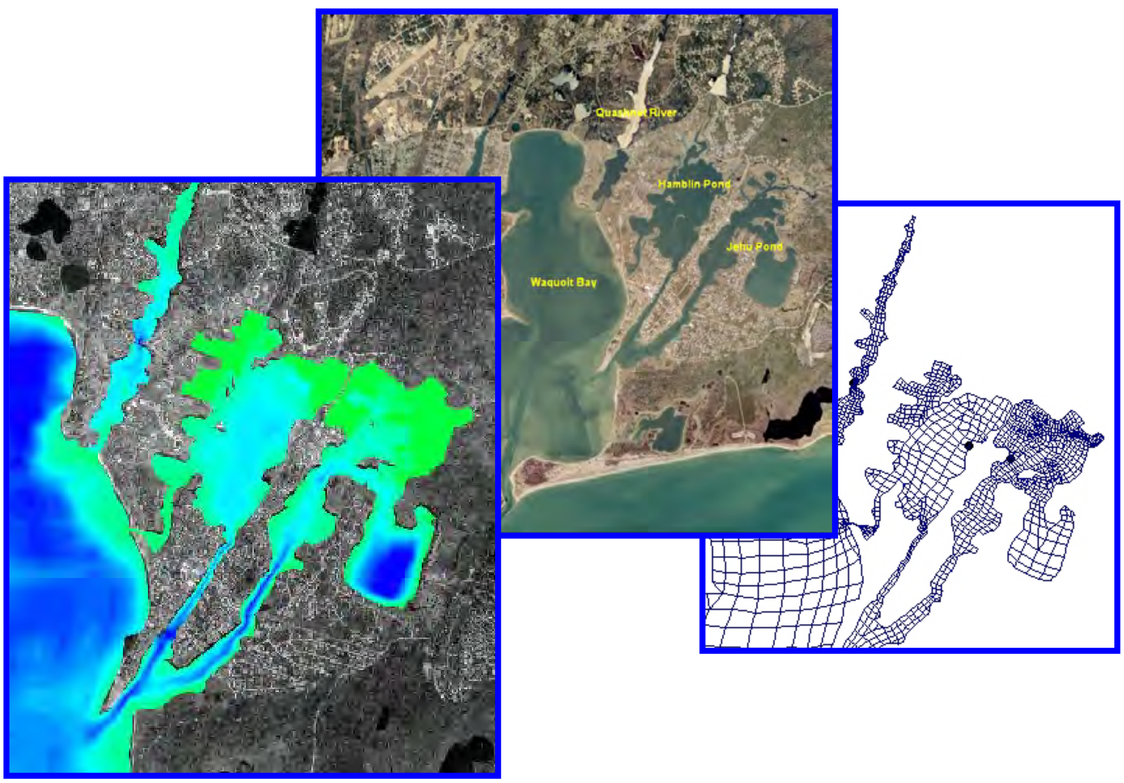
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Quashnet River, Hamblin Pond, and Jehu Pond, in the Waquoit Bay System of the Towns of Mashpee and Falmouth, MA



University of Massachusetts Dartmouth
School of Marine Science and Technology

Massachusetts Department of
Environmental Protection

FINAL REPORT – JANUARY 2005

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LINKED WATERSHED-EMBAYMENT MODEL TO DETERMINE CRITICAL NITROGEN LOADING THRESHOLDS FOR THE QUASHNET RIVER, HAMBLIN POND, AND JEHU POND, IN THE WAQUOIT BAY SYSTEM OF THE TOWNS OF MASHPEE AND FALMOUTH, MA

FINAL REPORT – JANUARY 2005

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I. INTRODUCTION

The Quashnet River Estuary, Hamblin Pond/Little River Estuary, and Jehu Pond/Great River Estuary are 3 major tributary sub-embayments to the Waquoit Bay System and are located along its eastern shore. These three sub-estuaries were prioritized for initial analysis by the DEP/SMAST Massachusetts Estuaries Project (MEP) to support on-going nitrogen management planning by the Town of Mashpee. These systems will be revisited and fully integrated into the entire Waquoit Bay System synthesis and modeling effort, at the point that baseline monitoring data and appropriate nitrogen loading and cycling data are available for the greater system.

The eastern Waquoit Bay sub-embayments (Quashnet River, Hamblin Pond/Little River, and Jehu Pond/Great River) are located within the Towns of Falmouth (north & west) and Mashpee (east), on Cape Cod Massachusetts. To the south is a barrier beach that separates the Waquoit Bay System from adjacent Nantucket Sound (Figure I-1). At present, each of the three sub-estuaries exchanges tidal waters with the main basin of Waquoit Bay, which receives tidal flows from Nantucket Sound. The main Bay has two main openings to Nantucket Sound waters, a historically open inlet in the main Bay and an ephemeral inlet that connects Eel Pond to Nantucket Sound. More recently, Hurricane Bob in 1991 created a third inlet immediately east of the Eel Pond entrance; however, this inlet has closed over the past few years.

The primary ecological threat to the eastern Waquoit Bay sub-embayment (Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River) resources is degradation resulting from continuing nutrient enrichment. Although the watershed and the Bay have some organic contamination and bacterial contamination issues, these do not appear to be having large system-wide impacts. Organic contamination has been associated with groundwater recharged in the upper watershed, within the Massachusetts Military Reservation (MMR). Plumes enter two of the major freshwater ponds, Ashumet Pond and Johns Pond, which eventually provide freshwater to the eastern sub-embayments of the Waquoit Bay System. The Ashumet Pond plume is mainly secondarily treated wastewater previously discharged to groundwater infiltration beds at the former MMR Wastewater Treatment Facility. The John’s Pond plume stems from a relatively small input of organic contamination. However, it is unlikely that the organic contaminants associated with Johns Pond have any significant effect on Waquoit Bay, due to the passage through Johns Pond and the mode of transport (surface water flow). It is likely that some fraction of the nitrogen loading from the wastewater plume entering Ashumet Pond does contribute to the overall nitrogen loading to the Waquoit System. However, this potential load has decreased in recent years, due to MMR’s reduction of land disposal of treated wastewater. Moreover, relative to the longer term, MMR has relocated its disposal beds to an area near the Cape Cod Canal and the disposal area from where the nutrient rich plume originated has been abandoned. Also, the nitrogen within this plume appears to be significantly attenuated by pond ecosystem function. In addition, the wastewater plume is primarily moving through the Ashumet Valley to Great and Green Ponds in Falmouth, rather than towards the Waquoit System. This nitrogen source was included in the present MEP analysis.

Bacterial contamination causes closures of shellfish harvest areas periodically within the Bay System. Overall, the Waquoit Bay System is relatively free of bacterial levels requiring management activities, with levels of indicator bacteria exceeding management thresholds only periodically in small areas, generally associated with the smaller tributary systems (Quashnet River, Hamblin Pond, Little River). However, the mouth of the Moonakis (Quashnet) River is

Figure I-1. Major components of the Waquoit Bay Estuarine System. The study region for the present Massachusetts Estuaries Project analysis is the 3 major sub-embayments within the eastern portion of the Waquoit Bay System (Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River). Tidal waters from Nantucket Sound enter the main Bay through a single inlet in the barrier beach and a smaller inlet to the Eel Pond sub-embayment. Freshwaters enter the estuary primarily through two major surface water discharges (Childs River to Eel Pond and Quashnet River), several smaller streams (e.g. Red Brook), and direct groundwater discharge.

frequently closed to the harvest of shellfish due to bacterial contamination and the Department of Marine Fisheries area associated with Hamblin Pond (DSGA SC16.2) is classified "Prohibited". This area is located at the northern end of Hamblin Pond near the entry of Red

Brook waters. In addition, a small area associated with the Little River Boatyard is classified "Conditionally Approved", closed between May 1 and October 31, as a management precaution related to marina activities. But progress has been made in recent years to reduce bacterial contamination of Bay waters. In 1994, Waquoit Bay was designated a Federal No-Discharge Zone, which mandates that boaters must not discharge wastewaters anywhere within the Bay System. Smaller projects to reduce direct stormwater inflows have also been undertaken, for example improvements associated with the recently redesigned Meadow Neck Bridge over the mid-lower Quashnet River estuary.

In contrast to bacterial contamination, loading of the critical eutrophying nutrient, nitrogen, to the Bay waters has been greatly increased over the past few decades with further increases certain unless nitrogen management is implemented. The increasing rates of nitrogen loading to the Waquoit Bay Estuarine System, like almost all embayments in southeastern Massachusetts, have resulted from activities associated with a shift in watershed land-use from primarily pine/oak forest to residential development. The largest single nitrogen source associated with this shift is on-site septic disposal of domestic wastewater. The Towns of Mashpee and Falmouth have been among the fastest growing towns in the Commonwealth over the past two decades and do not have broad sewer service supported by centralized wastewater treatment; although two small facilities (Mashpee High School and Southport) operate within the watershed of one of the tributary embayments to eastern Waquoit Bay, the Quashnet River. Within the eastern Waquoit Bay sub-embayment watersheds, wastewater is returned to the aquifer almost entirely through individual on-site septic systems. As existing and probable increasing levels of nutrients impact Falmouth's and Mashpee's coastal embayments, water quality degradation will accelerate, with further declining health of their environmental resources.

The primary stakeholders for the three eastern tributary sub-embayments to the Waquoit Bay System are the Towns of Mashpee and Falmouth. These Towns have cooperative agreements relating to the resources of Waquoit Bay, for example shellfish resources are shared (cf. Town of Mashpee Shellfish Regulations 2004). Both communities are concerned about documented declines in System health. Initial concerns over habitat quality were followed by significant successful efforts of open space protection, most notably South Cape Beach, Washburn Island, and large portions of the Quashnet River watershed. These efforts both preserved habitat areas and reduced the amount of nitrogen likely to be added to Bay waters at watershed full development (build-out). However, these acquisitions do little to restore the nitrogen impaired waters of the Waquoit Bay System. Other notable management actions include designation as an Area of Critical Environmental Concern (ACEC) in 1979 and in 1988, admission into NOAA's National Estuarine Research Reserve Program (WBNERR, 1996). At present, the Waquoit Bay National Estuarine Research Reserve is jointly managed by NOAA and the Massachusetts Department of Conservation Resources (DCR), formerly the Massachusetts Department of Environmental Management (DEM). It should be noted, however, that implementation of nitrogen management strategies for restoration of this system is still primarily a municipal issue, which will require the efforts of citizens and managers primarily within the Towns of Mashpee and Falmouth.

Concern over declining habitat quality within the Waquoit Bay System continues to this day. Periodic macroalgal blooms have caused significant public attention, most recently in the summer of 2003, when massive *Cladophora* accumulations were observed over a large expanse of the nearshore of the main Bay (drift algae). While this "event" was dramatic, it only underscored the extent of nutrient overloading, as macroalgal accumulations have been a serious concern for more that three decades in this system (Curley et al., 1971).

Over the past two to three decades, both primary stakeholder communities (Falmouth and Mashpee) have examined potential management options for the Bay. At present both are undertaking Comprehensive Wastewater Facilities Planning, with an eye towards restoration of receiving marine waters. The Town of Mashpee is currently conducting planning for the watersheds of the eastern three sub-embayments and for the adjacent Popponesset Bay System. As part of this effort, the Town of Mashpee supported MEP data collection efforts and also supported the collection of the only nitrogen related water quality data available for these sub-embayments (and for the main Bay). The Mashpee Nutrient Monitoring Program will continue through summer 2004, since it is the only source of nitrogen baseline data for the whole of the Waquoit Bay System. Since it was becoming clear that nitrogen restoration of the Bay would likely require some traditional wastewater treatment approaches, the on-going ecological assessment and modeling project was combined with the Town of Mashpee's Wastewater Facilities Planning effort by the Mashpee Sewer Commission starting in 2000. Under the direction of the Mashpee Sewer Commission, the three eastern sub-embayments to Waquoit Bay were included in the first round prioritization of the Massachusetts Estuaries Project to provide state-of-the-art analysis and modeling. These data collection efforts by the Town of Mashpee were essential to the application of the MEP Linked Embayment-Watershed Approach to this estuarine system.

The present MEP effort builds upon the water quality monitoring program and previous hydrodynamic and water quality analyses, and includes high order biogeochemical analyses and water quality modeling necessary to develop critical nitrogen targets for each major sub-embayment. These critical nitrogen targets and the link to specific ecological criteria form the basis for the nitrogen threshold limits necessary to complete wastewater master planning as well as nitrogen management alternatives development needed by the Towns of Mashpee and Falmouth. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large numbers of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns of Mashpee and Falmouth to develop and evaluate the most cost effective nitrogen management alternatives to restore these valuable coastal resources that 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 nutrient sources 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 their assimilative capacity, the level where nutrients begin to cause declines in ecological health. The result is the loss of fisheries habitat, eelgrass beds, and a general disruption of benthic communities. At higher levels, enhanced nutrient loading from surrounding watersheds causes aesthetic degradation and inhibits even recreational uses of coastal waters. In addition to nutrient related ecological declines, an increasing number of embayments are being closed to swimming, shellfishing and other activities as a result of bacterial contamination. While bacterial contamination does not generally degrade the habitat, it restricts human uses. However, like nutrients, bacterial contamination is related to changes in land-use as a 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 Towns of Falmouth and Mashpee) are grappling with Comprehensive Wastewater Facilities 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 nitrogen management approach must directly link watershed and embayment nitrogen conditions. In addition any approach must be quantitative and directly support the evaluation of alternative management approaches, both from ecological and cost/benefit perspectives. 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 others including the Cape Cod Commission (CCC) have undertaken the task of providing a quantitative tool for watershed-embayment management for communities throughout Southeastern Massachusetts.

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

In appropriate estuaries, TMDLs for bacterial contamination are also being 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 assessment, synthesis, and modeling approach will be used to evaluate 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 model “alive” to address future regulatory needs.

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

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

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

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

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 eastern Waquoit Bay sub-embayments (Quashnet River, Hamblin Pond/Little River, and Jehu Pond/Great River) are located within the Towns of Falmouth (north & west) and Mashpee (east), on Cape Cod Massachusetts. The southern shore is a barrier beach that separates the Waquoit Bay System from adjacent Nantucket Sound (Figure I-1). At present, each of the three sub-estuaries exchanges tidal waters with the main basin of Waquoit Bay, which receives tidal flows from Nantucket Sound. The main Bay has two main openings to Nantucket Sound, a historically open inlet in the main Bay and an ephemeral inlet that connects Eel Pond to Nantucket Sound. More recently, Hurricane Bob in 1991 created a third inlet immediately east of the Eel Pond entrance; however, this inlet has closed over the past few years.. The inlet to the main Bay has been fixed with jetties initially in 1918 (east) and 1937 (west), with subsequent lengthening and enhancements. This second inlet has been generally open over the past 50 years. The opening of the second inlet significantly increased the tidal range and flows within the Waquoit Bay System and caused important ecological shifts to its tidal wetlands and possibly other estuarine habitats (Orson and Howes, 1992). In recent years, Hurricane Bob (1991) opened a third inlet close to the second inlet to Eel Pond, helping to maintain the recent Waquoit Bay tidal range and circulation pattern. This important “natural” hydrodynamic shift coupled to anthropogenic alteration of the watershed support a recently highly altered estuarine habitat. Within the Quashnet River, Hamblin Pond, and Jehu Pond sub-embayments geomorphic and hydrologic alterations include the damming of the Quashnet (Moonakis) River to drive mills and alteration of riparian zone for cranberry agriculture, and creation of roadways altering circulation around Monomascoy Island. However, the over-riding change affecting these sub-systems appears to have been the shift from pine/oak forest to farming to current residential land-uses, with its associated large increases in watershed nitrogen loading to the estuarine system.

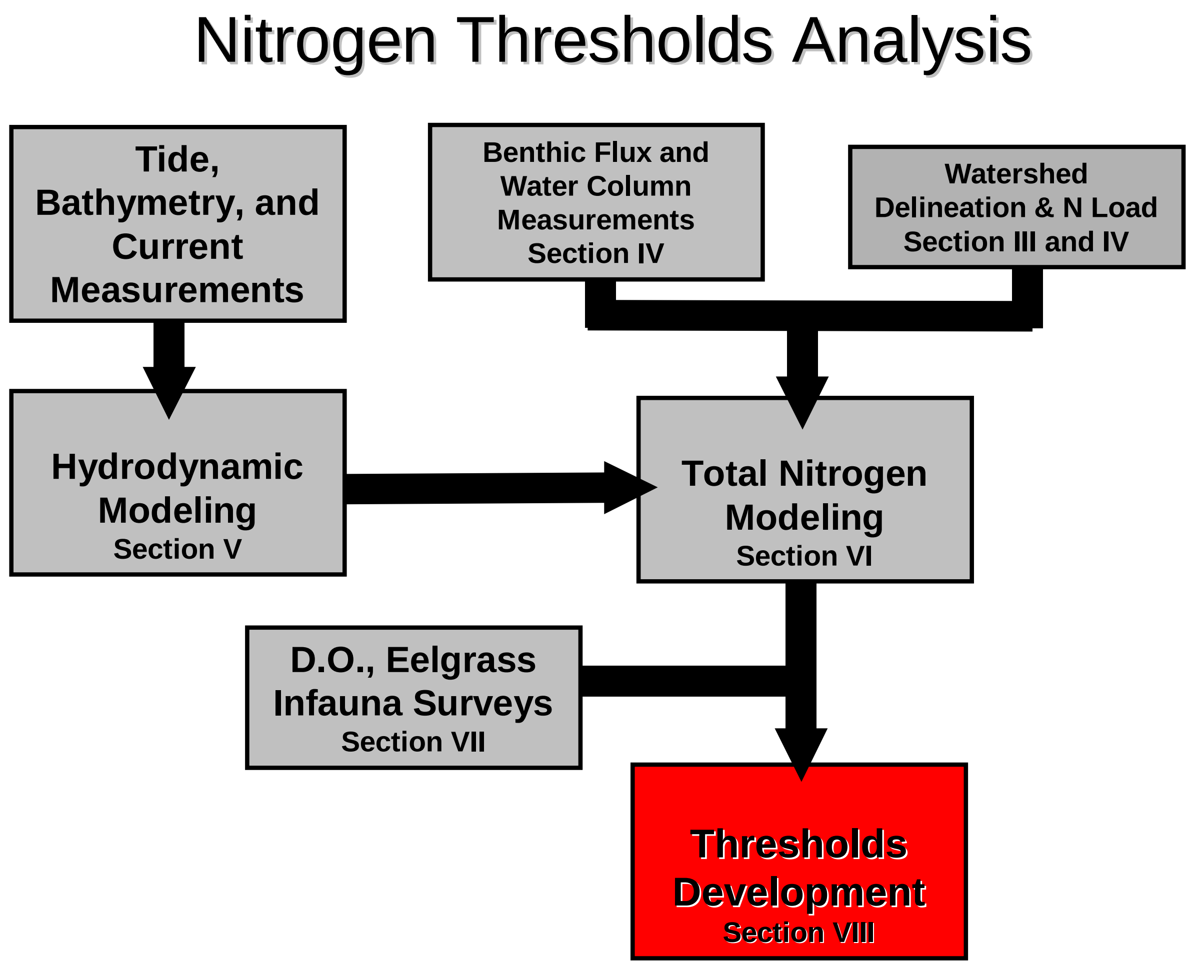


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

The Bay's watershed is distributed among the Towns of Falmouth and Mashpee, with a small portion of the upper-most region of the watershed located in Sandwich. The eastern sub-embayments are located in the Mashpee Pitted Outwash Plain that supports numerous kettle ponds (Oldale 1992). The Quashnet River Estuary is a drowned river valley estuary resulting from rising sea-level flooding the lower reaches of the Quashnet River. Hamblin and Jehu Pond appear to be drowned kettles currently exchanging tidal flows with Waquoit Bay through tidal rivers, Little River and Great River, respectively. Both the Hamblin Pond and Jehu Pond sub-systems support significant saltwater wetland resources.

The tidal reach of the Quashnet River Estuary is located within the Town of Falmouth while much of the freshwater region of the Quashnet River and its watershed is found in the Town of Mashpee. The river is one of the two major surface water inflows to the Waquoit Bay System and originates in John's Pond. Hamblin Pond is divided between the Towns of Falmouth and Mashpee, while Jehu Pond is entirely situated within the Town of Mashpee. The Waquoit Bay system is composed of a main bay with multiple associated sub-embayments (Quashnet River, Hamblin Pond, Jehu Pond, Eel Pond, Childs River). These sub-embayments constitute important components of the region's natural and cultural resources. In addition, the large number of sub-embayments greatly increases the System's shoreline 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 Waquoit Bay system and its sub-embayments along the Falmouth and Mashpee shores are at risk of eutrophication from high nitrogen loads in the groundwater and runoff from their watersheds. As will be presented in this report, numerous lines of evidence indicate that much of the Waquoit Bay System and the three sub-systems that this MEP Report focuses upon (Quashnet River, Hamblin Pond, Jehu Pond) are currently beyond their nitrogen loading threshold and are currently showing various levels of nitrogen related habitat decline.

Within the eastern Waquoit Bay System, the tidal portions of the major sub-estuaries (Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River, and Sage Lot Pond) show clear estuarine characteristics, with extensive salt marsh area, tidal flats and large salinity fluctuations. In contrast, the open water portion of eastern Waquoit Bay shows more typical characteristics of open water areas, having only fringing salt marshes, relatively stable salinity gradients and a large basin volume relative to tidal prism. The tidal forcing for these sub-systems is generated from Nantucket Sound. Nantucket Sound adjacent the inlets in South Cape Beach and the southern shore of Washburn Island, exhibits a moderate to low tide range, with a mean range of about 2.5 ft. Since the water elevation difference between Nantucket Sound and Waquoit Bay is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed into and out of the Bay System during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, Wellfleet Harbor is ~10 ft).

Tidal damping (reduction in tidal amplitude) through an embayment can range from negligible, indicating "well-flushed" conditions, or show tidal attenuation caused by constricted channels and marsh plains, indicating a "restrictive" system where tidal flow and the associated flushing are inhibited. Tidal data indicate only minimal tidal damping through Waquoit Bay inlet. It appears that the tidal inlet is operating efficiently, possibly due to the active inlet maintenance program. Similarly, within the eastern Waquoit Bay System, the tide generally propagates through the three focal sub-embayments with little attenuation, consistent with relatively unrestricted tidal exchanges.

Given the present hydrodynamic characteristics (well flushed) of the Waquoit Bay System, it appears that estuarine habitat quality is more dependent on nutrient loading to bay waters than tidal characteristics within the component sub-embayments. Due to the relatively well flushed conditions observed in the three sub-embayment systems that are the focus of this investigation, habitat degradation is therefore mostly a result of the exceedingly high nutrient loads currently being documented in these systems, not tidal damping.

Nitrogen loading to the eastern Waquoit Bay System was determined relative to the major eastern shore sub-embayments: Quashnet River, Hamblin Pond/Little River, and Jehu Pond/Great River (Sage Lot Pond is also included). The watershed for this estuarine system contains approximately 10,250 acres, the predominant land use based on area being public service/government, including the Massachusetts Military Reservation and protected open space along the Quashnet River. Public service occupies 54% of the total watershed area to eastern Waquoit Bay (see Figure IV-2). In contrast, while single-family residences occupy approximately 15% of the total watershed area to eastern Waquoit Bay, this land use class represents 61% of all the parcels. Commercial properties are fairly limited within the watershed, with two small clusters located on Route 28 and Route 151.

Relative to the Waquoit Bay System's eastern sub-embayments, residential land-uses primarily in the southern portion of Falmouth and in the Mashpee region create the major nutrient load. Approximately one half of the nitrogen load from single-family dwellings enters the Quashnet River sub-embayment, with almost all of the remainder entering the tidal reaches of Hamblin Pond and Jehu Pond Estuaries. The Sage Lot Pond watershed contains almost no residential development and is primarily a salt marsh with a central shallow pond.

As management alternatives are being developed and evaluated, it is important to note that eastern Waquoit Bay is presently a relatively dynamic and significantly man-altered estuarine system.

I.3 NITROGEN LOADING

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

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

Each embayment system maintains a capacity to assimilate watershed nitrogen inputs without degradation. However, as loading increases a point is reached at which the capacity (termed assimilative capacity) is exceeded, nutrient related water quality degradation occurs. Continuing increases in nitrogen inputs beyond this threshold level result in further declines in habitat quality. 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 is done in the MEP effort). 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 total nitrogen concentration throughout each of the 3 eastern sub-embayment to the Waquoit Bay System monitored by the Mashpee Water Quality Monitoring Program with site-specific habitat quality data (D.O., eelgrass, phytoplankton blooms, benthic animals). The integration of site specific nitrogen data with site specific habitat quality data allows the MEP 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 eastern Waquoit Bay sub-embayment systems (including Waquoit Bay) are near or beyond their ability to assimilate additional nutrients without impacting their ecological health. Nitrogen levels are elevated throughout the System and a marked reduction in eelgrass coverage has been observed in the Hamblin Pond and Jehu Pond sub-estuaries (Short and Burdick, 1996). Eelgrass has not been observed in the Quashnet River sub-embayment, instead high levels of macroalgae have been documented (Curley et al., 1971, Valiela et al., 1992). The result is that nitrogen management for each of the three sub-embayments to the Waquoit Bay System covered in this MEP Report must focus on 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, it is specified as “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 Waquoit Bay’s sub-embayments could potentially occur without man’s influence and must be considered in the nutrient threshold analysis. While this finding would not change the need for restoration, it would change the approach and potential targets for management. As part of future restoration efforts, it is important to understand that it may not be possible to turn each embayment into a “pristine” system.

I.4 WATER QUALITY MODELING

Evaluation of upland nitrogen loading provides important “boundary conditions” for water quality modeling of the eastern Waquoit Bay sub-embayment 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 eastern Waquoit Bay System, focusing on the tributary sub-embayments of Quashnet River, Hamblin Pond/Little Pond, Jehu Pond/Great River, and Sage Lot Pond. A two-dimensional depth-averaged hydrodynamic model based upon the tidal currents and water elevations was employed for each of the systems. Once the hydrodynamic properties of each estuarine system were computed, two-dimensional water quality model simulations were used to predict the dispersion of the nitrogen at current loading rates.

Using standard dispersion relationships for estuarine systems of this type, the water quality model and the hydrodynamic models were then integrated in order to generate estimates regarding the spread of total nitrogen from the site-specific hydrodynamic properties. The distributions of nitrogen loads from watershed sources were determined from land-use analysis, based upon watershed delineations by USGS using a modification of the West Cape model for sub-watershed areas designated by MEP. Almost all nitrogen entering east Waquoit Bay is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Waquoit Bay System were taken from the Mashpee Water Quality Monitoring Program (supported by the Town of Mashpee in association with the Coastal Systems Program at SMAST). Measurements of nitrogen and salinity distributions throughout estuarine waters of the System were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the east Waquoit Bay sub-embayment system for the Towns of Falmouth and Mashpee. 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 Nantucket Sound (Section IV). Intrinsic to the calibration and validation of the linked-watershed embayment modeling approach is the collection of background water quality monitoring data (conducted by municipalities) as discussed in Section IV. Results of hydrodynamic modeling of embayment circulation are discussed in Section V and nitrogen (water quality) modeling, as well as an analysis of how the measured nitrogen levels correlate to observed estuarine water quality are described in Section VI. This analysis includes modeling of current conditions, conditions at watershed build-out, and with removal of anthropogenic nitrogen sources. In addition, an ecological assessment of the component sub-embayments was performed that included a review of existing water quality information and the results of a benthic analysis (Section VII). The modeling and assessment information is synthesized and nitrogen threshold levels developed (Section VIII) for restoration of the sub-embayments to east Waquoit Bay. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined sub-embayments threshold for restoration. This latter assessment represents only one of many solutions and is produced to assist the Town(s) in developing a variety of alternative nitrogen management options for this system.

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

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

In most marine and estuarine systems, such as Waquoit Bay and its associated tributary sub-embayments (Quashnet River, Hamblin Pond, and Jehu Pond) that are the focus of this nutrient threshold analysis, the limiting nutrient, and thus the nutrient of primary concern, is nitrogen. In large part, if nitrogen addition is controlled, then eutrophication is controlled. This approach has been formalized through the development of tools for predicting nitrogen loads from watersheds and the concentrations of water column nitrogen that may result. Additional development of the approach generated specific guidelines as to what is to be considered acceptable water column nitrogen concentrations to achieve desired water quality goals (e.g., see Cape Cod Commission, 1991, 1998; Howes et al., 2002).

Many of the previously developed tools for predicting loads and concentrations tend to be generic in nature, and overlook some of the specific characteristics of a given water body. The present Massachusetts Estuaries Project (MEP) study focuses on linking water quality model predictions, based upon watershed nitrogen loading and embayment recycling and system hydrodynamics, to actual measured values for specific nutrient species. The linked watershed-embayment model is built using embayment specific measurements, thereby enabling calibration of the prediction process for specific conditions in each of the coastal embayments of southeastern Massachusetts, including the eastern sub-embayments to the Waquoit Bay (Quashnet River, Hamblin Pond, and Jehu Pond Estuaries).

The eastern tributary sub-embayments to Waquoit Bay are part of the Waquoit Bay National Estuarine Research Reserve (WBNERR). The National Estuarine Research Reserve System (NERR) was established to select “representative” estuarine systems associated with the coastal waters of the United States to support research and long-term monitoring of estuarine change. WBNERR joined the National System in 1988. Over the intervening 15 years, research has been conducted on organisms, land-use, and effects of nitrogen on embayment habitats. In addition, a land-use nitrogen model was developed to assess nitrogen loading. The various scientific publications and technical reports that have been produced were reviewed as part of the MEP assessment to garner quantitative data and qualitative information of use to the present data collection, synthesis, and modeling effort. A brief review of previous studies as relates to their utilization by the MEP approach is given below.

Data collected by Curley et al. (1971) indicated that as far back as the late 1960's, there was early evidence of nutrient related habitat decline within the eastern region of the Waquoit Bay System. This was confirmed and expanded upon two decades later in the first major scientific publication on the Waquoit Bay System (Valiela et al., 1990). This latter study documented eelgrass decline occurring within the Bay and its tributary systems, shifts in benthic species, and the linkage to increasing nitrogen loading from the associated watersheds. Further investigations have supported the detrimental effects on eelgrass (Valiela et al., 1992, Short and Burdick, 1996), enhancement of macroalgal accumulations (Hauxwell et al., 1998, Thompson and Valiela, 1999), system respiration (D'Avanzo et al., 1996), and potential moderate shifts in fish abundance and growth (Tober et al., 2000).

Coupled to these investigations of biological response to nitrogen loading, has been an attempt to determine watershed nitrogen loading rates. This approach has been termed the Waquoit Bay Nitrogen Loading Model (Valiela et al., 1992, Valiela et al., 2000). This approach is aimed at producing a research model which tracks nitrogen from all sources and uptake within the watershed, and attempts to predict the nitrogen discharges to the estuary. The approach is similar to other land-use loading models including the MEP watershed module. From the available information, it has been difficult to determine the various factors employed in the Waquoit Bay Nitrogen Loading Model and particularly difficult to rectify differences in watershed areas, nitrogen loads, and freshwater discharge volumes from the various reports and papers. In addition, validation of the model was based upon groundwater well point measurements which did not sample the full cross-section of the groundwater discharge boundary. Since no fractionation of the groundwater nitrogen pool or any salinity data is presented, it is not possible to evaluate whether the sampling at the "high tide mark at the seepage face" is representative of the groundwater flow. Limitations in this approach to measurement of groundwater nitrogen discharges are underscored by the very large discrepancy in the Sage Lot Pond sub-system which receives little anthropogenic loading (modeled versus measured from Valiela et al., 2000, Table 2, 147 versus 846 kg N yr^{-1}, respectively). In addition, the "measured" loads to Hamblin Pond, Jehu Pond, and Quashnet River using the watershed areas presented in Valiela et al., 2000 yield agreements to modeled loading of 54%, 73% and 118% respectively (see Table 2 in Valiela et al., 2000). Based on a general review of the Waquoit Bay Nitrogen Loading Model results published to date, there appeared to be significant bias in the model at higher nitrogen mass loadings.

In addition to the concerns regarding the groundwater measurement approach, the differences presented above need to be evaluated relative to changes in watershed area found by the MEP/USGS watershed delineation effort that was based on an updated groundwater model and improved parameterization as described in Chapter IV. It should be noted that the modeled Quashnet River Watershed nitrogen load is based upon freshwater discharges. In the earlier work, Quashnet River watershed total freshwater discharge was calculated from watershed area and recharge and compared to measured discharges (Valiela et al., 1992). The two estimates differed by only ~13%. Examination of the USGS discharge data during the likely period of this study (1989-1992) showed annual total river discharges of 1.17 to 1.32 10^7 m^3 yr^{-1}, compared to 1.1 10^7 m^3 yr^{-1} in the study, indicating excellent agreement. This river discharge was estimated to account for >80% of the total freshwater discharge from the Quashnet watershed. However, the 2657 ha watershed area upon which the freshwater flow values were based is ~30% larger than the watershed upon which the nitrogen loading comparison is based, 2055 ha (Valiela, et al., 1992). Interestingly, the USEPA in evaluating the Waquoit Bay nitrogen loading relied upon the "measured" nitrogen inputs from the well point samplers and the estimated groundwater discharges (see Figure 2-1 in USEPA, 2002). It is likely that these estimates will change significantly given the shift in watershed delineations (hence watershed area) and recent improvements in the USGS's groundwater recharge estimates.

A recent approach to evaluate nitrogen levels in Waquoit Bay and subsequent impacts on the Bay in response to watershed nitrogen loading has also been proposed (USEPA, 2002). This approach is not suited for the evaluation of nitrogen management alternatives at this time, as the approach is not robust, is calibrated to inorganic nitrogen concentrations (which generally represent a small fraction of the total nitrogen pool), and does not account for circulation or dispersion of nitrogen within the receiving waters.

Based upon the above concerns and shortcomings related to previous nitrogen loading estimates, and especially the new USGS watershed delineations, the MEP Technical Team was not able to directly assimilate these previous watershed nitrogen loading estimates. Comparison to previous nitrogen loading studies has focused primarily on the watershed delineation aspects.

As part of its mission of long-term monitoring, WBNERR has conducted both a volunteer monitoring program (BayWatcher) and formal monitoring program (System Wide Monitoring Program or SWMP). The WBNERR BayWatcher Program conducts a variety of water quality assays (Secchi Depth, salinity, temperature, dissolved oxygen, and chlorophyll a). Nutrients are also assayed, but only the inorganic forms (ammonium, nitrate, nitrite, ortho-phosphate, silicate). The more formal program (SWMP) is part of the NERR System and employs moored instrumentation to measure dissolved oxygen, salinity, temperature, pH, depth, and turbidity at four sites (upper Waquoit Bay, Childs River, lower Eel Pond, Sage Lot Pond). Organic nitrogen (particulate or dissolved) is not assayed in either monitoring program. Both programs are conducted under the supervision of the WBNERR Staff and the SWMP program is fully vetted through the NERR System. Therefore, the dissolved oxygen and chlorophyll a data collected by both WBNERR Programs has been included in this MEP analysis.

A major component of the MEP nutrient analysis is the evaluation of hydrodynamics within the estuarine system. Although previous hydrodynamic modeling efforts have been performed (e.g. Aubrey et al., 1993 and Valiela et al., 1998), information regarding these analyses are limited. A one-dimensional hydrodynamic model of Waquoit Bay was developed by Aubrey et al. (1993) to study the hydrodynamic effects of both the two and three inlet morphology. Bathymetry data were collected in the main basin of Waquoit Bay, Seapit River, Childs River, and the lower portion of the Quashnet River. Unfortunately, the digital data was not available and the datum described on the depth contour map could not be verified; therefore, the bathymetric information could not be utilized for the present study. If tide gage measurements were made to parameterize the model, results were not included in Aubrey et al., 1993. In Valiela et al. (1998), results of a circulation model are presented; however, there is no indication whether any physical measurements were performed to parameterize, calibrate, or validate the modeling effort. Again, this effort focused on changes to estuarine flushing with regard to formation of the third inlet by Hurricane Bob in 1991. Similar to Aubrey, et al. (1993), Valiela et al. (1998) conclude that the influence of the third inlet on tidal flushing is relatively minor. The MEP analysis presented in this report provides a comprehensive analysis of circulation for the entire Waquoit Bay System and an analysis of water quality within the tributary sub-embayments to the Waquoit Bay estuary (Quashnet River, Hamblin Pond and Jehu Pond); therefore, results from the earlier generation 1994 analysis have been superseded.

For the MEP modeling analysis, the data from the previous studies were evaluated relative to the needs of this project. Bathymetric data associated with Aubrey, et al. (1993) was cursory and was not collected relative to a known tidal datum (e.g. NGVD29). The Town of Mashpee through their designee contacted the Boston University Marine Program (BUMP) and the Waquoit Bay National Estuarine Research Reserve (WBNERR). Unfortunately, no data associated with physical processes (e.g. tide, current, or bathymetry information) was available for MEP use. For these reasons, it was necessary to collect both bathymetry and tide data to support the MEP analysis.

The MEP Technical Team conducted an extensive review of the nitrogen related studies of the Waquoit Bay System, including published articles, technical reports and discussions with WBNERR Staff (September 25, 2003 meeting). As a partner in the MEP, the Town of Mashpee through its wastewater engineering consultant also gathered information including discussions with Boston University Marine Program researchers (Dr. I. Valiela et al.). These data mining efforts determined that total nitrogen measurements were not available for the waters of Waquoit Bay System. Previous measurements of nitrogen in estuarine and surface freshwaters included only assays of inorganic nitrogen species (ammonium, nitrate, nitrite) with ground water assays sometimes including dissolved organic nitrogen. Total nitrogen is required for validation of the MEP Linked Watershed-Embayment Model and other high order estuarine nitrogen models, as nitrogen is rapidly transformed from one species to another. In estuarine systems, like the eastern sub-embayments to Waquoit Bay, inorganic nitrogen entering from the watershed is rapidly transformed to organic forms. The result is that it is not possible to balance the nitrogen budget for these systems without a full accounting of the nitrogen pool, especially since the inorganic forms account for only a minor fraction of the nitrogen pool in these estuarine waters (generally <5%).

As a result of the absence of water column total nitrogen data, the Town of Mashpee with the Coastal Systems Programs at SMAST-UMD conducted surveys of nitrogen levels in the eastern sub-embayments to Waquoit Bay and associated waters. The specific goal of the water quality surveys was to capture the nitrogen gradients within these estuaries to support the MEP Linked Watershed-Embayment Modeling effort. Sampling by the Mashpee Water Quality Monitoring Program was conducted as a joint effort between private citizens, the Mashpee Shellfish Department, Mashpee Harbor Master, Mashpee Waterways Commission, Mashpee Watershed Nutrient Management Committee, and SMAST. Water quality monitoring was conducted during the summer when eutrophication impacts are generally the greatest in Cape Cod embayments. The major findings were that nitrogen levels within both the sub-embayments and the main basin of Waquoit Bay were significantly elevated over adjacent Nantucket Sound waters. In addition, the sub-systems showed gradients in both nitrogen and salinity typical of estuaries.

After extensive review and evaluation of previous studies conducted in Waquoit Bay, the MEP Technical Team has attempted to incorporate all appropriate data from all historical studies. The objective of the in depth review of previous studies was to enhance the determination of nitrogen thresholds for the eastern sub-embayments to the Waquoit Bay System and to reduce costs to the Towns of Mashpee and Falmouth.

III. DELINEATION OF WATERSHEDS

III.1 BACKGROUND

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

In the present investigation, the USGS was responsible for the application of its groundwater modeling approach to define the watershed or contributing area to the eastern Waquoit Bay System under evaluation by the Project Team. The eastern Waquoit Bay estuarine system is composed of: the Quashnet River and its tidal waters, Hamblin Pond and the tidal waters of Little River connecting Hamblin Pond to Waquoit Bay, Jehu Pond including the tidal waters of Great River connecting Jehu Pond to Waquoit Bay and to the tidal Sage Lot Pond/Flax Pond salt marshes. Further watershed modeling was undertaken to sub-divide the overall watershed to the eastern portion of the Waquoit Bay System into functional sub-units based upon: (a) defining inputs from contributing areas to each major sub-embayment within the embayment system (for example Hamblin Pond tributary to the Waquoit Bay System), (b) defining contributing areas to major freshwater aquatic systems which generally attenuate nitrogen passing through them on the way to the estuary (lakes, streams, wetlands), and (c) defining 10 year time-of-travel distributions within each sub-watershed as a procedural check to gauge the potential mass of nitrogen from “new” development, which has not yet reached the receiving estuarine waters. The three-dimensional numerical model employed is also being used to define the contributing areas to public water supply wells in the Sagamore flow cell on Cape Cod. Model assumptions for calibration were matched to surface water inputs and flows from current (2002 to 2003) stream gage information.

The relatively transmissive sand and gravel deposits that comprise most of Cape Cod create a hydrologic environment where watershed boundaries are usually better defined by elevation of the groundwater and its direction of flow, rather than by the land surface topography (Cambareri and Eichner 1998, Millham and Howes 1994a,b). This is particularly true for embayments formed within outwash plains, such as those of eastern Waquoit Bay located within the Mashpee Pitted Outwash Plain. Freshwater discharge to estuaries is usually composed of both 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 discharging 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 (Howes, et al., 1996; DeSimone, et al., 1996). 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 within surficial sediments. Both removal mechanisms can occur simultaneously within any ecosystem, generally associated with the surficial 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 hydrologic features directly connected to the groundwater system, which receive groundwater inflow through up-gradient 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 the eastern portion of the Waquoit Bay System 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 four sub-embayments of the eastern Waquoit Bay System and also to determine portions of recharged water that may flow through freshwater ponds and streams prior to discharging into coastal water bodies.

The Sagamore Flow Model grid consists of 246 rows, 365 columns and 20 layers. The horizontal model discretization, or grid spacing, is 400 by 400 feet. The top 17 layers of the model extend to a depth of 100 feet below 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 eastern Waquoit Bay system resides, water elevations are generally less than +40 ft and, therefore, over much of the study area the uppermost layers are inactive. 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 watershed to the Waquoit Bay System. 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. In general, within the watershed to Waquoit Bay, the upper layers are composed of sand and gravel with a shift to fine sand, silt and clay at 0-50 ft below current sea level. In the upper watershed, this deposit is underlain by basal till (clay, silt, sand and gravel). In the lower watershed and beneath the Bay, a layer of sand and gravel is interspersed. Bedrock is found at about -130 feet within the upper watershed grading to about –380 feet beneath the Bay, relative to current sea level (Cambareri et al. 1993). Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water level and streamflow data collected in May 2002. For the Quashnet River, the USGS also used long-term flow data from its gauging station located above the tidal reach of the estuary.

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 most of Mashpee is unsewered, 85% of the water pumped from wells was modeled as being returned to the ground via on-site septic systems or outdoor use.

III.3 MASHPEE CONTRIBUTORY AREAS

Revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for the eastern Waquoit Bay sub-embayment system (Quashnet River, Hamblin Pond, Jehu Pond, and Sage Lot Pond) (Figure III-1). Model outputs of MEP watershed boundaries were “smoothed” to (a) correct for the grid spacing, (b) to enhance the accuracy of the characterization of the shoreline, and (c) to more closely match the sub-embayment segmentation of the tidal hydrodynamic model. The smoothing refinement was a collaborative effort between the USGS and the rest of the MEP Technical Team. Overall, 36 sub-watershed areas were delineated within the watershed to eastern Waquoit Bay sub-embayment system. Table III-1 provides the daily discharge volumes for various watersheds as calculated by the groundwater model; these volumes were used to assist in the salinity calibration of the tidal hydrodynamic models and for comparison to measured surface water discharges. The MEP delineation includes subwatershed delineations to five ponds and public drinking water supply wells and 10 yr time of travel boundaries. Contributing areas for fresh ponds were delineated if the pond covered most of three-groundwater model grid cells (400 ft X 400 ft each) generally about 10 acres. The decision to use 3 model grid cells (1 cell is 400 x 400 feet) as a minimum size criterion 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 significant 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. Smaller ponds were considered by USGS to not significantly intercept groundwater flows and from a modeling point of view, including ponds less than 10 acres in size added several degrees of computational complexity thereby making the groundwater models unwieldy.

The delineations completed for the MEP project are the second delineation for this portion of the Waquoit Bay estuary. Figure III-2 compares the delineation completed under the current effort with the delineation completed by the Cape Cod Commission in 1991 (Cambareri and Eichner, 1998). The delineation completed in 1991 was defined based on water table measurements collected in December 1991; these water table readings were at a period of low

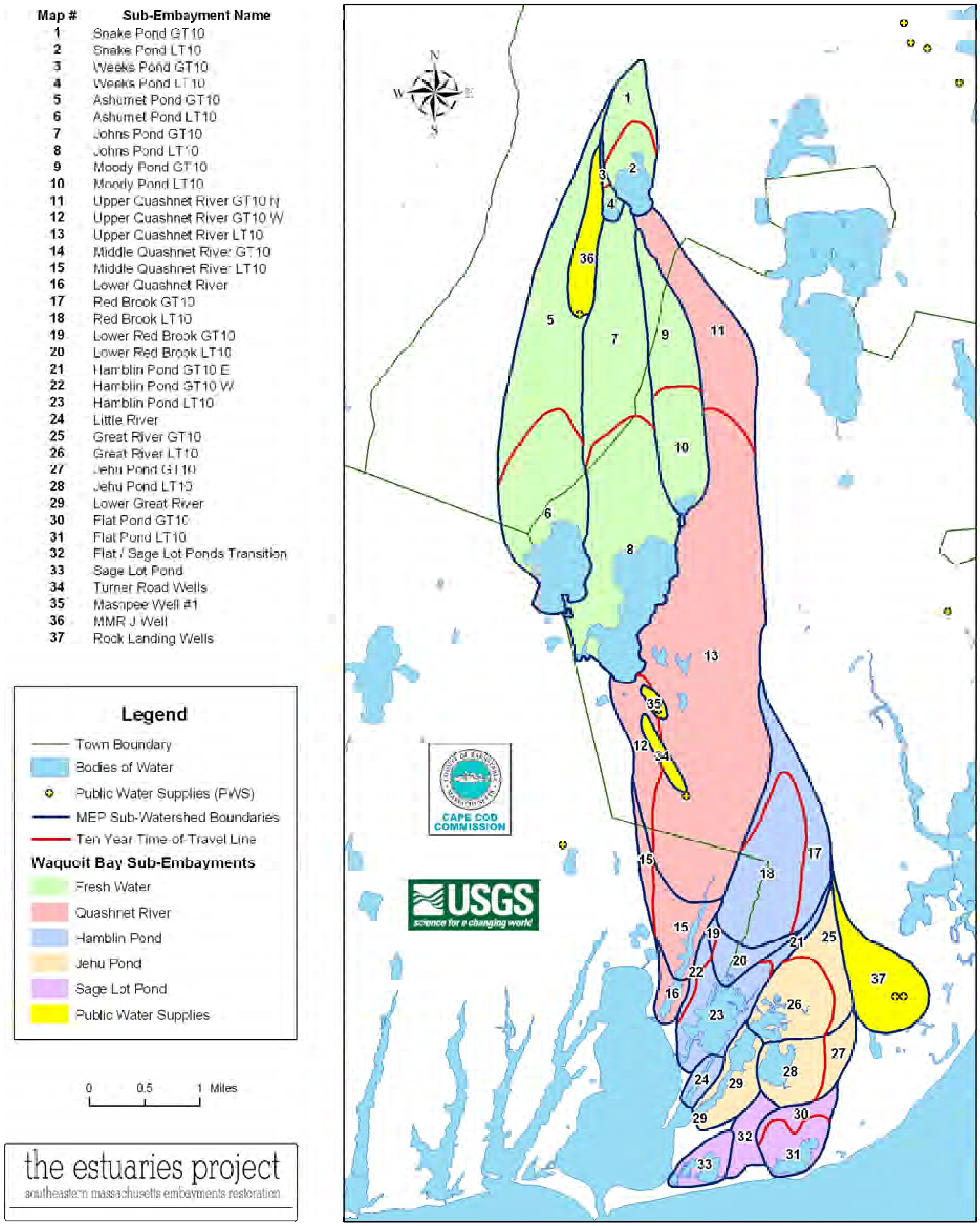


Figure III-1. Watershed and sub-watershed delineations for the Eastern Waquoit Bay estuary system. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the figure legend (left). Sub-watersheds to embayments were selected based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

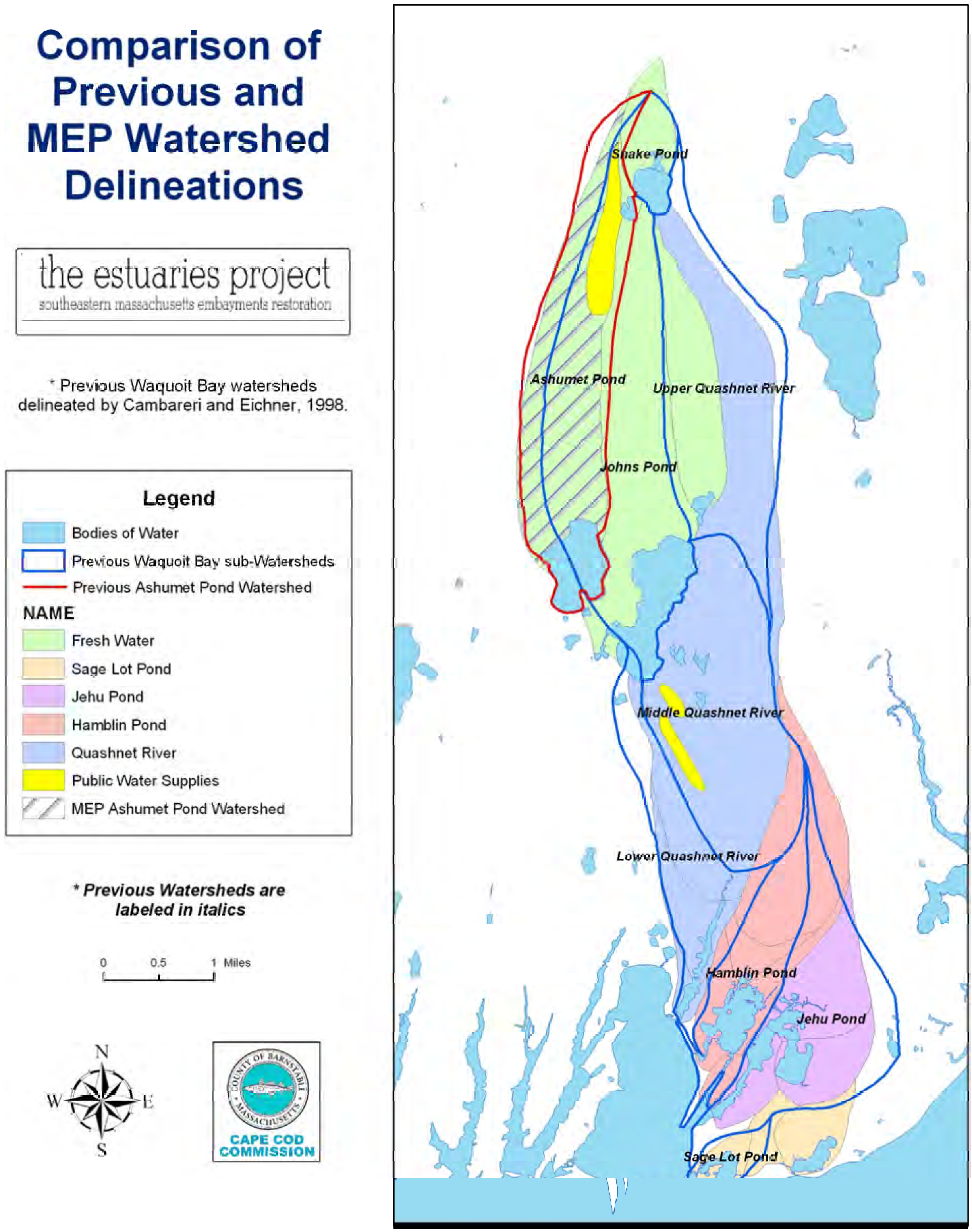


Figure III-2. Comparison of previous and current Eastern Waquoit Bay watershed and subwatershed delineations.

Table III-1. Daily groundwater discharge to each of the sub-embayments in the East Waquoit Bay system, as determined from the USGS groundwater model.

Watershed	Discharge		Watershed	Discharge	
	ft^3/day	m^3/day		Ft^3/day	m^3/day
Upper Quashnet River	1,649,859	46,715	Jehu Pond	65,473	1,854
Middle Quashnet River	65,937	1,867	Great River	110,631	3,133
Lower Quashnet River	14,910	422	Lower Great River	39,070	1,106
Red Brook	215,913	6,115	Sage Lot/Flat Pond	92,654	2,624
Lower Red Brook	31,425	890	Total System	2,365,217	66,984
Hamblin Pond	66,615	1,887			
Little River	13,082	370			

regional water table (Cambareri et al. 1993). The 2002 delineation using updated hydrogeologic information was completed by the USGS using a previous iteration of the Sagamore Lens groundwater model.

Table III-2 summarizes the percent difference in selected embayment watershed areas between 1991 watershed delineations of eastern Waquoit Bay system and the newly delineated watersheds. The MEP watershed delineation for the eastern Waquoit Bay area as a whole is 5% bigger (470 acres) than the 1991 CCC delineation; most of this change is attributable to the inclusion of a watershed to Flat Pond (302 acres). Flat Pond is connected by a surface water tidal channel to Sage Lot Pond, so it is functionally part of a greater Sage Lot/Flat Pond estuary. The changes in the delineation result from a slight movement of the regional groundwater divide toward the south and a slightly more eastern location for the divide between the Popponesset and Waquoit Bay systems. This latter change in the watershed boundary to the southwest near Nantucket Sound is significant as it relates both to nitrogen loading (area is significantly developed) and to potential groundwater sites which discharge directly to Nantucket Sound.

In contrast, internal subwatershed delineations showed significant differences relative to previous sub-watershed delineations; most changes are between ±20-47%. This is very important relative to nitrogen management, as the sub-systems are the functional targets for nitrogen management, within the context of the hydrodynamics of the greater Bay System. The shifts in the delineations for the Hamblin and Jehu Pond sub-estuaries results in a significant shift in the watershed nitrogen loads to these systems, compared to previous delineations. For example, 598 acres are now included in the Hamblin Pond subwatershed (44% areal increase) that had been associated with the Quashnet River and Jehu Pond subwatersheds. The Jehu Pond subwatershed lost 47% (453 acres); this loss was the combination of expansion of the Hamblin Pond subwatershed, changes in the dividing line between Waquoit Bay and Popponesset Bay, and inclusion of a subwatershed to the Rock Landing public water supply wells. Other watershed delineations based upon earlier models and significantly less hydrogeologic data than the USGS's present effort (Valiela et al. 2000, also Valiela et al. 1992), showed similarly large areal differences for Hamblin Pond (52%), Jehu Pond (9%) and Sage Lot/Flat Pond (30%). These delineations showed 14%-68% differences from the prior Cape Cod Commission delineations, as well. More detailed analysis of these previously developed sub-watershed delineations is not possible at this time as the underpinning data is not available to MEP Staff or the Town of Mashpee.

Though the overall watershed area for the eastern portion of the Waquoit Bay System has not changed significantly (5 %) from the original 1991 CCC watershed delineation, the changes in the sub-watershed delineations for each of the sub-embayments to eastern Waquoit Bay are noteworthy. The changes in sub-watershed delineations define the extent to which watershed based nutrient loads are transported to a specific embayment. The changes in the areal extent of a sub-watershed means that nutrient loads that were previously allocated to one sub-embayment are actually being introduced into another sub-watershed. This change in loading from one sub-embayment to another greatly affects the development of the nutrient thresholds for each sub-embayment as well as the load allocations developed under the nutrient TMDL and the management alternatives considered for any given sub-embayment.

The evolution of the watershed delineations for the eastern Waquoit Bay System have built one on another to increase the underlying hydrologic data supporting the modeling, thus increasing accuracy. This is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in proportional errors in nitrogen loading, errors in loading depend upon the land-uses that are included/excluded within the contributing areas. Small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the down-gradient estuary. In the case of the Waquoit Bay System, the present level of development and the areas of refinement indicate that the current and build-out nitrogen loading estimates were significantly improved through the use of the new delineation.

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

Eastern Waquoit Bay System	MEP2002	CCC 1991	% difference
WATERSHED	acres	acres	
Snake Pond	322	240	26%
Weeks Pond	40		
J Well	169		
Ashumet Pond	1,434	1,472	-3%
Johns Pond	1,647	1,376	16%
Quashnet River+Moody Pond	3,910	4,377	-12%
Hamblin Pond/Little River	1,348	751	44%
Jehu Pond/Great River	955	1,408	-47%
Sage Lot Pond	117	153	-31%
Flat Pond	302		
Entire System	10,246	9,776	5%

IV. WATERSHED NITROGEN LOADING TO EMBAYMENT: LAND USE, STREAM INPUTS, SEDIMENT NITROGEN FLUX AND 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 Waquoit Bay System. Determination of watershed nitrogen inputs to the eastern Waquoit Bay embayment system and its sub-embayments 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 wetlands. This latter natural attenuation process is conducted by biological systems 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 that 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 the determination of summertime nitrogen load.

The Massachusetts Estuaries Project (MEP) team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff the CCC staff conducted the land use analysis and integrated potable water-use for the development of nitrogen loading rates (Section IV.1) within each of the 36 subwatersheds to the eastern Waquoit Bay embayment system (Section III). The initial task in the MEP land use analysis is to gauge whether or not nitrogen discharges within 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. A ten year time of travel was selected for analysis. The time of travel represents the number of years required from the time water recharges the aquifer or enters a pond, until it reaches the down-gradient estuarine waters. After reviewing the percentage of nitrogen loading in the less than 10 year time of travel and greater than 10 year time of travel watershed regions (Table IV-1), reviewing 1994 and 2001 Mashpee land use development within these watershed regions, and reviewing water quality modeling, it was concluded that adjustments for time of travel would not substantially improve the analysis in this system. Although the percentage of nitrogen loads in the less than 10-year subwatersheds ranges between 38 and 100% of total watershed load, 81% of the overall system load is within 10 years flow to Eastern Waquoit Bay. A similar analysis conducted for the entire Waquoit Bay watershed suggests that the system is almost in steady state, with annual watershed inputs equaling outputs (Brawley et al. 2000). Therefore, the 10 year time of travel subwatersheds were eliminated and the number of subwatersheds was reduced to 21. In this analysis, it is important to note that even with Mashpee'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 nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to ponds and embayments.

Table IV-1. Percentage of nitrogen loads in less than 10 time of travel subwatersheds to Eastern Waquoit Bay

	LT10	GT10	TOTAL	%LT10
WATERSHED	kg/yr	kg/yr	kg/yr	
Upper Quashnet River Total	9926	2491	12417	80%
Middle Quashnet River	987	144	1131	87%
Lower Quashnet River	449		449	100%
Red Brook	2806	818	3624	77%
Lower Red Brook	639	94	734	87%
Hamblin Pond	2086	281	2367	88%
Little River	572		572	100%
Jehu Pond	1262	622	1885	67%
Great River	374	121	495	76%
Lower Great River	1637		1637	100%
Flat Pond	320	520	840	38%
Flat/Sage Pond Transition	269		269	100%
Sage Pond	188		188	100%
TOTAL SYSTEM	21516	5091	26608	81%

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 Eastern Waquoit Bay, the model used Mashpee, Falmouth, and Sandwich-specific land-use data transformed to nitrogen loads using both regional nitrogen load factors and local site-specific data (such as 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 Quashnet River. Attenuation during transport through each of the major fresh ponds was determined through (a) comparison with other Cape Cod lake studies (Eichner, E.M., et al., 1998; Eichner, E.M., et al., 2003) and (b) data collected on each pond. Internal nitrogen recycling was also determined within the Eastern Waquoit Bay embayment system; 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

Project staff obtained digital parcel and tax assessors data from the Towns of Mashpee, Falmouth, and Sandwich. Mashpee and Falmouth's land use data is from 2001, while Sandwich's data is from 2000. The parcel and assessors databases from the three towns were integrated by using the Cape Cod Commission Geographic Information System (GIS) for the MEP analysis. 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. New development outside of this travel time is included within the projected build-out nitrogen load. 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 seven land use categories: 1) residential, 2) commercial, 3) industrial, 4) undeveloped, 5) mixed use, 6) golf course, and 7) public service, including road rights-of-way. Within the Eastern Waquoit Bay subwatersheds, the predominant land use on an areal basis is public service/government, which includes a portion of the Massachusetts Military Reservation and protected open space along the Quashnet River. Public service occupies 54% of the total watershed area to eastern Waquoit Bay (Figure IV-2). In contrast, while single-family residences occupy approximately 15% of the total area of the watershed to eastern Waquoit Bay, this land use type includes 61% of all the parcels. Commercial properties are fairly limited in the watershed with two small clusters located on Route 28 and Route 151.

In order to estimate wastewater flows within the study area, MEP staff also obtained 1997 through 1999 water use information from the Mashpee Water District, 2000 water use information from the Town of Falmouth, and 1998 through 2000 water use information from the Sandwich Water Department. Water use information was linked to the parcel and assessors data using GIS techniques. In addition to water use information, flow, effluent quality, and service area information was obtained from the Town of Mashpee and the Massachusetts Department of Environmental Protection for the two wastewater treatment facilities (WWTFs) in the watershed: Mashpee High School and Southport (Table IV-2). This information was used instead of water use information to calculate nitrogen loads for parcels within the service areas to these facilities.

Table IV-2. Private Wastewater Treatment Facilities in the Eastern Waquoit Bay Watershed

System Name	Average Effluent Characteristics	
	Flow (gallons per day)	Total Nitrogen Concentration (mg/liter)
Mashpee High School	1,632	7.01
Southport	17,811	5.48

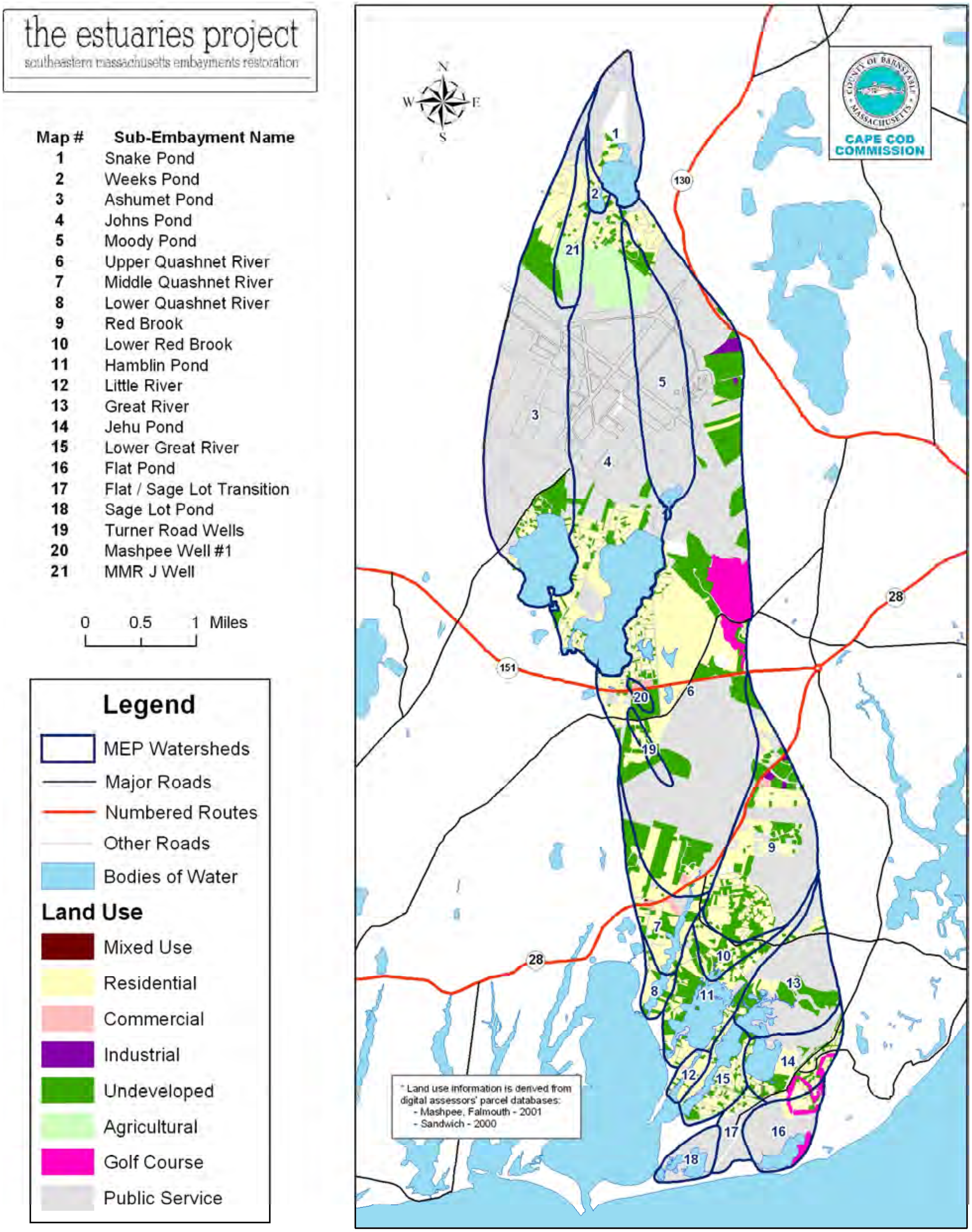


Figure IV-1. Land-use coverage in the Eastern Waquoit Bay watershed. Watershed data encompasses portions of the Towns of Mashpee, Falmouth, and Sandwich, MA.

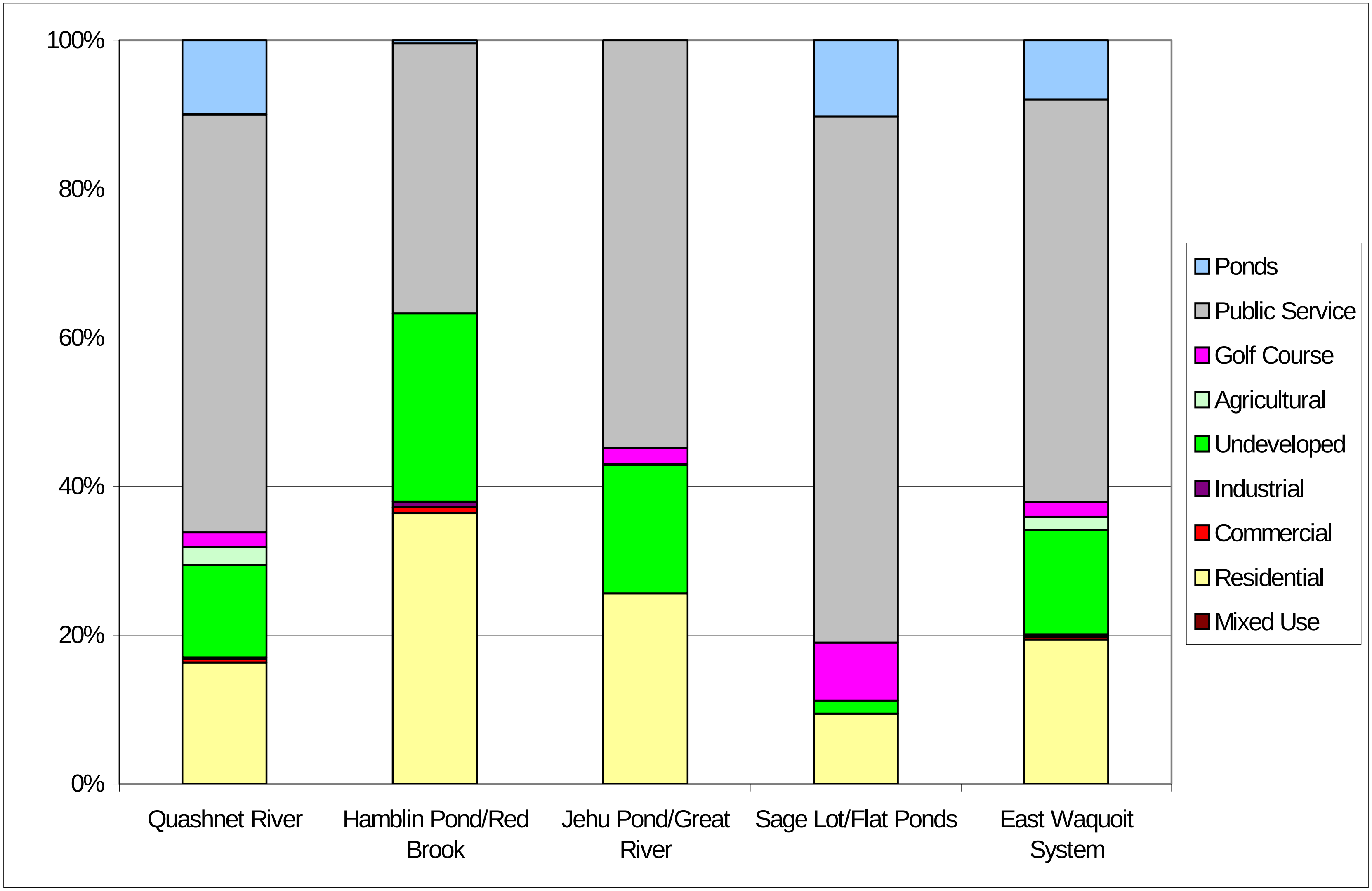


Figure IV-2. Distribution of land-uses within the major subwatersheds to the Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River estuaries and the entire watershed to the eastern portion of the Waquoit Bay System.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

All wastewater is returned to the aquifer within the eastern Waquoit Bay watershed either through the two private WWTFs or individual on-site septic systems. Wastewater within the watershed is predominantly treated through on-site septic systems. Approximately 31% of the residential, commercial, and industrial area developed in the watershed is connected to the two private wastewater treatment facilities. Both WWTFs are located in the Quashnet River subwatershed (Figure IV-3). Both WWTF's are providing significant nitrogen removal from their respective wastewater flows with total nitrogen in effluent of 5.5 and 7.0 mg N/L for Southport and Mashpee High School, respectively. These effluent levels compare favorably relative to the significantly higher levels (generally 4-5 times higher) in raw wastewater (e.g. the present MMR WWTF has a long-term average total nitrogen level of 35 mg N/L).

Wastewater based nitrogen loading from the residential properties 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%) conducted at DEP's Alternative Septic System Test Center at MMR, found that nitrogen removal within the septic tank was small (1%-3%), with most of the removal within the soil adsorption system (Costa et al. 2001).

In order to check the reliability of parcel linked water use as a proxy for wastewater flow, average influent flow at two nearby WWTFs (Mashpee Commons and Willowbend) was compared to average parcel water use (based upon water supply data) within the respective service areas. Wastewater engineering studies conventionally assume 90% of water used in a town is converted to wastewater (*e.g.*, Stearns and Wheler, 1999, Weiskel and Howes, 1991). This was consistent with the MEP findings in the Town of Chatham, based upon comparison public water supply flows versus return to the Town's WWTF. Within the Eastern Waquoit Bay watershed, the extensive mix of land uses connected to a municipal treatment facility is not available, so data from the two private WWTFs in the adjacent Popponesset Bay watershed were examined. Average WWTF flow data was used to gauge whether the 90% return flow is an appropriate assumption for this locale. Based on average flows, 79% of the Mashpee Commons water use is returned as wastewater to the associated WWTF, while 87% of the Willowbend water use is returned as wastewater to its WWTF. Given the land-uses tied to each of these WWTF's, this analysis supports the use of a 90% return flow estimate for adjustment of public water supply water use records to wastewater flows within the sub-watersheds to the eastern Waquoit Bay sub-embayments.

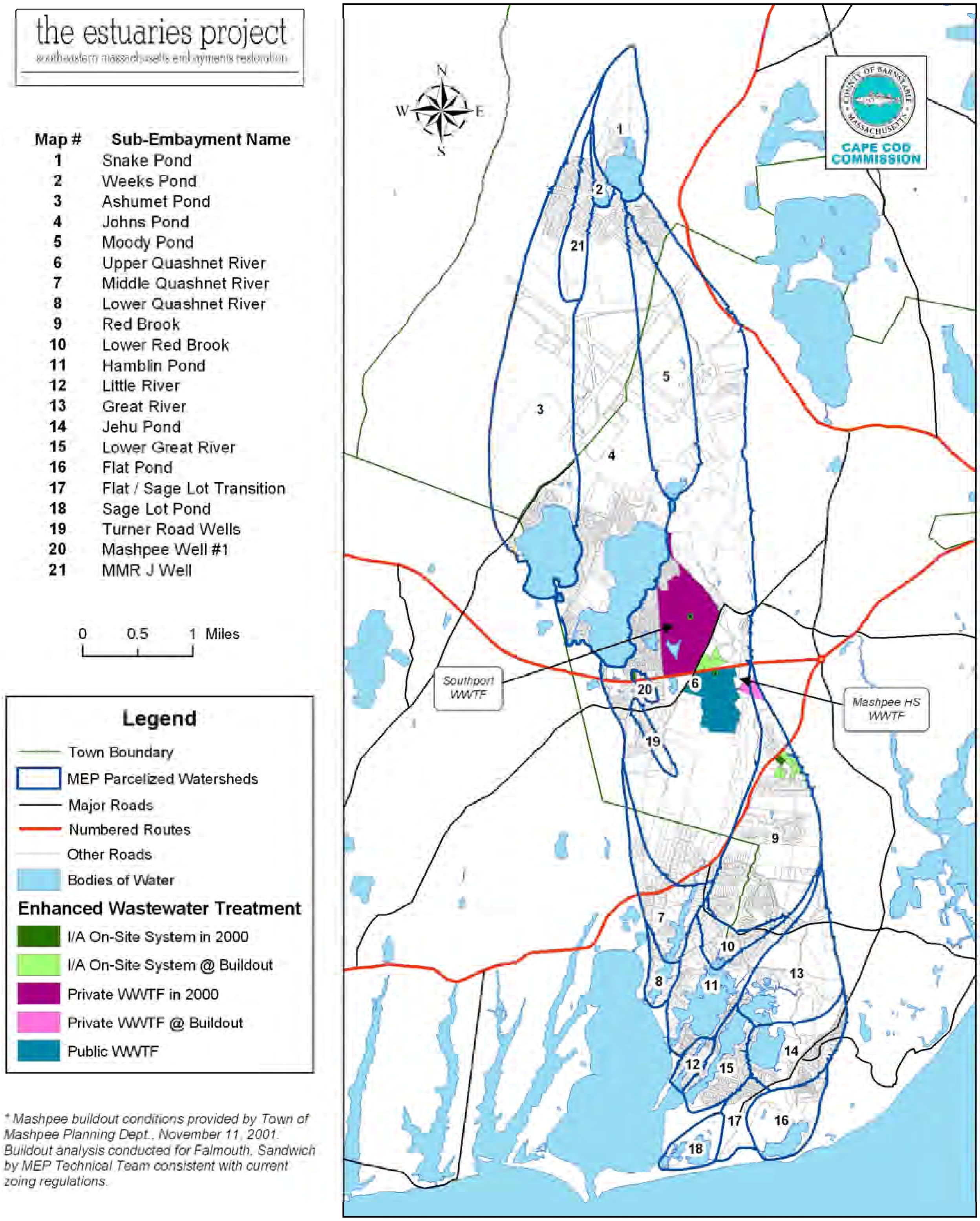


Figure IV-3. Parcels, Parcelized Watersheds, and Wastewater Treatment Facilities within the watershed to the eastern portion of the Waquoit Bay System.

The adjustment for 10% consumptive water use (i.e. 90% wastewater return flow) is an appropriate proxy for wastewater flows on the 76% of parcels with measured water use. However, 646 (24%) of the developed parcels in the eastern Waquoit Bay watershed do not have water use in the available database. These parcels are assumed to utilize private wells. A water use estimate for these parcels was developed based on measured water use from similar land uses. Of the 646 parcels without water use data, virtually all (634 or 99%) are classified as residential parcels or condominium parcels (land use codes 101 to 112), 10 are commercial (land use codes 300 to 389) and 2 are industrial (land use codes 400 to 439). In order to estimate water use by these parcels, MEP reviewed water use for residential, commercial, and industrial properties within the watershed that had measured water use (Table IV-3). Because water use information also forms the basis for evaluation of nitrogen loads at watershed build-out and about ¼ of the residential properties utilize wells, MEP staff reviewed other factors to assess whether average or median water use was most appropriate for residential land use estimates.

Table IV-3. Water Use in Popponesset Bay Watershed

Land Use	State Class Codes	# of Parcels	Water Use (gallons per day)		
			Average	Median	Range
Residential	101	2,053	154	118	1 to 3,177
Commercial	300 to 389	23	407	277	0 to 2,096
Industrial	400 to 439	4	62	5	0 to 237

As a check on water use data, MEP evaluated population data from the US Census. Average occupancy within the Town of Mashpee during the 2000 US Census was 2.46 people per household, while Falmouth was 2.36 and Sandwich was 2.75. If the Census occupancies are weighted based on the portion of the Eastern Waquoit Bay watershed that each town occupies, the Bay watershed average occupancy is 2.52. The Massachusetts on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd). Therefore, based on these regulations each person would generate 55 gpd. Using the Census occupancy data and these flow data, it is possible evaluate the use of average versus median flow data from the water use records. If the median water use of 118 gpd is multiplied by 0.9 (correcting for 10% consumptive use) and then divided by 55 gpd, the resulting occupancy is 1.93. In contrast, if the same procedure is applied to the average water use, the resulting occupancy is 2.52, which is the same as the Bay watershed average occupancy.

In order to provide a further check whether average residential water use was appropriate for build-out analysis and for parcels with private wells, project staff also reviewed annual water use for the Mashpee Water District between 1988 and 1998 (Earth Tech, 1999). Although the number of service connections more than doubled between 1988 and 1998 (from 1,956 to 5,695), the average annual water use per service connection generally fluctuated over a fairly narrow range (146.9 to 194.8 gpd). The overall average over this period is 161 gpd, while the average for 1998, which is the middle year of those reviewed for this analysis, was 153.7 gpd. The overall average is within 5% of the average water use determined by the MEP analysis.

Based on this analysis, MEP staff concluded that average residential water use was most appropriate for use in the nitrogen loading calculations for developed residential parcels that did not have parcel specific water use information and for future residences determined from the buildout assessment. Note that the nitrogen load modeling does not rely on the accuracy of the population data or on corrections for seasonality, but directly on the water use and the correction for consumptive use (90% return as wastewater). Similar comparisons were not available for commercial or industrial water uses, which have a much wider range of activities.

Commercial and industrial building footprints were made available to MEP staff as part of an impervious surface GIS coverage provided by the Mashpee Planning Department (virtually all of these land-uses are within the Town of Mashpee). Project staff used this data to review 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 average commercial and industrial water use is 81.5 gpd/1,000 ft^2 of building. This value was used to determine water use for all existing commercial and industrial buildings without water use and for all commercial and industrial additions as a result of the buildout analysis. Buildout building areas were determined by the Mashpee Planning Department. Based on a review of zoning, no commercial or industrial buildout additions were included for either the Falmouth or Sandwich portions of the Eastern Waquoit Bay watershed.

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 undertook an assessment of lawn fertilizer application rates and a review of leaching rates for inclusion in the land-use Nitrogen Loading Sub-Model.

The initial effort was to determine nitrogen fertilization rates for residential lawns within selected embayment watersheds within 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 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 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 Eastern Waquoit Bay are listed in Table IV-4.

Table IV-4. Primary Nitrogen Loading Factors used in Eastern Waquoit Bay MEP analysis. General factors are from the MEP modeling evaluation (Howes & Ramsey 2001). Site-specific factors are derived from Mashpee, Sandwich, and Barnstable data. *Data from MEP lawn study in Falmouth, Mashpee & Barnstable 2001.

<table>
<tr><td>Nitrogen Concentrations:</td><td>mg/l</td><td>Recharge Rates:</td><td>in/yr</td></tr>
<tr><td>Wastewater</td><td>35</td><td>Impervious Surfaces</td><td>40</td></tr>
<tr><td>Road Run-off</td><td>1.5</td><td>Natural and Lawn Areas</td><td>27.25</td></tr>
<tr><td>Roof Run-off</td><td>0.75</td><td colspan="2">Water Use/Wastewater:</td></tr>
<tr><td>Direct Precipitation on Embayments and Ponds</td><td>1.09</td><td>For Parcels wo/water accounts:</td><td>Gpd</td></tr>
<tr><td>Natural Area Recharge</td><td>0.072</td><td>Single Family Residence</td><td>154</td></tr>
<tr><td>Fertilizer:</td><td></td><td rowspan="2">Commercial & Industrial Properties</td><td rowspan="2">81.5 per 1,000 ft^2 of building</td></tr>
<tr><td>Average Residential Lawn Size (ft^2)*</td><td>5,000</td></tr>
<tr><td>Residential Watershed Nitrogen Rate (lbs/lawn)*</td><td>1.08</td><td rowspan="2">For Parcels w/water accounts:</td><td rowspan="2">Measured annual water use</td></tr>
<tr><td colspan="2">Nitrogen Fertilizer Rate for golf courses, cemeteries, and public parks determined by site-specific information</td></tr>
<tr><td colspan="2">Private WWTF flow and effluent nitrogen:
see Table IV-2</td><td colspan="2">Wastewater determined by multiplying water use by 0.9</td></tr>
</table>

IV.1.3 Calculating Nitrogen Loads

Once all the land and water use information was linked to the parcel coverages, parcels were assigned to the various sub-watersheds using GIS methodologies. Parcels divided by the watershed “line” were assigned to the sub-watershed which contained more than 50% of the land area. Following the assigning of these boundary parcels, all large parcels were further examined individually 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. Information for individual 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) and 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 sub-embayment to eastern Waquoit Bay. However, the effort was undertaken to better define the sub-embayment loads to enhance the use of the Linked Watershed-Embayment Model for the analysis of management alternatives.

Following the assignment of all parcels to individual watersheds, tables were generated for each of 36 sub-watersheds to summarize water use, parcel area, sewer connections, private wells, and road area. As mentioned above, these tables were then condensed to 21 subwatersheds based upon the results of the time of travel analysis (<10 yr vs. > 10 yr) discussed above.

The 21 individual sub-watershed assessments were then integrated to generate annual nitrogen loading rates to the estuarine waters of the Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River, and Sage Lot/Flat Pond sub-embayments, as well as the overall Eastern Waquoit Bay system. 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 support the development of nitrogen management alternatives. Within the eastern Waquoit Bay System the major types of nitrogen loads are: wastewater (septic systems and the WWTF), fertilizer, impervious surfaces, direct atmospheric deposition to water surfaces, and recharge from natural areas (Table IV-5). The output of the watershed nitrogen loading model is the annual mass (kilograms N) of nitrogen to each sub-watershed to the various sub-embayments (and freshwater ponds and streams), by land use category (Figures IV-4 a-e). 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.

IV.1.3.1 J Well Correction to Nitrogen Loads

As a result of the groundwater modeling completed for the MEP analysis, contributing areas were delineated for the three drinking water supply wells in the Eastern Waquoit Bay watershed. As a general rule, water withdrawn for drinking water supplies is distributed within the area close to where it is withdrawn; the water is then recharged back to the aquifer via septic systems along with any nitrogen in the original water withdrawn and additional nitrogen associated with its residential or commercial usage. However, the J Well in the eastern Waquoit Bay System watershed distributes the water it withdraws to facilities on the Massachusetts Military Reservation (MMR) and the wastewater generated at these facilities is treated at the MMR WWTF and discharged near the Cape Cod Canal.

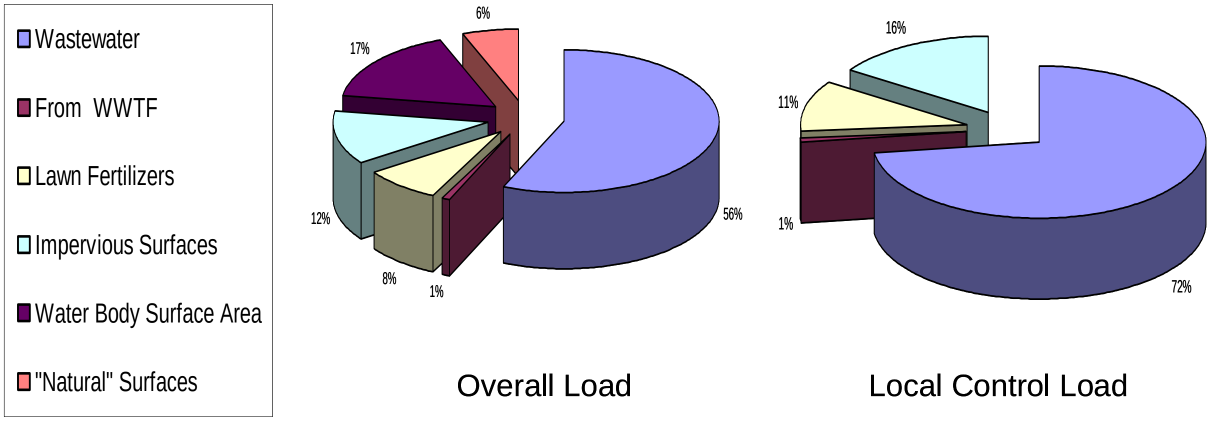
In order to address this removal of nitrogen from the eastern Waquoit Bay watershed, MEP staff estimated the nitrogen in water pumped from the J Well. This load is a combination of the load associated with recharge and land-use discharges to the aquifer within the watershed to the J Well (Watershed ID #21) and 30% of the load being recharged from Weeks Pond (including a portion of the load being recharged from Snake Pond into Weeks Pond). The estimated annual load removed by the J Well is 1,061 kg (see Table IV-5).

Since nitrate-nitrogen can be monitored in the J Well, MEP staff had an opportunity to verify the estimate of annual N removal. The watershed to the J well is 169 acres (see Figure III-1). Using USGS's annual recharge rate for the Eastern Waquoit watershed (27.25 inches), the watershed would capture 45,693 ft^3/d. According to USGS data used in the watershed delineations for this project, average pumping of the J Well between 1995 and 2000 is 40,614 ft3/d. On 4/16/02 a water sample collected from the J Well had a nitrate-nitrogen concentration of 3.14 mg/l . Using this concentration and the two estimates of pumping/recharge, an estimate of the annual nitrogen load captured by the J Well is 1,318 to 1,483 kg compared to the 1,061 kg from the land-use analysis. This analysis generally supports the nitrogen loading estimate for the J Well; further monitoring data from the well would be necessary to reliably assess variations in the nitrogen load captured by the J Well. For the MEP nitrogen loading analysis to the eastern sub-embayments to the Waquoit Bay System, a removal of 1,061 kg N yr-1 was used for the J Well, since it is based upon a more rigorous data set and is a conservative estimate.

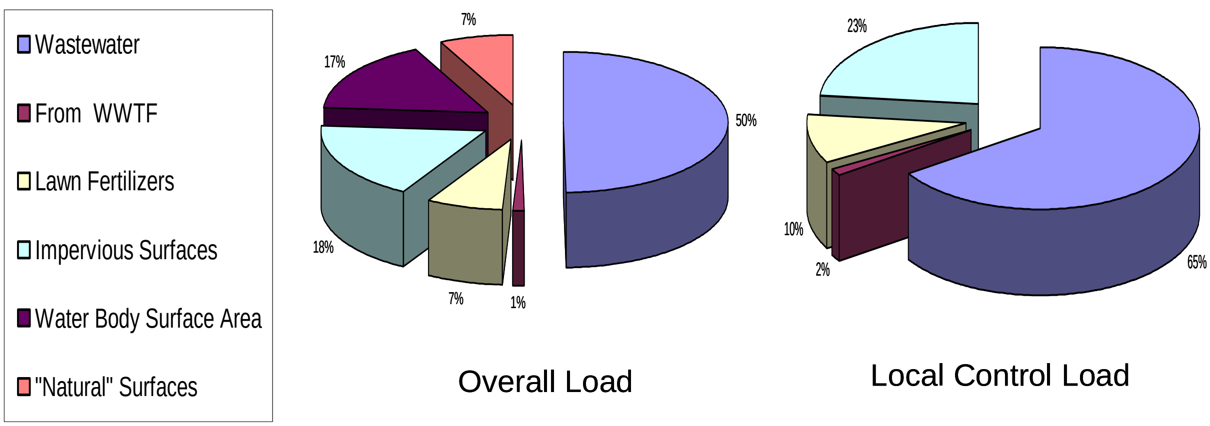
Table IV-5. Eastern Waquoit Bay System Nitrogen Loads. Build-out is based on current zoning and represents additional un-attenuated N expected in the future.

*All values in kilograms/year		Mashpee N Loads by Input:							% of Pond Outflow	Present N Loads		
Name	Watershed ID#	From Septic Systems	From WWTF	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load
E Waquoit Bay	**1 to 20 (minus 21)**	**13517**	**159**	**1981**	**2910**	**3959**	**1413**	**13105**		**23938**		**18823**
Quashnet River	1 to 8, 19, 20 (-21)	6759	159	993	2419	2254	996	10095		13580		9882
Upper Quashnet River	1 to 6, 19, 20 (-21)	5893	159	888	2366	2040	943	9743		12289		8591
Moody Pond (MP)	5	87	0	6	184	35	44	101	44%	356	50%	178
Turner Road Well No. 5	19	95	0	0	2	0	10	51		108		108
Mashpee Well No. 1	20	62	0	0	20	0	4	27		87		87
Johns Pond Summary (JPS)	4 + AP, WP, SNP, MP	1643	7	164	1375	1562	270	1318	67%	5021	50%	2113
Johns Pond (JP)	4	1805	0	189	1412	1499	207	497		5111		5111
Snake Pond (SNP)	1	3	0	2	1	73	10	6	21%	89	50%	44
Moody Pond (MP)	5	108	0	7	231	44	55	126	56%	445	50%	223
Weeks Pond (WP)	2 + SNP	43	0	5	8	86	9	19	70%	151	50%	63
Ashumet Pond (AP)	3	474	10	40	387	614	120	1305	65%	1645	50%	822
Removed from AP watershed by J Well												
MMR J Well	21	679	0	48	47	0	30	66		804		
Weeks Pond (WP)	2 + SNP	19	0	2	3	37	4	8	30%	54	50%	27
Middle Quashnet River	7	658	0	42	44	122	45	315		911		911
Lower Quashnet River	8	208	0	63	9	92	8	37		380		380
Hamblin Pond/Red Brook	9 to 12	4386	0	319	278	635	217	2330		5835		4417
Red Brook	9	2366	0	165	157	0	146	1701		2835	50%	1418
Lower Red Brook	10	459	0	45	37	20	20	205		581		581
Hamblin Pond	11	1227	0	73	57	558	44	366		1958		1958
Little River	12	334	0	37	26	57	7	59		461		461
Jehu Pond/Great River	13 to 15	1956	0	368	178	723	139	629		3365		3365
Great River	13	131	0	16	21	202	81	329		451		451
Jehu Pond	14	966	0	239	72	246	39	146		1563		1563
Lower Great River	15	858	0	113	85	275	20	154		1351		1351
Sage Lot/Flat Pond	16 to 18	416	0	300	35	346	61	51		1159		1159
Flat Pond	16	253	0	278	17	174	34	0		756		756
Flat / Sage Lot Ponds Transition	17	163	0	22	19	0	12	51		215		215
Sage Lot Pond	18	0	0	0	0	172	16	0		188		188

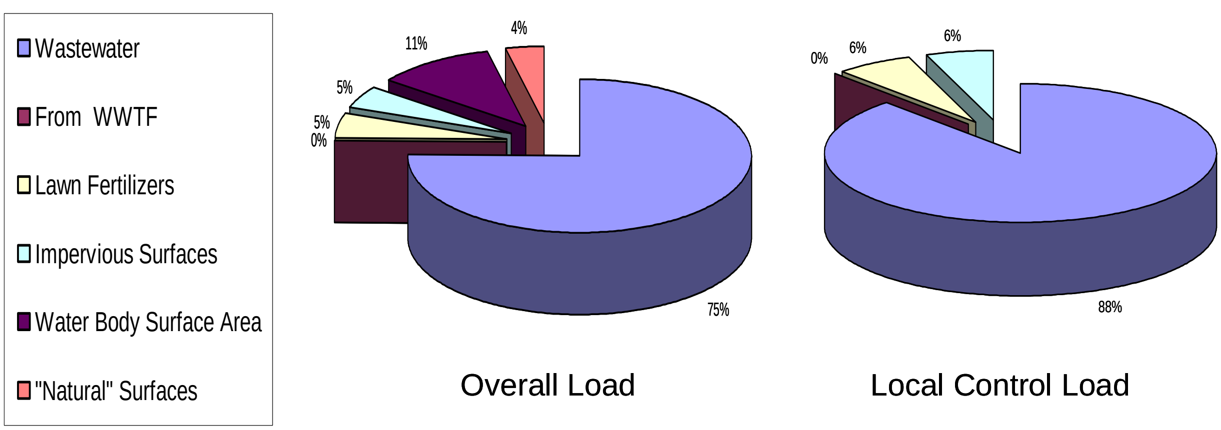
Note: Unattenuated buildout nitrogen load is sum of additions to wastewater (septic system and WWTF), fertilizers, and impervious surface caused by the development of undeveloped parcels within the subwatersheds.



a. Eastern Waquoit Bay

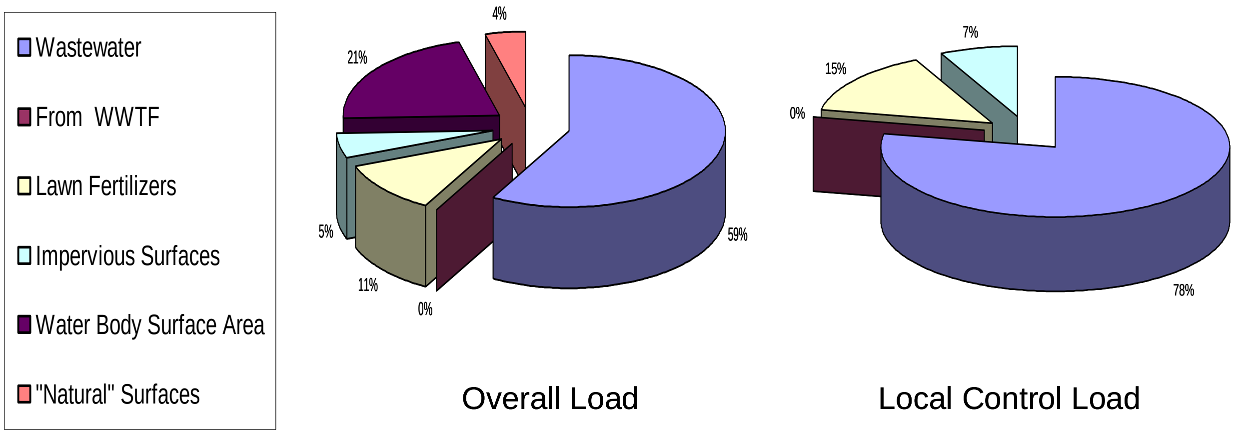


b. Quashnet River

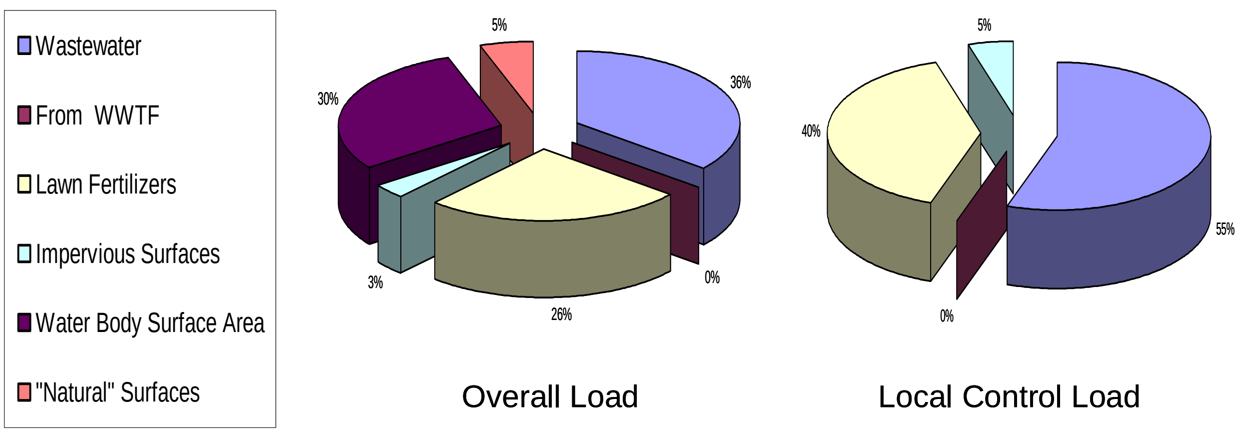


c. Hamblin Pond/Red Brook

Figure IV-4. (a-c). Land use-specific unattenuated nitrogen load (by percent) to the Eastern Waquoit Bay system, Quashnet River, and Hamblin Pond/Red Brook.



d. Jehu Pond/Great River



e. Sage Lot/Flat Pond

Figure IV-4. (d-e). Land use-specific unattenuated nitrogen load (by percent) to Jehu Pond/Great River and Sage Lot/Flat Pond.

Freshwater Pond Nitrogen Loads

Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table creating 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 pondwater 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 ecosystems that 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 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. Table IV-5 N Load summary 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.

Nitrogen attenuation was estimated directly, based using watershed nitrogen loading rates to the ponds coupled with pond residence time and nitrogen concentrations. Pond water quality information was collected from a couple of sources. One source is data collected during late August in both 2001 and 2002 under the Cape Cod Pond and Lake Stewardship (PALS) program, which is a collaborative Cape Cod Commission/SMAST Coastal Systems Program effort. Citizen volunteers in Mashpee and Sandwich collected dissolved oxygen and temperature profiles, Secchi disk depth readings and water samples at various depths within the following ponds: Snake, Johns, and Moody (Figure IV-1). Water samples were analyzed at the SMAST Coastal Systems Analytical Facility for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. This data was supplemented with data collected on Ashumet, Johns, and Snake ponds through various Massachusetts Military Reservation (MMR) monitoring programs (e.g., AFCEE, 1998).

In order to estimate nitrogen attenuation by the ponds, physical and chemical data for each pond was collected for each great pond (>10 acres). 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. 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-6 summarizes the pond attenuation estimates calculated from land-use modeled nitrogen inflow loads and nitrogen loads which appear to be recharged to the downgradient aquifer or to outflow streams from each pond based on pond characteristics and measured nitrogen levels. Nitrogen attenuation within these ponds appears to vary between 51 and 89%. 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 studies of other southeastern Massachusetts freshwater systems including Ashumet Pond (AFCEE, 2000) and Agawam/Wankinco River Nitrogen Discharges (CDM, 2001) have supported 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, annual measurements of the nitrogen discharge from the Quashnet River, that receives water from the aquifer, a major portion of which has passed through a pond system, documents a very large natural attenuation during transport through the upper watershed (see Section IV-2, below). This site-specific data supports a pond attenuation in the 50%-60% range as a conservative estimate for the watershed to the eastern Waquoit Bay System.

Table IV-6. Nitrogen attenuation by Freshwater Ponds in the Eastern Waquoit Bay watershed 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 Eastern Waquoit Bay analysis using the MEP Linked N Model 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 %
Ashumet	MA-808	218	19.2	1.6	41%
Johns	MA-818	338	18.9	2.1	84%
Moody	MA-793	19	2.6*	0.1*	93%
Snake	SA-568	83	10.1	2.0	51%
				Mean	64%
*estimated from PALS sampling data				s.d.	23%

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 apportionment was based on the percentage of pond discharging shoreline bordering each downgradient sub-watershed. The percentages of shoreline are shown in Table IV-5.

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 to the eastern Waquoit Bay system was determined from the GIS database (Figure IV-5). Buildout of parcels within the Town of Mashpee portion of the Eastern Waquoit Bay watershed were determined by the Mashpee Planning Department, including commercial and industrial parcel estimates. Buildout of parcels within the portions of the watershed within the Town of Sandwich and Falmouth were based on sub-divisions using minimum lot size included in current zoning. All municipal overlay districts (*e.g.*, water resource protection districts) 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-4 and discussed above. A summary of potential additional nitrogen loading from build-out is presented as unattenuated and attenuated loads in Table IV-5. 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 Bay waters under any realistic physical conditions.

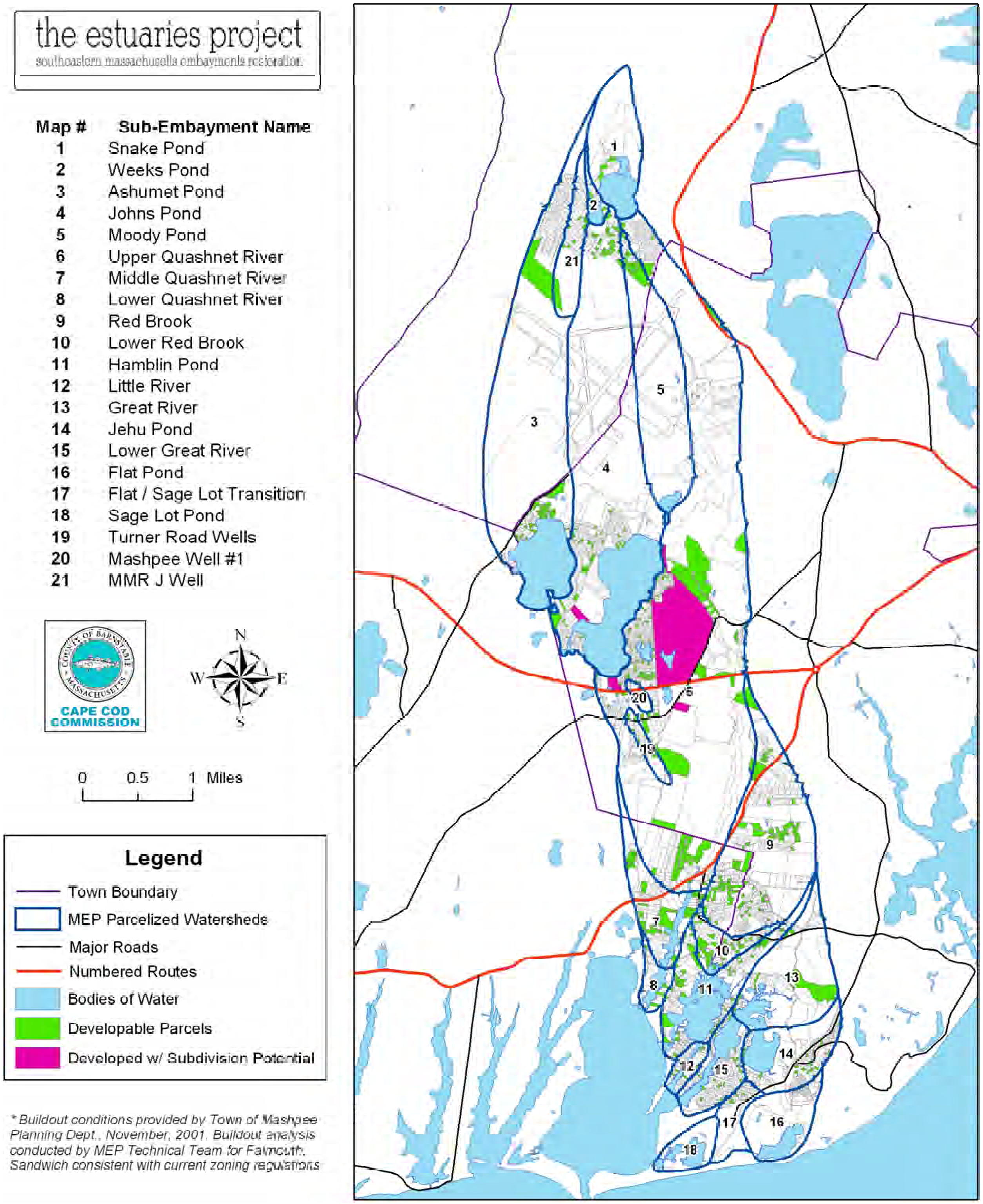


Figure IV-5. Distribution of present parcels which are potentially developable within the Eastern Waquoit Bay watershed.

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 or sewering analysis) 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 sub-embayment (Hamblin Pond, Jehu Pond and Quashnet River) of the overall Waquoit Bay embayment system 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 is through groundwater 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 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 which 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. Within the eastern Waquoit Bay System Watershed most of freshwater flow and transported nitrogen passes through a surface water system and frequently multiple systems, producing the opportunity for significant nitrogen attenuation.

Failure to determine the attenuation of watershed derived nitrogen overestimates the nitrogen load to receiving 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, in 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 discharge from the WWTF was attenuated by a small salt marsh prior to reaching Harbor waters. Similarly, the small tidal basin of Frost Fish Creek in the Town of Chatham showed ~20% nitrogen attenuation or watershed nitrogen load prior to discharge to Ryders Cove. Clearly, proper development and evaluation of nitrogen management options requires determination of the nitrogen loads reaching an embayment, not just loaded to the watershed.

Given the importance of determining accurate nitrogen loads to embayments for developing effective management alternatives and the potentially large errors associated with ignoring natural attenuation, direct integrated measurements were undertaken as part of the

MEP Approach. MEP conducted a study on natural attenuation relating to one sub-embayment of the eastern Waquoit Bay System in addition to the natural attenuation measures by fresh kettle ponds, addressed above. The additional site-specific study was conducted in the major surface water flow system, (i.e. the Quashnet River discharging to the tidal portion of the Quashnet River sub-embayment). This river carries the majority of the freshwater inflow to the eastern Waquoit Bay System, so that it provides a significant check on the nitrogen loading rate to this entire system.

Quantification of watershed based nitrogen attenuation is contingent upon being able to compare nitrogen load to the embayment system directly measured in freshwater stream flow (or in tidal marshes, net tidal outflow) to nitrogen load as derived from the detailed land use analysis (Section IV.1). Measurement of the Quashnet River (at Route 28) provided 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 eastern Waquoit Bay System. Flow and nitrogen load were measured at the gauging site for 16 months of record (Figure IV-6). During study period, velocity profiles were completed on the river every month to two months. Periodic measurement of flows over the entire stream gauge period of record allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the continuously record of stream stage by the US Geological Survey. At the start of the MEP nutrient threshold analysis of the Quashnet River sub-embayment to Waquoit Bay, a stream gauge was deployed proximal to the USGS gauging station. Though the USGS has been collecting river stage continuously since 1988, the MEP chose to deploy a stream gauge in the same location as the USGS in order to confirm the accuracy of MEP stage measurements relative to an independent measure. The gauge was deployed in June of 2002 and measured at a 10-minute frequency until the first week of August 2002 when it was stolen. A second gauge was not deployed due to the likelihood of theft or vandalism. The MEP Technical Team concluded that using the USGS stage data, in conjunction with an MEP developed rating curve for the Quashnet River, would yield satisfactory results and enable the MEP to meet its objectives of accurately determining nitrogen attenuation within the Quashnet River watershed. Though only a short term (June 2002 – August 2002) stage record was measured by the MEP for the Quashnet River, a comparison of MEP measured stage to the USGS measured stage was still possible and showed that both stage records agreed well. Both stage records showed similar peaks and magnitude of peaks indicating that the MEP gauge was functioning as an accurate measure of river stage.

A complete annual record of stream flow (365 days) was generated for the Quashnet River. The annual flow record for the river was merged with the nutrient data set generated through the weekly water quality sampling to determine nitrogen loading rates to the tidally influenced portion of the Quashnet River.

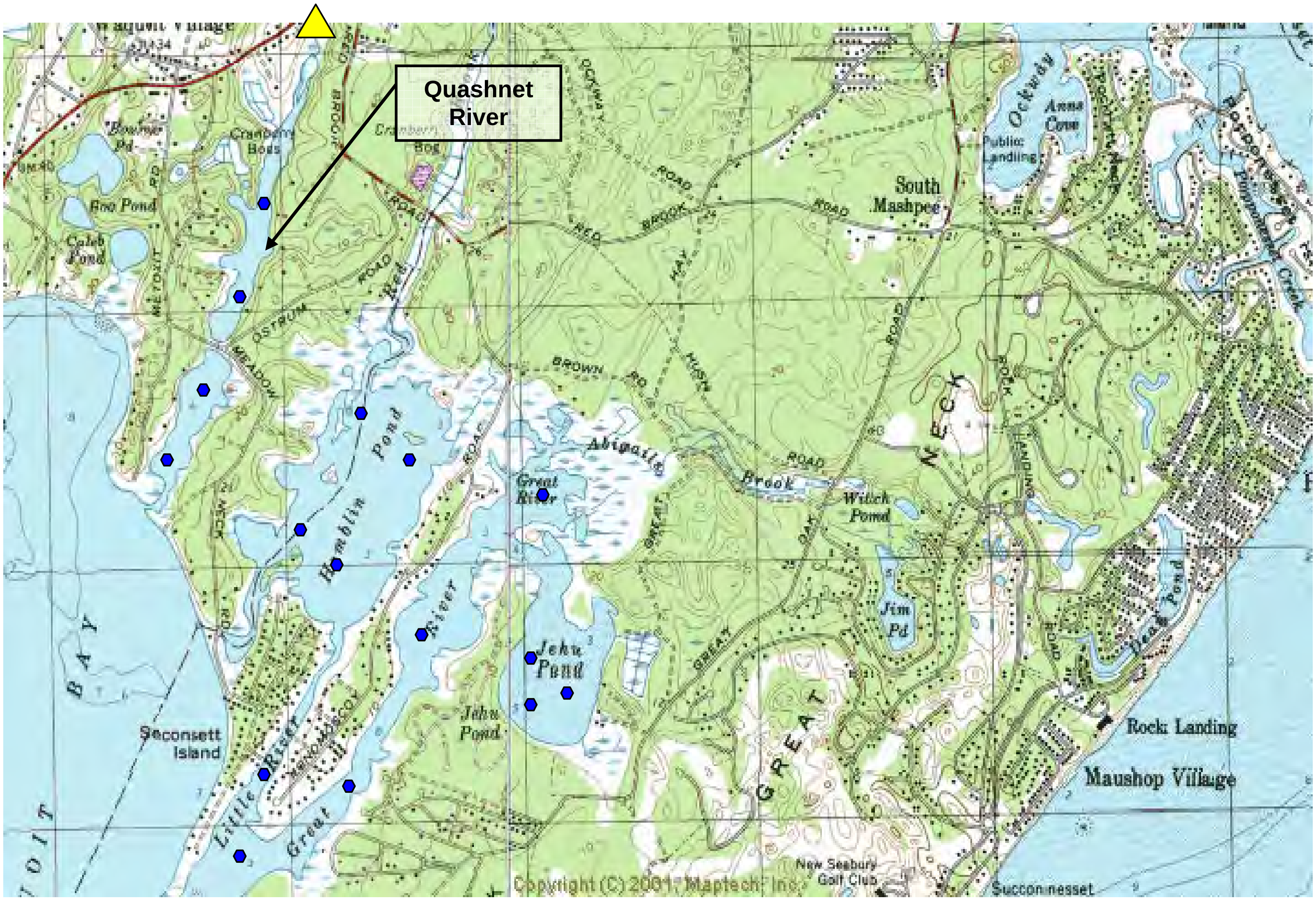


Figure IV-6. Location of Stream gauge (yellow triangle) and benthic coring locations (blue hexagons) in the Quashnet River sub-embayment to the Waquoit Bay system.

IV.2.2 Surface Water Discharge and Attenuation of Watershed Nitrogen: Quashnet River to Quashnet River Estuary

John's Pond (and an associated network of down stream cranberry bogs) is one of the larger ponds within the study area and unlike many of the freshwater ponds, John's Pond has stream outflow rather than discharging solely to the aquifer on the down-gradient shore. This stream outflow, the Quashnet 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 stream bed associated with the Quashnet 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 Quashnet River above the gauge site and the measured annual discharge of nitrogen to the tidal portion of the Quashnet River, Figure IV-6.

A water quality sampling station was established at the USGS stream gaging location within the outflow stream (Quashnet River) from John's Pond, which is also fed by groundwater inflow within its lower reaches Quashnet River (primarily sub-watershed #6, Figure IV-5). A rating curve was developed for the cross section of the Quashnet River that is situated upgradient of Route 28 prior to the discharge of the Quashnet River into the tidally influenced portion of the Quashnet River as depicted in Figure IV-6.

River flow (volumetric discharge) was measured monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Quashnet River site based upon these measurements and measured water levels at the USGS 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 transport to the estuarine portion of the Quashnet River (Figure IV-8, Figure IV-9, Table IV-7 and Table IV-8). 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 the gauge site (Figure IV-7). Comparison of measured and predicted discharge is used to confirm that the stream is capturing the entire recharge to its up-gradient contributing area. This comparison also can be used as a check on the watershed area, although it is limited in that the stream flow estimate from the watershed area is a long-term average and the MEP gauge estimate is over 12-16 months. In the MEP study, the 2 estimates were only ~10% apart (i.e. good agreement). This freshwater balance is also important for supporting the nitrogen attenuation calculations.

The final stream gauge record available for this analysis of freshwater stream flow and associated attenuated nitrogen load covers a period of 365 days for the discharge of the Quashnet River to the tidally influenced portion of the lower Quashnet River prior to discharge to Waquoit Bay. Using the available flow measurements, a stream flow record for a complete year was constructed for the freshwater portion of the Quashnet River from which annual and average daily freshwater flow to the Quashnet River Estuary was determined (Figures IV-8 and IV-9 and Table IV-7). The annual freshwater flow record for the Quashnet River, as developed using USGS measured stage and the stage – discharge relation developed by the MEP, was compared to the modeled flows as determined by the USGS and were found to be within two percent of each other indicating excellent agreement (Table IV-8).

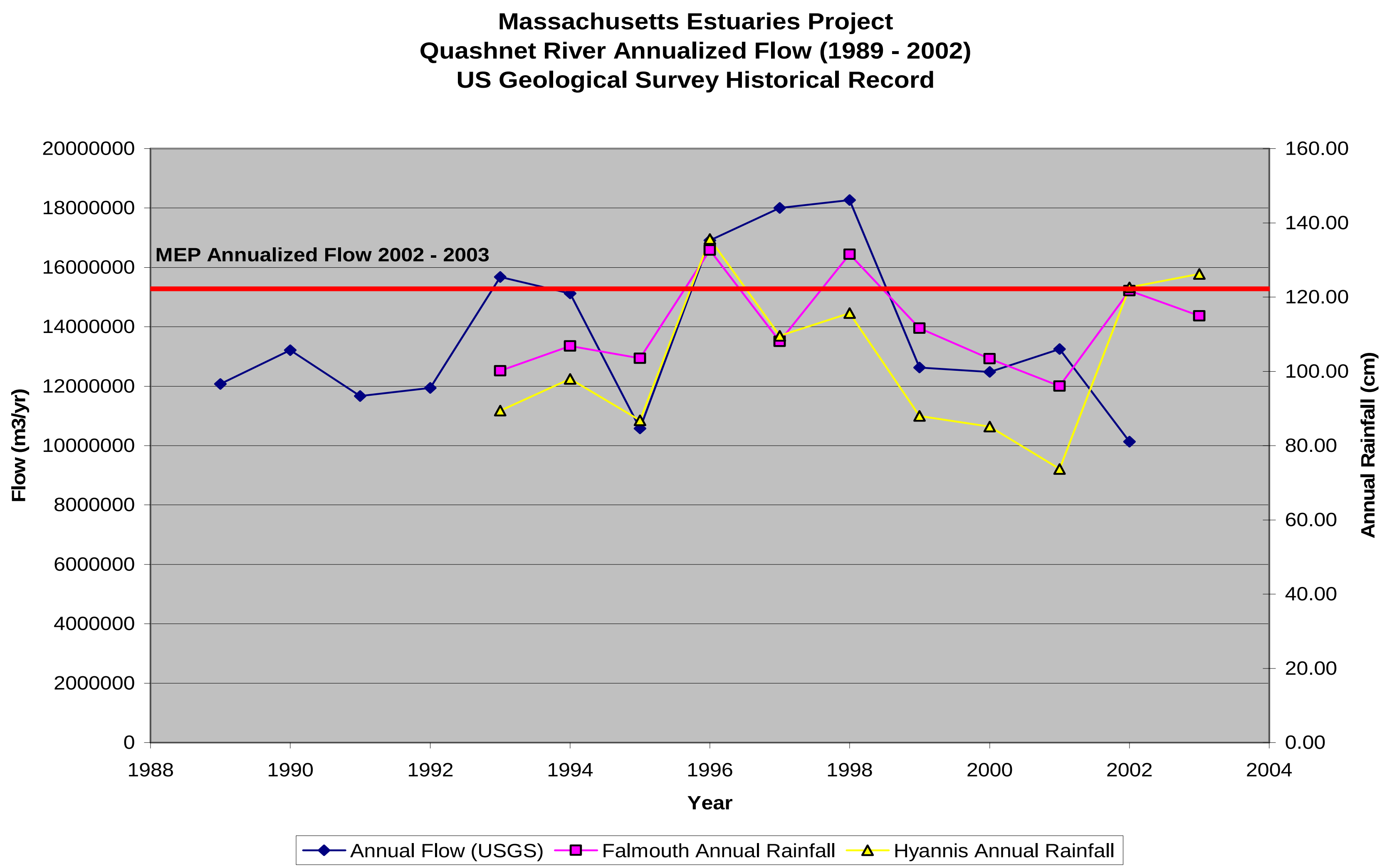


Figure IV-7. Comparison of historical Quashnet River flows as determined by the US Geological Survey (1989 – 2002) and the annualized flow developed by the MEP (2003) all relative to annual rainfall from meteorological stations in Falmouth and Hyannis, MA.

Table IV-7. Comparison of water flow and nitrogen discharges from the Quashnet River to the lower estuarine reach of the Quashnet River. The "Stream" data is from the MEP stream gauging effort. Watershed data is based on the MEP watershed modeling effort by the USGS.

Stream Discharge Parameter	Discharge Quashnet River to Waquoit Bay[b] (MEP)	Discharge Quashnet River to Waquoit Bay (USGS)	Data Source
Total Days of Record[a]	365	365	(1)
Flow Characteristics			
Stream Average Discharge (m3/day)	41529	40712	(1) / (2)
Contributing Area Average Discharge (m3/day)	46715	46715	(3)
Proportion Discharge Stream vs. Contributing Area (%)	89%	87%	
Nitrogen Characteristics			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.204	0.204	(1)
Stream Average Total N Concentration (mg N/L)	0.497	0.497	(1)
Nitrate + Nitrite as Percent of Total N (%)	41%	41%	(1)
Total Nitrogen (TN) Average Measured Stream (kg/d)	20.66	20.29	(1)
TN Average Contributing Area Attenuated Load (kg/d)	23.30	23.30	(3)
TN Average Contributing Area UN-attenuated Load (kg/d)	33.67	33.67	(4)
Attenuation of Nitrogen in Pond/Stream (%)	39%	39%	(5)

[a] from September 10, 2002 to September 10, 2003
[b] Flow and N load to Quashnet River including John's Pond Contributing Area

(1) MEP developed stream rating curve used in conjunction with USGS stage data.
(2) USGS stage and flow data.
(3) Calculated from MEP watershed delineations to John's Pond; the fractional flow path from each sub-watershed which contribute to Quashnet River and the annual recharge rate. This represents a long-term average flow estimate.
(4) As in footnote #3, with the addition of pond and stream conservative attenuation rates.
(5) Calculated based upon the measured TN discharge from the river vs. the unattenuated watershed load.

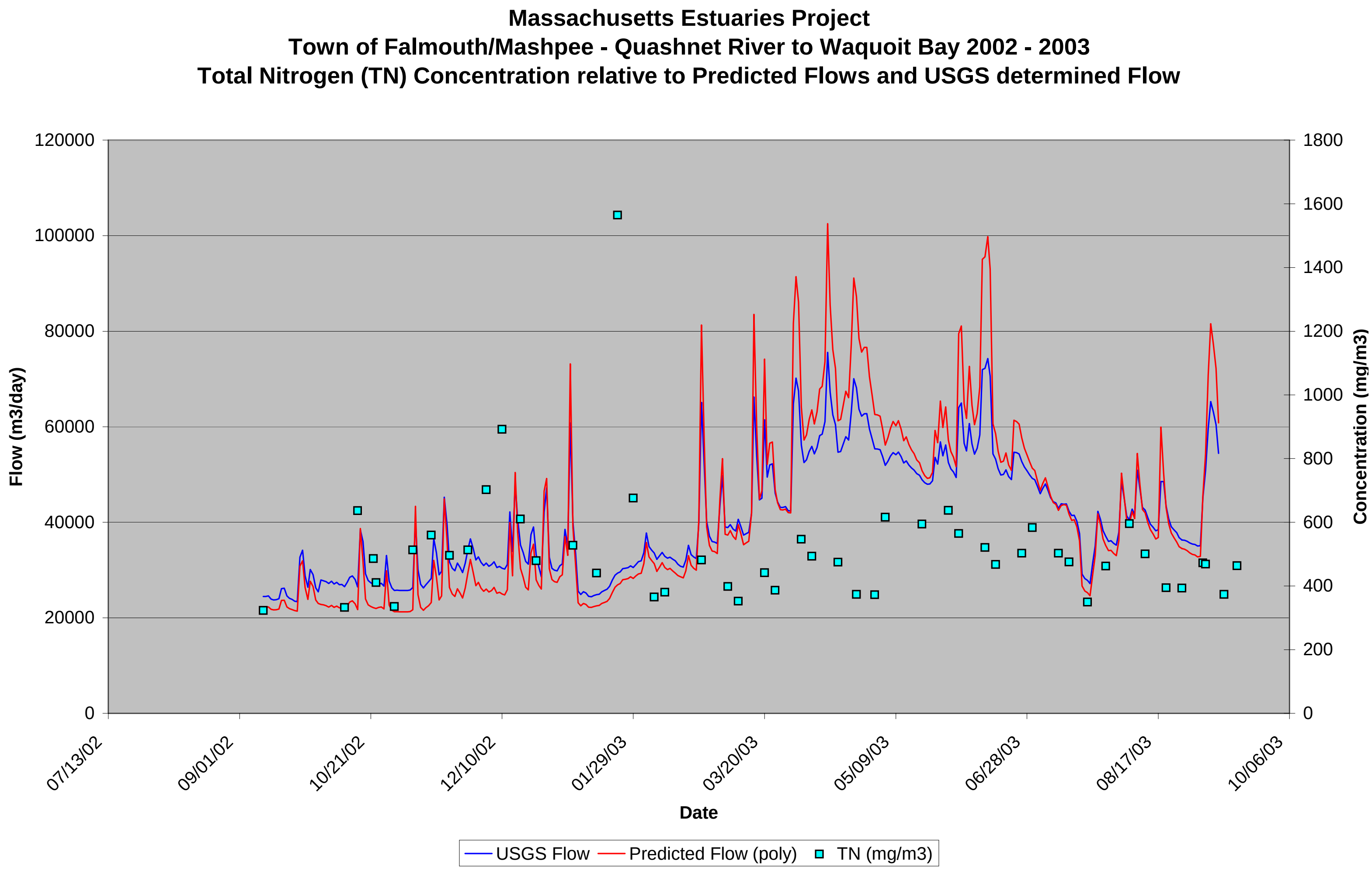


Figure IV-8. Quashnet River annual discharge developed from a stream gauge maintained above the tidal reach of the lower Mashpee River estuarine waters. Nutrient samples were collected weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-7).

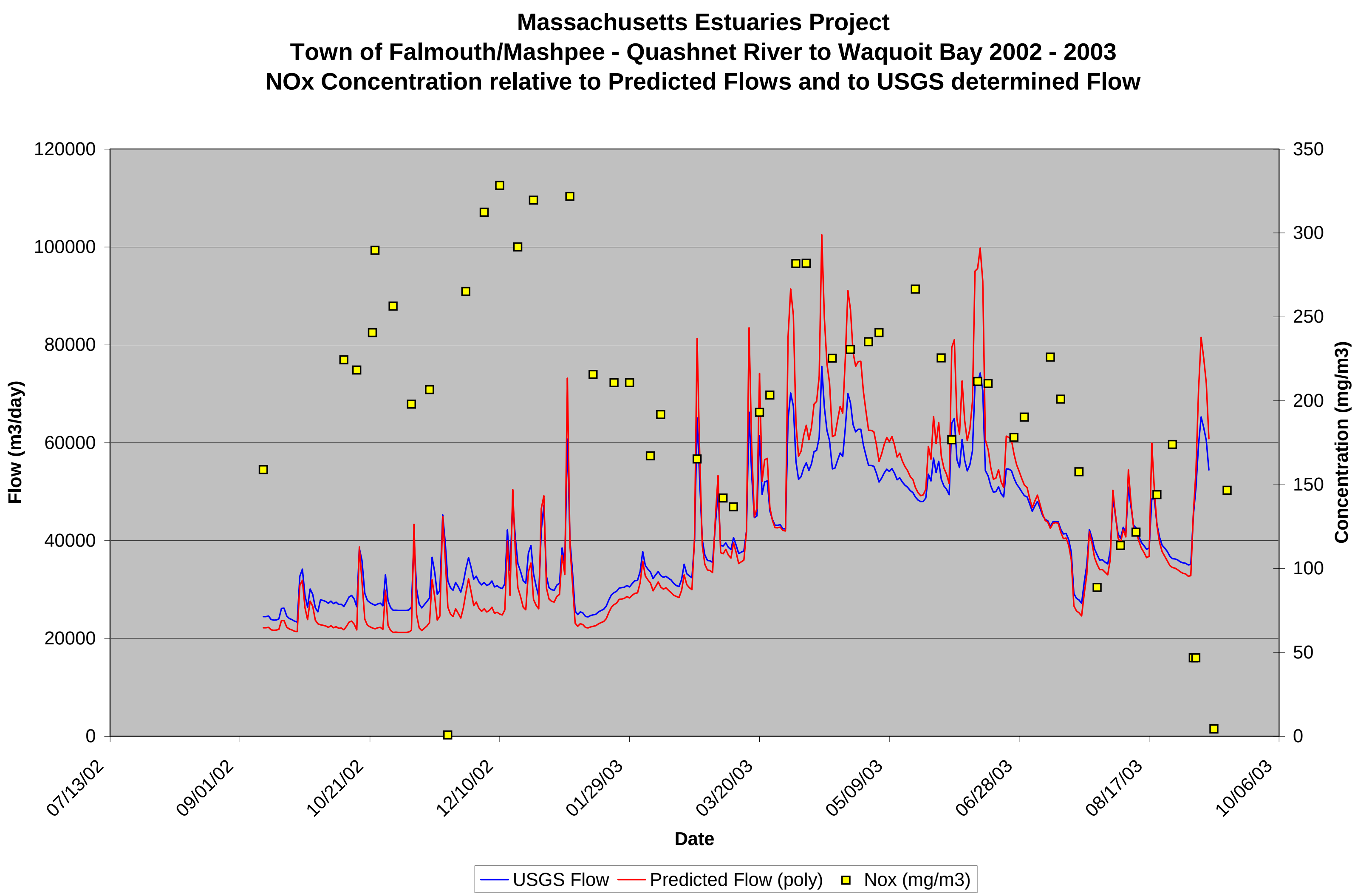


Figure IV-9. Nitrate + Nitrite (Nox) concentration and Quashnet River annual discharge developed from a stream gauge maintained in the outflow from John’s Pond discharging to tidally influenced portion of Quashnet River. Nutrient samples were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-7).

Table IV-8. Summary of Flow and Nutrient loads from both the Quashnet River discharging to tidally influenced Quashnet River estuarine reach.

Quashnet River	PERIOD	DISCHARGE (m3/yr)	ATTENUATED LOAD (Kg/yr)	
			Nox	TN
MEP-Gauge	September 10, 2002 to September 10, 2003	**15157967**	**3088**	**7540**
USGS-Gauge	September 10, 2002 to September 10, 2003	**14860060**	**3028**	**7407**
MEP-Water Balance *	Long-term Annual	**17050974**	**3478**	**8491**

*** Based upon watershed area (2464 hectares) and 27.25 in/yr recharge.**

Total nitrogen concentrations within the Quashnet River outflow were relatively high, 0.497 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 20,658 g/day (20.66 kg/d) and a measured total annual TN load of 7,540 kg/yr. In the Quashnet River, nitrate was the predominant form of nitrogen (41%), 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 Quashnet 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 (20.66 kg N d^{-1}, 7540 kg yr^{-1}) discharged from the freshwater Quashnet River and the nitrogen mass entering from the associated watershed (33.67 kg N d^{-1}, 12,289 kg yr^{-1}) the integrated measure of nitrogen attenuation by the pond/river ecosystem is 39%. This is consistent with the land-use model which yielded and integrated nitrogen attenuation of 31%, since pond and stream attenuation in the watershed model use conservative attenuation factors (see Table IV-6). The directly measured nitrogen loads from the Quashnet 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 Task was to quantify the summertime exchange of nitrogen, between the sediments and overlying waters within each major basin area within the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River sub-embayments to Waquoit Bay. 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 Hamblin Pond/Little River, Jehu Pond/Great River and Quashnet River estuaries 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 Waquoit Bay or Nantucket Sound). However, some of these phytoplankton particles are grazed by zooplankton or filtered from the water by shellfish and other benthic animals. 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 small enclosed basins (e.g. Hamblin Pond, Jehu Pond, etc). 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 the MEP Technical Team, 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 Hamblin Pond/Little River, Jehu Pond/Great River and Quashnet River estuaries, in order to determine the contribution of sediment regeneration to nutrient levels, sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected, during the most sensitive summer interval (July-August), from 16 sites (Figure IV-10) in 2001 as part of the Mashpee Sewer Commission investigation supporting wastewater facility planning. Measurements of total dissolved nitrogen, nitrate + nitrite, ammonium were made in time-series on each incubated core sample. As part of a separate research investigation, the rate of oxygen uptake was also determined and measurements of sediment bulk density, organic nitrogen, and carbon content were taken. These measurements were made by the MEP Technical Team members in the Coastal Systems Program at SMAST-UMD working with the Town of Mashpee.

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 a small boat. 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. Sampling was distributed throughout each sub-embayment (Figure IV-10) and the results for each site combined for calculating the net nitrogen regeneration rates for the water quality modeling effort.

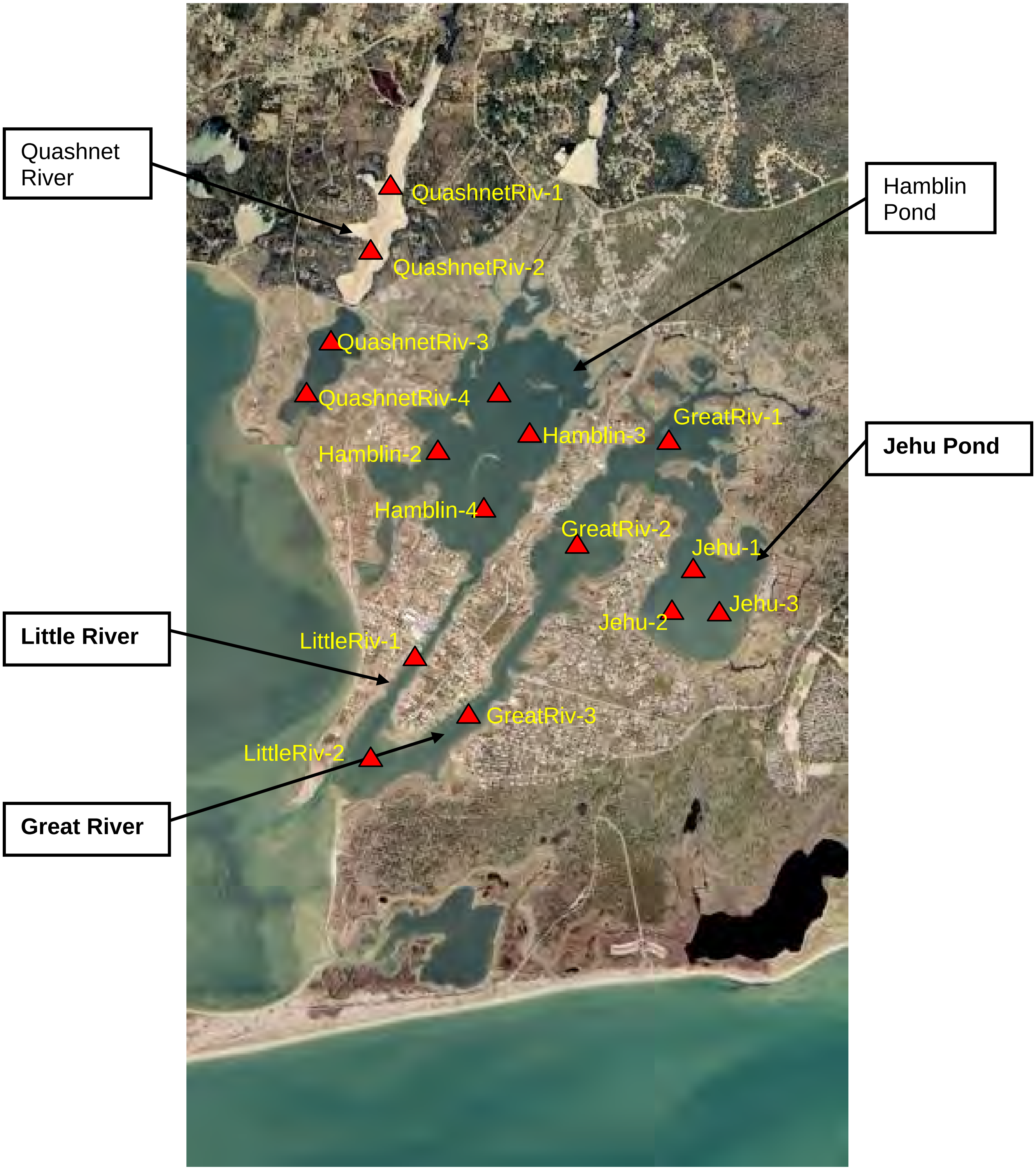


Figure IV-10. Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River estuaries to the Waquoit Bay System, locations (red triangles) of sediment sample collection for determination of nitrogen regeneration rates. Numbers are for reference to Table IV-9.

Table IV-9. Rates of net nitrogen return from sediments to the overlying waters of eastern Waquoit Bay sub-embayments (Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River). 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, standard deviation is based upon the linear regression of six samples per mean.		
Sub-Embayment	Sediment Nitrogen Release	
	Mean mg N m^{-2} d^{-1}	s.d. mg N m^{-2} d^{-1}
JEHU POND		
JP 1	83.7	30.2
JP 2	35.9	34.2
JP 3	36.2	27.8
GR 1	50.9	0.7
GR 2	64.0	8.3
GR 3	165.8	5.2
HAMBLIN POND		
HP 1	99.4	19.6
HP 2	-26.3	26.0
HP 3	-14.6	20.5
HP 4	-21.0	26.5
LR 1	34.9	6.9
LR 2	20.8	9.8
QUASHNET RIVER		
QR Mid-Up 1	101.4	14.8
QR Mid 2	49.9	47.0
QR Low 3	42.3	20.2
QR Low 4	75.2	17.8

Sediment-watercolumn exchange follow 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 near shore of Waquoit Bay), 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) assay was conducted within 24 hours and the remaining sample 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 and has passed review by DEP/USEPA for chemical analysis for MEP.

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

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

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

Overall, coastal sediments are not overlain by nitrate rich waters and the major nitrogen input is via phytoplankton grazing or direct settling. In these systems, on an annual basis, the amount of nitrogen input to sediments is generally higher than the amount of nitrogen release. This net sink results from the burial of reworked refractory organic compounds, sorption of inorganic nitrogen and generally a relatively large loss through 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).

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

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

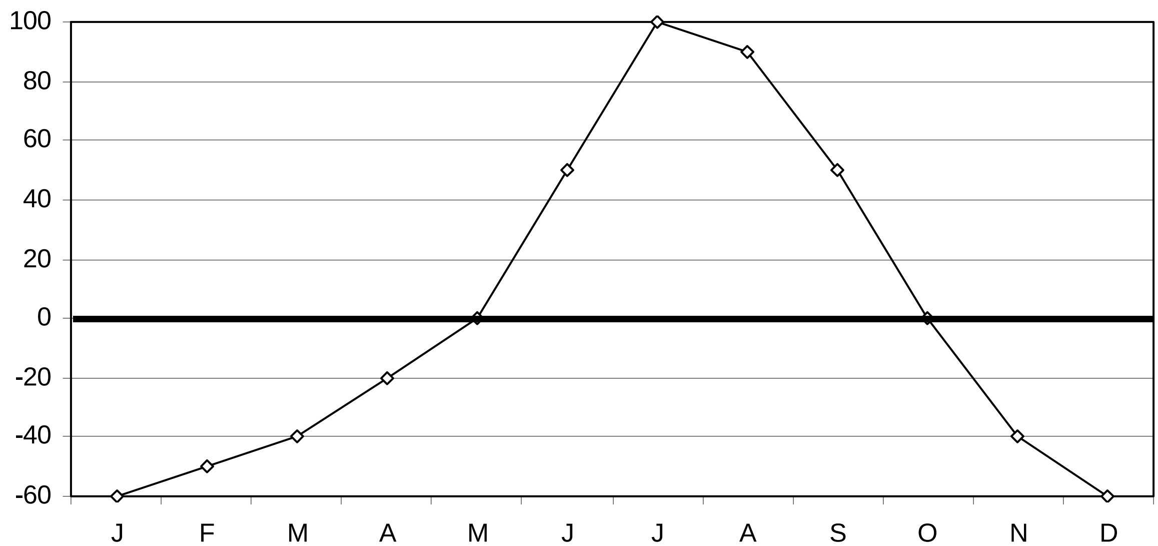


Figure IV-11. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

Sediment sampling was conducted within sub-embayments of the eastern Waquoit Bay System (Quashnet River, Hamblin Pond/Little River, Jehu Pond/Great River) in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-9). 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 bulk density 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 a 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 which 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) which would be accounted for by daily particulate carbon settling. This analysis indicated that sediment metabolism in the highly organic rich sediments of the wetlands and depositional basins is driven primarily by stored organic matter (ca. 90%). Also, in the more open lower portions of larger embayments, storage appears to be low and a large proportion of the daily carbon requirement in summer is met by particle settling (approximately 33% to 67%). This range of values and their distribution is consistent with ecological theory and field data from shallow embayments.

Net nitrogen release or uptake from the sediments within the eastern Waquoit Bay sub-embayments for use in the water quality modeling effort (Chapter VI) are presented in Table IV-9. It is clear that the sediments within the more nitrogen loaded regions show the highest nitrogen sources to the overlying waters. Within each estuary, the uppermost portion has the highest nitrogen release. Hamblin Pond showed net nitrogen uptake in some areas, most likely due to its generally oxidized nature. In contrast, Jehu Pond consistently showed net nitrogen release, possibly related to its periodic anoxia resulting in its reducing sediments (Chapter VII). In addition, Jehu Pond is a more restricted basin, likely with less wind-driven mixing (based on its fetch), than is Hamblin Pond also consistent with the nature of its sediments. The Quashnet River estuary receives more than 2 times the nitrogen loading (on a system area basis) than the other estuaries and has significant organic matter accumulations due to phytoplankton and macroalgal production. In addition, the lower basin has a “sill” formed by the flood tidal delta which enhances deposition which is reflected in the nitrogen release rate. The extensive salt marshes within the Great River and Little River estuarine regions are likely associated with the net nitrogen release from the sediments of these tidal rivers which support them. The observed sediment release rates within these estuaries is similar to the rates and distribution measured by MEP within adjacent Popponesset Bay. Most notably is the similarity between the estuarine reaches of the Quashnet River (mean=67 mg N m^{-2} d^{-1}) and Mashpee River (mean=72 mg N m^{-2} d^{-1}) with their generally similar hydrologic and physical characteristics. The observed sediment nitrogen release rates were used to determine spatially distributed nitrogen inputs from the sediments within each estuary in the Water Quality Model.

V. HYDRODYNAMIC MODELING

V.1. INTRODUCTION

To support the Town of Mashpee with their Comprehensive Wastewater Management Planning (CWMP), an evaluation of tidal flushing has been performed for the Waquoit Bay complex. Specifically, the Town is concerned with the influence of nitrogen load on coastal embayments within the borders of Mashpee (Jehu and Hamblin Pond, as well as the upper watershed of the Quashnet River). The field data collection and hydrodynamic modeling effort contained in this report, provides the first step towards evaluating the water quality of these estuarine systems, as well as understanding nitrogen loading "thresholds" for Jehu Pond, Hamblin Pond, and the Quashnet River. The hydrodynamic modeling effort serves as the basis for the total nitrogen (water quality) model, which will incorporate upland nitrogen load, as well as benthic regeneration within bottom sediments. Although the primary foci of the hydrodynamic and water quality analysis are the three Mashpee sub-embayments listed above, hydrodynamic modeling was performed for the entire Waquoit Bay complex. For logistics reasons, the MEP performed hydrodynamic data collection and modeling for the Waquoit Bay system, rather than limiting the evaluation to the Mashpee sub-embayments. It is anticipated that limited additional data collection for hydrodynamic analysis (e.g. current measurements through two inlets) will be required when the complete water quality evaluation is performed for Waquoit Bay.

Shallow coastal embayments are the initial recipients of freshwater flow and the nutrients they carry. An 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 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 nutrient and pollutant loading and the processes that help flush nutrients and pollutants from the estuary (e.g., tides and biological processes). Relatively low nutrient and pollutant loading and efficient tidal flushing are indicators of high water quality. The ability of an estuary to flush nutrients and pollutants is proportional to the volume of water exchanged with a high quality water body (i.e. Nantucket Sound). Several embayment-specific parameters influence tidal flushing and the associated residence time of water within an estuary. For Waquoit Bay and its sub-embayments, the most important parameters are:

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

The Waquoit Bay estuarine system (Figure V-1) is a tidally dominated embayment open to Nantucket Sound. The system separates the towns of Mashpee and Falmouth along the south coast of Cape Cod, Massachusetts. The system consists principally of sub-embayments Eel Pond, Childs River, Waquoit Bay, Moonakis/Quashnet River, Hamblin Pond, and Jehu Pond, as

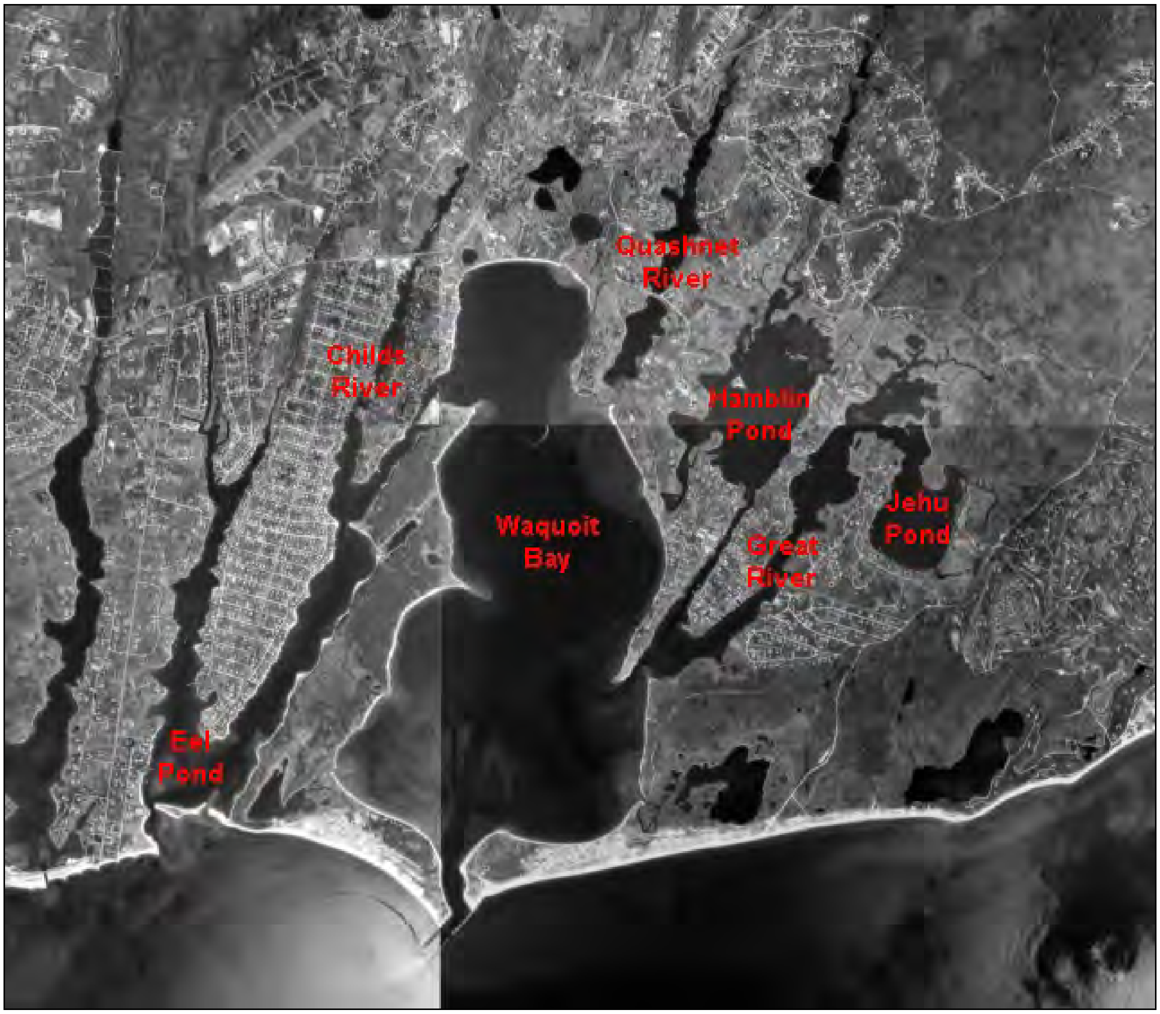


Figure V-1. Aerial photograph of the Waquoit Bay system.

well as numerous other smaller coves, creeks, and marshes. It is relatively shallow on average, exceptions being deeper channels that provide flow paths between the Nantucket Sound and the embayments. The approximate tidal range within the system is 1.5 feet, with Nantucket Sound tidal variations providing the hydraulic forcing that drives water movement throughout the system.

The objective of hydrodynamic modeling is to develop a numerical model to simulate accurately the hydrodynamic characteristics of the Waquoit Bay system. The calibrated model can be used to understand tidal circulation, as well as be extended to calculate system flushing rates. Further, the hydrodynamic model provides basis for water quality modeling, enabling the Towns (Mashpee and Falmouth) to understand how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

The Waquoit Bay sub-embayment is the largest body of water in the system, covering approximately 900 acres. Eel Pond is to the west of Waquoit Bay. It is connected to Waquoit Bay by Childs and Seapit Rivers, along with an opening onto Nantucket Sound. Great River and Hamblin Pond are located to the east of Waquoit Bay and contain the largest area of salt marshes in the system, approximately 100 acres. Moonakis River empties into the northeast portion of Waquoit Bay. The system is generally shallow, with a mean depth of 2.5 feet. Waquoit Bay contains the deepest section of water with a mean depth of 4.8 feet.

As described above, circulation in the system is dominated by tidal exchange with Nantucket Sound. From measurements made in the course of this study, the average tide range in Waquoit Bay is approximately 1.5 feet. The flow restrictions caused by natural restrictions and bridge abutments, produces minor reductions in the tide range in upper portions of the system. The reductions are on the order of 0.1-0.2 feet.

This hydrodynamic study proceeded as two component efforts. In the first portion of the study, bathymetry and tide data were collected in order to accurately characterize the physical system, and to provide data necessary for the modeling portion of the study. The bathymetry surveys of Waquoit Bay, Eel Pond, Childs River, Great River, Hamblin Pond, and Quashnet/Moonakis River were performed to determine the variation of embayment and channel depths throughout the system. This survey addressed the previous lack of adequate bathymetry data for this area. In addition to the survey, tides were recorded at six locations within the system for 29 days. These tide data were necessary to run and calibrate the hydrodynamic model of the system.

A numerical hydrodynamic model of the system was developed in the second portion of this study. Using the bathymetry survey data, a model grid mesh was generated for use with the RMA-2 hydrodynamic code. The tide data from a gage located offshore of the entrance to the Eel Pond entrance was used to define the open boundary conditions that drive the circulation of the model for the inlets to Waquoit and Eel Pond. Data from the six tide gages within the system were used to calibrate and verify model performance to ensure that it accurately represents the dynamics of the system.

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

V.2. GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE SYSTEM

The southern coast of Cape Cod in the vicinity of Waquoit Bay is a moderately dynamic region, where natural wave and tidal forces continue to reshape the shoreline. As beaches continue to migrate, episodic breaching of the barrier beach system can create new inlets that alter the pathways of water entering the estuary. Storm-driven inlet formation often leads to hydraulically efficient estuarine systems, where seawater exchanges more rapidly with water inside the estuary. However, this episodic inlet formation often is balanced by the gradual wave-driven migration of sediment along the barrier beach. If significant littoral drift exists, this longshore transport process typically will cause inlets to migrate in a downdrift direction. Armoring of inlet channels will “fix” the position of an inlet, preventing natural migration. Along the southern shore of Falmouth and Mashpee, including the two entrances to Waquoit Bay, coastal engineering structures control the position of the inlets.

As described in Aubrey et al. (1993), storms and human development have altered the number of inlets in the Waquoit Bay system through time. The general trends of shoreline movement and inlet opening/closing over the past century is shown in Figure V-2. Prior to 1938, the Waquoit Bay system was characterized by a single inlet at the boundary between the Towns of Falmouth and Mashpee. Following the September 1938 hurricane, a new inlet formed between Nantucket Sound and Eel Pond along the western side of the system. This inlet was filled by the U.S. Army in 1941. During the World War II, a road was constructed from the mainland to Washburn Island across the barrier beach. In addition, a series of groins were constructed along the Nantucket Sound shoreline south of Eel Pond during this time period. In 1944, the U.S. Army Corps of Engineers re-opened the 1938 inlet to Eel Pond. Aubrey et al. (1993) indicated that by 1955 the spit created by the breach had retreated into Eel Pond in response to a reduced sediment supply resulting from updrift coastal engineering structures. The retreating shoreline in the vicinity of the Eel Pond inlet has stranded three groins several hundred feet offshore. A 1964 report by the U.S Army Corps of Engineers indicated that the retreat of the western portion of Washburn Island was initially offset by accretion along the eastern end of the island (west of the Waquoit Bay jetty). Initial equilibration of the barrier beach to the new inlet configuration may be responsible for the initial shoreline retreat observed along the western portion of Washburn Island.

Following approximately 40 years of beach stability, Hurricane Bob in 1991 created a third inlet just east of the previously existing Eel Pond inlet (Figure V-2 and Figure V-3). This inlet has gradually infilled over the past 12 years. As of April 2004, the inlet was closed and the system has returned to its two inlet configuration.

To augment existing shoreline change information, a differential GPS shoreline was performed for the Falmouth and Mashpee coastline fronting Waquoit Bay during April and May 2004. In addition, an updated 1938 shoreline recently digitized by NOAA (http://www.ngs.noaa.gov/newsys_ims/shoreline/index.cfm) also was incorporated into the analysis. The 1938 to 2004 timeframe includes the large-scale change associated with the Eel Pond inlet formation and the associated shoreline equilibration. Although a statewide shoreline change analysis already exists (see Thieler et al., 2001 for more information), the shoreline change analysis incorporated in the present study utilizes a more accurate 1938 shoreline and a surveyed 2004 shoreline. Shorelines surveyed with differential GPS equipment significantly reduce the error associated with more remote techniques (e.g. interpretation of aerial photographs or maps). Therefore, this updated shoreline analysis provides more accurate long-term change data needed to effectively manage both the tidal inlets and the adjacent shorelines.

Figure V-4 illustrates the shoreline change between 1938 and 2004 As described above, the shoreline change immediately east of the Eel Pond inlet has been significant over the past 66 years. Formation of the 1938 inlet caused long-term reorientation of the shoreline. As described by the U.S. Army Corps of Engineers (1964), from 1891 to 1942, the shoreline of Washburn Island receded about 200 feet at its west end and moved seaward about 300 feet at the Waquoit Bay entrance. This trend continued between 1942 and 1961, where the west end receded approximately 500 feet and accretion adjacent to the Waquoit Bay entrance was an additional 100 feet. In the immediate vicinity of the Eel Pond entrance, the initial shoreline equilibration appeared to be relatively rapid (see Figure V-2 between 1941 and 1955). The reorientation of the shoreline between the two inlets required more time; however, the overall shoreline orientation has been stable since the mid-1970s. Shoreline change along the eastern half of Washburn Island and the regions to the east and west of the Waquoit Bay system are relatively minor, with change rates typically less than 1 ft/year. Exceptions to the low change rates exist in the area immediately east (downdrift) of the Waquoit Bay jetties. In this area, the local rate of change is up to 2 feet per year.

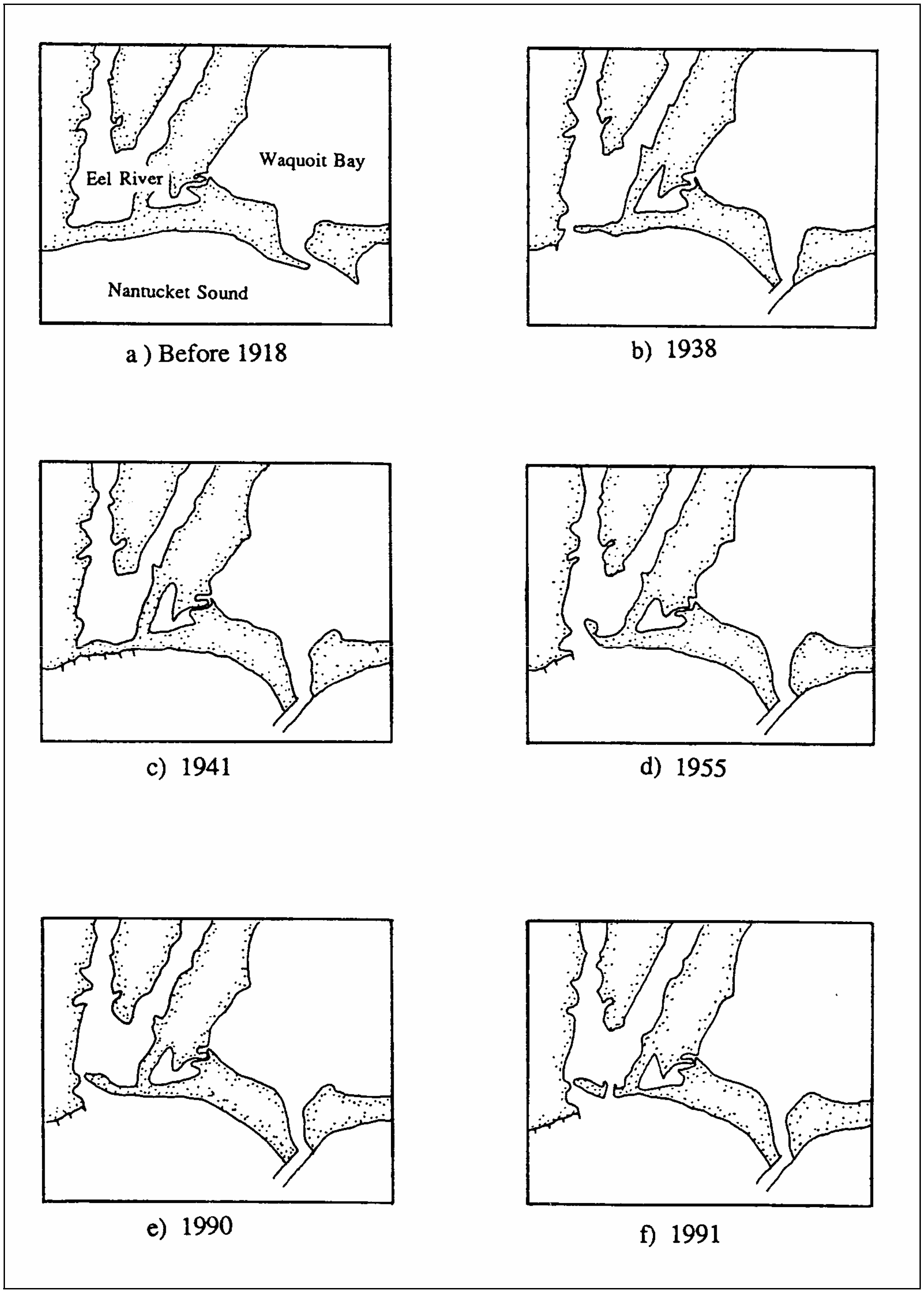


Figure V-2. Inlet changes within the Waquoit Bay system between the early 1900's and 1991 (from Aubrey et al., 1993).

Figure V-3. Photograph of the third inlet through the western portion of Washburn Island. The photograph was taken within a week of Hurricane Bob in 1991 (from Waquoit Bay National Estuarine Research Reserve website).

Based on tidal hydrodynamics alone, present-day conditions represent a more efficient flow pathway than the single inlet system that existed prior to the 1938 hurricane. As described in Aubrey et al. (1993), the two primary inlets interact essentially with different portions of the embayment. A simple stability analysis indicated that the two inlet system was stable. This hypothesis has been supported by the existence of the two inlet system for several decades. It was determined that the three inlet system was unstable under all conditions. As noted above, the third inlet has closed as of April 2004.

Manmade coastal structures along the Falmouth shoreline consist primarily of seawalls and/or revetments along the updrift shoreline (west of Eel Pond entrance). These structures likely have reduced the natural littoral sediment supply to the barrier beach system. In effect, this reduction in sediment supply may be partially responsible for the significant landward migration of the western portion of Washburn Island. In addition, this loss in local sediment supply likely was responsible for the slow rate of infilling observed for the third inlet formed by Hurricane Bob in 1991.

As depicted in Figure V-2, jetties were constructed at the Waquoit Bay entrance in 1918. This jetty system effectively prevents sediment from migrating across the inlet due to the regional east-to-west littoral drift. Although coastal engineering structures have altered the sediment dynamics in the Waquoit Bay system, they have maintained a consistent two inlet system. From the perspective of estuarine water quality, the stabilized two inlet system provides a healthier estuary than either the one inlet system or an estuarine system dominated by ephemeral inlets. In addition to improving tidal flushing, the formation of the 1938 inlet may have also contributed directly to the observed expansion of the *Spartina alterniflora* across much of the Waquoit Bay system over the 50 year period following the inlet's formation (Orson and Howes, 1992).

Figure V-4. Results of a shoreline change analysis utilizing the NOAA 1938 shoreline and a differential GPS shoreline measured in April/May 2004. Although large-scale retreat is limited to the western half of Washburn Island, other nearby areas of the Falmouth and Mashpee shoreline contain "hot spots" where erosion rates exceed 1 ft/year.

Coastal and estuarine management strategies need to account for both the natural and anthropogenic changes that have occurred in the Waquoit Bay system. Due to increased nitrogen load throughout the Waquoit Bay watershed, the two inlet system likely is required to maintain existing water quality conditions. In addition to reductions in nitrogen load to the embayments, management strategies should ensure that the two inlet system remains intact. As described above the existing form of the inlets likely is critical to both water quality and salt marsh health.

V.3 FIELD DATA COLLECTION AND ANALYSIS

A precise description of embayment geometries and hydrodynamic forcing processes is required for the development of numerical models. To support the hydrodynamic and water quality modeling effort in Waquoit Bay and the surrounding rivers and ponds, bathymetry of the embayments and water elevation variations were measured. The Waquoit Bay system consists of Waquoit Bay, Eel Pond, Childs River, Seapit River, Moonakis River, Hamblin Pond, Great River, and Jehu Pond (Figure V-5).

Bathymetry data was collected in regions where the coverage of historical bathymetry data lacked accuracy and/or detail necessary for evaluation of tidal hydrodynamics. Detailed bathymetric surveys of Eel Pond, Childs River, Seapit River, Moonakis/Quashnet River, Great

River, and the two inlets were conducted. Waquoit Bay, Hamblin Pond, and Jehu Pond were more sparsely sampled to identify general characteristics. The depth measurements were supplemented by existing NOAA bathymetry datasets to create computational grids of each system. Tidal elevation measurements within selected embayments were used for both forcing conditions and to evaluate tidal attenuation through each estuarine system.

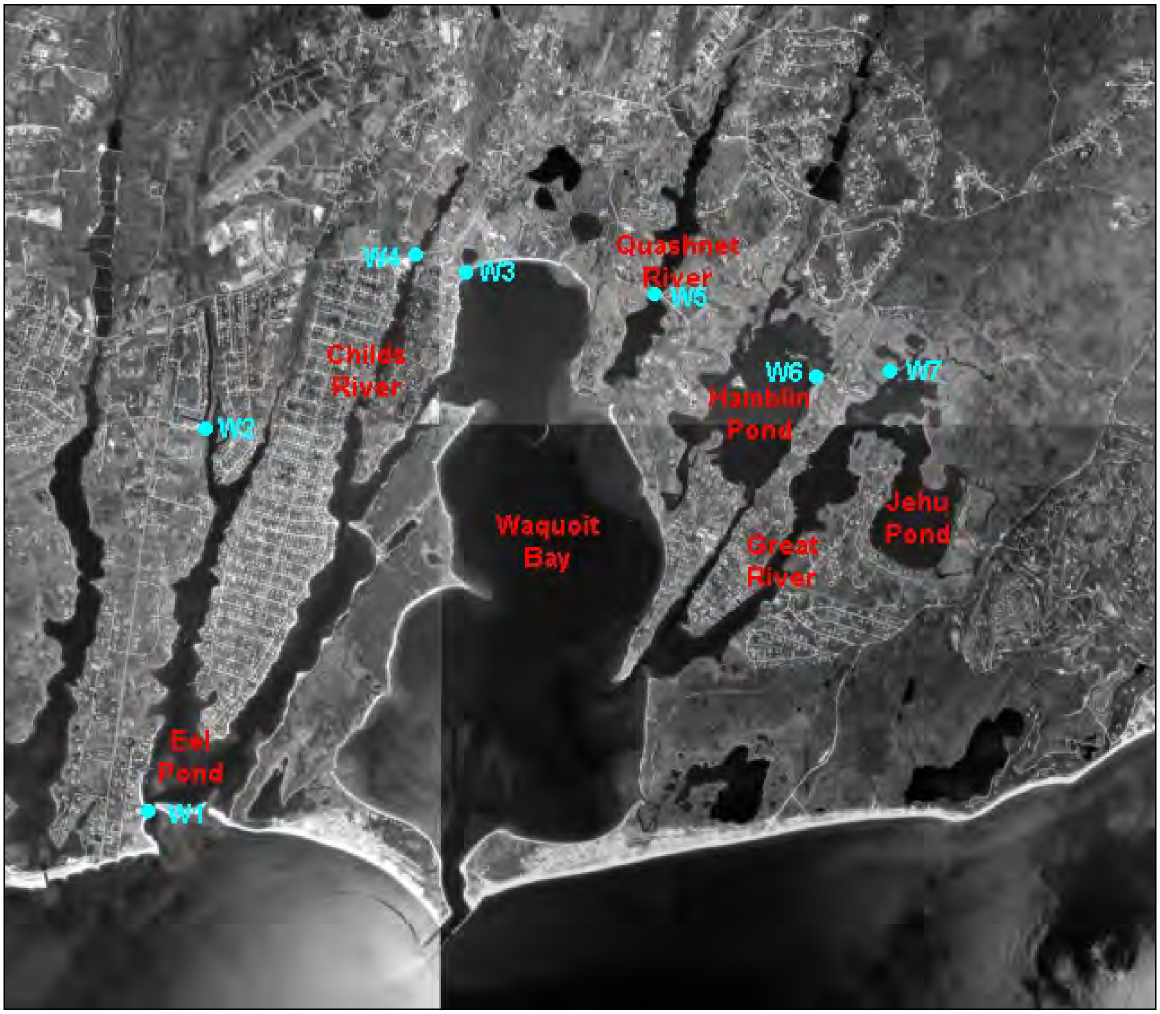


Figure V-5. Waquoit Bay system with tide gage locations labeled as W1-W7.

V.3.1 Bathymetry

Bathymetry, or depth, of Waquoit Bay, Eel Pond, Childs River, Moonakis River, Hamblin Pond, Great River, and Jehu Pond was measured during field surveys in January 2002. 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 x-y position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder and GPS were logged to a laptop computer.

GPS positions and echo sounder measurements were merged to produce data sets consisting of water depth as a function of x-y horizontal position (in Massachusetts Mainland State Plane, 1983). The data were combined with water surface elevations to obtain the vertical elevation of the bottom (z) relative to the NGVD 1929 vertical datum (NGVD29). The resulting xyz files were input to mapping software to calculate depth contours for the system shown in Figure V-6. The bathymetry collected by Applied Coastal was supplemented by existing data from NOAA collected in 1942.

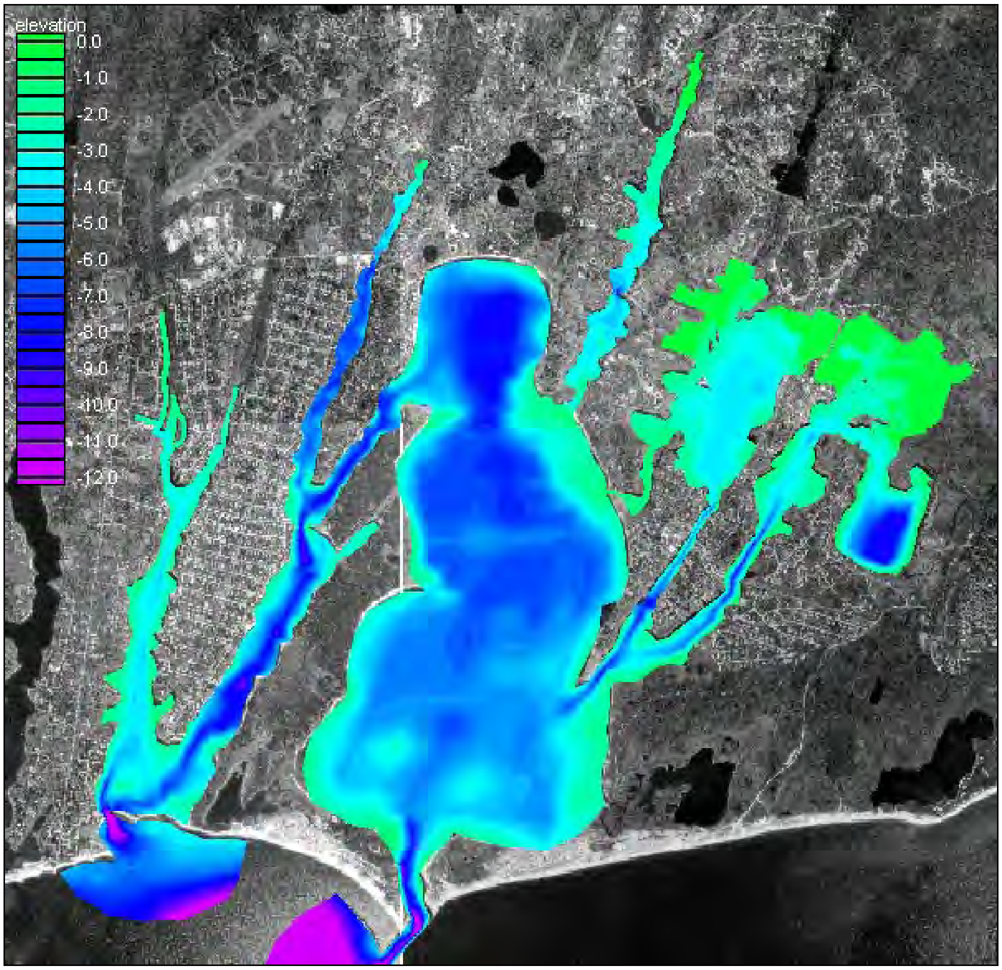


Figure V-6. Depth contour plots of the numerical grid for the Waquoit Bay region at 0.5-foot contour intervals relative to NGVD29.

V.3.2 Water Elevation Measurements and Analysis

Changes in water surface elevation were measured using internal recording tide gages. These tide gages were installed on fixed platforms (such as pier pilings) to record changes in water pressure over time. Variations in the water surface can be due to tides, wind set-up, or other low frequency oscillations of the sea surface. The tide gages were installed in 7 locations in the Waquoit Bay region (Figure V-5) on January 18, 2002 and recovered on February 19, 2002. Data records span at least 29 days to yield an adequate time period for resolving the primary tidal constituents.

The tide gages used for the study consisted of Brancker TG-205, Brancker XR-420 TG, and Global Water WL-15 instruments. Data were set for 10-minute intervals, with each observation resulting from an average of 60 1-second pressure measurements on 10-minute intervals. Each of these instruments use strain gage transducers to sense variations in pressure, with resolution on the order of 1 cm (0.39 inches) head of water. Each gage was calibrated prior to installation to assure accuracy.

Once the data were downloaded from each instrument, the water pressure readings were corrected for variations in atmospheric pressure. Hourly atmospheric readings were recorded by the Global Water WL-15 gage, interpolated to 10-minute intervals, and subtracted from the pressure readings, resulting in water pressure above the instrument. Further, a (constant) water density value of 1025 kg/m^3 was applied to the readings to convert from pressure units (psi) to head units (for example, feet of water above the tide gage). Several of the sensors were surveyed into local benchmarks to provide vertical rectification of the water level; these survey values were used to adjust the water surface to a known vertical datum. The result from each gage is a time series representing the variations in water surface elevation relative to NGVD29. Figures V-7 and V-8 present the water levels at each gage location.

Figure V-7 shows the tidal elevation for the period January 18 through February 19, 2002 at four locations: offshore Menahaunt Beach in Nantucket Sound (Location W1), Waquoit Bay (Location W2), Eel Pond (Location W3), and Childs River (Location W4). Tidal elevations are shown for the next three locations in Figure V-8: Moonakis/Quashnet River (Location W5), Hamblin Pond (Locations W6), and Great River (Locations W7). The curves have a predominant 12.42-hour variation around the lunar semi-diurnal (twice-a-day), or M_2, tidal constituent. Modulation of the lunar and solar tides, results in the spring-neap fortnightly cycle, typically evidence by a gradual increase and decrease in tide range. Water elevations in the Waquoit Bay System are strongly influenced by wind set-up resulting in a lowering of the water surface, clearly seen on February 2. The spring-neap cycle variation is masked by sudden changes in water surface elevation as a result of wind events. The neap (or minimum) tide range was approximately 1.8 feet, occurring January 20. The spring (maximum) tide range was approximately 3 feet, and occurred on January 31.

Analyses of the tide 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.

Harmonic analyses were performed on the time series from each gage location. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 known tidal constituents result from this procedure. Table V-1 presents the amplitudes of the eight largest tidal constituents. The M2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 0.68 feet at the offshore gage. The range of the M2 tide is twice the amplitude, or 1.36 feet. The diurnal tides, K1 and O1, possess amplitudes of approximately 0.25 feet. The N2 (12.66-hour period) semi-diurnal tide, also contributes significantly to the total tide signal with an amplitude of 0.22 feet. The M4 and M6 tides are higher frequency harmonics of the M2 lunar tide (exactly half the period of the M2 for the M_4, and one third of the M_2 for the M_6), results from frictional attenuation of the M_2 tide in shallow water. The M_4 is approximately 20% of the amplitude of the M_2 in the offshore gage (about 0.16 feet). The M_6 amplitude is relatively small throughout the system (less than 0.06 feet). The M_{sf} is a lunarsolar fortnightly constituent with a period of approximately 14 days, and is the result of the periodic conjunction of the sun and moon. The observed astronomical tide is therefore the sum of several individual tidal constituents, each with a particular amplitude and frequency.

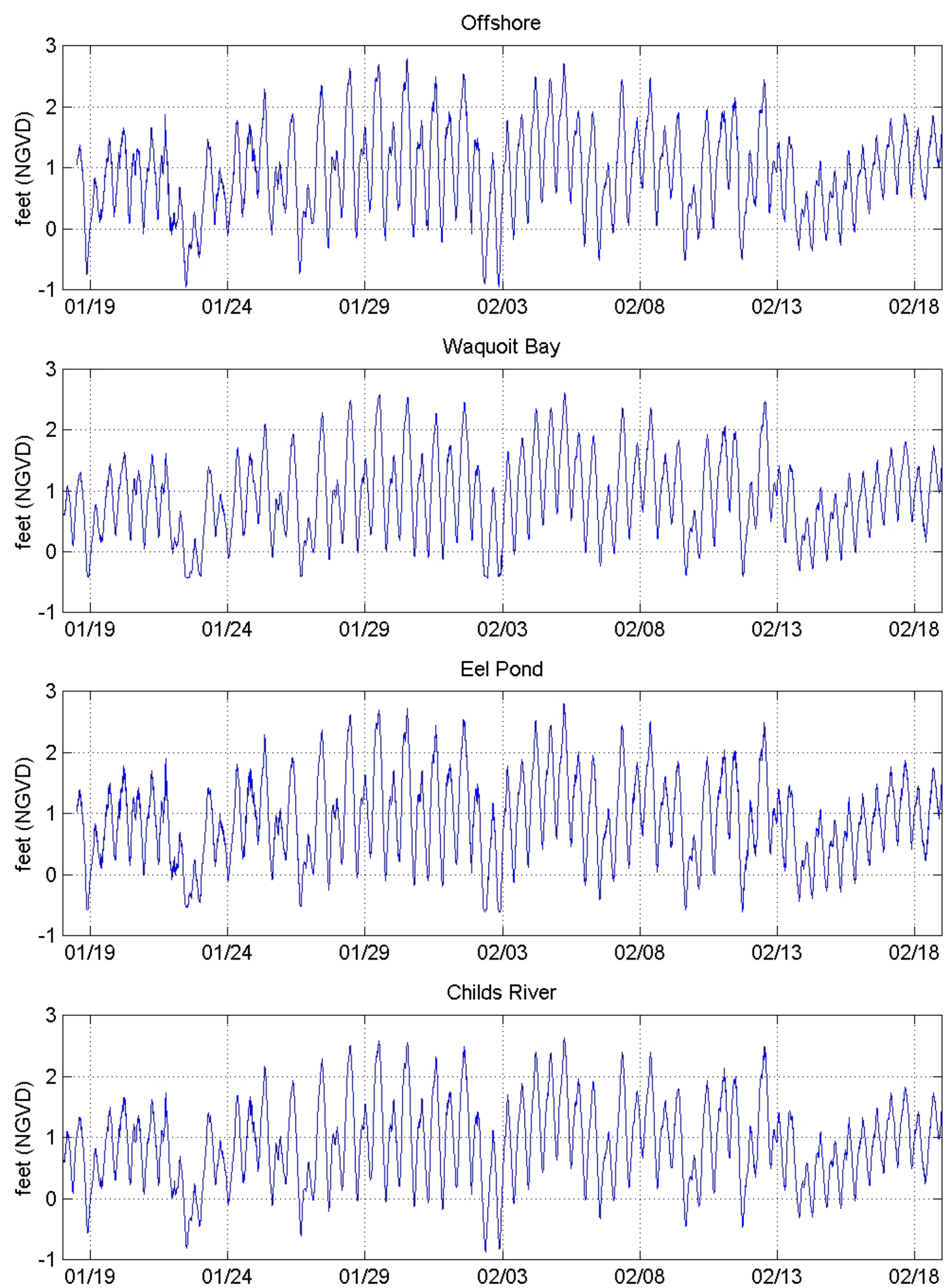


Figure V-7. Tidal elevation observations for offshore Menauhant Beach (location W1), Waquoit Bay (location W2), Eel Pond (location W3), Childs River (location W4).

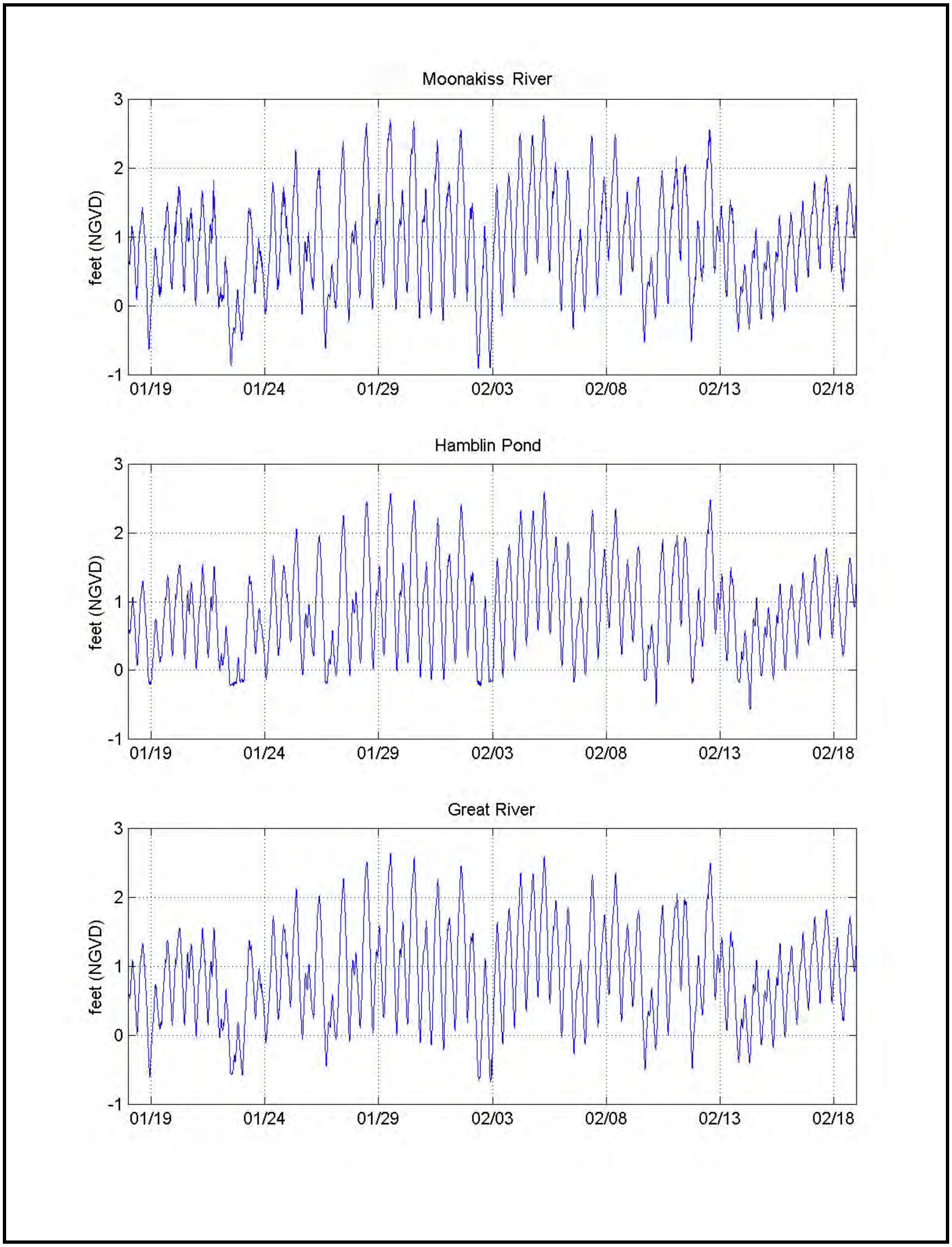


Figure V-8. Tidal elevation observations for Moonakis/Quashnet River (location W5), Hamblin Pond (location W6), and Great River (location W7).

Table V-1. Tidal Constituents, Waquoit Bay System January-February 2002								
	AMPLITUDE (feet)							
	M2	M4	M6	S2	N2	K1	O1	Msf
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Offshore	0.68	0.16	0.06	0.06	0.24	0.25	0.23	0.89
Waquoit Bay	0.64	0.10	0.04	0.05	0.22	0.25	0.23	0.80
Eel Pond	0.65	0.12	0.05	0.05	0.22	0.25	0.23	0.80
Childs River	0.67	0.10	0.06	0.05	0.23	0.27	0.24	0.86
Moonakis River	0.65	0.09	0.05	0.05	0.22	0.25	0.23	0.81
Hamblin Pond	0.63	0.07	0.05	0.05	0.21	0.25	0.23	0.78
Great River	0.67	0.13	0.06	0.06	0.23	0.27	0.23	0.89

Table V-1 also shows how the constituents vary as the tide propagates into the estuaries. The most significant reduction in the M_2 amplitude occurs between the Nantucket Sound (offshore) gage and the upper reaches of Hamblin Pond. Usually, a portion of the energy lost from the M_2 tide is transferred to higher harmonics, and is observed as an increase in the amplitude of the M_4 and M_6 constituents over the length of the estuary. However, in the Waquoit Bay system M_2, M_4 and M_6 are all clearly smaller than the amplitudes at the inlet. This is likely because the tidal attenuation through the two inlet channels is much stronger than the damping from frictional drag through tidal channels.

Table V-2 presents the phase delay of the M_2 tide at all tide gage locations compared to the offshore gage in Nantucket Sound. Phase delay is another indication of tidal damping, and results with a later high tide at inland locations (Figure V-9). The greater the frictional effects, the longer the delay between locations. The delay in Eel Pond (23.3 minutes) is the smallest, as a result of its proximity to the offshore gage location. In general, the delays increase with increasing distance from the offshore gage. The most significant damping is seen in Hamblin Pond and Great River with delays of 75.0 and 66.5 minutes, respectively. The larger delays in these embayments are a combination of the increased distance away from the offshore gage and the flooding of tidal flats around Hamblin Pond and Great River.

Table V-2. M_2 Tidal Attenuation, Waquoit Bay, January-February 2002 (Delay in minutes relative to Nantucket Sound)	
Location	Delay (minutes)
Offshore (Nantucket Sound)	--
Waquoit Bay	48.01
Eel Pond	23.31
Childs River	34.43
Moonakis River	35.29
Hamblin Pond	74.99
Great River	66.54

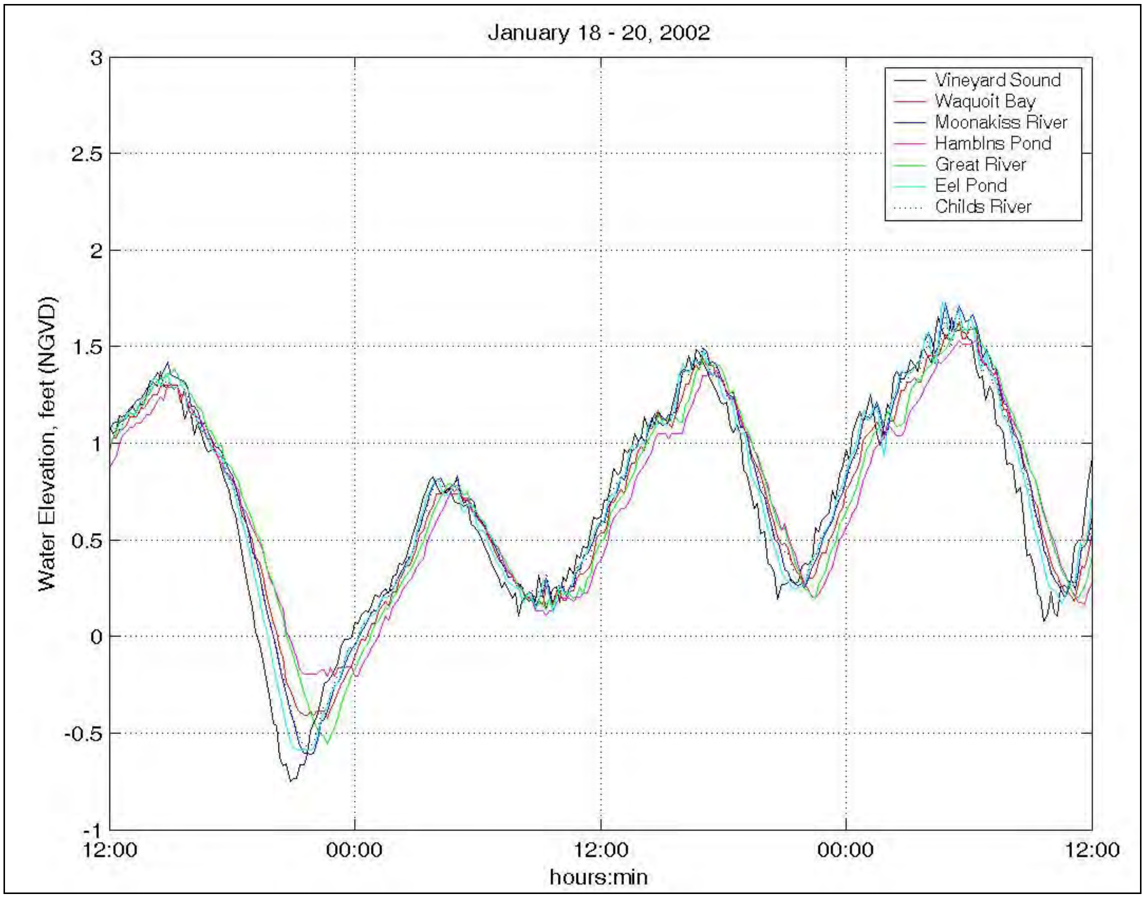


Figure V-9. Comparison of water surface elevation observations for Nantucket Sound (offshore), and six locations within the Waquoit Bay system. Damping effects are seen as a decrease in the tidal amplitude, as well as a lag in the time of high and low tides from Nantucket Sound.

In addition to the tidal analysis, the data were further evaluated to determine the importance of tidal versus non-tidal processes to changes in water surface elevation. These other processes include wind forcing (set-up or set-down) within the estuary, as well as sub-tidal oscillations of the sea surface. Nantucket Sound is a relatively shallow semi-enclosed basin; therefore, the water surface responds readily to wind-forcing. Variations in water surface elevation can also be affected by freshwater discharge into the system, if these volumes are relatively large. This analysis calculated the energy (or variance) of the original water elevation time series, and compared these energy values to that of the purely tidal signal (re-created by summing the contributions from the 23 known harmonic constituents). Subtracting the tidal signal from the original elevation time series resulted with the non-tidal, or residual, portion of the water elevation changes. The energy of this non-tidal signal is compared to the tidal signal, and yields a quantitative measure of how important these non-tidal physical processes can be to hydrodynamic circulation within the estuary. The results of this analysis for the Waquoit Bay system are presented in Table V-3.

Table V-3. Percentages of Tidal versus Non-Tidal Energy, Waquoit, 2002				
	Total Variance ($ft^2 \cdot sec$)	Total (%)	Tidal (%)	Non-tidal (%)
Offshore	0.51	100	72.3	27.7
Waquoit Bay	0.45	100	70.7	29.3
Eel Pond	0.50	100	71.3	28.7
Childs River	0.46	100	70.5	29.5
Moonakis River	0.50	100	70.9	29.1
Hamblin Pond	0.42	100	71.3	28.7
Great River	0.46	100	70.8	29.2

During the period of tidal analysis, the variability analysis showed that less than three-quarters of the changes in water surface elevation in Nantucket Sound and the Waquoit Bay system were due to tidal processes. More than one-quarter of the energy in Nantucket Sound water elevations was the result of non-tidal processes. The percentage of non-tidal energy increases and the percentage of tidal energy decreases as the residual signal propagates into the system. As mentioned previously, this is in part due to tidal damping through the inlets. The observed variations from strictly tidal processes indicates a variety of atmospheric forcing including local winds (short-term fluctuations in the residual tide), as well as wind set-up caused by major wind events further offshore and direct changes in regional atmospheric pressure. Results of the tidal elevation analysis indicate that hydrodynamic circulation in each of the embayments is dependent primarily upon tidal processes, with a secondary contribution from atmospheric forcing.

V.4. HYDRODYNAMIC MODELING

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

V.4.1 Model Theory

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

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

The time dependence of the governing equations is incorporated within the solution technique needed to solve the set of simultaneous equations. This technique is implicit; therefore, unconditionally stable. Once the equations are solved, corrections to the initial estimate of velocity and water elevation are employed, and the equations are re-solved until the convergence criterion is met. For this modeling analysis, the convergence criterion was set at 0.01 feet for water elevation change.

V.4.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using the shorelines from 1994 digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of Waquoit Bay and Eel Pond based on the tide gauge data collected near the Waquoit Yacht Club, at the entrance to Eel Pond. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several (20+) model calibration simulations for each system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.4.2.1 Grid Generation

The grid generation process was aided by the use of the SMS package. A 1994 digital aerial orthophoto and the bathymetry survey data were imported to SMS, and a finite element grid was generated to represent the embayments and waterways within the estuary. The aerial photograph was used to determine the land boundary of the system, as well as determine the surface coverage of salt marshes. The bathymetry data was interpolated to the developed finite element mesh of the system. The completed grid consists of 8708 nodes, which describe 2,573 total 2-dimensional (depth averaged) quadratic elements. The maximum nodal depth is -30.55 ft (NGVD 29), along the offshore boundary to Waquoit Bay, and the maximum modeled marsh plain elevation is 2.0 ft. In the model grid, a typical marsh plain elevation of +1.5 ft (NGVD 29) was used, based on spot surveys across the marsh. The model marsh topography was varied to provide a monotonically sloping surface, in order to enhance the stability of the hydrodynamic model. The completed grid mesh of the Waquoit Bay system is shown in Figure V-10.

The finite element grid for each system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties of the Waquoit Bay system. Areas of marsh were included in the model because they represent a large portion of the total area within Hamblin Pond and Great River, and have a significant effect on hydrodynamics within those systems. Fine resolution was required to simulate the numerous channel constrictions that significantly impact the estuarine hydrodynamics, such as the bridge abutments, as well as the marsh creeks. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary.

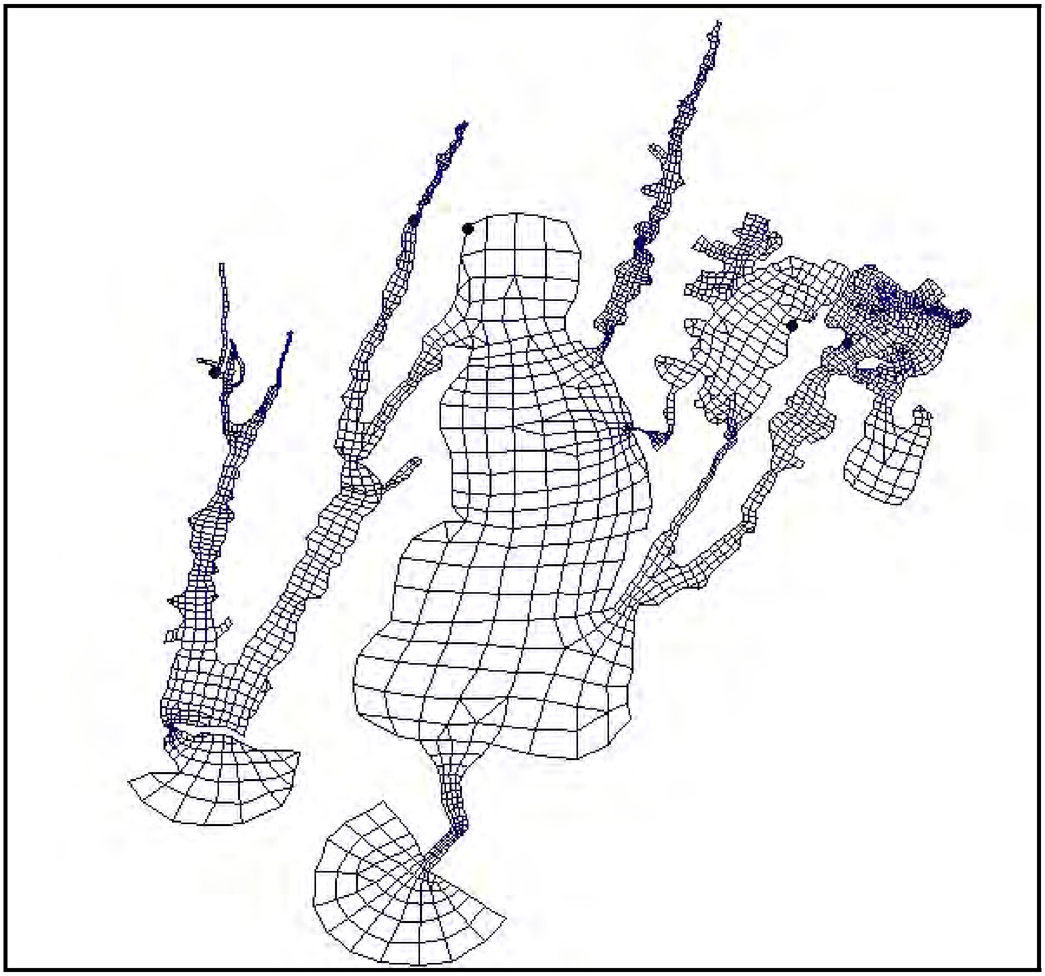


Figure V-10. Plot of hydrodynamic model grid mesh for the Waquoit Bay system.

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability of the system. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in marsh creeks and channels was designed to provide a more detailed analysis in these regions of rapidly varying flow. Widely spaced nodes were often employed in areas where flow patterns are not likely to change dramatically, such as in Waquoit Bay and the offshore boundaries. 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

Two types of boundary conditions were employed for the RMA-2 model of the Waquoit Bay system: 1) "slip" boundaries, and 2) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Tidal boundary conditions were specified at the inlets to Waquoit Bay and Eel 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 in this system. For the boundaries a dynamic (time-varying) water surface elevation condition was specified every model time step (10 minutes) to represent the tidal forcing.

V.4.2.3 Calibration

After developing the finite element grids, and specifying boundary conditions, the model for the Waquoit Bay system was calibrated. The calibration procedure ensures that the model predicts accurately what was observed in nature during the field measurement program. Numerous model simulations are required (typically 20+) for an estuary model, specifying a range of friction and eddy viscosity coefficients, to calibrate the model.

Calibration of the hydrodynamic model requires a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (i.e., from the TDR deployments). Initially, the model was calibrated to obtain visual agreement between modeled and measured tides. Once visual agreement was achieved, an approximate five-day period (10 tide cycles) was modeled to calibrate the model based on dominant tidal constituents discussed in Section V.3.2. The five-day period was extracted from a longer simulation to avoid effects of model spin-up, and to focus on average tidal conditions. Modeled tides for the calibration time period were evaluated for time (phase) lag and height damping of dominant tidal constituents

The calibration was performed for a five-day period beginning January 27, 2002 at 1000 EDT. This representative time period included the spring tide range of conditions, where the tide range and tidal currents are greatest. The Waquoit Bay system exhibits relatively small (<2.5 feet) forcing tides. For estuarine systems of this type, the influence of non-tidal forcing (wind, atmospheric pressure, etc.) often can strongly influence tidal circulation (see the longer-term fluctuations in water levels shown in Figures V-7 and V-8). Due to the high degree of non-tidal forcing observed in this system, a time-period with little non-tidal response was selected to most accurately simulate typical tide conditions. In addition, it is also important to realize that the diurnal inequality (the difference in the two high tide heights observed in a single 24-hour period) is significantly larger than the difference between spring and neap tide heights. The 5-day simulation period is representative of typical tide conditions within the Waquoit Bay system.

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

V.4.2.3.1 Friction Coefficients

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

To improve model accuracy, friction coefficients were varied throughout the model domain. First, the Manning's coefficients were matched to bottom type. For example, lower friction coefficients were specified for the smooth sandy channels found in Eel Pond, versus the rock lined channel in the inlet to Hamblin Pond, which provides greater flow resistance. Final model calibration runs incorporated various specific values for Manning's friction coefficients, depending upon flow damping characteristics of separate regions within each estuary. Manning's values for different bottom types were initially selected based ranges provided by the Civil Engineering Reference Manual (Lindeburg, 1992), and values were incrementally changed when necessary to obtain a close match between measured and modeled tides. Final calibrated friction coefficients are summarized in the Table V-4. The extents of each material type are shown in Figure V-11.

Table V-4. Manning's Roughness coefficients used in simulations of modeled embayments. These embayment delineations correspond to the material type areas shown in Figure V-8.

System Embayment	Bottom Friction
Waquoit Bay	0.025
Eel Pond	0.025
Moonakis River	0.020
Seapit River	0.025
Marsh Plains	0.070
Hamblin Pond	0.025
Great River	0.025
Child's River	0.025
Bridges	0.050
Rock lined channel	0.040

V.4.2.3.2 Turbulent Exchange Coefficients

Turbulent exchange coefficients approximate energy losses due to internal friction between fluid particles. The significance of turbulent energy losses increases where flow is swifter, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). In most cases, the modeled systems were relatively insensitive to turbulent exchange coefficients because there were no regions of strong turbulent flow. Typically, model turbulence coefficients were set between 80 and 100 lb-sec/ft^2.

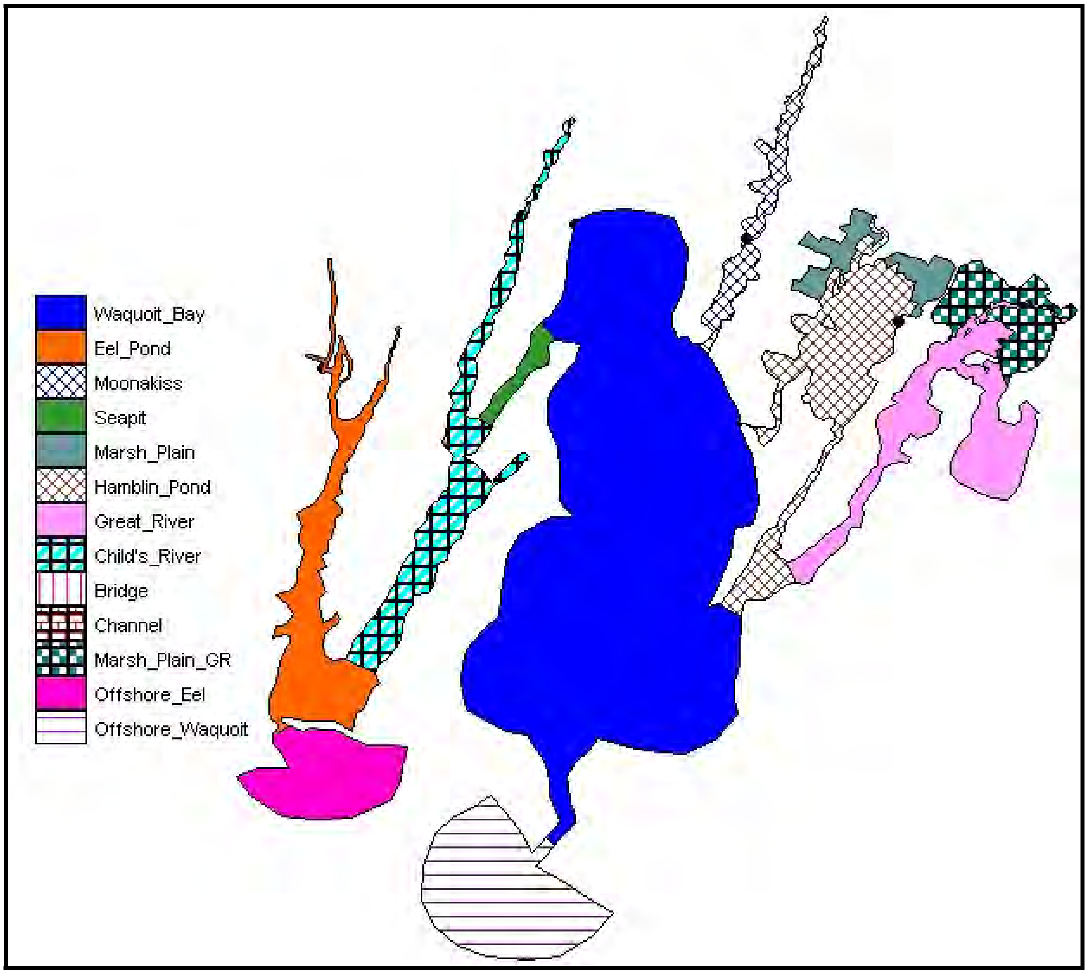


Figure V-11. Hydrodynamic model grid material properties utilized by the RMA-2 model. Color patterns designate the different model material types used to vary model calibration parameters and compute flushing rates.

V.4.2.3.3 Marsh Porosity Processes

Modeled hydrodynamics were complicated by wetting/drying cycles on the marsh plain included in the model within Hamblin Pond and Great River. 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-12 through V-17 illustrate the five-day calibration simulation, for Eel Pond, Childs River, Waquoit Bay, Moonakis River, Hamblin Pond, and Great River. Modeled (solid line) and measured (dotted line) tides are illustrated at each model location with a corresponding TDR.

Although visual calibration achieved reasonable modeled tidal hydrodynamics, further tidal constituent calibration was required to quantify the accuracy of the models. Calibration of M_2 was the highest priority since M_2 accounted for a majority of the forcing tide energy in the modeled systems. Due to the duration of the model runs, four dominant tidal constituents were selected for constituent comparison: K_1, M_2, M_4, and M_6. Measured tidal constituent heights (H) and time lags (ϕ_{lag}) shown in Table V-5 for the calibration period differ from those in Table V-2 because constituents were computed for only the five-day section of the longer time series represented in Table V-2. Table V-5 compares tidal constituent height and phase for modeled and measured tides at the TDR locations.

The constituent calibration resulted in excellent agreement between modeled and measured tides. The largest errors associated with tidal constituent amplitude were on the order of 0.04 ft in Moonakis River, which is of the same order of the accuracy of the tide gages (0.032 ft). Time lag errors were typically less than the time increment resolved by the model (0.10 hours or 10 minutes), indicating good agreement between the model and data. Moonakis/Quashnet River had the largest time lag errors (approximately 24 minutes). The small tide range combined with the relative importance of wind during the tidal measurement period likely is responsible for errors in the modeled tidal phase.

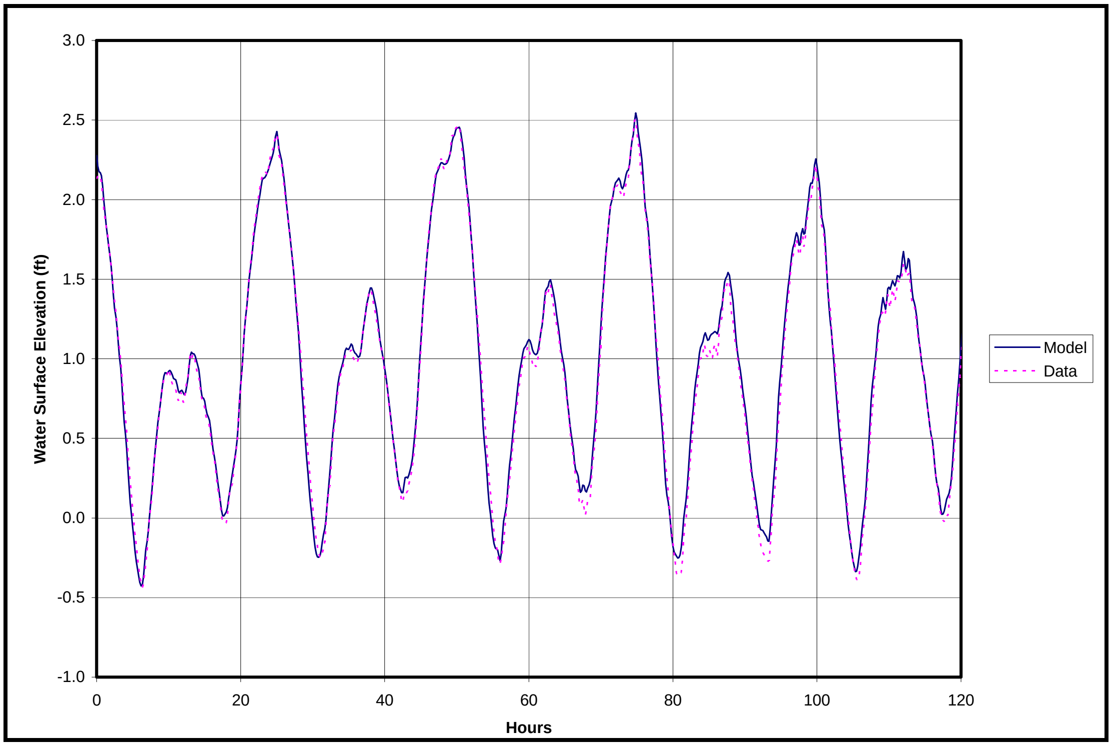


Figure V-12. Comparison of model output and measured tides for the TDR location in Eel Pond.

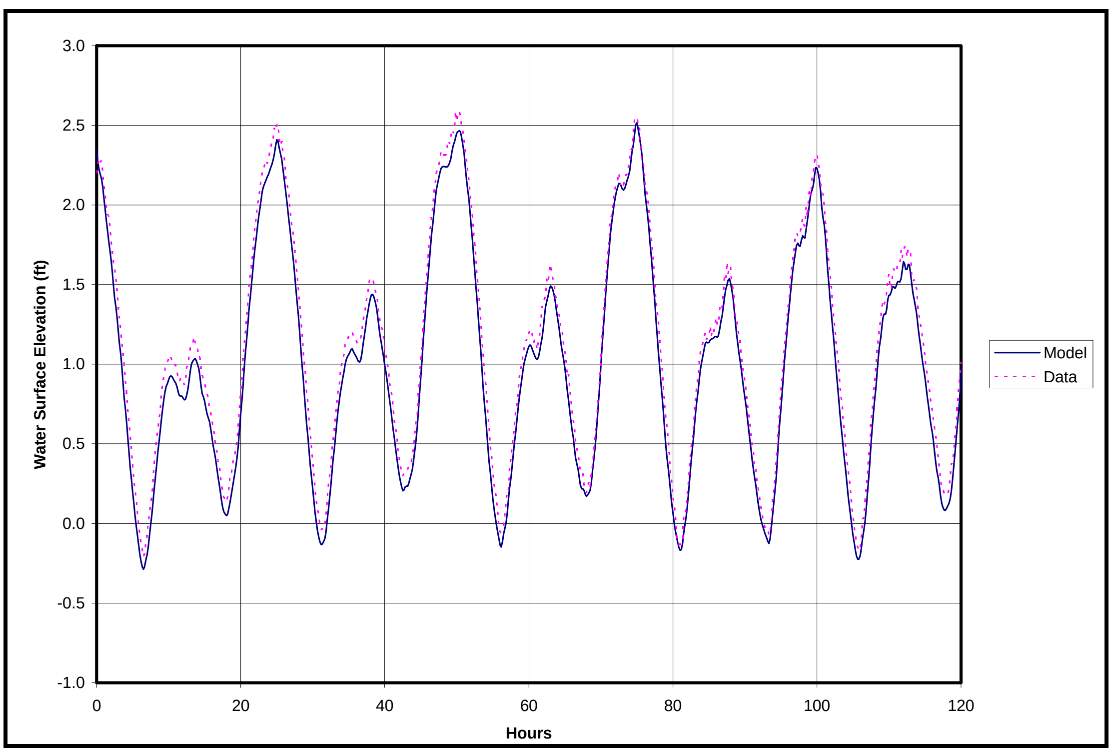


Figure V-13. Comparison of model output and measured tides for the TDR location in Childs River.

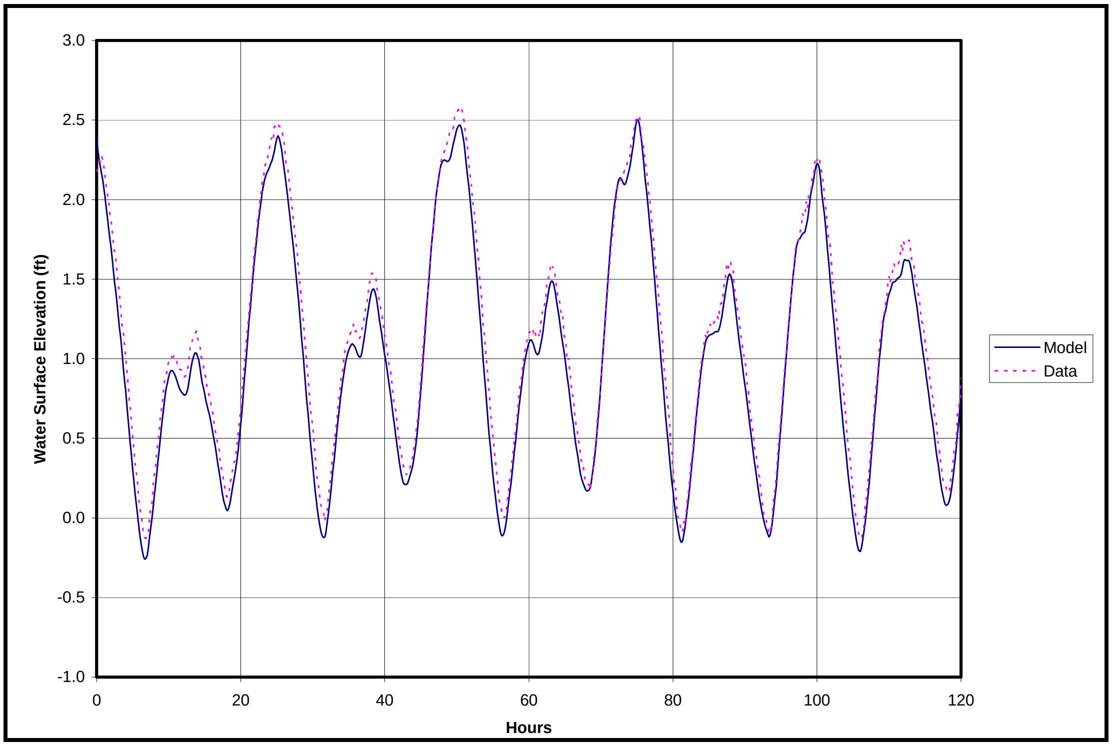


Figure V-14. Comparison of model output and measured tides for the TDR location in Waquoit Bay.

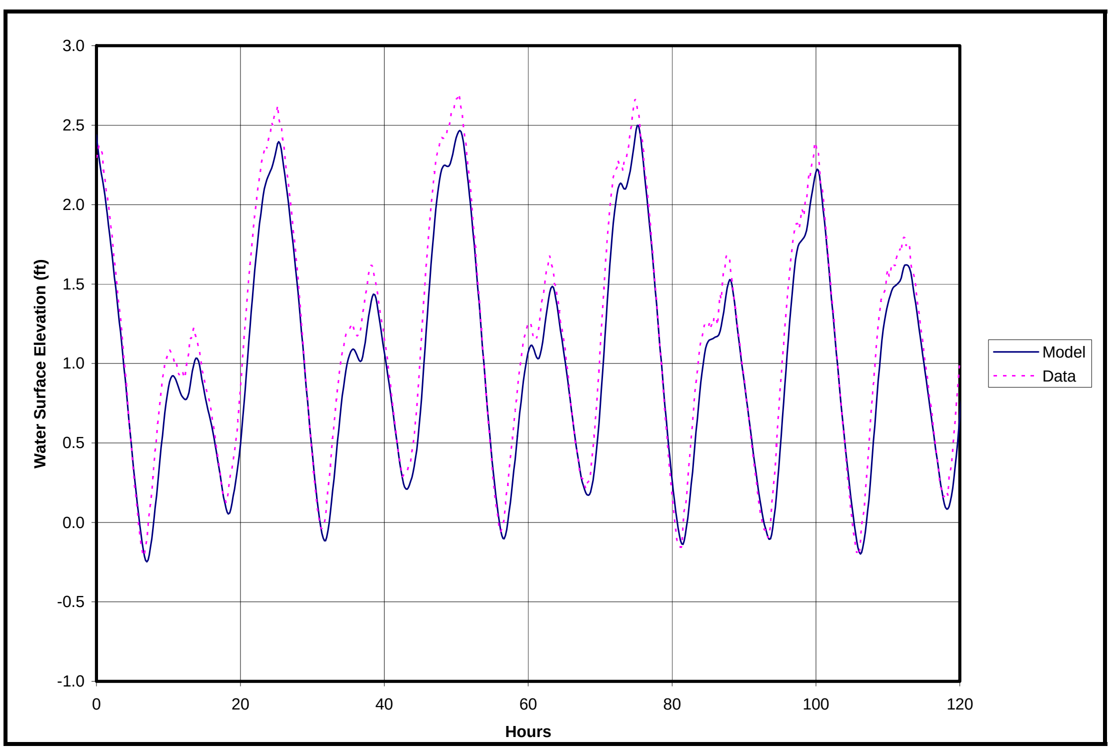


Figure V-15. Comparison of model output and measured tides for the TDR location in Moonakis/Quashnet River.

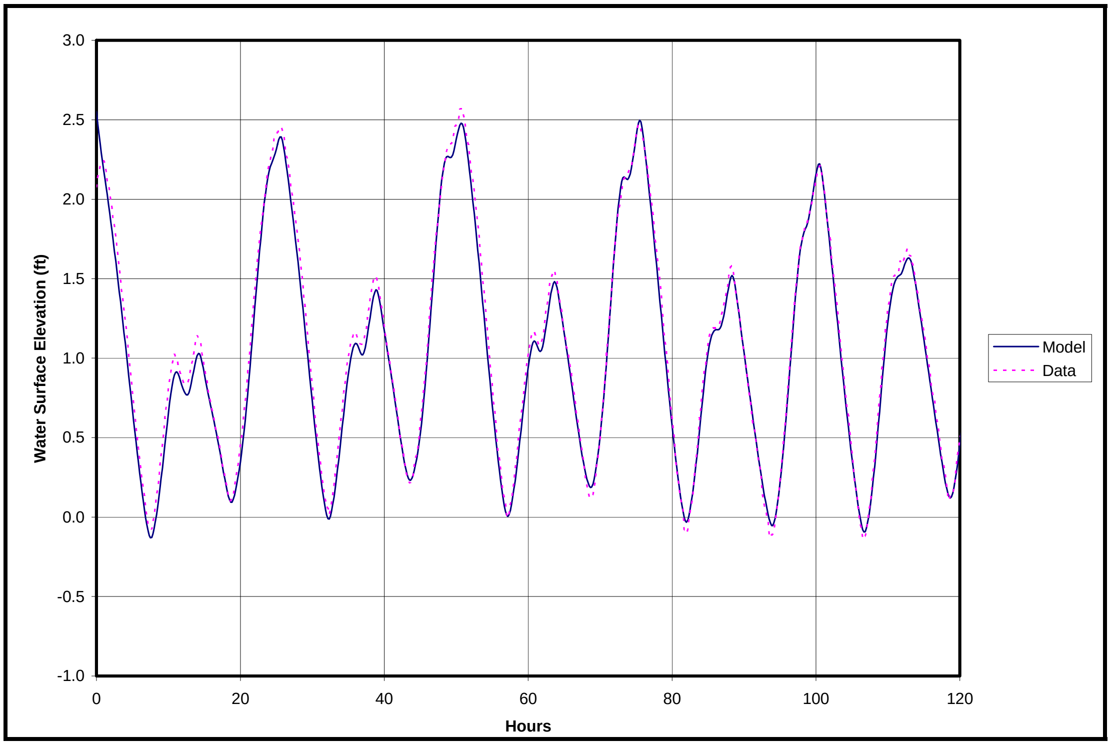


Figure V-16. Comparison of model output and measured tides for the TDR location in Hamblin Pond.

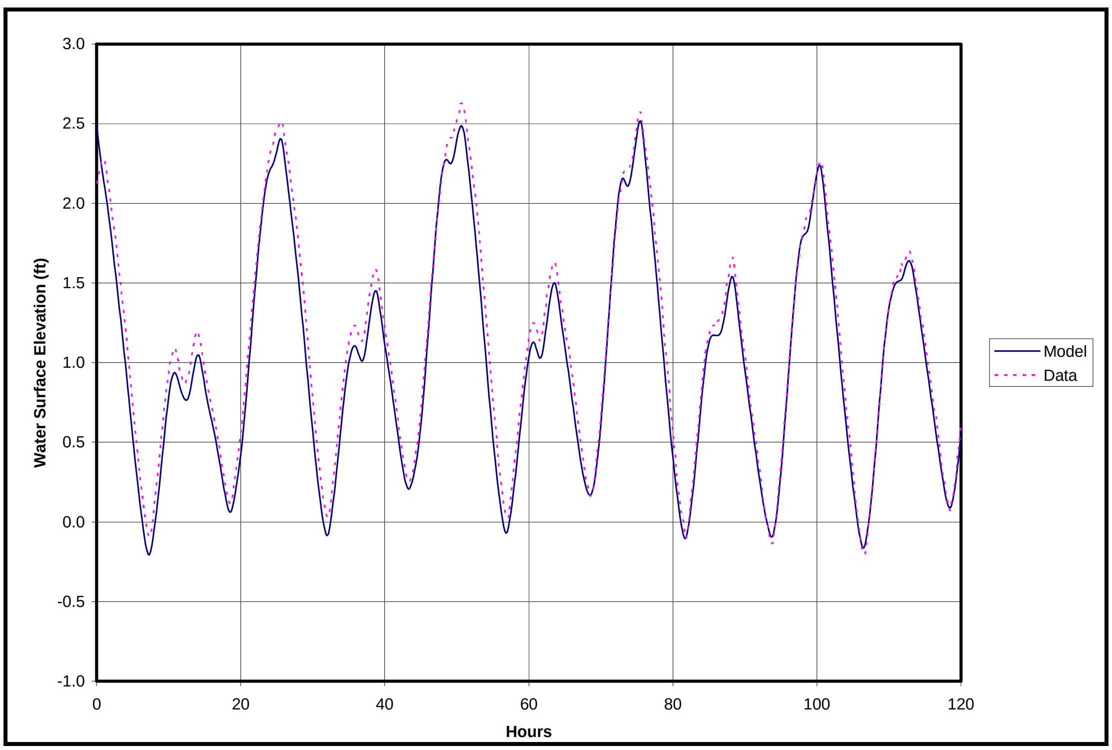


Figure V-17. Comparison of model output and measured tides for the TDR location in Great River.

Table V-5. Tidal constituents for measured water level data and calibrated model output for northern embayments.						
Model calibration run						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Eel Pond	0.91	0.21	0.08	0.45	49.9	-92.7
Childs River	0.89	0.16	0.08	0.45	57.8	-74.6
Waquoit Bay	0.88	0.15	0.08	0.45	62.7	-61.8
Moonakis River	0.88	0.14	0.08	0.45	67.9	-47.1
Hamblin Pond	0.85	0.09	0.06	0.45	81.0	-8.5
Great River	0.87	0.12	0.08	0.45	73.8	-31.1
Measured tide during calibration period						
Location	Constituent Amplitude (ft)				Phase (deg)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Eel Pond	0.91	0.21	0.1	0.47	51.6	-83.8
Childs River	0.87	0.17	0.08	0.44	56.5	-75.5
Waquoit Bay	0.87	0.15	0.07	0.44	64.1	-60.8
Moonakis River	0.92	0.18	0.09	0.47	59.1	-70.2
Hamblin Pond	0.86	0.12	0.07	0.45	78.8	-19.5
Great River	0.88	0.14	0.07	0.45	74.2	-35.5
Error						
Location	Error Amplitude (ft)				Phase error (min)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Eel Pond	0.00	0.00	0.02	0.02	3.5	9.2
Childs River	-0.02	0.01	0.00	-0.01	-2.7	-0.9
Waquoit Bay	-0.01	0.00	-0.01	-0.01	3.0	1.0
Moonakis River	0.04	0.04	0.01	0.02	-18.1	-23.9
Hamblin Pond	0.01	0.03	0.01	0.00	-4.5	-11.4
Great River	0.01	0.02	-0.01	0.00	0.7	-4.5

V.4.2.4 Model Circulation Characteristics

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

Examining the results from the model run of the Waquoit Bay shows flood velocities in the channels are slightly larger than velocities during maximum ebb. The maximum velocities occur in the entrance channels to Waquoit Bay and Eel Pond. At the entrance channel to Waquoit Bay, maximum depth-averaged flood velocities in the model are approximately 5.8 feet/sec, while maximum ebb velocities are about 6.1 feet/sec. In the inlet channel to Eel Pond, maximum depth averaged flood velocities are approximately 2.9 feet/sec, and maximum ebb velocities are 3.6 feet/sec. A close-up of the model output is presented in Figure V-18, which shows contours of velocity magnitude, along with velocity vectors which indicate the direction of flow, for a single model time-step, at the portion of the tide where maximum flood velocities occur.

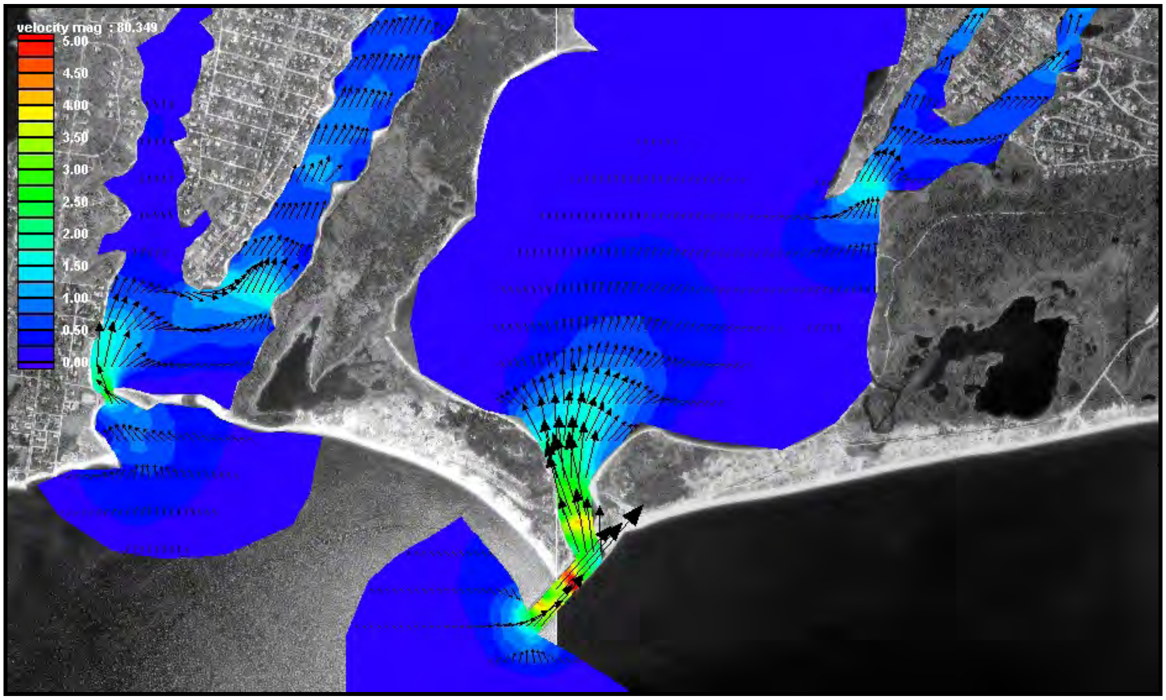


Figure V-18. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate velocity magnitude, and vectors indicate the direction of flow.

In addition to depth averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. For the flushing analysis in the next section, flow rates where computed across six separate transects in the Waquoit Bay system: at entrance to Waquoit Bay, the entrance to Eel Pond, the channel going to Hamblin Pond and Great River, the entrance to Moonakis/Quashnet River, a transect across Childs River, and a transect across Eel Pond near the interest of Eel Pond and Childs River. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-19. Although the highest velocities at a single point occur during the ebb portion of the tidal cycle, maximum flow rates across both of the inlets occur during flood tides in this system. This is an indication that the Waquoit Bay system is flood dominant, and likely a sediment sink (a system that accumulates sediment). During spring tides, the maximum flood flow rates through the Waquoit Bay inlet reach 8,000 ft^3/sec and the maximum ebb flow rates are only about 6,000 ft^3/sec. The flood or ebb dominance of the Eel Pond inlet is not clear, where both the spring flood and ebb flows slightly exceed 4,000 ft^3/sec.

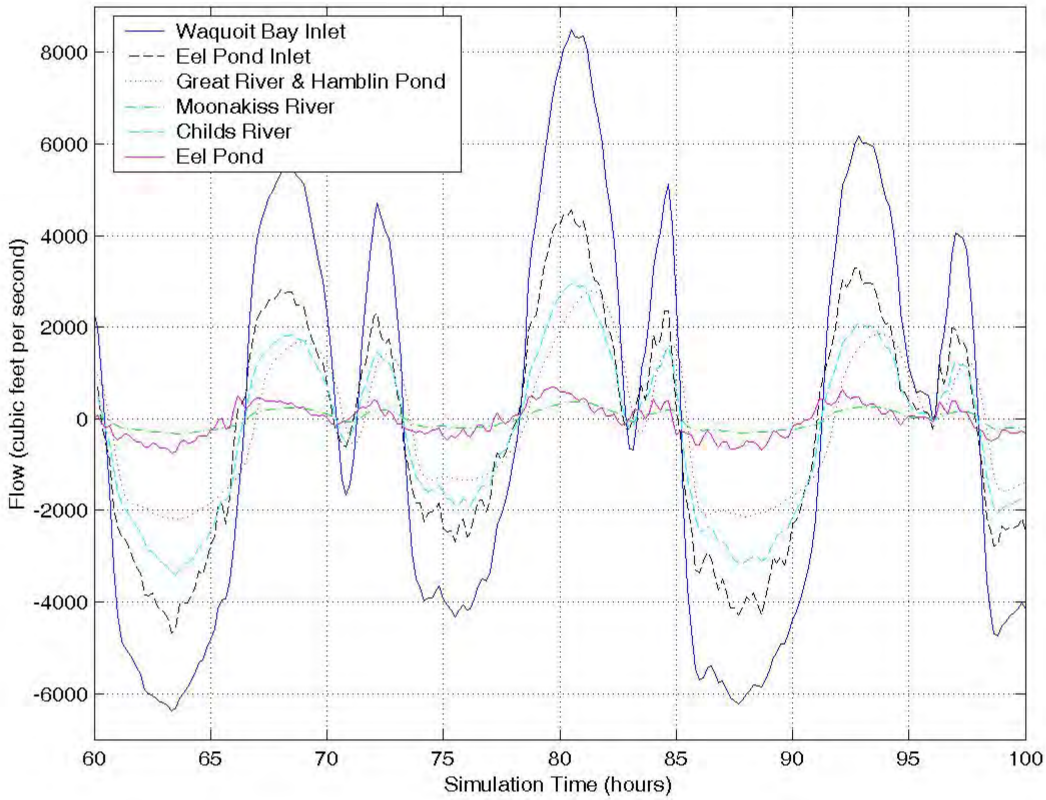


Figure V-19. Time variation of computed flow rates for six transects in the Waquoit Bay system. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Plotted time period represents three tide cycles (12.42 h cycle). Positive flow indicated flooding tide, while negative flow indicates ebbing tide.

V.5. FLUSHING CHARACTERISTICS

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

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

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

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

In addition to system residence times, a second residence, the **local residence time**, was defined as the average time required for a water parcel to migrate from a location within a sub-embayment to a point outside the sub-embayment. Using Moonakis River as an example, the **system residence time** is the average time required for water to migrate from Moonakis River, through Waquoit Bay, and into Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Moonakis River to just Waquoit Bay (not all the way out of the system). Local residence times for each sub-embayment are computed as:

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

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

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

Since the calibrated RMA-2 model simulated accurate two-dimensional hydrodynamics in the system, model results were used to compute residence times. Residence times were computed for the entire estuary, as well the main sub-embayments within the system. In addition, **system** and **local residence times** were computed to indicate the range of conditions possible for the system. Residence times were calculated as the volume of water (based on the mean volumes computed for the simulation period) in the entire system divided by the average volume of water exchanged with each sub-embayment over a flood tidal cycle (tidal prism).

Units then were converted to days. The volume of the entire estuary was computed as cubic feet.

Residence times were averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and are listed in Table V-6. The modeled time period used to compute the flushing rates was different from the modeled calibration period, and included the transition from neap to spring tide conditions. Model divisions used to define the system sub-embayments (Figure V-20) include 1) the entire Waquoit Bay system, 2) Eel Pond (west branch), 3) Great River 4) Jehu Pond, 5) Hamblin Pond, 6) Moonakis/Quashnet River, and 7) Childs River. The model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume. Since the 7.25-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

Table V-6. Embayment mean volumes and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
System	380,320,000	97,247,064
Eel Pond (West Branch)	13,080,500	3,339,810
Great River	30,244,000	9,436,062
Jehu Pond	13,011,000	2,892,063
Hamblin Pond	29,237,000	9,050,124
Moonakis/Quashnet River	8,840,800	2,800,916
Childs River	9,821,500	1,481,343

Table V-7. Computed System and Local residence times for embayments in the Waquoit Bay system.

Embayment	System Residence Time (days)	Local Residence Time (days)
System	2.02	
Eel Pond (West Branch)	58.93	2.03
Great River	20.86	1.66
Jehu Pond	68.05	2.33
Hamblin Pond	21.75	1.67
Moonakis/Quashnet River	70.27	1.63
Childs River	132.86	3.43

The computed flushing rates for the Waquoit system show that system takes approximately 2.0 days for the volume of the system to be exchanged. This residence time is relatively large, due primarily to a large embayment volume relative to its mean tidal prism. In general, the small tide range in western Nantucket Sound provides limited forcing for tidal exchange. Smaller sub-embayments have large system residence times; however, these residence times would only be appropriate for an indication of estuarine health if the receiving waters were of low quality. For example, Moonakis/Quashnet River has a system residence time of approximately 70 days. The local residence time for this embayment is only about 1.6 days, indicating that the Moonakis/Quashnet River exchanges water quite readily with Waquoit Bay. The long system residence time indicates that much of the water entering the Moonakis/Quashnet River resided in Waquoit Bay during the previous tidal cycle.

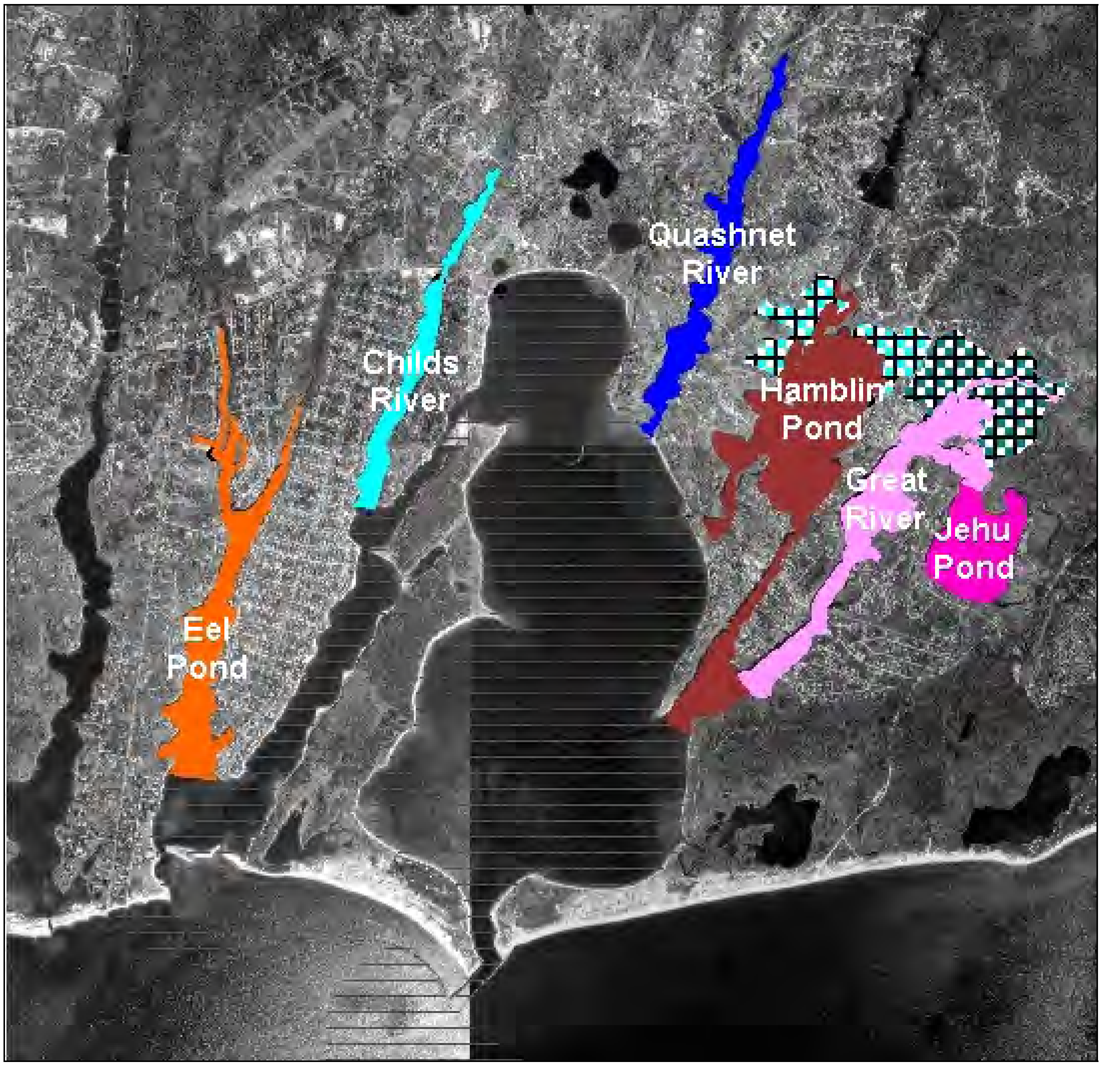


Figure V-20. Basins used to compute residence times for the Waquoit Bay system.

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

VI. WATER QUALITY MODELING

As was mentioned previously in Chapter V, the Hamblin Pond/Jehu Pond and Quashnet River sub-systems of Waquoit Bay were a part of the larger hydrodynamic model of the Waquoit Bay estuarine system. For the water quality modeling portion of this study, the Hamblin Pond/Jehu Pond and Quashnet River Sub-systems where modeled as separate systems, using hydrodynamic input developed from the larger Waquoit Bay model. The primary reason for modeling only the sub-systems is that that watershed loading analysis had only been performed for the Waquoit Bay sub-systems that are within the Town of Mashpee. A future MEP analysis will encompass the entire Waquoit Bay system.

VI.1 DATA SOURCES FOR THE MODEL

Several different data types and calculations are required to support the water quality modeling effort. 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 two embayment systems (Hamblin Pond/Jehu Pond and Quashnet River) modeled in the Waquoit Bay estuary. 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 grids used for the hydrodynamic models were also the computational grid used for the water quality models. The period of hydrodynamic output for the water quality model calibration was a 12-tidal cycle period in winter 2002 that includes the fortnightly variation between spring and neap tide ranges. For each modeled scenario (e.g., present conditions, buildout) the model was run for a 30-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 Waquoit Bay sub-embayments, consisting of the background concentrations of total nitrogen in the waters entering from Nantucket Sound and the main basin of Waquoit Bay. This load is represented as a constant concentration along the seaward boundary of the model grid.

VI.1.3 Measured Nitrogen Concentrations in the 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 of the Hamblin Pond/Jehu Pond and Quashnet River 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

Table VI-1. Measured and modeled Nitrogen concentrations for the Waquoit Bay system, at stations shown in Figure VI-2, and used in the model calibration plots. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of all total nitrogen data.

Sub-Embayment	monitoring station	2001 mean	2002 mean	2003 mean	data mean	s.d. all data	N	model min	model average	model max
Jehu Pond - JHP	WB 1	0.701	0.581	0.576	0.590	0.065	11	0.577	0.595	0.614
Upper Great River - GRu	WB 2	0.763	0.611	0.558	0.606	0.110	12	0.454	0.557	0.614
Great/Little River - GRl	WB 3	0.774	0.576	0.505	0.569	0.122	12	0.395	0.453	0.570
Hamblin Pond - HPu	WB 4	-	0.585	0.460	0.539	0.086	11	0.498	0.529	0.551
FW Red Brook - RBfw	WB 5	0.662	0.645	0.000	0.647	0.026	7	-	-	-
Hamblin Pond Drain - HPcut	WB 10	-	0.551	0.570	0.559	0.066	10	0.477	0.512	0.612
Seapit River	WB 11	0.484	0.501	0.504	0.500	0.049	12	-	-	-
Upper Waquoit Bay - WBu	WB 12	0.576	0.482	0.447	0.478	0.078	12	-	-	-
Lower Waquoit Bay - WBl	WB 13	0.497	0.392	0.376	0.395	0.072	12	-	-	-
FW Quashnet River - QRfw	WB 6	0.734	0.493	0.471	0.503	0.105	25	-	-	-
Upper Quashnet River - QRu	WB 7	1.587	0.674	0.892	0.830	0.444	10	0.736	0.787	0.842
Mid Quashnet River - QRm	WB 8	0.667	0.830	0.668	0.771	0.279	11	0.683	0.773	0.839
Lower Quashnet River - QRl	WB 9	-	0.560	0.525	0.546	0.091	10	0.465	0.560	0.690

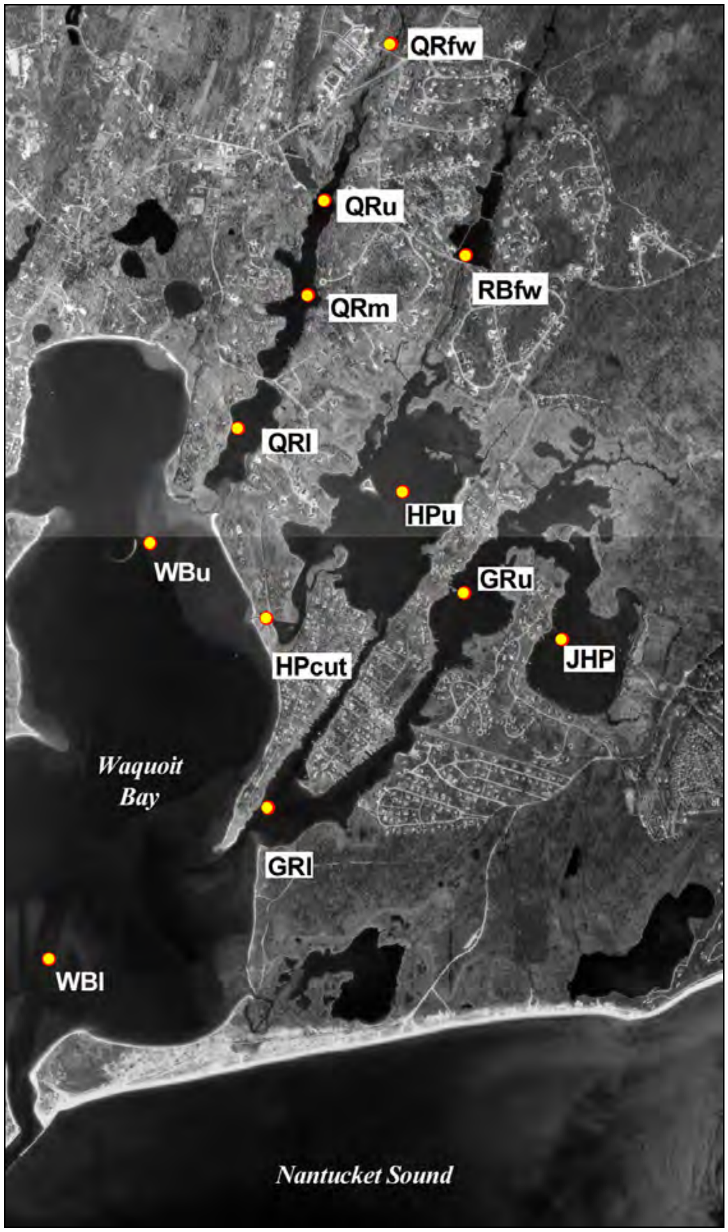


Figure VI-1. Estuarine water quality monitoring station locations in Waquoit Bay. Station labels correspond to those provided in Table VI-1.

years of baseline field data is the minimum required to provide a baseline for MEP analysis, and typically, three years of data were available for stations monitored by SMAST in the Waquoit Bay estuarine system.

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Hamblin Pond/Jehu Pond and Quashnet River system of Waquoit Bay. 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 Waquoit Bay 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 West Falmouth Harbor and the "finger" ponds of Falmouth, MA (Ramsey et al., 2000), embayment systems in Chatham, MA (Howes et al., 2003), and the Popponesset Bay system in Mashpee, MA (Howes et al., 2004).

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 each Waquoit Bay subembayment system.

VI.2.1 Model Formulation

The formulation of the model is for two-dimensional depth-averaged systems in which concentration in the vertical direction is assumed uniform. The depth-averaged assumption is justified since vertical mixing by wind and tidal processes prevent significant stratification in the Waquoit Bay 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.

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

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 modeled sub-systems also were used for the water quality constituent modeling portion of this study.

Based on measured flowrates from SMAST and groundwater recharge rates from the USGS, both the Hamblin Pond/Jehu Pond and Quashnet River hydrodynamic models were set-up to include the latest estimates of surface water flows from Red Brook (to Hamblin Pond) and Moonakis River (to the Quashnet River). Surface freshwater inputs from these streams are significant compared to the tidal prisms of the embayments to which they discharge. The Moonakis River has a measure flowrate of 15.65 ft^3/sec (38,290 m^3/day), which is 48.3% of the volume exchanged daily by the tide in the estuarine portion of the River. In Hamblin Pond, Red Brook has an estimated average daily flowrate of 2.5 ft^3/sec (6,120 m^3/day), which is 2.3% of the tidal prism volume of the 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 (30 day) spin-up period. At the end of the spin-up period, the model was run for an additional 6 tidal-day (150 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 from the Hamblin Pond\Jehu Pond and Quashnet River RMA-2 models.

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 Moonakis River and Red Brook. Nitrogen loads from each separate sub-embayment watershed were distributed across the sub-embayment. For example, the combined watershed, direct atmospheric deposition, and benthic regeneration loads for Jehu Pond were evenly distributed at grid cells that formed the perimeter of the embayment.

The loadings used to model present conditions in the Hamblin Pond/Jehu Pond system are given in Table VI-2, and load for the Quashnet River system are presented in Table VI-3. 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 Tables VI-2 and VI-3).

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., Jehu Pond) have approximately twice the N loading rate from benthic regeneration as from the watershed. For other sub-embayments (e.g., Hamblin Pond), the benthic flux is relatively low or negative indicating a net uptake of nitrogen in the bottom sediments.

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Hamblin Pond	3.84	1.53	-5.54
Hamblin Pond Cut	-	-	2.06
Upper Hamblin Pond	1.54	0.06	-4.98
Little River	1.11	0.16	3.53
Lower Great River	2.95	0.75	10.06
Upper Great River	0.68	0.55	9.55
Jehu Pond	3.61	0.67	10.43
Surface Water Sources			
Red Brook	3.88	-	-

Table VI-3. Sub-embayment and surface water loads used for total nitrogen modeling of the Quashnet River system, with total watershed N loads, atmospheric N loads, and benthic flux. These load represent **present loading conditions** for the listed sub-embayments.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Upper Quashnet River	2.16	0.33	8.55
Middle Quashnet River	-	-	1.50
Lower Quashnet River	0.72	0.25	4.78
Surface Water Sources			
Moonakis River	23.00	-	-

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary was specified. The models use concentrations at the open boundary during the flooding tide periods of the model simulations. Constituent concentrations of the incoming water are set at the value designated for the open boundary. The boundary concentrations inside Waquoit Bay were set at 0.395 mg/L at the Great River and Quashnet River Inlets (based on SMAST data from the Bay at station WB-13), and 0.478 mg/L at the Hamblin Pond cut inlets (based on data from monitoring station WB-12). The open boundary total nitrogen concentration set in the models represent long-term average summer concentrations.

VI.2.4 Model Calibration

Calibration of the Hamblin Pond/Jehu Pond and Quashnet River 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 large riverine estuary systems characterized by relatively wide channels (compared to channel depth) with moderate currents. Generally, the relatively quiescent Waquoit Bay sub-embayments are small compared to the riverine estuary systems evaluated by Fischer, et al., (1979); therefore the values of *E* also are relatively lower. 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-4. These values were used to develop the “best-fit” total nitrogen model calibration. For the case of TN modeling, “best fit” can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

Comparisons between model output and measured nitrogen concentrations are shown in Figure VI-2 for the Hamblin Pond/Jehu Pond system, and Figure VI-3 for the Quashnet River system. In the plot, 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. Because the water samples are taken during ebbing tides, calibration targets in each sub-embayment were set such that the means of the measured data would fall within the range between the modeled maximum and modeled mean concentration, for stations where there is a wide range of modeled concentrations. This technique was used on embayments like the stations in Great River. At other locations (e.g., the Quashnet River), where the model exhibited less variability than the measured data, a calibration target near the mean of the water column data was selected.

Calibrated model output is shown in Figure VI-4 and VI-5 for the Hamblin Pond/Jehu Pond and Quashnet River systems, respectively. In these figures, color contours indicate nitrogen concentrations throughout the model domain. Output in these figures show average total nitrogen concentrations, computed using the full 6-tidal-day model simulation output period.

VI.2.5 Model Salinity Verification

In addition to the model calibration based on nitrogen loading and water column measurements, numerical water quality model performance is typically verified by modeling salinity. This step was performed for the Waquoit Bay sub-systems using salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of Hamblin Pond/Jehu Pond and the Quashnet River, in addition to the RMA2 hydrodynamic model output, were salinities at the model open boundary and at the freshwater stream discharges. The open boundary salinity for both models was set at 29.9 ppt. For the steam inputs, salinities were set at 0.1 ppt. Fresh water flow rates for the streams were the same as those used for the total nitrogen model, as presented earlier in this section.

A comparison of modeled and measured salinities is presented in Figures VI-6 (Hamblin Pond/Jehu Pond) and VI-7 (Quashnet River), with contour plots of model output shown in Figures VI-8 and VI-9 for each modeled system. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model adequately represents salinity gradients in the Hamblin Pond/Jehu Pond and Quashnet River systems. The salinity verification provides a further independent confirmation that model dispersion coefficients and represented freshwater inputs to the model correctly simulate the real physical system.

Table VI-4. Values of longitudinal dispersion coefficient, E, used in calibrated RMA4 model runs of salinity and nitrogen concentration for the Hamblin Pond/Jehu Pond and Quashnet River sub-systems of Waquoit Bay.

Embayment Division	E m^2/sec
Hamblin Pond/Jehu Pond	
Waquoit Bay	5.0
Hamblin Pond Marsh	5.0
Hamblin Pond	35.0
Hamblin Pond Cut	5.0
Jehu Pond Marsh	10.0
Upper Great River	15.0
Jehu Pond	15.0
Great River	17.0
Little River	15.0
Hamblin Pond Cut Culvert	0.5
Quashnet River	
Upper Quashnet River	5.0
Middle Quashnet River	15.0
Lower Quashnet River	1.0
Waquoit Bay	1.0

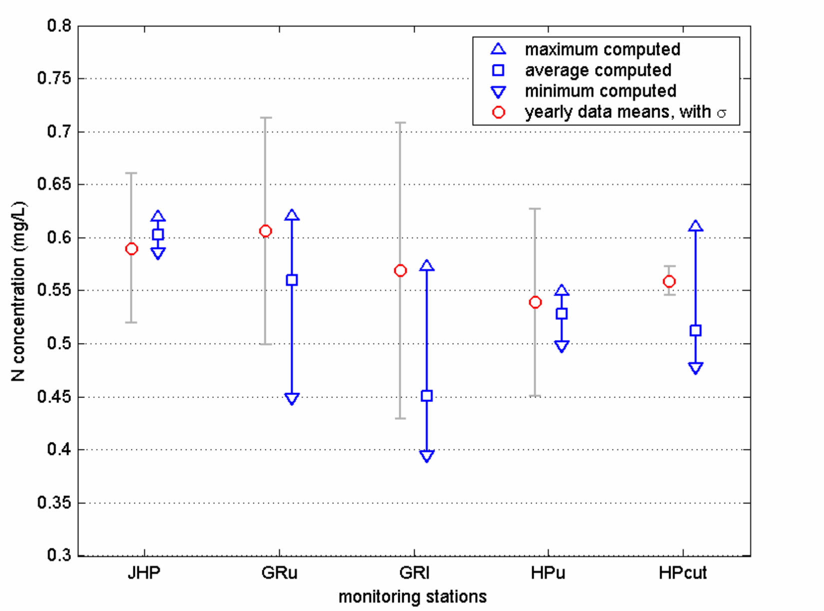


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Hamblin Pond/Jehu Pond system of Waquoit Bay. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed 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 annual data means.

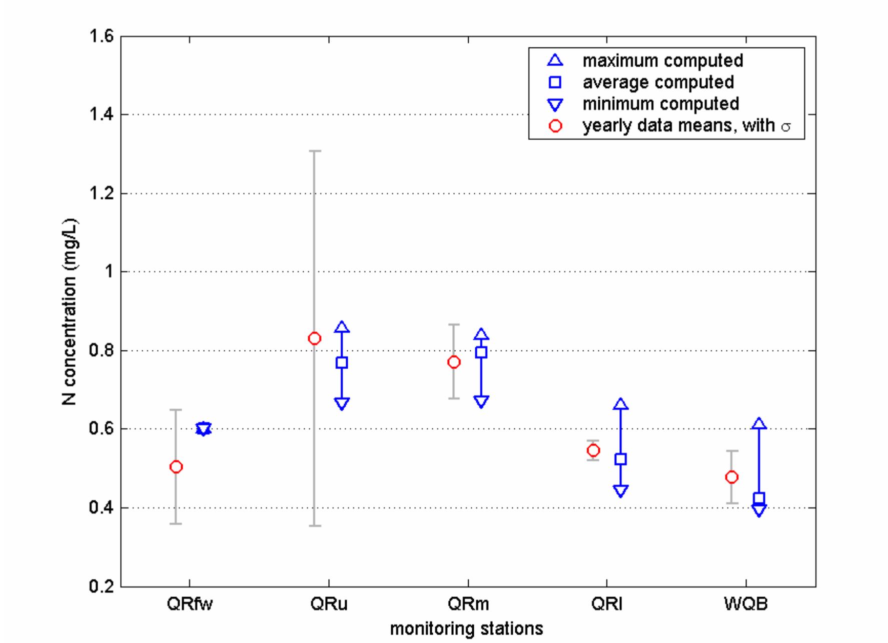


Figure VI-3. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Quashnet River system of Waquoit Bay. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed 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 annual data means.

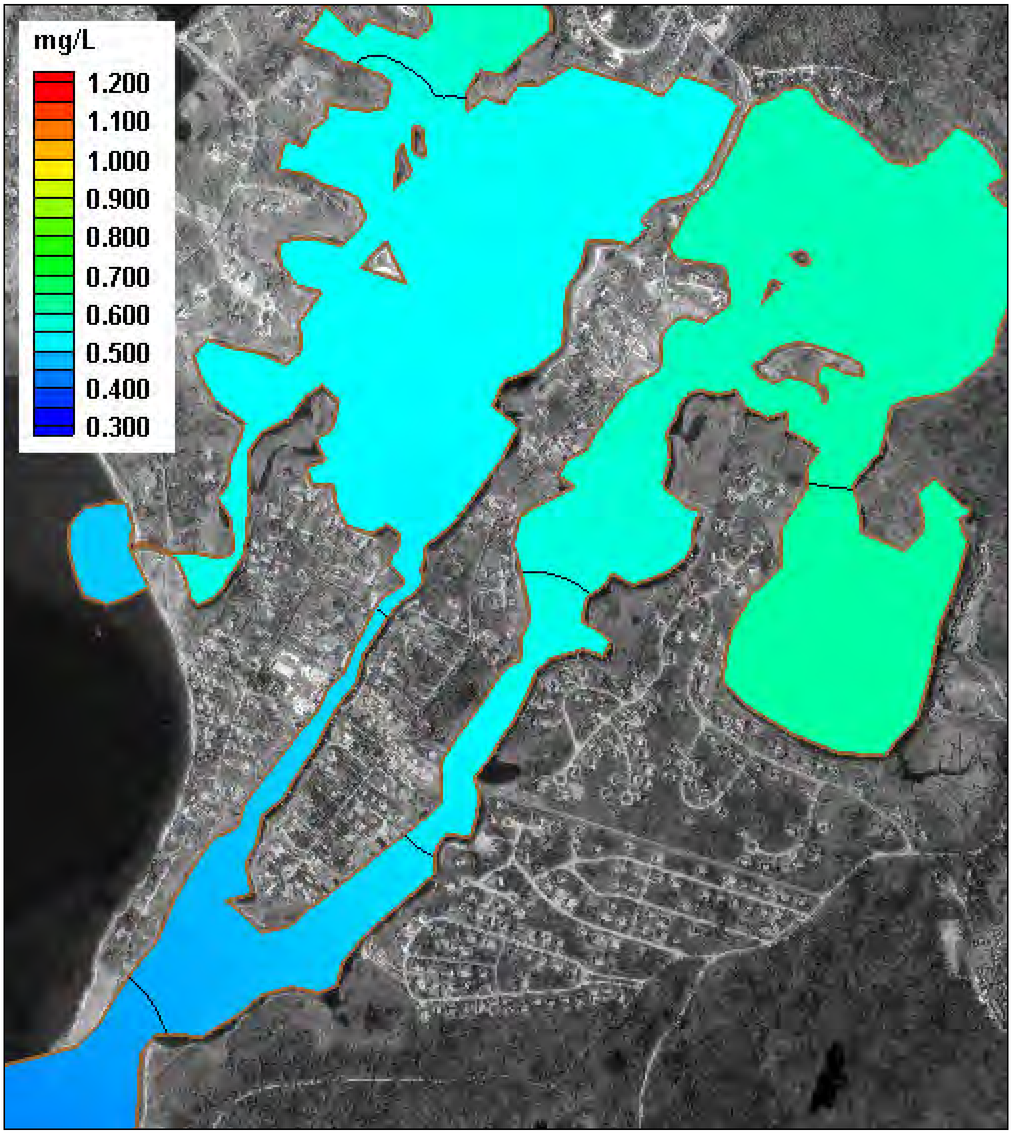


Figure VI-4. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for the Hamblin Pond and Jehu Pond sub-embayments of Waquoit Bay.

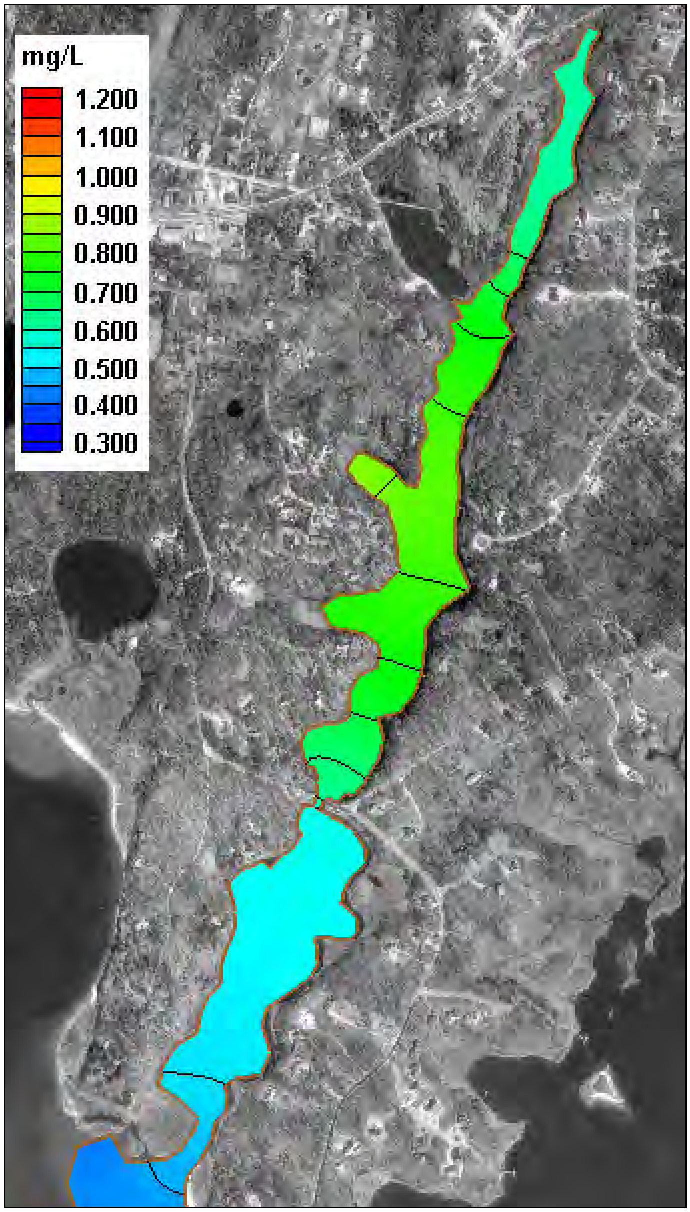


Figure VI-5. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Quashnet River sub-embayment system of Waquoit Bay.

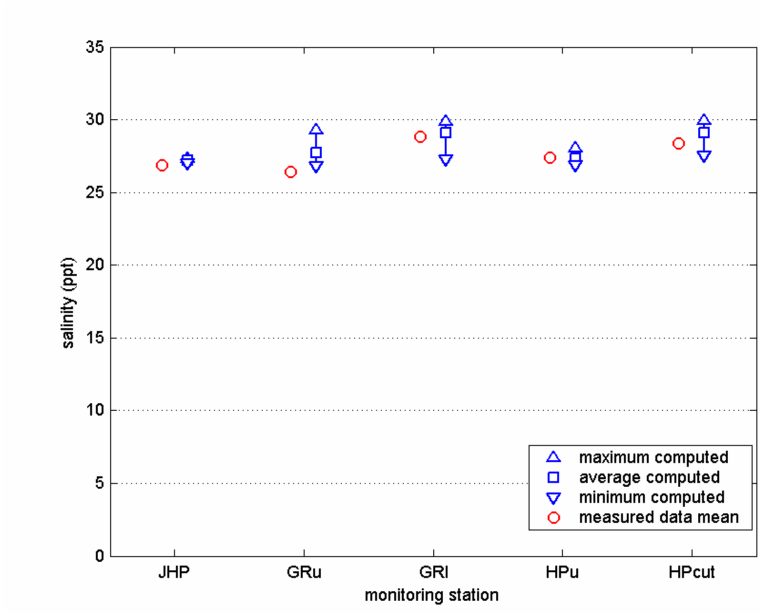


Figure VI-6. Comparison of measured salinity and calibrated model output at stations in the Hamblin Pond/Jehu Pond system. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (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).

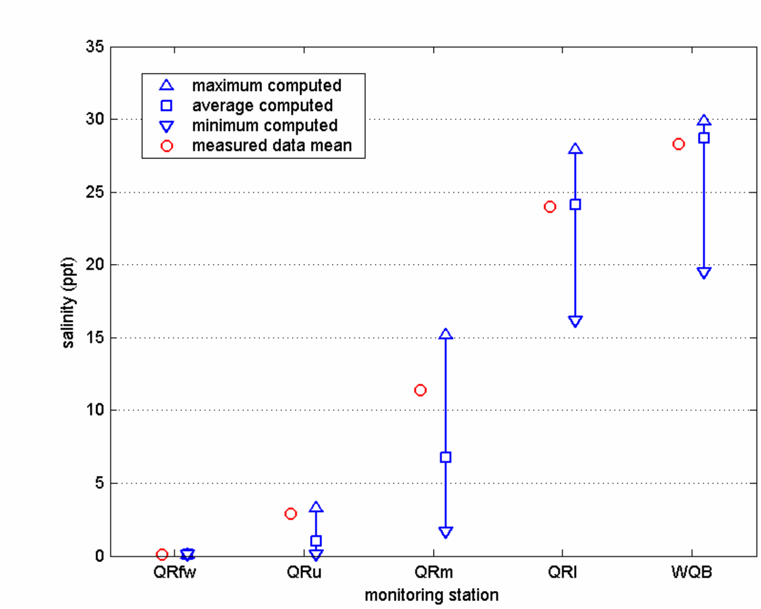


Figure VI-7. Comparison of measured salinity and calibrated model output at stations in the Quashnet River system. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle 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).

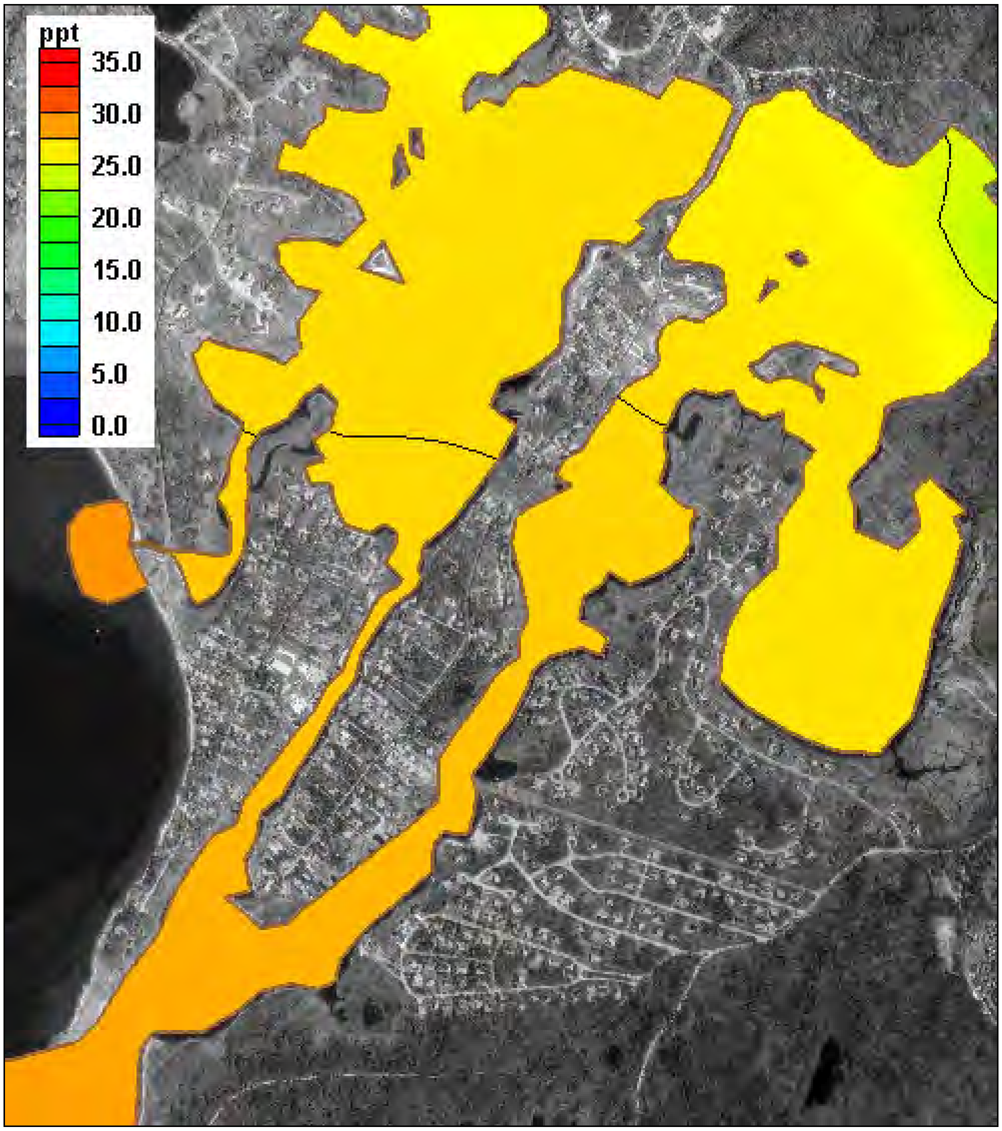


Figure VI-8. Contour Plot of modeled salinity (ppt) in the Hamlin Pond and Jehu Pond sub-embayment system of the Waquoit Bay estuarine system.

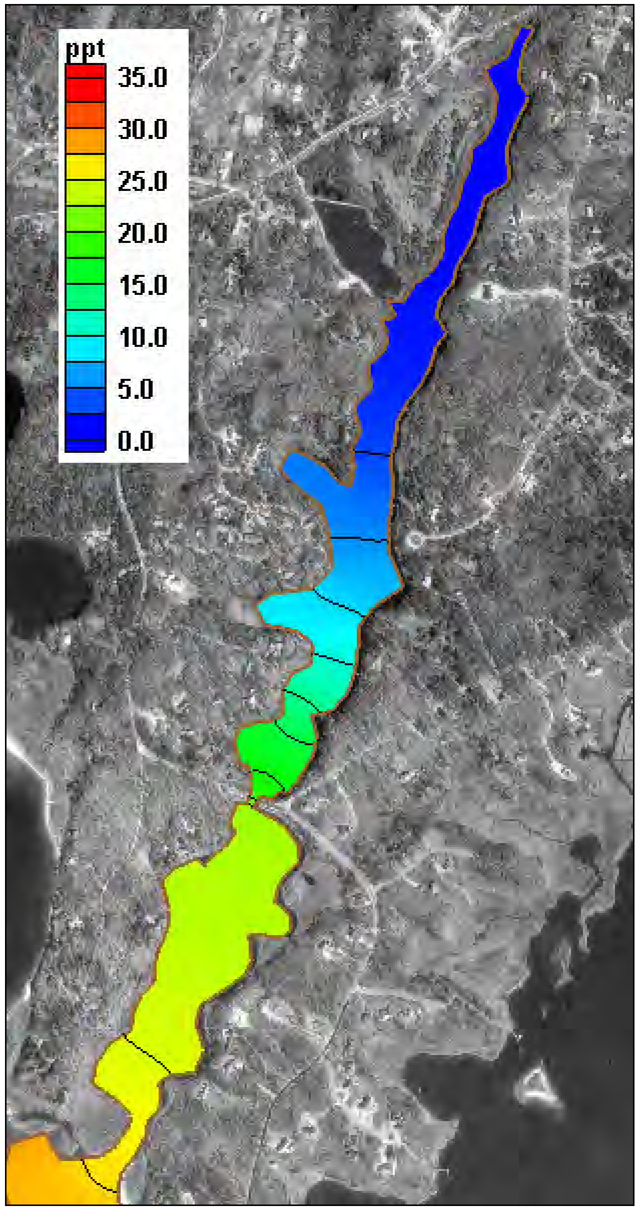


Figure VI-9. Contour Plot of modeled salinity (ppt) in the Hamlin Quashnet River sub-embayment system of Waquoit Bay.

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-5 (Hamblin Pond/Jehu Pond) and Table VI-6 (Quashnet River). 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-5. Comparison of sub-embayment watershed loads used for modeling of present, build out, and no-anthropogenic (“no-load”) loading scenarios of the Hamblin Pond/Jehu Pond system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present load (kg/day)	build out (kg/day)	build out % change	no load (kg/day)	no load % change
Hamblin Pond	3.84	4.84	+26.1%	0.13	-96.7%
Upper Hamblin Pond	1.54	2.10	+36.4%	0.06	-96.3%
Little River	1.11	1.27	+14.4%	0.02	-98.1%
Lower Great River	2.95	3.37	+14.3%	0.07	-97.8%
Upper Great River	0.68	1.58	+132.1%	0.22	-67.1%
Jehu Pond	3.61	4.01	+11.1%	0.12	-96.8%
Surface Water Sources					
Red Brook	3.88	7.29	+59.9%	0.42	-94.9%

Table VI-6. Comparison of sub-embayment watershed loads used for modeling of present, build out, and no-anthropogenic (“no-load”) loading scenarios of the Quashnet River system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms).

sub-embayment	present load (kg/day)	build out (kg/day)	build out % change	no load (kg/day)	no load % change
Upper Quashnet River	2.16	3.026	+39.9%	0.13	-94.0%
Lower Quashnet River	0.79	0.89	+12.5%	0.02	-97.2%
Surface Water Sources					
Moonakis River	23.00	46.82	+103.6%	4.16	-81.9%

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there only would be a 14% increase in watershed nitrogen load to the lower portion of the Hamblin Pond/Jehu Pond system as a result of potential future development. Other watershed areas would experience much greater load increases, for example the loads to the upper portion of Hamblin Pond (Lower Red Brook) would increase 36% from the present day loading levels. A maximum increase in watershed loading resulting from future development would occur in for the freshwater section of the Quashnet River (Moonakis River), where the increase would be about 23.8 kg/day or nearly 104% more load. 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 sub-embayment is shown in Table VI-7 (Hamblin Pond/Jehu Pond) and Table VI-8 (Quashnet River). 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 (positive) in benthic flux. Therefore, the benthic flux for the build-out scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase in benthic flux (i.e., a positive change in the absolute value of the flux), and *vise versa*.

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

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

where the projected PON concentration is calculated by,

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

using the watershed load ratio,

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

and the present PON concentration above background,

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

Following development of the nitrogen loading estimates for the build out scenario, the water quality model was run to determine nitrogen concentrations within each sub-embayment. Total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from build out was relatively large as shown in Table VI-9 and VI-10, with greater than 35% increases in total nitrogen concentrations in the upper portions of the Quashnet River system and in Hamblin Pond. Color contours of model output for the build-out scenario are presented in Figures VI-10 and VI-11. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figures VI-4 and VI-5, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-7. Buildout sub-embayment and surface water loads used for total nitrogen modeling of the Hamblin Pond/Jehu Pond system, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Hamblin Pond	4.84	1.53	-6.49
Hamblin Pond Cut	-	-	2.41
Upper Hamblin Pond	2.10	0.06	-5.83
Little River	1.27	0.16	4.03
Lower Great River	3.37	0.75	11.00
Upper Great River	1.58	0.55	10.45
Jehu Pond	4.01	0.67	11.34
Surface Water Sources			
Red Brook	6.21	-	-

Table VI-8. Buildout sub-embayment and surface water loads used for total nitrogen modeling of the Quashnet River system, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Upper Quashnet River	3.02	0.33	14.74
Middle Quashnet River	-	-	2.58
Lower Quashnet River	0.89	0.25	7.70
Surface Water Sources			
Moonakis River	46.82	-	-

Table VI-9. Comparison of model average total N concentrations from present loading and the buildout scenario, with percent change, for the Hamblin Pond/Jehu Pond system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

Sub-Embayment	monitoring station	present (mg/L)	buildout (mg/L)	% change
Jehu Pond (JHP)	WB 1	0.603	0.678	+12.4%
Upper Great River (GRu)	WB 2	0.560	0.628	+12.2%
Lower Great River (GRl)	WB 3	0.451	0.496	+9.9%
Hamblin Pond (HPu)	WB 4	0.528	0.713	+35.1%
Hamblin Pond cut (HPcut)	WB 10	0.512	0.577	+12.6%

Table VI-10. Comparison of model average total N concentrations from present loading and the buildout scenario, with percent change, for the Quashnet River system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

Sub-Embayment	monitoring station	present (mg/L)	buildout (mg/L)	% change
Moonakis River	WB 6	0.601	1.222	+103.3%
Upper Quashnet River	WB 7	0.768	1.484	+93.2%
Middle Quashnet River	WB 8	0.794	1.434	+80.6%
Lower Quashnet River	WB 9	0.523	0.700	+33.8%

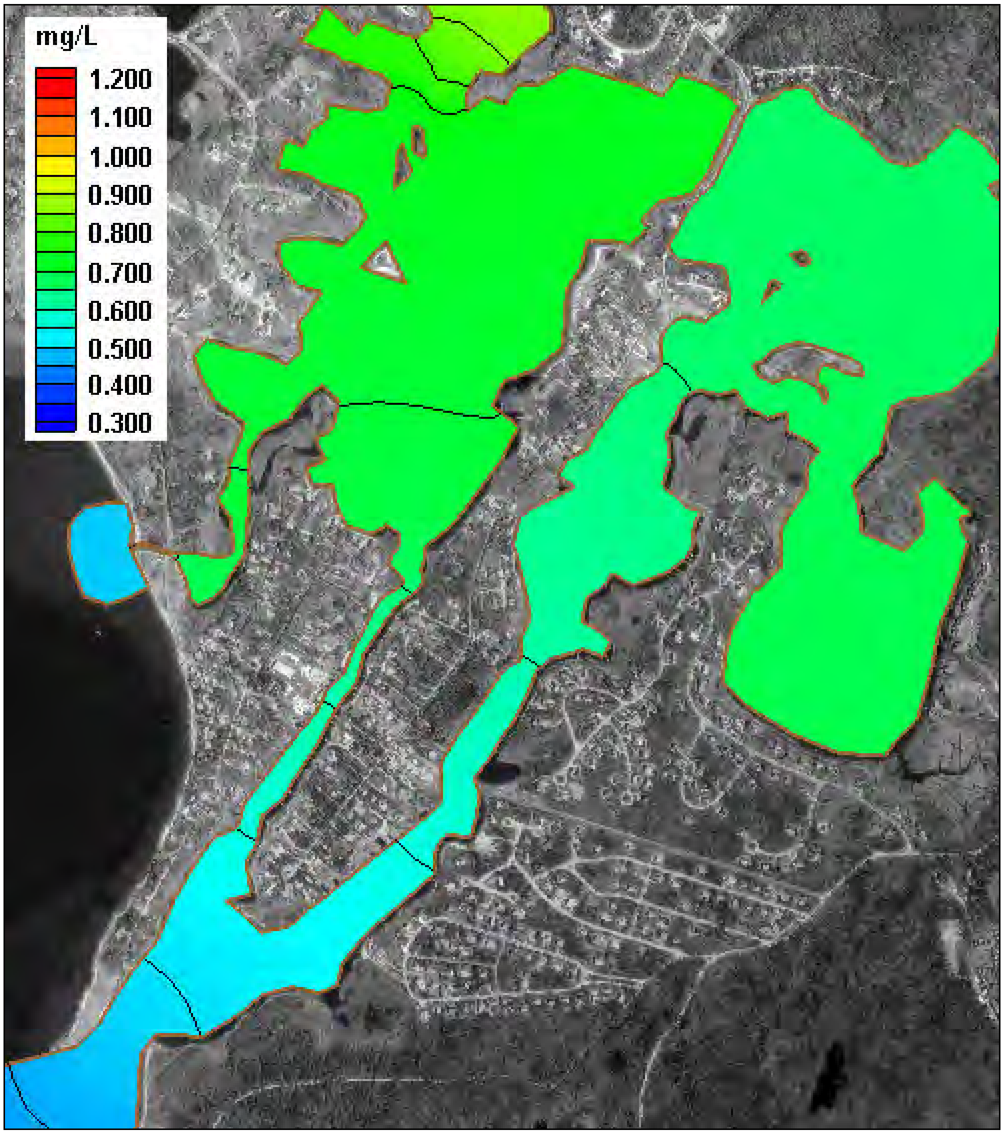


Figure VI-10. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Hamblin Pond/Jehu Pond embayment system of Waquoit Bay, for projected build out loading conditions.

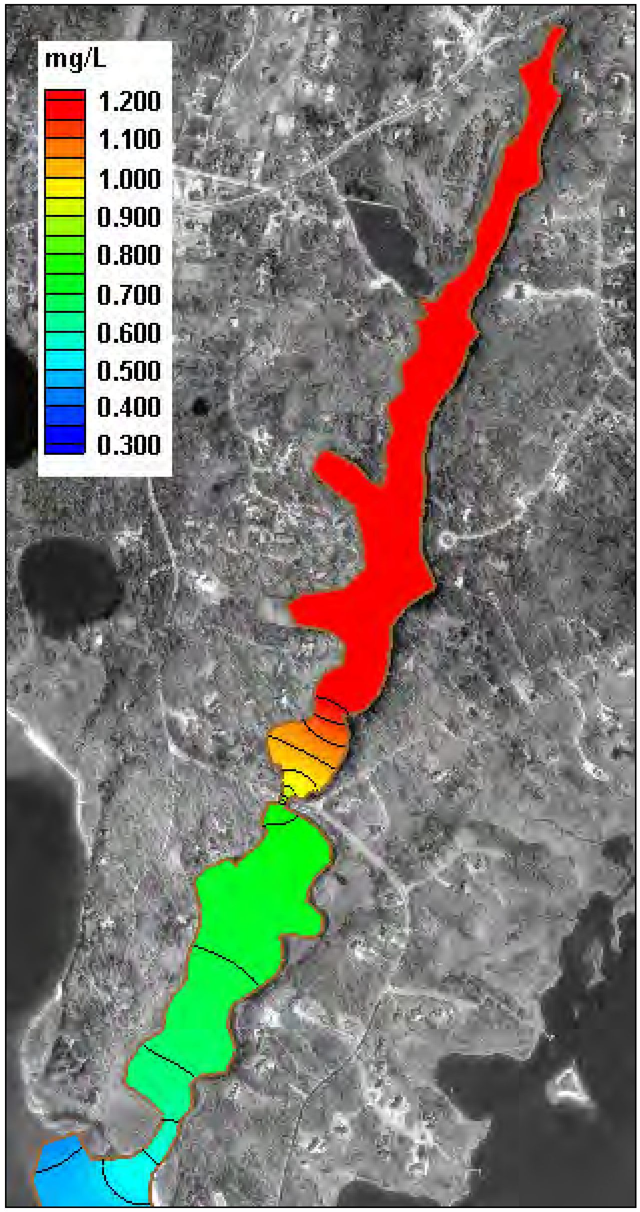


Figure VI-11. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Quashnet River embayment system of Waquoit Bay, for projected build out loading conditions.

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load scenarios is shown in Tables VI-11 and VI-12, for the Hamblin Pond/Jehu Pond and Quashnet River embayment systems, respectively. Benthic flux inputs to each embayment model was reduced (toward zero) based on the reduction in the watershed load. 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.

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. In contrast to the build out scenario, a Waquoit Bay boundary N concentration of 0.30 mg/L was used for this "forested watershed" scenario, to reflect likely pristine conditions in the main basin of Waquoit Bay. The relative change in total nitrogen concentrations resulting from "no load" loading conditions was significant, as is shown in Tables VI-13 and VI-14, with reductions greater than 60% occurring the upper portions of the systems (e.g., Hamblin Pond and the Quashnet River. These results are shown pictorially in Figures VI-12 and VI-13 for each of the modeled Waquoit sub-systems.

Table VI-11. No-anthropogenic-load scenario sub-embayment and surface water loads used for total nitrogen modeling of the Hamblin Pond/Jehu Pond system, with total watershed N loads, atmospheric N loads, and benthic flux. A Waquoit Bay boundary N concentration of 0.30 mg/L was used for this "forested watershed" scenario.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Hamblin Pond	0.13	1.53	-3.26
Hamblin Pond Cut	-	-	1.20
Upper Hamblin Pond	0.06	0.06	-2.93
Little River	0.02	0.16	2.31
Lower Great River	0.07	0.75	6.32
Upper Great River	0.22	0.55	6.00
Jehu Pond	0.12	0.67	6.85
Surface Water Sources			
Red Brook	0.20	-	-

Table VI-12. No-anthropogenic-load scenario sub-embayment and surface water loads used for total nitrogen modeling of the Quashnet River system, with total watershed N loads, atmospheric N loads, and benthic flux. A Waquoit Bay boundary N concentration of 0.30 mg/L was used for this "forested watershed" scenario.

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Upper Quashnet River	0.13	0.33	2.83
Middle Quashnet River	-	-	0.49
Lower Quashnet River	0.02	0.25	2.09
Surface Water Sources			
Moonakis River	4.26	-	-

Table VI-13. Comparison of model average total N concentrations from present loading and the no anthropogenic load ("no load") scenario, with percent change, for the Hamblin Pond/Jehu Pond system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). A Waquoit Bay boundary N concentration of 0.30 mg/L was used for this "forested watershed" scenario.

Sub-Embayment	monitoring station	present (mg/L)	no load (mg/L)	% change
Jehu Pond (JHP)	WB 1	0.595	0.357	-40.1%
Upper Great River (GRu)	WB 2	0.557	0.340	-39.0%
Lower Great River (GRl)	WB 3	0.453	0.294	-35.2%
Hamblin Pond (HPu)	WB 4	0.529	0.205	-61.2%
Hamblin Pond cut (HPcut)	WB 10	0.512	0.281	-45.2%

Table VI-14. Comparison of model average total N concentrations from present loading and the no anthropogenic load ("no load") scenario, with percent change, for the Quashnet River system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions). A Waquoit Bay boundary N concentration of 0.30 mg/L was used for this "forested watershed" scenario.

Sub-Embayment	monitoring station	present (mg/L)	no load (mg/L)	% change
Moonakis River	WB 6	0.601	0.109	-81.9%
Upper Quashnet River	WB 7	0.768	0.170	-77.9%
Middle Quashnet River	WB 8	0.794	0.229	-71.2%
Lower Quashnet River	WB 9	0.523	0.294	-43.8%

For the no load scenario, the sub-embayment concentrations are generally governed by the total nitrogen concentrations observed in Waquoit Bay. There is a negative gradient in total nitrogen concentrations from the Great River inlet to Hamlin Pond, and also within the Quashnet River. This is a major difference from the modeled present and buildout conditions, where concentrations increase from the inlet to the upper reaches of the system. The slight negative gradients in the modeled "no-load" scenario result because the surface freshwater inputs (i.e., the Moonakis River and Red Brook) have little load themselves, and therefore dilute concentrations at the head of the Quashnet River and in Hamblin Pond.

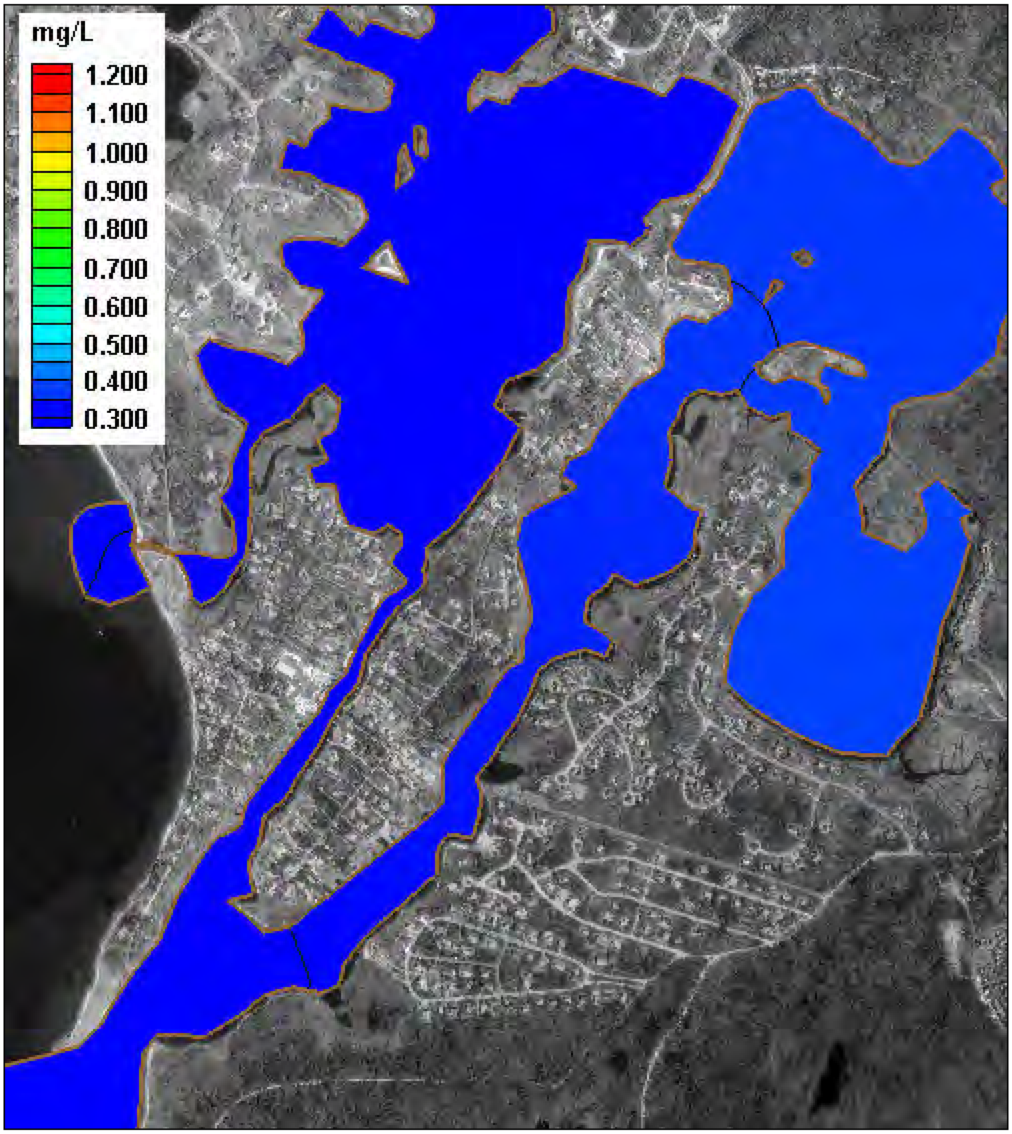


Figure VI-12. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Hamblin Pond/Jehu Pond sub-system of Waquoit Bay, for no anthropogenic loading conditions.

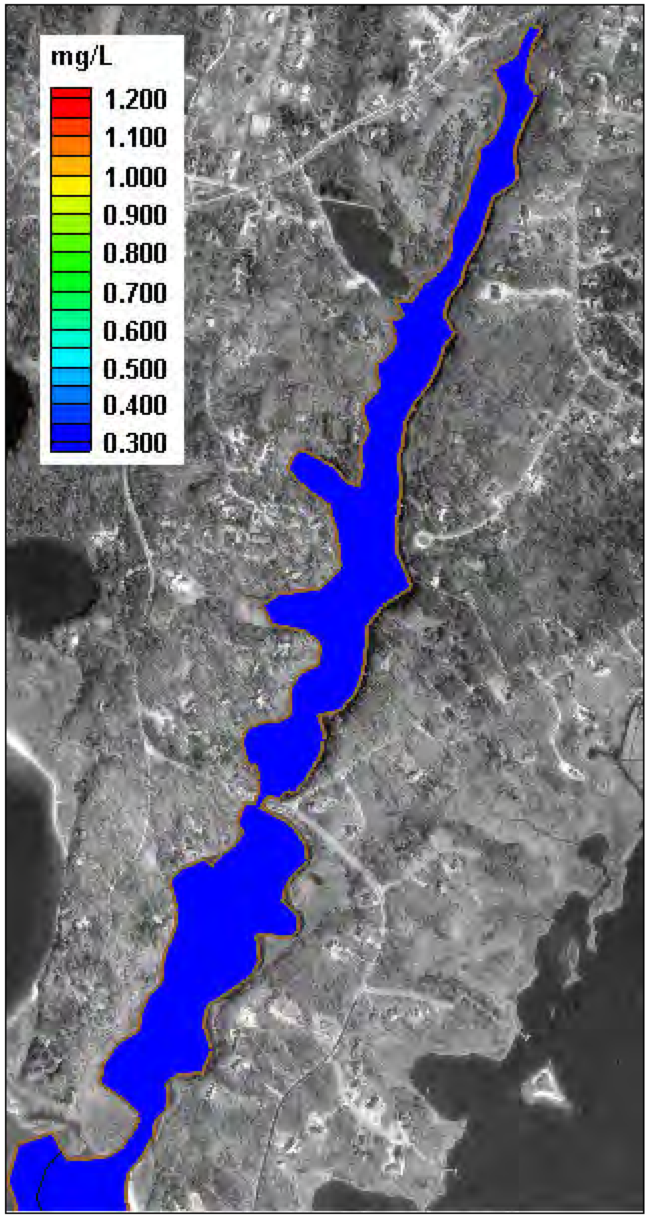


Figure VI-13. Contour Plot of modeled total nitrogen concentrations (mg/L) in the Quashnet River embayment system of Waquoit Bay, for no anthropogenic loading conditions.

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, as well as the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River Estuaries tributary to the Waquoit Bay System in the Towns of Falmouth and Mashpee, Cape Cod, MA, our assessment is based upon data from the water quality monitoring database and our surveys of eelgrass distribution, benthic animal communities and sediment characteristics, in addition to dissolved oxygen records obtained during the summers of 2001 and 2002. The water quality data (e.g. chlorophyll) was collected by the Mashpee Water Quality Monitoring Program and the WBNERR sponsored BayWatcher Program. These data form the basis of an assessment of this system's present health, and when coupled with a full water quality synthesis and projections of future conditions based upon the water quality modeling effort, will support complete nitrogen threshold development for these systems (Chapter VIII).

VII.1 OVERVIEW OF BIOLOGICAL HEALTH INDICATORS

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

As a basis for a nitrogen thresholds determination, MEP focused on major habitat quality indicators: (1) bottom water dissolved oxygen and chlorophyll a (Section VII.2), (2) eelgrass distribution over time (Section VII.3) and (3) benthic animal communities (Section VII.4). Dissolved oxygen depletion is frequently the proximate cause of habitat quality decline in coastal embayments (the ultimate cause being nitrogen loading). However, oxygen conditions can change rapidly, and frequently show strong tidal and diurnal patterns. Even severe levels of oxygen depletion may occur only infrequently, yet have important effects on system health. To capture this variation, the MEP Technical Team deployed dissolved oxygen sensors within the upper tributary sub-embayments (Quashnet River and Hamblin Pond) to record the frequency and duration of low oxygen conditions during the critical summer period. A dissolved oxygen sensor was also deployed in Jehu Pond, but failed to yield usable data. However, anoxic conditions were measured in Jehu Pond during previous MEP field data collection, indicating that severe oxygen depletion was occurring periodically in this basin.

The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within the eastern Waquoit Bay System was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). In addition, results of mapping studies of these estuaries conducted during 1987-1992 were also used for evaluating temporal trends (Short and Burdick 1996). 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 eastern Waquoit Bay sub-embayments, temporal changes in eelgrass distribution provides a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing due to new inlet formation) 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 or "Significantly Degraded" (high organic matter loading, low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, benthic population studies in Buzzards Bay (Woods Hole Oceanographic Institution) and New Bedford (SMAST), and more recently the Woods Hole Oceanographic Institution Nantucket Harbor Study (Howes et al., 1997). These data are coupled with the level of diversity (H') and evenness (E) of the benthic community and the total number of individuals to determine the infaunal habitat quality.

VII.2 BOTTOM WATER DISSOLVED OXYGEN

Dissolved oxygen levels near atmospheric equilibration are important for maintaining healthy animal and plant communities. Short-duration oxygen depletions can significantly affect communities even if they are relatively rare on an annual basis. For example, for the Chesapeake Bay it was determined that restoration of nutrient degraded habitat requires that instantaneous oxygen levels not drop below 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 Waquoit Bay System, including the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River estuaries, 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 within the sub-embayment system (Figure VII-2). 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 about biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the instrument mooring data from the Quashnet River and Hamblin Pond sub-embayments were collected during the summer of 2002. Since the moored instrument in

Jehu Pond did not yield usable data, the MEP analysis of this basin had to rely on traditional "grab" samples for dissolved oxygen (and chlorophyll a). These samples are typically collected in the early morning, when oxygen levels are at or near their lowest point for a day. These oxygen data were collected by WBNERR's Baywatch Program and the Mashpee Water Quality Monitoring Program overseen by Coastal Systems Program-SMAST Staff.

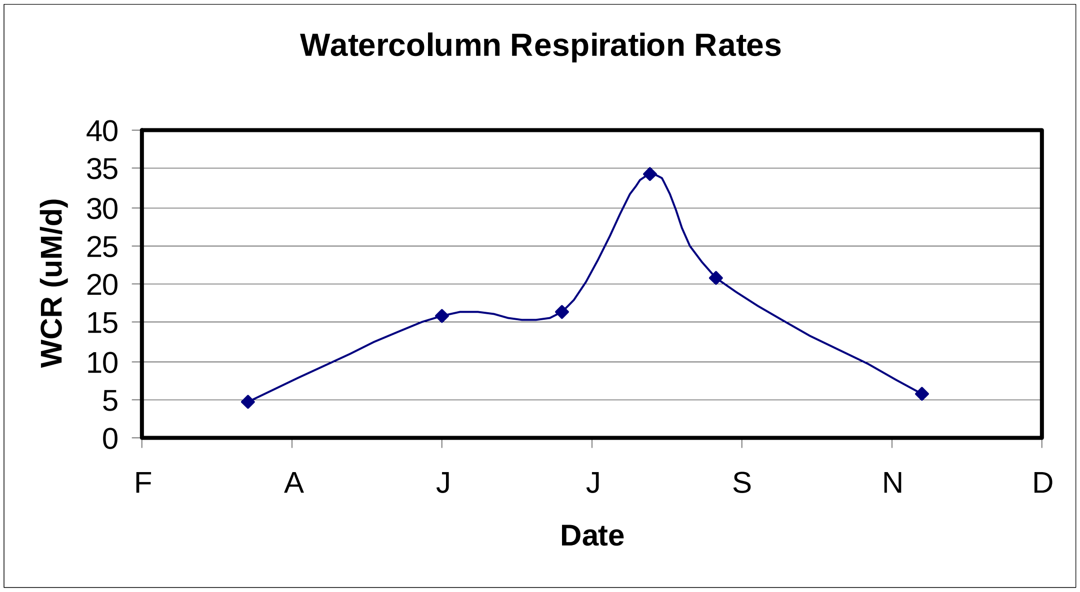


Figure VII-1. Average watercolumn respiration rates from water collected throughout the Popponesset Bay System (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Similar to other embayments in southeastern Massachusetts, the sub-embayments to the overall Waquoit Bay System evaluated in this assessment showed high frequency variation, apparently related to diurnal and sometimes tidal influences. Nitrogen enrichment of embayment waters generally manifests itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. The high degree of temporal variation in bottom water dissolved oxygen concentration at each mooring site underscores the need for continuous monitoring within these systems. More important, both the level of oxygen depletion and the magnitude of daily oxygen excursion indicate nutrient enriched waters and impaired habitat quality at both mooring sites (Quashnet River, Figure VII-3 and Hamblin Pond, Figure VII-4).

The dissolved oxygen records for the tidally influenced lower Quashnet River and the upper region of Hamblin Pond indicate that these sub-embayments currently maintain a high and moderate level of oxygen stress, respectively. Jehu Pond showed a high level of oxygen depletion, at a level which will impair habitat quality, with dissolved oxygen levels periodically approaching anoxia. Nitrogen enrichment of embayment waters can manifest itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. This phenomenon is best seen in the Quashnet River record, where dissolved oxygen levels frequently drop to less than 4 mg L^{-1} during the night and reach levels in excess of atmospheric saturation during the day time (Figure VII-3).

Figure VII-2. Aerial Photograph of the Waquoit Bay system in Falmouth/Mashpee showing locations of Dissolved Oxygen mooring deployments conducted in summer 2002.

Dissolved oxygen records were analyzed to determine the percent of the deployment time (29-37 days) that oxygen was below various benchmark concentrations (Table VII-1). The data collected by the water quality monitoring programs for Jehu Pond was of sufficient size to allow a frequency analysis similar to that for the moored instruments in Quashnet River and Hamblin Pond. These data indicate not just the minimum or maximum levels of this critical nutrient related constituent, but the intensity of the low oxygen circumstances. 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. From the oxygen records it is clear that, after Jehu Pond, the Quashnet River has the greatest extent of oxygen depletion and the oxygen excursion indicates a high degree of nutrient enrichment (as is supported by the chlorophyll a data, as described later in this Section). Note that this data are from the lower part of this system, which has the highest water quality, but still the oxygen levels are <4 mg L^{-1} almost 10% of the time. However, use of only the duration of oxygen below for example 4 mg/L^{-1} would underestimate oxygen stress in this system. The effect of nitrogen enrichment is to cause oxygen depletion; however, with increased phytoplankton (or epibenthic algae), oxygen levels will rise in daylight to above atmospheric equilibration levels in shallow systems. The Quashnet River data indicates that daily excursions of 15 mg L^{-1} in bottom water oxygen do occur. This is the case in the Quashnet River and to a lesser extent in Hamblin Pond.

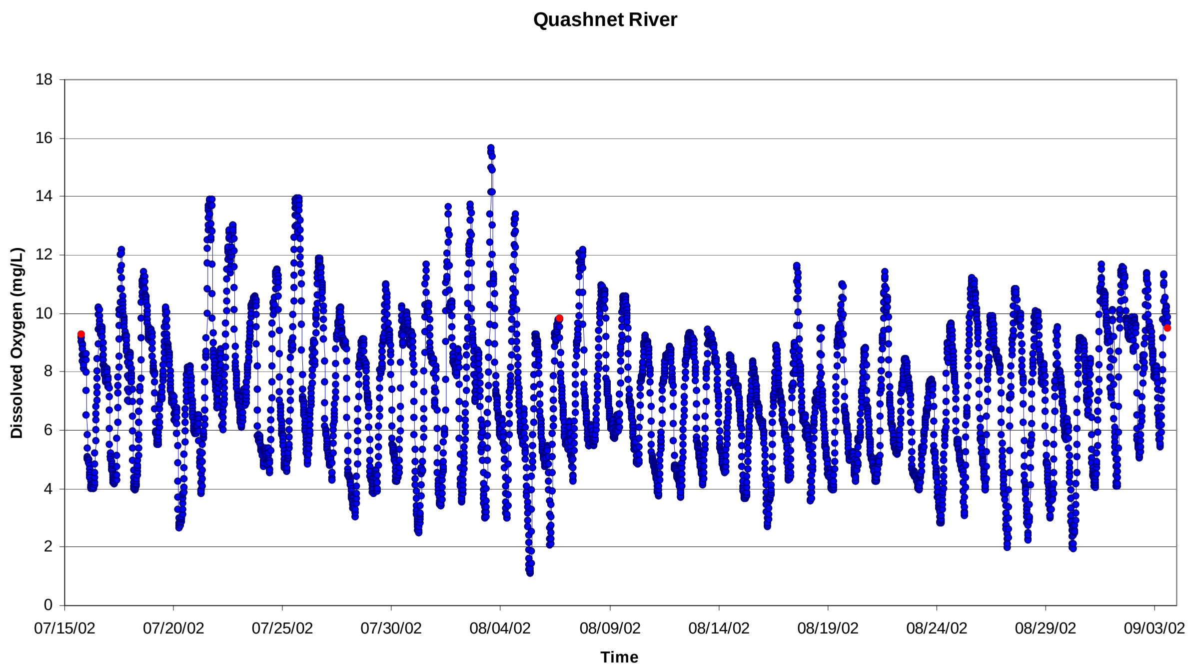


Figure VII-3. Bottom water record of dissolved oxygen (top panel) in the Quashnet River Estuary (lower basin), summer 2002. Calibration samples represented as red dots.

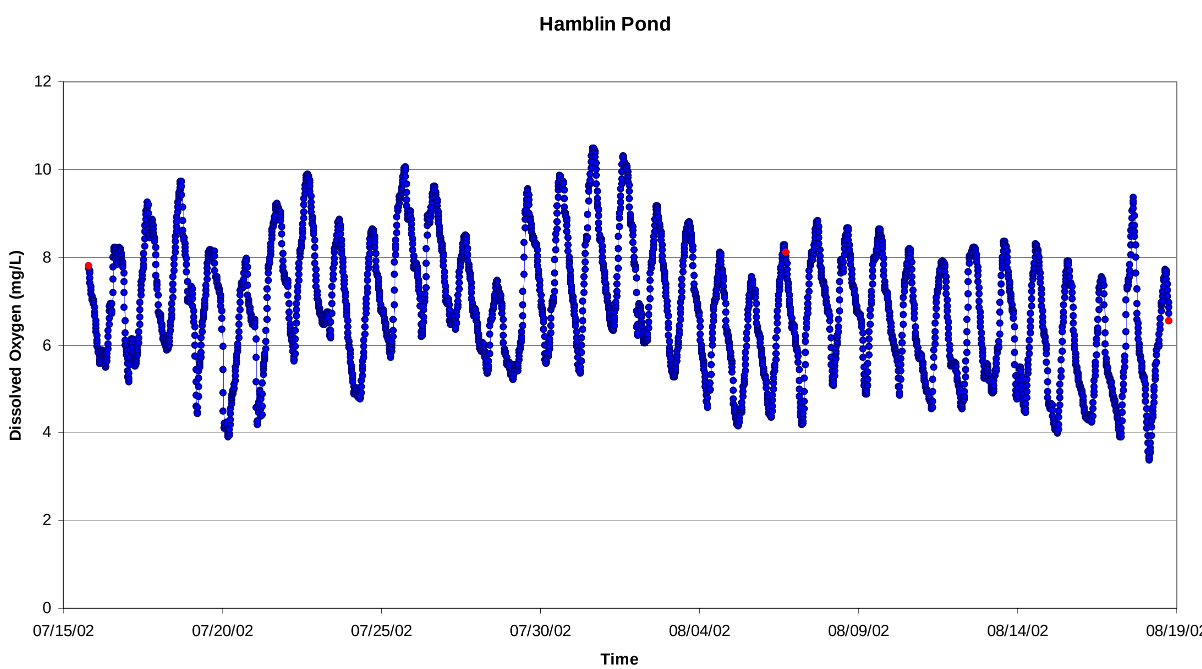


Figure VII-4. Bottom water record of dissolved oxygen (bottom panel) in Hamblin Pond, summer 2002. Calibration samples represented as red dots.

Table VII-1. Percent of time during deployment of *in situ* sensors or of traditional grab sampling events that bottom water oxygen levels were below various benchmark oxygen levels.

Massachusetts Estuaries Project Town of Mashpee: 2002					
Waquoit Bay Sub-Embayments	**Dissolved Oxygen: Summer**				
	Total Days	**<6 mg/L (% of days)**	**<5 mg/L (% of days)**	**<4 mg/L (% of days)**	**<3 mg/L (% of days)**
Continuous Record: 2002					
Hamblin Pond	29	31%	11%	1%	0%
Quashnet River (lower)	37	36%	21%	8%	2%
Grab Samples 1994-2003+					
Jehu Pond	43	81%	65%	37%	14%
Quashnet River (mid)	68	66%	46%	28%	13%

+ **Composite of Mashpee/SMAST and WBNERR (from NERR Web Site) grab sampling data; days = Number of sampling dates.**

Chlorophyll a data for each of the three estuaries collected by the water quality monitoring program was of sufficient size to allow a frequency analysis similar to that for dissolved oxygen (Table VII-2). The difference between the chlorophyll levels assayed by the Baywatch Program and Mashpee Program cannot be definitively explained. However, some difference was expected as the Mashpee Program assays for total chlorophyll a pigment (sum of chlorophyll a and its immediate breakdown product, pheophythin a) which is a better indicator of bloom conditions. The Mashpee/SMAST data were used for this MEP analysis, but the Baywatch data are presented for comparison, as it is a longer dataset. Both data sets show similar patterns of nitrogen related habitat quality. It is clear that the Quashnet River is highly eutrophic with total chlorophyll a levels in the upper and mid regions averaging >20 ug L^{-1} (SMAST data presented in Figure VII-5). The moored chlorophyll sensor showed similarly high values (Table VII-3). Phytoplankton blooms appear to be generated within the upper and mid basins of the Quashnet, most likely as a result of the high nitrogen loading to the headwaters via the Quashnet River freshwater discharge. It is interesting that on three sampling events the upper station showed exceedingly large blooms (>140 ug L^{-1}), while on three separate events the mid station showed very large blooms (>40 ug L^{-1}). The pattern seems to indicate potentially separate points of origination (upper versus mid), although a flushing out of an upper bloom cannot be discounted in the observed mid bloom events.

Jehu and Hamblin Ponds support lower total chlorophyll levels, averaging 11.9 and 7.4 µg L^{-1}, respectively. Jehu Pond appears to be showing more nutrient enrichment than Hamblin Pond both on average and in the size of the blooms (Table VII-2, maximum values). The high phytoplankton biomass in Jehu Pond is consistent with the observation of oxygen stress in this system. The moderate total chlorophyll levels in Hamblin Pond are consistent with its moderately good oxygen status. The agreement between the chlorophyll and oxygen levels in these Pond basins is likely the result of their physical structure. At first glance the Quashnet River did not show the same relationship. However, this likely results in part from the placement of the oxygen mooring in the lower basin which supports lower phytoplankton levels than the mid and upper stations above the bridge (Figure VII-6). However, traditional "grab" sampling data are also available for the mid station. These data indicate a high degree of oxygen depletion with almost one third of the sampling dates showing oxygen levels <4 mg L^{-1}. This pattern is also seen in the limited oxygen data from the upper region of this system. Taken in whole, it appears that the Quashnet River Estuary is showing oxygen stress throughout its reach and it is likely that the level of depletion is higher in the upper and mid reaches than in the lower basin, consistent with the distribution of phytoplankton biomass.

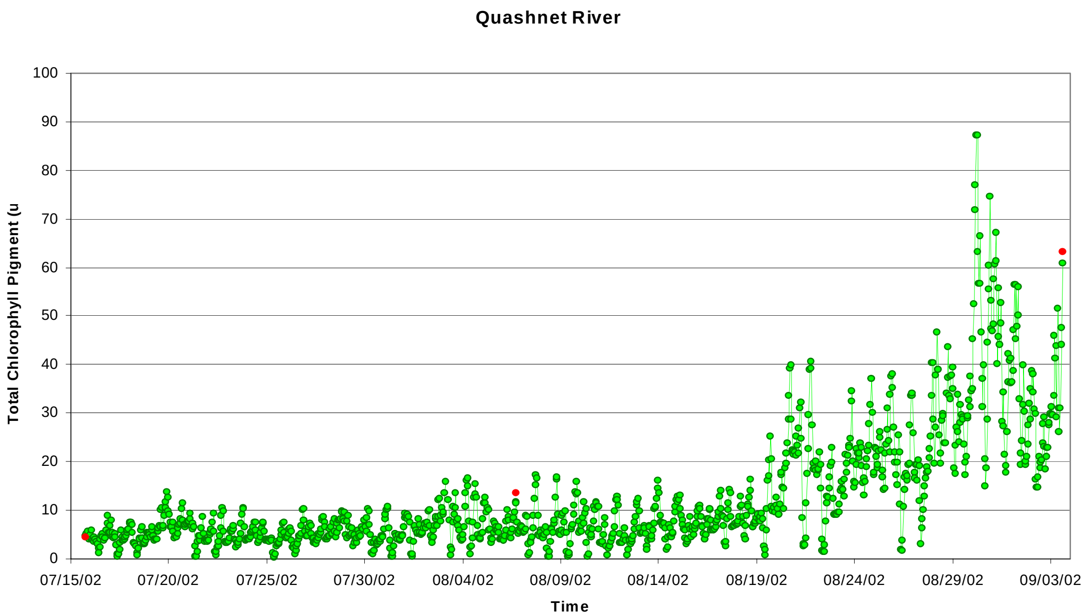


Figure VII-5. Bottom water record of chlorophyll-*a* (bottom panel) in the Quashnet River Estuary (lower basin), summer 2002. Calibration samples represented as red dots

Combining the dissolved oxygen and chlorophyll a data yield a clear pattern of nutrient related habitat quality. A further analysis incorporating eelgrass and infaunal indicators is included later in this Section. At present, the Quashnet River estuary is showing poor oxygen status (based upon depletions, daily excursions, mooring in lower basin) and large phytoplankton blooms. While this system appears to be stressed throughout, there is a clear gradient from hypereutrophic in the upper regions to eutrophic in the lower basin. Jehu Pond is also showing nitrogen enriched conditions, with periodic hypoxia/anoxia in the basin and high phytoplankton biomass. Hamblin Pond is showing the best nutrient related habitat quality, based both upon its moderately good oxygen conditions and moderate phytoplankton biomass. Based upon the dissolved oxygen and chlorophyll data the nutrient related habitat quality of the three estuarine sub-embayments to eastern Waquoit Bay can be classified is as follows:

- Quashnet River estuary – Significantly Impaired
- Jehu Pond – Moderately/Significantly Impaired
- Hamblin Pond – Moderately Impaired

Table VII-2. Levels of chlorophyll a pigments within the Town of Mashpee sub-embayments to Waquoit Bay. All data were collected by grab samples from June-September. Data collected by the Waquoit Bay BayWatcher Program (WBNERR) and by Popponesset Bay Water Quality Monitoring Program and Coastal Systems Program, SMAST (SMAST). Geometric averages were used to estimate "average" conditions, given the periodic phytoplankton blooms. WBNERR data (June-September) is from the BayWatcher samplings garnered from NERR Web site.

	Sampling			Statistics				
	Source	Station	Year	Geo Mean ug/L	Geo Stdev ug/L	Max ug/L	Min ug/L	N
Waquoit Bay Sub-Embayments								
Hamblin Pond								
Mid	WBNERR	Site 3	1998-2002	2.1	2.6	9.5	0.2	29
Mid	SMAST	WB-4	2001-2003	7.4	1.7	28.3	3.2	12
Jehu Pond								
Mid	WBNERR	Site 4	1998-2002	2.8	2.1	9.2	0.7	25
Mid	SMAST	WB-1	2001-2003	11.9	2.0	47.1	4.2	12
Quashnet River								
Upper	WBNERR	--	--	--	--	--	--	--
Upper	SMAST	WB-07	2001-2003	22.7	4.1	168.8	2.7	11
Mid	WBNERR	Site 5	1998-2002	4.6	3.6	80.2	0.6	34
Mid	SMAST	WB-08	2001-2003	20.1	2.1	53.2	5.5	11
Lower	WBNERR	--	--	--	--	--	--	--
Lower	SMAST	WB-09	2001-2003	9.7	2.0	44.5	4.8	12

Table VII-3. Frequency (number of events during deployment) and duration (total number of days over deployment) of chlorophyll a levels above various benchmark levels from MEP continuous records from Hamblin Pond and Quashnet River.

	Start Date	End Date	Total Deployment (Days)	Duration (cumulative days) >5 ug/L (Days)	>10 ug/L (Days)	>15 ug/L (Days)	>20 ug/L (Days)	>25 ug/L (Days)	Frequency (# events) >5 ug/L (#)	>10 ug/L (#)	>15 ug/L (#)	>20 ug/L (#)	>25 ug/L (#)
Waquoit Bay Sub-Embayments													
Hamblin Pond	15-July 2002	18-Aug 2002	34.0	**Sensor Failure**									
			Mean										
			Min										
			Max										
			S.D.										
Quashnet River	15-July 2002	3-Sept 2002	49.8	35.17	18.17	12.92	9.38	6.63	76	45	22	30	25
			Mean	0.46	0.40	0.59	0.31	0.27					
			Min	0.04	0.04	0.04	0.04	0.04					
			Max	7.25	7.17	3.04	0.96	0.92					
			S.D.	0.94	1.14	0.81	0.30	0.26					

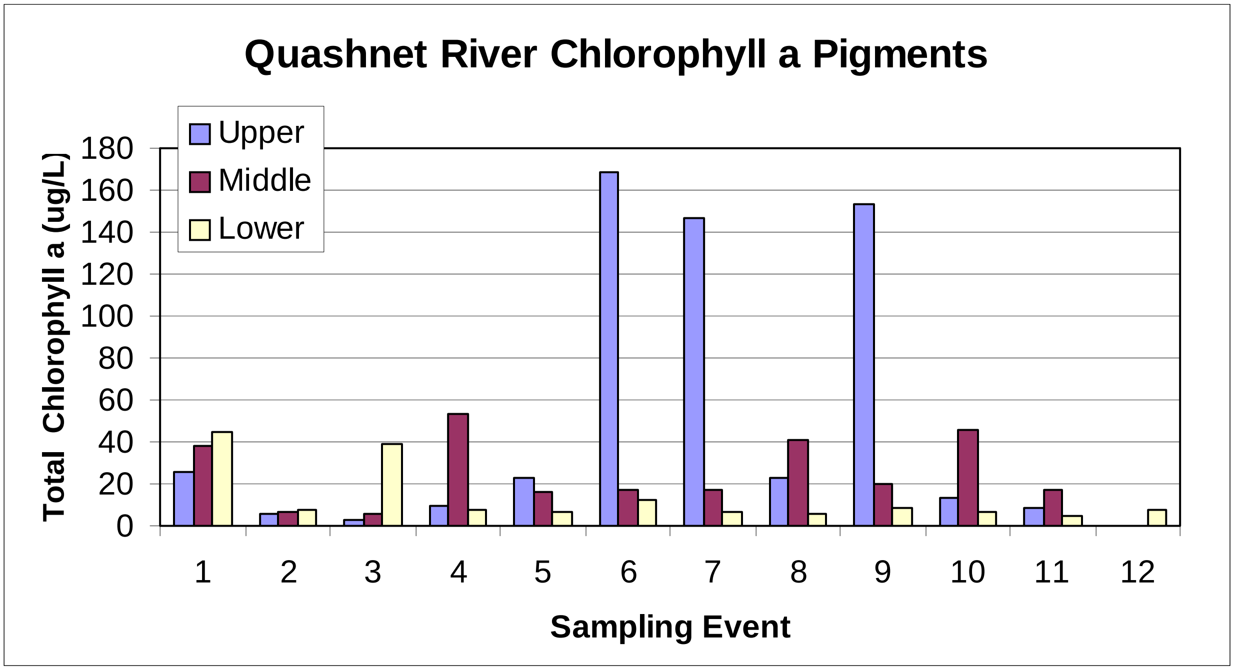


Figure VII-6. Distribution of chlorophyll a pigments within the Quashnet River Estuary from grab sampling by the Mashpee Water Quality Monitoring Program-Coastal Systems Program (SMAST) 2001 - 2003.

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data were conducted for the Popponesset Bay System by the DEP Eelgrass Mapping Program as part of the MEP Technical Team. Surveys were conducted in 1995 and 2001, as part of this program. Additional analysis of available high resolution aerial photos from 1951 was used to reconstruct the eelgrass distribution when the watershed was relatively undeveloped (estimated at <25% of today, Brawley et al. 2000). The 1951 data were only anecdotally validated, while the 1995 and 2001 maps were field validated. Additional high quality eelgrass coverage information for the eastern Waquoit Bay embayments from 1987-1992 was used in the temporal analysis of eelgrass distribution (Short and Burdick 1996). The primary use of the temporal data are 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 1987, 1988, 1989, 1992, 1995 to 2001 (Figures VII-7 and VII-8); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information is also used to determine the stability of the eelgrass community.

At present, eelgrass is not present within the Quashnet River estuary, nor was there evidence of eelgrass beds in 1951. This is consistent with observations in the 1960's of nutrient enriched conditions and macroalgae within this sub-embayment (Curley et al. 1971). In contrast, Hamblin Pond/Little River and Jehu Pond/Great River were almost completely colonized by eelgrass in the period 1951-1987 (Figures VII-7 and VII-8). The data suggest that during the 1980's eelgrass in these tributary embayments to Waquoit Bay began to significantly decline in coverage. The decline continued and by 2001 only 5%-10% of the beds remained (Table VII-4). More recent observations indicate that the residual beds are still declining in area,

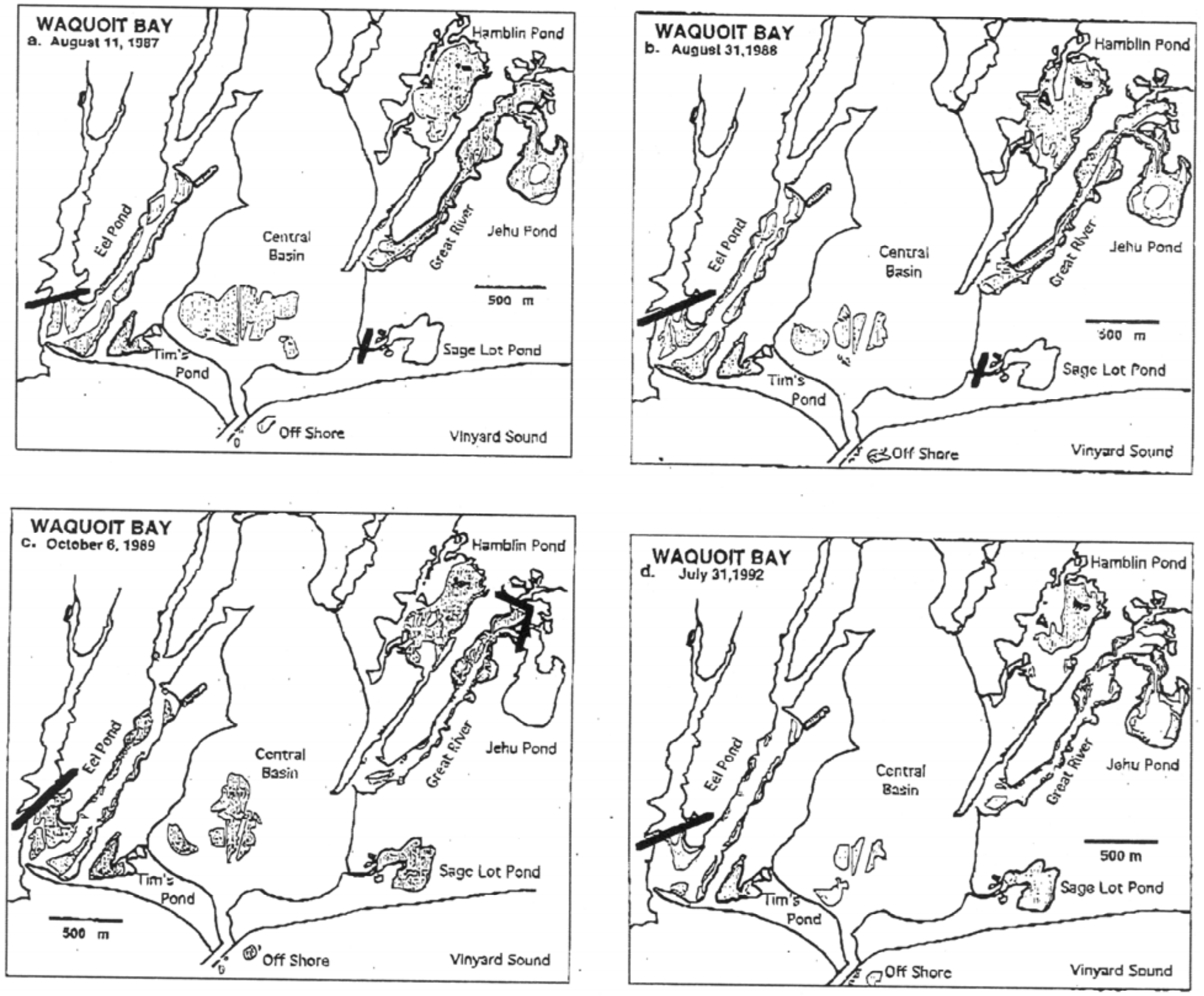


Figure VII-7. Eelgrass distribution (1987, 1988, 1989 and 1992) within the Waquoit Bay System determined with field observations (Short and Burdick 1996). Rate of loss of eelgrass is rapid in Jehu and Hamblin Ponds during this interval and continued over the next decade (Figure VII-8). Note the “hole in the Jehu Pond coverage is in the deep basin.

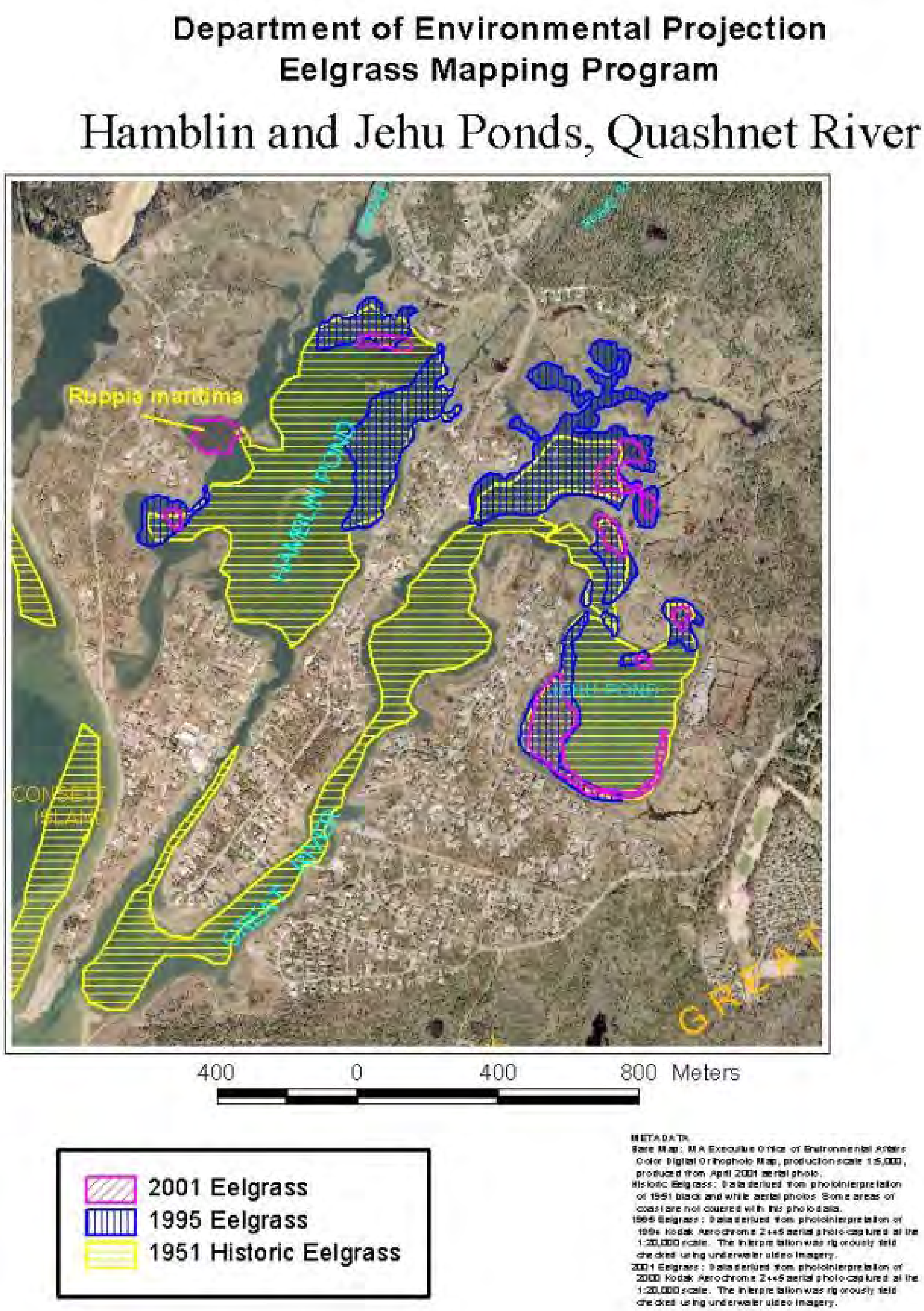


Figure VII-8. Eelgrass bed distribution within the Hamblin and Jehu Pond sub-embayment systems. The 1951 coverage is depicted by the yellow outline inside of which circumscribes the eelgrass beds. The blue (1995) and purple (2001) areas were mapped by DEP. All data were provided by the DEP Eelgrass Mapping Program.

with only marginal areas remaining. In addition, to the on-going DEP mapping, the more recent bed loss (since 2001) has been confirmed by the multiple MEP staff conducting sampling and the mooring studies. It appears that as these systems became nutrient enriched, that they could no longer support eelgrass beds. The proximate cause of loss is most likely related to nutrient related shifts in habitat quality, most significantly increased phytoplankton biomass as seen by high chlorophyll a (turbidity/shading), resulting in decreased light penetration through the watercolumn. However, it is likely that if nitrogen loading were to decrease, eelgrass could be restored in these basins to the 1951 pattern. This is supported by the fact that small areas still remain and that the decline from “full” coverage has been recent.

Table VII-4. Changes in eelgrass coverage in the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River estuaries within the Waquoit Bay System of the Towns of Mashpee and Falmouth over the past half century (DEP, C. Costello). Values base upon data in Figure VII-8.

Embayment	**1951 (acres)**	**1995 (acres)**	**2001 (acres)**	**% Difference (1951 to 2001)**
Hamblin Pond / Little River	**92.27**	**25.81**	**4.22**	95%
Jehu Pond / Great River	**115.01**	**48.1**	**12.98**	89%
Waquoit Bay & Seapit	**158.95**	**7.1**	**2.53**	98%
*No Eelgrass in the Following Embayment Areas: Quashnet River.				

It is significant that eelgrass was not detected in the Quashnet River Estuary in the 1951 data. The upper reaches of this estuary are highly altered, but the lower basin with direct communication to the Bay also did not support beds. Part of the reason, as suggested above, may be related to higher historical nitrogen loading to this estuary, but other causes such as tidal restriction cannot be evaluated at this time.

In systems like Hamblin Pond/Little River and Jehu Pond/Great River, the general pattern is for highest nitrogen levels to be found within the innermost basins, with concentrations declining moving toward the tidal inlet. This pattern is also observed in nutrient related habitat quality parameters, like phytoplankton, turbidity, oxygen depletion, etc. The consequence is that eelgrass bed decline typically follows a pattern of loss in the innermost basins (and sometimes also from the deeper waters of other basins) first. The temporal pattern is a “retreat” of beds toward the region of the tidal inlet. However, the Hamblin Pond and Jehu Pond basins also present a modification of this general pattern, in that eelgrass beds are typically lost from the deeper waters first, due to shading effects resulting from the increased phytoplankton production. This pattern is clearly seen in Jehu Pond, where coverage was virtually complete in 1951 (Figure VII-8) but a “hole” was clearly present in 1987 (Figure VII-7), which expanded through 1992, 1995, and 2001. The two patterns of loss combine to generate the overall shifts in eelgrass distribution in these systems. Lowering of nitrogen loads to these estuaries would likely result in a reversal of this pattern with the shallower areas being the first to recolonize.

Other factors which influence eelgrass bed loss in embayments may also be in the Hamblin Pond and Jehu Pond estuaries, though the loss seems completely in-line with nitrogen enrichment. 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 the Ponds support few 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 in the regions of recent loss.

Overall the mapping data indicate that nitrogen management of the Hamblin Pond and Jehu Pond estuaries should target eelgrass restoration. Based upon the 1951-1987 coverage data, it appears that on the order of 200 acres of eelgrass might be potentially recoverable in these estuarine sub-embayments, if nitrogen management alternatives were implemented (Table VII-4).

The relative pattern of these data are consistent with the results of the benthic infauna analysis and the observed eelgrass loss is typical of nutrient enriched shallow embayments.

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 13 locations throughout the eastern sub-embayments to Waquoit Bay (Figure VII-9). In all areas and particularly those that do not support eelgrass beds (hence most of the study areas), benthic animal indicators can be used to assess the level of habitat health from healthy (low organic matter loading, high D.O.) to highly stressed (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of the habitat in which they live. Benthic animal species from sediment samples are identified and ranked as to their association with nutrient related stresses, such as organic matter loading, anoxia, and dissolved sulfide. The analysis is based upon life-history information and animal-sediment relationships (Rhoads and Germano 1986). Assemblages are classified as representative of healthy conditions, transitional, or stressed conditions. Both the distribution of species and the overall population density are taken into account, as well as the general diversity and evenness of the community. It should be noted that, given the loss of eelgrass beds, the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River estuaries are clearly impaired by nutrient overloading throughout their tidal reaches. However, to the extent that a system can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

Analysis of the evenness and diversity of the benthic animal communities was also used to support the density data and the natural history information (Table VII-5). The evenness statistic can range from 0-1 (one being most even), while the diversity index does not have a theoretical upper limit. Highest quality habitat areas, as shown by the oxygen and chlorophyll records and eelgrass coverage, generally have the highest diversity (generally >3) and evenness (~0.7). The converse is also true, with poorest habitat quality found where diversity is <1 and evenness is <0.5.

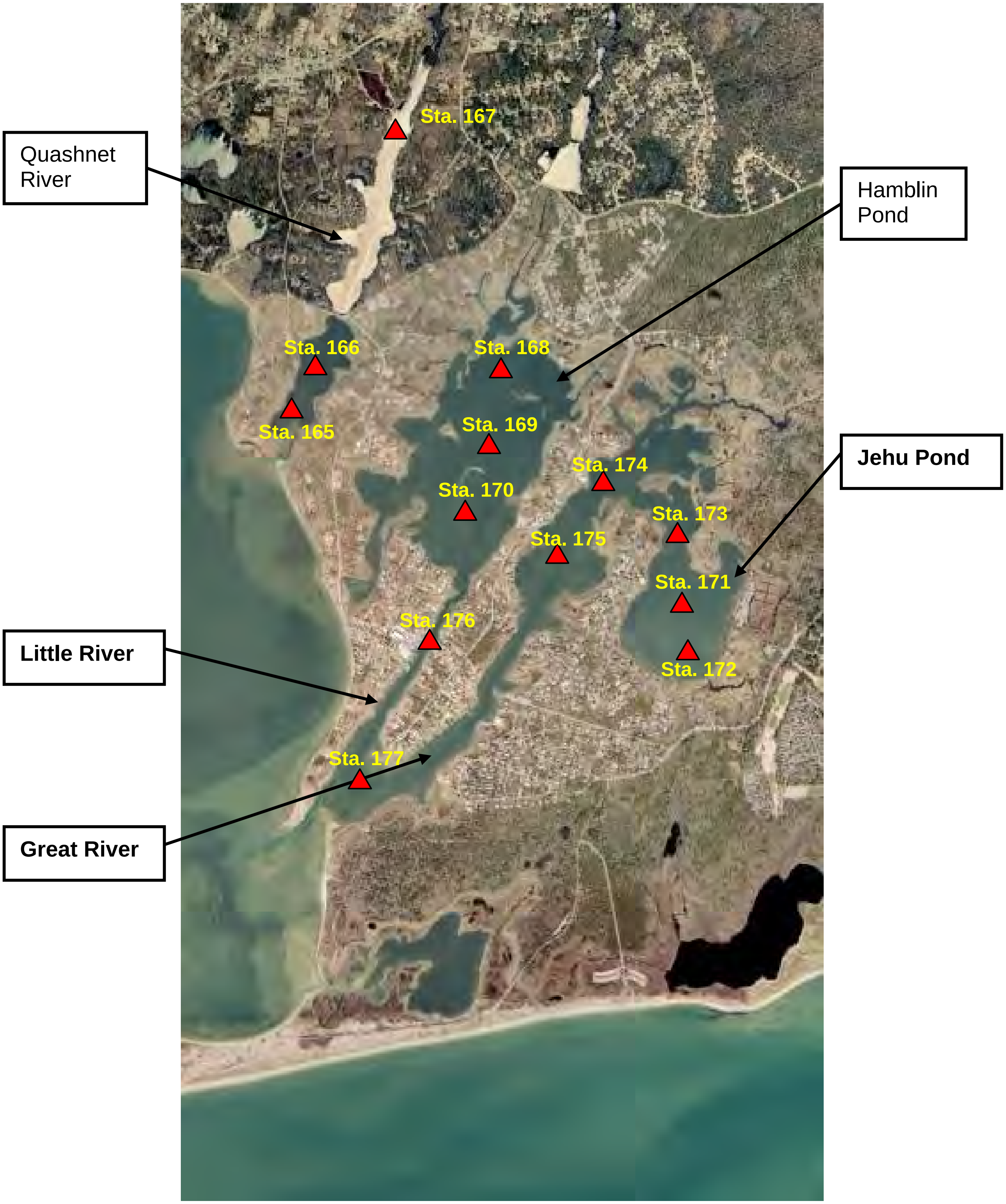


Figure VII-9. Aerial photograph of the eastern embayments within the Waquoit Bay System showing location of benthic sampling stations (red symbols) for infaunal community assessments.

Table VII-5. Benthic infaunal community data (May 2003) for the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River estuaries within the Waquoit Bay System. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.0625 m2).

Estuary	Location	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Waquoit Bay System						
Quashnet R. Upper	Station 165	1	18	N/A	0	N/A
Quashnet R. Lower	Station 166	1	4	N/A	0	N/A
Quashnet R. Lower	Station 167	0	0	N/A	N/A	N/A
Hamblin P. Upper	Station 168	10	496	9	2.39	0.72
Hamblin P. Mid	Station 169	4	26	N/A	1.57	0.79
Hamblin P. Lower	Station 170	18	793	9	2.42	0.58
Little River Mid	Station 176	19	3170	10	2.74	0.65
Jehu P. Upper	Station 171	4	34	N/A	1.74	0.87
Jehu P. Mid	Station 172	6	144	N/A	1.79	0.69
Jehu P. Lower	Station 173	4	401	4	1.38	0.69
Great R. Upper	Station 174	10	1068	8	2.13	0.64
Great R. Upper/Mid	Station 175	9	2148	6	1.81	0.57
Grt/Little Confluence	Station 177	4	14	N/A	1.84	0.92

Clearly, the Quashnet River Estuary is consistent with the above metrics with only a single species being found, hence a diversity equal to 0. The severely degraded nature of this habitat is underscored by the virtual absence of an infaunal community with only 18, 4, and 0 individuals being found at the three sites, compared to 100's to 1000's being found at healthy sites. The Jehu Pond and Hamblin Pond systems showed infaunal community habitats ranging from healthy to significantly impaired. There appears to be a gradient in habitat quality within the Jehu Pond/Great River Estuary. The basin of Jehu Pond supported a low number of species (4-6) and total individuals <150 at two of three stations and low diversity at all stations (<1.8). However, the Great River showed markedly better habitat, with 9-10 species and >1000 individuals per sample at each station, and slightly higher diversity. Hamblin Pond/Little River showed a similar pattern, although with much better habitat quality. Only the mid basin of Hamblin Pond was significantly impaired with all of the other stations showing 10-19 species and 500-3200 individuals per sample. Diversity was also high, generally ≥2.4. Most likely deposition within the mid basin of Hamblin Pond and subsequent organic matter loading effects are responsible for the observations at this station. However, the other areas of this system appear to support healthy benthic habitat (Lower Hamblin Pond and Little River) or habitat that is only moderately impaired (Upper Hamblin Pond).

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 the 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 by the MEP Team to support threshold development for the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River Estuaries within the Waquoit Bay System, and were discussed in Chapter VII. Nitrogen threshold development builds on these data and links habitat quality to summer water column nitrogen levels from the nitrogen modeling and baseline Mashpee Water Quality Monitoring Program (Chapter VI). At present these three estuaries are generally showing impaired habitat quality resulting from nitrogen enrichment (Chapter VII, Table VIII-1).

Eelgrass: Both the Hamblin Pond/Little River and Jehu Pond/Great River were almost completely colonized by eelgrass in the period 1951-1987 (Figures VII-7, VII-8). The data suggest that during the 1980's eelgrass in these tributary embayments to Waquoit Bay began to significantly decline in coverage. The decline has continued, with less than 5%-10% of the beds remaining today (Table VII-4). It appears that as these systems became nutrient enriched, these sites could no longer support eelgrass beds. The proximate cause of loss is most likely due to nutrient related shifts in habitat quality, most significantly the high chlorophyll a (turbidity/shading) and low dissolved oxygen levels. However, it is likely that if nitrogen loading were to decrease, eelgrass beds could be restored in these basins. Based upon the 1951-1987 coverage data, it appears that on the order of 200 acres of eelgrass might be recoverable in these estuaries, if nitrogen management alternatives were implemented (Table VII-4). This is supported by the fact that small areas of eelgrass still remain and that the decline from "full" coverage has been recent (~20 yrs). Given the significant loss of coverage, but the persistence of small patches of eelgrass in both of these systems, it appears that these estuaries are moderately impaired by nitrogen enrichment based upon this indicator alone. It is clear that nitrogen threshold development for the Hamblin Pond/Little River and Jehu Pond/Great River Estuaries should target restoration and maintenance of eelgrass habitat.

At present, eelgrass is not present within the Quashnet River estuary, nor was there evidence of eelgrass beds in 1951. This is consistent with observations in the 1960's of nutrient enriched conditions and macroalgae within this sub-embayment (Curley et al. 1971). In fact, large macroalgal accumulations occur within this estuary today and are indicative of severe degradation by nitrogen enrichment. The upper reaches of the Quashnet River Estuary have been highly man-altered which may relate to historical absence of eelgrass, but the lower basin proximal to the Bay also did not historically support beds. Part of the reason, as suggested above, may be related to higher historical nitrogen loading to this estuary, but other causes such as tidal restriction cannot be evaluated at this time. The Quashnet River inlet to Waquoit Bay has significant sediment movement which may periodically restrict tidal flows. To the extent that this has occurred in the past, it may also partially relate to the lack of historical eelgrass beds in this lower basin. It may also be that this system is not supportive of this type of habitat due to its physical properties, and stronger estuarine circulation than the other sub-embayments. Based upon history of nitrogen enrichment, the absence of eelgrass in the 1951, the Quashnet

River Estuary appears to be significantly impaired/degraded relative to eelgrass. However, given the uncertainties and the lack of historical support for eelgrass in this system, it is not prudent to target restoration thresholds on this parameter. Habitat quality for infaunal communities appears to be the threshold based upon the available data and uncertainties.

Water Quality: The water quality indicators that are central to evaluating the nutrient related habitat health for eelgrass and benthic infaunal communities are the degree of oxygen depletion in bottom waters and the level of phytoplankton biomass (blooms) as determined from total chlorophyll a measurements.

The dissolved oxygen records for the tidally influenced lower Quashnet River and the upper region of Hamblin Pond indicate that these sub-embayments currently maintain a high and moderate level of oxygen stress, respectively. Jehu Pond showed a high level of oxygen depletion, at a level which will impair habitat quality, with dissolved oxygen levels periodically approaching anoxia. Nitrogen enrichment of embayment waters can manifest itself in the dissolved oxygen record, both through oxygen depletion and through the magnitude of the daily excursion. This phenomenon is best seen in the Quashnet River record, where dissolved oxygen levels frequently become significantly depleted during the night and reach levels in excess of atmospheric saturation during the day time (Figure VII-3). From the oxygen records it is clear that, after Jehu Pond, the Quashnet River has the greatest extent of oxygen depletion. Additionally, the oxygen excursion indicates a high degree of nutrient enrichment (as is supported by the chlorophyll a data). Note that this data is from the lower part of this system, which has the highest water quality, but still the oxygen levels are <4 mg L^{-1} almost 10% of the time.

Based upon measured total chlorophyll a pigments (sum of chlorophyll a and its immediate breakdown product, pheophythin a, as a better indicator of bloom conditions) it is clear that the Quashnet River is highly eutrophic with total chlorophyll a levels in the upper and mid regions averaging >20 ug L^{-1} (Table VII-2 SMAST data). The moored chlorophyll sensor showed similarly high values (Table VII-3). Phytoplankton blooms appear to be generated within the upper and mid basins of the Quashnet, most likely as a result of the high nitrogen loading to the headwaters via the Quashnet River freshwater discharge. Exceedingly large blooms were observed within the upper Quashnet River basin(>140 ug L-1), with very large blooms (>40 ug L-1) also being observed in the mid reach of the estuary (bridge divides lower from mid reaches). Based upon all of the chlorophyll and oxygen data it appears that the Quashnet River Estuary is showing oxygen stress throughout its reach and it is likely that the level of depletion is higher in the upper and mid reaches than in the lower basin, consistent with the distribution of phytoplankton biomass.

Jehu and Hamblin Ponds support moderate to high total chlorophyll levels, averaging 11.9 and 7.4 ug L-1, respectively. Jehu Pond appears to be showing more nutrient enrichment than Hamblin Pond, both on average and relative to the size of the blooms (Table VII-2, maximum values). The high phytoplankton biomass in Jehu Pond is consistent with the observation of oxygen stress in this system. The moderate total chlorophyll levels in Hamblin Pond are consistent with its moderately good oxygen status. The agreement between the chlorophyll land oxygen levels in these Pond basins is likely the result of their physical structure.

The dissolved oxygen and chlorophyll a data alone indicate a clear pattern of nutrient related habitat quality. At present, the Quashnet River estuary is showing poor oxygen status (based upon depletions, daily excursions, and the mooring in the lower basin) and large phytoplankton blooms. While it appears to be stressed throughout, there is a clear gradient from hyper-eutrophic in the upper regions to eutrophic in the lower basin. Jehu Pond is also showing nitrogen enriched conditions, with periodic hypoxia/anoxia in the basin and high phytoplankton biomass. Hamblin Pond is showing the best nutrient related habitat quality, based both upon its moderately good oxygen conditions and moderate phytoplankton biomass. Based only upon the dissolved oxygen and chlorophyll data the nutrient related habitat quality of the three estuaries to eastern Waquoit Bay can be classified is as follows: Quashnet River estuary - Significantly Impaired, Jehu Pond - Moderately/Significantly Impaired, Hamblin Pond - Moderately Impaired.

Infaunal Communities: Clearly, the Quashnet River Estuary is a severely degraded habitat relative to supporting benthic infaunal communities supportive of only a single species, hence a diversity equal to 0. The poor quality of this habitat is underscored by the virtual absence of an infaunal community with only between 0 and 18 individuals being found per 0.0625 m^2, compared to 100's to 1000's being found at healthy sites. The Jehu Pond and Hamblin Pond systems showed infaunal community habitats ranging from healthy to significantly impaired. There appears to be a gradient in habitat quality within the Jehu Pond/Great River Estuary. The basin of Jehu Pond supported a low number of species (4-6) and total individuals <150 at 2 of 3 stations and low diversity at all stations (<1.8). However, the Great River was markedly better habitat with 9-10 species and >1000 individuals per sample at each station with slightly higher diversity. Hamblin Pond/Little River showed a similar pattern, although with much better habitat quality. Only the mid basin of Hamblin Pond was significantly impaired with all of the other stations showing 10-19 species and 500-3200 individuals per sample and high diversity ($\geq$2.4). The other areas of this system appear to support healthy benthic habitat (Lower Hamblin Pond, Little River) or habitat that is only moderately impaired (upper Hamblin Pond).

Overall, all of the indicators show consistent patterns within each of estuaries, Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River within the Waquoit Bay System. The results of the assessment of nutrient related habitat quality for each estuary is summarized in Table VIII-1.

Table VIII-1. Summary of Nutrient Related Habitat Health for the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River estuaries within the Waquoit Bay System, based upon assessment data presented in Chapter VII.

Health Indicator	Estuary within the Waquoit Bay System					
	Quashnet River		Hamblin Pond/Little R.		Jehu Pond/Great R.	
	Upper	Lower	Hamblin Pond	Little River	Jehu Pond	Great River
Dissolved Oxygen	SI	SI	MI	MI	SI	MI
Chlorophyll	SD	SI	MI		MI/SI	
Macroalgae	SD	SD				
Eelgrass	SI/SD	SI/SD[1]	MI	MI	MI	MI
Infaunal Animals	SD	SD	MI	H	SI	MI
Overall:	**SD**	**SI/SD**	**MI**	**H/MI**	**SI**	**MI**

1 – eelgrass lost prior to 1951
H = healthy habitat conditions; MI = Moderate Impairment; SI = Significant Impairment; SD = Severe Degradation

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.

Within the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River Estuaries within the Waquoit Bay System, it was necessary to select 3 sentinel locations. The Quashnet River Estuary operates independent from the Hamblin Pond and Jehu Pond Estuaries, except as they share common source waters from Waquoit Bay. Their interaction is primarily through their effect on the nitrogen level with Waquoit Bay. The sentinel system within the Quashnet River Estuary was set within the upper/mid basin (region above the bridge). Achieving the nitrogen threshold at this station will also improve benthic habitat in the lower basin. Since there is no historical evidence that the Quashnet River Estuary supported eelgrass, the threshold nitrogen concentration was based upon restoring benthic habitat at the sentinel station. The target nitrogen concentration to restore infaunal habitat is based upon the high quality infaunal sites in lower Hamblin Pond and in Little River (Stations 176 and 170, Figure VII-9). The tidally averaged nitrogen levels at these sites are 0.498 and 0.524 mg N L^{-1}, respectively. These values are consistent with the infaunal guidance levels within the Popponesset Bay sub-embayments of 0.5 to 0.4 mg N L^{-1} (0.5 mg N L^{-1} being the upper threshold value). Based upon these data a conservative estimate for the infaunal threshold for the Quashnet River Estuary is 0.50 mg N L^{-1}, with 0.52 likely to represent a slight stress, but still high quality habitat.

Within the Hamblin Pond/Little River and Jehu Pond/Great River Estuaries the sentinel locations were placed within the pond basins. The target nitrogen threshold focuses on eelgrass restoration of these systems. Given that the nitrogen gradients, with the ponds having the highest nitrogen levels within their respective estuarine sub-embayment, achieving the nitrogen target in the ponds will necessarily result in high quality habitat in the down-gradient reaches. However, setting the threshold for these ponds is not straight-forward. In other systems, a target nitrogen level of 0.38 mg N L^{-1} has been supported by the on-site data and assessments. It appears that this level would be restorative of eelgrass in the Jehu Pond and Hamblin Pond estuaries, as the few diminishing eelgrass patches in the main basin of Waquoit Bay, near the inlet persist at 0.395 mg N L^{-1}, relatively consistent with this threshold. A threshold of 0.38 mg N L-1 is being evaluated for the main basin of Waquoit Bay and will be thoroughly addressed when the whole of the system is re-addressed by MEP. However, eelgrass was almost completely lost from the main basin prior to significant loss from the Hamblin Pond and Jehu Pond Estuaries in the 1980's (Figures VII-7 and VII-8). Therefore, another approach to developing the threshold nitrogen level for these Ponds relates to the nitrogen level in the main bay, which is also their source water (boundary condition). Based upon a main bay boundary condition of 0.38 mg N L^{-1} (upper eelgrass threshold) the nitrogen levels in the Ponds would necessarily have been >0.38 mg N L^{-1}, given the gradients established by the interplay of loading and hydrodynamics. This is consistent with the existence of a few diminishing small patches of eelgrass at nitrogen levels on the order of 0.5 mg N L^{-1} in these ponds in 2001-2003. Note that since eelgrass can persist at nitrogen levels that are non-supportive of healthy beds, a value of 0.5 mg N L-1 is beyond the supportive nitrogen threshold.

To refine the nitrogen threshold for Jehu and Hamblin Ponds, modeling was conducted. The goal of this effort was to reconcile nitrogen levels to historical shifts in eelgrass distribution. The concept was to use conservative estimates of nitrogen loads and concentrations to estimate nitrogen levels prior to the eelgrass loss in the main bay and ponds. The details of the assumptions and modeling are presented in Section VIII.3. Based upon the modeling it appears that Jehu Pond could support eelgrass at a nitrogen threshold of 0.446 mg N L^{-1}. This is above the 0.38 mg N L-1 threshold likely for the main bay (and utilized for Stage Harbor and Popponesset Bay), but lower than the 0.527-0.552 found in the Bassing Harbor System. This level for Jehu Pond is also consistent with the pattern and timing of eelgrass loss throughout the Waquoit Bay System. Although Hamblin Pond is similar to Jehu Pond in gross structure, it has very different loading and attenuation characteristics. The result is that the structure of the system produces much lower nitrogen levels so a threshold of 0.38 mg N L-1 was selected to allow for uncertainties.

As will be discussed below, it will not be able to achieve the target nitrogen levels for the Quashnet River, Hamblin Pond/Little River or Jehu Pond/Great River Estuary without lowering the nitrogen level within the main basin of Waquoit Bay. At present the flooding waters from Waquoit Bay are sufficiently nitrogen enriched that even modest nitrogen loads from the watersheds to these estuaries exceed nitrogen targets. In fact, the flood waters from the main basin currently exceed the 0.38 mg N L^{-1}.

VIII.3 DEVELOPMENT OF TARGET NITROGEN LOADS

The tidally averaged total nitrogen thresholds derived in Section VIII-2 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel region for the Quashnet River, Hamblin Pond/Little River and Jehu Pond/Great River Estuaries. 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. It is also important to note that each of the three estuaries will be re-evaluated and integrated into the assessment and modeling of the whole of the Waquoit Bay System.

The scenario approach to Quashnet River, Hamblin Pond/Little River, and Jehu Pond/Great River estuarine sub-embayments which exchange tidal waters with Waquoit Bay is different than for estuaries which receive tidal waters directly from Nantucket Sound, Buzzards Bay, the Atlantic Ocean, or other open waters. In the case of these tributary estuaries or sub-embayments to Waquoit Bay, the incoming tidal waters are nitrogen enriched to the point that the Waquoit Bay main basin no longer supports healthy eelgrass beds. This contrasts with an open water boundary which typically has high quality waters with a low nitrogen burden. Therefore the first Scenario (A) was conducted to determine if the thresholds could be met with the current nitrogen level in the incoming tidal waters from the main basin and a total removal of watershed nitrogen loading from septic systems, i.e. septic loading set at “0”. The nitrogen loads associated with Scenario A for Jehu Pond and Hamblin Pond Estuaries are presented in Tables VIII-2, VIII-3, VIII-4, and VIII-5). Scenario A was conducted to determine the role of the Waquoit Bay water quality to the health of these systems. Scenario A (Figures VIII-1 and VIII-2) indicates that for Jehu Pond the nitrogen level is lowered from 0.603 mg N L^{-1} (present) to 0.466 mg N L^{-1} (all septic load removed), while only the lower Great River achieved a potential eelgrass threshold (0.38 mg N L^{-1}). However, note that nitrogen levels supportive of healthy infaunal habitat would be achieved throughout the Jehu Pond sub-embayment. Based upon this modeling scenario it appears that the only realistic mechanism for reaching 0.38 mg N L^{-1} within Jehu Pond would require nitrogen management relative to the Waquoit Bay basin in concert with nitrogen reductions within this sub-watershed.

Table VIII-2. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of **Present Conditions** and **Scenarios A and B** loading to the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg/day)	Scen. A septic load (kg/day)	Scen. A septic load % change	Scen. B septic load (kg/day)	Scen. B septic load % change
Hamblin Pond	3.36	0.00	-100%	0.87	-74%
Upper Hamblin Pond	1.26	0.00	-100%	0.33	-74%
Little River	0.92	0.00	-100%	0.24	-74%
Lower Great River	2.35	0.00	-100%	0.00	-100%
Upper Great River	0.36	0.00	-100%	0.00	-100%
Jehu Pond	2.65	0.00	-100%	0.00	-100%
Surface Water Sources					
Red Brook	3.24	0.00	-100%	0.81	-75%

Table VIII-3. Comparison of sub-embayment attenuated ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of **Present** and loading **Scenarios A and B** to the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.

sub-embayment	present N load (kg/day)	Scen. A N load (kg/day)	Scen. A load % change	Scen. B N load (kg/day)	Scen. B load % change
Hamblin Pond	3.84	0.47	-88%	1.34	-65%
Upper Hamblin Pond	1.54	0.28	-82%	0.61	-60%
Little River	1.11	0.19	-83%	0.43	-61%
Lower Great River	2.95	0.60	-80%	0.60	-80%
Upper Great River	0.68	0.32	-53%	0.32	-53%
Jehu Pond	3.61	0.96	-73%	0.96	-73%
Surface Water Sources					
Red Brook	3.88	0.64	-83%	1.45	-63%

Table VIII-4. Modeling **Scenarios A** sub-embayment and surface water loads used for total nitrogen modeling of the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg N/day)	direct atmospheric deposition (kg N/day)	benthic flux (kg N/day)
Hamblin Pond	0.47	1.53	-3.53
Hamblin Pond Cut	-	-	1.30
Upper Hamblin Pond	0.28	0.06	-3.17
Little River	0.19	0.16	2.45
Lower Great River	0.59	0.75	7.12
Upper Great River	0.32	0.55	6.75
Jehu Pond	0.96	0.67	7.64
Surface Water Sources			
Red Brook	0.64	-	-

Table VIII-5. Modeling **Scenario B** sub-embayment and surface water loads used for total nitrogen modeling of the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg N/day)	direct atmospheric deposition (kg N/day)	benthic flux (kg N/day)
Hamblin Pond	1.34	1.53	-4.04
Hamblin Pond Cut	-	-	1.50
Upper Hamblin Pond	0.61	0.06	-3.63
Little River	0.43	0.16	2.73
Lower Great River	0.60	0.75	7.12
Upper Great River	0.32	0.55	6.75
Jehu Pond	0.96	0.67	7.64
Surface Water Sources			
Red Brook	1.45	-	-

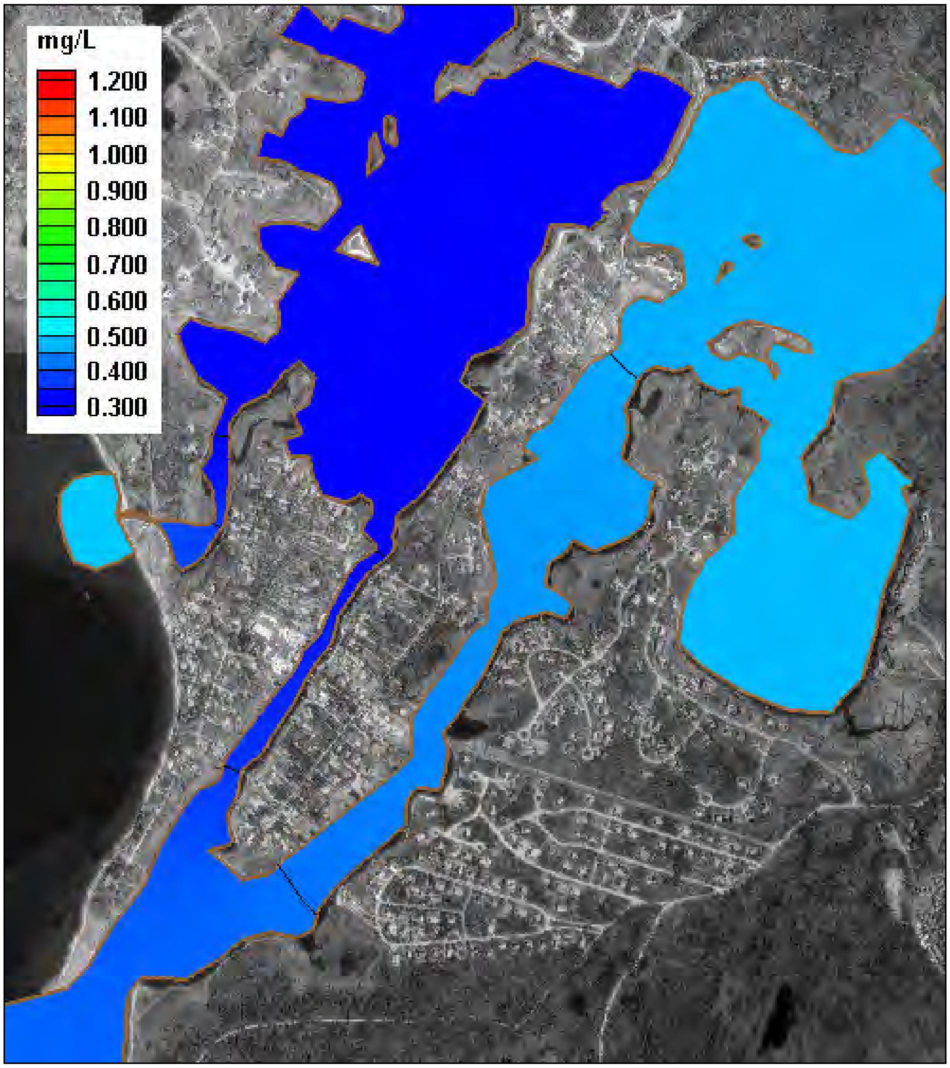


Figure VIII-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System, for **Scenario A** loading conditions (100% septic removal, with present Waquoit Bay boundary condition).

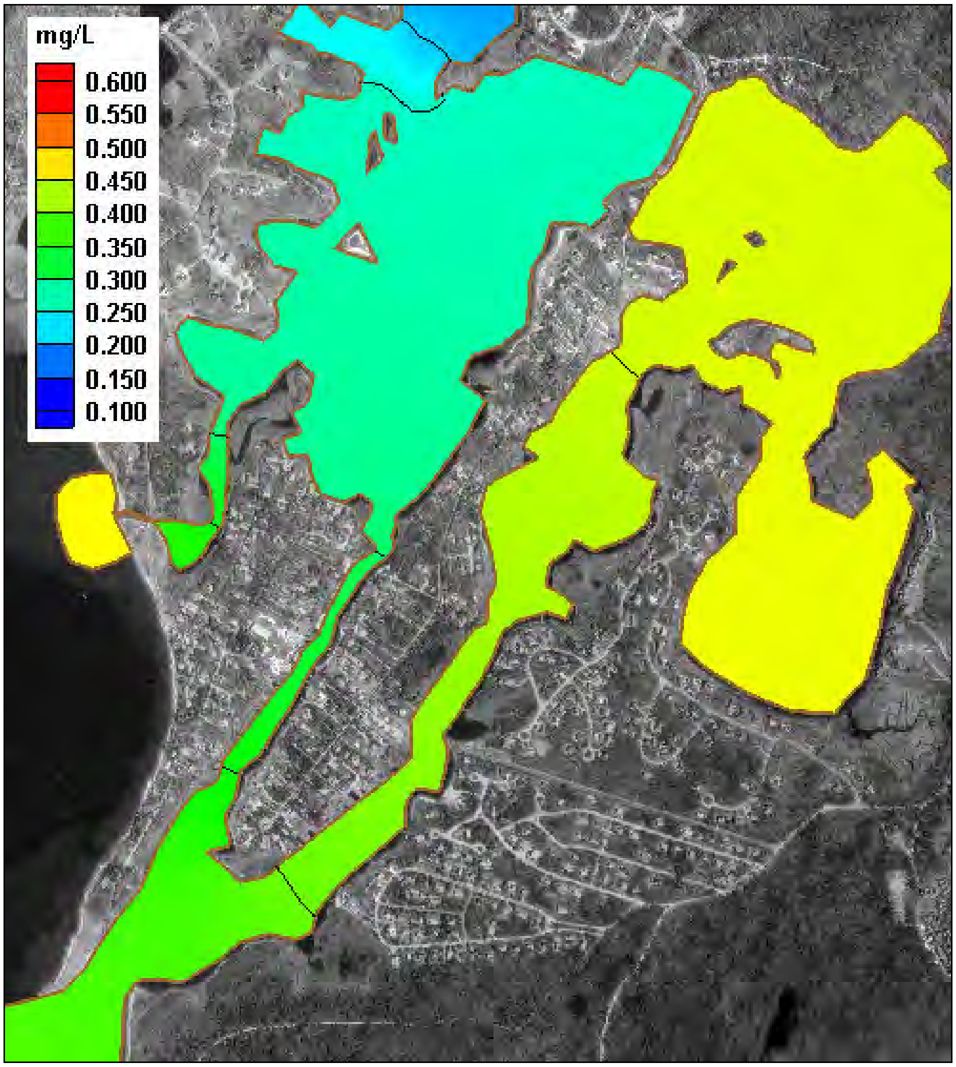


Figure VIII-2. Same results as for Figure VIII-1, but shown with finer contour increments for emphasis. Contour plot of modeled total nitrogen concentrations (mg/L) in the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System, for **Scenario A** loading conditions (100% septic removal, with present Waquoit Bay boundary condition).

As stated in Section VIII.2 above, a higher nitrogen threshold (i.e. > 0.38 mg N L^{-1}) for the Jehu Pond sentinel location is supported by the pattern of historical eelgrass distribution and loss. Since eelgrass was first lost from the central basin of Waquoit Bay it is possible to derive a higher yet still very conservative threshold for Jehu Pond. The approach (Scenario B shown in Figures VIII-3 and VIII-4) taken was to set the boundary condition in the main basin of Waquoit Bay at 0.35 mg N L^{-1}, a level unquestionably supportive of eelgrass. The second step was to reduce the watershed nitrogen load by about two-thirds of present day loading (Table VIII-3). This loading reduction is supported by a temporal analysis conducted by Brawley et al. (2002), where it was found that in the 1960-1975 time frame, loading to these estuaries and to the Waquoit Bay System was about half of present watershed loading. The additional removal in Scenario B was to account for the different watershed delineations and watershed modeling used in the Brawley et al. (2002) study. Under these very conservative conditions, the nitrogen level attained in Jehu Pond is 0.446 mg N L^{-1} (Table VIII-6). Therefore, we conclude that the nitrogen target restorative of eelgrass within this estuary is 0.446 mg N L^{-1}. Upon review of Scenario B it appears that the 0.38 mg N L-1 target can still be applied to Hamblin Pond, since these conditions will be met in targeting the Jehu Pond eelgrass restoration. A refinement of the balancing of nitrogen loads between the Jehu Pond and Hamblin Pond sentinel stations (there is some mixing of waters) should be conducted as part of the Alternatives Analysis underway with the Town of Mashpee/DEP.

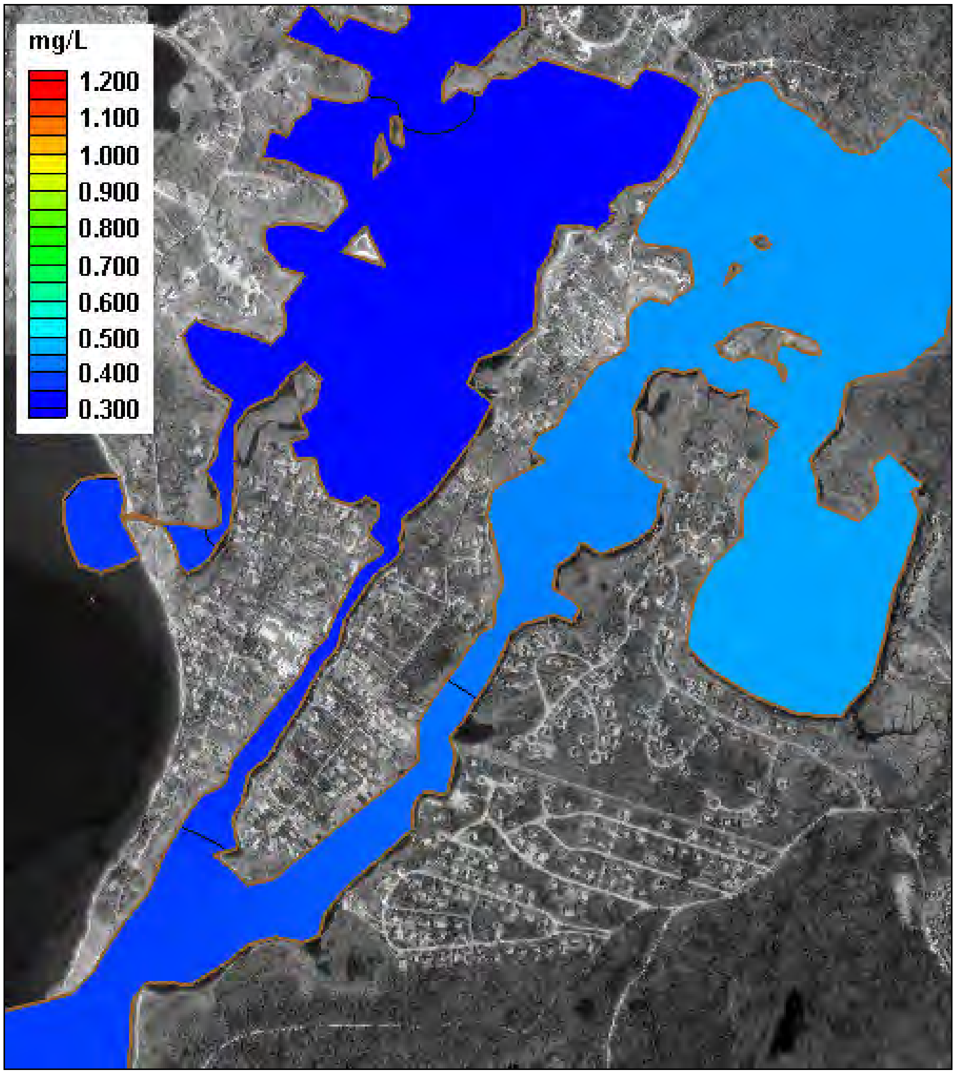


Figure VIII-3. Contour plot of modeled total nitrogen concentrations (mg/L) in the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System, for **Scenario B** loading conditions (75% septic removal in Hamblin Pond, 100% septic removal in Jehu Pond, with 0.350 mg/L Waquoit Bay boundary condition).

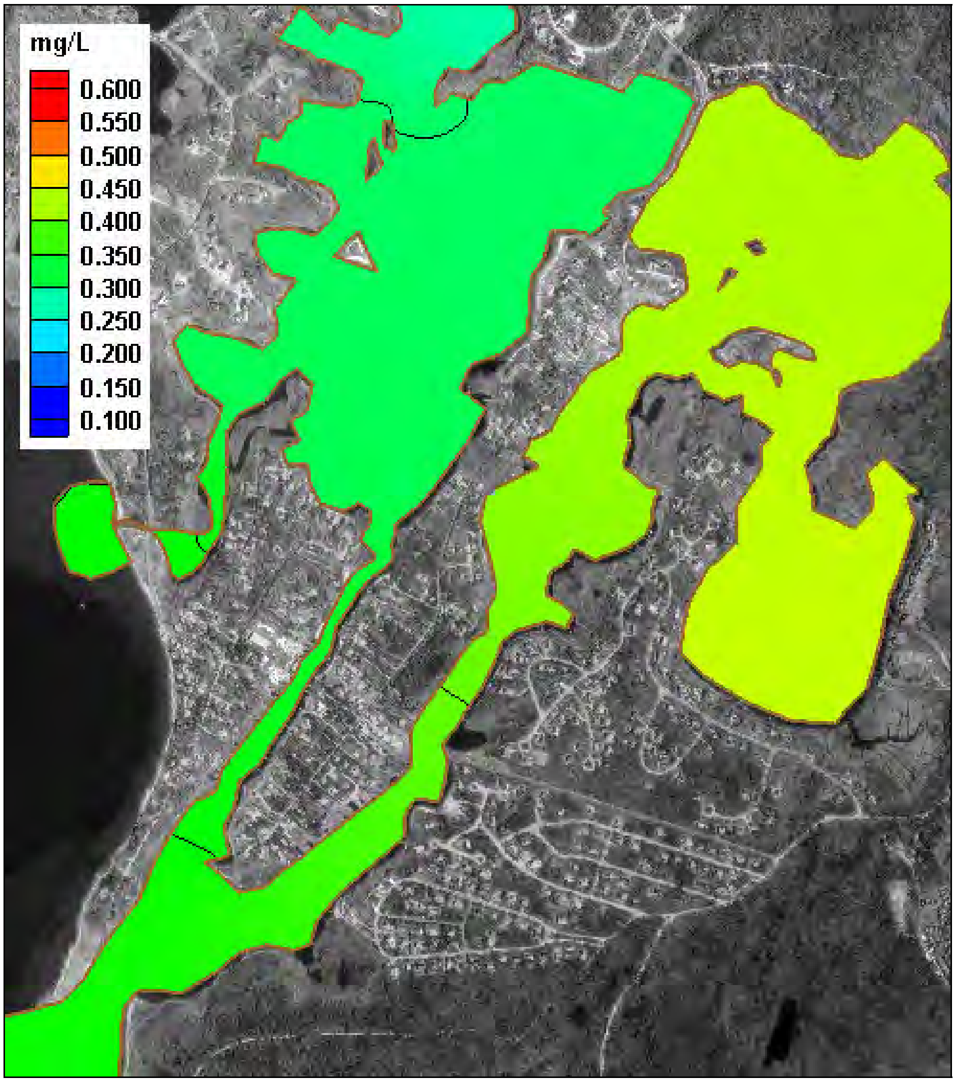


Figure VIII-4. Same results as for Figure VIII-3, but shown with finer contour increments for emphasis. Contour plot of modeled total nitrogen concentrations (mg/L) in the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System, for **Scenario B** loading conditions (75% septic removal in Hamblin Pond, 100% septic removal in Jehu Pond, with 0.350 mg/L Waquoit Bay boundary condition).

Table VIII-6. Modeled TN concentrations and percent change for present conditions and nitrogen loading **Scenarios A and B**, for the Jehu Pond and Hamblin Pond Estuaries within the Waquoit Bay System. Percent change represents the change in total N concentration.

	Present	Scenario A		Scenario B	
	(mg/L)	(mg/L)	% change	(mg/L)	% change
Jehu Pond	0.603	0.466	-22.7%	0.446	-26.0%
upper GR	0.560	0.442	-21.1%	0.421	-24.8%
lower GR	0.451	0.383	-15.1%	0.357	-20.8%
upper HP	0.528	0.265	-49.8%	0.305	-42.3%
Hamblin P cut	0.512	0.418	-18.4%	0.349	-31.7%

The Quashnet River Estuary nitrogen loading analysis was conducted in the same manner as Jehu Pond and Hamblin Pond. However, the target nitrogen level was based upon restoring healthy habitat for infaunal communities (0.5 to <0.52 mg N L^{-1}). In Scenario A (current boundary condition and removal of all septic nitrogen loading from the watershed, Table VIII-7, VIII-8, VIII-9, and VIII-10) the target nitrogen level was achieved (Figures VIII-5 and VIII-6). Therefore it was possible to move to Scenario B (Figures VIII-7 and VIII-8), where the boundary condition is lowered to 0.35 mg L-1 (as was required to meet the threshold for Jehu Pond) and then removed nitrogen loading step-wise until the infaunal habitat target for the upper/mid basin was achieved (Table VIII-11). The average of the upper and mid stations was used for this analysis. Achieving this target should restore infaunal habitat in the lower basin and possibly eelgrass to the extent that the structure and sediments of this system will support it. A refinement of the nitrogen loads to the Quashnet River Estuary based upon a range of nitrogen levels in the tidal waters should be assessed as part of the Alternatives Analysis underway with the Town of Mashpee/DEP. However, allowing for a higher nitrogen level in inflowing tidal waters will require a greater reduction in the watershed nitrogen load to meet the target levels in the sentinel basin.

Table VIII-7. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of **Present Conditions** and modeling **Scenarios A and B** loading of the Quashnet River Estuary within the Waquoit Bay System. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present septic load (kg N/day)	Scen. A septic load (kg N/day)	Scen. A septic load % change	Scen. B septic load (kg N/day)	Scen. B septic load % change
Upper Quashnet River	1.80	0.00	-100%	0.59	-67.0%
Lower Quashnet River	0.57	0.00	-100%	0.19	-67.0%
Surface Water Sources					
Moonakis River	12.59	0.00	-100%	4.16	-67.0%

Table VIII-8. Comparison of sub-embayment attenuated ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of **Present** and modeling **Scenarios A and B** loading to the Quashnet River Estuary within the Waquoit Bay System. These loads do not include direct atmospheric deposition (onto the sub-embayment surface), benthic flux, runoff, or fertilizer loading terms.

sub-embayment	present load (kg/day)	Scen. A load (kg/day)	Scen. A load % change	Scen. B load (kg/day)	Scen. B load % change
Upper Quashnet River	2.16	0.36	-83.3%	0.95	-55.8%
Lower Quashnet River	0.79	0.22	-72.2%	0.41	-48.3%
Surface Water Sources					
Moonakis River	23.00	10.41	-54.8%	14.56	-36.7%

Table VIII-9. Modeling **Scenario A** sub-embayment and surface water loads used for total nitrogen modeling of the Quashnet River Estuary within the Waquoit Bay System, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg N/day)	direct atmospheric deposition (kg N/day)	benthic flux (kg N/day)
Upper Quashnet River	0.36	0.33	4.73
Middle Quashnet River	0.00	-	0.83
Lower Quashnet River	0.22	0.25	2.99
Surface Water Sources			
Moonakis River	10.41	-	-

Table VIII-10. Modeling **Scenario B** sub-embayment and surface water loads used for total nitrogen modeling of the Quashnet River Estuary within the Waquoit Bay System, with total watershed N loads, atmospheric N loads, and benthic flux.

sub-embayment	watershed load (kg N/day)	direct atmospheric deposition (kg N/day)	benthic flux (kg N/day)
Upper Quashnet River	0.95	0.33	5.99
Middle Quashnet River	0.00	-	1.05
Lower Quashnet River	0.41	0.25	3.58
Surface Water Sources			
Moonakis River	14.56	-	-

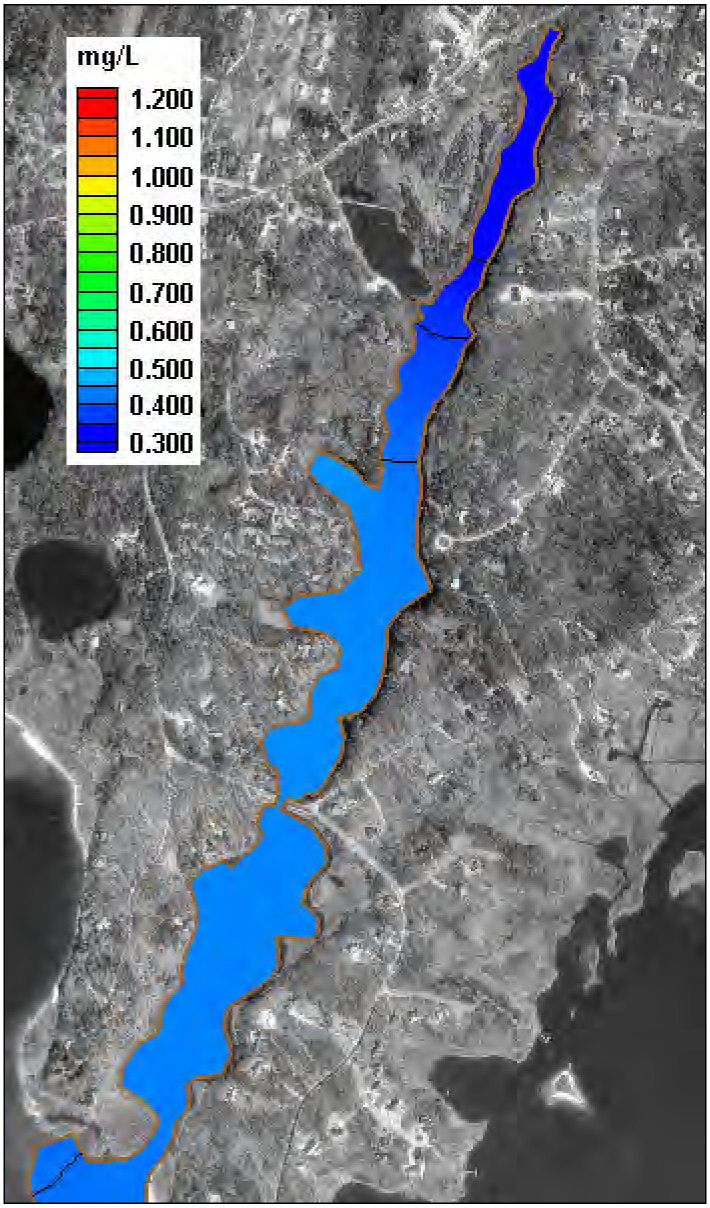


Figure VIII-5. Contour plot of modeled total nitrogen concentrations (mg/L) in the Quashnet River Estuary within the Waquoit Bay System, for **Scenario A** loading conditions (100% septic removal, with present Waquoit Bay boundary condition).

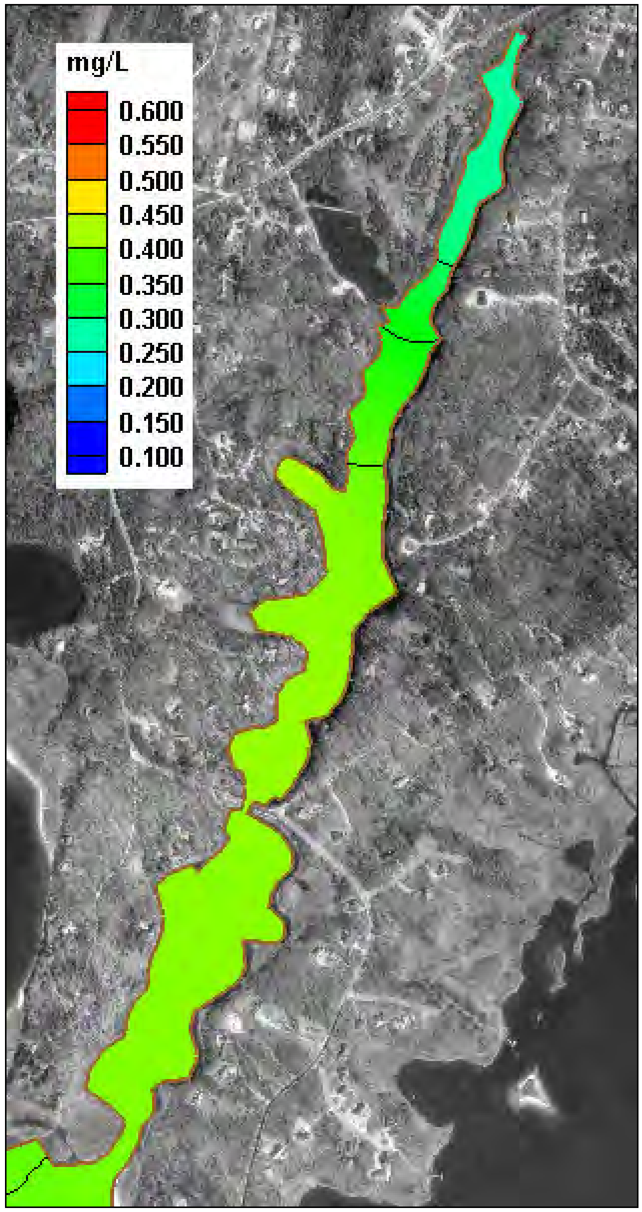


Figure VIII-6. Same results as for Figure VIII-5, but shown with finer contour increments for emphasis. Contour plot of modeled total nitrogen concentrations (mg/L) in the Quashnet River Estuary within the Waquoit Bay System, for **Scenario A** loading conditions (100% septic removal, with present Waquoit Bay boundary condition).

For all of these estuaries, additional scenarios using a watershed nitrogen removal strategy focusing on areas where groundwater is flowing directly into the estuary has merit relative to efficient wastewater planning. Nutrient loads entering the sub-embayments through surface flow, natural attenuation in freshwater bodies (i.e., streams and ponds) can significantly reduce the load that finally reaches the estuary. Future nitrogen management should take advantage of natural nitrogen attenuation to ensure the most cost-effective nitrogen reduction strategies. The lower freshwater reaches of the Quashnet River and Red Brook provide opportunities for enhancing natural attenuation of their nitrogen loads.

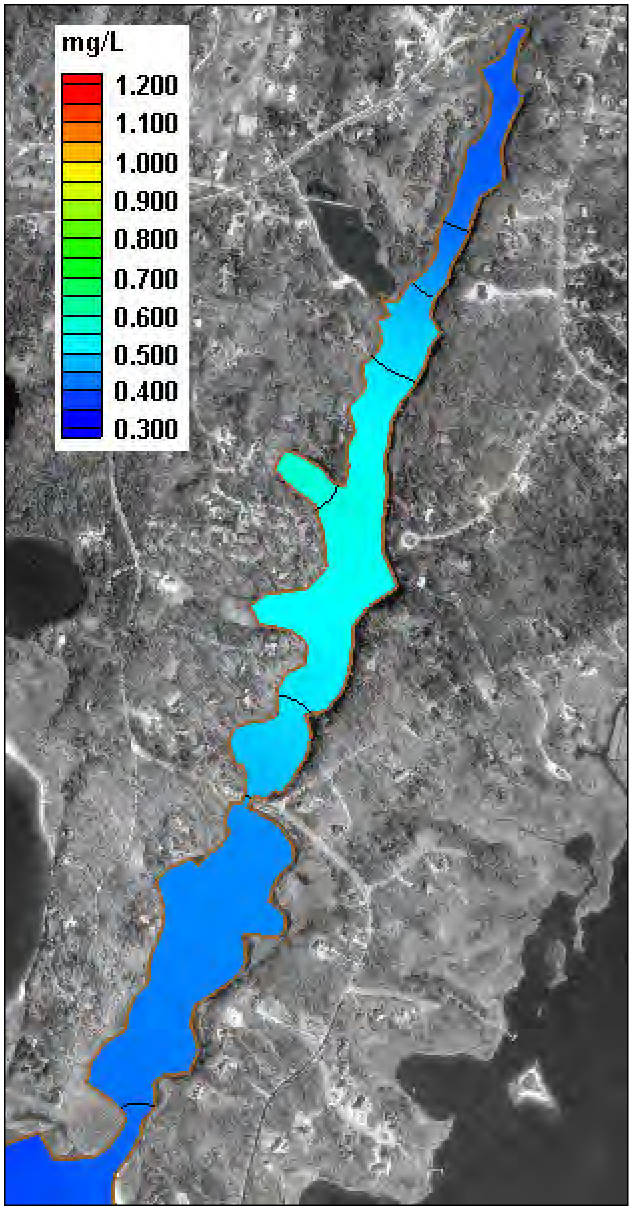


Figure VIII-7. Contour plot of modeled total nitrogen concentrations (mg/L) in the Quashnet River Estuary within the Waquoit Bay System, for **Scenario B** loading conditions (67% septic removal, with 0.350 mg/L Waquoit Bay boundary condition).

Although the above modeling results provide one manner of achieving the selected threshold levels for the sentinel site within this estuarine system, 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. As the restoration process continues, the MEP will work with the Towns of Mashpee and Falmouth to develop additional specific water quality modeling scenarios, to be run to evaluate other nitrogen removal strategies. The existing MEP analysis and model provides for the evaluation of nitrogen loading reduction alternatives and potential discharge sites relative to the amount of improvement of the nutrient related habitat quality within these estuarine sub-embayments.

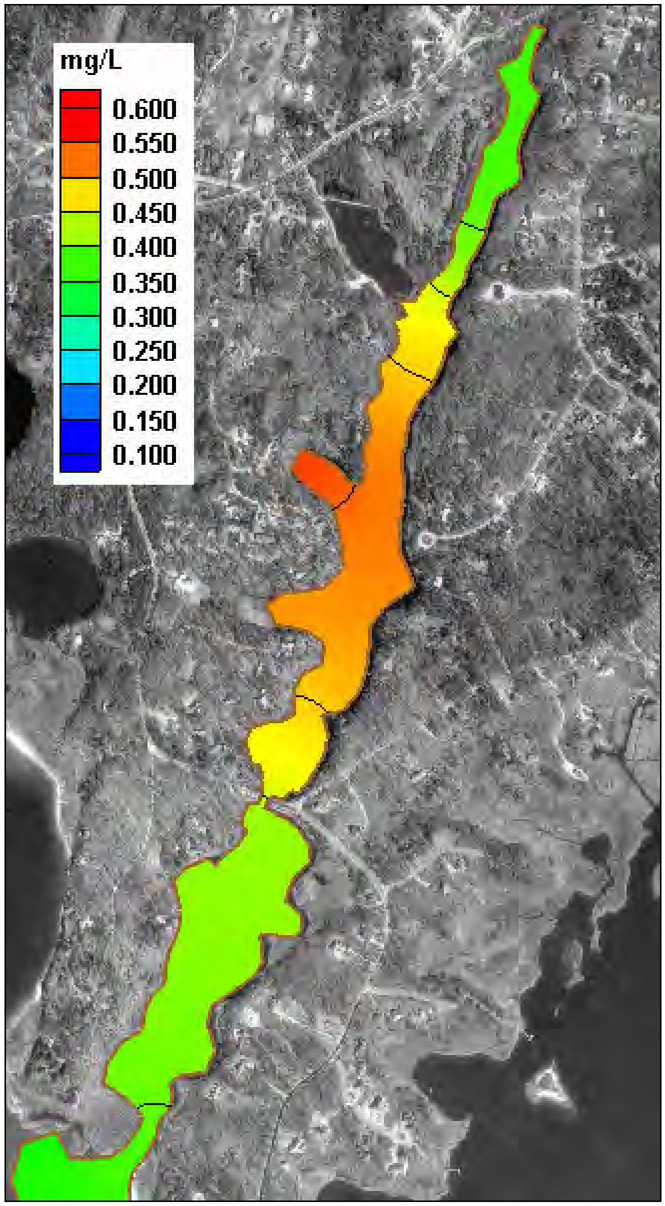


Figure VIII-8. Same results as for Figure VIII-7, but shown with finer contour increments for emphasis. Contour plot of modeled total nitrogen concentrations (mg/L) in the Quashnet River Estuary within the Waquoit Bay System, for **Scenario B** loading conditions (67% septic removal, with 0.350 mg/L Waquoit Bay boundary condition).

Table VIII-11. Modeled TN concentrations and percent change for present conditions and nitrogen loading **Scenarios A and B**, for the Quashnet River Estuary within the Waquoit Bay System.

	present	Scenario A		Scenario B	
	(mg/L)	(mg/L)	% change	(mg/L)	% change
QR fw	0.601	0.272	-54.7%	0.380	-36.8%
QR u	0.768	0.361	-53.0%	0.493	-35.8%
QR m	0.794	0.418	-47.4%	0.532	-33.1%
QR l	0.523	0.417	-20.3%	0.416	-20.5%

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