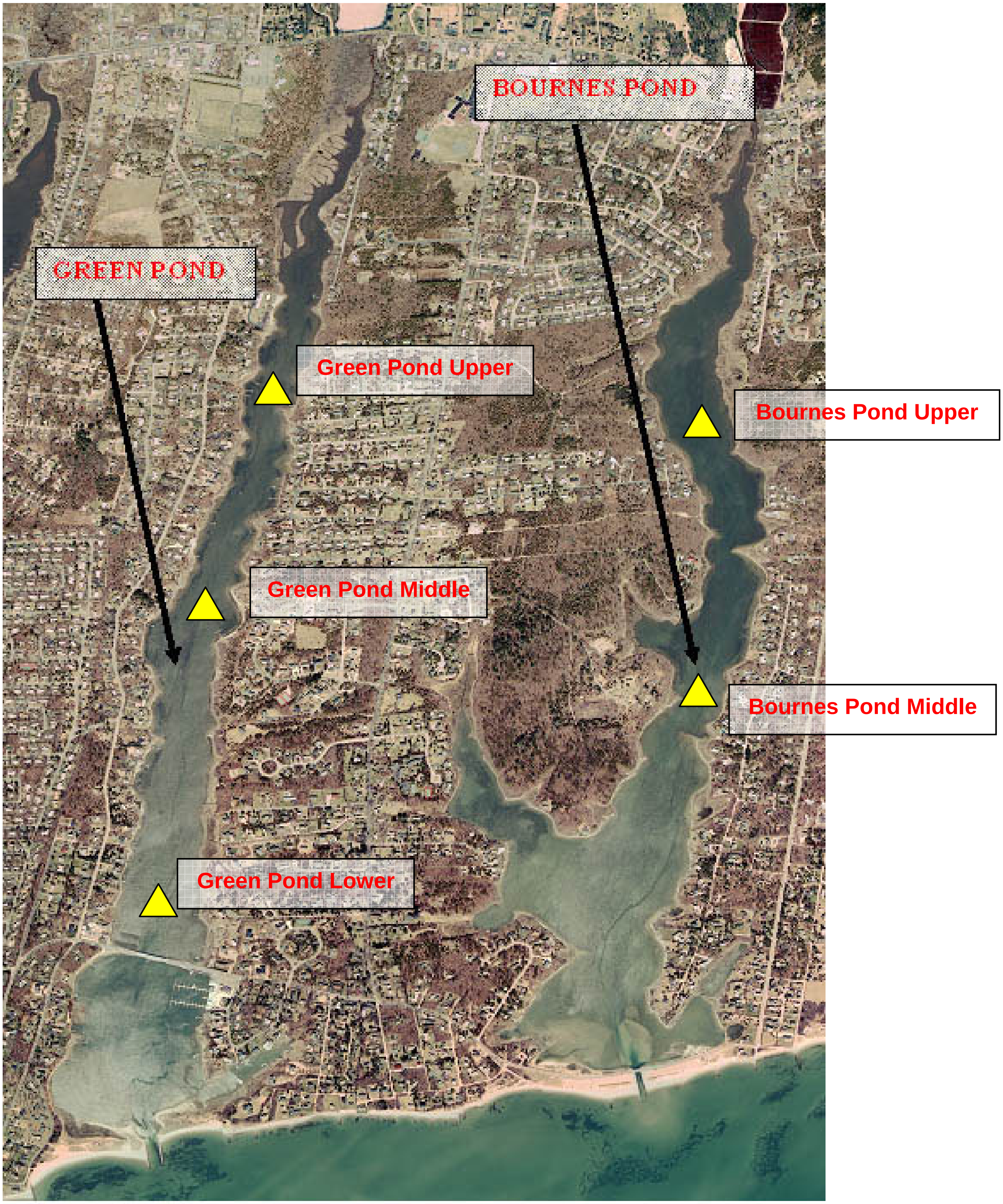


Figure VII-2b. Aerial Photograph of the Green and Bournes Pond systems in Falmouth showing locations of Dissolved Oxygen mooring deployments conducted in Summer 2002.

The dissolved oxygen records indicate that the upper regions of Great Green and Bournes Ponds 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, >25 µg/L 44%, 65% and 22% of the time, respectively (Table VII-2). Oxygen conditions and chlorophyll a levels improved in each system with decreasing distance to the tidal inlet, although all systems showed oxygen depletions below 5 mg L-1 and generally to <4 mg L^{-1} at the most southern stations measured in 2002. In all systems there was a clear gradient in chlorophyll a, with highest levels in the uppermost reaches and lowest levels near the tidal inlet to Vineyard Sound. The embayment specific results are as follows:

Great/Perch Pond (Figures VII-3,4,5,6): The "upper" mooring was placed one-third of the distance from Rt. 28 (upper extent of the tides) to the main Great Pond basin and the "lower" mooring in the middle of the upper half of the main basin (Figure VII-2a). Both the oxygen and chlorophyll a records from the upper mooring indicate a highly nitrogen enriched estuarine reach. Dissolved oxygen levels routinely declined below 2 mg L^{-1} coupled with very high chlorophyll a levels, in excess of 20 µg L^{-1} over 64% of the time and with 3 bloom periods in excess of 40 µg L^{-1}. These conditions are consistent with anecdotal reports of periodic summer fish kills within this upper tidal reach (reported in the Falmouth Enterprise and Falmouth PondWatch Reports). The lower mooring also showed conditions of nitrogen enrichment. This mooring was placed in the central region of the main basin, within the area that did not have eelgrass in the 1951 surveys (see Section VII-3, below). While chlorophyll a levels were significantly lower than in the upper tidal reach, only exceeding 15 µg L^{-1} only 20% of the deployment, oxygen conditions were still indicative of nitrogen related stress. This central basin condition is consistent with the infaunal community indicators, which show a significant stress in the central region with moderate quality habitat surrounding (see Section VII-4). The central basin differs from the surrounding larger basin area by having soft sediments and accumulations of drift macroalgae (*Cladophora, Ulva*) and a surface algal mat. The high frequency variation in the early August oxygen record is consistent with drift algal accumulations (D. Schlezinger, personal observation). An oxygen mooring was not placed in the Perch Pond basin as MEP Technical Staff had measured anoxic conditions (<0.5 mg L^{-1} at 0.5 meter depth) and high chlorophyll a (39.2 µg L^{-1}) in mid-August 2001. Previous investigations had found summer hypoxia/anoxia in this basin (Anderson and Hampson unpublished data).

Green Pond (Figures VII-7 through VII-12): The upper and middle sensors within Green Pond generally showed similar patterns of oxygen depletion with daily declines to below 5 mg L^{-1} and approaching or below 4 mg L^{-1} on more than one-third of the days. These conditions appear to be slightly better than would account for the anecdotal reports of periodic summer fish kills within this upper tidal reach (reported in the Falmouth Enterprise and Falmouth PondWatch Reports). However, since these events are infrequent, it is likely that interannual variation is the cause. This is supported by Falmouth PondWatch reports of hypoxia in the upper reaches of Green Pond (Howes and Goehringer, 1996). The lower station showed generally less oxygen depletion with all records >4 mg L^{-1} and only brief depletions below 5 mg L^{-1} (2% of total time). The spatial distribution of oxygen depletion was consistent with the pattern of phytoplankton biomass. Chlorophyll a levels within the upper and middle regions were similar and high (~20 µg L^{-1}) with a large bloom in early August (>60 µg L^{-1}). The lower station supported significantly less chlorophyll a, with July levels (~10 µg L^{-1}), half that of the upper stations, and smaller bloom levels. The mooring data indicates a system which is significantly nitrogen impaired throughout its upper half, based primarily upon the very high chlorophyll a levels and periodic oxygen declines. In contrast, the lower mooring indicated healthier conditions (moderately impaired/significantly impaired) based upon both the level and duration of observed oxygen depletion and chlorophyll a levels. The infaunal surveys were consistent with nitrogen enriched conditions within the upper and middle regions, but indicated poorer conditions (significant impairment) in the region of the lower mooring.

Bournes Pond (Figures VII-13, 14, 15): Sensors were placed in the upper reach of Bournes Pond (Rt. 28 to the main basin), as well as at one-third ("upper") and two-thirds ("middle") of the length of this estuarine reach (Figure VII-2b). The moorings indicated a nitrogen enriched system based upon both the dissolved oxygen and chlorophyll a records. However, there is a steep gradient in nitrogen related habitat impairment along this upper estuarine reach of Bournes Pond. The upper mooring found large oxygen depletions (to 4 and 3 mg L^{-1}) over long periods, 47% and 34% of deployment, respectively. The level of oxygen depletion was consistent with the high phytoplankton biomass, which was >20 µg L^{-1} for 32% of the deployment. However, simultaneous readings at the middle mooring location showed a much reduced level of nitrogen related habitat impairment. There is a strong gradient in habitat quality along this reach of the estuary, with oxygen levels within the lower third showing only brief excursions below 2 mg L^{-1} on 3 of 27 days sampled. Similarly, the lower mooring found that only 9% of the time was dissolved oxygen levels below 4 mg L^{-1}, compared to 47% of time at the upper mooring. Nevertheless, the mooring data indicate that the whole of the upper estuarine reach is severely degraded/significantly impaired by nitrogen enrichment. The gradients in the upper tidal reach of Bournes Pond are consistent with higher quality habitat in the lower main basin, as evidenced by eelgrass and infaunal communities (see Sections VII-3 and VII-4, below).

Based upon the available dissolved oxygen and chlorophyll data the ranking of the more sensitive regions of the Great Pond, Green Pond and Bournes Pond Estuaries is as follows:

- Great Pond:
  - Upper Reach – significantly impaired to severely degraded
  - Perch Pond – significantly impaired to severely degraded
  - Upper Central region of main basin – moderately to significantly impaired

- Green Pond:
  - Upper Region -- significantly impaired to severely degraded
  - Middle Region – moderately to significantly impaired

- Bournes Pond:

  Upper Region – significantly impaired to severely degraded

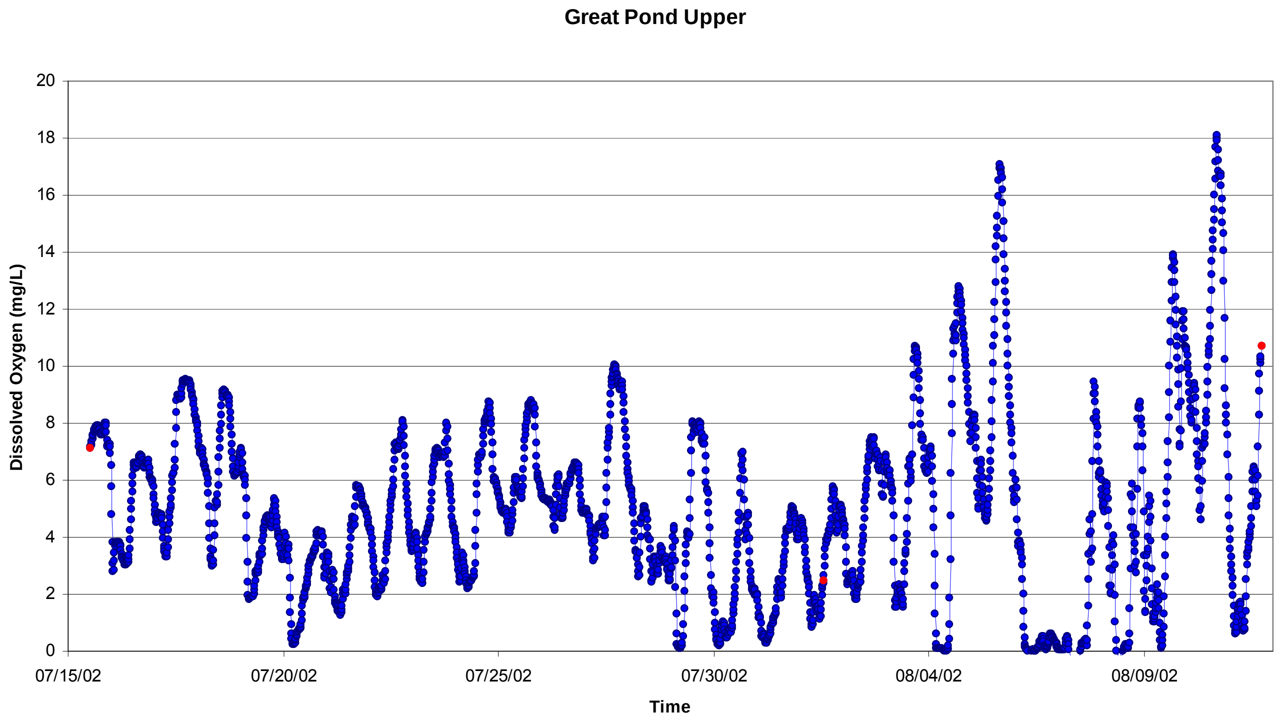


Figure VII-3. Bottom water record of dissolved oxygen at Great Pond Upper station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

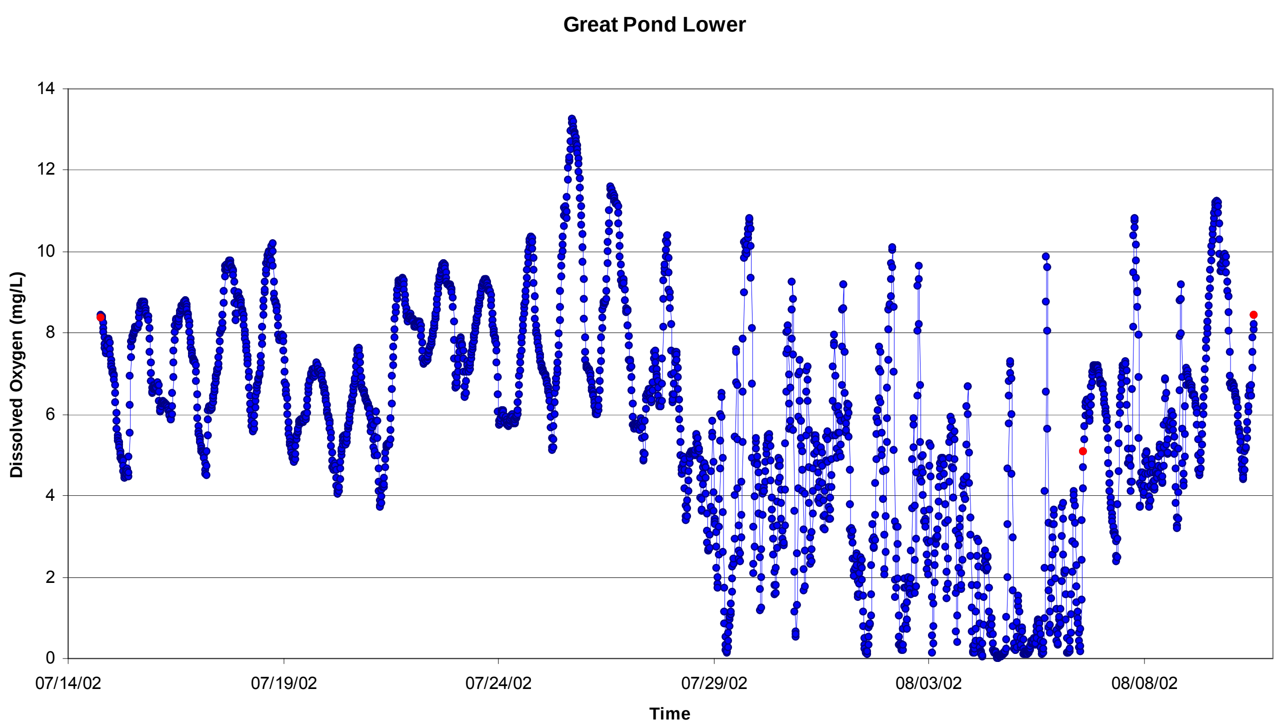


Figure VII-4. Bottom water record of dissolved oxygen at Great Pond Lower station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

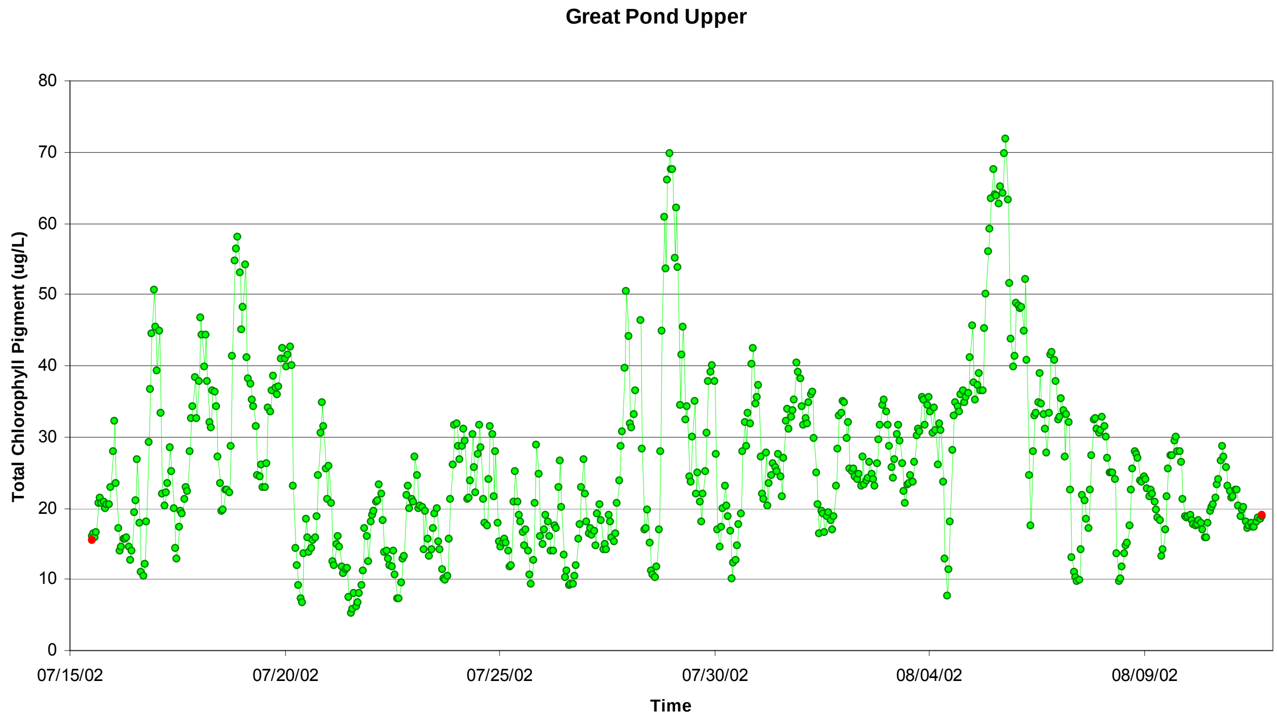


Figure VII-5. Bottom water record of chlorophyll-*a* at Great Pond Upper station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

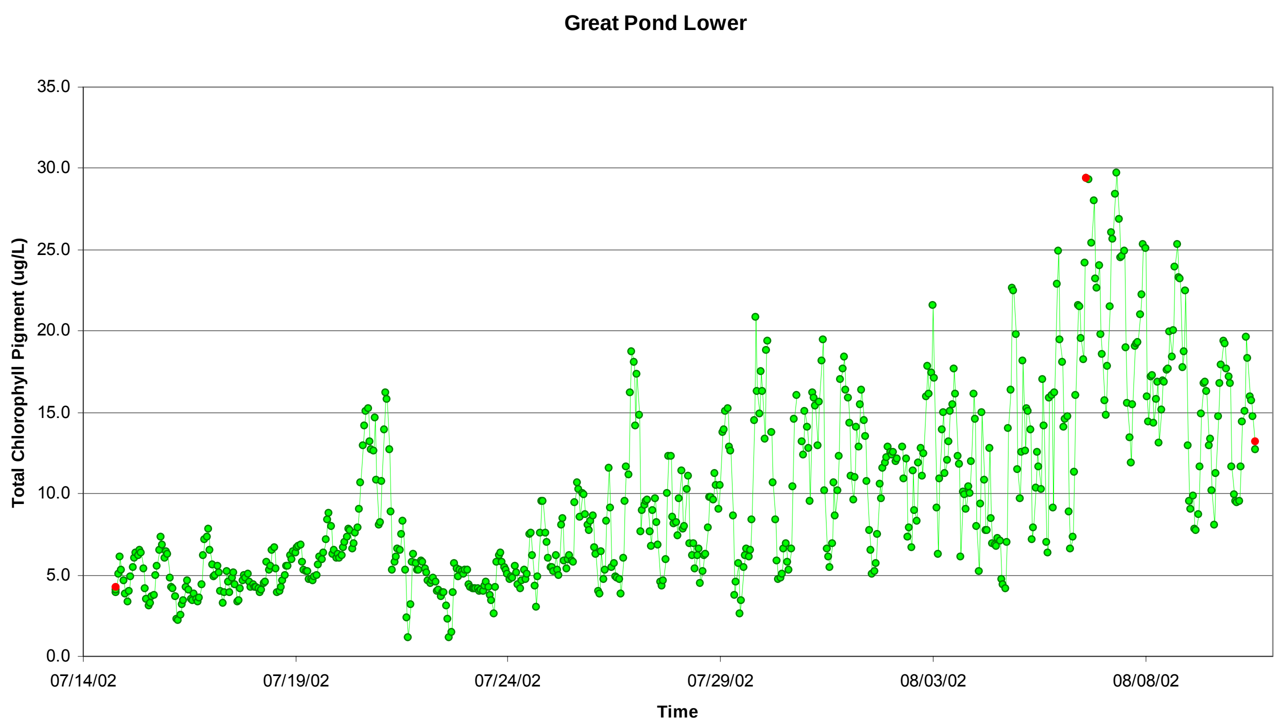


Figure VII-6. Bottom water record of chlorophyll-*a* at Great Pond Lower station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

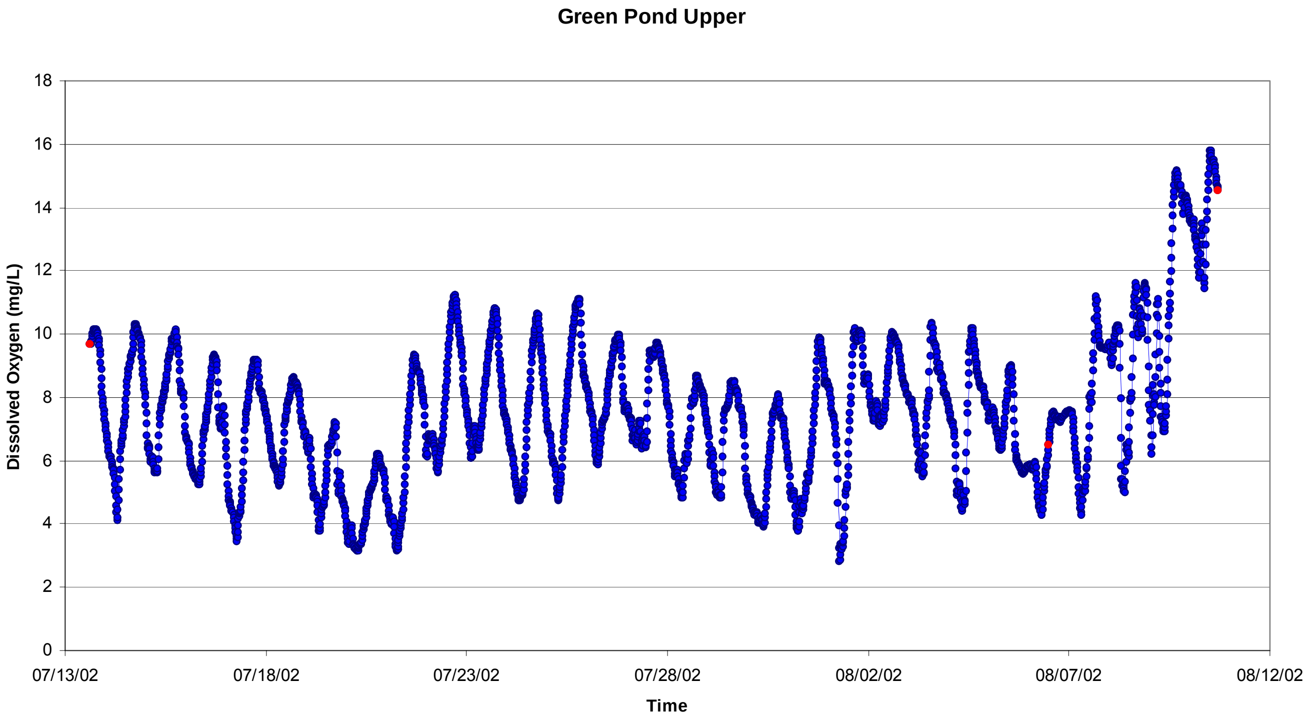


Figure VII-7. Bottom water record of dissolved oxygen at the Green Pond Upper station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

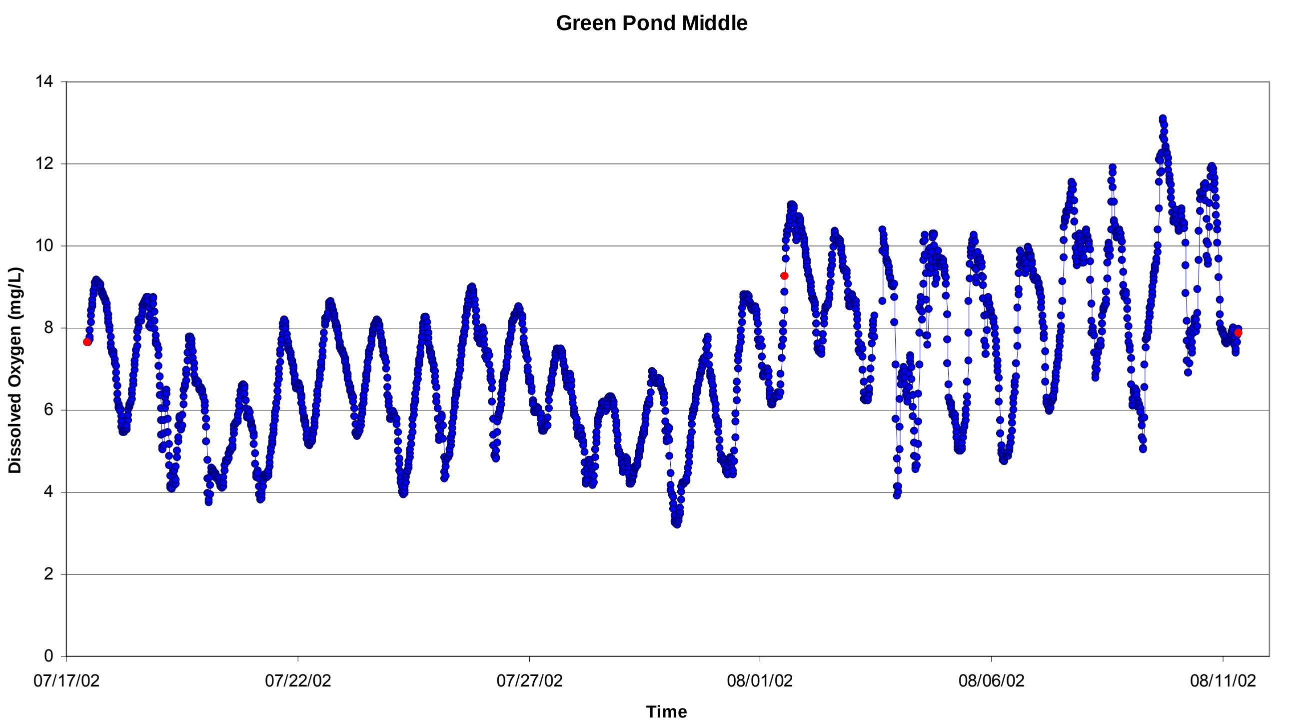


Figure VII-8. Bottom water record of dissolved oxygen at the Green Pond Middle station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

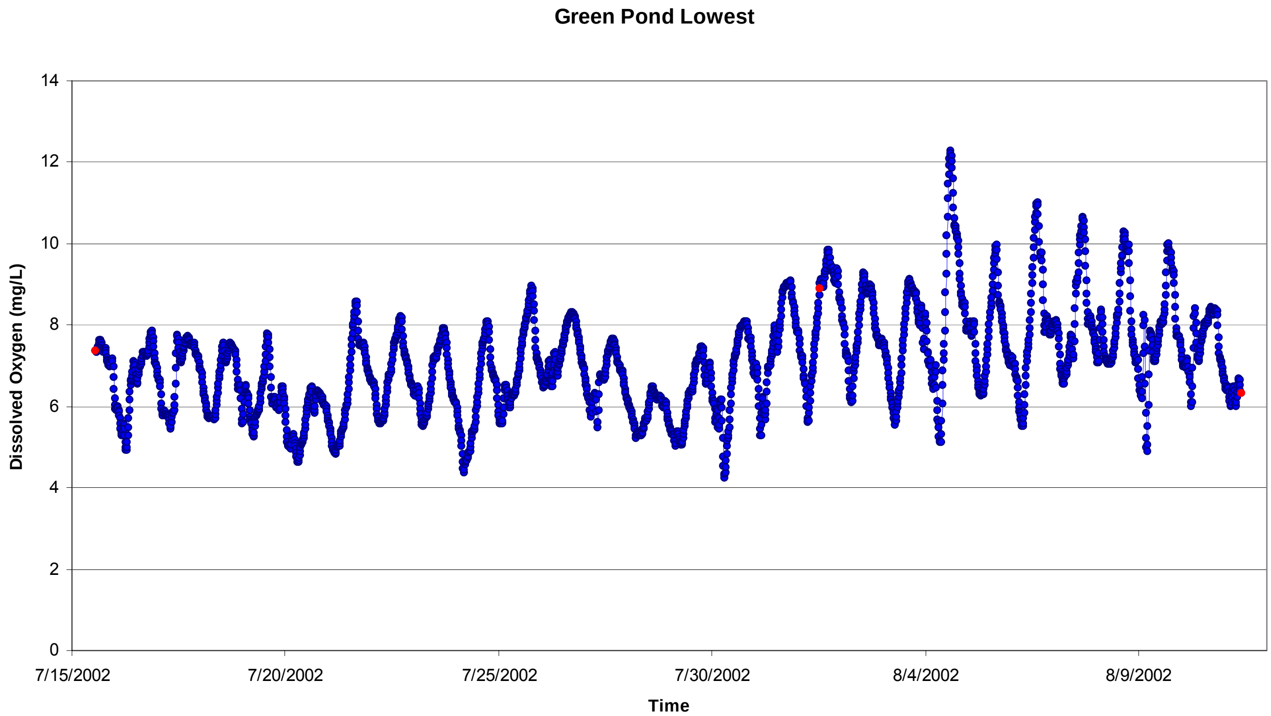


Figure VII-9. Bottom water record of dissolved oxygen at the Green Pond Lower station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

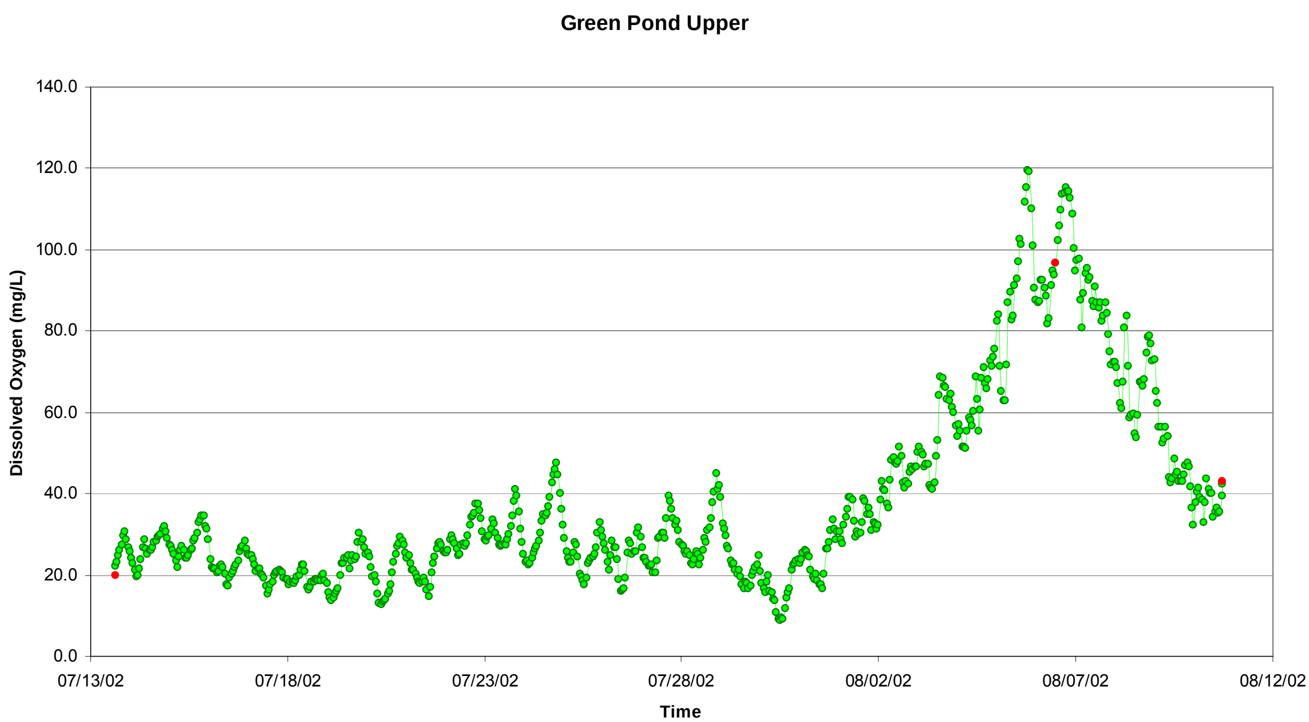


Figure VII-10. Bottom water record of chlorophyll-*a* at the Green Pond Upper station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

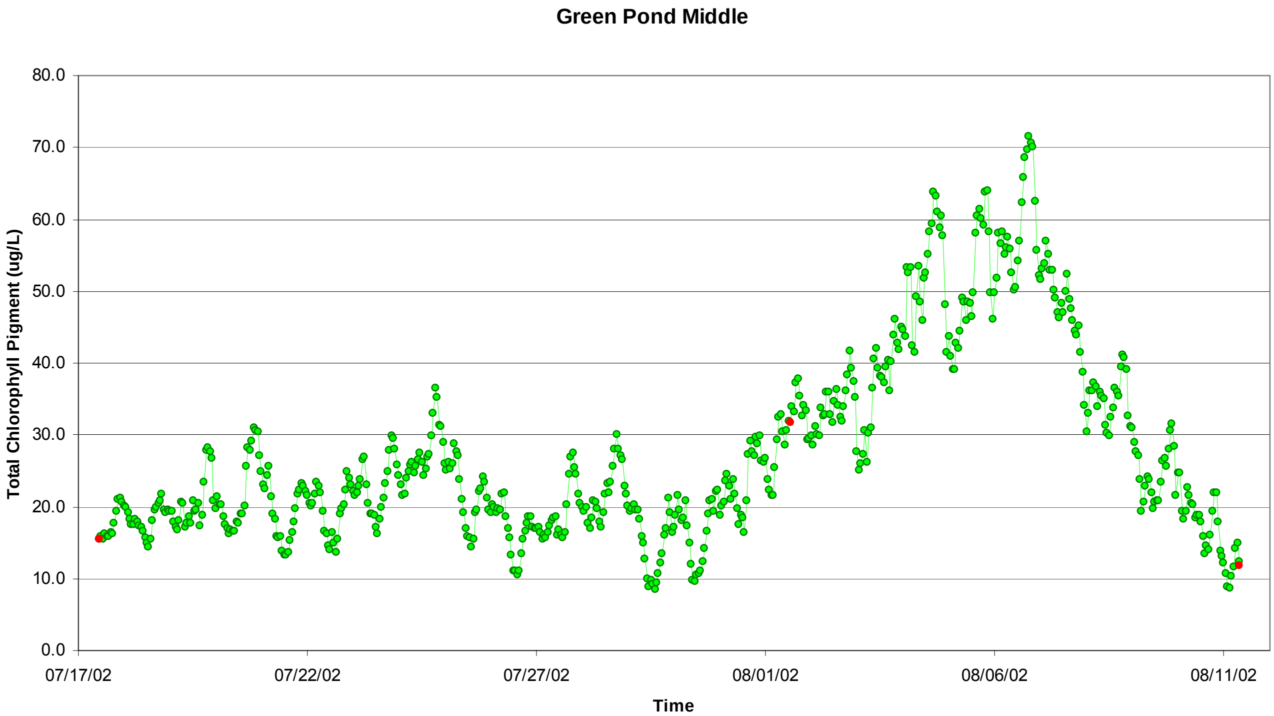


Figure VII-11. Bottom water record of chlorophyll-*a* at the Green Pond Middle station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

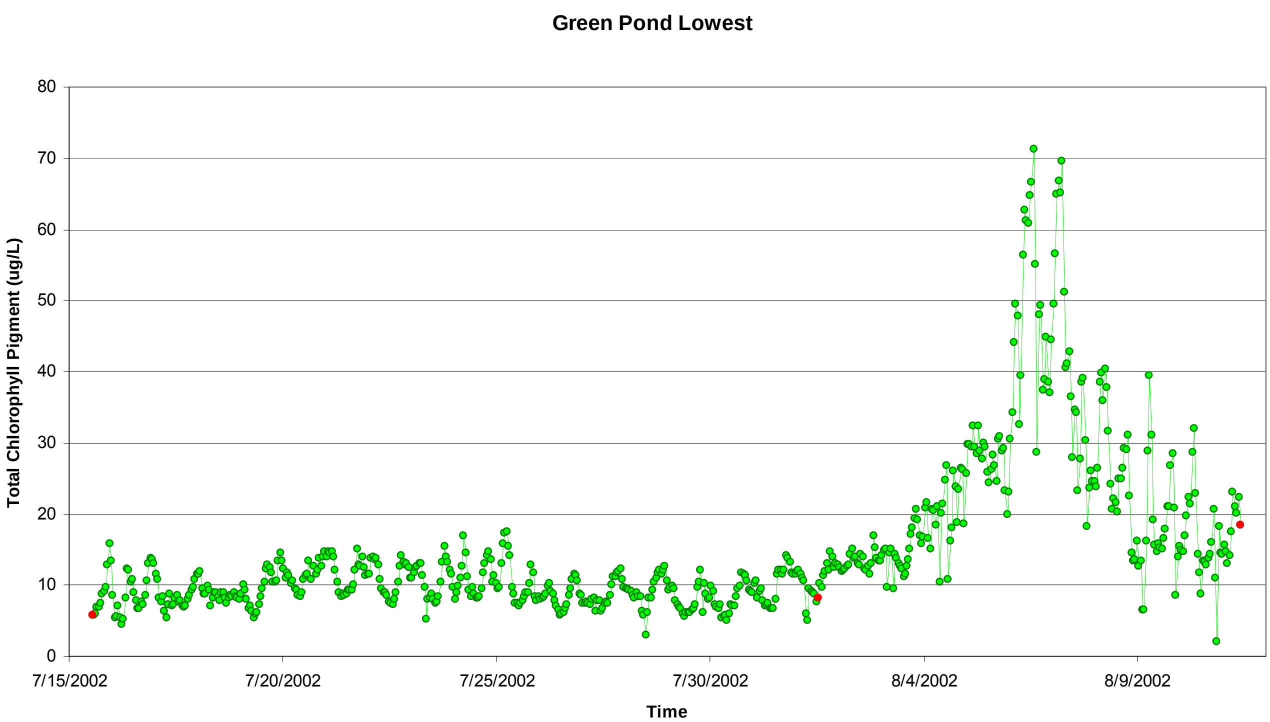


Figure VII-12. Bottom water record of chlorophyll-*a* at the Green Pond Lower station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

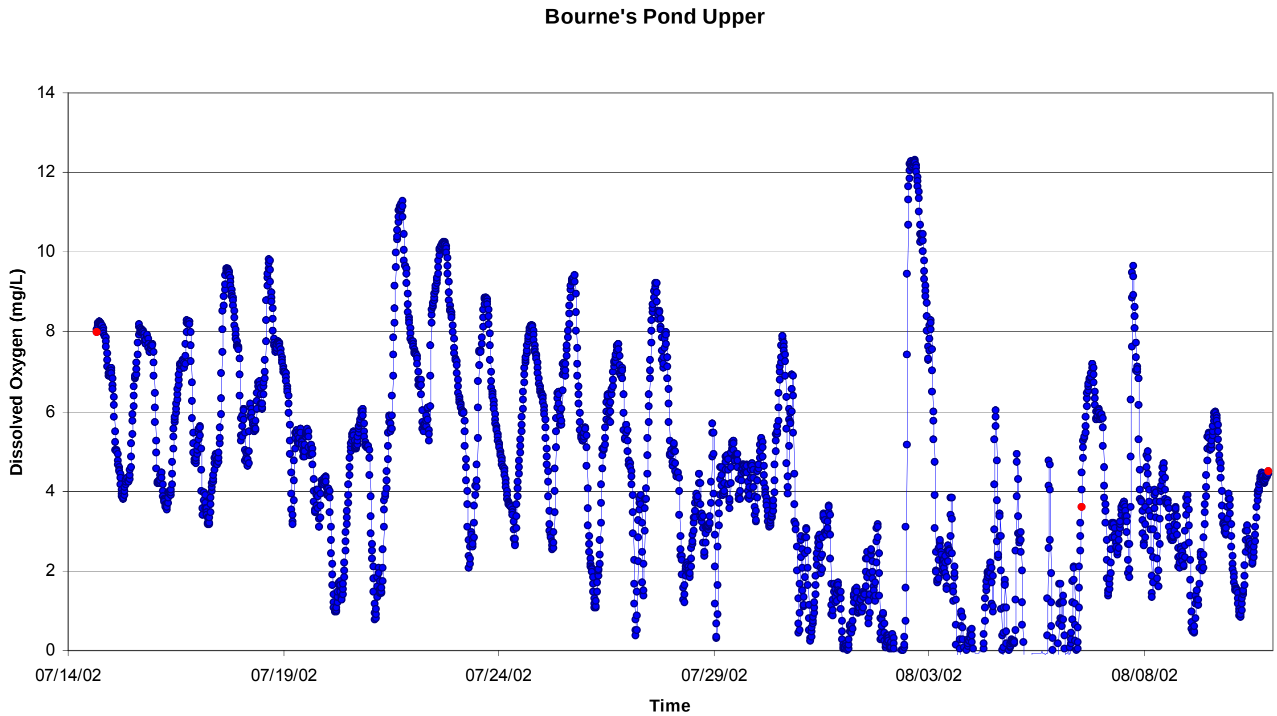


Figure VII-13. Bottom water record of dissolved oxygen at the Bournes Pond Upper station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

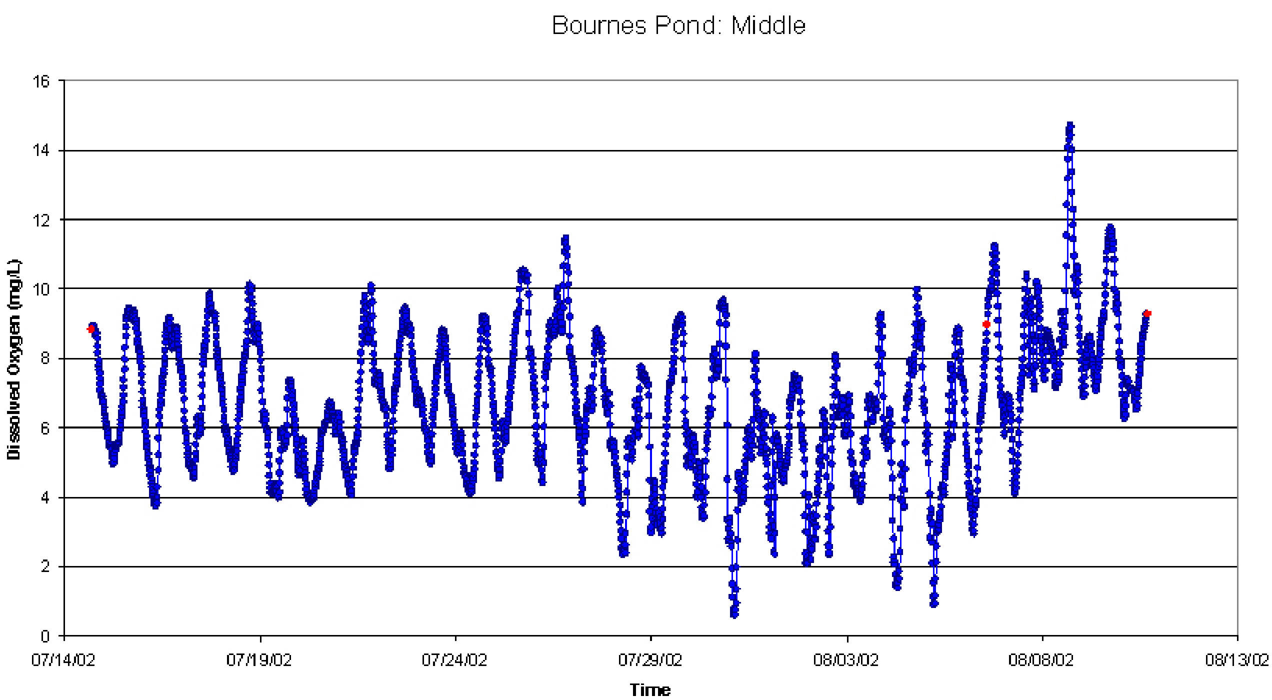


Figure VII-14. Bottom water record of dissolved oxygen at the Bournes Pond Middle station, within the upper estuarine reach, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

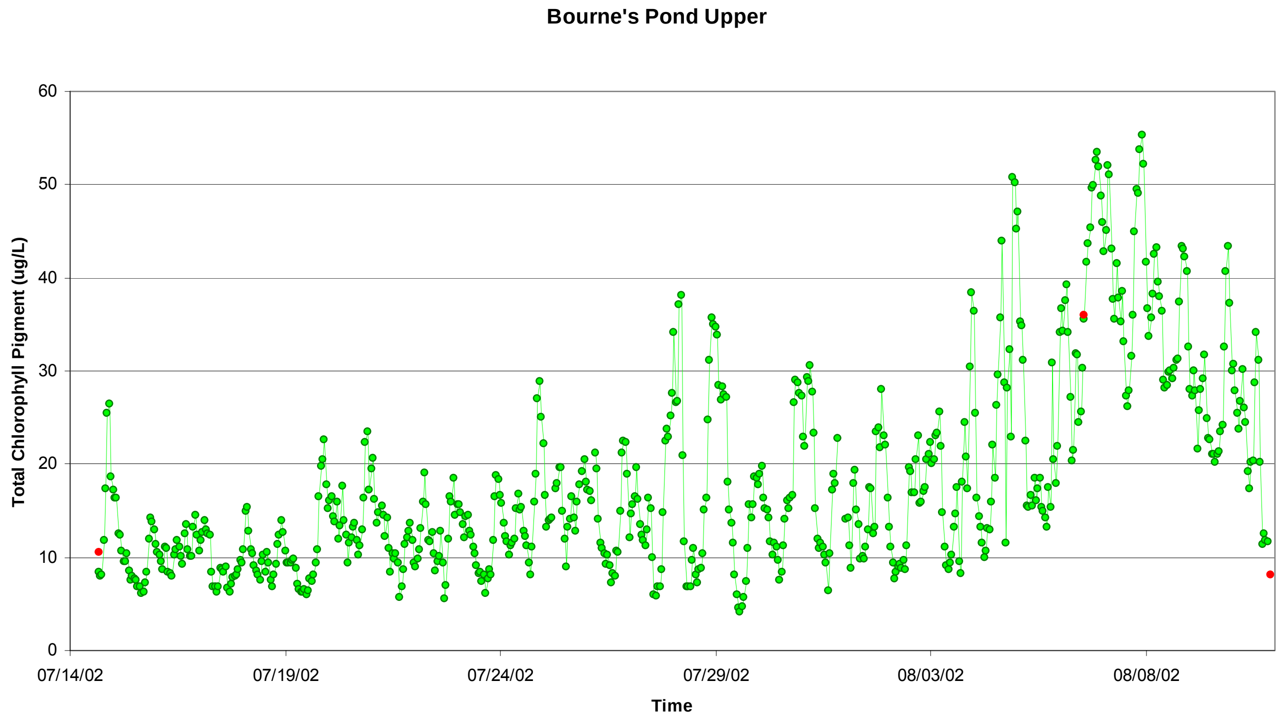


Figure VII-15. Bottom water record of chlorophyll-*a* at the Bournes Pond Upper station, Falmouth, MA., Summer 2002. Calibration samples represented as red dots.

Table VII-1. Percent of time during deployment that bottom water oxygen levels recorded by the *in situ* sensors were below various benchmark oxygen levels.

Embayment System	Start Date	End Date	Total Deployment (Days)	<6 mg/L Duration (Days)	<5 mg/L Duration (Days)	<4 mg/L Duration (Days)	<3 mg/L Duration (Days)
Great Pond							
Upper	7/15/2002	8/11/2002	27.2	66%	55%	41%	29%
Lower	7/15/2002	8/10/2002	26.8	48%	33%	22%	17%
Green Pond							
Upper	7/13/2002	8/10/2002	28.1	27%	13%	4%	0%
Middle	7/17/2002	8/11/2002	24.9	29%	13%	2%	0%
Lower	7/15/2002	8/11/2002	26.8	20%	2%	0%	0%
Bournes Pond							
Upper	7/14/2002	8/10/2002	27.0	73%	61%	47%	34%
Middle	7/14/2002	8/10/2002	27.0	39%	22%	9%	4%

Table VII-2. Duration (% of deployment time) that chlorophyll a levels exceed various benchmark levels within the Great / Perch, Green and Bournes Pond embayment systems. "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)
Great Pond								
Upper	7/15/2002	8/11/2002	27.2	**100%**	**96%**	**84%**	**64%**	**44%**
		Mean		13.58	2.38	0.73	0.53	0.29
		S.D.		1.30	2.04	1.06	0.61	0.34
Lower	7/15/2002	8/10/2002	26.8	**78%**	**39%**	**20%**	**5%**	**2%**
		Mean		0.62	0.26	0.15	0.12	0.10
		S.D.		0.87	0.36	0.14	0.10	0.06
Green Pond								
Upper	7/13/2002	8/10/2002	28.1	**100%**	**99%**	**97%**	**84%**	**65%**
		Mean		14.04	9.31	4.56	1.48	0.87
		S.D.		12.67	6.50	2.98	1.76	1.44
Middle	7/17/2002	8/11/2002	24.9	**100%**	**98%**	**92%**	**64%**	**44%**
		Mean		24.88	6.13	2.28	0.55	0.84
		S.D.		N/A	6.52	3.21	1.59	2.15
Lower	7/15/2002	8/11/2002	26.8	**100%**	**60%**	**26%**	**19%**	**13%**
		Mean		5.35	0.51	0.32	0.34	0.25
		S.D.		5.39	1.02	0.89	0.51	0.40
Bourne's Pond								
Upper	7/14/2002	8/10/2002	27.0	**99%**	**78%**	**49%**	**32%**	**22%**
		Mean		3.34	0.50	0.36	0.36	0.29
		S.D.		2.22	0.73	0.68	0.75	0.55
Middle	7/14/2002	8/10/2002	27.0	**Sensor Failure**				
		Mean						
		S.D.						

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was conducted by the DEP Eelgrass Mapping Program for the Great, Green and Bournes Pond embayment systems. The DEP Eelgrass Mapping Program is part of the MEP Technical Team. Aerial photography and field surveys were conducted in 1995 and 2001, as part of this program. Additional analysis of available high-resolution aerial photos from 1951 was used to reconstruct the eelgrass distribution prior to any substantial development of the watershed. The 1951 data were only anecdotally validated for Green and Bournes Ponds. However, within the Great Pond System it was possible to ground-truth the early photo-interpretation based upon available detailed macrophyte surveys from 1952 (Conover 1958). The primary use of the eelgrass distribution 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 (Figure VII-18); the interval over which most of the watershed nitrogen loading increases have occurred. This temporal information is also used to determine the stability of the eelgrass community.

Currently, there are no eelgrass beds within the Great Pond and Green Pond systems, other than a few small sparse isolated patches within the lower basin of Great Pond and adjacent the tidal inlet in Green Pond, with these residual patches within the boundary of the 1951 beds. Bournes Pond currently supports eelgrass beds within its lower reach, primarily in the region at the mouth of Israel's Cove. However, the Bournes Pond beds appear to be diminishing rapidly. Specifically, the assessment of the eelgrass data indicates the following for each of the embayment systems:

Great/Perch Pond System: Great Pond supported extensive eelgrass beds throughout the bulk of its lower basin in 1951 (Figure VII-18a). In contrast, the upper tributary and the Perch Pond basin showed no eelgrass in the 1951 aerial photography or the 1952 field surveys or subsequent DEP surveys. The distribution of eelgrass beds in the 1951 maps are independently supported by detailed field surveys conducted throughout 1952 (Conover 1958; Figure VII-16). Both the 1951 DEP map and the 1952 surveys show eelgrass beds throughout the lower basin, with the exception of a narrow zone in the center. However, the 1952 field surveys also found limited eelgrass penetrating into the lower region of the upper tributary. However, between 1951 and 1995 almost 90% of the Great Pond beds had been lost, with the remainder lost by 2001. Only a few sparse eelgrass patches currently remain within the lower basin of the Great/Perch Pond System. The recent loss of beds and the residual patches indicate that the lower basin of estuary likely is still close to its nitrogen loading threshold, supportive of eelgrass. The system's pattern of eelgrass bed loss and its current lack of eelgrass beds, indicates that the lower basin is moderately impaired and the upper tributary is significantly impaired eelgrass habitat. Since the depth of Perch Pond may preclude eelgrass growth and there is no record that this tributary basin has supported eelgrass, it is not possible to assess the nitrogen related health of this basin by this indicator at this time. Based upon the data, the lower basin (1951 distribution) should be the focus of eelgrass restoration in this system.

Green Pond System: Similar to the Great and Bournes Pond Systems, Green Pond also had significant eelgrass coverage based on the historical 1951 aerial photography. However, eelgrass habitat within Green Pond appears to have been restricted to the lower 1/3 of the estuary, primarily within the lower basin and along the shallower margins of the lower region of the middle reach (Figure VII-18a). All of the eelgrass beds were lost from the Green Pond

System by the 1995 DEP survey, although field surveys found eelgrass in the lowermost portion of the lower basin adjacent the tidal inlet. The observation of residual eelgrass patches adjacent tidal inlets appears to be the typical pattern even in eutrophied estuaries, likely resulting from the very high quality of the inflowing tidal water which overlies these sites for about half of each tidal cycle. However, it should be noted that eelgrass was observed between 1987-1990 in the central region of the lower basin of Green Pond (below the Menauhant Bridge) by Falmouth PondWatch, although no data on density or distribution is available.

It is significant that eelgrass was not detected in middle and upper portions of Green Pond in the 1951 data. It appears that these sections of the Green Pond system are not supportive of this type of habitat. These upper and middle regions of Green Pond are similar to the upper tributaries to Great and Bournes Ponds which also do not appear supportive of eelgrass beds. It is likely that the cause is similar in all systems and relates ultimately to the interplay of tidal flushing and nitrogen loading rates (see Great Pond discussion, above). In addition, the lack of eelgrass in the uppermost portions of Green Pond is also consistent with the predominance of salt marsh near the Pond's head.

The early loss of eelgrass beds (most likely prior to 1990) and with only residual patches indicate that the most of the lower region of this estuary is beyond its nitrogen loading threshold, supportive of eelgrass. The system's timing and pattern of eelgrass bed loss and its current lack of eelgrass beds (only residual patches), indicates that the lower 1/3 of the estuary is currently significantly impaired eelgrass habitat grading to moderately impaired adjacent the tidal inlet and that the upper and middle regions have not historically been supportive of eelgrass habitat. Therefore, the lower basin and not the upper and middle regions should be the focus of eelgrass restoration.

Bournes Pond System: Similar to the Great and Green Pond Systems, Bournes Pond supported eelgrass beds throughout most of its lower basin, with smaller beds within the lower region of Israel's Cove. But eelgrass beds did not penetrate significantly into the upper tributary, a pattern also similar to Great and Green Pond. However, unlike the other 2 estuaries, Bournes Pond currently supports eelgrass beds, although they are significantly reduced in areal coverage. The current distribution of eelgrass beds within the Bournes Pond System shows an approximately 64% reduction in overall coverage from 1951 to 2001, with 40% of the 49 acres lost occurring between the 1995 and 2001 surveys. This evaluation is supported by field observations by MEP Technical Team members conducting sampling throughout Bournes Pond in 2002 and 2003 and by similar point data collected in 1979 (Figure VII-17).

It is not possible to determine a quantitative rate of change in eelgrass coverage from the mapping data for Great and Green Pond, since there is only one survey with eelgrass. However, in the case of Bournes Pond, the 1995 survey and the 2001 survey do show eelgrass with a reduction in bed acreage from 1995 to 2001 equal to approximately 19.5 acres. If bed loss in Bournes Pond was uniform from year to year this would translate to bed loss of approximately 3.9 acres per year from 1995 to 2001.

Based upon the temporal and spatial patterns of bed loss and the continuing presence of eelgrass beds, it appears that the lower basin of the estuary is just beyond its nitrogen loading threshold, supportive of eelgrass. Since it appears that the upper tributary has not historically been supportive of eelgrass beds, it appears to be structurally non-supportive of this habitat. The data indicate that the lower basin and Israels Cove are moderately impaired and that the lower basin (1951 distribution) and not the upper tributary should be the focus of eelgrass restoration in this system.

Inter-System Comparisons and Nitrogen: In all three estuaries, eelgrass habitat appears to be restricted to primarily the lower basins even in the 1951 coverage. This is likely the result of the sensitivity of the upper reaches of these systems to nitrogen inputs, primarily due to the poor tidal flushing resulting from the small driving tide in Vineyard Sound (see Section V). As a result, the upper reaches likely were unable to support eelgrass even at the 1951 nitrogen input levels. Some support for this contention stems from the prevalence of macroalgae in the 1951 surveys by Conover. Restriction of 1951 eelgrass coverage to the lower basin was also observed in the MEP analysis of nearby Popponesset Bay (MEP). In addition, the pattern of these beds is consistent with the pattern of nitrogen related habitat quality that is currently observed within the systems. It appears that as embayments became nutrient enriched, the upper regions are the first to lose their eelgrass beds, with the lower basins containing the tidal inlets being the last to lose their beds. This is the general pattern of nitrogen mediated eelgrass loss in southeastern Massachusetts estuaries. The general pattern is for highest nitrogen levels to be found within the innermost/uppermost 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. The consequence is that eelgrass bed decline typically follows a pattern of loss in the innermost/uppermost 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. Conversely, it is likely that if nitrogen loading were to decrease, eelgrass could first be restored in the lower portions of the main basins, and with further reductions, be restored to the 1951 pattern. Based upon the substantial area of the main basins of Great and Bournes Pond and the lower 1/3 of Green Pond which supported eelgrass beds in 1951, implementation of nitrogen management alternatives for Great, Green and Bournes Ponds might be expected to restore as much as 170, 40, and 50 acres, respectively (Table VII-3).

Other factors which influence eelgrass bed loss in embayments may also be at play in Great, Green and Bournes Pond, although the loss of beds in Great and Green Pond and the significantly reduced eelgrass distribution in Bournes Pond is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within these estuarine areas (Tables VII-1, VII-2). However, a brief listing of non-nitrogen related factors is useful. Eelgrass bed loss does not seem to be directly related to mooring density, as only Great Pond supports significant moorings. Similarly, pier construction and boating pressure may be adding additional stress in nutrient enriched areas, but do not seem to be the overarching factor. It is not possible at this time to determine the potential effect of shellfishing on eelgrass bed distribution, although it must be small as there is little shellfishing on an areal basis in the Ponds.

The relative pattern of these 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).

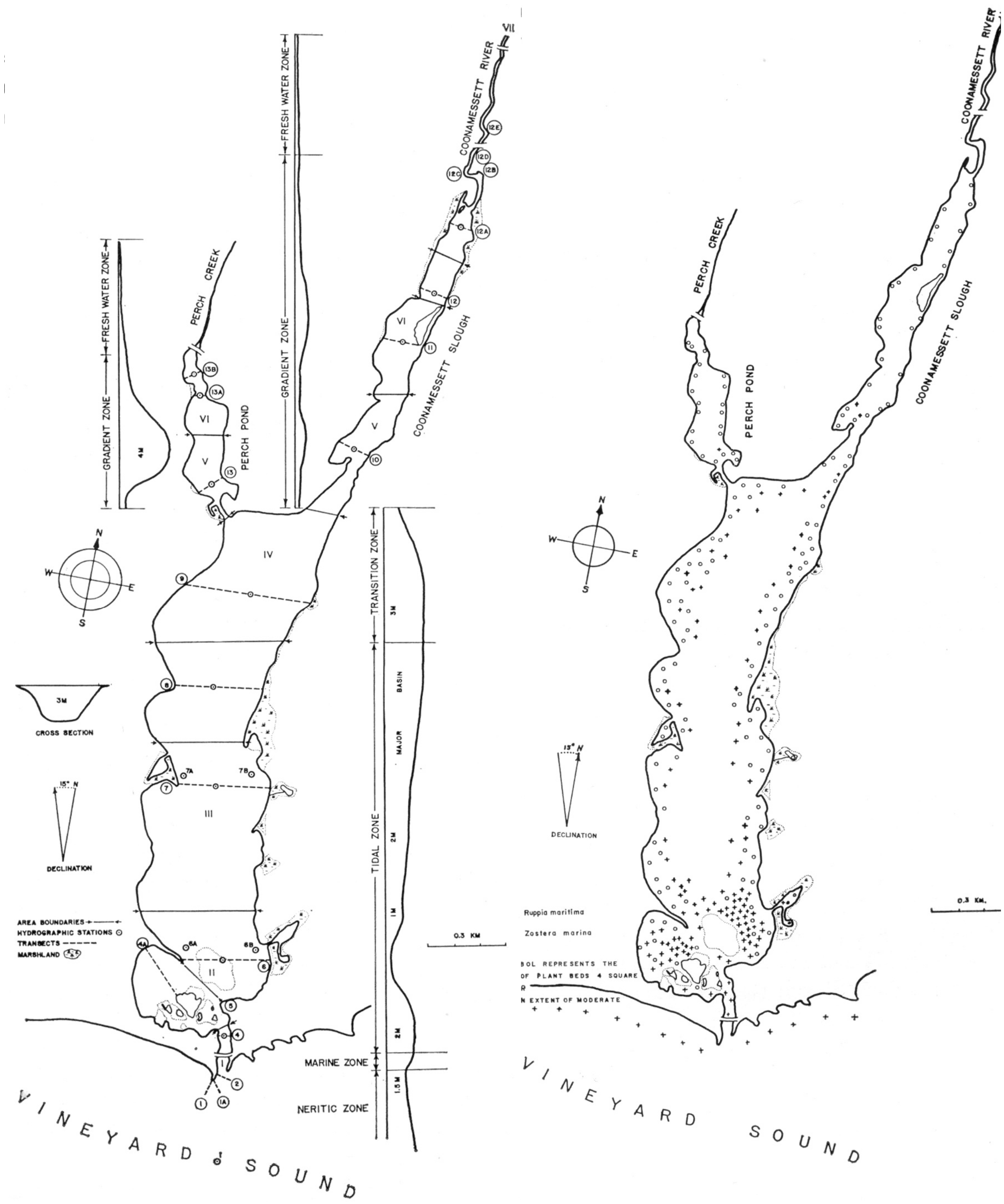


Figure VII-16. Map of Great/Perch Pond System, Falmouth, MA 1992 (Conover 1958). (Left) Map of topography, environmental zones, area boundaries, sections and locations of hydrographic stations. (Right) Map of **+** = eelgrass (*Zostera marina*) and **o** = widgeon grass (*Ruppia maritima*), from on-site surveys in 1952.

Figure VII-17. Map of eelgrass distribution in 1979 survey of sites A-R from Environmental Impact Statement by Weston and Sampson related to the new inlet/bridge which was installed in 1986. Eelgrass density is shown as: (+) no eelgrass, (open circles) sparse eelgrass, (hatched circle) moderate density eelgrass, and (full circle) dense eelgrass.

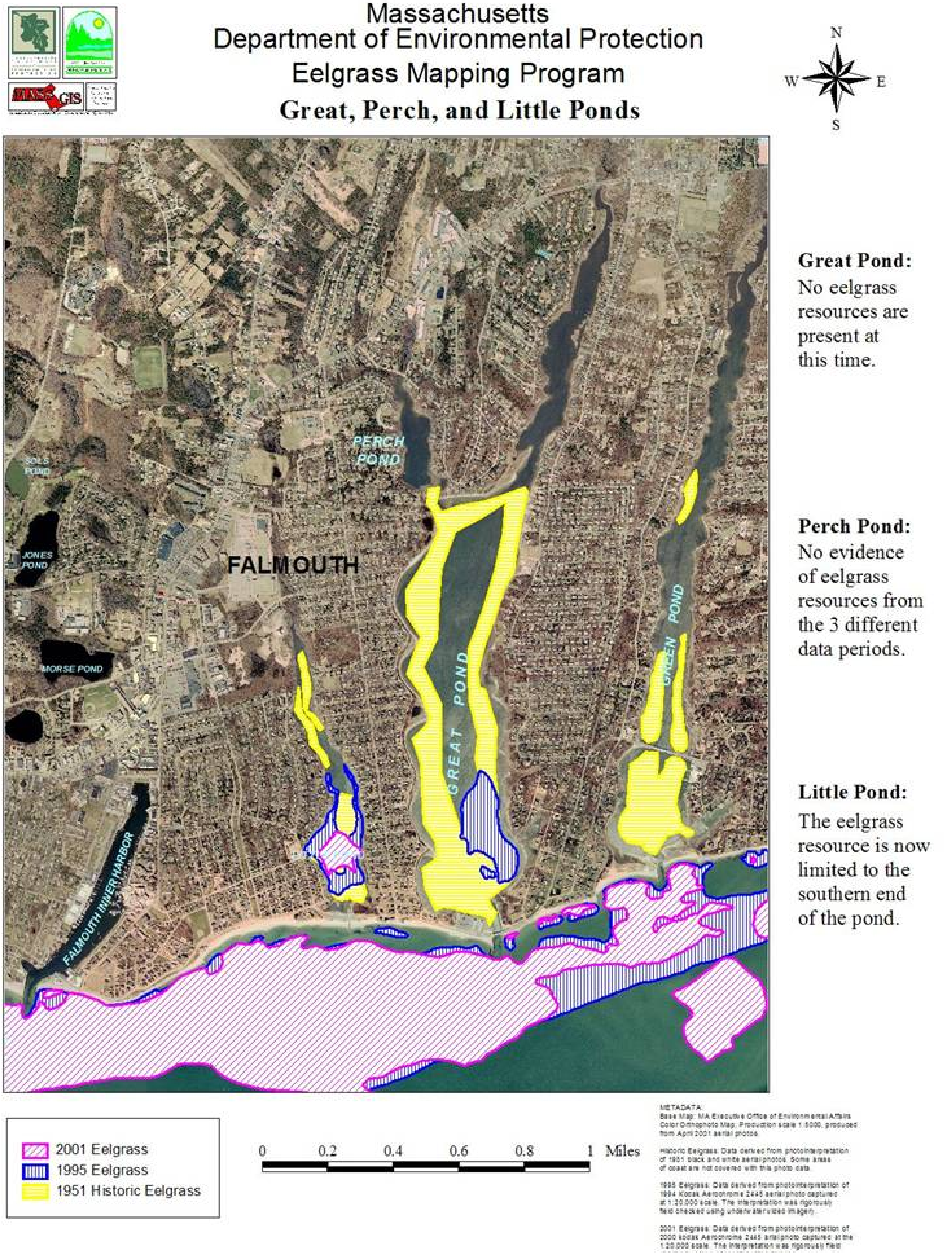


Figure VII-18a. Eelgrass bed distribution within Great and Green Ponds. The 1951 coverage is depicted by the yellow outline inside of which circumscribes the eelgrass beds. The blue (1995) and purple (2001) areas were mapped and ground-truthed by DEP.

Department of Environmental Protection
Eelgrass Mapping Program

Bournes Pond Historical Eelgrass Distribution

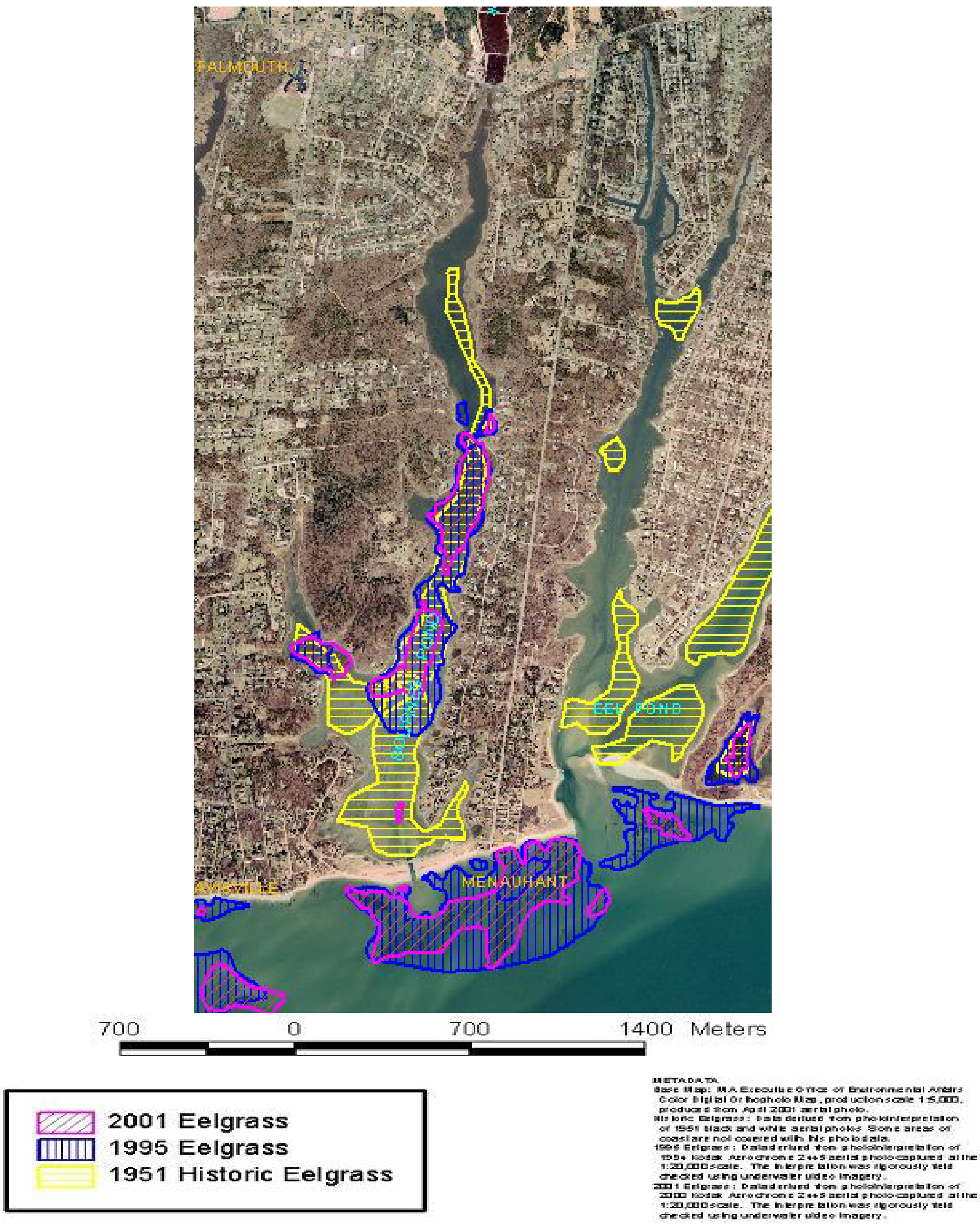


Figure VII-18b. Eelgrass bed distribution within Bournes Pond. The 1951 coverage is depicted by the yellow outline inside of which circumscribes the eelgrass beds. The blue (1995) and purple (2001) areas were mapped and ground-truthed by DEP.

Table VII-3. Changes in eelgrass coverage in the Great, Green and Bournes Ponds systems within the Town of Falmouth over the past half century (C. Costello).

Embayment	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Great Pond [1]	171.7	19.7	0	100%
Green Pond [2]	41.2	0	0	100%
Bournes Pond	77.0	47.3	27.9	63.8%

[1] There is no record of present or historical eelgrass beds in Perch Pond (tributary to Great Pond) either from aerial mapping or field surveys. The lower basin of Great Pond did support very sparse eelgrass in patches observed in the 2001 field survey, but no "beds".
[2] The lower basin of Green Pond did support very sparse eelgrass patches circa 1990, but no beds were observed in the 2001 field survey.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at locations throughout the Great/Perch Pond (6 locations, 24 samples), Green Pond (6 locations, 19 samples), and Bournes Pond (8 locations, 32 samples) (Figure VII-19a, b). At each location samples were collected by Young modified Van Veen Grab (0.0625 m^2) generally along transects established perpendicular to the long axis of the estuary. In all areas and particularly those that do not support eelgrass beds (hence Great Pond, Green Pond and most of Bournes Pond), 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. Stress indicators tend to be small fast growing opportunistic species, frequently small polychaete worms (e.g. Capitellids, Spionids, etc.). Transitional species tend to be larger and penetrate deeper into the sediments (e.g. amphipods, small bivalves). Species indicative of high quality environments tend to be long lived (years), larger, and deep burrowing (e.g. bivalves, large polychaetes, etc.). In addition there can be a shift from deposit feeding to filter feeding communities (at the lower levels of organic enrichment). Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, throughout Great and Green Ponds and the upper regions of Bournes Pond, these systems are clearly currently impaired by nutrient overloading. However, to the extent that they 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 is 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 Surveys conducted throughout the Great/Perch Pond, Green Pond and Bournes Pond Estuaries showed similar spatial patterns of nitrogen related impairment, although the magnitude of habitat decline varied significantly between systems. All systems showed significant nitrogen related habitat quality impairment within the uppermost reaches and within tributary deep basins (e.g. Perch Pond in Great Pond System). Mid and Lower reaches showed higher habitat health. However, only the lower basin of Bournes Pond exhibited moderately healthy benthic animal habitat. The nutrient related benthic infaunal habitat quality throughout each of the 3 estuaries as determined from the 2003 surveys indicated the following:

- *Great/Perch Pond System* (Table VII-4a), the upper estuarine reach (from the main basin to Rt. 28) showed limited infaunal species (mean 10) which 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, these species can occur in relatively high densities. Similarly, Perch Pond, a drowned kettle pond with tidal connection to the main basin of Great Pond, showed benthic animal habitat severely degraded-significantly impaired by nitrogen enrichment. As for the upper tidal reach, the Perch Pond infaunal community was also dominated by opportunistic stress tolerant species, but at much lower numbers of organisms distributed among fewer species indicating a higher level of stress in this system. It is likely that the poor habitat quality stems in part from the geomorphology of the Perch Pond basin, which is made from an enclosed deep kettle basin with a shallow tidal inlet. This structure increases the sensitivity of this tributary basin to nitrogen enrichment as it provides for periodic stratification and low oxygen levels in bottom waters. The sediments within this basin are sulfidic, with either a thin or absent oxidized surface layer. In contrast, most of the large main basin of Great Pond was found to currently support only a moderate level of nitrogen impairment. The sediments tend to be soft consolidated muds, sometimes sulfidic below, with some sand areas near the tidal inlet. The level of impairment is based primarily upon the number of species present and their relatively high density and the dominance by amphipods (transitional species) and mollusks. The designation of moderately impaired is consistent with both the average diversity (2.2) and evenness (0.60) scores. However, this large lower basin is not homogeneous. The deeper central region has significantly lower numbers of species and more stress indicator species than the large marginal basin (general pattern of eelgrass distribution map of 1951, Figure VII-10). Three quarters of the total basin samples averaged 13 species, range 10-17 species, and these were from the region surrounding the central basin, representing the bulk of the basin area. In contrast, the narrow central basin region averaged 5 species, range 3-7, indicative of a lower quality habitat (e.g. significantly impaired). The relative importance of depositional environment, depth, and boat traffic in creating the habitat distribution in the central region of the lower basin cannot be evaluated at this time. The overall results indicate an estuarine system which is presently showing significant impairment of its benthic habitat within its upper reach and in Perch Pond and moderate impairment throughout the bulk of the lower main basin under present nitrogen loading conditions.

- *Green Pond System* (Table VII-4b), the entirety of the Green Pond estuary showed significantly impaired and sometimes significantly degraded benthic habitat quality. The average number of species was less than 8 in all reaches, with low diversity (<1.45). The uppermost stations (GP-1A,B) were within the head region of the pond, where salt marsh processes predominate and the sediments contained visible peat deposits. However, stations GP-2 & 2A were well within the embayment portion of the pond. The upper region was 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). The upper reach also had low species diversity (H'=1.10) and low evenness (E=0.50). The apparently poor infaunal habitat quality is consistent with the high chlorophyll *a* levels and periodic low oxygen in bottom waters, as well as periodic fish kills within this region of the estuary. The middle reach of the estuary supports similar diversity and evenness estimates, but has lower animal densities. However, the dominance by amphipods, rather than polychaetes indicates a slightly higher quality environment for infauna. It should be noted that amphipods are also tolerant of organic matter enrichment and do not indicate healthy conditions. The survey clearly indicated that habitat impairment from nitrogen enrichment has reached well into the lower basin of Green Pond (Menauhant Bridge to inlet). The poor quality of the infaunal habitat within this lower basin is likely related to its increased depth and the highly enriched waters entering from the upper and mid estuary. The lower basin was severely impoverished in both infauna species (4) and numbers of organisms (25). The sediments in this region were sulfidic with little to no oxidized surface layer. In addition, while grain-size analysis was not conducted, it appeared that some of the surface sediments within the lower basin are soft muds. The overall results indicate an estuarine system that is currently supporting significantly impaired infaunal habitat throughout its tidal reach, and which borders on significantly degraded conditions in the uppermost reach and lower basin. Lower basin conditions are of particular concern as this region is closest to the tidal inlet and is most capable of supporting diverse healthy communities.

- *Bournes Pond System* (Table VII-4c), the upper estuarine reach (from the main basin to Rt. 28) showed limited infaunal species (mean 9) at only moderate densities (mean 124 individuals) living in soft muds. However, the diversity (H'=2.10) and evenness (E=0.70) indices were indicative of moderate/significant impairment of the benthic animal habitat, and this estuarine reach supported only moderate numbers of stress indicator (egg. small polychaete worms) and transitional (e.g. amphipods) species. The infaunal communities appear to reflect other habitat quality indicators such as chlorophyll a (moderate) and bottomwater dissolved oxygen (moderate to low). Israels Cove showed similar conditions to the upper estuarine reach, indicating a similar level of moderate to significant impairment for this tributary. While both the upper reach and Israels Cove showed nitrogen related impairment these regions support higher quality habitat than the similar regions of adjacent Great/Perch Pond and Green Pond. The ultimate cause of this higher habitat quality is almost certainly the much lower watershed nitrogen loading to the Bournes Pond Estuary as compared to the neighboring systems. The lower basin of Bournes Pond showed healthy to slightly impaired benthic habitat quality. This region of the estuary supported high diversity (H'=2.7), evenness (E=0.70) and numbers ~400. In addition, while there were some species indicative of nitrogen enrichment these species did not dominate the community. Similar to the spatial distribution of habitat health within the overall pond, the lower basin showed healthiest conditions in the lower half of the basin nearest the inlet (mean 16 species, H'=2.86, E=0.73) versus the upper half of the basin near the confluence of Israels Cove and the upper reach (mean 10 species, H'=2.40, E=0.74). The overall results indicate an estuarine system capable of supporting diverse healthy communities in the lower basin, with most of the system having infaunal habitat that is moderately to significantly impaired under present nitrogen loading conditions.

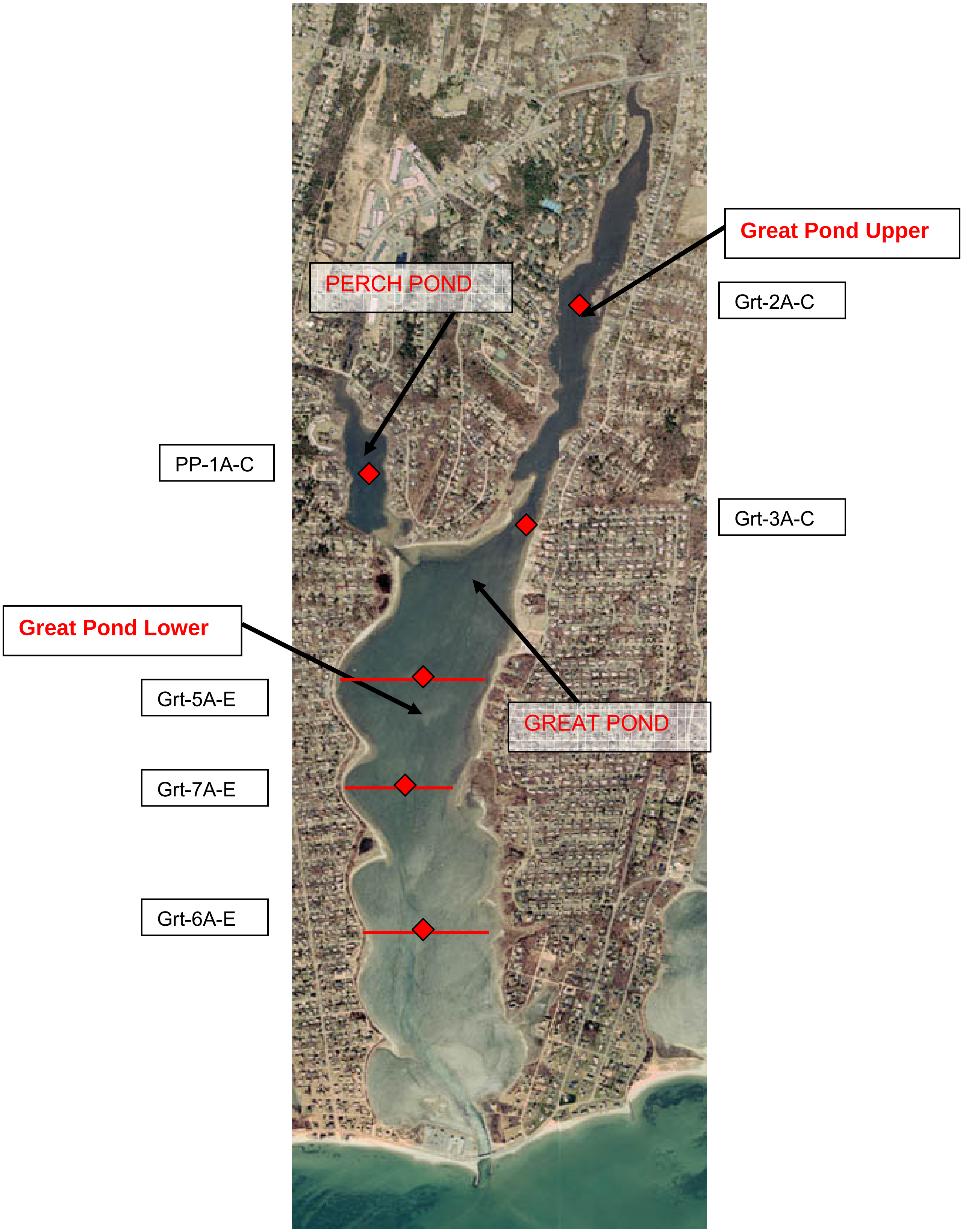


Figure VII-19a. Aerial photograph of Great / Perch Pond showing location of benthic infaunal sampling stations (red symbol). Lines represent horizontal transects sampled.

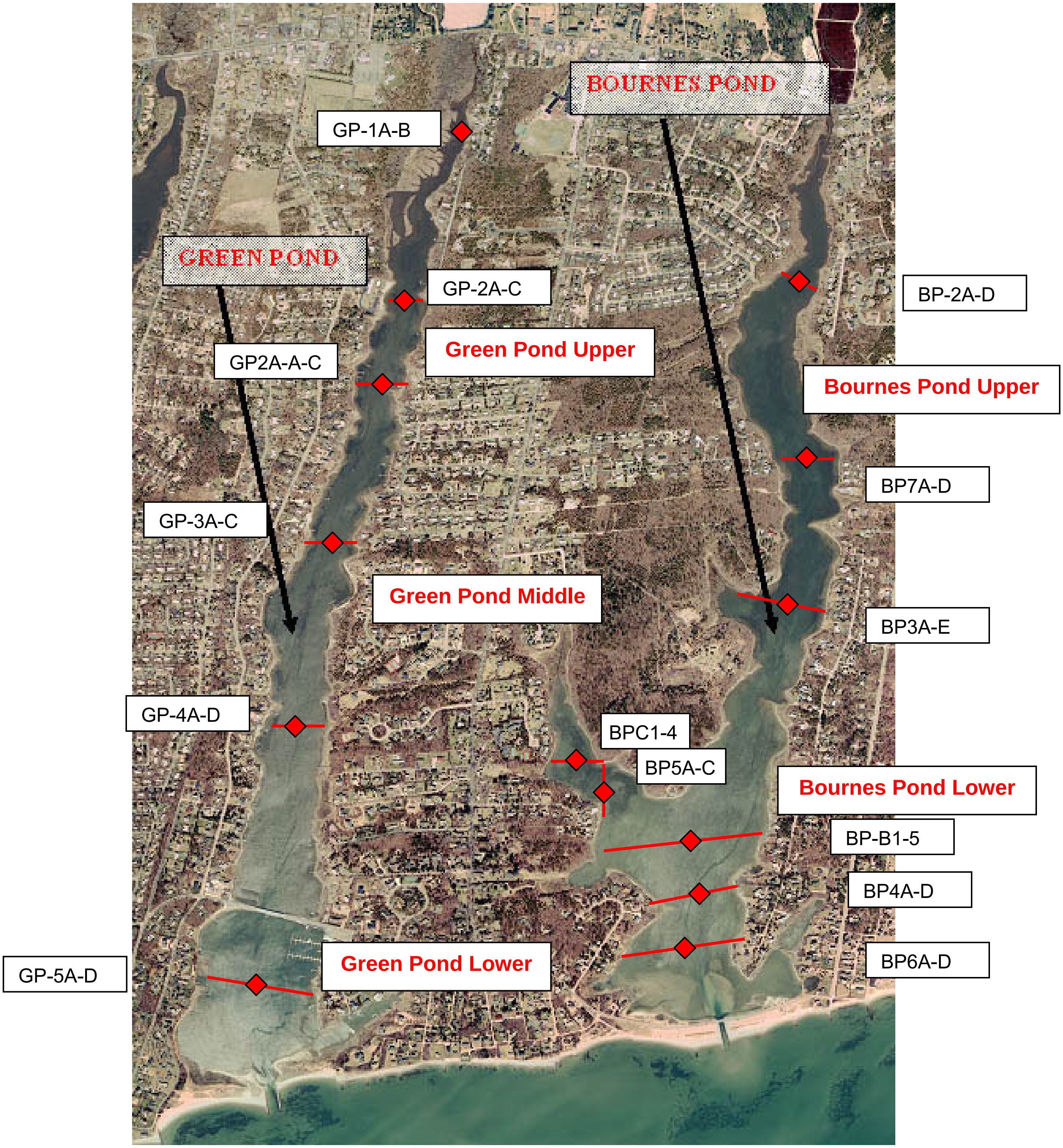


Figure VII-19b. Aerial photograph of Green and Bournes Ponds showing location of benthic infaunal sampling stations (red symbol). Lines represent horizontal transects sampled.

Table VII-4a. Benthic infaunal data for the Great/Perch Pond Estuarine System collected in 2003. 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 collect sediment area of 0.0625 m2. Individual values and mean and standard error (S.E.) of major estuarine regions are presented.

Embayment	Sampling Station	Total Species	Total Individuals	Species per 75 Individuals	Weiner Diversity (H')	Evenness (E)
Great Pond Main Estuary						
Upper	Grt-2A	11	603	9	2.21	0.64
Upper	Grt-2B	8	668	6	1.66	0.55
Upper	Grt-2C	10	1608	7	2.27	0.68
Upper	Grt-3A	13	708	8	2.21	0.60
Upper	Grt-3B	9	336	8	1.79	0.56
Upper	Grt-3C	11	403	8	2.19	0.63
Great Pond	**Mean**	**10**	**721**	**8**	**2.05**	**0.61**
Upper Reach	**S.E.**	**0.7**	**187.0**	**1**	**0.11**	**0.02**
Lower	Grt-5A	12	756	7	1.64	0.46
Lower	Grt-5B	10	372	9	2.48	0.75
Lower	Grt-5C	3	2184	1	0.11	0.07
Lower	Grt-5D	13	432	10	2.62	0.71
Lower	Grt-5E	12	96	10	3.13	0.87
Lower	Grt-6A	13	933	7	1.73	0.47
Lower	Grt-6B	7	76	NA	2.63	0.94
Lower	Grt-6C	5	81	1	1.84	0.79
Lower	Grt-6D	16	320	14	3.24	0.81
Lower	Grt-6E	14	632	12	2.93	0.77
Lower	Grt-7A	12	473	9	2.01	0.56
Lower	Grt-7B	12	228	11	2.89	0.80
Lower	Grt-7C	3	416	1	0.16	0.10
Lower	Grt-7D	17	400	13	3.09	0.76
Lower	Grt-7E	14	304	12	2.31	0.61
Great Pond	**Mean**	**11**	**514**	**NA**	**2.20**	**0.60**
Lower Basin	**S.E.**	**1.1**	**134.6**	**NA**	**0.30**	**0.10**
Perch Pond						
Perch	PP-1A	9	364	6	1.36	0.43
Perch	PP-1B	7	74	NA	1.82	0.65
Perch	PP-1C	4	47	NA	1.37	0.68
Perch Pond	**Mean**	**7**	**162**	**NA**	**1.51**	**0.59**
Main Basin	**S.E.**	**1.5**	**101**	**NA**	**0.15**	**0.08**

Table VII-4b. Benthic infaunal data for the Green Pond Estuarine System collected in 2003. 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 collect sediment area of 0.0625 m^2. Individual values and mean and standard error (S.E.) of major estuarine regions are presented.

Embayment	Sampling Event	Total Species	Total Individuals	Species per 75 Individuals	Weiner Diversity (H')	Evenness (E)
Green Pond Estuarine System						
Upper	GP-1A	5	172	3	0.43	0.19
Upper	GP-1B	4	48	NA	1.06	0.53
Upper	GP-2A	4	264	3	0.85	0.42
Upper	GP-2B	2	14	NA	0.59	0.59
Upper	GP-2C	14	994	10	2.83	0.74
Upper	GP-2A-A	5	41	NA	0.99	0.43
Upper	GP-2A-B	7	1130	5	1.14	0.41
Upper	GP-2A-C	11	357	7	1.20	0.35
Green Pond	**Mean**	**7**	**378**	**NA**	**1.10**	**0.50**
Upper Reach	**S.E.**	**1.4**	**155.6**	**NA**	**0.26**	**0.06**
Mid	GP-3A	4	18	NA	1.75	0.87
Mid	GP-3B	8	202	6	1.41	0.47
Mid	GP-3C	1	54	NA	0.00	0.00
Mid	GP-4A	3	52	NA	1.25	0.79
Mid	GP-4B	6	260	4	1.14	0.44
Mid	GP-4C	8	362	4	0.63	0.21
Mid	GP-4D	7	32	NA	2.24	0.80
Green Pond Middle	**Mean**	**5**	**140**	**NA**	**1.20**	**0.50**
Reach	**S.E.**	**1.0**	**51.0**	**NA**	**0.28**	**0.12**
Lower	GP-5A	6	91	4	1.83	0.71
Lower	GP-5B	5	6	NA	2.25	0.97
Lower	GP-5C	1	1	NA	0.00	0.00
Lower	GP-5D	2	2	NA	1.00	1.00
Green Pond	**Mean**	**4**	**25**	**NA**	**1.30**	**0.70**
Lower Basin	**S.E.**	**1.2**	**22.0**	**NA**	**0.50**	**0.23**

Table VII-4c. Benthic infaunal data for the Bournes Pond Estuarine System collected in 2003. 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 collect sediment area of 0.0625 m2. Individual values and mean and standard error (S.E) of major estuarine regions are presented.

Embayment	Sampling Event	Total Species	Total Individuals	Species per 75 Individuals	*Weiner* Diversity (H')	Evenness (E)
Bournes Pond Estuarine System						
Upper	BP-2A	5	140	6	2.18	0.84
Upper	BP-2B	9	276	10	2.32	0.61
Upper	BP-2C	10	140	10	2.46	0.71
Upper	BP-2D	15	197	14	3.36	0.82
Upper	BP-7A	12	441	10	2.75	0.77
Upper	BP-7B	6	96	5	1.77	0.68
Upper	BP-7C	9	218	9	2.55	0.81
Upper	BP-7D	25	35	NA	0.72	0.72
Upper	BP-3A	9	50	NA	2.42	0.76
Upper	BP-3B	11	85	8	1.75	0.51
Upper	BP-3C	8	151	6	1.39	0.46
Upper	BP-3D	6	31	NA	1.90	0.74
Upper	BP-3E	8	72	NA	2.82	0.94
Bournes Pond	**Mean**	**9**	**124**	**NA**	**2.10**	**0.70**
Upper Reach	**S.E.**	**0.9**	**30.1**	**NA**	**0.18**	**0.04**
Israels Cove	BP-C-1	7	159	6	1.69	0.60
Israels Cove	BP-C-2	7	97	6	2.10	0.75
Israels Cove	BP-C-3	8	87	5	2.29	0.76
Israels Cove	BP-C-4	4	35	NA	1.71	0.85
Israels Cove	BP-5A	10	79	3	3.11	0.94
Israels Cove	BP-5B	7	69	NA	2.41	0.86
Israels Cove	BP-5C	7	60	NA	2.69	0.96
Israels Cove	**Mean**	**7**	**84**	**NA**	**2.30**	**0.80**
	S.E.	**0.7**	**14.7**	**NA**	**0.19**	**0.05**
Lower Basin	BP-B2	7	84	4	2.31	0.82
Lower Basin	BP-B3	9	92	7	2.54	0.80
Lower Basin	BP-B4	9	228	8	1.80	0.57
Lower Basin	BP-B5	15	440	13	2.96	0.76
Lower Basin	BP-4A	24	871	15	3.46	0.75
Lower Basin	BP-4B	19	238	15	3.36	0.79
Lower Basin	BP-4C	19	574	12	2.41	0.57
Lower Basin	BP-4D	14	642	11	2.10	0.55
Lower Basin	BP-6A	9	37	NA	2.58	0.81
Lower Basin	BP-6B	21	767	15	3.24	0.74
Lower Basin	BP-6C	9	105	8	2.74	0.86
Lower Basin	BP-6D	15	512	12	2.98	0.76
Bournes Pond	**Mean**	**14**	**383**	**NA**	**2.70**	**0.70**
Lower Basin	**S.E.**	**1.6**	**83.7**	**NA**	**0.15**	**0.03**

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 Great Pond, Green Pond and Bournes Pond Systems 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, the upper reaches of all three estuaries (including Perch Pond) are showing significantly impaired habitat quality which generally improves in the lower basins, with decreasing distance from the tidal inlets (Chapter VII). In the Green Pond Estuary, only the region adjacent the inlet is presently supportive of moderate quality habitat. The bulk of the area of the large lower Great Pond Basin is generally supportive of moderate quality habitat. Most of the lower region of Bournes Pond (including Israels Cove) is supportive of moderate quality habitat, grading to healthy conditions within most of the lower main basin. A summary of nutrient related habitat health within the Great Pond, Green Pond and Bournes Pond is provided in Table VIII-1, based upon assessment data presented in Chapter VII.

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

	ESTUARY								
	Great Pond			Green Pond			Bournes Pond		
Health Indicator	**Upper**	**Perch Pond**	**Lower**	**Upper**	**Mid**	**Lower**	**Upper**	**Israels Cove**	**Lower**
Dissolved Oxygen	SI/SD	SI/SD[4]	SI[5]	SI	MI	--	SI/SD	--	--
Chlorophyll	SI	--	MI	SD	SI	--	SI	--	--
Macroalgae	SI	SI	--				SI[7]		
Eelgrass	SI[3]	--[2]	MI	--[2]	SI[3]	SI/MI[6]	SI[3]	MI	MI
Infaunal Animals	SI	SD/SI	MI	SI/SD	SI	SI/SD	MI/SI	MI/SI	H
Overall:	**SI**	**SI/SD**	**MI**	**SI/SD**	**SI**	**SI**	**SI**	**MI**	**H**

1 – eelgrass beds can be supported, but beds declining or lost circa1951.
2 – no evidence that estuarine reach is supportive of eelgrass.
3 – no evidence that upper region of this estuarine reach is supportive of eelgrass, but lower region of reach supported eelgrass in 1951 which was completely lost by 1995.
4 – hypoxia reported in PondWatch Reports
5 – mooring in deeper central region = SI, but border region not assessed.
6 – most of the basin devoid of eelgrass (SI), but remaining patch near the inlet (MI).
7 – accumulations in upper 1/3 of reach.
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment;
SD = Severe Degradation
-- = not applicable to this estuarine reach

Eelgrass: Currently, there are no eelgrass beds within the Great Pond and Green Pond systems, other than a few small sparse isolated patches within the lower basin of Great Pond and adjacent the tidal inlet in Green Pond. These residual patches lie within the boundary of the 1951 beds. Bournes Pond currently supports eelgrass beds within its lower reach, primarily in the region of the mouth of Israel's Cove. However, the Bournes Pond beds appear to be diminishing rapidly.

Great Pond supported extensive eelgrass beds throughout the bulk of its lower basin in 1951, with the exception of a narrow zone in the center (Figure VII-18a). In contrast, the upper tributary and the Perch Pond basin showed no eelgrass in the 1951 aerial photography, the 1952 field surveys, or subsequent surveys. The 1952 field surveys did observe limited eelgrass penetrating into the lower region of the upper tributary. However, between 1951 and 1995 almost 90% of the Great Pond beds had been lost, with the remainder lost by 2001. Only a few sparse eelgrass patches currently remain within the lower basin of the Great/Perch Pond System. The recent loss of beds and the residual patches indicate that the lower basin of estuary likely is still close to its nitrogen loading threshold, supportive of eelgrass. The system's pattern of eelgrass bed loss and its current lack of eelgrass beds indicate that the lower basin is moderately impaired and the upper tributary is significantly impaired eelgrass habitat. Since the depth of Perch Pond may preclude eelgrass growth and there is no record that this tributary basin has supported eelgrass, it is not possible to assess the nitrogen related health of this basin by this indicator at this time. Based upon the data, the lower basin (1951 distribution) and not the upper tributary and Perch Pond should be the focus of eelgrass restoration in this system.

Green Pond, similar to the Great and Bournes Pond Systems, also had significant eelgrass coverage based on the historical 1951 aerial photography. However, eelgrass habitat within Green Pond appears to have been restricted to the lower 1/3 of the estuary, primarily within the lower basin and along the shallower margins of the lower region of the middle reach (Figure VII-18a). All of the eelgrass beds were lost from the Green Pond System by the 1995 DEP survey, although 2002 field surveys found eelgrass in the lower-most portion of the lower basin adjacent to the tidal inlet. The observation of residual eelgrass patches adjacent to tidal inlets appears to be the typical pattern even in eutrophied estuaries, likely resulting from the very high quality of the inflowing tidal water which overlies these sites for about half of each tidal cycle.

It is significant that eelgrass was not detected in middle and upper portions of Green Pond in 1951. It appears that these sections of the Green Pond system by nature are not supportive of this type of habitat. This is consistent with the uppermost reach (upper 1/6 of pond) of Green Pond supporting salt marsh habitat with the tidal channel serving primarily as a salt marsh creek. These upper and middle regions of Green Pond are similar to the upper tributaries to Great and Bournes Ponds, which also do not appear supportive of eelgrass beds. The lack of eelgrass in the uppermost portions of Green Pond is also consistent with the predominance of salt marsh near the Pond's head.

The early loss of eelgrass beds (most likely prior to 1990) and with only residual patches remaining in 2002, indicate that the most of the lower region of this estuary is beyond its nitrogen loading threshold, supportive of eelgrass. The system's timing and pattern of eelgrass bed loss and its current lack of eelgrass beds (only residual patches) indicate that the lower 1/3 of the estuary is currently significantly impaired eelgrass habitat, grading to moderately impaired adjacent to the tidal inlet. The upper and middle regions have not historically been supportive of eelgrass habitat. Therefore, the lower basin and not the upper and middle regions should be the focus of eelgrass restoration.

Bournes Pond supported significant eelgrass beds throughout most of its lower basin, with smaller beds within the lower region of Israel's Cove in 1951. Eelgrass beds did not penetrate fully into the upper tributary, a pattern also similar to Great and Green Ponds. The eelgrass coverage in the upper tributary in the 1951 record shows good bed coverage in the lower 1/3 of the tributary, with a grading to patches restricted to the margins in the middle 1/3 of the tributary and then no eelgrass in the upper 1/3 of the tributary. Unlike Great Pond and Green Pond, Bournes Pond currently supports eelgrass beds, although they are significantly reduced in aerial coverage. The current distribution of eelgrass beds within the Bournes Pond System shows an approximately 64% reduction in overall coverage from 1951 to 2001, with 40% of the 49 acres lost occurring between the 1995 and 2001 surveys (Figure VII-17). If recent eelgrass bed loss in Bournes Pond was uniform from year to year this would translate to bed loss of approximately 3.9 acres per year from 1995 to 2001.

Based upon the temporal and spatial patterns of bed loss and the continuing presence of eelgrass beds, it appears that the lower basin of Bournes Pond is currently just beyond its nitrogen loading threshold, supportive of eelgrass. Since it appears that the upper tributary has not historically been supportive of eelgrass beds, it appears to be structurally non-supportive of this habitat. The data indicate that the lower basin and Israels Cove are moderately impaired and that the lower basin (1951 distribution) and not the upper tributary should be the focus of eelgrass restoration in this system.

Water Quality: The upper regions of Great, Green, and Bournes Ponds are currently under seasonal oxygen stress, consistent with nitrogen enrichment (Table VII-1). The cause of this oxygen stress likely is eutrophication, as supported by the high levels of chlorophyll a, >25 ug/L 44%, 65% and 22% of the time for Great, Green, and Bournes Ponds, respectively (Table VII-2). Oxygen conditions and chlorophyll a levels improved in each system with decreasing distance from the tidal inlet, although all systems showed oxygen depletions below 5 mg L-1 and generally to <4 mg L-1 at the most southern stations measured in 2002. In all systems there was a clear gradient in chlorophyll a, with highest levels in the uppermost reaches and lowest levels near the tidal inlet to Vineyard Sound.

In Great Pond both the oxygen and chlorophyll a records in the upper tributary indicate a highly nitrogen enriched estuarine reach. Dissolved oxygen levels routinely declined below 2 mg L^{-1} coupled with very high chlorophyll a levels, in excess of 20 ug L^{-1} over 64% of the time and with three bloom periods in excess of 40 ug L^{-1}. These conditions are consistent with anecdotal reports of periodic summer fish kills within this upper tidal reach (reported in the Falmouth Enterprise and Falmouth PondWatch Reports). The lower basin also indicated some nitrogen enrichment, in the area that did not have eelgrass in the 1951 surveys. Chlorophyll a levels were significantly lower than in the upper tidal reach. This central basin condition is consistent with the infaunal community indicators, which show a significant stress in the central region with moderate quality habitat in the surrounding areas (see Section VII-4). The central region of the lower basin differs from the surrounding larger basin area, as it consists primarily of soft sediments, as well as accumulations of drift macroalgae (*Cladophora, Ulva*) and a surface algal mat. Previous investigations have found summer hypoxia/anoxia in the Perch Pond basin (PondWatch and Anderson and Hampson, unpublished data). Macroalgal accumulations and a thin benthic algal mat were observed in Perch Pond in the 2002 MEP surveys.

Green Pond water quality data indicates a system which is significantly nitrogen impaired throughout its upper half, based primarily upon the very high chlorophyll a levels and periodic oxygen declines. In contrast, the lower reaches support healthier conditions (moderately impaired/significantly impaired) based upon both the level and duration of observed oxygen depletion and chlorophyll a levels. The infaunal surveys were consistent with nitrogen enriched conditions within the upper and middle regions and within the bulk of the lower basin. Only the portion of the lower basin adjacent to the inlet supports moderate quality habitat at present.

The upper tributary to Bournes Pond is currently nitrogen enriched based upon both the dissolved oxygen and chlorophyll a records. However, there is a steep gradient in nitrogen related habitat impairment along this upper estuarine reach of Bournes Pond. The upper mooring found large oxygen depletions (to 4 and 3 mg L-1) over long periods, 47% and 34% of deployment, respectively. The level of oxygen depletion was consistent with the high phytoplankton biomass, which was >20 ug L^{-1} for 32% of the deployment. However, simultaneous readings at the middle mooring location showed a much reduced level of nitrogen related habitat impairment. There is a strong gradient in habitat quality along this reach of the estuary, with oxygen levels within the lower third showing only brief excursions below 2 mg L^{-1} on 3 of 27 days sampled. Similarly, the lower mooring found that only 9% of the time were dissolved oxygen levels below 4 mg L^{-1}, compared to 47% of time at the upper mooring. Never-the-less, the mooring data indicate that the entire upper estuarine reach is severely degraded/significantly impaired by nitrogen enrichment. The gradients in the upper tidal reach of Bournes Pond are consistent with higher quality habitat in the lower main basin, as evidenced by eelgrass and infaunal communities (see Sections VII-3,4).

Based upon the available dissolved oxygen and chlorophyll data the ranking of the more sensitive regions of the Great Pond, Green Pond, and Bournes Pond Estuaries is as follows:

- Great Pond:
    - Upper Reach – significantly impaired to severely degraded
    - Perch Pond – significantly impaired to severely degraded
    - Upper Central region of main basin – moderately to significantly impaired

- Green Pond:
    - Upper Region -- significantly impaired to severely degraded
    - Middle Region – moderately to significantly impaired

- Bournes Pond:
    - Upper Region – significantly impaired to severely degraded
    - Lower Basin and Israels Cove – moderately impaired to healthy

Infaunal Communities: Infauna Communities within the Great/Perch Pond, Green Pond and Bournes Pond Estuaries showed similar spatial patterns of nitrogen related impairment, although the magnitude of habitat decline varied significantly between systems. All systems showed significant nitrogen related habitat quality impairment within the uppermost reaches and within tributary deep basins (e.g. Perch Pond in Great Pond System). Mid and Lower reaches showed higher habitat health. However, only the lower basin of Bournes Pond exhibited moderately healthy benthic animal habitat.

The Great/Perch Pond System's upper estuarine reach (from the main basin to Rt. 28) showed limited infaunal species and a prevalence of 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. Similarly, Perch Pond, a drowned kettle pond with tidal connection to the main basin of Great Pond, showed benthic animal habitat severely degraded/significantly impaired by nitrogen enrichment. Similar to the upper tidal reach, the Perch Pond infaunal community was dominated by opportunistic stress tolerant species, but at much lower numbers of organisms distributed among fewer species indicating a higher level of stress in this system. It is likely that the poor habitat quality stems in part from the geomorphology of the Perch Pond basin, which is made from an enclosed deep kettle basin with a shallow tidal inlet. This structure increases the sensitivity of this tributary basin to nitrogen enrichment as it provides for periodic stratification and low oxygen levels in bottom waters. The sediments within this basin are sulfidic, with either a thin or absent oxidized surface layer. In contrast, most of large main basin of Great Pond was found to currently support only a moderate level of nitrogen impairment. The sediments tend to be soft consolidated muds, sometimes sulfidic below, with some sand areas near the tidal inlet. The designation of moderately impaired is consistent with both the average diversity (2.2) and evenness (0.63) scores. However, this large lower basin is not homogeneous. The deeper central region has significantly lower numbers of species and more stress indicator species than the large marginal basin (consistent with the general pattern of eelgrass distribution map of 1951, Figure VII-10). Three quarters of the total basin samples averaged 14 species, range 10-17 species, and these were from the region surrounding the central basin, representing the bulk of the basin area. In contrast, the narrow central basin region, averaged 5 species, range 3-7, indicative of a lower quality habitat (e.g. significantly impaired). The relative importance of depositional environment, depth, and boat traffic in creating the habitat distribution in the central region of the lower basin cannot be evaluated at this time. The overall results indicate an estuarine system, which is presently showing significant impairment of its benthic habitat within its upper reach and in Perch Pond and moderate impairment throughout the bulk of the lower main basin under present nitrogen loading conditions.

The entirety of the Green Pond estuary showed significantly impaired and sometimes significantly degraded benthic habitat quality. The average number of species was less than 8 in all reaches, with low diversity (<1.45). The uppermost stations were within the head region of the pond, where salt marsh processes predominate and the sediments contained visible peat deposits. However, just below this reach, within the embayment portion of the pond was 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). The upper reach also had low species diversity (H'=1.35) and low evenness (E=0.50). The apparently poor infaunal habitat quality is consistent with the high chlorophyll *a* levels, periodic low oxygen in bottom waters, and periodic fish kills within this region of the estuary. The middle reach of the estuary supports similar diversity and evenness estimates, but has lower animal densities. However, the dominance by amphipods, rather than polychaetes indicates a slightly higher quality environment for infauna. It should be noted that amphipods are also tolerant of organic matter enrichment and do not indicate healthy conditions. The survey clearly indicated that habitat impairment from nitrogen enrichment has reached well into the lower basin of Green Pond (Menauhant Road Bridge to the inlet). The poor quality of the infaunal habitat within the bulk of this lower basin is likely related to its increased depth and the highly enriched waters entering from the upper and mid estuary. The lower basin was severely impoverished in both infauna species (4) and numbers of organisms (31). The sediments in this region were sulfidic, with little to no oxidized surface layer. In addition, while grain-size analysis was not conducted, it appeared that some of the surface sediments within the lower basin are soft muds. The overall results indicate an estuarine system that is currently supporting significantly impaired infaunal habitat throughout its tidal reach, and which borders on significantly degraded conditions in the uppermost reach and lower basin. The lower basin conditions are of particular concern as this region is closest to the tidal inlet and is most capable of supporting diverse healthy communities.

The upper estuarine reach of Bournes Pond showed limited infaunal species (mean 10) at only moderate densities (mean 150 individuals) living in soft muds. However, the diversity (H'=2.23) and evenness (E=0.72) indices were indicative of moderate/significant impairment of the benthic animal habitat, and this estuarine reach supported only moderate numbers of stress indicator (e.g. small polychaete worms) and transitional (e.g. amphipods) species. The infaunal communities appear to reflect other habitat quality indicators such as chlorophyll a (moderate) and bottom water dissolved oxygen (moderate to low). Israels Cove showed similar conditions to the upper estuarine reach, indicating a similar level of moderate to significant impairment for this tributary. While both the upper reach and Israels Cove showed nitrogen related impairment, they support higher quality habitat than the similar regions of adjacent Great/Perch Pond and Green Pond. The ultimate cause of this higher habitat quality is almost certainly the much lower watershed nitrogen loading to the Bournes Pond Estuary than to its neighboring systems. The lower basin of Bournes Pond showed healthy habitat quality with the upper regions indicating some moderate impairment. This region of the estuary supported high diversity (H'=2.8), evenness (E=0.74), and numbers ~400. Similar to the spatial distribution of habitat health within the overall pond, the lower basin showed healthiest conditions in the lower half of the basin nearest the inlet (mean 17 species, H'=2.93, E=0.74) versus the upper half of the basin near the confluence of Israels Cove and the upper reach (mean 11 species, H'=2.51, E=0.73). The overall results indicate an estuarine system capable of supporting diverse healthy communities in the lower basin, with most of the system having infaunal habitat that is moderately to significantly impaired under present nitrogen loading conditions.

VIII.2. THRESHOLD NITROGEN CONCENTRATIONS

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.

Great/Perch Pond Estuarine System: Within the Great/Perch Pond Estuary the most appropriate sentinel station was the upper station within the large main basin (Station GT5 in Figure VIII-1). This location was selected because (1) it was in the upper region where eelgrass bed coverage was documented in the 1952 surveys, (2) restoration of nitrogen conditions supportive of eelgrass at this location will necessarily result in even higher quality conditions throughout the entire lower basin, (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), and (4) Perch Pond habitat will be improved by the nitrogen reduction at the inlet to Perch Pond (the boundary condition for this sub-embayment). For embayment restoration, an additional requirement within Perch Pond and upper Great Pond was to ensure that TN in these sub-systems has been reduced to levels supportive of healthy infauna habitat when the eelgrass threshold was met for the main basin of Great Pond.

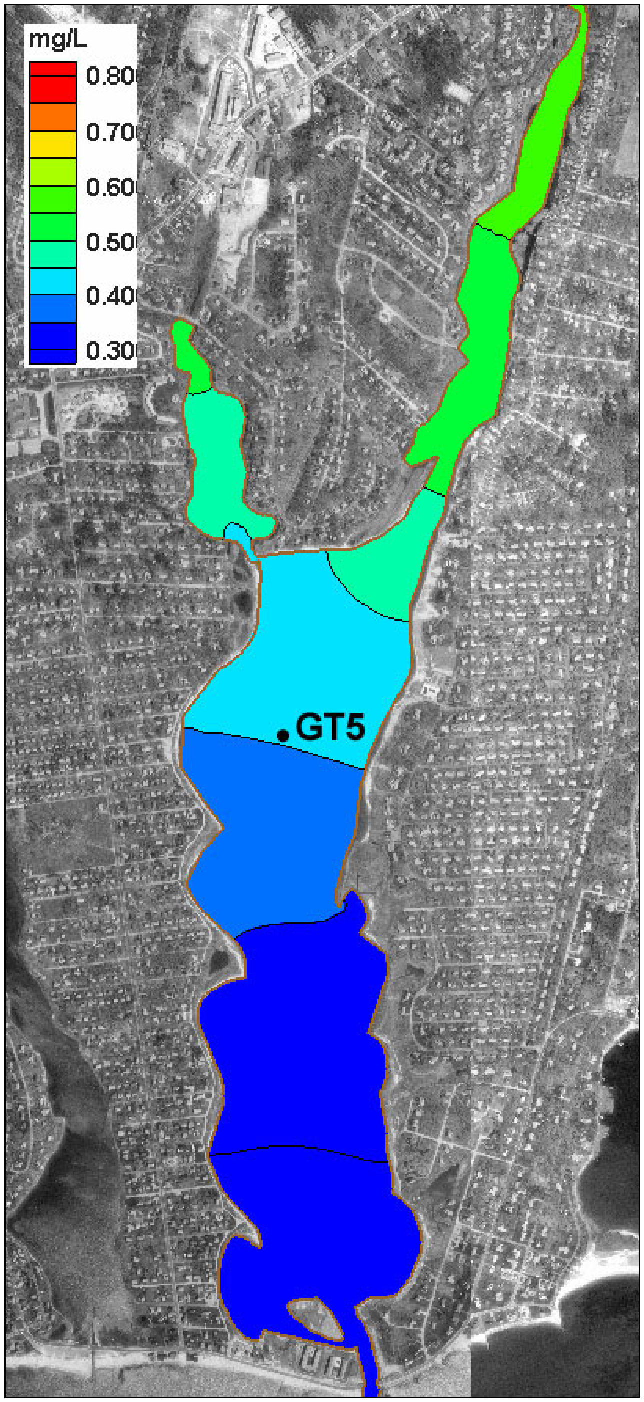


Figure VIII-1. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Great Pond system, for threshold conditions (0.40 mg/L at water quality monitoring station GT5). The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.40 mg TN L^{-1} within the large main basin (Station GT5). 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 0.591 mg TN L^{-1} at GT5 (2) sparse eelgrass can still be found adjacent to the inlet at tidally averaged TN of 0.34 mg TN L^{-1}, and (3) eelgrass beds are not supported at similar depths within the lower basin of Great or Green Ponds at a tidally averaged TN of 0.409 mg TN L^{-1} and (4) the eelgrass beds in Bournes Pond (threshold 0.45 mg TN L^{-1}, discussed below) are in much shallower water which is important for light penetration. Based upon these data and the deeper waters of Great Pond, the threshold TN level was set at 0.40 mg TN L^{-1}, lower than for Bournes Pond (0.42-0.45 mg TN L^{-1}). Based upon the infauna data from Great Pond and Bournes Pond, it appears that the TN levels for Bournes Pond are appropriate metrics for Great Pond, below 0.6 mg TN L^{-1} to support moderate to healthy habitat, with healthy infauna habitat requiring TN <0.5 mg TN L^{-1}. The result also is the same found to support moderate to healthy habitat by MEP for Popponesset Bay. This is consistent with the present condition of the lower main basin, which shows moderately impaired infaunal habitat (i.e. over the threshold). The moderately impaired upper region of the basin currently supports a tidally corrected average concentration of 0.59 mg TN L^{-1}. The significantly impaired upper tributary and Perch Pond have much higher TN levels, >0.78 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 0.40 mg L^{-1} and the lower main basin <0.30 mg TN L^{-1}. This indicates that significant eelgrass habitat restoration would occur within the regions of the 1951 eelgrass coverage. It is possible also to evaluate the response in benthic infaunal habitat. At present, the regions supporting the highest quality infaunal habitat have tidally averaged concentrations (mg TN L^{-1}) from 0.6 in the moderately impaired regions of the upper main basin to 0.34 in the healthy region adjacent to the tidal inlet. Upon reaching the TN threshold at the sentinel station, the upper tributary which is currently significantly impaired to severely degraded habitat (>0.78 mg TN L^{-1}) will have TN levels <0.5 mg L^{-1} in the lower reach and within Perch Pond, with only slightly higher levels in the uppermost region of Great Pond (<0.55 mg TN L^{-1}). It should be noted 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 Great Pond main basin (1951 distribution) and restorative of infaunal habitat throughout the estuary, including the upper tributary and Perch Pond.

Green Pond Estuary: Within the Green Pond Estuary the most appropriate sentinel station was about 2/3 of the distance from the headwaters to the tidal inlet (G4 in Figure VIII-2). 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 2/3 of the estuary was to ensure that TN in this region has been reduced to levels supportive of healthy infauna habitat when the eelgrass threshold was met for the lower 1/3 of the embayment.

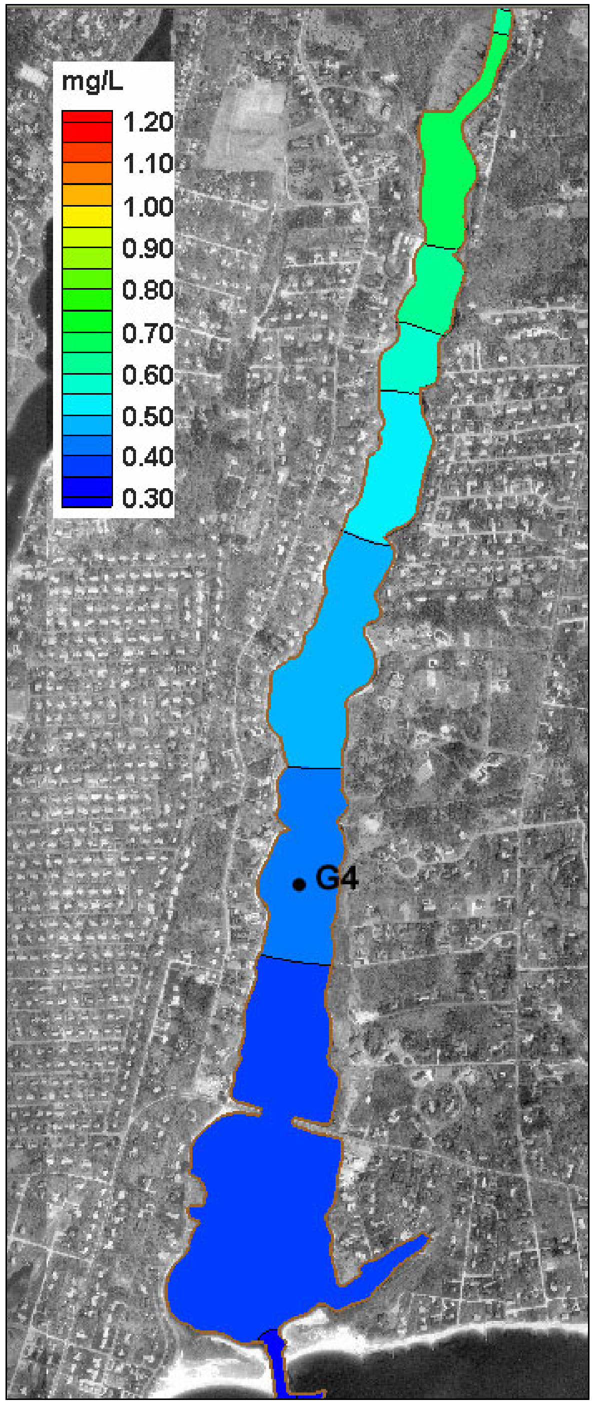


Figure VIII-2. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Green Pond system, for threshold conditions (0.42 mg/L at water quality monitoring station G4, with average concentrations less than 0.70 mg/L within the entire system). The approximate location of the sentinel threshold station for Green Pond (G4) is shown.

The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.42 mg TN L^{-1} for Station G4 and 0.4 mg TN L^{-1} in the lower basin (below the bridge). This threshold level is consistent with the findings that (1) eelgrass beds have been lost in the lower basin which currently supports at tidally averaged TN of 0.53 mg TN L^{-1} at G4 and

0.41 mg TN L^{-1} below the bridge (G5), (2) sparse eelgrass can be still be found adjacent to the inlet at tidally averaged TN of 0.41 mg TN L^{-1}, (3) the eelgrass beds in Bournes Pond (threshold 0.45 mg TN L^{-1}) at shallower water depths which is important for light penetration, and (4) the restriction of eelgrass beds to the margins in the region of the sentinel station (G4) in 1951 with more complete coverage in the lower basin. Based upon these data, the threshold TN level was set at 0.40 mg TN L^{-1} for complete coverage of the lower basin and 0.42 mg TN L^{-1} at the Sentinel Station to re-establish the marginal beds (both conditions are required in this system). These marginal beds north of the Menauhant Road Bridge existed outside of the relatively deep central channel within this portion of Green 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 the bridge should occur when TN levels are lowered to 0.42 mg TN L^{-1}.

Given the lack of healthy infaunal habitat in Green Pond, the infaunal requirements are based upon Great Pond and Bournes Pond, where TN levels below 0.6 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 Green Pond currently have very high TN levels, >0.7 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 0.42 mg L^{-1} and the lower main basin <0.36 mg TN L^{-1}. This indicates that significant eelgrass habitat restoration would occur within the regions of the 1951 coverage. Most significantly the lower basin achieves a TN level that is generally thought to be highly supportive of eelgrass beds and is well below the TN threshold of 0.40 mg TN L^{-1} set for this basin. It also is possible to evaluate the response in benthic infaunal habitat within the upper regions of the estuary. Upon reaching the TN threshold at the sentinel station, the upper tributary (G2a and lower) below the salt marsh reach (at G2) which is currently significantly impaired to severely degraded infaunal habitat (>0.8 mg TN L^{-1}) will have TN levels <0.6 mg L^{-1} and in the mid reach (G3-G4) <0.5 mg TN L^{-1}. These values indicate that restoration of infaunal habitat below the salt marsh dominated region in the upper reach will be restored. It should be noted 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 Green Pond main basin and marginal beds above the bridge (1951 distribution) and restorative of infaunal habitat throughout the estuary.

Bournes Pond Estuary: Within the Bournes Pond Estuary the most appropriate sentinel station was 2/3 of the way down the upper tributary (Station B3 in Figure VIII-3). This location was selected because (1) it was the upper extent of the full channel eelgrass bed coverage in 1951 (and is slightly above the eelgrass record for 1979), (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 Israels Cove, and (3) restoration of nitrogen concentrations at this site should result in conditions similar to 1951 within the upper 2/3 of the upper tributary, which will be supportive of high quality habitat for benthic infaunal communities (confirmed as described below).

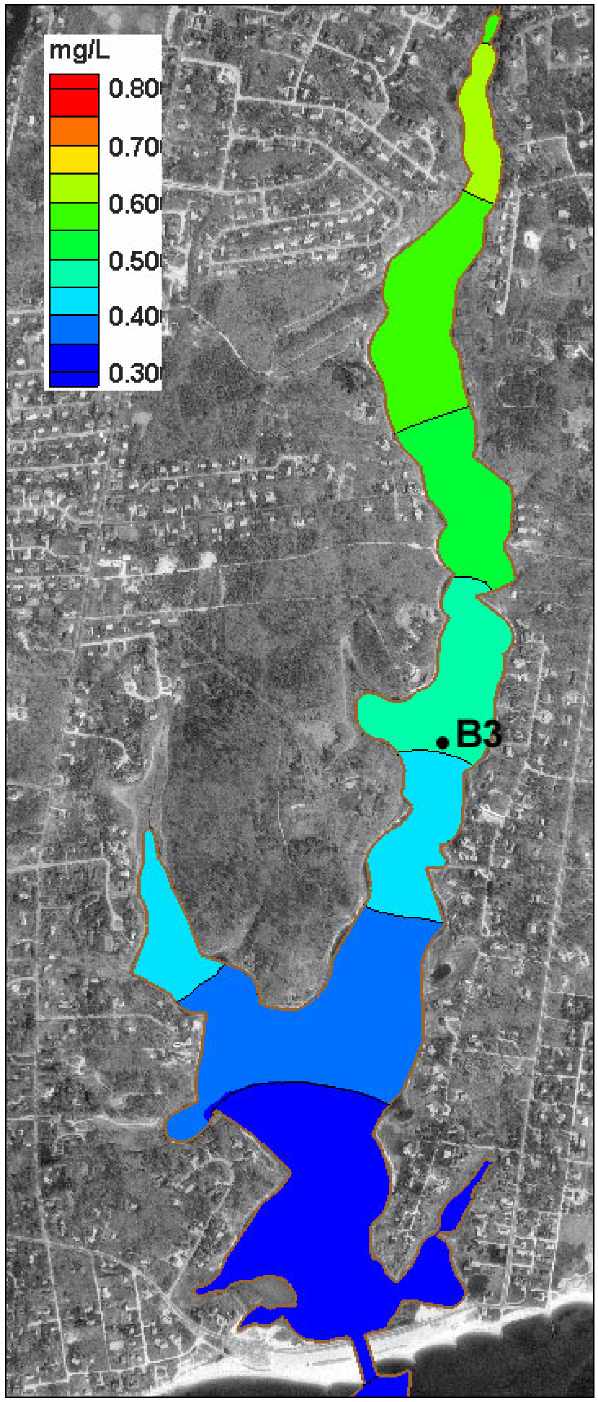


Figure VIII-3. Contour plot of modeled average total nitrogen concentrations (mg/L) in the Bournes Pond system, for threshold conditions (0.45 mg/L at water quality monitoring station B3). The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.45 mg TN L^{-1} within the lower 1/3 of the tributary (Station B3), 0.31 mg TN L^{-1} within the lower basin adjacent to the inlet and 0.42 mg TN L^{-1} within Israels Cove. Although there is only one sentinel station (B3), the thresholds analysis placed an additional requirement that the TN level in the upper region of the lower basin (Station B4) was supportive of healthy infauna habitat when the eelgrass threshold was met for the lower 1/3 of the embayment. These levels are consistent with the findings that (1) eelgrass loss was documented between 1995 and 2001 in the lower basin and that healthy eelgrass beds are currently observed in the upper portion of the lower basin at tidally averaged TN's of 0.426 mg TN L^{-1}, (2) eelgrass can be still be found at the mouth of the upper tributary at tidally averaged TN of 0.481 mg TN L^{-1}, and (3) eelgrass beds in the lower region of Israel's Cove exist at a tidally averaged TN of 0.429 mg TN L^{-1}. Based upon the infauna data, it appears that TN levels should be below 0.6 mg TN L^{-1} to 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 sentinel station (B3) under present loading conditions supports a tidally corrected average concentration of 0.643 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}, the lower basin <0.355 mg TN L^{-1}, and Israels Cove 0.42 mg TN L^{-1}. This indicates that significant eelgrass habitat restoration would occur within the regions of the 1951 coverage. Due to the shallow depths of lower Bournes Pond and Israels Cove, as well as 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, based on the site-specific eelgrass and TN data for Bournes Pond, restoration of eelgrass beds within Israels Cove should occur when TN levels are lowered to 0.42 mg TN L^{-1} and restoration of eelgrass beds within the lower 1/3 of the estuary (from B3 south) should occur when TN levels are lowered to 0.45 mg TN L^{-1} at the sentinel station.

It also is possible to evaluate the response in benthic infaunal habitat. At present, the regions supporting the highest quality infaunal habitat have tidally averaged concentrations (mg TN L^{-1}) from 0.6 in the moderate-significantly impaired regions of Israels Cove to <0.426 in the moderate to healthy lower basin. The Upper 2/3 of the tributary which are currently significantly impaired have TN levels of 0.43 mg L^{-1}. The data suggest that there is likely a range of total nitrogen which can support healthy infauna within this system. It should be noted that these 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 Bournes Pond main basin, most of Israels Cove, and the lower region of the upper tributary. In addition, achieving the nitrogen target at the threshold station is 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 Great Pond, Green Pond ,and Bournes Pond Estuaries. 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 each of Great, Green, and Bournes Ponds. 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 communities. 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 each system necessary to achieve the threshold nitrogen concentrations were highest in Great and Bournes Ponds, with 100% removal of septic load (associated with direct groundwater discharge to the embayment) required for the systems' lower watersheds. In addition, a portion of the septic load entering these two ponds also must be removed to meet the threshold nitrogen concentrations. For the load reduction scenario evaluated, the Coonamessett River (Great Pond) and Bournes Brook sub-watersheds required removal of approximately 50% and 55% of their septic load, respectively. Nitrogen removed in Green Pond required to meet threshold limits is a smaller percentage of the present load to the Pond, with a 73% reduction in the load from the lower watershed, and no reduction needed for the upper watershed. Distributions of tidally-averaged nitrogen concentrations associated with the above thresholds analysis are shown in Figures VIII-1 through VIII-3 for each pond separately.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Ashumet Valley systems. 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
Great Pond	21.28	0.00	-100.0%
Perch Pond	4.47	0.00	-100.0%
Green Pond	16.62	4.43	-73.4%
Bournes Pond	8.30	0.00	-100.0%
Israels Cove	1.78	0.00	-100.0%
Surface Water Sources			
Coonamessett River (Great Pond)	15.08	7.54	-50.0%
Backus Brook (Green Pond)	2.08	2.08	0.0%
Bournes Brook (Bournes Pond)	2.41	1.08	-55.0%

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 Great and Bournes Ponds results in an 85% reduction in total nitrogen load. In Green Pond, a reduction of 77% in the lower watershed septic load resulted in a 66% reduction in its total watershed load. For the Coonamessett River and Bournes Brook, septic load reductions of 50% and 55% resulted in total attenuated watershed load reductions of 33% and 40%. 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 Ashumet Valley systems. 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
Great Pond	25.00	3.72	-85.1%
Perch Pond	5.38	0.90	-83.2%
Green Pond	18.55	6.35	-65.8%
Bournes Pond	9.61	1.31	-86.4%
Israels Cove	2.05	0.27	-86.8%
Surface Water Sources			
Coonamessett River (Great Pond)	22.63	15.09	-33.3%
Backus Brook (Green Pond)	3.81	3.81	0.0%
Bournes Brook (Bournes Pond)	3.29	1.97	-40.3%

Table VIII-4. Threshold sub-embayment loads and attenuated surface water loads used for total nitrogen modeling of the Ashumet Valley systems, 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)
Great Pond	3.72	3.22	0.47
Perch Pond	0.90	0.22	-0.53
Green Pond	6.35	1.61	34.49
Bournes Pond	1.31	1.61	19.28
Israels Cove	0.27	0.26	-0.14
Surface Water Sources			
Coonamessett River (Great Pond)	15.09	-	-
Backus Brook (Green Pond)	3.81	-	-
Bournes Brook (Bournes Pond)	1.97	-	-

Comparison of model results between existing loading conditions and the selected loading scenario to achieve the target TN concentrations at the sentinel stations are shown in Table VIII-5. To achieve the threshold nitrogen concentrations at the sentinel stations, a reduction in TN concentration of approximately 32%, 20%, and 29% is required for Great, Green, and Bournes Ponds, respectively.

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 to ensure the most cost-effective nitrogen reduction strategies. However, “planned” use of natural systems has to be done carefully and with the full analysis to ensure that degradation of these systems will not occur. One clear finding of the MEP has been the need for analysis of the potential of 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 systems, as well. Cranberry bogs, other freshwater wetland resources and freshwater ponds that exist in the upper watersheds of each of these three ponds provide opportunities for enhancing natural attenuation of their nitrogen loads. Restoration or enhancement of wetlands and ponds associated with the lower ends of rivers and streams discharging to estuaries is 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 Ashumet Valley systems. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	threshold (mg/L)	% change
Coonamessett River (estuarine)	GT2	0.875	0.549	-37.3%
Great Pond - upper	GT3	0.782	0.496	-36.6%
Perch Pond	GT4	0.859	0.471	-45.2%
Great Pond - mid	**GT5**	**0.591**	**0.404**	**-31.7%**
Great Pond - lower	GT6	0.339	0.302	-11.0%
Green Pond - upper	G2	0.932	0.646	-30.6%
Green Pond - upper	G2a	0.792	0.570	-28.1%
Green Pond - mid	G3	0.642	0.487	-24.2%
Green Pond - mid	**G4**	**0.526**	**0.421**	**-19.9%**
Green Pond - lower	G5	0.409	0.355	-13.3%
Bournes Pond - upper	B2	0.901	0.566	-37.2%
Bournes Pond - mid	**B3**	**0.643**	**0.454**	**-29.3%**
Bournes Pond - lower	B4	0.426	0.355	-16.7%
Israels Cove	B5	0.633	0.424	-33.0%
Bournes Pond - lower	B6	0.340	0.312	-8.2%

Although the above modeling results provide one manner of achieving the selected threshold levels for the sentinel sites within these estuarine systems, the specific examples do not represent the only method for achieving this goal. However, the thresholds analysis provides general guidelines needed for the nitrogen management of this embayment. The Town of Falmouth has already provided sewering scenarios to the MEP that have been evaluated using the linked-watershed nitrogen model developed for each of the three Ashumet Valley systems. The model results and water quality implications of the Town’s sewering scenarios are discussed in the next chapter (Chapter IX).

IX. ALTERNATIVES TO IMPROVE TIDAL FLUSHING AND WATER QUALITY

IX.1 SEWERING ALTERNATIVES DEVELOPED BY THE TOWN

Following a meeting with MEP staff, the Town of Falmouth Wastewater Superintendent requested that the MEP evaluate three alternative scenarios involving sewering of selected areas within the Great, Green, and Bournes Ponds watersheds. The scenarios, including the extent of the sewered areas and the discharge locations, were provided by the Town of Falmouth. Figure IX-1 shows the extent of the proposed sewering for each the three alternatives. Based on the discussion with Falmouth, wastewater would be collected within selected sewer areas (A, B, and C), which are described along with the discharge locations and other details in Table IX-1. Scenario A assumes that all collected wastewater is discharged outside of the Ponds' watersheds, while the other two scenarios assume effluent treated to 3 ppm total nitrogen is discharged evenly among three golf course parcels (*i.e.*, Sites 1, 2, and 3). Site 1 is located in the Green Pond watershed, roughly half of Site 2 is located in the Bournes Pond watershed, and Site 3 and the other half of Site 2 are located outside of the study area watersheds (Figure IX-2). It should also be noted that portions of the sewered areas under each scenario collect wastewater from outside of the watersheds to Great, Green, and Bournes Ponds.

Evaluation of the nitrogen loading for each of the scenarios involved determining water use within each of the sewer areas and adjusting the wastewater-associated nitrogen loads within the associated sub-watersheds. Because these scenarios only address wastewater, there are no changes in other components of the nitrogen load (*e.g.*, lawn fertilizer). In Scenarios 2 and 3, nitrogen load from wastewater is removed from selected sub-watersheds and a lower load is added to the sub-watersheds where the treated effluent is discharged. The three golf course parcels selected by the Town for effluent discharge cross a number of existing watershed boundaries. Based on the discussion with the Town, it has been assumed that the effluent is equally distributed among the three parcels. Based on the watershed boundaries, the effluent discharged at Site 1 is a third of the total flow and completely within the Green Pond watershed. Site 2 is equally split between the Bournes and Eel Ponds watersheds; therefore, 50% (16.7% of the total) of the effluent discharged at Site 2 was placed in the Bournes Pond watershed and the remainder (16.7% of the total) in the Eel Pond watershed. Site 3 is assigned the remaining third and is located outside of the study area watersheds, where the entire effluent discharge would be to the Eel Pond watershed. Due to the watershed boundaries, distribution of the wastewater effluent evenly across the three golf course parcels (Sites 1, 2, and 3) will cause 50% of the effluent to be discharged into the Eel Pond watershed. Table IX-2 shows the changes in existing unattenuated wastewater nitrogen loads under each of the scenarios.

This analysis should be combined with a more refined analysis of the effluent discharge at the golf course sites. The discharge of 0.54 to 0.89 million gallons per day (MGD) at these three sites has the potential to alter groundwater flow paths and; therefore, change the watershed delineations and where the associated nitrogen loads are discharged. An analysis of groundwater impacts is being completed under an existing contract that the USGS has through the Barnstable County Wastewater Implementation Committee. Using the same groundwater model that was used to delineate the subwatersheds used in the MEP analysis, USGS can use the model's particle tracking capabilities to determine the percentage of flow ending up at ponds, estuaries, and rivers from effluent discharged at the golf course site. Following an evaluation of wastewater flows estimated by the MEP analysis, appropriate effluent peaking factors, likely discharge locations on the golf course sites, and flow-appropriate USGS particle tracking model runs, the Town could then use the MEP models to develop refined scenarios of potential water quality benefits and impacts.

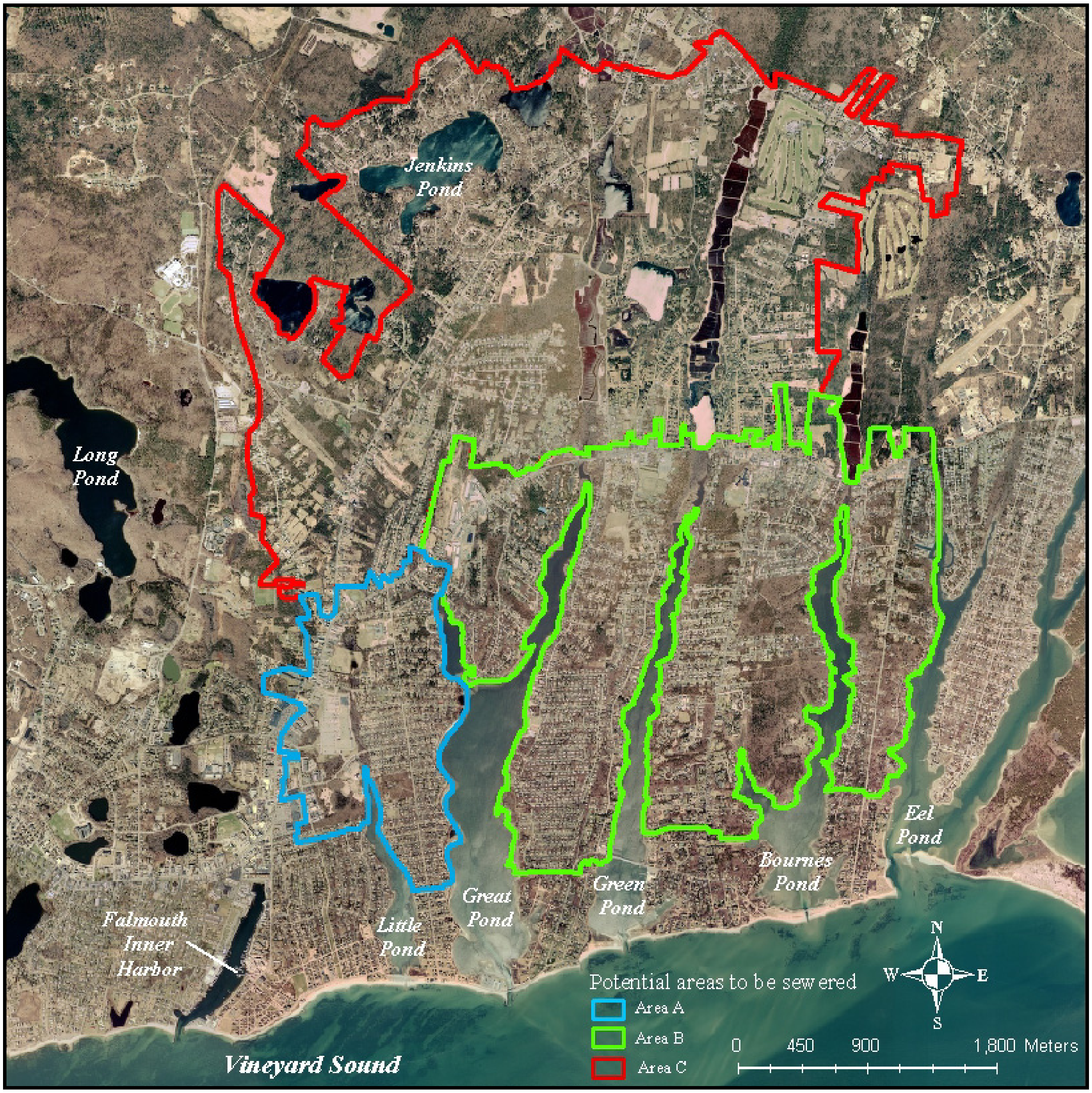


Figure IX-1. Sewer areas used to evaluate alternative nitrogen loading scenarios in the Great Pond, Green Pond, and Bournes Pond watersheds.

Table IX-1. Alternative Scenarios for Great, Green, and Bournes Pond					
Scenario #	Sewer Area[1]	Description	Discharge Location[1]	Effluent Concentration	Total Flow[2]
1	A	Interim Sewering – northern and mid portions of Maravista peninsula	Outside of Great, Green, and Bournes Pond watersheds	N/A	0.175 MGD
2	A & B	Scenario 1 plus northern and mid portions of peninsulas on both sides of Great, Green, and Bournes Ponds, south of Route 28	Discharge of Area B flows evenly distributed three golf course parcels; Area A flows discharged outside of watersheds	3 ppm TN	0.572 MGD
3	A, B, & C	Scenario 2 plus Sewer Area C (north of Route 28 in the Great, Green, and Bournes Pond watersheds	Discharge of Area B and C flows evenly distributed three golf course parcels; Area A flows discharged outside of watersheds	3 ppm TN	0.938 MGD

[1]Sewer Areas and discharge location shown in Figure X-1.
[2]Effluent flow within Sewer Areas based on existing water use (X 0.9 for consumptive loss); MGD = million gallons per day

For the purpose of this evaluation, it has been assumed that wastewater discharge would not influence watershed boundaries. In addition, reductions in nitrogen load are based on sewering the present development. Following determination of the unattenuated nitrogen loads from each of the three sewering scenarios, attenuated loads were developed for input to the water quality model. Determination of attenuated loads followed the method outlined in Chapter IV.

Scenario 1

The first alternative considered Falmouth's existing proposal to sewer most of the Maravista peninsula between Little and Great Ponds (Area A in Figure IX-1) and remove the wastewater from the watersheds of Great, Green, and Bournes Ponds. Since the watershed divide along the lower portion of Maravista is close to the eastern edge of the peninsula (as shown in Figure IX-2), the reduction in nitrogen load to Great Pond was expected to be relatively low for Scenario 1. However, a significant portion of the Perch Pond lower watershed would be affected by the proposed sewer project; therefore, the nitrogen load directly entering this sub-embayment would be significantly reduced. Tables IX-3 and IX-4 show a comparison of the nitrogen load entering the Great Pond system for present and Scenario 1 conditions. Since Scenario 1 would only affect the Great Pond system, water quality modeling of this scenario was limited to this single system. Daily total nitrogen loads utilized in the water quality modeling are shown in Table IX-5.

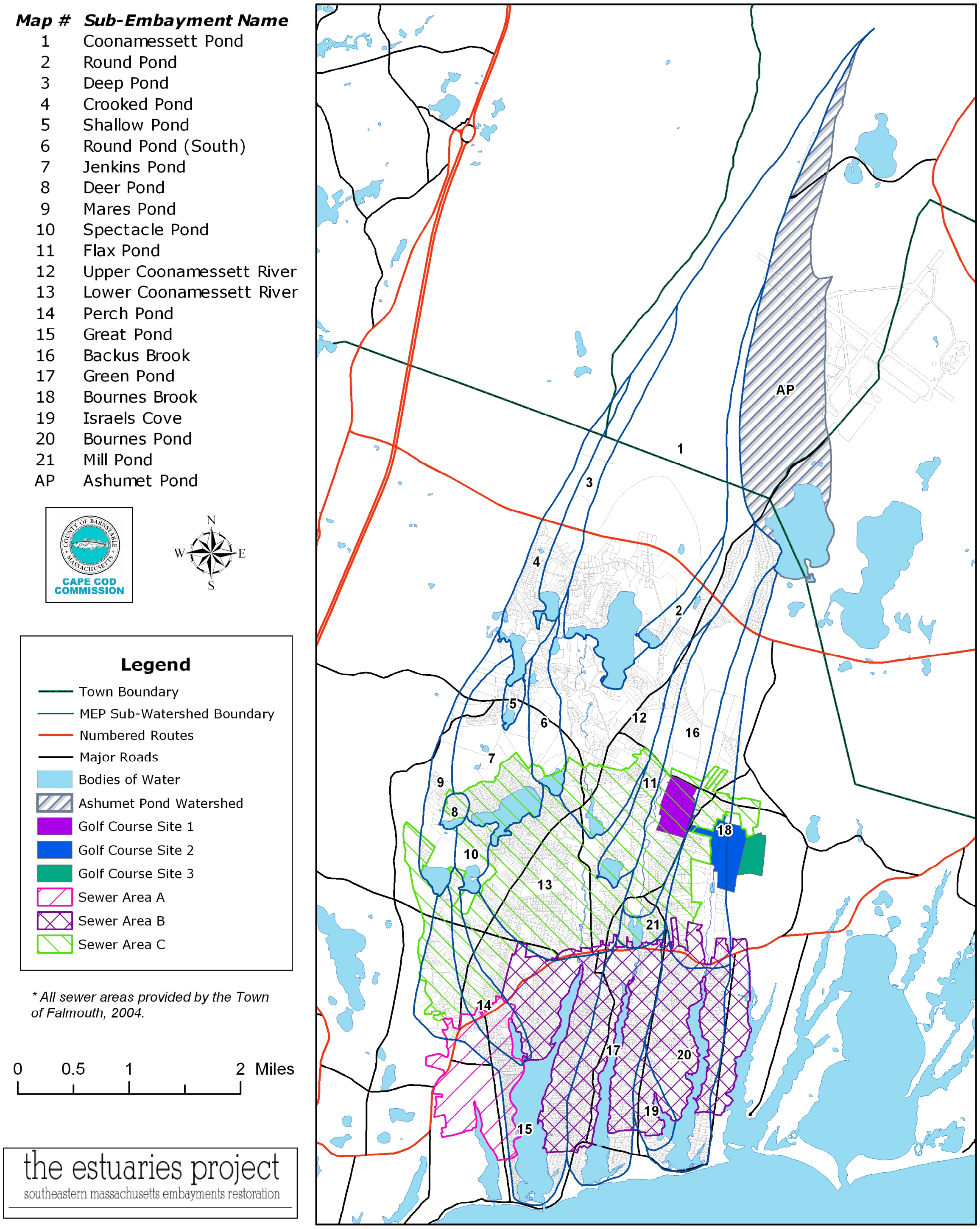


Figure IX-2. Sewer areas and effluent discharge locations used to evaluate alternative nitrogen loading scenarios in the Great Pond, Green Pond, and Bournes Pond watersheds.

Table IX-2. Net changes in unattenuated nitrogen loads in Great, Green, and Bournes Ponds subwatersheds resulting from alternative scenario analyses.

		Net Change in Nitrogen Load (kg/yr)		
Watershed Name	*Watershed #*	Scenario A	Scenario B	Scenario C
Coonamessett Pond	1	0	0	0
Round Pond	2	0	0	0
Deep Pond	3	0	0	0
Crooked Pond	4	0	0	0
Shallow Pond	5	0	0	0
Round Pond (South)	6	0	0	-21
Jenkins Pond	7	0	0	-687
Deer Pond	8	0	0	-142
Mares Pond	9	0	0	-67
Spectacle Pond	10	0	0	-151
Flax Pond	11	0	0	-490
Upper Coonamessett River	12	0	0	-725
Lower Coonamessett River	13	0	-283	-5823
Perch Pond	14	-832	-917	-1530
Great Pond	15	-1456	-5765	-7324
Backus Brook	16	0	+545	-568
Green Pond	17	0	-5121	-5239
Bournes Brook	18	0	-125	-750
Israels Cove	19	0	-594	-594
Bournes Pond	20	0	-2476	-2501
Mill Pond	21	0	-3	-703

Table IX-3. Comparison of sub-embayment watershed ***septic total nitrogen loads*** (attenuated) used for modeling of present and Scenario 1 loading conditions of the Ashumet Valley systems (Maravista, Great 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)	Scenario 1 septic load (kg/day)	septic load % change
Great Pond	21.28	17.29	-18.7%
Perch Pond	4.47	2.19	-50.9%
Surface Water Sources			
Coonamessett River (Great Pond)	15.08	15.08	0.0%

The total nitrogen modeling indicated only a minor reduction (<5%) in total nitrogen concentrations as a result of Scenario 1 within the main body of Great Pond (Table IX-6). However, a reduction in total nitrogen concentration of more than 12% would be achieved in Perch Pond using Scenario 1. Unfortunately, this reduction likely would not lower the nitrogen concentration in Perch Pond to a level that would be restorative to benthic habitat. In addition, Scenario 1 alone will not achieve the desired threshold nitrogen concentration (0.40 mg/L) at the sentinel threshold station within Great Pond, GT5. Figure IX-3 illustrates the tidally-averaged nitrogen conditions modeled for Scenario 1.

Table IX-4. Comparison of sub-embayment ***total nitrogen watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and Scenario 1 loading conditions of the Ashumet Valley systems. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present load (kg/day)	Scenario 1 load (kg/day)	load % change
Great Pond	25.00	21.01	-16.0%
Perch Pond	5.38	3.10	-42.4%
Surface Water Sources			
Coonamessett River (Great Pond)	22.63	22.63	0.0%

Table IX-5. Scenario 1 sub-embayment and surface water loads used for total nitrogen modeling of the Ashumet Valley systems, 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)
Great Pond	21.01	3.22	0.32
Perch Pond	3.22	0.22	-0.90
Surface Water Sources			
Coonamessett River (Great Pond)	22.63	-	-

Table IX-6. Comparison of model average total N concentrations from present loading and the modeled Scenario 1, with percent change, for the Great Pond system. The sentinel threshold station is in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	Scenario 1 (mg/L)	% change
Coonamessett River (estuarine)	GT2	0.875	0.839	-4.1%
Great Pond - upper	GT3	0.782	0.747	-4.5%
Perch Pond	GT4	0.859	0.753	-12.3%
Great Pond - mid	**GT5**	**0.591**	**0.564**	**-4.5%**
Great Pond - lower	GT6	0.339	0.332	-1.9%

Scenario 2

The second alternative included the sewer option described in Scenario 1, as well as sewering a majority of the watershed areas to Great, Green, and Bournes Ponds south of Route 28 (Area B in Figure IX-1). As described above, the wastewater generated from Area B would be treated and discharged into the three golf course parcels. Since much of the septic system nitrogen load north of Route 28 enters a stream and/or a pond before discharging into estuarine waters, some natural attenuation occurs within the upper (northern) portions of each watershed. Therefore, removal of a septic system nitrogen load that flows directly through groundwater to an estuary will have a larger effect on water quality than removal of the nitrogen load from an equivalent septic system that is in the upper watershed, where the nitrogen load will flow through a stream and/or ponds prior to entering the estuary. Scenario 2 includes sewer construction within most of the areas where septic system loads flow directly through groundwater to the estuary. Tables IX-7 and IX-8 show a comparison of the nitrogen load entering the Great, Green, and Bournes Pond systems for present and Scenario 2 conditions. Significant reductions in septic load result from implementation of Scenario 2 for almost all water bodies. However, a marked increase in nitrogen load to Backus Brook results from discharging a portion of the treated wastewater back into this watershed. Daily total nitrogen loads utilized in the water quality modeling are shown in Table IX-9.

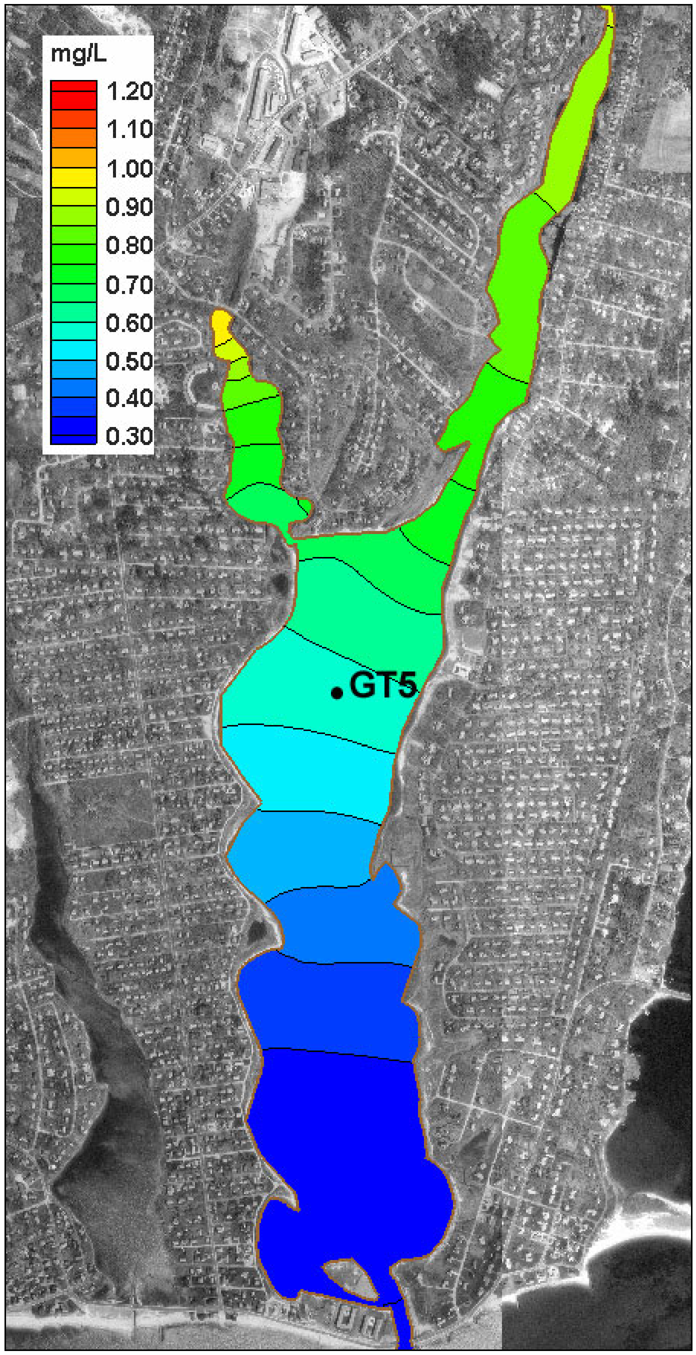


Figure IX-3. Contour plot of average total nitrogen concentrations from results of the Scenario 1 loading conditions for Great Pond. The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

Table IX-7. Comparison of sub-embayment watershed ***septic total nitrogen loads*** (attenuated) used for modeling of present and Scenario 2 (lower watersheds) loading conditions of the Ashumet Valley systems. 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)	Scenario 2 septic load (kg/day)	septic load % change
Great Pond	21.28	5.49	-74.2%
Perch Pond	4.47	1.96	-56.2%
Green Pond	16.62	2.52	-84.8%
Bournes Pond	8.30	1.52	-81.7%
Israels Cove	1.78	0.16	-91.2%
Surface Water Sources			
Coonamessett River (Great Pond)	15.08	14.70	-2.5%
Backus Brook (Green Pond)	2.08	2.57	+23.5%
Bournes Brook (Bournes Pond)	2.41	2.25	-6.6%

Table IX-8. Comparison of sub-embayment ***total nitrogen watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and Scenario 2 (lower watersheds) loading conditions of the Ashumet Valley systems. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	Scenario 2 load (kg/day)	load % change
Great Pond	25.00	9.20	-63.2%
Perch Pond	5.38	2.86	-46.8%
Green Pond	18.55	4.44	-76.0%
Bournes Pond	9.61	2.83	-70.6%
Israels Cove	2.05	0.43	-79.2%
Surface Water Sources			
Coonamessett River (Great Pond)	22.63	22.25	-1.7%
Backus Brook (Green Pond)	3.81	4.29	+12.8%
Bournes Brook (Bournes Pond)	3.29	3.13	-4.8%

Table IX-9. Scenario 2 (lower watersheds) sub-embayment and surface water loads used for total nitrogen modeling of the Ashumet Valley systems, 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)
Great Pond	9.20	3.22	0.39
Perch Pond	2.86	0.22	-0.90
Green Pond	4.44	1.61	30.03
Bournes Pond	2.83	1.61	21.80
Israels Cove	0.43	0.26	-0.16
Surface Water Sources			
Coonamessett River (Great Pond)	22.25	-	-
Backus Brook (Green Pond)	4.29	-	-
Bournes Brook (Bournes Pond)	3.13	-	-

As presented in Table IX-10, the total nitrogen modeling indicated substantial reductions in total nitrogen concentrations as a result of Scenario 2, especially in the upper reaches of Green and Bournes Ponds where nitrogen concentrations were lowered by 35% and 28%, respectively. Although the nitrogen load entering the estuarine systems at the northern end (through the streams) remained similar or increased slightly, the effect of tidal dispersion caused the maximum reduction in nitrogen concentration to occur in the upper reaches of each pond.

For Green Pond, the total nitrogen level modeled for Scenario 2 is lower than the threshold value determined for this embayment. The threshold nitrogen concentration at the sentinel station (G4) is 0.42 mg/L and the modeled nitrogen concentration for Scenario 2 was 0.403 mg/L. For Great and Bournes Ponds, the total nitrogen level modeled for Scenario 2 would not meet the threshold values at the sentinel stations. For Great Pond, the threshold nitrogen concentration at the sentinel station (GT5) is 0.40 mg/L and the modeled nitrogen concentration for Scenario 2 was 0.49 mg/L. For Bournes Pond, the threshold nitrogen concentration at the sentinel station (B3) is 0.45 mg/L and the modeled nitrogen concentration for Scenario 2 was 0.50 mg/L. Figures IX-4 through IX-6 illustrate the tidally-averaged nitrogen conditions modeled for Scenario 2 in Great, Green, and Bournes Ponds, respectively.

Table IX-10. Comparison of model average total N concentrations from present loading and the modeled Scenario 2 (lower watersheds) scenario, with percent change, for the Ashumet Valley systems. Sentinel threshold stations are in bold print.

Sub-Embayment	monitoring station	present (mg/L)	Scenario 2 (mg/L)	% change
Coonamessett River (estuarine)	GT2	0.875	0.723	-17.4%
Great Pond - upper	GT3	0.782	0.635	-18.8%
Perch Pond	GT4	0.859	0.643	-25.2%
Great Pond - mid	**GT5**	**0.591**	**0.486**	**-17.8%**
Great Pond - lower	GT6	0.339	0.317	-6.3%
Green Pond - upper	G2	0.932	0.589	-36.8%
Green Pond - upper	G2a	0.792	0.525	-33.8%
Green Pond - mid	G3	0.642	0.457	-28.9%
Green Pond - mid	**G4**	**0.526**	**0.403**	**-23.3%**
Green Pond - lower	G5	0.409	0.348	-14.9%
Bournes Pond - upper	B2	0.901	0.648	-28.0%
Bournes Pond - mid	**B3**	**0.643**	**0.497**	**-22.7%**
Bournes Pond - lower	B4	0.426	0.371	-12.9%
Israels Cove	B5	0.633	0.456	-28.0%
Bournes Pond - lower	B6	0.340	0.319	-6.1%

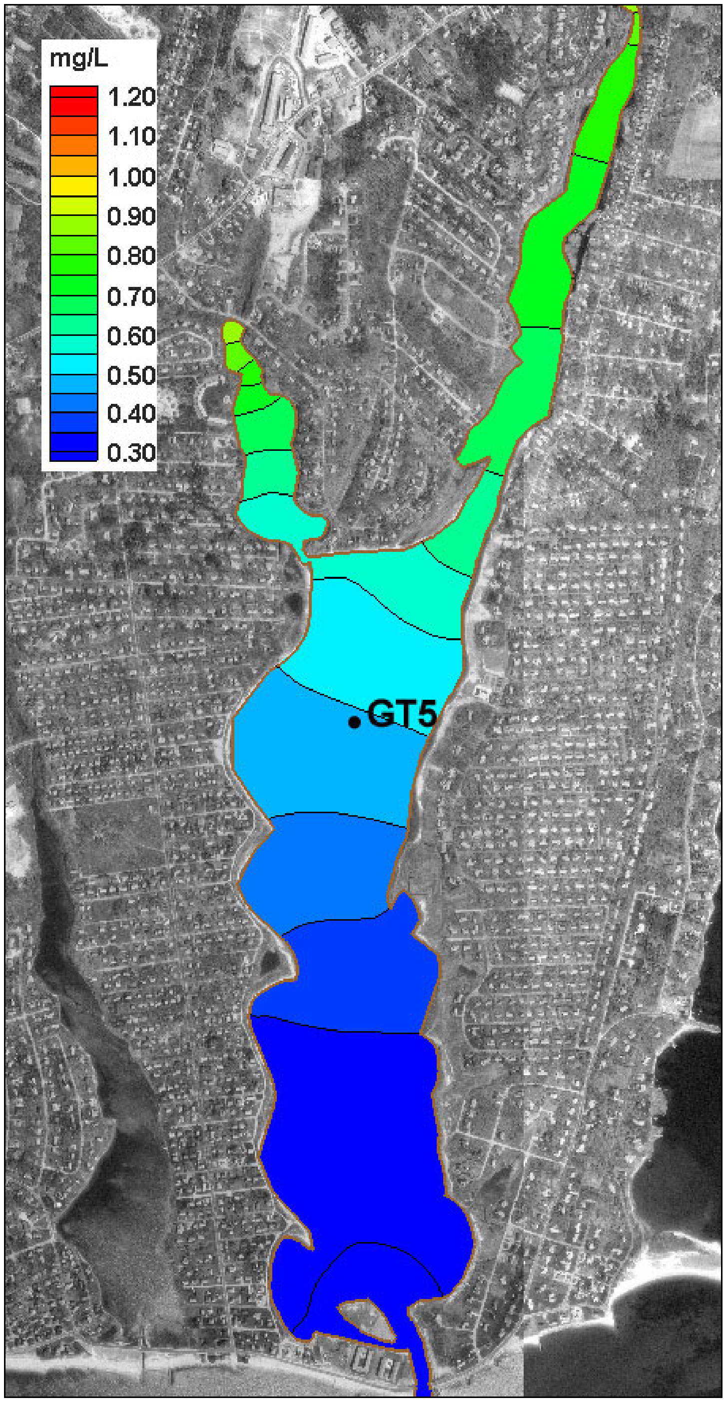


Figure IX-4. Contour plot of average total nitrogen concentrations from results of the Scenario 2 loading condition for Great Pond. The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

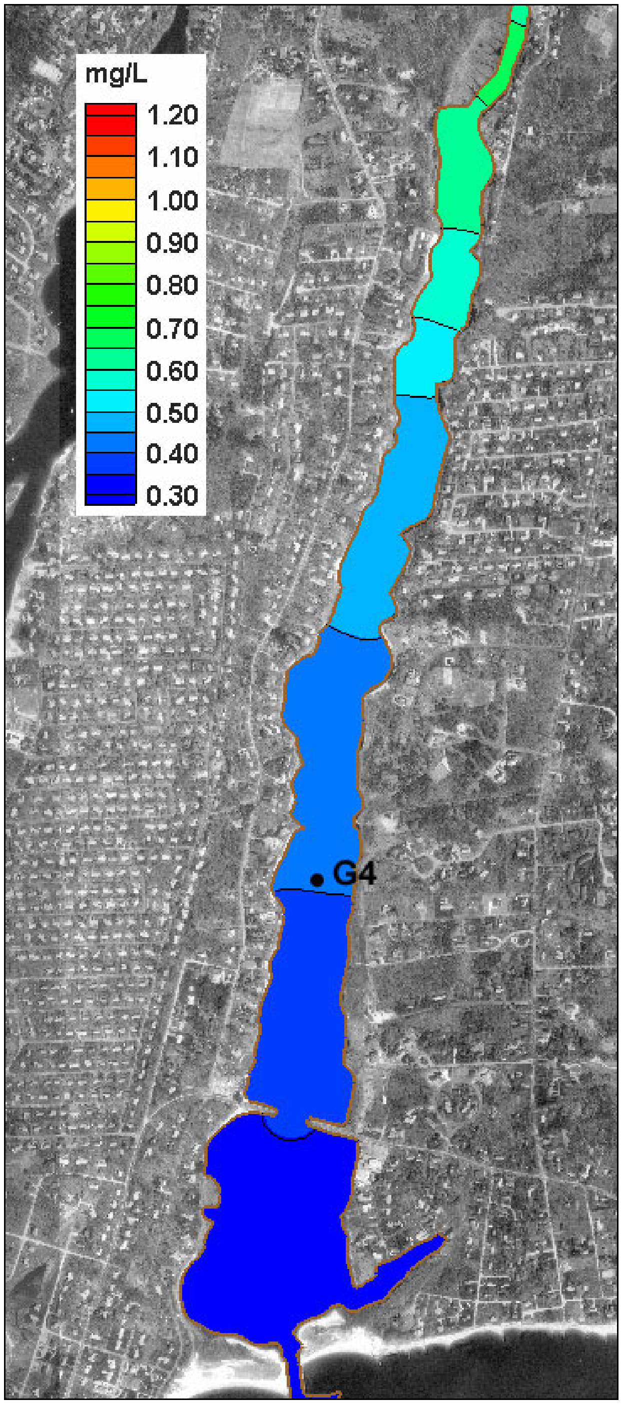


Figure IX-5. Contour plot of average total nitrogen concentrations from results of the Scenario 2 loading condition for Green Pond. The approximate location of the sentinel threshold station for Green Pond (G4) is shown.

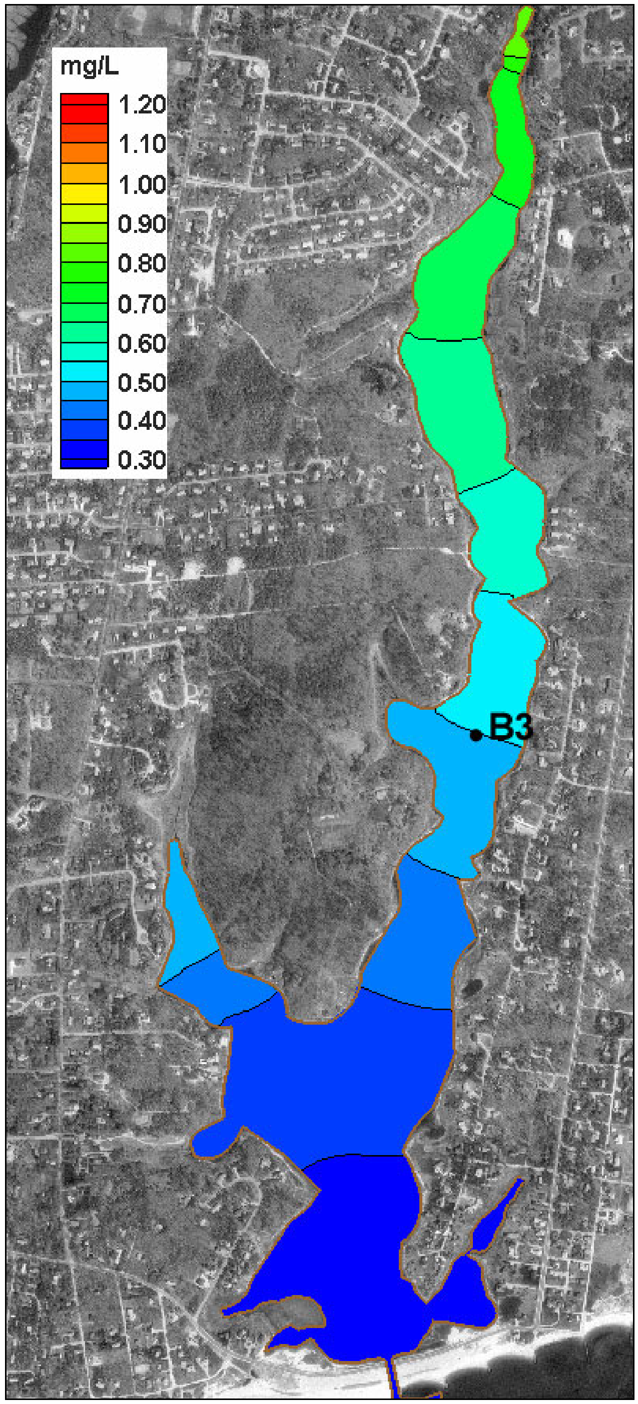


Figure IX-6. Contour plot of average total nitrogen concentrations from results of the Scenario 2 loading condition, for Bournes Pond. The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

Scenario 3

The third alternative included the sewer option described in Scenario 2, as well as sewering a substantial portion of the developed watershed areas to Great, Green, and Bournes Ponds north of Route 28 (Area C in Figure IX-1). As described above, the wastewater generated from Areas B and C would be treated and discharged into the three golf course parcels. Area C includes sewer construction within areas where groundwater typically flows through a pond and/or a stream prior to entering the estuary; therefore, sewering in these areas will be slightly less efficient with regard to nutrients entering the estuary, since natural attenuation would remove a portion of the nitrogen generated by a septic systems. Tables IX-11 and IX-12 show a comparison of the nitrogen load entering the Great, Green, and Bournes Pond systems for present and Scenario 3 conditions. Significant reductions in septic load result from implementation of Scenario 3 for almost all water bodies. However, a small increase in nitrogen loads to Backus Brook results from discharging a portion of the treated wastewater back into this watershed. Daily total nitrogen loads utilized in the water quality modeling are shown in Table IX-13.

Table IX-11. Comparison of sub-embayment watershed ***septic total nitrogen loads*** (attenuated) used for modeling of present and Scenario 3 (upper and lower watersheds) loading conditions of the Ashumet Valley systems. 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)	septic load % change
Great Pond	21.28	1.01	-95.3%
Perch Pond	4.47	0.12	-97.4%
Green Pond	16.62	2.20	-86.8%
Bournes Pond	8.30	1.45	-82.5%
Israels Cove	1.78	0.16	-91.2%
Surface Water Sources			
Coonamessett River (Great Pond)	15.08	4.85	-67.8%
Backus Brook (Green Pond)	2.08	1.57	-25.5%
Bournes Brook (Bournes Pond)	2.41	1.48	-38.7%

Table IX-12. Comparison of sub-embayment ***total nitrogen watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and Scenario 3 (upper and lower watersheds) loading conditions of the Ashumet Valley systems. 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)	% change
Great Pond	25.00	4.72	-81.1%
Perch Pond	5.38	1.02	-81.1%
Green Pond	18.55	4.12	-77.8%
Bournes Pond	9.61	2.76	-71.3%
Israels Cove	2.05	0.43	-79.2%
Surface Water Sources			
Coonamessett River (Great Pond)	22.63	12.40	-45.2%
Backus Brook (Green Pond)	3.81	3.3	-13.4%
Bournes Brook (Bournes Pond)	3.29	2.36	-28.3%

Table IX-13. Scenario 3 (upper and lower watersheds) sub-embayment and surface water loads used for total nitrogen modeling of the Ashumet Valley systems, 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)
Great Pond	4.72	3.22	0.48
Perch Pond	1.02	0.22	-0.55
Green Pond	4.12	1.61	27.98
Bournes Pond	2.76	1.61	21.01
Israels Cove	0.43	0.26	-0.16
Surface Water Sources			
Coonamessett River (Great Pond)	12.40	-	-
Backus Brook (Green Pond)	3.3	-	-
Bournes Brook (Bournes Pond)	2.36	-	-

As presented in Table IX-14, the total nitrogen modeling indicated substantial reductions in total nitrogen concentrations as a result of Scenario 3, especially in the upper reaches of the ponds, where nitrogen concentrations were lowered by 43%, 40% and 32% for Great, Green, and Bournes Ponds, respectively. Although the nitrogen load entering the estuarine systems at the northern end (through the streams) remained significant or increased slightly, the effect of tidal dispersion caused the maximum reduction in nitrogen concentration to occur in the upper reaches of each pond.

For Green Pond, the total nitrogen level modeled for Scenario 3 is significantly lower than the threshold value determined for this embayment. The threshold nitrogen concentration at the sentinel station (G4) is 0.42 mg/L and the modeled nitrogen concentration for Scenario 3 was 0.39 mg/L. For Great Pond, the total nitrogen level modeled for Scenario 3 meets the threshold value determined for this embayment. The threshold nitrogen concentration at the sentinel station (GT5) is 0.40 mg/L and the modeled nitrogen concentration for Scenario 3 was 0.39 mg/L. Unfortunately, in Bournes Pond, the total nitrogen level modeled for Scenario 3 does not meet the threshold values at the sentinel station. For Bournes Pond, the threshold nitrogen concentration at the sentinel station (B3) is 0.45 mg/L and the modeled nitrogen concentration for Scenario 3 was 0.48 mg/L. The tidal restriction at the Bournes Pond inlet likely is responsible for the limited effect of nitrogen removal on this system. The influence of the inlet configuration on water quality is evaluated in the following section. Figures IX-7 through IX-9 illustrate the tidally-averaged nitrogen conditions modeled for Scenario 3 in Great, Green, and Bournes Ponds, respectively.

Table IX-14. Comparison of model average total N concentrations from present loading and the modeled Scenario 3 (upper and lower watersheds), with percent change, for the Ashumet Valley systems. Sentinel threshold stations are in bold print.				
Sub-Embayment	monitoring station	present (mg/L)	Scenario 3 (mg/L)	% change
Coonamessett River (estuarine)	GT2	0.875	0.504	-42.5%
Great Pond - upper	GT3	0.782	0.463	-40.8%
Perch Pond	GT4	0.859	0.451	-47.5%
Great Pond - mid	**GT5**	**0.591**	**0.386**	**-34.8%**
Great Pond - lower	GT6	0.339	0.299	-11.8%
Green Pond - upper	G2	0.932	0.543	-41.8%
Green Pond - upper	G2a	0.792	0.497	-37.3%
Green Pond - mid	G3	0.642	0.441	-31.4%
Green Pond - mid	**G4**	**0.526**	**0.393**	**-25.3%**
Green Pond - lower	G5	0.409	0.343	-16.2%
Bournes Pond - upper	B2	0.901	0.611	-32.2%
Bournes Pond - mid	**B3**	**0.643**	**0.479**	**-25.4%**
Bournes Pond - lower	B4	0.426	0.365	-14.3%
Israels Cove	B5	0.633	0.448	-29.3%
Bournes Pond - lower	B6	0.340	0.317	-6.7%

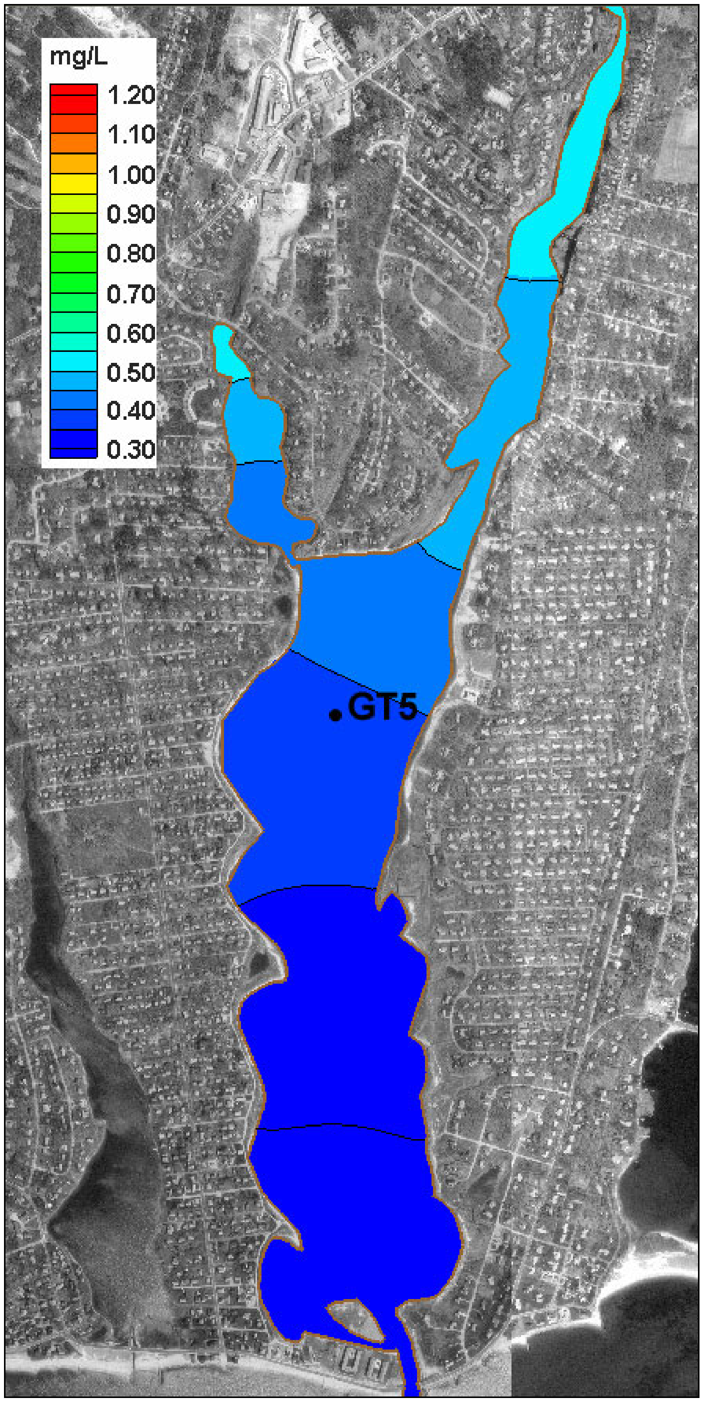


Figure IX-7. Contour plot of average total nitrogen concentrations from results of the Scenario 3 loading condition, for Great Pond. The approximate location of the sentinel threshold station for Great Pond (GT5) is shown.

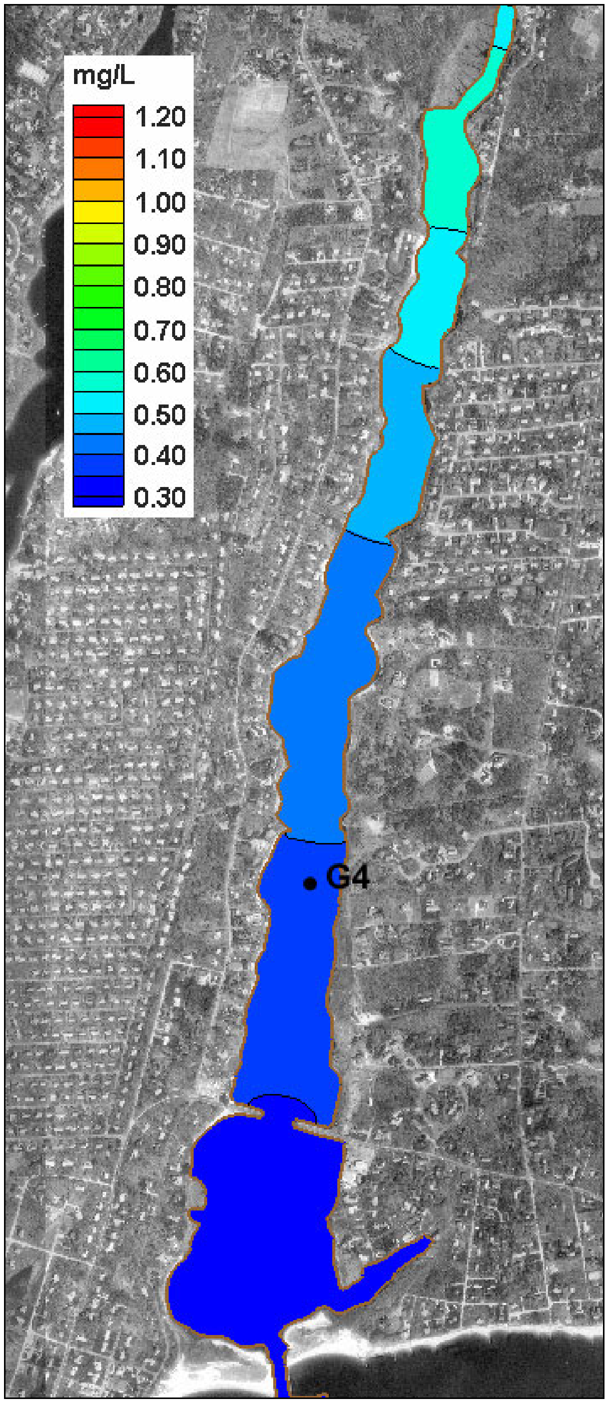


Figure IX-8. Contour plot of average total nitrogen concentrations from results of the Scenario 3 loading condition, for Green Pond. The approximate location of the sentinel threshold station for Green Pond (G4) is shown.

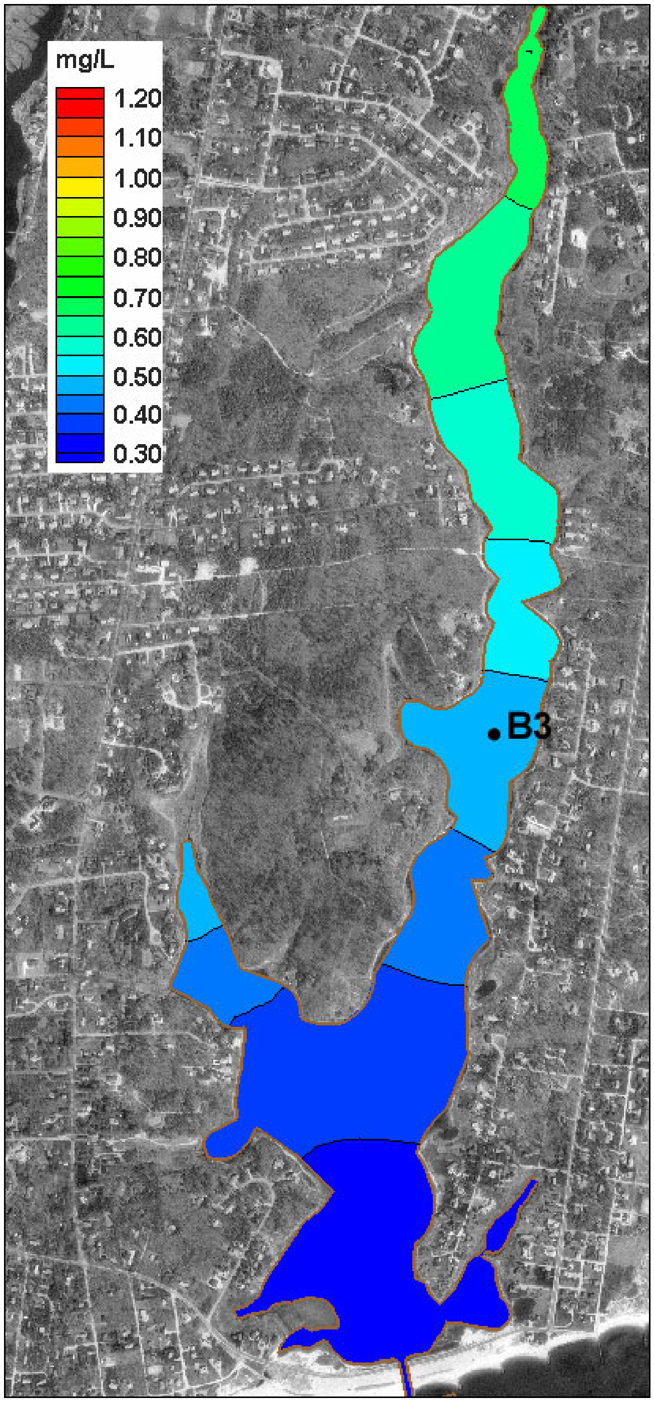


Figure IX-9. Contour plot of average total nitrogen concentrations from results of the Scenario 3 loading condition, for Bournes Pond. The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

IX.2 FLUSHING IMPROVEMENTS TO BOURNES POND BY MODIFICATIONS TO THE INLET

In addition to sewering, water quality improvements may be possible by improving tidal exchange in an estuary. For the Ashumet Valley embayments, the single system which could benefit possibly from flushing improvements is Bournes Pond. Tidal attenuation is greatest for this system, where the average tide range is less than 88% 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.

Improvements to tidal flushing would be possible if Bournes Pond inlet were widened. A model simulation was executed to simulate Bournes Pond hydrodynamics with an improved 100 ft-wide inlet, which is twice the width of the existing jettied inlet. This proposed larger Bournes Pond inlet would have the same width as the existing Green Pond inlet; therefore, a similar reduction in tidal attenuation was expected.

Though present flushing conditions through Bournes Pond Inlet are not ideal, they do represent a great improvement over those that existed before the Inlet was relocated in 1985. The pre-1985 Inlet was approximately 290 ft west of its present position. The historical inlet to Bournes Pond was not structured, and over time was becoming less efficient as it slowly filled with sediment. Historical data show that the spring tide range in the pond was at times as small as 0.6 ft before the inlet relocation (Weston and Sampson, 1981). Though a wider 90 ft inlet (at the pre-1985 location) was investigated as a possible alternative to the present 50 ft inlet, it was rejected primarily due to maintenance dredging costs estimated at the time of the project feasibility study (Weston and Sampson, 1981). The 1981 study concluded that the optimum high-tide cross-sectional area (based on an inlet stability analysis) of the inlet was 220 ft^2, with a bridge span of 50 ft. However, the recent maintenance dredging history of Green Pond and Bournes Pond Inlets suggest that Bournes Pond Inlet could be widened without any increase in annual maintenance costs. This estimate is based on the observation that though Green Pond Inlet is twice as wide as Bournes Pond inlet and the tide prisms of the two ponds are similar, the annual dredged volumes of the two inlet are also similar (~1000 yd^3/yr, from personal communication with Barnstable County Dredge personnel, 2004).

Hydrodynamic model results for existing and improved inlet conditions are presented in Figure IX-10. 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 dramatically reduced for the proposed 100 ft-wide inlet, to the point where there is little difference between the range and phase of both tide signals.

Base on model output, the average tide prism increases by 28% with the improved inlet. Average volumes of Bournes Pond for existing conditions and for the 100 ft-wide inlet scenario are presented in Table IX-15. As a result of the increased tide prism volume and the reduced mean tide volume of the system, the computed system residence time decreases to 0.82 days from 1.09 days for existing conditions.

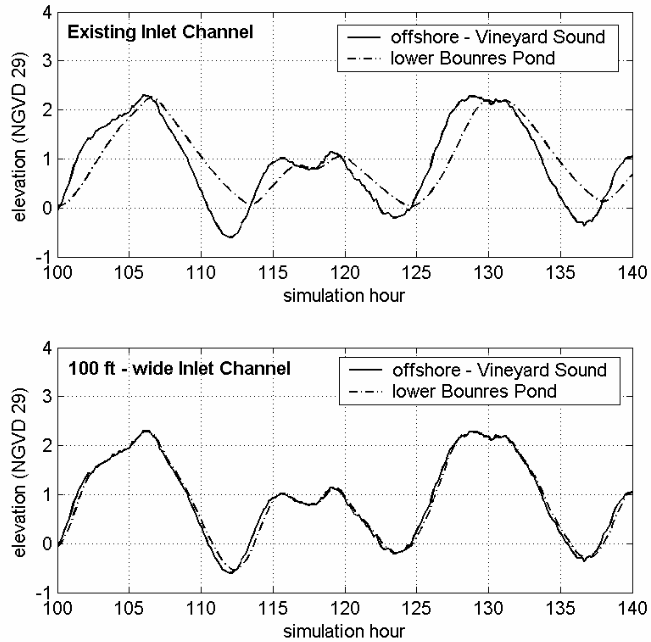


Figure IX-10. Plots showing a comparison of typical tides for modeled existing conditions (top plot) and proposed improved 100 ft-wide inlet (bottom plot) to Bournes Pond.

Table IX-15. Average high, mid and low tide volumes, with mean tide prism for Bournes Pond, for existing inlet conditions, and for the proposed 100 ft-wide inlet modification.

	existing inlet ft^3	100 ft-wide inlet ft^3	% change
Mean High Tide Volume	27,718,500	28,291,900	+2.1%
Mean Tide Volume	22,408,300	21,508,000	-4.0%
Mean Low Tide Volume	17,098,100	14,724,200	-13.9%
Mean Prism Volume	10,620,400	13,567,700	+27.8%

Water quality model runs were performed using the hydrodynamic model output of the proposed 100 ft-wide Bournes Pond inlet. First, present loading conditions were modeled. Second, loading conditions for Scenario 2 were modeled. Results from the present loading conditions with the improved hydrodynamics of the widened inlet are presented in Table IX-16, and plotted in Figure IX-11. Although TN concentrations are significantly reduced (i.e., up to an 11% reduction in the mid-to-lower portion of the Pond), the reduction is not large enough to meet the threshold limits set for Bournes Pond (0.45 mg/L at water quality monitoring station B3).

However, threshold concentrations are achieved using the Scenario 2 loading (presented in Table IX-17 and Figure IX-12) with the widened inlet. Therefore, a combination of sewering the lower watershed and increasing the width of the inlet would improve the system to a level that meets the selected restoration threshold. It should be noted that even sewering the upper and lower watersheds would not achieve conditions restorative of eelgrass with the existing inlet configuration (Section IX-1). Widening the inlet would certainly make the threshold limit more practically attainable, where significantly less nitrogen load 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-16. Comparison of model average total N concentrations from present loading and the widened inlet channel (100 ft) scenario with present loading, with percent change, for the Ashumet Valley systems.

Sub-Embayment	monitoring station	present (mg/L)	Channel mod, present (mg/L)	% change
Bournes Pond - upper	B2	0.875	0.829	-7.9%
Bournes Pond - mid	**B3**	**0.782**	**0.569**	**-11.4%**
Bournes Pond - lower	B4	0.859	0.378	-11.2%
Israels Cove	B5	0.591	0.581	-8.3%
Bournes Pond - lower	B6	0.339	0.323	-4.7%

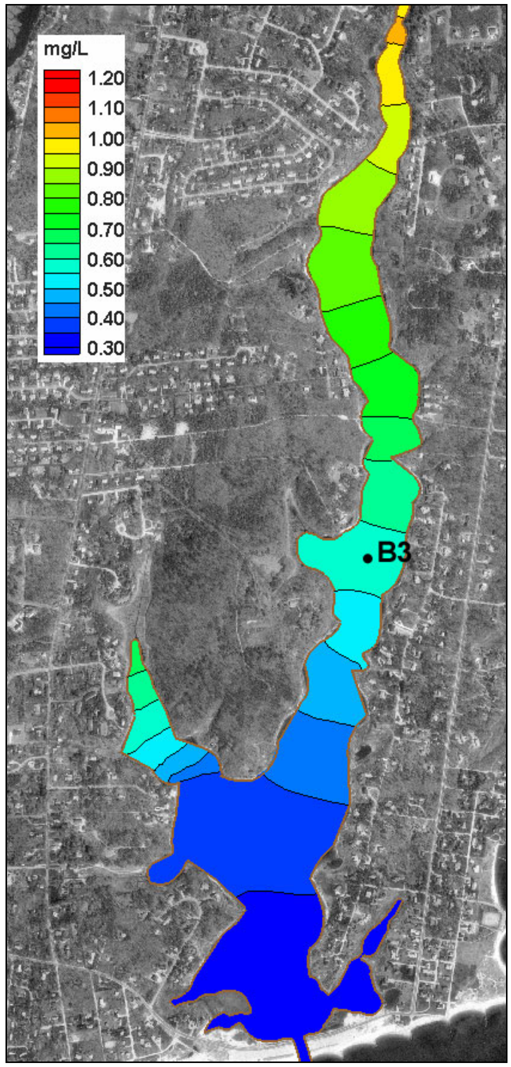


Figure IX-11. Contour Plot of modeled total nitrogen concentrations (mg/L) in Bournes Pond, for present loading conditions, and widened inlet channel (100 ft). The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

Table IX-17. Comparison of model average total N concentrations from present loading and the modeled Scenario 2 (lower watershed) with widened inlet channel (100 ft) scenario, with percent change, for the Ashumet Valley systems.

Sub-Embayment	monitoring station	present (mg/L)	Channel mod, 100-ft wide (mg/L)	% change
Bournes Pond - upper	B2	0.875	0.605	-32.8%
Bournes Pond - mid	**B3**	**0.782**	**0.452**	**-29.6%**
Bournes Pond - lower	B4	0.859	0.341	-20.0%
Israels Cove	B5	0.591	0.429	-32.3%
Bournes Pond - lower	B6	0.339	0.308	-9.2%

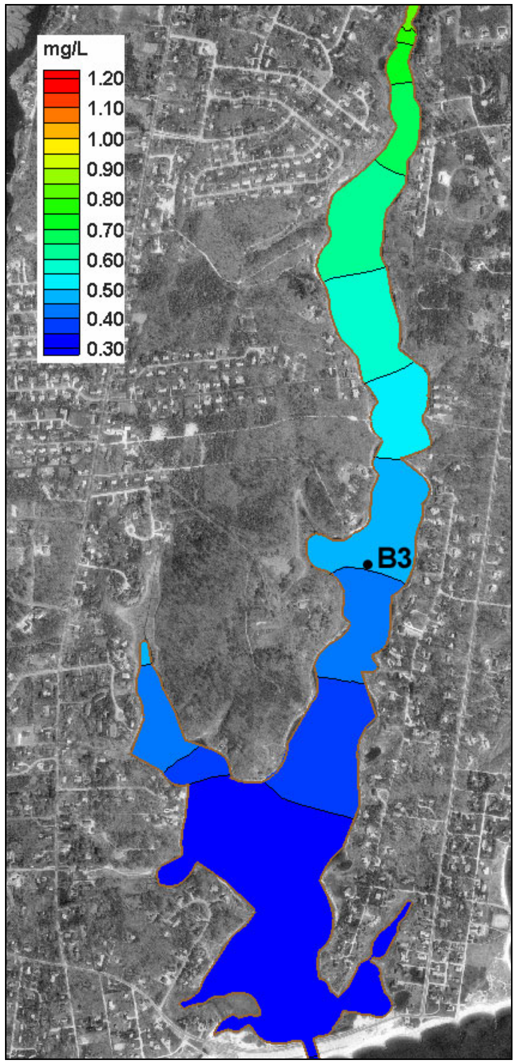


Figure IX-12. Contour Plot of modeled total nitrogen concentrations (mg/L) in Bournes Pond, for Scenario 2 (lower watershed) loading conditions, and widened inlet channel (100 ft). The approximate location of the sentinel threshold station for Bournes Pond (B3) is shown.

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