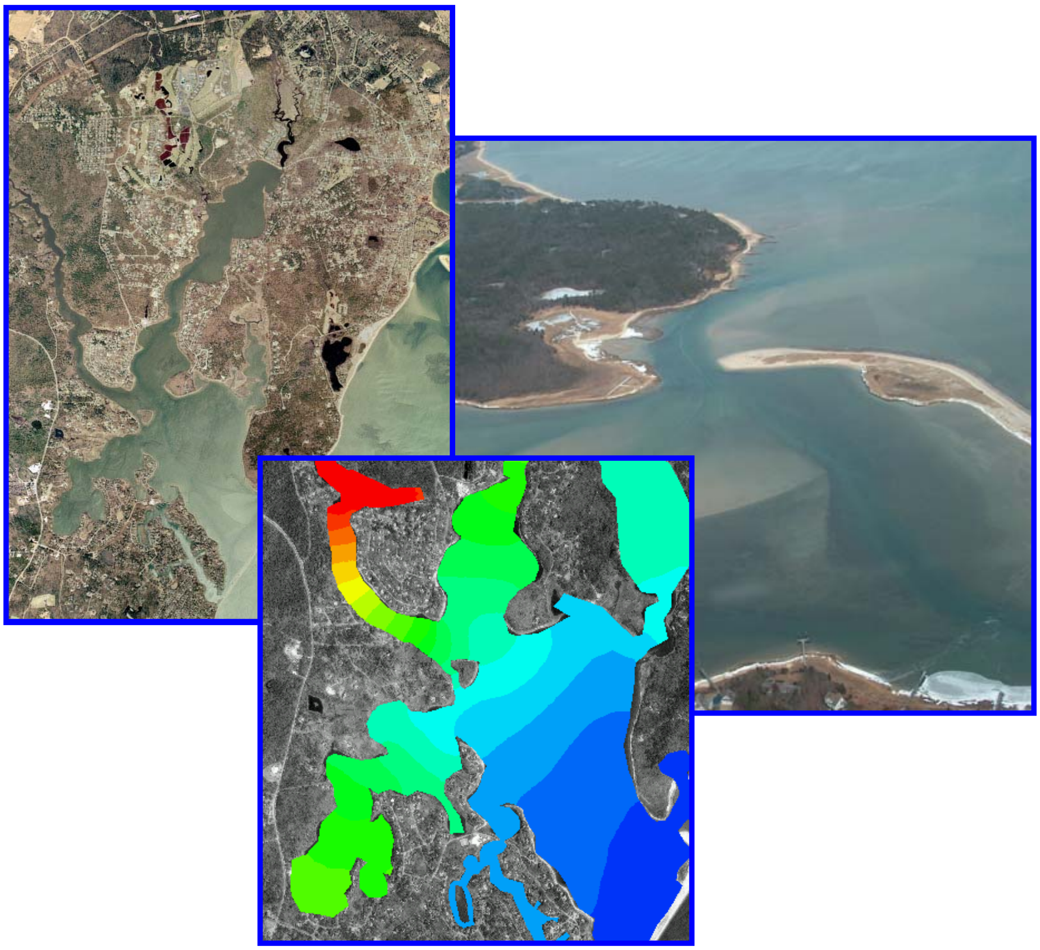
Massachusetts Estuaries Project

Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Popponesset Bay, Mashpee and Barnstable, Massachusetts



University of Massachusetts Dartmouth
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Massachusetts Department of
Environmental Protection

FINAL REPORT – SEPTEMBER 2004

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I. INTRODUCTION

The Popponesset Bay System is located within the Towns of Mashpee (north & west) and Barnstable (east), on Cape Cod Massachusetts with a southern shore bounded by water from Nantucket Sound (Figure I-1). The Bay's watershed is distributed among the Towns of Mashpee, Barnstable and Sandwich. It should be noted that Town of Sandwich does have jurisdiction over land and associated land uses in the uppermost portions of the overall watershed to Popponesset Bay. Specifically, portions of the Popponesset Bay watershed that exist within the Town of Sandwich are generally situated above the Mashpee-Wakeby Pond system with the exception of a small area immediately above Wakeby Pond that lies within the Town of Mashpee. As such, in order to achieve effective restoration of Popponesset Bay, it is critical that all three towns (Barnstable, Mashpee, and Sandwich) constituting the total Popponesset Bay watershed be involved in nutrient management discussions. Land uses closest to the embayment are likely to have greater impact than those in the upper portions of the watershed which are subject to nitrogen attenuation during transport through natural aquatic systems (e.g. ponds, rivers, wetlands etc.) prior to discharge to the embayment.

The present Bay results from tidal flooding of drowned river valleys formed primarily by the Mashpee and Santuit Rivers as a result of rising sea level. The Bay is separated from Nantucket Sound by a barrier spit, which grew from the southwestern shore. The spit, Popponesset Beach, as a barrier spit, is a very dynamic geomorphic feature. The Bay exchanges tidal water with Nantucket Sound through a single maintained inlet. The shore to the north of the inlet has been stabilized with riprap, as is the heavily residential southern portion of Popponesset Beach. The current spit is significantly shorter than seen in 1880 Barnstable County or 1938 USGS topographic maps, where the tip of the spit extended north to Rushy Marsh.

The estuarine region of the Popponesset Bay System is composed of a large lower basin, Popponesset Bay, and multiple tributary sub-embayments (Ockway Bay, Pinquickset Cove, Shoestring Bay, Mashpee River, Popponesset Creek). These sub-embayments constitute important components of the Town'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 Popponesset Bay system and its sub-embayments along the Mashpee and Barnstable shores are at risk of eutrophication (over enrichment) from high nitrogen loads in the groundwater and runoff from their watersheds.

The primary ecological threat to Popponesset Bay embayment system as a coastal resource is degradation resulting from 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 identified associated with an abandoned junkyard in Forestdale (J. Braden Thompson site) where a groundwater plume containing trichloroethylene and tetrachloroethylene is discharging to the surface waters of Mashpee-Wakeby Pond in the upper watershed to the Bay. In addition, a small volatile organic compound plume associated with the former Augat site (on Rt. 28) is discharging

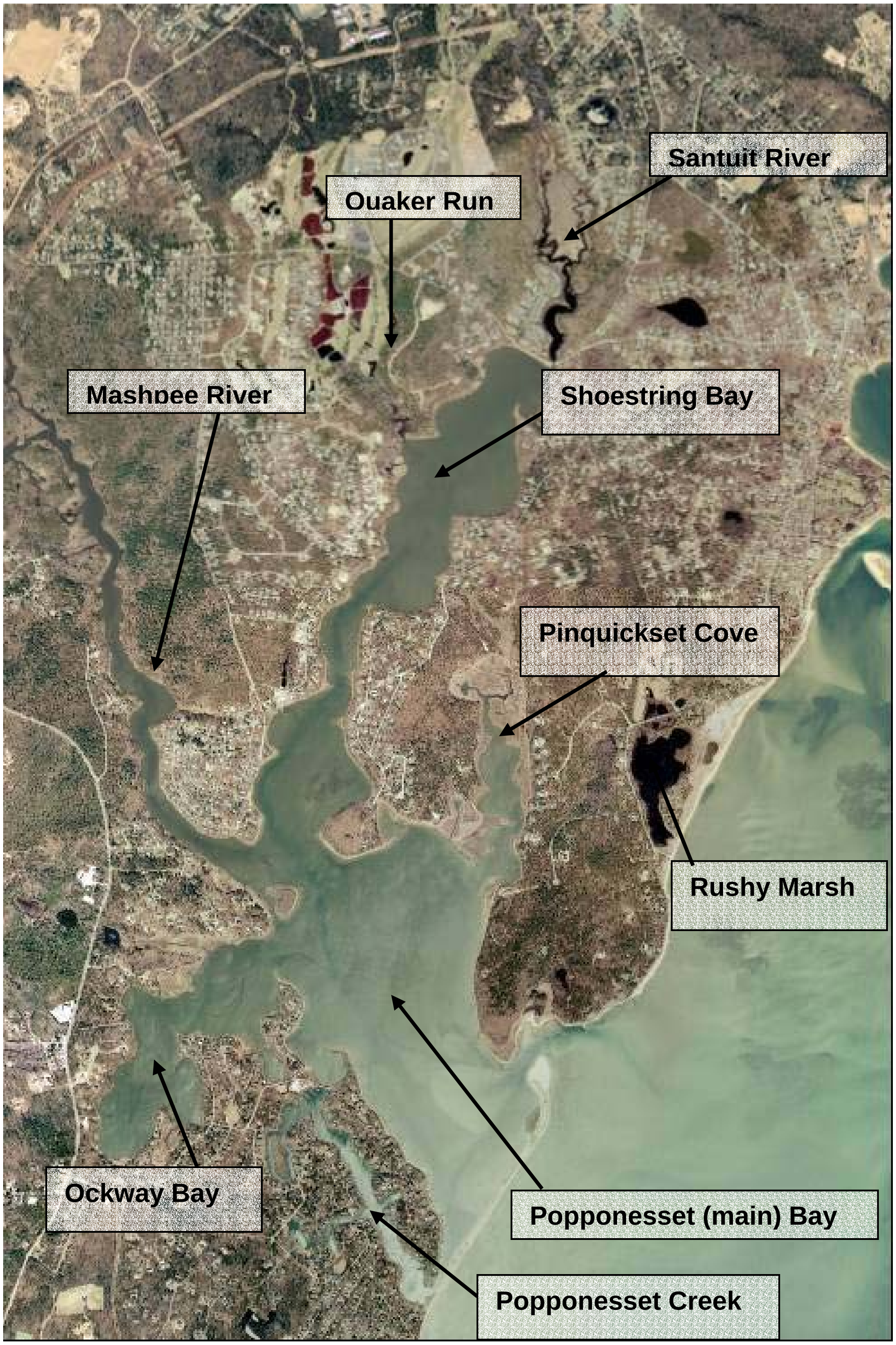


Figure I-1. Study region for the Massachusetts Estuaries Project analysis of the Popponesset Bay System. Tidal waters enter the Bay through the single inlet from Nantucket Sound. Freshwaters enter from the watershed primarily through 3 surface water discharges (Mashpee River, Santuit River, Quaker Run) and direct groundwater discharge. Rushy Marsh is a separate embayment with a direct tidal connection to Nantucket Sound.

directly to Shoestring Bay. Bacterial contamination causes closures of shellfish harvest areas periodically within the Bay System. In contrast, 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 nitrogen loading to the Bay, like almost all embayments in southeastern Massachusetts, results primarily from on-site disposal of wastewater. The Town of Mashpee has been among the fastest growing towns in the Commonwealth over the past two decades and does not have centralized wastewater treatment; although several small privately operated facilities operate within the Popponesset Bay watershed. As existing and probable increasing levels of nutrients impact Mashpee's coastal embayments, water quality degradation will accelerate, with further harm to invaluable environmental resources.

As the primary stakeholder to the Popponesset Bay System, the Town of Mashpee was the first community to become concerned over perceived degradation of Bay waters. The concern over declining habitat quality followed significant on-going efforts to preserve open space within the Mashpee River sub-watershed, most recently related to the Mashpee National Wildlife Refuge (1995). This concern led to one of the first ecological studies of contamination within the estuary, by KV Associates completed in 1991. This effort attempted to develop a plan for managing contamination in the Mashpee and Shoestring Bay estuaries. By the mid-1990's phytoplankton and macroalgal blooms had raised the declining quality of the Bay into the realm of general discussion. The Town of Mashpee through its Board of Selectman, Watershed Management Committee, Waterways Commission and Shellfish Department began the Popponesset Bay Water Quality Monitoring Program in July 1997, in concert with the Cotuit Waders of the Town of Barnstable and SMAST (then the Center for Marine Science and Technology). Initial results from 1997 and 1998, indicated nutrient, chlorophyll a and dissolved oxygen conditions were consistent with significant eutrophication within the Mashpee River, Ockway Bay and Shoestring Bay (Howes and Schlezinger 1998).

The Monitoring Program was then expanded (in recent years with formal Town of Barnstable participation) and has continued through summer 2003 to provide baseline water quality data for the MEP. Preliminary land-use analysis of the watershed to the Popponesset Bay embayment system supported the view that the habitat decline within this large estuarine system was being caused by increased nitrogen inputs from the surrounding watershed due to expanding commercial and residential development (Cape Cod Commission 1998). In 1998 and 1999 the Town of Mashpee allocated funds for a project to quantitatively assess nutrient sources and model nitrogen levels within the System with SMAST scientists. 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 wrapped into the Town's Wastewater Facilities Planning effort by the Mashpee Sewer Commission. Under the direction of the Mashpee Sewer Commission and the Town of Barnstable DPW, the Popponesset Bay System was included in the first round prioritization of the Massachusetts Estuaries Project to provide state-of-the-art analysis and modeling. However, given that the MEP was able to fully integrate the Towns' on-going data collection and modeling effort, no additional municipal funds were required for MEP tasks.

The common focus of the Mashpee and Barnstable effort has been to gather site-specific data on the current nitrogen related water quality throughout the Popponesset Bay System and determine its relationship to watershed nitrogen loads. This seven-year effort has provided the baseline information required for determining the link between upland loading, tidal flushing, and estuarine water quality. The MEP effort builds upon the Water Quality Monitoring Program, and 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 and nitrogen management alternatives development needed by the Towns of Mashpee and Barnstable. While the completion of this complex multi-step process of rigorous scientific investigation to support watershed based nitrogen management has taken place under the programmatic umbrella of the Massachusetts Estuaries Project, the results stem directly from the efforts of large number of Town staff and volunteers over many years. The modeling tools developed as part of this program provide the quantitative information necessary for the Towns of Mashpee and Barnstable to develop and evaluate the most cost effective nitrogen management alternatives to restore this valuable coastal resource which is currently being degraded by nitrogen overloading.

I.1 THE MASSACHUSETTS ESTUARIES PROJECT APPROACH

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

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

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

The Massachusetts Estuaries Project represents the next generation of watershed based nitrogen management approaches. The Massachusetts Department of Environmental Protection (MA DEP), the University of Massachusetts – Dartmouth School of Marine Science and Technology (SMAST), and 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 and municipalities with technical guidance to support policies on nitrogen loading to embayments. In addition, the technical reports prepared for each embayment system will serve as the basis for the development of Total Maximum Daily Loads (TMDLs). Development of TMDLs is required pursuant to Section 303(d) of the Federal Clean Water Act. TMDLs must identify sources of the pollutant of concern (in this case nitrogen) from both point and non-point sources, the allowable load to meet the state water quality standards and then allocate that load to all sources taking into consideration a margin of safety, seasonal variations, and several other factors. In addition, each TMDL must contain an outline of an implementation plan. For this project, the DEP recognizes that there are likely to be multiple ways to achieve the desired goals, some of which are more cost effective than others and therefore, it is extremely important for each Town to further evaluate potential options suitable to their community. As such, DEP will likely be recommending that specific activities and timelines be further evaluated and developed by the Towns (sometimes jointly) through the Comprehensive Wastewater Management Planning process.

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

The major Project goals are to:

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

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

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

The Linked Model has been applied for watershed nitrogen management in approximately 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 facilitates the evaluation of nitrogen management alternatives relative to meeting water quality targets within a specific embayment. The Linked Watershed-Embayment Model also enables Towns to evaluate improvements in water quality relative to the associated cost. In addition, once a model is fully functional it can be “kept alive” and updated for continuing changes in land-use or embayment characteristics (at minimal cost). In addition, since the Model uses a holistic approach (the entire watershed, embayment and tidal source waters), it can be used to evaluate all projects as they relate directly or indirectly to water quality conditions within its geographic boundaries.

Linked Watershed-Embayment Model Overview: The Model provides a quantitative approach for determining an embayment’s: (1) nitrogen sensitivity, (2) nitrogen threshold loading levels (TMDL) and (3) response to changes in loading rate. The approach is 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 Popponesset Bay embayment system exchanges tidal water with Nantucket Sound through a single maintained inlet at the tip of Popponesset Beach. For the MEP analysis, the Popponesset Bay estuarine system has been partitioned into five general sub-embayment groups: the 1) Popponesset (main) Bay, 2) Pinquickset Cove, 3) Ockway Bay, 4) Mashpee River (lower or tidal region) and 5) Shoestring Bay (see Figure I-1). Popponesset Creek was considered as part of the Popponesset (main) Bay in the modeling and thresholds analysis.

Within the Popponesset Bay System, the tidal portion of the Mashpee River shows the clearest estuarine characteristics, with extensive salt marsh area, tidal flats and large salinity fluctuations. In contrast, Popponesset Bay, Shoestring Bay and Ockway Bay show more typical embayment characteristics dominated by open water areas, having only fringing salt marshes, relatively stable salinity gradients and relatively large basin volumes relative to tidal prism. Although the four sub-embayment systems bounding the main open water portion of Popponesset Bay (Pinquickset Cove, Ockway Bay, Mashpee River lower, and Shoestring Bay) exhibit different hydrologic characteristics (river dominated versus tidally dominated), the tidal forcing for these systems is generated from Nantucket Sound. Nantucket Sound, adjacent Popponesset Beach, 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 Popponesset Bay is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle (note the tide range off Stage Harbor Chatham is ~4.5 ft, Wellfleet Harbor is ~10 ft).

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

Given the present hydrodynamic characteristics of the Popponesset 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.

Nitrogen loading to the Popponesset Bay System was determined relative to five (5) sub-embayments: Pinquickset Cove, Ockway Bay, Mashpee River (lower or tidal region), Shoestring Bay, and Popponesset Bay. The watershed for this estuarine system contains approximately 13,000 acres, dominated by single-family residences. Commercial and residential land-uses primarily in the southern portion of Mashpee and in the Barnstable region create a large nutrient load to the Popponesset Bay System. The nitrogen loading from the more heavily populated areas of the Town of Mashpee is focused on the northern reaches of the estuarine system.

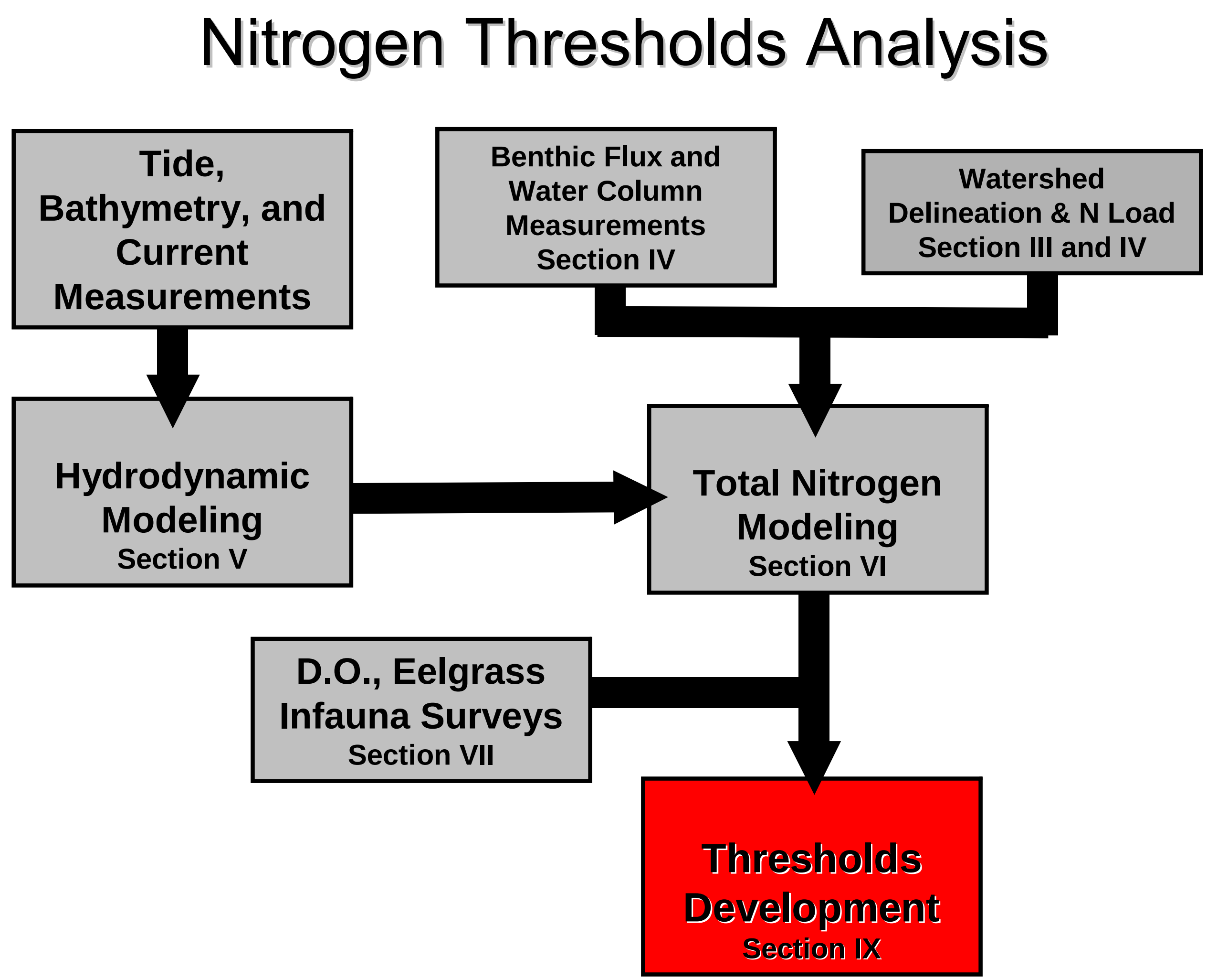


Figure I-2. Massachusetts Estuaries Project Critical Nutrient Threshold Analytical Approach

System wide, approximately three quarters of the nitrogen load from single-family dwellings enters the Shoestring Bay sub-embayment as well as the tidally influenced lower reach of the Mashpee River.

As management alternatives are being developed and evaluated, it is important to note that Popponesset Bay is a relatively dynamic system. The spit forming Popponesset Beach is continually expanding and eroding, once nearly reaching the inlet channel to the Three Bays System to the north. The spit frequently experiences periodic over wash (Aubrey and Gaines 1982). The present inlet position is relatively new, resulting from a breach of the spit in the hurricanes of 1954. Similarly, within the main Bay, several islands apparent 50 -100 years ago have been incorporated into other landforms with unquantified effects on the circulation of Bay waters. Thatch Island and Little Thatch Island within the lower main Bay have “joined” with the spit, most likely due to a combination of the natural processes of overwash of the barrier beach and shoreline retreat. Daniels Island, at the entrance to Ockway Bay, has been joined to the mainland by filled causeways, apparently filling salt marshes and changing the local circulation pattern.

Hydrodynamics have also been altered within Popponesset Creek due to dredging and channelization of wetlands. Within the watershed there have been changes to the freshwater systems which attenuate nitrogen during transport to bay waters. Most notable have been the modification to riparian zones either through channelization, restriction, or filling of freshwater wetlands and, in some cases, transformation to cranberry agriculture. Most of the alterations have reduced the nutrient buffering capacity of these systems, magnifying the nitrogen loading to the bay. However, the predominant watershed alteration has been the shifting of fields and pine-oak forest to residential and commercial development, with its resultant increasing nitrogen input to the watershed, aquifer and ultimately bay waters. This recent shift in land-use has likely resulted in this estuary receiving its highest rates of nitrogen loading than at any period over the past 400 years. Previous large shifts in land-use, primarily from forest to agriculture did not have the same resultant enhancement in nitrogen loading as agriculture generally recycled nitrogen (as opposed to commercial fertilizers) and the population was <10% of today. The present year-round population per square mile is greater than the entire town population of 50 years ago (total population based on 2000 census for Towns of Mashpee, Sandwich, and Barnstable are 12,946, 20,136 and 47,821 respectively). It appears that the nitrogen attenuation capacity of the freshwater systems may have been reduced, as the need to intercept the nitrogen loading to the watershed has increased. While this may be a partial cause of the present estuarine decline, it may also represent a potential opportunity for restoration of bay systems.

I.3 NUTRIENT LOADING

Surface and groundwater flows are pathways for the transfer of land-sourced nutrients to coastal waters. Fluxes of primary ecosystem structuring nutrients, nitrogen and phosphorus, differ significantly as a result of their hydrologic transport pathway (i.e. streams versus groundwater). In sandy glacial outwash aquifers, such as in the watershed to the Popponesset 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 Popponesset 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 and nutrient related water quality degradation occurs. Because nearshore coastal salt ponds and embayments are the primary recipients of nutrients carried via surface and groundwater transport from terrestrial sources, it is clear that activities within the watershed, often miles from the water body itself, can have chronic and long lasting impacts on these fragile coastal environments.

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

Unfortunately, almost all of the estuarine reaches within the Popponesset Bay System (including Popponesset Bay) are near or beyond their ability to assimilate additional nutrients without impacting their ecological health. Nitrogen levels are elevated throughout the System and eelgrass has not been observed for over a decade. The result is that nitrogen management of the primary sub-embayments is aimed at restoration, not protection or maintenance of existing conditions. In general, nutrient over-fertilization is termed “eutrophication” and when the nutrient loading is primarily from human activities, “cultural eutrophication”. Although the influence of human-induced changes has increased nitrogen loading to the systems and contributed to the degradation in ecological health, it is sometimes possible that eutrophication within Popponesset 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” (e.g. watershed derived and offshore nutrient inputs) for water quality modeling of the Popponesset Bay 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 Popponesset Bay System, including the tributary sub-embayments of Mashpee River, Ockway Bay, Shoestring Bay, Pinquickset Cove and the Popponesset Bay central basin. 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 Popponesset Bay is transported by freshwater, predominantly groundwater. Concentrations of total nitrogen and salinity of Nantucket Sound source waters and throughout the Popponesset Bay System were taken from the Popponesset Bay Water Quality Monitoring Program (supported by the Towns of Mashpee and Barnstable, associated with the Coastal Systems Program at SMAST). Measurements of current salinity and nitrogen and salinity distributions throughout estuarine waters of the System were used to calibrate and validate the water quality model (under existing loading conditions).

I.5 REPORT DESCRIPTION

This report presents the results generated from the implementation of the Massachusetts Estuaries Project linked watershed-embayment approach to the Popponesset Bay System for the Towns of Mashpee (lead) and Barnstable. 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 for restoration of the Bay in Section VIII. Additional modeling is conducted to produce an example of the type of watershed nitrogen reduction required to meet the determined Bay threshold for restoration. This latter assessment represents only one of many solutions and is produced to assist the Town in developing a variety of alternative nitrogen management options for this system. Finally, analyses of the Popponesset Bay System was relative to potential alterations of circulation and flushing, including an analysis to identify hydrodynamic restrictions and an examination of dredging options to improve nitrogen related water quality. The results of the nitrogen modeling for each scenario have been presented (Section IX).

II. PREVIOUS STUDIES RELATED TO NITROGEN MANAGEMENT

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

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

A major component of the MEP nutrient analysis is the evaluation of hydrodynamics within the estuarine system. A two-dimensional hydrodynamic and water quality model was previously developed by Aubrey Consulting, Inc. (ACI, 1994). The purpose of this modeling effort was to assess potential impacts of nitrogen loading resulting from the proposed expansion of a sewage treatment plant. Field measurements of water elevations and bathymetry were taken to parameterize the hydrodynamic modeling effort; however, dispersion coefficients for the water quality modeling portion of the study were based upon previous studies of similar estuaries. The water quality modeling portion of the analysis utilized simplified assumptions regarding the incremental effects of increasing nitrogen loads to the estuarine system. It did not include a rigorous evaluation of all nitrogen sources to the estuary and did not include nitrogen sinks. The MEP analysis presented in this report provides a comprehensive analysis of nutrients within the Popponesset Bay estuary; therefore, results from the less rigorous 1994 analysis have been superceded.

Results from the 1994 hydrodynamic modeling study of flushing rates within the Popponesset Bay estuary indicate that central Popponesset Bay is relatively well flushed, since Popponesset Bay is generally shallow and the tide range is significant relative to embayment depth. At the time of this pilot hydrodynamic study greater than 50 percent of the water within the estuary was exchanged during a typical tidal cycle. The sub-embayments (located within the upper portions of the estuary system) to Popponesset Bay, however, show long residence times and receive a high percentage of the nutrient load to the Popponesset Bay system.

Following the initial hydrodynamic modeling effort, the Town of Mashpee, through the Mashpee Waterways Commission, funded a hydrodynamic study focusing on the effects of dredging on tidal flushing within the tidal portion of the Mashpee River (Hamilton, 1996 and 1998). Additional data was utilized to parameterize this model, including updated tide data from 1997 and updated bathymetry data from 1996. Initial modeling efforts (Hamilton, 1996) indicated a measurable reduction in the Mashpee River residence time as a result of dredging, indicating a potential water quality improvement. In later communications (Hamilton, 1998), this conclusion was changed to indicate that feasible dredging scenarios do not significantly benefit Mashpee River flushing. Although the 1998 study indicated minor improvements to the hydrodynamic model, it is unclear how these modifications were responsible for the substantial change in model results.

For the MEP modeling analysis, the data from the previous studies were evaluated relative to the needs of the Linked Watershed-Embayment Model. Bathymetric data associated with the 1994 study was cursory and was not collected relative to a known tidal datum (e.g. NGVD29) as required for MEP. In addition, the tidal information also was not related to a known tidal datum, rather the tide data was related to a computed mean tide level, which is the average water elevation from the 30-day record. These data shortcomings and recent alterations to the system bathymetry (specifically in the vicinity of Popponesset Bay inlet) necessitated the collection of both bathymetry and tide data to support the MEP analysis.

Based on the above findings, a revised hydrodynamic analysis of the Popponesset Bay system, biological and chemical measurements, and a water quality model were developed that used the tidal flushing inputs and simulated the calculated and measured nitrogen loads to the embayments. This model was then calibrated in a process that rationalizes the resulting calculated water column concentrations with measured values from monitoring programs over the past four years. The water quality model then becomes a predictive tool for evaluating the effects of various nitrogen loading scenarios on nitrogen concentrations in the embayments.

The concern about excessive nitrogen loading to the water bodies in the Mashpee study area is evidenced by the number of studies and analyses conducted over the past 10 years. As early as 1984 attention was being given to possible water quality problems within Popponesset Bay whereby James Begley of the D.E.Q.E. Shellfish Sanitation Section identified excessive levels of coliform bacterial contamination in the Mashpee River. This finding promptly led to closure of the Mashpee River to shellfishing. Contamination problems in Popponesset Bay were further investigated by K-V Associates, Inc. on behalf of the Mashpee Planning Department and Planning Board. Initial concerns over contamination problems in Popponesset Bay resulted in the development of a Interim Report (October 1987) entitled "Sources of Bacterial and Nutrient Contamination into the Mashpee River, Santuit River and Shoestring Bay." This initial report was followed by a second report also completed by K-V Associates, Inc. in 1988 that examined storm discharges (under winter conditions) to Popponesset Bay as well as undertook recharge zone delineations for the Mashpee River, Quaker Run and the Santuit River. In addition, data on Mashpee River flow and water quality was developed and compiled by Goldberg-Zoino and Associates in a July 1988 report prepared in conjunction with the Mashpee Sewer Commission's work on a sub-regional wastewater treatment facility proposed to be located adjacent to the former Mashpee landfill. It was clear from the initial studies that the Popponesset Bay System is nutrient overloaded. Based upon water quality indicators (chlorophyll a, total nitrogen, bottom water dissolved oxygen) much of the System would be classified as eutrophic (KV Associates 1984, Howes and Schlezinger 1997, 1998). This section summarizes these studies in chronological order to help put the present study in historical perspective.

One of the first identified studies that address nutrient contamination problems in Popponesset Bay is a Cumulative Impact Assessment performed by K-V Associates, Inc. (1991). The analysis presented in the K-V assessment (1991) supported a plan to reduce and control sources of contamination in the Mashpee River and Santuit River/Shoestring Bay estuaries to Popponesset Bay. However, the overall nutrient data was somewhat limited and suffered from inadequate method detection limits. In addition, the significant development that has occurred in the intervening years suggests that these data do not reflect current conditions. In addition, this study focused primarily upon the upper bay sub-embayments and the rivers. It did not include a comprehensive land-use analysis and did not account for nitrogen dynamics within the aquatic systems. However, it did point out many of the nutrient issues that continue to be relevant and are to be examined through the MEP analysis.

The Cape Cod Commission (CCC) undertook the Cape Cod Coastal Embayment Project that indicated that nutrient loading to the Popponesset Bay system, which includes the Mashpee River, Shoestring Bay, and Ockway Bay, is a significant problem. The data was based upon the 1996 watershed delineations. Due to the difference in watershed areas, updating of the land-use analysis and refinement of the watershed nitrogen loading model component of the MEP approach, the results from the MEP are different and supersede those of this earlier study.

The most recent survey of nutrient related water quality in the Popponesset Bay embayment system was performed by the University of Massachusetts – Dartmouth, School for Marine Science and Technology (SMAST) (Howes and Schlezinger, 1997) The goal of the 1997 water quality survey was to evaluate the relative nutrient related ecological health of the major component embayments to the Popponesset Bay system and determine if there was nutrient related degradation of the sub-systems to Popponesset Bay. Sampling for the survey was conducted during the summer when eutrophication impacts are generally the greatest in Cape Cod embayments as a joint effort by the Town of Mashpee, SMAST, and private citizen volunteers. The survey was conducted during the summer of 1997 and involved 5 periodic field sampling events through the period of July 31 to September 12, 1997. Major findings of the 1997 water quality survey indicate: 1) nitrogen levels within the Popponesset Bay system are significantly higher than the incoming water from Nantucket Sound with resultant enhancement of phytoplankton biomass, 2) both biomass and total nitrogen (TN) are more than 10 and 2 fold higher, respectively, than the high quality water from Nantucket Sound, 3) there is a distinct nutrient and phytoplankton biomass (chlorophyll-a) gradient within the Popponesset Bay system with highest levels for each being Mashpee River>Shoestring Bay>Ockway Bay>Central Bay>Nantucket Sound, 4) oxygen depletions of bottom waters of the sub-embayments to Popponesset Bay is relatively wide spread and frequent within the Mashpee River, Ockway Bay, and Shoestring Bay. At the time of the 1997 survey the central portion of Popponesset Bay still exhibited relatively high water quality.

The water quality data from this preliminary water quality study have been incorporated with data collected in subsequent years by the same group, the Popponesset Bay Water Quality Monitoring Program, which includes private citizens, the Mashpee Shellfish Department, Mashpee Harbor Master, Mashpee Waterways Commission, Mashpee Watershed Management Committee, Cotuit Waders, and Barnstable DPW (Nutrient Management Committee). The MEP has incorporated all appropriate data from all previous studies to enhance the determination of nitrogen thresholds for the Popponesset Bay System and to reduce costs to the Towns of Barnstable and Mashpee.

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 Popponesset Bay System.

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 Popponesset Bay System under evaluation by the Project Team. The Popponesset Bay estuarine system is composed of: the main body of Popponesset Bay, Pinquickset Cove, Ockway Bay, Mashpee River (tidal region), and Shoestring Bay. Further watershed modeling was undertaken to sub-divide the overall watershed to the Popponesset 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 Shoestring Bay tributary to the Popponesset 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 on the Sagamore flow cell on Cape Cod as part of a separate Massachusetts DEP effort. 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 1994 a, b). Freshwater discharge to estuaries is usually composed of surface water inflow from streams, which receive much of their water from groundwater base flow, and direct groundwater discharge. For a given estuary, differentiating between these two water inputs and tracking the sources of nitrogen that they carry requires determination of the portion of the watershed that contributes directly to the stream and the portion of the groundwater system that discharges directly into the estuary as groundwater seepage.

Biological attenuation of nitrogen (natural attenuation) occurs primarily within surface aquatic ecosystems (streams, wetlands, ponds) with little occurring within the main aquifer. Biological attenuation of nitrogen is predominantly through denitrification, sometimes directly from nitrate and sometimes indirectly after uptake by plants and remineralization and oxidation back to nitrate in the surface sediments. Burial of decayed plant matter containing nitrogen is almost always much less important than denitrification in reducing nitrogen transport. The freshwater ponds on Cape Cod provide important environments for the biological attenuation of nitrogen entering them and therefore also require that their contributing areas be delineated. Fresh ponds are hydrologic features directly connected to the groundwater system, which receive groundwater inflow through upgradient shores and discharge water into the aquifer in downgradient areas. 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 Popponesset 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 Popponesset Bay System and also to determine portions of recharged water that may flow through 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 Popponesset Bay System resides, water elevations are greater than 60 ft at the top of the lens and therefore these uppermost layers are required for model operation. At depth within the aquifer, layer 18 has a thickness of 40 feet and layer 19 extends to 240 feet below sea level. The bottom layer, layer 20, extends to the bedrock surface and has a variable thickness depending upon site characteristics.

The glacial sediments that comprise the aquifer of the Sagamore flow cell consist of gravel, sand, silt, and clay that were deposited in a variety of depositional environments. The sediments generally show a fining downward sequence with sand and gravel deposits deposited in glaciofluvial (river) and near-shore glaciolacustrine (lake) environments underlain by fine sand, silt and clay deposited in deeper, lower-energy glaciolacustrine environments. While there are glacial morainal deposits comprising some regions of the aquifer of the Sagamore flow cell, these are generally located adjacent to Buzzards Bay and are not found within the watershed to the Popponesset 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. Final aquifer parameters were determined through calibration to observed water levels and stream flows. Hydrologic data used for model calibration included historic water-level data obtained from USGS records and local Towns and water-level and streamflow data collected in May 2002.

The model simulates steady state, or long-term average, hydrologic conditions including a long-term average recharge rate of 27.25 inches/year and the pumping of public-supply wells at average annual withdrawal rates for the period 1995-2000 with a 15% consumptive loss. This recharge rate is based on the most recent USGS information. Large withdrawals of groundwater from pumping wells may have a significant influence on water tables and watershed boundaries and therefore the flow and distribution of nitrogen within the aquifer. Since almost all of the Popponesset Bay System watershed is unsewered, 85% of the water pumped from wells was modeled as being returned to the ground via on-site septic systems.

III.3 MASHPEE CONTRIBUTORY AREAS

Revised watershed and sub-watershed boundaries were determined by the United States Geological Survey (USGS) for each of the Popponesset Bay System's five major component sub-embayments (the main body of Popponesset Bay, Pinquickset Cove, Ockway Bay, Mashpee River (estuarine portion), and Shoestring Bay) (Figure III-1). Model outputs of MEP watershed boundaries are "smoothed" (a) to 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, 28 sub-watershed areas were delineated within the watershed to the Popponesset Bay 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 and water quality models. 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 criteria for ponds to which contributing areas would be developed was based partly on nitrogen attenuation considerations as well as computational complexity. Ponds with a surface area greater than or equal to 10 acres are likely to have the potential for nitrogen attenuation and as such warrant developing a sub-watershed delineation and performing a land use analysis in order to quantify the level of nitrogen attenuation. From a modeling point of view, including ponds less than 10 acres in size adds several degrees of computational complexity thereby making the groundwater models unwieldy with little if any measurable improvement in the watershed nitrogen loading analysis.

The delineations completed for the MEP project are the third delineation in less than 10 years; each delineation has been based on more and better data and has included more subwatersheds. Figure III-2 compares the MEP delineation with the delineations completed for the Cape Cod Commission in 1996 (Eichner, *et al*., 1998) and 2002 (Eichner, *et al*., 2002). The delineation completed in 1996 was based on a water table map developed by the Cape Cod Commission from long-term measurements of groundwater elevations, while the 2002 delineation was completed by the USGS using a previous iteration of the Sagamore Lens groundwater model.

Table III-2 summarizes the differences in watershed areas determined for the Popponesset Bay System from the 3 available delineations. As might be expected, the current MEP delineation agrees quite well with the previous USGS modeling effort in 2002. Overall, the MEP delineation for the System is 7% smaller (900 acres) than the 2002 USGS delineation. 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, the boundary between Popponesset and Three Bays is in the same location.

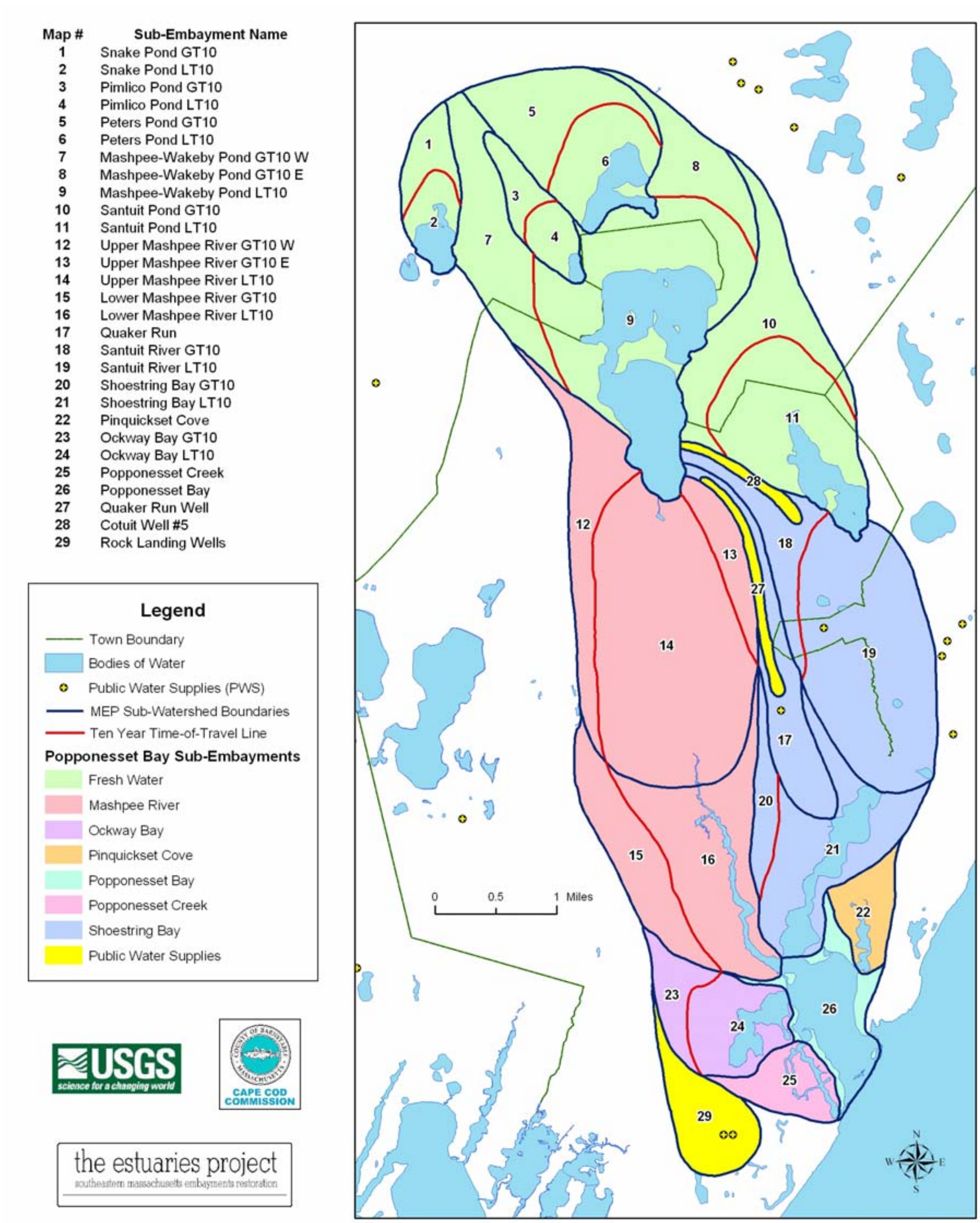


Figure III-1. Watershed and sub-watershed delineations for Popponesset Bay. Approximate ten year time-of-travel delineations were produced for quality assurance purposes and are designated with a "10" in the figure legend (above at left). Sub-watersheds to embayments were selected based upon the functional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

Table III-1. Long-term average daily groundwater discharge to each of the sub-embayments in the Popponesset Bay system, as determined from the USGS groundwater model.

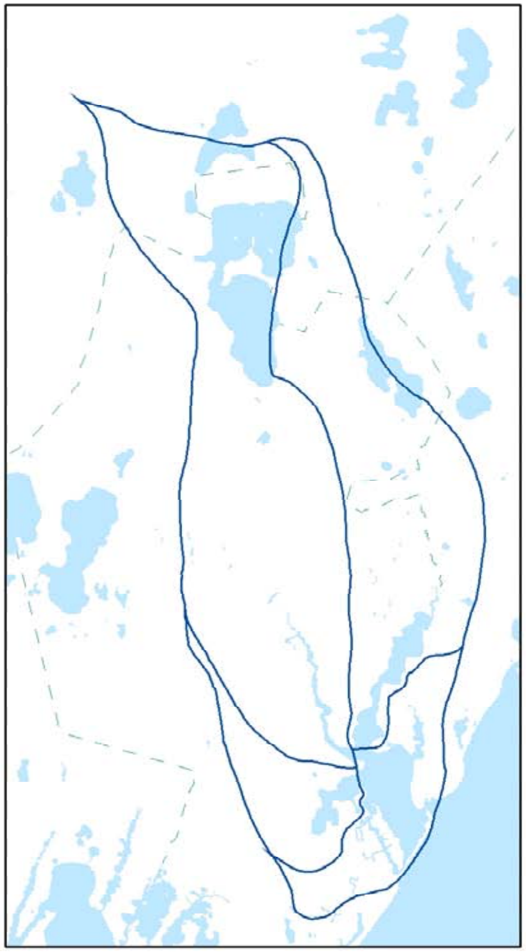
Watershed	Discharge		Watershed	Discharge	
	ft³/day	m³/day		ft³/day	m³/day
Upper Mashpee River	1,597,053	45,220	Santuit River	709,625	20,093
Lower Mashpee River	204,105	5,780	Quaker Run	131,724	3,730
Ockway Bay	75,887	2,149	Shoestring Bay	146,455	4,148
Pinquickset Cove	54,914	1,555	Popponesset Bay	41,496	1,175
Popponesset Creek	60,596	1,716			

While the MEP and the 2002 USGS delineation generally agree, they are significantly different from the 1996 delineation, both in coverage and acreage. The 2002 delineation expanded the overall area of the system watershed by approximately 2,400 acres as compared to the 1996 delineation. This expansion is mostly due to a more northern location for the regional groundwater divide, which expanded the watersheds to the major ponds (Mashpee-Wakeby, Santuit, and Snake).

Internal subwatershed delineations generally changed ±10-15%, although some of the smaller watersheds had much higher percent changes. For example, the Quaker Run subwatershed was reduced by 53% (253 acres); most of this area was lost to the Mashpee River subwatershed. Ockway Bay subwatershed was reduced by 34% (183 acres); most area was lost to the subwatershed of the Rock Landing public water supply wells. While these shifts do not change the specific sources of nitrogen within the watershed to the Popponesset Bay System, the shifting does potentially affect the amount of natural attenuation of nitrogen during transport. This further enhances the success of future nitrogen management options.

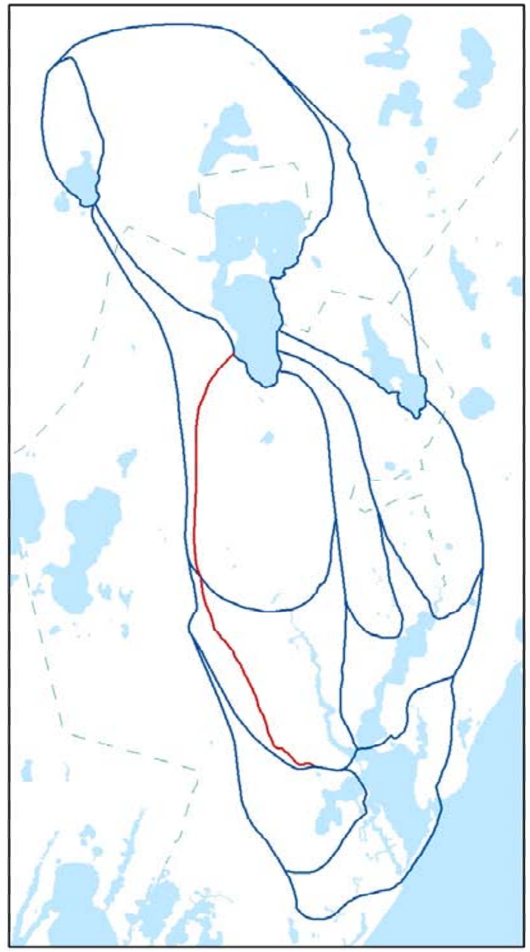
The evolution of the watershed delineations for the Popponesset Bay System have built one on another to increase the underlying hydrologic data underpinning the modeling, thereby increasing the accuracy. This is important as it decreases the level of uncertainty in the final calibrated and validated linked watershed-embayment model used for the evaluation of nitrogen management alternatives. Errors in watershed delineations do not necessarily result in significant errors in nitrogen loading. For example, small errors in watershed area can result in large errors in loading if a large source is counted in or out. Conversely, large errors in watershed area that involve only natural woodlands have little effect on nitrogen inputs to the downgradient estuary. In the case of the Popponesset Bay System, the present level of development and the areas of refinement in the watershed delineations indicate that the current and build-out nitrogen loading estimates were made more accurate through the use of the new delineations.

CCC, 1996.



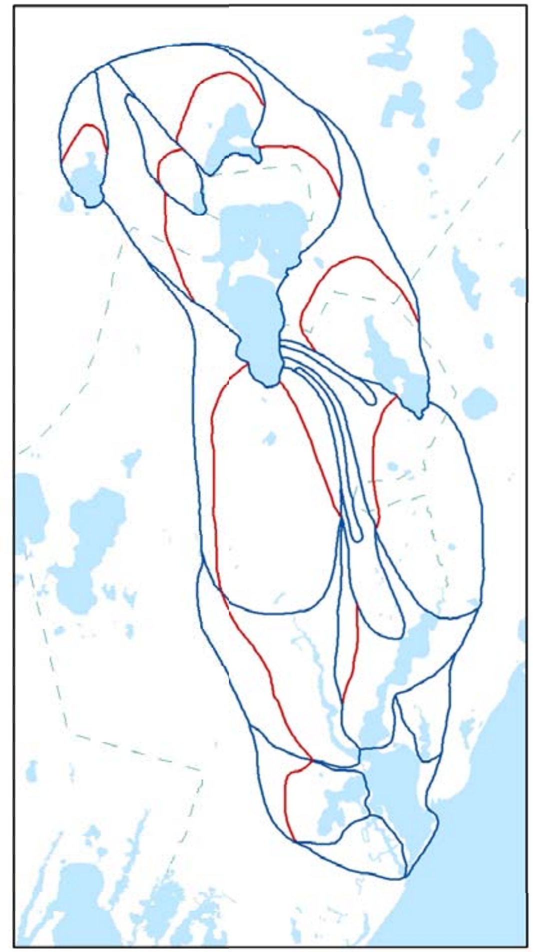
- Used in the 1996 & 2001 Regional Policy Plans. (Eichner, et al., 1998)

USGS, 2002.



- Delineated by USGS (Eichner, et al., 2002.)

MEP, 2004.



- Delineated by USGS for MEP analysis.

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

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

Table III-2. Percent difference in delineated embayment watershed areas between old and newly revised delineations. Note only 14% of the Snake Pond recharge enters the Popponesset Bay watershed, equivalent to ~20 acres.

Popponesset Bay System	MEP Watershed	2002 Watershed	% difference MEP vs. 2002	1996 Watershed	% difference MEP vs. 1996
WATERSHED	acres	acres		acres	
Mashpee Wakeby Pond	3,474	4,018	-16%	6,042	15%
Upper Mashpee River	2,374	2,616	-10%		
Lower Mashpee River	1,231	1,409	-14%		
Quaker Run	486	742	-53%	3,586	18%
Santuit Pond	1,408	1,396	1%		
Santuit River	1,618	1,422	12%		
Shoestring Bay	879	742	16%		
Ockway Bay	543	726	-34%	1,773	-20%
Lower Popponesset Bay (including Pinquickset Cove and Popponesset Creek)	929	772	17%		
Entire System (excluding Snake Pond)	12,942	13,843	-7%	11,401	12%

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 Popponesset Bay System. Determination of watershed nitrogen inputs to the Popponesset Bay embayment system 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 by examining groundwater travel times, 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 which may be released to the overlying waters or lost to denitrification within the sediments. Burial of nitrogen is generally small relative to the amount cycled. Sediment nitrogen regeneration can be a seasonally important source of nitrogen to embayment waters and leads to errors in predicting water quality if it is not included in determination of summertime nitrogen load.

The MEP project team includes technical staff from the Cape Cod Commission (CCC). In coordination with other MEP technical team staff, CCC staff developed nitrogen loading rates (Section IV.1) within each of the 28 subwatersheds to the Popponesset Bay embayment system (Section III). After reviewing the percentage of nitrogen loading in the less than 10 year time of travel and greater than 10 year time of travel watersheds (Table IV-1), reviewing Mashpee land use development in 1994 (CCC, 1998) and 2001 in the time of travel watersheds, and reviewing water quality modeling, the 10 year time of travel subwatersheds were eliminated and the number of subwatersheds was reduced to 16. Although the percentage of nitrogen loads in the less than 10-year subwatersheds ranges between 47 and 100%, more than three quarters (76%) of the overall system load is within 10 years flow to Popponesset Bay. The nitrogen loading effort also involved further refinement of watershed delineations to accurately reflect shoreline areas to ponds and embayments.

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 Popponesset Bay, the model used Mashpee, Barnstable, 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.

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

	LT10	GT10	TOTAL	%LT10
WATERSHED	kg/yr	kg/yr	kg/yr	
Mashpee-Wakeby Pond total	4066	2589	6655	61%
Upper Mashpee River	11275	3308	14583	77%
Lower Mashpee River	2728	3071	5799	47%
Santuit Pond	2770	977	3747	74%
Santuit River	8001	2168	10169	79%
Quaker Run	2708		2708	100%
Shoestring Bay	4120	879	4998	82%
Pinquickset Cove	454		454	100%
Popponesset Creek	2285		2285	100%
Popponesset Bay	2316		2316	100%
Ockway Bay	1641	190	1831	90%
TOTAL SYSTEM	42365	13181	55547	76%

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 Mashpee River and the Santuit River. Attenuation during transport through each of the major fresh ponds was determined through (a) comparison with other Cape Cod lake studies and (b) data collected on each pond. Internal nitrogen recycling was also determined within the Popponesset 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, Barnstable, and Sandwich. Mashpee's land use data is from 2001, while Sandwich and Barnstable's data is from 2000. The parcel and assessors databases from the three towns were combined by using the Cape Cod Commission Geographic Information System (GIS) for the MEP analysis.

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. "Public service" is the land classification assigned by the Massachusetts

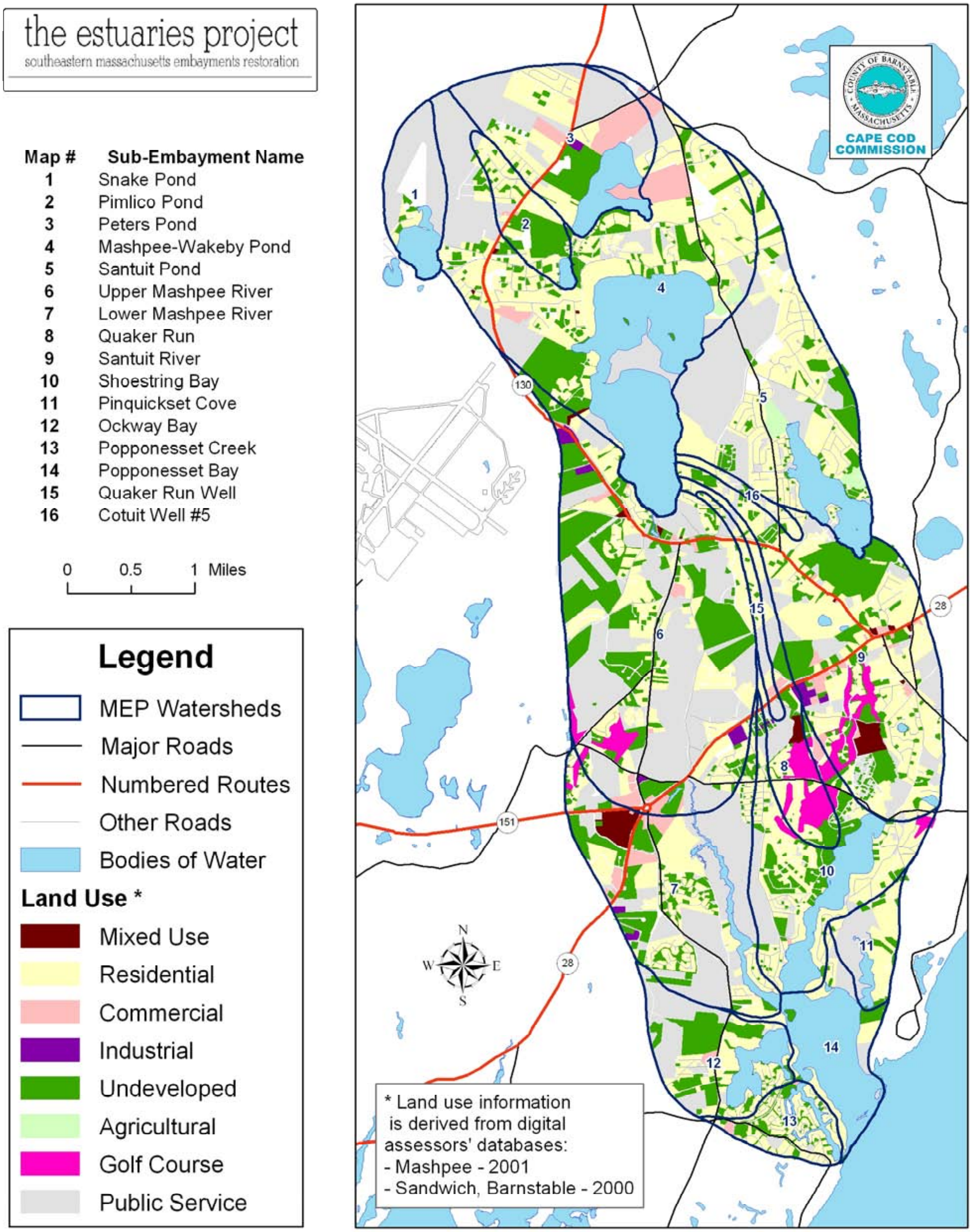


Figure IV-1. Land-use coverage in the Popponesset Bay watershed. Watershed data encompasses portions of the Towns of Mashpee, Barnstable, and Sandwich, MA.

Department of Revenue to tax exempt properties, including lands owned by government (e.g., wellfields, schools, open space, roads) and private groups like churches and colleges. Within the Popponesset Bay subwatersheds, the predominant land use is residential, most of which are single family residences. Single-family residences occupy approximately 13% of the total watershed area to Popponesset Bay and are 67% of the total parcels (Figure IV-2). Commercial properties are located throughout the watershed, with most parcels along Routes 28 and 130. Note that land-use determinations were made within the contributing sub-watersheds to major ponds, river and estuarine basins and to major water supply wells (Quaker Run Well, Cotuit Well #5). In these latter cases, nitrogen withdrawn from the aquifer for potable water distribution was applied as a loss in the nitrogen loading analysis. This nitrogen mass was very small and was redistributed through the water supply.

In order to estimate wastewater flows within the study area, MEP staff also obtained 1997 through 1999 Mashpee Water District water use information from the Mashpee Sewer Commission**,** 1998 through 2000 water use information from the Town of Barnstable, 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 the service area information was obtained from the Town of Mashpee and the state Department of Environmental Protection for the four wastewater treatment facilities (WWTFs) operating in the watershed in 1999 to 2000: Mashpee Commons, Willowbend, Stratford Ponds, and Forestdale School (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. The WWTFs at Windchime Point and Southcape were constructed after 2000 and, as such, are not included in the nitrogen loads for existing conditions, but are included in the buildout loads.

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

System Name	Average Effluent Characteristics		
Facility Name	Flow (gallons per day)	Total Nitrogen Concentration (mg/liter)	Annual Nitrogen Load (kg N/yr)
Mashpee Commons	16,392[a]	2.37[b]	54
Willowbend	14,408[a]	3.15[b]	63
Stratford Ponds	8,902[a]	8.96[b]	110
Forestdale School	951[c]	35[d]	46
Windchime Point[e]	12,700[f]	10[g]	175

Notes: [a] average flow (2000-2002); [b] flow-weighted average concentration (2001-2002); [c] average flow (2000-2003); [d] No apparent TN limit (personal communication, B. Dudley, DEP); [e] Prior to 8/01 all flows treated through on-site septic systems, WWTF information used in buildout analysis; [f] estimated average flow at buildout (153 units) based on flows during 2003 and 2004; [g] state permit concentration. Review of performance data indicates effluent concentrations at estimated buildout flow will be 9.3 mg/l, but given the uncertainty of ramping up the flow, it was determined that the regulatory permit concentration was appropriate for a buildout projection.

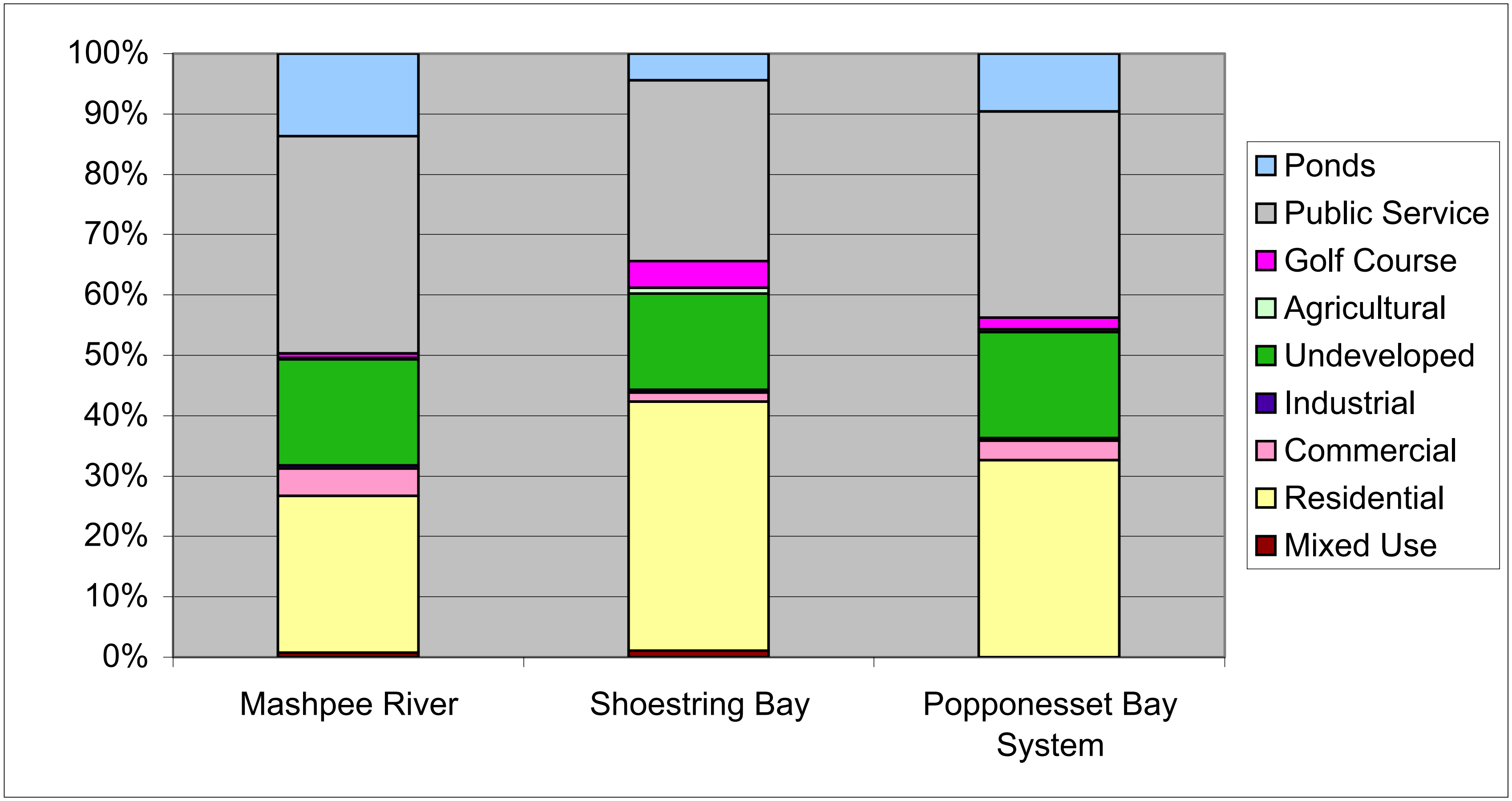


Figure IV-2. Distribution of land-uses within the Mashpee River and Shoestring Bay subwatersheds and the entire Popponesset Bay watershed.

IV.1.2 Nitrogen Loading Input Factors

Wastewater/Water Use

All wastewater is returned to the aquifer within the Popponesset Bay watershed either through individual on-site septic systems or the four WWTFs. Wastewater within the watershed is predominantly treated through on-site septic systems; 97% of the parcels use on-site septic systems. Measured water use is used as a proxy for wastewater, which is assumed to have a nitrogen concentration of 35 mg N/L with 25% nitrogen loss within the septic tank and soil adsorption system. Loss in passage through the septic system is consistent with other regional studies (Howes and Ramsey 2000, Weiskel and Howes 1991, Costa et al. 2001, Brawley et al. 2000). The best local quantitative information on Title 5 septic system nitrogen removals has been conducted at DEP's Alternative Septic System Test Center at the Massachusetts Military Reservation and has found that nitrogen removal in the septic tank is small (1-3%) with most of the removal (20-22%) within the soil adsorption system (Costa et al. 2001).

Only 3% of the parcels within the watershed are connected one of the four wastewater treatment facilities. The Mashpee Commons WWTF is located in the Mashpee River subwatershed, while the Stratford Ponds and Willowbend WWTFs are located within the Shoestring Bay subwatershed. The Forestdale School WWTF is located with the Mashpee-Wakeby Pond subwatershed and the Windchime Point WWTF, which is only included in the buildout scenario, is located in the Mashpee River subwatershed (Figure IV-3). It should be noted that the among these WWTF effluent nitrogen concentrations vary across a wide range.

In order to check the reliability of parcel water use as a proxy for wastewater flow, average influent flow at the Mashpee Commons and Willowbend WWTF was compared to average parcel water use 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). Within the Popponesset Bay watershed, the extensive mix of land uses connected to a municipal treatment facility is not available, but average flows from the two private WWTF are available to gauge whether the 90% return flow is an appropriate assumption. Based on average flows, 79% of the Mashpee Commons water use is returned to the WWTF, while 87% of the Willowbend water use is returned to its WWTF. This analysis supports the use of 90% return flow as an appropriate general adjustment for converting water use to wastewater flows in the nitrogen loading assessment within the Popponesset Bay watershed.

Although this adjustment is an appropriate proxy for wastewater flows on parcels with measured water use, 2,318 (28%) of the parcels in the Popponesset 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 available measured water use from similar land uses. Of the 2,318 parcels without water use data, 2,272 (98%) are classified as residential parcels or condominium parcels (land use codes 101 to 112), 29 are commercial (land use codes 300 to 389) and 9 are industrial (land use codes 400 to 439). In order to address the nitrogen load from these parcels, MEP staff reviewed existing water use for residential, commercial, and industrial properties with measured water use (Table IV-3). Within each of these land use categories are numerous different types of uses. For example, within the commercial category are low water users, like small offices or retail with one or two employees, and large water users, like small motels with a dozen or more rooms. The ranges in Table IV-3 are very similar to those observed in the MEP analysis of water use in Chatham.

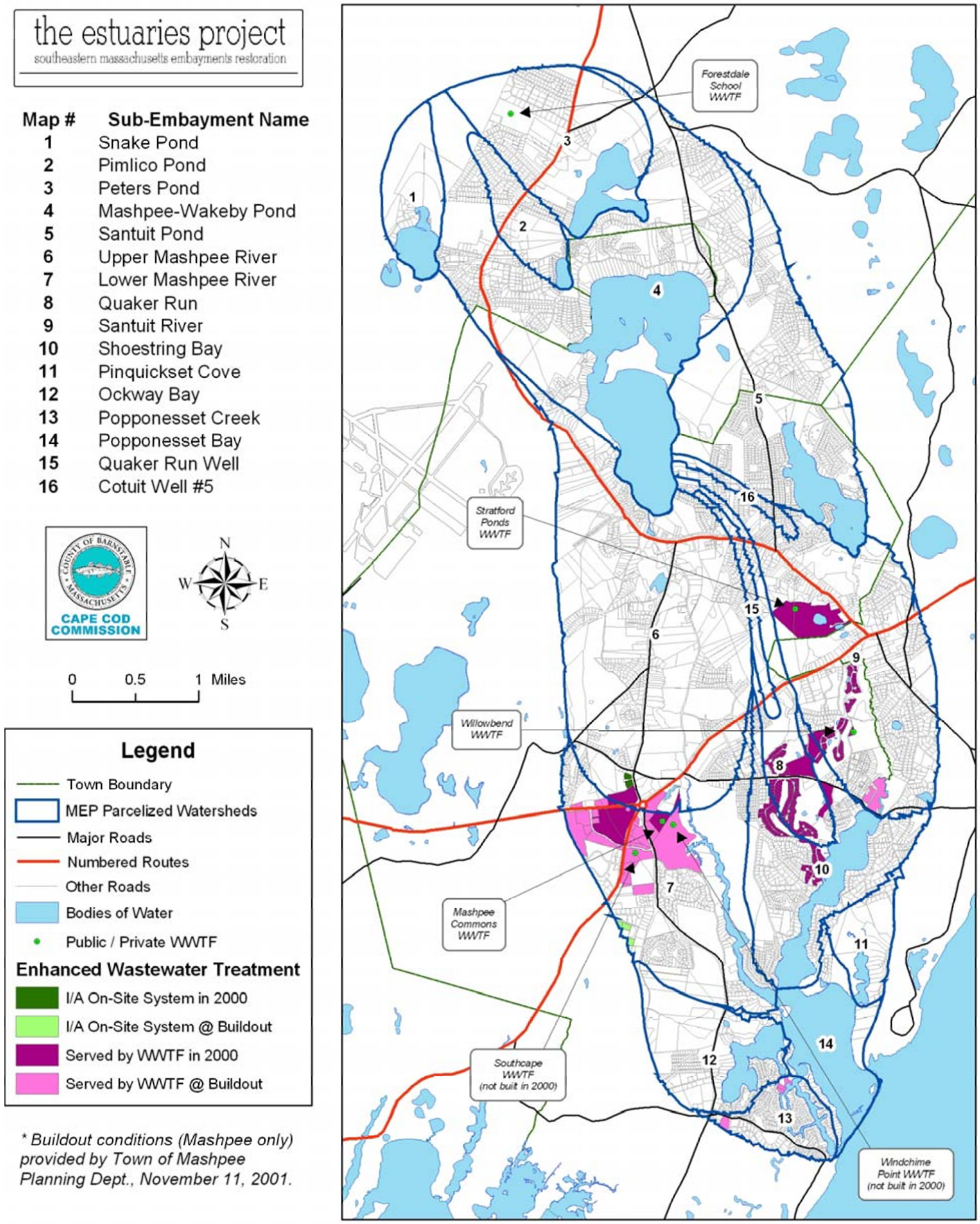


Figure IV-3. Parcels, Parcelized Watersheds, and Private Wastewater Treatment Facilities in the Popponesset Bay watershed.

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	3,462	154	127	0.9 to 3,177
Commercial	300 to 389	47	502	92	12 to 7,343
Industrial	400 to 439	5	286	68	11 to 1,079

Because water use information also forms the basis for evaluation of buildout nitrogen loads and the relatively high percentage of residential properties utilizing wells, MEP staff reviewed other factors to assess whether mean or median water use estimates is most appropriate for residential land uses. The state on-site wastewater regulations (*i.e.*, 310 CMR 15, Title 5) assume that two people occupy each bedroom and each bedroom has a wastewater flow of 110 gallons per day (gpd). Therefore, based on these regulations each person would generate 55 gpd. Average occupancy within the Town of Mashpee during the 2000 US Census was 2.46 people per household, while Barnstable was 2.44 and Sandwich was 2.75. If these occupancies are weighted based on the portion of the Popponesset Bay watershed that each town occupies, the Bay watershed average occupancy is 2.54. If the median water use of 127 gpd is multiplied by 0.9 to correct it to wastewater flows and then divided by 55 gpd, the resulting calculated occupancy is 2.14. In contrast, if the same procedure is applied to the average water use, the resulting occupancy is 2.51, which is approximately the same as the Bay watershed average occupancy. In order to provide a further check whether the average residential water use was appropriate for buildout and 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 the MEP analysis. Based on these analyses, project staff felt that the average residential water use was most appropriate for use in the nitrogen loading calculations for developed residential parcels without water use information and for new residential parcels determined from the buildout assessment.

Similar comparisons were not available for the commercial or industrial water uses, which have a much wider range of land uses, but only represent less than 0.5% of the parcels. However, commercial and industrial building footprints were made available to project staff as part of an impervious surface GIS coverage provided by the Mashpee Planning Department. Project staff used this data to review water use for these properties based on square footage of building and to determine the percentage of each commercial or industrial lot that is occupied by a building. 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 in Mashpee and for all buildout additions. 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 Barnstable or Sandwich portions of the Popponesset 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 in the Towns of Falmouth, Mashpee and Bourne, and related to inland, fresh ponds and embayments sub-watershed regions. 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 fertilize at all, 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 in the survey to have the higher rate of fertilization (loss to groundwater of 3 lb/lawn/yr).

Nitrogen Loading Input Factors: Other

The nitrogen loading factors for 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 Popponesset Bay are listed in Table IV-4. Impervious surfaces in Mashpee (e.g. road, parking, and building areas) were determined from impervious surface coverages provided by the Mashpee Planning Department.

In an early study of the Mashpee River, leaf fall was proposed as an important nitrogen source to the freshwater reach (K-V Associates 1991). We assessed the importance of this potential nitrogen source by evaluating the nitrogen mass delivered to vegetation from rainwater (as the sole source of nitrogen for leaf production). If 100% of the rainfall nitrogen is taken up by plants and converted to leaves in the 10 m (30 ft) swath on both banks to the Mashpee River and if 100% of these leaves fall into the River, then the amount of nitrogen added is less than 0.4% of the watershed loading to the freshwater reach of the River. Since these assumptions are gross over estimates of likely leaf fall, leaf nitrogen inputs to the river were not included in the analysis below. Note that observations of higher N and P in river waters in fall versus spring/summer are easily accounted for by fall plant senescence (particularly in river bank and channel vegetation) and from the observed increases associated with rainfall.

Table IV-4. Primary Nitrogen Loading Factors used in Popponesset 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">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 various watersheds based initially on whether at least 50% or more of the land area of each parcel was located within a respective watershed. Following the assigning of boundary parcels, all large parcels were examined separately and were split (as appropriate) in order to obtain less than a 2% difference between the total land area of each watershed and the sum of the area of the parcels within each watershed. The resulting “parcelized” watersheds are shown in Figure IV-3. This review of individual parcels straddling watershed boundaries included corresponding reviews and individualized assignment of nitrogen loads associated with lawn areas, septic systems, and impervious surfaces. Individualized information for parcels with atypical nitrogen loading (small public water supplies, golf courses, etc.) were also assigned at this stage. DEP and Town records were reviewed to determine water use for small public water supplies (*e.g.*, non-community public water supplies) and golf course superintendents for two golf courses in the study area were contacted to determine fertilizer application rates.

Following the assignment of all parcels to individual watersheds, tables were generated for each of 28 sub-watersheds to summarize water use, parcel area, frequency, sewer connections, private wells, and road area. As mentioned above, these tables were then condensed to 16 subwatersheds following the elimination of the 10-year time of travel subwatersheds.

The 16 individual sub-watershed assessments were then integrated to generate nitrogen loading tables relating to the Mashpee River and Shoestring Bay subembayments, as well as the overall Popponesset 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 separated into various nitrogen sources to support potential nitrogen mitigation alternative development: 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 effort is the kg N per year loaded into each sub-embayment's contributing area, by land use category (Figures IV-4 a-c), which is then adjusted for natural nitrogen attenuation during transport before use in the Linked Model.

Freshwater Pond Nitrogen Loads

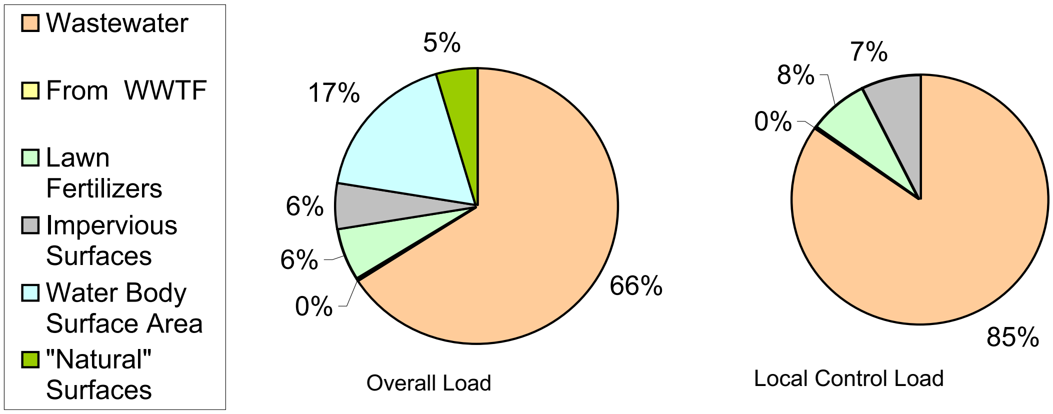
Freshwater ponds on Cape Cod are generally kettle hole depressions that intercept the surrounding groundwater table revealing what some call "windows on the aquifer." Since the ponds are connected to the aquifer, the ecosystems in these ponds have the opportunity to alter the nitrogen loads flowing into them via groundwater flow. This reduction in the nitrogen load takes place as a result of biological interaction within the pond. Following this reduction, the loads flow back into the groundwater system along the downgradient side of the pond or through a stream outlet and eventual discharge into the downgradient embayment. Table IV-5 N Load summary includes both the unattenuated (nitrogen load to each subwatershed) and attenuated nitrogen loads. Nitrogen attenuation in the ponds was assumed to be 50%.

This attenuation assumption was checked through the use of pond water quality information 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 Program. 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, Pimlico, Peters, Mashpee-Wakeby, Santuit, Ashumet, Johns, and Moody (Figure IV-1). Water samples were analyzed at the SMAST laboratory for total nitrogen, total phosphorus, chlorophyll *a*, alkalinity, and pH. This data was supplemented with data collected on Ashumet, Johns, Peters, Mashpee-Wakeby, and Snake ponds through various Massachusetts Military Reservation (MMR) monitoring programs (*e.g.*, AFCEE, 1998).

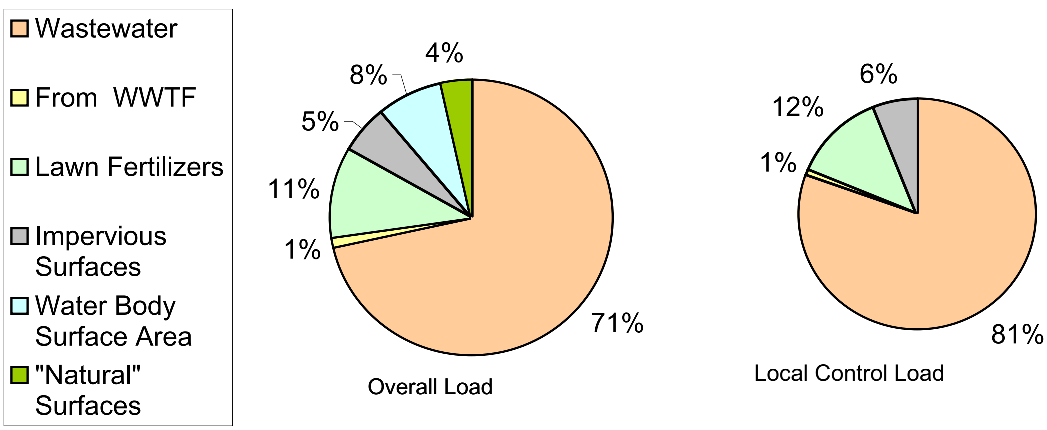
In order to estimate nitrogen attenuation in the ponds physical and chemical data for each pond was assessed. Available bathymetric information was reviewed relative to measured pond temperature profiles to determine the epilimnion (*i.e.*, well mixed, homothermic, upper portion of the water column) 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-5. Popponesset Bay System Nitrogen Loads.

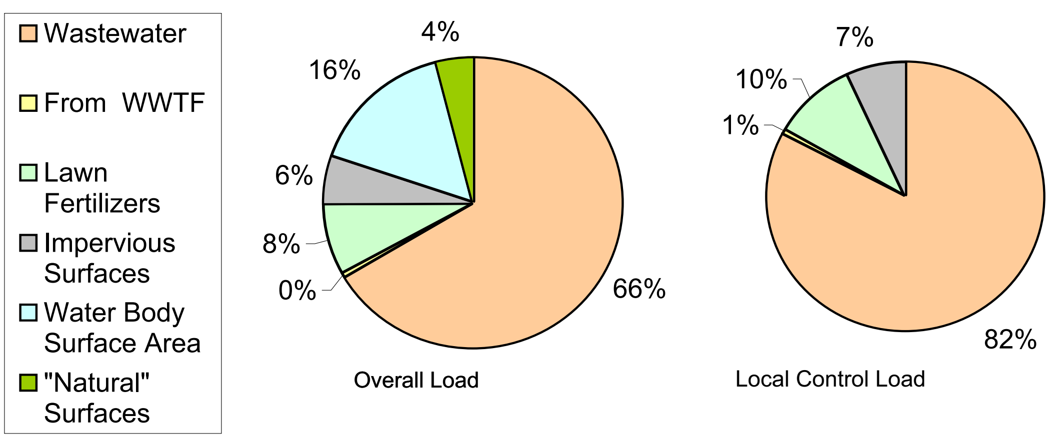
*All values in kilograms/year		Popponesset Bay Subwatershed N Loads by Input:							% of Pond Outflow	Present N Loads			Buildout N Loads		
Name	Watershed ID#	Wastewater	From WWTF	Lawn Fertilizers	Impervious Surfaces	Water Body Surface Area	"Natural" Surfaces	Buildout		UnAtten N Load	Atten %	Atten N Load	UnAtten N Load	Atten %	Atten N Load
Popponesset Bay System	**11, 12, 13, 14 + Shoestring Bay + Mashpee River**	**32300**	**227**	**3765**	**2668**	**7584**	**1971**	**9394**		**48513**		**31885**	**57804**		**39644**
Mashpee River	6, 7 + MWP	16199	54	1458	1411	4238	1153	6643		24512		13010	31051		18796
Upper Mashpee River	6 + MWP	12692	0	1206	1016	3996	941	4566		19851	30%	8349	24314	30%	12058
Mashpee-Wakeby Pond (MWP)	1, 2, 3, 4	7828	0	676	513	3989	503	771	100%	13509	50%	5585	14176	50%	5781
Direct to MWP	4	4572	0	421	321	3212	307	117		8833			8950		
Snake Pond (SNP)	1	5	0	3	2	121	17	10	34%	148	50%	74	54	50%	27
Pimlico Pond (PIP)	2	1139	0	90	63	74	42	563	100%	1407	50%	704	1971	50%	985
Peters Pond (PEP)	3	2112	0	163	126	582	138	80	100%	3121	50%	1560	3201	50%	1601
Lower Mashpee River	7	3507	54	251	395	242	212	2077		4661		4661	6737		6737
Shoestring Bay	8, 9, 10, 15, 16 + SAP	12986	173	1978	986	1379	636	2058		18139		13012	20196		14293
Santuit River	9, 16	8853	173	1075	675	564	440	1623		11780	30%	6653	13403	30%	7500
Cotuit Well No. 5	16	329	0	33	16	0	11	66		388		388	454		454
Quaker Run	8, 15	1598	0	405	99	0	82	232		2183		2183	2415		2415
Quaker Run Wells	15	659	0	37	23	0	18	51		736		736	787		787
Santuit Pond (SAP)	5	4386	0	429	326	758	216	1112	100%	6114	50%	3057	7226	50%	3613
Ockway Bay	12	874	0	103	90	399	83	402		1549		1549	1951		1951
GW Flow to Popponesset Ba	11, 13, 14	2241	0	226	180	1568	98	290		4314		4314	4604		4604
Pinquickset Cove	11	210	0	19	10	106	40	80		385		385	465		465
Popponesset Creek	13	1455	0	171	146	0	31	151		1803		1803	1954		1954
Popponesset Bay	14	576	0	36	24	1462	28	59		2126		2126	2185		2185



a. Mashpee River



b. Shoestring Bay



c. Popponesset Bay System

Figure IV-4. Land use specific unattenuated watershed based nitrogen load (by percent) to Mashpee River, Shoestring Bay, and entire Popponesset Bay system.

Table IV-6 summarizes the pond attenuation estimates calculated from land-use modeled nitrogen inflow loads and nitrogen loads 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 varies 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% 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).

Table IV-6. Nitrogen attenuation by Freshwater Ponds in the Popponesset 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 Popponesset 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 %
Mashpee-Wakeby	MA-634	725.8	29.0	4.5	86%
Peters	SA-526	130.6	17.4	1.8	80%
Pimlico	SA-615	16.4	7.6	0.3	89%
Santuit	MA-718	170.5	2.5	0.3	75%
Snake	SA-568	83.5	10.1	2.0	51%
				Mean	73%
				s.d.	16%

Since groundwater outflow from a pond can enter more than one downgradient sub-watershed, the length of shoreline on the downgradient side of the pond was used to apportion the attenuated nitrogen load to respective downgradient watersheds. The apportionment was based on the percentage of pond discharging shoreline bordering each downgradient sub-watershed. The percentages of shoreline from larger ponds are shown in Table IV-5. For the present analysis, all of the outflow from Santuit Pond was discharged to the Santuit River. A small amount of Santuit Pond water is directed though cranberry bog operations to Lovells Pond, outside of the Popponesset Bay watershed. However, water withdrawals for bog operations are only periodic and are generally small compared to the total annual Santuit River flows. Major withdrawals for bog operations generally occur in fall for harvest and in winter for frost protection. Additional data could be collected to yield a precise estimate of this watershed nitrogen export.

Buildout

In order to gauge potential future nitrogen loads resulting from continuing development, the potential number of future residential, commercial, and industrial lots within each subwatershed was determined from the GIS database (Figure IV-5). Buildout of parcels within the Town of Mashpee portion of the Popponesset 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 Towns of Sandwich and Barnstable were based on subdivisions 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 buildout is presented as unattenuated and attenuated loads in Table IV-5. Only attenuated loads were used in the water quality modeling. Buildout loads, or any alternative future load scenarios, can be used to evaluate water quality impacts of current or alternative zoning and/or other land use regulations.

During the course of discussion of the nitrogen loading analysis with town representatives, MEP staff agreed to provide a limited evaluation of the land use changes that have occurred since the acceptance of the land use databases used in the nitrogen loading. The nitrogen loading analysis discussed above uses Town of Barnstable assessor and land use information from the year 2000. Figure IV-6 shows the changes that occurred in the Barnstable portion of the Popponesset Bay watershed between 2000 and 2004; undeveloped lands in 2000 are outlined in green with 2004 land uses shown. Of the 261 acres in this portion of the watershed classified as undeveloped in 2000, 13 acres (5% of 261) were converted to residential land use by 2004. These 13 acres were turned into 29 lots or an increase of 4% from the number of lots in 2000. These differences are slight, would result in even smaller percentage increases in the overall nitrogen loads, and are within the margin of error in the overall linked model results.

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 of the overall Popponesset Bay embayment system under study 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 Popponesset 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.

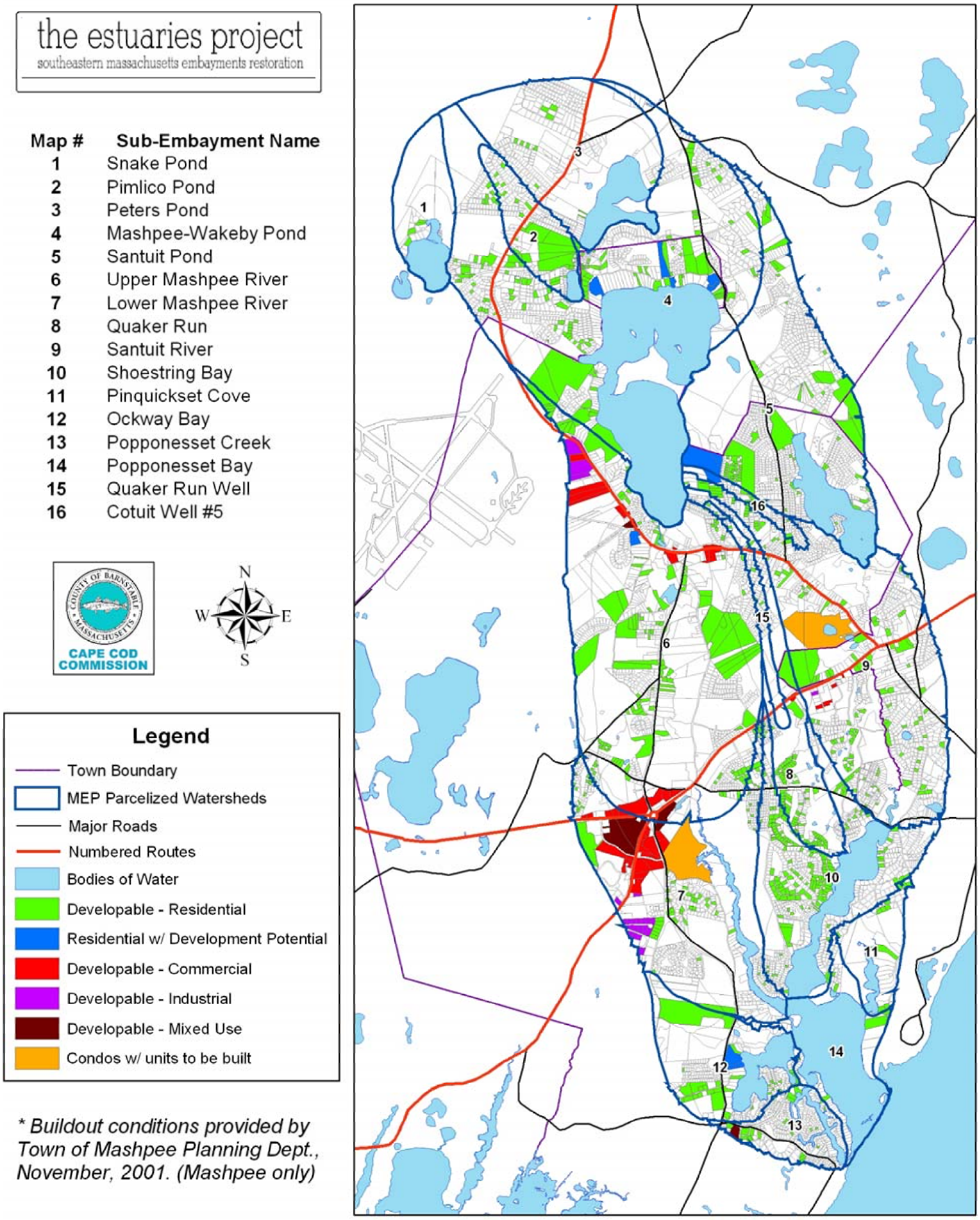


Figure IV-5. Distribution of present parcels that are potentially developable within the Popponesset Bay watershed.

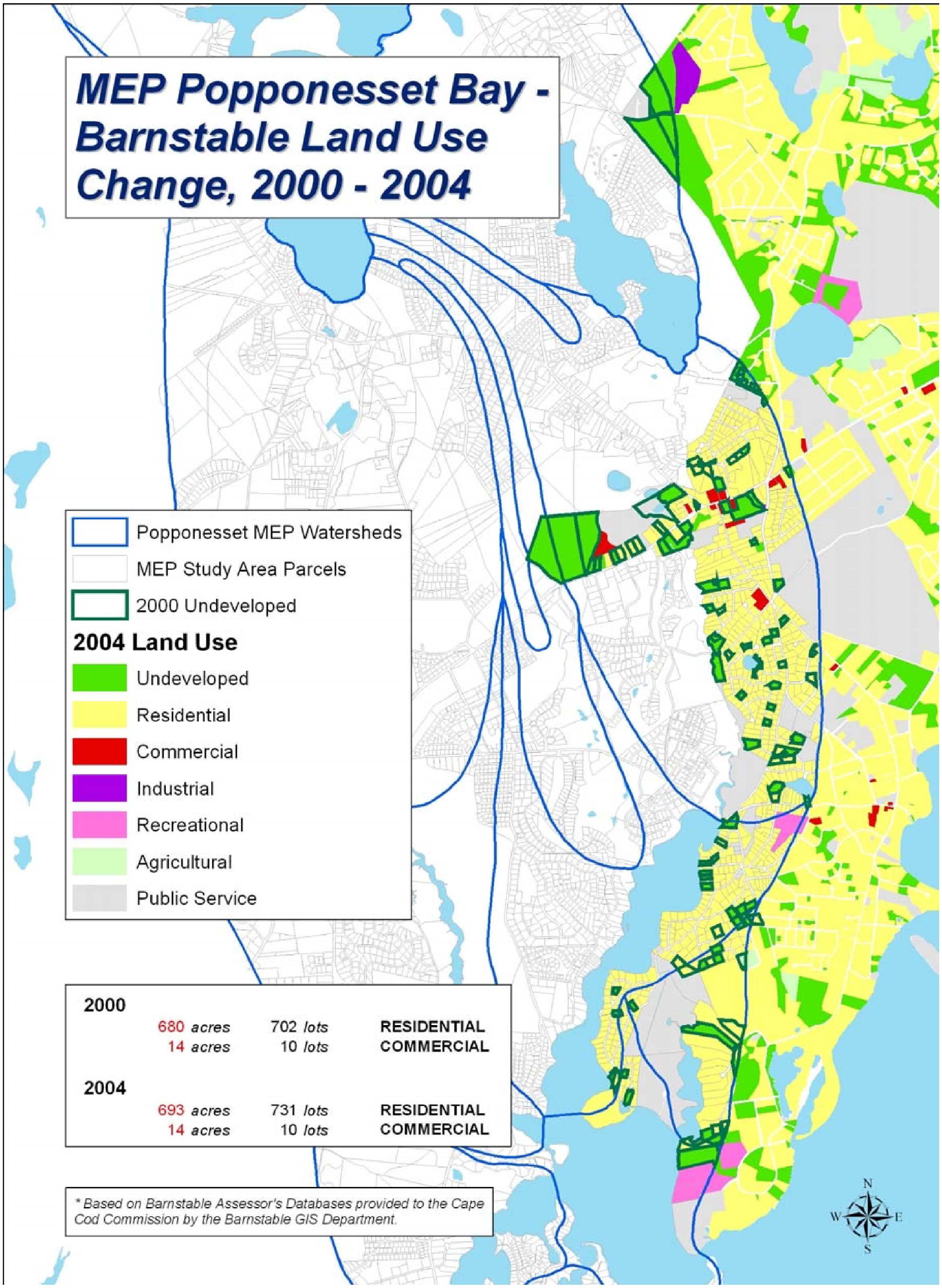


Figure IV-6. Change in Town of Barnstable parcel and land use in the Popponesset Bay watershed between year 2000 and 2004. 2004 land use is shown with 2000 undeveloped parcels outlined in green.

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 multiple studies on natural attenuation relating to sub-embayments of the Popponesset Bay System in addition to the natural attenuation measures by fresh kettle ponds, addressed above. These additional site-specific studies were conducted in each of the 2 major surface water flow systems, i.e. the Mashpee River discharging to the tidal portion of the Mashpee River sub-embayment and the Santuit River discharging to Shoestring Bay).

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 Mashpee River (at Route 28) and Santuit River (at the tidal reach) provide a direct integrated measure of all of the processes presently attenuating nitrogen in the sub-watersheds upgradient from the gauging sites. These upper watershed regions account for more than half of the entire watershed area to the Popponesset Bay System. Flow and nitrogen load were measured at each site for 16 months of record (Figure IV-7). During the study period, velocity profiles were completed on each river every month to two months. Periodic measurement of flows over the entire stream gauge deployment period allowed for the development of a stage-discharge relationship (rating curve) that could be used to obtain flow volumes from the detailed record of stage measured by the continuously recording stream gauges. A complete annual record of stream flow (365 days) was generated for both the Mashpee River and the Santuit River. The annual flow records for both rivers were merged with the nutrient data sets generated through the weekly water quality sampling to determine nitrogen loading rates to the tidally influenced portion of the Mashpee River and to the headwaters of Shoestring Bay. Comparing these measured nitrogen loads based on stream flow and water quality sampling to predicted loads based on the land use analysis allowed for the determination of the degree to which natural biological processes within the watershed currently reduce (percent attenuation) nitrogen loading to the Popponesset Bay embayment system.

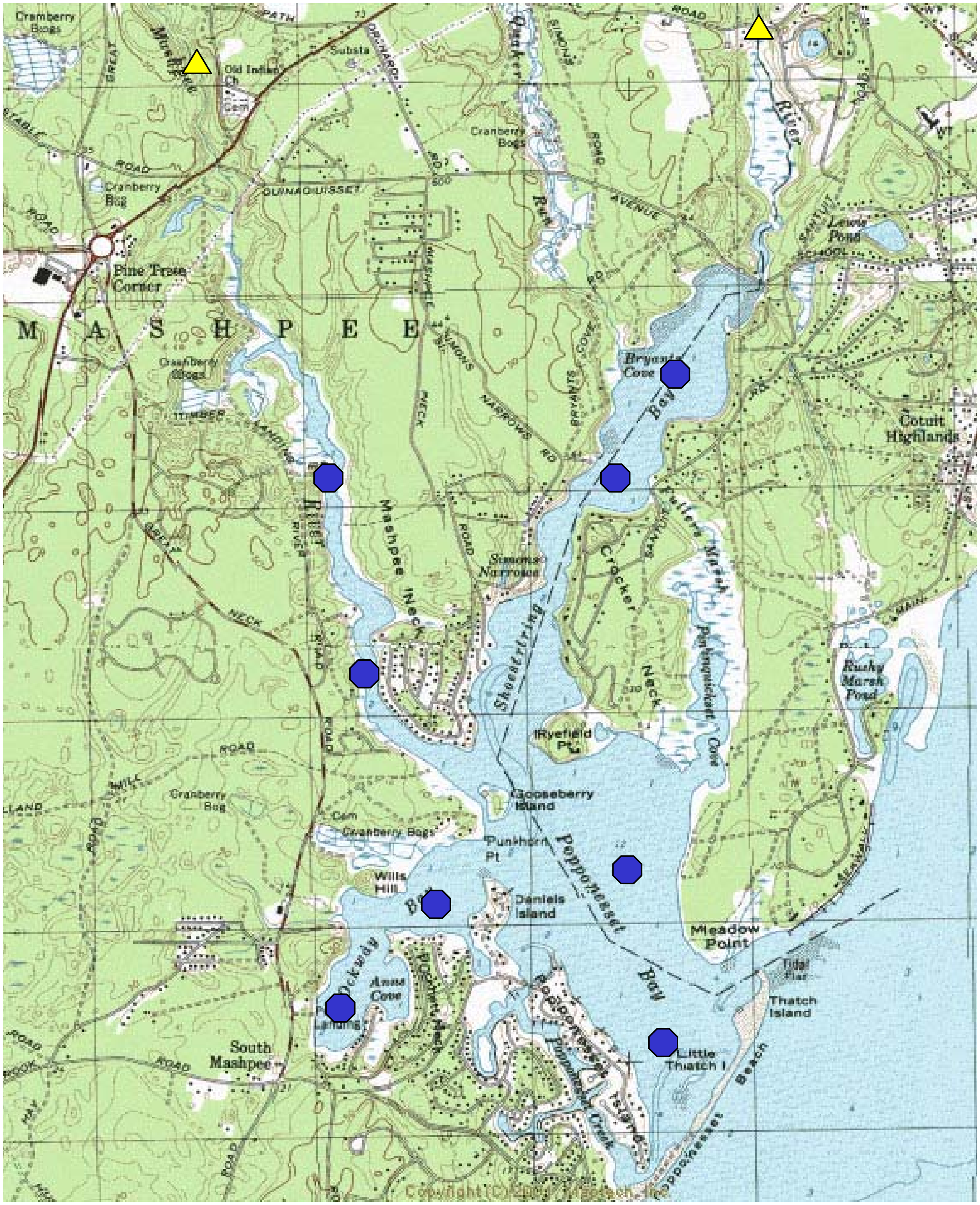


Figure IV-7. Location of Stream gauges (yellow triangles) and benthic coring locations (blue dots) in the Popponesset Bay System.

An additional analysis of flow was undertaken relative to the USGS long term record (1989 – 2002) of flow in the Quashnet River, adjacent to the Popponesset Bay watershed, in order to gage the degree to which the 2003 water year (2002 baseflow period to 2003 baseflow period) in this geographic region was representative of average hydrologic conditions. Using the USGS daily flow record for the Quashnet River for 1989 to 2002, average flow during the 6-month period (April to September) was calculated for the 15-year period 1989 to 2002. The mean flow for the 6-month period, based on the 15-year record of daily flow, was 41,248 m3/day. Considering the USGS flow data for 2003, the mean flow in the Quashnet River for the 6-month period was 49,983 m3/day which is 21 percent higher than the 15 year mean flows for these months. It is therefore likely that the MEP determined flows in the Mashpee River and the Santuit River are also about 20% higher than the long-term average for the same period, since these river systems are proximal to the Quashnet River watershed. Therefore, the percent attenuation determined by the MEP for both the Mashpee River and the Santuit River is likely to be slightly conservative (lower) as residence times in each river system may be a bit shorter and the nitrate concentrations lower than under the average conditions of lower flow.

A similar analysis of USGS historical long term daily flows in the Quashnet River was completed for the 6-month period (October 2002 to March 2003) to ascertain the degree to which flows in the Mashpee and Santuit Rivers may be below mean flow for that period. It was determined that flow in the Quashnet River was approximately 2 percent below mean flow during the period October 2002 to March 2003 and 30 percent below mean flow conditions for the previous period April 2002 to September 2002. Given these low flow conditions in the Quashnet River during the hydrologic year prior to the acquisition of the MEP flow record (September 2002 to September 2003), it is likely that the MEP determined flows in the Mashpee River and the Santuit River are also lower than the long-term average for the same period, since these river systems are proximal to the Quashnet River watershed. As such, percent attenuation during the lower flow periods may be slightly higher than during average flow conditions when stream flow is relatively higher and residence times are shorter. From the perspective of overall nutrient loading to the Popponesset Bay system on an annualized basis, the potentially higher attenuation during the period when stream flow is below average flow conditions is likely to be offset by the lower attenuation rates when the stream flow record appears to be above average flow conditions. As such, the annual attenuated nitrogen load to the Popponesset Bay embayment system during the 2002- 2003 study period is considered representative of average loading conditions.

IV.2.2 Surface Water Discharge and Attenuation of Watershed Nitrogen: Mashpee River to Mashpee River (lower)

Mashpee – Wakeby Pond is one of the largest ponds on Cape Cod and unlike many of the freshwater ponds, this pond has stream outflow rather than discharging solely to the aquifer along its down-gradient shore. This stream outflow, the Mashpee 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 Mashpee 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 Mashpee River above the gauge site and the measured annual discharge of nitrogen to the tidal portion of the Mashpee River, Figure IV-7.

At the Mashpee River gauge site, a continuously recording vented calibrated water level gauge was installed to yield the level of water in the freshwater portion of the Mashpee River that carries the flows and associated nitrogen load to the Bay. Calibration of the gauge was checked monthly. The gauge on the Mashpee River was installed on December 3, 2001 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to an instrument upgrade in July of 2002, instrument failure, and vandalism, stage data collection was extended until October 3, 2003. The 12 month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Mashpee River site based upon these measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to the estuarine portion of the Mashpee River (Table IV-7 and Figure 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 each gauge site.

The annual freshwater flow record for the Mashpee River and Santuit River (see below), determined from measured stage and the stage – discharge relation developed by the MEP, was compared to the flows determined by the USGS modeling effort (Table III-1). The measured freshwater discharge from the Mashpee River and Santuit River were both ~1/3 lower than the long-term average modeled flows. The lower values are consistent with the extremely low groundwater levels during the initial months of the study period. Given that the streamflows are significantly groundwater fed, and the fact that the ratio of the Mashpee River/Santuit River flows were consistent between the measured (2.00) and modeled (2.25) discharges, the watershed and river datasets appear to be in balance.

Total nitrogen concentrations within the Mashpee River outflow were relatively high, 0.593 mg N L^{-1}, yielding an average daily total nitrogen discharge to the estuary of 15,562 g/day (15.6 kg/d) and a measured total annual TN load of 5,680 kg/yr. In the Mashpee River, nitrate was the predominant form of nitrogen (54%), 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 Mashpee 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 (15.6 kg N d^{-1}, 5680 kg yr^{-1}) discharged from the freshwater Mashpee River and the nitrogen mass entering from the associated watershed (57.4 kg N d^{-1}, 20,941 kg yr^{-1}) the integrated measure of nitrogen attenuation by the pond/river ecosystem is 71%. This is consistent with the land-use model which yielded and integrated nitrogen attenuation of 52%, since pond and stream attenuation in the watershed model use conservative attenuation factors (see Table IV-6). The directly measured nitrogen loads from the rivers were used in the Linked Watershed-Embayment Modeling of water quality (see Chapter VI, below).

Table IV-7. Comparison of water flow and nitrogen discharges from Mashpee River to Mashpee River (lower) and Santuit River to Shoestring Bay. The "Stream" data is from the MEP stream gauging effort. Watershed data is based upon the MEP watershed modeling effort by USGS.

Stream Discharge Parameter	Mashpee River Discharge to Mashpee River (lower)[b]	Santuit River Discharge to Shoestring Bay[c]	Data Source
Total Days of Record [a]	365	365	(1)
Flow Characteristics:			
Stream Average Discharge (m3/d)	26,223	13,164	(1)
Contributing Area Long-term Average Discharge (m3/d)	45,220[b]	20,093[c]	(2)
Discharge Stream 2002-03 vs. Long-term Discharge	58%	66%	
Nitrogen Characteristics:			
Stream Average Nitrate + Nitrite Concentration (mg N/L)	0.318	0.702	(1)
Stream Average Total N Concentration (mg N/L)	0.593	1.184	(1)
Nitrate + Nitrite as Percent of Total N (%)	54%	59%	(1)
Total Nitrogen (TN) Average Measured Stream Discharge (kg/d)	15.56	15.58	(1)
TN Average Contributing Area Attenuated Load (kg/d)	22.59	18.06	(2)
TN Average Contributing Area UN-attenuated Load (kg/d)	53.89	32.04	(3)
Attenuation of Nitrogen in Pond/Stream (%)	71%	51%	(4)

[a] from 09/24/02 to 09/24/03 (Mashpee River and Santuit River gauges)
[b] flow and N load to Mashpee River include Mashpee – Wakeby Pond Contributing Area,
[c] flow and N load to Santuit River include Santuit Pond Contributing Area

(1) MEP gauge site data
(2) Calculated from MEP watershed delineations to Mashpee – Wakeby Pond, and Santuit Pond; the fractional flow path from each sub-watershed which contribute to Mashpee River and Santuit River Flow; and the annual recharge rate.
(3) As in footnote #2, with the addition of pond and stream conservative attenuation rates.
(4) Calculated based upon the measured TN discharge from the river vs. the unattenuated watershed load.

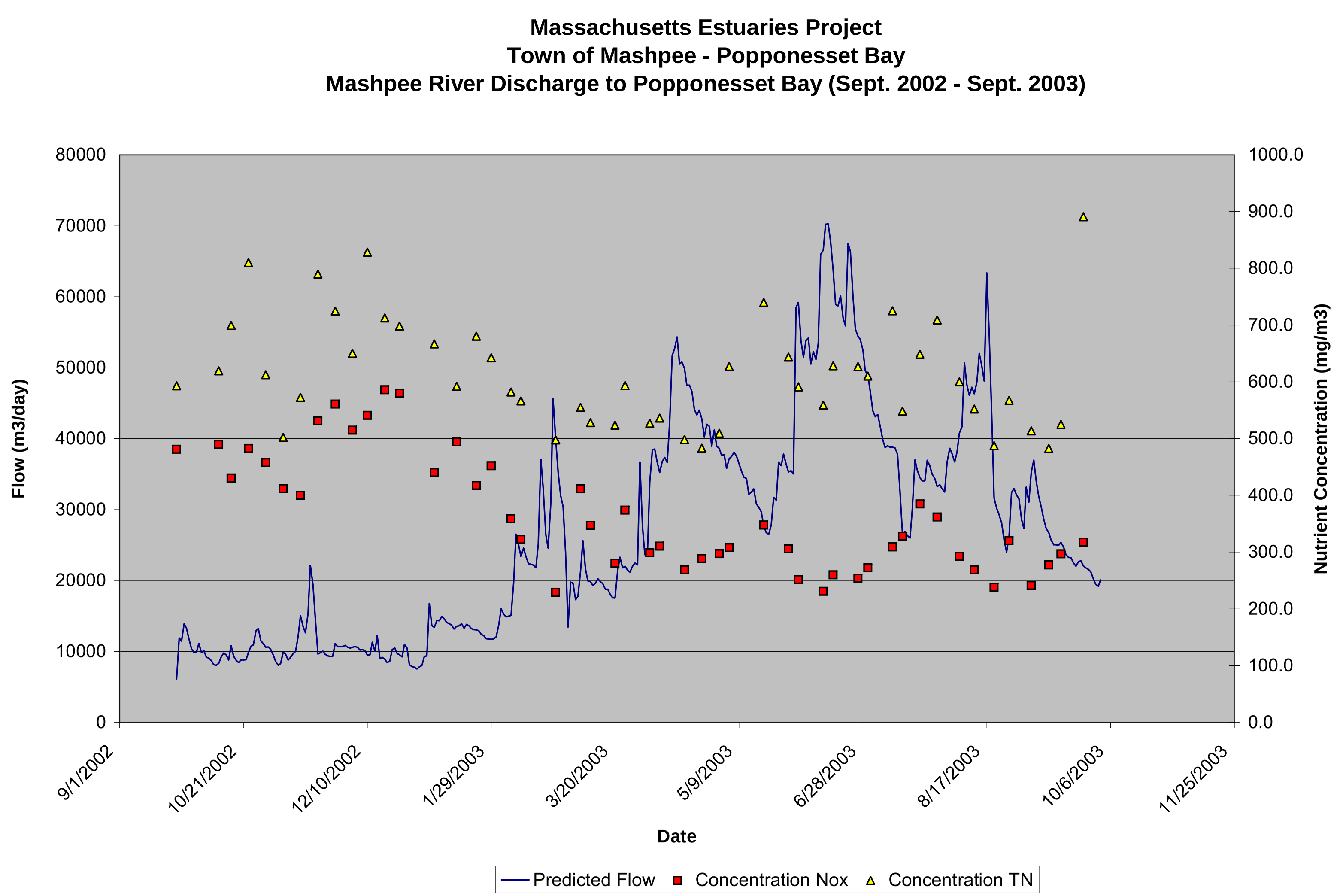


Figure IV-8. Mashpee River annual discharge developed from a stream gauge maintained above the tidal reach of the lower Mashpee River estuarine waters. Nutrient samples (Nox – Nitrate+Nitrite) were collected weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine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

IV.2.3 Freshwater Discharge and Attenuation of Watershed Nitrogen: Santuit River to Shoestring Bay

Santuit Pond is one of the larger ponds within the study area and unlike many of the freshwater ponds, the Santuit Pond has stream outflow to the Santuit River, rather than discharging solely to the aquifer on the down-gradient shore. As for the Mashpee River (see IV.2.2 above) this stream outflow may serve to decrease the pond attenuation of nitrogen, but it also provides for a direct measurement of the nitrogen attenuation. Nitrogen attenuation also occurs within the wetlands and stream-bed associated with the Santuit 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 Santuit River above the gauge site and the measured annual discharge of nitrogen.

At the Santuit River gauge site (Figure IV-7), a continuously recording vented calibrated water level gauge was installed to yield the level of water for the determination of freshwater flow. Calibration of the gauge was checked monthly. The gauge on the Santuit River was installed on December 3, 2001 and was set to operate continuously for 16 months such that two summer seasons would be captured in the flow record. Due to the desire to have simultaneous measurement of river discharge from the Mashpee and Santuit Rivers, stage data collection was extended until October 3, 2003 (to match the Mashpee River). The 12 month uninterrupted record used in this analysis encompasses the summer 2003 field season.

River flow (volumetric discharge) was measured monthly using a Marsh-McBirney electromagnetic flow meter. A rating curve was developed for the Santuit River site based upon these measurements and measured water levels at the gauge site. The rating curve was then used for conversion of the continuously measured stage data to obtain daily freshwater flow volume. Water samples were collected weekly for nitrogen analysis. These measurements allowed for the determination of both total volumetric discharge and nitrogen mass discharge to the headwaters of Shoestring Bay (Figure IV-9 and Table IV-8).

Total nitrogen concentrations within the Santuit River outflow were relatively high, 1.18 mg N L^{-1} (2 times that observed in the Mashpee River). However the total nitrogen load was similar to the Mashpee River given that the flow was ~1/2 as high. Average daily total nitrogen discharge from the Santuit River to the estuary was 15,584 g/day (15.6 kg/d) with a measured total annual TN load of 5,688 kg/yr. As in the Mashpee River, nitrate was the predominant form of nitrogen (59%), 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 Santuit 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 Shoestring Bay. Based upon the lower measured nitrogen load (15.6 kg N d^{-1}, 5688 kg yr^{-1}) discharged from the Santuit River and nitrogen mass entering from the associated watershed (40.0 kg N d^{-1}, 14,615 kg yr^{-1}), the integrated measure of nitrogen attenuation by the pond/river ecosystem is 51%. This is consistent with the land-use model which yielded and integrated nitrogen attenuation of 44%, since pond and stream attenuation in the watershed model use conservative attenuation factors (Table IV-6). Directly measured nitrogen loads from the rivers were used in the Linked Watershed-Embayment Modeling of water quality (Chapter VI).

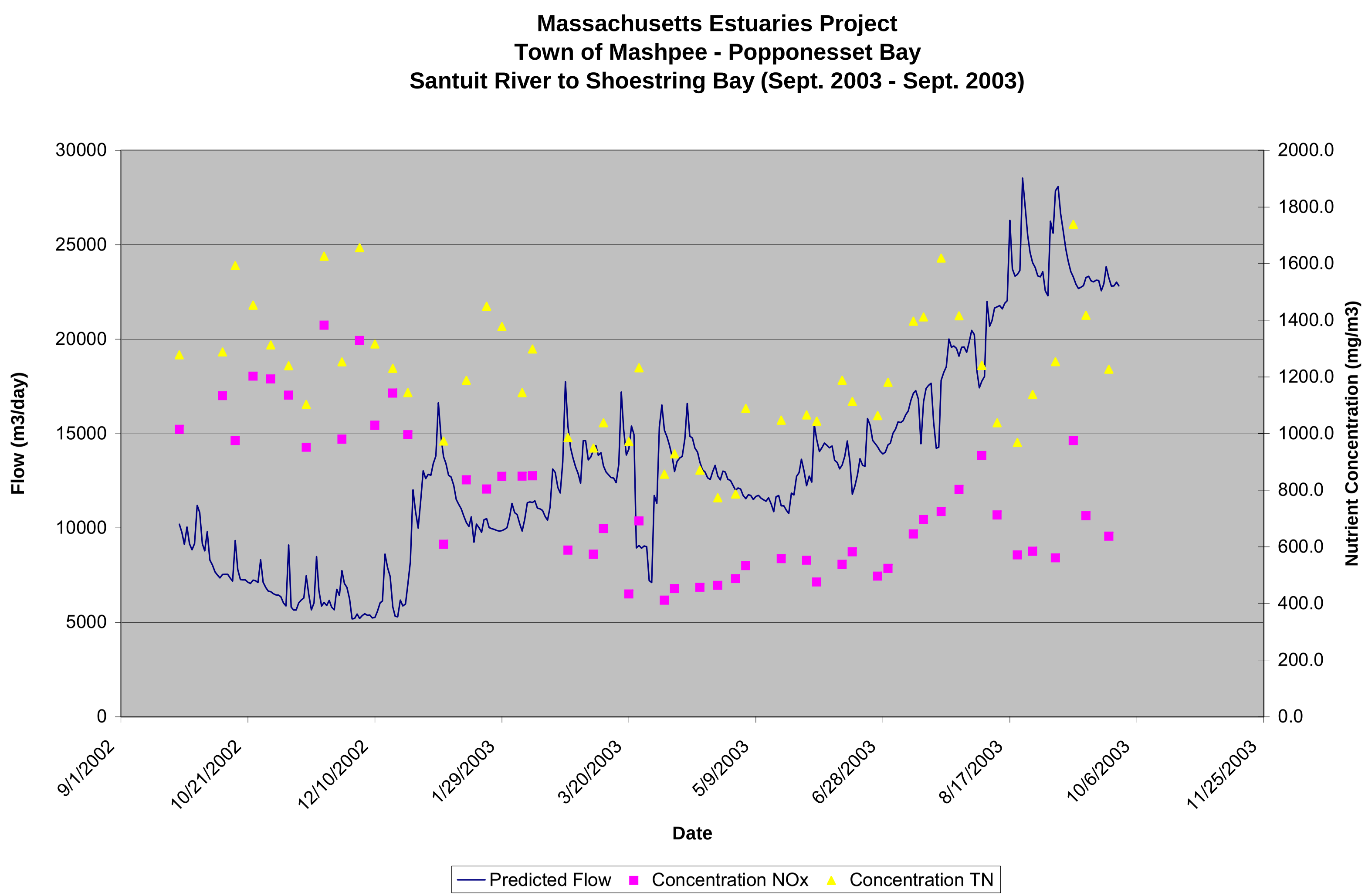


Figure IV-9. Santuit River annual discharge developed from a stream gauge maintained in the outflow from Santuit Pond discharging to Shoestring Bay. Nutrient samples (Nox – Nitrate+Nitrite) were collected approximately weekly and analyzed for inorganic and organic nitrogen species. These data were used to determine both annual flow and total nitrogen transport for determining nitrogen attenuation (see Table IV-5).

Table IV-8. Summary of Flow and Nutrient loads from both the Mashpee River discharging to tidally influenced Mashpee River (lower) and the Santuit River discharging to Shoestring Bay

SYSTEM	PERIOD	DISCHARGE (m3/yr)	ATTENUATED LOAD (Kg/yr)	
			Nox	**TN**
MASHPEE RIVER	September 24, 2002 - September 24, 2003	**9571405**	**3045**	**5680**
SANTUIT RIVER	September 24, 2002 - September 24, 2003	**4805008**	**3374**	**5688**

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

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 Popponesset Bay embayment predominantly in highly bioavailable forms from the surrounding upland watershed and more refractory forms in the inflowing tidal waters. If all of the nitrogen remained within the watercolumn (once it entered), then predicting watercolumn nitrogen levels would be simply a matter of determining the watershed loads, dispersion, and hydrodynamic flushing. However, as nitrogen enters the embayment from the surrounding watersheds it is predominantly in the bioavailable form nitrate. This nitrate and other bioavailable forms are rapidly taken up by phytoplankton for growth, i.e. it is converted from dissolved forms into phytoplankton "particles". Most of these "particles" remain in the watercolumn for sufficient time to be flushed out to a downgradient larger waterbody (like 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. Ockway Bay, Shoestring Bay, 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 we have investigated, 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 Popponesset Bay System, in order to determine the contribution of sediment regeneration to nutrient levels during the most sensitive summer interval (July-August), sediment samples were collected and incubated under *in situ* conditions. Sediment samples were collected from 8 sites (Figure IV-10) in August 1998, June, July, and early September 1999. 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. These measurements were made by 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. The number of core samples from each site (see Figure IV- 10) per incubation were as follows:

- Station 1 – 3 cores (Popponesset Bay main bay)
- Station 2 – 3 cores (Popponesset Bay main bay)
- Station 3 – 1 core (Ockway Bay)
- Station 4 – 2 cores (Ockway Bay)
- Station 5 – 2 cores (Mashpee River)
- Station 6 – 2 cores (Mashpee River)
- Station 7 – 2 cores (Shoestring Bay)
- Station 8 – 1 core (Shoestring Bay)

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

Sediment-watercolumn exchange 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 on the shores of Ockway 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) and ortho-phosphate (Murphy and Reilly 1962) assays were 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.

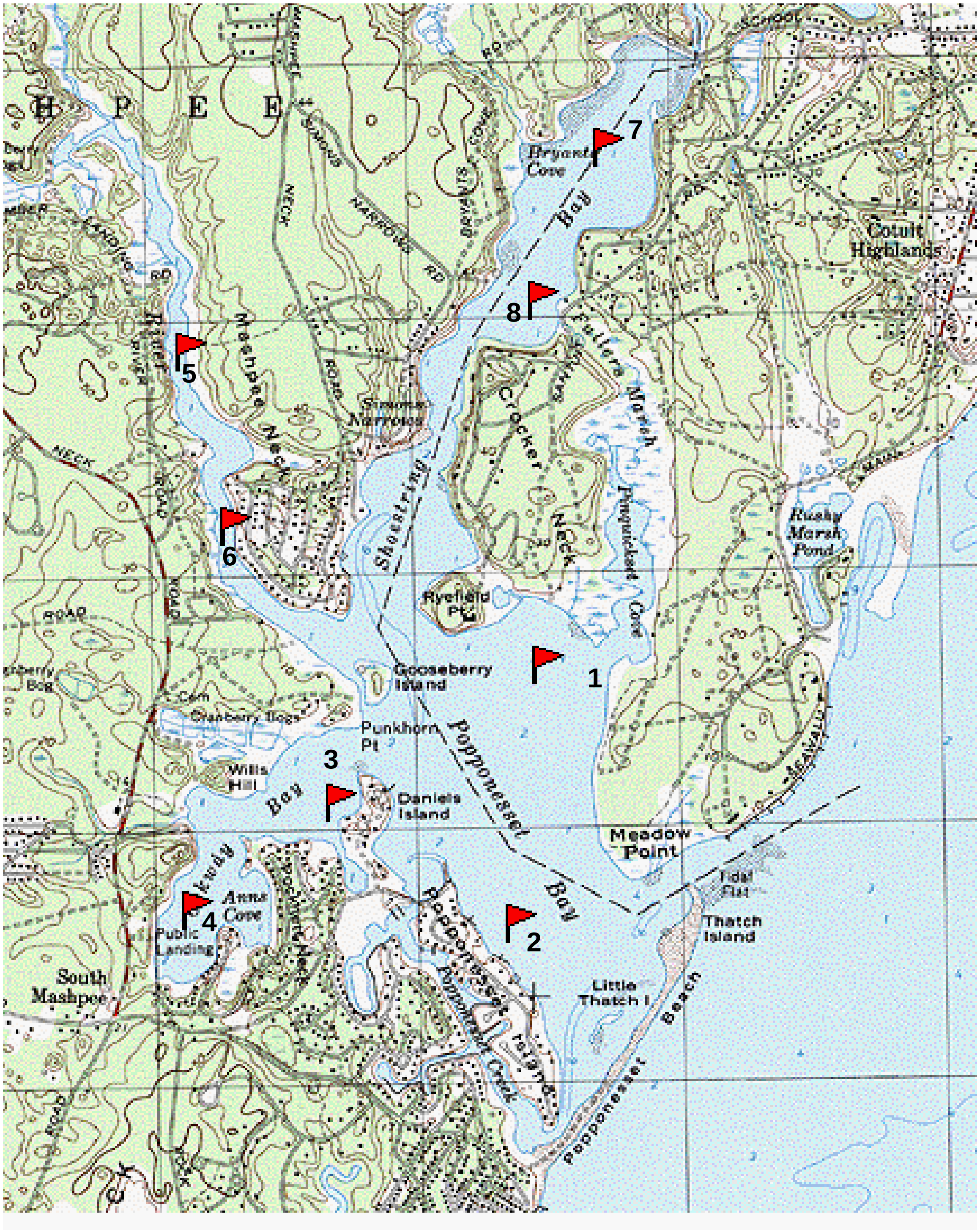


Figure IV-10. Popponesset Bay System locations (red flags) of sediment sample collection for determination of nitrogen regeneration rates. Numbers are for reference in Table IV-9.

Table IV-9. Rates of net nitrogen return from sediments to the overlying waters of the Popponesset Bay System. These values are combined with the basin areas to determine total nitrogen mass in the water quality model (see Chapter VI). Measurements represent June – early September rates.

		Sediment Nitrogen Release		
Sub-Embayment	Station	Mean mg N m^{-2} d^{-1}	std. dev. mg N m^{-2} d^{-1}	N
Mashpee River, Upper-Mid	5	85.40	10.52	8
Mashpee River, Lower	6	59.22	36.42	8
Shoestring Bay, Upper	7	-13.81	15.83	8
Shoestring Bay, Lower	8	-17.05	20.20	4
Ockway Bay, Upper	4	15.85	25.14	8
Ockway Bay, Lower	3	-11.45	9.67	3
Popponesset Bay, Upper	1	4.37	2.98	12
Popponesset Bay, Lower	2	-12.52	12.42	12

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 some denitrification of produced inorganic nitrogen before it can “escape” to the overlying waters. However, this net sink evaluation of coastal sediments is based upon annual fluxes. If seasonality is taken into account, it is clear that sediments undergo periods of net input and net output. The net output is generally during warmer periods and the net input is during colder periods. The result can be an accumulation of nitrogen within late fall, winter, and early spring and a net release during summer. The conceptual model of this seasonality has the sediments acting as a battery with the flux balance controlled by temperature (Figure IV-11).

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

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

Sediment sampling was conducted within each of the sub-embayments of the Popponesset Bay System in order to obtain the nitrogen regeneration rates required for parameterization of the water quality model (Figure IV-10). The distribution of cores was established to cover gradients in sediment type, flow field and phytoplankton density. The rate measurements conducted on the 4 sampling dates were averaged. 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.

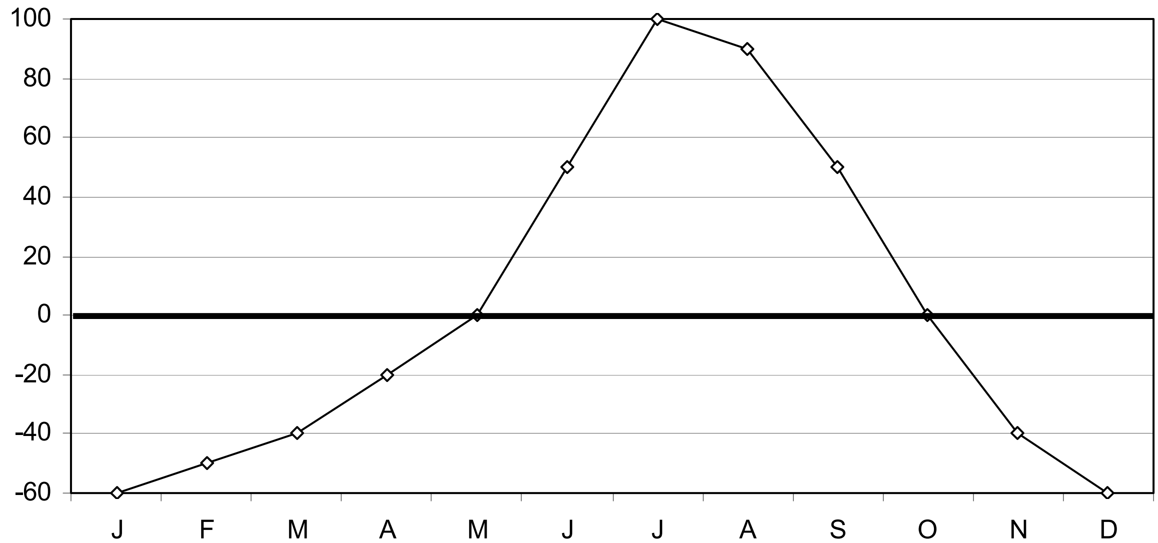


Figure IV-11. Conceptual diagram showing the seasonal variation in sediment N flux, with maximum positive flux (sediment output) occurring in the summer months, and maximum negative flux (sediment up-take) during the winter months.

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 Popponesset Bay System for use in the water quality modeling effort (Chapter VI) are presented in Table IV-9. It is clear that the sediments within the tidal reach of the Mashpee River represent a significant summer source of nitrogen to the overlying waters. This partially reflects the high phytoplankton production within (Chapter VII) and high nitrogen loading to this sub-embayment. In addition, the Mashpee River appears to function as a salt marsh system with a large single tidal channel. Other basins, more typical of embayments, showed relative small positive or negative net nitrogen fluxes. This appears to result from the relatively high mass of particulate nitrogen settling within this system, due to the high phytoplankton production in the nitrogen rich embayment waters.

Higher nitrogen net fluxes from sediments of the Mashpee River versus the other basins likely results in part from differences in basin depth and tidal exchange (cf. Table V-9 for local residence times). There is also an indication that the very reducing (anoxic) nature of the Mashpee River sediments may be increasing the percentage of nitrogen which is released from the sediments versus the amount of nitrogen being lost to denitrification via the pathway of mineralization→ nitrification → denitrification. The coupled nitrification-denitrification step in the pathway is significantly influenced by the availability of oxygen within the surficial sediments for nitrifying bacteria. That the anoxic/sulfidic nature of the Mashpee River sediment is affecting enhancement of nitrogen release is supported by estimates of potential nitrogen loss versus the amount of measured loss. Using this rough approximation, more nitrogen is released from the Mashpee River sediments than from the other sites. Note that this approach yields general patterns and cannot be used to determine accurate nitrogen removal rates

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 Popponesset Bay estuarine system. 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 each system. 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.

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 increase the secondary impacts of nutrients recycled from the sediments. Degradation of coastal waters and development are tied together through inputs of pollutants in runoff and groundwater flows, and to some extent through direct disturbance, i.e. boating, oil and chemical spills, and direct discharges from land and boats. Excess nutrients, especially nitrogen, promote phytoplankton blooms and the growth of epiphytes on eelgrass and attached algae, with adverse consequences including low oxygen, shading of submerged aquatic vegetation, and aesthetic problems.

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 Popponesset Bay, the most important parameters are:

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

The Popponesset Bay estuarine system (Figure V-1) is a tidally dominated embayment open to Nantucket Sound. The system separates the towns of Mashpee and Barnstable along the south coast of Cape Cod, Massachusetts. The system consists principally of subembayments Popponesset Bay, Ockway Bay, Mashpee River, and Shoestring Bay, as 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 2.5 feet, with Nantucket Sound tidal variations providing the hydraulic forcing that drives water movement throughout the system.

The objective of this analysis is to develop a numerical model to simulate accurately the hydrodynamic characteristics of the Popponesset 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

Figure V-1. Map of the Popponesset Bay estuary (from United States Geological Survey topographic map, Cotuit quadrangle).

(Mashpee and Barnstable) to understand how pollutant loadings into the estuary will affect the biochemical environment and its ability to sustain a healthy marine habitat.

Since the water elevation difference between Nantucket Sound and each of the estuarine systems is the primary driving force for tidal exchange, the local tide range naturally limits the volume of water flushed during a tidal cycle. Tidal damping (reduction in tidal amplitude) through the Popponesset Bay system is negligible indicating “well-flushed” systems. Based on the tidal characteristics alone, this might indicate that the Popponesset Bay embayments (e.g. the Mashpee River) are “healthy” relative to embayments with more occluded inlets; however, land development in the watershed serving the estuarine system has created significant nutrient loading, especially in the northern half of the system. Consequently, estuarine water quality is more dependent on nutrient loading than tidal characteristics for the Popponesset Bay system.

In addition to tidal forcing characteristics, the regional geomorphology influences flushing characteristics within the Popponesset Bay system. Offshore shoal migration and alongshore sediment transport patterns along the south shore of Mashpee and Cotuit (e.g., beach sand movement along Popponesset Beach spit) have caused numerous changes to the inlet over the past 50 to 70 years.

This section summarizes the development of hydrodynamic models for the Popponesset Bay estuarine system. For the estuarine system, the calibrated model offers an understanding of water movement through the estuary. Tidal flushing information will be utilized as the basis for a quantitative evaluation of water quality. Nutrient loading data combined with measured environmental parameters within the various sub-embayments become the basis for an advanced water quality model based on total nitrogen concentrations. This type of model provides a tool for evaluating existing estuarine water quality, as well as determining the influence of various methods for improving overall estuarine health.

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

To calibrate the hydrodynamic model, field measurements of water elevations and bathymetry were required. For the Popponesset Bay system tide data was acquired within Nantucket Sound (two gages were installed offshore of the groin field to the east of the inlet) and within major sub-embayments of the estuary. All temperature-depth recorders (TDRs or tide gages) were installed for a 30-day period to measure tidal variations through an entire neap-spring cycle. In this manner, attenuation of the tidal signal as it propagates through the various sub-embayments was evaluated accurately.

V.2 GEOMORPHIC AND ANTHROPOGENIC EFFECTS TO THE ESTUARINE SYSTEM

The southern coast of Cape Cod in the vicinity of Popponesset 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 creates 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 is balanced by the gradual wave-driven migration of the barrier beach separating the estuaries from the ocean. As beaches elongate, the inlet channels to the estuaries often become long, sinuous, and hydraulically inefficient. Periodically, overwash from storm events will erode the barrier beach enough at a point to allow again the formation of a new inlet. It is then possible that the new inlet will stabilize and become the main inlet for the system, while the old inlet eventually fills in. Several examples of this process along the Massachusetts coast include Allen's Pond (Westport), New Inlet/Chatham Harbor/Pleasant Bay (Chatham), and Nauset inlet (Orleans).

As described in Aubrey and Goud (1983), the loss of nearly one-half of the barrier beach between 1954 and the early 1980s led to concerns regarding future barrier spit migration. Figures V-2 and V-3 illustrate changes to the barrier spit over the 30 year period between 1951 and 1981. According to Aubrey and Gaines (1982), the present spit length has been historically the stable configuration. It wasn't until after about 1860 that the spit began to grow past its present location. The USGS map from 1893 (Figure V-4) shows the barrier beach in a condition where the spit is elongated slightly beyond present day conditions, where the flood shoal is emergent (Thatch Island).

Based on tidal hydrodynamics alone, present-day conditions represent a more efficient flow pathway than the elongated channel that existed in the three decades preceding the 1954 hurricane. Similar to most tidal inlets, the natural position of the inlet is a balance between hydrodynamic efficiency and littoral transport along the open coast. As the barrier spit elongated between the early 1900s and the mid-1950s as a result of regional littoral drift, the inlet channel become less efficient, where the tide height within Popponesset Bay decreased and the lag time between high tide in the estuary and Nantucket Sound increased. This increase in tidal attenuation was remedied in 1954, when a hurricane breached the barrier spit, creating an efficient inlet in the vicinity of the present inlet. Once the spit had breached, the remnants of the spit east of the inlet gradually overwashed and rejoined the shoreline (primarily in the vicinity of Rushy Marsh). This inlet spit growth and breaching process has been documented extensively for the southeastern coast of Massachusetts (e.g. Fitzgerald, 1993).

In addition to natural phenomena affecting estuarine hydrodynamics, man-made alterations have impacted tidal exchange in the Popponesset Bay system. Examples of anthropogenic modifications include the 1916 dredging within the main portion of Popponesset Bay, as well as the 1962 dredging associated with the large-scale development at New Seabury. The location of existing and proposed dredge channels within the Popponesset Bay system is shown in Figure V-5. Since the inlet has no jetties, the position of the main inlet migrated naturally for much of the past 100 years. Over the past decade, maintenance dredging likely has stabilized the inlet position at the present location.

Manmade coastal structures along the Mashpee shoreline consist primarily of seawalls and/or revetments along the updrift shoreline (west of Popponesset Beach). These structures likely have reduced the natural littoral sediment supply to the barrier beach system. In effect, this reduction in sediment supply may decrease spit growth and the associated needs for maintenance dredging. Long-term plans for the New Seabury shoreline facing Nantucket Sound include a large-scale beach nourishment project aimed at offsetting the impact of coastal structures on the local littoral system. If designed properly, this proposed beach nourishment program should have a negligible impact on inlet stability.

Although man has modified much of the Mashpee coastline, most of the large-scale changes to the estuarine systems have been caused by nature. For example, the 1954 breach of Popponesset Beach created a much more efficient inlet channel. Most of the manmade modifications to Popponesset Bay or the adjacent coastline have caused small changes to overall estuarine health. While past dredging efforts may have had a slight positive impact to tidal flushing, this influence is minor relative to natural large-scale changes.

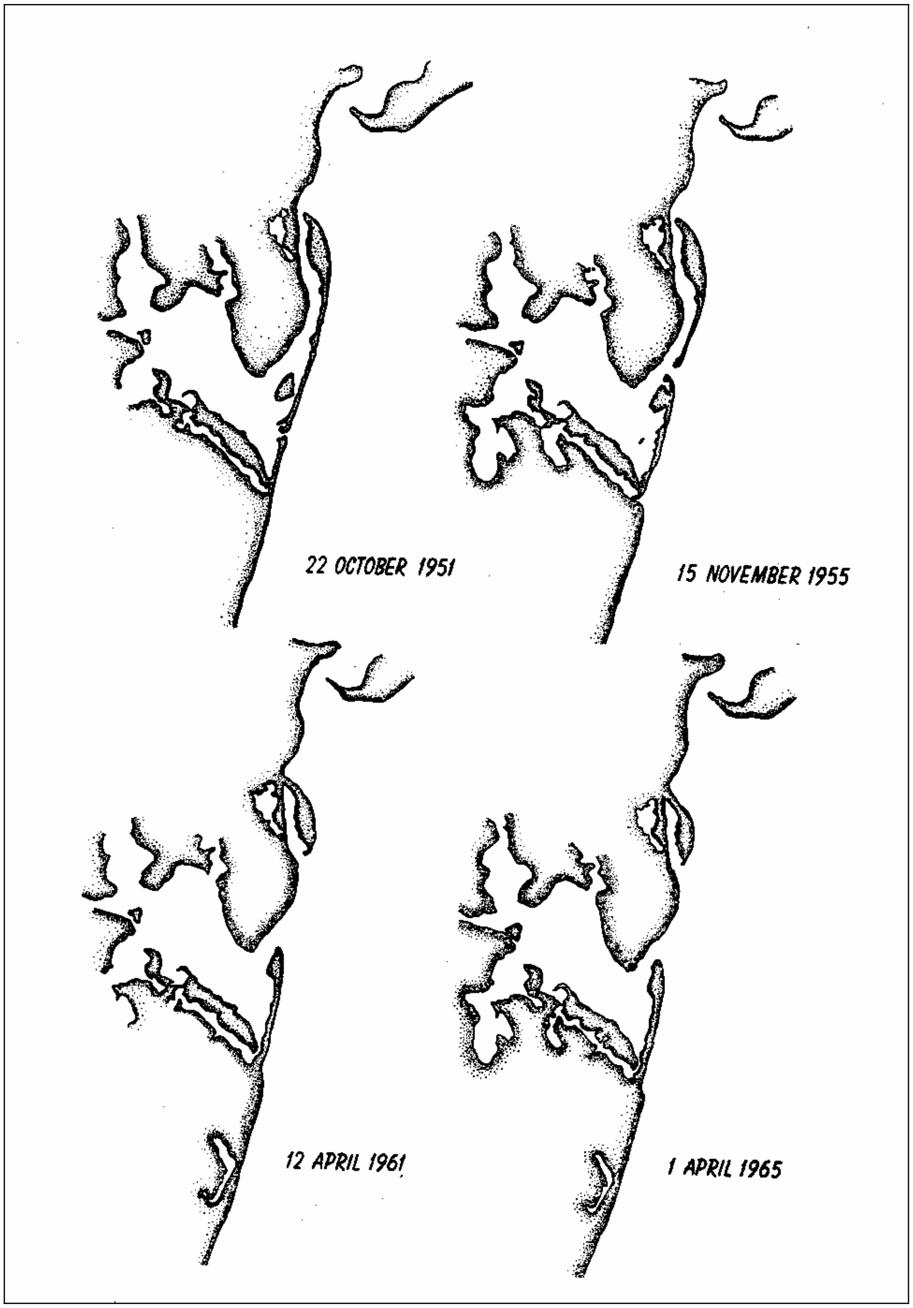


Figure V-2. Outlines of vertical aerial photographs illustrating stages of shoreline evolution in the Popponesset Spit region between 1951 and 1965 (from Aubrey and Goud, 1983).

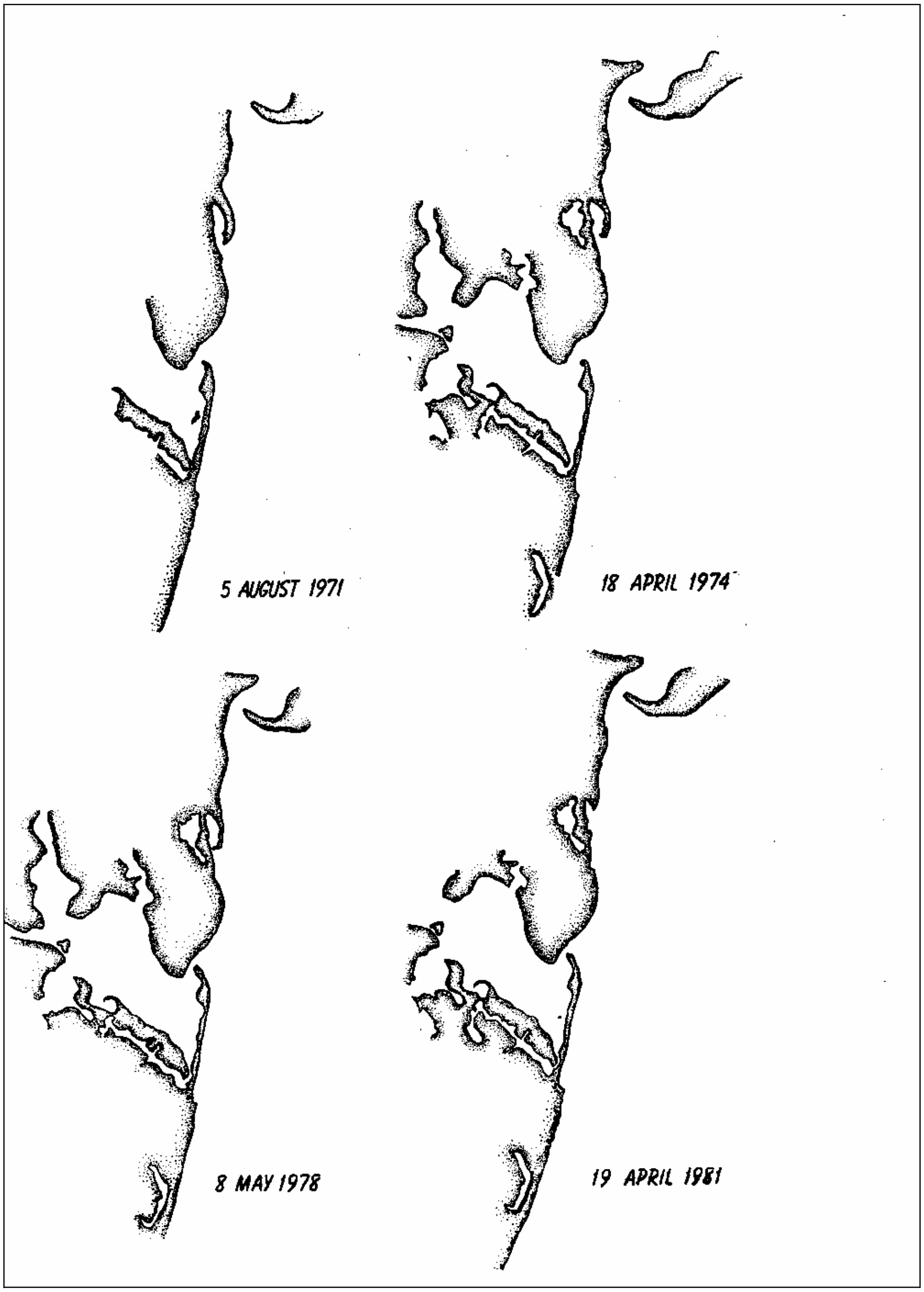


Figure V-3. Outlines of vertical aerial photographs illustrating stages of shoreline evolution in the Popponesset Spit region between 1971 and 1981 (from Aubrey and Goud, 1983).

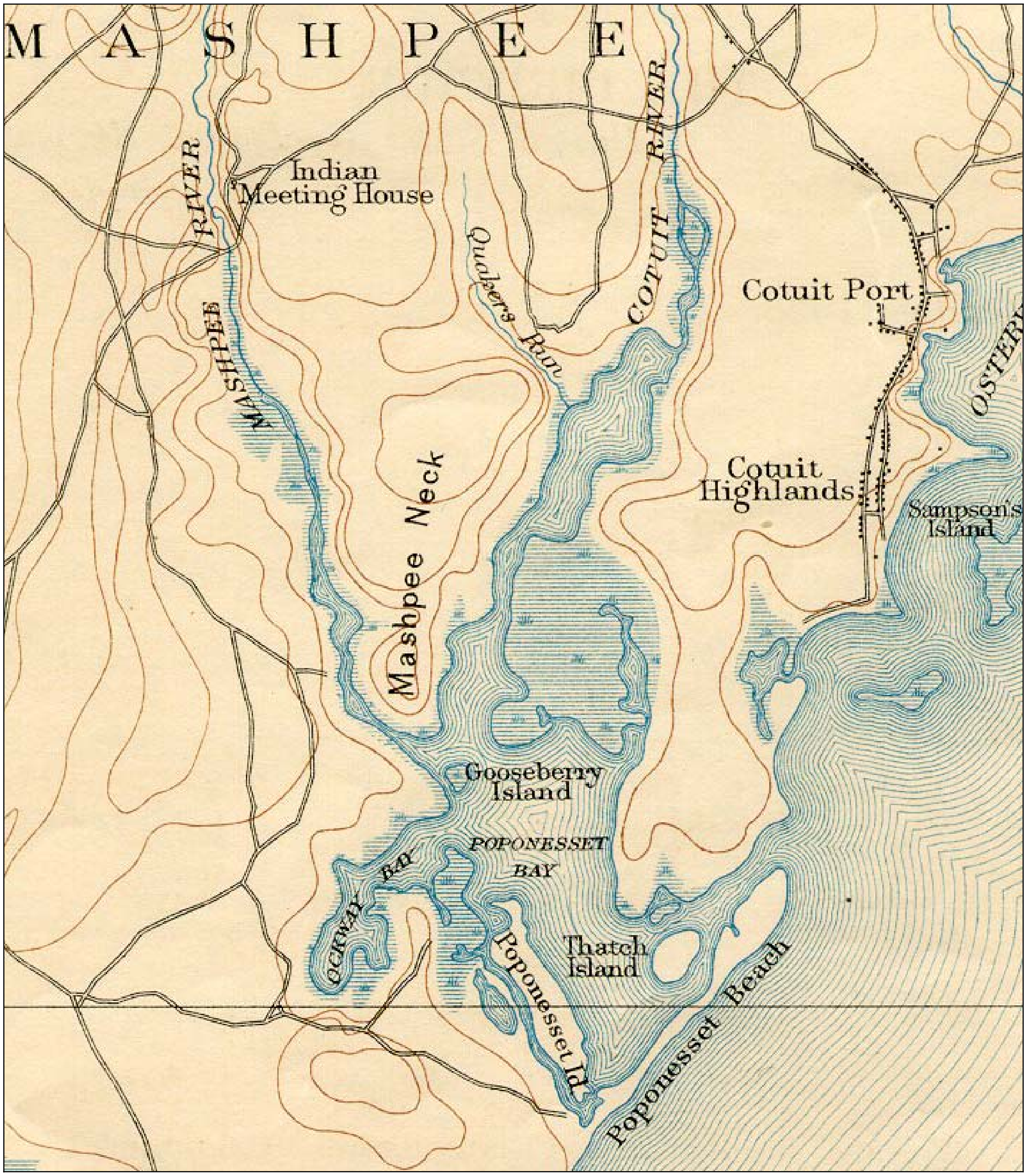


Figure V-4. Portion of the 1893 USGS topographic map (Cotuit Quadrangle) showing the position of the inlet at a similar location as the present inlet.

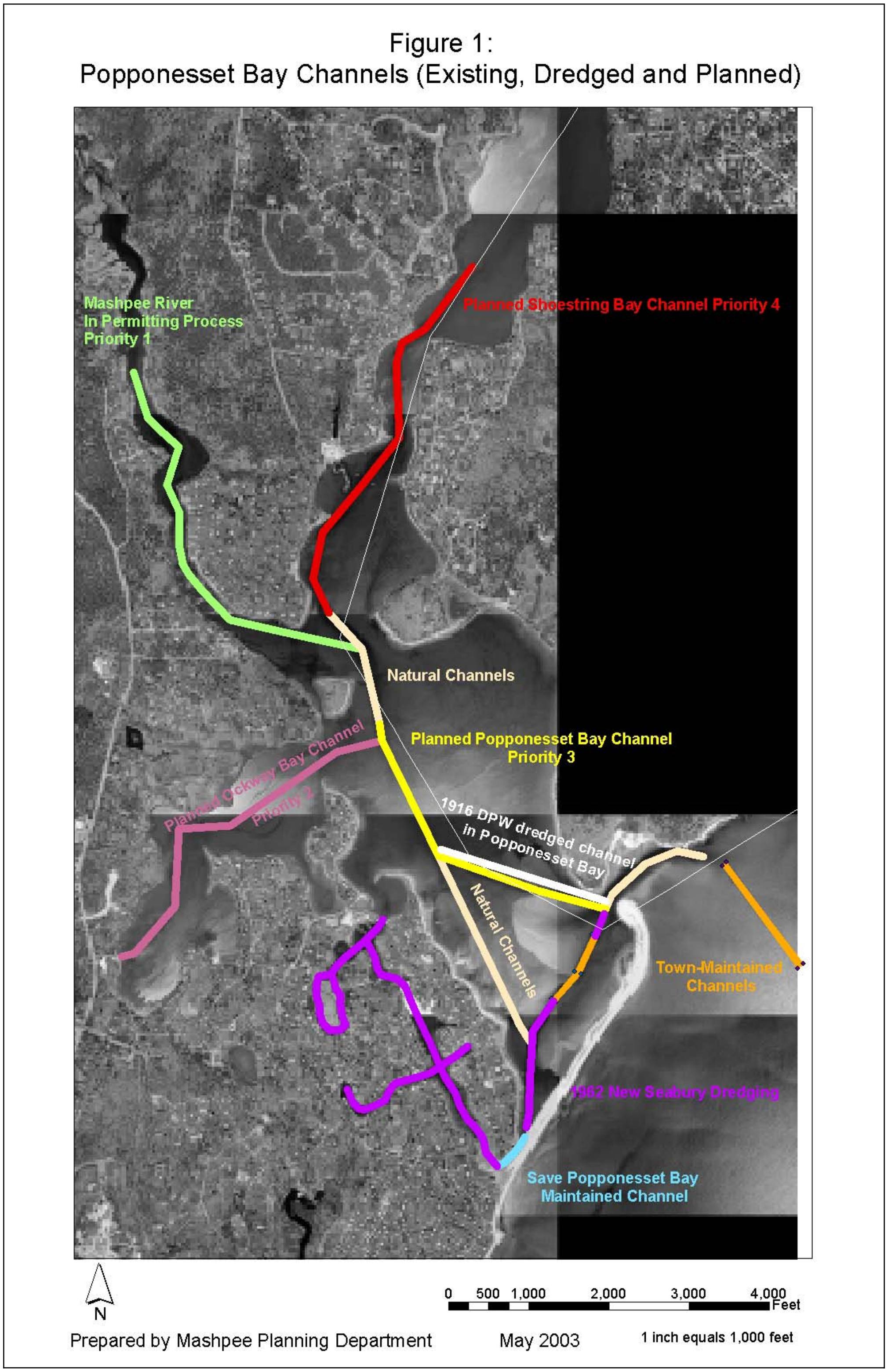


Figure V-5. Past and proposed future dredged channels within the Popponesset Bay estuarine system. Much of the previous improvement dredging within the system was performed during the 1960s development of New Seabury (shown in purple).

V.3 FIELD DATA COLLECTION AND ANALYSIS

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

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

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

Boundary conditions for the numerical model consist of variations of water surface elevation in Nantucket Sound. These variations result principally from tides, and provide the dominant hydraulic forcing for the system. A pressure sensor was installed near the mouth of Popponesset Bay to measure the Nantucket Sound tides. This tidal function was used as the principal forcing function, or boundary condition, to the model.

Additional pressure sensors were installed at selected interior locations to measure variations of water surface elevation within subembayments. These measurements were used to calibrate and verify the model results, and to assure that the important physics were properly simulated.

V.3.1 Data Acquisition

V.3.1.1 Water Elevation

Variations in water surface elevation were measured at five locations around Popponesset Bay Estuary (Figure V-6):

- Offshore of the inlet to Popponesset Bay (location #1)
- Popponesset Island (location #2)
- Ockway Bay (location #3)
- Mashpee River (location #4)
- Shoestring Bay (location #5)

These variations were measured using small, self-contained pressure/ temperature sensors (typically referred to as tide gauges). These sensors use electronic recording circuits to sense external temperature and pressure, and write the measurements to internal memory recorders. The units are installed rigidly to pier pilings or other fixed objects, and remain in place throughout a monthly tidal cycle (more than 29 days).

The units were installed in early October, 1999, and recovered in early November, 1999. Data presented in this report spans October 4 to November 3, 1999. All recorders captured 100% of the data. Tide gauges in Shoestring Bay and Popponesset Island were surveyed into

Figure V-6. Map of the study region identifying locations of the tide gauges used to measure water level variations throughout the system. Five (5) gauges were deployed for one month between October 4 and November 5, 1999. Each black square represents the approximate locations of the tide gauges (1) Offshore of the inlet to the northeast of Thatch Island, (2) on the eastern shore of Popponesset Island, (3) within Anns Cove (Ockway Bay), (4) in the Mashpee River, and (5) on the western shore of Shoestring Bay.

local vertical datum using standard engineering rod-and-level techniques. Local benchmarks were obtained from FEMA Flood Insurance maps and National Geodetic Survey data sheets. Surveying at one gauge to a known datum allows estimation of the elevation of the remaining gauges.

Two types of sensors were utilized of this study: Brancker TG-205 recorders (at locations 1, 2, 4, and 5), and Coastal Leasing MicroTide recorders (location 3). In addition, a Coastal Leasing MicroTide sensor was used to record atmospheric pressure variations. The Brancker recorders utilize a 0-30 psia Druck strain gauge pressure sensor; resolution is 0.03% full scale and its accuracy is 0.1% full scale, or about 0.03 psi. This translates to an accuracy of about 0.8 inches of seawater. The MicroTide units possess the same accuracy standards as the Brancker units.

Two problems with the measurements were identified upon recovery; these problems did not affect the accuracy of the study. The first problem resulted from galvanic corrosion of the pressure port plumbing on the Brancker gauge installed in the Mashpee River. Corrosion caused the port to become slightly clogged during the last four-to-five days of the study, inhibiting the response of the gauge to changes in water elevation. Data returned from this gauge for the first four weeks of the study showed no adverse effects from this problem. The second problem was installation of the Ockway Bay gauge. While the gauge was mounted to a pier along the shoreline, the pressure port was located approximately 1 foot above the seabed. The gauge became exposed during extreme low-water events, hence the tidal curve becomes truncated during extreme low tides. This will affect the tidal constituent analysis, reporting harmonic constituents that may be decreased by as much as 3%-5%.

Upon recovery, the raw data were transferred from the instrument recorders to PC hard disks for processing and analysis. The raw data were first converted to engineering units (pounds per square inch, or psi) using each sensors factory-supplied calibration coefficients. Once in pressure units, the data were corrected for variations due to atmospheric pressure. These atmospheric pressure observations were collected near Ockway Bay using a Coastal Leasing MicroTide pressure sensor. After correction for atmospheric pressure variations, the data were then converted to head-of-water units, assuming a constant water density value of 1025 kg/m^3. These water elevation variation values for Shoestring Bay and Popponesset Island were then rectified to the NGVD 1929 vertical datum using survey measurements. The measurements obtained at each location are presented as Figure V-7.

Tide records of greater than 29 days ensure a complete evaluation of spring and neap tidal conditions within the estuarine system. Although a one-month record does not necessarily include extreme high or low tides, it does provide an accurate basis for typical tidal conditions governed by both lunar and solar gravitational attraction. In Nantucket Sound, additional attenuation of the tidal signal is caused by the geomorphology of the nearshore region. For numerical modeling of hydrodynamics, the typical tide conditions associated with a one-month record are appropriate for driving tidal flows within the estuarine system.

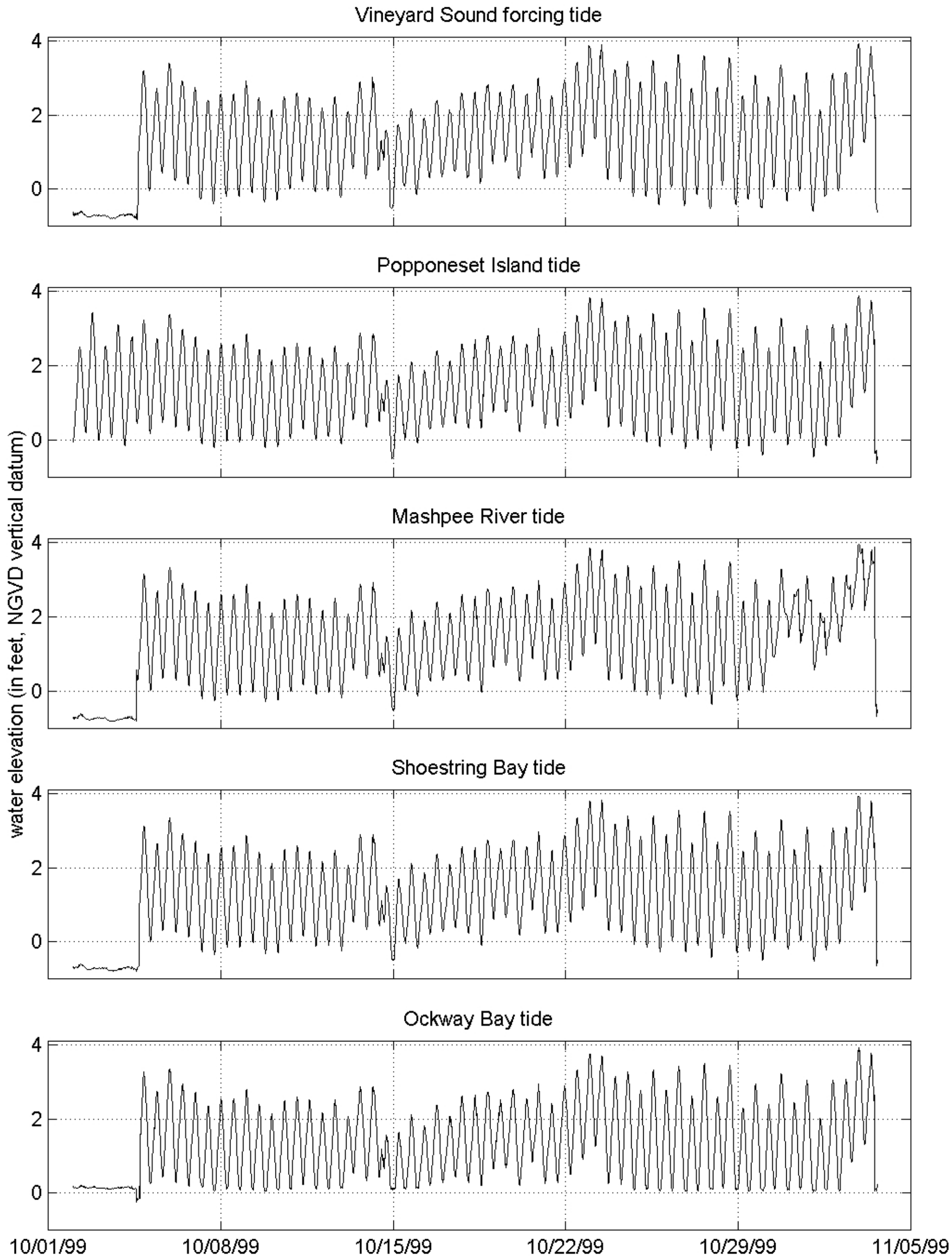


Figure V-7. Water elevation variations as measured at the five locations within the Popponesset Bay estuary. Atmospheric effects have been removed from the records. The gauge at Popponesset Island was deployed approximately three days prior to the other gauges.

V.3.1.2 Bathymetry

Bathymetry, or depth, of each subembayment was measured during a field survey October 28, 1999. The survey was completed using a small vessel equipped with a precision fathometer interfaced to a differential GPS receiver. The fathometer had a depth resolution of approximately 0.1 foot, and the differential GPS provides position measurements accurate to approximately 1-3 feet. Digital data output from both the echo sounder and GPS were logged to a laptop computer, which integrated the data to produce multiple data sets consisting of water depth as a function of geographic position (latitude/longitude). The surveys were performed within each sub-embayment to develop plan view contour maps of system geometry.

The data files of water depth as a function of geographic position were merged with water surface elevation measurements to correct the measured depths to the NGVD 1929 vertical datum. Once corrected, the data were then merged into larger 'xyz' files containing x-y horizontal position (in Massachusetts State Plan 1983 coordinates) and vertical elevation of the bottom (z) relative to NGVD29. These xyz files were then interpolated into the finite element mesh used for the hydrodynamic simulations. The interpolated bathymetric data is presented in Figure V-8. The bathymetry survey and tidal measurements were performed after the 1999 dredging of the Popponesset Bay inlet.

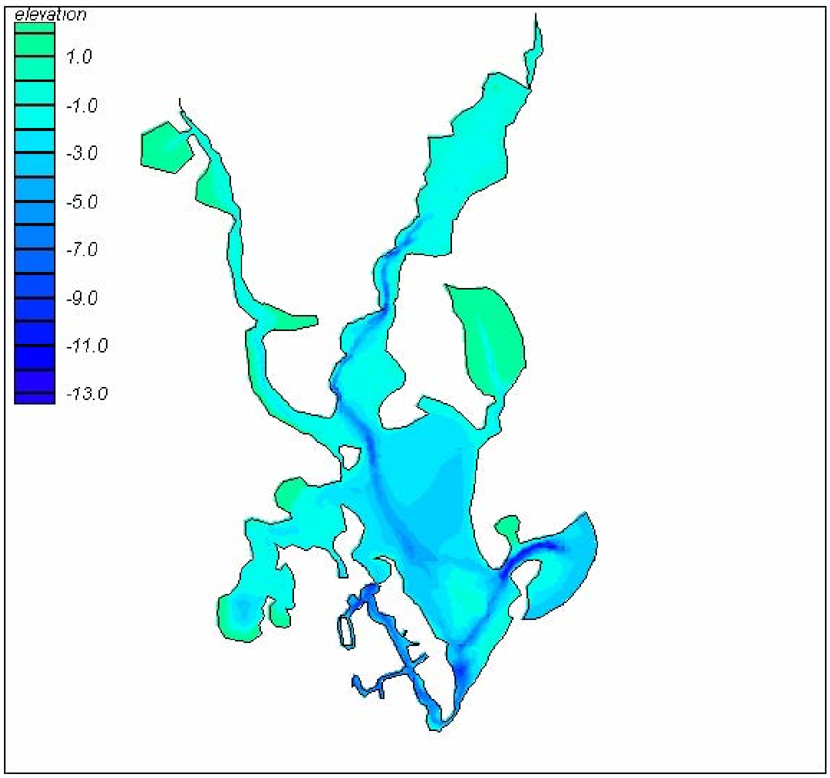


Figure V-8. Bathymetric data interpolated to the finite element mesh of hydrodynamic model.

V.3.2 Discussion of Results

V.3.2.1 Bathymetry Analysis

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

The Popponesset Bay system is open to Nantucket Sound. The inlet is affected significantly by longshore sand transport (the predominant transport is west to east), which tends to accrete Popponesset Spit towards Cotuit Highlands. This longshore sediment transport can impede hydrodynamic exchange at the mouth. Groins constructed along the Cotuit side of the shoreline armor the downdrift side of the inlet; the Popponesset Beach side (updrift side) is unarmored. The offshore region near the entrance to the system is quite shallow, with mild slopes extending offshore. The entrance channel is narrow and relatively deep, approximately 10 to 15 feet below NGVD and features strong tidal currents. Inside the system, the Popponesset Bay embayment possesses relatively deep water along the western edge, and shallower depths on the eastern edge. Popponesset Creek, on backside of Popponesset Island has been modified by dredging; in some areas the creek depth is more than 8 feet below NGVD. The northeast portion of Popponesset Bay embayment splits into Ockway Bay, a shallow embayment of soft sediments and sluggish flow, and the entrances to Shoestring Bay (Santuit River) and Mashpee River. The Mashpee River is long and narrow, with relatively shallow depths in the channel of order 3 to 5 feet below NGVD. A deep channel (6 to 10 feet below NGVD) along the western edge of Shoestring Bay leads to a northern basin where the depths are shallow and sediments relatively soft. Marsh areas exist within the system, most significantly the Pinquickset Marsh on the northeast corner of the embayment, but also in areas of Ockway Bay and upper (northern) portions of the Mashpee River.

V.3.2.2 Tidal Harmonic Analysis

Harmonic analyses were performed on the time series from each gage location in an effort to separate the various tidal components and identify the important properties. In addition, it allows an understanding of the relative contribution that various physical processes (i.e. tides, winds, etc.) have on water level variations within the estuary. Harmonic analysis is a mathematical procedure that fits sinusoidal functions of known frequency to the measured signal. The amplitudes and phase of 23 tidal constituents result from this procedure.

Table V-1 presents the amplitudes of the eight largest tidal constituents. The M_2, or the familiar twice-a-day lunar semi-diurnal, tide is the strongest contributor to the signal with an amplitude of 1.2 feet in Nantucket Sound. The range of the M_2 tide is twice the amplitude, or about 2.4 feet. The diurnal tides, K_1 (solar) and O_1 (lunar), possess amplitudes of approximately 0.15-0.20 feet. The N_2 tide, a lunar constituent with a semi-diurnal period, rivals the diurnal constituents with an amplitude of 0.30 feet. The M_4 tide, a higher frequency harmonic of the M_2 lunar tide (twice the frequency of the M_2), results from frictional dissipation of the M_2 tide in shallow water. The M_4 is significant in Vineyard and Nantucket Sounds, and is responsible for the unusual 'double high' tide signature prominent along the Falmouth shoreline to the west. This M_4 constituent tends to decrease eastward in Nantucket Sound, but at Popponesset Bay is still 0.18 feet, approximately one-fifth the amplitude of the M_2.

Table V-1. Tidal Constituents for Popponesset Bay System 1999.

	Amplitude (feet)							
Constituent	M_2	M_4	M_6	S_2	N_2	K_1	O_1	Msf
Period (hours)	12.42	6.21	4.14	12.00	12.66	23.93	25.82	354.61
Nantucket Sound (Inlet)	1.20	0.18	0.06	0.10	0.30	0.15	0.20	0.25
Popponesset Island	1.14	0.08	0.06	0.08	0.28	0.14	0.19	0.24
Ockway Bay	1.10	0.07	0.05	0.08	0.27	0.13	0.18	0.24
Shoestring Bay	1.15	0.06	0.07	0.07	0.29	0.13	0.19	0.24
Mashpee River	1.05	0.04	0.07	0.15	0.22	0.11	0.18	0.27

The observed astronomical tide is therefore the sum of several individual tidal constituents, with a particular amplitude and frequency. For demonstration purposes a graphical example of how these constituents add together is shown in Figure V-9.

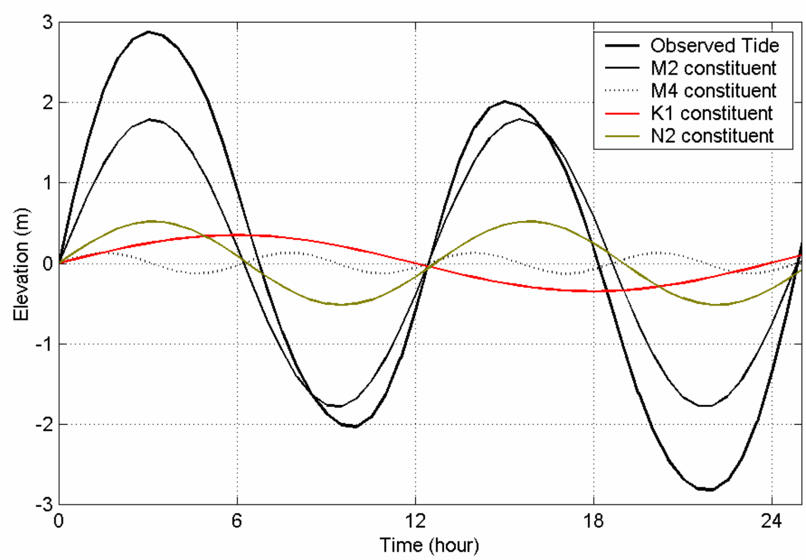


Figure V-9. Example of observed astronomical tide as the sum of its primary constituents.

Table V-1 also shows how the constituents vary as the tide propagates into the estuaries. Most estuaries exhibit tidal damping, that is, a reduction of the tide range relative to the offshore forcing tide. Note the reduction in the M_2 amplitude between Nantucket Sound and Popponesset Island (M_2 amplitude of 1.2 feet in Nantucket Sound versus 1.14 feet at Popponesset Island, just inside the inlet, a reduction of 5%). Tidal amplitude decreases are also shown in Ockway Bay and Mashpee River. In general, the amplitude reduction of the M_2 constituent is relatively small through the system, with the largest reduction across the inlet.

To better quantify the changes to the tide from the inlet to inside the system, the standard tide datums were computed from the 29-day records. These datums are presented in Table V-2. For most NOAA tide stations, these datums are computed using 19 years of tide data, the definition of a tidal epoch. For this study, a significantly shorter time span of data was available; however, these datums still provide a useful comparison of tidal dynamics within the system. The Mean Higher High Water (MHHW) and Mean Lower Low Water (MLLW) levels represent the mean of the daily highest and lowest water levels. The Mean High Water (MHW) and Mean Low Water (MLW) levels represent the mean of all the high and low tides of a record, respectively. The Mean Tide Level (MTL) is simply the mean of MHW and MLW.

The tides in Nantucket Sound are semi-diurnal, meaning that there are typically two tide cycles in a day. There is usually a small variation in the level of the two daily tides. This variation can be seen in the differences between the MHHW and MHW, as well as the MLLW and MLW levels. With the relatively small tide range of the Popponessett Bay, the influence of atmospheric forcing also can be seen in the tide records. For example, the maximum tide reading in the Mashpee River is higher than the remainder of the system, where this anomalous reading likely was due to local wind set-up.

Table V-2. Tide datums computed from records collected in Popponesset Bay from October 4 to November 3, 1999. Datum elevations are given relative to NGVD.					
Tide Datum	Offshore (feet)	Popponesset Island (feet)	Shoestring Bay (feet)	Mashpee River (feet)	Ockway Bay (feet)
Maximum Tide	3.94	3.88	3.93	3.96	3.91
MHHW	3.02	2.98	2.96	2.97	2.95
MHW	2.77	2.74	2.70	2.73	2.69
MTL	1.51	1.51	1.45	1.56	1.46
MLW	0.14	0.23	0.14	0.35	0.29
MLLW	-0.59	0.03	-0.07	0.12	0.16
Minimum Tide	-0.83	-0.51	-0.64	-0.50	0.03

Table V-3 presents the phase delay of the M_2 tide at all tide gauge locations, in other words, the travel time required for the tidal wave to propagate throughout the system. The data suggest that it takes approximately 5 minutes for the tide wave to travel from Popponesset Island to Ockway Bay, about 7-8 minutes to go from Popponesset Island to Shoestring, and more than 11 minutes to propagate from Popponesset Island to the Mashpee River. The important result from Table V-3 is that it takes about a half-hour for the tide to move from Nantucket Sound through the inlet. This suggests that the inlet may be responsible for significant modification of the tide wave. Modification of the wave, or tidal distortion, can alter the hydrodynamic characteristics of a system and resulting efficiency with which the system flushes pollutants. Long delays signify reduced hydrodynamic exchange, while small delays indicate an efficient, well-flushed system. The results from Table V-3 suggest that hydrodynamic circulation is quite efficient within the system, that is, between Popponesset Bay and the subembayments; however, the relatively longer delay between Nantucket Sound and Popponesset Island suggests exchange through the inlet is the primary source of inefficiency.

Table V-3. M_2 Phase Delays from Nantucket Sound through the Popponesset Bay System	
Location	Delay (minutes)
Popponesset Bay	28.75
Ockway Bay	33.57
Shoestring Bay	36.48
Mashpee River	40.37

Table V-4 shows the relative variance of tidal versus non-tidal (or residual) processes at different locations in the systems. Variance is directly proportional to energy. Non-tidal processes include wind responses, for example wind set-up and set-down, or sub-tidal oscillations originating in Nantucket Sound. In addition, the water levels within the estuary can be affected by freshwater input, either through groundwater or surface runoff during rain events. The table shows the percentage of non-tidal energy at various points within the estuary, and that the relative percentage increases with increasing distance into the system. At Popponesset Island, only about 14% of the signal variance can be attributed to non-tidal events. In the

Mashpee River, nearly 25% of the signal was due to non-tidal processes. The larger influence of non-tidal processes within the Mashpee River likely is due to local effects of wind which can produce significant non-tidal variations of the sea surface, hence increasing the energy of non-tidal processes. Fresh water inflow also could produce variations, especially after substantial precipitation, although these effects are difficult to quantify through a constituent analysis of short-term tidal data. The results from Table V-4 indicate that hydrodynamic circulation in each of the subembayments is dependent primarily upon tidal processes, yet wind and other non-tidal effects are of significant concern as well. For the hydrodynamic modeling effort described below, the actual tide signal in Nantucket Sound was used to force the model; therefore, the effects of non-tidal energy are included in the modeling analysis.

Table V-4. Percentages of Tidal versus Non-Tidal Variance Popponesset Bay System (units of ft^2)

Location	Total Variance	Tidal Variance	Residual Variance	% Residual
Nantucket Sound	1.024	0.889	0.135	13.2%
Popponesset Bay	0.903	0.778	0.124	13.8%
Ockway Bay	0.856	0.732	0.124	14.4%
Shoestring Bay	0.933	0.792	0.141	15.1%
Mashpee River	0.887	0.668	0.219	24.7%

Figure V-10 shows the results of the tidal versus the non-tidal separation procedure for the tide gauge at Popponesset Island. While the measurements show that tides dominate the variations of water level within the estuary, it was clear that non-tidal processes also affected changes in the water surface. While the tidal range was nearly four feet (maximum), the measurements suggest that non-tidal processes, probably winds, can produce water surface variations of +/- 1 foot, a significant fraction of the tide range. These processes can have important hydrodynamic repercussions, specifically a major influence in horizontal mixing within the estuary. In a sense, major wind storms provide benefits for estuaries with limited tide ranges, such as those along the southwest Cape shore (i.e. Falmouth finger ponds, Waquoit and Popponesset Bays). Strong winds have energy sufficient to increase substantially horizontal circulation and improve flushing in areas where tidal-induced circulations are poor.

Table V-5 presents additional analytical information regarding the hydrodynamic behavior of the estuary. The amplitude ratio and relative phase values can indicate the degree of tidal distortion, or modification, of the tide entering the estuary, and provide insight into the physical processes responsible for the observed signals (Freidrichs and Aubrey, 1985). Two results of Table V-5 deserve attention: the first is the reduction of the M_4/M_2 ratio with distance into the estuary, and second are relative phase values in the vicinity of 270°. Both of these values suggest Popponesset Bay estuary can be described as an ‘ebb-dominant’ estuary, in contrast to many other estuaries on Cape Cod that can be described as ‘flood-dominant’ estuaries.

Table V-5. Amplitude Ratio and Relative Phase

Location	M_4/M_2 ratio	$2M_2$-M_4 phase
Nantucket Sound	0.148	262°
Popponesset Bay	0.072	270°
Ockway Bay	0.065	294°
Shoestring Bay	0.053	284°
Mashpee River	0.042	291°

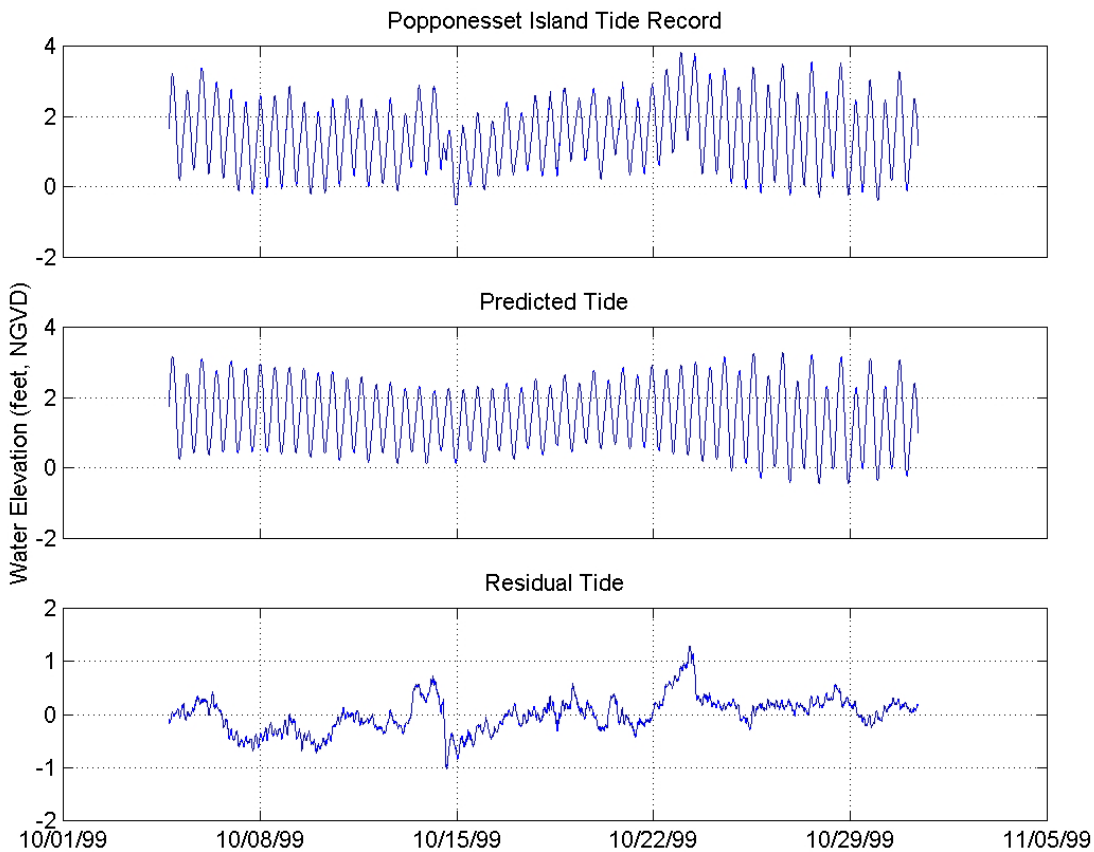


Figure V-10. Results of the harmonic analysis and the separation of the tidal from the non-tidal, or residual, signal measured at Popponesset Island.

A detailed scientific explanation of these parameters is presented in Freidrichs and Aubrey (1985). In general, the results of this analysis indicate that the Popponesset Bay system is ebb-dominant, where the estuary tends to have a flood tide that is longer in duration than the ebb tide. Due to this asymmetry in the tide phases, and the need to conserve volume within the estuary, ebb currents will tend to be stronger than flood currents because the same (or similar) volume must pass through the inlet over a shorter time period. The 'ebb-dominant' estuary, with its stronger ebb currents, will tend to have a net transport of sediments out of the system and into Nantucket Sound. Flood-dominant systems tend to accumulate sediments.

V.4 HYDRODYNAMIC MODELING

The focus of this study was the development of a numerical model capable of accurately simulating hydrodynamic circulation within this estuary. Once calibrated, the model was used to calculate water volumes for selected subembayments (e.g., Ockway Bay, Mashpee River, and Shoestring Bay) as well as determine the volumes of water exchanged during each tidal cycle. These parameters are used to calculate system residence times, or flushing rates. Use of a calibrated numerical model is the most accurate and reliable method to determine system flushing rates.

V.4.1 Model Theory

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

In its original form, RMA-2 was developed by William Norton and Ian King under contract with the U.S. Army Corps of Engineers (Norton et al., 1973). Further development included the introduction of one-dimensional elements, state-of-the-art pre- and post-processing data programs, and the use of elements with curved borders. Recently, the graphic pre- and post-processing routines were updated by Brigham Young University through a package called the Surfacewater Modeling System or SMS (BYU, 1998). SMS is a front- and back-end software package that allows the user to easily modify model parameters (such as geometry, element coefficients, and boundary conditions), as well as view the model results and download specific data types. While the RMA model is essentially used without cost or constraint, the SMS software package requires site licensing for use.

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

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

V.4.2 Model Setup

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

- Grid generation
- Boundary condition specification
- Calibration

The extent of each finite element grid was generated using digital aerial photographs from the MassGIS online orthophoto database. A time-varying water surface elevation boundary condition (measured tide) was specified at the entrance of the system based on the tide gauge data collected in Nantucket Sound. Once the grid and boundary conditions were set, the model was calibrated to ensure accurate predictions of tidal flushing. Various friction and eddy viscosity coefficients were adjusted, through several (15+) model calibration simulations for each system, to obtain agreement between measured and modeled tides. The calibrated model provides the requisite information for future detailed water quality modeling.

V.4.2.1 Grid Generation

The grid generation process for the model was simplified by the use of the SMS package. The digital shoreline and bathymetry data were imported to SMS, and a finite element grid was generated to represent the estuary with 3584 elements and 10227 nodes (Figure V-11). All regions in the system were represented by two-dimensional (depth-averaged) elements. The finite element grid for the system provided the detail necessary to evaluate accurately the variation in hydrodynamic properties within the estuary. Fine resolution was required to simulate the numerous channel constrictions that significantly impact the estuarine hydrodynamics. The SMS grid generation program was used to develop quadrilateral and triangular two-dimensional elements throughout the estuary. Reference water depths at each node of the model were interpreted from bathymetry data obtained in the field surveys. The model computed water elevation and velocity at each node in the model domain.

Grid resolution was governed by two factors: 1) expected flow patterns, and 2) the bathymetric variability in each region. Relatively fine grid resolution was employed where complex flow patterns were expected. For example, smaller node spacing in each creek and/or channel was designed to provide a more detailed analysis in these regions of rapidly varying flow. Also, elements through channels were designed to account for the rapid changes in bathymetry caused by shoaling and scour processes. Widely spaced nodes were defined for much of the marsh and inter-tidal flats, where flow patterns did not change dramatically. Appropriate implementation of wider node spacing and larger elements reduced computer run time with no sacrifice of accuracy.

V.4.2.2 Boundary Condition Specification

Three types of boundary conditions were employed for the RMA-2 model: 1) "slip" boundaries, 2) freshwater inflow, and 3) tidal elevation boundaries. All of the elements with land borders have "slip" boundary conditions, where the direction of flow was constrained shore-parallel. The model generated all internal boundary conditions from the governing conservation equations. Freshwater recharge was specified at the upper end of the Mashpee River and Shoestring Bay (Santuit River), although these values were quite small relative to the tidal prism.

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

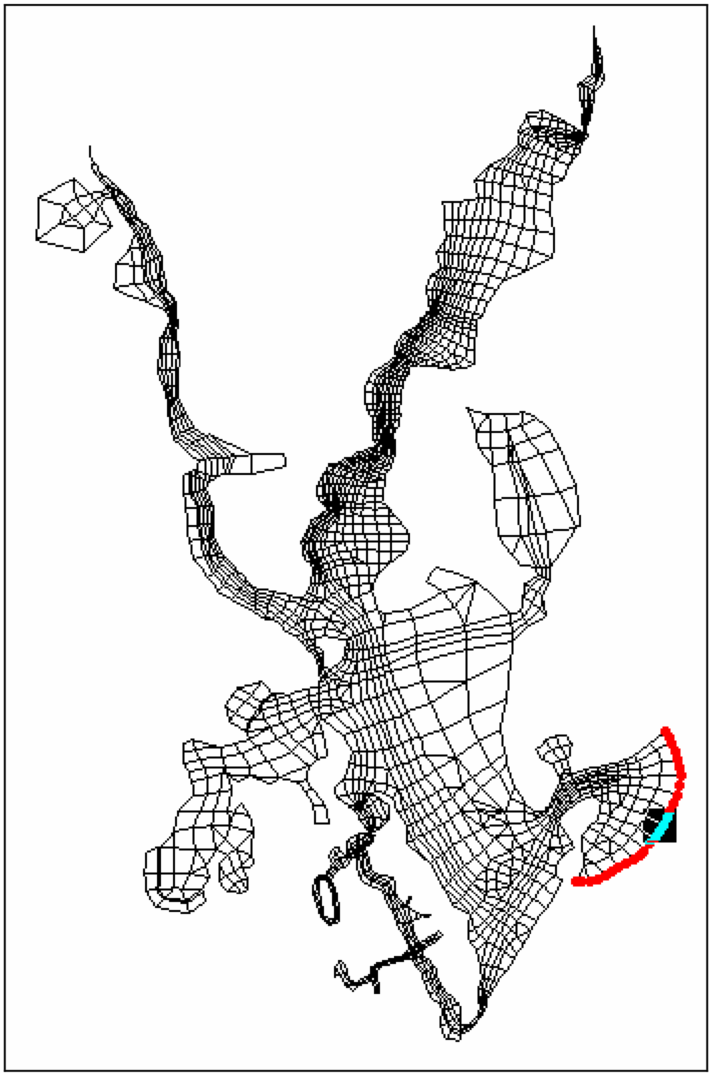


Figure V-11. The model finite element mesh developed for Popponesset Bay. The model seaward boundary (red bold line) was specified with a forcing function consisting of water elevation measurements obtained in Nantucket Sound.

V.4.2.3 Calibration

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

Calibration of the flushing model required a close match between the modeled and measured tides in each of the sub-embayments where tides were measured (e.g. Mashpee River, Shoestring Bay, etc). Initially, a seven-day period was calibrated to obtain visual agreement between modeled and measured tides. To refine the calibration procedure, water elevations were outputted from the simulation at the same locations in the estuary where tide gauges were installed. The two data curves, model and measured, were overlaid on a graph for visual comparison. In addition, the data were processed to calculate harmonic constituents (of both data sets) over the seven-day period. The amplitude and phase of four constituents (M_2, M_4, M_6, and K_1) were compared and the corresponding errors for each were calculated. In addition, the standard error between the two curves was calculated. The intent of the calibration procedure is to minimize the standard error between the curves, as well as to minimize the error in amplitude and phase of the individual constituents. In general, minimization of the M_2 amplitude and phase becomes the highest priority, since this is the dominant constituent. Emphasis is also placed on the M_4 constituent, as this constituent has the greatest impact on the degree of tidal distortion within the system, and provides the unique shape of the modified tide wave at various points in the system.

The calibration was performed for a seven-day period, beginning 1100 hours EST October 23, 1999 and ending October 30, 1999. This representative time period was selected because it included tidal conditions where the wind-induced portion of the signals (i.e. the residual) was minimal, hence more typical of purely tidal circulation within the estuary. The selected time period also spanned the transition from neap (bi-monthly minimum) to spring (bi-monthly maximum) tide ranges, which is representative of average tidal conditions in the embayment system. Throughout the selected seven-day period, the tide ranged approximately 3.5 feet from low to high tides. The ability to model a range of flow conditions is a primary advantage of a numerical tidal flushing model. Modeled tides were evaluated for time (phase) lag and height damping of dominant tidal constituents. The calibrated model was used to analyze existing detailed flow patterns and compute residence times.

V.4.2.3.1 Friction Coefficients

Friction inhibits flow along the bottom of estuary channels or other flow regions where water depths can become shallow and velocities relatively high. Friction is a measure of the channel roughness, and can cause both significant amplitude attenuation and phase delay of the tidal signal. Friction is approximated in RMA-2 as a Manning coefficient. First, the Manning's coefficients were matched to bottom type. Manning's friction coefficient values of 0.025 were specified for all elements. These values correspond to typical Manning's coefficients determined experimentally in smooth earth-lined channels with no weeds (low friction) to winding channels with pools and shoals with higher friction (Henderson, 1966). On the marsh plains, damping of flow velocities typically is controlled more by "form drag" associated with marsh plants than the bottom friction described above. However, simulation of this "form drag" is performed using Manning's coefficients as well, with values ranging from 2-to-10 times friction coefficients used in channels. Final calibrated friction coefficients for the marsh were selected as 0.05. Small changes in these values did not change the accuracy of the calibration.

Variation of the friction parameters during the calibration effort showed that the system was not sensitive to small changes in friction values. A greater change to the estuary response was observed in the model when turbulent exchange values were increased within the inlet channel. The strong flow speeds within the entrance channel will be sensitive to changes in turbulent exchange. As turbulent exchange coefficients increase, tidal energy is removed from the system. Increases in turbulent exchange values through the inlet were found to increase the phase delay of the tide (i.e. high tide arrived later in the Bay) and to decrease the tidal amplitude. Final calibrated friction coefficients are summarized in Table V-6.

Table V-6. Manning's Roughness coefficients used in simulations of modeled embayments.

System	Embayment	Bottom Friction
Popponesset Bay	Nantucket Sound	0.025
	Inlet	0.025
	Meadow Point	0.050
	Popponesset Bay	0.025
	Popponesset Creek	0.025
	Pinquicket Cove	0.025
	Pinquicket Marsh	0.050
	Shoestring Bay	0.025
	Mashpee River	0.025
	Mashpee River Marsh	0.050
	Ockway Bay	0.025
	Ockway Bay Marsh	0.050

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 swift, such as inlets and bridge constrictions. According to King (1990), these values are proportional to element dimensions (numerical effects) and flow velocities (physics). The model was mildly sensitive to turbulent exchange coefficients, specifically in the entrance channel of strong turbulent flow. In other regions where the flow was generally weak, for example broad regions of eastern Popponesset Bay, the model was insensitive to changes in turbulent exchange coefficients. Typically, model turbulence coefficients were set between 50 and 100 lb-sec/ft^2. Higher values (up to 200 lb-sec/ft^2) were used on the marsh plain, to ensure solution stability.

The calibration procedure proved that, in addition to changes in friction and turbulence coefficients, the model was also quite sensitive to changes in system geometry. This fact is not unexpected, and is the reason why accurate bathymetry and topography data are required for these models. While the bathymetry data set obtained for this study was extensive: spatial coverage and vertical resolution of upland marshes and creek areas were not well documented. For example, it was found that the changes in the marsh surface area would affect model response. Similarly, variations in the elevation of the marsh plain could also elicit model changes.

V.4.2.3.3 Wetting and Drying/Marsh Porosity Processes

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

V.4.2.3.4 Comparison of Modeled Tides and Measured Tide Data

Many experimental model runs were performed to determine how changes to various parameters (e.g. friction and turbulent exchange coefficients) affected the model results. These trial runs achieved excellent agreement between the model simulations and the field data, with standard errors on the order of 1 inch. Examples of the simulated tide curves for each of the four inner-estuary locations are shown in Figure V-12.

Once the model was calibrated, a validation model run was performed to test the accuracy of the calibrated model. A new time period was selected for the validation run, from October 4 through October 11, 1999. The period was selected because it included different tidal conditions than contained in the calibration time period. During the validation time period, the tide range was approximately 3 feet or less, slightly reduced relative to the calibration time period in late October. Comparison of the model output and data observations for this simulation is shown in Figure V-13.

A further tidal constituent verification was undertaken for the validation period. The constituent calibration values are presented in Table V-7, and reveal excellent agreement between modeled and measured tides. Standard errors between the model and observed conditions were less than 1 inch on average for all locations, suggesting the model accurately predicts tidal hydrodynamics within the Popponesset Bay system. Measured tidal constituent amplitudes and time lags (ϕ_{lag}) for the validation time period differ from those in Tables V-1 and V-3 because constituents were computed for only seven days, rather than the entire thirty-day period represented in Tables V-1 and V-3. Errors associated with tidal constituent height were on the order of hundredths of feet, which was of the same order of magnitude as the accuracy of the tide gage (0.03 ft). The greatest errors were noticed in Ockway Bay. These errors were likely a result of the deployment problem (i.e. the sensor becoming dry at extreme low tides and, therefore not measuring the full range of the tide) and not due to model inaccuracies.

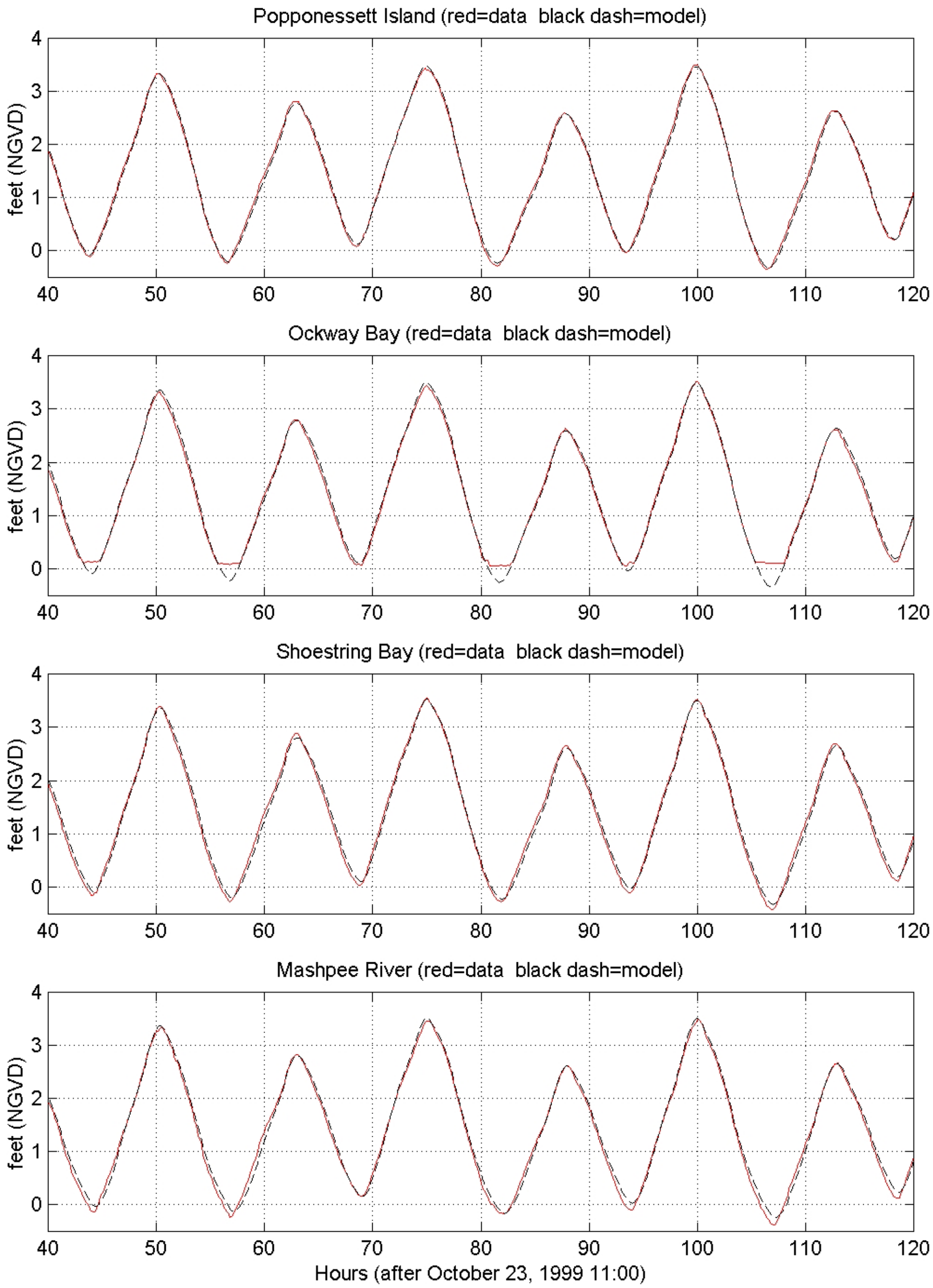


Figure V-12. Comparison of water surface variations simulated by the model (dashed black line) to those measured within the system (red solid line) for the calibration time period at four interior locations.

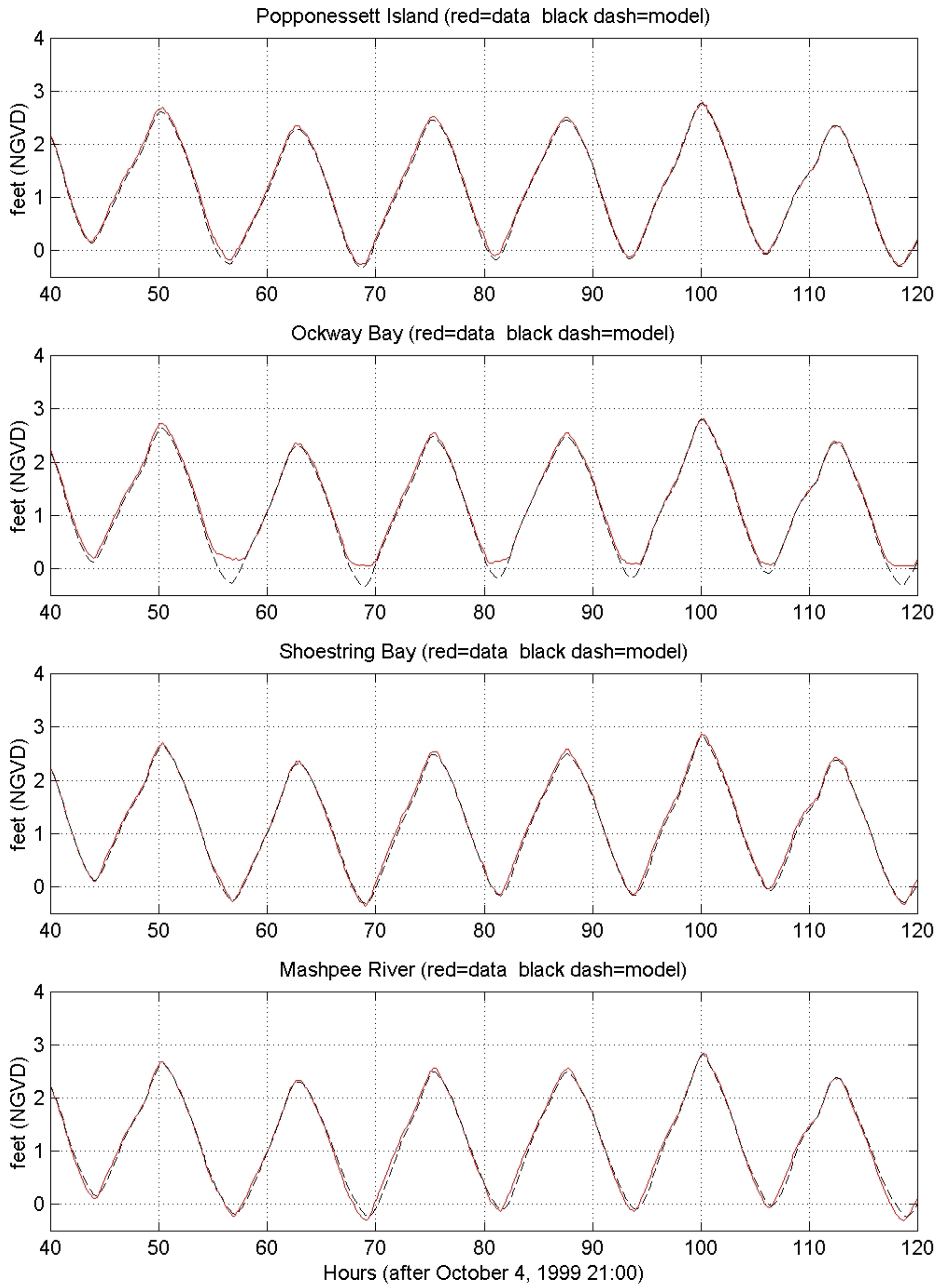


Figure V-13. Comparison of water surface variations simulated by the model (dashed black line) to those measured within the system (red solid line) for the verification time period at four interior locations.

Table V-7. Comparison of Tidal Constituents calibrated RMA2 model versus measured tidal data for the period October 4-11, 1999.

Model Verification Run						
Location	Constituent Amplitude (ft)				Phase (degrees)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Popponesset Island	1.188	0.119	0.080	0.031	17.1	120.3
Ockway Bay	1.193	0.103	0.089	0.032	19.4	124.9
Shoestring Bay	1.193	0.072	0.091	0.033	22.9	126.5
Mashpee River	1.177	0.035	0.081	0.034	24.9	111.1
Measured Tidal Data						
Location	Constituent Amplitude (ft)				Phase (degrees)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Popponesset Island	1.179	0.115	0.084	0.041	12.5	111.0
Ockway Bay	1.154	0.090	0.067	0.043	15.4	100.6
Shoestring Bay	1.195	0.086	0.100	0.044	16.5	116.5
Mashpee River	1.186	0.082	0.096	0.043	17.2	115.7
Error						
Location	Constituent Amplitude (ft)				Phase (minutes)	
	M_2	M_4	M_6	K_1	ΦM_2	ΦM_4
Popponesset Island	0.009	0.004	0.004	0.010	9.5	9.6
Ockway Bay	0.039	0.013	0.022	0.011	8.3	25.2
Shoestring Bay	0.002	0.014	0.009	0.011	13.4	10.4
Mashpee River	0.009	0.047	0.015	0.009	15.9	4.8

V.4.2.3.5 Model Circulation Characteristics

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

From the model run of the Popponesset Bay system, ebb velocities in the channels are slightly larger than velocities during maximum flood. In the inlet to Popponesset Bay from Nantucket Sound, the maximum depth-averaged flood velocities in the model are approximately 2.6 feet/sec, while maximum ebb velocities are about 3.6 feet/sec. A close-up of the model output is presented in Figure V-14, which shows contours of flow velocity, along with velocity vectors which indicate the direction and magnitude of flow, for a single model time-step, at the portion of the tide where maximum flood velocities occur.

In addition to depth averaged velocities, the total flow rate of water flowing through a channel can be computed with the hydrodynamic model. For the flushing analysis in the next section, flow rates were computed across four separate transects in the Popponesset Bay system: at the inlet channel to Popponesset Bay, at the inlet to Mashpee River, and at the inlet to Shoestring Bay, and at the inlet to Ockway Bay. The variation of flow as the tide floods and ebbs is seen in the plot of system flow rates in Figure V-15. Maximum flow rates occur during ebb tides in this system, an indication that this estuary system is ebb dominant. During spring tides, the maximum flood flow rates reach 5200 ft^3/sec at the Popponesset Bay inlet. Maximum ebb flow rates during spring tides are slightly higher, or about 6400 ft^3/sec.

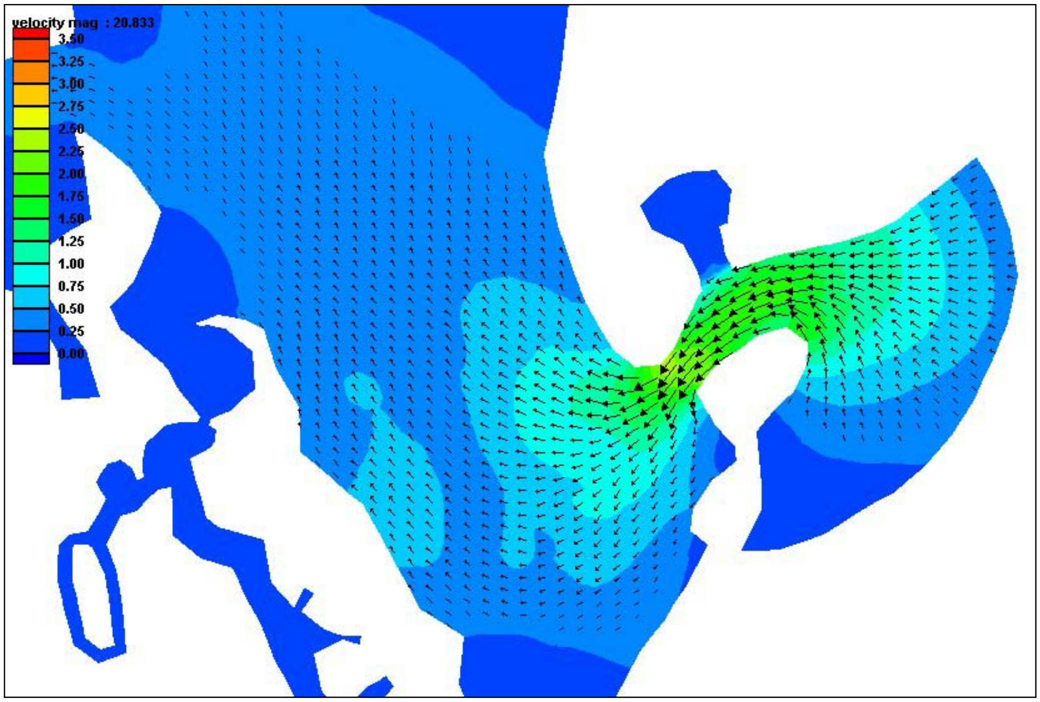


Figure V-14. Example of hydrodynamic model output for a single time step where maximum flood velocities occur for this tide cycle. Color contours indicate flow velocity, and vectors indicate the direction and magnitude of flow.

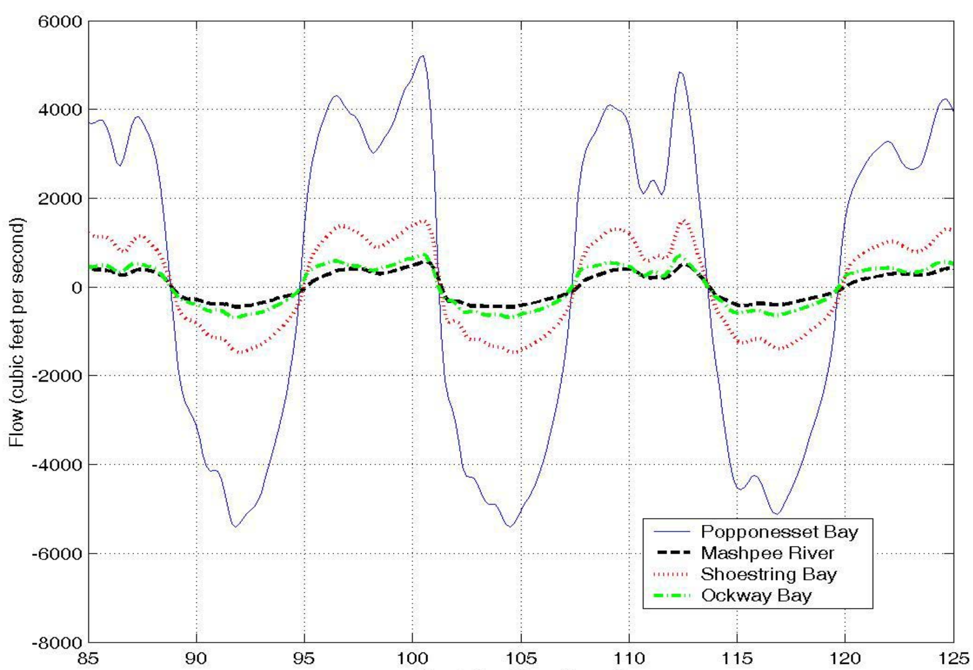


Figure V-15. Time variation of computed flow rates for three transects in the Popponesset Bay system. Model period shown corresponds to spring tide conditions, where the tide range is the largest, and resulting flow rates are correspondingly large compared to neap tide conditions. Positive flow indicates 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 Popponesset 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 Shoestring Bay as an example, the **system residence time** is the average time required for water to migrate from Shoestring Bay, through Popponesset Bay, and into Nantucket Sound, where the **local residence time** is the average time required for water to migrate from Shoestring Bay to just Popponesset Bay (not all the way to the Sound). Local residence times for each sub-embayment are computed as:

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

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

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

The rate of pollutant/nutrient loading and the quality of water outside the estuary both must be evaluated in conjunction with residence times to obtain a clear picture of water quality. Efficient tidal flushing (low residence time) is not an indication of high water quality if pollutants and nutrients are loaded into the estuary faster than the tidal circulation can flush the system. Neither are low residence times an indicator of high water quality if the water flushed into the estuary is of poor quality. Advanced understanding of water quality will be obtained from the calibrated hydrodynamic model by extending the model to include a total nitrogen dispersion model (Section VI). The water quality model will provide a valuable tool to evaluate the complex mechanisms governing estuarine water quality in the Popponesset Bay 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 three 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 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 each sub-embayment, as well as their respective tidal prisms, were computed as cubic feet (Table V-8).

Residence times were averaged for the tidal cycles comprising a representative 7.25 day period (14 tide cycles), and are listed in Table V-9. 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 include 1) the entire system, 2) Mashpee River, 3) Shoestring Bay, and 4) Ockway Bay. The model calculated flow crossing specified grid lines for each sub-embayment to compute the tidal prism volume. Since the 7-day period used to compute the flushing rates of the system represent average tidal conditions, the measurements provide the most appropriate method for determining mean flushing rates for the system sub-embayments.

Table V-8. Embayment mean volumes and average tidal prism during simulation period.

Embayment	Mean Volume (ft^3)	Tide Prism Volume (ft^3)
Popponesset Bay System	119,443,200	86,946,000
Mashpee River	6,562,500	6,875,000
Shoestring Bay	25,195,000	21,707,000
Ockway Bay	12,457,800	9,755,000

Table V-9. Computed System and Local residence times for embayments in the Popponesset Bay system.

Embayment	System Residence Time (days)	Local Residence Time (days)
Popponesset Bay System	0.71	0.71
Mashpee River	8.99	0.49

Shoestring Bay	2.85	0.60
Ockway Bay	6.34	0.66

The computed flushing rates for the Popponesset Bay system show that as a whole, the system flushes well. A flushing time of 0.71 days for the entire estuary indicates that on average, water is resident in the system less than one day. Mashpee River has the greatest system residence time, approximately 9 days. By the definition of system residence time, smaller sub-embayments have longer residence times; therefore, residence times may be misleading for small, remote parts of the estuary. Instead, it is useful to compute a local residence time for each sub-embayment. A local residence time represents the time required for a water parcel to leave the particular sub-embayment. For instance, the local residence time for Ockway Bay represents the time required for a water parcel to be flushed from the sub-embayment into Popponesset Bay. Local residence times are computed as the volume of the sub-embayment divided by the tidal prism of that sub-embayment, and units are converted to days.

Local residence times are significantly lower than residence times based on the volume of the entire estuary. For example, flow entering Shoestring Bay on an average tidal cycle flushes through Popponesset Bay inlet in 6.3 days, but flushes into Popponesset Bay in 0.6 days. Generally, a local residence time is only useful where the adjacent embayment has high water quality. For embayments located in the upper reaches of the system (Ockway Bay, Mashpee River, and Shoestring Bay), the receiving waters that exchange tidal flow with the various sub-embayments show signs of ecological stress, indicative of poor water quality. Therefore, system residence times may be more appropriate for future planning scenarios.

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. For the purposes of this study, a coastal area exhibiting “strong littoral drift” is defined as a system that has noticeable longshore tidal currents and frequent wave-induced mixing of estuarine waters entering the coastal region. 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 Nantucket Sound shorelines in Mashpee/Barnstable typically is strong and 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

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 five embayment systems. Files of node locations and node connectivity for the RMA-2V model grids were transferred to the RMA-4 water quality model; therefore, the computational grid for the hydrodynamic model also was the computational grid for the water quality model. The period of hydrodynamic output for the water quality model calibration was a 12-tidal cycle period in autumn 1999 that includes the fortnightly variation between spring and neap tide ranges. For each modeled scenario (e.g., present conditions, build-out) the model was run for a 30-day spin-up period, to allow the model to reach a dynamic "steady state", and ensure that model spin-up would not affect the final model output.

VI.1.2 Nitrogen Loading to the Embayments

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

VI.1.3 Measured Nitrogen Concentrations in the Embayments

In order to create a model that realistically simulates the total nitrogen concentrations in a system in response to the existing flushing conditions and loadings, it is necessary to calibrate the model to actual measurements of water column nitrogen concentrations. The refined and approved data for each monitoring station used in the water quality modeling effort are presented in Table VI-1. Station locations are indicated in Figure VI-1. The multi-year averages present the "best" comparison to the water quality model output, since factors of tide, temperature and rainfall may exert short-term influences on the individual sampling dates and even cause inter-annual differences. Three years of baseline field data is the minimum required to provide a baseline for MEP analysis. For the Popponesset Bay System water quality data was collected by the Popponesset Bay Water Quality Monitoring Program, which was established by the Town of Mashpee with the Coastal Systems Program at SMAST-UMASS-Dartmouth in 1997. The multiple departments and groups taking part in this effort were coordinated by the Mashpee Waterways Commission (see Chapter II). For most of the 15 sampling stations, seven years of data were available. Average summer total nitrogen within the waters at each station was used to calibrate the model. Extremely high or low values (<5% of samples), as determined by a departure criterion of >3 standard deviations from the mean, were not included in the averages. Deletion of data outliers was deemed appropriate, since the water quality module of the Linked Watershed-Embayment Model represents typical or average summertime conditions throughout the embayment.

Table VI-1. Measured and modeled Nitrogen concentrations for the Popponesset Bay system used in the model calibration plots of Figure VI-2. All concentrations are given in mg/L N. "Data mean" values are calculated as the average of the separate yearly means. Overall mean is presented as "data mean" with the standard deviation (s.d.) and number of total samples (N).

Sub-Embayment	monitoring station	1997 mean	1998 mean	1999 mean	2000 mean	2001 mean	2002 mean	2003 mean	data mean	s.d. all data	N	model min	model max	Model avg
Mashpee River - head (MRh)	PB 1	0.798	0.633	0.787	0.831	0.990	1.137	0.756	0.859	0.200	26	0.690	0.893	0.771
Mashpee River - Upper (MRu)	PB2	1.006	0.726	1.022	0.798	1.082	1.153	0.892	0.958	0.242	24	0.774	0.944	0.855
Mashpee River - Mid (MRm)	PB3	0.651	0.764	0.669	0.596	0.740	1.120	0.733	0.739	0.216	25	0.510	0.941	0.783
Mashpee River - Lower (MRl)	PB4	0.603	0.668	0.564	0.485	0.695	0.694	0.736	0.627	0.134	25	0.341	0.889	0.561
Shoestring Bay - head (SBh)	SR 5	-	1.193	0.860	0.878	1.278	1.132	1.377	1.135	0.380	20	0.956	1.166	1.067
Shoestring Bay - upper (SBu)	PB5	0.730	0.878	0.606	0.594	0.678	0.580	0.870	0.690	0.169	26	0.644	0.754	0.692
Shoestring Bay - mid (SBm)	PB 6	0.617	0.695	0.644	0.671	0.668	0.707	0.884	0.688	0.140	28	0.498	0.713	0.620
Shoestring Bay - lower (SBl)	PB 7	0.518	0.551	0.467	0.506	0.507	0.527	0.592	0.520	0.113	27	0.379	0.670	0.524
Ockway Bay - upper (OBu)	PB 9	0.552	0.569	0.498	0.486	1.003	0.785	0.734	0.677	0.217	27	0.537	0.570	0.551
Ockway Bay - lower (OBl)	PB 10	0.485	0.508	0.426	0.467	0.765	0.592	0.512	0.536	0.177	27	0.328	0.556	0.464
Popponesset Bay – head (PBh)	PB 8	0.476	0.589	0.444	0.592	0.772	0.595	0.732	0.581	0.151	24	0.306	0.667	0.451
Popponesset Bay – upper (PBu)	PB 11	0.307	0.521	0.434	0.417	0.579	0.567	0.506	0.485	0.111	23	0.289	0.533	0.374
Popponesset Bay - mid (PBm)	PB 12	0.343	0.492	0.393	0.473	0.539	0.554	0.486	0.456	0.102	25	0.282	0.485	0.325
Popponesset Creek (POC)	PB 13	0.369	0.376	0.351	0.486	0.591	0.456	0.479	0.422	0.107	23	0.355	0.376	0.366
Pinquickset Cove (PQC)	PB 15	-	-	-	-	-	0.470	0.640	0.527	0.097	6	0.331	0.447	0.407
Nantucket Sound (NAS)	PB 14	0.260	0.282	0.297	0.326	0.368	0.351	0.375	0.315	0.055	22	0.283	0.320	0.288

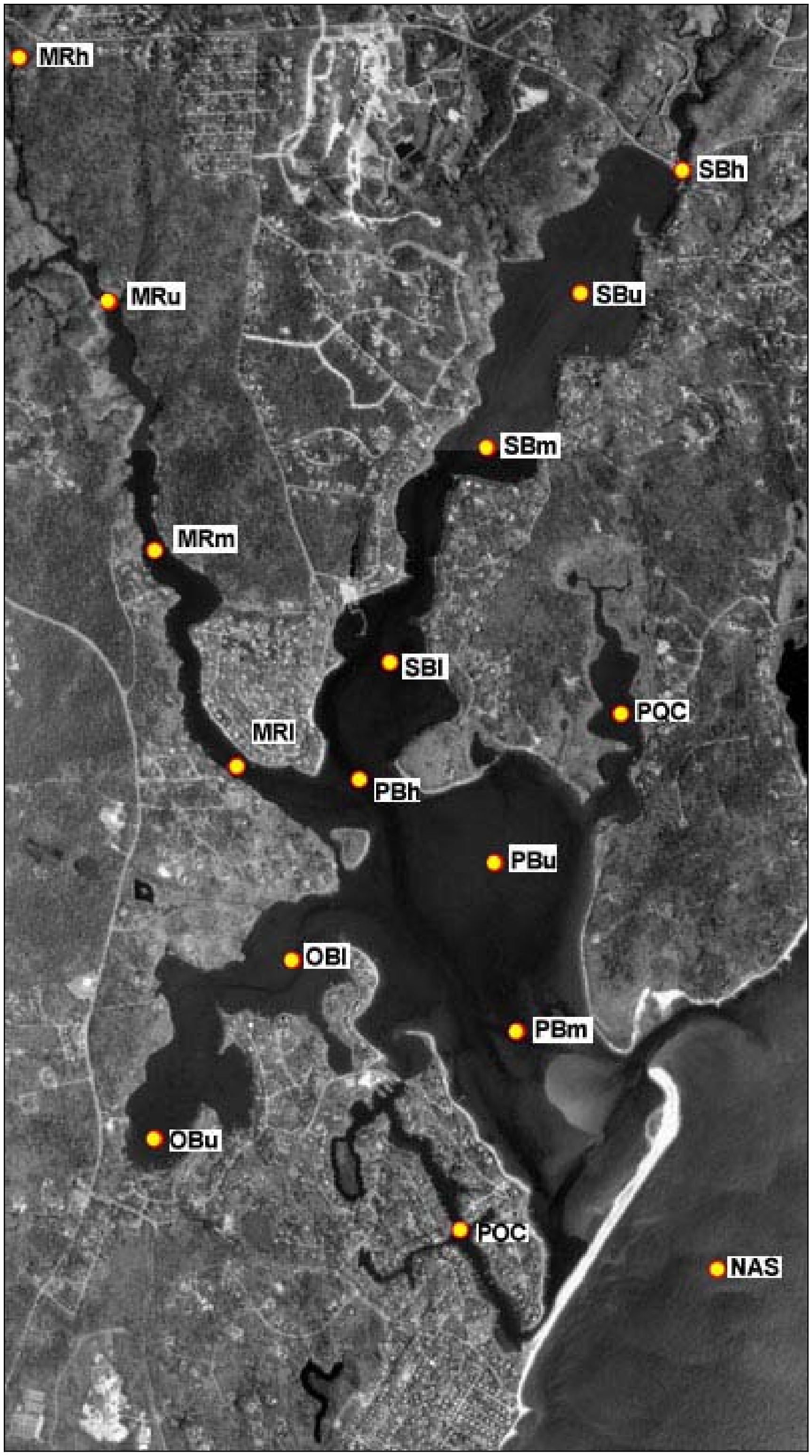


Figure VI-1. Estuarine water quality monitoring station locations in Popponesset Bay. Station labels correspond to those provided in Table VI-1.

VI.2 MODEL DESCRIPTION AND APPLICATION

A two-dimensional finite element water quality model, RMA-4 (King, 1990), was employed to study the effects of nitrogen loading in the Popponesset estuarine system. The RMA-4 model has the capability for the simulation of advection-diffusion processes in aquatic environments. It is the constituent transport model counterpart of the RMA-2 hydrodynamic model used to simulate the fluid dynamics of the Popponesset system. Like RMA-2 numerical code, RMA-4 is a two-dimensional, depth averaged finite element model capable of simulating time-dependent constituent transport. The RMA-4 model was developed with support from the US Army Corps of Engineers (USACE) Waterways Experiment Station (WES), and is widely accepted and tested. Applied Coastal staff have utilized this model in water quality studies of other Cape Cod embayments, including West Falmouth Harbor and the “finger” ponds of Falmouth, MA (Ramsey *et al*., 2000), and embayment systems in Chatham, MA (Howes *et al*., 2003).

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

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 Popponesset 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 Popponesset Bay system.

VI.2.2 Water Quality Model Setup

Required inputs to the RMA-4 model include a computational mesh, computed water elevations and velocities at all nodes of the mesh, constituent mass loading, and spatially varying values of the dispersion coefficient. Because the RMA-4 model is part of a suite of integrated computer models, the finite-element meshes and the resulting hydrodynamic simulations previously developed for Popponesset Bay 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, the Popponesset Bay hydrodynamic model was set-up to include the latest estimates of surface water flows from Mashpee River, Quaker Run River and Santuit River. Surface freshwater inputs from these streams are significant compared to the tidal prisms of the embayments to which they discharge. The Mashpee River has a measured flowrate of 10.7 ft^3/sec (26,223 m^3/day), which is 7.0% of the volume exchanged daily by the tide in the estuarine portion of the River. The Santuit and Quaker Run Rivers have average flows of 5.4 ft^3/sec and 1.5 ft^3/sec (13,164 m^3/day and 3,730 m^3/day) respectively, which total to 1.9% of the daily tidal exchange in Shoestring Bay.

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 for Popponesset Bay.

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 Santuit and Mashpee Rivers. 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 Shoestring Bay were evenly distributed at grid cells that formed the perimeter of the embayment.

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

In addition to mass loading boundary conditions set within the model domain, concentrations along the model open boundary was specified. The model uses 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 concentration in Nantucket Sound was set at 0.285 mg/L, based on SMAST data from the Nantucket Sound. The open boundary total nitrogen concentration represents long-term average summer concentrations found within Nantucket Sound.

The boundary condition for Popponesset Bay that was used in the Draft Final Report was 0.305 mg N/L. The reason that 0.315 mgN/L (Station PB14) was not used was that (1) the data were highly variable from year to year and (2) there was concern from a many of Technical Staff the station was within the ebb tidal "plume" from Popponesset Bay. Therefore, a lower offshore boundary condition (0.305mgN/L) was used, based on a variety of estimates. However, subsequent to completion of the Draft Final Report, the core Technical Group convened to specifically address this issue. The consensus was that (1) the "plume" concern about station PB14 cannot be discounted and (2) the "best" estimate of the Nantucket Sound Boundary was the long-term monitoring station off of Green Pond. This station has been monitored since 1987 and has consistently yielded a value of 0.285 mgN/L. Therefore, the modeling in the Final Report reflects this new boundary condition. Note that the boundary condition basically lowers the nitrogen background, but does not affect the other modeling parameters (a change is boundary concentration is basically subtracting a constant from all values).

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

sub-embayment	watershed load (kg/day)	direct atmospheric deposition (kg/day)	benthic flux (kg/day)
Popponesset Bay	1.82	4.01	-5.04
Popponesset Creek	4.94	-	-0.64
Pinquickset Cove	0.76	0.29	-0.33
Ockway Bay - lower	-	-	-1.60
Ockway Bay - upper	3.15	1.09	3.37
Mashpee River	12.11	0.66	11.47
Shoestring Bay	9.21	2.23	-11.85
Surface Water Sources			
Mashpee River	15.56	-	-
Santuit River (Shoestring Bay)	15.58	-	-
Quaker Run River (Shoestring Bay)	5.98	-	-

VI.2.4 Model Calibration

Calibration of the Popponesset Bay total nitrogen model proceeded by changing model dispersion coefficients so that model output of nitrogen concentrations matched measured data. Generally, several model runs of each system were required to match the water column measurements. Dispersion coefficient (*E*) values were varied through the modeled 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 Popponesset 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-3. These values were used to develop the "best-fit" total nitrogen model calibration. For the case of TN modeling, "best fit" can be defined as minimizing the error between the model and data at all sampling locations, utilizing reasonable ranges of dispersion coefficients within each sub-embayment.

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

Embayment Division	E m^2/sec
Popponesset Bay	2.0
Popponesset Inlet	5.0
Nantucket Sound	5.0
Lower Mashpee River	10.0
Upper Mashpee River	0.5
Lower Shoestring Bay	10.0
Upper Shoestring Bay	4.0
Lower Ockway Bay	1.0
Upper Ockway Bay	0.25
Popponesset Creek	5.0
Pinquickset Cove	0.25
Santuit River	0.4
Quaker Run River	0.4

Comparisons between model output and measured nitrogen concentrations are shown in Figure VI-2. 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 Mashpee River. At other locations (e.g., Ockway Bay), 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-3. In this figure, color contours indicate nitrogen concentrations throughout the model domain. The output in these figures show average total nitrogen concentrations, computed using the full 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 Popponesset Bay using Salinity data collected at the same stations as the nitrogen data. The only required inputs into the RMA4 salinity model of Popponesset bay, 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 was set at 28.5 ppt. For the stream inputs, salinities were set at 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 Figure VI-4, with a contour plot of model output shown in Figure VI-5. Though model dispersion coefficients were not changed from those values selected through the nitrogen model calibration process, the model adequately represents salinity gradients in the Popponesset Bay system. 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.

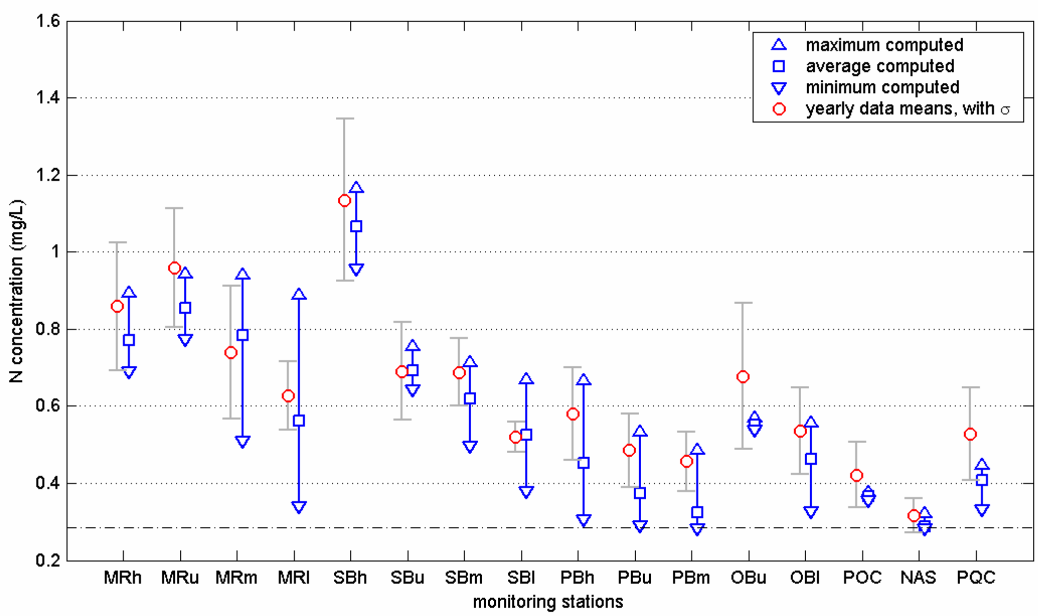


Figure VI-2. Comparison of measured total nitrogen concentrations and calibrated model output at stations in the Popponesset Bay system. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), along with the average computed 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. The background concentration (0.285 mg/L) in Nantucket Sound is indicated by the black dot-dashed line.

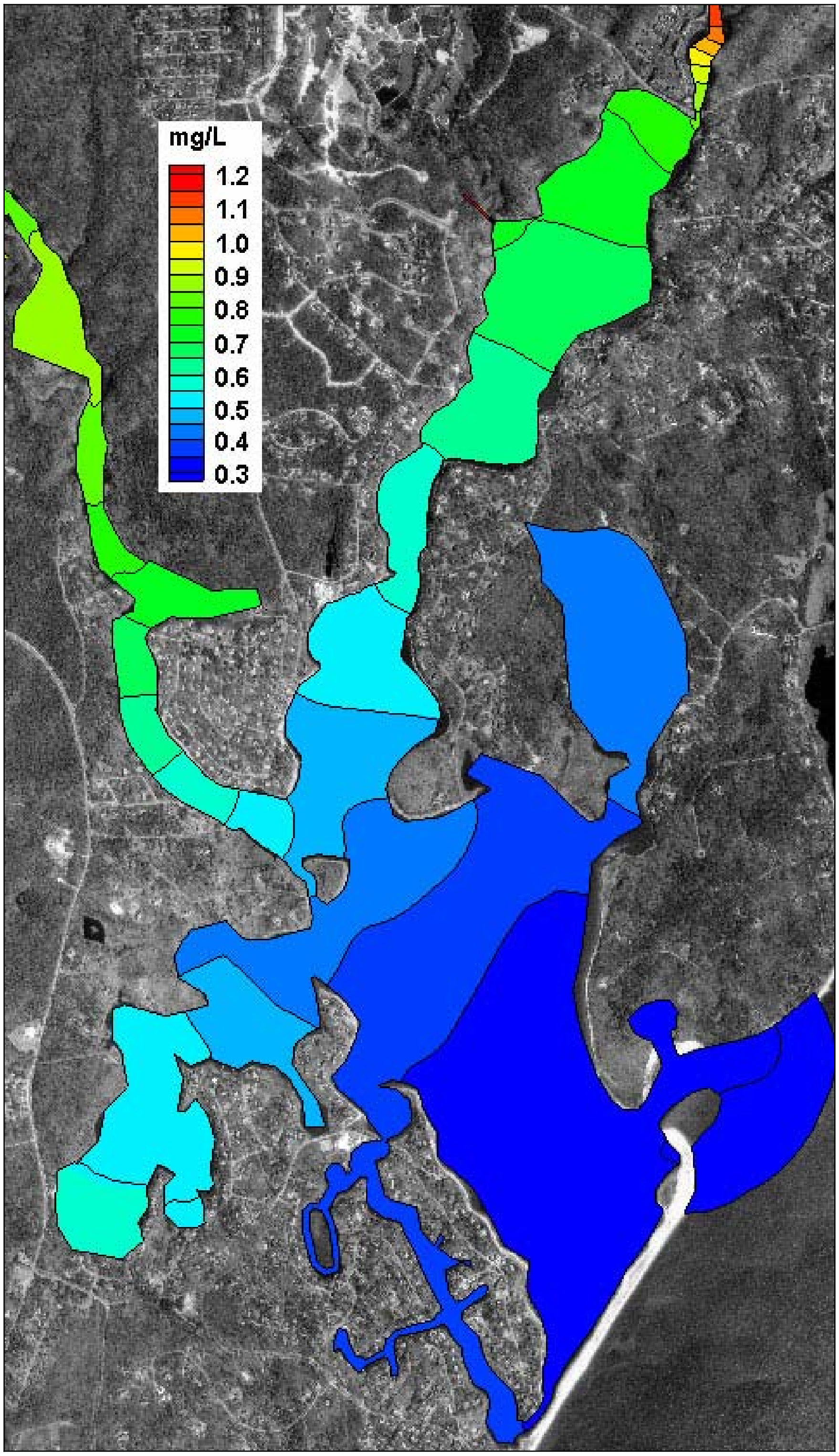


Figure VI-3. Contour plot of average total nitrogen concentrations from results of the present conditions loading scenario, for Popponesset Bay.

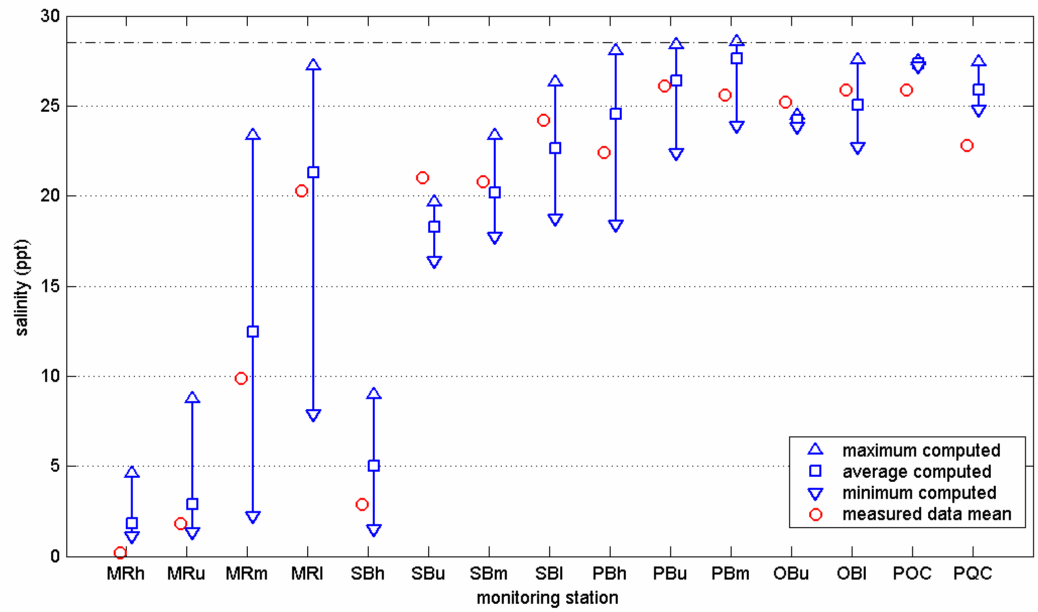


Figure VI-4. Comparison of measured and calibrated model output at stations in the Popponesset Bay system. Stations labels correspond with those provided in Table VI-1. Model output is presented as a range of values from minimum to maximum values computed during the simulation period (triangle markers), 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). The background salinity (28.5 ppt) in Nantucket Sound is indicated by the black dot-dashed line.

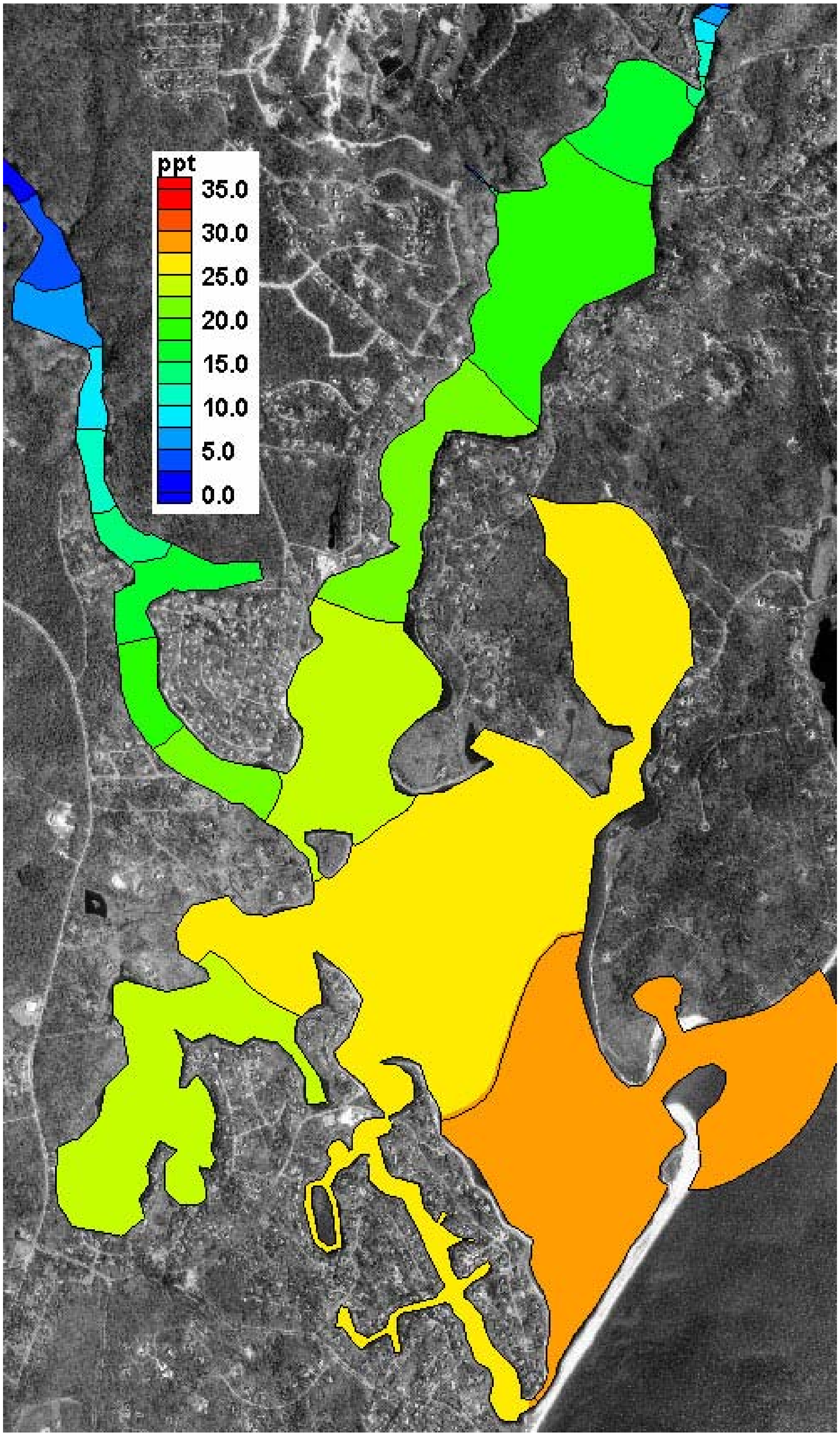

ppt
35.0
30.0
25.0
20.0
15.0
10.0
5.0
0.0

Figure VI-5. Contour Plot of modeled salinity (ppt) in Popponesset 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-4. Loads are presented in kilograms per day (kg/day) in this Section, since it is inappropriate to show benthic flux loads in kilograms per year due to seasonal variability.

In general, certain sub-embayments would be impacted more than others. The build-out scenario indicates that there would be less than a 9% increase in watershed nitrogen load to the lower portion of the Popponesset Bay system as a result of potential future development. Other watershed areas would experience much greater load increases, for example the loads to Shoestring Bay and Ockway Bay would increase 25% and 35% respectively from the present day loading levels. A maximum increase in watershed loading resulting from future development would occur in the freshwater section of the Mashpee River, where the increase would be 139%. 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%.

Table VI-4. Comparison of sub-embayment watershed loads used for modeling of present, build out, and no-anthropogenic ("no-load") loading scenarios of the Popponesset Bay 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
Popponesset Bay	1.82	1.98	+8.9%	0.08	-95.6%
Popponesset Creek	4.94	5.35	+8.4%	0.10	-97.9%
Pinquickset Cove	0.76	0.98	+28.7%	0.11	-85.7%
Ockway Bay	3.15	4.25	+35.0%	0.24	-76.0%
Mashpee River	12.11	17.57	+45.1%	0.62	-79.4%
Shoestring Bay	9.21	11.47	+24.5%	0.34	-75.5%
Surface Water Sources					
Mashpee River	15.56	37.15	+138.7%	4.68	-69.9%
Santuit River (Shoestring Bay)	15.58	21.46	+37.7%	1.27	-91.8%
Quaker Run River (Shoestring Bay)	5.98	6.62	+10.6%	0.24	-96.0%

For the build out scenario, a breakdown of the total nitrogen load entering each sub-embayment is shown in Table VI-5. The benthic flux for the build-out scenarios is assumed to vary proportional to the watershed load, where an increase in watershed load will result in an increase (positive) in benthic flux.

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-6, with greater than 30% increases in total Nitrogen concentrations in the upper portions of the Popponesset

Bay system. Color contours of model output for the build-out scenario are present in Figure VI-6. The range of nitrogen concentrations shown are the same as for the plot of present conditions in Figure VI-3, which allows direct comparison of nitrogen concentrations between loading scenarios.

Table VI-5. Build-out sub-embayment and surface water loads used for total nitrogen modeling of the Popponesset Bay 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)
Popponesset Bay	1.98	4.01	-5.13
Popponesset Creek	5.35	-	-0.65
Pinquickset Cove	0.98	0.29	-0.33
Ockway Bay - lower	-	-	-1.81
Ockway Bay - upper	4.25	1.09	3.89
Mashpee River	17.57	0.66	21.30
Shoestring Bay	11.46	2.23	-13.34
Surface Water Sources			
Mashpee River	37.14	-	-
Santuit River (Shoestring Bay)	21.46	-	-
Quaker Run River (Shoestring Bay)	6.62	-	-

Table VI-6. Comparison of model average total N concentrations from present loading and the build-out scenario, with percent change, for the Popponesset Bay system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

Sub-Embayment	monitoring station	present (mg/L)	build-out (mg/L)	% change
Mashpee River - head (MRh)	PB 1	0.771	1.650	113.9%
Mashpee River - Upper (MRu)	PB2	0.855	1.736	102.9%
Mashpee River - Mid (MRm)	PB3	0.783	1.361	73.8%
Mashpee River - Lower (MRl)	PB4	0.561	0.818	45.9%
Shoestring Bay - head (SBh)	SR 5	1.067	1.473	38.1%
Shoestring Bay - upper (SBu)	PB5	0.692	0.955	38.0%
Shoestring Bay - mid (SBm)	PB 6	0.620	0.856	38.1%
Shoestring Bay - lower (SBl)	PB 7	0.524	0.701	33.8%
Ockway Bay - upper (OBu)	PB 9	0.551	0.714	29.4%
Ockway Bay - lower (OBl)	PB 10	0.464	0.586	26.3%
Popponesset Bay - head (PBh)	PB 8	0.451	0.578	28.2%
Popponesset Bay - upper (PBu)	PB 11	0.374	0.442	18.1%
Popponesset Bay - mid (PBm)	PB 12	0.325	0.354	9.1%
Popponesset Creek (POC)	PB 13	0.366	0.407	11.4%
Pinquickset Cove (PQC)	PB 15	0.407	0.495	21.5%

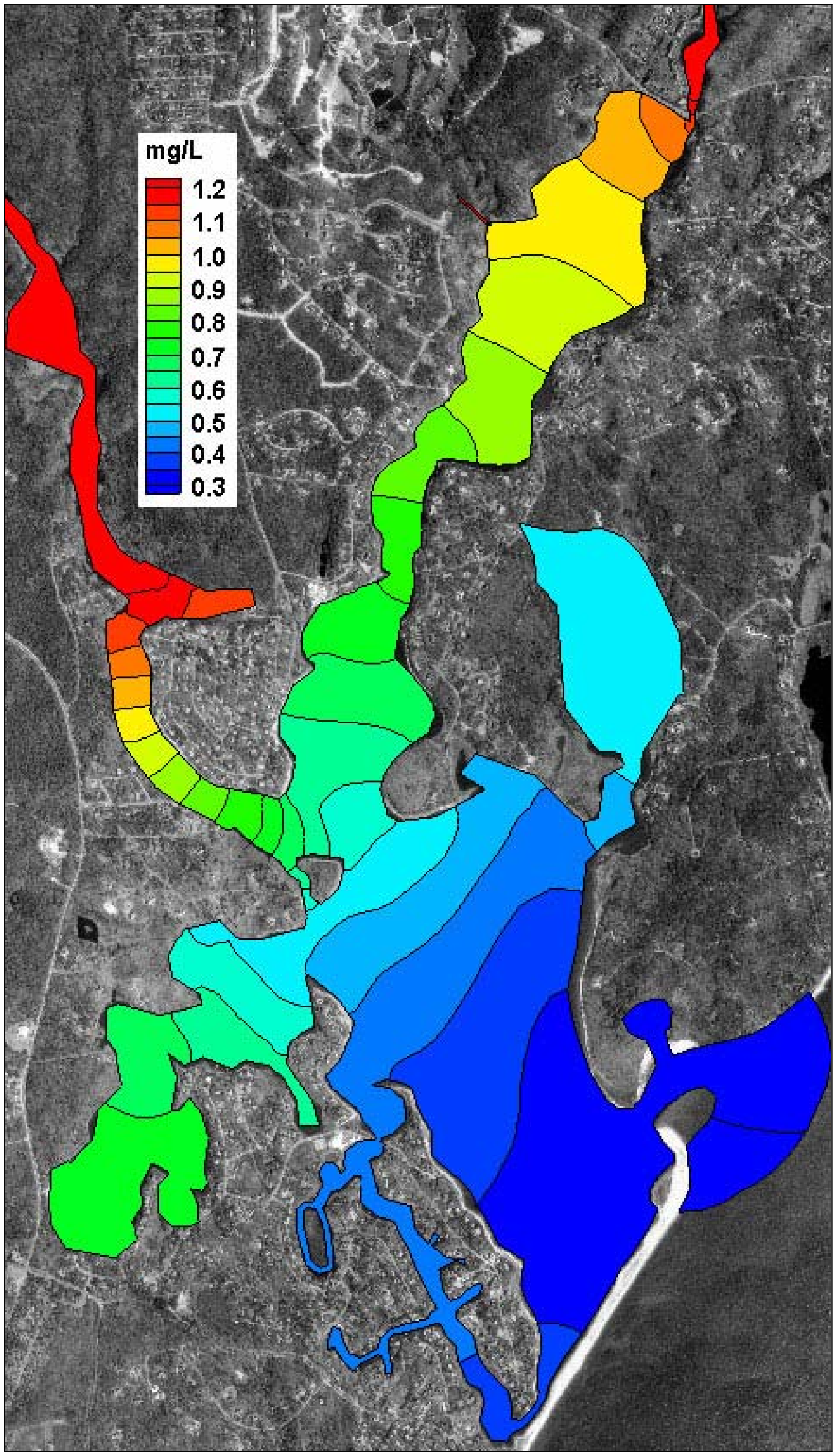


Figure VI-6. Contour Plot of modeled total nitrogen concentrations (mg/L) in Popponesset Bay, for projected build out loading conditions.

A breakdown of the total nitrogen load entering each sub-embayment for the no anthropogenic load scenarios is shown in Table VI-7. The benthic flux input to each embayment 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. Again, total nitrogen concentrations in the receiving waters (i.e., Nantucket Sound) remained identical to the existing conditions modeling scenarios. The relative change in total nitrogen concentrations resulting from "no load" was significant as shown in Table VI-8, with reductions greater than 35% occurring the upper portions of the system. These results are shown pictorially in Figure VI-7.

For the no load scenario, the sub-embayment concentrations are generally governed by the total nitrogen concentrations observed in Nantucket Sound. There is a negative gradient in total nitrogen concentrations from the inlet to Popponesset Bay to the upper reaches. This is different from the modeled present and build-out conditions, where concentrations increase from the inlet to the upper reaches of the system. The slight negative gradient in the modeled "no-load" scenario results because the surface freshwater inputs have little load themselves, and dilute concentrations at the heads of Shoestring Bay and the Mashpee River.

Table VI-7. "No anthropogenic loading" sub-embayment and surface water loads used for total nitrogen modeling of the Popponesset Bay 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)
Popponesset Bay	0.08	4.01	-4.83
Popponesset Creek	0.10	-	-0.61
Pinquickset Cove	0.11	0.29	-0.31
Ockway Bay - lower	-	-	-1.03
Ockway Bay - upper	0.24	1.09	2.00
Mashpee River	0.62	0.66	6.56
Shoestring Bay	0.34	2.23	-4.43
Surface Water Sources			
Mashpee River	4.68	-	-
Santuit River (Shoestring Bay)	1.27	-	-
Quaker Run River (Shoestring Bay)	0.24	-	-

Table VI-8. Comparison of model average total N concentrations from present loading and the no anthropogenic ("no load") scenario, with percent change, for the Popponesset Bay system. Loads are based on atmospheric deposition and a scaled N benthic flux (scaled from present conditions).

Sub-Embayment	monitoring station	present (mg/L)	no load (mg/L)	% change
Mashpee River - head (MRh)	PB 1	0.771	0.235	-69.6%
Mashpee River - Upper (MRu)	PB2	0.855	0.264	-69.2%
Mashpee River - Mid (MRm)	PB3	0.783	0.294	-62.5%
Mashpee River - Lower (MRl)	PB4	0.561	0.291	-48.2%
Shoestring Bay - head (SBh)	SR 5	1.067	0.129	-87.9%
Shoestring Bay - upper (SBu)	PB5	0.692	0.244	-64.8%
Shoestring Bay - mid (SBm)	PB 6	0.620	0.261	-58.0%
Shoestring Bay - lower (SBl)	PB 7	0.524	0.274	-47.8%
Ockway Bay - upper (OBu)	PB 9	0.551	0.319	-42.2%
Ockway Bay - lower (OBl)	PB 10	0.464	0.296	-36.1%
Popponesset Bay - head (PBh)	PB 8	0.451	0.281	-37.7%
Popponesset Bay - upper (PBu)	PB 11	0.374	0.283	-24.3%
Popponesset Bay - mid (PBm)	PB 12	0.325	0.285	-12.3%
Popponesset Creek (POC)	PB 13	0.366	0.282	-22.8%
Pinquickset Cove (PQC)	PB 15	0.407	0.284	-30.2%

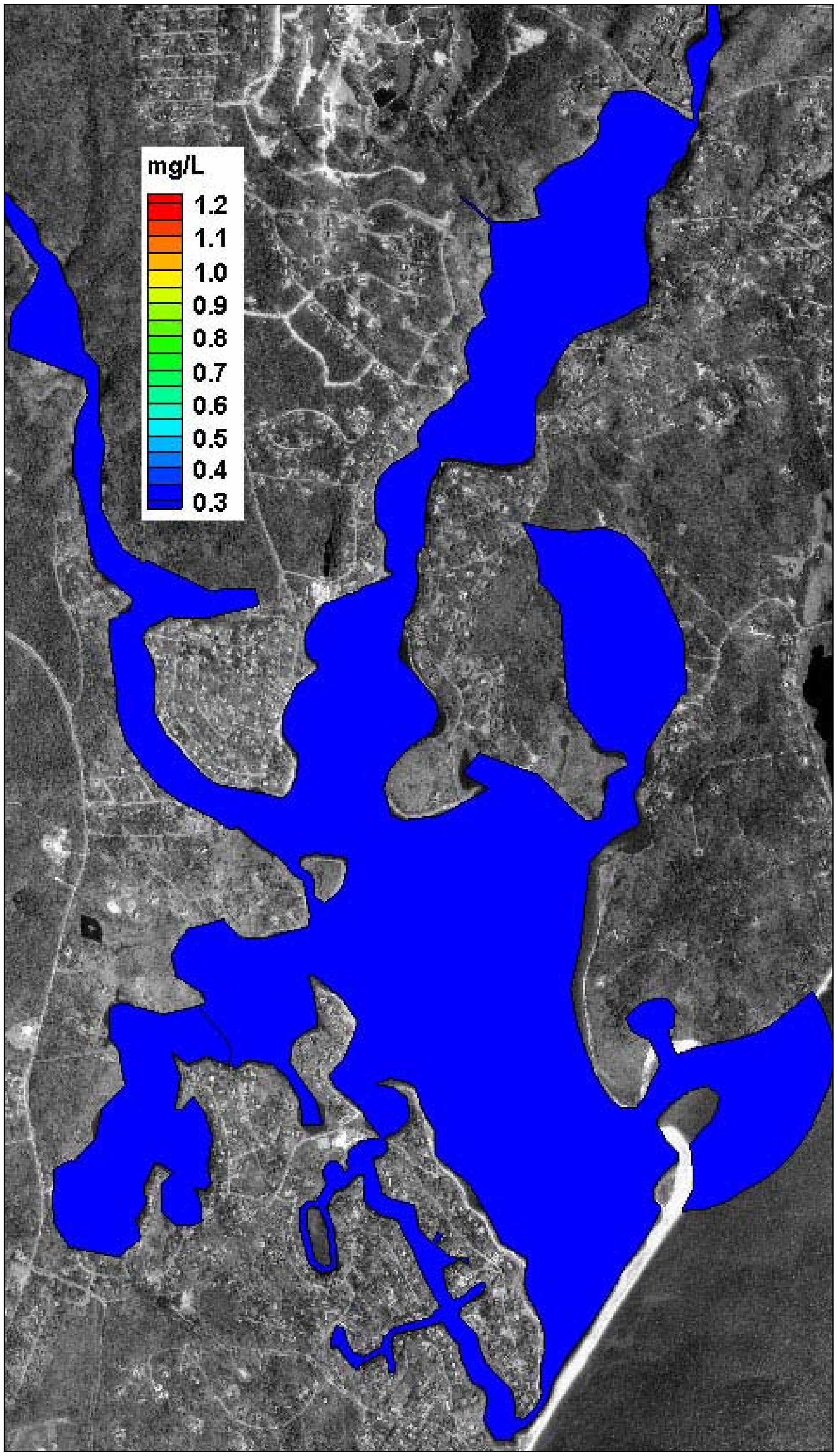


Figure VI-7. Contour Plot of modeled total nitrogen concentrations (mg/L) in Popponesset Bay, for no anthropogenic loading conditions.

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 and the plant (eelgrass, macroalgae) and animal communities (fish, shellfish, infauna) which it supports. For the Popponesset Bay embayment system in the Towns of Mashpee and Barnstable, 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, and dissolved oxygen records conducted during the summers of 1998 and 1999. 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, as well as closer to the inlet to Popponesset Bay, to record the frequency and duration of low oxygen conditions during the critical summer period. The MEP habitat analysis uses eelgrass as a sentinel species for indicating nitrogen over-loading to coastal embayments. Eelgrass is a fundamentally important species in the ecology of shallow coastal systems, providing both habitat structure and sediment stabilization. Mapping of the eelgrass beds within the Popponesset Bay System was conducted for comparison to historic records (DEP Eelgrass Mapping Program, C. Costello). Temporal trends in the distribution of eelgrass beds are used by the MEP to assess the stability of the habitat and to determine trends potentially related to water quality. Eelgrass beds can decrease within embayments in response to a variety of causes, but throughout almost all of the embayments within southeastern Massachusetts, the primary cause appears to be related to increases in embayment nitrogen levels. Within the Popponesset Bay System, temporal changes in eelgrass distribution provides a strong basis for evaluating recent increases (nitrogen loading) or decreases (increased flushing-new inlet) in nutrient enrichment.

In areas that do not support eelgrass beds, benthic animal indicators were used to assess the level of habitat health from "healthy" (low organic matter loading, high D.O.) to "highly stressed" (high organic matter loading-low D.O.). The basic concept is that certain species or species assemblages reflect the quality of their habitat. Benthic animal species from sediment samples were identified and the environments ranked based upon the fraction of healthy, transitional, and stressed indicator species. The analysis is based upon life-history information on the species and a wide variety of field studies within southeastern Massachusetts waters, including the Wild Harbor oil spill, 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 Popponesset Bay System are currently listed under this Classification as SA. It should be noted that the Classification system represents the water quality that the embayment should support, not the existing level of water quality. It is through the MEP and TMDL processes that management actions are developed and implemented to keep or bring the existing conditions in line with the Classification.

Dissolved oxygen levels in temperate embayments vary seasonally, due to changes in oxygen solubility, which varies inversely with temperature. In addition, biological processes that consume oxygen from the water column (water column respiration) vary directly with temperature, with several fold higher rates in summer than winter (Figure VII-1). It is not surprising that the largest levels of oxygen depletion (departure from atmospheric equilibrium) and lowest absolute levels (mg L^{-1}) are found during the summer in southeastern Massachusetts embayments when water column respiration rates are greatest. Since oxygen levels can change rapidly, several mg L^{-1} in a few hours, traditional grab sampling programs typically underestimate the frequency and duration of low oxygen conditions within shallow embayments (Taylor and Howes, 1994). To more accurately capture the degree of bottom water dissolved oxygen depletion during the critical summer period, autonomously recording oxygen sensors were moored 30 cm above the embayment bottom within key regions of the Popponesset Bay System (Figure VII-2). The sensors (YSI 6600) were first calibrated in the laboratory and then checked with standard oxygen mixtures at the time of initial instrument mooring deployment. In addition periodic calibration samples were collected at the sensor depth and assayed by Winkler titration (potentiometric analysis, Radiometer) during each deployment. Each instrument mooring was serviced and calibration samples collected at least biweekly and sometimes weekly during a minimum deployment of 30 days within the interval from July through mid-September. All of the mooring data from the Popponesset Bay embayment system was collected during 1997, 1998 and 1999.

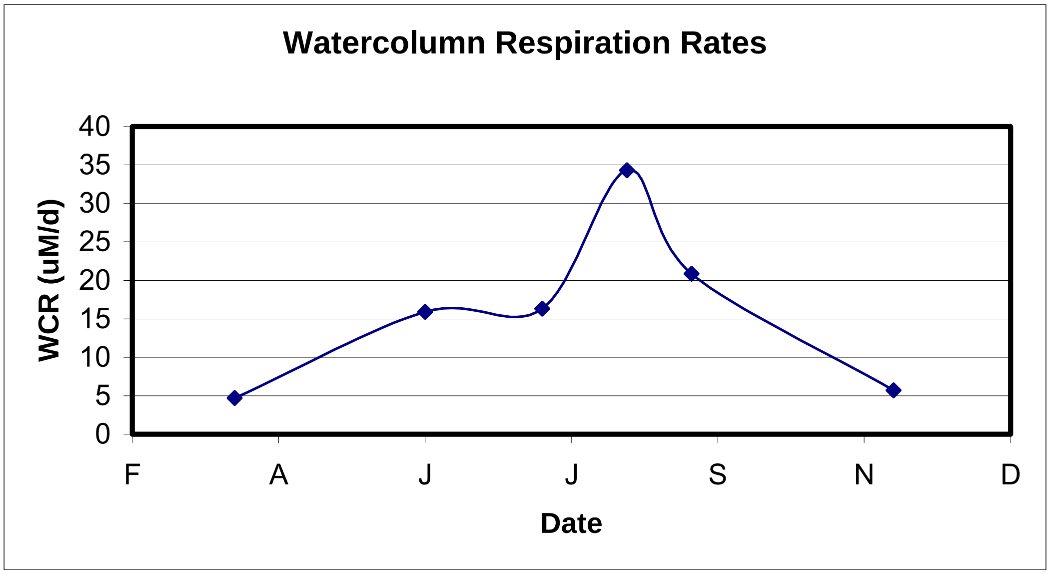


Figure VII-1. Average watercolumn respiration rates (micro-Molar/day) from water collected throughout the Popponesset Bay System at the stations shown in Figure VII-11 (Schlezinger and Howes, unpublished data). Rates vary ~7 fold from winter to summer as a result of variations in temperature and organic matter availability.

Similar to other embayments in southeastern Massachusetts, the sub-embayments to the overall Popponesset 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 highly nutrient enriched waters and impaired habitat quality at all but the lower-most mooring site (Popponesset Upper-Daniels Island, Figure VII-3 versus Figures VII-4 through VII-9).

The dissolved oxygen records indicate that the upper region of the Popponesset Bay Central Basin (the area generally associated with the Popponesset Bay Outer – Daniels Island DO mooring station) is currently maintaining adequate oxygen levels. However, the other regions of the Bay show significant depletions during summer. The oxygen depletion data indicate that the upper-most region of the central basin (south of Gooseberry Island), the region defined by the entrances to the Mashpee River and Shoestring Bay, and Shoestring Bay are exhibiting similar levels of oxygen depletion. Ockway Bay is showing greater levels of oxygen depletion, while the lower and upper regions of the Mashpee River are exhibiting oxygen excursions and depletion levels consistent with extremely high levels of organic matter loading.

Dissolved oxygen records were analyzed to determine the percent of the deployment time (18-45 days) that oxygen was below various benchmark concentrations (Table VII-1). 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

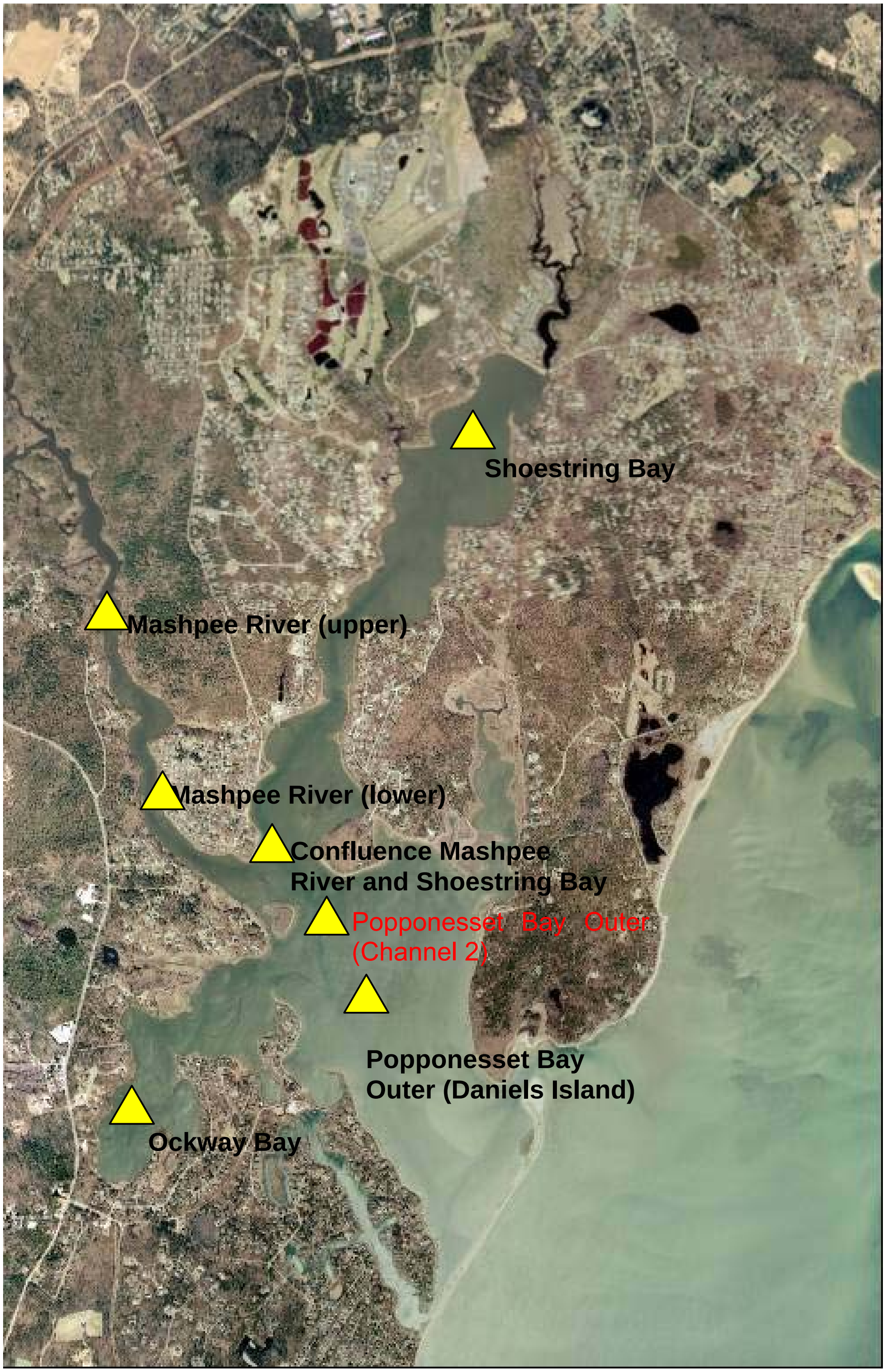


Figure VII-2. Aerial Photograph of the Popponesset Bay system in Mashpee showing locations of Dissolved Oxygen mooring deployments conducted in Summer 1998 and 1999.

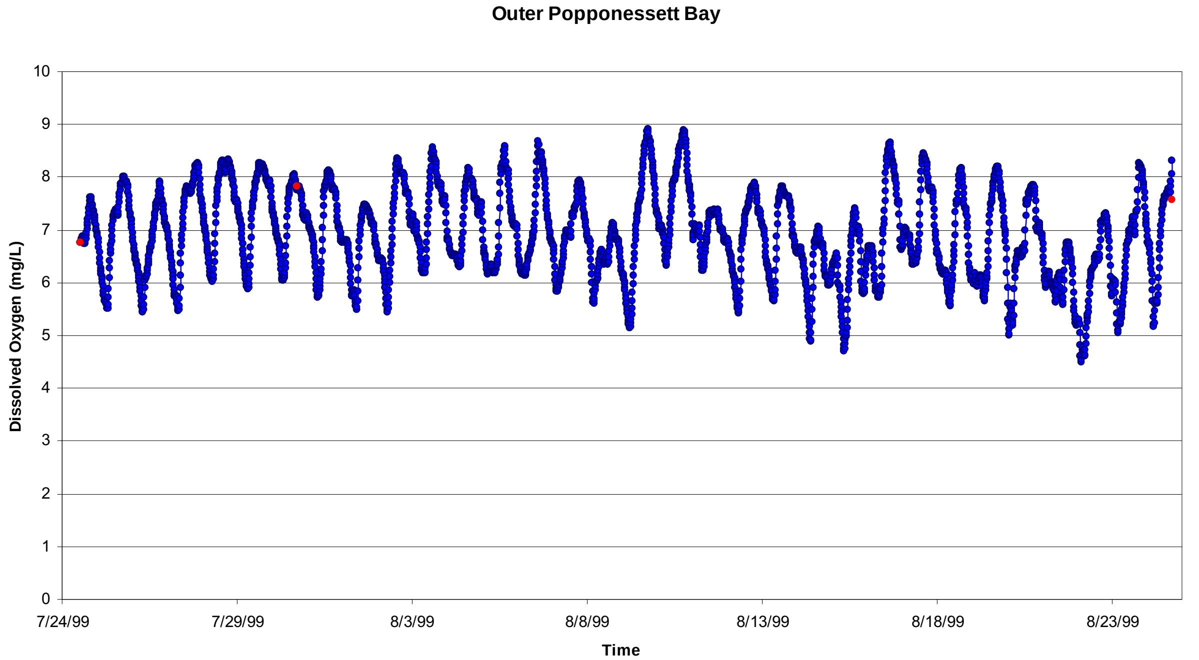


Figure VII-3. Bottom water record of dissolved oxygen at Outer Popponesset Bay station by Daniels Island, Summer 1999. Calibration samples represented as red dots.

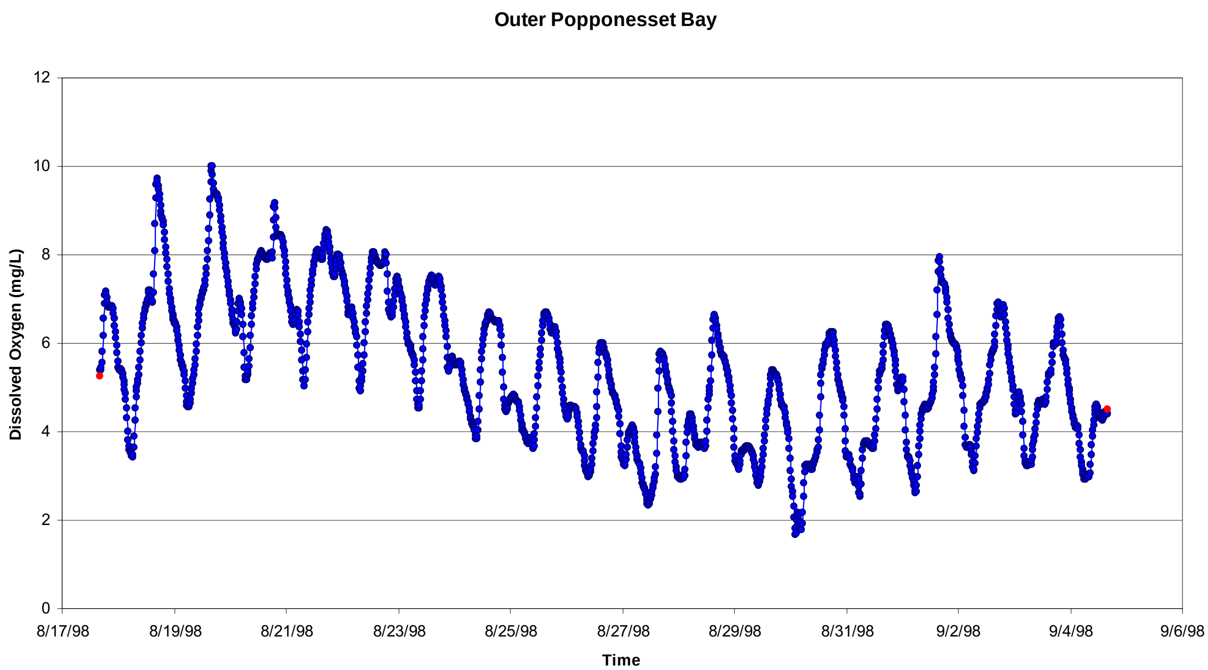


Figure VII-4. Bottom water record of dissolved oxygen in Outer Popponesset Bay station in Channel 2, Summer 1998. Calibration samples represented as red dots.

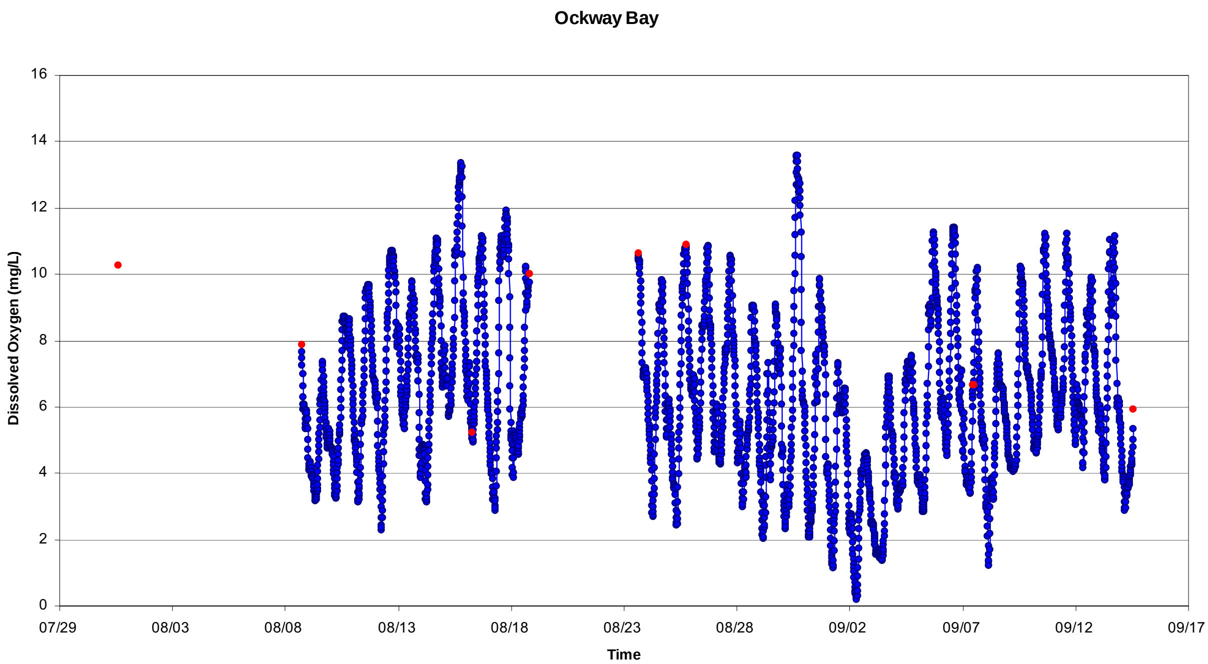


Figure VII-5. Bottom water record of dissolved oxygen in Ockway Bay, Summer 1997. Calibration samples represented as red dots.

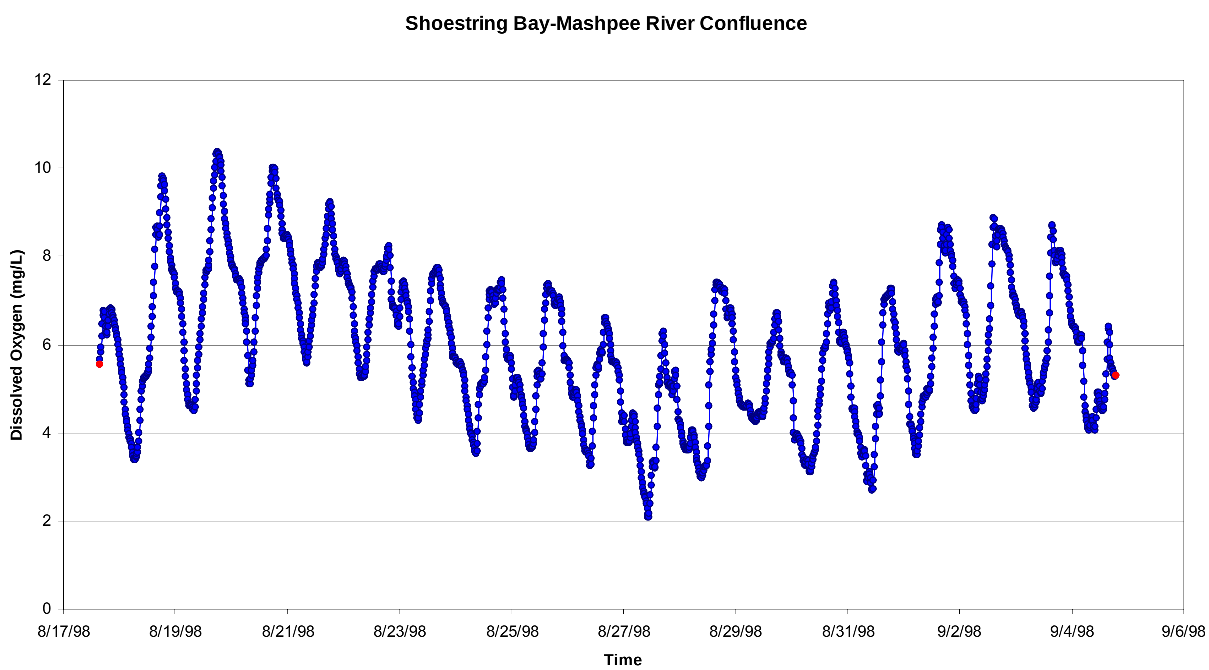


Figure VII-6. Bottom water record of dissolved oxygen at the confluence of the Mashpee River (lower) and Shoestring Bay, Summer 1998. Calibration samples represented as red dots.

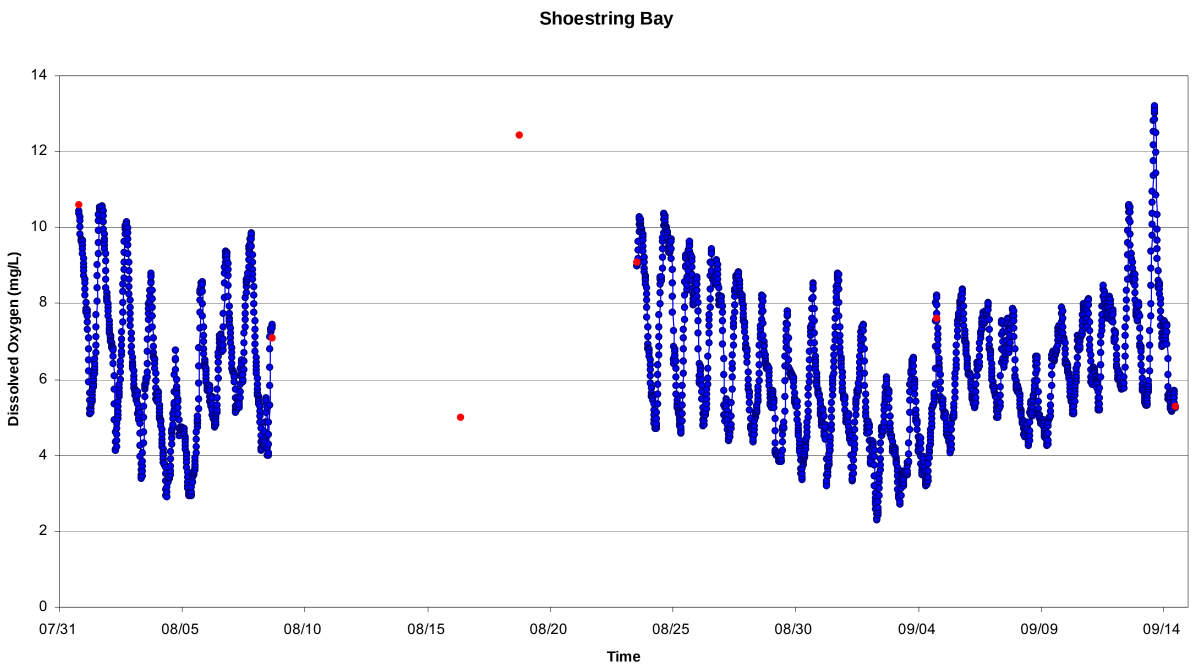


Figure VII-7. Bottom water record of dissolved oxygen in Shoestring Bay, Summer 1997. Calibration samples represented as red dots.

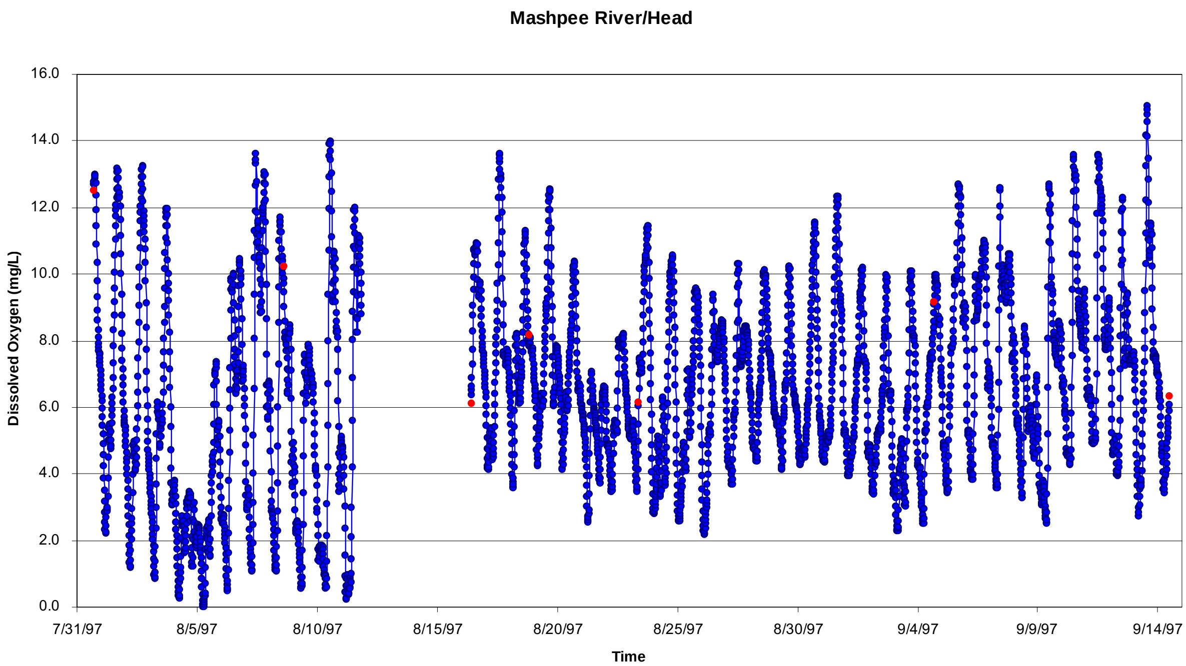


Figure VII-8. Bottom water record of dissolved oxygen in upper Mashpee River, Summer 1997. Calibration samples represented as red dots.

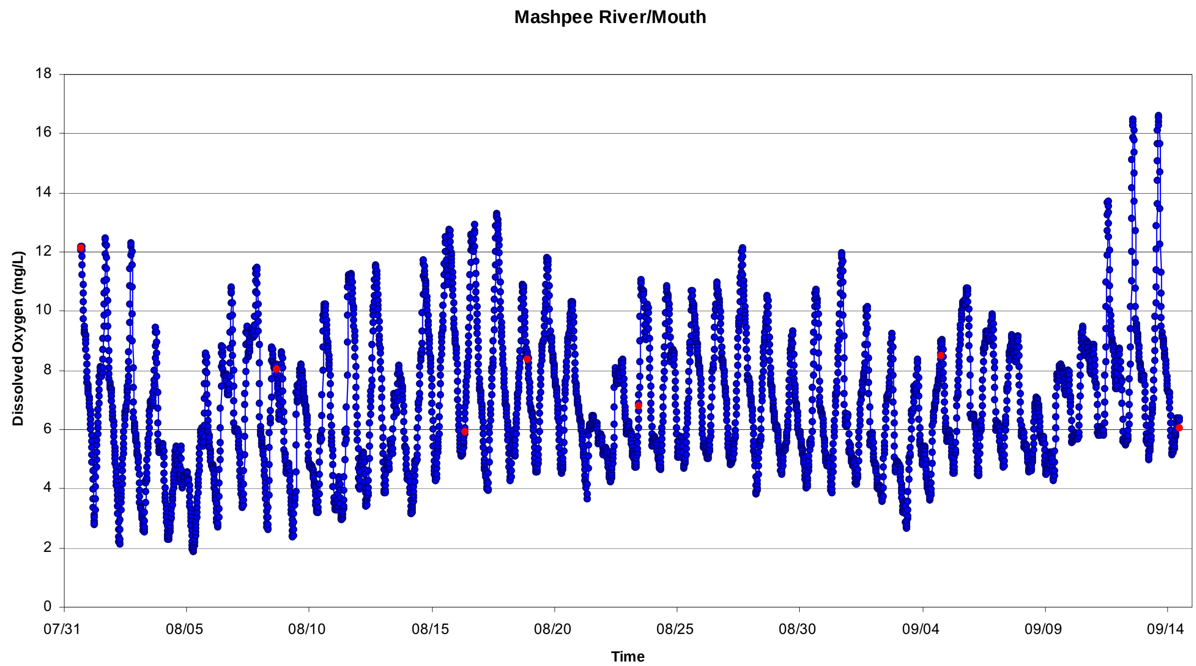


Figure VII-9. Bottom water record of dissolved oxygen in Mashpee River (lower) at the mouth, Summer 1999. Calibration samples represented as red dots.

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

Massachusetts Estuaries Project Town of Mashpee: 1998-1999	**Dissolved Oxygen: Continuous Record, Summer 1997-1999**				
	Deployment Days	**<6 mg/L (% of days)**	**<5 mg/L (% of days)**	**<4 mg/L (% of days)**	**<3 mg/L (% of days)**
Mashpee River Upper	44.8	51%	40%	27%	20%
Mashpee River/ Mouth	44.8	43%	22%	8%	2%
Mashpee River/Shoestring Bay	18.2	52%	31%	13%	2%
Ockway Bay	32.0	47%	33%	19%	8%
Shoestring Bay	29.8	48%	25%	9%	1%
Popponesset Bay Channel 2	18.0	60%	41%	24%	6%
Popponesset Bay Daniels Isle	31.2	15%	1%	0%	0%

clear that the Mashpee 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, see below). 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. This is the case in the Mashpee River and to a lesser extent in the other basins.

Chlorophyll a data 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). It is clear that the upper regions of the Popponesset Bay System periodically have large phytoplankton blooms and that the general gradient in chlorophyll levels is consistent with the observed distribution in oxygen depletion of bottom waters. The Mashpee River and Shoestring Bay supported phytoplankton blooms >15 ug L^{-1} on about one third of the sampling dates (sum of frequencies 15-20, 20-25, >25 ug L^{-1}, Table VII-2). Ockway Bay and upper Popponesset Bay showed lesser blooms and the central region of Popponesset Bay exhibited only moderate-low chlorophyll levels.

Combining the dissolved oxygen and chlorophyll a data yield a clear pattern of nutrient related habitat quality (based on these parameters only, see eelgrass and infaunal indicators below). At present, the central basin of Popponesset Bay supports relatively healthy habitat conditions of consistently high bottom water dissolved oxygen and modest phytoplankton blooms during summer. In contrast, the other regions of the System have moderate to high levels of nitrogen related impairment. Shoestring Bay shows both periodic oxygen declines and significant phytoplankton blooms, while Ockway Bay has similar oxygen declines, but apparently less phytoplankton biomass. Dissolved oxygen measurements in the Mashpee River also indicate nutrient impairment, with extreme oxygen excursions and night-time oxygen depletion on a consistent basis, and significant phytoplankton blooms. The major issue with the Mashpee River is the extent to which its structure as a salt marsh system ameliorates the impact of these water quality features. However, even as a salt marsh, these levels of chlorophyll a and oxygen excursion indicate a moderate level of impairment. Based upon the dissolved oxygen and chlorophyll data the ranking of the Popponesset Bay System components is as follows:

- Popponesset Bay Central Basin – high quality
- Popponesset Bay upper/confluence, Shoestring & Ockway Bays – significantly impaired
- Mashpee River – significantly impaired to degraded (relative to embayments)
  -- moderately to significantly impaired (relative to salt marshes).

Table VII-2. Frequency of grab samples for summer chlorophyll a levels above various benchmark levels within each of the tributary sub-embayments to the Popponesset Bay System. Data collected by the Popponesset Bay Water Quality Monitoring Program and Coastal Systems Program, SMAST. Geometric averages were used to estimate "average" conditions, given the periodic phytoplankton blooms.

		Frequency						Statistics		
Sub-Embayment	**Sta ID**	**<5 ug/L %**	**5-10 ug/L %**	**10-15 ug/L %**	**15-20 ug/L %**	**20-25 ug/L %**	**>25 ug/L %**	**Geo Mean ug/l**	**Geo s.d. ug/L**	**N**
Popponesset Bay System										
Mashpee River	1997-2003									
Upper (PB2)	PB2	23	27	19	12	4	15	11.2	3.1	26
Mid	PB3	11	30	26	19	4	11	12.8	2.7	27
Lower	PB4	14	46	25	7	4	4	8.8	2.0	28
Shoestring Bay	1997-2003									
Upper	PB5	11	30	26	11	4	19	12.3	2.0	27
Mid	PB6	7	39	25	18	7	4	11.2	1.6	28
Lower	PB7	14	57	7	7	14	0	8.2	1.7	28
Ockway Bay	1997-2003									
Upper	PB9	17	55	19	6	4	0	7.7	1.6	27
Lower	PB10	29	54	7	7	0	4	6.5	1.8	28
Pinquickset Cove	2002-2003									
Mid	PB15	57	0	14	0	14	14	7.6	2.7	7
Popponesset Bay	1997-2003									
Upper	PB8	12	67	7	2	5	7	8.0	1.8	27
Mid	PB11	29	58	4	0	0	8	6.8	2.1	24
Lower	PB12	48	41	7	0	0	4	5.1	1.8	27
Popponesset Crk	PB13	50	43	4	0	0	4	5.3	1.8	28

VII.3 EELGRASS DISTRIBUTION - TEMPORAL ANALYSIS

Eelgrass surveys and analysis of historical data was 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 prior to any substantial development of the watershed. The 1951 data were only anecdotally validated, while the 1995 and 2001 maps were field validated. The primary use of the data is to indicate (a) if eelgrass once or currently colonizes a basin and (b) if large-scale system-wide shifts have occurred. Integration of these data sets provides a view of temporal trends in eelgrass distribution from 1951 to 1995 to 2001 (Figure VII-10); the period in which watershed nitrogen loading significantly increased to its present level. This temporal information can be used to determine the stability of the eelgrass community.

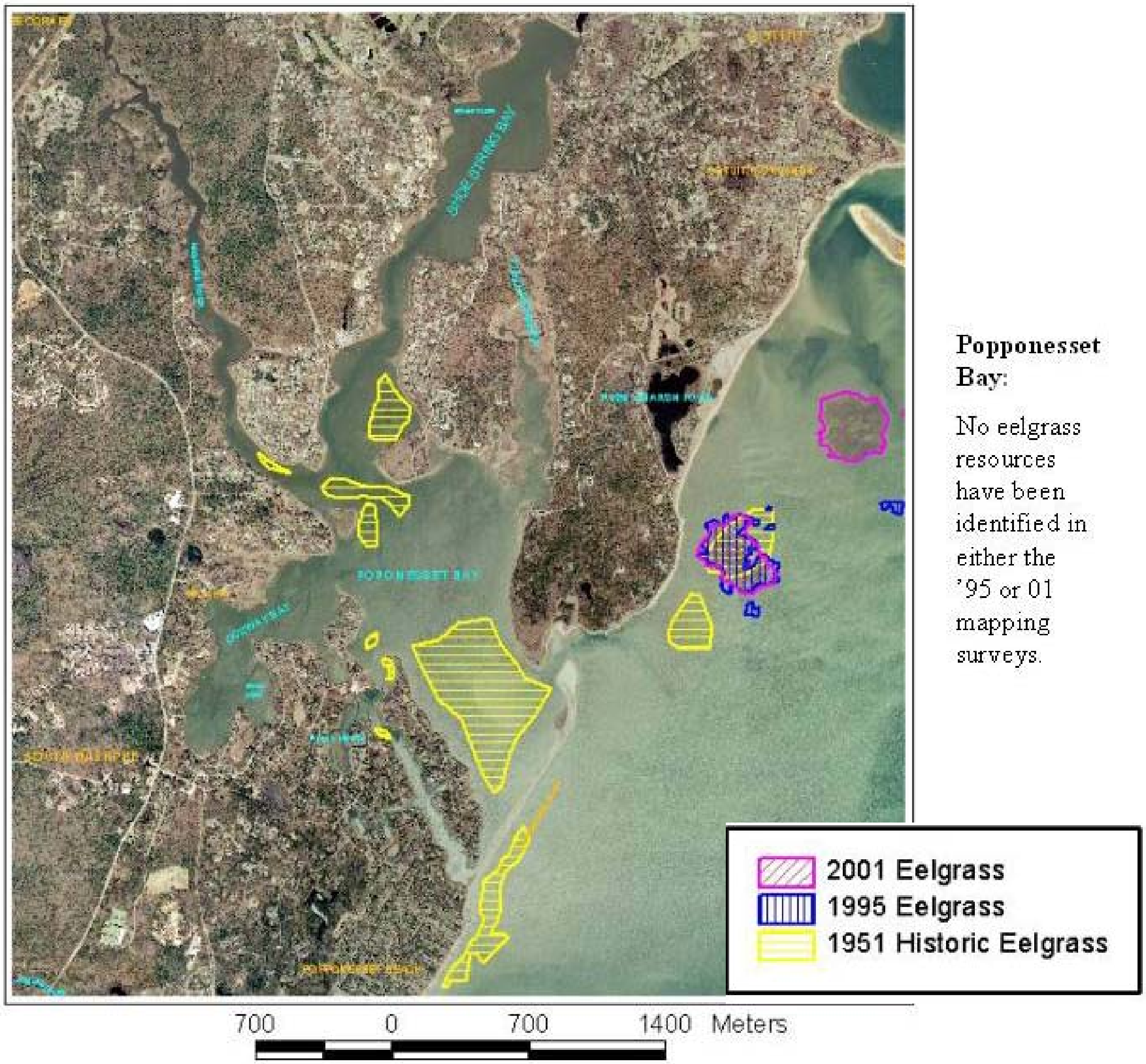
At present, eelgrass is not present within the Popponesset Bay System. In addition, to the DEP mapping, this has been confirmed by the multiple MEP staff conducting the infaunal and sediment sampling and the mooring studies. The current lack of eelgrass beds is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within this system. However, it appears that a substantial area of the central basin supported eelgrass beds in 1951. In addition, there were smaller beds within the upper region of the main basin, at the mouth to Shoestring Bay. The pattern of these beds is consistent with the pattern of nitrogen related habitat quality which is currently observed within the System. It appears that as the Bay became nutrient enriched, that these sites could no longer support eelgrass beds. However, it is likely that if nitrogen loading were to decrease that eelgrass could first be restored in the lower portion of the main basin and with further reductions, be restored to the 1951 pattern.

It is significant that eelgrass was not detected in Shoestring Bay or Ockway Bay in the 1951 data. It appears that these systems are not supportive of this type of habitat. Given the structure of these sub-embayments and their sediment types, it appears that these are natural depositional basins and may not be conducive to supporting rooted macrophytes. The lack of eelgrass in the Mashpee River is consistent with its role as a salt marsh system, which drains completely at low tide in some regions and which is "naturally" organic rich. For these reasons, salt marshes typically do not support eelgrass beds within their main channels.

In systems like Popponesset Bay, 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.

Other factors which influence eelgrass bed loss in embayments may also be at play in Popponesset Bay, 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 Bay supports few boat moorings. Similarly, pier construction and boating pressure may be adding additional stress in nutrient enriched areas, but do not seem to be the overarching factor. It is not possible at this time to determine the potential effect of shellfishing on eelgrass bed distribution, although it must be small as there is little shellfishing on an areal basis in the Bay.

As for the apparent decline along Popponesset Spit and just off the inlet, there are 2 major likely causes. First, coastal processes in this area are highly dynamic. This is particularly true in the nearshore to the spit, where there have been several storm overwash events and the need for beach nourishment to maintain the spit. There was a major hurricane in 1954 just after the 1951 survey, which surely affected the area. Directly off the inlet there has been periodic maintenance dredging, without which the tidal flushing of the Bay would be reduced, magnifying the eutrophic conditions of the entire system. Note in Figure V-2 the significant reduction in the length of Popponesset Spit through the area in question, since 1951. The second cause may be related to the nitrogen plume emanating from Popponesset Bay on the ebb tide. This plume can result in elevated nitrogen levels, though the elevated nitrogen plume levels would be lower than what would typically lead to eelgrass bed loss. Boat traffic cannot be ruled out either.



Popponesset Bay:

No eelgrass resources have been identified in either the '95 or 01 mapping surveys.

Figure VII-10. Eelgrass bed distribution within the Popponesset Bay System. 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 was provided by the DEP Eelgrass Mapping Program.

It is not possible to determine a general idea of short- and long-term rates of change in eelgrass coverage from the mapping data, since there is only one survey with eelgrass. However, it is possible to utilize the 1951 coverage data as an indication that an eelgrass bed might be recovered on the order of 100 acres, if nitrogen management alternatives were implemented (Table VII-3). Even more area is possible, if the upper portion of the central basins, which did not contain eelgrass in 1951, could also be restored. Note that restoration of this habitat will necessarily result in restoration of other resources in Shoestring and Ockway Bays and in the region of the mouth to the Mashpee River.

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

Table VII-3. Changes in eelgrass coverage in the Popponesset Bay sub-systems within the Town of Mashpee over the past half century (C. Costello).

Embayment	1951 (acres)	1995 (acres)	2001 (acres)	% Difference (1951 to 2001)
Popponesset Main Bay	**85.41**	**0**	**0**	**100 %**
Shoestring Bay	**10.64**	**0**	**0**	**100 %**
Mashpee River	**0.83**	**0**	**0**	**100 %**
Ockway Bay	**0**	**0**	**0**	
Pinquickset Cove	**0**	**0**	**0**	
There is presently no eelgrass within the Popponesset Bay System				

VII.4 BENTHIC INFAUNA ANALYSIS

Quantitative sediment sampling was conducted at 8 locations throughout the Popponesset Bay System (Figure VII-11). In some cases multiple assays were conducted. In all areas and particularly those that do not support eelgrass beds (hence all of the Popponesset Bay System), 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 Popponesset Bay System is clearly impaired by nutrient overloading. However, to the extent that it can still support healthy infaunal communities, the benthic infauna analysis is important for determining the level of impairment (moderately impaired→significantly impaired→severely degraded). This assessment is also important for the establishment of site-specific nitrogen thresholds (Chapter VIII).

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

The Infauna Study indicated that all areas with the exception of the lower station within the central basin of Popponesset Bay are presently moderately to severely degraded (Table VII-4). Upper Ockway Bay was found to support the poorest infaunal communities within the system, based upon the very low number of species and individuals. Although the 2 species found (compared to 31 in the central basin) were indicative of healthy conditions, the low numbers (20's compared to 400-500 typically) indicated that this system cannot support a community. The indication of better habitat in the Mashpee River, although listed as significantly impaired, results from it supporting a community indicative of a riverine salt marsh. The stress indicator species present were dominated by *Cyathura polita,* which is tolerant of the salinity stress and helps to define this as a marsh system. However, the total numbers of individuals and diversity tends to be low, indicative of a significantly impaired resource. Shoestring Bay and the uppermost portion of the Popponesset Bay central basin both showed a resource that was categorized between moderate and significant impairment. The numbers of individuals was generally high (500-600 per 0.018 m^2) representing a moderate number of species. Diversity was also moderate to high and distributed between indicators of healthy and stressed conditions (Table VII-5), again indicative of moderate impairment. In contrast, the Lower Popponesset Bay station supports a relatively healthy infaunal community, with nearly double the species of other sites and high numbers of individuals (~500 per 0.018 m^2). The high diversity and general evenness are consistent with a healthy community. The indication of moderate impairment stems from the presence of stress indicator species. The overall results indicate a system capable of supporting diverse healthy communities in the region nearest the tidal inlet, with most of the system having infaunal habitat that is significantly impaired under present nitrogen loading conditions.

Figure VII-11. Aerial photograph of Popponesset Bay showing location of benthic infaunal sampling stations (red symbol).

Table VII-4. Benthic Infaunal Community Assessment for Popponesset Embayments. Samples collected Fall of 1998. All data is represented as per 0.018 m^2. Indicator assessment based upon life history information. Classification is based upon DEP-SMAST proposed thresholds. Some species are unclassified so the Total Species do not always equal the Indicators.

Popponesset Bay System			Benthic Infaunal Community - Indicators						Classification
Sub-Embayment	Total #Species	Total #Individuals	Healthy #Species	Healthy #Individual	Transition #Species	Transition #Individuals	Stressed #Species	Stressed #Individuals	
Mashpee River									
Mid - B5 Rep 1	6	98	0	0	3	57	3	41	Significant Impairment
Rep 2	7	195	1	1	2	79	4	115	Significant Impairment
Lower – B6 Rep 1	12	223	6	31	2	41	4	151	Significant Impairment
Ockway Bay									
Inner - B4 Rep 1	2	26	2	26	0	0	0	0	Severe Degradation[1]
Rep 2	2	10	2	10	0	0	0	0	Severe Degradation[1]
Rep 3	2	11	2	11	0	0	0	0	Severe Degradation[1]
Outer – B3 Rep 1	12	68	6	18	2	5	3	30	Significant Impairment
Rep 2	14	126	6	30	3	26	3	60	Significant Impairment
Shoestring Bay									
Inner - B7 Rep 1	15	666	6	161	3	92	4	405	Moderate-Significant Imp[2]
Rep 2	16	645	6	274	3	50	5	312	Moderate-Significant Imp[2]
Rep 3	16	474	7	171	4	95	4	205	Moderate-Significant Imp[2]
Mid – B8 Rep 1	15	534	5	133	6	225	4	176	Moderate-Significant Imp[2]
Popponesset Bay									
Upper – B1 Rep 1	8	500	2	6	3	332	2	160	Moderate-Significant Imp[2]
Rep 2	6	642	1	2	3	340	2	300	Moderate-Significant Imp[2]
Rep 3	9	962	3	22	3	394	2	544	Moderate-Significant Imp[2]
Rep 4	11	88	5	16	2	5	3	57	Moderate-Significant Imp[2]
Lower - B2 Rep 1	31	489	18	105	5	180	5	200	Healthy – Moderate Imp[3]

1 Based upon the virtual absence of animals in numbers of individuals and species.
2 Based upon the total number of species and individuals and the large contribution of stress species
3 Based upon the total number of species (healthy) and total number of individuals and the fraction of transition and stress species

Table VII-5. Benthic infaunal community data for the Popponesset Bay embayment system. Estimates of the number of species adjusted to the number of individuals and diversity (H') and Evenness (E) of the community allow comparison between locations (Samples represent surface area of 0.018 m2).

Sub-Embayment	Location	Total Actual Species	Total Actual Individuals	Species Calculated @75 Indiv.	Weiner Diversity (H')	Evenness (E)
Popponesset Bay System						
Mashpee River	Mid - PB5					
	Rep 1	6	98	4	1.59	0.61
	Rep 2	7	195	6	1.75	0.62
	Lower - PB6					
	Rep 1	12	223	8	1.76	0.49
Ockway Bay	Inner - PB4					
	Rep 1	2	26	N/A	0.24	0.24
	Rep 2	2	10	N/A	0.47	0.47
	Rep 3	2	11	N/A	0.44	0.44
	Outer - PB3					
	Rep 1	12	68	N/A	3.05	0.85
	Rep 2	15	128	14	3.02	0.77
Shoestring Bay	Inner - PB7					
	Rep 1	15	666	11	2.69	0.69
	Rep 2	17	645	10	2.70	0.66
	Rep 3	16	474	10	2.55	0.64
	Mid - PB8					
	Rep 1	15	534	9	2.71	0.69
Popponesset Bay	Upper - PB1					
	Rep 1	8	500	6	1.63	0.54
	Rep 2	6	642	5	1.66	0.64
	Rep 3	9	962	6	1.76	0.55
	Rep 4	11	88	10	2.48	0.72
	Lower - PB2					
	Rep 1	31	489	17	3.39	0.68

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

VIII.1. ASSESSMENT OF NITROGEN RELATED HABITAT QUALITY

Determination of site-specific nitrogen thresholds for an embayment requires integration of key habitat parameters (infauna and eelgrass), sediment characteristics, and nutrient related water quality information (particularly dissolved oxygen and chlorophyll a). Additional information on temporal changes within each sub-embayment and its watershed further strengthen the analysis. These data were collected to support threshold development for the Popponesset Bay System by MEP Team and were discussed in Chapter VII. Nitrogen threshold development builds on this data and links habitat quality to summer water column nitrogen levels from the long-term baseline water quality monitoring program. At present the bulk of the Popponessett Bay System is showing significantly impaired habitat quality (Chapter VII).

Eelgrass: Since the system once supported eelgrass beds (~100 acres in 1951) and now has lost all eelgrass coverage, the highest habitat evaluation currently possible is moderate impairment (Howes et. al 2003). The level of impairment after the loss of eelgrass can be determined from the dissolved oxygen and phytoplankton biomass (chlorophyll) and infaunal communities. The current lack of eelgrass beds is expected given the high chlorophyll a and low dissolved oxygen levels and watercolumn nitrogen concentrations within this system. However, it appears that a substantial area of the central basin supported eelgrass beds in 1951. In addition, there were smaller beds within the upper region of the main basin, at the mouth to Shoestring Bay. The pattern of these beds is consistent with the pattern of nitrogen related habitat quality, which is currently observed within the System. It appears that as the Bay became nutrient enriched, that these sites could no longer support eelgrass beds. However, it is likely that if nitrogen loading were to decrease, eelgrass could first be restored in the lower portion of the main basin and with further reductions, be restored to the 1951 pattern.

It is significant that eelgrass was not detected in Shoestring Bay and Ockway Bay in the 1951 data. It appears that these systems are not supportive of this type of habitat. Given the structure of these sub-embayments and their sediment types, it appears that these are natural depositional basins and may not be conducive to supporting rooted macrophytes. It is also possible that the tidal flushing of the Popponesset Bay System has historically varied from unrestricted to restricted as the inlet has migrated. This variation may have created nutrient related habitat quality issues within Ockway and Shoestring Bays, even under the low watershed nitrogen loading levels generally associated with the 1951 population. The lack of eelgrass in the Mashpee River is consistent with its role as a salt marsh system, which drains completely at low tide in the upper regions and which is “naturally” organic rich. For these reasons, salt marshes typically do not support eelgrass beds within their main channels.

Water Quality: At present, the central basin of Popponesset Bay supports relatively healthy habitat conditions of consistently high bottom water dissolved oxygen and modest phytoplankton blooms during summer. In contrast, the other regions of the System have moderate to high levels of nitrogen related impairment. Shoestring Bay shows both periodic oxygen declines and significant phytoplankton blooms, while Ockway Bay has similar oxygen declines, but apparently less phytoplankton biomass. Dissolved oxygen measurements in the Mashpee River also indicate nutrient impairment, with extreme oxygen excursions and night-time oxygen depletion on a consistent basis, and significant phytoplankton blooms. The major issue with the Mashpee River is the extent to which its structure as a salt marsh system ameliorates the impact of these water quality features. However, even as a salt marsh, these levels of chlorophyll a and oxygen excursion indicate a moderate level of impairment. Based upon the dissolved oxygen and chlorophyll data the ranking of the Popponesset Bay System components is as follows:

- Popponesset Bay Central Basin – high quality
- Popponesset Bay upper/confluence, Shoestring & Ockway Bays – significantly impaired
- Mashpee River – significantly impaired to degraded (relative to embayments)
  -- moderately to significantly impaired (relative to salt marshes).

Infaunal Communities: The Infauna Study indicated that all areas, except for the lower station within the central basin of Popponesset Bay, are presently moderately to severely degraded (Table VII-5). Upper Ockway Bay was found to support the poorest infaunal communities within the system, based upon the very low number of species and individuals. Although the 2 species found (compared to 31 in the central basin) were indicative of healthy conditions, the low numbers (20's compared to 400-500 typically) indicated that this system is not supporting a productive or diverse benthic community. The indication of better habitat, although listed as significantly impaired, is seen in the Mashpee River results. The River is currently supporting a community indicative of a riverine salt marsh. However, the total numbers of individuals and diversity tend to be low, indicative of an impaired resource (significant impaired on the overall classification scale). Shoestring Bay and the uppermost portion of the Popponesset Bay central basin both showed a resource between moderate and significant impairment. The numbers of individuals was generally high and were distributed among a moderate number of species. Diversity was also moderate to high and distributed between indicators of healthy and stressed conditions (Table VII-5), again indicative of moderate impairment. In contrast the Lower Popponesset Bay station supports a relatively healthy infaunal community, with nearly double the species of other sites and high numbers of individuals. The high diversity (H') and general evenness (E) are consistent with a healthy community. The indication of moderate impairment stems from the presence of stress indicator species. The overall results indicate an embayment system capable of supporting diverse healthy communities in the region nearest the tidal inlet, with most of the system having infaunal habitat that is significantly impaired under present nitrogen loading conditions.

Overall, all of the indicators show a consistent pattern of moderate impairment of the lower portion of the central basin of Popponesset Bay, primarily based upon its loss of eelgrass. While the upper central bay and Shoestring Bay are moderately to significantly impaired and Ockway Bay is significantly impaired to severely degraded based primarily upon the infaunal community data and the extent and duration of bottom water dissolved oxygen depletion. The Mashpee River appears to be functioning as a riverine salt marsh. However, due to its impoverished benthic community in the upper reach and the extreme dissolved oxygen excursions and phytoplankton blooms, it appears to be nutrient overloaded at present.

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 is determined, the Linked Watershed-Embayment Model is used to sequentially adjust nitrogen loads until the targeted nitrogen concentration is achieved.

Within the Popponesset Bay System the region between the upper portion of the central basin to Popponesset Bay and the mouth of Shoestring Bay was selected as the sentinel region (PBh in Figure VI-1). This location was selected because (1) it was the upper extent of the eelgrass coverage in 1951, (2) restoration of nitrogen conditions supportive of eelgrass at this location will necessarily result in even higher quality conditions throughout the whole of the central basin, and (3) restoration of nitrogen concentrations at this site should result in conditions similar to 1951 within Shoestring and Ockway Bays. Shoestring Bay and Ockway Bay should then be supportive of high quality habitat for benthic infaunal communities (confirmed as described below).

The target nitrogen concentration for restoration of eelgrass in this system was determined to be 0.38 mg TN L^{-1}. It was not possible to make this determination based upon an analysis of the relationship of measured nitrogen levels to existing eelgrass beds in Popponesset Bay, as all beds have been lost. Instead, the value stems from (1) the analysis of Stage Harbor, Chatham which also exchanges tidal water with Nantucket Sound (for which a MEP target has already been set), (2) analysis of nitrogen levels within the eelgrass bed in adjacent Waquoit Bay, near the inlet (measured TN of 0.395 mg N L^{-1}, tidally corrected <0.38 mg N L^{-1}), and (3) a similar analysis in West Falmouth Harbor. The sentinel station under present loading conditions supports a measured nitrogen level at mid-ebb tide of 0.581 mg TN L^{-1} and a tidally corrected average concentration of 0.451 mg TN L^{-1}. Based upon sequential reductions in watershed nitrogen loading in the analysis described in the section below (VIII-3), the sentinel station achieved an average TN level of 0.371 mg L^{-1} , the mouth of Ockway Bay, 0.376 mg TN L^{-1} and the whole of the Popponesset Bay basin <0.331 mg TN L^{-1}. This indicates that significant eelgrass habitat restoration would occur within the regions of the 1951 coverage. It is possible also to evaluate the response in benthic infaunal habitat. At present, the regions supporting the highest quality infaunal habitat have tidally averaged concentrations (mg TN L^{-1}) from 0.692 in the moderate-significantly impaired Shoestring Bay sites, to 0.451 in the similar upper Popponesset Bay site to 0.325 at the watercolumn site closest to the infaunal sampling site in the lower Bay basin. This latter concentration is likely too low and results from the fact that the model results are at the lower end of the allowable fit to the measured data at this site. The measured TN concentration is currently 0.456 mg TN L^{-1}. In any case, the data suggest that there is likely a range of total nitrogen that can support healthy infauna within this system. Based upon the infaunal analysis (Chapter VII) coupled with the nitrogen data (measured and modeled), nitrogen levels on the order of 0.4 to 0.5 mg TN L-1 are likely supportive of high infaunal habitat quality in this system. It should be noted that these values were not used for setting nitrogen thresholds in this embayment system. These values merely provide a check on the acceptability of conditions in Shoestring and Ockway Bays, as well as Lower Mashpee River, at the point that the threshold level is attained at the sentinel station. The results of the Linked Watershed-Embayment modeling, when the nitrogen threshold is attained (Section VIII-3), yield TN levels in these regions within the acceptable range: upper to lower Shoestring Bay, 0.522 to 0.412 mg TN L^{-1}; upper Ockway Bay, 0.421 mg TN L^{-1}; and mid to lower Mashpee River, 0.525 to 0.422 mg TN L^{-1}. Therefore, it appears that achieving the nitrogen target at the sentinel location is restorative of eelgrass habitat throughout the Popponesset Bay central basin and restorative of infaunal habitat throughout Shoestring and Ockway Bays, as well as the lower portion of the Mashpee River.

VIII.3. DEVELOPMENT OF TARGET NITROGEN LOADS

The tidally averaged total nitrogen thresholds derived in Section VIII-1 were used to adjust the calibrated constituent transport model developed in Section VI. Watershed nitrogen loads were sequentially lowered, using reductions in septic effluent discharges only, until the nitrogen levels reached the threshold level at the sentinel region for the Popponesset Bay System. It is important to note that load reductions can be produced by reduction of any or all sources or by increasing the natural attenuation of nitrogen within the freshwater systems to the embayment. The load reductions presented below represent only one of a suite of potential reduction approaches that need to be evaluated by the communities. The presentation is to establish the general degree and spatial pattern of reduction that will be required for restoration of this nitrogen impaired embayment.

As shown in Table VIII-1, the nitrogen load reductions within the Popponesset Bay System necessary to achieve the threshold nitrogen concentrations were relatively high, 100% removal of septic load (associated with direct groundwater discharge to the embayment) required within four sub-embayments (Popponesset Creek, Ockway Bay, Mashpee River, and Shoestring Bay). In addition, a portion of the septic load entering the estuarine system via Mashpee and Santuit Rivers also must be removed to meet the threshold nitrogen concentrations. For the load reduction scenario evaluated, the Mashpee River and Santuit River required removal of approximately 41% and 35% of their septic load, respectively. The distribution of tidally-averaged nitrogen concentrations associated with the above thresholds analysis is shown in Figures VIII-1 and VIII-2.

Tables VIII-2 and VIII-3 provide additional loading information associated with the thresholds analysis. Table VIII-2 shows the change to the total watershed loads, based upon the removal of septic loads depicted in Table VIII-1. In general, removal of 100% of the septic load from Popponesset Creek, Ockway Bay, Mashpee River, and Shoestring Bay results in an 80% to 85% reduction in total nitrogen load from these sub-watersheds. Table VIII-3 shows the breakdown of threshold sub-embayment and surface water loads used for total nitrogen modeling. For Table VIII-3, loading rates are shown in kilograms per day, since benthic loading varies throughout the year and the values shown represent ‘worst-case’ summertime conditions. The benthic flux for this modeling effort is reduced from existing conditions based on the load reduction and the observed particulate organic nitrogen (PON) concentrations within each sub-embayment relative to Nantucket Sound.

The basis for the watershed nitrogen removal strategy utilized to achieve the embayment thresholds may have merit, since this example nitrogen remediation effort is focused on watersheds where groundwater is flowing directly into the estuary. For nutrient loads entering the 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. Presently, this attenuation is occurring due to natural ecosystem processes and the extent of attenuation being determined by the mass of nitrogen which discharges to these systems. The nitrogen reaching these systems is currently “unplanned”, resulting primarily from the widely distributed non-point nitrogen sources (e.g. septic systems, lawns, etc.). Future nitrogen management should take advantage of natural nitrogen attenuation to ensure the most cost-effective nitrogen reduction strategies. However, “planned” use of natural systems has to be done carefully and with the full analysis to ensure that degradation of these systems will not occur. One clear finding of the MEP has been the need for analysis of the potential of restored wetlands or ecologically engineered ponds/wetlands to enhance nitrogen attenuation. Attenuation by ponds in agricultural systems has also been found to work in some cranberry systems, as well. The lower freshwater and salt water reaches of the Mashpee and Santuit Rivers provide opportunities for enhancing natural attenuation of their nitrogen loads. Restoration or enhancement of wetlands and ponds associated with the lower ends of rivers and streams discharging to estuaries is seen as providing a dual service of lowering infrastructure costs associated with wastewater management and increasing aquatic resources associated within the watershed and upper estuarine reaches.

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 Barnstable to develop additional specific water quality modeling scenarios, to be run to evaluate other nitrogen removal strategies. One such proposed scenario, removing the discharges from the existing wastewater facilities from the watershed (pipeline), was partially evaluated by the MEP Team. At present only a tiny fraction (<0.5%) of the watershed nitrogen loading is discharged by the existing treatment facilities. Removing this load would have a very small impact. However, with increased sewering and treatment of wastewater, discharge within the groundwatershed directly discharging to Nantucket Sound has merit. The existing MEP analysis and model provides for the determination of potential discharge sites and the concomitant improvement of the nutrient related habitat quality within the Popponesset Bay System.

Table VIII-1. Comparison of sub-embayment watershed ***septic loads*** (attenuated) used for modeling of present and threshold loading scenarios of the Popponesset 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)	threshold septic load (kg/day)	threshold septic load % change
Popponesset Bay	1.58	1.58	0.0%
Popponesset Creek	4.00	0.00	-100.0%
Pinquickset Cove	0.58	0.58	0.0%
Ockway Bay	2.39	0.00	-100.0%
Mashpee River	9.61	0.00	-100.0%
Shoestring Bay	6.94	0.00	-100.0%
Surface Water Sources			
Mashpee River	9.96	5.85	-41.3%
Santuit River (Shoestring Bay)	11.69	7.58	-35.2%
Quaker Run River (Shoestring Bay)	4.69	4.69	0.0%
TOTAL	51.12	19.96	-61.0%

Table VIII-2. Comparison of sub-embayment ***total watershed loads*** (including septic, runoff, and fertilizer) used for modeling of present and threshold loading scenarios of the Popponesset Bay system. These loads do not include direct atmospheric deposition (onto the sub-embayment surface) or benthic flux loading terms.			
sub-embayment	present load (kg/day)	threshold load (kg/day)	threshold % change
Popponesset Bay	1.82	1.82	0.0%
Popponesset Creek	4.94	0.95	-80.7%
Pinquickset Cove	0.76	0.76	0.0%
Ockway Bay	3.15	0.76	-76.0%
Mashpee River	12.11	2.50	-79.4%
Shoestring Bay	9.21	2.26	-75.5%
Surface Water Sources			
Mashpee River	15.56	11.45	-26.4%
Santuit River (Shoestring Bay)	15.58	11.47	-26.4%
Quaker Run River (Shoestring Bay)	5.98	5.98	0.0%
TOTAL	69.11	37.95	-45.2%

Table VIII-3. Threshold sub-embayment and surface water loads used for total nitrogen modeling of the Popponesset Bay 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)
Popponesset Bay	1.82	4.01	-4.91
Popponesset Creek	0.95	-	-0.62
Pinquickset Cove	0.76	0.29	-0.33
Ockway Bay - lower	-	-	-1.13
Ockway Bay - upper	0.76	1.09	2.24
Mashpee River	2.50	0.66	9.47
Shoestring Bay	2.26	2.23	-8.73
Surface Water Sources			
Mashpee River	11.45	-	-
Santuit River (Shoestring Bay)	11.47	-	-
Quaker Run River (Shoestring Bay)	5.98	-	-
TOTAL	37.9	8.28	

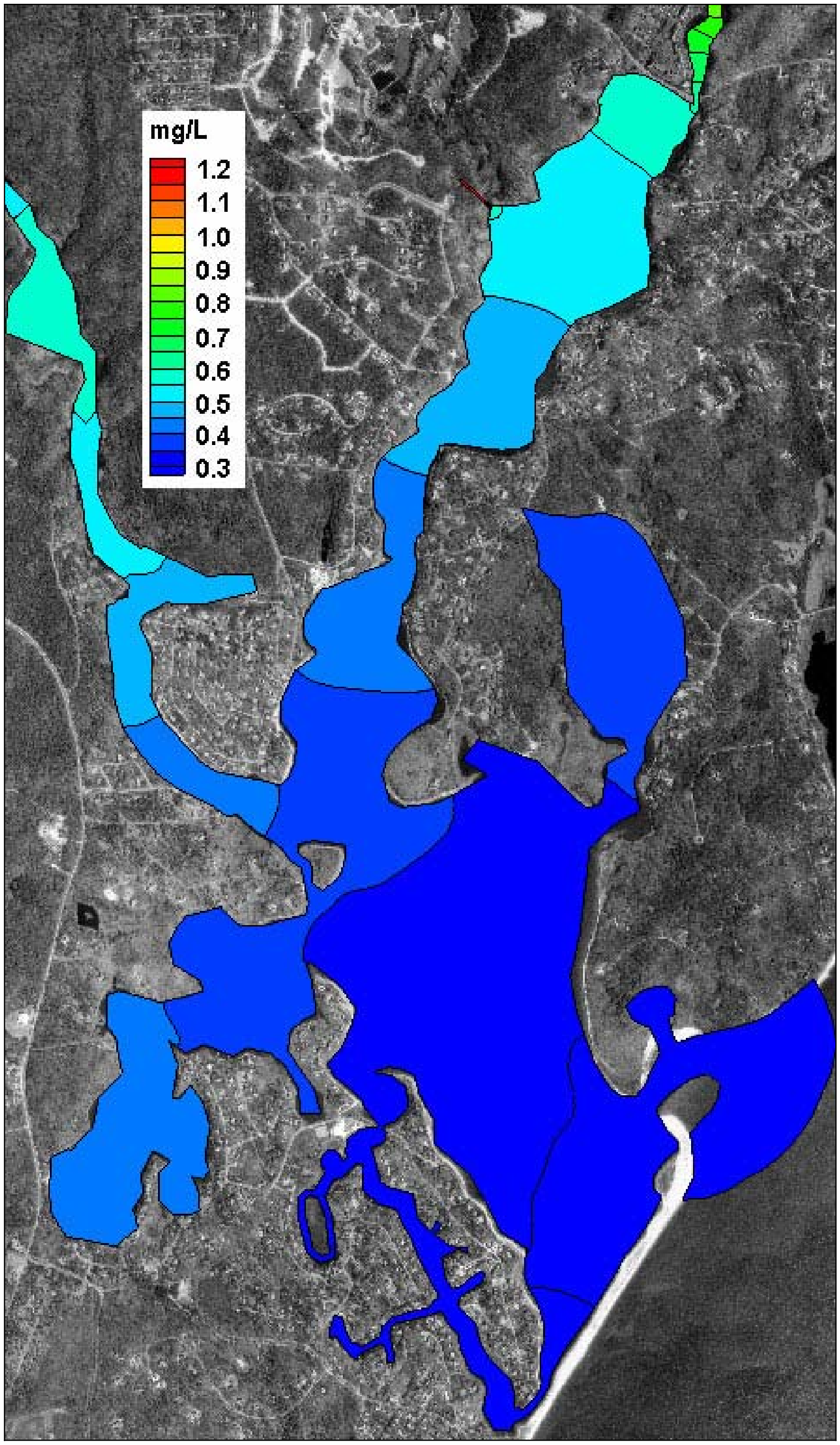


Figure VIII-1. Contour plot of modeled total nitrogen concentrations (mg/L) in the Popponesset Bay system, for threshold conditions (0.38 mg/L at lower Mashpee River, Lower Shoestring Bay, and mid Ockway Bay).

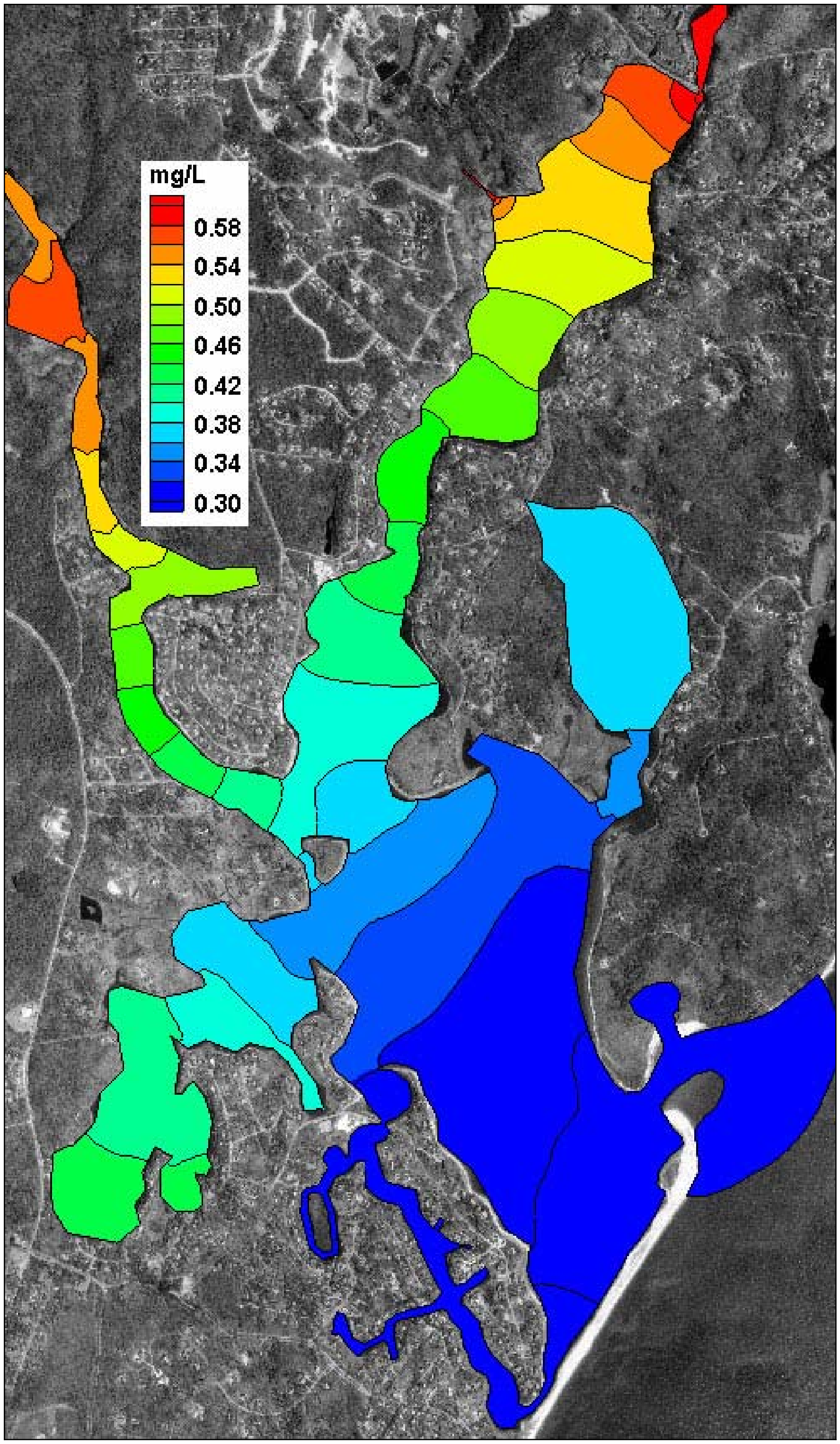


Figure VIII-2. Same results as for Figure VIII-1, but shown with finer contour increments for emphasis. Contour plot of modeled total nitrogen concentrations (mg/L) in the Popponesset Bay system, for threshold conditions (0.38 mg/L at lower Mashpee River, Lower Shoestring Bay, and mid Ockway Bay).

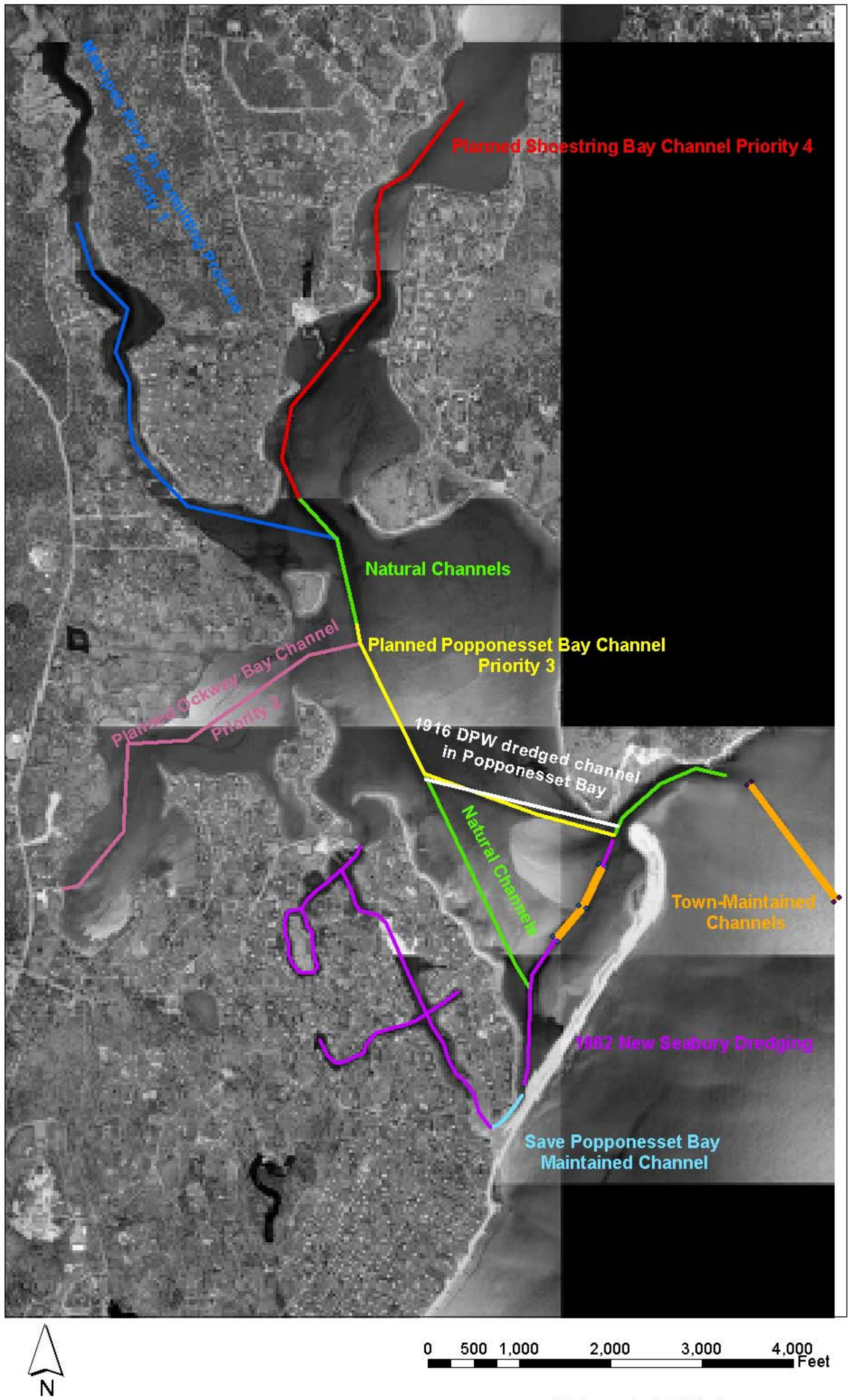
IX. DREDGING IMPACTS TO WATER QUALITY

Keystones of the MEP include modeling support upon request of municipalities and the analysis of non-traditional approaches to nitrogen mitigation. In the case of the Popponesset Bay System, channel dredging was evaluated to determine potential effects on nutrient related water quality. Dredging of inlets and channels is conducted for a variety of reasons throughout southeastern Massachusetts and even within a single system. For example, within New Bedford Inner Harbor dredging has recently been conducted in one region for removal of PCB contaminated sediments and in another region for navigation associated with port activities. In the case of the Popponesset Bay System, maintenance dredging of the inlet has been conducted over the past several years for navigation and for the general health of the system and additionally for beach nourishment (Popponesset Spit). The Mashpee Waterways Commission has developed various dredging plans for dredging within the Bay System, some of which are currently underway, some for which permits are being sought and some for future design (Figure IX-1). The various dredging projects are conducted by both municipal and private entities.

The Popponesset Bay System has had a significant amount of dredging over the past century. In 1916 the DPW dredged a channel from the inlet to mid-bay (Figure IX-1) for navigation. Vestiges of this channel remain today. One of the major dredging projects was conducted by New Seabury in 1962. This effort was related to the development of the Popponesset Creek Watershed and significantly increased the depth within this part of the system, which remains today. Over the past 50 years, dredging has been more for maintenance purposes. Both for the tidal inlet (Town-maintained channels) and related to navigation between Popponesset Bay and Popponesset Creek (Save Popponesset Bay channel). Disposal of appropriate dredged material has been used to stabilize Popponesset Spit, which is subject to erosion and storm overwash. At present, the maintenance projects continue, but there has been an effort by the Town of Mashpee (Waterways Commission) to develop a dredging plan for the interior basins, due to the very shallow nature of Popponesset Bay. The dredge plan has prioritized navigational dredging into priority channels of (1) Mashpee River (in permitting process), (2) Ockway Bay (to public boat ramp), (3) Popponesset Bay central channel, and (4) Shoestring Bay. It is worth noting that the major logistical issue to be resolved in this effort is the disposal of the fine organic enriched dredged material and the major environmental issue involves the effects on the System's nutrient related habitat quality. In this Chapter we address the latter issue.

An evaluation of two dredging alternatives was performed using the hydrodynamic and water quality models developed for Popponesset Bay. The evaluation was performed to determine potential impacts to water quality throughout the Popponesset Bay system resulting from dredging the proposed navigation channels. As a result of discussions with the Mashpee Waterways Commission (Hanks, 2004), the two modeled dredging alternatives selected were 1) dredging an approximately 50 ft wide channel to -3 ft MLW in the Mashpee River (Alternative 1); and 2) dredging an additional -5 ft MLW channel between the lower Mashpee River and Popponesset Bay inlet (Alternative 2). Channel layouts for these two alternatives are shown in Figure IX-2. The navigation channel layout in the lower portion of Popponesset Bay for Alternative 2 follows the approximate route of a channel first dredged in 1916.

Popponesset Bay Channels (Existing, Dredged and Proposed)



Prepared by Mashpee Planning Department

Figure IX-1. Historical and proposed dredging plans for the Popponesset Bay system.

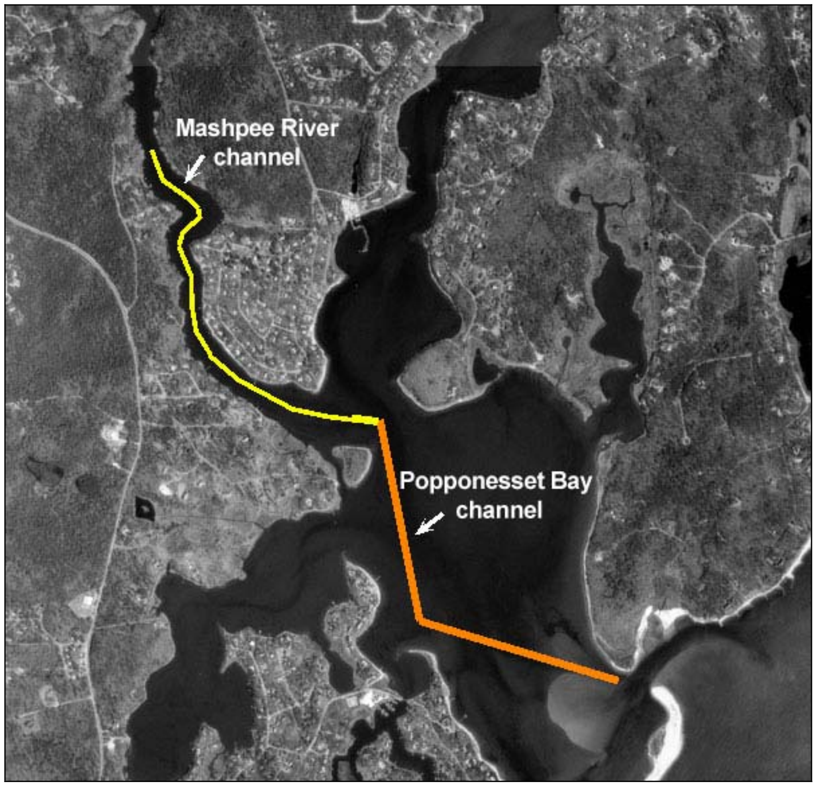


Figure IX-2. Channel layouts for two dredging alternatives in Popponesset Bay: 1) Alternative 1 (yellow line), dredging a -3 ft MLW channel in the Mashpee River, and 2) Alternative 2 (yellow and orange lines), dredging an additional -5 ft MLW channel between the lower Mashpee River and Popponesset Bay inlet.

Modeling these two alternatives first required that the computational grid of Popponesset Bay be modified to incorporate the dredged channels. Next, the hydrodynamic model was rerun using the grid developed for each separate alternative. With the updated hydrodynamic solutions, the total nitrogen model was then rerun, using the same diffusion coefficients used for the modeling of present conditions (Section VI).

Model results showing N concentration changes resulting from Alternatives 1 and 2 are presented in Figures IX-3 and IX-4. Output from the two modeled scenarios shows that the two dredging alternatives have essentially the same effect on N concentrations in Popponesset Bay. Changes in average modeled N concentrations along the Mashpee River for the two modeled alternatives are presented in Figure IX-5, and Table IX-1.

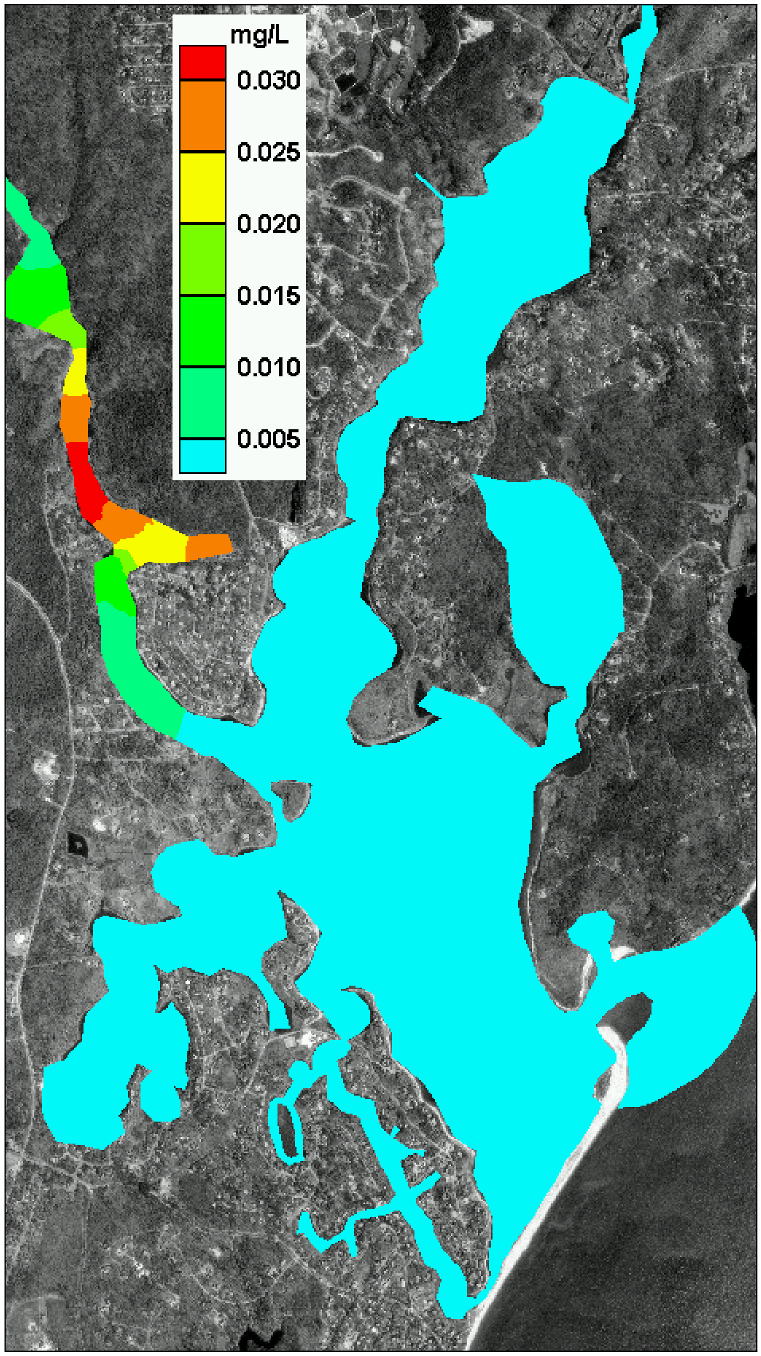


Figure IX-3. Color contours indicating change in nitrogen concentrations (mg/L) resulting from dredging the Mashpee River (Alternative 1). Contours greater than zero indicate N concentration increases over present modeled conditions.

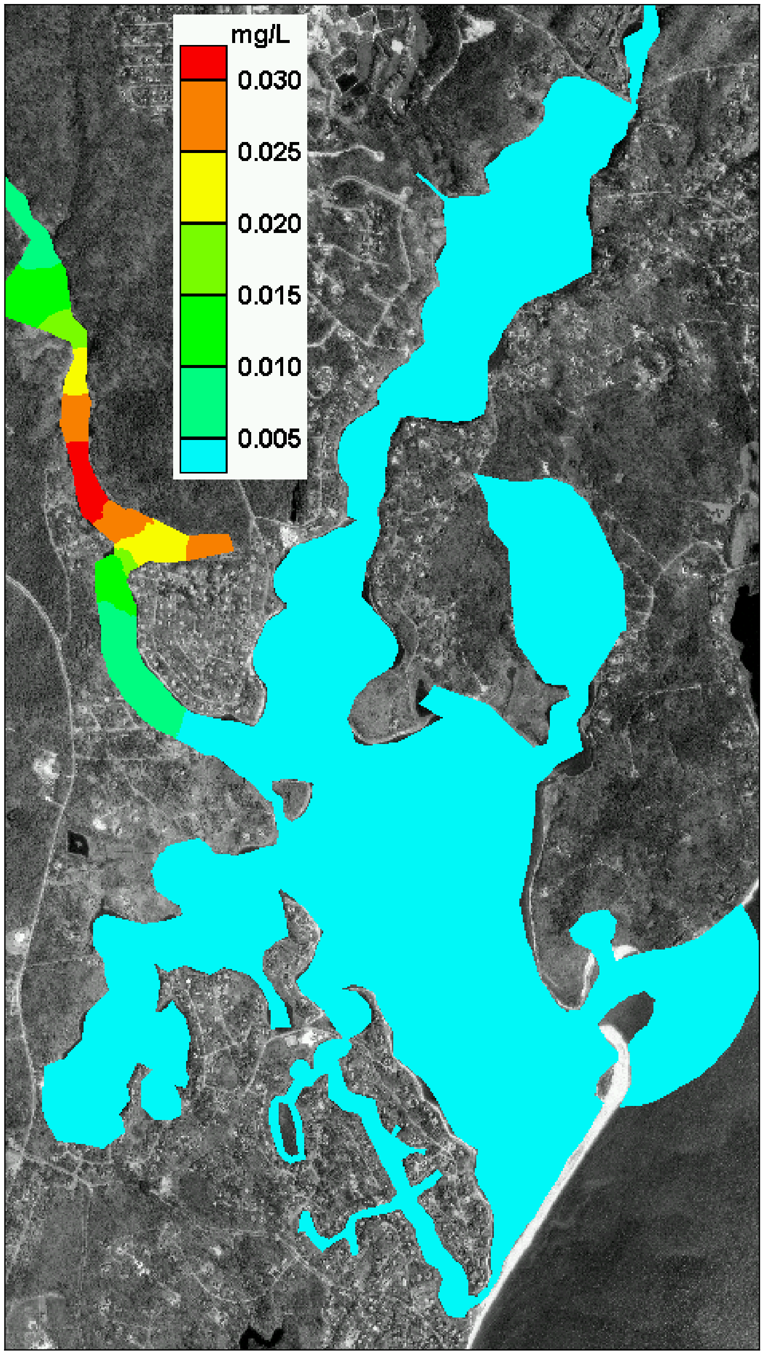


Figure IX-4. Color contours indicating change in nitrogen concentrations (mg/L) resulting from dredging the Mashpee River together with the 1916 channel in Popponesset Bay. (Alternative 2). Contours greater than zero indicate N concentration increases over present modeled conditions.

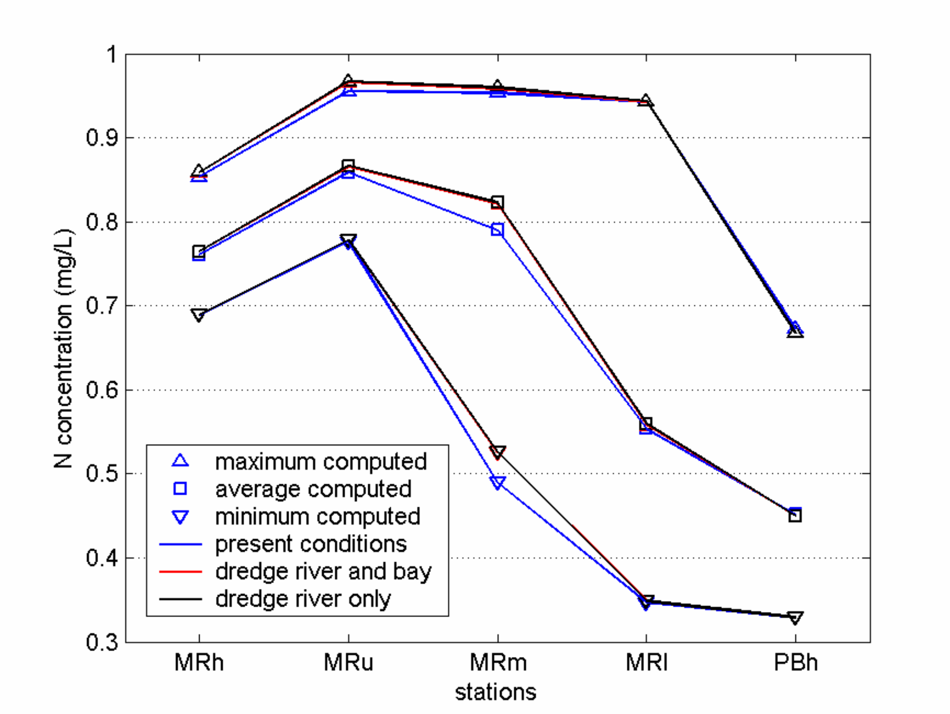


Figure IX-5. Comparison of modeled maximum, average and minimum total N concentrations for present conditions (blue line), and two dredging alternatives (Alternative 1, dredging Mashpee River inlet, black line; Alternative 2, dredging Mashpee River and Popponesset Bay channels). Stations are along the Mashpee River, from the Mashpee River headwater (MRh) to the upper reach of Popponesset Bay (PBh), see Figure VI-1. Because the two dredging alternatives cause the same concentration changes in the Mashpee River, the lines representing conditions for Alternative 1 and 2 are indistinguishable from each other in this plot.

Table IX-1. Comparison of changes in average computed average total N concentrations resulting from two dredging alternatives: 1) Alternative 1, dredging a -3 ft MLW channel in the Mashpee River, and 2) Alternative 2, dredging an additional -5 ft MLW channel between the lower Mashpee River and Popponesset Bay inlet. Positive values indicate increased concentration over present conditions. Stations references are the same as Figure VI-1.

Station Name	SMAST Station Indicator	Alternative 1: dredge Mashpee River only	Alternative 2: dredge River and Bay
		N % change	N % change
Mashpee River head - MRh	PB1	+0.5%	+0.5%
Mashpee River upper - MRu	PB2	+0.9%	+0.8%
Mashpee River mid - MRm	PB3	+4.2%	+4.0%
Mashpee River lower - MRl	PB4	+1.0%	+0.8%
Popponesset Bay head - PBh	PB8	+0.4%	+0.5%

Model results indicate that dredging the proposed channel in the Mashpee River will increase N concentrations along the river, with the greatest increase occurring at the mid-way point of the tidal portion of the river. The reason that nitrogen concentrations increase is due to the increase in mean tide volume of the Mashpee River sub-embayment. Because the tide prism change from pre- and post-dredge conditions is negligible compared to the change in mean tide volume of the sub-embayment (i.e., a 9.0% increase in mean tide volume results from dredging the channel, but there is only a 0.5% increase in tidal prism), tidal excursion (i.e., the distance a parcel of water travels during a half tide cycle) in the river decreases. As a result, residence times in the river increase, which, in turn, increases N concentrations in the river. The change is greatest at the mid-point of the estuarine reach of the river because of the diluting effects of the large freshwater discharge at the head of the river and the lower concentration waters from Popponesset Bay at the opposite end of the sub-embayment, combined with the watershed and benthic N loads which are distributed along the length of the river (which have a concentrating effect).

Results for Alternative 2 indicate that the proposed channel in the lower portion of the bay would have negligible effects on nitrogen concentrations in the main basin of Popponesset Bay. The reason there is little change is that the dredged channel does not significantly affect tidal exchange (i.e., change the volume of the tidal prism) in the Popponesset Bay system. In order for there to be significant reductions in nitrogen concentrations in a coastal embayment, it is usually necessary to increase the tide prism volume. Embayments that would benefit most from dredging likely would have significant attenuation of tidal energy, possibly caused by an under-sized inlet to the ocean. For the Popponesset Bay system, possible improvements to water quality as a result of dredging are small, since tidal attenuation through the system is minimal (see Section V).

Although the results of the modeling for the two dredging scenarios indicated a slight increase in total nitrogen concentrations, it likely is possible to develop dredging scenarios that have no increase in total nitrogen concentrations. In general, the dredging conditions evaluated caused a more significant increase in the mean sub-embayment volume than the increase in tidal flushing could offset, allowing high-nutrient water to reside longer within the Mashpee River. Therefore, to avoid potential negative impacts to water quality, future dredging efforts within the Popponesset Bay system (except for the inlet) should focus on keeping the overall sub-embayment volume fixed at present-day conditions. Dredging scenarios that do not increase sub-embayment volume would require dredged material to remain within the sub-embayment. One option would be to utilize dredge spoils for wetland restoration.

The conditions described above are appropriate for sub-embayments where tidal attenuation is negligible. For the Popponesset Bay system, it is unlikely that dredging will improve water quality within the three main sub-embayments: Ockway Bay, Shoestring Bay, and Mashpee River. However, the hydrodynamic modeling indicated that tidal attenuation is small, but not negligible, across the main inlet to Popponesset Bay. Therefore, the main channel connecting to Nantucket Sound should be maintained to its existing cross-section to avoid further degradation of estuarine health. Although no dredging alternatives were evaluated through the main inlet, the combined hydrodynamic and water quality model provides an ideal tool for evaluating future proposed dredging scenarios.

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